ENDANGERED SPECIES ACT SECTION 7 CONSULTATION
BIOLOGICAL OPINION

Action Agency: National Marine Fisheries Service, Southwest Region, Sustainable Fisheries Division and Protected Resources Division

Activity: Adoption of (1) proposed Highly Migratory Species Fishery Management Plan; (2) continued operation of Highly Migratory Species fishery vessels under permits pursuant to the High Seas Fishing Compliance Act; and (3) Endangered Species Act regulation on the prohibition of shallow longline sets east of the 150° West longitude.

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

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Abstract. To comply with the requirements of the Endangered Species Act of 1973, the National Marine Fisheries Service (NOAA Fisheries) has prepared a biological opinion on the proposed adoption of 1) proposed Highly Migratory Species (HMS) Fishery Management Plan (FMP); 2) continued operation of HMS fishery vessels under permits pursuant to the High Seas Fishing Compliance Act; and 3) Endangered Species Act regulation on the prohibition of shallow longline sets east of the 150° West longitude. The biological opinion considers the effects of longline, troll, drift gillnet, small vessel purse seine, rod and reel, and harpoon fisheries based in California, Oregon, and Washington on threatened and endangered species and critical habitat.

Based on previous patterns of interactions between the fisheries and threatened and endangered marine mammals, the biological opinion concludes that the proposed fisheries are not likely to adversely affect the following marine mammals or critical habitat that has been designated for them: blue whale, *Balaenoptera musculus*; Pacific right whale, *Eubalaena japonica*, sei whale, *Balaenoptera borealis*, Guadelupe fur seal *Arctocephalus townsendii*, and Steller sea lion *Eumetopias jubatus*. The fisheries are likely to adversely affect humpback whale, *Megaptera novaeangliae*; fin whale *Balaenoptera physalus*, and sperm whale, *Physeter macrocephalus*. NOAA Fisheries has based this conclusion on previous patterns of marine mammals that have been captured, injured, or killed through interactions with the gear used in the fisheries.

Based on previous patterns of interactions between the fisheries and threatened and endangered Pacific salmonids, the biological opinion concludes that the proposed fisheries are not likely to adversely affect any Pacific salmonid or critical habitat that has been designated for them. The harvest of salmon in commercial and recreational ocean fisheries is managed under the Pacific Salmon FMP, which has been analyzed in prior Section 7 consultations. Impacts to these species have already been assessed in previous consultations and take of listed salmonids in ocean fisheries is already covered under separate incidental take statements.

Based on previous patterns of interactions between the fisheries and threatened and endangered sea turtles, the biological opinion concludes that the proposed fisheries are likely to adversely affect green turtles (*Chelonia mydas*), leatherback turtles (*Dermochelys coriacea*), loggerhead turtles (*Caretta caretta*), and olive ridley turtles (*Lepidochelys olivacea*). Hawksbillsea turtles have not been reported interacting with any of the HMS fisheries assessed in this Opinion are not expected to be adversely affected by the proposed actions. NOAA Fisheries has based this conclusion on previous patterns of turtles that have been captured, injured, or killed through interactions with the gear used in the fisheries.

Only limited, quantitative information on the marine mammal and turtle species was available for NOAA Fisheries' analyses. To conduct its jeopardy analyses for sea turtles, NOAA Fisheries used conceptual models, life stage matrix analysis, and limited application of the Dennis model (Dennis *et al.,* 1991) to those nesting aggregations of sea turtles where sufficient census information was available. These assessments considered the information available on the population dynamics of the sea turtle species and the numbers of sea turtles captured, injured, or killed in the U.S. Pacific HMS fisheries to determine if these injuries or deaths could be expected to reduce a species' reproduction, numbers, or distribution in a way that would be expected to appreciably reduce the species' likelihood of surviving and recovering in the wild. As part of these analyses, NOAA Fisheries made assumptions about the number of adult, female sea turtles that might be captured, injured, or killed in the HMS fisheries. NOAA Fisheries then considered the probable effects on turtle mortalities in the fisheries on the species' population structure, the status and trends of the various populations, the vital rates, and the relationship between vital rates and the population's status and trend (that is, the population's rates of increase). Specifically, NOAA Fisheries considered whether mortalities associated with the fisheries are a significant or chronic source of (a) reduced fecundity in the breeding population of these turtles or (b) decreased rates of survival in one or more life history stage of these sea turtles.

Based on these analyses, NOAA Fisheries concluded that the numbers of loggerhead turtles captured, injured, or killed in the proposed fisheries as they would proceed under the proposed HMS FMP would reduce the numbers and reproduction of this species in a way that would be expected to appreciably reduce its' likelihood of surviving and recovering in the wild. However, the Opinion also assessed the effects of a proposed regulation that would avoid most of the expected captures and mortalities of these species. When viewed in combination, the entire proposed action was not expected to reduce appreciably the likelihood of both the survival and recovery of loggerhead sea turtles. Also based on these analyses, NOAA Fisheries concluded that the numbers of fin whales, humpback whales, sperm whales, and green, leatherback, and olive ridley turtles captured, injured, or killed in the proposed fisheries would not reduce the numbers and reproduction of those species in a way that would be expected to appreciably reduce their likelihood of surviving and recovering in the wild. The Opinion also provides an Incidental Take Statement that includes measures to minimize the impact of residual captures and deaths on all the sea turtles and marine mammals.

Introduction

Section 7(a)(2) of the Endangered Species Act (ESA; 16 U.S.C. § 1531 *et seq.*) requires that each Federal agency shall ensure that any action authorized, funded, or carried out by such agency 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 the action of a Federal agency may affect a protected species or critical habitat, that agency is required to consult with either the National Marine Fisheries Service (NOAA Fisheries) or the U.S. Fish and Wildlife Service, depending upon the protected species or critical habitat that may be affected. For the actions described in this document, the action agency is the Sustainable Fisheries Division of NOAA Fisheries. The consulting agency is the Protected Resources Division, also of NOAA Fisheries.

This document represents NOAA Fisheries' biological Opinion (Opinion) based on our review of: (1) the Fishery Management Plan and Environmental Impact Statement for U.S. West Coast Fisheries for Highly Migratory Species Fisheries (HMS FMP) as proposed by the Pacific Fishery Management Council (Council) for approval and implementation by the Secretary of Commerce; (2) the proposed rule to prohibit shallow longline sets of the type normally targeting swordfish on the high seas in the Pacific Ocean east of 150EW longitude by vessels that are not already prohibited from making such sets; and (3) the west coast pelagic longline fishery on the high seas that is permitted by NOAA Fisheries under the High Seas Fishing Compliance Act, and the effects of these actions on fin whales (*Balaenoptera physalus*), humpback whales (*Megaptera novaeangliae*), sperm whales (*Physeter macrocephalus*), green turtles (*Chelonia mydas*), hawksbill turtles (*Eretmochelys imbricata*), leatherback turtles (*Dermochelys coriacea*), loggerhead turtles (*Caretta caretta*), and olive ridley turtles (*Lepidochelys olivacea*) in accordance with section 7 of the ESA.

This Opinion is based on information provided in the August 2003 HMS FMP (Council 2003) and a September 24, 2003, biological assessment; the proposed ESA rule package, including a January 15, 2004, biological assessment, recovery plans for the whales and sea turtles, the most current marine mammal stock assessment reports, past and current research and population dynamics modeling efforts, observer and logbook data on fishery effort and protected species interactions within the fisheries covered under the FMP, and past biological Opinions for these and other relevant fisheries. A complete administrative record of this consultation is on file at the NOAA Fisheries Southwest Regional Office, Long Beach, California.

I. CONSULTATION HISTORY

A. Past Consultations on Fisheries Included in the HMS FMP

1. ETP Tuna Purse Seine Fishery

On July 6, 1990, NOAA Fisheries completed a section 7 consultation which evaluated the impacts of the ETP large vessel (Class V and VI vessels under IATTC classification) tuna purse seine fishery on sea turtle populations (NOAA Fisheries, 1990). NOAA Fisheries concluded in its biological opinion that U.S. ETP tuna purse seine fishing operations would not likely jeopardize the continued existence of the sea turtles species. Based on past take levels and anticipated fishing levels in the ETP, NOAA Fisheries authorized the annual take of no more than 180 turtles by the U.S. fleet. Of these turtles, no more than 20 comatose animals could be taken each year (10 olive ridleys, 3 greens, 3 loggerheads, 2 hawksbills, and 2 leatherbacks), and no more than 12 mortalities were allowed each year (8 olive ridleys, 1 green, 1 loggerhead, 1 hawksbill, and 1 leatherback).

On December 8, 1999, NOAA Fisheries issued a biological opinion which superceded the previous consultation. This Opinion assessed the effects of an interim final rule to continue authorization of the U.S. tuna purse seine fishery in the ETP under the Marine Mammal Protection Act (MMPA), as revised by the International Dolphin Conservation Program Act, and its effects on listed sea turtle species. In the Opinion, NOAA Fisheries concluded that the proposed action was not likely to jeopardize the continued existence of sea turtle species. NOAA Fisheries authorized U.S tuna purse seiners in the ETP (large vessels only) to take annually: 3 loggerheads (1 mortality every 7 years), 2 leatherbacks (1 mortality every 10 years), 35 greens (2 mortalities annually), 2 hawksbills (1 mortality every 10 years), and 133 olive ridleys (7 mortalities annually). This Opinion and its associated incidental take statement with non-discretionary reasonable and prudent measures is still in effect.

2. <u>CA/OR Drift Gillnet Fishery</u>

On September 30, 1997, NOAA Fisheries completed a formal section 7 consultation on the final regulations to implement the Pacific Offshore Cetacean Take Reduction Plan (PCTRP). At that time, NOAA Fisheries concluded that the implementation of mandatory pingers, extender lengths, and skipper workshops would most likely reduce the incidental take of listed species. The Opinion and Incidental Take Statement exempted take, in the form of annual entanglement and mortality of 18 and 3 loggerheads, respectively, and 30 and 19 leatherbacks, respectively. Since implementation of the PCTRP on October 30, 1997, 1 loggerhead mortality was observed in 1998, resulting in an estimated mortality of six loggerheads (based on 17.5 percent observer coverage). Because this estimated mortality exceeded the authorized annual mortality of 3 loggerheads, NOAA Fisheries reinitiated consultation. In addition, the fishery interacted with three new listed species not previously considered affected by the fishery. One fin whale and one green turtle were observed killed in 1999, and one olive ridley was observed entangled (released alive) during that same year. Therefore, section 7 consultation was reinitiated to assess the possible causes of the new species interactions and to determine if the action was likely to jeopardize any of the affected species.

On October 23, 2000, NOAA Fisheries issued an Opinion on the issuance of a permit under section 101(a)(5)(E) of the MMPA for the incidental take of marine mammal species listed under the ESA during commercial fishing operations. In that consultation, NOAA Fisheries estimated that the fishery was likely to capture 6 green turtles and olive ridley turtles (over the three year period of the proposed permit) and kill 2 of each species; a maximum of 27 leatherback turtles per year, killing 17; and 33 loggerhead turtles, killing 11, during El Niño events. After reviewing the available scientific and commercial data, the current status of Pacific leatherback and loggerhead turtles, the environmental baseline for the action area, the effects of the proposed action and the cumulative effects, the Opinion concluded that the issuance of section 101(a)(5)(E) permits and the associated continued operation of the CA/OR drift gillnet fishery, as regulated under the PCTRP was likely to jeopardize the continued existence of Pacific leatherback and loggerhead turtles. Based on this Opinion, NOAA Fisheries implemented regulations that eliminate drift gillnet effort from August 15 through November 15 north of Point Conception in an area described in the proposed action to reduce the likelihood of interactions with leatherback turtles (66 FR 44549, August 24, 2001). In addition, fishing effort east of the 120EW longitude is prohibited during El Niño events from June 1 through August 31, in order to reduce the likelihood of an interaction with loggerhead turtles (68 FR 69962, December 16, 2003). The Incidental Take Statement exempted the following incidental take of listed species:

Species	**Estimated Entanglement**	**Estimated Mortality**	**Total Expected Observations**
Fin whale	6 in 3 years	6 in 3 years	1
Humpback	6 in 3 years	0	1
Sperm whale	6 in 3 years	4 in 3 years	1
Steller sea lion	5 in 3 years	5 in 3 years	1
Green turtle	6 in 3 years	2 in 3 years	1
Leatherback turtle	9 in 3 years	6 in 3 years	1
Loggerhead turtle	5 per El Niño year	2 per El Niño year	1 per El Niño year
Olive ridley turtle	6 in 3 years	2 in 3 years	1

3. West Coast-based pelagic longline fishery

NOAA Fisheries has not previously consulted on the effects of the West Coast-based longline fishery on listed species. This fishery is permitted under the High Seas Fishing Compliance Act of 1995 (HSFCA). The purpose of the HSFCA is to license U.S. vessels fishing on the high seas and to implement the Agreement to Promote Compliance with International Conservation and Management Measures by Fishing Vessels on the High Seas. In May 2001, the Center for Biological Diversity and Turtle Island Restoration Network filed a complaint with the U.S. District Court for the Northern District of California alleging that NOAA Fisheries unlawfully failed to conduct an ESA section 7 consultation on the West Coast-based pelagic longline fishery when issuing the HSFCA permits for this fishery. On November 27, 2001, the court denied the plaintiffs' motion and ruled in favor of NOAA Fisheries. In response, the plaintiffs appealed the decision to the Ninth Circuit Court of Appeals in December, 2001. On August 21, 2003, the Ninth Circuit Court of Appeals ruled that NOAA

Fisheries is required to engage in consultation under section 7 of the ESA regarding the effects on sea turtles and other listed species when issuing fishing permits under the HSFCA to West Coast-based pelagic longline fishing vessels. Inasmuch as the FMP will regulate this fishery, the effects of issuance of HSFCA permits to authorize fishing by West Coast-based pelagic longline vessels will be considered in this biological opinion through an analysis of the effects on listed species from the fishing that will occur pursuant to the permits and the proposed FMP.

B. The Current Consultation on the Proposed Actions

On September 24, 2003, the NOAA Fisheries-Southwest Region's Sustainable Fisheries Division (Sustainable Fisheries Division) requested initiation of consultation on the proposed HMS FMP. On January 15, 2004, NOAA Fisheries-Southwest Region's Protected Resources Division (Protected Resources Division) requested initiation of consultation on a proposed rule to prohibit shallow longline sets of the type normally targeting swordfish on the high seas in the Pacific Ocean east of 150EW longitude by vessels that are not already prohibited from making such sets. The analysis of the effects of these two actions, as well as the continued operation of vessels in the FMP fisheries under their HSFCA permits and under the proposed FMP, have been combined into one biological opinion due to the overlap in areas and type of effect, species affected, and the interrelated and interdependent nature of the FMP and ESA rule actions.

Throughout the consultation period, Protected Resources Division and Sustainable Fisheries Division staff met to discuss and clarify the information available for the consultation. In addition, staff from NOAA Fisheries' Office of Protected Resources and Protected Resources Division staff met to identify common approaches to the assessments for the section 7 consultations on the proposed management regimes for the Western Pacific Pelagics and Pacific Highly Migratory Species fisheries management plans. Due to the similar nature of the actions and the overlap in the affected environment and listed species between the FMPs, a consistent analytical approach was required to ensure consistency, the use of the best available information, and a coordinated approach to analyzing the impacts of the fisheries on listed species, such as sea turtles. Office of Protected Resources and Protected Resources Division staff also met with staff from the Southwest Fisheries Science Center to discuss the overall approach and gather additional information on the biogeography and genetics of loggerhead and leatherback sea turtles in the Pacific Ocean.

As a result, two January 16, 2004, memos from the Protected Resources Division to the Sustainable Fisheries Division presented information on the ongoing and expected steps in the consultation and effects analysis. On January 21, 2004, representatives of the Office of Protected Resources and the Southwest Region's Protected Resources Division met with representatives of the Pacific Island Region's and Southwest Region's Sustainable Fisheries Divisions, NOAA Fisheries' Pacific Islands Fisheries Science Center, the Hawaii Longline Association, and the Western Pacific Regional Fisheries Management Council. At that meeting, representatives of the Office of Protected Resources provided a detailed description of the approach, as described in the January 16, 2004, memos, that the Office of

Protected Resources and the Southwest Region Protected Resources Division were using to assess the effects of the pelagic fisheries of the western Pacific region and the highly migratory species fisheries of the U.S. west coast on threatened and endangered species.

A draft of this biological opinion was circulated for review by the action agency, the Sustainable Fisheries Division, on February 3, 2004.

II. DESCRIPTION OF THE ACTIONS

II-A. Sustainable Fisheries Division Action - Adoption of the Proposed HMS FMP and ongoing operation of vessels with High Seas Fishing Compliance Permits

NOAA Fisheries' Sustainable Fisheries Division, Southwest Region, requested ESA Section 7 consultation on the fisheries to be managed under the HMS FMP as proposed by the Council for approval and implementation by the Secretary of Commerce. Therefore, the management regime, as described in the proposed FMP and proposed for adoption by the Secretary of Commerce, constitutes the main action being considered in this Opinion. Among the fisheries being considered are those that operate (even occasionally) on the high seas and that consist of vessels with HSFCA permits issued by NOAA Fisheries. These include the West Coast-based longline fishery that operates exclusively on the high seas, the albacore troll fishery, and the ETP purse seine fishery, all of which fish on the high seas as well as occasionally within the EEZ off the U.S. west coast.

The purpose of fishery management plans, including the HMS FMP, has been established by the Magnuson-Stevens Fishery Conservation and Management Act (MSA; 16 U.S.C. 1801 *et seq*.). The stated purpose of the HMS FMP is to maximize the net benefits of the fisheries to the nation and to the eastern Pacific region. Background information on Federal fisheries policy and management under the MSA, fishery management plan development process, and HMS FMP is described in the August 2003 HMS FMP (See Chapter 1). The FMP is a “framework” FMP which includes some fixed elements and a process for implementing or changing regulations without amending the FMP. Changes to any of the fixed elements in the FMP require an FMP amendment. The framework procedures are described in Chapter 8 of the FMP.

The HMS FMP may also provide an implementing mechanism for the U.N. Agreement to Promote Compliance with International Conservation and Management Measures by Fishing Vessels on the High Seas (Agreement), which was adopted by the U.N. Food and Agriculture Organization (FAO) in November 1993. The Agreement established the responsibility of each nation for the actions of vessels fishing under that nation’s flag on the high seas. The Agreement requires that vessels have specific authorization from their flag nation to participate in high seas fishing. Furthermore, nations must maintain a registry of authorized vessels, ensure that those vessels are marked for identification, and ensure that they report sufficient information on their fishing activities. As mentioned briefly in the previous section, the HSFCA is the domestic legislation enacted in 1995 to provide authority to the Secretary of Commerce to implement the FAO Agreement.

NOAA Fisheries has implemented regulations requiring U.S. vessel operators fishing on the high seas to maintain and submit records of catch and effort on their high seas fishing activities. The reporting requirement would be met if a vessel operator is reporting in compliance with regulations under another Federal statute (e.g., MSA requirements). Thus, longline vessel operators fishing outside of the EEZ but based on the West Coast must maintain and file an HSFCA logbook, and West Coast albacore trollers must maintain and file a troll HSFCA logbook. NOAA Fisheries provides the required forms or logbooks. Fishermen are not required to report catch and effort within the EEZ under this requirement. The HMS FMP would supercede the HSFCA reporting requirements and thus provide a mechanism to harmonize eastern and western Pacific fishery reporting and monitoring mechanisms.

The proposed management regime under the HMS FMP would regulate commercial and charter recreational fisheries for tuna, billfish, dorado (mahi mahi) and selected sharks. The gear types under the FMP are longline, drift gillnet (mesh >14" stretched mesh), purse seine, troll/jigboat, harpoon, and charter/commercial passenger carrying vessels. Existing State conservation and management programs would for the most part be maintained; this is most important for nearshore time/area closures affecting certain gear types. Initial conservation and management measures under the HMS FMP would be limited to new controls on pelagic longline fishing and maintaining (but under MSA authority) existing measures to limit drift gillnet fishing. All fisheries would be subject to Federal permit and reporting requirements and observer coverage.

The HMS FMP includes initial estimates of maximum sustainable yields (MSY) for fished stocks and sets overfishing control rules under MSA for these fisheries in the EEZ.

A. Final Rules to Implement the HMS FMP

The measures that would be implemented under the HMS FMP are:

1. Owners and operators of vessels registered for use of longline gear may not use longline gear to fish for or target swordfish *(Xiphias gladius*) west of 150E W. long. and north of the equator (0E N. lat.).

2. A person aboard a vessel registered for use of longline gear fishing for HMS west of 150E W. long. and north of the equator (0E N. lat.) may not possess or deploy any float line that is shorter than or equal to 20 m (65.6 ft or 10.9 fm). As used here, float line means a line used to suspend the main longline beneath a float.

3. From April 1 through May 31, owners and operators of vessels registered for use of longline gear may not use longline gear in waters bounded on the south by 0E lat., on the north

by 15E N. lat., on the east by 145E W. long., and on the west by 180E long.

4. From April 1 through May 31, owners and operators of vessels registered for use of longline gear may not receive from another vessel HMS that were harvested by longline gear in waters bounded on the south by 0E lat., on the north by 15E N. lat., on the east by 145E W. long., and on the west by 180E long.

5. From April 1 through May 31, owners and operators of vessels registered for use of longline gear may not land or transship HMS that were harvested by longline gear in waters bounded on the south by 0E lat., on the north by 15E N. lat., on the east by 145E W. long., and on the west by 180E long.

6. No light stick may be possessed on board a vessel registered for use of longline gear during fishing trips that include any fishing west of 150E W. long. and north of the equator (0E N. lat.). A light stick as used in this paragraph is any type of light emitting device, including any florescent glow bead, chemical, or electrically powered light that is affixed underwater to the longline gear.

7. When a conventional monofilament longline is deployed in waters west of 150E W. long. and north of the equator (0E N. lat.) by a vessel registered for use of longline gear, no fewer than 15 branch lines may be set between any two floats. Vessel operators using basket-style longline gear must set a minimum of 10 branch lines between any 2 floats when fishing in waters north of the equator.

8. Longline gear deployed west of 150E W. long. and north of the equator (0E N. lat.) by a vessel registered for use of longline gear must be deployed such that the deepest point of the main longline between any two floats, i.e., the deepest point in each sag of the main line, is at a depth greater than 100 m (328.1 ft or 54.6 fm) below the sea surface.

9. Owners and operators of longline vessels registered for use of longline gear may land or possess no more than 10 swordfish from a fishing trip where any part of the trip included fishing west of 150E W. long. and north of the equator (0E N. lat.).

10. Fishing vessels that use longline gear to catch managed species beyond the EEZ and east of 150° W. longitude are not prohibited from making shallow water sets of the type used to target swordfish and are not subject to the limitations of items 2, 6, 7, 8, and 9 above.

The proposed regulations also include measures intended to protect against or mitigate interactions with sea turtles and seabirds to the extent they are encountered, as follows:

Sea turtle take mitigation measures.

1. Owners and operators of vessels registered for use of longline gear must carry aboard their vessels line clippers meeting the minimum design standards specified in (2) of this section, dip nets meeting minimum standards specified in (3) of this section, and wire or bolt cutters capable of cutting through the vessel's hooks. These items must be used to disengage any hooked or entangled sea turtles with the least harm possible to the sea turtles and as close to the hook as possible in accordance with the requirements specified in (4) through (6) of this section.

2. Line clippers are intended to cut fishing line as close as possible to hooked or entangled sea turtles. NOAA Fisheries has established minimum design standards for line clippers. The Arceneaux line clipper (ALC) is a model line clipper that meets these minimum design standards and may be fabricated from readily available and low-cost materials. The minimum standards are as follows:

> (a) The cutting blade must be curved, recessed, contained in a holder, or otherwise afforded some protection to minimize direct contact of the cutting surface with sea turtles or users of the cutting blade.
> (b) The blade must be capable of cutting 2.0-2.1 mm monofilament line and nylon or polypropylene multi-strand material commonly known as braided mainline or tarred mainline.
> (c) The line clipper must have an extended reach handle or pole of at least 6 ft (1.82 m).
> (d) The cutting blade must be securely fastened to the extended reach handle or pole to ensure effective deployment and use.

3. Dip nets are intended to facilitate safe handling of sea turtles and access to sea turtles for purposes of cutting lines in a manner that minimizes injury and trauma to sea turtles. The minimum design standards for dip nets that meet the requirements of this section are:

> (a) The dip net must have an extended reach handle of at least 6 ft (1.82 m) of wood or other rigid material able to support a minimum of 100 lbs (34.1 kg) without breaking or significant bending or distortion.
> (b) The dip net must have a net hoop of at least 31 inches (78.74 cm) inside diameter and a bag depth of at least 38 inches (96.52 cm). The bag mesh openings may be no more than 3 inches x 3 inches (7.62 cm 7.62 cm).

4. All incidentally taken sea turtles brought aboard for dehooking and/or disentanglement must be handled in a manner to minimize injury and promote post-hooking survival.

> (a) When practicable, comatose sea turtles must be brought on board immediately, with a minimum of injury, and handled in accordance with the procedures specified in paragraphs (5) and (6) of this section.

(b) If a sea turtle is too large or hooked in such a manner as to preclude safe boarding without causing further damage/injury to the turtle, line clippers described in paragraph (2) of this section must be used to clip the line and remove as much line as possible prior to releasing the turtle.
(c) If a sea turtle is observed to be hooked or entangled by longline gear during hauling operations, the vessel operator must immediately cease hauling operations until the turtle has been removed from the longline gear or brought on board the vessel.
(d) Hooks must be removed from sea turtles as quickly and carefully as possible. If a hook cannot be removed from a turtle, the line must be cut as close to the hook as possible.

5. If the sea turtle brought aboard appears dead or comatose, the sea turtle must be placed on its belly (on the bottom shell or plastron) so that the turtle is right side up and its hindquarters elevated at least 6 inches (15.24 cm) for a period of no less than 4 hours and no more than 24 hours. The amount of the elevation depends on the size of the turtle; greater elevations are needed for larger turtles. A reflex test, performed by gently touching the eye and pinching the tail of a sea turtle, must be administered by a vessel operator, at least every 3 hours, to determine if the sea turtle is responsive. Sea turtles being resuscitated must be shaded and kept damp or moist but under no circumstance may be placed into a container holding water. A water-soaked towel placed over the eyes, carapace, and flippers is the most effective method to keep a turtle moist. Those that revive and become active must be returned to the sea in the manner described in paragraph (6) of this section. Sea turtles that fail to revive within the 24-hour period must also be returned to the sea in the manner described in paragraph (6)(a) and (b) of this section.

6. Live turtles must be returned to the sea after handling in accordance with the requirements of paragraphs (4) and (5) of this section:

(a) By putting the vessel engine in neutral gear so that the propeller is disengaged and the vessel is stopped, and releasing the turtle away from deployed gear; and
(b) Observing that the turtle is safely away from the vessel before engaging the propeller and continuing operations.

7. In addition to the above sea turtle mitigation requirements, a vessel operator shall perform sea turtle handling and resuscitation techniques consistent with Title 50, Code of Federal Regulations, Section 223.206 (d)(1), as appropriate.

Seabird mitigation measures

1. Seabird mitigation techniques. Owners and operators of vessels registered for use of longline gear must ensure that the following actions are taken when fishing north of 23E N. lat.:

    (a) Employ a line setting machine or line shooter to set the main longline when making deep sets west of 150E W. long. using monofilament main longline;
    (b) Attach a weight of at least 45 g to each branch line within 1 m of the hook when making deep sets using monofilament main longline;
    (c) When using basket-style longline gear, ensure that the main longline is deployed slack to maximize its sink rate;

2. Use completely thawed bait that has been dyed blue to an intensity level specified by a color quality control card issued by NOAA Fisheries;

3. Maintain a minimum of two cans (each sold as 0.45 kg or 1 lb size) containing blue dye on board the vessel;

4. Discharge fish, fish parts (offal), or spent bait while setting or hauling longline gear, on the opposite side of the vessel from where the longline gear is being set or hauled;

5. Retain sufficient quantities of fish, fish parts, or spent bait, between the setting of longline gear for the purpose of strategically discharging it in accordance with paragraph (a)(6) of this section;

6. Remove all hooks from fish, fish parts, or spent bait prior to its discharge in accordance with paragraph 4 of this section; and

7. Remove the bill and liver of any swordfish that is caught, sever its head from the trunk and cut it in half vertically, and periodically discharge the butchered heads and livers in accordance with paragraph 6 of this section.

8. If a short-tailed albatross is hooked or entangled by a vessel registered for use of longline gear, owners and operators must ensure that the following actions are taken:
    (a) Stop the vessel to reduce the tension on the line and bring the bird on board the vessel using a dip net;
    (b) Cover the bird with a towel to protect its feathers from oils or damage while being handled;
    (c) Remove any entangled lines from the bird;
    (d) Determine if the bird is alive or dead.
        (i) If dead, freeze the bird immediately with an identification tag attached

directly to the specimen listing the species, location and date of mortality, and band number if the bird has a leg band. Attach a duplicate identification tag to the bag or container holding the bird. Any leg bands present must remain on the bird. Contact NOAA Fisheries, the Coast Guard, or the U.S. Fish and Wildlife Service at the numbers listed on the Short-tailed Albatross Handling Placard distributed at the NOAA Fisheries protected species workshop, inform them that you have a dead short-tailed albatross on board, and submit the bird to NOAA Fisheries within 72 hours following completion of the fishing trip.
(ii) If alive, handle the bird in accordance with paragraphs 9 through 14 of this section.

9. Place the bird in a safe enclosed place;

10. Immediately contact NOAA Fisheries, the Coast Guard, or the U.S. Fish and Wildlife Service at the numbers listed on the Short-tailed Albatross Handling Placard distributed at the NOAA Fisheries protected species workshop and request veterinary guidance;

11. Follow the veterinary guidance regarding the handling and release of the bird.

12 . Complete the short-tailed albatross recovery data form issued by NOAA Fisheries.

13. If the bird is externally hooked and no veterinary guidance is received within 24-48 hours, handle the bird in accordance with paragraphs 17(d) and (e) of this section, and release the bird only if it meets the following criteria:

(a) Able to hold its head erect and respond to noise and motion stimuli;
(b) Able to breathe without noise;
(c) Capable of flapping and retracting both wings to normal folded position on its back;
(d) Able to stand on both feet with toes pointed forward; and
(e) Feathers are dry.

14. If released under paragraph 13 of this section or under the guidance of a veterinarian, all released birds must be placed on the sea surface.

15. If the hook has been ingested or is inaccessible, keep the bird in a safe, enclosed place and submit it to NOAA Fisheries immediately upon the vessel's return to port. Do not give the bird food or water.

16. Complete the short-tailed albatross recovery data form issued by NOAA Fisheries.

17. If a seabird other than a short-tailed albatross is hooked or entangled by a vessel

registered for use of longline gear, owners and operators must ensure that the following actions are taken:

(a) Stop the vessel to reduce the tension on the line and bring the seabird on board the vessel using a dip net;
(b) Cover the seabird with a towel to protect its feathers from oils or damage while being handled;
(c) Remove any entangled lines from the seabird;
(d) Remove any external hooks by cutting the line as close as possible to the hook, pushing the hook barb out point first, cutting off the hook barb using bolt cutters, and then removing the hook shank;
(e) Cut the fishing line as close as possible to ingested or inaccessible hooks;
(f) Leave the bird in a safe enclosed space to recover until its feathers are dry; and
(g) After recovered, release seabirds by placing them on the sea surface.

To ensure full understanding of the sea turtle and sea bird protection measures and enhance their effectiveness, each year both the owner and the operator of a vessel registered for use of longline gear must attend and be certified for completion of a workshop conducted by NOAA Fisheries on mitigation, handling, and release techniques for turtles and seabirds and other protected species. A protected species workshop certificate will be issued by NOAA Fisheries annually to any person who has completed the workshop. An owner of a vessel registered for use of longline gear must have on file a valid protected species workshop certificate or copy issued by NOAA Fisheries in order to maintain or renew their vessel registration. An operator of a vessel registered for use of longline gear must have on board the vessel a valid protected species workshop certificate issued by NOAA Fisheries or a legible copy thereof.

The proposed rule also requires that longline vessels be equipped with vessel monitoring system units, as in the western Pacific.

Drift Gillnet Fishery Controls

The proposed regulations would not affect the gear restrictions resulting from the Pacific Offshore Cetacean Take Reduction Plan established under the authority of the Marine Mammal Protection Act of 1972. These measures can be found at 50 CFR 229.31.

The proposed regulations would maintain, but under MSA authority, conservation and management measures now in place under the authority of the Endangered Species Act and the State of California Fish and Game Code as follows:

1. The maximum length of a drift gillnet on board a vessel shall not exceed 6,000 feet.
2. Up to 1,500 feet of drift gillnet in separate panels of 600 feet may be on board the vessel in

a storage area.

Protected Resource Area Closures:

1. No person may fish with, set, or haul back drift gillnet gear in U.S. waters of the Pacific Ocean from August 15 through November 15 in the area bounded by straight lines connecting the following coordinates in the order listed:

(a) Pt. Sur at 36E 18.5' N. lat., to
(b) 34E 27' N. lat. 123E 35' W. long.;
(c) 34E 27' N. lat. 129E W. long.;
(d) 45E N. lat. 129E W. long., thence
(e) to the point where 45E N. lat. intersects the Oregon coast.

2. No person may fish with, set, or haul back drift gillnet gear in U.S. waters of the Pacific Ocean east of 120E W. long. during the months of June, July, and August, during a forecasted or occurring El Niño event off Southern California. The Assistant Administrator will publish a notification in the Federal Register that an El Niño event is occurring off, or is forecast for off, the coast of southern California and the requirement for time area closures in the Pacific loggerhead conservation zone. The notification will also be announced in summary form by other methods as the Assistant Administrator determines necessary and appropriate to provide notice to the California/Oregon drift gillnet fishery. The Assistant Administrator will rely on information developed by NOAA offices that monitor El Niño events, such as NOAA's Coast Watch program, and developed by the State of California, to determine if such a notice should be published. The requirement for the area closures from June 1 through August 31 will remain effective until the Assistant Administrator issues a notice that the El Niño event is no longer occurring.

Mainland area closures:

The following areas off the Pacific coast are closed to driftnet gear:

1. Within the U.S. EEZ from the United States-Mexico International Boundary to the California-Oregon border from February 1 through April 30.

2. In the portion of the U.S. EEZ within 75 nautical miles (nm) from the mainland shore from the United States-Mexico International Boundary to the California-Oregon border from May 1 through August 14.

3. In the portion of the U.S. EEZ within 25 miles of the coastline from December 15 through January 31 of the following year from the United States-Mexico International Boundary to the California-Oregon border.

4. In the portion of the U.S. EEZ from August 15 through September 30 within the area bounded by line extending from Dana Point to Church Rock on Santa Catalina Island, to Point La Jolla.

5. In the portion of the U.S. EEZ within 12 nautical miles from the mainland shore north of a line extending west of Point Arguello to the California-Oregon border.

6. In the portion of the U.S. EEZ within the area bounded by a line from the lighthouse at Point Reyes, California to Noonday Rock, to Southeast Farallon Island to Pillar Point.

7. In the portion of the U.S. EEZ off the Oregon coast east of a line approximating 1000 fathoms as defined by the following coordinates:

42E 00' 00" N. lat.	125E 10' 30" W. long.
42E 25' 39" N. lat.	124E 59' 09" W. long.
42E 30' 42" N. lat.	125E 00' 46" W. long.
42E 30' 23" N. lat.	125E 04' 14" W. long.
43E 02' 56" N. lat.	125E 06' 57" W. long.
43E 01' 29" N. lat.	125E 10' 55" W. long.
43E 50' 11" N. lat.	125E 19' 14" W. long.
44E 03' 23" N. lat.	125E 12' 22" W. long.
45E 00' 06" N. lat.	125E 16' 42" W. long.
45E 25' 27" N. lat.	125E 16' 29" W. long.
45E 45' 37" N. lat.	125E 15' 19" W. long.
46E 04' 45" N. lat.	125E 24' 41" W. long.
46E 16' 00" N. lat.	125E 20' 32" W. long.

8. In the portion of the U.S. EEZ north of 46E 16' N. latitude (Washington coast).

Channel Islands area closures:

The following areas off the Channel Islands are closed to driftnet gear:

1. San Miguel Island closures.
   (a) Within the portion of the U.S. EEZ north of San Miguel Island between a line extending 6 nm west of Point Bennett and a line extending 6 nm east of Cardwell Point.
   (b) Within the portion of the U.S. EEZ south of San Miguel Island between a line extending 10 nm west of Point Bennett and a line extending 10 nm east of Cardwell Point.

2. Santa Rosa Island Closure. Within the portion of the U.S. EEZ north of San Miguel Island between a line extending 6 nm west from Sandy Point and a line extending 6 nm east of Skunk

Point from May 1 through July 31.

3. San Nicolas Island closure. In the portion of the U.S. EEZ within a radius of 10 nm of 33E 16' 41" N. lat., 119E 34' 39" W. long. (west end) from May 1 through July 31.

4. San Clemente Island closure. In the portion of the U.S. EEZ within 6 nm of the coastline on the easterly side of San Clemente Island within a line extending 6 nm west from 33E 02' 16" N. lat., 118E 35' 27" W. long. and a line extending 6 nm east from the light at Pyramid Head

Regulations in place under the MMPA would be unchanged. The TRT process would continue to be the principal mechanism for considering regulatory changes to meet MMPA requirements.

No new conservation and management measures are proposed for purse seine, harpoon, surface hook-and-line, or charter and recreational fisheries except the permit and logbook requirements.

However, large vessels (>400 st) in the U.S. purse seine fleet must also abide by the following conditions required under the Incidental Take Statement for the December 8, 1999 biological opinion on the interim final rule to continue authorization of the U.S. tuna purse seine fishery in the ETP under the Marine Mammal Protection Act (MMPA), as revised by the International Dolphin Conservation Program Act (regulations at 50 CFR 300.29(e) Bycatch reduction measures. (66 FR 49320):

(3) All purse seine vessels must apply special sea turtle handling and release procedures, as follows:

(i) Whenever a sea turtle is sighted in the net, a speedboat shall be stationed close to the point where the net is lifted out of the water to assist in the release of the turtle;

(ii) If a turtle is entangled in the net, net roll shall stop as soon as the turtle comes out of the water and shall not resume until the turtle has been disentangled and released;

(iii) If, in spite of the measures taken under paragraphs (e)(3)(i) and (ii) of this section, a turtle is accidentally brought onboard the vessel alive and active, the vessel's engine shall be disengaged and the turtle shall be released as quickly as practicable;

(iv) If a turtle brought on board under paragraph (e)(3)(iii) of this section is alive but comatose or inactive, the resuscitation procedures described in 223.206(d)(1)(i)(B) of this title shall be used before release of the turtle.

B. HMS Fisheries by West Coast Vessels

This section provides a general descriptive overview of the physical, economic and social environment for HMS fisheries off the West Coast.

The HMS fisheries consist of the fish stocks and participants involved in their commercial harvest, commercial use, recreational harvest, and recreational use. The principal HMS harvested by vessels based on the West Coast and fishing in the EEZ or beyond the EEZ include: north Pacific albacore (*Thunnus alalunga*), yellowfin tuna (*T. albacares*), bigeye tuna (*T. obesus*), skipjack tuna (*Katsuwonus pelamis*), northern bluefin tuna (*T. orientalis*), swordfish (*Xiphias gladius*), common thresher shark (*Alopias vulpinus*), pelagic thresher shark (*A. pelagicus*), bigeye thresher shark *(A. superciliosus*), shortfin mako shark (*Isurus oxyrinchus*), blue shark (*Prionace glauca*), striped marlin (*Tetrapturus audax*) and dorado (*Coryphaena hippurus*).

HMS are taken directly in fisheries that use many types of gears and vessels. Gears used to harvest HMS by directed commercial fisheries are primarily: surface hook-and-line, drift gillnet, harpoon, purse seine, and pelagic longline.

The recreational fishery for HMS targets albacore, yellowfin, skipjack, bigeye and northern bluefin tunas, striped marlin, swordfish, dorado, and mako, blue and thresher sharks using hook-and-line gear. The fisheries are composed of both private angler vessels and charter vessels (also known as "head boats" and commercial passenger fishing vessels (CPFV)).

Most HMS and the fisheries they support are distributed internationally with components in the EEZs of Canada and Mexico as well as in international waters outside of any country's EEZ. U.S. vessels' participation may reflect not only changes in domestic fishery conditions, but also changes in conditions, including the status of stocks, resulting from international fishing. Also, landings may be affected as much by market conditions as by stock conditions. These factors give rise to considerable variability in annual U.S. landings of HMS and corresponding exvessel revenues.

Over the 1981-99 period, the most important HMS in terms of landings by all gear types were yellowfin, skipjack, and albacore tunas, swordfish, and common thresher shark. In recent years, the most important HMS have been albacore tuna, swordfish, and common thresher shark. By the end of the 1990s landings of yellowfin and skipjack tuna were substantially less than the amounts landed in the early 1980s. Bluefin tuna landings during the period were characterized by a high degree of variability. Through the 1980s and into the early 1990s albacore landings fell sharply, but by the late 1990s they had returned to relatively high levels of the late 1970s. Swordfish landings declined during the 1980s, but were on the rise through most of the 1990s. Common thresher shark landings followed a pattern similar to that for swordfish over the period. Landings of shortfin mako shark exhibited a fairly sharp decline over the 1981-99 period. Landings of pelagic thresher, bigeye thresher and blue sharks as well as dorado were relatively minor during the 1981-99 period. The bulk of the HMS fisheries occur off

California. These fisheries make up a small portion of the HMS fisheries of the Pacific Ocean, accounting for less than 5 percent of total catches.

1. Characteristics of the Domestic Fisheries

This section describes the characteristics of the domestic fisheries: (a) the albacore fishery using surface hook-and-line gear by trolling and bait-fishing techniques; (b) the longline fishery based in California fishing for swordfish, tuna, and sharks beyond the EEZ; (c) the swordfish and shark drift gillnet fishery (>14 inch stretched mesh nets); (d) the tropical tuna fisheries using purse seine, including the coastal purse seine fishery (small vessels) that concentrates on small pelagic species, especially northern anchovy and Pacific sardine, but which also harvests northern bluefin and yellowfin tuna when they migrate into the Pacific EEZ; (e) the swordfish and shark harpoon fishery; and, (f) the charter and private boat HMS sport fisheries.

a. Albacore Surface Hook-and-Line Fishery

The west coast-based U.S. albacore fishery is comprised of vessels that predominantly troll for albacore using jigs and, to a lesser extent, live bait. Together, these gears and other hook and line gears used to target albacore are known as surface hook-and-line gear and account for the bulk of West Coast albacore landings and exvessel revenues. U.S. troll vessels have fished for albacore in the north Pacific since the early 1900s.

The basic troll vessel gear consists of between 8 and 12 (a few vessels use more) lines towed up to 30 meters (m) behind the vessel. Lateral spacing of the lines is accomplished by using outriggers or long poles extended to each side of the vessel with fairleads spreading 3 or more lines to each side, with the remainder attached to the stern. Terminal gear is generally chrome-headed jigs with varying colored plastic fringed skirts and a double barbless undulated hook. The gear is relatively inexpensive. Retrieval is done by hand or by powered gurdies, similar to salmon troll vessels. Fishing effort continues throughout the day, with lines retrieved when target species or bycatch are hooked.

Albacore may be discarded because they are undersized. Albacore troll vessels catch minor amounts of other fish species, usually while in transit to or from the fishing grounds. The primary species caught incidentally include skipjack tuna, bluefin tuna, yellowfin tuna, dorado, billfish, and sharks.

A few troll vessels carry small amounts of live bait, which is chummed under some conditions to aggregate albacore and improve catches. Very few vessels operate with bait only. Transshipment at sea is used by some vessels to extend the effective length of a fishing trip which might otherwise be limited due to carrying capacity. Catches are landed at ports along the U.S. West Coast, in Hawaii, or at canneries in American Samoa or Tahiti. Transshipped fish is generally landed in American Samoa. The bulk of the U.S. catch is canned as white meat tuna at canneries in American Samoa and Puerto Rico. A small amount of the catch finds its way into the fresh fish trade, which is a significant income to these participants.

In recent years, the north Pacific albacore troll season has begun as early as mid-April in areas northwest of Midway Island. In July and August, the fleet moves eastward, fishing near 45E N latitude, 150E W longitude and along the West Coast of North America from Vancouver Island to southern California. Fishing can continue into November if weather permits and sufficient amounts of albacore remain available to troll gear.

The troll fleet is composed of a varying number of vessels ranging from 16 ft to over 100 ft in length. The vast majority of vessels are 25 ft or greater. Both big and small vessels fish inside the EEZ. The smaller vessels more likely make short trips (1-3 days) and fish 25-100 miles from shore; the larger vessels may fish on trips across the north Pacific and well beyond the EEZ for 90 days or more. The total estimated number of vessels landing albacore peaked at more than 2,000 in the mid-1970s. Fewer vessels have been active in recent years with 741 reporting landings in 1996; 1,244 in 1997; 913 in 1998; and 775 in 1999. The number of larger vessels (greater than 50 ft) is relatively steady, ranging from 285 to 372 in the 1996 to 1998 period.

The north Pacific albacore stock has rebounded from low levels in the 1980s, but it is not known how long the increased availability of albacore to the West Coast fleet will continue. Albacore (like most tunas) have variable recruitment dependent in part on environmental conditions, and their migratory patterns may bring them closer to shore in some years than in others. Further, the industry’s occasional difficulty marketing its catch when canneries have large supplies and thus offer low prices, is buffered by more of the catch entering the fresh and frozen product market. Overall, it does not appear likely that the West Coast albacore fishery will change substantially in the next few years. The fishery is expected to remain fairly stable with a fleet of 800-1,000 vessels making total landings averaging about 10,000 metric tons (mt), and valued at about $20 million per year.

The top five albacore ports in California based on average annual landings during the 1981-99 period were Terminal Island, Moss Landing, San Francisco Bay area, Eureka and San Diego. Through the U.S.-Canadian albacore treaty, U.S. vessels can fish in Canadian waters and land in certain Canadian ports. A reciprocal arrangement holds for Canadian vessels. Thus, in any given year, U.S. troll vessels may fish a portion of the year in the U.S. EEZ, a portion on the high seas, and a portion in Canada’s EEZ. For example, in 1997, effort by U.S. trollers took place 29 percent in the U.S. EEZ, 4 percent in Canada’s EEZ and 67 percent on the high seas (from Table 2-16 in the HMS FMP).

b. West Coast-based Longline Fishery

The High Seas Fishing Compliance Act, passed to implement the Agreement to Promote Compliance with International Conservation and Management Measures by Fishing Vessels on the High Seas and adopted by the United Nations in 1993, requires logbooks and permits for U.S. vessels fishing beyond the EEZ but no management measures are currently proposed pursuant to that statute.

The longline fishery based on the West Coast fishes outside the EEZ and targets mainly swordfish. Under California law, longline fishing in the EEZ off California is prohibited. However, California-registered vessels are allowed to land longline caught fish in California ports as long as fishing takes place outside of the EEZ. Oregon does allow the harvest of swordfish and blue shark within the EEZ under a developmental fishery permit; however, no landings have occurred under the permits. Up to 10 permits are allowed for blue shark and 20 for swordfish. Since 1995, the number of blue shark permits issued in a year has ranged from none to six and the number of swordfish permits issued has ranged from one to nine. Permit stipulations restrict the harvest from within 25 miles of the shore. Washington prohibits longline fishing by State vessels in the EEZ.

In 1991, there were three longline vessels that fished beyond the EEZ targeting swordfish and bigeye tuna and unloaded their catch and re-provisioned in California ports. In 1993, a Gulf Coast fish processor set up an infrastructure at Ventura Harbor, California to provide longline vessels with ice, gear, bait, and fuel, and fish offloading and transportation services. Consequently, longline vessels seeking an alternative to the Gulf of Mexico longline fishery, and precluded from entering the Hawaii fishery, began arriving in Southern California. By 1994, 31 vessels comprised this California based fishery, fishing beyond the EEZ, and landing swordfish and tunas into California ports. These vessels fished side-by-side with Hawaiian vessels in the area around 135E W longitude in the months from September through May.

In 1995, only six longline vessels made a high seas trip from a California port, although 22 vessels made at least one longline landing. The group of vessels that came to California from the Gulf of Mexico in 1993 and 1994 left the California-based fishery and either returned to the Gulf of Mexico fishery, or acquired Hawaiian longline permits in order to have fishery options for the months from June to September, when fishing within range of California ports drops off substantially. Many of the vessels that had participated in the California fishery had discovered productive swordfish fishing grounds in the fall and winter that were farther east than the Hawaiian fleet usually operated. Therefore, these vessels continued to move east later in the year, and operated out of California ports only when it became closer to the point of the end of the fishing trip than the distance to return to Hawaii. These vessels fished from California until about January, when the pattern of fishing moved to the west, and operating from Hawaii again became more convenient. Consequently, beginning in the latter part of 1995, a number of vessels from the Hawaiian fleet began a pattern of fishing operations that moved to California in the fall and winter and then back to Hawaii in the spring and summer. This pattern continued until 2001, when the swordfish targeting prohibition and other restrictions were implemented for Hawaii vessels. Because of the prohibition, about 20 vessels removed themselves from their western Pacific longline limited entry permit and shifted to California.

The number of boats comprising the West Coast fleet total has ranged between 21 and 44 vessels since 1997. Table II-1 shows the fleet composition and effort since 1995.

Table II-1. Western Pacific longline logbook summary for 1995 through 2002.

	1995	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
# vessels	10	15	25	28	37	44	39	21
# trips	36	71	55	70	101	137	128	91
# sets	311	678	663	922	1,430	2,104	1,937	1,294
# hooks set	251,704	550,420	518,841	738,739	1,143,066	1,608,593	1,443,029	948,657
# light sticks used	137,756	193,050	90,140	206,960	150,369	170,135	487,525	352,834

Source: http://www.NOAA Fisheries.hawaii.edu/fmpi/fmep/hilong/westcoast.htm

The number of longline vessels fishing in the West Coast fleet has ranged between 21 and 44 vessels since about 1997. Vessels vary in length from 20 to 35 m. Longline fishing gear consists of a main line strung horizontally across 1-100 kilometers (km) (< 1-62 mi) of ocean, supported at regular intervals by vertical float lines connected to surface floats. Descending from the main line are branch lines, each ending in a single, baited hook. The main line droops in a curve from one float line to the next and bears some number (2-25) of branch lines between floats. Fishing depth is determined by the length of the floatlines and branchlines, and the amount of sag in the main line between floats. The depth of hooks affects their efficiency at catching different species. When targeting swordfish, vessels typically fish 24 to 72 km (15-45 mi) of 600 to 1,200 pound test monofilament mainline per set. Mainlines are rigged with 22 m branch lines at approximately 61 m intervals and buoyed every 1.6 km (1 mi). Between 800 and 1,300 hooks are deployed per set. Large squid (*Illex spp.*) are used for bait; various colored light sticks are also used. The mainline is deployed in 4 to 7 hours and left to drift (unattached) for 7 to 10 hours. Radio beacons are attached to the gear for recovery. Retrieval requires 7 to 10 hours. Fishing occurs primarily during the night when more swordfish are available in surface waters. Longline gear targeting swordfish is set at sunset at depths less than 100 m, and hauled at sunrise.

A typical longliner carries a crew of six, including the captain, although some of the smaller vessels operate with a four-man crew. Fishing trips last around 3 weeks. Some vessels do not have built-in refrigeration equipment, limiting their trip length. They take on ice at the docks, but this only supports relatively shorter trips (10-14 days). Some vessels have ice-making equipment such that they can refresh ice supplies and maintain fish quality with iced brine for long periods (up to 60 days).

Swordfish has been the principal target species. Other marketable species in the longline catch include opah (*Lampris regius*), dolphin fish (*Coryphaena hippurus*), and escolar (*Lepidocybium flavobrunneum*). Relatively few sharks, in proportion to those caught, have been marketed from this fishery. The major shark bycatch is blue shark, which is discarded. Other bycatch includes striped marlin, turtles, birds, and marine mammals.

The West Coast-based longline fishery mainly targets swordfish and is estimated to have fished a total of about 1 million hooks in 2002. This estimate is based on dock observation records of vessel departures and returns, from which estimates of days at sea and days fishing are derived based on NOAA Fisheries observers' records of search and fishing time by observed vessels. Effort has been deployed over a large area, including occasional trips from California to waters north of Hawaii. The principal fishing area is east of 150EW longitude (Figure 1).

Swordfish landings by California-based longline vessels increased from 28 mt in 1991 to 497 mt in 1994, with swordfish accounting for 78 percent, tunas 9 percent and sharks 6 percent of total longline landings in 1994. The overall trend for West Coast longline landings since 1991 is decidedly increasing, with total landings ranging from a 1991 low of 56 mt to the 1999 high of 1,524 mt, and swordfish landings increasing from 28 mt to 1,287 mt. There is a developmental pelagic longline fishery authorized off Oregon, but it has produced negligible landings. California receives virtually all of the high seas longline catch. In 1994, West Coast swordfish landings by California-based longline vessels represented 35 percent of total swordfish exvessel revenues; by 1999 this share had risen to 56 percent.

As indicated, the longline fishery targets swordfish. However, it is conceivable that these vessels could attempt to fish for tuna as the proposed adoption of the FMP and associated regulations restrict swordfish opportunities. Tuna-target longline fishing is also known as deep-set longline fishing. A line shooter is used on deep-sets to deploy the mainline faster than the speed of the vessel, thus allowing the longline gear to sink to its target depth (400 m for bigeye tuna[1]). Deep-set longline gear is set in the morning and hauled in the afternoon. The main line is typically 30 km to 100 km (18 nm to 60 nm) long. A minimum of 15, but typically 20 to 30, branch lines (gangions) are clipped to the mainline at regular intervals between the floats. Each gangion terminates with a single baited hook. The branch lines are typically 11 to 15 meters (35 to 50 feet) long. *Sanma* (saury) or sardines are used for bait. No lightsticks are attached to the gangions on this type of longline set. A typical deep-set (one day of fishing) consists of 1,200 to 1,900 hooks. Data from observations on Hawaii-based vessels indicate that sea turtle interactions are much less frequent for deep-sets than for swordfish sets.

There are no observer records of sea turtle or seabird interaction rates for longline sets targeting tuna in the eastern Pacific. It seems unlikely, based on current information, that tuna targeting can provide an economically viable alternative to swordfish in the eastern Pacific. However, it cannot be ruled out. The FMP would require placement of observers at levels that provide statistically valid estimates of bycatch and protected species interactions. If tuna fishing occurs, NOAA Fisheries will be able to determine if bycatch of protected species is a problem that needs to be addressed.

NOAA Fisheries began placing observers on the West Coast-based swordfish longline fishery in October 2001 and began a mandatory observer program in August 2002, pursuant to the MMPA.

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[1] 400 meters is the average deepest depth, ranging from 100 to 400 meters.

From October 2001 through November, 2003, 391 sets were observed. The purpose of the observer program for this fishery is to document the incidental take of marine mammals, sea turtles, seabirds, target and non-target fish species, and to collect biological specimens. Observers also collect socio-economic data from vessel owners/operators. During the 2002-03 fishing season, observer coverage was approximately 12 percent of total fleet coverage[2]

Effort in the West coast-based longline fishery

As shown in Table II-1, the West coast-based longline fleet has comprised between 10 and 44 vessels since 1995. In 2002, 21 vessels actively fished, deploying nearly 1 million hooks. Effort for 2003 was similar, with 21 vessels actively fishing (D. Petersen, NOAA Fisheries, personal communication, December, 2003), based on high seas longbook data, PacFin landings, and observer contractor fishing effort determinations. The HMS FMP states that current fishing effort by the fleet is 1.5 million hooks. However, based on fishing effort during the last two years, NOAA Fisheries has determined that an expected effort of 1 million hooks is more representative. NOAA Fisheries does not expect this effort to change in the coming years.

c. *West Coast Drift Gillnet Fishery*

The California/Oregon (CA/OR) drift gillnet fishery targets swordfish and thresher shark. The fishery has been observed by NOAA Fisheries since July, 1990, and observer coverage has ranged from 4.4 percent in 1990 to an estimated 22.9 percent in 2000. Between July 1990 and January 31, 2003, NOAA Fisheries has observed a total of 6,720 sets. The fishery occurs primarily within 200 nautical miles of the California coastline and to a lesser extent off the coast of Oregon (Figure 2).

Drift gillnets capture by entanglement. Typically, besides an appropriate vessel, drift gillnet gear required for this fishery includes a net, 45 to 60 large inflatable ball buoys, a spar buoy called a “high flyer” affixed with a radar reflector and strobe light, a deck mounted hydraulically powered reel on which to store the net, and a reel mounted level wind to assist in deploying, and retrieving the net. A large net guard of one of two basic styles, either resembling a catchers mitt or resembling a football helmet’s face guard, is affixed to the stern of the vessel and lowered into the water during retrieval to keep the net from becoming entangled in the propeller. A stern roller reduces net wear. Each net is custom-made from component parts that are often purchased separately from different suppliers. The basic components of a net include the webbing, a small diameter lead-cored braided line (leadline), a large diameter braided or three-strand buoyant line (floatline), small diameter braided hollow-core poly line (buoyline), and a large quantity of seizing twine to attach it all together. Nets are most commonly constructed with one size of twisted nylon strand meshes that typically measure 18 to 20 inches

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[2] Unit and definition of fishing effort for purpose of estimating coverage: longline vessels in this fleet make a single gear haul (set) each day. The unit of effort is defined as the number of hooks deployed (i.e. 1,000 hooks - 1 unit of effort).

between opposing knots when the mesh is stretched together. The curtain of webbing ranges from 80 to 160 meshes deep (90 to 170 ft), and from 4,800 ft long to the legal maximum of 6,000 ft finished length. Webbing is hung loosely, much like a drapery, between the floatline at the top, and the leadline at the bottom. The looseness, or “slack,” gives the net its entanglement properties and is built into the net by adjusting the amount of net captured with the hangings that attach the top of the webbing to the floatline so that the finished length of the net is about 40 to 50 percent less than the total length of webbing used if it were stretched out. A fisher chooses the depth/length combination for his net based on the size reel that it would require, and the amount of vessel stability sacrificed by carrying the weight of reel and a wet net. The net is suspended below the sea surface by the ball buoys to a depth equal to the length of the buoylines. This depth has historically ranged from 18 ft to as much as 90 ft, but is currently limited by regulations enacted under the MMPA to a minimum depth of 36 feet below the sea surface.

The length of drift gillnet trips range from one night to one month, but typically last 5 to 15 days. Fish availability, market price, weather conditions, phase of the moon, vessel fishing range, and fish-cooling capabilities dictate the timing, and length of fishing trips. Crew size is typically two or three persons, including the captain. Around sunset, the net is usually deployed starting at the upwind position of the set. The high flyer is attached to the end of the net and both are lowered into the water. The vessel proceeds slowly in a downwind direction reeling off net as it goes. As a series of buoylines that are attached to the floatline about 100 ft apart unwind from the reel, a ball buoy is attached to the buoyline and thrown overboard. At the end of the set, the vessel stops and drifts with the net attached throughout the night. Typically before sunrise, retrieval of the net begins. Fish-cooling capabilities vary widely from none to ice, spray brine, or blast refrigeration.

Fishers locate where to fish by looking for temperature fronts between cooler and warmer water masses, or turbidity fronts between green and blue water masses. Using prearranged high frequency radio channels, drift gillnet fishers often communicate in coded messages with other members of loosely organized “code-groups.” However, in recent years, the accessibility of high-resolution satellite generated sea surface temperature data has greatly reduced the importance of code-group communications for locating the temperature fronts where swordfish are typically found.

California’s drift gillnet permits are issued to individual fishers rather than to vessels. This practice separates the value of the permit from the value of the vessel, keeps the value of vessels from becoming inflated and allows permit holders to buy new vessels as needed. Permit holders are required to be onboard during fishing operations, and fishers are required to declare the fishing vessel being used.

Fishing effort has varied from season to season. Effort peaked in the 1986-87 season with over 11,000 sets, quickly declined to about 4,500 sets by 1990, and averaged about 3,500 sets per year through 1998 (Enriquez 2000 *in* a NOAA Fisheries working paper: Observed Catch of HMS in the California/Oregon Drift Gillnet Fishery). However, effort has been declining annually since then, with only about 350 trips and 1,948 sets in the 2001-2002 fishing year.

Historically, the CA/OR drift gillnet fishery has occurred along much of the West Coast. The fishery now operates primarily outside of state waters to about 150 miles offshore, ranging from the U.S Mexico border in the south to northward of the Columbia River, depending on sea temperature conditions. Because of seasonal fishing restrictions, and the seasonal migratory pattern of swordfish, about 90 percent of the annual fishing effort occurs between August 15 and December 31. Depending on where they fish, drift gillnet vessels primarily land fish in San Diego, San Pedro, Ventura, Morro Bay, Monterey, Moss Landing, and San Francisco Bay area ports where it is sold in the fresh fish market providing high quality, locally-caught fish for the restaurant trade.

The principal species of thresher sharks caught in this fishery are common, bigeye and pelagic thresher. Shortfin mako also constitutes an important incidental catch. They are not so abundant as to attract directed effort, but their market quality and ex-vessel value are good. Blue sharks are rarely landed or marketed. The incidental catch of non-target species in the drift gillnet fishery varies by year, but some of the predictable and saleable species include albacore and bluefin tunas, Pacific bonito (*Sarda chiliensis*), opah *(Lampris guttatus)*, and louvar *(Luvarus imperialis)*.

Bycatch (discarded fish) in the drift gillnet fishery is mainly comprised of ocean sunfish *(Mola mola)* and blue shark. In the period 1990-1998, ocean sunfish amounted to 26.1 percent of the total observed catch of which 80.6 percent were returned alive, and blue shark amounted to 15.2 percent of the total observed catch of which 14.5 percent were returned alive (Holts and Rasmussen 1999).

In order to protect gray whales, in 1985, California adopted a closure within 25 miles of the mainland coastline from December 15 through the season's end on January 31. Due to high marine mammal interactions, the drift gillnet fishery was listed as a Category I fishery under the MMPA. Placement in this category required the formation of the Pacific Offshore Cetacean Take Reduction Team in 1996 to develop a Take Reduction Plan (TRP) for the drift gillnet fishery aimed at reducing the level of marine mammal interactions to specified levels. In 1997, regulations implementing the TRP required all drift gillnet fishers to attach a number of acoustic "pingers" to the top and bottom of the net, lower the top of the net to a minimum of 36 ft below the sea surface, and attend annual "skipper workshops" to facilitate the exchange of information with NOAA Fisheries regarding marine mammal interactions in the fishery.

In the fall of 2000, NOAA Fisheries conducted an ESA-required section 7 consultation to examine the impacts of issuing an MMPA permit under section 101(a)(5)(E) for the incidental taking of ESA-listed marine mammals to the drift net fishery. The resulting biological opinion concluded that the issuance of the permit and the associated operation of the drift gillnet fishery was likely to jeopardize the continued existence of the leatherback and loggerhead sea turtles. The reasonable and prudent alternative required the imposition of additional time and area closures. Based on this Opinion, NOAA Fisheries implemented regulations that eliminated drift gillnet fishing effort from August 15th through November 15th in the area bounded by straight lines from Point Sur (34E18.5' N) to 34E27' N 123E35' W, to 34E27' N 129E W, to 45E N 129E W, to the point 45E N intersects land in order to reduce impacts to leatherback sea turtles. If an El Niño condition is predicted to occur, or is occurring, the area south of

Point Conception will be closed to drift gillnet fishing from June 1st through August 31st to reduce impacts to loggerhead sea turtles. NOAA Fisheries published the final rule for this action on December 16, 2003.

Effort in the California/Oregon drift gillnet fishery

NOAA Fisheries does not expect additional drift gillnet vessels to enter the CA/OR drift gillnet fishery in the future because it is a limited entry fishery. Therefore, only a maximum of 185 permits for California and 10 permits for Oregon will be re-issued each year.

Fishing effort in the CA/OR drift gillnet fishery peaked (more than 11,000 sets per season) in the mid-1980s (Hanan *et al*., 1993) and decreased to less than 3,000 sets per year in 1999 (CDFG, unpublished data). Legislation passed in 1982 established the fishery as a limited entry fishery with a maximum of 150 permits (California Code of Regulations, Title 14, §106). Because the legislation allowed those already involved in the fishery to continue fishing, the actual number of permittees initially exceeded the established cap of 150 permits. Consequently, no new entrants could enter the fishery until the number of permittees dropped to below 150. In 1984, an additional 35 permits, referred to as experimental swordfish permits, were established for taking swordfish north of Point Arguello (Hanan *et al*., 1993). There were over 210 active permittees (those that caught and landed fish) participating in the fishery in the 1986-87 season (NOAA Fisheries, 1997b). In 1989, the 35 experimental swordfish permits were combined with the 150 permits (185 permits). The number of drift gillnet permits issued by the California Department of Fish and Game (CDFG) has decreased from 167 permits in 1997 to 139 permits in 1999 (R. Read, CDFG, personal communication, June 2000). This number is expected to drop further as CDFG continues not to issue new permits and permits lapse because of retirement, illness, injury, and death.

The overall fishing effort trend has continued to decline during the last 16 years with the lowest fishing effort occurring in 2001 with only 1,667 total sets. Based on this trend, NOAA Fisheries anticipates that overall fishing effort for any of the next three calendar years will not exceed 2,000 sets. This annual estimate is supported by the fact that the fishing effort average for calendar years 2000 - 2002, is equal to approximately 1,800 sets per year. Furthermore, the number of vessels that have obtained Marine Mammal Authorization Certificates during the past three years have decreased from 126 vessels in 1997 to 97 vessels in 2000 to 92 vessels in 2002 (D. Petersen, NOAA Fisheries, pers. comm., January 2004). This reduction in the number of fishing vessels since the mid-1990s can be attributed partly to the larger vessels (greater than 50 feet) switching from fishing swordfish using a drift gillnet to fishing squid using a purse seine net and other vessels switching to longline gear. In addition, the number of fishing days was further reduced during the mid-1990s when many of the larger vessels began targeting albacore tuna during the summer months and into late September rather than target swordfish using drift gillnet gear. This reduction in the number of fishing vessels participating in the fishery, the reduced fishing days by vessels targeting albacore, and the number of permits lapsing because of retirement, illness, injury, and death is expected to keep the overall fishing effort by the

CA/OR drift gillnet fishery to below 2,000 sets for each subsequent calendar year.

d. Eastern Tropical Pacific Tuna Purse Seine Fishery

The ETP is designated as the area bounded by 40EN latitude, 40ES latitude, 160EW longitude, and the west coast of the Americas. U.S. vessels primarily fish in the area between San Diego, California and 20ES and from the Central American coast out to 150EW or 160EW (see Figure 3; A. Coan, NOAA Fisheries, personal communication, January 2004). The target species sought by the U.S. ETP tuna purse seine fishery are yellowfin and skipjack tuna, although bigeye has also become an important component in the fishery in recent years. Purse seining is currently the most efficient method of catching tuna. Tuna purse seine vessels, for the purposes of this analysis, typically vary in size from 400 to 1700 short tons (st) carrying capacity. Exceptions to this size range are rare; however, an occasional U.S. purse seine vessel less than 400 st may target tuna in the ETP year-round. The majority of these smaller U.S. purse seine vessels based on the West Coast focus on coastal pelagic species and only target tuna when they are seasonally available.

The HMS FMP includes the purse seine fishery within the ETP. However, except for smaller vessels and one or two large vessels, the remainder of the fleet operates outside of the U.S. EEZ, does not land catches at U.S. ports, and does not therefore require permits under the HMS FMP. In addition, NOAA Fisheries has previously consulted on the operations of the large vessel purse seine fleet. The proposed FMP makes no changes to that fishery. As a result, the effects of the large vessel ETP purse seine fleet are described in the *Status of the Species and Environmental Baseline*. The operations of small vessels are not covered under any previous consultations, and these vessels fall under the jurisdiction of the HMS FMP.

Purse seines are large nets that encircle the target species. Depending on the size of vessels, nets generally vary from 1/4 mile to one mile in circumference, and from 300 to 700 feet in depth. During deployment of gear, the net forms a circular wall of webbing around the school of fish. The net must be deep enough to reduce the likelihood of fish escaping underneath, and the encircling must be done rapidly enough to prevent the fish from escaping before the bottom is secured ("pursed") shut.

A set is initiated when a skiff is released from the stern of the purse seiner, anchoring one end of the seine. The targeted fish are contained in a vertical cylinder of webbing after the seine vessel encircles the targeted school and rejoins the skiff. The bottom of the net is then pursed by hauling the cable that is threaded through rings on the bottom of the net. After the net is pursed, it is retrieved until the diameter of the net compass and the volume of water inside the net decreases to a point when, in both space and time, fish are sufficiently concentrated that they can be hydraulically scooped ("brailed") into wells onboard the vessel.

For reasons that are still not clear, yellowfin tuna over 55 pounds are often found in association with schools of dolphin in the ETP. Tuna fishermen have taken advantage of this association between yellowfin tuna and dolphins by using the more easily detected dolphin schools to help find fish. Dolphin sets (which generally can only be carried out by large purse seine vessels with the capacity to carry speed boats and appropriate nets) yield relatively large yellowfin tuna and result in low bycatch relative to other types of sets: log sets and school sets. Log sets (sets on tuna schools associated with floating logs or fish aggregating devices (FADs)) tend to yield relatively small, pre-reproductive yellowfin tuna or skipjack tuna (or a mixture of both tuna), together with a wide variety and large quantity of other biota, including sea turtles, sharks, billfish, other sportfish, and a variety of other small non-commercial tunas. School sets (sets on tuna schools not associated with either floating objects or with dolphins) target free-swimming schools of yellowfin or mixed yellowfin and skipjack tuna that are generally moderately small, and result in relatively less bycatch than log sets. Traditionally, dolphin sets have been preferred by the majority of large vessel tuna fishermen because they yield large quantities of large yellowfin tuna that are economically valuable, relatively easy to locate and capture, not associated with unwanted fish, and generally receive a higher price per pound than the smaller tuna associated with school or log sets. Currently, no U.S. Class 6 tuna purse seine vessels in the ETP are setting on dolphins, and only one or two vessels occasionally fish in the EEZ or make landings into a West Coast port.

The bycatch of dolphins associated with large yellowfin tuna by purse seiners in the ETP prompted the United States to initiate action within the Inter-American Tropical Tuna Commission (IATTC), a regional fisheries management organization of which the United States is a member, to establish a program to address the tuna-dolphin issue. The IATTC, whose Convention is implemented domestically by the Tuna Conventions Act of 1950, is responsible for developing measures to conserve and manage tuna resources in the ETP, and also provides the Secretariat for the International Dolphin Conservation Program. A schedule of progressively decreasing annual limits on dolphin mortality was implemented and a research program was approved.

Vessel captains helped develop the “backdown” procedure, along with other techniques and gear modifications, in the 1970's to promote the safe release of dolphins encircled in the tuna purse seine fishery. The objective of performing the backdown is to allow the safe release of encircled dolphins without loss of tuna. Backdown is a complex technique that may vary from set to set, depending on the specific conditions (e.g., currents, winds) present at any given time. Backdown occurs after the net has been pursed (rings along the bottom of the net are brought aboard the vessel, or “rings up”) and consists of six mains steps (Coe *et al.*, 1984; NOAA Fisheries, 1986):

(1) Tie down at pre-established marks;
(2) With the wind at port beam, use the skiff and bow thruster to move the stern away from the net, then shift the vessel in reverse;
(3) Reverse slowly as the backdown channel (long narrow channel between the port bow of the purse seine vessel and the apex of the net) forms, then increase speed to sink the apex of the corkline;
(4) If fish move toward the apex, slow to allow the corks to rise. When the fish turn toward the vessel, shift back into reverse;

(5) Continue backdown until it is no longer possible to remove live marine mammals from within the net;
(6) Complete backdown with the wind on the port beam.

Backdown sinks the corkline of the seine net at the apex, which allows, with the aid of crewmembers deployed to the water and speed boats that hold the backdown channel open, dolphins to swim out over the top of the net and tuna to be retained. In many situations, the sunk corkline is actually pulled out from under dolphins, rather than the dolphins actively swimming out of the net.

In 1997, the U.S. Congress passed, and the President signed, the International Dolphin Conservation Program Act (IDCPA) of 1997. This legislation required changes to the dolphin-safe labeling standard in the Dolphin Protection Consumer Information Act. Under the proposed standard, dolphin-safe catches of yellowfin tuna would be identified on a per-set basis rather than on a per-fishing-trip basis, as under the current standard. Dolphin-safe would indicate the absence of dolphin mortality or serious injury in a set. Interim regulations carrying out the IDCPA are in effect, although court action has resulted in retention of the previous dolphin-safe standard for the time being; dolphin-safe tuna are those caught on a fishing trip during which no dolphins were intentionally encircled and no dolphins were seriously injured or killed. As a practical matter, changes in the dolphin-safe labeling standard will not significantly affect U.S. HMS purse seine fisheries, as no fishing on dolphin is occurring and all U.S.-caught tuna is dolphin-safe under the previous standard. However, it is important to note that as a result U.S. vessels are setting on free swimming schools or those associated with floating objects. Available data indicate that these two methods of purse seining for tuna result in higher rates of bycatch than setting on dolphin (Hall, 1998; IATTC, 2002).

The IATTC classifies vessels according to their carrying capacity into the following size classes: Class 1 = less than 51 st; Class 2 = 51-100 st; Class 3 = 101-200 st; Class 4 = 201-300 st; Class 5 = 301-400 st, Class 6 = more than 400 st (362.8 mt).

The U.S. fleet of purse seiners in the ETP reached approximately 144 vessels in 1979, but by 1999 it had decreased to 10 vessels of Class V or VI size. In 2002, only five U.S. Class 5 or 6 vessels actively participated in the fishery and were listed on the IATTC register of vessels qualified to purse seine for tuna in the ETP (Table II-2). Until the 1990's, most of the U.S. purse seiners operating in the ETP were Class 6 vessels, targeting tuna year-round. However, in the mid-1990's smaller Class 1-5 purse seine vessels began to outnumber Class 6 vessels. Generally, Class 1-5 purse seine vessels only occasionally target tunas when they are seasonally available and their effort is focused on coastal pelagic species, so Class 6 vessels still comprise the majority of purse seine vessels targeting tuna in the ETP.

Most Class 6 vessels that previously fished in the ETP have either re-flagged or are active in the western Pacific Ocean, where a treaty with the south Pacific islands (the South Pacific Regional Tuna Treaty, signed in 1988) provides the U.S. fleet with access to richer fishing grounds. For purely economic reasons, the trend is not likely to change. In general, western Pacific tuna fishermen catch more tuna per set compared to ETP tuna fishermen, and thus make fewer and shorter trips. However, it should be noted that yellowfin tuna are the target of the purse seine fishery in the ETP, whereas skipjack tuna are targeted in the western Pacific Ocean. No association is known to occur between skipjack tuna and dolphins.

Table II-2. Estimates of the number of U.S. purse seine vessels fishing in the ETP by year and size class.

	Size class						
Year	**1**	**2**	**3**	**4**	**5**	**6**	**Total**
1990	0	12	4	0	1	29	46
1991	0	5	5	0	1	13	24
1992	0	6	5	0	1	8	20
1993	0	10	5	0	2	8	25
1994	0	12	4	0	2	9	27
1995	0	7	4	0	2	5	18
1996	1	10	4	0	2	6	23
1997	1	12	4	0	2	6	25
1998	0	13	4	0	2	6	25
1999	0	4	3	0	2	5	14
2000	0	3	2	0	2	6	13
2001	0	0	1	0	2	5	8

(Source: IATTC, 1991, 1992, 1993, 1994, 1995, 1996 1997, 1998, 1999, 2000, 2001, 2002).

Tropical tuna caught in the U.S. purse seine fishery are canned as light meat tuna. Catches have historically been delivered or transshipped to canneries in California, Puerto Rico, American Samoa, other canneries in the Pacific rim or to Europe. Today only four U.S. plants are in operation, two in America Samoa (conventional canneries) and one in Puerto Rico, with a small plant in California that cans only imported tuna loins.

Landings and corresponding exvessel revenues at West Coast ports have greatly decreased since the 1980s, when the major West Coast canneries began relocating overseas. Most of the tropical tuna landings on the West Coast are now made by “wetfish” (sardine, mackerel, anchovy) purse seiners that catch relatively small quantities of tropical tunas only when they are seasonally available. As noted above, only one or two large purse seine vessels now fish in the EEZ at any time or make landings into a West Coast port.

Significant growth in the West Coast purse seine fishery for tuna is not expected, and declines seem more likely, but changes are difficult to predict with so many variables. Tropical tunas are not significantly abundant in the U.S. EEZ or available to current commercial fishing gear off the West Coast. U.S. vessels continue to be excluded from Mexico's waters where fishing is more productive. States' area closures and other fishery restrictions will likely remain in place without the FMP. Within the U.S. EEZ, the expected baseline for this fishery is no more than 5 part-time, small purse seine vessels with total landings of 1,000 mt or less valued at $1.5 million or less per year. Total employment in this fishery is expected to remain below 50 persons, with the fishery still centered in southern California.

In addition to the 5 small purse seine vessels that are expected to participate in the tuna fishery when fish are available in the U.S. EEZ, a maximum of 6 Class 6 U.S. purse seine vessels are likely to fish for tuna in the broader ETP but rarely in the U.S. EEZ. These large vessels target tuna on the high seas on a full-time basis.

Effort in the ETP purse seine fishery

As shown in Table II-2, between 1996 and 2001, between 5 and 6 large U.S. vessels actively fished in the ETP, mainly on the high seas and landing their catch in foreign ports. During that same period, the number of small U.S. purse seiners in this fishery ranged between a low of 3 to a high of 19. Between 1999 and 2001, the number of small vessels making landings into West Coast ports has declined, from 9 in 1999, to 7 in 2000, and finally to 3 small purse seine vessels in 2001.

NOAA Fisheries does not expect additional large U.S. purse seine vessels to enter the ETP tuna purse seine fishery in the future because of historical trends in vessel participation and the high start-up costs for a new large vessel to enter the fishery, and fishing in the EEZ on a regular basis would not be expected. In the late 1980's and early 1990's, with the passage of the South Pacific Regional Tuna Treaty, most U.S. large purse seiners either re-flagged or moved to the richer fishing grounds of the central-western Pacific Ocean. With little incentive to fish in the ETP, NOAA Fisheries does not expect a future influx of large U.S. purse seine vessels. A recent IATTC resolution which set fleet limits and a voluntary U.S. commitment to limit participation of domestic vessels to a total 8,969 mt carrying capacity (Chris Fanning, NOAA Fisheries, pers. comm., January 14, 2004) are also expected to limit or preclude future increases in large U.S. purse seine vessels.

NOAA Fisheries does not expect a significant influx of smaller vessels into the ETP tuna purse seine fishery. The coastal pelagic fishery is a limited entry fishery. Therefore, any small (# 400 st) purse seine vessels that potentially would enter the ETP tuna fishery would either be a brand-new purse seine vessel or a purse seine vessel that normally targets squid–squid is not a limited entry fishery. Squid purse seine vessels that originate from Washington generally fish for more profitable salmon in Washington and Alaska in the summertime, not for tuna in the ETP. Squid purse seine vessels also operate out of the ports at San Pedro and Monterey, California.

e. California Harpoon Fishery

The harpoon fishery for swordfish in California dates back 3,000 years when Native Americans fished with stone and wooden harpoons from driftwood canoes. The modern harpoon fishery off California began in the early 1900s, was the primary gear for swordfish from the early 1900s to the 1980s, and declined in 1980, when drift gillnet fishing started. Many vessels converted to drift gillnet fishing gear or obtained permits to use both types of gear. Today, only a handful of vessels continues to participate in the harpoon fishery.

The harpoon fishery targets swordfish, although small quantities of shark are also landed by harpoon gear, most often common thresher and shortfin mako. There have been infrequent reports of blue, hammerhead (*Sphyrna spp*), soupfin (*Galeorhinus zyopterus*), and white (*Carcharodon carcharias*) sharks being recorded as taken with harpoon gear[3].

Harpoon vessels are from 6 m to 26 m (20-87 ft) in length with a 6 m to 8 m bow plank and hold capacities up to 100 mt (Coan et al. 1998). When a fish is spotted, the plank is positioned above the swordfish and the harpoon thrown from the end of the plank. The fish is stored over ice for the rest of the trip. The hand-held harpoon consists of a 10-16 foot metal and/or wood pole attached to a 2-foot long metal shank and tipped with a 4-inch tethered bronze or iron dart. After harpooning, the handle is pulled free from the dart, and the mainline, marker flag, and floats are thrown overboard, leaving the fish to tire itself. The vessel then proceeds to search for and/or harpoon other fish. After the fish is tired, in approximately two hours, the vessel returns to retrieve it.

The harpoon fishing season typically begins in May, peaks in July to September, and ends in December, coincident with the annual northwesterly movement of the North Equatorial Countercurrent and during months of calm sea conditions that harpoon fishing generally requires. Fishing usually concentrates in the Southern California Bight (SCB) off San Diego early in the season and shifts to areas as far north as Oregon later in the season, especially in El Niño years. Swordfish are usually sighted basking at the surface of the water in temperatures between 12E to 26EC. In El Niño years, the range of water temperatures where the majority of swordfish sightings occur narrows and favors warmer temperatures between 20E and 22EC. Harpoon is legal gear in California and Oregon, but is not defined as legal gear in Washington.

Harpoon vessels work in conjunction with an airplane to spot swordfish basking at the surface beyond binocular range from a vessel or sub-surface swordfish. Spotter planes were introduced in the early 1970s. Spotter planes were banned by California Department of Fish and Game (CDFG) for one year during 1976. In 1984, spotter airplanes were allowed full-time in the fishery.

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[3] Shark catches by harpoon gear are highly suspect according to industry and Coan et al. (1998).

Confinement of the fishery to a relatively small area, principally the calm waters of the SCB, leaves it vulnerable to changing environmental conditions and competition from other gears. Environmental effects during El Niño events lead to decreased catches and CPUE. Competition from the drift gillnet fishery since 1980 has also led to decreased harpoon catches. Prices received for harpoon-caught swordfish generally exceed those of drift gillnet-caught swordfish, since the harpoon-caught swordfish do not spend the time in the net that the drift gillnet-caught swordfish do, and thereby generally allowing a fresher product. The harpoon season tends to taper off when the drift gillnet season begins because the substantial increase in swordfish volume lowers the ex-vessel swordfish price for harpoon-caught swordfish. The effects (if any) from recent increases in offshore longline fisheries are not yet seen.

f. Charter and Private Boat Recreational Fishing

Recreational fishing for large, migratory pelagic species began off southern California and Baja California, Mexico in the late 1800s. This fishery now operates year round with peaks in activity for tuna, billfish and pelagic sharks during the spring and summer and lasting into the fall. The fleet is composed of charter vessels, party boats, and head boats, collectively called commercial passenger fishing vessels (CPFV), and privately owned vessels. The HMS recreational fisheries off the Washington and Oregon coasts are solely targeting albacore tuna using hook-and-line gear. A recreational fishing license is not required to fish for albacore tuna in Washington but is required in Oregon. The Washington and Oregon recreational fishery is open year-round and there is no minimum size limit. In Washington, there is no catch or possession limit. In Oregon, albacore tuna come under the catch limit of 25 miscellaneous fish.

Biological and socioeconomic data available for HMS recreational fisheries pale in comparison to those for HMS commercial fisheries. State administered logbook programs are an important source of recreational fishing catch and effort data for CPFV patrons, including those participating on long-range trips aboard California based CPFVs into Mexican waters. NOAA Fisheries conducts the Marine Recreational Fishing Statistical Survey (MRFSS) which routinely collects recreational catch and effort data from West Coast marine anglers, including those targeting HMS from CPFVs and privately owned vessels, as well as occasional add-on surveys to collect angler socioeconomic data. The data available from MRFSS and the state recreational fishery monitoring programs are provided to the coastwide recreational fishery network data system (RecFIN) where they are integrated into a comprehensive coastwide marine recreational fishery data base. Recreational billfish fishery data are also collected by the Southwest Fisheries Science Center (SWFSC) through its Billfish Angler Survey and the Billfish Tagging Programs. The data from these programs are published annually in the Billfish Newsletter (Holts and Prescott 2001).

West coast recreational fishing activity directed towards large, migratory pelagic species emanates mainly from CPFVs and privately owned vessels departing sportfish landings, marinas and launch ramps dotting the southern California coast from Los Angeles to San Diego, California. The Sportfishing Association of California (SAC) is the major industry organization representing nearly 200

CPFVs operating out of 23 landings from Morro Bay to San Diego. This fleet carries almost 1 million passengers annually to local and Mexican fishing grounds. The fleet and supporting shoreside facilities represent a monetary investment totaling close to $80 million, and a labor force of about 4,000 persons. In 2000, there were an estimated 876,000 trips taken aboard southern California based CPFVs resulting in a total catch of 2,941,000 fish, a 44 percent and 30 percent increase respectively from 1999 (RecFIN). Approximately 429,000, 49 percent, of all southern California based CPFV trips in 2000 accounted for total HMS catches of 99,000 fish, 3 percent of the total CPFV catch. This is 12 times the number of trips, and a 21 percent increase in HMS catch compared to 1999.

A large number of southern California based privately owned vessels are used to recreationally fish for HMS, upwards of 6,000 annually. These vessels cover a wide range of sizes and types, ranging in length from 17 ft skiffs to 90 ft or greater luxury yachts, with many vessels under 30 ft. In 2000, private vessels made approximately 1,760,000 fishing trips, of which 1,318,000, 75 percent, resulted in HMS catches. This was an increase of 51 percent and 100 percent in total trips and HMS trips from 1999 (RecFIN). The estimated total recreational catch of southern California based private vessels in 2000 was 2,594,000 fish of which 57,000, 2 percent, were HMS (RecFIN), up 37 percent and 150 percent respectively from 1999. Southern California based private vessels accounted for 75 percent of the total (CPFV plus private vessel) number of HMS trips, and 37 percent of total HMS catches in 2000, a decrease of 21 percent and an increase of 68 percent respectively from 1999.

(1) Charter/Party Boat Fleet. Tropical tunas, billfish and sharks become available off the West Coast as they move seasonally eastward from oceanic waters and northward from Mexico. Except during periods of warm water, recreational catches of these species are almost exclusively from waters off southern California (Table 2-60 in the HMS FMP). Albacore and northern bluefin (more temperate water species) move into the coastal waters along the West Coast from more temperate waters offshore. The timing and extent of the species appearance is dependent on seasonal development of environmental and oceanographic conditions such as water temperature, coastal up welling, strength of the California Current, El Niño episodes and possibly longer decadal cycles. Albacore are one of the most important species caught by the West Coast charter and CPFV fleet.

The CPFV fleet offers short trips from one to two days and long-range trips of up to 15 days into Mexican waters. The fleet is made up about 300 vessels from about 8 to 40 m in length and target large pelagic species when quantities occur within their range.

The smaller and faster California sport fishers licensed to carry six passengers or less are called “six-packs.” Six-pack vessels target tunas, billfish and coastal pelagic species on one or two-day trips. These vessels are more likely to spend the extra time necessary to catch billfish if requested by their clientele. The larger CPFV vessels may carry 40 or more passengers and target albacore, bluefin, yellowfin, skipjack, dorado and coastal pelagic species on long-range trips into Mexico and shorter trips of one or two days within the SCB. Few CPFV vessels with more than six passengers will take the time necessary to catch billfish or pelagic sharks because it limits fishing activity of other passengers.

In California, charter vessels are required to submit logbooks from each trip detailing the number of anglers and catch by species to Department of Fish and Game. Oregon and Washington do not require CPFV logbooks, but Washington does have a voluntary CPFV logbook. The state agencies also conduct occasional angler interviews to supplement catch and effort data. In addition, a specialized sector of this California fishery is the long-range and multi-day fleet that fishes extensively off Mexico. Mexico provides special permits, subject to payment of fees, certain port call requirements, and observer and reporting requirements.

California's CPFV catch for 1998, by CDFG block number indicates highest catches in the Southern California Bight, and south of San Clemente Island for albacore, yellowfin, bluefin, bigeye, skipjack, and dorado (see Figures 2-2 and 2-3 and Table 2-58 in the HMS FMP). Average catch of albacore was 90,000 fish annually over the 1980- 1998 period of which 80 percent were taken off Mexico. Strong El Niño conditions and possibly decadal shifts in oceanographic conditions have a strong influence on albacore distribution and movement patterns. Reported albacore CPUE increased in the 1980s and late 1990s when El Niño conditions were present. California CPFV vessels also conduct night fishing trips for blue and mako sharks during the spring and summer and daytime trips for thresher sharks in coastal waters when supported by adequate passengers/client interest.

The San Diego Bay long-range charter vessel fleet is comprised of approximately 57 vessels. The fleet is based at three sport fishing landings: H&M Landing, with 26 vessels; Point Loma Sport Fishing, with 13 vessels; and Fisherman's Landing, with 18 vessels (London Group 1999). The typical fishing season is March through October. During the off-season (November to February), about 15 percent of the vessels fish in more northerly waters and the remaining 85 percent remain in San Diego for repair and maintenance for the upcoming season.

A total of 154,567 fishers visited the three sportfish landings in San Diego Bay in 1998 (London Group 1999). Approximately 66,355 fished in U.S. waters and the remaining 88,212 fished the waters off of Mexico.

In Washington, the major port for charter vessels is Westport, which has seven charter offices with an average of fifteen charter vessels that routinely fish for albacore tuna in the summer months. The importance of albacore tuna to this fleet has risen in the last decade as other fishery opportunities (e.g. salmon and rockfish) have declined. Based on information from charter vessel operators, the Washington recreational fishery has been fairly stable, with increases in catch in recent years. The distance from shore varies from year-to-year (in 2000, the average distance was 64 nm) and charter vessels often take two-day fishing trips for albacore. According to one charter operator, the number of anglers reserving tuna trips on his vessel nearly doubled from 1992 to 1998. The amount of tuna caught has also increased in proportion to the number of anglers, from about 1,300 in 1992 to about 3,000 in 1998.

Washington has a voluntary program for charter/party logbooks, which was instituted in 2000 with a 69 percent compliance rate. Based on the 2000 Washington logbook data, over 8,000 albacore were caught by over 1,300 anglers. The average number of albacore caught per person is six with an average weight of 14.5 pounds. Oregon does not have a logbook program.

In Oregon, it is difficult to separate the charter/party boat fishery from the private vessel recreational fishery (see the private sport description below for additional details). Albacore sport fishing off Oregon has increased in recent years due to improvements in navigational aids and marine equipment and greater appreciation of albacore as game fish. Depending upon the availability of albacore nearshore, recreational landings have ranged from 11 mt to about 80 mt in recent years, accounting for up to 2 percent of the total Oregon albacore harvest. Charter vessels account for 60-70 percent of the total recreational catch. The majority of effort and catch is concentrated along the central part of the Oregon coast, though landings occur in ports coast wide. The majority of the charter effort is out of Depot Bay and Newport, with less effort out of Garibaldi and Brookings.

(2) Private Sport Fishing Fleet. The California recreational rod-and-reel fishery for tuna, striped marlin and swordfish developed about the turn of the century. The Tuna Club of Avalon, Santa Catalina Island, California was established in 1898, and set the standard for big game fishing in waters off California which is widely adhered to today, “fair play to game fishes” (United Anglers of Southern California 2001: From brochure Recreational Fishing in Southern California). To this end, strict rules were designed to give the fish an even chance, and these rules became the foundation for the International Game Fish Association’s regulations for fish to qualify for its record books.

Highly migratory species continue to be highly prized by the recreational fishing community, although their catches of tuna and swordfish are relative low in quantity compared to the commercial catch. Swordfish and striped marlin were listed as game fish in 1931 and required a sport-fishing license issued by the CDFG. The California State legislature banned the use of harpoons to take striped marlin in 1935 and further curtailed the sale and import of striped marlin in 1937 thus preserving that southern California fishery entirely for recreational anglers. Private vessel anglers are not required to report their fishing activity or catches. Catch data from the private sport vessels are obtained through occasional CDFG monitoring and the MRFSS. There is little opportunity to recreationally fish for marlins and swordfish north of San Francisco. Most striped marlin fishing is from privately owned vessels based in local southern California marinas.

Many private vessel owners also possess Mexican fishing licenses and travel south looking for schools of tuna and billfish. Sport fishing vessels will target tuna when they move into southern California and northern Baja California waters. The estimated number of private vessels in southern California fishing large pelagic fish is 4,000 to 6,000 annually, although accurate census and economic information is currently unavailable for this fishery.

The rod-and-reel season for striped marlin and swordfish can begin as early as May and continue through November, although most fish are taken from July to October. Fishing locations are primarily in the SCB from Santa Barbara, south and into Mexico. Many California anglers will fish the productive waters around Mexico's Coronado Islands for tuna, marlin, dorado and coastal pelagic species. A few private vessel owners travel as far south as Magdalena Bay and Cabo San Lucas in the fall and winter.

California recreational anglers were allowed the use of hand-held harpoons to take swordfish until 1971. Catching swordfish with a rod-and-reel is difficult because they are usually not receptive to bait or artificial lures while finning at the surface. A few anglers now successfully target swordfish at night using techniques adapted from the East Coast that employ the use of light-sticks.

In Oregon, it is difficult to distinguish the charter/party boat fishery from the private vessel recreational fishery. Private vessels make up approximately 30-40 percent of the total recreational catch. The majority of effort and catch is concentrated along the central part of the Oregon coast, though landings occur in ports coast wide. The majority of private vessel effort is from Garibaldi to Newport, and Coos Bay and Brookings.

Most recreational albacore fishing in Oregon occurs within 50 miles of shore with most private vessels staying much closer. Fishing is usually limited to mid-July through early October, with most of the effort and catch occurring from mid-August through early September. Anglers fishing for albacore off Oregon will usually troll "tuna" jigs near the surface at 5-8 knots, and will concentrate their effort in waters with surface temperatures of 60^o F or higher.

C. Observer Program

1. California/Oregon Drift Gillnet Fishery

An observer program was mandated by the California state legislation for the developing drift gillnet fishery in 1980, and observations began in October of that year through the CDFG. From 1980-86, observers recorded detailed fishing information, including numbers of each species in the catch, for a total of 443 sets, or only approximately 1 percent of the total effort. There were no systematic observations during the 1986-87 through 1989-90 fishing seasons, after which NOAA Fisheries established an observer program as mandated by the 1988 amendments to the Marine Mammal Protection Act (MMPA) (Hanan *et al*, 1993).

Since 1990, fishing effort has been observed from the waters off San Diego to the waters off Oregon, and out beyond 200 miles from shore. Observers record bycatch by taxon for fish, marine mammals, and sea turtles, collect specimens, and record data on environmental conditions and over 10 different net characteristics (NOAA Fisheries, 1997b). From 1990-2002, the percentage of observer coverage was 4.0 percent, 9.9 percent, 13.2 percent, 13.5 percent, 18.0 percent, 15.6 percent, 13.0 percent, 22.8 percent, 17.5 percent, 20.0 percent, 22.9 percent, 20.4 percent, and 20.2 percent for an annual average of approximately 17.25 percent from 1991-2002 (full year; CDFG unpublished data and D. Petersen, NOAA Fisheries pers comm., January 13, 2004). Between July, 1990, and January 31, 2003, NOAA Fisheries has observed 6,720 sets. Observer coverage is distributed equally along the coast based on expected effort. The observer coverage is representative of the effort occurring off the west coast. Vessels are selected on an opportunistic basis. A vessel is required to carry an observer about 20 percent of the time. Therefore, if a boat just had an observer, they are not required to carry another observer until it would approach their 20 percent requirement. Vessels are notified of this obligation when they report their arrival or departure information - or at the docks, by an observer monitoring vessel activity.

2. West coast-based longline fishery

NOAA Fisheries began placing observers on longline vessels based on the west coast of the U.S. in October, 2001. During that time, observer coverage was approximately 5 percent of the total effort. In January 2002, the fishery was categorized as a "Category II" fishery; subsequently, NOAA Fisheries had mandatory authority to place observers on west coast-based longline vessels. Between October 2001 and November 2003, NOAA Fisheries has observed 391 sets (295,904 hooks, Table II-3).

Table II-3. Observed sets in the West coast-based longline fishery (D. Petersen, NOAA Fisheries, pers. comm., January 13, 2004).

Year	Percent Observed	Hooks Observed	Sets Observed
2001-2002	5	49,150	59
2002-2003	12	161,210	221
2003-Nov 2003[1]	20 (projected)	85,544	111
Total		**295,904**	**391**

[1] 4 vessels with observers still at sea as of 12/31/03

3. ETP purse seine fishery

The 1999 Agreement on the International Dolphin Conservation Program (AIDCP), implemented domestically by the International Dolphin Conservation Program Act (IDCPA) of 1997, requires 100 percent observer coverage on trips by purse seiners with carrying capacities greater than 400 short tons (362.8 metric tons) that fish for tunas in the eastern tropical Pacific Ocean (see Tables II-4 and II-5 for the number of observed sets in this fishery). These large purse seine vessels comprised approximately 90 percent of the total well volume of the surface gear operating in the ETP during 2001.

This mandate for 100 percent observer coverage of large tuna purse seine vessels operating in the ETP is carried out by the AIDCP On-Board Observer Program, made up of the IATTC's international observer program and the national observer programs of Ecuador, the European Union, Mexico, and Venezuela. In 2002, the On-Board Observer Program covered over 99 percent of all trips for which observer coverage is mandated by the AIDCP. In 2003, the observer programs of the European Union, Mexico, and Venezuela sampled approximately half, and that of Ecuador approximately one-third, of the trips by vessels of their respective fleets, while IATTC observers sampled the remainder of those trips. IATTC observers cover the balance of all trips by vessels required to carry observers that are registered in other nations.

Observers keep counts of stock-specific dolphin mortalities and serious injuries that occur during fishing operations and provide the data to the IATTC. Data are also recorded on herds of dolphins sighted, which may be used to estimated relative dolphin abundance. In addition, since 1992, observers have recorded data on fish and other animals released or discarded from purse seine operations at sea.

Table II-4. Number of observed sets in the international (U.S. and foreign) tuna purse seine fishery operating in the ETP (Source: Inter-American Tropical Tuna Commission 2002)

Year	Dolphin	Floating object	Unassociated school	Total
1997	8,977	5,653	4,693	**19,323**
1998	10,645	5,481	4,631	**20,757**
1999	8,648	4,620	6,143	**19,411**
2000	9,235	3,916	5,482	**18,633**
2001	9,577	5,659	2,973	**18,209**

Table II-5. Number of observed sets in the domestic (U.S. only) tuna purse seine fishery operating in the ETP (Source: Pers. Comm., January 14, 2004, Inter-American Tropical Tuna Commission)

Year	Dolphin[1]	Floating object	Unassociated school	Total
1997	0	600	232	**832**
1998	0	290	221	**511**
1999	0	263	115	**378**
2000	0	255	193	**448**
2001	0	262	131	**393**

[1] No intentional sets on dolphins or other marine mammals were made.

II-B. Protected Resources Division Action - Proposed Rule to Prohibit Shallow Longline Sets East of 150EW longitude.

NOAA Fisheries, Protected Resources Division, Southwest Region proposes to use Secretarial authority under 11(f) of the ESA (as well as under the HSFCA, 16 U.S.C. 5503(d)) to promulgate regulations in the West Coast-based longline fishery (as described above in section II-A) to ensure the fishery complies with the ESA. Specifically, Protected Resources proposes (68 FR 70219, December 17, 2003) to implement Alternative 3 in the Pacific Council's FMP Environmental Impact Statement and Regulatory Impact Review/Initial Regulatory Flexibility Analysis. This would prohibit West Coast-based longline vessels from making shallow longline sets on the high seas in the Pacific Ocean east of

150E West longitude to conserve leatherback and loggerhead sea turtles. This proposed action is in addition to the sea turtle conservation measures under the proposed rule to implement the FMP which requires vessels to have on board and to use dip nets, line cutters, and wire or bolt cutters capable of cutting through the vessel's hooks to release sea turtles with the least harm possible to the sea turtles and the prohibition of shallow longline sets west of 150E West longitude (68 FR 68834, December 10, 2003). This rule would become effective in March 2004 at the same time as the regulations implementing the FMP.

According to observer data collected on longline vessels fishing shallow sets east of the 150E West longitude, overall take rates of sea turtles in the West Coast-based longline fishery appear to be similar to those in the western Pacific. If this is the case, there could be large numbers of sea turtle takes under the Pacific Council's preferred alternative of allowing shallow sets east of the 150E West longitude. Based on an estimate that the West coast-based longline vessels set approximately 1,000,000 hooks per year and using observed capture rates (Carretta 2003; J. Carretta, NOAA Fisheries, personal communication, January 2004), this fishery is expected to capture between 16 and 51 leatherback turtles, between 81 and 144 loggerhead turtles, and up to 11 olive ridley turtles annually. Of these captured turtles, 2 to 23 leatherback turtles, 23-67 loggerhead turtles, and up to 2 olive ridley turtles would be killed using mortality rates in the January 22, 2004 draft "*Criteria for Estimating Post-Hooking Mortality in Marine Turtles as a Result of Interactions with Longline Gear for Purposes of Conducting the Februrary 2004 Section 7 Consultations on Pelagic Fisheries in the Pacific.*" memorandum from Laurie Allen, NOAA Fisheries Office of Protected Resources.

The estimated takes and mortalities of leatherback and loggerhead sea turtles by the West Coast-based longline fishery under the FMP exceed those that are already exempted under the Endangered Species Act (ESA) in other Pacific fisheries. As an example, the incidental take statement in the 2002 biological opinion on the western Pacific pelagic fisheries authorized mortalities in the Hawaii-based longline fishery of only 3 leatherback and 8 loggerhead turtles per year. Based on the severe decline and lack of recovery in leatherback and loggerhead sea turtle populations and the estimated mortality of leatherback and loggerhead sea turtles, NOAA Fisheries is concerned that the take levels expected to occur in the West coast-based longline fishery under the Pacific Council's FMP might be likely to jeopardize the continued existence of these species. As a result, Protected Resources Division proposes to issue regulations intended to avoid this possibility as a possible companion measure to the FMP regulations.

II-C. Description of the Action Areas

The action area is defined as all areas affected directly or indirectly by the Federal action and not merely the immediate area involved in the action (50 CFR 402.02). The following fisheries operate within the areas described below (in some instances, maps are provided as attachments) and their effects occur primarily within these areas. Some effects from fishing effort, such as oil and gas leaks, trash and gear discards, or vessel noise impacts may occur outside of these areas but NOAA Fisheries cannot determine the geographic extent of these effects or if they do in fact travel beyond the general fishing area boundaries described below.

A. Albacore Hook and Line Fishery

North Pacific albacore troll effort occurs in areas northwest of Midway Island and eastward, fishing near 45E N latitude, 150E W longitude and along the West Coast of North America from Vancouver Island to southern California. Fishing gear is found primarily at the surface or very shallow depths.

B. West Coast-based Longline Fishery

Longliners based out of the west coast of the US fish outside of the EEZ (it is illegal for them to fish within the EEZ off California and Washington). Principal fishing area is east of 150EW longitude, although trips north of Hawaii have occurred (Figure 1). Under Protected Resource Division's proposed action, the area east of 150EW longitude would be closed to shallow-set longline gear. Vessels may switch to deep-set longline gear, however the action area is expected to stay approximately the same. Shallow-set longline gear typically fishes at depths less than 100 meters. Deep set longlining gear typically fishes at depths greater than 100 meters.

C. CA/OR Drift Gillnet Fishery

Fishing effort for swordfish by the CA/OR drift gillnet fishery primarily occurs in waters off San Diego, north to San Francisco, and within 300 miles of shore. Small numbers of swordfish are also caught between San Francisco and the California-Oregon border and within 125 miles of shore, and very few swordfish catches are made north of Oregon. Fishing effort for swordfish usually peaks in October and November and tapers off in December and January (Holts and Sosa-Nishizaki, 1998). Thresher shark are mainly targeted within 9 miles (8 nm) of the coast or near the Channel Islands, where mean water depth is approximately 400 fathoms. Thus the action area, for the purposes of this Opinion, is the body of water delineated by the California-Mexico border to the south (30EN latitude), the Oregon-Washington border to the north (45EN), extending as far west as 129EW (Julian and Beeson, 1998; Figure 2) excluding those areas that are closed to protect sea turtles between August 15 and November 15 and closed in June, July, and August during a forecasted or occurring El Nino event. Drift gillnet gear fishes at depths below 36 feet. Nets may be between 90 and 170 feet "deep."

D. ETP Purse Seine Fishery and Coastal Pelagic Purse Seine Fishery

The ETP is described in section II.B.1.d above. Large U.S. vessels, which do not currently set on dolphin, primarily fish beyond the EEZ though one or two vessels occasionally fish in the EEZ in the area between San Diego, California and 20ES and from the Central American coast out to 150EW or 160E W (see Figure 3). Smaller U.S. purse seine vessels fish primarily within the EEZ off southern

California when tunas are available. Purse seine gear fishes at depths between the surface and 300 to 700 feet deep.

E. Swordfish and Shark Harpoon Fishery

Harpoon fishing typically concentrates in the Southern California Bight (SCB) off San Diego early in the season and shifts to areas as far north as Oregon later in the season, especially in El Niño years. Swordfish are usually sighted basking at the surface of the water in temperatures between 12E to 26EC. In El Niño years, the range of water temperatures where the majority of swordfish sightings occur narrows and favors warmer temperatures between 20E and 22EC. Harpoon is legal gear in California and Oregon, but is not defined as legal gear in Washington.   Harpoon gear is thrown from the fishing vessel to the target species basking on the surface of the ocean.

F. Charter Boat and Private Boat HMS Sport Fisheries.

Tropical tunas, billfish and sharks become available off the West Coast as they move seasonally eastward from oceanic waters and northward from Mexico. Except during periods of warm water, recreational catches of these species are almost exclusively from waters off southern California (Table 2-60 in the HMS FMP). Albacore and northern bluefin (more temperate water species) move into the coastal waters along the West Coast from more temperate waters offshore. California's CPFV fishing effort occurs primarily in the Southern California Bight, and south of San Clemente Island.   In addition, a significant amount of albacore fishing effort occurs off Mexico.

In Washington, the major port for charter vessels is Westport. The trip distance from shore varies from year-to-year (in 2000, the average distance was 64 nm) and charter vessels often take two-day fishing trips for albacore. The majority of effort and catch in Oregon is concentrated along the central part of the Oregon coast, though landings occur in ports coast wide. The majority of the charter effort is out of Depot Bay and Newport, with less effort out of Garibaldi and Brookings.

Many private vessel owners possess Mexican fishing licenses and travel south looking for schools of tuna and billfish. Sport fishing vessels also target tuna in southern California and northern Baja California waters. Fishing locations are primarily in the SCB from Santa Barbara, south and into Mexico. Many California anglers will fish the productive waters around Mexico's Coronado Islands for tuna, marlin, dorado and coastal pelagic species. A few private vessel owners travel as far south as Magdalena Bay and Cabo San Lucas in the fall and winter.

The majority of private vessel effort and catch in Oregon is concentrated along the central part of the Oregon coast, though landings occur in ports coast wide. The majority of private vessel effort is from Garibaldi to Newport, and Coos Bay and Brookings. Most recreational albacore fishing in Oregon occurs within 50 miles of shore with most private vessels staying much closer. Gear for these fisheries is fished at or very near the surface of the water.

III. APPROACH TO THE ASSESSMENT

Section 7(a)(2) of the Endangered Species Act of 1973, as amended (16 U.S.C. §1536), requires federal agencies to ensure that their actions are not likely to jeopardize the continued existence of any listed species or result in the destruction or adverse modification of critical habitat that has been designated for those species. Regulations that implement section 7(b)(2) of the ESA define *jeopardize the continued existence of* as engaging in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species (50 CFR 402.02). With respect to threatened and endangered species, then, federal actions are required to ensure that their actions would not be reasonably expected to appreciably reduce the species' likelihood of both surviving and recovering in the wild, by reducing the species' reproduction, numbers, or distribution.

The regulations that defined destruction or adverse modification were vacated by the Court in *Sierra Club* v *U.S. Fish and Wildlife Service and National Marine Fisheries Service* (Services; Fifth Circuit Court of Appeals; CA No. 98-3788-K-2 E.D. La). Until the Services promulgate a new regulatory definition, the Services apply the statutory definition of critical habitat: "(i) the specific areas within the geographical area occupied by the species, at the time it is listed ,..., on which are found those physical or biological features (ii) essential to the conservation of the species and (ii) which may require special management considerations or protection; and (ii) specific areas outside the geographical area occupied by the species at the time it is listed,..., upon a determination by the Secretary that such areas are essential for the conservation of the species" (16 U.S.C. 1533(5)(A)) to their determination of destruction or adverse modification. If the Services determine that a proposed action is likely to render these areas or physical and biological features unuseable or inaccessible or degrade their conditions so that the listed species can no longer rely upon them for their conservation, then the proposed action is likely to destroy or adversely modify that critical habitat.

By law, the National Marine Fisheries Service (NOAA Fisheries) issues biological opinions to help federal agencies comply with the requirements of section 7 of the Endangered Species Act. This biological opinion is designed to help the Sustainable Fisheries Division of NOAA Fisheries' Southwest Regional Office ensure that the proposed management regime of the U.S. fisheries for highly migratory species in the Pacific Ocean is not likely to jeopardize the continued existence of threatened or endangered species. Because the proposed fisheries have been determined to be not likely to adversely affect critical habitat that has been designated in the action area for this consultation (see Section IV. *Status of the Species and Environmental Baseline*), this Opinion will focus only on jeopardy analyses.

1. Method

After receiving a complete description of the proposed management regime for the fisheries from

Southwest Regional Office's Sustainable Fisheries Division, we conducted our assessment of the effects of the proposed fisheries and fishery management regime using four discrete steps:

1. Our first step of our assessment deconstructed the proposed fisheries management plans into their constituent parts (using our agency's prior experience with the fisheries and published information) to allow us to distinguish the effects of different fisheries and different fishing strategies on listed resources.

2. The second step of our assessment consisted of exposure analyses which identify the listed species and designated critical habitat that are likely to co-occur with different components of those fisheries in space and time and any important attributes of that co-occurrence that might help explain the potential risks the fisheries pose to the species.

3. The third step of our assessment consisted of response analyses which identify how listed resources are likely to respond once exposed to the Action's stressors. These analyses distinguished between turtles that are captured and released, unharmed; captured and released with injuries that prove fatal later, and sub-lethal effects. As part of these analyses, we considered new information on sea turtle mortalities following their release after having been captured by longline gear.

4. The final step of our assessment used the analyses from the previous two steps identify the number of individuals of each species that are likely to be exposed to the proposed fisheries (as well as other information like their age or life history stage) and what is likely to happen to those individuals given exposure. In the final step of our assessment we ask (1) what is likely to happen to different nesting aggregations given the exposure and responses of individual members of those aggregations and (2) what is likely to happen to the populations or species those nesting aggregations comprise (Table X provides the details of the risk analyses we have conducted for these consultations).

In this consultation, our analyses focused on four specific measures of a species' extinction risk: (a) estimated times to quasi-extinction; (b) probabilities of quasi-extinction in 25, 50, and 100-year time intervals to capture the short-term, mid-term, and long-term risks the fisheries may pose to listed resources; (c) mean times to quasi-extinction; and (d) median times to quasi-extinction. To assess the probability of regional extinction (for example, the probability of leatherback turtles becoming extinct in the Pacific Ocean), we consider a regional probability of ultimate extinction. We consider probabilities of extinction over multiple time horizons because the results of most population models have a log-normal or right-skewed distribution, species have higher short-term risks of extinction and lower long-term extinction risks. At the same time, the long-lives of species like turtles can often dampen their extinction risks over time so longer-term projections may allow us to detect the dampening influence of their long lives.

As the preceding paragraph suggests, our analyses focused on the risks of species falling below quasi-extinction thresholds rather than declining to zero. We used quasi-extinction thresholds instead of true extinction for several reasons. First, most populations or species that have become extinct since the passage of the Endangered Species Act became extinct because their populations had declined to levels were demographic stochasticity — or variation in the number of births and the number of deaths in a population — dominated their population dynamics.

By its nature, demographic stochasticity, which occurs in all populations, can be fatal to small populations because it can cause their populations to oscillate into extinction. Quasi-extinction thresholds can help prevent species from declining to levels where demographic stochasticity makes their extinction almost certain. For these reasons, recent literature on conservation biology and population modeling recommends using quasi-extinction thresholds instead of true extinction for population viability analyses (for example, see Burgman et al. 1993, Morris and Doak 2002).

In this Opinion, we respond to this challenge by projecting the effects of the proposed fisheries over 1-, 10-, and 20-year time horizons, assuming that the fisheries will continue as long as they can sustainably harvest target fish species and as long as there are economic markets for those species. From those projections, we then estimate probabilities of quasi-extinction over 25-, 50-, and 100-year time intervals to capture the short-term, mid-term, and long-term risks the fisheries may pose to threatened and endangered species. We will also consider scenarios that are based on the time it would take individuals born in the current year (2004) to enter the adult population and breed. This is consistent with approaches population biologists normally use when addressing life tables, which follow a cohort's patterns of survival and fecundity from birth to death (for age-based models) or from eggs to adults (for stage-based approaches). Since these life tables form the foundation for quantitative assessments of a population's risk of extinction or explosion such as population viability analyses (see Burgman *et al*., 1993; Caughley and Gunn, 1999; Heppell *et al*., 1999), we will use the same approach for our qualitative assessment. We apply this approach by assessing the effects of the proposed fisheries on the turtles' survival and fecundity over the time it would take the 2004 cohort of hatchlings to recruit into the adult, breeding population.

2. Relationship Between these Analyses and Jeopardy Determinations

We begin our analyses with an implicit understanding that the sea turtles considered in this Opinion are threatened with global extinction by a wide array of human activities and natural phenomena; we have outlined many of those activities in the *Status of the Species* section of this Opinion. We also recognize that some of these other human activities and natural phenomena pose a much larger and more serious threat to the survival and recovery of threatened and endangered species than the HMS fisheries. For example, many foreign fishing fleets have substantially larger, adverse effects on threatened and endangered sea turtle populations in the Pacific Ocean than U.S. fishing fleets. We recognize that we will not be able to recover threatened and endangered species without addressing the full range of human activities and natural phenomena that have caused these species to decline or could cause these species to become extinct in the foreseeable future (USFWS and NOAA Fisheries 1997). Recovering threatened and endangered sea turtles, as with other imperilled marine species, will require an international, cooperative effort that addresses the full suite of threats to those species.

Nevertheless, our task in this consultation is not to identify the various risks contributing to the endangerment of listed marine species, rank them according to their relative significance, and address them according to their ranked order. Our task in a consultation is simpler: identify the direct and indirect effects of the HMS fisheries managed under the HMS FMP to determine if the proposed management regime is likely to *contribute* to the endangerment of threatened and endangered species by appreciably reducing their likelihood of both surviving and recovering in the wild. We reach our conclusions by adding the fisheries' effects to the effects of other human activities and natural phenomena on the species' status and trend as described in the *Status* and *Environmental Baseline* section of this Opinion.

All of the affected turtle species are represented by populations within the Pacific and Atlantic Oceans. We will treat these sea turtle populations in the Pacific Ocean as distinct populations from the Atlantic Ocean populations for the purposes of this consultation although our final jeopardy determination will be made at the scale of the listing for the affected turtles rather than at the distinct population scale.. For the purposes of this analysis, we will assume that anything that places sea turtle populations in the Pacific Ocean at greater risk of extinction, also places the entire species at a greater risk of extinction; or, in other words, reduces the species' likelihood of survival and recovery. This assumption is reasonable based on the relationship between local and regional persistence in species (see Gotelli, 2001 for a simple explanation of this relationship). Based on this relationship, the risk of regional extinction is lower than the risk of local extinction; however, as local probabilities change, the probability of regional persistence changes correspondingly.

For this assessment, we consider several scenarios that represent various assumptions about which nesting aggregations of the different species of sea turtles are likely to be exposed to the proposed fisheries and their responses upon exposure. We use these scenarios as the starting point of our risk assessment (see Table 2). Using those scenarios as reference points, we evaluate the evidence we have assembled to determination if reductions in reproduction, numbers, or distribution of threatened or endangered species, if there are any, would reasonably be expected to reduce a species' likelihood of surviving and recovering in the wild.. Our conclusions about whether the proposed fisheries are or are not likely to jeopardize the continued existence of listed species relies on the strength of the assembled evidence using our general understanding of population dynamics and the processes by which other populations and species have already become extinct.

3. Evidence Available for the Assessment

Detailed background information on the status of these species and critical habitat has been published in a number of documents including recent status reviews of sea turtles (NOAA Fisheries and USFWS, 1995; USFWS, 1997); recovery plans for the eastern Pacific green turtle (NOAA Fisheries and USFWS, 1998a), U.S. Pacific populations of hawksbill sea turtles (NOAA Fisheries and USFWS, 1998b), loggerhead sea turtle (NOAA Fisheries and USFWS, 1991), leatherback sea turtle (NOAA Fisheries and USFWS, 1992), and U.S. Pacific populations of olive-ridley sea turtles (NOAA Fisheries and USFWS, 1998c); and reports on interactions between sea turtles and gear used in pelagic fisheries (Bolten *et al*., 1996). In addition, Crouse *et al*. (1987), Crowder *et al*. (1994), Heppell (1998), Heppell *et al*. (1996, 1999, and 2000) published results from population models, sensitivity analyses, and elasticity analyses for various species of marine turtles, although most models are based on data on loggerhead sea turtles in the Atlantic Ocean.

In the past two years, significant new information on the biology and ecology has become available and has begun to answer some of the major questions that remain about the biology and ecology of sea turtles. For example, combining the information provided by Kamezaki et al. (2003) on the structure, status, and trends of the loggerhead sea turtle nesting aggregation in Japan with earlier work on the geography of a rare haplotype in Japanese loggerhead sea turtles and the incidence of that haplotype in the area fished by the Hawaii-based longline fisheries allows us to distinguish between the nesting aggregations on Yakushima Island and other nesting aggregations in Japan. Polovina et al. (2004) provides substantial new information on the migratory patterns of loggerhead and olive ridley sea turtles in the Pacific Ocean and their relationships with oceanographic phenomena like eddies and currents.

Lutz et al. (2001) edited a volume on the biology of sea turtles that integrated and synthesized substantial amounts of new information on the general ecology and biology of sea turtles generally, including their reproductive ecology, population dynamics, biogeography, and threats. Bolten and Witherington (2003) edited a volume that further integrated the state of scientific knowledge on the biology and ecology of loggerhead sea turtles, including their biology, distribution, population structure, and population dynamics

Despite the availability of this new information, our knowledge of the biogeography, migratory patterns, life history and population dynamics, and their response to environmental and other variation remains rudimentary and limits the precision of our assessments. The data available are distinct from the numerous statistical and computerized methods that are available to analyze data or develop simulations. The number of different statistical procedures available to analyze data fills entire sections of university libraries and continues to evolve. The number of different kinds of software and computerized procedures available to analyze population information or to conduct simulations is almost as extensive. Each of these analytical procedures is based on specific sets of assumptions and they all have their strengths and weaknesses; they can help make sense of data and they can provide information, but they may or may not capture the true state of nature. Although conservation biology distinguishes between qualitative and quantitative assessments, we cannot make that distinction in this consultation because truly quantitative models require large amounts of data on the survival, growth, and fecundity of the different life stages of species and populations and the effects of environmental variability on these parameters (Feiberg and Ellner 2000, Groom and Pascual 1998). Without robust, long-term data, “quantitative” models can lead to highly-biased estimates of the extinction risks facing populations and species (Beissinger and Westphal 1998, Feiberg and Ellner 2000, Heppell et al. 2003, Ludwig 1996, Ludwig 1998, Taylor 1995).

4. Simulations Using the Dennis Model

To help assess the status of the various species of sea turtles, we evaluated census data for different nesting aggregations, when those data were available, using the density-independent form of the Dennis model (Dennis et al. 1991, Morris and Doak 2002). This model uses a diffusion estimation equation to estimate demographic variables for a population and probable population trends. We chose the Dennis model because the available data allows us to meet most, if not all, of the model’s data requirements, while the data required to conduct more complex models (for example, population matrices) are not available for all but a few species of sea turtles or nesting aggregations (for example, stage- or age-specific survival rates, growth rates, and any variance associated with these parameters).

Truly quantitative models require large amounts of data on the survival, growth, and fecundity of the different life stages of species and populations and the effects of environmental variability on these parameters (Feiberg and Ellner 2000, Groom and Pascual 1998). Without robust, long-term data, “quantitative” models can lead to highly-biased estimates of the extinction risks facing populations and species (Beissinger and Westphal 1998, Feiberg and Ellner 2000, Heppell et al. 2003, Ludwig 1996, Ludwig 1998, Taylor 1995). With the exception of long-term datasets for loggerhead sea turtles in Australia, the kind of information these models require is not available for this consultation and are not likely to become available in the near future.

The Dennis model, however, uses time series of census counts to estimate several demographic variables that provide important insights into a population’s (or subpopulation’s) status and future trend. Despite its simplicity, this model allows us to make full use of the data in hand: time series of census counts of the number of nests or nesting females of different species. When the only data available were estimates of the number of nests, we converted those estimates into estimates of the number of adult females in a particular nesting aggregation (which we treat as a equivalent to a subpopulation) using published conversion methods. When the only data available were estimates of the number of females that nested in a particular year, we converted those estimates into estimates of the number of adult females in a particular nesting aggregation using published estimates of remigration intervals for the different species.

The equation is represented as:

$$G(T|d,m,s^{2}) = j\left(\frac{-d-mT}{\sqrt{s^{2}T}}\right) + \exp\left(-2md/s^{2}\right)j\left(\frac{-d+mT}{\sqrt{s^{2}T}}\right)$$

Where G(T|d, μ, ó^2) = the cumulative probability of reaching the quasi-extinction threshold at time T.

ö = is the standard normal cumulative distribution function (produced by the NORMDIST function in Excel)

d = log N_c - Log N_x or the difference between the log of the current population size (N_c) and the log of the quasi-extinction threshold (N_x)

µ = the mean of the log population growth rate

ó^2 = the variance of the log population growth rate

Table III-1 lists some of the products of our analyses, which are described in narrative form below. Anyone interested in more detailed discussion of this method, the interpretation of model results, and the application of this method to endangered species should refer to Dennis et al. (1991) and Morris and Doak (2002).

Table III-1. Results of analyses using the discrete-time, density-independent diffusion estimation model described by Dennis et al. (1991)

Demographic Parameter
Mean log growth rate (µ)
Upper 95% confidence interval
Lower 95% confidence interval
Variance in mean log growth rate (ó^2)
Upper 95% confidence interval
Lower 95% confidence interval
Finite rate of population increase (ë)
Upper 95% confidence interval
Lower 95% confidence interval

A population's mean log growth rate, which is equal to the natural logarithm of the population's geometric mean growth rate, is measure of the population's stochastic growth over time (Dennis et al. 1991, Lande and Orzack 1988, Morris and Doak 2002). If someone forecast a population's stochastic growth over time, some trajectories would increase, some would remain somewhat stable, while others would decrease. The mean log growth rate is a measure of the population's "average" growth rate assuming that some trajectories will increase, some will remain stable, and others will decrease (here, "average" is a geometric mean rather than an arithmetic mean because forecasts of population growth multiply a starting value by a rate; averages of multiplicative processes are best represented by geometric means). If a population's mean log growth rate, $µ > 0$, then most population trajectories will increase; if $µ < 0$, then most population trajectories will decline (Morris and Doak 2002).

The variance in a population's mean log growth rate (ó^2) captures the rate at which the variance around the distribution of the population's growth rate changes over time (Lande and Orzack 1988, Morris and Doak 2002). This parameter is important because even populations that are growing have some risk of falling to low levels or becoming extinct simply because of variation in growth rates. As a population's growth rate varies from year to year as a result of environmental variation, the population's variance will increase accompanied by an increase in the range of population sizes in the future.

A population's finite rate of increase (ë) captures a population's growth rate or the amount by which a population size multiplies from year to year. In the face of stable environmental conditions, this growth rate would be constant and a population would increase geometrically (ë > 1), decrease geometrically (ë < 1), or remain the same (ë = 1). However, in changing environments, a population's birth and death rates will vary and the population's growth rate will vary as well. Where the appropriate census data were available, we used the Dennis model to assess the status of the different species of sea turtles and report the results of our analyses in the narratives for the species located in the Status of the Species section of this Opinion.

To assess the consequences of mortalities associated with the HMS fisheries on listed sea turtles we applied the procedures prescribed by Dennis et al. (1991) and Morris and Doak (2002) to these estimates to approximate several demographic variables for different nesting aggregations of leatherback and loggerhead sea turtles: the population's mean log growth rate, the variance in the population's mean log growth rate, continuous rate of increase (r), and finite rate of population increase (lambda). Using these parameters, we calculated the risks of population quasi-extinction (probability of threshold), mean time to threshold, median time to threshold, and modal time to threshold. In all instances, we used a quasi-extinction threshold of 50 adult females. We also applied the procedures Dennis et al. (1991) and Morris and Doak (2002) to estimate the upper and lower 95% confidence intervals for all of these parameters.

Although these assessments provide numerical estimates of different demographic variables, it is important to note that simple models like the Dennis model produce qualitative rather than quantitative predictions. Despite this requirement, these estimates provide important insights into the probable status and future trend of the different sea turtle species.

IV. STATUS OF THE SPECIES AND ENVIRONMENTAL BASELINE

The following endangered and threatened species occur in the action area, as defined above, and may be affected by the proposed action:

Marine Mammals	**Status**
Blue whale (*Balaenoptera musculus*)	Endangered
Fin whale (*Balaenoptera physalus*)	Endangered
Guadalupe fur seal (*Arctocephalus townsendii*)	Threatened

Humpback whale (*Megaptera novaeangliae*)	Endangered
Pacific Right whale (*Eubalaena japonica*)	Endangered
Sei whale (*Balaenoptera borealis*)	Endangered
Sperm whale (*Physeter macrocephalus*)	Endangered
Steller sea lion - eastern population (*Eumetopias jubatus*)	Threatened

Sea turtles

Green turtle (*Chelonia mydas*)	Endangered/Threatened
Hawksbill turtle (*Eretmochelys imbricata*)	Endangered
Leatherback turtle (*Dermochelys coriacea*)	Endangered
Loggerhead turtle (*Caretta caretta*)	Threatened
Olive ridley turtle (*Lepidochelys olivacea*)	Endangered/Threatened

Salmonids

Chinook salmon - Puget Sound (*Oncorhynchus tshawytscha*)	Threatened
Chinook salmon - Lower Columbia River	Threatened
Chinook salmon - Upper Columbia River spring	Endangered
Chinook salmon - Upper Willamette River	Threatened
Chinook salmon - Central Valley spring	Threatened
Chinook salmon - Sacramento River winter	Endangered
Chinook salmon - Snake River spring/summer	Threatened
Chinook salmon - Snake River fall	Threatened
Chum salmon - Columbia River (*O. keta*)	Threatened
Chum salmon - Hood Canal summer	Threatened
Coho salmon - Central California Coast (*O. kisutch*)	Threatened
Coho salmon - Oregon Coast	Threatened
Coho salmon - Southern Oregon/Northern Coastal California	Threatened
Sockeye salmon - Ozette Lake (*O. nerka*)	Endangered
Sockeye salmon - Snake River	Endangered
Steelhead - Upper Columbia River (*O. mykiss*)	Endangered
Steelhead - Middle Columbia River	Threatened
Steelhead - Lower Columbia River	Threatened
Steelhead - Upper Willamette River	Threatened
Steelhead - Snake River Basin	Threatened
Steelhead - Northern California	Threatened
Steelhead - California Central Valley	Threatened
Steelhead - Central California Coastal	Threatened
Steelhead - South Central California	Threatened
Steelhead - Southern California	Endangered

Of the fisheries proposed to be managed under the HMS FMP, only the California/Oregon drift gillnet fishery has been observed or is known to interact with listed marine mammals. Observers have recorded the incidental take of three species of large whales: fin whale, humpback whale, and sperm whale. From 1990 to the present, observers recorded the entanglement and mortality of one fin whale, in 1999, off southern California. During the same period, two humpback whales were observed taken: one in 1994 (prior to the implementation of the PCTRP) and one in 1999. Both were released alive and uninjured. Prior to the implementation of the PCTRP (October 30, 1997), the CA/OR drift gillnet fishery was observed to take seven sperm whales (3 dead, 3 alive and uninjured, and 1 injured). Since the implementation of the PCTRP, only one sperm whale has been observed incidentally taken, in 1998.

In 1997, one "unidentified baleen whale" was reported accidently killed in the ETP tuna purse seine fishery (IATTC, 1999). No information is available to determine whether the whale killed in 1997 was a listed species. Therefore, because both listed and non-listed baleen whales occur in the ETP (e.g. minke whale (*Balaenoptera acutorostrata*) and Bryde's whale (*B. edeni*)), it is not possible to determine whether the whale reported killed was listed under the ESA. NOAA Fisheries has no other observer reports of baleen whales accidentally killed in the ETP tuna purse seine fishery.

Based on observer reports in the Hawaii-based longline fishery, vessels using longline gear to target highly migratory species have on rare occasions interacted with humpback whales and sperm whales. These species interacted with longliners based out of Hawaii in areas of the north Pacific Ocean where longliners based out of the west coast also fish. Therefore, the possibility exists that longliners based out of the west coast could interact with these marine mammal species, although there have been no reports from observers or fishermen in their logbooks.

Steller sea lions are rarely taken in the CA/OR drift gillnet fishery. In the 13 years that NOAA Fisheries observers have been collecting data, Steller sea lions have been observed entangled and killed in two instances in the CA/OR drift gillnet fishery, one in 1992, off central California (net extenders were 20 feet), and one in 1994, off the California/Oregon border (net extenders were 30 feet). No Steller sea lions have been observed taken or reported since the implementation of the PCTRP, in October, 1997.

Although the Steller sea lion and the CA/OR drift gillnet fishery are known to co-occur in areas off the California and Oregon coast, the implementation of the PCTRP appears to have reduced the incidental take of pinnipeds. In addition, the northern closure of the fishery during the late summer and fall also protects Stellers during this time and likely reduces or eliminates overall incidental take levels. Based on all of the above, NOAA Fisheries expects the entanglement of Stellers in this fishery to be a rare event, and therefore, the proposed action is not likely to adversely affect Steller sea lions. This species will not be considered further in this Opinion.

Although blue whales, right whales, sei whales, Guadalupe fur seals and hawksbill sea turtles are found within the action area and could potentially interact with the fisheries managed under the HMS FMP, there have been no reported or observed incidental takes of these species in any of the fisheries. Therefore, the proposed action is not likely to adversely affect blue whales, northern right whales, sei whales, Guadalupe fur seals, or hawksbill sea turtles and these species will not be considered further in this Opinion.

All listed species of Pacific salmonids may occur within the action area during the pelagic stage of their life history. The harvest of salmon in commercial and recreational ocean fisheries is managed under the Pacific Salmon FMP, which has been analyzed in prior Section 7 consultations. Impacts to these species have already been assessed in previous consultations and take of listed salmonids in ocean fisheries is already covered under separate incidental take statements. There are no records indicating any instances of takes of listed salmon in any HMS fisheries. Based on gear types, location of effort, and methods, it is unlikely that vessels targeting HMS would interact with salmonids. Therefore, Pacific salmonids will not be considered further in this Opinion.

Three fisheries proposed to be managed under the HMS FMP have been observed or are reported to take listed sea turtles, including green turtles, leatherback turtles, loggerhead turtles, and olive ridley turtles. Therefore, the effects of the proposed action on these species will be analyzed in this Opinion.

The term "critical habitat" is defined in the ESA to mean: (1) the specific areas within the geographic area occupied by the species at the time it is listed in accordance with the provisions of section 4 of this Act, on which are found those physical or biological features (a) essential to the conservation of the species and (b) which may require special management consideration or protection; and (2) the specific areas outside of the geographical area occupied by the species at the time it is listed in accordance with the provisions of section 4 of this Act, upon a determination by the Secretary that such areas are essential to the conservation of the species.

Critical habitat for the fin, humpback, and sperm whale has not been designated or proposed within the action area. In addition, critical habitat for the green, hawksbill, leatherback, loggerhead, and olive ridley turtle has not been designated or proposed within the action area.

The following subsections are synopses of the current state of knowledge on the life history, distribution, and population trends of marine mammal and sea turtle species adversely affected by the action. These subsections focus primarily on the Pacific Ocean populations of these species as these are the populations directly affected by the proposed action. However, NOAA Fisheries recognizes that many of these species are listed as global populations (e.g. leatherback and loggerhead turtles and large whales), and the global status and trends of these species are included as well in order to provide a basis for our final determination of the effects of the proposed action on the species as listed under the ESA. Although, the Status of the Species and the Environmental Baseline are typically two separate sections in Biological Opinions, they are combined here because the status of the species in the action area and the factors affecting them throughout their range are virtually the same.

A. Status of Marine Mammals

Most large whales are listed as endangered species under the ESA because their populations were depleted by whalers in the nineteenth and twentieth centuries. Currently, ship strikes and incidental take in commercial fishing operations (domestic and international) are most likely the greatest threat to the recovery of large cetaceans. Under the 1994 amendments to the Marine Mammal Protection Act (MMPA), NOAA Fisheries was required to produce stock assessment reports (SAR) for all marine mammal stocks that occur in U.S. waters. These reports include information on the status and trends of marine mammals and assessments of all human-caused mortality and serious injury of the listed marine mammal stocks. Information on humpback whales, fin whales, and sperm whales was obtained from both final and draft SARs and is presented below, along with other relevant information (sources identified therein).

1. Fin Whale

a. Listing status

In the North Pacific, the International Whaling Commission first protected fin whales in 1976 (Allen, 1980). Fin whales were listed as endangered under the Conservation of Endangered Species and Other Fish or Wildlife on June 2, 1970 and later listed as endangered when the ESA was passed in 1973. Critical habitat has not been designated for this species.

b. Species description and distribution

Fin whales are widely distributed in the world's oceans and are the second largest member of the family Balaenopteridae, reaching lengths of between 20 and 29 meters at adulthood (Aguilar and Lockyer, 1987). Fin whales are dark gray dorsally and white underneath, with a long, slender body and a prominent dorsal fin about two-thirds of the way back on their body (Agler *et al.,* 1990, *in* Reeves, *et al.*, 1998). Like other baleen whales, fin whales have fringed baleen plates and ventral grooves, which expand during feeding.

Fin whales have been known to associate with steep contours, either because tidal and current mixing along such gradients drives high biological production, or because changes in depth aid their navigation. Depending on food supply, fin whale groups may exhibit seasonal migration patterns to high latitudes in summer for feeding, and to low latitudes in winter, when they may be fasting. Other groups may remain in a particular area, depending on food supply. Thus, the local distribution of fin whales during much of the year is probably governed by prey availability. Although there has been considerable discussion of interspecific competition among mysticete whales for prey, there has been no conclusive evidence to demonstrate that it occurs (Clapham and Brownell, 1996, *in* Reeves, *et al.*, 1998). At present, NOAA Fisheries recognizes three stocks: 1) Alaska (North Pacific); 2) California/Oregon, and Washington; and 3) Hawaii (Hill and DeMaster 1998).

Fin whales have a complex migratory behavior that appears to depend on their age or reproductive state as well as their "stock" affinity. Movements can be either inshore-offshore or north-south. In the North Pacific in summer, fin whales are present in the Bering and Chukchi Seas and are concentrated along the coast of Alaska, around the Aleutian Islands and off Baja California. Historical accounts from the 1960s indicate that the fin whale was formerly the most abundant large whale off California in spring and summer. Fin whales seem to be resident in the Gulf of California year round, with peak numbers in the summer and fall. Peak numbers of fin whales have also been seen during the summer off Oregon and in summer and fall in the Gulf of Alaska and southeastern Bering Sea (*in* Perry, *et al*, 1999). Rice (1974) reported that several fin whales tagged from November to January off southern California were later killed by whalers in May to July off central California, Oregon, and British Columbia and in the Gulf of Alaska, suggesting possible southern California wintering areas and summering areas further north. Although fin whale abundance is lower in winter/spring off California, and higher in the Gulf of California, further research and surveys need to be conducted in order to determine whether fin whales found off southern and central California migrate to the Gulf of California for the winter (Forney, *et al.*, 2000). In the west, fin whales may be seen from the coast of Japan, north to the Sea of Okhotsk in summer. In winter, they are distributed southward to the Sea of Japan, Yellow Sea and Philippine Sea.

In the North Atlantic, fin whales spend the summer in a broad region between North America and the Arctic, around Greenland, Iceland, Northern Norway, Spitsbergen and the Barents Sea. In the winter, they are distributed between the ice edge, to the Caribbean, Gulf of Mexico and British Isles, to the Bay of Biscay and Mediterranean. Some fin whales are present in the Mediterranean year round.

c. *Life history information*

Fin whales calve and mate in winter (November to March, with a peak in December and January), mostly in temperate waters. The gestation period of fin whales is probably somewhat less than a year, and calves are nursed for 6-7 months. The average calving interval has been estimated at about two years. Fin whales in populations near carrying capacity may not attain sexual maturity until ten years of age or older, whereas those in exploited populations may mature as early as six or seven years of age. Ohsumi (1986) analyzed age at sexual maturity for a large sample of fin whales killed in the eastern North Pacific from the mid-1950s to 1975, and found that age at sexual maturity declined markedly with time, from 12 to 6 years in females and from 11 to 4 years in males, interpreted as a density-dependent response to heavy exploitation of the stock during much of the twentieth century. Fin whales reach their maximum size at 20-30 years of age (Aguilar and Lockyer, 1987, *in* Reeves, *et al.*, 1998). The life span of a fin whale may be 85 to 90 years. The largest fin whales reported in the catch off California (during the whaling era) were a 24.7 meter (81 feet) female and a 22.9 meter (75 feet) male (Clapham, 1997, *in* Reeves, *et al.*, 1998). Shark and killer whale attacks are presumed to occur on fin whales, although no such events have been documented (Reeves, *et al.*, 1998).

Fin whales feed on planktonic crustacea, some fish, and cephalopods. The diet varies between areas and seasons. Herring, capelin and other shoaling fish are eaten in both the North Atlantic and North Pacific along with squid, euphausiids and copepods. In addition to euphausiids and copepods in the North Pacific Ocean, fin whales also feed on schooling fish such as herring, walleye pollock, and capelin (Reeves, *et al.*, 1998). In the Southern Hemisphere, euphausiids (*Euphausia superba*) are the major prey item, however other species of euphausiids and amphipods (such as *Parathemisto gaudichaudi*) are eaten in lower latitudes and when seasonally abundant. Sergent (1977, *in* Reeves, et al, 1998) suggested that euphasiids were the basic food of fin whales, but that they took advantage of fish when sufficiently concentrated, particularly in the pre-spawning, spawning, and post-spawning adult stages on the continental shelf and in coastal waters.

Fin whales usually feed using the swallowing technique. They often feed on their sides at the surface, scooping up prey and water in their expanded buccal cavity. The amount of food consumed by fin whales per day has been calculated as 1- 1.5 tons in the North Pacific, 2.8 tons in the Antarctic, and 0.533 tons off the northeast coast of the US.

d. Diving and social behavior

Fin whales dive to depths of at least 230 meters. They are reportedly one of the fastest of the big whales, possibly reaching burst speeds in excess of 32 km per hour (Leatherwood and Reeves, 1983). When they are moving leisurely at the surface, fin whales expose the dorsal fin shortly after the appearance of the blowholes. When they are surfacing from a deeper dive, however, they emerge at a steeper angle, blow, submerge the blowholes, and then arch the back and dorsal fin high above the surface. Duration of dives ranges from 25 seconds to 15 minutes and mean blow intervals were found to be about 50 seconds for fin whales feeding at the surface.

Fin whales may be found alone, or in pairs, but often form larger pods of 3 to 20 animals, which may be part of a wider group of hundreds of individuals spread over a broad area, especially on feeding grounds. Although they are more commonly seen in small groups than other rorqual whales (which are often solitary), little is known of fin whale herd behavior or group composition during the reproductive season in the winter. Differences in group size may result from the presence of different prey types in different areas, although geographical segregation by sex or age -class might also influence group size.

Observations of fin whales off Newfoundland and Labrador revealed that the bonds between pairs and groups of fin whale were variable over periods of hours, indicating that long-term pair bonds, presumed by earlier scientists, are probably not common. Fin whales probably associate with many different individuals, forming fluid associations in feeding areas. Apparent sexual behavior has been observed on occasions, including excited chases at the surface. Lunging activity, when several whales are present, has been interpreted as antagonistic behavior in some cases, although it may also represent feeding behavior.

Photo-identification studies have indicated that spatial segregation of sexes and age classes may occur in the Gulf of Maine. During migration, fin whales (as with other species of baleen whale) are segregated by sex as well as age: males migrate first and pregnant females migrate in advance of other sexual classes, with immature whales at the rear.

e. Population status and trends

Prior to whaling, the total north Pacific fin whale population was estimated to be between 42,000 and 45,000, based on catch data and a population model (Ohsumi and Wada, 1974, *in* Perry, *et al.*, 1999). In 1973, the North Pacific population was estimated to have been reduced to 13,620-18,680 (Ohsumi and Wada, 1974), of which 8,520-10,970 were estimated to belong to the eastern Pacific stock. Of this, the component of the population centered in waters east of 180E longitude was estimated to be 25,000-27,000. However, there is still insufficient information to accurately determine population structure in fin whales in the North Pacific (Carretta *et al.* 2003). Since fin whale abundance appears lower in winter/spring in California (Dohl *et al.* 1983; Forney *et al.*, 1995) and in Oregon (Green *et al.*, 1992), it is likely that the distribution of the California/Oregon/Washington stock extends seasonally outside coastal waters. Recent genetic studies of fin whales have shown that the population in the Gulf of California is isolated from fin whales in the rest of the eastern North Pacific and is an evolutionary unique population (Bérubé *et al.*, 2002).

Based on shipboard surveys conducted in summer/autumn of 1996 (Barlow and Taylor, 2001) and 2001 (Barlow 2003), 3,279 (CV=0.31) fin whales, with a minimum of 2,541 whales, were estimated off California, Oregon, and Washington. This is probably a slight underestimate because it almost certainly excludes some fin whales which could not be identified in the field.

An increasing trend in the abundance of fin whales between 1979-80 and 1993 is suggested by the available survey data, but it is not statistically significant (Barlow, 1997). No data are available on the estimated abundance of the Hawaiian stock or the Northeast Pacific (Alaska) stock of fin whales (Forney, *et al*., 2000; Hill and DeMaster, 1999). However, results of surveys in 1999 and 2000 in the central-eastern Bering Sea and southeastern Bering Sea provided provisional estimates of 3,368 (CV=0.29) and 683 (CV=0.32), respectively. These estimates are considered provisional because they have not been estimated for animals missed in the trackline, animals submerged when the ship passed, and responsive movement (Angliss and Lodge, 2003). Only one fin whale was seen on vessel cruises in the eastern tropical Pacific Ocean from 1986 through 1990; therefore, no abundance estimates were available for this region (Wade and Gerrodette, 1993).

For the western North Atlantic stock, the minimum population estimate is 2,362 fin whales, with the best estimate of abundance of 2,814 (CV=0.21) whales. There is insufficient data to determine population trends for this species. Current and maximum productivity rates are unknown for this stock. Based on photographically identified whales, Agler *et al.* (1993) estimated that the gross annual reproductive rate was at 8% with a mean calving interval of 2.7 years. The maximum productivity rate was assumed to be 0.04.

2. Humpback Whale

a. Listing status

The International Whaling Commission first protected humpback whales in the North Pacific in 1965. Humpback whales were listed as endangered under the Conservation of Endangered Species and Other Fish or Wildlife on June 2, 1970 and later listed as endangered when the ESA was passed in 1973. They are also protected by the Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES) and the MMPA. Critical habitat has not been designated for this species.

b. Species description and distribution

The humpback whale is of medium size relative to other large whales, with females and males reaching an average length of around 14 meters and 13 meters, respectively (Nitta and Naughton, 1989) and a weight of about 34 metric tons at maturity (Johnson and Wolman, 1984 *in* Perry *et al.*, 1999). They are characterized by wing-like pectoral flippers that are from one-fourth to one-third of their total body length and their heads are covered in tubercles, and tail flukes with individually identifiable trailing-edge patterns. Like other balaenopterids, they have fringed baleen plates, which allow for the filtering of small crustaceans and fish.

The humpback whale is distributed worldwide in all ocean basins. They typically migrate between tropical/sub-tropical and temperate/polar latitudes, occupying tropical areas during winter months when they are breeding and calving, and polar areas during the spring, summer, and fall, when they are feeding. Humpbacks primarily feed on small schooling fish and krill (Caldwell and Caldwell, 1983). It is believed that minimal feeding occurs in wintering grounds, such as the Hawaiian Islands (Balcomb, 1987; Salden, 1987). Humpback whales summer throughout the central and western portions of the Gulf of Alaska, including Prince William Sound, around Kodiak Island (including Shelikof Strait and the Barren Islands), and along the southern coastline of the Alaska Peninsula. The few sightings of humpback whales in offshore waters of the central Gulf of Alaska are usually attributed to animals migrating into coastal waters (Morris *et al.*, 1983), although it has been suggested that they may use offshore banks for feeding. The continental shelf of the Aleutian Islands and Alaska Peninsula was once considered the center of the North Pacific humpback whale population (Berzin and Rovnin, 1966; Nishiwaki, 1966). The northern Bering Sea, Bering Strait, and the southern Chukchi Sea along the Chukchi Peninsula appear to form the northern extreme of the humpback whale's range (Nikulin, 1946, Berzin and Rovnin, 1966). NOAA Fisheries recognizes three stocks of humpback whales in the North Pacific basin, based on genetic and photo-identification studies: an Eastern North Pacific stock, a Central North Pacific stock and a Western Pacific stock (Hill and DeMaster, 1998).

Humpback whales are often found along or in groups of two or three, but throughout their breeding and feeding ranges, they may congregate in groups of up to fifteen animals. In general, they are distributed over shallow banks and in shelf waters (Leatherwood and Reeves, 1983). Cow and calf pairs appear to prefer very shallow water less than 18 meters deep (10 fm [60 ft]) (Glockner and Venus, 1983). Cows with calves may select shallow nearshore water, at least partially to minimize encounters with courting adults. Some results suggest that habitat use patterns of nearshore waters by females and calves near the island of Maui in Hawaii may have changed (decreased), potentially due to increasing vessel and other human activities (Salden, 1988; Glockner-Ferrari and Ferrari, 1990).

In the Atlantic Ocean, humpback whales feed in the northwestern Atlantic during the summer months and migrate to calving and mating areas in the Caribbean. During the winter, humpbacks mate and calve primarily in the West Indies, where spatial and genetic mixing among subpopulations occurs (Katona and Beard, 1990; Clapham *et al.*, 1993; Palsboll *et al.*, 1997; Stevick *et al.*, 1998). A few whales of unknown origin migrate to the Cape Verde Islands (Reiner *et al.*, 1996). In the West Indies, the majority of whales are found in the waters of the Dominican Republic, notably on Silver Bank, on Navidad Bank, and in Samana Bay (Balcomb and Nichols 1982; Whitehead and Moore 1982; Mattila *et al.* 1989, 1994). Humpback whales are also found at much lower densities throughout the remainder of the Antillean arc, from Puerto Rico to the coast of Venezuela (Winn *et al.*, 1975; Levenson and Leapley, 1978; Price, 1985; Mattila and Clapham, 1989).

Feeding is the principle activity of humpbacks in New England waters, and their distribution in this region has been largely correlated to prey species and abundance, although behavior and bottom topography are factors in foraging strategy (Payne *et al.*, 1986, 1990). Humpback whales also use the

Mid-Atlantic as a migratory pathway and apparently as a feeding area, at least for juveniles. Since 1989, observations of juvenile humpbacks in that area have been increasing during the winter months, peaking January through March (Swingle *et al.*, 1993). Biologists theorize that non-reproductive animals may be establishing a winter feeding range in the Mid-Atlantic since they are not participating in reproductive behavior in the Caribbean.

c. Life history information

Humpback whale reproductive activities occur primarily in winter. They become sexually mature at age four to six. Annual pregnancy rates have been estimated at about 0.40–0.42 (NOAA Fisheries, unpublished; Nishiwaki, 1959), and female humpback whales are believed to become pregnant every two to three years. Gestation averages around 12 months, and lactation lasts nearly a year. Cows will nurse their calves for up to 12 months. The age distribution of the humpback whale population is unknown, but the proportion of calves in various populations has been estimated at about 4–12% (Chittleborough, 1965; Herman *et al.*, 1980; Whitehead, 1982, Bauer, 1986; and Clapham and Mayo, 1987). The information available does not identify natural causes of death among humpback whales or their number and frequency over time, but potential causes of natural mortality are believed to include parasites, disease, predation (killer whales, false killer whales, and sharks), biotoxins, and entrapment in ice.

Humpback whales exhibit a wide range of foraging behaviors, and feed on a range of prey types including small schooling fishes, euphausiids, and other large zooplankton. Fish prey in the North Pacific include herring, anchovy, capelin, pollock, Atka mackerel, eulachon, sand lance, pollock, Pacific cod, saffron cod, arctic cod, juvenile salmon, and rockfish. In the waters west of the Attu Islands and south of Amchitka Island, Atka mackerel were preferred prey of humpback whales (Nemoto, 1957). Invertebrate prey include euphausiids, mysids, amphipods, shrimps, and copepods. Humpback whales use a wide variety of fishing techniques, at times involving more than one individual and resembling a form of cooperative participation. The two most observable techniques are lob-tail feeding (Weinrich *et al.*, 1992) and bubble-cloud feeding (Ingebrigtsen, 1929; Jurasz and Jurasz, 1979; Hain *et al.*, 1982). Recently, there has also been documentation of bottom-feeding by humpback whales in the Atlantic (Swingle *et al.*, 1993); however, it is currently unknown whether humpback whales in the Pacific feed in this manner.

d. Diving and social behavior

Maximum diving depths for humpbacks are approximately 150 meters (492 ft) (but usually <60 meters [197 ft]), with a very deep dive (240 meters [787 ft]) recorded off Bermuda (Hamilton *et al.*, 1997). They may remain submerged for up to 21 min (Dolphin, 1987). Dives on feeding grounds ranged from 2.1 - 5.1 min in the north Atlantic (Goodyear, unpublished manuscript). A study of dive behaviors of humpback whales in Alaska found that 66 percent of the dives were made to depths of between 0 and 20 meters (~65 feet), while only 15 percent of the dives extended beyond 60 meters (Dolphin, 1986).

In southeast Alaska average dive times were 2.8 min for feeding whales, 3.0 min for non feeding whales, and 4.3 min for resting whales (Dolphin, 1987). In the Gulf of California humpback whale dive times averaged 3.5 min (Strong, 1990). Because most humpback prey is likely found in waters shallower than 300 m most humpback dives are probably relatively shallow.

Clapham (1996) reviewed the social behavior of humpback whales. They form small unstable groups during the breeding season. During the feeding season they form small groups that occasionally aggregate on concentrations of food. Feeding groups are sometimes stable for long periods of times. There is good evidence of some territoriality on feeding grounds (Clapham, 1994; 1996), and on wintering ground (Tyack, 1981). On the breeding grounds, males sing long complex songs directed towards females, other males, or both. The breeding season can best be described as a floating lek or male dominance polygyny (Clapham, 1996). Intermale competition for proximity to females can be intense as expected by the sex ratio on the breeding grounds, which may be as high as 2.4:1.

Humpbacks produce a wide variety of sounds. During the breeding season males sing long, complex songs, with frequencies in the 25-5000 Hz range and intensities as high as 181 dB (Payne, 1970; Winn *et al.*, 1970a; Thompson *et al.*, 1986). Source levels average 155 dB and range from 144 to 174 dB (Thompson *et al.*, 1979). The songs appear to have an effective range of approximately 6 to 12 mi (10 to 20 km). Animals in mating groups produce a variety of sounds (Tyack, 1981; Tyack and Whitehead, 1983, Silber, 1986). Sounds are produced less frequently on the summer feeding grounds. Feeding groups produce distinctive sounds ranging from 20 Hz to 2 kHz, with median durations of 0.2-0.8 sec and source levels of 175-192 dB (Thompson *et al.*, 1986). These sounds are attractive and appear to rally animals to the feeding activity (D'Vincent *et al.*, 1985; Sharpe and Dill, 1997). In summary, humpback whales produce at least three kinds of sounds: (1) complex songs with components ranging from at least 20 Hz to 4 kHz with estimated source levels of 144 - 174 dB; these are mostly sung by males on the breeding grounds (Payne, 1970; Winn *et al.*, 1970a; Richardson *et al.*, 1995); (2) social sounds in the breeding areas that extend from 50 Hz to more than 10 kHz with most energy below 3 kHz (Tyack and Whitehead, 1983, Richardson *et al.*, 1995); (3) feeding area vocalizations that are less frequent, but tend to be 20 Hz - 2 kHz with estimated sources levels in excess of 175 dB (Thompson *et al.*, 1986; Richardson *et al.*, 1995). Sounds often associated with possible aggressive behavior by males (Tyack, 1981; Silber, 1986) are quite different from songs, extending from 50 Hz to 10 kHz (or higher), with most energy in components below 3 kHz. These sounds appear to have an effective range of up to 9 km (Tyack and Whitehead, 1983).

e. Population status and trends

Currently, there are no statistically reliable estimates of humpback whale population abundance for the entire North Pacific Ocean. Based on aerial, vessel, and photo-identification surveys, and genetic analyses, within the EEZ, there are at least three relatively separate populations that migrate between their respective summer/fall feeding areas and their winter/spring calving and mating areas: 1) winter/spring populations in coastal Central America and the Pacific coast of Mexico, which migrate to the coast of California and north to southern British Columbia in the summer/fall, referred to as the Eastern Pacific stock; 2) winter/spring populations off the Hawaiian Islands which migrate to northern British Columbia/Southeast Alaska and Prince William Sound west to Kodiak, referred to as the Central North Pacific stock; and 3) winter/spring populations of Japan which probably migrate to waters west of the Kodiak Archipelago (Bering Sea and Aleutian Islands), referred to as the Western North Pacific Stock. Winter/spring populations of humpback whales also occur in Mexico's offshore islands (i.e. Revillagigedo Archipelago), but the migratory destination of these whales is not well known (Forney, *et al.*, 2000). Connections between humpback whales in the Hawaiian and Mexican breeding areas and the North Pacific feeding areas have been observed (Darling and Jurasz, 1983; Baker *et al.*, 1990; Calambokidis, *et al.*, 1997), although fewer genetic differences were found between the two breeding areas than the two feeding areas (California and Alaska) (Baker, 1992). Humpback whales were also identified west of Kodiak Island, Alaska, in 1999 and later documented in Hawaii, Japan, and Mexico (B. Witeveen, unpublished report).

Based on the results of photo-identification studies of humpback whales in their wintering areas, the current population estimate for the Central North Pacific stock is 4,005 (CV=0.095) (Calambokidis *et al.*, 1997; Ferrero *et al.*, 2000), with a minimum estimate of 3,698 whales. The most recent abundance estimate for the Western North Pacific stock of humpback whales is 394 (CV=0.084) animals, with a minimum estimate of 367 (Hill and DeMaster, 1999). Barlow (2003) estimated 1,314 (CV=0.30) humpbacks in California, Oregon, and Washington waters (Eastern North Pacific stock), based on summer/fall vessel surveys in 1996 and 2001. The minimum population estimate for the Eastern North Pacific stock is 681 animals (Carretta *et al.*, 2003). Combining all three stocks yields a total abundance estimate of approximately 5,713 (minimum 4,746) humpback whales for the entire North Pacific. This estimate does not include the Mexican breeding stock abundance estimates, because most of these animals are included in the estimates of the Eastern North Pacific stock.

Ship surveys provide some indication that humpback whales increased in abundance in California coastal waters between 1979/80 and 1991 (Barlow, 1994) and between 1991 and 1996 (Barlow, 1997); however estimates declined between 1996 and 2001 (Barlow, 2003). In general, population estimates for the entire North Pacific have increased from 1,200 in 1996 to approximately 6,000-8,000 animals (Calambokidis *et al.*, 1997; Cerchio, 1998; Mobley *et al.*, 1999b). Mark-recapture population estimates have increased from 1988-90 to 1997-98 for the Eastern North Pacific stock at about 8% per year (Forney *et al.*, 2000). Other than these estimates of the size of the humpback whale population, the available information is not sufficient to determine population trends.

Waite *et al.* (1999) identified 127 individual humpback whales in the Kodiak Island region between 1991 and 1994 and estimated there were 651 whales in this region (95% CI:356-1,523). Waite *et al.* (1999) also estimated that 200 humpback whales regularly feed in Prince William Sound. The humpback whale population is believed to have increased since whaling ceased, although the rate of increase is unknown.

Cerchio (1998) estimated that about 4,000 animals visit Hawaii annually. Aerial surveys conducted between 1976 and 1990 found a significant increase in sighting rates of humpbacks over that time (Mobley *et al.*, 1999a), consistent with the increase in photographic estimates. Finally, aerial surveys using line-transect methodologies were conducted in 1993, 1995 and 1998. Hawaii population estimates derived from the sighting data show an increase from 2,717 (+/- 608) in 1993, to 3,284 (+/- 646) in 1995 and 3,852 (+/- 777) in 1998 (Mobley *et al.*, 1999b).

The overall North Atlantic population (including the Gulf of Maine) of humpbacks was estimated from genetic tagging data collected by the YONAH project in the breeding range at 4,894 males (95%CI = 3,374-7,123) and 2,804 females (95%CI = 1,776-4,463) (Palsboll *et al.*, 1997). Since the sex ratio in this population is known to be even (Palsboll *et al.*, 1997), it is assumed that the excess males is due to sampling bias. The best estimate of abundance for the Gulf of Maine humpback whales is 902 (CV=0.41). The minimum population estimate for this stock is 647. Current data indicates that the Gulf of Maine whale stock is steadily increasing in size. Current and maximum productivity rates are unknown for the North Atlantic population overall. However, Stevick *et al.* (2001) calculated an average population growth rate of 3.2% (SE=0.005) for the period of 1979-1993.

3. Sperm Whale

a. Listing status

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). Sperm whales were listed as endangered under the Conservation of Endangered Species and Other Fish or Wildlife on June 2, 1970 and later listed as endangered when the ESA was passed in 1973. They are also protected by CITES and the MMPA. Critical habitat has not been designated for sperm whales.

b. Species description and distribution

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 of marine mammals worldwide, between 60EN and 70ES (Leatherwood and Reeves, 1983). A sperm whale's head is blunt and squared off and can take up to 40 percent of its body length. It has a small underslung jaw, and its eyes are relatively small. Their bodies are a dark brownish gray with a rounded or triangular hump followed by knuckles along its spine. It has the largest brain of any animal on Earth and its blunt snout houses a large reservoir of spermaceti, a high-quality oil.

Sperm whales are found throughout the North Pacific and are distributed broadly from tropical and temperate waters to the Bering Sea as far north as Cape Navarin. Mature female and immature sperm whales of both sexes are found in more temperate and tropical waters from the equator to around 45°N throughout the year. These groups of adult females and immature sperm whales are rarely found at latitudes higher than 50°N and 50°S (Rice, 1989; Reeves and Whitehead, 1997). Sexually mature males join these groups throughout the winter. During the summer, mature male sperm whales are thought to move north into the Aleutian Islands, Gulf of Alaska, and the Bering Sea. They are often concentrated around oceanic islands in areas of upwelling, and along the outer continental shelf and mid-ocean waters.

Sperm whales have a strong preference for the 1,000 meter (3,280-ft) depth contour and seaward. Berzin (1971) reported that they are restricted to waters deeper than 300 m (984 ft), while Watkins (1977) and Reeves and Whitehead (1997) reported that they are usually not found in waters less than 1,000 m (3,281 ft) deep. While deep water is their typical habitat, sperm whales have been observed near Long Island, New York, in waters of 41-55 m (135-180 ft) (Scott and Sadove, 1997). When found relatively close to shore, sperm whales are usually associated with sharp increases in bottom depth where upwelling occurs and biological production is high, implying the presence of a good food supply (Clarke, 1956). They can dive to depths of at least 2000 m (6562 ft), and may remain submerged for an hour or more (Watkins *et al.*, 1993).

A 1997 survey to investigate sperm whale stock structure and abundance in the eastern temperate North Pacific area did not detect a seasonal distribution pattern between the U.S. EEZ off California and areas farther west, out to Hawaii (Forney *et al.*, 2000). However, sperm whales have been sighted around several of the Northwestern Hawaiian Islands (Rice, 1960; Barlow, 2003) and off the main Island of Hawaii (Lee, 1993; Mobley *et al.*, 2000). Sperm whales sighted during aerial surveys conducted in Hawaiian waters from 1993 through 1998 tended to be on the outer edge of a 50 - 70 km distance from the Hawaiian Islands, indicating that presence may increase with distance from shore (Mobley, personal communication, 2000).

In the Atlantic Ocean, NOAA Fisheries' most recent stock assessment report notes that sperm whales are distributed in a distinct seasonal cycle, concentrated east-northeast of Cape Hatteras in winter and shifting northward in spring when whales are found throughout the Mid-Atlantic Bight. Distribution extends further northward to areas north of Georges Bank and the Northeast Channel region in summer and then south of New England in fall, back to the Mid-Atlantic Bight.

In the Mediterranean Sea sperm whales are found from the Alboran Sea to the Levant Basin, mostly over steep slope and deep offshore waters. Sperm whales are rarely sighted in the Sicilian Channel, and are vagrant in the northern Adriatic and Aegean Seas (Notarbartolo di Sciara and Demma, 1997). In the Italian seas, sperm whales are more frequently associated with the continental slope off western Liguria, western Sardinia, northern and eastern Sicily, and both coasts of Calabria.

c. Life history information

Both males and females are thought to reach sexual maturity at approximately 10 years of age (Kasuya, 1991 *in* Perry *et al.*, 1999). However, males may not reach social maturity (i.e. sufficient size to compete for breeding rights) for another decade (Kasuya, 1991). Adult females give birth after about 15 months gestation and nurse their calves for 2 –3 years. The calving interval is estimated to be about four to six years (Kasuya, 1991). The age distribution of the sperm whale population is unknown, but sperm whales are believed to live at least 60 to 70 years (Rice, 1978). Estimated annual mortality rates of sperm whales are thought to vary by age, but previous estimates of mortality rate for juveniles and adults are now considered unreliable (IWC 1980 *in* Perry *et al.*, 1999). Potential sources of natural mortality in sperm whales include killer whales and papilloma virus (Lambertson *et al.*, 1987).

Sperm whales are known for their deep foraging dives (in excess of 3 km). It is thought that the sperm whale lacks good vision, given their small eye size and the lack of light in its deep-water hunting grounds. Instead of depending on their vision, sperm whales use echolocation to find their food. In general, they feed primarily on slow-moving squid (Clark *et al.*, 1993), including the giant squid, *Architeuthis* sp. but may occasionally eat octopus and a variety of fish, including salmon (*Oncorhynchus spp*.), rockfish (*Sebastes spp.*), lingcod (*Ophiodon elongatus*) and skates (*Raja spp.*) (Caldwell and Caldwell, 1983; Leatherwood and Reeves, 1983; Perry *et al*., 1999). Fish eaten in the North Pacific included salmon, lantern fishes, lancetfish, Pacific cod, pollock, saffron cod, rockfishes, sablefish, Atka mackerel, sculpins, lumpsuckers, lamprey, skates, and rattails (Tomilin, 1967, Kawakami, 1980, Rice, 1986b). How sperm whales find and catch their prey can only be inferred, because it has never been possible to observe them feeding. Feeding probably takes place at night, and at great depth, so that vision would be of little use to them, except for locating luminous prey species (luminous species of squid comprised 0-97% of the sperm whale's diet in different areas (Clarke, 1980, *in* Rice, 1989). Daily food consumption rates for sperm whales ranges from 2 - 4% of their total body weight (Lockyer, 1976b; Kawakami, 1980).

d. Diving and social behavior

Sperm whales are likely the deepest and longest diving mammal. Sperm whales are generally found in waters deeper than 180 meters (Leatherwood and Reeves, 1983), and dives of over 2 hours and as deep as 3,000 meters have been recorded (Clarke, 1976; Watkins *et al.*, 1985). Typical foraging dives last 40 minutes and descend to about 400 meters followed by approximately 8 minutes of resting at the surface (Gordon, 1987; Papastavrou *et al.*, 1989). Descent rates recorded from echo-sounders were approximately 1.7 meters/second and nearly vertical (Goold and Jones, 1995). There are no data on diurnal differences in dive depths in sperm whales. However, like most diving vertebrates for which there is data (e.g. rorqual whales, fur seals, chinstrap penguins), sperm whales probably make relatively shallow dives at night when deep scattering layer organisms move towards the surface.

Groups of closely related females and their offspring develop dialects specific to the group (Weilgart and Whitehead, 1997), and females other than birth mothers will guard young at the surface (Whitehead, 1996) and will nurse young calves (Reeves and Whitehead, 1997). Sperm whales produce loud broad-band clicks from about 0.1 to 20 kHz (Weilgart and Whitehead, 1993, 1997;

Goold and Jones, 1995). These have source levels estimated at 171 dB re 1FPa (Levenson, 1974). Current evidence suggests that the disproportionately large head of the sperm whale is an adaptation to produce these vocalizations (Norris and Harvey 1972; Cranford, 1992). This suggests that the production of these loud low frequency clicks is extremely important to the survival of individual sperm whales. The function of these vocalizations is relatively well-studied (Weilgart and Whitehead, 1993, 1997; Goold and Jones, 1995). Long series of monotonous regularly spaced clicks are associated with feeding and are thought to be produced for echolocation. Distinctive, short, patterned series of clicks, called codas, are associated with social behavior and intragroup interactions. They are thought to be for intra-specific communication, perhaps to maintain social cohesion with the group (Weilgart and Whitehead, 1993).

e. Population status and trends

Sperm whales are distributed in all of the world's oceans. Several authors have recommended three or more stocks of sperm whales in the North Pacific for management purposes (Bannister and Mitchell, 1980; Kasuya, 1991,). However, the IWC's Scientific Committee designated two sperm whale stocks in the North Pacific: a western and eastern stock (Donovan 1991). The line separating these stocks has been debated since their acceptance by the IWC's Scientific Committee. For stock assessment purposes, NOAA Fisheries recognizes three discrete population "centers" of sperm whales: (1) Alaska; (2) California/Oregon/Washington; and (3) Hawaii.

Mesnick *et al*. (1999) analyzed the genetic relationships of animals in the eastern Pacific and found that the mtDNA and microsatellite DNA of animals sampled in the California Current is significantly different from animals sampled further offshore, although the line of delineation is unknown. It is likely somewhere between the North American coast and halfway to Hawaii (B. Taylor, NOAA Fisheries - SWFSC, personal communication, March, 2000). Mesnick *et al*. (1999) also found that genetic differences appeared larger in an east-west direction than in a north-south direction. This is confirmed by tagging studies conducted by Rice (1974), who documented three whales tagged in San Francisco and later caught by whalers as far north as British Columbia. Based on differences in gene samples between sperm whales in the Gulf of California, and coastal California, the California-Mexico border is probably near the southern limit of the U.S. west coast stock; however, scientists cannot state with certainty how far west or north the stock may range (B. Taylor, NOAA Fisheries-SWFSC, personal communication, March, 2000).

Current estimates for population abundance, status, and trends for the Alaska stock of sperm whales are not available (Hill and DeMaster, 1999). Approximately 258,000 sperm whales in the North Pacific were harvested by commercial whalers between 1947 and 1987 (Hill and DeMaster, 1999). In particular, the Bering Sea population of sperm whales (consisting mostly of males) was severely depleted (Perry *et al.*, 1999). Catches in the North Pacific continued to climb until 1968, when 16,357 sperm whales were harvested. Catches declined after 1968 through limits imposed by the IWC.

A combined visual and acoustic line-transect survey conducted by NOAA Fisheries in 1997 estimated the population of sperm whales in the eastern temperate North Pacific to be between 24,000 (CV=0.46) sperm whales based on visual sightings, and 39,200 sperm whales (CV=0.60) based on acoustic detections and visual group size estimates (Forney *et al.*, 2000). An analysis for the eastern tropical Pacific estimates abundance at 22,700 sperm whales (95% C. I. = 14,800-34,000; Forney *et al.*, 2000). However, it is not known whether any or all of these animals routinely enter the EEZ of Hawaii. A 2002 shipboard line-transect survey of the entire Hawaiian EEZ resulted in an abundance estimate of 7,082 (CV=0.30) with a minimum estimate of 5,532 sperm whales (Barlow, 2003 *in* Pacific Scientific Review Group Meeting, November 17-20, 2003) including a correction factor for missed diving animals. This is currently the best available abundance estimate for this stock, but it does not include animals that may be found in other U.S. waters of the central Pacific Ocean. Barlow and Taylor (2001) estimated 1,407 (CV=0.39) sperm whales along the coasts of California, Oregon, and Washington during summer/fall, based on ship line transect surveys in 1993 and 1996. The most recent abundance estimate is based on summer/autumn shipboard surveys conducted within 300 nm of the coasts of California, Oregon, and Washington in 1996 (Barlow, 1997) and 2001 (Barlow, 2003). The combined weighted estimate for the 1996 and 2001 surveys is 1,233 (CV=0.41) sperm whales (Barlow, 2003).

The best abundance estimate that is currently available for the western North Atlantic sperm whale population is 2,698 (CV=0.67) animals, and the minimum population estimate used for NOAA Fisheries management purposes is 1,617 (CV=0.67) (Waring *et al.*, in prep.). Due to insufficient data, no information is available on population trends at this time for the western North Atlantic sperm whale stock. No information is available either on Mediterranean sperm whale population size or on the population relationship between sperm whales in the Mediterranean and the North Atlantic. However, the frequent observation of neonates in the Mediterranean and the scarcity of sightings from the Gibraltar area (Bayed and Beaubrun, 1987) points to the possibility that sperm whales in the Mediterranean, like fin whales, may form a resident, reproductively isolated population.

B. Status of Listed Sea Turtles

For the purposes of this consultation, this Opinion focuses on the effects of the HMS FMP fisheries on 1) sea turtle nesting aggregations affected by the proposed actions, 2) the sea turtle population in the Pacific Ocean as distinct from their, as listed, global distribution, and 3) the species as they are listed. The loss of sea turtle populations in the Pacific Ocean would result in a significant gap in the distribution of each turtle species, thus making these populations biologically significant. Finally, the loss of these sea turtle populations in the Pacific Ocean would dramatically reduce the distributions and population abundances of these species and would, by itself, appreciably reduce all species' likelihood of surviving and recovering in the wild. However, despite primarily focusing on the Pacific Ocean populations, NOAA Fisheries must make its final determination of the effect of the HMS FMP fisheries on the species as they are listed, or their global populations. To that end, the following discussions include information on the global status and trends of the sea turtles as well as more detailed information on the

Pacific Ocean populations. In addition, green turtles and olive ridley turtles on the Pacific coast of Mexico are listed separately as endangered species, rather than the threatened status assigned to the remainder of their global populations. These two formally recognized distinct endangered populations will receive separate final determinations from their threatened counterparts.

Populations persist as individuals survive from eggs to adults that successfully reproduce. Populations increase as survivorship rates consistently exceed mortality rates; mortality rates that consistently exceed survivorship rates result in declines in population abundance and may result in a population's eventual extirpation (Mangel and Tier, 1994). As summarized in the *Global Status* and *Factors Affecting Sea Turtles in the Pacific Ocean* sections that follow, both natural and anthropogenic (human-caused) activities affect the abundance and survivorship rates of each life- stage. Turtles that survive from one life stage to the next must survive the rigors of that stage and subsequent stages before they can reach sexual maturity and reproduce. In general, most anthropogenic activities have negatively affected each life stage, resulting in the observed declines in abundance of most sea turtle populations.

Except for nesting aggregations of olive ridley sea turtles and the threatened Hawaiian green sea turtles, nesting aggregations of the other sea turtle species that interact with the HMS FMP fisheries are declining. These population declines are primarily the result of a wide variety of human activities, including legal harvests and illegal poaching of adults, immatures, and eggs; incidental capture in fisheries (coastal and high-seas); and loss and degradation of nesting and foraging habitat as a result of coastal development, including predation by domestic dogs and pigs foraging on nesting beaches associated with human settlement and commercial development of coastal areas (Heppell et al. 2003a, Lutcavage et al. 1997). Increased environmental contaminants (e.g. sewage, industrial discharge) and marine debris, which adversely impact nearshore ecosystems that turtles depend on for food and shelter, including sea grass and coral reef communities, also contribute to the overall decline. While turtle biologists and others generally accept that these factors are the primary cause of turtle population declines, the limited amount of quantitative data on the risks posed by these different activities makes it difficult to rank the absolute risks these different activities pose to listed turtles.

Green, leatherback, loggerhead, and olive ridley sea turtles are highly migratory or have a highly migratory phase in their life history, which makes them susceptible to being incidentally caught by fisheries operating throughout the Pacific Ocean. The collective fisheries proposed to be managed under the HMS FMP are known to interact with all of these species, although varying in degree. In addition to anthropogenic factors, natural threats to nesting beaches and marine habitats such as coastal erosion, seasonal storms, predators, temperature variations, and phenomena such as El Niño also affect the survival and recovery of sea turtle populations. More information on the status of these species along with an assessment of overall impacts are found in this section as well as the Pacific Sea Turtle Recovery Plans (NOAA Fisheries and USFWS, 1998a-e) and are reviewed extensively in Eckert (1993).

1. Green Turtles

a. Global status

Green turtles are listed as threatened under the ESA, except for breeding populations found in Florida and the Pacific coast of Mexico, which are listed as endangered. Using a precautionary approach, Seminoff (2002) estimates that the global green turtle population has declined by 34% to 58% over the last three generations (approximately 150 years) although actual declines may be closer to 70% to 80%. Causes for this decline include harvest of eggs, subadults and adults, incidental capture by fisheries, loss of habitat, and disease.

b. Taxonomy

The genus *Chelonia* is composed of two taxonomic units at the subspecies/subspecific level: the east Pacific green turtle (also known as the "black turtle," *C. mydas agassizii*), which ranges (including nesting) from Baja California south to Peru and west to the Galapagos Islands, and the nominate *C. m. mydas* in the rest of the range (insular Pacific, including Hawaii).

c. Physical Description

Green turtles are distinguished from other sea turtles by their smooth carapace with four pairs of lateral scutes, a single pair of prefrontal scales, four post-orbital scales, and a serrated upper and lower jaw. Adult green turtles have a light to dark brown carapace, sometimes shaded with olive, and can exceed one meter in carapace length and 200 kilograms (kg) in body mass. Females nesting in Hawaii averaged 92 cm in straight carapace length (SCL), while at the Olimarao Atoll in Yap, females averaged 104 cm in curved carapace length (CCL) and approximately 140 kg. Eastern Pacific green turtles are conspicuously smaller and lighter than their counterparts in the central and western Pacific. At the rookeries of Michoacán, Mexico, females averaged 82 cm in CCL, while males averaged 77 cm CCL (*in* NOAA Fisheries and USFWS, 1998a). Nesting females at the Bramble Cay rookery in Queensland, Australia averaged 105.9 cm CCL (Limpus *et al*., 2001).

d. General Distribution

Green turtles are found throughout the world, occurring primarily in tropical, and to a lesser extent, subtropical waters. The species occurs in five major regions: the Pacific Ocean, Atlantic Ocean, Indian Ocean, Carribean Sea, and Mediterranean Sea. These regions can be further divided into nesting aggregations within the eastern, central, and western Pacific Ocean; the western, northern, and eastern Indian Ocean; Mediterranean Sea; and eastern, southern, and western Atlantic Ocean, including the Carribean Sea. Primary nesting aggregations of green turtles (i.e. sites with greater than 500 nesting females per year) include: Ascension Island (south Atlantic Ocean), Australia, Brazil, Comoros Islands, Costa Rica, Ecuador (Galapagos Archipelago), Equatorial Guinea (Bioko Island), Guinea-Gissau (Bijagos Archipelago), Iles Eparses Islands (Tromelin Island, Europa Island), Indonesia, Malaysia, Myanmar, Oman, Philippines, Saudi Arabia, Seychelles Islands, Suriname, and United States (Florida)

(Seminoff, 2002).

Smaller nesting aggregations include: Angola, Bangladesh, Bikar Atoll, Brazil, Chagos Archipelago, China, Costa Rica, Cuba, Cyprus, Democratic Republic of Yemen, Dominican Republic, d'Entrecasteaux Reef, French Guiana, Ghana, Guyana, India, Iran, Japan, Kenya, Madagascar, Maldives Islands, Mayotte Archipelago, Mexico, Micronesia, Pakistan, Palmerston Atoll, Papua New Guinea, Primieras Islands, Sao Tome é Principe, Sierra Leone, Solomon Islands, Somalia, Sri Lanka, Taiwan, Tanzania, Thailand, Turkey, Scilly Atoll, United States (Hawaii), Venezuela, and Vietnam (Seminoff, 2002).

Green turtles appear to prefer waters that usually remain around 20EC in the coldest month. During warm spells (e.g., El Niño), green turtles may be found considerably north of their normal distribution. Stinson (1984) found green turtles to appear most frequently in U.S. coastal waters with temperatures exceeding 18EC. An east Pacific green turtle equipped with a satellite transmitter was tracked along the California coast and showed a distinct preference for waters with temperatures above 20EC (Eckert, unpublished data).

Additionally, it is presumed that drift lines or surface current convergences are preferential zones due to increased densities of likely food items. In the western Atlantic, drift lines commonly contain floating *Sargassum* capable of providing small turtles with shelter and sufficient buoyancy to raft upon (NOAA Fisheries and USFWS, 1998a). Underwater resting sites include coral recesses, the underside of ledges, and sand bottom areas that are relatively free of strong currents and disturbance from natural predators and humans. Available information indicates that green turtle resting areas are in proximity to their feeding pastures (NOAA Fisheries, 2000e).

Molecular genetic techniques have helped researchers gain insight into the distribution and ecology of migrating and nesting green turtles. Throughout the Pacific, nesting assemblages group into two distinct regional clades: 1) western Pacific and South Pacific islands, and 2) eastern Pacific and central Pacific, including the rookery at French Frigate Shoals, Hawaii. In the eastern Pacific, greens forage coastally from San Diego Bay, California in the north to Mejillones, Chile in the South. Based on mtDNA analyses, green turtles found on foraging grounds along Chile’s coast originate from the Galapagos nesting beaches, while those greens foraging in the Gulf of California originate primarily from the Michoacan nesting stock. Green turtles foraging in San Diego Bay and along the Pacific coast of Baja California originate primarily from rookeries of the Islas Revillagigedos (Dutton, 2003).

e. Life Cycle and Population Dynamics

Figure IV-1 illustrates the basic life cycle of green turtles (based on Chaloupka, 2002). This cycle is broken into six life stages: (1) egg/neonate; (2) pelagic juvenile; (3) benthic juvenile; (4) sub-adult; (5) maturing adult; and (6) adult, each with their own expected survival rate (Table IV-1). Arrows along the bottom represent the probability of each ageclass surviving and remaining in the ageclass. Arrows between each ageclass represent the probability of the ageclass surviving and growing to the next ageclass, and the arrows along the top represent the ageclass-specific fertility. The thickness or length of the lines do not indicate a level of probability or fecundity. Available information on the behavior, physiology, and biological requirements of these stages is summarized below.

Figure IV-1. **Life-cycle graph of the green turtle**

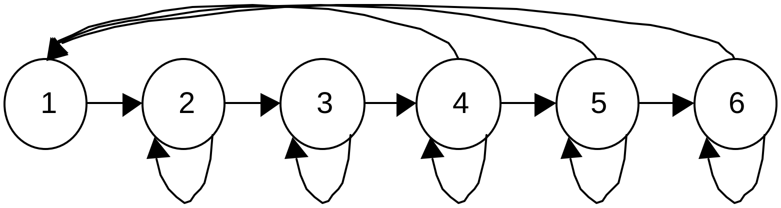


Table IV-1: Stage specific demographic information for the southern Great Barrier Reef green turtle (Chaloupka 2002)

Life Stage	Name	Mean Stage Duration (# years)	Stable Stage Structure	Survival Probability (lx)	Fecundity (eggs/female)
1	Egg-neonate	1	38.0%	0.4394	0
2	Pelagic Juvenile	4	38.8%	0.6445	0
3	Benthic Juvenile	11	18.1%	0.8804	0
4	Subadult	19	4.4%	0.8474	.2488
5	Maturing Adult	5	0.1%	0.9482	40.59
6	Adult	19	0.45%	0.9482	68.84

Numerical analyses of the survival rates, transition rates, and fecundities in Table III-1 indicated that the southern Great Barrier Reef green turtle population has a finite population growth rate (ë) of approximately 1, which suggests a population that is stationary – neither increasing nor declining. This nesting aggregation has not been seriously exposed to incidental capture in fisheries or direct harvest and has shown no evidence of a population decline (Chaloupka, 2002) and therefore may be viewed as a surrogate example of green turtle population dynamics in the absence of anthropogenic activities. The stable stage structure for this nesting aggregation of green turtles is typical of long-lived species with delayed maturity – a life history with large numbers of early stage individuals (as a result of high fecundity in the adult life stages) of which relatively few survive through the rigors of natural mortality from predation, environmental variation, and individual fitness to older reproductive stages (Crouse, 1999). The earliest life stages (Stages 1 and 2) have the highest proportion of individuals but the lowest survival probabilities. Despite low abundances in these life stages, mature individuals have more chances to reproduce and replace themselves. Consequently, changes in the survival rates of adult would be expected to have significant effect on the growth and persistence of this population.

A review of the elasticity, or proportional effect of a change in the vital rates of a stage on ë, of this stage structure confirms the general relationships in this life cycle. Table IV-2 includes the elasticities of the vital rates of each life stage in the green turtle life cycle.

Table IV-2. Stage elasticities (Chaloupka, 2002)

Life Stage	Survival Rate	Transition Rate	Fecundity
1	0	0.0277	0
2	0.0367	0.0277	0
3	0.1466	0.0277	0
4	0.1457	0.0268	0.0008
5	0.0942	0.0227	0.0041
6	0.4166	0	0.0228

Based on these data, a change in the survival rate of an adult green turtle (or the proportion of the stage population that survives as a reproductive adult another year) will have the highest proportional change on the population’s finite growth rate (ë). Changes in the survival rates of the 3rd, 4th, and 5th life stages have the next highest proportional effect on ë, followed by smaller proportional effects due to changes in the survival of pelagic juveniles (Stage 2), transition rates between all stages, or fecundity. The growth, decline, or persistence of the population is determined by the survival rate of reproductive adults, sub-adults, and benthic juveniles. This is not particularly surprising given that these are the longest duration stages for this species. Persistence of long-lived species with delayed maturity would be most vulnerable to impacts that preclude individuals from attaining age and sexual maturity.

The observed declines in the green turtle populations attest to the effect of changing these survival rates on species' persistence. Green turtles have long survived natural fluctuations in environmental conditions (environmental stochasticity) such as changes in climate, coastal erosion, or destruction of nesting beaches by hurricanes and typhoons. Green turtles have survived these phenomena by evolving a life history strategy that allows their populations to withstand periodic, and often significant, losses in the life stages that would be most vulnerable to environmental change (that is, eggs, hatchlings, and juveniles) while buffering the adult life stages from these environmental changes through ocean dispersal. Although adult females on nesting beaches are also vulnerable to phenomena like beach erosion, hurricanes, and typhoons, the reproductive pattern in which adult females only nest every two or more years exposes only a small portion of the breeding population to these risks. Conversely, most anthropogenic activities such as harvest and poaching of eggs and adults, incidental capture in fisheries, or human destruction or encroachment of nesting habitat place these populations under constant pressure, can affect entire regions in short periods of time, and can affect all life stages simultaneously.

For example, green turtle eggs and hatchlings are vulnerable to many of the same factors affecting other sea turtle populations: beach erosion, human or wildlife poaching and predation, and widely fluctuating beach temperatures. Once the green turtles transition into the oceanic environment, however, individual life stages are vulnerable to different impacts based on the habitats they inhabit. Pelagic individuals are incidentally captured in pelagic fisheries such as longline. Benthic life stages are injured or killed by coastal fisheries and other hazards associated with the nearshore environment. While relatively few green turtles are taken by HMS fisheries, based on past observations in the Hawaii-based longline fishery, sub-adult and adult green turtles are the life stage most commonly captured and injured or killed. Because changes in the survival rates of these stages have the highest proportional effect on a population's finite growth rate (ë), the consequences of these fisheries on the survival and recovery of green turtle populations would be significant, particularly when these losses are added to losses in other life stages. The combined effect of these activities, which affect most or all life stages of most green turtle populations, would cause these populations to have ës significantly lower than the southern Great Barrier Reef green turtle population, meaning that these populations would be declining.

f. Biological Characteristics

Diet

Although most green turtles appear to have a nearly exclusive herbivorous diet, consisting primarily of sea grass and algae (Wetherall *et al*., 1993; Hirth, 1997), those along some areas of the east Pacific coast seem to have a more carnivorous diet. Analysis of stomach contents of green turtles found off Peru revealed a large percentage of molluscs and polychaetes, while fish and fish eggs, and jellyfish and commensal amphipods comprised a lesser percentage (Bjorndal, 1997). Black turtles studied in the Magdalena Bay region of the Baja California Peninsula were found to feed predominantly on red algae, *Gracilariopsis*, and to a lesser extent, sea lettuce (*Ulva lactuca*) (Hilbert *et al*., 2002). These turtles locate algae in the rocky coasts and marine grasses plentiful in the shallow waters of the coastal areas, including lagoons and bays (Millan and Carrasco, 2003). Black turtles foraging in areas adjacent to Magdalena Bay fed primarily on sea grass. The stomach contents of one turtle in this area contained more than 82% red crabs (*Plueroncodes planipes*), perhaps the first record of this species feeding predominantly on crustaceans (Mendilaharsu *et al*., 2003). In the Hawaiian Islands, green turtles are site-specific and consistently feed in the same areas on preferred substrates, which vary by location and between islands (*in* Landsberg, *et al*., 1999).

Diving Behavior

Based on the behavior of post-hatchlings and juvenile green turtles raised in captivity, it is presumed that those in pelagic habitats live and feed at or near the ocean surface, and that their dives do not normally exceed several meters in depth (NOAA Fisheries and USFWS, 1998a). The maximum recorded dive depth for an adult green turtle was 110 meters (Berkson, 1967, *in* Lutcavage and Lutz, 1997), while subadults routinely dive 20 meters for 9-23 minutes, with a maximum recorded dive of 66 minutes (Brill, *et al*., 1995, *in* Lutcavage and Lutz, 1997).

Life History/ Reproduction

Compared to all other sea turtles, green turtles exhibit particularly slow growth rate, and age to maturity appears to the longest. Based on age-specific growth rates, green turtles are estimated to attain sexual maturity beginning at age 25 to 50 years (Limpus and Chaloupka, 1997, Bjorndal *et al*., 2000, Chaloupka *et al*., in press, *all in* Seminoff, 2002, Zug *et al*., 2002). Dobbs (2002) estimated the age at first breeding of green turtles in Australia to be 46 years of age. The length of reproductivity has been estimated to range from 17 to 23 years (Carr *et al*., 1978, Fitzsimmons *et al*., 1995 *in* Seminoff, 2002). In Hawaii, green turtles lay up to six clutches of eggs per year (mean of 3.7), and clutches consist of about 100 eggs each. Females migrate to breed only once every two or possibly many more years. On the Hawaiian Archipelago, females nest every 3 to 4 years (Balazs and Chaloupka, in press). Eastern Pacific green turtles have reported nesting between two and six times during a season, laying a mean of between 65 and 86 eggs per clutch, depending on the area studied (Michoacán, Mexico and Playa Naranjo, Costa Rica) (*in* Eckert, 1993 and NOAA Fisheries and USFWS, 1998a). Mean observed and estimated clutch frequency for green turtles nesting at Colola beach (Michoacan, Mexico) was 2.5 and 3.2, respectively (Arias-Coyotl *et al*., 2003). At the Bramble Cay rookery in Queensland, Australia, green turtles laid an estimated 6.2 clutches per season, with an average clutch containing 102.2 eggs. The renesting interval was 12.4 days (Limpus *et al*., 2001).

Migration

The nonbreeding range of green turtles is generally tropical, and can extend thousands of miles from shore in certain regions. Hawaiian green turtles monitored through satellite transmitters were found to travel more than 1,100 km from their nesting beach in the French Frigate Shoals, south and southwest against prevailing currents to numerous distant foraging grounds within the 2,400 kilometer span of the archipelago (Balazs, 1994; Balazs, *et al*., 1994; Balazs and Ellis, 1996). Three green turtles outfitted with satellite tags on the Rose Atoll (the easternmost island at the Samoan Archipelago) traveled on a southwesterly course to Fiji, approximately 1,500 km distance (Balazs, *et al*., 1994).

Tag returns of eastern Pacific green turtles establish that these turtles travel long distances between foraging and nesting grounds. In fact, 75 percent of tag recoveries from 1982-90 were from turtles that had traveled more than 1,000 kilometers from Michoacán, Mexico. Even though these turtles were found in coastal waters, the species is not confined to these areas, as indicated by 1990 sightings records from a NOAA research ship. Observers documented green turtles 1,000-2,000 statute miles from shore (Eckert, 1993). The east Pacific green is also the second-most sighted turtle in the east Pacific during tuna fishing cruises; they are frequent along a north-south band from 15EN to 5ES along 90EW, and between the Galapagos Islands and Central American Coast (NOAA Fisheries and USFWS, 1998a). In a review of sea turtle sighting records from northern Baja California to Alaska, Stinson (1984) determined that the green turtle was the most commonly observed sea turtle on the U.S. Pacific Coast, with 62% reported in a band from southern California and southward. The results of genetic studies and satellite telemetry of black turtles off Baja California suggest a strong connection to rookeries on Michoacan, and to a lesser extent rookeries on Isla Revillagigedo (Nichols, 2003).

The northernmost reported resident population of green turtles occurs in San Diego Bay, where about 50-60 mature and immature turtles concentrate in the warm water effluent discharged by a power plant (McDonald, *et al*., 1994). These turtles appear to have originated from east Pacific nesting beaches and the Revillagigedo Islands (west of Baja California), based on morphology, genetic analyses, and tagging data (*in* NOAA Fisheries and USFWS, 1998a; P. Dutton, NOAA Fisheries, personal communication, March, 2002); however, the possibility exists that some are from Hawaii (P. Dutton, NOAA Fisheries, personal communication, January, 2001). In order to reach nesting beaches in late fall and winter, sea turtles in this area must depart these feeding areas by late summer, returning to the area again in early spring (Nichols, 2003).

g. Population Status and Trends

While some nesting populations of green turtles appear to be stable and/or increasing in the Atlantic Ocean (e.g. Bujigos Archipelago (Guinea-Bissau), Ascension Island, Tortuguero (Costa Rica), Yucatan Peninsula (Mexico), and Florida), declines of over 50% have been documented in the eastern (Bioko Island, Equatorial Guinea) and western Atlantic (Aves Island, Venezuela). Nesting populations in Turkey (Mediterranean Sea) have declined between 42% and 88% since the late 1970s. Population trend variations also appear in the Indian Ocean. Declines greater than 50% have been documented at Sharma (Republic of Yemen) and Assumption and Aldabra (Seychelles), while no changes have occurred at Karan Island (Saudi Arabia) or at Ras al Hadd (Oman). The number of females nesting annually in the Indian Ocean has increased at the Comoros Islands, Tromelin and maybe Europa Island (Iles Esparses) (*in* Seminoff, 2002).

Despite international conservation efforts to protect green turtles in all areas of the world, threats to their survival continue. In the Atlantic and Indian Oceans and the Mediterranean Sea, intentional harvest continues. Egg collection is ongoing at nesting beaches in the eastern Atlantic, western Atlantic and in the Caribbean, while nesting females continue to be killed in the Caribbean, eastern Atlantic and Indian Ocean. High numbers of juveniles and adults are intentionally captured at foraging habitats in the eastern Atlantic, Caribbean, Indian Ocean, and in the Mediterranean (*in* Seminoff, 2002).

Green turtles are thought to be declining throughout the Pacific Ocean, with the exception of Hawaii, as a direct consequence of a historical combination of overexploitation and habitat loss (Eckert, 1993; Seminoff, 2002). A more detailed description of the abundance, distribution, and population trends for green turtles in the Pacific Ocean is presented in the following subsections.

<u>Pacific Ocean</u>

Western Pacific - Distribution and Abundance of Green Turtles, including Nesting Females

In the western Pacific, the only major (> 2,000 nesting females) populations of green turtles occur in Australia and Malaysia.

<u>Australia</u>
In Queensland, Australia there are four distinct subpopulations of green turtles. The southern Great Barrier Reef subpopulation (located at the Capricorn/ Bunker group of islands) has an average annual nesting population of 8,000 females; the northern Great Barrier Reef subpopulation (Raine Island and

Moulter Cay) consists of an average of 30,000 nesting females; the Gulf of Carpenteria (nesting concentrated around Wellesley) averages 5,000 nesting females; and the Coral Sea Islands Territory averages around 1,000 nesting females (Dobbs, 2002).

Threats to green turtles in this area include boat strikes, indigenous harvest of adults and eggs, increased incidence of disease, ingestion of synthetic materials, incidental catch in a shark control program and by commercial fisheries, predation of eggs at nesting beaches, and tourism (*in* Dobbs, 2001). In a study conducted between 1985 and 1992 on foraging greens near southern Great Barrier Reef waters, researchers documented an 11% per year increase in the resident green turtle population, while the female nesting population increased at 3% per year. In 1992, the resident green turtle population was estimated to be comprised of 1,300 individuals (Chaloupka and Limpus, 2001).

Malaysia

In Malaysia, green turtles are widely distributed. Major rookeries are located in Sabah, in the Turtle Islands, where there are about 10,000 nests (increasing trend); in the Sipadan Islands - 800 nests (decreasing trend); in the Sarawak Turtle Islands, about 2,500 nests (stabilized since 1970); in Terengganu, with 2,500 nests (stabilized since 1984); and minor rookeries in Pahang (250 nests) and Perak (200 nests) (Liew, 2002).

Although there are no current estimates available, Pulau Redang, a coral fringed island located approximately 45 kilometers off the coast of Terengganu, Malaysia contains one of the largest green turtle rookeries in peninsular Malaysia, and a 1 nautical mile no-fishing zone has been established around the island to prevent interactions between fishing gear and internesting females (Liew and Chan, 1994).

French Polynesia

Smaller colonies of green turtles occur in the islands of French Polynesia. Although green turtles used to nest in large numbers at Scilly, Motu-one, and Mopelia, located in the western limits of French Polynesia, their populations have declined in recent decades due mainly to commercial exploitation for markets in Tahiti (Balazs, *et al*., 1995). Currently, Scilly is the only known sea turtle nesting site of any magnitude throughout the 130 islands and atolls that comprise French Polynesia. Although residents of Scilly are allowed to harvest 50 adult turtles annually, Balazs *et al*. (1995) estimates that the number of green turtles nesting annually in 1991 is approximately 300-400 turtles, similar to what Lebeau (1985 *in* Balazs, *et al*., 1995) estimated several years earlier.

Indonesia

In Indonesia, green turtles are widely distributed throughout the archipelago; however, many of the largest rookeries have decreased over the last 50 years, primarily due to over-harvest of eggs. Since the Indonesian Government Legislation No. 7/1999 was formally promulgated, all sea turtles in Indonesia, including green turtles, are listed as a protected species. Green turtles reportedly nest in high numbers in the Berau district of East Kalimantan province, the Aru and Kei islands in the Malukkas, and other smaller and more remote islands throughout the country (Dermawan, 2002).

Throughout the Barau district, there are five major nesting sites for green turtles, including the islands of Sangalaki, Mataha, Belambangan, Bilang-bilangan, Balikukup, and Sambit. During 2000, over 1.5 million green turtle eggs were collected in this district, according to the Berau Fisheries Department. Once collected for subsistence, green turtle eggs are now sold to local businessmen, who sell the eggs to distant markets throughout the country as well as illegally export them to Singapore, Brunei, and Sarawak, Malaysia (Dermawan, 2002). Sangalaki Island in the Berau region of East Kalimantan, Indonesia contained one of the largest known nesting populations of green turtles in the Sulawesi Sea. During the post-World War II period, nearly 200 turtles reportedly nested per night. In 1993-94, 20-50 turtles nested per night, while during 2000-2001, 10 turtles on average nested nightly. In the past, egg collectors collected 100% of the eggs. In February, 2001, the Turtle Foundation instituted measures to protect approximately 20% of the eggs laid by female green turtles (approximately 2000 eggs saved per week), and the latest information from the Foundation is that as of January 1, 2002, Bupati and the government of Berau stopped granting licenses to collect turtle eggs on Sangalaki (Turtle Foundation, 2002). At Pulau Banyak (Sumatra, Indonesia), green turtle nesting has been monitored since 1997. The main nesting site is at Amandangan beach. Several thousand clutches are laid annually by several hundred nesting females (Stringell, *et al*., 2000).

Thailand

In Thailand, green turtles nest at the Khram Islands, in the Gulf of Thailand. Here, the nesting areas have been protected and controlled since 1950, so nesting populations have not declined significantly. While peak nesting years for greens showed almost 1,000 nests (late 1980s), since 1994, there has been a steady trend of approximately 200 nests per year at the Khram Islands.

Fiji

In a recent study of migratory patterns of green turtles in the central South Pacific, Craig *et al*. (2004) concluded that about half of the turtle migrations they studied were specifically headed to Fiji and that the seagrass and algae beds associated with Fiji are a regionally-significant food resource for green turtles in that region. However, in Fiji, there is very little information on population trends of green turtles. Although 4,000-5,000 green turtles are found foraging or migrating in Fijian waters, only 30-40 green turtles nest in Fiji. The only nesting sites are located on the islands of Heemskereq Reef and Ringgold reefs. Threats to green turtles in this country are not well known, although green turtles are the most prized food of the Fijians, and they are used as important ceremonial gifts (Rupeni, *et al*., 2002).

Commonwealth of the Northern Mariana Islands

Greens and hawksbills make up most of the composition of sea turtle species in the Pacific island groups under U.S. jurisdiction. Unfortunately, there is a serious shortage of information on the population sizes, distribution, and migration patterns of these turtles, which can hamper recovery efforts. Recently, an assessment of resident sea turtles and their nearshore habitats on two islands of the Commonwealth of the Northern Mariana Islands (CNMI) was conducted. The study took place from

March 12-21, 2001 on the islands of Tinian and Aguijan. An estimated 351 individual green turtles were observed in surveys covering approximately 59% of Tinian's total shore and outer reef perimeter, while only 14 greens were observed during tow surveys covering 95% of Aguijan's shore and reef perimeter. Most of the turtles sighted were juveniles, suggesting recent and continuing recruitment at both islands. Based on data from surveys of four of the five CNMI southern arc islands, Kolinski (2001) also projected sea turtle densities and abundance in these areas and concluded that "the small uninhabited islands of Farallon de Medinilla and Aguijan sustain tens of turtles, turtle numbers around the larger inhabited islands of Saipan and Tinian range in the hundreds, while the CNMI portion of the southern arc (which includes Rota) likely supports between 1,000 and 2,000 resident green turtles." The Division of Fish and Wildlife (2002) report that sea turtles in the Northern Marianas still face problems such as poaching, disturbance of nesting habitat, and the Carolinian and Chamorros (natives) have put in a request to take a limited number of turtles for culture practices.

Guam

In Guam, nesting surveys have been conducted since 1973, more consistently since 1990, and most reliably for the 2000 and 2001 nesting seasons. Trend data since 1990 show that the number of nesting females range between a few to approximately 60 annually, with a general increasing trend over the last 12 years. Aerial surveys from 1990-2000 also show an increasing trend in the number of green turtles sighted around Guam (Cummings, 2002).

Based on limited data, green turtle populations in the Pacific islands have declined dramatically, due foremost to harvest of eggs and adults by humans. In the green turtle recovery plans, directed take of eggs and turtles was identified as a "major problem" in American Samoa, Guam, Palau, CNMI, Federated States of Micronesia, Republic of the Marshall Islands, Wake, Jarvis, Howland, Baker, and Midway Islands, Kingman Reef, Johnston and Palmyra Atoll. Severe overharvests have resulted in modern times from a number of factors: 1) the loss of traditional restrictions limiting the number of turtles taken by island residents; 2) modernized hunting gear; 3) easier boat access to remote islands; 4) extensive commercial exploitation for turtle products in both domestic markets and international trade; 5) loss of the spiritual significance of turtles; 6) inadequate regulations; and 7) lack of enforcement (NOAA Fisheries and USFWS, 1998a). Confirming this, Cummings (2002) reports that in Guam, there is still a high level of illegal take for cultural reasons, particularly during fiestas for the patron saints of villages. Based on anecdotal information, nesting females and eggs are also likely harvested.

Taiwan

Scattered low density nesting of green turtles occur on beaches in Taiwan. Here, Cheng and Chen (1996) report that between 1992 and 1994, green turtles were found nesting on 9 of 11 beaches on Wan-An Island (Peng-Hu Archipelago). The numbers, however, were small, between 8 and 14 females nested during each of these 3 years. Cheng (2002) recently reported similar numbers of nesting greens for those areas: 2-19 nesters on Wan-An Island and 4 to 11 nesters on Lanyu Island.

Vietnam

In Vietnam, researchers have only recently been documenting green turtle nesting populations on their beaches; however, anecdotal reports are that the population has declined sharply, due in part to the harvest of turtles, egg collection for food and wildlife trade, and coastal development. Sea turtles were considered an economic resource until the mid-1990s, when the World Wildlife Fund helped educate the government in the importance of protecting sea turtles and their habitat. Presently, Con Dao National Park is the most important sea turtle nesting site in Vietnam. Data from 1995 through October, 2001 show that for all years except one (1996) over 200 green turtles nested on 14 beaches. Limited numbers of green turtles (23 nests in 2001) have also been documented nesting in Nui Chua Nature Preserve (Hien, 2002).

Japan

In Japan, the Ogasawara Islands, located approximately 1,000 km south of Tokyo, serve as the northern edge of green turtles rookeries in the western Pacific. In the late 1800s, when Japan first colonized the islands, the government encouraged a sea turtle fishery. Declines in catch were steady from 1880-1890s (1,000-1,800 adults taken annually) through the mid-1920s (250 taken annually). Data from 1945-1972 (American occupation) indicate that 20-80 turtles were taken annually, and since then, annual harvests have fluctuated from 45-225 turtles per year (Horikoshi, *et al*., 1994). Suganuma, *et al*. (1996) estimates 100 mating adults are speared by fishermen annually. Beach census data from 1985-93 indicate that 170-649 clutches were deposited each year (43 to 162 nesting females, assuming a female deposited 4 clutches during a nesting season). The Ogasawara population has declined in part due to past commercial exploitation, and it is likely to continue if fishery effort continues (Horikoshi, *et al*., 1994).

Central Pacific - Hawaii

Green turtles in Hawaii are considered genetically distinct and geographically isolated although a nesting population at Islas Revillagigedos in Mexico appears to share the mtDNA haplotype that commonly occurs in Hawaii. In Hawaii, green turtles nest on six small sand islands at French Frigate Shoals, a crescent-shaped atoll situated in the middle of the Hawaiian Archipelago (Northwestern Hawaiian Islands) (Balazs, 1995). Ninety to 95% percent of the nesting and breeding activity occurs at the French Frigate Shoals, and at least 50% of that nesting takes place on East Island, a 12-acre island. Long-term monitoring of the population shows that there is strong island fidelity within the regional rookery.

Researchers have monitored East Island since 1973 and have collected information on numbers of females nesting annually, and have conducted tagging studies (Balazs, 2002). Since the establishment of the ESA in 1973, and following years of exploitation, the nesting population of Hawaiian green turtles has shown a gradual but definite increase (Balazs, 1996; Balazs and Chaloupka, in press). In three decades the number of nesting females at East Island increased from 67 nesting females in 1973 to 467 nesting females in 2002 (Figure IV-2). At this rookery, "... nester abundance increased rapidly during the early 1980s, leveled off during the early 1990s before again increasing rapidly during the late 1990s and up to the present. This trend is very similar to the underlying trend in the recovery of the much larger green turtle population that nests at Tortuguero, Costa Rica (Bjorndal *et al*., 1999). The stepwise increase of the long-term nester trend since the mid-1980s is suggestive, but not conclusive, of a density-dependent adjustment process affecting sea turtle abundance at the foraging grounds (Bjorndal *et al*., 2000)" (Balazs and Chaloupka, in press). This increase can likely be attributed to increased female survivorship since harvesting of turtles in the foraging grounds was prohibited (in the mid-1970s), and cessation of habitat damage at the nesting beaches since the early 1950s (Balazs and Chaloupka, in press). Low level nesting also occurs at Laysan Island, Lisianki Island and on Pearl and Hermes Reef (NOAA Fisheries and USFWS, 1998a).

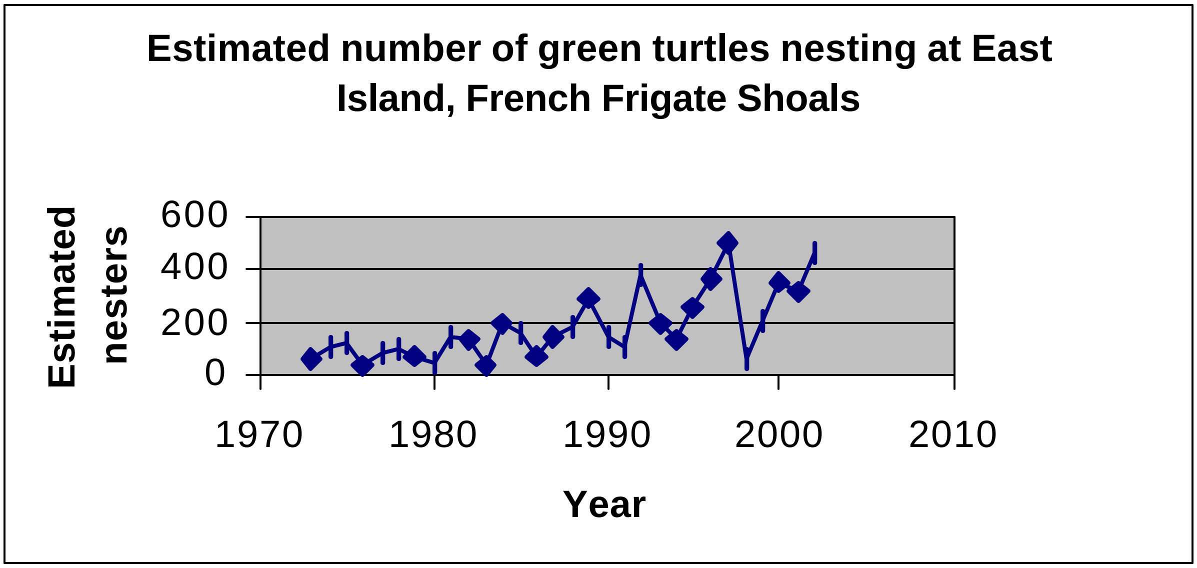


Figure IV-2. Estimated number of female green turtles nesting at East Island, French Frigate Shoals, Hawaiian Archipelago.

Important resident areas of green turtles have been identified and are being monitored along the coastlines of Oahu, Molokai, Maui, Lanai, Hawaii, and at nesting areas in the reefs surrounding the French Frigate Shoals, Lisianski Island, and Pearl and Hermes Reef (Balazs, 1982; Balazs *et al*., 1987).

Unfortunately, the green turtle population in the Hawaiian Islands area is afflicted with a tumor disease, fibropapilloma, which is of an unknown etiology and often fatal, as well as spirochidiasis, both of which are the major causes of stranding of this species (G. Balazs, NOAA Fisheries, personal communication, 2000). The presence of fibropapillomatosis among stranded turtles has increased significantly over the past 17 years, ranging from 47-69 percent during the past decade (Murakawa, *et al*., 2000). Green turtles captured off Molokai from 1982-96 showed a massive increase in the disease over this period, peaking at 61% prevalence in 1995 (Balazs, *et al*., 1998). Preliminary evidence suggests that there is an association between the distribution of fibropapillomatosis in the Hawaiian Islands and the distribution of toxic benthic dinoflagellates (*Prorocentrum* spp.) known to produce a tumor promoter, okadaic acid (Landsberg, *et al*., 1999). Fibropapillomatosis is considered an inhibiting factor to the full recovery of the Hawaiian green turtle populations, and the incidence of decreased growth rates in afflicted turtles is a minimum estimate of the impact of the disease (Balazs, *et al*., 1998). Stranding reports from the Hawaiian Islands from 1982-1999 indicate that the green turtle is the most commonly stranded sea turtle (96.5 percent, compared to other species), averaging around 150 per year (2,689 total/18 years). Despite recent increases in this disease, increases in nester abundance in the Hawaiian Archipelago has continued to occur (Aguirre *et al*., 1998 *in* Balazs and Chaloupka, in press).

Eastern Pacific - Distribution and Abundance of Nesting Females

Analysis using mitochondrial DNA (mtDNA) sequences from three key nesting green turtle populations in the eastern Pacific indicate that they may be considered distinct management units: Michoacán, Mexico; Galapagos Islands, Ecuador, and Islas Revillagigedos, Mexico (Dutton, 2003).

The primary green turtle nesting grounds in the eastern Pacific are located in Michoacán, Mexico, and the Galapagos Islands, Ecuador (NOAA Fisheries and USFWS, 1998a). Here, green turtles were widespread and abundant prior to commercial exploitation and uncontrolled subsistence harvest of nesters and eggs. Sporadic nesting occurs on the Pacific coast of Costa Rica.

<u>Mexico</u>

In the Mexican Pacific, the two main nesting beaches for female green turtles occur in Michoacán and include Colola, which is responsible for 70% of total green turtle nesting in Michoacán (Delgado and Alverado, 1999), and Maruata. These nesting beaches have showed a dramatic decline, particularly in the early 1980s, decreasing from 5,585 females in 1982 to 940 in 1984 which represents about a 90%

decline in two years. On Colola, an estimated 500-1,000 females nested nightly in the late 1960s. In the 1990s, that number dropped to 60-100 per night, or about 800-1,000 turtles per year (Eckert, 1993).

Since their decline in the 1980s from about 5,500 nesting females per year, the number of nesting females arriving at Colola Beach in Mexico has fluctuated widely between lows of 171 and highs of 880, until recently when about 2,100 female turtles returned to nest in 2001 (see Figure IV-3).

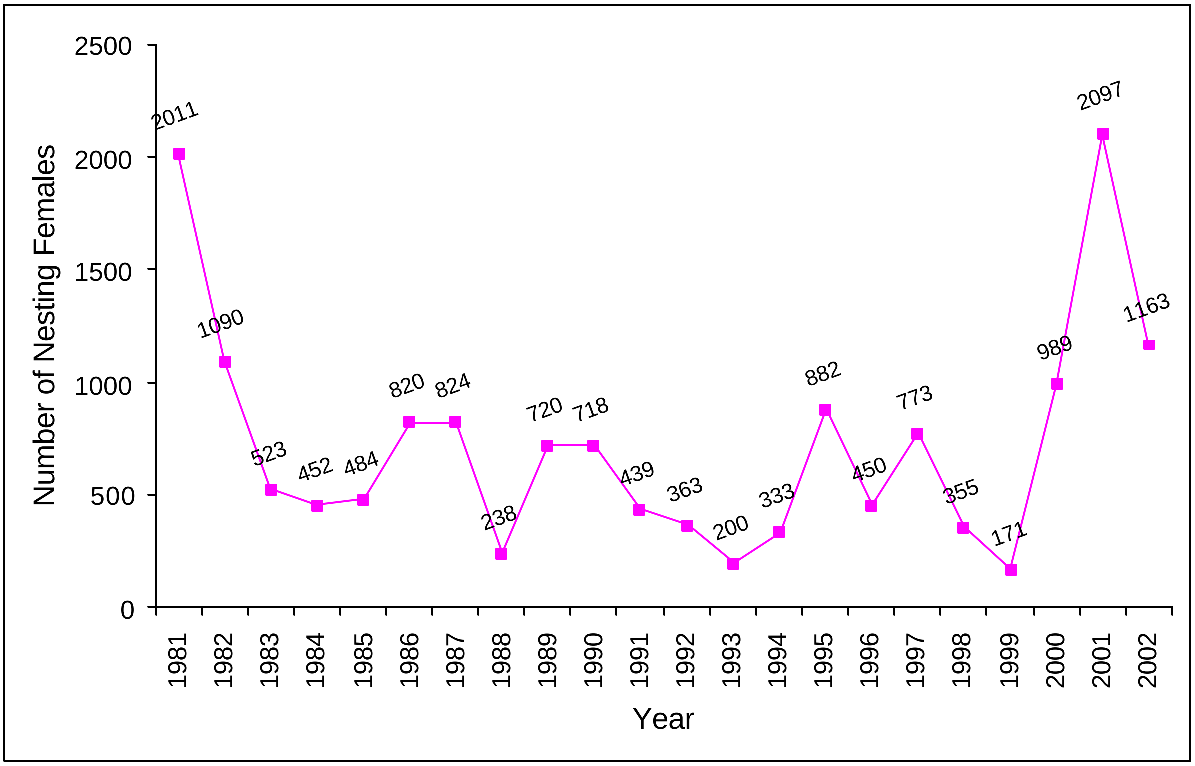
Table IV-3. Results of an assessment of the Colola, Mexico, nesting aggregation of green sea turtles using a discrete-time, density-independent diffusion estimation model

Demographic Parameter	**Estimate**
Mean log growth rate (µ)	-0.026078
Upper 95% confidence interval	0.321947
Lower 95% confidence interval	-0.374102
Variance in mean log growth rate (ó^2)	0.584556
Upper 95% confidence interval	0.637932
Lower 95% confidence interval	0.342150
Finite rate of population increase (ë)	1.304997
Upper 95% confidence interval	1.892563
Lower 95% confidence interval	0.899847

Our analyses of estimates of the number of female green turtles that nest at Colola Beach suggest that the population’s growth rate is, on average, positive (ë = 1.30, which is greater than 1), but the lower confidence interval around this estimate suggests that the population may, in fact, be declining (0.90 which is less than 1; Table IV-3). The mean of the population’s log growth rate (µ = -0.261) supports this conclusion: most trajectories of this population would be expected to decline slightly. The wide fluctuations in the number of nesting females that return from year-to-year could present a more serious problem for this population as those fluctuations bring the population to very low levels that, over time, would be expected to create weak year-classes of recruits into the adult, female population. Although the increases in nesting females in 2000 and 2001 provide cause for optimism, historical numbers of this species nesting during the 1960s show that the population is still below its natural level (Alvarado-Diaz and Trejo, 2003; Alvarado-Diaz, personal communication, October, 2003). The small size of this nesting population, relative to its historic levels, leaves this population with a moderate risk of extinction: projecting over 25-, 50-, and 100-year intervals suggest that this population has a low risk of declining to extinction in any interval of time, but has a moderate risk of declining to 100 or 500 individuals in about 50 years.

Historically in the Mexican Pacific, more than 165,000 turtles were harvested from 1965 to 1977. In the early 1970s nearly 100,000 eggs per night were collected from these nesting beaches (*in* NOAA Fisheries and USFWS, 1998a). Despite long-term protection of females and their eggs at these sites since 1990, the population continues to decline, and it is believed that adverse impacts (including incidental take in various coastal fisheries as well as illegal directed take at forage areas) continue to prevent recovery of endangered populations (P. Dutton, NOAA Fisheries, personal communication, 1999; Nichols, 2002). In addition, the black market for sea turtle eggs in Mexico has remained as brisk as before the ban (Delgado and Alvarado, 1999). Although only about 5% of the nests were poached at Colola during this season, approximately 50% of the nests at Maruata were poached, primarily because of difficulties in providing protections as a result of political infighting (Delgado and Alvarado, 1999).

Figure IV-3. Number of Nesting Female Green Turtles at Colola beach, Michoacan, Mexico (from Alvarado-Diaz, personal communication, October, 2003).



The Archipelago of Revillagigedo, an isolated group of islands offshore from the Pacific Mexican coast also hosts a nesting and foraging population of green turtles. Monitoring studies from 1999-2001 on three beaches on Clarión Island and five beaches on Socorro showed a small but productive population of nesting greens. Nesting occurs year-round but peaks during October and November. Genetic analyses on these turtles show the Revillagigedo population to be a genetically distinct stock distributed throughout Baja California and the western United States (Juarez-Ceron, *et al*., 2003).

Ecuador

There are few historical records of abundance of green turtles from the Galapagos. Investigators documented nesting females during the period 1976-1982 and recorded an annual average of 1,400 nesting females. At this time, only residents were allowed to harvest turtles for subsistence, and egg poaching occurs only occasionally (NOAA Fisheries and USFWS, 1998a) The main documented threats that were registered in the past was the presence of feral pigs (*Sus scrofa*), an introduced species to the islands, and the beetle (*Omorgus suberosus*), a native species. Both of these combined to reduce turtle hatchling success during earlier monitoring years (Zárate *et al*., 2003). After nearly twenty years of limited data, a field study commenced in 2002 to assess the status of green turtles nesting in the main nesting sites of the Galapagos Archipelago. The most important nesting beaches are Quinta Playa and Bahía Barahona, both on Isabela Island, Las Bachas, Santa Cruz Island, Las Salinas, Seymour Island, and Espumilla, Santiago Island. All are protected as either national parks, tourist sites, or are under military jurisdiction (e.g. Seymour Island). Monitoring sites included all of the above-listed nesting beaches except Espumilla. Nesting activity was monitored for nearly 4 months in Las Bachas and approximately 3 months on the remaining sites. During the season, a total of 2,756 females were tagged, with the highest numbers in Las Bachas (925 females). This total outnumbers the highest values recorded in previous studies (1,961 females tagged in 1982) (Table IV-4). Researchers observed few feral pigs and they were only observed in Qunita Playa. There were few documented beetle observations, although feral cats were observed predating on hatchlings as they emerged from the nest (Zàrate *et al*. 2003).

Table IV-4. Historic data of turtles tagged in Galapagos between 1975-1983 and 2002.

Year	Total Number of Tagged Females
1975	102
1976	478
1977	526
1978	1,087
1979	827
1980	1,411
1981	1,639
1982	1,961
1983	89
2002	2,756

Source: Data from 1979-1983 from Hurtado (1984); Data from 1975 from Cifuentes (1975); data from 1976-79 from Green (1984); data from 1980-83 in Hurtado (1984) *all in* Zàrate *et al*. (2003).

Costa Rica

Green turtles also nest sporadically on the south Pacific coast of Costa Rica, and have been monitored in Caña Blanca and Punta Banco. The total number of nests recorded in Caña Blanca from 1998-2001 ranged from 47 to 106 annually, while the total nests recorded in Punta Banco from 1996 to 2001 ranged from 73 to 233 nests (Lopez and Arauz, 2003). At Playa Naranjo, the population of nesting green turtles was estimated to be between 125 and 175 (Cornelius, 1976 *in* NOAA Fisheries and USFWS, 1998a).

Green turtles encountered by U.S. vessels managed under the HMS FMP may originate from a number of known proximal, or even distant, breeding colonies in the Pacific Ocean. No green turtles have been observed taken in the west coast-based longline fishery. Genetic analyses conducted on the one green turtle observed taken in the CA/OR drift gillnet fishery was found to originate from eastern Pacific stock, most likely a Mexican nesting beach (P. Dutton, personal communication, January, 2000). Green turtles taken in the ETP purse seine fishery likely originate from eastern Pacific nesting beaches; however, genetic sampling has not been conducted. Genetic sampling of green turtles taken by the Hawaii-based longline fishery, in which the area of fishing effort overlaps with much of the fishing grounds of the west coast-based longline fleet, indicates representation from nesting beaches on Hawaii (French Frigate Shoals) and the eastern Pacific (Mexico - both Revillagigedos and Michoacan and Galapagos). Preliminary genetic analysis has revealed that of 14 green turtles sampled by observers in the Hawaii-based longline fishery from 1994 to 2001, six were of eastern Pacific (Mexico) stock origin, five were of eastern Pacific or Hawaiian nesting stock origin, three were of Hawaii stock origin, and one was of unknown origin, although it is most likely to be of eastern Pacific stock due to similarities in mtDNA sequence. (P. Dutton, NOAA Fisheries, personal communication, December, 2003).

2. Leatherback Turtles

a. Global Status

The leatherback turtle is listed as endangered under the ESA throughout its global range. Increases in the number of nesting females have been noted at some sites *in the Atlantic*, but these are far outweighed by local extinctions, especially of island populations, and the demise of once large populations *throughout the Pacific,* such as in Malaysia and Mexico. Spotila *et al.* (1996) estimated the *global* population of female leatherback turtles to be only 34,500 (confidence limits: 26,200 to 42,900) nesting females; however, the eastern Pacific population has continued to decline since that estimate, leading some researchers to conclude that the leatherback is now on the verge of extinction in the Pacific Ocean (e.g. Spotila, *et al*., 1996; Spotila, *et al*., 2000).

b. Physical Description

Leatherback turtles are the largest of the marine turtles, with a CCL often exceeding 150 cm and front flippers that are proportionately larger than in other sea turtles and may span 270 cm in an adult (NOAA Fisheries and USFWS, 1998c). In view of its unusual ecology, the leatherback is morphologically and physiologically distinct from other sea turtles. Its streamlined body, with a smooth, dermis-sheathed carapace and dorso-longitudinal ridges may improve laminar flow of this highly pelagic species. Leatherbacks nesting in the western Pacific are considerably larger than those nesting in the eastern Pacific. Adult females nesting in Michoacán, Mexico averaged 145 cm CCL (Sarti, unpublished data, *in* NOAA Fisheries and USFWS, 1998c), while adult female leatherback turtles nesting in eastern Australia averaged 162 cm CCL (Limpus, *et al*., 1984, *in* NOAA Fisheries and USFWS, 1998c). Leatherbacks in Papua, Indonesia and Papua New Guinea averaged 161 cm CCL (Starbird and Suarez, 1994; Hirth *et al*., 1993, respectively).

c. General Distribution

Leatherback turtles are widely distributed throughout the oceans of the world. The species is found in four main regions of the world: the Pacific, Atlantic, and Indian Oceans, and the Caribbean Sea. Leatherbacks also occur in the Mediterranean Sea, although they are not known to nest there. The four main regional areas may further be divided into nesting aggregations. Leatherback turtles are found on the western and eastern coasts of the Pacific Ocean, with nesting aggregations in Mexico and Costa Rica (eastern Pacific) and Malaysia, Indonesia, Australia, the Solomon Islands, Papua New Guinea, Thailand, and Fiji (western Pacific). In the Atlantic Ocean, leatherback nesting aggregations have been documented in Gabon, Sao Tome and Principe, French Guiana, Suriname, and Florida. In the Caribbean, leatherbacks nest in the U.S. Virgin Islands and Puerto Rico. In the Indian Ocean, leatherback nesting aggregations are reported in India and Sri Lanka.

Leatherback turtles have the most extensive range of any living reptile and have been reported circumglobally from 71EN to 47ES latitude in the pelagic Pacific and in all other major pelagic ocean habitats (NOAA Fisheries and USFWS, 1998c). For this reason, however, studies of their abundance, life history and ecology, and pelagic distribution are exceedingly difficult. Leatherback turtles lead a completely pelagic existence, foraging widely in temperate waters except during the nesting season, when gravid females return to tropical beaches to lay eggs. Males are rarely observed near nesting areas, and it has been proposed that mating most likely takes place outside of the tropical waters, before females move to their nesting beaches (Eckert and Eckert, 1988). Leatherbacks are highly migratory, exploiting convergence zones and upwelling areas in the open ocean, along continental margins, and in archipelagic waters (Morreale, *et al*., 1994; Eckert, 1998; Eckert, 1999a). In a single year, a leatherback may swim more than 10,000 kilometers (Eckert, 1998).

The distribution of juvenile leatherback turtles has long been a mystery. However, a recent compilation and analysis of sighting and stranding data for the species has provided some insight into the developmental habitats of this species at earlier life stages. It appears that young leatherback turtles (carapace length <100 cm) reside only in waters warmer than 26EC, which should generally place them outside of areas in which longline swordfish fleets operate (Eckert, 1999b; Eckert, 2002).

d. Life Cycle and Population Dynamics

Figure IV-4 illustrates the basic life cycle of the leatherback turtle (based on estimates provided by Chaloupka (2001) for western Pacific leatherback nesting aggregations). This cycle is broken into seven life stages based on age: (1) egg/hatchling; (2) neonate; (3) warm water juvenile, (4) cool water juvenile, (5) immature, (6) sub-adult, and (7) adult, each with their own expected survival rate (Table IV-5). Arrows along the bottom represent the probability of each ageclass surviving and remaining in the ageclass. Arrows between each ageclass represent the probability of the ageclass surviving and growing to the next ageclass, and the arrows along the top represent the ageclass-specific fertility. The thickness or length of the lines do not indicate a level of probability or fecundity. Available information on the behavior, physiology, and biological requirements of these stages is summarized below.

Figure IV-4. **Life-cycle graph of the leatherback turtle (based on Chaloupka, 2001)**

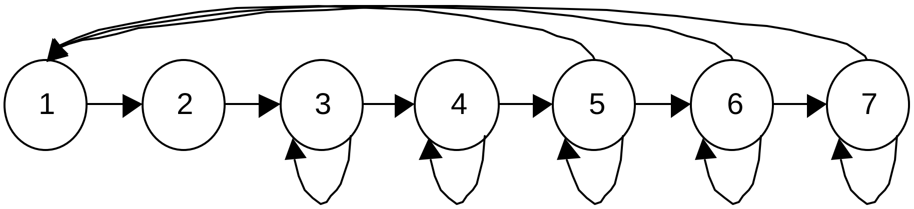


Table IV-5: Stage structure and vital rates for leatherback turtles (Chaloupka, 2001; 2002)

Stage	Name	Age	Stable Stage Structure[1]	Survival Probability (lx)	Fecundity[2]
1	Egg-hatchling	0	not estimated	various	0
2	Neonate	1	65.12%	0.25	0
3	Warm Water Juvenile	2-4	21.38%	0.75	0
4	Cool Water Juvenile	4-5	4.02%	0.75	0
5	Immature	5-9	5.99%	0.85	79-90
6	Sub-adult	10-14	1.46%	0.9	79-90
7	Adult	15+	1.97%	0.9	79-90

[1] Stable age structure proportions estimated from Chaloupka (2002) leatherback simulation model parameters for initial abundance of western Pacific population in 1900.

[2] Eckert (2000).

Because leatherback turtles spend most of their lives in pelagic environments, it is very difficult to gather the basic information on their abundance, life history and ecology, and pelagic distribution. In the absence of these data, several investigators have constructed conceptual models, simulations, or thought experiments to estimate possible stable age structures and stage-specific survival probabilities for leatherback turtles (Chaloupka 2001, 2002; Spotila et al. 1996, 2000). The results of these efforts help frame the direction of future research, but the degree to which they reliably describe the actual vital rates of different leatherback turtle populations is unknown.

However, the data that are available suggest that leatherback turtles follow patterns that are similar to other long-lived species that delay the age at which they become mature (Chaloupka 2001, 2002; Crouse 1999; Heppell et al. 1999, 2003a; Meylen and Ehrenfeld 2000; Spotila et al. 1996, 2000). That is, leatherback turtles can be expected to have low and variable survival in the egg and hatchling stages and high and relatively constant annual survival in the subadult and adult life stages (Heppell et al. 2003). An undisturbed population of leatherback sea turtles is likely to have rates of increase that are fairly stable. For example, green turtles in the southern Great Barrier Reef have a finite rate of increase (ë) of approximately 1, indicating a stationary population, or one that is neither increasing nor decreasing over time intervals covering several years.

In addition, growth rates of leatherback turtle populations are probably more sensitive to changes in the survival rate of juvenile, sub-adult, and adult turtles than other stages. As a result, the survival rate of reproductive adults, sub-adults, and juvenile leatherback turtles will largely determine the growth, decline, or maintenance of the population (Crouse 1999; Heppell et al. 1999, 2003a; Meylen and Ehrenfeld 2000; Spotila et al. 1996, 2000). Conversely, the population's rates of increase or decrease would be relatively insensitive to changes in the survival rates of eggs or hatchlings; this does not imply that other life stages can be disregarded, but does imply that the species has evolved to withstand low survival rates at these stages as well as large amounts of year-to-year variation (Heppell et al. 2003a). Persistence of long-lived species with delayed maturity would be most vulnerable to impacts that preclude individuals from (1) attaining age and sexual maturity, or (2) surviving to produce sufficient offspring to replace themselves.

Finally, like other sea turtles, high site fidelity in nesting females implies that once a nesting aggregation declines to a few individuals or becomes extinct, it will not be “rescued” by adult females from other nesting aggregations. As a result, the loss of a nesting aggregation is final and irreversible.

The dynamics of most leatherback turtle populations today are certain to reflect the effects of numerous anthropogenic activities which have caused or exacerbated the declines in abundance noted in many leatherback nesting aggregations, such as those documented in Malaysia. As a result, the rates of population increase or decrease, and life stage elasticities of these populations are likely indicative of declining populations (that is rates of increase or ës less than 1, and changed proportional importance of different life stage elasticities on a population’s rate of growth). For an example of the changed dynamics of a declining sea turtle population, see the *Life Cycle and Population Dynamics* discussion for loggerhead turtles below. In a disturbed population, the survival rates of adult turtles may still have the highest elasticities, typical for long-lived species with delayed maturity. However, the survival rates of life stages relatively undisturbed by chronic or significant sources of mortality increase in importance as the population relies upon these stages to supply enough individuals to survive the rigors of subsequent life stages and reach sexual maturity. In the case of a population where the survival of all life stages has been decreased by anthropogenic activities, stage elasticities may change such that the proportional effect of a change in survival rate in any stage can have significant effect on the rates at which their populations grow over time.

Leatherback populations currently face high probabilities of extinction as a result of both environmental and demographic stochasticity. Demographic stochasticity, or chance variation in the birth or death of an individual of the population, is facilitated by the increases in mortality rates of leatherback populations due in part to harvest of individuals (either eggs or reproductive females on nesting beaches) or incidental capture and mortality of individuals in various fisheries. Environmental stochasticity, or random environmental changes that deteriorate or degrade environmental quality, is facilitated by destruction of nesting beach habitat or changes in nest temperature from loss of shade on nesting beaches. Loss of habitat or deterioration in habitat quality can reduce egg survival or even change the sex ratios of produced hatchlings. In both cases, the variation in rate or ratio due to environmental stochasticity exacerbate demographic stochasticity through increased mortality, or decreased breeding probability as individuals in a sex-skewed population have more difficulty finding members of the opposite sex. Increases in demographic stochasticity tend to increase the variance in the population growth rate (Gilpin and Soule, 1986). As this variance increases, a population's probability of extinction due to chance events increases. As a result, declines in a species' abundance due to increased mortality or the loss of some resource (nesting habitat, prey, etc.,) that might otherwise lead to extinction through deterministic processes also increase a species' chance of extinction via other random occurrences (Gilpin and Soule, 1986).

Based on past observations, the leatherback turtles that are captured and killed in U.S. fisheries operating in the Pacific are primarily sub-adult and adult leatherback turtles (see the discussion in the *Effects of the Action* section). Over the history of these fisheries, the effect of these annual deaths would significantly reduce the survival rates of individuals in these life stages in the nesting aggregations that interact with these fisheries. From our analyses, these reductions would be expected to have a significant, adverse affect on the trend of those nesting aggregations, particularly if these losses are added to losses in other life stages. The combined effect of these activities, which affect most or all life stages of most leatherback turtle populations, would significantly reduce the population growth rates of the nesting aggregations that interact with these fisheries.

e. Biological Characteristics

Diet

Satellite telemetry studies indicate that adult leatherback turtles follow bathymetric contours over their long pelagic migrations and typically feed on cnidarians (jellyfish and siphonophores) and tunicates (pyrosomas and salps), and their commensals, parasites and prey (NOAA Fisheries and USFWS, 1998c). Because of the low nutritive value of jellyfish and tunicates, it has been estimated that an adult leatherback would need to eat about 50 large jellyfish (equivalent to approximately 200 liters) per day to maintain its nutritional needs (Duron, 1978, *in* Bjorndal, 1997). Compared to greens and loggerheads, which consume approximately 3-5% of their body weight per day, leatherback turtles may consume perhaps 20-30% of their body weight per day (Davenport and Balazs, 1991).

Surface feeding by leatherbacks has been reported in U.S. waters, especially off the west coast (Eisenberg and Frazier, 1983), but foraging may also occur at depth. Based on offshore studies of diving by adult females nesting on St. Croix, U.S. Virgin Islands, Eckert *et al*. (1989) proposed that observed internesting[4] dive behavior reflected nocturnal feeding within the deep scattering layer (strata comprised primarily of vertically migrating zooplankton, chiefly siphonophore and salp colonies, as well as medusae). Hartog (1980, *in* NOAA Fisheries and USFWS, 1998c) also speculated that foraging may occur at depth, when nematocysts from deep water siphonophores were found in leatherback stomach samples. Davenport (1988, *in* Davenport and Balazs, 1991) speculated that leatherback turtles may locate pyrosomas at night due to their bioluminescence; however direct evidence is lacking.

Diving Behavior

The maximum dive depths for post-nesting female leatherbacks in the Carribean have been recorded at 475 meters and over 1,000 meters, with routine dives recorded at between 50 and 84 meters. The maximum dive length recorded for such female leatherback turtles was 37.4 minutes, while routine dives ranged from 4-14.5 minutes (*in* Lutcavage and Lutz, 1997). Leatherback turtles also appear to spend almost the entire portion of each dive traveling to and from maximum depth, suggesting that maximum exploitation of the water column is of paramount importance to the leatherback (Eckert, *et al*., 1989).

A total of six adult female leatherback turtles from Playa Grande, Costa Rica were monitored at sea during their internesting intervals and during the 1995 through 1998 nesting seasons. The turtles dived continuously for the majority of their time at sea, spending 57-68% of their time submerged. Mean dive depth was 19 ± 1 meters and the mean dive duration was 7.4 ± 0.6 minutes (Southwood, *et al*., 1999). Similarly, Eckert (1999a) placed transmitters on nine leatherback females nesting at Mexiquillo Beach and recorded dive behavior during the nesting season. The majority of the dives were less than 150 meters depth, although maximum depths ranged from 132 meters to over 750 meters. Although the dive durations varied between individuals, the majority of them made a large proportion of very short dives (less than two minutes), although Eckert (1999a) speculates that these short duration dives most likely represent just surfacing activity after each dive. Excluding these short dives, five of the turtles had dive durations greater than 24 minutes, while three others had dive durations between 12-16 minutes.

Migrating leatherback turtles also spend a majority of time at sea submerged, and they display a pattern of continual diving (Standora, *et al*., 1984, *in* Southwood, *et al*., 1999). Based on depth profiles of four leatherbacks tagged and tracked from Monterey Bay, California in 2000 and 2001, using satellite-linked dive recorders, most of the dives were to depths of less than 100 meters and most of the time was spent shallower than 80 meters. Based on preliminary data analysis, 75-90% of the time the leatherback turtles were at depths less than 80 meters (Peter Dutton, NOAA Fisheries, personal communication, January 2004).

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[4] Internesting – time spent between laying clutches of eggs during a single nesting season.

Life History/Reproduction

Using a small sample size of leatherback sclerotic ossicles, analysis by Zug and Parham (1996) suggested that mean age at sexual maturity for leatherback turtles is around 13 to 14 years, giving them the highest juvenile growth rate of all sea turtle species. Zug and Parham (1996) concluded that for conservation and management purposes, 9 years is a likely minimum age for maturity of leatherback turtles, based on the youngest adult in their sample. The natural longevity of leatherback turtles has not been determined (NOAA Fisheries and USFWS, 1998c), although there are recorded documentations of post-maturation survival on the order of about 20 years (Pritchard, 1996).

On the Pacific coast of Mexico, female leatherback turtles lay an average of 4 clutches per season, with clutch size averaging 64 yolked eggs per clutch (García and Sarti, 2000) (each clutch contains a complement of yolkless eggs[5], sometimes comprising as much as 50 percent of total clutch size, a unique phenomenon among leatherback turtles and some hawksbills (Hirth and Ogren, 1987)). Each clutch is laid within a 9.3 day interval (García and Sarti, 2000). In Las Baulas, Costa Rica, the average clutch size is also 64.7 ± 1.4 yolked eggs. Reproductive output ranged from 4.3 ±0.2 to 7.9 ±0.3 clutches per female per nesting season (Reina *et al*., 2002). Clutch sizes in Terengganu, Malaysia, and in Pacific Australia were larger, averaging around 85-95 yolked eggs and 83 yolked eggs, respectively (*in* Eckert, 1993).

Females are believed to migrate long distances between foraging and breeding grounds, at intervals of typically two or four years (García and Sarti, 2000). Spotila *et al*. (2000), found the mean re-nesting interval of females on Playa Grande, Costa Rica to be 3.7 years, while in Mexico, 3 years was the typical reported interval (L. Sarti, Universidad Naçional Autonoma de Mexico (UNAM), personal communication, 2000). In Mexico, the nesting season generally extends from November to February, although some females arrive as early as August (Sarti *et al*., 1989). Most of the nesting on Las Baulas takes place from the beginning of October to the end of February (Reina *et al*., 2002). In the western Pacific, nesting peaks on Jamursba-Medi Beach (Papua, Indonesia) from May to August, on War-Mon Beach (Papua) from November to January (Starbird and Suarez, 1994), in peninsular Malaysia in June and July (Chan and Liew, 1989), and in Queensland, Australia in December and January (Limpus and Riemer, 1984).

Migration

Migratory routes of leatherback turtles originating from eastern and western Pacific nesting beaches are not entirely known. However, satellite tracking of post-nesting females and genetic analyses of leatherback turtles caught in U.S. Pacific fisheries or stranded on the west coast of the U.S. present some strong insight into at least a portion of their routes and the importance of particular foraging areas. Aerial surveys conducted during the late summer and fall months of 1990-2001 reveal that leatherbacks forage off central California, generally at the end of the summer, when upwelling relaxes and sea surface

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[5] Bell *et al*. (2003) note that "yolkless eggs" is an incorrect nomenclature, since they do not contain a 1 N nucleous with an associated yolk that together make up a gamete or oöcyte.

temperatures increase. Leatherbacks were most often spotted off Point Reyes, south of Point Arena, in the Gulf of the Farallones, and in Monterey Bay. These areas are upwelling “shadows,” regions where larval fish, crabs, and jellyfish are retained in the upper water column during relaxation of upwelling. Researchers estimated an average of 170 leatherbacks (95% CI = 130-222) were present between the coast and roughly the 50 fathom isobath off California. Abundance over the study period was variable between years, ranging from an estimated 20 leatherbacks (1995) to 366 leatherbacks (1990) (Benson *et al*., 2003).

Current data from genetic research suggest that Pacific leatherback stock structure (natal origins) may vary by region. Due to the fact that leatherback turtles are highly migratory and stocks mix in high seas foraging areas, and based on genetic analyses of samples collected by both Hawaii-based and west coast-based longline observers, leatherback turtles inhabiting the northern and central Pacific Ocean are comprised of individuals originating from nesting assemblages located south of the equator in the western Pacific (e.g. Indonesia, Solomon Islands) and in the eastern Pacific along the Americas (e.g., Mexico, Costa Rica) (Dutton, *et al*., 2000).

For female leatherback turtles nesting at Mexiquillo Beach, Mexico, the eastern Pacific region has been shown to be a critical migratory route. Nine females outfitted with satellite transmitters in 1997 traveled along almost identical pathways away from the nesting beach. These individuals moved south and, upon encountering the North Equatorial Current at about 8EN, diverted west for approximately 800 km and then moved east/southeast towards the waters off Peru and Chile (Eckert, 1999a). In addition, four leatherback turtles recovered from Chilean fishing vessels from 1988-91 had been tagged on nesting beaches in Costa Rica and Mexico (Brito-Montero, 1995, *in* Donoso, 2000). A leatherback tagged at Agua Blanca in Baja California in 2000 began migrating south to approximately 370 kilometers from where it was tagged (Pinal *et al*., 2002).

Morreale *et al*. (1994) demonstrated that satellite tagged, post-nesting leatherback turtles leaving Costa Rica followed precisely defined, long-distance migratory pathways after nesting. Despite differences in dates of departure from the nesting areas, nesting cohorts followed along nearly identical pathways. All 6 leatherback turtles’ (from the Pacific and Caribbean coasts of Costa Rica) movements paralleled deepwater bathymetric contours ranging from 200-3,500 meters. When a turtle’s path intersected an abyssal plain, it veered along the outer slope, and when an abyssal plain was unavoidable, the turtle crossed it at its narrowest point. These studies underscore the importance of this offshore habitat and migratory corridors and the likelihood that sea turtles are present on fishing grounds, particularly for large commercial fishing fleets south of the equator (Eckert, 1997). Eckert (1999a) speculates that leatherback turtles leaving the nesting areas of Mexico and Costa Rica may be resource-stressed by a long reproductive season with limited food and the high energetic requirements brought about by the demands of reproduction, elevated water temperatures, or both. When they leave, their greatest need is to replenish energy stores (e.g. fat) and they must move to areas where food is concentrated (e.g. upwelling areas). Most of these eastern Pacific nesting stocks migrate south, although one genetic sample from a leatherback turtle caught south of the main Hawaiian Islands by the

Hawaii-based longline fishery indicated representation from eastern Pacific nesting beaches (P. Dutton, NOAA Fisheries, personal communication, October 2002).

In the last three years, researchers have discovered two important migratory corridors of leatherback turtles originating from western Pacific nesting beaches. Initially, genetic analyses of stranded leatherbacks found along the western U.S. mainland determined that the turtles had originated from western Pacific nesting beaches. Furthermore, genetic analysis of samples from leatherback turtles taken off California and Oregon by the CA/OR drift gillnet fishery and in the northern Pacific, taken by the California-based longline fishery, revealed that all originated from western Pacific nesting beaches (i.e. Indonesia/Solomon Islands/Malaysia; P. Dutton, NOAA Fisheries, personal communication, December, 2003).

Observations of tracked leatherbacks captured and tagged off the west coast of the United States have revealed an important migratory corridor from central California, to south of the Hawaiian Islands, leading to western Pacific nesting beaches. In September, 2000, researchers captured their first two leatherbacks off Monterey, California. Of two females, one was of a size normally associated with the western Pacific nesting stock, which are, on average, 10-20 centimeters larger than eastern Pacific nesting stocks (Zug and Parham, 1996). Both headed on a southwest migratory path, appearing to be heading to the western Pacific nesting beaches (Dutton and Eckert in press). In 2001, a male and female leatherback were captured and tagged. The male headed north of the "migratory corridor" taken by the two females the year before and stopped transmitting on 12/17/01, while the female traveled north to the Farallon Islands and then headed west, where transmissions stopped on 10/11/01 (D. Parker and P. Dutton, NOAA Fisheries, personal communication, June, 2002). Genetic analysis confirmed that all four of these leatherbacks tagged and outfitted with transmitters were from the western Pacific stock (P. Dutton, NOAA Fisheries, personal communication, October 2002). Since then, eight leatherbacks (6 females, 2 males) were captured in 2002, and six (5 females, 1 male) were captured in 2003. All were outfitted with satellite tags and tracked. Most followed the southwest migratory corridor, heading towards western Pacific nesting beaches. Two that have been tracked for an extended period of time did not arrive on the nesting beaches, instead heading north and east, back towards the northen part of Hawaii. One leatherback did not follow a southwest track out of Monterey and instead headed southeast, along Baja California, Mexico, and into the Gulf of California. All leatherbacks captured off central California have been found to originate from western Pacific nesting beaches (P. Dutton, NOAA Fisheries, personal communication, December, 2003).

Researchers have also begun to track female leatherbacks tagged on western Pacific nesting beaches, both from Jamursba-Medi, Papua, and from the Morobe coast of Papua New Guinea. Most of the females that have been tagged in Papua have been tracked heading on an easterly pathway, towards the western U.S. coast. One female headed north and is currently meandering in the East China Sea and the Sea of Japan, generally between Japan and South Korea. Another female headed north and then west of the Philippines. Meanwhile, all the leatherbacks tagged off Papua New Guinea have traveled on a southeasterly direction, in the south Pacific Ocean (P. Dutton, NOAA Fisheries, personal communication, December, 2003).

Genetic markers in 16 of 17 leatherback turtles sampled to date from the central North Pacific (captured in the Hawaii-based longline fishery) have identified those turtles as originating from nesting populations in the southwestern Pacific; the other specimen, taken in the southern range of the Hawaii fishery, was from nesting beaches in the eastern Pacific (Dutton and Eckert, in press). All 3 leatherbacks taken in the California-based longline fishery were found to originate from western Pacific nesting beaches, based on genetic analyses.

f. Population Status and Trends

Leatherback turtles are widely distributed throughout the oceans of the world, and are found in waters of the Atlantic, Pacific, and Indian Oceans, the Caribbean Sea, and the Gulf of Mexico (Ernst and Barbour, 1972). Globally, leatherback turtle populations have been decimated worldwide. In 1980, the leatherback population was estimated at approximately 115,000 (adult females) globally (Pritchard, 1982b). By 1995, this global population of adult females had declined to 34,500 (Spotila *et al.* 1996). Populations have declined in Mexico, Costa Rica, Malaysia, India, Sri Lanka, Thailand, Trinidad, Tobago, and Papua New Guinea. Throughout the Pacific, leatherbacks are seriously declining at all major nesting beaches. The decline can be attributed to many factors, including fisheries interactions, direct harvest, egg collection, and degradation of habitat. On some beaches, nearly 100% of the eggs laid have been harvested. Eckert (1996) and Spotila *et al*. (1996) note that adult mortality has also increased significantly, particularly as a result of driftnet and longline fisheries.

<u>Atlantic Ocean/Caribbean Sea</u>

In the Atlantic and Caribbean, the largest nesting assemblages of leatherbacks are found in the U.S. Virgin Islands, Puerto Rico, and Florida. Since the early 1980s, nesting data has been collected at these locations. Populations in the eastern Atlantic (*i.e.* off Africa) and Caribbean appear to be stable; however, information regarding the status of the entire leatherback population in the Atlantic is lacking and it is certain that some nesting populations (*e.g.,* St. John and St. Thomas, U.S. Virgin Islands) have been extirpated (NOAA Fisheries and USFWS, 1995). Data collected in southeast Florida clearly indicate increasing numbers of nests for the past twenty years (9.1-11.5% increase), although it is critical to note that there was also an increase in the survey area in Florida over time (NOAA Fisheries SEFSC, 2001). However, the largest leatherback rookery in the western North Atlantic remains along the northern coast of South America in French Guiana and Suriname. Recent information suggests that Western Atlantic populations declined from 18,800 nesting females in 1996 (Spotila *et al.,* 1996) to 15,000 nesting females by 2000 (Spotila, personal communication *in* NOAA Fisheries SEFSC, 2001). The nesting population of leatherback turtles in the Suriname-French Guiana trans-boundary region has been declining since 1992 (Chevalier and Girondot, 1998). Poaching and fishing gear interactions are, once again, believed to be the major contributors to the decline of leatherbacks in the area (Chevalier *et al.* in press; Swinkels *et al.* in press). While Spotila *et al.* (1996) indicated that turtles may have been shifting their nesting from French Guiana to Suriname due to beach erosion, analyses show that the overall area trend in number of nests has been negative since 1987 at a rate of 15.0 -17.3 % per year (NOAA Fisheries SEFSC, 2001). If turtles are not nesting elsewhere, it appears that the Western Atlantic portion of the population is being subjected to mortality beyond sustainable levels, resulting in a continued decline in numbers of nesting females.

Leatherbacks are exposed to commercial fisheries in many areas of the Atlantic Ocean. For example, leatherback entanglements in fishing gear are common in Canadian waters where Goff and Lien (1988) reported that 14 of 20 leatherbacks encountered off the coast of Newfoundland/ Labrador were entangled in fishing gear including salmon net, herring net, gillnet, trawl line and crab pot line. Leatherbacks are reported taken by the many other nations that participate in Atlantic pelagic longline fisheries (see NOAA Fisheries SEFSC 2001, for a complete description of take records), including Taiwan, Brazil, Trinidad, Morocco, Cyprus, Venezuela, Korea, Mexico, Cuba, U.K., Bermuda, People's Republic of China, Grenada, Canada, Belize, France, and Ireland. Leatherbacks are known to drown in fish nets set in coastal waters of Sao Tome, West Africa (Castroviejo *et al.*, 1994; Graff, 1995). Gillnets are one of the suspected causes for the decline in the leatherback turtle population in French Guiana (Chevalier *et al.*,1999), and gillnets targeting green and hawksbill turtles in the waters of coastal Nicaragua also incidentally catch leatherback turtles (Lagueux *et al.*, 1998). Observers on shrimp trawlers operating in the northeastern region of Venezuela documented the capture of six leatherbacks from 13,600 trawls (Marcano and Alio, 2000). An estimated 1,000 mature female leatherback turtles are caught annually off of Trinidad and Tobago with mortality estimated to be between 50-95% (Eckert and Lien, 1999). However, many of the turtles do not die as a result of drowning, but rather because the fishermen butcher them in order to get them out of their nets (NOAA Fisheries SEFSC 2001). There are known to be many sizeable populations of leatherbacks nesting in West Africa, possibly as many as 20,000 females nesting annually (Fretey 2001). In Ghana, nearly two thirds of the leatherback turtles that come up to nest on the beach are killed by local fishermen.

Pacific Ocean - general

Based on published estimates of nesting female abundance, leatherback populations are declining at all major Pacific basin nesting beaches, particularly in the last two decades (Spotila *et al*., 1996; NOAA Fisheries and USFWS, 1998c; Spotila, *et al*., 2000). Declines in nesting populations have been documented through systematic beach counts or surveys in Malaysia (Rantau Abang, Terengganu), Mexico and Costa Rica. In other leatherback nesting areas, such as Papua New Guinea, Indonesia, and the Solomon Islands, there have been no systematic consistent nesting surveys, so it is difficult to assess the status and trends of leatherback turtles at these beaches. In all areas where leatherback nesting has been documented, however, current nesting populations are reported by scientists, government officials, and local observers to be well below abundance levels of several decades ago. The collapse of these nesting populations was most likely precipitated by a tremendous overharvest of eggs coupled with incidental mortality from fishing (Sarti *et al*., 1996; Eckert, 1997).

Eastern Pacific Nesting Populations of Leatherbacks

Leatherback nesting populations are declining at a rapid rate along the Pacific coast of Mexico and Costa Rica (see Appendix B, Table 1). Three countries which are important to leatherbacks nesting in the eastern Pacific include Costa Rica, which has the highest abundance and density in this area, Mexico, with several important nesting beaches, and Nicaragua, with two important nesting areas. Leatherbacks have been documented nesting as far north as Baja California Sur and as far south as Panama, with few areas of high nesting (Sarti, 2002).

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. As shown in Figure IV-5, during the 1988-89 season (July-June), 1,367 leatherback turtles nested on this beach, and by the 1998-99 season, only 117 leatherback turtles nested (Spotila, 2000). The 1999-2000 and 2000-01 season showed increases in the number of adult females nesting here, with 224 and 397 leatherbacks nesting, respectively. The last two nesting seasons have shown major declines, with only 69 nesting females during the 2001-02 season, and only 55 nesting females during the 2002-03 season. Scientists speculate that the low turnout during 2002-03 may be due to the "better than expected season in 2000-01 which temporarily depleted the reproductive pool of adult females in reproductive condition following the El Niño/La Niña transition" (R. Reina, Drexel University, personal communication, September, 2003).

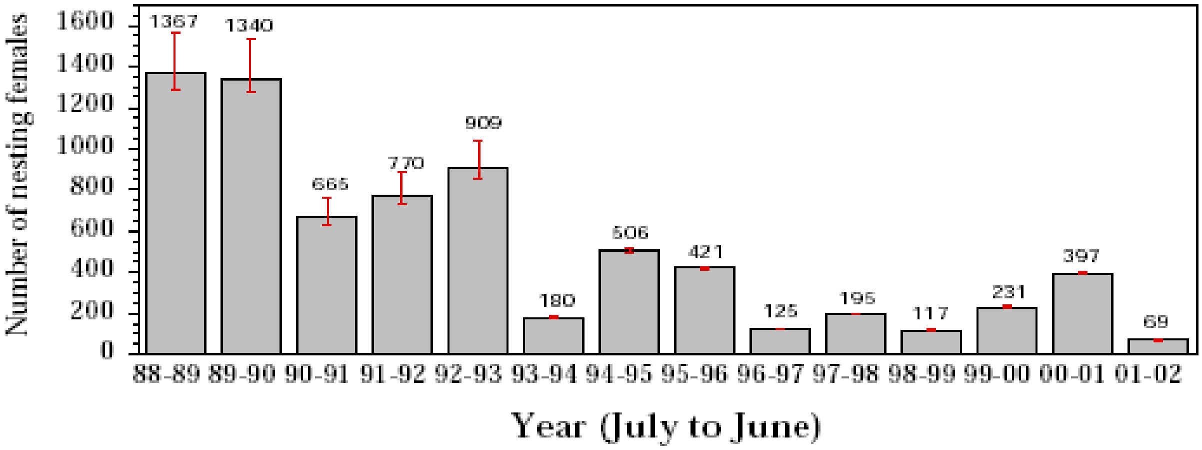

Figure IV-5. Number of female leatherbacks nesting at Playa Grande (Las Baulas, Costa Rica) from 1988-2002. (Source: R. Reina, Drexel University, personal communication, September, 2003).

Researchers began tagging females at Playa Grande in 1994. Since then, tagged leatherbacks have had a low return rate - 16% and 25% in the five or six years following tagging. Spotila *et al*. (2000)

calculated a mean annual mortality rate of 35% for leatherbacks nesting at Las Baulas. At St. Croix, US Virgin Islands nesting grounds, female leatherbacks returned approximately 60% over the same period (McDonald and Dutton, 1996 *in* Reina *et al*., 2002) and annual mortality rates ranged from 4-10% (Dutton *et al*., 1999 *in* Reina *et al*., 2002). Thus, comparatively few leatherback turtles are returning to nest on east Pacific nesting beaches and it is likely that leatherback turtles are experiencing abnormally high mortalities during non-nesting years. Since 1993, environmental education and conservation efforts through active law enforcement has greatly reduced egg poaching in Costa Rica (Chaves, *et al*., 1996). For example, during the 1993-94 nesting season, poaching accounted for only 1.3 percent of the loss of nests on Playa Grande. Other losses were due to predation, tidal effects and failure in egg development or infestation by maggots (Schwandt, *et al*., 1996). Bell *et al*. (2003) found that while leatherbacks at Playa Grande had a high rate of fertility (mean = 93.3% ± 2.5%), embryonic death was the main cause of low hatchling success in this population. Researchers at Playa Grande have also found that temperature of the sand surrounding the egg will determine the sex of the hatchlings during a critical phase of their embryonic development. At this beach, temperatures above 29.5EC produce female hatchlings, while below 29.5EC, the hatchlings are male.

Since the late 1980s, the number of leatherback turtles nesting on the beaches of Playa Grande has declined from about 1,300 nesters per year to less than 400. The nesting aggregation appears to have fluctuated between about 400 and 70 individuals throughout most of the 1990s and early 2000s which suggests an instability in the population. This is consistent with the reports on the infertility of females in this population, high female mortalities between breeding intervals, and changing beach temperatures, all of which increase the variance in a population.

Table IV-6. Results of an assessment of the Playa Grande nesting aggregation of leatherback sea turtles using a discrete-time, density-independent diffusion estimation model

Demographic Parameter	**Lower Census Estimate**	**Upper Census Estimate**
Mean log growth rate (µ)	-0.048485	-0.048439
Upper 95% confidence interval	0.174896	0.194803
Lower 95% confidence interval	-0.271865	-0.291680
Variance in mean log growth rate (ó^{2})	0.226610	0.268697
Upper 95% confidence interval	0.653270	0.774597
Lower 95% confidence interval	0.113718	0.134839
Finite rate of population increase (ë)	1.066967	1.089708
Upper 95% confidence interval	1.327670	1.387804
Lower 95% confidence interval	0.8574557	0.855642

The results of our analyses (Table IV-6) of the lower and upper estimates of the number of female leatherback turtles that nest at Playa Grande suggest that the population's growth rate is, on average, positive (ë = 1.17, which is greater than 1), but the lower confidence interval around this estimate suggests that the population may, in fact, be declining (0.667 which is less than 1). The mean of the population's log growth rate ($\mu = -0.229501$) supports this conclusion: most trajectories of this population would be expected to decline. Projecting these results over 25-, 50-, and 100-year intervals suggest that this population has a high risk of extinction (declining to 1 or 0 females) in the one human generation (about 20 years) if no action is taken. As we have discussed previously, different nesting aggregations of sea turtles are effectively isolated from one another, the female leatherback turtles nesting at Playa Grande will not be "rescued" by migrants from other nesting beaches. If this nesting aggregation becomes extinct, it will remain extinct.

There have been anecdotal reports of leatherbacks nesting at Playa Caletas and Playa Coyote. Playa Caletas is an 8 km beach on the Nicoya Peninsula on the Pacific Coast of Costa Rica. It is separated from Playa Coyote to the north. Locals report that in the mid-1990s, approximately 20 leatherbacks emerged to nest each night, while during the 1997-98 nesting season, 30-40 leatherback nesting incidences were observed. A monitoring study in this area during October 1 through December 11, 1999 noted only five leatherback body pits and one possible leatherback body pit on Playa Caletas (Squires, 1999).

Mexico

The decline of leatherback subpopulations is even more dramatic off the Pacific coast of Mexico. Surveys indicate that the eastern Pacific Mexican population of adult female leatherback turtles has declined from 70,000[6] in 1980 (Pritchard, 1982b, *in* Spotila *et al*., 1996) to approximately 60 nesting females during the 2002-03 nesting season, the lowest seen in 20 years (L. Sarti, UNAM, personal communication, June, 2003).

Leatherbacks nesting in Mexico nest from October through March. According to reports from the late 1970s and early 1980s, three beaches located on the Pacific coast of Mexico (Bahiá de Chacahua, Oaxaca, Tierra Colorada, Guerrero and Mexiquillo, Michoacán) sustained a large portion of all global nesting of leatherback turtles, perhaps as much as one-half. Because nearly 100% of the clutches in these areas were poached by local people, a monitoring plan was implemented to evaluate the nesting population and establish measures for the protection of eggs. From aerial surveys, daily beach surveys, and nightly patrols, the following information has been determined for nesting leatherbacks on the Pacific coast of Mexico:

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[6] This estimate of 70,000 adult female leatherback turtles comes from a brief aerial survey of beaches by Pritchard (1982), who has commented: "I probably chanced to hit an unusually good nesting year during my 1980 flight along the Mexican Pacific coast, the population estimates derived from which (Pritchard, 1982b) have possibly been used as baseline data for subsequent estimates to a greater degree than the quality of the data would justify" (Pritchard, 1996).

(1) Four main nesting beaches: Mexiquillo, Michoacán; Tierra Colorada, Guerrero; and Cahuitan and Barra de la Cruz, in Oaxaca, comprise from 40-50% of total nests along the Mexican Pacific;
(2) Four secondary nesting beaches: Chacahua, Oaxaca; La Tuza, Oaxaca; Playa Ventura, Guerrero, and Agua Blanca, Baja California Sur;
(3) All eight beaches comprise approximately 75-80% of the total annual nests of the Mexican Pacific (Sarti, personal communication, December, 2003).

Monitoring of the nesting assemblage at Mexiquillo, Mexico has been continuous since 1982. During the mid-1980s, more than 5,000 nests per season were documented along 4 kilometers of this nesting beach. By the early 1990s (specifically 1993), less than 100 nests were counted along the entire beach (18 kilometers) (Sarti, 2002). According to Sarti *et al.* (1996), nesting declined at this location at an annual rate of over 22 percent from 1984 to 1995. Sarti *et al*. (1998) reports:

> "While reporting the results for the 1995-96 nesting season (Sarti *et al*., 1996), we regarded beaches having densities higher than 50 nests per kilometer as the most important. In the present season [1997-98] no beach reached such density values: the main beaches had 5 or more nests per kilometer, and none were higher than 25. This is evidence of the large decrement witnessed from the start of the aerial surveys, and may indicate that the nesting population still has a declining trend despite the protection efforts in the major beaches."

Censuses of four index beaches in Mexico during the 2000-2001 nesting season showed a slight increase in the numbers of females nesting compared to the all-time lows observed from 1996 through 1999 (Sarti *et al*. in prep). However, the number of nestings during the last two nesting seasons (2001-02 and 2002-03) is the lowest ever recorded, as shown in Table IV-7.

Table IV-7. Annual number of leatherback nestings from 2000-2003 on primary and secondary nesting beaches.

Index beach	**2000-2001**	**2001-2002[1]**	**2002-2003[2]**
Primary Nesting Beaches (40-50% of total nesting activity)			
Mexiquillo	624	20	36
Tierra Colorada	535	49	8
Cahuitan	539	52	73
Barra de la Cruz	146	67	3
Secondary Nesting Beaches			
Agua Blanca	113	no data	no data

Total - all index beaches	**1,957**	**188**	**120**
Total - Mexican Pacific	**4,513**	**658**	**not available yet**

[1] Source: Sarti, pers. comm, March, 2002 – index beaches; Sarti *et al*., 2002 for totals;
[2] Source: Sarti, pers. comm, December, 2003 – index beaches, totals.

A summary of total leatherback nestings counted and total females estimated to have nested along the Mexican coast from 1995 through 2003 is shown in Table IV-8.

Table IV-8. Total leatherback nestings counted and total number of females estimated to nest along the Mexican Pacific coast per season.

Season	Nestings	Females
1995-1996	5,354	1,093
1996-1997	1,097	236
1997-1998	1,596	250
1998-1999[1]	799[1]	67[2]
1999-2000	1,125	225
2000-2001	4,513	991
2001-2002	658	109-120
2002-2003	not available	not available

[1] Value corrected for E1 (error due to track and bodypit aging) and E2 (error due to difficulty of observation from the air) only.
[2] Number of females only includes tagged females at the key beaches.
Source - Sarti *et al*., 2000 (1995-1999 data), Sarti *et al.*, 2002 (2001-02 data), Sarti, personal communication, June, 2003 (2002-03 data).

Furthermore, Sarti, *et al*. (2000) notes that during the 1980s, 30% of the nesting females per season were remigrants, but since the mid-1990s, there has been very little evidence of remigration, even with more efficient tagging methods. Sarti (2002) reported that during the 1999-2000 and 2000-01 nesting seasons, only a small increment in the number of remigrant turtles was observed.

Although the causes of the decline in the eastern Pacific nesting populations are not entirely clear, Sarti *et al*. (1998) surmises that the decline could be a result of intensive egg poaching on the nesting beaches, incidental capture of adults or juveniles in high seas fisheries, and natural fluctuations due to changing environmental conditions. Although leatherback turtles are not generally captured for their meat or skin in Mexico, the slaughter of female leatherback turtles has been detected on beaches such as Piedra de Tiacoyunque, Guerrero (Sarti, *et al*., 2000). Nichols (2002) notes that leatherbacks were once harvested off Baja California but their meat is now considered inferior for human consumption. In addition, there is little information on incidental capture of adults due to coastal fisheries off Mexico, but entanglement in longlines and driftnets probably account for some mortality of leatherback turtles. Eckert (1997) speculates that the swordfish gillnet fisheries in Peru and Chile have contributed to the decline of the leatherback in the eastern Pacific. The decline in the nesting population at Mexiquillo, Mexico occurred at the same time that effort doubled in the Chilean driftnet fishery.

Most conservation programs aimed at protecting nesting sea turtles in Mexico have continued since the early 1980s, and there is little information on the degree of poaching prior to the establishment of these programs. However, Sarti *et al.* (1998) estimates that as much as 100% of the clutches were taken from the Mexican beaches. Since protective measures have been in place, particularly emergency measures recommended by a joint U.S./Mexico leatherback working group meeting in 1999, there has been greater nest protection and nest success (Table IV-9). Mexican military personnel were present during the 1999-2000 season at three of the primary nesting beaches in Mexico (Llano Grande, Mexiquillo, and Tierra Colorado), responsible for approximately 34% of all nesting activity in Mexico. Of 1,294 nests documented, 736 were protected (57%), resulting in a total of 25,802 hatchlings. Monitoring and protection measures at two secondary nesting beaches resulted in the protection of 67% and 10% at Barra de la Cruz and Playa Ventura, respectively. Beginning in 2000, the primary management objective has been to protect over 95% of nests laid at the three index beaches (includes protecting nesting females, eliminating illegal egg harvest, and relocating nests to protected hatcheries) and to maximize protection of all the secondary nesting beaches over the next three years. NOAA Fisheries has committed funding for three years to help implement these objectives (Dutton *et al.*, 2002).

Table IV-9. Nest protection at index beaches on the Pacific coast of Mexico (Source: Sarti *et al*., personal communication, December, 2003)

Nesting Season	Number of clutches laid	Number of clutches protected	Percentage of clutches protected
1996-97	445	86	19.3
1997-98	508	101	19.9
1998-99	442	150	33.9
1999-00	1590	943	58.7
2000-01	1,732	933	57.04
2001-02	171	116	67.9

The most recent results for 2000-01 indicate that nearly 58% of clutches laid in key beaches in Mexico were relocated to hatcheries. This is a significant increase since 1996, when only 12% of nests were relocated. Although data are not available, most of the nests that were not moved are believed to have survived in situ in 2000-01, unlike previous years when it is assumed that all nests that are not relocated are taken by poachers. This has been due to successful involvement of community leaders in Cahuitan, the most important leatherback nesting beach in the nest protection program. At this beach 24,797 eggs representing 80% of the nests laid were protected, producing a total of 12,275 hatchlings (L. Sarti, INP Preliminary Report).

Nicaragua

In Nicaragua, small numbers of leatherbacks nest on Playa El Mogote, and Playa Chacocente, both beaches within 5 kilometers of one another and located in the Rio Escalante Chacocente Wildlife Refuge. From October through December,1980, 108 leatherbacks were sighted nesting on Playa Chacocente, while during January, 1981, 100 leatherbacks reportedly nested in a single night on Playa El Mogote (*in* Arauz, 2002). Similar to many of the leatherback nesting beaches along the eastern Pacific, the abundance of nesting females has decreased. An aerial survey conducted during the 1998-1999 season estimated a nesting density in Playa El Mogote of only 0.72 turtles per kilometer (Sarti *et al.*, 1999 *in* Arauz, 2002). During the 2000-01 nesting season, community members near Playa El Mogote noted that 210 leatherback nests had been deposited. Of these, 31 nests produced hatchlings, while the rest were poached (85% poaching rate). During the 2001-02 nesting season (monitored from October through March), leatherbacks successfully nested 29 times. Of these, 6 nests were protected in a hatchery and 23 were poached (79.3% poaching rate) (Arauz, 2002).

Guatemala

On the Pacific coast of Guatemala, leatherbacks nest in limited numbers (2-3 nests per night from November to December), primarily on the beach at Hawaii. Since an average nest can bring in one quarter of the monthly income of a typical agricultural worker or fishermen, most leatherback eggs are collected (Juarez and Muccio, 1997), and in the Hawaii area, "it is very rare that a nest is laid without being detected by an egg collector" (Muccio, 1998).

Total Abundance Estimates of Nesting Females in Eastern Pacific

From tagging and aerial surveys, Spotila *et al.* (2000) have estimated that there are currently 687 adult females and 518 subadults comprising the Central American population of leatherback turtles. With an estimated Mexican population of 1,000 adults and 750 subadults (by Spotila *et al*., 2000), the entire east Pacific leatherback population has been estimated by Spotila *et al.* (2000) to contain approximately 2,955 females (1,687 adults and 1,268 subadults); however, insufficient foundation was given for these estimates (i.e. derivation of estimates are unclear, and models rely on theoretical assumptions that need further evaluation and testing).

Based on aerial surveys and ground censuses during the 2000-2001 season and using an estimated clutch frequency of 5.8, Sarti *et al*. (in preparation) estimated the total number of female leatherbacks (nesters only) in the eastern Pacific:

> (a) primary beaches in Mexico - 396 females;
> (b) total Mexico (without primary beaches) - 452 females;
> (c) Central America (including data from Costa Rica) - 751 females; and (d) grand total - 1,599 females.

Western Pacific Nesting Populations of Leatherback Turtles

Similar to their eastern Pacific counterparts, leatherback turtles originating from the western Pacific are also threatened by poaching of eggs, killing of nesting females, human encroachment on nesting beaches, incidental capture in fishing gear, beach erosion, and egg predation by animals. Little is known about the status of the western Pacific leatherback nesting populations but once major leatherback nesting assemblages are declining along the coasts of Malaysia, Indonesia and the Solomon Islands. Low density and scattered nesting of leatherback turtles occurs in Fiji, Thailand, and Australia (primarily western and to a lesser extent, eastern).

Malaysia

The decline of leatherback turtles is severe at one of the most significant nesting sites in the western Pacific region - Terengganu, Malaysia, with current nesting representing less than 2 percent of the levels recorded in the 1950s, and the decline is continuing. The nesting population at this location has declined from 3,103 females estimated nesting in 1968 to 2 nesting females in 1994 (Chan and Liew, 1996; Table IV-10). With one or two females reportedly nesting each year, this population has essentially been eradicated (P. Dutton, personal communication, 2000). Years of excessive egg harvest, egg poaching, the direct harvest of adults in this area, as well as incidental capture in various fisheries in territorial and international waters, have impacted the Malaysian population of leatherback turtles. There were two periods in which there were sharp declines in nesting leatherback turtles at this location: 1972-74 and 1978-80. Between 1972 and 1974, the number of females nesting declined 21% and coincided with a period of rapid development in the fishing industry, particularly trawling, in Terengganu (Chan *et al*., 1988 *in* Chan and Liew, 1996). Between 1978 and 1980, nestings dropped an average of 31% annually, and coincided directly with the introduction of the Japanese high seas squid fishery of the North Pacific in 1978 (Yatsu *et al*., 1991, *in* Chan and Liew, 1996). Because tagged individuals from Rantau Abang have been recovered from as far away as Taiwan, Japan and Hawaii, this fishery, as well as fisheries operating within the South China Sea, may have impacted the Malaysian leatherback population (Chan and Liew, 1996). After 1980, rates of decline averaged 16% annually, suggesting continuing threats from fisheries (Chan and Liew, 1996).

Table IV-10. **Number of nesting females per year in Terengganu, Malaysia (summarized in Spotilla, *et al*., 1996)**

1968	1970	1972	1974	1976	1978	1980	1984	1987	1988	1993	1994
3,103	1,760	2,926	1,377	1,067	600	200	100	84	62	20	2

In the 1960s, the leatherback turtles nesting on the beaches in Terengganu represented one of the larger remaining nesting aggregations for this species in the Pacific Ocean. Since then, the population has declined to a handful of individual, nesting females. Although the implications of such a decline are readily apparent and should require no further analyses, we evaluated the census data for this population using the density-independent form of the Dennis model (Dennis et al. 1991) to assess the probable trend for this population (see Table IV-11 for results).

Table IV-11. Results of an assessment of the Terengganu nesting aggregation of leatherback sea turtles using a discrete-time, density-independent diffusion estimation model

Demographic Parameter	**Estimate**
Mean log growth rate (µ)	-0.229501
Upper 95% confidence interval	0.302985
Lower 95% confidence interval	-0.761988
Variance in mean log growth rate (ó^{2})	0.776462
Upper 95% confidence interval	2.115806
Lower 95% confidence interval	0.399266
Finite rate of population increase (ë)	1.172021
Upper 95% confidence interval	2.060818

Lower 95% confidence interval	0.666547

The results of our analyses of the number of female leatherback turtles that nest at the Terengganu supports the conclusion that the population's growth rate has been, on average, negative (ë = 0.97, which is less than 1, and µ = -0.282579), a conclusion that is supported by a casual observation of the counts. Projecting these results over 25-, 50-, and 100-year intervals suggest that this population has a very high risk of extinction (declining to 1 or 0 females) in the short-term (less than a decade) if no action is taken. As we have discussed previously, different nesting aggregations of sea turtles are effectively isolated from one another, the female leatherback turtles nesting at Terengganu will not be "rescued" by migrants from other nesting beaches. If this nesting aggregation becomes extinct, it will remain extinct.

Indonesia

In Indonesia, leatherbacks have been protected since 1978 and low density nesting occurs along western Sumatra (200 females nesting annually) and in southeastern Java (50 females nesting annually), although the last known information is from the early 1980s (*in* Suarez and Starbird, 1996a; Dermawan, 2002). However the largest leatherback rookery can be found on the north coast of Papua, and information on population status and trends are reviewed extensively below.

Leatherback nesting generally takes place on two major beaches, located 30 km apart, on the north Vogelkop coast of the State of Papua: Jamursba-Medi (18 km) and War-Mon beach (4.5 km) (Starbird and Suarez, 1994). In 1984, the World Wildlife Fund (WWF) began a preliminary study to assess the status of the leatherback nesting population and found at least an estimated 13,000 nests on Jamursba Medi. A subsequent survey undertaken in 1992 reported a decline of nesting levels to 25% of the 1984 levels (Table III-9). A near total collection of eggs during this time period may have contributed to this decline. Commercial exploitation of turtle eggs on this beach was intense for a long time; for example, during 1984-1985, four to five fishermen boats were observed visiting the beach weekly and returning with 10,000 - 15,000 eggs per boat (Hitipeuw, 2003a). Out of concern for the rapid declines in nestings, the WWF proposed the designation of five beaches as protected areas - Sauapor (14 km), Wewe-Kwoor (20 km), Jamursba-Medi (28 km), Sidei-Wibain (18 km) and Mubrani-Kaironi (20 km). These beaches are monitored for leatherback nesting activities and patrolled for potential poaching activities (Hitipeuw and Maturbongs, 2002).

Leatherbacks nest on Jamursba-Medi during April through September, with a peak in July and August (Suarez *et al*., in press). A summary of data collected from leatherback nesting surveys from 1981 to 2003 for Jamursba-Medi has been compiled, re-analyzed, and standardized and is shown in Table IV-12 (Hitipeuw and Maturbongs, 2002; Hitipeuw, 2003b). The number of nests were adjusted to correct for the days or months of the survey missed during the nesting season, and the average number of nests per female is assumed to range between 4.4 to 5.8 (see footnotes in Table IV-12). Gaps in the data for the year 1998 and 2000 were due to lack of financial support and transition of management changes of WWF Indonesia, which has been helping to monitor the leatherback nesting populations at these beaches since the early 1980s.

Table IV-12. **Estimated numbers of female leatherback turtles nesting on Jamursba-Medi Beach, along the north coast of the State of Papua (Summarized by Hitipeuw and Maturbongs, 2002 and Hitipeuw, 2003b)**

Survey Period	# of Nests	Adjusted # Nests	Estimated # of Females[3]
Jamursba-Medi Beach:			
September, 1981	4,000+	7,143[1]	1,232 - 1,623
April - Oct. 1984	13,360	13,360	2,303 - 3,036
April - Oct. 1985	3,000	3,000	658 - 731
June - Sept. 1993	3,247	4,091[2]	705 - 930
June - Sept. 1994	3,298	4,155[2]	716 - 944
June - Sept. 1995	3,382	4,228[2]	729 - 961
June - Sept., 1996	5,058	6,373[2]	1,099 - 1,448
May - Aug., 1997	4,001	4,481[4]	773 - 1,018
May - Sept. 1999	2,983	3,251	560 - 739
April - Dec., 2000	2,264	No	390 - 514
March - Oct., 2001	3,056	No	527 - 695
March - Aug., 2002	1,865	1,921	331 - 437
March - July, 2003 (ongoing)	2,109	2,459	424 - 559

[1] The total number of nests reported during aerial surveys were adjusted to account for loss of nests prior to the survey. Based on data from other surveys on Jamursba-Medi, on average 44% of all nests are lost by the end of August.

[2] The total number of nests have been adjusted based on data from Bhaskar's surveys from 1984-85 from which it was determined that 26% of the total number of nests laid during the season (4/1-10/1) are laid between April and May.

[3] Based on Bhaskar's tagging data, an average number of nests laid by leatherback turtles on Jamursba-Medi in 1985 was 4.4 nests per female. This is consistent with estimates for the average number of nests by leatherback turtles during a season on beaches in Pacific Mexico, which range from 4.4 to 5.8 nests per female (Sarti *et al.*, unpub. report). The range of the number of females is estimated using these data.

[4] Number adjusted from Bhaskar (1984), where percentage of nests laid in April and September is 9% and 3%, respectively, of the total nests laid during the season.

Suarez *et al*. (in press) has also compiled information on the estimated number of nests lost due to both natural and anthropogenic causes. For example, during 1984 and 1985, on Jamursba-Medi, 40-60%

of nests were lost to inundation and erosion, while 90% of those nests not taken by poachers[7] or by the sea were destroyed by feral pigs (*Sus scrofa*). Eggs from poached nests were commercially harvested for sale in the Sarong markets until 1993, when the beaches first received protection by the Indonesian government (J. Bakarbessy, personal communication, *in* Suarez and Starbird, 1996a). During the 1993-96 seasons, environmental education activities in nearby villages and protection measures on this same beach were put into place, with unreported results. Again, approximately 90% of those nests not taken by poachers or the sea[8] were destroyed by pigs (Suarez *et al.* in press). In addition to natural erosion, logging activity in the area also threatens the nesting beach habitat. Current nearby logging activities include lumber harvest and transportation and the construction of a log pond and base camp. Such activities may remove vegetation, change drainage patterns and increase human presence, which may also increase poaching of eggs. Logs washed up on the beach may impair females coming ashore to nest and hatchlings from reaching the ocean (Hitipeuw, 2003a).

Nesting of leatherbacks on War-Mon beach takes place during October through February, with a peak in December (Suárez *et al*., in press). Recently, the beach was monitored during the nesting season and documented 1,442 nests (Hitipeuw, 2003b), which may equate to several hundred females (249-328 females, given 4.4 to 5.8 nests per female). Given shorter monitoring periods in past studies, it is difficult to analyze any trends for this nesting beach (see Table IV-13).

Table IV-13. Number of leatherback turtle nests observed along War-Mon Beach

Monitoring Period	# nests	Source
Nov. 23-Dec. 20, 1984 and Jan. 1-24, 1985	1,012	Starbird and Suárez, 1994; Suárez *et al*., in press
Dec. 6-22, 1993	406	Starbird and Suárez, 1994; Suárez *et al*., in press
Dec. 2002 - May, 2003	1,442	Hitipeuw, 2003b

Egg poaching for subsistence on War-Mon beach accounted for over 60% of total nest loss during 1993-94, and total loss of nests due to pig predation was 40% (because there are more people in this region, there is more pig hunting; hence less pig predation of leatherback eggs (Starbird and Suárez, 1994)). In 2001 and 2002, conservation measures have reduced predation of eggs by pigs (P. Dutton, NOAA Fisheries, personal communication, October 2002), and coastal patrols are currently being conducted to prevent disturbance and exploitation of the beach (Hitipeuw, 2003b).

The leatherback turtles nesting on the beaches in the State of Papua represent one of the largest remaining nesting aggregations for this species in the Pacific Ocean. The nesting aggregation appears to

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[7] Suarez, *et al*. (in press) provided no information on the estimated percentage of nests lost to poachers.

[8] No information on percentage of nests lost to poachers or the sea were given, except that it was "noted."

be relatively large and has fluctuated between 400 and 1,000 individuals throughout most of the 1990s and early 2000s and could suggest that the population is stable or slightly increasing. However, we evaluated the census data for this population using the density-independent form of the Dennis model (Dennis et al. 1991) to better assess the probable trend for this population (see Table IV-14 for results).

Table IV-14. Results of an assessment of the Jamursba-Medi nesting aggregation of leatherback sea turtles using a discrete-time, density-independent diffusion estimation model

Demographic Parameter	**Lower Census Estimate**	**Upper Census Estimate**
Mean log growth rate (µ)	-0.048485	-0.048439
Upper 95% confidence interval	0.174896	0.194803
Lower 95% confidence interval	-0.271865	-0.291680
Variance in mean log growth rate ($ó^2$)	0.226610	0.268697
Upper 95% confidence interval	0.653270	0.774597
Lower 95% confidence interval	0.113718	0.134839
Finite rate of population increase (ë)	1.066967	1.089708
Upper 95% confidence interval	1.327670	1.387804
Lower 95% confidence interval	0.8574557	0.855642

The results of our analyses of the lower and upper estimates of the number of female leatherback turtles that nest at the Jamursba-Medi suggest that the population's growth rate is, on average, positive (ë = 1.06 to 1.09, which are both greater than 1), but the confidence intervals around these estimates suggest that the population may, in fact, be declining (0.857 to 0.856, which are both less than 1). The mean of the population's log growth rate (µ = -0.0484) supports this conclusion: most population trajectories are declining slightly. Projecting these results over 25-, 50-, and 100-year intervals suggest that this population has a low risk of extinction (declining to 1or 0 females), but the population has a high risk of declining to 100 or 50 females. Our assessment suggests that this population has a 50 percent probability of declining to 100 females in about 30 years or 50 females within 40 years. At these smaller population sizes, this nesting aggregation would have an increased risk of extinction from stochastic events like changes in the ratio of males to females, the probability of an adult female dying before giving birth, or difficulties in finding mates.

Given the current, serious threats to all life stages of the Indonesian leatherback populations, these forecasts are not surprising. As human populations in Indonesia increase, the need for meat and competition between the expanding human population and turtles for space increases, all leading to more direct takes of leatherback turtles or incidental take by local fisheries. There is no evidence to indicate that the threats discussed earlier in this narrative are not continuing today, as problems with nest destruction by feral pigs, beach erosion, and harvest of adults in local waters have been reported (Suarez et al., unpublished report). This forecast is also consistent with the observations of local Indonesian villagers who have reported dramatic declines in local sea turtle populations (Suarez, 1999) and agrees with Suarez *et al*. (in press) who, when writing about the Papuan population of nesting leatherback turtles, concluded that "Given the high nest loss which has occurred along this coast for over thirty years it is not unlikely that this population may also suddenly collapse. Nesting activity must also continue to be monitored along this coast, and nest mortality must be minimized in order to prevent this population of leatherback turtles from declining in the future." Without adequate protection of nesting beaches, emerging hatchlings, and adults, this population will continue to decline.

Papua New Guinea

In Papua New Guinea, leatherbacks nest primarily along the coast of the Morobe Province, mostly between November and March, with a peak of nesting in December. There are no current estimates of the number of nesting females in this area, but researchers are analyzing all known data to determine status and trends[9]. Based on data from surveys conducted during the 1980s, researchers estimated that between 200-300 females were estimated to nest annually in an area between the two villages of Labu Tali and Busama (approximately 19 kilometers along the Morobe Province; Quinn and Kojis (1985) and Bedding and Lockhart (1989), both *in* Hirth *et al*., 1993). While leatherback meat is not consumed in this area, leatherback eggs are an important source of protein for the local people, and eggs are also sold in towns such as Lae. In addition, when rivers break through a berm in the area, leatherback eggs are exposed and destroyed by inundation (Hirth *et al*., 1993). Egg collection continues in this country, although the extent is unknown (P. Dutton, NOAA Fisheries, personal communication, March, 2002) but "significant" (M. Philip, Office of Environment and Conservation, Papua New Guinea, personal communication, December, 2003). The Kamiali nesting beach (also in the Morobe Province and within the Kamiali Wildlife Management Area) is approximately 11 km long and is an important nesting area for leatherbacks. Currently, Kamiali contains approximately 150 nesting females producing 500-600 clutches per season. Due to increasing awareness and concern about the local declines in nesting leatherbacks, the Kamiali community agreed to a 100 meter no-take zone in 1999, increased to a 1 km no-take zone in 2000, and 0.5 km was added in 2001 (1.5 km total). The no-take zone is effective from December to February (nesting season). Although very few adults are killed, 99% of the eggs are collected outside of the no-take zone (Philip, 2002).

In January, 2004, NOAA Fisheries plans to conduct aerial surveys to locate nesting areas and assess the significance of this area for leatherback nesting females (S. Benson, NOAA Fisheries-SWFSC, personal communication, December, 2003).

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[9] Philip (2002) reports an estimated 1,000 to 1,500 females nesting (very approximate) along the Morobe coast between Labu Butu and Busama beach, but without an ongoing monitoring project in place, these numbers are very speculative and probably should not be used until a full study and analysis has been conducted. Researchers are currently analyzing the data to determine a trend, but so far there has not been a comprehensive analysis.

Solomon Islands

In the Solomon Islands, the rookery size is estimated to be less than 100 females nesting per year (D. Broderick, personal communication, *in* Dutton, *et al*., 1999). Past studies have identified four important nesting beaches in Isabel Province: Sasakolo, Lithoghahira, Lilika, and Katova. While Leary and Laumani (1989 *in* Ramohia *et al*., 2001) reported that leatherback nesting throughout Isabel Province doubled since 1980, there have been few monitoring studies to substantiate this reported trend. From November 28, 2000 through January 21, 2001, a monitoring study was conducted on one of the nesting beaches, located on Sasakolo Beach. This period represented approximately two-thirds of the known peak-breeding season. During this time, leatherbacks appeared 192 times, with 132 clutches laid. A total of 27 nesting turtles were encountered: 26 were new nesting individuals and 1 had been tagged in 1995. Egg harvest by humans has been reported in the past. In addition, lizards and iguanas have been documented predating on leatherback eggs (Rahomia, *et al*., 2001).

Fiji

In Fiji, leatherbacks are uncommon, although there are recorded sightings and 4 documented nesting attempts on Fijian beaches. They have been seen in the Savusavu region, Qoma, Yaro passage, Vatulele and Tailevu, and researchers estimate approximately 20-30 individual leatherbacks in Fijian waters (Rupeni, *et al*., 2002).

Australia

In Australia, leatherback nesting is sporadic, less than 5 per year, generally outside of Great Barrier Reef in southeast Queensland. Human related threats are listed as: incidental capture in fisheries and ingestion and entanglement in marine debris (Dobbs, 2002).

Conclusion on Status of Eastern and Western Pacific leatherback turtles

Although quantitative data on human-caused mortality are scarce, the available information suggests that leatherback mortality on many nesting beaches remains at unsustainable levels (Tillman, 2000). Published assessments of the extinction risks of leatherback turtles in the Pacific Ocean have concluded that these turtles have a very high risk of disappearing from the Pacific Ocean within one or two human generations (Spotila et al. 1996, 2002). Our assessments of three nesting aggregations support this conclusion: if no action is taken to reverse their decline, leatherback turtles nesting in the Pacific Ocean either have high risks of extinction in a single human generation (for example, nesting aggregations at Terrenganu and Costa Rica) or they have a high risk of declining to levels where more precipitous declines become almost certain (for example Jamursba-Medi). As we have discussed previously, different nesting aggregations of sea turtles are effectively isolated from one another, the female leatherback turtles nesting at this different beaches will not be “rescued” by migrants from other nesting beaches. If a nesting aggregation becomes extinct, it will remain extinct.

Although these assessments have focused on adult, female leatherback turtles, as the extinction of the

Dusky seaside sparrow (*Ammodramus maritimus nigrescens*) demonstrated, no animal population will survive for more than a generation without adult females. That species became functionally extinct when the last female in the population produced only male young, then died. The species' final extinction occurred when the last male died in 1987.

Except for elimination of fishing mortality in the now-defunct high-seas driftnet fisheries in the North and South Pacific, and reductions of effort in a few other fisheries (e.g. the Hawaii-based longline fishery and the CA/OR drift gillnet fishery), many of the risks that caused these populations to decline remain. Leatherback turtles still experience harvests of their eggs, they are still killed for subsistence purposes, their beaches continue to erode, and adult and sub-adult leatherback turtles are still captured and killed in fisheries interactions. The dire predictions of sea level rise and associated increases in beach erosion and subsidence present new risks to this declining species. Finally, the small sizes of leatherback turtle populations places this species at high risk of extinction in the Pacific; with such small population sizes, each pre-mature death of an adult or sub-adult turtles reduces the number of breeding adults in the population over time. As the number of breeding adults declines, the number of eggs each generation produces declines and reduces the population's ability to recover.

Conservation efforts during the last few years at nesting beaches in Mexico and Costa Rica have led to increased survival of eggs, and therefore greater hatchling production per nesting female. This has the potential for increasing future recruitment if post-hatchling survival is not further reduced; however, since numbers of nests are so low, and post-hatchling and juvenile natural mortality are assumed to be high, this increase in hatchling production may only result in the addition of a few adults annually. In western Pacific populations, particularly Papua, nest destruction by beach erosion and feral pig predation is widespread, and hatchling production is likely to be low relative to the numbers of nests laid. Overall, both eastern and western Pacific populations appear to have low female abundance as a result of legal harvest of eggs and nesting females, poaching, and incidental take in fisheries. Representation in the various age classes of female leatherback turtles is most likely unbalanced as a result of losses of adult females, juveniles and eggs and sub-adults and adults as a result of on-going fisheries and the now-defunct high seas driftnet fisheries. Gaps in age structure may cause sudden collapse of nesting populations when age classes with few individuals recruit into the reproductive population as older individuals die or are removed.

Leatherbacks have been observed taken by the ETP purse seine fishery; however, genetic data was not collected from any turtles in this fishery. Genetic analysis of samples taken from two leatherbacks incidentally captured in the CA/OR drift gillnet fishery revealed that they both originated from western Pacific nesting beaches (i.e. Indonesia/Solomon Islands, Malaysia). Similarly, all three leatherbacks taken in the California-based longline fishery were found to originate from western Pacific nesting beaches (P. Dutton, personal communication, 2003). Because a leatherback taken in the Hawaii-based longline fishery is thought to originate from an eastern Pacific nesting beach, the north Pacific Ocean may be a foraging area for leatherbacks from both sides of the Pacific. As Dutton *et al*. (2000) note, the predominance of western Pacific turtles may be an artifact of small sample size or may reflect the relative abundance of the two subpopulations. However, recent data from satellite-tagged leatherbacks at nesting beaches in the eastern Pacific indicate that these turtles migrated into the southern Pacific Ocean after leaving the beach, whereas leatherbacks tagged of Monterey, California and nesting beaches in the western Pacific all traveled within the northern Pacific. Further, even turtles tagged on western Pacific beaches showed differences in migratory behavior, with leatherbacks tagged in Papua New Guinea migrating into the southern hemisphere and leatherbacks tagged at Jarmusba-Medi migrating in the northern Pacific. As a result, fisheries occurring in the northern Pacific Ocean may encounter very few leatherbacks from eastern Pacific beaches or some western Pacific beaches.

3. Loggerhead Turtles

a. Global Status

The loggerhead turtle is listed as threatened under the ESA throughout its range, primarily due to direct take, incidental capture in various fisheries, and the alteration and destruction of its habitat.

b. Physical Description

The loggerhead is characterized by a reddish brown, bony carapace, with a comparatively large head, up to 25 cm wide in some adults. They usually have five pairs of costal scutes, and three inframarginals without pores. Adult males have comparatively narrow shells, gradually tapering posteriorly, and long thick tales, extending well beyond the edge of the carapace. Adults typically weigh between 80 and 150 kg, with average CCL measurements for adult females worldwide between 95-100 cm CCL (*in* Dodd, 1988) and adult males in Australia averaging around 97 cm CCL (Limpus, 1985, *in* Eckert, 1993). Juveniles found off California and Mexico measured between 20 and 80 cm (average 60 cm) in length (Bartlett, 1989, *in* Eckert, 1993). Skeletochronological age estimates and growth rates were derived from small loggerheads caught in the Pacific high-seas driftnet fishery. Loggerheads less than 20 cm were estimated to be 3 years or less, while those greater than 36 cm were estimated to be 6 years or more. Age-specific growth rates for the first 10 years were estimated to be 4.2 cm/year (Zug, *et al*., 1995).

c. General Distribution

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 (*in* NOAA Fisheries and USFWS, 1998d).

Loggerheads can be divided into five regions: the Atlantic Ocean, Pacific Ocean, Indian Ocean, Caribbean Sea and Mediterranean Sea. These regions may be further divided into nesting aggregations. In the Pacific Ocean, loggerhead turtles are represented by a northwestern Pacific nesting aggregation (located in Japan) which may be comprised of separate nesting groups (Hatase, *et* *al.*, 2002) and a smaller southwestern nesting aggregation that occurs in Australia (Great Barrier Reef and Queensland), New Caledonia, New Zealand, Indonesia, and Papua New Guinea. In the western Atlantic Ocean, NOAA Fisheries recognizes five major nesting aggregations: (1) a northern nesting aggregation that occurs from North Carolina to northeast Florida, about 29° N; (2) a south Florida nesting aggregation, occurring from 29° N on the east coast to Sarasota on the west coast; (3) a Florida panhandle nesting aggregation, occurring at Eglin Air Force Base and the beaches near Panama City, Florida; (4) a Yucatán nesting aggregation, occurring on the eastern Yucatán Peninsula, Mexico; and (5) a Dry Tortugas nesting subpopulation, occurring in the islands of the Dry Tortugas, near Key West, Florida (NOAA Fisheries SEFSC, 2001). In addition, Atlantic and Caribbean nesting aggregations are found in Honduras, Colombia, Panama, the Bahamas, and Cuba. In the Mediterranean Sea, nesting aggregations in Greece, Turkey, Israel, Italy, and several other sites have been recorded. One of the largest loggerhead nesting aggregations in the world is found in Oman, in the Indian Ocean.

d. Life Cycle and Population Dynamics

Figure IV-6 illustrates the basic life cycle of the loggerhead turtle (based on data presented by Crouse, *et. al.* (1987) for females of the western Atlantic nesting aggregations). This cycle is broken into seven life stages based on age: (1) egg/hatchling; (2) small juveniles (pelagic); (3) large juveniles (benthic); (4) subadults; (5) novice breeders; (6) first year remigrants; (7) and mature breeders, each with their own expected survival rate (Table IV-15). Arrows along the bottom represent the probability of each ageclass surviving and remaining in the ageclass. Arrows between each ageclass represent the probability of the ageclass surviving and growing to the next ageclass, and the arrows along the top represent the ageclass-specific fertility. The thickness or length of the lines do not indicate the level of probability or fecundity. Available information on the behavior, physiology, and biological requirements of these stages is summarized below.

Figure IV-6. Life-cycle graph of the loggerhead turtle (Crouse *et. al.,* 1987)

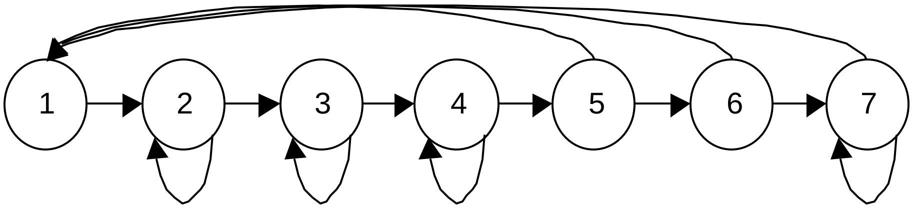

Table IV-15: Stage Structure and Vital Rates for Loggerhead Turtles (Crouse, *et. al.*, 1987 as modified by Bolten 2003 and Heppell et al. 2003)

Stage	Name	Size	Stable Stage Structure	Survival Probability (lx)	Fecundity (eggs/female)

1	Egg-Hatchling	0	20.66%	0.6747	0
2	Oceanic Juvenile	5-45 cm	66.97%	0.745 (0.875)	0
3	Small Neritic Juvenile	45-72 cm	11.46%	0.6758 (0.7)	0
4	Large Neritic Juvenile	72-92 cm	0.66%	0.7425 (0.8)	0
5	Breeding Adult	>92 cm	0.04%	0.8091 (0.85)	127
6	Non-nesting Females	-	0.03%	0.8091 (0.85)	4
7	(Mature Breeder)	-	0.18%	0.8091	80

Numerical analyses of the survival rates, transition rates, and fecundities in Table III-11 indicated that the modeled loggerhead turtle population has a finite population growth rate (ë) of approximately 0.95, which suggests a population that is declining at a rate of approximately 5 percent per year. The stage structure of this population is atypical for long-lived species with delayed maturity[10] and may reflect the effects of various human activities. For example, the survival rates of stage 1 individuals (eggs and hatchlings) probably reflect the combined effects of habitat degradation, feral and wild predators, and beach erosion (NRC 1990). The survival rates of the benthic stages, including adults returning to breed, probably reflect the effects of incidental capture in coastal fisheries and habitat degradation (NRC 1990). By contrast, the survival rate of pelagic juveniles (stage 2) would not have been affected by human activities on land or in coastal waters, which is why a higher proportion of individuals are in that stage. However, this population structure is probably an appropriate surrogate of other declining loggerhead populations because of its atypical structure.

Elasticity analyses for the stage matrix of this population support these general conclusions (Table IV-16). The survival rates of adult loggerhead turtles have the highest elasticities, which is typical for long-lived species with delayed maturity. However, the survival rates of pelagic juveniles had the second highest elasticities, which is atypical but, as we discussed previously, may be an artifact of a population whose structure has been modified by various human activities and natural phenomena. These results suggest that changes in the survival rates of mature, reproductive, adults and pelagic juveniles would

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[10] The typical population structure for these species has the largest proportion of individuals and the highest mortality rates in the earliest stages; with proportions and mortality declining through successive stages and the smallest proportion of the total population in the adult stages, which also have the lowest mortality rates. For further discussion of these population structures, see Stearns (1992) and Crouse (1999).

have the largest proportional effect on this population's trend; increasing those survival rates would help the population recover from its decline, while decreasing those survival rates would exacerbate the population's rate of decline.

Table IV-16. Stage elasticities

Stage	Survival Rate	Transition Rate	Fecundity
1	0	0.051	0
2	0.1851	0.051	0
3	0.1186	0.051	0
4	0.1384	0.051	0
5	0	0.039	0.0120
6	0	0.039	0.0003
7	0.2298	0	0.0386

In contrast to the rates provided in Crouse, *et al.* (1987; Table IV-16), Chaloupka and Limpus (2002) reported higher survival rates for immature (benthic juvenile and sub-adult) and adult loggerhead turtles at one large coral reef in the southern Great Barrier Reef (Table IV-17). Although this population is exposed to a high risk of incidental capture in coastal Australia otter-trawl fisheries (Chaloupka and Limpus 2002), these higher survival rates are more representative of the expected dynamics of a long-lived, delayed maturity species.

Table IV-17. Expected age-class-specific survival probability estimates for southern Great Barrier Reef loggerhead turtles (Chaloupka and Limpus 2002)

Age Class	Survival (Mean)	Survival (95% CI)
Immature[1]	0.8588	0.828-0.885
Adult	0.8749	0.835-.0906

[1] Immature turtles in Chaloupka and Limpus (2001) correspond to stages 3 and 4 of the Crouse model (benthic juveniles and sub-adults).

Based on past observations, the loggerhead turtles that are captured and killed in Hawaii-based longline fisheries are primarily pelagic, juvenile loggerhead turtles (see the discussion in the *Effects of the Action* section). Over the history of these fisheries, the effect of these annual deaths would significantly reduce the survival rates of individuals in this stage in the nesting aggregations that interact with these fisheries. From our analyses, these reductions would be expected to have a significant, adverse affect on the trend of those nesting aggregations, particularly if these losses are added to losses in other life stages. The combined effect of these activities, which affect most or all life stages of most loggerhead turtle populations, would significantly reduce the growth rates of the nesting aggregations that interact with these fisheries.

e. Biological Characteristics

Diet

For their first years of life, loggerheads forage in open ocean pelagic habitats. Both juvenile and subadult loggerheads feed on pelagic crustaceans, mollusks, fish, and algae. The large aggregations of juveniles off Baja California have been observed foraging on dense concentrations of the pelagic red crab, *Pleuronocodes planipes* (Pitman, 1990; Nichols, *et al*., 2000). A high percentage of loggerheads sampled off Baja California Sur have had exclusively pelagic red crab in their stomachs, revealing the importance of this area and this prey species for loggerheads (Peckham and Nichols, 2003). Similarly, examinations of the gut contents of 70 loggerheads stranded off North Africa revealed a large presence of bentho-pelagic crab, *Polybius henslowii* during all seasons. Loggerheads in this area are found coincident with the high abundance of crabs during spring and summer (Ocaña and García, 2003).

Data collected from stomach samples of turtles captured in North Pacific driftnets indicate a diet of gastropods (*Janthina* sp.), heteropods (*Carinaria* sp.), gooseneck barnacles (*Lepas* sp.), pelagic purple snails (*Janthina* sp.), medusae (*Vellela* sp.), and pyrosomas (tunicate zooids). Other common components include fish eggs, amphipods, and plastics (Parker, *et al*., in press). These loggerheads in the north Pacific are opportunistic feeders that target items floating at or near the surface, and if high densities of prey are present, they will actively forage at depth (Parker, *et al*., 2002). As they age, some loggerheads begin to move into shallower waters, where, as adults, they forage over a variety of benthic hard- and soft-bottom habitats (reviewed *in* Dodd, 1988).

Diving Behavior

Studies of loggerhead diving behavior indicate varying mean depths and surface intervals, depending on whether they were located in shallow coastal areas (short surface intervals) or in deeper, offshore areas (longer surface intervals). The maximum recorded dive depth for a post-nesting female was 211-233 meters, while mean dive depths for both a post-nesting female and a subadult were 9-22 meters. Routine dive times for a post-nesting female were between 15 and 30 minutes, and for a subadult, between 19 and 30 minutes (Sakamoto, *et al*., 1990 *in* Lutcavage and Lutz, 1997). Two loggerheads tagged by Hawaii-based longline observers in the North Pacific and attached with satellite-linked dive recorders were tracked for about 5 months. Analysis of the dive data indicate that most of the dives were very shallow - 70% of the dives were no deeper than 5 meters. In addition, the loggerheads spent approximately 40% of their time in the top meter and nearly all of their time in waters shallower than 100 meters. For only 5% of the days, the turtles dove deeper than 100 meters; the deepest daily dive recorded was 178 meters (Polovina *et al*., 2003). A recent study (Polovina *et al.*, 2004) found that tagged turtles spent 40 percent of their time at the surface and 90 percent of their time at depths shallower than 40 meters. On only five percent of recorded dive days loggerheads dove to depths greater than 100 meters at least once. In the areas that the loggerheads were diving, there was a shallow thermocline at 50 meters. There were also several strong surface temperature fronts the turtles were associated with, one of 20EC at 28EN and another of 17EC at 32EN.

Life History/Reproduction

For loggerheads, the transition from hatchling to young juvenile occurs in the open sea, and evidence from genetic analyses and tracking studies show that this part of the loggerhead life cycle involves trans-Pacific developmental migration. The size structure of loggerheads in coastal and nearshore waters of the eastern and western Pacific suggest that Pacific loggerheads have a pelagic stage similar to the Atlantic. This is supported by the fact that the high seas driftnet fishery, which operated in the Central North Pacific in the 1980s and early 1990s, incidentally caught juvenile loggerheads (mostly 40-70 cm in length) (Wetherall, *et al*., 1993). In addition, large aggregations (numbering in the thousands) of mainly juveniles and subadult loggerheads are found off the southwestern coast of Baja California, over 10,000 km from the nearest significant nesting beaches (Pitman, 1990; Nichols, *et al*., 2000). Genetic studies have shown these animals originate from Japanese nesting subpopulation (Bowen *et al*., 1995), and their presence reflects a migration pattern probably related to their feeding habits (Cruz, *et al*., 1991, *in* Eckert, 1993). While these loggerheads are primarily juveniles, carapace length measurements indicate that some of them are 10 years old or older. Dobbs (2002) reports that loggerheads off Australia recruit from the open ocean pelagic habitat at around 10 to 15 years of age, or approximately 78 cm in carapace length.

Based on skeletochronological and mark-recapture studies, mean age at sexual maturity for loggerheads ranges between 25 to 35 years of age, depending on the subpopulation (*in* Chaloupka and Musick, 1997). Dobbs (2002) reports that loggerheads originating from Australian beaches mature at around age 25, although Frazer *et al*. (1994 *in* NOAA Fisheries and USFWS, 1998d) determined that maturity of loggerheads in Australia occurs between 34.3 and 37.4 years of age.

Upon reaching maturity, adult female loggerheads migrate long distances from resident foraging grounds to their preferred nesting beaches. Clutch size averages 110 to 130 eggs, and one to six clutches of eggs are deposited during the nesting season (Dodd, 1988). The mean number of clutches deposited are 1.1 for females at Miyazaki, Japan, 2.06 for females at Yakushima Island, Japan (both *in* Schroeder *et al*., 2003), and 3.4 clutches per season estimated for loggerheads in eastern Australia (Limpus and Limpus, 2003). The average renesting interval for eastern Australian loggerheads is 14 days (Limpus and Limpus, 2003). The average re-migration interval is between 2.6 and 3.5 years (*in* NOAA Fisheries and USFWS, 1998d) (average 3.8 years for eastern Australian loggerheads (Limpus and Limpus, 2003)), and adults can breed up to 28 years (Dobbs, 2002). Nesting is preceded by offshore courting, and individuals return faithfully to the same nesting area over many years.

Migration

Loggerhead hatchlings on nesting beaches in Japan undertake developmental migrations in the North Pacific, using the Kuroshio and North Pacific Currents. Tagging programs to study migration and movement of sea turtles and genetic analyses provide evidence that loggerhead turtles undergo trans-

Pacific migrations and have been found foraging off Baja California. For example, loggerheads tagged in Mexico and California with flipper and/or satellite transmitters have been monitored returning to Japanese waters (Resendiz, *et al*., 1998a-b). In addition, genetic analyses of all loggerheads caught and sampled in the Hawaii-based and the west coast-based longline fishery indicated that all originated from Japanese nesting stock (P. Dutton, NOAA Fisheries, personal communication, December, 2003). Most loggerheads taken in the Hawaii-based longline fishery are non-adults, suggesting that loggerheads in the Pacific are pelagic until they become sexually mature, returning to nesting beaches and subsequently begin a benthic existence (Parker *et al*., 2003).

After reaching sexual maturity, female loggerheads exhibit precise natal homing and nearly all return to their nesting beach. Following nesting, females undertake seasonal breeding migrations between foraging grounds and the same nesting beach every few years (*in* Hatase, *et al*., 2002).[11]

Loggerheads originating from south Pacific nesting stocks have been documented foraging in the waters off southern Peru and northern Chile. Genetic analyses conducted on three specimens incidentally taken by Peruvian artisanal fisheries confirmed them to be loggerheads originating from Australian nesting stocks (Alfaro-Shigueto, *et al*., in press). In eastern Australia, nesting females have been documented migrating to feeding areas spread over a 2,600 kilometer radius throughout eastern and northern Australia, eastern Indonesia, Papua New Guinea, the Solomon Islands, and New Caledonia (Limpus and Limpus, 2003).

In the north Pacific Ocean, satellite telemetry studies show that loggerhead turtles tend to follow 17E and 20EC sea surface isotherms north of the Hawaiian Islands (Polovina, *et al*., 2000; Eckert, unpublished data). Relationships between other turtle species and sea surface temperatures have also been demonstrated, with most species preferring distinct thermal regimes (Stinson, 1984). After capture in the Hawaii-based longline fishery, six satellite transmitter-equipped loggerheads traveled westward along two convergent oceanic fronts, against prevailing currents and associated with a "cool" front characterized by sea surface temperature (17EC), surface chlorophyll and an eastward geostrophic current of about 4 centimeters/second (cm/sec). Three others were associated with a warmer front (20EC), lower chlorophyll levels, and an eastward geostrophic flow of about 7 cm/sec. This study supports a theory that fronts are important juvenile habitat (Polovina, *et al*., 2000).

Recent telemetry studies have described the oceanic habitat of loggerheads in more detail. Polovina *et al.* (2004) tagged 26 loggerheads captured in Hawaii-based longline fishery. All of these turtles came from Japanese nesting beaches. Three of the 26 loggerhead turtles tagged may have been sexually mature based on straight carapace lengths, the remainder with immature turtles. These turtles tended to migrate west following interactions. The turtles also shifted seasonally north and south between 28EN and 40EN. During January through June the loggerheads were found in the southern portion of this

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[11] For example, of 2,219 tagged nesting females, only 5 females relocated their nesting sites (0.2 percent) (Kamezaki, *et al*., 1997 *in* Hatase *et al*., 2002).

range, shifting to the northern end during July though December. The turtles also associated with areas with sea surface temperatures (SSTs) between 15E and 25E C. The loggerhead turtles were found in cooler waters during winter and spring, warmer waters in summertime.

Loggerhead turtles appear to utilize surface convergent forage habitat to capture their primary prey organisms which float along currents and congregate at fronts. Based on oceanographic conditions, the loggerheads were associated with fronts, eddies, and geostrophic currents (Polovina *et al.* 2004). The turtles moved with the seasonal movements of the Transition Zone Chlorophyll Front (TZCF), although they tended to remain south of the front itself, and were found along the southeastern edge of the Kuroshio Extension Current (KEC) and the northern edge of the Subtropical Gyre. The TZCF and KEC appear to be important forage habitat for loggerhead turtles as these areas contain colder, plankton-rich waters. The study indicates that loggerheads may spend months at the edge of eddies in these areas. As this area has also been found to be an important foraging habitat for juvenile bluefin tuna (Ingake *et al.* 2001 *in* Polovina *et al.* 2004), overlaps between fisheries targeting these fish and others with similar habitat associations are likely to also encounter loggerhead sea turtles.

f. Population status and trends

Based on genetic analyses conducted at nesting sites, there are five distinct subpopulations of loggerheads in the western Atlantic: (1) a northern nesting subpopulation that occurs from North Carolina to northeast Florida, about 29º N (approximately 7,500 nests in 1998); (2) a south Florida nesting subpopulation, occurring from 29º N on the east coast to Sarasota, Florida on the west coast (approximately 83,400 nests in 1998); (3) a Florida panhandle nesting subpopulation, occurring at Eglin Air Force Base and the beaches near Panama City, Florida (approximately 1,200 nests in 1998); (4) a Yucatán nesting subpopulation, occurring on the eastern Yucatán Peninsula, Mexico (TEWG, 2000); and (5) a Dry Tortugas nesting subpopulation, occurring in the islands of the Dry Tortugas, near Key West, Florida (approximately 200 nests per year) (NOAA Fisheries SEFSC, 2001). The status of the northern population based on the number of loggerhead nests has been classified as stable or declining (TEWG, 2000). Although nesting data from 1990 to the present for the northern loggerhead subpopulation suggests that nests have been increasing annually (2.8 - 2.9%) (NOAA Fisheries SEFSC, 2001), there are confidence intervals about these estimates that include no growth[12]. Adding to concerns for the long-term stability of the northern subpopulation, genetics data has shown that, unlike the much larger south Florida subpopulation which produces predominantly females (80%), the northern subpopulation produces predominantly males (65%; NOAA Fisheries SEFSC 2001).

The diversity of the loggerheads' life history renders them susceptible to many natural and human

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[12] Meta-analyses conducted by NOAA Fisheries' Southeast Fisheries Science Center to produce these estimates were unweighted analyses and did not consider a beach's relative contribution to the total nesting activity of a subpopulation. Consequently, the results of these analyses must be interpreted with caution.

impacts, including impacts while they are on land and in the ocean, including both the benthic and the pelagic environment. Hurricanes are particularly destructive to sea turtle nests. Sand accretion and rainfall that result from these storms as well as wave action can appreciably reduce hatchling success. For example, in 1992, all of the eggs over a 90-mile length of coastal Florida were destroyed by storm surges on beaches that were closest to the eye of Hurricane Andrew (Milton *et al.,* 1994). Other sources of natural mortality include cold stunning and biotoxin exposure. Anthropogenic factors that impact hatchlings and adult female turtles on land, or the success of nesting and hatching include: beach erosion, beach armoring and nourishment; artificial lighting; beach cleaning; increased human presence; recreational beach equipment; beach driving; coastal construction and fishing piers; exotic dune and beach vegetation; and poaching. An increased human presence at some nesting beaches or close to nesting beaches has led to secondary threats such as the introduction of exotic fire ants, feral hogs, dogs, and an increased presence of native species (*e.g.,* raccoons, armadillos, and opossums), which raid and feed on turtle eggs. Although sea turtle nesting beaches are protected along large expanses of the northwest Atlantic coast, other areas along these coasts have limited or no protection. Sea turtle nesting and hatching success on unprotected high density east Florida nesting beaches from Indian River to Broward County are affected by all of the above threats (NOAA Fisheries SEFSC, 2001). Loggerhead turtles are affected by a completely different set of anthropogenic threats in the marine environment. These include oil and gas exploration, coastal development, and transportation; marine pollution; underwater explosions; hopper dredging, offshore artificial lighting; power plant entrainment and/or impingement; entanglement in debris; ingestion of marine debris; marina and dock construction and operation; boat collisions; poaching, and fishery interactions. In the pelagic environment, loggerheads are exposed to a series of longline fisheries that include the U.S. Atlantic tuna and swordfish longline fisheries, an Azorean longline fleet, a Spanish longline fleet, and various fleets in the Mediterranean Sea (Aguilar *et al.,* 1995, Bolten *et al.,* 1994, Crouse, 1999). In the benthic environment in waters off the coastal U.S., loggerheads are exposed to a suite of fisheries in federal and state waters including trawl, purse seine, hook and line, gillnet, pound net, longline, and trap fisheries.

g. Distribution and Abundance of Nesting Females in the Pacific Ocean

In the Pacific Ocean, loggerhead turtles are represented by a northwestern Pacific nesting aggregation (located in Japan) and a smaller southwestern nesting aggregation that occurs in eastern Australia (Great Barrier Reef and Queensland) and New Caledonia (NOAA Fisheries SEFSC, 2001). There are no reported loggerhead nesting sites in the eastern or central Pacific Ocean basin.

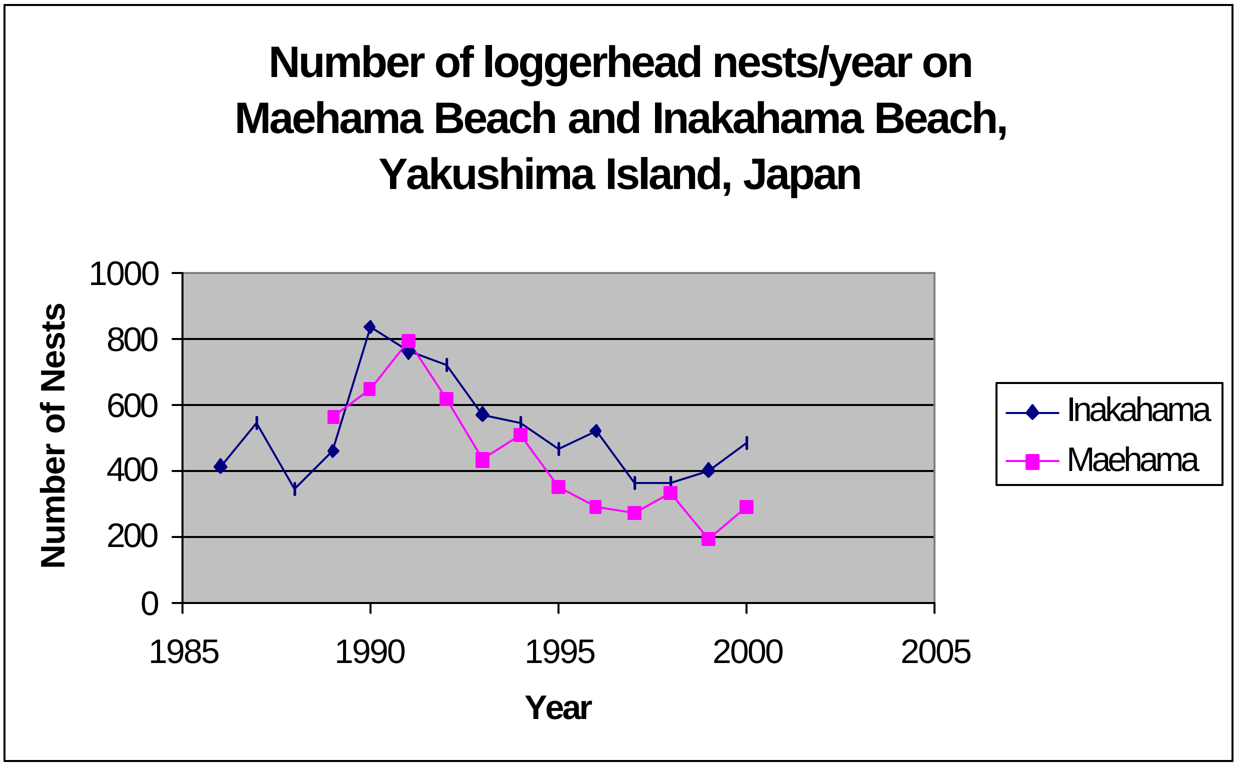
Japan

In the western Pacific, the only major nesting beaches are in the southern part of Japan (Dodd, 1988). Balazs and Wetherall (1991) speculated that 2,000 to 3,000 female loggerheads nested annually in all of Japan. From nesting data collected by the Sea Turtle Association of Japan since 1990, the latest estimates of nesting females on almost all of the rookeries are as follows: 1998 - 2,479 nests; 1999 - 2,255 nests; 2000 - 2,589 nests. Considering multiple nesting estimates, Kamezaki *et al*. (2003) estimates that approximately less than 1,000 female loggerheads return to Japanese beaches per nesting season.

In Japan, loggerheads nest on beaches across 13 degrees of latitude (24EN to 37EN), from the mainland island of Honshu south to the Yaeyama Islands, which appear to be the southernmost extent of loggerhead nesting in the western North Pacific. Researchers have separated 42 beaches into five geographic areas: (1) the Nansei Shoto Archipelago (Satsunan Islands and Ryukyu Islands); (2) Kyushu; (3) Shikoku; (4) the Kii Peninsula (Honshu); and (5) east-central Honshu and nearby islands. There are nine “major nesting beaches” (defined as beaches having at least 100 nests in one season within the last decade) and six “submajor nesting beaches” (defined as beaches having 10-100 nests in at least one season within the last decade), which contain approximately 75% of the total clutches deposited by loggerheads in Japan (Kamezaki *et al*., 2003).

Two of the most important beaches, Inakahama Beach and Maehama Beach, located on Yakushima Island in the Nansei Shoto Archipelago, account for approximately 30% of all loggerhead nesting in Japan. Monitoring on Inakahama Beach has taken place since 1985. Figure IV-7 shows the abundance and trend of loggerhead nests on these two beaches. Monitoring on some other nesting beaches has been ongoing since the 1950s, while other more remote beaches have been only recently monitored, since the 1990s. Sea turtle conservation and research is growing in Japan, resulting in more widespread beach summaries; however, there are limited reports describing the trends and status of loggerheads in this country (Kamezaki *et al*., 2003).

Figure IV-7. Abundance and Trend of Loggerhead Sea Turtle Nests on Yakushima Island.



According to the latest status and trend information, as reviewed in Kamezaki *et al*. (2003):

> "In the 1990s, there has been a consistent decline in annual nesting, especially in Hiwasa Beach (89% decline) and Minabe (74% decline) [both of these are 2 of 9 major nesting beaches]. For most beaches, the lowest nesting numbers recorded have been during the recent period of 1997-1999.
>
> In the 1980s, there were increases in nesting numbers. However, nesting at the beginning of the 1980s was in most instances greater than nesting at the same beach some 20 years later at the end of the 1990s.
>
> There are indications that the 1970s was a period of approximate population stability with respect to breeding numbers.
>
> For the one population with census data extending back to the 1950s (Kamouda Beach) [one of 6 submajor nesting beaches], there is a clear indication that the population has greatly declined."

In general, during the last 50 years, loggerhead nesting populations have declined 50-90%. (Also see Figure 1 in Appendix B; N. Kamezaki, Sea Turtle Association of Japan, personal communication, August, 2001). Recent genetic analyses on female loggerheads nesting in Japan suggest that this "subpopulation" is comprised of genetically distinct nesting colonies (Hatase, *et al.*, 2002) with precise natal homing of individual females. As a result, Hatase, *et al.* (2002) indicate that loss of one of these colonies would decrease the genetic diversity of Japanese loggerheads; recolonization of the site would not be expected on an ecological time scale.

While loggerhead meat is generally not consumed by Japanese, except in some local communities, there has been a black market for sea turtle eggs. However, egg poaching has nearly disappeared due to conservation efforts and research throughout the country. As mentioned in the “Threats” section, coastal fisheries off Japan, particularly gillnets, poundnets, and intensive trawl fisheries for anchovies operating offshore of the major rookeries, may be impacting loggerhead populations. The Sea Turtle Association (2002) reports that approximately 80 mature loggerheads strand every year. This may be significant if they are pre- or post-nesting females. The most serious problem, however, may be a lack of nesting habitat due to beach erosion from upstream dams and dredging, and obstruction by sea walls. The extent of this impact has not been quantitatively studied to evaluate the impact to the loggerhead population (Kamezaki *et al.*, 2003).

Australia

In eastern Australia, Limpus and Riemer (1994) reported an estimated 3,500 loggerheads nesting annually in during the late 1970s. Since that time, there has been a substantial decline in nesting populations at all sites. Currently, less than 500 female loggerheads nest annually in eastern Australia, representing an 86% reduction within less than one generation (Limpus and Limpus, 2003).

Loggerheads originating from eastern Australia nest on nearly all beaches along the mainland and large barrier sand islands from South Stradbroke Island (27.6ES) northwards to Bustard Head (24.0ES) and islands of the Capricorn Bunker Group and Swain reefs in the southern Great Barrier Reef and on Bushy Island in the central Great Barrier Reef. Within this area, there are five major rookeries which account for approximately 70% of nesting loggerheads in eastern Australia.

Long-term census data has been collected at some rookeries since the late 1960s and early 1970s, and nearly all the data show marked declines in nesting populations since the mid-1980s (Limpus and Limpus, 2003). For example, in southern Great Barrier Reef waters, nesting loggerheads have declined approximately 8% per year since the mid-1980s (Heron Island), while the foraging ground population has declined 3% and were comprised of less than 40 adults by 1992. Researchers attribute the declines to perhaps recruitment failure due to fox predation of eggs in the 1960s and mortality of pelagic juveniles from incidental capture in longline fisheries since the 1970s (Chaloupka and Limpus, 2001). Wreck Island has seen a 70 to 90% decline over the last few decades. The decline of loggerheads in Australia can generally be attributed to incidental catch in trawl, net and drumline fisheries, boat strikes, ingestion/ entanglement of marine debris, and fox predation of mainland nests (Dobbs, 2002).

New Caledonia

Although loggerheads are the most common nesting sea turtle in the Île de Pins area of southern New

Caledonia, there is no quantitative information available, and surveys in the late 1990s failed to locate regular nesting. However, anecdotal information from locals indicate that there may be more substantial loggerhead nesting occurring on peripheral small coral cays offshore of the main island. Limpus and Limpus (2003) estimate that the annual nesting population in the Île de Pins area may be in the "tens or the low hundreds."

Other Countries

Scattered loggerhead nesting has also been reported on Papua New Guinea, New Zealand, Indonesia (NOAA Fisheries and USFWS, 1998d); however, Limpus and Limpus (2003) state that reports have not been confirmed, and in some cases, sea turtles species have been misidentified. The authors state that it is very unlikely for one to encounter nesting loggerheads north of Australia.

There are no records of nesting loggerheads in the Hawaiian Islands (Balazs, 1982), or in any of the islands of Guam, Palau, the Northern Mariana Islands (Thomas, 1989), the Federated States of Micronesia (Pritchard, 1982b), Fiji (Rupeni *et al*., 2002), or American Samoa (Tuato'o-Bartley, *et al*., 1993). In addition, loggerheads are not commonly found in U.S. Pacific coastal waters, and there has only been one documented stranding of a loggerhead in the Hawaiian Islands in the past 20 years (1982-2002 stranding data, G. Balazs, NOAA Fisheries, personal communication, 2002). There are very few records of loggerheads nesting on any of the many islands of the central Pacific, and the species is considered rare or vagrant on islands in this region (NOAA Fisheries and USFWS, 1998d).

Loggerhead mortality from human activities in the Pacific Ocean is not well-documented except for estimates based on NOAA Fisheries observer data in the Hawaii-based longline fishery, CA/OR drift gillnet fishery, and recent ongoing studies in Baja California, Mexico (Nichols, *et al*., 2000; Nichols, 2002). A high mortality in the North Pacific high-seas driftnet fisheries of Japan, Republic of Korea, and Taiwan was estimated in the 1980s and 1990s, but those fisheries no longer operate (Wetherall, *et al*., 1993). Mortality of loggerheads in the East China Sea and other benthic habitats of this population are a concern and thought to be "high," but have not been quantified (Kamezaki, personal communication, *in* Tillman, 2000).

Of the loggerheads taken in the California-based longline fishery and the CA/OR drift gillnet fishery, all were determined to have originated from Japanese nesting beaches, based on genetic analyses (P. Dutton, NOAA Fisheries, personal communication, December, 2003). Therefore, this fishery is impacting a subpopulation that consists of approximately 1,000 females nesting annually. Loggerheads taken in the ETP purse seine fishery have not been sampled for genetic data; however, because loggerheads originating from Japanese nesting beaches have been tracked to foraging areas off Baja California, Mexico, it is likely that any loggerheads taken in this area by purse seiners originated from Japan.

4. Olive Ridley Turtle

a. Global Status

Although the olive ridley is regarded as the most abundant sea turtle in the world, olive ridley nesting populations on the Pacific coast of Mexico are listed as endangered under the ESA; all other populations are listed as threatened.

b. Physical Description

Olive ridleys are the smallest living sea turtle, with an adult carapace length between 60 and 70 cm, and rarely weighing over 50 kg. They are olive or grayish green above, with a greenish white underpart, and adults are moderately sexually dimorphic (NOAA Fisheries and USFWS, 1998e). They have an unusually broad carapace, a medium-sized head that is triangular in planar view, five to nine pairs of costal scutes and four inframarginals with pores.

c. General Distribution

Olive ridley turtles occur throughout the world, primarily in tropical and sub-tropical waters. The species is divided into three main populations, with distributions in the Pacific Ocean, Indian Ocean, and Atlantic Ocean. Nesting aggregations in the Pacific Ocean are found in the Marianas Islands, Australia, Indonesia, Malaysia, and Japan (western Pacific), and Mexico, Costa Rica, Guatemala, and South America (eastern Pacific). In the Indian Ocean, nesting aggregations have been documented in Sri Lanka, east Africa, Madagascar, and there are very large aggregations in Orissa, India. In the Atlantic Ocean, nesting aggregations occur from Senegal to Zaire, Brazil, French Guiana, Suriname, Guyana, Trinidad, and Venezuela.

d. Life Cycle and Population Dynamics

Figure IV-8 illustrates the basic life cycle of the olive ridley turtle (based on general life history information presented by Marquez (1994) for the Kemp's ridley sea turtle, a surrogate species for the lesser known olive ridley turtle). This cycle is broken into six life stages: (1) egg/hatchling; (2) pelagic juvenile; (3) sub-adult; (4) neophyte breeder; (5) remigrant; and (6) adult. Arrows along the bottom represent the probability of each ageclass surviving and remaining in the ageclass. Arrows between each ageclass represent the probability of the ageclass surviving and growing to the next ageclass, and the arrows along the top represent the ageclass-specific fertility. The thickness or length of the lines do not indicate the level of probability or fecundity. Information on the life stage survival rates and fecundities of olive ridley turtles is sparse. Table IV-18 includes the available information on the Kemp's ridley turtle (*Lepidochelys kempi*). Available information on the behavior, physiology, and biological requirements of the olive ridley turtle is summarized below.

Figure IV-8. **Life-cycle graph of the olive ridley**

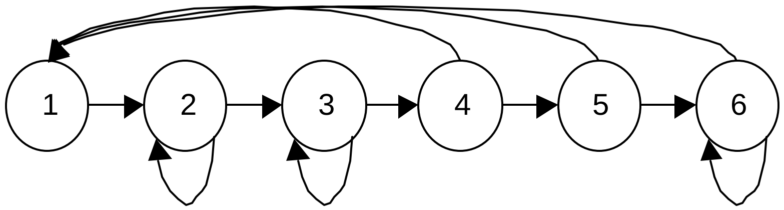


Table IV-14: **Stage structure and vital rates for olive ridleys (based on Marquez (1981, 1994) data for Kemp's ridley turtles)**

Stage	Name	Age	Survival Probability (lx)	Fecundity
1	Egg-hatchling	0	0.609	0
2	Pelagic juvenile	1	0.565	0
3	Sub-adult	na	0.445	0
4	Neophyte	8	0.421	80
5	Remigrant	9	0.421	84
6	Mature breeder	10-	0.421 (0.48)[1]	81

[1] Marquez *et al.* (1982a, *in* Chaloupka and Limpus 2002) report a survival rate of 0.48 for adult female olive ridley turtles.

Data on the transition rates between life stages are unavailable; olive ridleys spend most of their life in the pelagic environment which makes studies of their abundance, life history and ecology, and pelagic distribution difficult. As a result, we were unable to analyze the stage structure of this population to estimate its finite population growth rate (ë) or the elasticities of the various life stages. The typical population structure for long-lived, late-maturing species like olive ridley turtles has the largest proportion of individuals and the highest mortality rates in the earliest stages; proportions and mortality decline through successive stages with the smallest proportion of the total population in the adult stages, which also have the lowest mortality rates. For further discussion of these population structures, see Crouse (1999) and Stearns (1992).

The dynamics of some olive ridley turtle populations today are certain to reflect the effects of the various anthropogenic activities which have caused or exacerbated the declines in abundance noted in some olive ridley nesting aggregations, such as those in areas of India, Malaysia, Costa Rica, and Guatemala. As a result, the ë and life stage elasticities of these populations are likely indicative of declining populations (ës less than 1, and changed proportional importance of different life stage elasticities on ë). For an example of the changed dynamics of a declining sea turtle population, see the *Life Cycle and Population Dynamics* discussion for loggerhead turtles above. In a disturbed population the survival rates of adult turtles may still have the highest elasticities, typical for long-lived species with delayed maturity. However, the survival rates of life stages relatively undisturbed by chronic or significant sources of mortality increase in importance as the population relies upon these stages to supply enough individuals to survive the rigors of subsequent life stages and reach sexual maturity. In the case of a population where the survival of all life stages has been decreased by anthropogenic activities, stage elasticities may change such that the proportional effect of a change in survival rate in any stage can have significant effect on the population's growth rate.

Based on past observations, the olive ridley turtles that are captured and killed in Pacific Ocean longline fisheries are primarily sub-adults and adults (see the discussion in the *Effects of the Action* section). As a result, olive ridley nesting aggregations affected by the HMS FMP fisheries could experience declines in adult and sub-adult life stage survival rates, with a corresponding proportional effect on the growth rate of that aggregation. Depending on the magnitude of the change in survival rates and ë, some of these aggregations could slow their rate of increase, begin to decline, or increase the rate of their decline.

e. Biological Characteristics

Diet

Olive ridleys feed on tunicates, salps, crustaceans, other invertebrates and small fish. Montenegro *et al*. 1986 (*in* NOAA Fisheries and USFWS, 1998e) found a wide variety of prey in olive ridleys from the eastern Pacific. Adult males fed primarily on fishes (57%), salps (38%), crustaceans (2%) and molluscs (2%), while adult females fed primarily on salps (58%), and a lesser degree on fishes (13%), molluscs (11%), algae (6%), crustaceans (6%), bryozoans, sea squirts, sipunculid worms and fish eggs (all individually less than 1%). Similar to loggerheads, olive ridleys off western Baja California may feed exclusively on pelagic red crabs (Marquez, 1990 *in* NOAA Fisheries and USFWS, 1998e).

Dive Behavior

Olive ridleys have been caught in trawls at depths of 80-110 meters (NOAA Fisheries and USFWS, 1998e), and a post-nesting female reportedly dove to a maximum depth of 290 meters. The average dive length for an adult female and adult male is reported to be 54.3 and 28.5 minutes, respectively (Plotkin, 1994, *in* Lutcavage and Lutz, 1997).

The most common prey of olive ridley turtles are salps and pyrosomes, similar to leatherback turtles. These prey organisms occur sub-surface and migrate within the water column as part of the deep scattering layer. As a result, olive ridley turtles tend to dive deeper, spending 20 percent of the time at the surface and 40 percent of their time at depths greater than 40 meters. On 25 percent of the recorded dive days, olive ridley turtles dove to depths greater than 150 meters at least once (Polovina *et al.*, 2004). Daily dives of 200 meters have been observed, and one dive was recorded at 254 meters (Polovina *et al*., 2003). The dive habitat of the tagged olive ridley turtles had a deep thermocline at 100 meters and minimal horizontal surface temperature fronts (Polovina *et al.*, 2004).

In 1999 eight olive ridley turtles (4 adult females, 3 adult males, and 1 juvenile) were tagged using satellite-linked dive recorders during a research survey in the eastern tropical Pacific Ocean. Sixty percent of the dives were of two minutes or less in duration. The average of the longest dive time for females was 120-180 minutes, 75 minutes for males, and 45-60 minutes for the one juvenile. A diurnal dive behavior was seen where most turtles spent more time near the surface during daylight hours, which were between 9 a.m. to 2 p.m., between 22-56% (mean of 37%) of the total dive time was spent near the surface during this 6-hour period. Female olive ridleys in this study spent significantly more time at 40 to 80 meters than did the males, and the thermocline is an important foraging area for the olive ridley as both male and female turtles spent a significant amount of time in the region of the thermocline. Mated females and males did not make dives greater than 150 meters, while a non-mated pelagic male and female both made dives greater than 150 meters, with a number of dives over 250 meters (Parker *et al*., 2003).

Life History/Reproduction

Olive ridleys are famous for their synchronized mass nesting emergences, a phenomenon commonly known as "arribadas." While arribadas occur only on a few beaches world-wide, the olive ridley's nesting range is far-reaching and is also comprised of solitary nesters. Thus, there are two clearly distinct reproductive behaviors within the species - some females are solitary nesters, while others are arribada nesters (Plotkin and Bernardo, 2003).

Olive ridley turtles begin to aggregate near the nesting beach two months before the nesting season, and most mating is generally assumed to occur in the vicinity of the nesting beaches, although copulating pairs have been reported over 100 km from the nearest nesting beach. Olive ridleys are considered to reach sexual maturity between 8 and 10 years of age, and approximately 3 percent of the number of hatchlings recruit to the reproductive population (Marquez, 1982 *in* Salazar, *et al*., 1998). The mean clutch size for females nesting on Mexican beaches is 105.3 eggs, in Costa Rica, clutch size averages between 100 and 107 eggs (*in* NOAA Fisheries and USFWS, 1998e). Research shows that arribada nesters produced larger clutches than solitary nesters, perhaps to offset the large number of predators near the arribada sites (Plotkin and Bernardo, 2003). Females generally lay 1.6 clutches of eggs per season in Mexico (Salazar, *et al*., 1998) and two clutches of eggs per season in Costa Rica (Eckert, 1993). Arribada nesters have high site fidelity and remain near the nesting beach during the internesting period and are relatively inactive (Plotkin and Bernardo, 2003). Solitary nesters appear to have low site fidelity (Kalb, 1999 *in* Plotkin and Bernardo, 2003). Data on the remigration intervals of olive ridleys in the eastern Pacific are scarce; however, in the western Pacific (Orissa, India), females showed an annual mean remigration interval of 1.1 years. Reproductive span in females of this area was shown to be up to 21 years (Pandav and Kar, 2000).

Migration

Like leatherback turtles, most olive ridley turtles lead a primarily pelagic existence (Plotkin *et al*., 1993), migrating throughout the Pacific, from their nesting grounds in Mexico and Central America to the north Pacific. While olive ridleys generally have a tropical to subtropical range, with a distribution from Baja California, Mexico to Chile (Silva-Batiz, *et al*., 1996), individuals do occasionally venture north, some as far as the Gulf of Alaska (Hodge and Wing, 2000). Surprisingly little is known of their oceanic distribution and critical foraging areas, despite being the most populous of north Pacific sea turtles. They appear to occupy a series of foraging areas geographically distributed over a very broad range within their oceanic habitat (Plotkin, *et al*., 1994).

Little is also known about the habitat of the juvenile olive ridleys, primarily because there have been few observations. While adult olive ridleys are the most abundant and widely distributed in the eastern tropical Pacific, no juveniles were seen during several years of observations (Pitman, 1990 *in* Juárez-Cerón and Sarti-Martínez, 2003). It has been hypothesized that depending on food sources, the distribution of juveniles may be similar to that of adults. Young olive ridleys may move offshore and occupy areas of surface current convergences to find food and shelter among aggregated floating objects until they are large enough to recruit to benthic feeding grounds of the adults. During four surveys carried out between Socorro Island of the Revillagigedo Archipelago and Bahia de Manzanillo between November 1999 and December 2000, researchers observed a number of juvenile olive ridleys (11), measuring around 29 cm CCL. All were found close together, and almost always in pairs. All were in a pelagic environment, characterized by deep water (land was not visible and there was no algae accumulation; Juárez-Cerón and Sarti-Martínez, 2003).

In the eastern Pacific Ocean, adult olive ridleys are found in warm, tropical waters, bounded on the north by the California Current and on the south by the Humboldt Current. There are few observations of olive ridleys west of 140EW. Olive ridleys appear to forage throughout the eastern tropical Pacific Ocean, often in large groups, or flotillas, and are occasionally found entangled in scraps of net or other floating debris. In a three year study of communities associated with floating objects in the eastern tropical Pacific, Arenas and Hall (1992) found sea turtles present in 15 percent of observations and suggested that flotsam may provide the turtles with food, shelter, and/or orientation cues in an otherwise featureless landscape. Olive ridleys comprised the vast majority (75%) of these sea turtle sightings. Small crabs, barnacles and other marine life often reside on the debris and likely serve as food attractants to turtles.

During seven research cruises conducted in the eastern tropical Pacific from 1989 to 2000, researchers opportunistically captured olive ridleys and recorded environmental information surrounding the capture location. This included distance to land, water depth, sea surface temperature and currents. Analyses of the data revealed high numbers of adults distributed on the continental shelf and slope (near major nesting beaches), next to the Pacific trench in upwelling regions. Adults were frequently found in shallow waters, with peak numbers between 0 and 1,000 meters. Juveniles were more often found in deeper waters (off the continental shelf; Kopitsky *et al*. 2003).

The post-nesting migration routes of olive ridleys tracked via satellite from Costa Rica traversed thousands of kilometers of deep oceanic waters, ranging from Mexico to Peru, and more than 3,000 kilometers out into the central Pacific (Plotkin, *et al*., 1993).

Tagging data from Orissa, India shows that olive ridleys that nest there migrate to southern Tamil Nadu and Sri Lanka during the non-breeding season. Four olive ridleys nesting in Orissa were outfitted with satellite transmitters and tracked. Three turtles moved in large circles off the coast and northern Andhra Pradesh, while one turtle swam south towards Sri Lanka, swimming 1,000 kilometers in 18 days. All turtles averaged about 25 to 30 kilometers per day (Shanker *et al*., 2003a).

Olive ridley turtles from both eastern and western Pacific nesting beaches were tagged in the Hawaii-based longline fishery (Polovina *et al.*, 2004). Two of the 10 olive ridleys may have been sexually mature based on straight carapace lengths, the remainder were immature turtles. These turtles migrated in areas between 8 and 31EN, with SSTs of 23E to 28EC (primarily in areas with SSTs of 24E or 27EC). Throughout the year, the olive ridley turtles had a less distinct pattern of distribution than loggerhead turtles tagged in this fishery. For example, olive ridley turtles were seen in the southern portion of their preferred range between October and December. Between April and September, the turtles were found between 14E and 28EN, but not in the area between 20E and 24E N. This middle area is where olive ridley turtles were most frequently found during January through March. The data was not separated by nesting beach origin, however, so some of these patterns may also be attributable to the different habitat associations between eastern and western Pacific olive ridley turtles.

Interestingly, olive ridley turtles from the east and west Pacific had different habitat associations. Western Pacific olive ridley turtles associated with major ocean currents, such as the southern edge of the KEC, the North Equatorial Current (NEC) and the Equatorial Countercurrent (ECC). Olive ridley turtles from the eastern Pacific were not associated with strong current systems, most of these turtles remained within the center of the Subtropical Gyre. These waters are warm, vertically stratified with deep thermoclines, and do not have strong surface temperature or chlorophyll gradients. Olive ridley turtles of either nesting aggregation origin were not associated with strong surface chlorophyll fronts. However, olive ridleys from the western Pacific were found in habitat characterized by wind-induced upwelling and shoaling of the thermocline, which may allow olive ridley turtles to forage more shallowly in these areas. Polovina *et al.* (2004) theorize that these conditions may provide an energetic advantage to turtles migrating across the Pacific to nesting beaches.

f. Population status and trends

As mentioned, the Mexican nesting population of olive ridley is listed as endangered, while all other populations of olive ridleys are listed as threatened. Since its listing in 1978, there has been a decline in abundance of this species, and it has been recommended that the olive ridley for the western Atlantic be reclassified as endangered. This is based on continued direct and incidental take of olive ridleys, particularly in shrimp trawl nets. Since 1967, the western North Atlantic (Surinam and adjacent areas) nesting population has declined more than 80 percent. In general, anthropogenic activities have negatively affected each life stage of the olive ridley turtle populations, resulting in the observed declines in abundance of some olive ridley turtle nesting aggregations. Other aggregations, however, have experienced significant increases in abundance in recent years, often as a result of decreased adult and egg harvest pressure, indicating populations in which the birth rates are now exceeding death rates.

Declines in olive ridley populations have been documented in Playa Nancite, Costa Rica; however, other nesting populations along the Pacific coast of Mexico and Costa Rica appear to be stable or increasing, after an initial large decline due to harvesting of adults. Historically, an estimated 10 million olive ridleys inhabited the waters in the eastern Pacific off Mexico (Cliffton, *et al*., 1982 *in* NOAA Fisheries and USFWS, 1998e). However, human-induced mortality led to declines in this population. Beginning in the 1960s, and lasting over the next 15 years, several million adult olive ridleys were harvested by Mexico for commercial trade with Europe and Japan. (NOAA Fisheries and USFWS, 1998e). Although olive ridley meat is palatable, it was not widely sought after; its eggs, however, are considered a delicacy, and egg harvest can certainly be considered one of the major causes for its decline. Fisheries for olive ridley turtles were also established in Ecuador during the 1960s and 1970s to supply Europe with leather (Green and Ortiz-Crespo, 1982).

In the Indian Ocean, Gahirmatha supports perhaps the largest nesting population; however, this population continues to be threatened by nearshore trawl fisheries. Direct harvest of adults and eggs, incidental capture in commercial fisheries, and loss of nesting habits are the main threats to the olive ridley’s recovery.

g. Distribution and Abundance of Nesting Females in the Pacific Ocean

Eastern Pacific Ocean

In the eastern Pacific Ocean, nesting occurs all along the Mexican and Central American coast, with large nesting aggregations occurring at a few select beaches located in Mexico and Costa Rica. Few turtles nest as far north as southern Baja California, Mexico (Fritts, *et al*., 1982) or as far south as Peru (Brown and Brown, 1982). As mentioned previously, where population densities are high enough, nesting takes place in synchronized aggregations known as arribadas. The largest known arribadas in the eastern Pacific are off the coast of Costa Rica (~475,000 - 650,000 females estimated nesting annually) and in southern Mexico (~800,000+ nests/year at La Escobilla, in Oaxaca (Millán, 2000).

Mexico

The nationwide ban on commercial harvest of sea turtles in Mexico, enacted in 1990, has improved the situation for the olive ridley. Surveys of important olive ridley nesting beaches in Mexico indicate increasing numbers of nesting females in recent years (Marquez, *et al*., 1995; Arenas, *et al*., 2000). Annual nesting at the principal beach, Escobilla Beach, Oaxaca, Mexico, averaged 138,000 nests prior to the ban, and since the ban on harvest in 1990, annual nesting has increased to an average of 525,000 nests (Salazar, *et al*., in press). At a smaller olive ridley nesting beach in central Mexico, Playon de Mismalayo, nest and egg protection efforts have resulted in more hatchlings, but the population is still "seriously decremented and is threatened with extinction" (Silva-Batiz, *et al*., 1996). Still, there is some discussion in Mexico that the species should be considered recovered (Arenas, *et al*., 2000).

Costa Rica

In Costa Rica, 25,000 to 50,000 olive ridleys nest at Playa Nancite and 450,000 to 600,000 turtles nest at Playa Ostional each year (NOAA Fisheries and USFWS, 1998e). In an 11-year review of the nesting at Playa Ostional, (Ballestero, *et al*., 2000) report that the data on numbers of nests deposited is too limited for a statistically valid determination of a trend; however, there does appear to be a six-year decrease in the number of nesting turtles. Under a management plan, the community of Ostional is allowed to harvest a portion of eggs. Between 1988 and 1997, the average egg harvest from January to May ranged between 6.7 and 36%, and from June through December, the average harvest ranged from 5.4 to 20.9% (Ballestero, *et al*., 2000). At Playa Nancite, concern has been raised about the vulnerability of offshore aggregations of reproductive individuals to "trawlers, longliners, turtle fishermen, collisions with boats, and the rapidly developing tourist industry" (Kalb, *et al*., 1996). The greatest single cause of olive ridley egg loss comes from the nesting activity of conspecifics on *arribada* beaches, where nesting turtles destroy eggs by inadvertently digging up previously laid nests or causing them to become contaminated by bacteria and other pathogens from rotting nests nearby. At a nesting site in Costa Rica, an estimated 0.2 percent of 11.5 million eggs laid during a single *arribada* produced hatchlings (*in* NOAA Fisheries and USFWS, 1998e). In addition, some female olive ridleys nesting in Costa Rica have been found afflicted with the fibropapilloma disease (Aguirre, *et al*., 1999).

Guatemala

In Guatemala, the number of nesting olive ridleys nesting along their Pacific coast has declined by 34% between 1981 and 1997. This is only based on two studies conducted 16 years apart, however: in 1981, the estimated production of olive ridley eggs was 6,320,000, while in 1997, only 4,300,000 eggs were estimated laid (*in* Muccio, 1998). Villagers also report a decline in sea turtles; where collectors used to collect 2-3 nests per night during the nesting season 15 years prior, now collectors may find only 2-4 nests per year due to fewer turtles and more competition. This decline most certainly can be attributed to the collection of nearly 95% of eggs laid, and the incidental capture of adults in commercial fisheries (Muccio, 1998).

Nicaragua

In Nicaragua, there are two primary *arribada* beaches: Playa La Flor and Playa Chacocente, both in the southern Department of Rivas. At Playa La Flor, the second most important nesting beach for olive ridleys on Nicaragua, Ruiz (1994) documented 6 *arribadas* (defined as 50 or more females nesting simultaneously). The main egg predators were domestic dogs and vultures (*Coragyps atratus* and *Cathartes aura*). During the largest *arribada*, 12,960 females nested from October 13-18, 1994 at Playa La Flor (*in* NOAA Fisheries and USFWS, 1998e). Von Mutius and Berghe (2002) reported that management of this beach includes a six-month open season for egg collection, during a time when the *arribadas* is small. During this time, all eggs are taken by locals, and during the "closed period," approximately 10-20% of eggs are given to the locals to consume or sell. At Playa Chacocente, approximately 5,000 to 20,000 females may nest over the course of five days (Camacho y Cáceres, 1994, *in* Arauz, 2002). Here, the harvest and commercialization of sea turtle eggs is allowed and somewhat controlled. During a monitoring project conducted on nearby Playa El Mogote from October, 2001 through March, 2002, researchers documented olive ridleys nesting 327 times. Of these, 99.7% of the nests were poached (Arauz, 2002).

Indian Ocean

In the eastern Indian Ocean, olive ridleys nest on the east coast of India, Sri Lanka, and Bangladesh.

India

In India, a few thousand olive ridleys nest in northern Tamil Nadu, Andhra Pradesh, and the Andaman and Nicobar Islands (*in* Shanker *et al*., 2003b). However, the largest nesting aggregation of olive ridleys in the world occurs in the Indian Ocean along the northeast coast of India (Orissa). Not surprisingly then, olive ridleys are the most common sea turtle species found along the east coast of India, migrating every winter to nest en-masse at three major rookeries in the state of Orissa: Gahirmatha, Devi River mouth, and Rushikulya (Shanker *et al*., 2003b). Sporadic nesting occurs between these mass nesting beaches.

The Gahirmatha rookery, located along the northern coast of Orissa, hosts the largest known nesting concentration of olive ridleys. Shanker *et al*. (2003b) provide a comprehensive report on the status and trends of olive ridleys nesting in Orissa since monitoring began in 1975. Table IV-19 shows the estimated number of olive ridleys nesting at Gahirmatha in the <u>largest</u> arribada during a season. No estimates are available for arribadas at the Devi River mouth and Rushikulya. Current population sizes are estimated to be between 150-200,000 nesting females per year. Based on analyses of the data, while there has been no drastic decline in the nesting population at Gahirmatha in the last 25 years, there are differences in trends between decades. For example, trend analyses suggest stability or increase in the size of the 1980s arribadas, which may be due to enforcement of legislation in the late 1970s, stopping the directed take of turtles. However, the 1990s data show that the population is declining or on the verge of a decline, which may be consistent with the recent increase in fishery related mortality and other threats (see below). No arribadas occurred on this nesting beach in 1997, 1998, and 2002, which is the highest documented incidence of failure since this rookery has been monitored (Shanker *et al*., 2003b).

Table IV-19. Consensus estimates for nesting populations in Gahirmatha, derived from multiple sources (Source: Shanker *et al*., 2003b)

Year	Consensus estimate[1]	Comments
1975-1976	158,000	
1976-1977	150,000	
1977-1978	150,000	1 arribada
1978-1979	133,000	1 arribada
1979-1980	218,000	1 arribada
1980-1981	191,000	1 arribada
1981-1982	0	no arribada
1982-1983	200,000	2 arribadas
1983-1984	300,000	2 arribadas
1984-1985	280,000	2 arribadas
1985-1986	50,000	1 arribada
1986-1987	386,000	2 arribadas
1987-1988	0	no arribada
1988-1989	300,000	1 arribada
1989-1990	200,000	1 arribada
1990-1991	350,000	2 arribadas
1991-1992	320,000	2 arribadas
1992-1993	350,000	?
1993-1994	350,000	2 arribadas
1994-1995	340,000	?
1995-1996	200,000	1 arribada
1996-1997	0	no arribada
1997-1998	0	no arribada
1998-1999	180,000	1 arribada
1999-2000	?	1 arribada
2000-2001	?	1 arribada
2001-2002	0	no arribada

[1] Estimates refer to the largest arribada during a nesting season, usually the first.

Uncontrolled mechanized fishing in areas of high sea turtle concentration, primarily illegally operated trawl fisheries, has resulted in large scale mortality of adults during the last two decades. Records of stranded sea turtles have been kept since 1993. Since that time, over 90,000 strandings (mortalities) of olive ridleys have been documented (*in* Shanker *et al*., 2003b), and much of it is believed to be due to illegal gillnet and shrimp trawl fishing in the offshore waters. Fishing in coastal waters off Gahirmatha was restricted in 1993 and completely banned in 1997 with the formation of a marine sanctuary around the rookery. Marine turtles in Orissa are protected by a prohibition of all mechanized fishing within 5 km of the coast and within 20 km of the Gahirmatha coast (~35 km). Despite these rules, mortality due to shrimp trawling reached a record high of 13,575 ridleys during the 1997-98 season, and none of the approximately 3,000 trawlers operating off the Orissa coast use turtle excluder devices in their nets (Pandav and Choudhury, 1999), despite mandatory requirements passed in 1997. “Operation Kachhapa” was developed in the late 1990s to protect sea turtles and their habitat by enabling strict enforcement of the 5 km non-mechanized fishing zone limit, as well as putting forward efforts to monitor nestings and educate local inhabitants and fishermen (Shanker and Mohanty, 1999). However, shrimp boats continue to fish close to shore within this protected zone and continue to not use turtle excluder devices. Current mortality rates are estimated to be ~15,000 turtles per year (B. Mohanty, personal communication, *in* Shanker *et al*., 2003b). Threats to these sea turtles also include artificial illumination from coastal development and unsuitable beach conditions, including reduction in beach width due to erosion (Pandav and Choudhury, 1999).

Genetic studies indicate that olive ridleys originating from the east coast of India are distinct from other ridleys worldwide, increasing the conservation importance of this particular population (Shanker *et al*., 2000 *in* Shanker *et al*., 2003b).

Western Pacific Ocean

In the western Pacific, olive ridleys are not as well documented as in the eastern Pacific, nor do they appear to be recovering as well. There are a few sightings of olive ridleys from Japan, but no report of egg-laying. Similarly, there are no nesting records from China, Korea, the Philippines, or Taiwan. No information is available from Vietnam or Kampuchea (*in* Eckert, 1993).

Indonesia

Indonesia and its associated waters also provides habitat for olive ridleys, and there are some recently documented nesting sites. The main nesting areas are located in Sumatra, Alas Purwo in East Java, Paloh-West Kalimantan and Nusa Tenggara. On Jamursba-Medi beach, on the northern coast of Papua, 77 olive ridley nests were documented from May to October, 1999 (Teguh, 2000 *in* Putrawidjaja, 2000). However, as mentioned in the leatherback subsection, extensive hunting and egg collection, in addition to rapid rural and urban development, have reduced nesting activities in this area. In Jayapura Bay, olive ridleys were often seen feeding, and in June, 1999, an estimated several hundred ridleys were observed nesting on Hamadi beach, despite heavy human population in the nearby area. Locals report daily trading and selling of sea turtles and their eggs in the local fish markets (Putrawidjaja, 2000). At Alas Purwo National Park, located at the eastern-most tip of East Java, olive ridley nesting was documented from 1992-96. Recorded nests were as follows: from September, 1993 to August, 1993, 101 nests; between March and October, 1995, 162 nests; and between April and June, 1996, 169 nests. From this limited data, no conclusions could be reached regarding population trends (Suwelo, 1999); however, recently, Dermawan (2002) reports that there were up to 250 females nesting at this site in 1996, with an increasing trend.

Malaysia

Olive ridleys nest on the eastern and western coasts of peninsular Malaysia; however, nesting has declined rapidly in the past decade. The highest density of nesting was reported to be in Terengganu, Malaysia, and at one time yielded 240,000 *eggs* (2,400 nests, with approximately 100 eggs per nest) (Siow and Moll, 1982, *in* Eckert, 1993)), while only 187 *nests* were reported from the area in 1990 (Eckert, 1993). In eastern Malaysia, olive ridleys nest very rarely in Sabah and in low numbers (Basintal, 2002), and only a few records are available from Sarak (*in* Eckert, 1993).

Thailand

In Thailand, olive ridleys occur along the southwest coast, on the Surin and Similan islands, and in the Andaman Sea. On Phra Thong Island, on the west coast of Thailand, the number of nesting turtles have declined markedly from 1979 to 1990. During a 1996-97 survey, only six olive ridley nests were recorded, and of these, half were poached, and one was predated by feral dogs. During the 1997-98 survey, only three nests were recorded. The main threats to turtles in Thailand include egg poaching, harvest and subsequent consumption or trade of adults or their parts (i.e. carapace), indirect capture in fishing gear, and loss of nesting beaches through development (Aureggi, *et al.*, 1999).

Central Pacific Ocean

There are no records of nesting on the unincorporated U.S. territories in the North Pacific. In the central Pacific, a single nesting was reported in September, 1985 on the island of Maui, Hawaii but the eggs did not hatch and the event was most likely an anomaly (Balazs and Hau, 1986 *in* NOAA Fisheries and USFWS, 1998e). In October 2002, an olive ridley turtle was reported to have nested on the shores of Hilo Bay, on the Island of Hawaii. If confirmed upon hatching, this nesting event marks the second recorded nesting of an olive ridley in the main Hawaiian Islands.

Based on genetic analyses, an olive ridley taken in the CA/OR drift gillnet fishery originated from an eastern Pacific stock (i.e. Costa Rica or Mexico) (P. Dutton, NOAA Fisheries, personal communication, October 2002). The one olive ridley observed taken in the California-based longline fishery was found to originate from the eastern Pacific (P. Dutton, NOAA Fisheries personal communication, December, 2003). Although genetic analyses are not executed on olive ridleys taken in the ETP purse seine fishery, captured olive ridleys likely originate from eastern Pacific nesting beaches. Research cruises in the ETP collected information on sighted olive ridleys and genetic analyses determined those turtles originated from eastern Pacific nesting beaches. Recent genetic information analyzed from 39 olive ridleys taken in the Hawaii-based longline fishery, which fishes in similar areas as the west coast-based longline fleet, indicate that 74% of the turtles (n=29) originated from the eastern Pacific (Mexico and Costa Rica) and 26% of the turtles (n=10) were from the Indian and western Pacific rookeries (P. Dutton, NOAA Fisheries, personal communication, January, 2001), indicating the animals from both sides of the Pacific converge in the north Pacific pelagic environment.

C. Factors Affecting Listed Marine Mammals

Because marine mammal species in the Pacific Ocean are subject to different impacts, the fisheries and non-fisheries related threats to the listed marine mammals in the Pacific Ocean that are affected by the HMS fisheries proposed to be managed under the FMP are discussed for each species. All large whales are vulnerable to the effects of marine pollution. Marine pollution from sewage outfalls, dumping at sea, bilge cleaning, discarded trash or fishing gear, etc., could adversely impact the habitat of listed marine mammal species by having a negative effect on their prey, causing entanglement or disrupting the digestive system through ingestion of foreign materials (e.g. occlusion of the digestive tract).

1. Fin Whales

Historic Whaling

As early as the mid-seventeenth century, the Japanese were capturing fin, blue, and other large whales using a fairly primitive open-water netting technique (Tonnessen and Johnsen 1982, Cherfas 1989). In 1864, explosive harpoons and steam-powered catcher boats were introduced in Norway, allowing the large-scale exploitation of previously unobtainable whale species. The North Pacific and Antarctic whaling operations soon added this ‘modern’ equipment to their arsenal. After blue whales were depleted in most areas, the smaller fin whale became the focus of whaling operations and more than 700,000 fin whales were landed in the twentieth century.

Fisheries Interactions

Because little evidence of entanglement in fishing gear exists, and large whales such as the fin whale may often die later and drift far enough not to strand on land after such incidents, it is difficult to estimate the numbers of fin whales killed and injured by gear entanglement. In addition, the injury or mortality of large whales due to interactions or entanglements in fisheries may go unobserved because large whales swim away with a portion of the net or gear. Fishers have reported that large whales tend to swim through their nets without entangling and causing little damage to nets (Barlow *et al.* 1997).

In 1997, the eastern tropical Pacific tuna purse seine fishery accidentally killed “one unidentified baleen whale,” although there is no information available to determine whether the whale was a listed species (IATTC, 1999). However, since 1993, the fishery has had 100 percent observer coverage, and in over 100,000 sets, only one baleen whale has been killed. Therefore, the likelihood of this fishery taking a large listed baleen whale, such as a fin whale, is considered to be extremely low.

Ship Strikes

Because little evidence of ship strikes exists, and large whales such as the fin whale may often die later and drift far enough not to strand on land after such incidents, it is difficult to estimate the numbers of fin whales killed and injured by ship strikes. In addition, a boat owner may be unaware of the strike when it happens. If they do strand, they may not have obvious signs of trauma. Most of the evidence that has been gathered demonstrates that such events are rare occurrences (Heyning and Lewis, 1990; Barlow, *et al*., 1997). However, in 2002, off the Pacific Northwest coast, three fin whales arrived into ports on the bow of vessels; of those three, one has been confirmed to have been hit while alive. Results of examinations on the remaining two whales are pending (B. Norberg, NOAA Fisheries, personal communication, 2002).

Other Threats

No major habitat concerns have been identified for the fin whale, and there is no evidence that levels of organochlorines, organotins or heavy metals in baleen whales generally (including the fin whale) are high enough to cause toxic or other damaging effects (O'Shea and Brownell, 1995, *in* Reeves, *et al.*, 1998). However, there is a growing concern that the increasing levels of anthropogenic noise in the ocean may be a habitat concern for whales, particularly for whales that use low frequency sound to communicate, such as baleen whales (Forney *et al.*, 2000).

2. Humpback whale

Historic Whaling

Between 1919 and 1929, approximately 7,300 humpback whales were taken along the west coast of North America, and between 1947 and 1987, commercial whalers killed approximately 7,700 humpbacks in the North Pacific. Catches between 1910 and 1965 totaled 2,000 taken off Oregon and Washington, 3,400 taken off California, and 2,800 taken off Baja California, Mexico (*in* Carretta *et al*. 2001).

Fisheries Interactions

Between 1990 and 1997, no humpback whale deaths have been attributed to interactions with groundfish trawl, longline and pot fisheries in the Bering Sea/Aleutian Islands, and Gulf of Alaska (Hill and DeMaster 1999). Humpback whales have been injured or killed elsewhere along the mainland U.S. and Hawaii (Barlow *et al*. 1997). For instance, the California salmon troll fishery has been responsible for snagging one humpback whale in 1997. The animal swam away with the hook and many feet of trailing monofilament. In addition, the deaths of two humpbacks that stranded off southern California have been attributed to entanglement in fishing gear, and a humpback whale was observed off Ventura, California in 1993 with a section of netting wrapped around it and trailing gear. In 1999, a cow-calf pair was observed entangled in a net off Big Sur, California (*in* Carretta *et al*. 2001).

In 1991, a humpback whale was observed entangled in longline gear and released alive (Hill *et al.* 1997). This interaction occurred inside what is now the protected species zone of the islands and atolls of the Northwestern Hawaiian Islands. Another humpback whale was reported entangled in longline gear off Lanai by Nitta and Henderson (1993) and by whalewatch operators off Maui in 1993 (Hill and DeMaster, 1999). Confirmation was not made as to whether the gear type was pelagic longline gear, and the reports were believed to be for the same whale. In 1995, a humpback whale in Maui waters was found trailing numerous lines (not fishery-related) and entangled in mooring lines. The whale was successfully released, but subsequently stranded and was attacked and killed by tiger sharks in the surf zone. In 1996, a humpback whale calf was found stranded on Oahu with evidence of vessel collision (propeller cuts; NOAA Fisheries unpublished data). Also in 1996, a vessel from Pacific Missile Range Facility in Hawaii rescued an entangled humpback, removing two crab pot floats from the whale; the gear was traced to a recreational fisherman in southeast Alaska. In 2001 and 2002, NOAA Fisheries observed a humpback whale entangled in the mainline of Hawaii-based longline vessels. The animals were released alive, although one may have had some trailing line attached (<30ft). No information is available on the number of humpback whales that have been killed or seriously injured by interactions with fishing fleets outside of U.S. waters in the North Pacific Ocean (NOAA Fisheries, 2002).

Ship Strikes

Humpback whales are killed by ship strikes along both coasts of the U.S. Off California, in 2000, one humpback was struck and killed by a vessel, and from 2000 to 2001, three unidentified whales were injured and one unidentified balaenopterid was killed due to vessel strikes (J. Cordaro, NOAA Fisheries, personal communication 2002). In addition, several humpbacks photographed off California have had large gashes in their dorsal surface which appeared to be from ship strikes. Additional mortality from ship strikes most likely goes unreported, particularly if a whale does not strand, or if they do, they may not have obvious signs of trauma. Large vessels may also not be aware that they have struck a whale.

Other Threats

Humpback whales seem to respond to moving sound sources, such as whale-watching vessels, fishing vessels, recreational vessels, and low-flying aircraft (Beach and Weinrich 1989, Clapham *et al.* 1993, Atkins and Swartz 1989). Their responses to noise are variable and have been correlated with the size, composition, and behavior of the whales when the noises occurred (Herman *et al.* 1980, Watkins *et al.* 1981, Krieger and Wing 1986). Several investigators have suggested that noise may have caused humpback whales to avoid or leave feeding or nursery areas (Jurasz and Jurasz 1979b, Dean *et al.* 1985), while others have suggested that humpback whales may become habituated to vessel traffic and its associated noise. Still other researchers suggest that humpback whales may become more vulnerable to vessel strikes once they habituate to vessel traffic (Swingle *et al.* 1993; Wiley *et al.* 1995). In Hawaii, regulations prohibit boats from approaching within 91 m of adult whales and within 274 m in areas protected for mothers with a calf. Likewise, in Alaska, the number of cruise ships entering Glacier Bay has been limited to reduce possible disturbance.

3. Sperm Whale

Historic and Current Whaling

Approximately 258,000 sperm whales in the North Pacific were harvested by commercial whalers between 1947 and 1987 (*in* Hill and DeMaster 1999). However, both Japan (Kasuya 1998) and the Soviet Union (Kasuya 1998) under-reported catches of sperm whales, so a total of at least 436,000 individuals of this species were taken between 1800 and 1987 (Carretta *et al*. 2001). Of this total, about 33,842 sperm whales were taken by Japanese and Soviet vessels in the eastern North Pacific between 1961 and 1976, and 965 were taken in by land-based operations on the west coast of the

United States between 1947 and 1971 (Ohsumi 1980). An additional
13 whales were taken by shore whaling stations between 1919 and 1926 (Clapham *et al.* 1997).

In 2000, the Japanese Whaling Association announced that it proposed to kill 10 sperm whales in the Pacific Ocean for research purposes, which was the first time sperm whales have been taken since the international ban on commercial whaling took effect in 1987. The implications of this action for the status and trend of sperm whales is uncertain.

Fisheries Interactions

NOAA Fisheries has observed one sperm whale interaction by the Hawaii-based longline fishery. The event occurred in May, 1999 inside the Northwestern Hawaiian Islands EEZ (about 140 nautical miles north of Raita Bank), and the vessel was targeting swordfish. According to the observer report, the sperm whale's pectoral fin was entangled in the mainline. The captain stopped the boat, let out more mainline, and then backed up until he could reach the other end of the mainline. At this point, both ends of the mainline, on each side of the sperm whale, were secured on the vessel. During this time, the whale broke the mainline and swam away without trailing gear. There have been no reported sperm whale interactions by fishers in their logbook submissions (NOAA Fisheries, 2002).

Observer data indicate interactions between longline fisheries and sperm whales in the Gulf of Alaska have been reported over the past decade (Rice, 1989, Hill and DeMaster, 1999). Observers aboard Alaskan sablefish and halibut longline vessels have documented sperm whales feeding on fish caught in longlines 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 longline gear is not yet clear.

D. Factors Affecting Sea Turtles in the Pacific Ocean

Because impacts to sea turtles in the Pacific Ocean are generally non-discriminatory insofar as the different species are concerned, the following is a description of known fisheries and non-fisheries-related threats to all sea turtles in the Pacific Ocean.

1. Fisheries impacts

Very few fisheries in the Pacific Ocean are observed or monitored for bycatch. Rough estimates can be made of the impacts of coastal, offshore, and distant water fisheries on sea turtle populations in the Pacific Ocean by extrapolating data collected on fisheries with known effort that have been observed to incidentally take sea turtles. However, it is important to note that a straight extrapolation of this data contains a large degree of uncertainty and variability. Sea turtles are not uniformly distributed, either by area, or by time of year. In addition, observer coverage of a fishery may be very low, observers may not always be randomly assigned to vessels, or they may be placed on vessels that use fishing strategy that may be uncharacteristic of the fleet. Also, surveys and logbooks may contain biased or incomplete information. Lastly, any take estimates are hampered by a lack of data on pelagic distribution of sea turtles.

This section will summarize known fisheries that have been observed or reported to incidentally or intentionally take sea turtles in the Pacific Ocean. The past effects of the fisheries of the HMS FMP are summarized in Section IV, the Effects of the Action section of this Opinion.

Appendix A provides a summary of current trends in fishing effort in the eastern and western Pacific Ocean, by year, and country. Estimates of total fishing effort are complicated by the fact that not all active vessels fish equivalent number of days per trip or annually, or use the same number of hooks, length of net, or mesh size, or have the same carrying capacity. However, even with minimum effort estimates, it is apparent that there is significant fishing effort in the Pacific Ocean for which NOAA Fisheries has no bycatch information for sea turtles.

a. North Pacific Driftnet Fisheries (before December 1992)

Because the effects of high seas driftnet fisheries operating prior to 1992 may still be evident in sea turtle population trends, it is important to summarize what little is known about the impact of the fisheries on sea turtles in the North Pacific Ocean. Foreign high-seas driftnet fishing in the north Pacific Ocean for squid, tuna and billfish ended with a United Nations moratorium in December, 1992.

Except for observer data collected in 1990-1991, there is virtually no information on the incidental take of sea turtle species by the driftnet fisheries prior to the moratorium. The high seas squid driftnet fishery in the North Pacific was observed in Japan, Korea, and Taiwan, while the large-mesh fisheries targeting tuna and billfish were observed in the Japanese fleet (1990-91) and the Taiwanese fleet (1990). A combination of observer data and fleet effort statistics indicate that 4,366 turtles, mostly loggerheads and leatherback turtles, were entangled by the combined fleets of Japan, Korea and Taiwan during June, 1990 through May, 1991, when all fleets were monitored (Table IV-20). Of these incidental entanglements, an estimated 1,009 turtles were killed (77 percent survival rate).

Table IV-20. Estimated annual bycatch and mortality of sea turtles in the North Pacific high-seas driftnet fishery for squid, tuna & billfish in 1990-91 (Wetherall, 1997).

Species	Estimated Annual Take	Estimated Annual Mortality
green	378	93
leatherback	1,002	111
loggerhead	2,986	805
TOTAL	4,366	1,009

Data on size composition of the turtles caught in the high-seas driftnet fisheries were also collected by observers. Green turtles and the majority of loggerheads measured by observers were immature, and most of the actual measured leatherback turtles were immature, although the size of leatherback turtles that were too large to bring on board were only estimated, and are therefore unreliable (Wetherall, 1997).

These rough mortality estimates for a single fishing season provide only a narrow glimpse of the past impacts of the driftnet fishery on sea turtles. A full assessment of impacts would consider the turtle mortality generated by the driftnet fleets over their entire history and geographical range. Unfortunately, comprehensive data are lacking, but the observer data does indicate the possible magnitude of past turtle mortality, given the best information available. Wetherall *et al*. (1993) speculate that "the minimum total turtle mortality in the North Pacific high-seas driftnet fisheries may have been on the order of 2,500 turtles per year during the late 1980s. The actual mortality was probably greater than this, but less than the estimated total driftnet bycatch of perhaps 9,000 turtles per year. Based on 1990 observer data, most of the mortalities would have been loggerheads taken in the Japanese and Taiwanese large-mesh fisheries."

While a comprehensive, quantitative assessment of the past impacts of the North Pacific driftnet fishery on turtles is impossible without a better understanding of turtle population abundance, stock origins, exploitation history and population dynamics, it is likely that the mortality inflicted by the driftnet fisheries in 1990 and in prior years was significant (Wetherall *et al*. 1993), and the effects may still be evident in sea turtle populations today. The high mortality of juvenile, pre-reproductive adults, and reproductive adults in the high-seas driftnet fishery has probably altered the current age structure (especially if certain age groups were more vulnerable to driftnet fisheries) and therefore diminished or limited the reproductive potential of affected sea turtle populations.

b. Japan

(1) Japanese tuna longliners in the Western Pacific Ocean and South China Sea - in the year 1978

Based on turtle sightings and capture rates reported in a survey of fisheries research and training vessels and extrapolated to total longline fleet effort by the Japanese fleet in 1978, Nishimura and Nakahigashi (1990) estimated that 21,200 turtles, including greens, leatherback turtles, loggerheads, olive ridleys and hawksbills, were captured annually by Japanese tuna longliners in the Western Pacific and South China Sea, with a reported mortality of approximately 12,300 turtles per year. Using commercial tuna longline logbooks, research vessel data and questionnaires, Nishimura and Nakahigashi (1990) estimated that for every 10,000 hooks in the Western Pacific and South China Sea, one turtle is captured, with a mortality rate of 42 percent. Although species-specific information is not available, vessels reported sightings of turtles in locations which overlap with commercial fishing grounds in the following proportions: loggerhead - 36 percent; green turtle - 19 percent; leatherback - 13.7 percent; hawksbill - 10.3 percent; olive ridley - 1.7 percent; and unknown - 19 percent.

Caution should be used in interpreting the results of Nishimura and Nakahigashi (1990), including estimates of sea turtle take rate (per number of hooks) and resultant mortality rate, and estimates of annual take by the fishery, for the following reasons: (1) the data collected was based on observations by training and research vessels, logbooks and a questionnaire (i.e. hypothetical), and do not represent actual, substantiated logged or observed catch of sea turtles by the fishery; (2) the authors assumed that turtles were distributed homogeneously; and (3) the authors used only one year (1978) to estimate total effort and distribution of the Japanese tuna longline fleet. Although the data and analyses provided by Nishimura and Nakahigashi (1990) are conjectural, longliners fishing in the Pacific have had, and (with the current level of effort) probably continue to have significant impacts on sea turtle populations.

(2) Japanese tuna longliners - in the year 2000

The most recent bycatch information for Japanese tuna longliners is based on data collected during 2000. At a recent bycatch working group meeting of the IATTC, held in Kobe, Japan on January 14-16, a member of the Japanese delegation stated that based on preliminary data from 2000, the Japanese tuna longline fleet was estimated to take approximately 6,000 turtles, with 50 percent mortality. Little information on species composition was given; however, all species of Pacific sea turtles were taken, and of an estimated 160 leatherbacks taken, 25 were dead. The Japanese are currently analyzing the data and plan to provide more information to the Secretariat of the IATTC at a later date (K. Hanafusa, Fisheries Agency of Japan, personal communication, January, 2004).

Recently, the Japanese have tested the use of circle hooks and mackerel bait to determine effectiveness of reducing sea turtle interaction and mortality rates. Preliminary analyses show that circle hooks and mackerel bait significantly reduced the catch rate of sea turtles, without affecting the catch rate of the target species (tuna). The experiments were conducted off Japan, between 30EN and 40EN. At the bycatch working group meeting, the Japanese proposed that longline fisheries in the eastern tropical Pacific should be required to use circle hooks and not use squid bait on gear set at depths less than 120 meters from the surface. Unfortunately, several countries would not agree to the proposal, generally stating that more research needed to take place before such proposals were implemented (T. Fahy, NOAA Fisheries, personal communication, January, 2004).

Table 1 in Appendix A provides a summary of the number of active Japanese longline vessels fishing mainly for tuna in the Central Western Pacific Ocean from 1990-2002. Okamoto *et al*. (1999) estimated the number of hooks deployed by Japan's offshore and distant water longline fleet in the western Pacific Ocean to average around 154 million hooks per year, based on effort data from 1990 to 1997.

(3). Japanese coastal fisheries

Off the coast of Japan, gillnets and pound nets are very common. In addition, there is an intense trawl fishery for anchovy operated off-shore of some major loggerhead rookeries during the nesting season. According to the Sea Turtle Association of Japan (2002), approximately 80 mature loggerheads strand every year in Japan - "these coastal fisheries might be strongly related with stranding." With less than 1,000 female loggerheads nesting annually in Japan, this number of strandings is not insignificant.

c. Taiwan

(1). Coastal setnet and gillnet fishery

Taiwanese have harvested sea turtles for many years for their meat, their bones for use in Chinese medicine, and eggs for profit. In Taiwan, sea turtle bycatch in fisheries occurs, although little quantitative information is available for fisheries operating in the Pacific Ocean (Cheng, 2002).

Researchers investigated the incidental capture of sea turtles by the coastal setnet and gillnet fisheries in the eastern waters of Taiwan from 1991 through 1995. Setnets used in the coastal waters off Taiwan are near-shore sedentary trap nets, and rarely extend below 20 meters. During the time of the study, there were 107 setnets in Taiwan, and they provided the second largest total fish yields, after gillnets. According to interviews with fishermen, incidentally caught sea turtles are either sold to dealers in the market or are butchered for meat (subsistence). Fishing grounds including set nets and gillnets were observed from 1991 through 1992, and the fish market was visited once or twice per month from 1991 through 1995 to corroborate bycatch data (Cheng and Chen, 1997).

Of the sea turtles caught, 82% were caught in setnets, and of these, all were alive. As shown in Table IV-21, green turtles accounted for 70% of the sea turtles taken, and captured turtles represented all age classes (large juvenile, subadult and adults). Most captured loggerheads were either subadults or adult females (only one male was unidentified), and most of the captured olive ridleys were subadults. The one captured leatherback was released alive. Not surprisingly, bycatch rate also increased with fishing effort, and most of the turtles taken were sold to temples for "religious release"[13] later. Of all captured turtles, 88% were sold to temples for Chinese religious ceremonies, 8% were stuffed or butchered, and 3% were released at the site (Cheng and Chen, 1997).

Table IV-21. Sea turtles incidentally caught in fishing gear off Taiwan from 1991-1995.

Year/Species	1991	1992	1993	1994	1995	Total
green	6	17	28	23	42	116
leatherback	1	0	0	0	0	1
loggerhead	1	4	5	15	1	26
olive ridley	9	0	1	0	4	14

Source: Cheng and Chen, 1997

d. Philippines

Near the Turtle Islands, a variety of fisheries interact with sea turtles, and Cruz (2002) reports and an increasing number of floating dead turtles have been observed in this area in the since 1999, most likely attributable to an increasing number of fishing vessels operating in the area, including purse seiners, shrimp trawlers, and hulbot-hulbot (demersal drive-in net). These vessels originate primarily from Sabah, Malaysia, and Manila, Philippines. There are also an increasing number of fishing vessels operating in Philippine waters that have originated from China. Aside from fishing illegally, the Chinese vessels are also catching sea turtles. In January, 2002, more than 58 sea turtles, primarily green turtles were discovered on four Chinese vessels in Tabbataha Marine Park, a UNESCO Natural Heritage Park, located in the Sulu Sea (Cruz, 2002).

e. Malaysia

Sea turtles are caught an a variety of fisheries in Malaysia, ranging from driftnets, lift nets, ray nets (similar to sunken driftnets with a large mesh to target rays and sharks), trawl nets, and purse seines. In 1994-95, a survey was conducted of fisherman to determine the percentage of them that had past experience incidentally capturing sea turtles. The results are presented in Table IV-22.

Table IV-22. Summary of 1994-95 sample survey of fishermen for incidental catch of sea turtles in

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[13] "Religious release" refers to the practice by which fishermen would sell live turtles to a temple. The temple master would then sell the turtles to believers for release back into the ocean several months later. Many turtles were kept in an unhealthy environment during captivity and died following release. This practice is not done anymore because any landing of live turtles is forbidden and violators will be reported to the police (I.J. Cheng, Institute of Marine Biology, Keelung, Taiwan, personal communication, November, 2003).

Terengganu, Malaysia.

Gear Type	**No. Fishermen Inteviewed**	**No. Of Fishermen with past experience incidentally capturing sea turtles (%)**
Hook and Line	77	0
Fish Traps	35	4 (11%)
Purse Seine	27	6 (22%)
Drift/Trammel Nets	23	3 (13%)
Longlines	20	0
Trawls	20	11 (55%)
Ray Nets	9	6 (67%)
Lift Nets	7	2 (27%)
Beach Seine	4	4 (100%)

Source: Liew (2002).

f. India

As summarized in the prior section on status of the olive ridley, thousands of these turtles nest in Orissa, India each year. With an increase in fishing intensity off Orissa's coast, there has been an increase in the mortality of olive ridleys, primarily due to illegal gillnet and trawl fishing in the offshore waters. While turtle excluder devices are mandatory in Orissa, the trawler community opposes their use and thus many trawlers do not use them. In addition, the Orissa Marine Fisheries Act (1982) and Rules (1983) prohibit all mechanized fishing within 5 kilometers of the coast and within 20 kilometers of the Gahirmatha coast (~35 km).

During the 1980s, a few hundred ridleys per year were reported killed incidentally in Orissa. By the 1990s, mortality increased from 5,000 per year in 1994 to 13,000 per year in 1999, a total of approximately 46,000 dead turtles along the coast of Orissa in six years (*in* Shanker *et al*., 2003b). The number of dead turtles counted during a survey correlated strongly with the number of mechanized fishing vessels operating in their respective coastal waters (Pandav, 2001). Since the late 1980s, there has been an increase in fishing intensity, from less than 1,000 mechanized boats to greater than 4,000 boats by 1996. Since 1994, an estimated 90,000 olive ridleys have been documented stranding, and current annual estimates of mortality are approximately 15,000 per year (Shanker *et al*., 2003b).

g. *Distant Water Fishing Nations Longline Fishing in the EEZ around the Federated States of Micronesia*

Heberer (1997) summarized the results of 51 distant-water fishing nation (DWFN) longline trips observed by Micronesian Maritime Authority fisheries observers from 1993 through 1995. Vessels from China, Taiwan, and Japan captured a total of 34 sea turtles. These turtles were reported as 15 olive ridleys, 8 green turtles, and 11 unidentified sea turtles. Thirty of the 34 turtles were released alive and the remainder were dead when landed (11.8% mortality rate). Data on hooking location or entanglement was not reported, nor was the condition of each turtle by species.

The Micronesia Fisheries Authority (previously Micronesian Maritime Authority) places observers aboard distant water fishing vessels fishing by longline in their EEZ. Table IV-23 shows the observed catch of sea turtles by these vessels from January 1, 1990 through December, 2000. While the overall data set represents a significant amount of effort - 971 sets and 1,272,000 hooks observed over a 10 year period, the rate of observer coverage is extremely low. From 1990 through 1997, observer coverage ranged from 1 to 3%.

Table IV-23. Observed captures of sea turtles aboard distant water longline vessels, January 1990 through December 2000. Source: Micronesian Fisheries Authority

Species	Number	Condition	
		% Alive	% Dead
Green	4	100	0
Hawksbill	1	100	0
Loggerhead	1	100	0
Olive ridley	8	100	0
Unidentified turtle	33	79	21
Total	47		

The information presented above is from two separate data sets, which may not have been coordinated. The study conducted by Heberer (1997) utilized observers specifically trained and directed to record bycatch information, whereas observers in this fishery typically prioritize the collection of target catch data over bycatch information. This information represents the best available information on bycatch in this fishery. Appendix A provides additional information on fishing effort. However, the above data cannot be compared or used to extrapolate expected rates of turtle bycatch based on small sample sizes, low rates of observer coverage, and prioritization of catch data.

h. Foreign tuna fisheries in the western and central Pacific Ocean

As described above, the western and central Pacific Ocean (area west of 150EW longitude, and between 10EN and 45ES) contains the largest industrial tuna fisheries in the world. Much of the effort takes place in the EEZs of Pacific-Island counties, in the western tropical Pacific area (10EN - 10ES). Annual tuna catches in this area have averaged around 1.5 million metric tons, with around 60% of the catch taken by purse seine vessels, and the rest taken by longline vessels and other gears (e.g. pole-and-line, troll, ring-net). As Table 1 in Appendix A indicates, approximately five thousand longliners operate throughout the western and central Pacific (45EN to 45ES), using up to 3,000 baited hooks per line to catch tuna. As shown in Table 2 in Appendix A, there are nearly 400 active purse seine vessels originating from a variety of countries and operating nearly exclusively in tropical waters.

Observers have been placed on both purse seiners and longliners in this area, and operate and report to the Oceanic Fisheries Programme of the Secretariat of the Pacific Community (SPC). While observers have covered most of the fleets, three fleets have not been observed: the Japanese and Korean distant-water longline fleets operating in the eastern areas and a recently established Australian swordfish fishery operating off eastern Australia.

Given the low observer coverage (<1%) for the longline fisheries, patterns of sea turtle observed interactions show that sea turtles are more likely to encounter gear in tropical waters and that they are much more likely (by an order of magnitude) to encounter gear that is shallow-set versus deep-set. When encountered on deep-set gear, sea turtles were likely to be taken on the shallowest hooks. From available observer data, the longline fisheries operating in the western and central Pacific is estimated to take 2,182 sea turtles per year, with 500-600 expected to die as a result of the encounter. From observer data, 1,490 are estimated taken by offshore/fresh tuna vessels using shallow-night sets, 129 are estimated taken by offshore/fresh tuna vessels on deep-day sets, and 564 are estimated taken by distant water freezer vessels on deep-day sets. The species observed taken include (ranked by highest occurrence first): olive ridley, green, leatherback, loggerhead and hawksbill. Given the low observer coverage, this estimate has very wide confidence intervals.

For the purse seine fisheries operating in this area, an estimated 105 sea turtles are taken per year, with approximately 17% mortality rate (less than 20 sea turtles dead per year). The species included green turtles, hawksbills and most often olive ridleys. Animal-associated, drifting log and anchored-FAD sets had the highest incidence of sea turtle encounter (1.115, 0.807, and 0.615 encounters per 100 sets, respectively). In contrast, drifting FAD sets were observed to have only 0.07 encounters per 100 sets. With less than 5% observer coverage, confidence intervals for these estimates are also very wide (Oceanic Fisheries Programme, SPC (draft report), 2001).

i. Chile

Although data on the incidental take of sea turtles in the Chilean swordfish fisheries are sparse, both green and leatherback turtles have been confirmed taken and killed, and olive ridleys and loggerheads may also be taken incidentally by the fishery (Weidner and Serrano, 1997). As described further in Appendix A, the Chilean swordfish fishery is comprised primarily of artisanal fishermen, averaging 500 boats (mainly driftnetters) from 1989 to 1991, and decreasing in numbers after 1991. Since 1991, approximately 20 large industrial (i.e. commercial) boats have fished swordfish in Chile, the effort is comprised of gillnets (27%), pelagic longliners (72%) and boats that switch gear. Effort by the artisanal fishery (including the driftnet fleet) increased from 5,265 days-at-sea in 1987 to 41,315 days-at-sea in 1994 (Barbieri, *et al*., 1998).

Adult female leatherback turtles tagged in Mexico have been taken in Chilean waters by gillnet *and* purse seine fisheries (Marquez and Villanueva, 1993). In addition, data were recorded opportunistically from the artisanal swordfish fishery (driftnetters primarily) for a single port (San Antonio) over a two year period. This partial record documented leatherback captures and sightings totaling 9 in 1988 and 21 in 1989. A rough estimate of 250 leatherback takes per year without differentiating between kills and total takes for vessels operating out of San Antonio was provided (Frazier and Brito Montero, 1990). A more recent estimated annual take of 500 leatherback turtles was provided by Montero (personal communication, 1997, *in* Eckert, 1997) which was not unreasonable, given the nearly ten-fold increase in fishing effort from 1987 to 1994.[14] As shown in Table IV-24, the take of sea turtles by the artisanal driftnet fishery in the late 1980s appeared to be comprised primarily of leatherback turtles.

Table IV-24. Chile – total turtle bycatch of artisanal driftnet fishermen, 1988-89 season.

Species	Number	Percentage of Total
Green turtle	42	28%
Leatherback	82	55%
Loggerhead	5	3%
Olive ridley	21	14%
Total	150	100%

Source: José Brito-Montero, personal communication, 3/3/97, *in* Weidner and Serrano, 1997

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[14] Based on all information from Chile and Peru, Eckert (1997) estimated that a minimum of 2,000 leatherback turtles are killed annually by Peruvian and Chilean swordfish operations, representing a major source of mortality for leatherback turtles originating from and returning to nesting beaches in Costa Rica and Mexico. Because swordfish fishing effort has declined significantly since the early 1990s, incidental take has most likely declined as well, although the current estimate is unknown.

Effort by the artisanal driftnet fishery for swordfish appears to be relatively constant through 1996, as shown in Table III-21. Given the total sea turtle take estimate from the 1988-89 season, and combining it with the total effort (days-at-sea) data from 1988-1996, and assuming effort was constant and in the same general area during all years, a simple calculation can be made to estimate the incidental take of turtles by the Chilean artisanal driftnet fishery for swordfish during subsequent years (third column in Table IV-25). Turtles reportedly began appearing in Chilean markets in 1987, just as the swordfish driftnet fishery was expanding, and Chilean observers have reported occasional individual sets with leatherback mortalities from 3-13 (*in* Weidner and Serrano, 1997). Assuming the current artisanal driftnet fishing effort is equivalent to 1996 and assuming the proportion of species taken is equivalent to data collected from the 1988-89 fishing season, this fishery would currently take an estimated 39 greens, 76 leatherback turtles, 4 loggerheads, and 29 olive ridleys annually. However, Donoso (personal communication, September 2002) reports that the artisanal fleet has declined to maybe a third of its size in recent years.

Table IV-25. Chile - artisanal (driftnet) swordfish effort, by year, from 1989-1996 and calculated (not actual or known) turtle take [note assumptions used in this Opinion].

Year	Effort (Days-at-sea)	Calculated Turtle Take*
1989	7,579	150*
1990	6,226	123
1991	11,450	227
1992	11,209	222
1993	10,755	213
1994	8,393	166
1995	8,152	161
1996	7,041	139

*Calculated turtle take was estimated by comparing effort for 1989 (7,579 days-at-sea) and a known turtle take of 150 (1988-89 season) with subsequent years for which effort was known, but turtle take is not known.

**Estimated take of turtles by Brito-Montero, for the 1988-89 season, and assuming 1989 data is equivalent in effort to 1988-89 effort, for the purpose of comparing year-to-year calculations of estimated turtle take. Source: Weidner and Serrano, 1997.

During 1996, there was a substantial expansion of Chilean longline fishing in offshore areas, but as there has been no collection of data on this fishery as of 1997 (Weidner and Serrano, 1997), the number of takes and mortalities and their effects on sea turtle stocks as a result in this change in fishing strategy are not known. Since effort for swordfish in the Chilean fishery or throughout the Pacific has declined significantly overall since 1994 (as a result of concerns about overfishing swordfish stocks), and populations of turtles have declined, the bycatch of sea turtles in this fishery has likely declined as well, although the extent of this decrease is currently unknown. There is very little information on lethal and non-lethal incidental catch per unit effort although new studies are underway to quantify bycatch. In addition to the swordfish fishery, Chile also has a substantial purse seine fleet, which has recently shifted from a reliance on coastal anchovy and sardines to a substantial take of jack mackerel further offshore, where turtle interactions may be more common (Weidner and Serrano, 1997). The extent of the impact of the Chilean purse seine fishery on sea turtles is unknown.

j. Peru

Since 1995, Peruvian law has prohibited the capture, trade, and consumption of sea turtles. Despite the law, sea turtles continue to be caught alive in artisanal fisheries as bycatch and are nearly always killed for “bushmeat.”

Appendix A contains a description of known domestic and foreign fisheries in Peru. Peruvian commercial longline fleets have had limited success in fishing for swordfish, so there is probably very little incidental catch of sea turtles in this fishery. Peruvian artisanal fishermen, however, also target fish species normally taken in commercial longline fisheries (especially shark) and have been more successful than the commercial longline fleet, so more turtles may be caught incidental to these artisanal fisheries (Weidner and Serrano, 1997).

From 1997-1999, the government agency IMARPE estimated that 8.02 tons of turtles were captured (Alfaro-Shigueto, in press). Kelez *et al*. (2003) report that sea turtles are commonly caught incidentally by artisanal fisherman, entangled by gillnets and hook-and-line. In general, fishermen from the smaller villages may release a turtle that is alive; however, if it is dying or dead, they will kill it. In the larger towns, fishermen will nearly always kill an incidentally caught turtle because of the demand for its meat. The carapaces of sea turtles are also sold in the department of Tumbes and in the northern part of the department of Piura, due to the tourist industry (Kelez *et al*., 2003). From January, 2001 through February, 2003, observers sampled eight ports in Peru to document sea turtle bycatch. During this time, observed turtle bycatch was 1,630 individuals, with total estimated bycatch to be 2,025 turtles (after extrapolation for days not observed). Ports sampled included Mancora (272 turtles), Constante (231 turtles), Parachique (337 turtles), San Jose (153 turtles), Salaverry (167 turtles), Chimbote (168 turtles), Pisco (77 turtles), and Morro Sama (620 turtles). Table IV-26 shows a breakdown of turtle bycatch by species.

Table IV-26. Estimated number of sea turtles captured in artisanal fisheries in Peru from January, 2001 - February, 2003.

Species	Estimated # captured
Green turtle	1,509
Loggerhead	354

Leatherback	103
Olive Ridley	51
Hawksbill	8
Total	2,025

Source: Alfaro-Shigueto, In press.

Foreign longline fleets are also active and extensive off Peru and the bycatch of sea turtles in these foreign fisheries has been considered significant (Weidner and Serrano, 1997).

k. Ecuador

Appendix A contains a description of known current commercial and artisanal fisheries in Ecuador. Currently, the artisanal longline fleet is composed of roughly an estimated 5,000 vessels, while the industrial longline fleet is composed of approximately 181 vessels (E. Everett, IATTC, personal communication, November, 2003).

Unfortunately, the composition of turtle species incidentally taken by Ecuadoran commercial and artisanal fisheries is unavailable. Prior to a ban on the commercial harvest for olive ridleys in 1986, artisanal fishermen prosecuted a directed turtle fishery as well as taking them incidentally. During 1985 and 1986, 124 and 715 metric tons of turtles, respectively, were reportedly taken (Table III-21). In 1990, the Ecuadoran government permanently ended the directed fishery, prohibiting the catch as well as domestic and export marketing. Incidental catches of sea turtles by tuna and swordfish longliners are reportedly very rare, but they do occur, and Ecuadoran authorities have seized turtle skins from Japanese longliners (*in* Weidner and Serrano, 1997).

l. Colombia

A description of known Colombian commercial fisheries is provided in Appendix A and summarized in Table 5 of the Appendix. No information is available on the sea turtle bycatch levels in the shrimp trawl fisheries and other fisheries operating out of Colombia. However, a turtle excluder device program has been initiated in the shrimp trawl fishery to reduce incidental catch. Artisanal fisheries in the past targeted turtles (Weidner and Serrano, 1997); however, no recent information on directed take is available.

m. Central American shrimp fishery

Shrimp fishery operations were initiated throughout Central America during the mid 1950s. On the Pacific, vessels pull one standard 50 to 75 foot headrope length two seam balloon trawl or one standard flat net from each outrigger. Target species include white and small shrimp in shallow waters (9-20 meters deep), pink and brown shrimp in water depths ranging from 55 to 85 meters, and deep shrimp “fidel” or “camello” in deeper waters (150-225 meters depth). A description of the shrimp fisheries on the Pacific coast of Central America is contained in Appendix A.

Beginning in 1996, the U.S. has required countries on the Pacific coast of Central America to meet the requirements of Section 609 of U.S. Public Law 101-162, including the adoption of a sea turtle protection program comparable in effectiveness to that of the U.S. in order to be certified to export shrimp from commercial fisheries. Though compliance with Section 609 has generally been good, it has been inconsistent for some countries. Costa Rica and Panama have both had certificates withdrawn or withheld in the past over concerns about the effectiveness of their program, though all Central American countries with commercial trawl fisheries in the Pacific are currently certified pursuant to Section 609 (D. Hogan, State Department, personal communication, January 2004).

Arauz (1995) estimated that over 60,000 sea turtles were taken by shrimp trawlers on the Pacific coast of Central America. Mortality rates were not estimated. Olive ridleys were the species most commonly taken, and foraging grounds for these turtles overlap with shrimp trawling grounds. Table IV-27 shows the estimated turtle catch by shrimp trawlers in Central America, by country, for 1993.

Table IV-27. Estimated turtle catch by shrimp trawlers for the Pacific coast of central America, 1993

Country	**# Vessels**	**Total CPUE turtles/hr**	**Turtles/year**
Guatemala	58	?	(10,000)
El Salvador	70	0.0511	21,280
Nicaragua	21	?	(8,000)
Costa Rica	55	0.0899	20,762
Total	**204**		**60,042**

Note: figures in parenthesis are estimated. Source: Arauz, 1995.

Observers have also been deployed on shrimp trawling operations off the Pacific coast of Costa Rica. During 2,556.5 hours of observation, 281 sea turtles were incidentally captured. Of those captured, 90% were olive ridleys (253 observed taken), 9.6% were Pacific greens (27 observed taken) and 0.4% were hawksbills (1 observed taken). The observed mortality rate for this species captured by this fishery was around 40%.

n. *Costa Rica*

Sea turtles are impacted by Costa Rican fisheries and by interaction with human activities. Several studies have been undertaken in recent years in order to document the incidental capture of sea turtles in Costa Rican longline fisheries. The longline fleet consists of a “medium” artisanal fishery, which targets mahi mahi and tunas within the country’s EEZ, and an “advanced” fleet, which targets billfish and tunas within and outside the EEZ. In 1999, the fleet was comprised of 678 registered vessels, with lengths varying between 6 and 29.9 meters. Two studies in 1997 and 1998 on two longline fishing cruises (one experimental) documented a high incidental take of sea turtles. On one cruise, a total of 34 turtles (55% olive ridleys and 45% east Pacific green turtles) were taken on two sets containing 1,750 hooks (1.42 turtles per 100 hooks). One additional set caught two leatherbacks. The second cruise documented the incidental take of 26 olive ridleys, with 1,804 hooks deployed (Arauz *et al.*, 2000).

An observer program was put in place from August, 1999 through February, 2000. Seventy seven longline sets were observed on 9 cruises. Of the nearly 40,000 hooks deployed, turtles represented 7.6% of the total catch, with a catch per unit effort of 6.364 turtles/1,000 hooks. The results are shown in Table IV-28. Immediate sea turtle mortality was 0%, and most of the hooks were removed prior to release (Arauz, 2001).

Table IV-28. Costa Rican longline fleet - observed number and condition of sea turtles taken on nine cruises, August, 1999 - February, 2000

Species/condition	**Number**
Olive ridley	
Hooked in mouth	216
Hooked in flipper	26
Hooked in neck	1
Entangled	4
Total	**247**
Green turtle	
Hooked in mouth	8
Hooked in flipper	4
Total	**12**

Source: Arauz, 2001.

From September to December of 2000, the Sea Turtle Restoration Project documented more than 400 dead turtles washed up along the north and central Pacific coast of Costa Rica. Of 423 dead turtles observed, 84 turtles showed “clear interaction with human activities, such as cracked skulls or carapaces due to collisions with boats, hooks imbedded in the mouth and throat, incisions in the groin to collect eggs, and digital fractures due to entanglement in gillnets. As of 2001, more than 130 dead turtles have been observed.” The Costa Rican Fishery Institute (INCOPESCA) has “declared itself incompetent to enforce sea turtle protection laws, and proposes that [the authorities of the Environment] MINAE should be responsible and apply the Wildlife Conservation Law (PESJ-1451-2000). However, while MINAE eludes responsibility, hundreds of dead sea turtles continue to wash up along the coast...” (Sea Turtle Restoration Project press release, 8/6/01[15]).

During a survey of three Costa Rican beaches (Nancite, Ostional and Grande or Baulas) from August 2000 and January 2001, stranded sea turtles were collected and assessed. Ninety three dead turtles were assessed, and of these, 78.5% were attributed to anthropogenic causes, including: “capture and forced immersion by shrimp nets, entanglement in nylon lines, cranial traumas, boat strikes that may cause injuries, and slaughter to harvest eggs and meat for consumption by humans.” Hooks were also found in the mouths and esophagus of sea turtles, primarily in olive ridleys (Vasquez and Morales, 2003).

o. Nicaragua

Incidental capture of sea turtles in Nicaraguan fisheries occurs; however, there has been little documentation. The primary concern is with the artisanal fleet, which is comprised of nearly 5,000 vessels. Every year, hundreds of dead olive ridleys are reported throughout the Pacific coast of Nicaragua. During 2001, over 100 olive ridleys and one juvenile leatherback were documented stranded in the Chacocente Wildlife Refuge. Of stranded turtles that were examined during a monitoring project in 2001-2002, 100% (12/12, all females) had been cut in the groin area (common practice by fishermen in search of eggs). Artisanal gillnetters and industrial shrimp trawlers routinely operate within the limits of the “no fishing zone” established around the nesting beaches (Arauz, 2002).

p. Mexican (Baja California) fisheries and direct harvest

Sea turtles have been protected in Mexico since 1990, when a federal law decreed the prohibition of the “extraction, capture and pursuit of all species of sea turtle in federal waters or from beaches within national territory ... [and a requirement that] ... any species of sea turtle incidentally captured during the operations of any commercial fishery shall be returned to the sea, independently of its physical state, dead or alive” (*in* Garcia-Martinez and Nichols, 2000). Despite the ban, studies have shown that sea turtles continue to be caught, both indirectly in fisheries and by a directed harvest of eggs, immatures, and adults. Turtles are principally hunted using nets, longlines and harpoons. While some killed immediately, others are kept alive in pens and transported in trucks, pick-ups, or cars. The market for sea turtles consists of two types: the local market (consumed locally) and the export market (sold to restaurants in cities such as Tijuana, Ensenada, Mexicali, and U.S. cities such as San Diego and Tuscon). Consumption is highest during holidays such as Easter and Christmas (Wildcoast, *et al*. 2003).

As discussed earlier, green turtle populations in the Mexican Pacific continue to decline. Based on a combination of analyses of stranding data, beach and sea surveys, tag-recapture studies and extensive interviews, all carried out between June, 1994 and January, 1999, Nichols (2002) conservatively

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[15] http://www.seaturtles.org/press_release2.cfm?pressID=107

estimated the annual take of sea turtles by various fisheries and through direct harvest in the Baja California, Mexico region.

Although there are no solid estimates of fisheries-related sea turtle mortality rates for the region, sea turtles are known to interact with (and be killed by) several fisheries in the area. As in other parts of the world, shrimp trawling off Baja California is a source of sea turtle mortality, although since 1996, shrimp fishermen are required to use turtle excluder devices. Prior to this requirement, Figueroa *et al*. (1992 *in* Nichols, 2002) reported that nearly 40% of known mortality of post-nesting green turtles tagged in Michoacán was due to shrimp trawlers. Based on stranding patterns, Nichols, *et al*. (2000) speculate that mortality of loggerheads due to local fishing in Baja California may primarily be due to a net-based fishery. None of the stranded turtles showed signs of hooking; however, the halibut (*Paralichthys californicus*) gillnet fishery, which reports regular loggerhead bycatch and coincides with the movement of pelagic red crab into the shallower continental shelf, may interact with loggerheads as they enter coastal waters in the spring and summer. Fishermen also report the incidental capture of sea turtles, primarily loggerheads, by pelagic longlines and hook sets used to catch sharks and pelagic fish. Lastly, sea turtles have occasionally been found by fishermen entangled in buoy and trap lines, although this is apparently a rare occurrence (Nichols, 2002). Although fishermen may release sea turtles alive after being entangled in or hooked by their gear, based on information on the directed harvest and estimated human consumption of sea turtles in this region, incidentally caught sea turtles are likely retained for later consumption.

Sea turtle mortality data collected between 1994 and 1999 indicate that over 90% of sea turtles recorded dead were either green turtles (85%) or loggerheads (8%; Table IV-29), and signs of human consumption were evident in over half of the specimens. Most of the loggerheads were immature, while size ranges for both green and olive ridleys indicated representation from both immature and mature life stages (Nichols, 2002).

Table IV-29. Recorded sea turtle mortality by species during 1994-1999 on the Gulf of California coast and the Pacific coast of Baja California, Mexico.

Species	Gulf of California	Pacific	Totals
green turtle	30	276	**306**
leatherback	1	0	**1**
loggerhead	3	617	**620**
olive ridley	1	35	**36**
unidentified	0	57	**57**
Total	**35**	**985**	**1,020**

Source: Nichols (2002).

A more focused study was conducted from June to December, 1999 in Bahia Magdalena, a coastal lagoon to determine the extent of sea turtle mortality. Researchers searched for sea turtle carapaces in local towns and dumps as well as coastal beaches. The majority (78%) of the carapaces were found in towns and dumps and green and loggerhead turtles most frequently observed. Both species found were generally smaller than the average size of nesting adults. Researchers estimated that the minimum sea turtle mortality rate for the Bahía Magdalena region was 47 turtles per month, or 564 turtles per year. Based on observations, approximately 52% were green turtles, 35% were loggerheads, 2% olive ridleys, and 1% hawksbills (10% unidentified) (Gardner and Nichols, 2002). A study conducted from 1995 to 2002 in Bahía de los Angeles, a large bay that was once the site of the greatest sea turtle harvest in the Gulf of California, revealed that the populations of green turtles in the area had decreased significantly since the early 1960s. Despite the 1990 ban, sea turtle carcasses were found at dumpsites, so human activities continue to impact green turtles in this important foraging site (Seminoff, *et al*., 2003).

Based on surveys conducted in coastal communities of Baja California, extrapolated to include the entire coastal peninsula, Nichols (2002) estimated the annual mortality of green turtles in this region to be *greater* than 7,800 turtles, impacting both immature and adult turtles. Results from a region-wide socioeconomic study conducted with The Universidad Autonoma de Baja California Sur preliminarily suggest that the actual annual harvest of green turtles may be three to four times higher than this estimate (i.e. approximately 23,000 - 31,000 green turtles taken per year). Mortality of loggerhead turtles, based on stranding and harvest rates, is estimated at 1,950 annually, and affects primarily immature size classes. The primary causes for mortality are the incidental take in a variety of fishing gears and direct harvest for consumption and [illegal] trade. With the local declines of green turtles, a market for loggerhead meat has developed in several Pacific communities. Olive ridleys are not found as commonly in Baja California waters as loggerheads and greens; however, they are consumed locally, occasionally strand on beaches, and have been found entangled in plastic debris. No annual mortality estimates of olive ridleys in the area were presented. Lastly, anecdotal reports of leatherbacks caught in fishing gear or consumed exist for the region; however, these instances are rare, and no annual mortality estimates of leatherbacks were presented (Nichols, 2002). A recent estimate by Wildcoast *et al*. (2003) reiterates that there is likely high mortality of turtles in the Californias[16], estimating 15,600 to 31,200 sea turtles consumed annually (no differentiation between species).

q. Foreign tuna purse seine fishery in the eastern tropical Pacific

The international fleet represents the majority of the fishing effort and carrying capacity in the ETP tuna fishery, with much of the total capacity consisting of purse seiners greater than 400 st. These large vessels comprised nearly 70 percent of the total fishing capacity operating in the ETP in 1996 (IATTC, 2002). An average of 122 foreign vessels with a carrying capacity greater than 400 st fished each year in the ETP during 1996 to 2001. In addition to these larger vessels, the foreign fleet contains smaller

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[16] "California" as defined here is the region encompassing the Gulf of California (including the coast of Sonora and Sinaloa, Mexico); Baja California and Baja California Sur, Mexico, and California, USA.

vessels less than 400 st that target tuna in the ETP. From 1996 to 2001, an average of 59 foreign vessels ranging from 45 to 400 st carrying capacity fished in the ETP each year (IATTC, 1999, 2001, 2002a-b).

Since 1999, seminars have been given by the IATTC to skippers and their crews to educate them on, among other items, status of sea turtles, and handling and recovery of turtles taken by purse seine. In addition, during their 70th meeting held in Antigua, Guatemala on June 24-27, 2003, the IATTC passed Resolution C-03-08. Under the resolution, purse seine fishermen are required to promptly release unharmed, to the extent practicable, all sea turtles. Crews are required to be trained in techniques for handling turtles to improve survival after release. Vessels are to encourage the release of sea turtles entangled in FADs and recover FADs when they are not being used in the fishery. Specific to the purse seine fishery operation, whenever a sea turtle is sighted in the net, all reasonable efforts should be made to rescue the turtle before it becomes entangled, including, if necessary, the deployment of a speedboat. If a sea turtle is entangled in the net, net roll should stop as the turtle comes out of the water and should not start again until the turtle has been disentangled and released. If a turtle is brought aboard the vessel, all appropriate efforts to assist in the recovery of the turtle should be made before returning it to see (IATTC Resolution C-03-08, Action #3).

Data from observers on both U.S. and foreign tuna purse seine vessels have been gathered collectively by the IATTC since the early 1990s. The most recent data from the IATTC indicate that an average of 136 sea turtles per year were killed by vessels over 400 st in the foreign ETP purse seine fishery (non-U.S.) from 1993-2002 (Table IV-30; M. Hall, IATTC, personal communication, December, 2003). The numbers of sea turtles killed by the fishery dropped significantly in 2002, likely as a result of increased awareness by fishermen through educational seminars given by the IATTC. Given the passing of the latest IATTC Resolution on Bycatch, the mortalities should continue to decrease.

Table IV-30. **Estimated sea turtle mortality by species for the foreign ETP tuna purse seine fishery (non-U.S.) from 1993-2002[1]**

Species/Year	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002
Green/black	15	16	13	13	20	9	11	6.1	7.6	2
Hawksbill	0	1.8	0	1	0	3	2	1	1.3	0
Leatherback	0	1	0	0	0	0	0	0	0	0
Loggerhead	3.6	1.8	2	0	4.6	1	4	1.8	1.3	0
Olive ridley	75.8	80	91.3	72.8	93.8	106.6	108.8	91.6	68.9	30.1
Unidentified	21	45.2	43	48.6	51	41	46	29.2	55.4	13.8
TOTAL	115.4	145.8	149.3	135.4	169.4	160.6	171.7	129.8	134.6	45.9

[1] (M. Hall, IATTC, personal communication, December, 2003)

The 1993-2002 data indicate that turtles killed by the entire tuna purse seine fishery were "unidentified," although the reasons for this were not given. Assuming that these unidentified turtle mortalities occurred in the same proportions as the identified turtle mortalities, 85% would be olive ridleys, 12% would be green turtles, 2% would be loggerheads, 1% would be a hawksbill, and 0% would be leatherbacks.

r. United States (non-HMS FMP fisheries)

(1) U.S. tuna purse seine fishery in the central and western Pacific Ocean

The vast majority of the U.S. western and central Pacific purse seine activity occurs in the highly productive fishing grounds of the equatorial western Pacific (principally in the EEZs surrounding Papua New Guinea, the Federated States of Micronesia and Kiribati) under a multilateral agreement entitled *Treaty on Fisheries Between the Governments of Certain Pacific Island States and the Government of the United States of America* or the South Pacific Tuna Treaty (SPTT). The treaty was signed by the United States and 16 Pacific Island Parties belonging to the Forum Fisheries Agency (FFA), and provides U.S. tuna purse seiners access to tunas in a 25.9 million km^2 area of the central-western Pacific Ocean in exchange for fishing fees and adherence to rules related to closed area, etc (Coan, *et al*., 1997). The treaty was renegotiated in 1992 for an additional 10 years.

Between 1988 and 1999, the number of licensed U.S. tuna purse seiners ranged from 35 to 51, although only between 31 and 49 vessels fished during those years. Between 71 and 241 trips were made during each calendar year (Coan, *et al*., 2000), and most of the fishing was conducted in the equatorial belt, extending from around 155EW to 140EE longitude, the traditional fishing zone for the U.S. fleet (Coan, *et al*., 1997). The U.S. fleet primarily lands their catch in American Samoa (Coan, *et al*, 1997, 2000). From 1988 to 1995, the fleet primarily set on free-swimming school sets and less on log sets; however, beginning in 1996, sets were increasingly made on floating aggregation devices (FADs), and in 1999, nearly 100 % of sets were on FADs (Coan, *et al*., 2000). Because turtles tend to congregate around floating objects in the open ocean, this change in fishing strategy may increase the likelihood of sea turtle interactions.

The U.S. fleet is required to take Fisheries Forum Agency (FFA) observers on a minimum of 20 percent of their fishing trips, and captains are responsible for recording catch and bycatch data in logbooks. Logbooks are verified by observers, if possible, and are sent to the FFA no later than 14 days after returning to port (K. Staisch, FFA, personal communication, February, 2001). Between 1997 and 1999, there was approximately 20-23% observer coverage (Forum Fisheries Agency, 1998; A. Coan, personal communication, February, 2001). Collecting data on target species (i.e. tuna) is a priority for observers; however, if possible, and when time permits, observers do collect bycatch data. Observers receive limited training on sea turtle identification and are trained to look for tags, but they do not collect information on length or take biopsies, as the turtles are generally released immediately from the net. The incidental catch of sea turtles is a “rare occurrence,” and any turtles observed taken have been released alive. Purse seine techniques normally allow turtles to surface for air during the pursing period, and based on observer reports, any turtles caught in nets are usually released as soon as possible. In addition, there have been no reports of turtles caught in the power block (K. Staisch, FFA, personal communication, February, 2001).

(2) U.S. tuna purse seine fishery in the eastern tropical Pacific Ocean (ETP)

The HMS FMP includes the purse seine fishery within the ETP. However, except for smaller vessels and one or two large vessels, the remainder of the fleet operates outside of the U.S. EEZ, does not land catches at U.S. ports, and does not therefore require permits under the HMS FMP. In addition, NOAA Fisheries has previously consulted on the operations of the large vessel purse seine fleet (described below). The proposed FMP makes no changes to that fishery. As a result, the large vessel ETP purse seine fleet is described here in the *Status of the Species and Environmental Baseline*. The operations of small vessels are not covered under any previous consultations, and these vessels fall under the jurisdiction of the HMS FMP. Their past operations (to the extent we ave information) are covered here. The effects of their future operations under the FMP are covered in the subsequent section *Effect of the Action*.

As shown in Table II-2, between 1996 and 2001, between 5 and 6 large U.S. vessels actively fished in the ETP. During that same period, the number of small U.S. purse seiners in this fishery ranged between a low of 3 to a high of 19. Between 1999 and 2001, the number of small vessels has declined, from 9 in 1999, to 7 in 2000, and finally to 3 small purse seine vessels in 2001.

Although all large tuna purse seine vessels fishing in the ETP for tuna have been required to carry observers since 1989 (100 percent coverage), smaller purse seine vessels are not required to carry observers. Most smaller tuna vessels fishing off southern California fish on tuna schools because the vessels are old, slow, and lack the resources (e.g. helicopters) needed to place and find floating objects (B. Jacobson, NOAA Fisheries, personal communication, 1999). Based on observer data from the large vessels, the chances of incidentally capturing a sea turtle during a school set are much less than incidentally capturing a sea turtle during floating object sets; therefore, the incidental take of sea turtles by the small vessel fleet is likely to be less than that of the larger purse seine vessels. However, with no observer coverage, data on sea turtle bycatch are not available for the small tuna purse seine vessels in the ETP.

Table IV-31a shows sea turtle interactions in the large U.S. tuna purse seine fleet from 1990 to 1997. Table IV-31b shows sea turtle interactions in the fleet from 1998 through 2001 - note that data for 2001 is incomplete. Observer data from 1990-97 indicated that sea turtles caught by the U.S. tuna purse seine fleet had a high survival rate. Approximately 90 percent of the sea turtles caught were released unharmed (1002 released unharmed/1104 total captured), 3.8 percent were released slightly injured (42 slightly injured/1104 total), and 0.6 percent were killed accidentally (7 killed/1104 total). The rest either escaped the net, or were previously dead. "Escapes" were often the result of turtles entangled outside the net and dropping free during the net roll. "Previously dead" recordings were for turtles that were obviously dead before they became entangled, and these were not recorded as "takes." Overall, approximately 95 percent of captured or entangled sea turtles were released unharmed, or uninjured, or escaped from the net.

Table IV-31a. Sea turtle interactions by U.S. tuna purse seine fleet in the ETP (1990 - 1997) - large vessels only*

Set Summary / by calendar year 1/1 - 12/30									
Cruise Year	**1990[1]**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**Total**
Number of sea turtles taken (mortality in parentheses) by species[2]									**Annual Average**
Olive ridley	113(2)	104	132	133(1)	69	69(1)	45(1)	95(1)	**96**
Green turtle	4	8	21	35	28	29	17	11	**19**
Leatherback	3	0	0	2	1	0	0	0	**0.8**
Loggerhead	0	1	0	0	3	0	0	2	**0.8**
Unidentified	36	37	25(1)	21	19	3	25	8	**22**
Totals	**156**	**150**	**178**	**191**	**120**	**101**	**87**	**116**	**137**
Condition of sea turtle when released (injury/mortality due to set)									**Annual Average**
Prev. dead	0	0	2	1	4	2	0	2	**1.4**
Released unharmed	126	137	168	181	115	92	73	110	**127**
Released slightly injured	13	5	7	1	3	6	5	2	**5.3**
Kill accidentally	2	0	1	1	0	1	1	1	**0.9**
Escaped net	11	5	3	6	2	0	7	3	**4.7**
Other/unknown	3	3	0	2	0	4	1	2	**1.9**
Totals	**156**	**150**	**181**	**192**	**124**	**105**	**87**	**120**	**141.1**

[1] First year of sea turtle data collection, did not began until 3/20. Summary reflects cruises from 3/20/90 - 12/30/90, when data was collected. 1,629 sets out of 1,814 for 1990 were observed for sea turtles.

[2] Mortalities are a subset of total incidental take.

*Note: there is some discrepancy between the numbers in the two parts of the table because previously dead turtles were not included in species estimates and hawksbill turtles were not included in the top part of the table and not accounted for it in the lower part

Table IV-31b. Sea turtle interactions by U.S. tuna purse seine fleet in the ETP (1998 - 2001) - large vessels only (note: 2001 data is incomplete).

Species/condition	1998	1999	2000	2001
Green turtle				
Released unharmed	3	5	2	2
Total	**3**	**5**	**2**	**2**
Loggerhead				
Released unharmed	0	1	5	0
Total	**0**	**1**	**5**	**0**
Olive ridley				
Released unharmed	38	27	3	16
Light injuries*	4	6	2	0
Grave injuries**	1	0	0	3
Escaped/evaded net	0	0	1	0
Total	**43**	**33**	**6**	**19**
Unidentified turtle				
Released unharmed	2	0	3	5
Light injuries*	0	0	0	1
Escaped/evaded net	2	1	1	0
Other***	1	0	0	1
Total	**5**	**1**	**4**	**7**

*Light injuries are considered to be those that would not be lethal to the turtle
**Grave injuries are considered to be those that would eventually cause death.
***”Other” refers to an unknown condition

In its December 8, 1999, biological opinion on the effects of the interim final rule for the continued authorization of the ETP U.S. tuna purse seine fishery on listed species, NOAA Fisheries estimated the maximum annual incidental takes and mortalities of sea turtles for 2000-2010: green - 35 taken, 2 killed; leatherback turtles - 2 taken, 1 killed every 10 years; loggerheads - 3 taken, 1 killed every 7 years; olive ridleys - 133 taken, 7 killed (NOAA Fisheries, 1999).

NOAA Fisheries does not expect additional large U.S. purse seine vessels to enter the ETP tuna purse seine fishery in the future because of historical trends in vessel participation and the high start-up costs for a new large vessel to enter the fishery. In the late 1980s and early 1990s, with the passage of the

South Pacific Regional Tuna Treaty, most U.S. large purse seiners either re-flagged or moved to the richer fishing grounds of the central-western Pacific Ocean. With little incentive to fish in the ETP, NOAA Fisheries does not expect a future influx of large U.S. purse seine vessels. A recent IATTC resolution which set fleet limits and a voluntary U.S. commitment to limit participation of domestic vessels to a total 8,969 metric ton capacity (Chris Fanning, NOAA Fisheries, pers. comm., January 14, 2004) are also expected to limit or preclude future increases in large U.S. purse seine vessels.

Although large (>400 st carrying capacity) U.S. tuna purse seine vessels are allowed to set on dolphins to catch tuna in the ETP, such fishing is not expected to occur because U.S. canneries have stated that they will not buy tuna that has been caught by setting on dolphins. United States purse seine vessels are expected to continue fishing on schools and floating objects at the same approximate levels as in the past. Based on data from 1993-1997, U.S. vessels fishing on floating objects and schools are capture sea turtles at the rate of just over 234 animals per 1,000 floating object sets, and 75 animals per 1,000 school sets. Therefore, NOAA Fisheries expects that there would be no change in the current number of sea turtles taken annually by U.S. vessels in the ETP. However, if U.S. purse seine vessels in the ETP shifted from floating object or school fishing to setting on dolphins to capture tuna, the level of sea turtle mortality would decline because the capture rate of sea turtles in dolphin sets (37 sea turtles per 1,000 sets) is much less than in log sets or school sets, and the survival rate would be expected to stay the same. Any decrease in sea turtle mortality as a result of shifting fishery operations is expected to benefit all affected sea turtle species.

(3) American Samoa-based longline fishery

For the American Samoa-based longline fishery, the federal logbooks from 1992 through 1999 indicate six interactions with sea turtles (i.e. hooking/entanglement). In 1992, one vessel interacted with a green turtle. In 1998, one vessel interacted with an unidentified sea turtle; it was released alive. In 1999, one vessel reported interactions with four sea turtles. Three turtles released alive were recorded as a hawksbill, a leatherback, and an olive ridley. One turtle, identified as a green, was reported to have died from its interaction with this vessel. None of the species' identification were validated by NOAA Fisheries' Southwest Fisheries Science Center; and NOAA Fisheries cannot attest to the local knowledge of fishermen regarding the identity of various turtle species, particularly hard-shelled turtles. However, all four species of sea turtles reportedly caught by the fishery do occur in the fishing grounds of this longline fishery. In addition, as discussed below, logbook data may not be a reliable method to measure sea turtle interaction in the fisheries. From 2000 through October 2002, there have been no reported interactions with sea turtles in this fishery (S. Pooley, NOAA Fisheries, personal communication, October 2002).

(4) Hawaii-based longline fishery (Post- 2000)

The Hawaii-based longline fishery now operates under management measures, described in the *preferred alternative* of the final Environmental Impact Statement (FEIS) completed on March 30, 2001, which were implemented to mitigate adverse impacts on sea turtles (67 FR 40232, June 12, 2002)[17]. The rules were adopted by the Council under the authority of the MSA, approved by the Secretary of Commerce, and implemented by NOAA Fisheries. They prohibit swordfish-targeted longline fishing, impose a seasonal closure in waters south of Hawaii (from the equator to 15°N and 145°W to 180°) during April and May, ban the possession of light sticks, and limit the possession of 10 swordfish per trip by any Hawaii-based longline vessel. The definition of swordfish-target or shallow-set longline gear is described in the March 2001 FEIS.

The Hawaii-based longline fishery is a limited access fishery, with a total of 164 permits that are transferable. Vessels active in this fishery are limited to 101 feet in length. The area fished ranges as close as 25 miles from Hawaii to thousands of miles from port. These Hawaii-based longline vessels compete with foreign distant water fishing fleets operating on the high seas. In 2001, 101 Hawaii–based longline vessels made 1,034 trips, almost all of which targeted tunas. Swordfish was a major target species of this fishery prior to 2001, but due to conservation measures to protect sea turtles this segment of the Hawaii-based longline fishery was phased out completely by the end of 2001.

Vessels based out of Hawaii targeting tuna deploy about 34 horizontal miles of main line in the water. Vessels targeting tuna typically use a line shooter. The line shooter increases the speed at which the main line is set which causes the main line to sag in the middle (more line between floats), allowing the middle hooks to fish deeper. The average speed of the shooter is 9 knots. The vessel speed is about 6.8 knots. No light sticks are used as the gear soaks. The float line length is about 22 meters (72 feet) and the branch line lengths are about 13 meters (43 feet). The average number of hooks deployed is about 1,690 hooks per set with about 27 hooks set between each float. There are approximately 66 floats used during each set. Deep set vessels use saury (sanma) as bait and the hook type used are "tuna" hooks. The average target depth is 167 meters. The gear is allowed to soak during the day and the total fishing time typically lasts about 19 hours, including setting and hauling of gear. This type of set is referred to below as "deep set."

Table IV-32 contains rough estimates of the annual capture and mortality of sea turtles in the Hawaii-based longline fishery, based on past interactions between July 1, 2001 and June 30, 2002.

Table IV-32. Annual sea turtle capture and mortality estimates in the Hawaii-based longline fishery.

Species	Incidental Take	Incidental Mortality[1]
Green	8	7
Leatherback	8	3
Loggerhead	14	8

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[17] Sea turtle mitigation measures for the Hawaii-based longline fishery were initially promulgated as an emergency interim rule on June 12, 2001 (66 FR 31561).

Olive Ridley	26	24

[1] The estimated incidental mortality is a subset of the estimated incidental take by hooking or entanglement.

2. Direct harvest

a. Solomon Islands

Between 1993 and 1996, Broderick (1997) investigated the subsistence harvest of green (and hawksbill) turtles by people from three different communities, Kia, Wagina, and Katupika on the Solomon Islands. At Kia, the majority of turtles are consumed for feasts, and the meat of the green turtle is more highly valued than that of the hawksbill. Broderick (1997) estimated that a minimum of 1,068 green turtles were harvested per year, and most were immature turtles.

b. Indonesia

In the Kai Islands (also spelled "Kei Islands"), located approximately 1,000 kilometers southwest of the Papua nesting beaches, adult leatherback turtles are traditionally hunted and captured at sea by local people. Villagers hunt leatherback turtles only for ritual and subsistence purposes, and, according to their beliefs (known as *adat*), they are forbidden to sell or trade the meat. However, due to population increase and deforestation of the area which has lead to the loss of forest resources such as deer, pigs, and birds, villagers are taking leatherback turtles more for their increased need for meat for subsistence than for traditional purposes (Suarez and Starbird, 1996b). The carapace is rendered for oil, and the meat from the plastron is shared among villagers (Starbird and Suarez, 1994). Based on a study conducted during October-November, 1994, Suarez and Starbird (1996a) estimated that approximately 87 leatherback turtles were taken annually by villagers in the Kai Islands, and this estimate did not include incidental take by local gill and shark nets. Locals report that sea turtle populations in the area have declined dramatically (Suarez, 1999). Overall, approximately 200 leatherback turtles, both adult males and females, were estimated killed per year in these traditional fisheries southwest of Kai Kecil during October-April (*in* Chan and Liew, 1996) (the Kai Islands take is assumed included in this estimate).

While takes of adult leatherbacks are continuing, approximately 20 leatherback turtles are currently taken per year, as villagers are reportedly too busy in village activities and local economy to be hunting (Hitipeuw, WWF, personal communication, December, 2003). In addition, a specialist from a local non-governmental organization is currently working with the eight villages of the Kai Islands to explore the potential for a community-based harvest monitoring as well as alternative substitutes for the traditional harvest of leatherbacks. The main strategy is to gain community support for sea turtle conservation. Harvest monitoring and research initiatives are scheduled to be implemented during the next hunting period (November, 2003-February, 2004) (Hitipeuw, 2003b).

c. Mexico

Because studies of sea turtle mortality in Mexico focused on both fisheries bycatch and directed harvest, a summary of estimated mortality due to harvest is contained above in section 1(p) (above).

d. Peru and Ecuador

The Ministerio de Pesqueria (MIPE), which is the Peruvian agency responsible for fisheries, prohibited the taking of all leatherback turtles and green turtles less than or equal to 80 cm in length through a resolution in January, 1977 (Weidner and Serrano, 1997). In 1995, the Peruvian government prohibited the capture, trade, and consumption of green turtles, leatherbacks, olive ridleys, and hawksbills. However, in many ports of Peru, this decree was and is poorly enforced, and sea turtles were widely caught for human consumption. Noted Peruvian ports included Pisco, Chincha, Pucusana, Callao, and Chimbote (Alfaro-Shigueto, *et al*., 2002).

Peru conducted directed commercial turtle harvests throughout the 1980s, and, as recently as 1990, over 100 metric tons of turtles were taken (Table IV-33; FAO, Yearbook of Fishery Statistics, 1994, *in* Weidner and Serrano, 1997). Species-specific information was not available. Based on a sighting of 167 leatherback carapaces in a canyon near the port of Pucusana in 1978, Brown and Brown (1982) estimated a minimum of 200 leatherback turtles killed per year at that time. Furthermore, central Peru was known to have had the largest leatherback fishery in the world, taking what appeared to be adults and subadults, thus representing a considerable number of reproductive and near reproductive individuals (*in* Brown and Brown, 1982).

Table IV-33. Ecuador and Peru - turtle catch in metric tons, 1985-95.

Year	Catch - Ecuador (metric tons)	Catch - Peru (metric tons)
1985	124	36
1986	715	9
1987	–	305
1988	–	32
1989	–	79
1990	–	101
1991	–	9
1992	–	30
1993	–	28

1994	–	6
1995	10*	4*

Source: FAO, Yearbook of Fishery Statistics, 1994, *in* Weidner and Serrano (1997)
*1995 data would not be found in the above source, yet Weidner and Serrano (1997) provide data for this year.

Researchers from the Peruvian Centre for Coastal Research also opportunistically collected data on sea turtle captures while collecting data on dolphin mortality. They present data on sea turtle mortality in two ports, Cerro Azul and Chimbote in 1993 and 1994, and compile data on leatherback capture along the Peruvian coast from 1984-1999. Sea turtles, particularly olive ridleys and green turtles, are commonly taken with "animaleros," which are large mesh drift gillnets targetting sharks and rays, but take dolphins and sea turtles as bycatch. Researchers provided a minimum estimate of 77 turtles taken in 11 months (1993) and 45 turtles taken in 8 months (1994) in Cerro Azul. In Chimbote, researchers estimated a minimum of 133 turtles taken in approximately 7 months (1993). Species composition of observed turtles taken included both olive ridleys and greens (83.2%) and leatherbacks (16.18%) (Alfaro-Shigueto, *et al*., 2002).

During 1985-1999, researchers observed at least 33 leatherbacks, alive and dead, along Peruvian beaches, in fishmarkets, or in dumps located in Pisco, Cerro Azul, Pucusana, Ancón, Chancay, Huacho, Chimbote and Salaverry. In addition, remains of at least two dozen leatherbacks were found in fish offal dumps in Pucusana in 1984, the same area where a large number of leatherback carapaces were found in 1978 (see above) (Alfaro-Shigueto, *et al*., 2002).

d. Vietnam

In Vietnam, there is a high demand for sea turtle products in the market, and as a result, green turtles and hawksbills have been harvested heavily to supply this demand. Direct harvest of sea turtles is common among the coastal communities, where turtles forage and breed. In addition, sea turtle eggs are collected for food. Poverty in the country and a lack of awareness of the conservation of resources are partially to blame for this exploitation; in addition, there are no regulations and little government support for sea turtle research and conservation efforts (Hien, 2002). Unfortunately, no quantitative estimates are available on the level of sea turtle mortality or the number of eggs taken.

e. Australasia (Bali, Torres Strait)

Bali appears to have the largest trade in live green turtles. Reports from WWF/IUCN (1984 *in* Dermawan, 2002) indicate that green turtles have been collected from all over Indonesia in order to supply Bali with up to 30,000 turtles. Turtles have been used as a standard source of food and in religious festivities in southern Bali (within the Balinese-Hindu culture) for many years, and the demand is increasing (Dermawan, 2002). While traditional religious ceremonies require the use of sea turtle meat, Hindu high priests have estimated that only 300 to 500 turtles annually should serve that purpose (*in* Dethmers and Broderick, 2003). The average demand for sea turtles in Bali alone is approximately 17,000 per year, although the government only permitted the harvest and slaughter of up to 3,000 turtles per year. With green turtles foraging near and nesting on Bali decreasing, the sea turtle fishery out of Bali has had to expand to more distant foraging and nesting populations throughout the Indonesian archipelago. This has required larger vessels and a network of hunters, traders, and shippers (Dethmers and Broderick, 2003).

In the Torres Strait, both a commercial fishery and a subsistence fishery operates, taking substantially fewer turtles than the Balinese fishery. In the subsistence fishery, Islanders use small aluminum dinghies and deploy small nets or use traditional gear, typically within a day's journey from their village. Sea turtles are consumed for subsistence or used in traditional feasts. In the late 1980s, the commercial fishery was estimated to take 5,000 and 10,000 sea turtles annually and is marketed through Daru in Papua New Guinea (Limpus and Parmenter, 1986 and Groombridge and Luxmoore, 1989, both *in* Dethmers and Broderick, 2003).

Based on analysis of genetic data collected from green turtles from the Bali and Torres Strait region as well as a feeding aggregation in Aru, researchers analyzed the extent of the fisheries' impact on genetic stocks. There are 17 genetic stocks throughout the Australasian region. Researchers found that the Bali fishery is impacting several green turtle stocks throughout the region, with few stocks unaffected, while the Torres Strait fishery, having a more local focus, affects the NGBR almost exclusively (Dethmers and Broderick, 2003).

Turtle meat is reportedly sold at several restaurants in Indonesia and has been exported to Japan, Hong Kong, South Korea, and Europe. In 2001, the Indonesian government began to more strictly implement the existing laws and confiscated several shiploads of live turtles and temporarily closed turtle slaughterhouses on the island (Dermawan, 2002).

f. Fiji

Of the main threats to sea turtle populations around Fiji, mortalities due to the traditional harvesting of adults for ceremonial purposes, and subsistence and commercial harvesting of adults, eggs, and shells are significant. Traditionally, sea turtles were consumed for special occasions; however, eggs were not used for such feasts. As the tradition has weakened, sea turtles have been considered more common property and have been harvested for general consumption as well as for sale in local markets and exports. For example, approximately 30,000 hawksbill shells were exported during the 1980s, with approximately 2,000 kilograms of shells exported in just 1989. In addition, eggs have also been harvested for subsistence and commercial purposes. Hunting for sea turtles in Fiji is relatively easy because it is generally unregulated and uncoordinated. Currently, Fijians are prohibited from taking turtles and their eggs during the breeding season (December through March), and there was a moratorium on the killing of turtles and poaching of eggs (including trade of turtle meat and eggs) through December, 2000. The Department of Fisheries is hoping to extend this moratorium (Rupeni *et al*. 2002).

g. Australia

Anecdotal information indicates that from 100 to up to 1,000 southern Great Barrier Reef sea turtles are taken by hunters for traditional purposes (K. Dobbs, 2002). It is unclear as to whether this number is "per year" or over what period, or what species are taken.

h. Philippines

In the Philippines, despite a significant increase in conservation awareness in the past decade, turtles are still killed and sold for their meat and eggs are also taken and sold. This primarily occurs in remote areas of the country and the reasons are the following: (1) lack of law-enforcement personnel in the area; (2) lack of implementation of existing local and national laws/ordinances/orders; (3) penalties are not enough to deter violators; (4) traditional use of turtles, especially during celebration of town fiestas and weddings; and (5) poverty. Each year, an estimated 1,000 nesters are being killed (Cruz, 2002), and given that greens primarily nesting in the Philippines, they are the likely species being killed.

3. Scientific Research Permits

a. Scientific Research Permit #1277

Scientific Research Permit #1277 was issued under an ESA Section 10, to the Southwest Fisheries Science Center, NOAA Fisheries, in part to study habitat use, home range, stock structure, and migration patterns of the leatherback turtle in the Pacific Ocean. During a routine capture of a 150 cm male leatherback in Monterey, California in August, 2003, the turtle died. Subsequent necropsy revealed that this animal had several chronic conditions believed to have compromised its health. Although this is an extremely rare event, it has prompted the need to collect baseline data on the health and physiology of leatherbacks (P. Dutton, NOAA Fisheries-SWFSC, personal communication, January, 2004).

b. Scientific Research Permit #1303

Between March 1, 2002 and July 31, 2002, NOAA Fisheries observed 16 trips, 194 sets, and 159,468 hooks. During this time period, 2 loggerheads and 1 leatherback were released alive and 1 sperm whale was released injured. These are totals for the experiment to date based on 100% observer coverage. This permit is currently suspended.

4. Other Impacts

Threats to sea turtles vary among the species, depending on their distribution and behavior. The value of their meat, eggs, shell or other parts plays an important role in the extent of directed harvest. All sea turtle life stages are vulnerable to human-induced mortality. On nesting beaches, direct exploitation of turtles for meat, eggs, skin or shell, and other products takes place for both commercial markets and local utilization, and to a much lesser degree for traditional ceremonies. Nesting beach and in-water habitat degradation and destruction have occurred due to many factors, including coastal development, dredging, vessel traffic, erosion control, sand mining, vehicular traffic on beaches, and artificial lighting, which repels the adults and disorients the hatchlings. In areas where recreational boating and ship traffic is intense, propeller and collision injuries are not uncommon. Human alteration of terrestrial habitats can also change the feeding patterns of natural predators, thereby increasing predation on marine turtle nests and eggs. In addition, the hawksbill's dependence on coral reefs for shelter and food link its well-being to the condition of the reefs. Destruction of reefs from vessels anchoring, striking or grounding is a growing problem.

Petroleum and other forms of chemical pollution (pesticides, heavy metals, and PCB's) affect turtles throughout their marine and terrestrial habitats and have been detected in turtles and eggs. Poisoning, as well as blockage of the gastrointestinal tract by ingested tar balls, has been reported. Low level chemical pollution, possibly causing immunosuppression has been suggested as one factor in the epidemic outbreak of a tumor disease (fibropapilloma) in green turtles. Plastics and other persistent debris discharged into the ocean are also recognized as harmful pollutants in the pelagic environment. Marine turtles such as leatherback turtles actively feed on jellyfish, and plastic bags floating in the water potentially resemble such prey in form, color and texture. Hawksbills also eat a wide variety of debris such as plastic bags, plastic and styrofoam pieces, tar balls, balloons and plastic pellets. Ingested plastics can occlude the gut, preventing or hampering feeding, and causing malnutrition or starvation. Both the entanglement in, and ingestion of, this synthetic debris have been documented (*in* NOAA Fisheries and USFWS, 1998a-e).

C. Status Summary of Sea Turtle Species

All listed sea turtle populations affected by the proposed action have been impacted by human-induced factors such as commercial fisheries, direct harvest of turtles and eggs, and modification or degradation of the turtle's terrestrial and marine habitats. Nesting beach habitat impacts have resulted in the loss of eggs and hatchlings as well as the deterrence of nesting females, resulting in decreased nesting success. In the marine environment, a significant anthropogenic impact is the incidental capture and mortality of subadult and adult sea turtles in various commercial fisheries. Generally, mortality resulting from the effects of marine pollution are important but less significant (NOAA Fisheries and USFWS, 1998a-e). Increased mortality from these anthropogenic sources at the egg and early life history stages has impacted the species' ability to maintain or increase their numbers by limiting the number of individuals that survive to sexual maturity. In addition, the human-induced mortality of adult females results in the loss of their future reproductive output. The age at sexual maturity of loggerheads may be as high as 35 years, while green turtles may not reach maturity until 30-60 years (*in* Crouse, 1999). Upon reaching maturity, female sea turtles generally lay between 100-130 eggs per clutch, minimally 2-3 clutches per year, every 2-4 years. Thus, in general, a female sea turtle will lay between 200-390 eggs per season over an average of 2-4 years.

The potential for an egg to develop into a hatchling, into a juvenile, and finally into a sexually mature adult sea turtle varies among species and populations, as well as the degree of threats faced during each life stage. Females killed prior to their first successful nesting will have contributed nothing to the overall maintenance or improvement of the species' status. Anthropogenic mortality and natural mortality of females (or males, for that matter) prior to the end of their reproductive life results in a serious loss of reproductive potential to the population. While quantitative data do not yet exist to provide a precise understanding of the effects of the loss of reproductive potential, the status and trends of the turtles themselves are the best evidence that sea turtle populations cannot withstand current mortality rates. In the face of current levels of mortality and extent of habitat degradation, nesting aggregations of green, leatherback, and loggerhead turtles have declined to levels that place them at a very high risk of extinction within the foreseeable future. Of the sea turtles considered in this Opinion, Hawaii green turtles are increasing, and olive ridley turtle nesting aggregations in the western Pacific appear to be somewhat stable or increasing slightly.

V. EFFECTS OF THE PROPOSED ACTIONS

Pursuant to Section 7(a)(2) of the ESA (16 USC §1536), federal agencies are directed to ensure that their activities are not likely to jeopardize the continued existence of any listed species or result in the destruction or adverse modification of critical habitat. During this consultation, NOAA Fisheries has analyzed the effects of the actions on the listed species to determine whether the actions, individually or cumulatively are likely to jeopardize the continued existence of that species. This analysis is done after a careful review of the listed species' status and the factors that affect the survival and recovery of that species, as described above.

The proposed action is the management of HMS fisheries, as described in the HMS FMP, as proposed by the Council for approval and implementation by the Secretary of Commerce. Therefore, the management regime, as described in the proposed FMP, constitutes the action being considered in this Opinion. Among the fisheries being considered is those that operate (even occasionally) on the high seas and that consist of vessels with HSFCA permits issued by NOAA Fisheries. These include the West Coast-based longline fishery that operates exclusively on the high seas, the albacore troll fishery, and the ETP purse seine fishery, both of which occasionally fish on the high seas. The Opinion also evaluates the likely impact of a proposed rule to implement longline fishery controls in exclusive economic zone (EEZ) waters east of 150° west longitude.

A. Marine Mammals

Of all the HMS fisheries, listed marine mammals have only been observed taken in the CA/OR drift gillnet fishery. The following is a discussion of the effects of drift gillnet gear and fishery operations on listed marine mammals.

1. General impacts to marine mammals from drift gillnet operations

In the CA/OR drift gillnet fishery, a wide variety of marine mammals are killed, which is most likely attributable to the large geographic range of many of the species, nonselectivity of gear, and the amount and location of fishing effort. For example, cetacean bycatch in the CA/OR drift gillnet fishery is greater and more diverse than for the California set net fishery because the area of driftnet effort contains more diverse habitat than the area of the set net fishery.

The probability that a marine mammal will initially survive an entanglement in fishing gear depends largely on the species and age of marine mammal involved. For instance, larger animals such as fin whales, humpback whales and sperm whales may become entangled in gillnet but often survive the initial contact with the gear. Such entanglement may cause considerable damage to the gear, as the large whales "punch" through and continue swimming. Such damage may be related to the type of net used, however, for fishermen do report that large blue and fin whales usually break through drift gillnets without entangling, and that very little damage is done to the net (Barlow, *et al*., 1997).

Marine mammals may also swim away with a portion of the gillnet wrapped around a pectoral fin, the tail stock, the neck or the mouth. For large whales, there are generally three areas of entanglement in a net: 1) the gape of the mouth, 2) around the flippers, and 3) around the tail stock (although this area is often difficult to view, as most balaenopterids do not fluke frequently). Documented cases have indicated that entangled animals may travel for extended periods of time and over long distances before either freeing themselves, being disentangled by an outside network, or dying as a direct result of the entanglement (Angliss and DeMaster, 1998). In most cases, it is unknown whether the injury is serious enough or debilitating enough to lead to death. If the debris fragments are heavy, the animal will most likely drown. Less heavy fragments may lead the animal to exhaustion, depletion and starvation due to the increased drag (Wallace, 1985). In addition, if an animal's appendage is caught in a mesh, the debris can debilitate the animal, especially if it is constricting, causes lacerations, or impairs swimming or feeding ability (Scordino, 1985). Younger animals are particularly at risk if the entangling gear is tightly wrapped, for as they continue to grow, the gear will likely become more constricting. The majority of large cetaceans that become entangled are juveniles (Angliss and DeMaster, 1998). Marine mammals that die as a result of entanglement in drift gillnets usually drown. With a typical soak time of 12-14 hours, the animal is unable to survive without oxygen, especially if it is entangled at the beginning of the set, or deep in the net.

Marine mammals may also be indirectly affected as a result of being captured in a drift gillnet. An entanglement may compromise the animal by causing cuts or impeding mobility or feeding, which may make the animal more susceptible to disease or predation. In addition, although marine mammals have evolved to handle a wide variety of stressors, including a saline environment, predation, food shortages, etc, only healthy animals have an optimal healing response. Cetaceans in particular have developed a very unique healing process, which requires salt and water to kill several cell layers to block penetration of additional salt water. After this process is completed, healing from within can begin. A sustained stress response, such as repeated or prolonged entanglement in gear, makes marine mammals less able to fight infection or disease (Angliss and DeMaster, 1998).

In the CA/OR drift gillnet fishery observers record detailed information on marine mammals entangled in the net. Animals that are released alive from the net with netting attached are classified as “injured.” Animals that release themselves or are released from the net by fishermen and can swim normally are recorded as “alive.” Marine mammals that have been entangled and are released alive usually only have minor abrasions as a result of interaction with the net. However, as discussed above, effects from the stress of capture may cause temporary and/or long-term effects that may not be visible upon release. Because long-term stress studies have not been conducted on the impacts of capture by a fishery on marine mammals, NOAA Fisheries is only able to make assumptions on the condition of marine mammals that have been released “unharmed” from a drift gillnet. Although marine mammals released “unharmed” do not have visible injuries, they may have been stressed from being caught or entangled in a net. This stress may cause an interruption in essential feeding behaviors or migration patterns; however, NOAA Fisheries believes this effect, if experienced, is likely to be temporary and short-term. For these reasons, NOAA Fisheries will assume that most of the marine mammals released and reported as “unharmed,” or uninjured, have not been harmed or harassed by their capture in a drift gillnet, and that latent effects are limited to short-term physiological stress or interruption of normal behavioral patterns.

All marine mammal species that forage or migrate by diving or swimming at depth in areas of concentrated fishing effort are vulnerable to drift gillnets. Susceptibility to capture largely depends on a species’ physical characteristics and behavior. Not surprisingly, survival rate likely varies among marine mammal species incidentally taken by the CA/OR drift gillnet fishery. This is due in part to variations in size and diving and foraging behavior, as well as location in the net and time of capture. With few observed marine mammal captures in the CA/OR drift gillnet fishery, it is difficult to speculate as to the survival rate of the three listed species observed taken in the fishery from 1990-2003. However, because the baleen whales (humpback and fin) and the sperm whale differ so greatly in the nature of their food and foraging behavior (e.g. the sperm whale is capable of diving to much greater depths than the baleen whales in order to find their preferred prey of squid, depending largely on oxygen storage and metabolism, while the baleen whales rely less on diving, if possible, and tend to skim and gulp for euphausiids at the surface or below) and their physiology, survival rates following gillnet entanglement most likely vary greatly as well. Of the 8 sperm whales entangled in CA/OR drift gillnet gear, 3 survived uninjured (37.5 percent), 1 was released injured (12.5 percent), and 4 were killed (50 percent). Of the 3 baleen whales entangled in drift gillnets from this fishery, 2 were released alive (both humpback whales) (66 percent), and one was killed (fin whale) (33 percent).

a. *Fin whale impacts*

The incidental take of fin whale in the CA/OR drift gillnet fishery is extremely rare. From July, 1990 until December, 2003, observers recorded the entanglement and mortality of only one fin whale by the fishery, in 1999, off southern California. The net had a full complement of pingers (40), and had 36 foot extenders, as required by the PCTRP.

The fin whale taken in 1999 was entangled southwest of San Clemente Island, in an area characterized by a generally counterclockwise current flow or gyre centered in the Gulf of Catalina. About the center of the current gyre, sea surface temperatures tend to be higher than temperatures found to the north or south of the Gulf of Catalina. These warmer temperatures attract subtropical species such as striped marlin and swordfish, as well as large whales, such as the fin whale. In addition, coastal upwelling areas are prime foraging areas for fish and marine mammals, attracted to the high primary productivity. The local distribution of fin whales during much of the year is probably governed by prey availability. Like swordfish, fin whales have been known to associate with steep bottom contours, most likely because tidal and current mixing along such gradients drives high biological production. During the year immediately following the 1997-98 El Niño event, zooplankton production was exceptionally high, primarily because this period saw a transition from the warm-water conditions associated with the El Niño event to cool water conditions which were still prevalent in coastal southern California in October, 1999. Because euphausiids are a fin whale's prey of choice, this fin whale was most likely taking advantage of the locally high biological productivity, either by surface feeding, or foraging by diving. From November, 1999 through January, 2000, an anomalously high upwelling event occurred off southern California, which most likely increased primary productivity and attracted large whales to the area. Observers also recorded the incidental take of one humpback and one minke whale (*Balaenoptera acutorostrata*), two other baleen whale species rarely taken by the fishery, on the same day (11/29/99) and in the same general area that the fin whale was taken, further indicating that high forage density may have played a role in the fin whale interaction.

Fin whales are very rarely taken in the CA/OR drift gillnet fishery. Based on a worst-case scenario, NOAA Fisheries estimates that a maximum of 4 fin whales ((1 fin whale observed entangled and killed in 1999/526 sets observed in 1999) x 2,000 maximum expected sets per year) could be captured by the CA/OR drift gillnet fleet and killed. Based on anecdotal reports from fishermen, who have evidence of large whales punching through their nets, fin whales have likely interacted with the CA/OR drift gillnet fishery before. However, because of their size and strength, fin whales likely punch through the net, and entanglement is a rare event. Entanglement, and any associated mortality, of fin whales is not anticipated to occur every year. Based on past fishery performance, fin whales were observed taken once in thirteen years, or once during the six years the PCTRP has been in place. However, following the implementation of protective measures to avoid the likelihood of jeopardy to leatherback and loggerhead sea turtles in the CA/OR drift gillnet fishery, no fin whales have been observed taken in the fishery.

b. *Humpback whale impacts*

From July, 1990 to October 29, 1997, the day before the effective date of the PCTRP, observers recorded the incidental entanglement of one humpback by the CA/OR drift gillnet fishery, in 1994, off southern California. This animal was released alive and uninjured. Following the implementation of the PCTRP, only one humpback was observed entangled, in 1999, off southern California; this animal was also released alive and uninjured. The net had a full complement of pingers (41) and 36 foot extenders.

Both humpback whales caught by this fishery were caught south of Point Conception during years immediately following El Niño events (1992-93 and 1997-98), and during the months (August and November) when humpback whales typically are found north of their breeding grounds, taking advantage of coastal upwelling events. Humpback whales feed both at the surface and at depths. Surface feeding is characterized by fast, short-duration dives, and rapid surface swim speeds compared with deep diving. Humpback whales observed off the California continental shelf from 1988-90 primarily fed on euphausiids; however their foraging behavior changed as environmental conditions changed. The whales fed at the surface 56% of the time in 1988 and 32% of the time in 1990, using primary lateral lunges to capture swarms of euphausiids. In 1989, however, no surface feeding was observed; instead, deep, long-duration dives were followed by extended surface intervals with many respirations. These 1989 observations coincided with increased prey depth as indicated by depth sounder records of diving whales and prey scattering layer. The increased prey depth and associated feeding behaviors were strongly associated with unusually high sea surface temperatures, calm seas, and changes in water circulation (Kieckhefer, 1992).

The humpback observed entangled in 1994 was taken in an area and during a time of the year (August) when the average monthly sea surface temperature was approximately 20EC, and about 0.5-1.0EC above normal (Coastwatch El Niño watch). Although there was coastal upwelling in the area, which could have brought food to the surface for the whale, the animal may have had to forage at depth, causing it to interact with the driftnet gear. The humpback observed entangled in 1999 was taken in an area and at a time (November) when the fishery was observed to capture a higher number of large whales and sea turtles than normal. The waters off southern California during this time period were characterized by an extremely strong and anomalous upwelling event. Marine mammals, sea turtles, and other pelagic species that feed on zooplankton and small fish were likely attracted to this concentrated food source, and because drift gillnet fishery effort in that area and during that time period is normally high, the concurrence of fishing effort and foraging animals caused more entanglements than normal.

Humpback whales are rarely taken by the CA/OR drift gillnet fishery, and of the two whales observed taken in the past thirteen years, both have survived uninjured. Based on a worst-case scenario, NOAA Fisheries estimates that a maximum of 4 humpback whales ((1 humpback observed taken in 1999/526 sets observed in 1999) x 2,000 maximum expected sets per year) in a given year could be captured by the CA/OR drift gillnet fleet. Fishermen have reported anecdotally evidence of large whales punching through their nets; therefore, humpback whales likely interact with the CA/OR drift gillnet fishery. However, because of their size and strength, humpback whales likely punch through the net, and entanglement is a rare event. Entanglement, and any associated mortality, of humpback whales is not anticipated to occur every year. Based on past fishery performance, humpback whales were observed taken twice in thirteen years, or once during the six years the PCTRP has been in place. However, following the implementation of protective measures to avoid the likelihood of jeopardy to leatherback and loggerhead sea turtles in the CA/OR drift gillnet fishery, no humpback whales have been observed taken in the fishery.

c. *Sperm whale impacts*

Prior to the implementation of the PCTRP on October 30, 1997, the CA/OR drift gillnet fishery was observed to incidentally take seven sperm whales; of these whales, three were dead (43%), three were released alive and uninjured (43%), and one was released injured and was not expected to survive (14%). In 1992 the CA/OR drift gillnet fishery was observed taking 3 sperm whales in one set off central California; two were alive and released uninjured, and one was dead. The net was suspended 36 feet below the surface. In 1993, 2 sperm whales were entangled in one set off southern California; one was alive and released uninjured, and one was dead. The extender length of the net was 60 feet. Also in 1993, one sperm whale was observed entangled and died in a drift gillnet off central California, with a net that was using 36 feet extenders. In 1996, one sperm whale was observed entangled and released injured (trailing gear, and wounded from ramming the vessel) off central California. The net was configured with 33 pingers, and was suspended 36 feet below the surface. Since the implementation of the PCTRP, only one sperm whale was observed incidentally taken in 1998. This animal died in a net off central California which did not have the full complement of pingers.

There is speculation that sperm whales tend to feed at nighttime, and because they often forage by diving to great depths, possibly with an open jaw, they may be more vulnerable to a drift gillnet than perhaps other large whales. In addition, because sperm whales often prey on luminous squid, they may be attracted to light sticks occasionally used by drift gillnetters, which may explain why the CA/OR drift gillnet fishery has been observed taking over twice as many sperm whales (eight) as it has fin and humpback whales combined (three).

All of the sperm whales incidentally taken in the CA/OR drift gillnet fishery were caught between October and December, in waters with an average sea surface temperature of between 13 and 18EC. Sperm whales are found in peak abundance off California from the end of August to mid-November, during the same time period when effort in the fishery increases. All but two (caught in the same net) of the sperm whales were taken in a concentrated area 50-75 miles west of Monterey Bay, California. Three of the sperm whales caught in this area were entangled in the same set, and based on their estimated length (12, 14, and 20 feet), they were likely subadults from a breeding school, beginning their south-bound migration down the coast. In addition, most (6/8) of the sperm whales taken were caught during the 1992-93 El Niño, when a lack of upwelling and unusually high sea surface temperatures resulted in animals having to forage at depth for longer periods of time for food, making them increasingly vulnerable to a drift gillnet. Sperm whales appear to be vulnerable to becoming entangled in uncomplicated gear, and this may be due to their foraging behavior, curiosity, or something unexplainable. Heezen (1957) documented 14 instances where sperm whales were entangled in deep sea cables, some as deep as 3,000 meters, along the ocean floor.

Of the eight sperm whales observed taken by the CA/OR drift gillnet fishery, three were released alive and uninjured (37.5 percent), one was released injured (12.5 percent), and four were killed (50 percent). Therefore, approximately 63 percent of captured sperm whales could be killed accidentally or injured (based on the mortality and injury rate of sperm whales observed taken by the U.S. fleet from 1990-2003). Based on past fishery performance, sperm whales are not observed taken in every year; they were observed taken in four out of the last thirteen years. During the six years the PCTRP has been in place, a sperm whale was observed taken only once (in a non-PCTRP compliant set). In addition, measures in place to protect leatherback sea turtles are also expected to reduce the likelihood of interactions between sperm whale and this fishery. Therefore, NOAA Fisheries conservatively anticipates that a sperm whale entanglement could be observed once every three years, and given the mortality and injury rate, the entangled whale might die as a result of the interaction.

B. Sea Turtles

1. Exposure of Sea Turtles to HMS Fisheries

a. Attractants on Gear

Floats and FADs. Sea turtles may be attracted to the floats used on driftnet and longline gear. Turtles show a preference for objects floating horizontally and nearly submerged and are strongly attracted to brightly colored objects (Arenas and Hall, 1992). Lab experiments have shown that sea turtles prefer bright colors (i.e. red and yellow) over dull or darker colors (i.e. black, green or blue) (e.g. Fontaine, *et al*., 1985). In the Southeast, controlled experiments and qualitative evaluations were conducted using captive reared sea turtles to evaluate their responses to various components of pelagic longlining gear and other stimuli. One experiment tested the attraction of sea turtles to orange and white colored longline floats in a 80' x 35' pen enclosure. Sea turtles were introduced into the pen with a single float treatment. Preliminary analysis of the results indicate that the test turtles may have been more attracted to orange colored floats than to white colored floats (J. Watson, SEFSC, personal communication, July, 2001). Floats typically used during swordfish-style sets are bright orange, bullet-shaped, and slightly submerged. Deep sets generally use larger cylindrical inflatable or rigid spherical buoys and floats, and these also are typically orange in color (L. Enriquez, NOAA Fisheries, personal communication, January, 2001; e.g. www.lindgren-pitman.com/floats.htm).

Mainline and hardware. The Southeast Fishery Science Center also conducted evaluations at their Panama City Laboratory which involved placing longline gear in open water pens with captive reared loggerhead turtles to investigate turtle entanglement with various longline gear components. During these experiments, scientists observed turtles tracking along the mainline and biting at the hardware (snaps). Turtles that were placed in a pool without longline gear (i.e. control) tended to track along the outside edges of the pool. These observations and observations of longline gear at sea by divers and remotely operated vehicles indicate that the standard mainline used by the fishing industry is highly visible and that turtles may be attracted to the mainline and hardware and may follow the mainline (J. Watson, SEFSC, personal communication, August, 2001).

Lightsticks. Sea turtles foraging at night may be attracted to the lightsticks, confusing them for prey. Lightsticks are often used by longliners targeting swordfish in order to attract the swordfish to the bait. Whether lightsticks attract swordfish directly or whether they attract baitfish, which in turn attract the swordfish, is not entirely clear; however, fishermen report higher takes of swordfish when they use lightsticks. Lightsticks are generally attached to every other branchline, approximately a meter above the hook. Researchers studying the prey and foraging habits of sea turtles have reported the ingestion of pyrosomas, the so-called "fiery bodies," by leatherbacks, loggerheads, and olive ridleys; however, there is little information on the actual ingestion of lightsticks by sea turtles. In addition, statisticians have not been able to find any correlation between sea turtle take and the proximity of a lightstick to the hook or branchline that the turtle was hooked on or entangled in.

Bait. Sea turtles may also be attracted to the bait used on longline gear. Four olive ridleys necropsied after being taken dead by Hawaii-based longliners were found with bait in their stomachs (Work, 2000). In addition, a leatherback has been documented ingesting squid (the bait typically used on the now prohibited gear targeting swordfish). The authors speculate that the lightsticks used on this gear type may initially have attracted the turtle, by simulating natural prey (Skillman and Balazs, 1992).

b. Exposure by Species

The narratives that follow describe the co-occurrence between the fisheries and listed sea turtles. The information regarding sea turtle presence and behavior in the action area stems from observer reports and other scientific information available on the foraging, diving behavior, and natal origin of the sea turtles known to be affected by the fisheries. The information presented below is based on past observed interactions between the HMS fisheries and sea turtles as well as information from other fisheries such as the Hawaii-based longline fishery which also operates in the action area. A similar analysis for the marine mammals adversely affected by the HMS fisheries was not done due to the extremely rare and random nature of interactions between the fisheries and marine mammals foraging and migrating through the action area.

We assessed the exposure of sea turtles to the HMS fisheries using four variables:

Demographic patterns of exposure. Interaction between fishing gear and marine species can be described using a wide variety of demographic variables, but three variables are particularly important for section 7 assessments: the number of individuals, the age or gender of those individuals, and the populations to which those individuals belong. Fishing gear under the FMP can interact with individuals from all populations of threatened and endangered species or they can interact with particular populations. To the extent that information was available, we described demographic patterns using all three variables.

Behavioral patterns of exposure. Interaction between fishing gear and marine species may be influenced by the behaviors of the sea turtles in the action area. Fishing gear under the FMP can interact with individuals that are foraging within or migrating through the action area. The turtles may have a specific behavioral response to the gear (presence of attractants such as light sticks, floats, or bait for example) or the course of normal behaviors may bring the animal into contact with the fishery. There are spatial and temporal components to these interactions as well, therefore, behavioral and spatial or temporal components sometimes overlap in the discussions below.

Spatial patterns of exposure. Interactions between fishing gear and marine species in space can be described by three dimensions: degrees of latitude, degrees of longitude, and vertically within the water column. The first two dimensions describe the patterns as seen from the surface. The third dimension, however, describes the interaction from the eyes of the turtle. To the extent that information was available, we described these patterns using all three dimensions.

Temporal patterns of exposure. Interactions between fishing gear and marine species through time can be described by three dimensions: annual patterns, time of year, and time of day. These dimensions describe the patterns as they interact with changes in fishing effort, changes in oceanographic conditions, and changes in behavior of target species as well as listed species taken as bycatch. To the extent that information was available, we described these patterns using all three dimensions.

1. Green Sea Turtles

Demographic Patterns of Exposure. Based on genetic data collected in HMS and other U.S. fisheries, most green turtles in the action area will probably be members of the Hawaiian (French Frigate Shoals) or Mexican (Pacific coast) nesting aggregations. Genetic analysis of one green turtle taken in the HMS fisheries indicated eastern Pacific origin.

Life history information collected by observers in other fisheries operating in the action area suggests that juvenile, subadult and adult green turtles occur on the fishing grounds. From those turtles for which genetic data were collected, turtles originating from Hawaiian nesting aggregations were represented by smaller animals (juvenile and sub-adult sizes); turtles from Mexican nesting aggregations were represented by larger animals (sizes that suggest they were probably adult turtles). There have been no observed captures of green turtles in the west coast-based longline fishery. One green turtle has been observed taken in the CA/OR drift gillnet fishery. This animal was an immature turtle from one of the eastern Pacific nesting aggregations.

Behavioral Patterns of Exposure. Although most green turtles appear to have a nearly exclusive herbivorous diet, consisting primarily of sea grass and algae (Wetherall et al., 1993; Hirth, 1997), those along some areas of the east Pacific coast seem to have a more carnivorous diet. Analysis of stomach contents of green turtles found off Peru revealed a large percentage of molluscs and polychaetes, while fish and fish eggs, and jellyfish and commensal amphipods comprised a lesser percentage (Bjorndal, 1997). Exclusively herbivorous green turtles may be less likely to be attracted to bait set on fishing gear reducing their chances of hooking.

Based on the behavior of post-hatchlings and juvenile green turtles raised in captivity, wild green turtles in pelagic habitats probably live and feed at or near the ocean surface, and their routine dives probably do not exceed several meters in depth (NOAA Fisheries and USFWS, 1998a). The maximum recorded dive depth for an adult green turtle was 110 meters (Berkson, 1967, *in* Lutcavage and Lutz, 1997), while subadults routinely dive 20 meters for 9-23 minutes, with a maximum recorded dive of 66 minutes (Brill, *et al*., 1995, *in* Lutcavage and Lutz, 1997). These dive depths are within the range of fishing gear such as longlines and drift gillnets that are set and left to fish for long periods. Green turtles may interact with nets or lines as they descend or ascend through the water column. Turtles resting or foraging at or near the surface could be accidentally hooked by trolling gear or encircled by nets.

Spatial Patterns of Exposure. Green turtles appear to prefer waters with temperatures above 18 - 20E Celsius. Green turtles in these areas are likely foraging in shallow waters or at shallow depths, or transiting to foraging grounds. During warm spells (e.g., El Niño), green turtles may be found considerably north of their normal distribution.

The nonbreeding range of green turtles is generally tropical, and can extend thousands of miles from shore in certain regions. Hawaiian green turtles monitored through satellite transmitters were found to travel more than 1,100 km from their nesting beach in the French Frigate Shoals (Balazs, 1994; Balazs, et al., 1994; Balazs and Ellis, 1996). Tag returns of eastern Pacific green turtles establish that these turtles travel long distances between foraging and nesting grounds. Observers on a NOAA research ship documented green turtles 1,000-2,000 statute miles from shore (Eckert, 1993).

In the Hawaii-based longline fishery, green turtles have been caught in the area bounded by 155EW and approximately 180EE longitude and between 0EN and 30EN latitude. For the green turtle taken in the CA/OR drift gillnet fishery environmental conditions appear to have changed some of their expected spatial distribution. A sea surface temperature warming trend (2-3EC) from October to November, 1999 off the west coast of the United States may have attracted more warm water species, such as the green turtle. An associated upwelling event probably increased the primary production in southern California, attracting large whales and sea turtles who prefer to feed on zooplankton, and increasing their vulnerability to becoming entangled by the CA/OR drift gillnet fishery. Future interactions between this species and the fisheries operating in the Bight are therefore more likely to occur when oceanographic variation produces temperature and forage conditions conducive to green turtles.

Temporal Patterns of Exposure. Green turtles have been captured in all months of the year except January and September in the Hawaii-based longline fishery. The green turtle taken in the CA/OR drift gillnet fishery was captured during November, 1999. Oceanographic conditions, as described above, likely played a role in this interaction.

Levels of Exposure in the Proposed Fisheries. There are no reported or observed interactions between the HMS fisheries and green sea turtles with the exception of the CA/OR drift gillnet fishery. The turtles and the fisheries overlap somewhat within the action area, but the nature of the gear and methods for most of these fisheries makes interactions unlikely. For example, the harpoon fishery would be unlikely to incidentally capture a green sea turtle as the harpoon is thrown after the target species (swordfish) is identified. Rod and reel fisheries are also unlikely to capture green turtles as the gear and bait may be unattractive to the turtles. Some of these fisheries may operate in areas outside of the normal foraging grounds of the green turtles. There have been no observed interactions between green turtles and longline gear, either west coast-based or Hawaii-based, in the areas east of 150EW.

Most smaller tuna purse seine vessels fishing off southern California fish on tuna schools. Based on observer data from large purse seine vessels, the chances of incidentally capturing a sea turtle during a school set are much less than incidentally capturing a sea turtle during floating object sets; therefore, the incidental capture of sea turtles by the small vessel fleet is likely to be small, if any interactions occur at all. There are three small vessels currently operating in the ETP. Each vessel may make as many as three trips per year for HMS species. Given the low level of effort, it is unlikely that any captures occur. Based on information from the large purse seine vessels, the mortality rate of sea turtles captured within sets is very low. If any incidental captures do occur, they are unlikely to result in mortalities.

Only the CA/OR drift gillnet fishery has documented an incidental capture of a green sea turtle. As discussed above, this interaction and future interactions are likely to be a result of oceanographic conditions such as those that occurred during November 1999. Based on the expected future effort in this fishery, up to 4 green turtles may be captured when conditions similar to those that occurred in November 1999 are present. Since implementation of measures to protect leatherback and loggerhead sea turtles, no green turtles have been observed captured in this fishery.

2. Leatherback Sea Turtles

Demographic Patterns of Exposure. Based on genetic analysis of leatherback turtles captured in the Hawaii-based longline fishery and the HMS fisheries, leatherback turtles found in the action area are from nesting aggregations in the western Pacific region (Papua New Guinea, Indonesia, and Solomon Islands) or the eastern Pacific region (Mexico and Costa Rica). All three leatherback turtles sampled in the HMS fisheries were from the western Pacific nesting aggregations. Similarly, all samples taken from stranded leatherbacks on the California coast have indicated representation from western Pacific nesting beaches (Dutton, *et al*., in press, and P. Dutton, personal communication, March, 2000).

Straight carapace lengths taken from a subset of the leatherback turtles caught in these fisheries suggest that some of them were subadults, representing both early and late pelagic stage, based on the stage structure for Malaysian turtles presented in Bolten, *et al*. (1996). Only five of the turtles caught in the CA/OR drift gillnet fishery were measured, all between 132 to 160 cm (sub-adults and adults). Most of the leatherbacks caught in these fisheries were not measured. Those leatherbacks that were not measured may have been too large to be safely brought on board; therefore they may have been adults. It appears that young leatherback turtles (carapace length <100 cm) reside only in waters warmer than 26EC, which should generally place them outside of areas in which longline swordfish fleets operate (Eckert, 1999b; Eckert, 2002).

Behavioral Patterns of Exposure. Their long pectoral flippers and extremely active behavior make Leatherback sea turtles particularly vulnerable to fishing gear and ocean debris. Observed leatherback sea turtle entanglements have primarily involved the front flippers and/or the neck and head region. Leatherback hatchlings studied in captivity for almost 2 years swam persistently without ever recognizing the tank sides as a barrier (Deraniyagala, 1939, *in* Wyneken, 1997). A leatherback entangled in a net will most likely continue trying to swim, expending valuable amounts of energy and oxygen. As available oxygen diminishes, anaerobic glycolysis takes over, producing high levels of lactic acid in the blood. Unlike the shelled turtles, leatherbacks lack calcium, which helps to neutralize the lactic acid build-up by building up bicarbonate levels. In addition, leatherbacks store an enormous amount of oxygen in their tissues, similar to marine mammals, and have comparatively high hematocrits, which is efficient for such a deep-diving turtle but means that they have relatively less oxygen available for submergence. Maximum dive duration for the species is substantially less than half that of other turtles. The disadvantage of this is that they are not able to hold their breath as long and are probably more vulnerable to drowning in long gear sets.

Leatherback turtles within the action area of the Hawaii-based longline fishery are probably foraging (at the surface or at depth, including the deep scattering layer – strata comprised primarily of vertically migrating zooplankton, chiefly siphonophore and salp colonies, as well as medusae) or migrating between their nesting, mating, and foraging areas. Davenport (1988, *in* Davenport and Balazs, 1991) speculated that leatherback turtles may locate pyrosomas at night due to their bioluminescence; however direct evidence is lacking. If they are tracking these prey via bioluminescence, fishing gear that uses lighsticks or other similar devices may attract leatherback sea turtles. Additionally, if gillnet gear collects medusae or pyrosomas during their sets, leatherback sea turtles may become entangled while foraging on this "catch."

Leatherback sea turtles are able to dive quite deep, but appear to spend most of their time (up to 90%) diving to depths shallower than 80 meters. Leatherback turtles also appear to spend almost the entire portion of each dive traveling to and from maximum depth, suggesting that maximum exploitation of the water column is of paramount importance to the leatherback (Eckert, *et al*., 1989). Migrating leatherback turtles also spend a majority of time at sea submerged, and they display a pattern of continual diving (Standora, *et al*., 1984, *in* Southwood, *et al*., 1999). They are highly migratory, exploiting convergence zones and upwelling areas in the open ocean, along continental margins, and in archipelagic waters (Morreale, *et al*., 1994; Eckert, 1998; Eckert, 1999a).

Recent information on leatherback sea turtles tagged off the west coast of the United States has revealed an important migratory corridor from central California, to south of the Hawaiian islands, leading to western Pacific nesting beaches (P. Dutton, NOAA Fisheries, personal communication, December 2003). This corridor runs through the areas typically fished by HMS fleets and supports genetic findings that most of the leatherback turtles caught in the fishery originate from western Pacific beaches. Most of the western Pacific leatherback sea turtles followed the southwest migratory corridor, heading towards western Pacific nesting beaches. Two that have been tracked for an extended period of time did not arrive on the nesting beaches, instead heading north and east, back towards the northen part of Hawaii where shallow-set longline fleets operate. One leatherback did not follow a southwest track out of Monterey and instead headed southeast, along Baja California, Mexico, and into the Gulf of California. Eastern Pacific leatherback turtles appear to migrate primarily to the south, into the fishing grounds of South American fishing nations, supporting the low observed interaction rate between the Hawaii-based longline fishery, west-coast based longline fishery, CA/OR drift gillnet fishery, and eastern Pacific leatherback turtles.

Researchers have also begun to track female leatherbacks tagged on western Pacific nesting beaches, both from Jamursba-Medi, Papua, and from the Morobe coast of Papua New Guinea. Most of the females that have been tagged in Papua have been tracked heading on an easterly pathway, towards the western U.S. coast. One female headed north and is currently meandering in the East China Sea and the Sea of Japan, generally between Japan and South Korea. Another female headed north and then west of the Philippines. Meanwhile, all the leatherbacks tagged off Papua New Guinea have traveled on a southeasterly direction, in the south Pacific Ocean (P. Dutton, NOAA Fisheries, personal communication, December, 2003).

These observations further support findings that the HMS fisheries in the northern Pacific Ocean are more likely to interact with leatherback sea turtles from western Pacific nesting aggregations. Within the western Pacific aggregations, turtles from the Indonesian beaches may be more likely to interact with north Pacific fisheries.

Spatial Patterns of Exposure. In the Hawaii-based longline fishery, leatherback turtles were captured in the area bounded by 170EE and 133EW longitude and between 5EN and 41EN latitude. Leatherback turtles captured in the west coast-based longline fishery were taken in the area bounded by 140EE and 134EW longitude and between 35EN and 40EN latitude. All of the leatherbacks observed taken by the CA/OR drift gillnet fishery, except for one, were located north of Point Conception. The leatherbacks were found in waters with an average monthly sea surface temperature of between 10 to 17.5EC, and the majority of them were found in areas of coastal upwelling and some were found on distinct temperature breaks. The highest density of leatherback sightings on the U.S. West Coast is in and around Monterey Bay, with a peak in sightings in August (Starbird, *et al.*, 1993).

Temporal Patterns of Exposure. Leatherback sea turtles have been captured in every month of the year, except August in the Hawaii-based longline fishery. All of the leatherback sea turtles observed taken by the CA/OR drift gillnet fishery were taken from September to January, with approximately 60% of the captures occurring in October. Leatherback sea turtles have been taken in the west coast-based longline fishery in October and November.

Levels of Exposure in the Proposed Fisheries. There are no reported or observed interactions between the HMS fisheries and leatherback sea turtles with the exception of the CA/OR drift gillnet fishery and west coast-based longline fishery. The turtles and the fisheries overlap within the action area, but the nature of the gear and methods for most of these fisheries makes interactions unlikely. For example, the harpoon fishery would be unlikely to incidentally capture a leatherback sea turtle as the harpoon is thrown after the target species (swordfish) is identified. Rod and reel fisheries are also unlikely to capture leatherback turtles as the gear and bait may be unattractive to the turtles.

Most smaller tuna purse seine vessels fishing off southern California fish on tuna schools. Based on observer data from large purse seine vessels, the chances of incidentally capturing a sea turtle during a school set are much less than incidentally capturing a sea turtle during floating object sets; therefore, the incidental capture of sea turtles by the small vessel fleet is likely to be small, if any interactions occur at all. There are three small vessels currently operating in the ETP. Each vessel may make as many as three trips per year for HMS species. Given the low level of effort, it is unlikely that any captures occur. Based on information from the large purse seine vessels, the mortality rate of sea turtles captured within sets is very low. If any incidental captures do occur, they are unlikely to result in mortalities.

Of the HMS fisheries, only the CA/OR drift gillnet fishery and west coast-based longline fishery have documented incidental captures of leatherback sea turtles. Based on past interaction data, the CA/OR drift gillnet fishery may capture as many as three leatherback turtles per year. However, following the implementation of protective measures to avoid the likelihood of jeopardy to leatherback and loggerhead sea turtles in the CA/OR drift gillnet fishery, no leatherback sea turtles have been observed taken in the fishery. Based on past interactions with U.S. shallow-set longline fleets operating east of 150EW, 23 to 57 leatherback turtles may be captured per year. Most, if not all of these leatherback turtles are expected to be from western Pacific nesting aggregations, and given observed migration patterns of satellite tagged sea turtles, most are expected to be from beaches in Papua such as Jamursba-Medi.

3. Loggerhead Sea Turtles

Demographic Patterns of Exposure. Based on genetic analyses, all of the loggerhead turtles captured in the Hawaii-based longline fishery and west coast-based longline fishery originated from

Japanese nesting aggregations (Dutton *et al*., 1998, P. Dutton, NOAA Fisheries, personal communication, October, 2002). Five percent of these turtles were from beaches on Yakushima Island. These turtles are identified by a rare haplotype distinct from other turtle nesting beaches in Japan. Available data on the length of the captured turtles indicate that the fishery captures oceanic-stage juvenile loggerhead turtles. These data are supported by the available information on the foraging and migrating patterns of loggerhead turtles. The transition from hatchling to young juvenile occurs in the open sea, and evidence is accumulating that this part of the loggerhead life cycle may involve a trans-Pacific developmental migration (Bowen, *et al*., 1995).

Behavioral Patterns of Exposure. For their first years of life, loggerheads forage in open ocean pelagic habitats. Both juvenile and subadult loggerheads feed on pelagic crustaceans, mollusks, fish, and algae. Large aggregations of juveniles off Baja California have been observed foraging on dense concentrations of the pelagic red crab. Loggerheads in the north Pacific are opportunistic feeders that target items floating at or near the surface, including gastropods, medusae, and pyrosomas. If high densities of prey are present, they will actively forage at depth (Parker, *et al.,* in press). Based on their foraging pattern, loggerhead turtles may be attracted to bait or lightsticks on longline gear. If gillnet gear collects medusae or pyrosomas during their sets, loggerhead sea turtles may become entangled while foraging on this “catch.”

A recent study (Polovina *et al.*, 2004) found that tagged turtles spent 40 percent of their time at the surface and 90 percent of their time at depths shallower than 40 meters. On only five percent of recorded dive days loggerheads dove to depths greater than 100 meters at least once. In the areas that the loggerheads were diving, there was a shallow thermocline at 50 meters. There were also several strong surface temperature fronts the turtles were associated with, one of 20EC at 28EN and another of 17EC at 32EN. These patterns suggest that loggerhead turtles are more likely to interact with fishing gear that fishes at the surface or shallow depths. This is borne out by observed patterns in longline and drift gillnet fisheries. In addition, two loggerheads were incidentally caught in the albacore surface hook and line fishery which fishes at the surface or very shallow depths.

Loggerhead hatchlings on nesting beaches in Japan undertake developmental migrations in the North Pacific, using the Kuroshio and North Pacific Currents. Loggerheads tagged in Mexico and California with flipper and/or satellite transmitters have been monitored returning to Japanese waters (Resendiz, et al., 1998a-b). Loggerheads appear to utilize surface convergent forage habitat to capture their primary prey organisms which float along currents and congregate at fronts.

Based on oceanographic conditions, the loggerheads were associated with fronts, eddies, and geostrophic currents. The turtles moved with the seasonal movements of the Transition Zone Chlorophyll Front (TZCF), although they tended to remain south of the front itself, and were found along the southeastern edge of the Kuroshio Extension Current (KEC) and the northern edge of the Subtropical Gyre (Polovina *et al.* 2004). The TZCF and KEC appear to be important forage habitat for loggerhead turtles as these areas contain colder, plankton-rich waters. The tagging studies indicate that loggerheads may spend months at the edge of eddies in these areas. As this area has also been found to be an important foraging habitat for juvenile bluefin tuna (Ingake *et al.* 2001 *in* Polovina *et al.* 2004), overlaps between fisheries targeting these fish and others with similar habitat associations are likely to also encounter loggerhead sea turtles.

Spatial Patterns of Exposure. Of 70 trips in the Hawaii-based longline fishery in which loggerheads were captured, 39 had captured loggerheads in more than one set, and several trips had captured multiple loggerheads in the same set. Three of seven trips in the west coast-based longline fishery captured loggerheads in more than one set. Two of these sets had multiple captures. These observations as well as data from observations off Baja, California suggest that juvenile loggerhead turtles forage or migrate in groups. In the Atlantic Ocean, 68.1% of the loggerhead turtles captured in longline gear were caught in sets with other loggerheads compared with 31.9% that were caught singly (Hoey, 1998).

Temporal Patterns of Exposure. As discussed above, loggerhead sea turtles shifted seasonally north and south between 28EN and 40EN with temperature fronts. During January through June, satellite tagged loggerheads were found in the southern portion of this range, shifting to the northern end during July though December. The turtles also associated with areas with sea surface temperatures (SSTs) between 15E and 25E C. The loggerheads were found in cooler waters during winter and spring, warmer waters in summertime (Polovina *et al.* 2004).

There are reports of loggerhead turtle captures in the Hawaii-based longline fishery in all months except May and June; most captures occurred during the fall and winter months, however, especially in January and February. In the west coast-based longline fishery, loggerhead turtles have been captured primarily October through February. This is the main portion of the fishing effort. One trip towards the end of May 2002 captured 6 loggerhead turtles in 5 sets. Since 1990, all of the loggerheads incidentally taken in the CA/OR drift gillnet fishery were located in a concentrated area south of San Clemente Island, and the majority of them were caught in the summertime, when sea surface temperatures are highest. All of the loggerheads were caught during El Niño years (1992-93, and 1997-98), when unusually warm sea surface temperatures and northward flowing equatorial currents bring hundreds of thousands of pelagic red crabs from Baja California north up the coast of California. Loggerheads taken by the fishery had most likely migrated north from waters off Baja California, Mexico, following their primary food source. No loggerheads were observed taken by the CA/OR drift gillnet fishery in non-El Niño years.

Levels of Exposure in the Proposed Fisheries. The only reported or observed interactions between the HMS fisheries and loggerhead sea turtles occurred in the CA/OR drift gillnet fishery, west coast-based longline fishery, and rarely, the albacore surface hook and line fishery. The turtles and the HMS fisheries overlap within the action area, but the nature of the gear and methods for the rod and reel, purse seine, and harpoon fisheries makes interactions unlikely. For example, the harpoon fishery would be unlikely to incidentally capture a loggerhead sea turtle as the harpoon is thrown after the target species (swordfish) is identified.

Most smaller tuna purse seine vessels fishing off southern California fish on tuna schools. Based on observer data from large purse seine vessels, the chances of incidentally capturing a sea turtle during a school set are much less than incidentally capturing a sea turtle during floating object sets; therefore, the incidental capture of sea turtles by the small vessel fleet is likely to be small, if any interactions occur at all. There are three small vessels currently operating in the ETP. Each vessel may make as many as three trips per year for HMS species. Given the low level of effort, it is unlikely that any captures occur. Based on information from the large purse seine vessels, the mortality rate of sea turtles captured within sets is very low. If any incidental captures do occur, they are unlikely to result in mortalities.

All of the loggerhead turtles captured in HMS fisheries are expected to be from Japanese nesting aggregations. Approximately five percent may be from the nesting beaches on Yakushima Island. Based on past interaction data, the CA/OR drift gillnet fishery may capture as many as five loggerhead turtles per year when an El Nino event is occurring. However, following the implementation of protective measures to avoid the likelihood of jeopardy to leatherback and loggerhead sea turtles in the CA/OR drift gillnet fishery, one loggerhead sea turtle has been observed taken in the fishery. Based on past interactions with U.S. shallow-set longline fleets operating east of 150EW, 126 to 195 loggerhead turtles may be captured per year. Additionally, two loggerheads have been observed captured in the albacore surface hook and line fishery. Based on the rarity of these interactions, it is unlikely that this fishery would capture more than one loggerhead turtle in any year. Based on the method of fishing in the albacore surface hook and line fishery, NOAA Fisheries expects that interactions will be incidental hooking or entanglement of the carapace or flippers as the gear travels through the water.

4. Olive Ridley Sea Turtles

Demographic Patterns of Exposure. Genetic analyses indicate that the Hawaii-based longline fishery catches olive ridley turtles from nesting aggregations in the eastern and western Pacific Ocean and the Indian Ocean (26 % were from the Indian Ocean or western Pacific Ocean and 74% were from the eastern Pacific). Length information collected by observers indicates that the fishery interacts with sub-adult and adult olive ridley turtles. Three olive ridley turtles taken in the HMS fisheries were determined to be of eastern Pacific origin (P. Dutton, NOAA Fisheries, personal communication, August 2003).

Behavioral Patterns of Exposure. The most common prey of olive ridleys are salps and pyrosomes, similar to leatherback turtles. These prey organisms occur sub-surface and migrate within the water column as part of the deep scattering layer. As a result, olive ridleys tend to dive deeper than species like loggerhead sea turtles, spending 20 percent of the time at the surface and 40 percent of their time at depths greater than 40 meters (Polovina *et al*. 2004). On 25 percent of the recorded dive days, olive ridleys dove to depths greater than 150 meters at least once. The dive habitat of the tagged olive ridleys had a deep thermocline at 100 meters and minimal horizontal surface temperature fronts. These dive patterns indicate that olive ridleys are diving and foraging in areas where longline, purse seine, and gillnet gear from HMS fisheries is active. If gillnet gear collects medusae or pyrosomas during their sets, olive ridley sea turtles may become entangled while foraging on this "catch."

Spatial Patterns of Exposure. In the eastern Pacific Ocean, adult olive ridleys are found in warm, tropical waters, bounded on the north by the California Current and on the south by the Humboldt Current. It has been hypothesized that depending on food sources, the distribution of juveniles may be similar to that of adults. Young olive ridleys may move offshore and occupy areas of surface current convergences to find food and shelter among aggregated floating objects until they are large enough to recruit to benthic feeding grounds of the adults.

Satellite monitoring of post nesting movements showed migration routes traversing thousands of kilometers over deep (>1000 m) oceanic water, distributed over a very broad range, suggesting that olive ridleys are nomadic and exploit multiple feeding areas, rather than migrate to one specific foraging area (*in* NOAA Fisheries and USFWS, 1998d).

Olive ridley turtles from both eastern and western Pacific nesting beaches were tagged in the HI longline fishery. These turtles occurred in habitat south of the loggerheads that were tagged in the fishery, in areas between 8 and 31EN, with SSTs of 23E to 28EC (primarily in areas with SSTs of 24E or 27EC). Olive ridleys from the east and west Pacific had different habitat associations. Western Pacific olive ridleys associated with major ocean currents, such as the southern edge of the KEC, the North Equatorial Current (NEC) and the Equatorial Countercurrent (ECC). Olive ridley turtles from the eastern Pacific were not associated with strong current systems, most of these turtles remained within the center of the Subtropical Gyre. These waters are warm, vertically stratified with deep thermoclines, and do not have strong surface temperature or chlorophyll gradients. Olive ridleys of either nesting aggregation origin were not associated with strong surface chlorophyll fronts. However, olive ridleys from the western Pacific were found in habitat characterized by wind-induced upwelling and shoaling of the thermocline, which may allow olive ridleys to forage more shallowly in these areas.

The Hawaii-based longline fishery has interacted with olive ridleys throughout the fishing area, with captures reported from as far north as 33EN to as far south as 7EN latitude, and from longitudes 143EW, west to 175EW. The olive ridley captured in the west coast-based longline fishery was taken near 29EN and 142EW.

In the CA/OR drift gillnet fishery, environmental conditions appear to have changed some of the expected spatial distribution of olive ridley sea turtles. A sea surface temperature warming trend (2-3EC) from October to November, 1999 off the west coast of the United States may have attracted more warm water species, such as the olive ridley turtle. An associated upwelling event probably increased the primary production in southern California, attracting large whales and sea turtles who prefer to feed on zooplankton, and increasing their vulnerability to becoming entangled by the CA/OR drift gillnet fishery, which target swordfish in the same area during that time period. Future interactions between this species and the fisheries operating in the Bight are therefore more likely to occur when oceanographic variation produces temperature and forage conditions conducive to olive ridley turtles.

Temporal Patterns of Exposure. Olive ridley turtles tagged in the Hawaii-based longline fishery were seen in the southern portion of their observed range (between 8 and 31EN) between October and December. Between April and September, the turtles were found between 14E and 28EN , but not in the area between 20E and 24E N. This middle area is where the turtles were most frequently found during January through March (Polovina *et al.* 2004). The data was not separated by nesting beach origin, however, so some of these patterns may also be attributable to the different habitat associations between eastern and western Pacific olive ridleys.

Olive ridley turtles have been captured in every month of the year, except February, in the Hawaii-based longline fishery with most of the captures occurring during warmer months (May to August). An olive ridley sea turtle was captured in the west coast-based longline fishery in May. As discussed above, an olive ridley sea turtle was captured in the CA/OR drift gillnet fishery during an El Nino event in November, 1999.

Levels of Exposure in the Proposed Fisheries. Although the olive ridley is widely regarded as the most abundant sea turtle in the world, they are very rarely caught in the CA/OR drift gillnet fishery, probably because the olive ridley prefers tropical and warm temperate waters. Of all sea turtle strandings in California from 1990-99, the olive ridley was the sea turtle most rarely found (J. Cordaro, NOAA Fisheries, personal communication, May, 2000).

There are no reported or observed interactions between the HMS fisheries and olive ridley sea turtles with the exception of the CA/OR drift gillnet fishery and west coast-based longline fishery. The turtles and the fisheries overlap within the action area, but the nature of the gear and methods for most of these fisheries makes interactions unlikely. For example, the harpoon fishery would be unlikely to incidentally capture a olive ridley sea turtle as the harpoon is thrown after the target species (swordfish) is identified. Rod and reel fisheries are also unlikely to capture olive ridley turtles as the gear and bait may be unattractive to the turtles.

Most smaller tuna purse seine vessels fishing off southern California fish on tuna schools. Based on observer data from large purse seine vessels, the chances of incidentally capturing a sea turtle during a school set are much less than incidentally capturing a sea turtle during floating object sets; therefore, the incidental capture of sea turtles by the small vessel fleet is likely to be small, if any interactions occur at all. There are three small vessels currently operating in the ETP. Each vessel may make as many as three trips per year for HMS species. Given the low level of effort, it is unlikely that any captures occur. Based on information from the large purse seine vessels, the mortality rate of sea turtles captured within sets is very low. If any incidental captures do occur, they are unlikely to result in mortalities.

Based on past interaction data, the CA/OR drift gillnet fishery may capture as many as four olive ridley turtles per year when conditions similar to those that occurred in November 1999 are present. Since the implementation of protective measures to avoid the likelihood of jeopardy to leatherback and loggerhead sea turtles in the CA/OR drift gillnet fishery, no olive ridley sea turtles have been observed taken in the fishery. Based on past interactions with U.S. shallow-set longline fleets operating east of 150EW, 1 to 11 olive ridley turtles may be captured per year. Most, if not all of these olive ridley turtles are expected to be from the eastern Pacific nesting aggregations. Based on genetic analyses from HMS fisheries, all of these turtles would be from the eastern Pacific. Using information from the Hawaii-based longline fishery, 74%, or 8 of the olive ridley turtles taken in the west coast-based longline fishery may be from the eastern Pacific.

2. Response of Sea Turtles to HMS Fisheries

As discussed in the Assessment Approach, once we have identified which listed resources are likely to be exposed to the proposed fisheries, we conduct response analyses to identify how listed resources are likely to respond once exposed to the fisheries.

The most significant hazard HMS fisheries present to sea turtles results from potential entanglement in or hooking by gear used in the fisheries which can injure or kill turtles. Turtles that are entangled in or hooked by gear can drown after being prevented from surfacing for air; alternatively, turtles that are hooked, but do not die from their wounds, can suffer impaired swimming or foraging abilities, altered migratory behavior, and altered breeding or reproductive patterns. Although survivability studies have been conducted on sea turtles captured in longline fisheries, such long-term effects are nearly impossible to monitor; therefore a quantitative measure of the effect of longlining on sea turtle populations is very difficult. Even if turtles are not injured or killed after being entangled or hooked, these interactions can be expected to elicit stress-responses in the turtles that can have longer-term physiological or behavioral effects. The following discussion summarizes the information on how sea turtles are likely to respond to these interactions with fishing gear.

a. Longline Gear

Entanglement in Longline Gear

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

Sea turtles have been found entangled in branchlines (gangions), mainlines and float lines. Longline gear is fluid and can move according to oceanographic conditions determined by wind and waves, surface and subsurface currents, etc.; therefore, depending on both sea turtle behavior, environmental conditions, and location of the set, turtles could be entangled in longline gear. Entanglement in monofilament line (mainline or gangion) or polypropylene (float line) could result in substantial wounds, including cuts, constriction, or bleeding on any body part. In addition entanglement could directly or indirectly interfere with mobility, causing impairment in feeding, breeding, or migration. Sea turtles entangled by longline gear are most often entangled around their neck and foreflippers, and, often in the case of leatherback entanglements, turtles have been found snarled in the mainline, floatline, and the branchline (e.g. Hoey, 2000).

Hooking (Longline Gear)

In addition to being entangled in a longline, sea turtles are also injured and killed by being hooked. Hooking can occur as a result of a variety of scenarios, some of which will depend on foraging strategies and diving and swimming behavior of the various species of sea turtles. For example, necropsied olive ridleys have been found with bait in their stomachs after being hooked; therefore, they most likely were attracted to the bait and attacked the hook. In addition, leatherbacks, loggerheads and olive ridleys have all been found foraging on pyrosomas which are illuminated at night. If lightsticks are used on a shallow set at night to attract the target species, the turtles could mistake the lightsticks for their preferred prey and get hooked externally or internally by a nearby hook. Similarly, a turtle could concurrently be foraging in or migrating through an area where the longline is set and could be hooked at any time during the setting, hauling, or soaking process.

Sea turtles are either hooked externally - generally in the flippers, head, beak, or mouth - or internally, where the animal has attempted to forage on the bait, and the hook is ingested into the gastro-intestinal tract, often a major site of hooking (E. Jacobson, *in* Balazs, *et al*., 1995). Even if the hook is removed, which is often possible with a lightly hooked (i.e. externally hooked) turtle, the hooking interaction is believed to be a significant event. Like most vertebrates, the digestive tract of the sea turtle begins in the mouth, through the esophagus, and then dilates into the stomach. The esophagus is lined by strong conical papillae, which are directed caudally towards the stomach (White, 1994). The existence of these papillae, coupled with the fact that the esophagus snakes into an s-shaped bend further towards the tail make it difficult to see hooks, especially when deeply ingested. Not surprisingly, and for those same reasons, a deeply ingested hook is also very difficult to remove from a turtle's mouth without significant injury to the animal. The esophagus is attached fairly firmly to underlying tissue; therefore, when a hook is ingested, the process of movement, either by the turtle's attempt to get free of the hook or by being hauled in by the vessel, can traumatize the internal organs of the turtle, either by piercing the esophagus, stomach, or other organs, or by pulling the organs from their connective tissue. Once the hook is set and pierces an organ, infection may ensue, which may result in the death of the animal.

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

Trailing Gear

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

Trailing line can also become snagged on a floating or fixed object, further entangling sea turtles or the drag from the float can cause the line to constrict around a turtle's appendages until the line cuts through the appendage. With the loss of a flipper a turtle's mobility is reduced, as is its ability to feed, evade predators, and reproduce. Observers on longliners that have captured (hooked) a turtle are directed to clip the line as close to the hook as possible in order to minimize the amount of trailing gear. This is difficult with larger turtles, such as the leatherback, which often cannot practicably be brought on board the vessel, or in inclement weather, when such action might place the observer or the vessel and its crew at risk. Turtles captured by vessels without observers may not have the line cut as close to the hook as possible because this is not required under the proposed action.

Post-hooking Survival

Research has been conducted in both the Atlantic and the Pacific to estimate post-hooking survival and behavior of sea turtles captured by longline. In the Pacific, from 1997 to late 2000, a total of 54 pelagic turtles hooked by the Hawaii-based longline fishery have had satellite transmitters attached to them in order to track their location and distance traveled following the interaction. Of these 54 turtles, 15 produced no transmissions, or their transmissions lasted less than a month - 11 had deeply ingested hooks (turtles had swallowed the hook, and it was not removed) and 4 were lightly hooked.(turtles had the hook lodged externally (beak or flipper), permitting easy removal) (D. Parker and G. Balazs, NOAA Fisheries, personal communication, April, 2002). No assumptions were made by the researchers regarding the fate of these turtles that failed to transmit or only transmitted for a short period of time. Assuming that the satellite transmitter was working correctly, there are a number of possible explanations for few or no transmissions, any of which could be correct. Following the hooking incident, including the forced submergence, hauling of the longline and subsequent capture by the vessel, the released turtle may not have had time to recover from its experience. As discussed above, turtles that expend energy as a result of increased activity, need time at the surface to process lactic acid loads. Sea turtles often appear to be moving fairly well and then just collapse, while they rebuild their energy stores or repay their oxygen debt (E. Jacobsen, *in* Balazs, *et al*., 1995). If a turtle does not have enough energy to remain afloat, it could submerge and die. In addition, injuries sustained as a result of the hooking incident, especially in incidents where the hook may have perforated an organ, may also result in death to the turtle. In both instances, the turtle sinks with the transmitter, and no signal is received. Whether or not these turtles remained submerged and therefore died, or the transmitters failed to transmit is a matter of speculation.

For the 34 turtles that did produce successful tracks for periods lasting more than a month, there were no significant differences ($P>0.05$) found for the duration of tracking (days) and the distance traveled between lightly hooked turtles ($n=15$) and turtles with deeply ingested hooks ($n=19$). Even when the 15 turtles that did not produce successful tracks were taken into account, no significant differences were found in terms of distance traveled and duration between the two groups (19 total lightly hooked, and 30 total deeply ingested). Furthermore, when species were analyzed individually for the two categories, no significant differences were found.

Polovina (NOAA Fisheries, personal communication, September, 2000) used a contingency table approach to analyze the transmission duration in intervals of 1 month for 34 loggerheads (including those w/ few or no transmissions), comparing lightly hooked versus deeply hooked turtles. While 43% of the deeply hooked turtles transmitted less than one month compared to 27% of the lightly hooked turtles, the chi-squared test found no significant difference between the transmission distributions for these two categories. When the data for all hard shell turtles are combined ($n=48$), 22% ($n=4$) lightly hooked and 37% ($n=11$) deeply hooked turtles transmitted less than one month. Again, the difference was not statistically significant between hooking categories based on a chi-square test.

Data were also analyzed to determine whether the length of the turtle (in straight carapace length) played any role in determining differences between deeply hooked turtles and those that were lightly hooked. Only all satellite tagged loggerheads (both with successful tracks and without ($n=35$)) showed a significant difference ($P=0.02$) in size between deeply ingested (mean size = 62.0 ± 10.9 cm) and lightly hooked (mean size = 53.0 ± 6.6 cm) (D. Parker and G. Balazs, NOAA Fisheries, personal communication, November, 2000).

In the eastern Atlantic, in the waters around the Azores, three juvenile loggerheads that had been lightly hooked by swordfish longline gear were instrumented with satellite-linked time-depth recorders in 1998. The number of dives performed by these hooked turtles was compared to five juvenile loggerheads that had been captured by dipnet and also instrumented. Turtles caught on longline fishing gear had significantly lower dive counts than turtle caught with dipnets during the normal (observed) period of most intense diving activity (from 9:00 am to 3:00 pm) (Bjorndal, *et al*., 1999). During a similar study in the summer of 2000, in the same area of the Atlantic, 10 pelagic juvenile loggerheads were instrumented - four were captured with dipnets (control), and six had been deeply hooked. In all periods of the 24-hour day (separated by 6-hour increments), the hooked turtles appeared to make longer and shallower dives than control turtles, but overall, dive behavior appeared similar between hooked and non-hooked turtles, having a diurnal component (shallowest dives occurring during 21:00 and 03:00) and a seasonal component (dive depth generally increased for most turtles from summer into fall) (Riewald, *et al*., 2000). Caution was given in interpreting both sets of data, as the studies were ongoing at the time of writing.

Forcible Submergence

Sea turtles can be forcibly submerged by drift gillnet or longline gear, or in the FADs deployed by purse seiners. Forcible submergence may occur through a hooking or entanglement event, where the turtle is unable to reach the surface to breathe. This can occur at any time during the set, including the setting and hauling of the gear, and generally occurs when the sea turtle encounters a net or line that is too deep below the surface, or or is too heavy to be brought up to the surface by a swimming sea turtle. For example, a sea turtle that is hooked on a 3 meter branchline attached to a mainline set at depth by a 6 meter floatline will generally not be able to swim to the surface unless it has the strength to drag the mainline approximately 3 more meters (discussed further below).

When interacting with longline gear, hooked sea turtles will sometimes drag the clip, attached to the branch line, along the main line. If this happens, the potential exists for a turtle to become entangled in an adjacent branch line which may have another species hooked such as a shark, swordfish, or tuna. According to observer reports, most of the sharks and some of the larger tuna such as bigeye are still alive when they are retrieved aboard the vessel, whereas most of the swordfish are dead. If a turtle were to drag the branch line up against a branch line with a live shark or bigeye tuna attached, the likelihood of the turtle becoming entangled in the branch line is greater. If the turtle becomes entangled in the gear, then the turtle may be prevented from reaching the surface. The potential also exists, that if a turtle drags the dropper line next to a float line, the turtle may wrap itself around the float line and become entangled.

During drift gillnet operations, a sea turtle may encounter a net when swimming or foraging at depth. The net is suspended below the sea surface by the ball buoys to a depth equal to the length of the buoylines. This depth has historically ranged from 18 ft to as much as 90 ft, but is currently limited by regulations enacted under the Marine Mammal Protection Act (MMPA) to a minimum depth of 36 feet below the sea surface. Since the CA/OR drift gillnet fishery is required to fish at this minimum depth, a sea turtle would have to either be swimming at or below this depth to encounter the net. On the other hand, a sea turtle could be entangled when swimming shallower than 36 feet during the hauling or setting of the gear.

Sea turtles have been observed entangled in FADs. Such entanglement has lead to mortality, likely through drowning. Sea turtles can become entangled in any part of the FAD, including the webbing or lines associated with it.

Response of Sea Turtles to Being Submerged

Sea turtles forcibly submerged for extended periods of time show marked, even severe, metabolic acidosis as a result of high blood lactate levels. With such increased lactate levels, lactate recovery times are long (even as much as 20 hours), indicating that turtles are probably more susceptible to lethal metabolic acidosis if they experience multiple captures in a short period of time, because they would not have had time to process lactic acid loads (*in* Lutcavage and Lutz, 1997). Kemp's ridley turtles that were stressed from capture in an experimental trawl (#7.3 minute forcible submergence) experienced significant blood acidosis, which originated primarily from non-repiratory (metabolic) sources. Visual observations indicated that the average breathing frequency increased from approximately 1-2 breaths/minute pre-trawl to 11 breaths/minute post-trawl (a 9 to 10-fold increase). Given the magnitude of the observed imbalance, complete recovery of acid-base homeostasis may have required 7 to 9 hours (Stabenau *et al*., 1991). Similar results were reported for Kemp's ridleys captured in entanglement nets - turtles showed significant physiological disturbance, and post-capture recovery depended greatly on holding protocol (Hoopes *et al*., 2000).

Presumably, however, a sea turtle recovering from a forced submergence would most likely remain resting on the surface (given that it had the energy stores to do so), which would reduce the likelihood of being recaptured by a submerged longline. Recapture would also depend on the condition of the turtle and the intensity of fishing pressure in the area. NOAA Fisheries has no information on the likelihood of recapture of sea turtles by HMS fisheries. However, in the Atlantic Ocean, turtles have been reported as captured more than once by longliners (on subsequent days), as observers reported clean hooks already in the jaw of captured turtles. Such multiple captures were thought to be most likely on three or four trips that had the highest number of interactions (Hoey, 1998).

Stabenau and Vietti (2003) studied the physiological effects of multiple forced submergences in loggerhead turtles. The initial submergence produced severe and pronounced metabolic and respiratory acidosis in all turtles. As the number of submergences increased, the acid-base imbalances was substantially reduced; although successive submergences produced significant changes in blood pH, $P\text{CO}_2$, and lactate. Increasing the time interval between successive submergences resulted in greater recovery of blood homeostatis. The authors conclude that as long as sea turtles have an adequate rest interval at the surface between submergences, their survival potential should not change with repetitive submergences.

Respiratory and metabolic stress due to forcible submergence is also correlated with additional factors such as size and activity of the sea turtle (including dive limits), water temperature, and biological and behavioral differences between species and will therefore also affect the survivability. For example, larger sea turtles are capable of longer voluntary dives than small turtles, so juveniles may be more vulnerable to the stress of forced submergence than adults. Gregory *et al*. (1996) found that corticosterone concentrations of small loggerheads captured were higher than those of large loggerheads captured during the same season. During the warmer months, routine metabolic rates are higher, so the impacts of the stress due to entanglement or hooking may be magnified (e.g. Gregory *et al*., 1996). In addition, disease factors and hormonal status may also play a role in anoxic survival during forced submergence. Any disease that causes a reduction in the blood oxygen transport capacity could severely reduce a sea turtle's endurance on a longline, and since thyroid hormones appear to have a role in setting metabolic rate, they may also play a role in increasing or reducing the survival rate of an entangled sea turtle (*in* Lutz and Lutcavage, 1997). Turtles necropsied following capture (and subsequent death) by longliners in this fishery were found to have pathologic lesions. Two of the seven turtles (both leatherbacks) had lesions severe enough to cause probable organ dysfunction, although whether or not the lesions predisposed these turtles to being hooked could not be determined (Work, 2000). As discussed further in the leatherback and loggerhead subsections below, some sea turtle species are better equipped to deal with forced submergence.

Sea turtles also exhibit dynamic endocrine responses to stress. In male vertebrates, androgen and glucocorticoid hormones (corticosterone (CORT) in reptiles) can mediate physiological and behavioral responses to various stimuli that influence both the success and costs of reproduction. Typically, the glucocorticoid hormones increase in response to a stressor in the environment, including interaction with fishing gear. "During reproduction, elevated circulating CORT levels in response to a stressor can inhibit synthesis of testosterone or other hormones mediating reproduction, thus leading to a disruption in the physiology or behavior underlying male reproductive success" (Jessop *et al*., 2002). A study in Australia examined whether adult male green turtles decreased either CORT or androgen responsiveness to a capture/restraint stressor to maintain reproduction. Researchers found that migrant breeders, which typically had overall poor body condition because they were relying on stored energy to maintain reproduction, had decreased adrenocortical activity in response to a capture/restraint stressor. Smaller males in poor condition exhibited a pronounced and classic endocrine stress response compared to the larger males with good body condition. The authors state: "We speculate that the stress-induced decrease in plasma androgen may function to reduce the temporary expression of reproductive behaviors until the stressor has abated. Decreased androgen levels, particularly during stress, are known to reduce the expression of reproductive behavior in other vertebrates, including reptiles." Small males with poor body condition that are exposed to stressors during reproduction and experience shifting hormonal levels may abandon their breeding behavior (Jessop *et al.,* 2002).

Female green turtles have also been studied to evaluate their stress response to capture/restraint. Studies showed that female green turtles during the breeding season exhibited a limited adrenocortical stress response when exposed to ecological stressors and when captured and restrained. Researchers speculate that the apparent adrenocortical modulation could function as a hormonal tactic to maximize maternal investment in reproductive behavior such as breeding and nesting (*in* Jessop, *et al.,* 2002).

Although a low percentage of turtles that are captured by longliners actually are reported dead, sea turtles can drown from being forcibly submerged. Such drowning may be either "wet" or "dry." In the case of dry drowning, a reflex spasm seals the lungs from both air and water. With wet drowning, water enters the lungs, causing damage to the organs and/or causing asphyxiation, leading to death. Before death due to drowning occurs, sea turtles may become comatose or unconscious. Studies have shown that sea turtles that are allowed time to stabilize after being forcibly submerged have a higher survival rate. This of course depends on the physiological condition of the turtle (e.g. overall health, age, size), time of last breath, time of submergence, environmental conditions (e.g. sea surface temperature, wave action, etc.), and the nature of any sustained injuries at the time of submergence (NRC, 1990).

Survival of Sea Turtles that Interact With Longline Gear

In February 2001, NOAA-Fisheries established a policy and criteria for estimating sea turtle survival and mortalities following interactions with longline fishing gear (NOAA Fisheries 2001b; see Table V-1). These criteria were based on the information that was available on the survival of leatherback sea turtles after they were captured and released from longline gear and were expected to be refined or revised once more information became available.

Table V-1. Sea turtle mortality rates based on level and type of interaction with longline fishing gear.
Source: NOAA Fisheries, 2001b

Interaction	Response	Injury	Mortality Rate
Entangled / no hook	Disentangled	No injury	0%
Entangled / external hook	Disentangled, no gear	Minor	27%
	Disentangled, trailing gear	Moderate	27%
	Dehooked, no gear	Minor	27%
Hooked in beak or mouth	Hook left, no gear	Moderate	27%
	Hook left, trailing gear	Serious	42%
	Dehooked, no gear	Moderate	27%
Hook swallowed	Hook left, no gear	Serious	42%
	Hook left, trailing gear	Serious	42%
Turtle Retrieved Dead	- - -	Lethal	100%

In 2003, NOAA-Fisheries' Office of Protected Resources was charged with conducting a review of NOAA-Fisheries' February 2001 post-hooking mortality criteria and recommending if and how the earlier criteria should be modified. As part of that review, the Office of Protected Resources convened a Workshop on Marine Turtle Longline Post-Interaction Mortality on 15-16 January 2004, during which seventeen experts in the areas of biology, anatomy/physiology, veterinary medicine, satellite telemetry and longline gear deployment presented and discussed the more recent data available on the survival and mortality of sea turtles subsequent to being hooked by fishing gear. Based on the information presented and discussed at the workshop and a comprehensive review of all of the information available on the issue, the Office of Protected Resources proposed the following changes to the earlier criteria (see Table V-2 for the criteria):

Categories. The February 2001 injury categories were expanded to better describe the specific nature of the interaction. The February 2001 criteria described two categories for mouth hooking: (1) hook does not penetrate internal mouth structure; and (2) mouth hooked (penetrates) or ingested hook. The new criteria divides the mouth hooking event into three components to reflect the severity of the injury and to account for the probable improvement in survivorship resulting from removal of gear, where appropriate, for each injury. The three components consist of: (1) hooked in esophagus at or below the heart (insertion point of the hook is not visible when viewed through the open mouth; (2) hooked in cervical esophagus, glottis, jaw joint, soft palate, or adnexa[18] (insertion point of the hook is visible when viewed through the open mouth); and (3) hooked in lower jaw (not adnexa). The new criteria, also, separates external hooking from mouth hooking, eliminates the ‘no injury’ category, and adds a new category for comatose/resuscitated.

Probable Improvement in Survivorship When Gear is Removed: The new criteria recognize that in most cases removal of some or all of the gear (except deeply-ingested hooks) is likely to improve the probability of survival. The categories for gear removal are: released with hook and with line that is greater than or equal to half the length of the carapace; released with hook and with line that is less than or equal to half the length of the carapace; and released with all gear removed. Turtles that have all or most of the gear removed are expected to have, on average, a higher probability of survival.

Species Difference: Species differences between hard-shelled turtles and leatherback turtles appears to play a role in post-interaction survival. The new criteria takes these differences into consideration and assign slightly higher rates of post-interaction mortality for leatherback turtles.

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[18] Subordinate part such as tongue, extraembryonic membranes

Table V-2. Criteria for assessing marine turtle post-interaction mortality after release from longline gear. Percentages are shown for hardshelled turtles, followed by percentages for leatherbacks (in parentheses).

Nature of Interaction	**Released with hook and with line greater than or equal to half the length of the carapace**	**Release with hook and with line less than half the length of the carapace**	**Released with all gear removed**
	Hardshell (Leatherback)	Hardshell (Leatherback)	Hardshell (Leatherback)
Hooked externally with or without entanglement	20 (30)	10 (15)	5 (10)
Hooked in lower jaw (not adnexa [19]) with or without entanglement	30 (40)	20 (30)	10 (15)
Hooked in cervical esophagus, glottis, jaw joint, soft palate, or adnexa (and the insertion point of the hook is visible when viewed through the mouth) with or without entanglement	45 (55)	35 (45)	25 (35)
Hooked in esophagus at or below level of the heart (includes all hooks where the insertion point of the hook is not visible when viewed through the mouth) with or without entanglement	60 (70)	50 (60)	n/a[20]
Entangled Only	Released Entangled 50 (60)		Fully Disentangled 1 (2)
Comatose/resuscitated	n/a[21]	70 (80)	60 (70)

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[19] Subordinate part such as tongue, extraembryonic membranes

[20] Per veterinary recommendation hooks would not be removed if the insertion point of the hook is not visible when viewed through the open mouth.

[21] Assumes that a resuscitated turtle will always have the line cut to a length less than half the length of the carapace, even if the hook cannot be removed.

Calculation of anticipated captures and mortalities of sea turtles in the west coast-based longline fishery

Estimation of captures and mortalities was crucial to completing our assessment of the risks the HMS fisheries pose to the listed sea turtles. Interaction rates for shallow longline sets occurring east of the 150E W longitude were calculated by NOAA Fisheries Southwest Fisheries Science Center (J. Carretta, NOAA Fisheries-SWFSC, personal communications, January and February 2004). These rates were based on interactions in both the west coast and Hawaii-based longline fisheries. In order to assess the range of possible captures of sea turtles in the west coast-based longline fishery, we used the upper and lower bounds of the 95% confidence interval to calculate the minimum and maximum number of turtles we would anticipate capturing in any given year.

From these estimated ranges of captures, mortalities were estimated via two methods. The first method used interaction information from the Pacific longline fleets to calculate the numbers of turtles that would fall within the categories of post-hooking survival described in the preceding section. Applying those mortality rates resulted in a range of possible mortalities that would occur in any given year. The second method used interaction information from longline fleets operating in the Atlantic Ocean, in the Northeast Distant (NED). Both methods used the same mathematical calculations to derive estimated ranges of mortalities. The only difference was in the proportions of animals assigned to the different categories.

There was an appreciable and sometimes large difference between the outcomes of the two methods. Each method had strengths and weaknesses. For example, the NED data set had large sample size and benefitted from detailed information collected at the time of fishery interactions with sea turtles. However, the Pacific information was more specific to the turtles and fisheries under consultation. A qualitative comparison of the difference between the two sets of data (NED and Pacific) indicated that there were differences between the types of interactions observed in the Atlantic and Pacific fisheries. However, these differences may also be an artifact of the sample size differences.

In order to fully assess the possible effects of the longline fishery, we used the estimated mortality ranges from both methods in our risk analyses to ensure against arbitrary over- or under-estimation of the risks. Table V-3 reports the values we calculated.

Table V-3. Estimated Captures and Mortalities of Sea Turtles in the West Coast-based Longline Fishery, as calculated via Pacific fishery and NED fishery interaction information.

Species	Captured (Range as determined by upper and lower bounds of 95% CI as calculated by Caretta (2004))	Estimated Mortalities using Pacific fishery interaction information	Estimated Mortalities using NED fishery interaction information
Green	0	0	0
Leatherback	23-57	7.4-24.5	3.2-7.9
Loggerhead	126-195	35.4-90.24	41.5-64.3
Olive Ridley	1-11	0.19-3.3	0.3-2.2

b. Purse Seines

In the ETP purse seine fishery, a sea turtle may be captured or entangled in net or gear used by the U.S. ETP tuna purse seine fishery. Sea turtles taken may be subsequently released alive and uninjured, released with light or grave injuries, or released dead. They may also evade or escape the net after capture.

Very few, if any, sea turtles are expected to be captured in the small vessel purse seine fishery. Injury or mortality to these turtles may result from being dropped on deck, excessive net abrasions or from being run through the power block as the net is hauled aboard. A turtle may become entangled in the webbing at any time during the set, including along the outside perimeter. Also, repeated capture of the same animals is likely to have an additive adverse effect, particularly if the fishing vessels are in an area where borderline animals have been resuscitated. Turtles that are recaptured may drown more easily if they have already been debilitated or weakened by the previous capture. Debilitated turtles also may be captured by other fisheries if the vessels are fishing in an area of high turtle density. Presumably, recapture would depend on the condition of the turtle and the intensity of fishing pressure in the area.

No stress studies have been conducted on sea turtles that have been released unharmed after being caught in a purse seine net. Stress and survivability studies have been conducted on the Hawaii longline fishery and the Atlantic shrimp trawl fishery. Sea turtles captured in the Hawaii longline fishery may suffer stress from internal or external hooking injuries and continued submergence. Sea turtles in the Atlantic shrimp trawl fishery are forcibly submerged by the trawls and kept submerged for long periods, often resulting in high mortalities. Contrary to these fisheries, turtles captured by the purse seines may suffer injuries from net entanglement or from being dropped on deck or run through the power block as the net is hauled aboard. The level and types of injuries suffered by turtles in the purse seine fishery, specifically the lack of incidents of forced submergence and eventual drowning, makes direct application of the results of these studies to turtles captured in the purse seine fishery difficult. Thus, NOAA Fisheries is only able to make assumptions on the condition of turtles that have been released "unharmed" from a purse seine net in the ETP. Although turtles released "unharmed" do not have visible injuries, they may have been stressed from being caught or entangled in a net. This stress may cause an interruption in essential feeding behaviors or migration patterns; however, NOAA Fisheries believes this effect, if experienced, is likely to be temporary and short-term. For these reasons, NOAA Fisheries will assume that any turtle released and reported as "unharmed," or uninjured, has not been harmed or harassed by its capture in the net and that latent effects are limited to short-term physiological stress or interruption of normal behavior patterns.

As discussed above, NOAA Fisheries does not anticipate any captures or mortalities of sea turtles in the small vessel purse seine fleet. Based on past information and expected levels of effort, it is unlikely that these vessels operate often enough in the same areas as turtles to make interactions possible. In addition, these vessels fish on schools of tuna instead of floating objects or FADs. School sets tend to have very low interaction rates with sea turtles.

c. Drift Gillnets

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

Potential impacts from the CA/OR drift gillnet fishery on sea turtles will generally be related to injury or mortality, although the entanglement episode, whether or not it develops into an injury or mortality, may also impact sea turtles. Injury or mortality of turtles entangled in a long-soaking drift gillnet may result from drowning due to forced submergence, and/or impairment or wounds suffered as a result of net entanglement.

While most voluntary dives by sea turtles appear to be aerobic, showing little if any increases in blood lactate and only minor changes in acid-base status (pH level of the blood), sea turtles that are stressed as a result of being forcibly submerged rapidly consume oxygen stores, triggering an activation of anaerobic glycolysis, and subsequently disturbing the acid-base balance, sometimes to lethal levels. It is likely that the rapidity and extent of the physiological changes that occur during forced submergence are functions of the intensity of struggling as well as the length of submergence (Lutcavage and Lutz, 1997). In a field study examining the effects of shrimp trawl tow times and sea turtle deaths, there was a strong positive correlation between the length of time of the tow and sea turtle deaths (Henwood and Stuntz, 1987, *in* Lutcavage and Lutz, 1997). Sea turtles forcibly submerged for extended periods of time show marked, even severe, metabolic acidosis as a result of high blood lactate levels. With such increased lactate levels, lactate recovery times are long (even as much as 20 hours), indicating that turtles are probably more susceptible to lethal metabolic acidosis if they experience multiple captures, because they would not have had time to process lactic acid loads (*in* Lutcavage and Lutz, 1997). Presumably, however, a sea turtle recovering from a forced submergence would most likely remain resting on the surface, which would reduce the likelihood of being recaptured in a drift gillnet submerged over 30 feet. Recapture would also depend on the condition of the turtle and the intensity of fishing pressure in the area. NOAA Fisheries has no information on the likelihood of recapture of sea turtles by the CA/OR driftnet fishery or other fisheries.

Additional factors such as size, activity, water temperature, and biological and behavioral differences between species also bear directly on metabolic rates and aerobic dive limits and will therefore also influence survivability in a gillnet. For example, larger sea turtles are capable of longer voluntary dives than small turtles, so juveniles may be more vulnerable to the stress of enforced submergence than adults. During the warmer months, routine metabolic rates are higher, so the impacts of the stress due to entanglement may be magnified. In addition, disease factors and hormonal status may also play a role in anoxic survival during forced submergence. Any disease that causes a reduction in the blood oxygen transport capacity could severely reduce a sea turtle's endurance in a net, and since thyroid hormones appear to have a role in setting metabolic rate, they may also play a role in increasing or reducing the survival rate of an entangled sea turtle (*in* Lutz and Lutcavage, 1997). As discussed further in the upcoming leatherback and loggerhead subsections, some sea turtle species are better equipped to deal with forced submergence.

No stress studies have been conducted on sea turtles that have been released alive after being caught in a drift gillnet. Survivability studies have been conducted on the Hawaii longline fishery and the Atlantic shrimp trawl fishery. Sea turtles captured in the Hawaii longline fishery may suffer stress and injury from entanglement and from internal or external hooking injuries and continued submergence. Sea turtles in the Atlantic shrimp trawl fishery are forcibly submerged by the trawls and kept submerged for long periods, often resulting in high mortalities. Similar to the Atlantic shrimp trawl fishery, turtles entangled in a long-soaking drift gillnet may drown due to forced submergence, or may suffer injuries from net entanglement. Thus, NOAA Fisheries is only able to make assumptions on the condition of turtles that have been released "unharmed" from a drift gillnet. Although turtles released "unharmed" do not have visible injuries, they may have been stressed from being caught or entangled in a net. This stress may cause an interruption in essential feeding behaviors or migration patterns; however, NOAA Fisheries believes this effect, if experienced, is likely to be temporary and short-term and that latent effects are limited to short-term physiological stress or interruption of normal behavior patterns.

Turtles that are involuntarily submerged experience an imbalance in blood homeostasis and require time to recover to normal ph, CO_2 , and lactate levels. If this recovery time is interrupted by additional forced submergence, the turtle may die as a result. This assumes that the "unharmed" turtle is able to recover. A loggerhead recovered from a shrimp trawl net was initially reported as normal, and subsequently became limp. The turtle was kept onboard and went through several periods of activity and lethargy. The turtle was transported to a laboratory facility and continued to exhibit periods of activity with alternate "limp periods" and was finally determined to have died (Stender, unpublished report, 2000). Thus, an apparent normal, active turtle that is returned to the water may subsequently drown.

Survival rates appear to be greater for hard-shelled turtles than for leatherbacks when forcibly submerged. For the purposes of this Opinion, the survival rates for the hard-shelled turtles (green, loggerhead and olive ridley) will be combined and the survival rate for the leatherback turtle will be calculated separately. Both survival rates are based on incidental capture data from July, 1990 to January, 2000 by the CA/OR drift gillnet fleet. Leatherbacks caught in this fishery had a survival rate of 39 percent (9 released unharmed/23 total captured), while the hard-shelled turtles had a combined survival rate of 68 percent (13 released unharmed/19 total captured). The total survival rate for all species combined is approximately 52 percent (22 released unharmed/42 total captured), 2.5 percent were released injured (1 injured/42 total), and 43 percent were killed accidentally (18 killed/42 total). The rest were unknown (1).

Based on expected future effort in the CA/OR drift gillnet fishery and the conditions under which sea turtle takes have been observed in the past, NOAA Fisheries anticipates that during years when the fishery operates in areas of warmer waters, or warm water intrudes into normal fishing grounds – such as occurred in November 1999, olive ridley and green sea turtles will be taken. During these years, as many as 4 of each species may be captured. Based on past survival rates, 1 of each of these species may be killed. Loggerhead sea turtles are only expected to be captured during El Nino events, based on past interactions. Based on measures to protect these turtles, NOAA Fisheries expects that as many as 5 loggerheads could be captured during an El Nino event, and 2 of these might die. Three leatherback turtles may be captured per year, and 2 of those might die, based on past interaction and kill data for this species.

d. Summary of Sea Turtle Responses to Interactions with the Fisheries

Table V-4 summarizes the numbers of estimated captures and mortalities of each species of sea turtle in the HMS fisheries. The table also assigns these captures and mortalities to the various turtle nesting aggregations based on the available information we have on the origin of sea turtles that interaction with HMS fisheries and other fisheries in the action area.

Green Sea Turtle. Assuming that patterns observed in the past represent future patterns, green sea turtles are expected to be captured by the CA/OR drift gillnet fishery. About 4 green turtles are expected to be captured by the fishery, primarily during El Nino events or when oceanographic conditions otherwise bring warmer waters into the fishing grounds. Of these turtles, 1 is expected to die as a result of the exposure. Two of these 4 might be endangered green turtles from nesting beaches in Mexico while the remaining 2 green turtles killed in an interaction with longline gear might have originated in either the Hawaiian nesting beaches or the Mexican nesting beaches.

Leatherback Sea Turtle. Leatherback sea turtles are expected to be captured in the west coast-based longline fishery and the CA/OR drift gillnet fishery. Between 26 and 60 leatherback turtles are expected to be captured by these fisheries each year. Three of these would be captured in the drift gillnet fishery. Of these 26 to 60 turtles, 5 to 27 are expected to die as a result of the exposure; 2 of these deaths would be expected in the drift gillnet fishery. One to 4 of those killed might be leatherback turtles from the eastern Tropical Pacific while 4 to 23 leatherback turtles killed in an interaction with longline gear would have originated in the western Pacific Ocean. Based on abundance patterns, we assume that most of the turtles that die as a result of exposure to longline and drift gillnet gear would have originated in Indonesia, although leatherback turtles from any of the smaller nesting aggregations in the western Pacific Ocean have a risk of being captured, injured, or killed by these fisheries in some years.

Loggerhead Sea Turtle. Loggerhead sea turtles are expected to be captured in the west coast-based longline fishery and the CA/OR drift gillnet fishery. Between 131 and 200 loggerhead turtles are expected to be captured by these fisheries each year. Five of these would be captured in the drift gillnet fishery, during El Nino events or when oceanographic conditions otherwise bring warmer waters into the fishing grounds. Of these 131 to 200 turtles, 37 to 92 are expected to die as a result of the exposure; 2 of these deaths would be expected in the drift gillnet fishery. Most of these loggerhead turtles would be oceanic juveniles originated from nesting beaches in southern Japan, while 2 to 5 of these oceanic juveniles could be from the two nesting beaches on Yakushima Island. Additionally, one loggerhead might be captured in any given year of the albacore surface hook and line fishery. Past interactions between loggerheads and this fishery have been very infrequent. NOAA Fisheries does not expect any loggerhead turtle to die as a result of an interaction with the albacore hook and line fishery.

Olive Ridley Sea Turtle. Assuming that patterns observed in the past represent future patterns, olive ridley sea turtles will be exposed to the longline and drift gillnet fisheries. Between 5 and 15 olive ridley turtles are expected to be captured by the HMS longline and drift gillnet fisheries each year; 4 of these turtles are expected to be taken in the drift gillnet fishery only during El Nino events or when oceanographic conditions otherwise attract olive ridley sea turtles to the fishing grounds. Of these 5 to 15 turtles, 1 to 4 are expected to die as a result of that exposure. One to 3 of the killed olive ridley turtles would be endangered turtles that have migrated from the eastern Tropical Pacific while the remaining olive ridley turtles killed in an interaction with longline gear would have originated in the western Pacific or Indian Ocean.

Table V-4. Annual estimates of the number of different species of sea turtles captured and killed by the HMS fisheries

Species and Scenario	**Exposure**		**Mortalities**			
	Lower Range	**Upper Range**	**Lower Range**	**Upper Range**	**NED Lower Range**	**NED Upper Range**
Green Sea Turtles	**4**		**1**		**0**	**0**
Scenario 1						
Eastern Tropical Pacific	2	-	0.5	-	0	0
Hawaiian population	0.48	-	0.12	-	0	0
Mixture	1.52	-	0.38	-	0	0
Rounded Values*						
Eastern Tropical Pacific	2.0	-	1.0	-	0.0	0.0
Hawaiian population	0.0	-	0.0	-	0.0	0.0
Mixture	2.0	-	0.0	-	0.0	0.0
Leatherback Sea Turtle	**26**	**60**	**9**	**27**	**5**	**10**
Scenario 1						
Eastern Tropical Pacific	3.64	8.4	1.26	3.78	0.7	1.4
Western Pacific nesting aggregations	22.36	51.6	7.74	23.22	4.3	8.6
Rounded Values*						
Eastern Tropical Pacific	4.0	8.0	1.0	4.0	1.0	1.0
Western Pacific nesting aggregations	22.0	52.0	8.0	23.0	4.0	9.0
Loggerhead Sea Turtle	**131**	**200**	**37**	**92**	**44**	**66**
Scenario 1						
Yakushima nesting aggregation	6.55	10	1.85	4.6	2.2	3.3
Other Japanese nesting aggregations	124.45	190	35.15	87.4	41.8	62.7
Rounded Values*						
Yakushima nesting aggregation	7.0	10.0	2.0	5.0	2.0	3.0
Other Japanese nesting aggregations	124.0	190.0	35.0	87.0	42.0	63.0
Olive Ridley Sea Turtle	**5**	**15**	**1.19**	**4.3**	**1.3**	**3.2**
Scenario 1						
Eastern Tropical Pacific	3.7	11.1	0.8806	3.182	0.962	2.368
Western Pacific and Indian Ocean	1.3	3.9	0.3094	1.118	0.338	0.832
Rounded Values*						
Eastern Tropical Pacific	4.0	11.0	1.0	3.0	1.0	2.0
Western Pacific and Indian Ocean	1.0	4.0	0.0	1.0	0.0	1.0

* Because of rounding, these values may produce estimates that, when summed, are higher than the initial total

3. Effects of the Proposed ESA Rule to Prohibit Shallow Longline Sets East of 150EW.

NOAA Fisheries, Protected Resources Division, Southwest Region proposes to use Secretarial authority under 11(f) of the ESA (as well as under the HSFCA, 16 U.S.C. 5503(d)) to promulgate regulations in the West Coast-based longline fishery (as described above in section II-A) to ensure the fishery complies with the ESA. Specifically, Protected Resources proposes (68 FR 70219, December 17, 2003) to prohibit West Coast-based longline vessels from making shallow longline sets on the high seas in the Pacific Ocean east of 150E W longitude to conserve leatherback and loggerhead sea turtles. This rule would become effective in March 2004 at the same time as the regulations implementing the FMP.

The proposed FMP would allow shallow longline sets east of the 150EW longitude. The number of leatherback, loggerhead, and olive ridley sea turtles that would be captured and killed as a result of this fishery are described in the section immediately preceding. Implementation of a rule that prohibited these sets, should result in avoidance of all of the estimated captures and mortalities of these species in the west coast-based longline fishery. All remaining captures and mortalities of sea turtles would be limited primarily to those occurring on the drift gillnet fishery.

If vessels in the west coast-based longline fishery choose to either relocate their effort to the Hawaii-based longline fishery, or abandon longline fishing entirely, the expected total captures and mortalities of sea turtles from HMS fisheries would be significantly less than that expected under the proposed HMS FMP (Table V-5).

Table V-5. Estimated captures and mortalities of sea turtles under the HMS FMP by all fisheries following implementation of the proposed ESA rule.

Species	Captured	Killed	Environmental Conditions
Green Turtles	4	1	SSTs in fishing area similar to Nov-99
Leatherback Turtles	3	2	
Loggerhead Turtles	5	2	Only during El Nino events
Olive Ridley Turtles	4	1	SSTs in fishing area similar to Nov-99

If west coast-based vessels that previously targeted swordfish with shallow-sets instead switched to deep sets to target tuna species, capture and mortality rates would decrease from those estimated for the shallow set fishery, but the total estimated captures and mortalities would be higher than those listed in Table V-5. Analyses of the deep-set longline fishery out of Hawaii indicate that far fewer turtles are captured on deep-sets. However, the mortality rate per interaction is higher in deep-sets possibly because hooked or entangled sea turtles cannot reach the surface to breathe and therefore drown.

Most vessels are expected to either return to the Hawaii-based fishery after renewing their limited entry permits, or a very few may experiment with deep-set longline fishing. Based on past effort and tuna catch data in deep-set fleets, the area east of 150EW longitude is not expected to support much deep set effort. As a result, captures and mortalities under the HMS FMP longline fishery would be minimal.

4. Risk Analyses

As discussed in the *Approach to the Assessment*, the final step of our assessment uses the results from our exposure and response analyses to ask (1) what is likely to happen to different nesting aggregations given the exposure and responses of individual members of those aggregations, and (2) what is likely to happen to the populations or species those nesting aggregations comprise. These analyses form the foundation for our jeopardy determinations, which are designed to determine if we would reasonably expect threatened or endangered species to experience reductions in reproduction, numbers, or distribution that would appreciably reduce the species' likelihood of surviving and recovering in the wild (since the proposed actions are not likely to adversely affect designated critical habitat, this Opinion did not conduct “destruction and adverse modification” analyses).

In the *Status of the Species and Environmental Baseline* sections of this Opinion, we discussed the various natural and human-related phenomena that caused the various sea turtle and marine mammal species to become threatened or endangered and continue to keep their populations suppressed. For sea turtles, this section of the Opinion examines the physical, chemical, and biotic effects of the fisheries associated with the HMS FMP to determine (a) if those effects can be expected to reduce the reproduction, numbers, or distribution of threatened or endangered species in the action area, (b) determine if any reductions in reproduction, numbers, or distribution would be expected to appreciably reduce the Pacific Ocean population's likelihood of surviving and recovering in the wild through impacts to nesting aggregations, and (c) if appreciable reductions in the Pacific Ocean population's likelihood of surviving and recovering in the wild would cause appreciable reductions in the species (as listed) likelihood of surviving and recovering in the wild.

For the purposes of this analysis, we will assume that anything that places sea turtle populations in the Pacific Ocean at greater risk of extinction, also places the entire species at a greater risk of extinction; or, in other words, reduces the species' likelihood of survival and recovery. This assumption is reasonable based on the relationship between local and regional persistence in species (see Gotelli, 2001 for a simple explanation). Based on this relationship, the risk of regional extinction is lower than the risk of local extinction; however, as local probabilities change, the probability of regional persistence changes correspondingly.

Marine Mammals

1. Fin Whale

Based on the available information on interactions between fin whales and the HMS fisheries, a single observed fin whale has been entangled and killed in drift gillnet fishing gear in 13 years of observation. These interactions appear to be rare occurrences likely triggered by oceanographic conditions such as the El Nino event ongoing at the time of the interaction. Because the HMS fisheries have a small probability of interacting with fin whales, those few interactions are not expected to reduce the reproduction, numbers, or distribution of fin whales. Since the implementation of measures in the drift gillnet fishery to protect leatherback and loggerhead sea turtles, no fin whales have been cpatured. As a result, NOAA Fisheries does not expect that HMS FMP impacts to fin whales will be likely to reduce the fin whales' likelihood of surviving and recovering in the wild.

2. Humpback Whale

Based on the available information on interactions between humpback whales and the HMS fisheries, humpback whales have been entangled in drift gillnet fishing gear. However, based on the information available on these previous interactions, the interactions appear to be rare occurrences and, when they have occurred, the humpback whales do not appear to have been injured or killed by the interaction. Because the HMS fisheries have a small probability of interacting with endangered humpback whales and, when they occur, they do not appear to kill or injure the whales, those interactions are not likely to reduce the reproduction, numbers, or distribution of humpback whales. As a result, they are not likely to reduce the humpback whales' likelihood of surviving and recovering in the wild.

3. Sperm Whale

Based on the available information on interactions between sperm whales and the HMS fisheries, sperm whales have been entangled in drift gillnet fishing gear. Based on the information available on these previous interactions, the interactions have been common occurrences when compared to other listed whales. Sperm whales were injured or killed during the interactions. However, in the 6 years following implementation of the Pacific Cetacean Take Reduction Plan (PCTRP) only one sperm whale has been captured and that capture was in a set that was not compliant with the requirements of the PCTRP. As a result, sperm whale interactions are now expected to be rare occurrences. Because the HMS fisheries have a small probability of interacting with and killing endangered sperm whales, those few interactions are not likely to reduce the reproduction, numbers, or distribution of sperm whales. As a result, they are not likely to reduce the sperm whales' likelihood of surviving and recovering in the wild.

Sea Turtles

The following discussions describe the expected risks to threatened and endangered sea turtles from the proposed HMS FMP fisheries and the proposed rule to prohibit shallow longline sets east of 150EW longitude. We first assess the risks associated with the proposed FMP, followed by a discussion of the effects of the proposed longline set prohibition in order to individually and then cumulatively evaluate the risks the proposed actions pose to listed sea turtles. For the FMP analyses we projected the effects of the fisheries on listed sea turtles over 1-, 10- , and 20-year time periods to assess the reasonable likelihood of the FMP actions appreciably reducing the likelihood of both the survival and recovery of these species in the wild.

1. Green Turtles

Assuming that patterns observed in the past represent future patterns, green sea turtles are expected to be captured by the CA/OR drift gillnet fishery only. About 4 green turtles are expected to be captured by the fishery, primarily during El Nino events or when oceanographic conditions otherwise bring warmer waters into the fishing grounds. Of these turtles, 1 is expected to die as a result of the exposure. Based on genetic data collected in HMS and other U.S. fisheries, most green turtles in the action area will probably be members of the Hawaiian (French Frigate Shoals) or Mexican (Pacific coast) nesting aggregations. Genetic analysis of one green turtle taken in the HMS fisheries indicated eastern Pacific origin. As a result, two of these 4 might be endangered green turtles from nesting beaches in Mexico while the remaining 2 green turtles killed in an interaction with longline gear might have originated in either the Hawaiian nesting beaches or the Mexican nesting beaches.

Life history information collected by observers in other fisheries operating in the action area suggests that juvenile, subadult and adult green turtles occur on the fishing grounds. From those turtles for which genetic data were collected, turtles originating from Hawaiian nesting aggregations were represented by smaller animals (juvenile and sub-adult sizes); turtles from Mexican nesting aggregations were represented by larger animals (sizes that suggest they were probably adult turtles). There have been no observed captures of green turtles in the west coast-based longline fishery. One green turtle has been observed taken in the CA/OR drift gillnet fishery. This animal was an immature turtle from one of the eastern Pacific nesting aggregations.

Several authors have demonstrated that long-lived species that have evolved low, adult mortality rates, and delayed maturity cannot sustain high adult or juvenile mortalities without having increased extinction risk. For example, Crouse (1999) discussed the importance of high adult and juvenile survival in long-lived species with delayed maturity; after examining the population ecology of a large number of these species (including leatherback and loggerhead sea turtles, and several species of sharks, rockfish, groundfish, albatross, and whales), she concluded that seemingly small numbers of deaths in these species, particularly of adults and juveniles, could have catastrophic effects on the health of population of these long-lived species. Crouse (1999), Heppell (1999), and Caswell (2001) demonstrated that changes in the survival of adult and sub-adult stages of loggerhead turtles can have significant, short-term effects on the status and trend of these turtle populations. Heppell *et al.* (1999) reached similar conclusions based on demographic evaluations of several species of sea turtles and sharks. Congdon *et al.* (1999) and Congdon and Dunham (1984) reached the same conclusions after conducting demographic simulations of several species of long-lived freshwater turtles and sea turtles. Caswell *et al.* (1999) concluded that the loss of small numbers of adult females would be sufficient to critically endanger the western Atlantic population of northern right whales (*Eubalaena glacialis*), which is another long-lived species with delayed maturity.

Because of the similarities between these life history patterns and those of green turtles (they are long-lived, have high adult survival rates, and delayed maturity), we assume that changes in the survival of adult and sub-adult stages of green turtles would have significant, short-term effects on the status and trend of these turtle populations. Because of their life history pattern, the long lives and high adult and sub-adult survival rates of sea turtles would mask changes in the survival rates of non-adult age classes. Nevertheless, we do not believe these mortalities (the periodic loss of 1 adult or sub- adult green turtle) would be expected to appreciably reduce the threatened or endangered green turtle's likelihood of surviving and recovering in the wild. This conclusion is based on the number of green turtles that are likely to be killed during interactions with the fishery relative to the size of the subpopulation to which those turtles probably belong and the changed conditions of the Environmental Baseline. We will discuss the status and trend of the two aggregations separately, then summarize our conclusions for both.

Eastern Pacific Green Turtle Population. As discussed in the *Status of the Species* section of this opinion, the primary green turtle nesting grounds in the eastern Pacific are located in Michoacán, Mexico, and the Galapagos Islands, Ecuador (NOAA Fisheries and USFWS, 1998a). The nesting aggregation at the two main nesting beaches in Michoacán, (Colola — which represents about 70% of the total green turtle nesting in Michoacán — and Maruata; Delgado and Alverado, 1999), decreased from 5,585 females in 1982 to 940 in 1984. On Colola, an estimated 500-1,000 females nested nightly in the late 1960s. In the 1990s, that number dropped to 60-100 per night, or about 800-1,000 turtles per year (Eckert, 1993). During the 1998-99 season, based on a comparison of nest counts and egg collection data, an estimated 600 green turtles nested at Colola.

In 1990, the government provided female green turtles and their eggs with long-term protection from poaching and other activities. During the 1998-99 season, only about 5% of the nests were poached at Colola, although about 50% of the nests at Maruata were poached because political infighting made it difficult to protect the turtles on this beach (Delgado and Alvarado, 1999). Nevertheless, despite the long-term protections, the nesting aggregation continues to decline, and investigators believe that human activities (including incidental take in various coastal fisheries as well as illegal directed take at forage areas) continue to prevent the aggregations from recovering (P. Dutton, NOAA Fisheries, personal communication, 1999; Nichols, 2002).

There are few historical records of abundance of green turtles from the Galapagos - only residents are allowed to harvest turtles for subsistence, and egg poaching occurs only occasionally. An annual average of 1,400 nesting females was estimated for the period 1976- 1982 in the Galapagos Islands (NOAA Fisheries and USFWS, 1998a). More current estimates of the status and trend of this population are not available.

The periodic loss of 1 adult or sub-adult green turtle from any of these nesting aggregations would reduce the number of animals in the sub-population. If we assume that some of the adult or sub-adult turtles that are killed during interactions with the fishery are female, this reduction in numbers would also reduce the number of adult turtles that reproduce each year.

Hawaiian Green Turtle Population. The green turtles in Hawaii are genetically-distinct and geographically isolated from other green turtle populations; therefore, we treat them as a discrete subpopulation. Ninety percent of the nesting and breeding activity of the Hawaiian green turtle occurs at

French Frigate Shoals, where 200-700 females were estimated to nest annually (NOAA Fisheries and USFWS, 1998a). The incidence of diseases such as fibropapilloma, and spirochidiasis, which are major causes of strandings of green turtles suggests that future declines in this population could reverse or eliminate the increases of recent decades (Murakawa *et al.*, 2000). Nevertheless, since the green turtles in Hawaii were first protected in the early 1970s, ending years of exploitation, the nesting population of green turtles in Hawaii has shown a definite increase (Balazs, 1996, Chaloupka and Balazs in press). For example, the number of green turtles nesting at an index study site at East Island has tripled since systematic monitoring began in 1973 (NOAA Fisheries and USFWS, 1998a).

Killing 1 of these green turtles periodically would reduce the abundance of this nesting aggregation. If we assume that some of the adult turtles that are killed in interactions with the Hawaii-based longline fisheries are females, then the fishery would also reduce the reproduction of this nesting aggregation.

Synthesis: Almost all of the green turtles that interact with the HMS fisheries are probably members of the eastern Pacific and Hawaiian nesting aggregations. If we assume that some of the adult turtles that are killed in interactions with the Hawaii-based longline fisheries are females, then the fishery would also reduce the reproduction of these nesting aggregations, although, the consequences of losing a female turtle on the dynamics of a turtle's population will vary depending on whether the adult female dies before or after she lays her eggs (if the turtle dies before laying her eggs, the potential effect on the population would be larger).

In the *Environmental Baseline* section of this opinion, we noted that green turtles are captured, injured, or killed in numerous Pacific fisheries including Japanese longline fisheries in the western Pacific Ocean and South China Seas; longline fisheries off the Federated States of Micronesia; commercial and artisanal swordfish fisheries off Chile, Columbia, Ecuador, and Peru; and purse seine fisheries for tuna in the eastern tropical Pacific Ocean. Because of limited available data, we cannot accurately estimate the number of green turtles captured, injured, or killed through interactions with these fisheries. However, an estimated 85 green turtles were estimated to have died between 1993 and 1997 in interactions with the tuna purse seine fishery in the eastern tropical Pacific Ocean; approximately 7,800 green turtles are estimated to die annually in fisheries and direct harvest off of Baja, California; and before 1992, the North Pacific driftnet fisheries for squid, tuna, and billfish captured an estimated 378 green turtles each year, killing about 93 of them each year. Little data on the life stage or sex of captured animals are available; however, we expect that both incidental and intentional takes affect the larger turtle life stages, sub-adults and adults, and generally the same nesting aggregations as are affected by the proposed actions. Given the population ecology of sea turtles in general, and green turtles in particular, these mortalities would be expected to reduce the numbers of these green turtles.

Although the mortalities associated with the HMS fisheries would reduce the numbers and may reduce the reproduction of both the eastern Pacific and Hawaiian nesting aggregations, the "jeopardy" standard requires us to consider those effects on a species' survival and recovery in the wild. Specifically, as discussed in the introduction to the *Effects of the Action* section, the "jeopardy" standard requires us to determine that reductions in a species' reproduction, numbers, or distribution would be expected to appreciably reduce a species' likelihood of surviving and recovering in the wild. We identify reductions in a species' likelihood of surviving and recovering in the wild by quantitatively or qualitatively analyzing the probable effect of changes in reproduction, numbers, and distribution based on our understanding of relationships between vital rates (for example, age- or stage-specific rates of survival or fecundity), variance in those rates over time and among different populations, a species' rates of increase (lambda), and a species' probability of quasi-extinction or persistence over time.

Nevertheless, we estimate that 1 adult or sub-adult female green turtle from the eastern Pacific nesting aggregations or 1 from the Hawaiian nesting aggregations could be periodically killed in the HMS drift gillnet fishery. Killing this number of green turtles would represent a small proportional change in the survival rates of female turtles in those life history stages. The quantitative analyses we conducted to assess the potential risks these mortalities might pose to the different nesting aggregations could not detect the effect of these mortalities on the extinction risk of either the endangered or threatened green sea turtles, although these results may say more about the power of the models than they says about the effect of these mortalities on the different nesting aggregations. With such small sample sizes, moderate amounts of observer bias, and wide confidence intervals, the ability of these quantitative methods to detect these small effects is very limited.

To approach the assessment qualitatively, we asked if the deaths associated with the proposed fisheries are likely to be exceeded by the number of younger turtles recruiting into the adult or sub-adult population. Although most populations are designed to withstand some level of mortality without increases in their risk of extinction, threatened and endangered species will often be incapable of recovering from even small numbers of deaths of some life stages. Further, most populations fluctuate over time, if a population is experiencing an increasing trend in a longer cycle, it is more likely to be able to withstand mortalities than if the population is experiencing a decreasing trend. The important consideration is whether the population appears to have a growth rate that would allow it to recover from small numbers of deaths.

The Hawaii nesting aggregation of green turtles has been increasing for several years and has the demographic characteristics of a population that is slowly recovering from historic declines (see the detailed assessment in Status). Similarly, our assessment of the female green turtles that nest at Colola Beach suggest that this nesting beach is also growing, on average, despite a lower confidence interval suggesting that the population may, in fact, be declining. The wide fluctuations in the number of nesting females that return from year-to-year could present a more serious problem for this population as those fluctuations bring the population to very low levels that, over time, would be expected to create weak year-classes of recruits into the adult, female population. Although the increases in nesting females in 2000 and 2001 provide cause for optimism, historical numbers of this species nesting during the 1960s show that the population is still below its natural level (Alvarado-Diaz and Trejo, 2003; Alvarado-Diaz, personal communication, October, 2003).

Conclusions for the proposed HMS FMP. Overall, both nesting aggregations appear to be resilient to the periodic loss of 1 adult or sub-adult female. Therefore, NOAA Fisheries does not expect that the impacts of the HMS fisheries are sufficient to reduce appreciably both the likelihood of survival and recovery of either the Hawaiian or eastern Pacific nesting aggregations or the two green turtle species as listed.

Assessment of the proposed rule prohibiting shallow longline sets. Green turtles have never been observed captured in shallow longline sets east of 150EW longitude. As a result, the proposed prohibition of these sets is unlikely to affect green sea turtles. The rule may have a positive effect by precluding future captures that could have occurred.

Summary of risks posed from the proposed actions taken together. The combined effects of the actions under consultation are not expected to reduce appreciably the likelihood of survival and recovery of threatened green sea turtles as they are listed globally or endangered green sea turtles as they are listed for nesting aggregations in the eastern Pacific.

2. Leatherback Turtles

Leatherback sea turtles are expected to be captured in the west coast-based longline fishery and the CA/OR drift gillnet fishery. Between 26 and 60 leatherback turtles are expected to be captured by these fisheries each year. Three of these would be captured in the drift gillnet fishery. Of these 26 to 60 turtles, 5 to 27 are expected to die as a result of the exposure; 2 of these deaths would be expected in the drift gillnet fishery. Although we estimate that the proposed HMS fisheries might kill as many as 27 leatherback turtles in the Pacific Ocean, that estimate was based on the most conservative assumptions we could make; the most likely estimates range from about 5 to 9 mortalities associated with these fisheries.

Based on genetic analysis of leatherback turtles captured in the Hawaii-based longline fishery and the HMS fisheries, leatherback turtles found in the action area are from nesting aggregations in the western Pacific region (Papua New Guinea, Indonesia, and Solomon Islands) or the eastern Pacific region (Mexico and Costa Rica). All three leatherback turtles sampled in the HMS fisheries were from the western Pacific nesting aggregations. Similarly, all samples taken from stranded leatherbacks on the California coast have indicated representation from western Pacific nesting beaches (Dutton, *et al*., in press, and P. Dutton, personal communication, March, 2000).

Therefore, one to 4 of those killed in the HMS fisheries might be leatherback turtles from the eastern Tropical Pacific while 4 to 23 leatherback turtles killed in an interaction with longline gear would have originated in the western Pacific Ocean. Based on abundance patterns, we assume that most of the turtles that die as a result of exposure to longline and drift gillnet gear would have originated in Indonesia, although leatherback turtles from any of the smaller nesting aggregations in the western Pacific Ocean have a risk of being captured, injured, or killed by these fisheries in some years.

Straight carapace lengths taken from a subset of the leatherback turtles caught in these fisheries suggest that some of them were subadults, representing both early and late pelagic stage, based on the stage structure for Malaysian turtles presented in Bolten, *et al*. (1996). Only five of the turtles caught in the CA/OR drift gillnet fishery were measured, all between 132 to 160 cm (sub-adults and adults). Most of the leatherbacks caught in these fisheries were not measured. Those leatherbacks that were not measured may have been too large to be safely brought on board; therefore they may have been adults. It appears that young leatherback turtles (carapace length <100 cm) reside only in waters warmer than 26EC, which should generally place them outside of areas in which longline swordfish fleets operate (Eckert, 1999b; Eckert, 2002).

Published estimates of the abundance of nesting, female leatherback turtles in the Pacific Ocean have established that leatherback populations have collapsed or are declining at all major Pacific basin nesting beaches over the past two decades (Spotila *et al*., 1996; NOAA Fisheries and USFWS 1998b; Sarti, *et al*. 2000; Spotila, *et al*. 2000). Leatherback turtles had disappeared from India before 1930, have been virtually extinct in Sri Lanka since 1994, and appear to be approaching extinction in Malaysia (Spotila *et al*. 2000). Leatherback turtle colonies throughout the eastern and western Pacific Ocean have been reduced to a fraction of their former abundance by the combined effects of human activities that have reduced the number of nesting females and reduced the reproductive success of females that manage to nest (for example, egg poaching). At current rates of decline, leatherback turtles in the Pacific basin are a critically endangered species with a low probability of surviving and recovering in the wild (see *Status of the Species and Environmental Baseline*).

As discussed previously, almost all of the leatherback turtles that interact with the Hawaii-based longline fisheries are probably members of the western Pacific nesting aggregation, which consists of nesting aggregations located in Indonesia, Papua New Guinea, and the Solomon Islands. In the *Status of the Species and Environmental Baseline* section of this Opinion, we established that in the western Pacific Ocean and South China Seas, leatherback turtles are captured, injured, or killed in numerous fisheries including Japanese longline fisheries. Leatherback turtles in the western Pacific are also threatened by poaching of eggs, killing of nesting females, human encroachment on nesting beaches, incidental capture in fishing gear, beach erosion, and egg predation by animals. As a result of these threats, the nesting assemblage at Terengganu - which was one of the most significant nesting sites in the western Pacific Ocean - has declined severely from an estimated 3,103 females in 1968 to 2 nesting females in 1994 (Chan and Liew, 1996). The size of the current nesting assemblage represents less than 2 percent of the size of the assemblage reported from the 1950s; with one or two females nesting in this area each year (P. Dutton, personal communication, 2000). Nesting assemblages of leatherback turtles along the coasts of the Solomon Islands, which supported important nesting assemblages historically, are also reported to be declining (D. Broderick, personal communication, *in* Dutton *et al*. 1999). In Fiji, Thailand, Australia, and Papua-New Guinea (East Papua), leatherback turtles have only been known to nest in low densities and scattered colonies.

The nesting aggregations of leatherback turtles in the western Pacific currently consist of about 1,000 adult females (between 400 and 500 nesting females per year in Indonesia, about 150 in the Solomon Islands, about 400 in Papua New Guinea, and a handful in Malaysia). Killing 5 to 9 of these adult or sub-adult leatherback turtles each year would reduce the numbers of these nesting aggregations (reductions in numbers are one of the considerations in jeopardy analyses). If we assume that all or most of the 5 to 9 turtles that are killed in interactions with the HMS fisheries are females (some of the turtles that have been captured in Pacific longline fisheries have been male, but adult females predominate), then the fishery would reduce the reproduction of this nesting aggregation.

Given that satellite tracking results of leatherback turtles tagged on nesting beaches indicate that tagged turtles from beaches in Papua New Guinea head into the southern Pacific Ocean and given the abundance of the Indonesia leatherbacks relative to the other nesting aggregations in the western Pacific, it is reasonable to assume that all of the western Pacific leatherback turtles captured in the HMS fisheries are from the State of Papua, Indonesia. Counts of adult, nesting females in the Indonesia nesting aggregation, which is one of the largest remaining nesting aggregations for leatherback turtles in the Pacific Ocean, have fluctuated between 400 and 1,000 individuals throughout most of the 1990s, but has fluctuated between 390 and 625 since the early 2000s. Our assessment of this population suggests that the long-term trend of this population is declining slightly, although it may be stable of slightly increasing in some years.

If we assume that, in most years, all of these turtles migrate into the action area from Indonesia, then our estimates of the average number of leatherback turtles that would be killed in this fishery (5 or 12-16, depending on how we estimate the mortality rates) would represent between 0.5 and 1.5 percent of current estimates of the number of females in the population (depending on assumptions about remigration rates). The upper limits of our mortality estimates would represent about 2.2 percent of the number of nesting females. If we assume that, in all or some years, leatherback turtles from Papua New Guinea, the Solomon Islands, or Malaysia may also be captured and killed by the fisheries, then the risks to the Indonesian nesting aggregation would be smaller, but the risks to these other nesting aggregations would be larger.

To assess the risks the HMS fisheries might pose to leatherback sea turtles, we conducted a series of simulations that considered how these additional deaths of adult or subadult leatherback turtles would affect the demographic variables and extinction risks of a population of leatherback sea turtles. Using the approach and the procedures described in the Approach to the Assessment section of this Opinion, we considered different scenarios using the following annual mortalities, which were taken from Table V-6: 4 (to assess the risks the mortalities would pose to the Playa Grande nesting aggregation), 7 (to assess the risks of the mortalities assuming mortality rates are equivalent to those reported for the Northeast Distant), and 23 (to assess the risks of the mortalities assuming mortality rates are equivalent to those reported for the Pacific Ocean).

Table V-6. Mortality estimates used in assessment scenarios for leatherback turtles that would be killed in the HMS fisheries

Nesting Aggregation	Estimates Using Pacific Rates		Estimates Using NED Rates	
	Lower Range	Upper Range	Lower Range	Upper Range
Eastern Tropical Pacific	1.0	4.0	1.0	1.0
Western Pacific	8.0	23.0	4.0	7.0

As with green sea turtles, the quantitative analyses we conducted to assess the potential risks these mortalities might pose to the different nesting aggregations could not detect the effect of these mortalities on the extinction risk of leatherback sea turtles in the Indonesian nesting aggregation. However, given limitations in the information available, the procedures we used had a limited ability to detect those effects and, at best, would underestimate any effects they detected. Given these limitations, we assume that any differences we could detect were much larger than our analyses would suggest.

For our assessment, we then considered the potential consequences of these mortalities on leatherback turtles qualitatively, relying on published information and our knowledge of the population dynamics of imperilled species, particularly sea turtles, and their response to mortalities. Specifically, we asked if the deaths associated with the proposed fisheries are likely to be exceeded by the number of younger turtles recruiting into the adult or sub-adult population. Although most populations are designed to withstand some level of mortality without increases in their risk of extinction, threatened and endangered species will often be incapable of recovering from even small numbers of deaths of certain life stages. Further, most populations fluctuate over time, if a population is experiencing an increasing trend in a longer cycle, it is more likely to be able to withstand mortalities than if the population is experiencing a decreasing trend. The important consideration is whether the population appears to have a growth rate that would allow it to recover from small numbers of deaths.

Spotila et al. (1996) and Spotila et al. (2000) also considered this question using information derived from Playa Grande, Costa Rica, which is the fourth largest leatherback nesting colony in the world. Based on their analyses, which included a number of assumptions about the vital rates of leatherback turtles, they concluded that leatherback turtle populations can withstand no more than 1.0 percent increase in annual mortality from human sources before their populations start to decline. They estimated that the eastern Pacific population of leatherback turtles could withstand the death of about 17 adult females per year and the western Pacific population could withstand the death of about 18 adult females per year.

Although these estimates were intended to address all sources of human mortality and were designed to facilitate the species' recovery, they seem reasonable guides for our assessments that are designed to prevent individual actions from making things worse for threatened and endangered species even if they do not facilitate the species' recovery. Although we estimate that the proposed HMS fisheries might kill as many as 27 leatherback turtles in the Pacific Ocean, that estimate was based on the most conservative assumptions we could make; the most likely estimates range from about 5 to 9 mortalities associated with these fisheries. Applying this criterion to the proposed HMS fisheries, we believe the mortalities (the annual mortality of between 5 and 9 adult or sub-adult leatherback turtles or between 0.5 and <1.0 percent of the total estimates) associated with the proposed would not appreciably reduce the leatherback sea turtle's likelihood of surviving and recovering in the Pacific Ocean.

Conclusions for the proposed HMS FMP. Based on careful analysis, relying on our qualitative and quantitative assessments of the potential effects of the HMS fisheries and our past assessments of the risk posed to leatherback sea turtles in other U.S. Pacific Ocean fisheries, we could detect no increases in the risks of extinction for one of the largest nesting aggregations in the Pacific Ocean. Therefore, NOAA Fisheries does not expect that the impacts of the HMS fisheries are sufficient to reduce appreciably both the likelihood of survival and recovery of either the eastern or western Pacific nesting aggregations or the leatherback turtles as listed.

Assessment of the proposed rule prohibiting shallow longline sets. Implementation of the proposed ESA regulation to prohibit shallow longline sets is expected to preclude the anticipated captures and mortalities of leatherback sea turtles in the west coast-based longline fishery. Therefore, we would expect that the remaining interactions in the CA/OR drift gillnet fishery would capture up to 3 leatherback turtles, and kill two of them, annually. These estimates are well below the levels of mortality expected to result in appreciable changes in the likelihood of survival and recovery of the species. This fishery has not taken any leatherback turtles since the implementation of protective measures approximately 3 years ago.

Summary of risks posed from the proposed actions taken together. The combined effects of the actions under consultation are not expected to reduce appreciably the likelihood of survival and recovery of the leatherback sea turtles as they are listed globally or in nesting aggregations in the eastern or western Pacific.

3. Loggerhead Turtles

Loggerhead sea turtles are expected to be captured in the west coast-based longline fishery and the CA/OR drift gillnet fishery. Between 131 and 200 loggerhead turtles are expected to be captured by these fisheries each year. Five of these would be captured in the drift gillnet fishery, during El Nino events or when oceanographic conditions otherwise bring warmer waters into the fishing grounds. Of these 131 to 200 turtles, 37 to 92 are expected to die as a result of the exposure; 2 of these deaths would be expected in the drift gillnet fishery. Most of these loggerhead turtles would be oceanic juveniles originated from nesting beaches in southern Japan, while 2 to 5 of these oceanic juveniles could be from the two nesting beaches on Yakushima Island. Additionally, one loggerhead might be captured in any given year by the albacore surface hook and line fishery. Past interactions between loggerheads and this fishery have been very infrequent. NOAA Fisheries does not expect any loggerhead turtle to die as a result of an interaction with the albacore hook and line fishery.

Killing between 37 and 92 oceanic juvenile loggerhead turtles each year would reduce the numbers of individuals in the species. Assuming that some of the loggerhead turtles captured and killed in the fishery would be females, we would also conclude that these deaths would reduce the number of female loggerhead turtles that recruit into the adult, breeding population, with future effects on the species' reproduction.

As we had discussed in the *Status of the Species and Environmental Baseline* section of this Opinion, loggerhead sea turtles in the Pacific Ocean are represented by a northwestern Pacific nesting aggregation (located in Japan) and a smaller southwestern nesting aggregation that occurs in Australia (Great Barrier Reef and Queensland), New Caledonia, New Zealand, Indonesia, and Papua New Guinea. Based on available information, the Japanese nesting aggregation is significantly larger than the southwest Pacific nesting aggregation. Data from 1995 estimated the Japanese nesting aggregation at 1,000 female loggerhead turtles (Bolten *et al*., 1996; Sea Turtle Association of Japan, 2002). Recent data reflect a continuing decline (see Table 2 in Appendix B; N. Kamezaki, Sea Turtle Association of Japan, personal communication, August, 2001). We have no recent, quantitative estimates of the size of the nesting aggregation in the southwest Pacific, but currently, approximately 300 females nest annually in Queensland, mainly on offshore islands (Capricorn-Bunker Islands, Sandy Cape, Swains Head; Dobbs, 2001).

In the *Status of the Species and Environmental Baseline* section of this Opinion, we established that loggerhead turtles are captured, injured, or killed in numerous Pacific fisheries including Japanese longline fisheries in the western Pacific Ocean and South China Seas; direct harvest and commercial fisheries off Baja California, Mexico, commercial and artisanal swordfish fisheries off Chile, Columbia, Ecuador, and Peru; and purse seine fisheries for tuna in the eastern tropical Pacific Ocean. In addition, the abundance of loggerhead turtles on nesting colonies throughout the Pacific basin has declined dramatically over the past 10 to 20 years. Loggerhead turtle colonies in the western Pacific Ocean have been reduced to a fraction of their former abundance by the combined effects of human activities that have reduced the number of nesting females and reduced the reproductive success of females that manage to nest (for example, egg poaching). Despite limited quantitative data on the effects of these fisheries and other natural and anthropogenic phenomena on the Japanese nesting population, the effects of the mortalities associated with the HMS FMP fisheries added to the current status and trend of the Japanese loggerhead population would increase the Japanese loggerhead population's rate of decline.

Given the population ecology of loggerhead turtles and the status and trend of loggerhead nesting populations in the Pacific, these results are not surprising. As we discussed in the Life Cycle and Population Dynamics section for loggerhead turtles in Section III, changes in the survival rates of adult and sub-adult loggerhead turtles are likely to have the highest proportional effect on their population growth. In addition, the survival rates of oceanic juveniles had the second highest proportional effect on population growth rate, which may be an artifact of a population whose structure has been modified by various human activities and natural phenomena. This dynamic suggests that changes in the survival rates of mature, reproductive, adults and oceanic juveniles would have the largest proportional effect on this population's trend: increasing those survival rates would help the population recover from its decline while decreasing those survival rates would exacerbate the population's rate of decline.

Other investigators have demonstrated that long-lived species that have evolved low, adult mortality rates, and delayed maturity cannot sustain high adult or juvenile mortalities without having increased extinction risk. For example, Crouse (1999) discussed the importance of high adult and juvenile survival in long-lived species with delayed maturity; after examining the population ecology of a large number of these species (including leatherback and loggerhead sea turtles, and several species of sharks, rockfish, groundfish, albatross, and whales), she concluded that seemingly small numbers of deaths in these species, particularly of adults and juveniles, could have catastrophic effects on the health of population of these long-lived species. Heppell et al. (1999) and Heppell et al. (2002) reached similar conclusions based on demographic evaluations of several species of sea turtles and sharks. Congdon et al. (1999) and Congdon and Dunham (1984) reached the same conclusions after conducting demographic simulations of several species of long-lived freshwater turtles and sea turtles. Caswell et al. (1999) concluded that the loss of small numbers of adult females was sufficient to critically endanger the western Atlantic population of northern right whales (*Eubalaena glacialis*), which is another long-lived species with delayed maturity. Fujiwara and Caswell (2001) concluded that the survival of one or two adult females could be sufficient to avoid the extinction of the western Atlantic population of right whales; increasing the mortalities in this species by the same amount would be expected to increase the species' risk of extinction.

We also considered the risks the HMS fisheries might pose to these loggerhead sea turtles quantitatively by conducting a series of simulations of how these additional deaths of large oceanic juveniles would affect the demographic variables and extinction risks of a population of loggerhead sea turtles. We were specifically interested in identifying the incremental effects of these mortalities on the turtle's demographic variables or extinction risks, rather than estimating absolute increases in risk.

For those analyses, we converted census estimates for Inakahama and Maehama Beaches, treating the time series of these counts as a "simulated" population of loggerhead sea turtles with appropriate levels of year-to-year variation. We then subtracted a constant level of mortality from this "population' and re-calculated the demographic variables for the "population" that had been affected by these mortalities – we considered using stochastic mortality rates, but decided that a constant rate would bracket the upper boundary of any stochastic estimate. Using the mean log growth rate and variance in mean log growth rate from this "population," we then compared the "population's" risk of quasi-extinction (probability of threshold), mean time to threshold, median time to threshold, and modal time to threshold with those of the population that had not experienced these mortalities. All of these scenarios used a quasi-extinction threshold of 50 adult, female loggerhead turtles.

We assumed that the other nesting aggregations in Japan were roughly equivalent in size to Yakushima and that, by assigning the mortalities to the total number of adult females, we approximate the consequences of randomly distributing the mortalities among the various nesting aggregations (since we don't have counts for the other nesting aggregations, this seemed a reasonable approximation of the risks). Because the proposed fisheries are likely to kill oceanic juveniles, we first converted the mortalities estimates from the fishery into their adult equivalents by multiplying by 0.8 and 0.7425 (the survival values for this stage from Heppell et al. 2003).

Using this general approach and the procedures described by Dennis et al. (1991) and Morris and Doak (2002), we assessed different scenarios using the following annual mortalities, which were taken from Table V-7: 5 (to assess the risks the mortalities would pose to the Yakushima nesting aggregations), 29 (to assess the consequences of killing 36 oceanic juveniles), 50 (to assess the consequences of killing 63 oceanic juveniles), and 70 (to assess the consequences of killing 88 oceanic juveniles).

Table V-7. Mortality estimates used in assessment scenarios for loggerhead turtles killed in the HMS fisheries

Nesting Aggregation	Estimates Using Pacific Rates		Estimates Using NED Rates	
	Lower Range	Upper Range	Lower Range	Upper Range
Yakushima Island	2.0	5.0	2.0	3.0
Other beaches	36.0	88.0	41.0	63.0

Five additional mortalities produced a very small, albeit measurable increase in the extinction risk. The results of these scenarios are presented in Table V-8. However, given limitations in the information available, the procedures we used had a limited ability to detect those effects and, at best, would underestimate any effects they detected. Given these limitations, we assume that our any differences we could detect were much larger than our analyses would suggest and that the risks to these turtles is higher.

Table V-8. Results of scenarios used to assess the potential effects of loggerhead turtles mortalities associated with the HMS fisheries on loggerhead nesting aggregations in the North Pacific. Scenarios are described in the text.

Measure	Scenario				
	No Mortality	+5	+36	+63	+88
Mean time to 50 (in years)	112	106	83	66	59
Modal time to 50 (in years)	32	32	29	26	24
Median time to 50 (in years)	75	73	59	51	45
Probability of reaching 50 in:					
25 years (probability %)	9	10	14	18	22
50 years (probability %)	33	34	42	49	56

100 years (probability %)	63	65	73	80	85

The results of analyses strongly suggest that the mortalities associated with the proposed fisheries can be expected to appreciably increase the extinction risks — or reduce the probability of persistence — of the Japanese nesting aggregations of loggerhead sea turtles. Mortalities like those expected from the proposed fisheries would be expected to reduce the persistence of loggerheads in Japan by between 20 and 40 percent (reductions in median time to 50 adult, females). Although these analyses relied on data on the number of adult, female loggerhead turtles, clearly this species cannot survive for long without adult females in the population.

These quantitative analyses support the conclusions we reached qualitatively: the mortalities associated with the proposed HMS fisheries would be expected to increase the extinction risks of nesting aggregations of loggerhead sea turtles. Increasing the extinction risk of these nesting aggregations would increase this species' extinction risk in the Pacific Ocean. Similarly, increasing the extinction risk of loggerhead turtles in the Pacific Ocean would appreciably reduce the loggerhead turtle's likelihood of surviving and recovering in the wild (that is, it would increase the loggerhead turtle's likelihood of global extinction).

Conclusions for the proposed HMS FMP. The nesting aggregations of loggerhead turtles in Japan represent one of the largest aggregations of this species in the Pacific Ocean. Given the relationship between the risk of local extinctions and regional persistence, as demonstrated by studies of metapopulation dynamics (for example, see Gotelli 2001), increasing the extinction risks of the loggerhead nesting aggregations in Japan would increase their extinction risk in the Pacific Ocean. Similarly, increasing the extinction risk of loggerhead turtles in the Pacific Ocean would appreciably reduce the loggerhead turtle's likelihood of surviving and recovering in the wild (that is, it would increase the loggerhead turtle's likelihood of global extinction). Therefore, the proposed HMS FMP is likely to jeopardize the continued existence of loggerhead sea turtles.

Assessment of the proposed rule prohibiting shallow longline sets. Implementation of the proposed ESA regulation to prohibit shallow longline sets is expected to preclude the anticipated captures and mortalities of loggerhead sea turtles in the west coast-based longline fishery. Therefore, we would expect that the remaining interactions with the CA/OR drift gillnet fishery would capture up to 5 loggerhead turtles, and kill two of them, annually. These estimates are well below the levels of mortality expected to result in appreciable changes in the likelihood of survival and recovery of the species. The CA/OR drift gillnet fishery has taken (observed) one loggerhead turtle since the implementation of protective measures approximately 3 years ago.

Summary of risks posed from the proposed actions taken together. The combined effects of the actions under consultation are not expected to reduce appreciably the likelihood of survival and recovery of loggerhead sea turtles.

4. Olive Ridley Turtle

Assuming that patterns observed in the past represent future patterns, olive ridley sea turtles will be exposed to the longline and drift gillnet fisheries. Between 5 and 15 olive ridley turtles are expected to be captured by the HMS longline and drift gillnet fisheries each year; 4 of these turtles are expected to be taken in the drift gillnet fishery only during El Nino events or when oceanographic conditions otherwise attract olive ridley sea turtles to the fishing grounds. Of these 5 to 15 turtles, 1 to 4 are expected to die as a result of that exposure. Genetic analyses indicate that the Hawaii-based longline fishery catches olive ridley turtles from nesting aggregations in the eastern and western Pacific Ocean and the Indian Ocean (26 % were from the Indian Ocean or western Pacific Ocean and 74% were from the eastern Pacific). Length information collected by observers indicates that the fishery interacts with sub-adult and adult olive ridley turtles. Three olive ridley turtles taken in the HMS fisheries were determined to be of eastern Pacific origin (P. Dutton, NOAA Fisheries, personal communication, August 2003). Therefore, 1 to 3 of the killed olive ridley turtles would be endangered turtles that have migrated from the eastern Tropical Pacific while the remaining olive ridley turtles killed in an interaction with longline gear would have originated in the western Pacific or Indian Ocean.

We do not believe these mortalities will appreciably reduce the olive ridley sea turtles' likelihood of surviving and recovering in the wild, because of the status and trend of olive ridley turtle populations in the Pacific basin. Historically, an estimated 10 million olive ridleys inhabited the waters in the eastern Pacific off Mexico (Cliffton, *et al*., 1982 *in* NOAA Fisheries and USFWS, 1998d). However, human-induced mortality led to declines in this population. Beginning in the 1960s, and lasting over the next 15 years, several million adult olive ridleys were harvested by Mexico for commercial trade with Europe and Japan. (NOAA Fisheries and USFWS, 1998d). Although olive ridley meat is palatable, it was not widely sought after; its eggs, however, are considered a delicacy. Fisheries for olive ridley turtles were also established in Ecuador during the 1960s and 1970s to supply Europe with leather (Green and Ortiz-Crespo, 1982).

In the eastern Pacific, nesting occurs all along the Mexico and Central American coast, with large nesting aggregations occurring at a few select beaches located in Mexico and Costa Rica. The largest known *arribadas* in the eastern Pacific are off the coast of Costa Rica (about 475,000 to 650,000 females estimated nesting annually) and in southern Mexico (about 800,000 or more nests per year at La Escobilla, in Oaxaca; Millán, 2000). The greatest single cause of olive ridley egg loss comes from the nesting activity of conspecifics on *arribada* beaches, where nesting turtles destroy eggs by inadvertently digging up previously laid nests or causing them to become contaminated by bacteria and other pathogens from rotting nests nearby.

The nationwide ban on commercial harvest of sea turtles in Mexico, enacted in 1990, appears to have improved the situation for the olive ridley. Surveys of important olive ridley nesting beaches in Mexico indicate increasing numbers of nesting females in recent years (Marquez, *et al*., 1995; Arenas, *et al*., 2000). Annual nesting at the principal beach, Escobilla Beach, Oaxaca, Mexico, averaged 138,000 nests prior to the ban, and since the ban on harvest in 1990, annual nesting has increased to an average of 525,000 nests (Salazar, *et al*., in press).

Olive ridleys are not as well documented in the western Pacific as in the eastern Pacific, nor do they appear to be recovering as well (with the exception of Orissa, India in recent years). There are a few sightings of olive ridleys from Japan, but no report of egg-laying. Nesting information from Thailand indicates a marked decline in olive ridley numbers primarily due to egg poaching, harvest and subsequent consumption or trade of adults or their parts (i.e. carapace), indirect capture in fishing gear, and loss of nesting beaches through development (Aureggi, *et al*., 1999). Extensive hunting and egg collection, in addition to rapid rural and urban development, have reduced nesting activities in Indonesia as well.

Olive ridley nesting is known to occur on the eastern and western coasts of Malaysia; however, nesting has declined rapidly in the past decade. The highest density of nesting was reported to be in Terengganu, Malaysia, and at one time yielded 240,000 eggs (2,400 nests, with approximately 100 eggs per nest) (Siow and Moll, 1982, *in* Eckert, 1993)), while only 187 nests were reported from the area in 1990 (Eckert, 1993).

In contrast, olive ridleys are the most common species found along the east coast of India, migrating every winter to nest en-masse at three major rookeries in the state of Orissa, Gahirmatha, Robert Island, and Rushikulya (*in* Pandav and Choudhury, 1999). The Gahirmatha rookery, located along the northern coast of Orissa, hosts the largest known nesting concentration of olive ridleys. Unfortunately, uncontrolled mechanized fishing in areas of high sea turtle concentration, primarily illegally operated trawl fisheries, has resulted in large scale mortality of adults during the last two decades. Fishing in coastal waters off Gahirmatha was restricted in 1993 and completely banned in 1997 with the formation of a marine sanctuary around the rookery. Threats to these sea turtles also include artificial illumination and unsuitable beach conditions, including reduction in beach width due to erosion (Pandav and Choudhury, 1999). According to Pandav and Choudhury (1999), the number of nesting females at Gahirmatha has declined in recent years, although after three years of low nestings, the 1998-99 season showed an increasing trend, and the 1999-2000 season had the largest recorded number of olive ridleys nesting in 15 years when over 700,000 olive ridleys nested at Nasi islands and Babubali island, on the Gahirmatha coast.

Given initial population sizes and increases in the Mexican and Costa Rican populations in recent years, the mortalities associated with the HMS fisheries are not likely to halt or reverse the increasing trend of those populations. Removing adult or sub-adult turtles from the eastern Pacific population could slow the recovery of the population that is occurring, although it is not clear if that reduction would be measurable given the size of the nesting population.

Population trends in the western Pacific are more difficult to discern, although it is clear that there are still large populations of olive ridleys nesting in India. Killing adult and sub-adult turtles in the western Pacific population could have more serious consequences, since this population continues to be affected by ongoing factors such as incidental take in fisheries, the harvest of eggs on nesting beaches, and inundation and erosion of beaches. By removing reproductive adults and pre-reproductive sub-adults from this declining population, the HMS fisheries could adversely affect this population's persistence, although it is unknown how much, or to what degree, this might impact the population's survival in light of the other factors currently affecting this population.

Conclusions for the proposed HMS FMP. The major populations of olive ridley turtles in the Pacific Ocean appear to be increasing, despite some residual, adverse effects of fishery-related mortalities and harvest of adults and eggs. Because of the population size, number of reproductive females, and the rates at which sub-adults are probably recruiting into the adult population, we believe nesting aggregations of this species can withstand the mortalities and reduced reproductive rates associated with the HMS fisheries without appreciable reductions in the olive ridley turtle's likelihood of the surviving and recovering in the wild.[22]

Assessment of the proposed rule prohibiting shallow longline sets. Implementation of the proposed ESA regulation to prohibit shallow longline sets is expected to preclude anticipated captures and mortalities of olive ridley sea turtles in the west coast-based longline fishery. Therefore, we would expect that the remaining interactions in the CA/OR drift gillnet fishery would periodically capture up to 4 olive ridley turtles, and kill 1 of them, annually. These estimates are well below the levels of mortality expected to result in appreciable changes in the likelihood of survival and recovery of the species. This fishery has not taken any olive ridley turtles since the implementation of protective measures approximately 3 years ago.

Summary of risks posed from the proposed actions taken together. The combined effects of the actions under consultation are not expected to reduce appreciably the likelihood of survival and recovery of the olive ridley sea turtles as they are listed globally or endangered green sea turtles as they are listed for nesting aggregations in the eastern Pacific.

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[22] Olive ridley turtles on the Pacific coast of Mexico are listed separately as endangered species, rather than the threatened status assigned to the remainder of their global populations. Under normal circumstances, we would analyze the effects of the proposed fisheries on the endangered populations separately from their threatened counterparts; however, using the information available, we cannot distinguish the effects of the fisheries on the different populations (because our data on interactions between the fisheries and these turtles cannot distinguish between the endangered turtles and the threatened turtles of these turtles). As a result, our analyses group the endangered populations and the threatened populations and treat them both as endangered to make certain that we afford the endangered turtles the additional protection warranted by their classification.

VI. CUMULATIVE EFFECTS

Cumulative effects include the effects of future State, tribal, local, or private actions that are reasonably certain to occur in the action area considered in this Opinion. 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.

Most of the fisheries described as occurring within the action area (Section IV. *Status of the Species and Environmental Baseline*), are expected to continue as described into the foreseeable future. Therefore, NOAA Fisheries is not aware of any proposed or anticipated changes in most of these fisheries that would substantially change the impacts each fishery has on the sea turtles covered by this Opinion.

In addition to fisheries, NOAA Fisheries is not aware of any proposed or anticipated changes in other human-related actions (e.g. poaching, habitat degradation) or natural conditions (e.g. over-abundance of land or sea predators, changes in oceanic conditions, etc.) that would substantially change the impacts that each threat has on the sea turtles or marine mammals covered by this Opinion. Therefore, NOAA Fisheries expects that the levels of take of sea turtles described for each of the fisheries and non-fisheries will continue at similar levels into the foreseeable future.

VII. CONCLUSION

After reviewing the available scientific and commercial data, current status of green turtles, the environmental baseline for the action area, the effects of the proposed action and the cumulative effects, it is NOAA-Fisheries' biological opinion that the continued authorization of HMS fisheries under the HMS FMP, operation of vessels in these fisheries under their HSFCA permits, and implementation of a regulation that would prohibit shallow longline sets east of the 150 W longitude are not likely to jeopardize the continued existence of endangered green sea turtles in the eastern Pacific, or threatened green sea turtles globally.

After reviewing the available scientific and commercial data, current status of leatherback turtles, the environmental baseline for the action area, the effects of the proposed action and the cumulative effects, it is NOAA-Fisheries' biological opinion that the continued authorization of HMS fisheries under the HMS FMP, operation of vessels in these fisheries under their HSFCA permits, and implementation of a regulation that would prohibit shallow longline sets east of the 150 W longitude are not likely to jeopardize the continued existence of endangered leatherback sea turtles.

After reviewing the available scientific and commercial data, current status of loggerhead turtles, the environmental baseline for the action area, the effects of the proposed action and the cumulative effects, it is NOAA-Fisheries' biological opinion that the continued authorization of HMS fisheries under the HMS FMP, operation of vessels in these fisheries under their HSFCA permits, and implementation of a regulation that would prohibit shallow longline sets east of the 150 W longitude are not likely to jeopardize the continued existence of threatened loggerhead sea turtles.

After reviewing the available scientific and commercial data, current status of olive ridley turtles, the environmental baseline for the action area, the effects of the proposed action and the cumulative effects, it is NOAA-Fisheries' biological opinion that the continued authorization of HMS fisheries under the HMS FMP, operation of vessels in these fisheries under their HSFCA permits, and implementation of a regulation that would prohibit shallow longline sets east of the 150 W longitude are not likely to jeopardize the continued existence of endangered olive ridley sea turtles in the eastern Pacific, or threatened olive ridley sea turtles globally.

VIII. INCIDENTAL TAKE STATEMENT

Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct. NOAA Fisheries further defines "harm" as an act which actually kills or injures fish or wildlife. Such an act may include significant habitat modification or degradation where it actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns including breeding, spawning, rearing, migrating, feeding or sheltering. 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 7(o)(2), taking that is incidental to and not intended as part of the proposed action is not considered to be prohibited taking under the Act provided that such taking is in compliance with this Incidental Take Statement.

The measures described below are nondiscretionary, and must be undertaken by NOAA Fisheries for the exemption in section 7(o)(2) to apply. NOAA Fisheries has a continuing duty to regulate the activity covered by this incidental take statement. If NOAA Fisheries (1) 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, NOAA Fisheries must monitor the progress of the action and its impact on the species as specified in the incidental take statement. (50 CFR §402.14(I)(3))

Section 7(b)(4) of the ESA requires that when a proposed agency action is found to be consistent with section 7(a)(2) of the ESA and the proposed action may incidentally take individuals of listed species, NOAA Fisheries will issue a statement that specifies the impact of any incidental taking of endangered or threatened species. It also states that reasonable and prudent measures, and terms and conditions to implement the measures, be provided that are necessary to minimize such impacts. Only incidental take resulting from the agency action and any specified reasonable and prudent measures and terms and conditions identified in the incidental take statement are exempt from the taking prohibition of section 9(a), pursuant to section 7(o) of the ESA.

A marine mammal species or population stock which is listed as threatened or endangered under the ESA is, by definition, also considered depleted under the MMPA. The ESA allows takings of threatened and endangered marine mammals only if authorized by section 101(a)(5) of the MMPA. Until the proposed action receives authorization for the incidental taking of marine mammals under section 101(a)(5)(E) of the MMPA, the incidental takes of marine mammals described below are not exempt from the taking prohibition of section 9(a), pursuant to section 7(o) of the ESA.

A. Amount or Extent of Take

Mortality and entanglement rates of marine mammals and sea turtles have been calculated based on past observed interaction rates (per set) multiplied by the number of sets expected under current predictions of future fishing effort. Mortality and entanglement rates vary from year to year, with some species observed captured and killed every year, and others observed captured and killed only when certain environmental conditions make interactions likely.

NOAA Fisheries has developed this incidental take statement (Table VIII -1) based on the premise that both proposed actions (HMS FMP and ESA rule prohibiting shallow longline sets) will be implemented. As a result, takes are only expected in the CA/OR drift gillnet fishery, with some rare captures occurring in the albacore surface hook and line fishery. Takes are not anticipated at this time for the west coast-based longline fishery as NOAA Fisheries has no information to suggest that longline fishing vessels will continue to operate after implementation of the proposed actions. In the event that longline vessels continue to operate by switching gear methods to deep sets, terms and conditions to this incidental take statement would require the placement of observers on all vessels in order to monitor the effect to listed species. This consultation would also likely be reinitiated in order to assess potential effects and incidental take levels.

Except where noted, the numbers below are annual estimates. For species like green, olive ridley, or loggerhead sea turtles, the number is applied over a calendar year when the environmental conditions conducive to expected takes in the CA/OR drift gillnet fishery have been present.

Table VIII-1. Anticipated incidental takes of listed species in the HMS fisheries.

Species	Estimated Entanglement	Estimated Mortality	Conditions Resulting in Take
Fin whale	4 in 3 years	2 in 3 years	
Humpback whale	4 in 3 years	0	
Sperm whale	4 in 3 years	2 in 3 years	
Green turtle	4	1	SSTs in fishing area similar to Nov-99
Leatherback turtle	3	2	
Loggerhead turtle	5	2	Only in “El Niño” years[1]
Olive ridley turtle	4	1	SSTs in fishing area similar to Nov-99

[1] final rule on December 16, 2003 (68 FR 69962) to prohibit fishing with drift gillnets in the California/Oregon (CA/OR) thresher shark/swordfish drift gillnet fishery in U.S. waters off southern California in waters east of the 120EW., for the months of June, July, and August

In addition, the albacore surface hook and line fishery may interact with sea turtles. Past observations were of either loggerhead or green sea turtles, but these were very infrequent (two observations in approximately 1,500 observed days of effort). Because gear is immediately retrieved in this fishery, turtles accidentally hooked or entangled by albacore hook and line gear are not expected to die.

Based on the method NOAA Fisheries currently uses to estimate incidental entanglement in the drift gillnet fishery, one observed entanglement usually results in an estimate of five entanglements. Therefore, if more than one entanglement is observed in a year (or over the course of three years for marine mammals), NOAA Fisheries is likely to determine that incidental take has been exceeded. In the past, NOAA Fisheries has also applied the same calculation to estimated mortalities – if one individual of a species is entangled and killed, then 5 individuals have been entangled and killed. However, this likely overestimates the mortal take because, based on past observations, most of the above species have an expected survival rate per entanglement. For example, hard shelled turtles are estimated to have a 68 percent survival rate. Of 4 green turtles, 3 are expected to survive their entanglement and 1 is expected to die. However, based on the random sampling of the species incidentally caught in the fishery by observed vessels, the one mortality observed is likely one of the instances of expected mortal incidental take and not grounds for reinitiation of the consultation. This is applicable to all species described in this Opinion except for the humpback whale. If a mortality of a humpback whale is observed, this would require reinitiation of this consultation because no humpback mortalities are expected.

B. Effect of the Take

In the accompanying biological opinion, NOAA Fisheries determined that this level of anticipated take is not likely to result in jeopardy to the fin whale, humpback whale, sperm whale, green turtle, leatherback turtle, loggerhead turtle, or olive ridley turtle.

C. Reasonable and Prudent Measures

NOAA Fisheries believes the following reasonable and prudent measures, as implemented by the terms and conditions, are necessary and appropriate to minimize impacts to minimize impacts to sea turtles from the fisheries considered in this Opinion. NOAA Fisheries has determined that the existing requirements of the PCTRP are adequate and appropriate to minimize the impact of the take on listed marine mammals and therefore no additional measures are necessary. The measures described below are non-discretionary, and must be undertaken by NOAA Fisheries for the exemption in section 7(o)(2) to apply. If NOAA Fisheries fails to adhere to the terms and conditions of the incidental take statement, the protective coverage of section 7(o)(2) may lapse. Thus, the following reasonable and prudent measures must be implemented to allow activities by the HMS fisheries to continue.

4. HMS fishery vessel operators and observers shall be educated on sea turtle biology and on methods that will reduce injury or mortality during fishing operations.

5. Live captured sea turtles shall be released uninjured from gear in a manner that minimizes the likelihood of further gear entanglement or entrapment.

6. NOAA Fisheries shall continue to collect data on capture, injury and mortality of sea turtles in addition to life history information.

4. Comatose and lethargic sea turtles shall be retained on board, handled, resuscitated, and released according to the procedures outlined in the 50 CFR 223.206(d)(1).

5. Dead sea turtles shall be disposed of at sea unless an observer requests retention of the carcass for sea turtle research.

D. Terms and Conditions

In order to be exempt from the prohibitions of Section 9 of the ESA, NOAA Fisheries must comply or ensure compliance with the following terms and conditions, which implement the reasonable and prudent measures described above. These terms and conditions are non-discretionary.

1. The following terms and conditions implement reasonable and prudent measure No. 1.

    1A. NOAA Fisheries will continue to provide skipper education workshops with a module on sea turtle resuscitation requirements, as outlined in 50 CFR §223.206(d)(1). These workshops shall be provided to skippers in the CA/OR drift gillnet and longline fisheries (for any vessels that switch to deep set longlining).

    1B. NOAA Fisheries will also include in skipper education workshops a module of information on sea turtle biology and ways to avoid and minimize sea turtle impacts.

    1C. NOAA Fisheries will encourage HMS permitholders to suggest additional strategies or techniques that might minimize impacts of fishing gear or practices on sea turtles.

    1D. NOAA Fisheries will include sea turtle resuscitation techniques and sea turtle biology information during observer training.

    1E. For these workshops, and as a reference that shall be provided to skippers of small purse seine vessels, albacore surface hook and line, and charter or private recreational fishing vessels, NOAA Fisheries shall produce a pamphlet describing sea turtle species, biology, and recommended techniques for releasing and resuscitating incidentally captured sea turtles.

2. The following terms and conditions implement reasonable and prudent measure No. 2.

    2A. As soon as practicable upon capture, vessel operators or observers shall disengage any hooked or entangled sea turtles with the least harm possible to the sea turtles. If a hook cannot be removed, the line should be cut as close to the hook as possible.

    2B. Sea turtles must not be dropped on to the deck or run through a power block.

    2C. Sea turtles seen within a purse seine net must be released over the corkline by a speedboat driver, swimmer, or raft operator, if possible. Turtles should not be lifted out of the water in a purse seine net that is being rolled aboard a vessel.

3. The following terms and conditions implement reasonable and prudent measure No. 3.

3A. NOAA Fisheries shall continue to maintain an observer program to collect data on the incidental take of marine mammals, sea turtles, and other protected species. Quarterly and annual reports summarizing protected species bycatch data collected for HMS fisheries shall be prepared and disseminated in a timely fashion to the Southwest Region-Protected Resources Division and to the Sea Turtle Coordinator in Silver Spring, Maryland.

Information collected shall include, at a minimum, the incidental capture, injury, and mortality of sea turtles by species, gear and set information in which each interaction occurred, and life history information.

3B. NOAA Fisheries shall continue to collect life history information on sea turtles, such as species identification, measurements, condition, skin biopsy samples, and the presence or absence of tags. NOAA Fisheries observers shall directly measure or visually estimate tail length on all sea turtles captured by the HMS fisheries.

3C. NOAA Fisheries shall place an observer aboard any longline vessel subject to the HMS FMP which may make deep sets in order to monitor the effects to listed species.

3D. NOAA Fisheries collected data and other available information shall be evaluated on an annual basis to determine whether estimated annual incidental injuries or mortalities of sea turtles has exceeded allowable removal levels. The report will be sent to the Sea Turtle Coordinator in Silver Spring, Maryland.

4. The following terms and conditions implement reasonable and prudent measure No. 4.

4A. HMS vessel operators shall bring comatose sea turtles aboard, if feasible, and perform resuscitation techniques according to the procedures described at 50 CFR §223.206(d)(1).

4B. If an observer is aboard the vessel, the observer shall perform resuscitation techniques on comatose sea turtles brought aboard the vessel.

5. The following term and condition implements reasonable and prudent measure No. 5.

5A. Dead sea turtles may not be consumed, sold, landed, offloaded, transhipped or kept below deck, but must be returned to the ocean after identification unless the observer requests the turtle to be kept for further study.

IX. CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the Act directs Federal agencies to utilize their authorities to further the purposes of the Act by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or 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. NOAA Fisheries should explore the possibility of developing or modifying existing gear to reduce the likelihood of gear interactions.

2. NOAA Fisheries should explore the possibility of developing or modifying existing technologies, such as sonar, to detect and alert fishers if sea turtles or marine mammals become entangled in their gear.

3. NOAA Fisheries should explore the feasibility of developing a system for fishermen to collect life history information on sea turtles.

X. REINITIATION NOTICE

This concludes formal consultation on the action outlined above. 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 the incidental take 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 causes an effect to the listed species or critical habitat not considered in this opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action. In instances where the amount or extent of incidental take is exceeded, the Sustainable Fisheries Division, Southwest Region, NOAA Fisheries, should immediately request initiation of formal consultation.

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