APPENDIX V

NMFS' February 23, 2004 *Biological Opinion on the Authorization of Pelagic Fisheries under the Fishery Management Plan for the Pelagic Fisheries of the Western Pacific Region*

ENDANGERED SPECIES ACT SECTION 7 CONSULTATION
BIOLOGICAL OPINION

Action Agency:	National Marine Fisheries Service, Pacific Islands Region, Sustainable Fisheries Division
Activity:	Proposed Regulatory Amendments to the Fisheries Management Plan for the Pelagic Fisheries of the Western Pacific Region.
Consulting Agency:	National Marine Fisheries Service, Office of Protected Resources
Approved By:	Laurie K. Allen
Date issued:	FEB 23 2004

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, 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 that may be affected. For the actions described in this document, the action agency is NOAA-Fisheries' Pacific Islands Region- Sustainable Fisheries Division. The consulting agency is NOAA-Fisheries' Office of Protected Resources.

This biological opinion represents NOAA-Fisheries' assessment of the effects of the Fishery Management Plan for the Pelagic Fisheries of the Western Pacific Region (Pelagics FMP) and associated fisheries threatened and endangered species. Specifically, the effects of the fisheries on endangered humpback whales (*Megaptera novaeangliae*), endangered sperm whales (*Physeter macrocephalus*), endangered Hawaiian monk seals (*Monachus schauinslandi*), endangered fin whales (*Balaenoptera physalus*), sei whales (*B. borealis*), endangered right whales (*Eubalaena japonica*), threatened and endangered green turtles (*Chelonia mydas*), endangered hawksbill turtles (*Eretmochelys imbricata*), endangered leatherback turtles (*Dermochelys coriacea*), threatened loggerhead turtles (*Caretta caretta*), and threatened and endangered olive ridley turtles (*Lepidochelys olivacea*), in accordance with section 7 of the ESA.

This Opinion considers information provided in NOAA-Fisheries' January 14, 2004, biological assessment, the January 8, 2004, regulatory amendment to the Fishery Management Plan for the Pelagics Fisheries of the Western Pacific Region, the draft supplemental Environmental Impact Statement on the fisheries, recovery plans for the humpback whale and Hawaiian monk seal, the most current marine mammal stock assessment reports, sea turtle recovery plans, past and current research and population dynamics modeling efforts, observer and logbook data on fishery effort and protected species interactions within the Hawaii-based longline fishery, and biological opinions for this and other relevant fisheries.

1.0 CONSULTATION HISTORY

Consultation histories for earlier consultations on the Fisheries Management Plan for the Pelagic Fisheries of the Western Pacific Region (Pelagics FMP) can be found in the November 3, 1998, biological opinion on the reinitiated consultation for the Pelagics FMP Hawaii North Central Pacific Longline Fishery (NOAA-Fisheries, 1998a). That opinion found that the proposed action was not likely to jeopardize the continued existence of listed sea turtles or Hawaiian monk seals, and established anticipated incidental take levels for sea turtles captured by the Hawaii-based longline fishery. The opinion also required continuation of the observer program for the fishery and required handling procedures for incidentally captured sea turtles and review of the circumstances surrounding the observed capture of any leatherback turtle.

Since then, the Hawaii-based longline fisheries have been the subject of several court-orders and was operating under a restricted fishing regime to protect listed sea turtles when NOAA-Fisheries completed a new consultation on the Pelagics FMP on March 29, 2001. In that opinion, NOAA-Fisheries determined that the proposed action was not likely to jeopardize the continued existence of the listed marine mammals or olive ridley sea turtles or destroy or adversely modify designated critical habitat affected by the fisheries. NOAA-Fisheries did determine, however, that the Pelagics FMP fisheries were likely to jeopardize the continued existence of green, leatherback, and loggerhead turtles. The March 29, 2001, opinion included a Reasonable and Prudent Alternative (RPA) designed to avoid the likelihood of jeopardy to these species and an Incidental Take Statement with terms and conditions designed to minimize the impact of any incidental take of all four species of turtles that would occur as a result of implementation of the RPA.

Subsequent to completion of the March 29, 2001, opinion, the Pelagics fisheries have been operating pursuant to the requirements of regulations issued to implement the RPA and to comply with the agency's ESA obligations to protect sea turtles (66 FR 31561, June 12, 2002). However, on December 12, 2001, NOAA-Fisheries reinitiated consultation on the Pelagics Fisheries to account for new information which may improve NOAA-Fisheries' ability to quantify and evaluate the effects of the United States' pelagic fisheries under the FMP and the reasonable and prudent alternative in the March 29, 2001, opinion on listed sea turtle populations. The agency issued the biological opinion resulting from that consultation on November 15, 2002.

On December 11, 2003, the Acting Regional Administrator of NOAA-Fisheries' Pacific Islands Regional Office sent a memorandum to the Director of NOAA-Fisheries' Office of Protected Resources that asked the OPR to reinitiate formal consultation on the Pelagics FMP, as amended by the Council in response to the court's vacating the sea turtle protection regulations.

On January 14, 2004, the Acting Regional Administrator of NOAA-Fisheries' Pacific Islands Regional Office delivered a copy of a January 14, 2004, biological assessment on the proposed fisheries, which was prepared by the Western Pacific Regional Fisheries Management Council on behalf of the Hawaii Longline Association, to the Director of NOAA-Fisheries' Office of Protected Resources.

On January 14, 2004, the Acting Regional Administrator of NOAA-Fisheries' Pacific Islands Regional Office delivered a copy of a January 8, 2004, regulatory amendment to the Fishery Management Plan for the Pelagics Fisheries of the Western Pacific Region, which was prepared by the Western Pacific Regional Fisheries Management Council, to the Director of NOAA-Fisheries' Office of Protected Resources. This proposed regulatory amendment included a draft supplemental Environmental Impact Statement.

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 comprehensive overview summary of the approach the office is using to assess the effects of the pelagic fisheries of the western Pacific region on threatened and endangered species.

On February 2, 2004, NOAA-Fisheries' Office of Protected Resources provided a partial draft biological opinion (everything except a draft incidental take statement, conservation recommendations, literature cited section, and appendices) to NOAA-Fisheries' Pacific Islands Regional Office, which transmitted the draft to the Hawaii Longline Association (as applicants) and representatives of the Western Pacific Regional Fisheries Management Council.

On February 4, 2004, representatives of NOAA-Fisheries' Office of Protected Resources, Southwest Regional Office (Protected Resources and Sustainable Fisheries Programs), Pacific Islands Regional Office, and Pacific Islands Area Science Center, the Hawaii Longline Association, and Council held a video- and teleconference to discuss NOAA-Fisheries' February 2, 2004, draft biological opinion. At the meeting, the Office of Protected Resources received initial comments on the draft opinion.

On February 6, 2004, NOAA-Fisheries' Office of Protected Resources provided a draft incidental take statement, conservation recommendations, literature cited section, and appendices to NOAA-Fisheries' Pacific Islands Regional Office, which transmitted the draft to the Hawaii Longline Association (as applicants) and representatives of the Western Pacific Regional Fisheries Management Council (including the sea turtle working group).

During the week of February 16-20, 2004, NOAA-Fisheries' Office of Protected Resources received written comments on the entire draft biological opinion from NOAA-Fisheries' Pacific

Islands Regional Office, Stoel-Rives LLP on behalf of the Hawaii Longline Association (as applicants) and representatives of the Western Pacific Regional Fisheries Management Council.

2.0 DESCRIPTION OF THE ACTION

The National Marine Fisheries Service's Pacific Islands Regional Office, Sustainable Fisheries Division proposes to approve regulations for fisheries managed under the Fisheries Management Plan for pelagic fisheries in the western Pacific Region (Pelagics FMP) and to manage those fisheries pursuant to those regulations. The purpose of fishery management plans, including the Pelagics FMP, has been established by the Magnuson-Stevens Fishery Conservation and Management Act (16 U.S.C. 1801 *et seq.*; MSA).

The U.S. pelagic fisheries in the central and western Pacific region are managed under the Pelagics FMP, as amended[1]. The Pelagics FMP and its amendments are developed by the Western Pacific Regional Fisheries Management Council (Council) under the authority of the MSA. The stated purpose of the Pelagics FMP is to maximize the net benefits of the fisheries to the western Pacific region and the Nation. Background information on federal fisheries policy and management under the MSA, the fishery management plan development process, and the Pelagics FMP is described in the March 2001 FEIS (Section 1.3, pages 11 - 34).

The current management regime under the Pelagics FMP primarily regulates the domestic pelagic longline fisheries, although certain permit, reporting, and sea turtle mitigation measures apply to non-longline pelagic fisheries in the region, such as the domestic troll, handline, and pole-and-line fisheries. NMFS has no specific regulations pursuant to the Pelagics FMP for the domestic tuna purse seine fishery operating in the western Pacific. If problems or issues arise in the future in this fishery, the Pelagics FMP can be adjusted as necessary.

As proposed, the fisheries managed under the Pelagics FMP would be prosecuted under the combination of the following existing and potential management measures:

1. All fishery management measures for the western Pacific pelagic fisheries in existence on the date of initiation of consultation, December 11, 2003, except those that will be eliminated as a result of the court orders of August 31 and October 6, 2003, in *Hawaii Longline Association* v. *NMFS* (D.D.C., Civ. No. 01-0765). These orders vacated, effective April 1, 2004, a set of regulations promulgated June 12, 2002, to mitigate the adverse effects of the western Pacific pelagic fisheries on sea turtles (67 FR 40232).

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[1] The U.S. tuna purse seine fishery is managed under the Treaty on Fisheries Between the Governments of Certain Pacific Island States and the Government of the United States of America (the South Pacific Tuna Treaty). Although they occur in the action area and are subject to management under the MSA, for the most part this fishery is not regulated under the Pelagics FMP (one regulation prohibits large vessels, including U.S. tuna purse seiners, from fishing in waters within approximately 50 nm of the islands of American Samoa). This fishery will be subject to separate section 7 consultation on federal options to implement the Treaty and will not be evaluated as part of the effects of this action.

2. The regulatory management measures in the proposed regulatory amendment to the Pelagics FMP, "Management Measures to Implement New Technologies for the Western Pacific Pelagic Longline Fisheries," dated January 8, 2004 (hereafter called "new technologies regulatory amendment"). The measures proposed in this regulatory amendment, together with the effects of the court orders cited above, are in a proposed rule published January 28, 2004 (69 FR 4098).

3. The management measures proposed in Pelagics FMP Amendment 11, which would establish a limited access program for longline fishing in the EEZ around American Samoa.

4. The following existing management measures, which will be eliminated by court order on April 1, 2004, but which are expected to be re-implemented through separate rule-making within the next year (hereafter called "future measures"):

   (a) A requirement that operators of vessels registered for use under longline general permits annually attend a NMFS-conducted protected species workshop and carry on board a valid certificate of completion of the workshop (see 50 CFR 660.34);

   (b) a requirement that owners and operators of vessels registered for use under longline general permits that have a freeboard of more than three feet carry line clippers and dip nets meeting certain minimum design standards and wire or bolt cutters capable of cutting through the vessel's hooks and use these items in specified manners to disengage sea turtles, and that certain turtle handling, resuscitation, and release methods be employed (see 50 CFR 660.32); and

   (c) a requirement that owners and operators of vessels registered for use under longline general permits that have a freeboard of three feet or less carry and use line clippers capable of cutting the fishing line or leader within about one foot of the eye of an imbedded hook and wire or bolt cutters capable of cutting through the vessel's hooks and use these items in specified manners to disengage sea turtles, and that certain turtle handling, resuscitation, and release methods be employed (see 50 CFR 660.32).

In summary, the management measures that constitute the action under consideration, along with their sources, are.

- Fishing for pelagic management unit species (PMUS) in the western Pacific EEZ with drift gillnets is prohibited *(52 FR 5987, March 23, 1987).*

- Fishing vessels that use longline gear to catch PMUS in the EEZ around American Samoa, Guam, Commonwealth of the Northern Mariana Islands, or the U.S. Pacific remote islands areas (PRIA), such as Palmyra and Johnston Atolls, Kingman Reef, Jarvis, Howland, Baker and Wake Islands, and vessels used to transport or land U.S.

longline-harvested PMUS shoreward of the outer boundary of these same EEZs, must be registered for use with longline general permits or Hawaii longline limited access permits, and must keep daily logbooks detailing species harvested, area of harvest, time of sets, and other information, including interactions with protected species. Also, longline gear must be marked with the official number of the permitted vessel that deploys the gear *(56 FR 24731, May 26, 1991).*

- Fishing vessels that use longline gear to catch PMUS in the EEZ around Hawaii, or are used to transport or land longline-harvested PMUS shoreward of the outer boundary of the EEZ around Hawaii, must keep daily logbooks detailing species harvested, area of harvest, time of sets, and other information, including interactions with protected species *(56 FR 24731, May 26, 1991).*

- Longline fishing for PMUS is prohibited in closed areas 50 nm around the center points of each of the Northwestern Hawaiian Islands, plus a 100 mile wide corridor connecting those circular closed areas that are non-contiguous (protected species zone) *(56 FR 52214, October 14, 1991).* In the main Hawaiian Islands longline fishing, except as exempted, is prohibited in areas approximately 75 nm around the islands of Kauai, Niihau, Kaula, and Oahu, and approximately 50 nm off the islands of Hawaii, Maui, Kahoolawe, Lanai, and Molokai. This prohibition is lessened from October 1 through January 30, when the longline closed areas decrease on the windward sides to approximately 25 nm off Hawaii, Maui, Kahoolawe, Lanai, Molokai, Kauai, Niihau, and Kaula, and approximately 50 nm off Oahu *(56 FR 28116, June 14, 1991).*

- Longline fishing is also prohibited in an area approximately 50 nm off Guam *(57 FR 7661, March 2, 1992).*

- Vessels registered for use under Hawaii longline limited access permits ("Hawaii-based longline vessels") must carry a NOAA-Fisheries' observer when directed to do so by NOAA-Fisheries *(58 FR 67699, December 22, 1993).*

- Fishing vessels that use longline gear to catch PMUS in the EEZ around Hawaii, or are used to transport or land longline-harvested PMUS shoreward of the outer boundary of the EEZ around Hawaii, must be less than 101 feet in length and be registered for use with one of 164 Hawaii longline limited access permits *(59 FR 26979, June 24, 1994).*

- As directed by NMFS, all vessels registered for use with Hawaii longline limited access permits (Hawaii longliner) must carry NMFS-owned "vessel monitoring system" transmitters *(59 FR 58789, November 15, 1994).*

- All Hawaii-based longline vessels and fishing vessels registered for use with longline general permits are required to employ sea turtle handling measures specified by NMFS, including mitigation gear, sea turtle resuscitation, and sea turtle release procedures, to maximize the survival of sea turtles that are accidentally taken by fishing gear *(65 FR 16346, March 28, 2000; future measures).*

- Domestic longline fishing vessels greater than 50 feet (length overall), except as exempted, are prohibited from fishing for PMUS within approximately 50 nm around the islands of American Samoa, including Tutuila, Manua, and Swains Islands, and Rose Atoll *(67 FR 4369, January 30, 2002).*

- Federal regulations that implemented the Shark Finning Prohibition Act prohibit any person under U.S. jurisdiction from engaging in shark finning, possessing shark fins harvested on board a U.S. fishing vessel without corresponding shark carcasses, or landing shark fins harvested without corresponding carcasses *(67 FR 6194, February 11, 2002).*

- Any domestic fishing vessel that employs troll or handline gear to catch PMUS in the EEZ around the U.S. Pacific remote islands areas, e.g., Palmyra and Johnston Atolls, Kingman Reef, Jarvis, Howland, Baker and Wake Islands, and Midway Atoll in the Main Hawaiian Islands, must be registered for use with a permit issued by NMFS and must also maintain daily logbooks detailing species harvested, area of harvest, fishing effort, and other information, including interactions with protected species *(67 FR 30346, May 6, 2002).*

- Hawaii-based longline vessels operating north of 23° N. must: when using traditional basket-style longline gear, ensure that the main longline is deployed slack to maximize its sink rate; when making deep sets using monofilament main longline, use a line-setting machine or line shooter and attach a weight of at least 45 gm to each branch line within 1 m of each hook; use thawed blue-dyed bait; and discharge offal strategically *(67 FR 34408, May 14, 2002).*

- The operator and crew of all Hawaii-based longline vessels that accidentally hook or entangle an endangered short-tailed albatross must employ specific handling procedures *(67 FR 34408, May 14, 2002).*

- Operators and owners of Hawaii-based longline vessels and operators of tered for use under longline general permits are required to attend annual protected species workshops conducted by NMFS that cover sea turtle and seabird conservation and mitigation techniques *(67 FR 34408, May 14, 2002; future measures).*

- There is an annual limit on the number of longline shallow-sets that may be collectively made north of the equator by Hawaii-based longline vessels, set at 2,120 shallow-sets per year, which is divided and distributed each calendar year in equal portions in the form of transferable single-set certificates to all holders of Hawaii longline limited access permits that respond positively to an annual solicitation of interest from NMFS. Shallow-setting means the deployment of longline gear with any float line less than 20 meters in length, with less than 15 branch lines between any two floats (except basket-style longline gear, the threshold for which is 10 branch lines between any two floats), with the use of light sticks, or resulting in the possession or landing of more than 10 swordfish at any time during a given trip. Hawaii-based longline vessels are required to have on board, and to submit to NMFS at the end of each trip, one valid shallow-set certificate for every shallow-set made north of the equator (*new technologies regulatory amendment*).

- Hawaii-based longline vessels, when making shallow-sets north of the equator, must use circle hooks sized 18/0 or larger with a 10-degree offset and only mackerel-type bait (*new technologies regulatory amendment*).
- There are annual limits on the numbers of interactions between leatherback and loggerhead sea turtles and Hawaii-based longline vessels while engaged in shallow-setting. The limit for each species is equal to the annual estimated incidental take for the species in the shallow-set component of the Hawaii-based fishery (either incidental captures or incidental deaths, whichever limit is reached first) as established in the prevailing biological opinion issued by NMFS pursuant to section 7 of the ESA. When either one of the turtle interaction limits is reached, as determined from estimates derived from vessel observer data, the shallow-set component of the Hawaii-based longline fishery is closed for the remainder of the calendar year, after giving 1 week advanced notice of such closure to all holders of Hawaii longline limited access permits (*new technologies regulatory amendment*).
- Operators of Hawaii-based longline vessels are required to notify the Regional Administrator in advance of every trip whether the trip will involve shallow-setting or deep-setting, and such vessels are required to make sets only of the type declared (*new technologies regulatory amendment*).
- Operators of Hawaii-based longline vessels are required to carry and use NMFS-approved de-hooking devices (*new technologies regulatory amendment*).
- Hawaii-based longline vessels, when making shallow-sets north of 23° N. lat., are required to start and complete the line-setting procedure during the nighttime, specifically, no earlier than one hour after local sunset and no later than local sunrise (*new technologies regulatory amendment*).

The governments of American Samoa, Guam, and the Commonwealth of the Northern Mariana Islands do not specifically regulate pelagic fishing activities, although fishing vessel registration is required. The State of Hawaii (State) prohibits the sale of yellowfin and bigeye tuna (both known in Hawaii as *ahi*) smaller than three pounds landed by all domestic fisheries. State statutes establishing longline area closures around the main Hawaiian islands and prohibiting shark finning activities[2] complement Federal fisheries regulations. The State also requires fishermen who sell any portion of their catch to hold a commercial marine license and file catch reports.

2.1. Pelagic Fisheries in Hawaii

This section covers pelagic fisheries of Hawaii managed under the Pelagics FMP. These fisheries employ a variety of techniques (longline, troll, handline, and pole-and-line) and operate in areas that range from near shore to beyond the U.S. EEZ. A detailed description of each of these fisheries is provided in the March 2001 FEIS.

2.1.1. Hawaii-based Pelagic Longline Fishery

The Hawaii-based longline fishery as it operated until March 2001 is described in detail in the March 2001 FEIS (Section 3.10.3.1, pages 195 to 256). Since 2000 the fishery has been operating in a highly dynamic regulatory environment, so the operational characteristics of the fishery have been quite dynamic, as well. The fishery's regulatory history is described in the March 2001 FEIS and the proposed new technologies regulatory amendment (WPRFMC, 2004).

The Hawaii-based longline fishery is a limited access fishery, with a total of 164 permits that are transferable (Table 2.1). 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.

Table 2.1. Summary Information on the Hawaii-based Longline Fishery, 2002. Source: NMFS, unpublished data, 2004

Area Fished	**U.S. EEZ around Hawaii and PRIAs; high seas in the central and mid-North Pacific regions**
Total Landings	17.1 million lb
Target Species	Bigeye, Yellowfin, and Albacore Tuna
Composition of Landings	72% tuna, 2.6% swordfish, 8.3% marlins, 2.3% sharks, 14.7% miscellaneous pelagic species
Season	Year round but highest during fall and winter
Active Vessels	100
Total Permits	164 (transferable, limited entry)
Total Trips	1162
Total Ex-Vessel Value	US$37.5 million

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[2] Hawaii Revised Statutes Chapter 188, enacted in June 2001, prohibit shark finning in State waters. All sharks caught by fishermen must be landed whole; that is, fins must be attached to the shark.

2.1.1.1. Hawaii-based Longline Tuna-target (Deep-Set) Gear Configuration

Tuna-target longline fishing is also known as deep-set longline fishing. In general, longline gear consists of a continuous main line that is set on the surface and supported in the water column horizontally by attaching floats. Longline fishing allows a vessel to distribute effort over a large area to harvest fish that are not concentrated in great numbers. Overall catch rates in relation to the number of hooks are generally low. Plastic floats are commonly used though radio buoys are also used to keep track of the mainline. 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 (400m for bigeye tuna). Deep-set longline gear is set in the morning and hauled in the afternoon (Ito and Machado, 2001). The main line is typically 30 to 100 km (18 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. Lightsticks are not typically 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.

2.1.1.2. Hawaii-based longline Swordfish target (shallow-set) Gear configuration

Swordfish-target fishing differs from tuna target fishing as it is set at a shallower depth, usually between (~30-90m) . Shallow-set longline gear is generally set at night, with luminescent light sticks, thought to attract swordfish, attached to the gangions. 4-6 gangions are typically clipped to the mainline between floats. A typical set for swordfish uses about 700-1,000 hooks. The historical swordfish fishery used squid as bait, but under the proposed action circle hooks with mackerel bait will be required for shallow-sets. The proposed action only allows for 2,120 shallow sets each year by the Hawaii longline fleet. These 2,120 sets will be distributed equally allocated between holders of Hawaii longline limited access permits

2.1.1.3. Vessel Activity

The Hawaii-based longline fishery is the largest commercial fishery in the western Pacific region. In 2002, 100 Hawaii-based longline vessels were active. The number of active longline vessels has decreased by about 25 since 2000. The decrease was due to Hawaii-based longline vessels that relocating to California to fish for swordfish as a result of turtle conservation and mitigation measures that prohibited Hawaii-based longline vessels from targeting swordfish. These vessels that relocated to California de-registered their vessels from their Hawaii longline limited access permits in order to continue fishing for swordfish. Approximately 35 vessels fished out of California in 2001, almost all of which had some history of fishing in the Hawaii-based longline fishery.

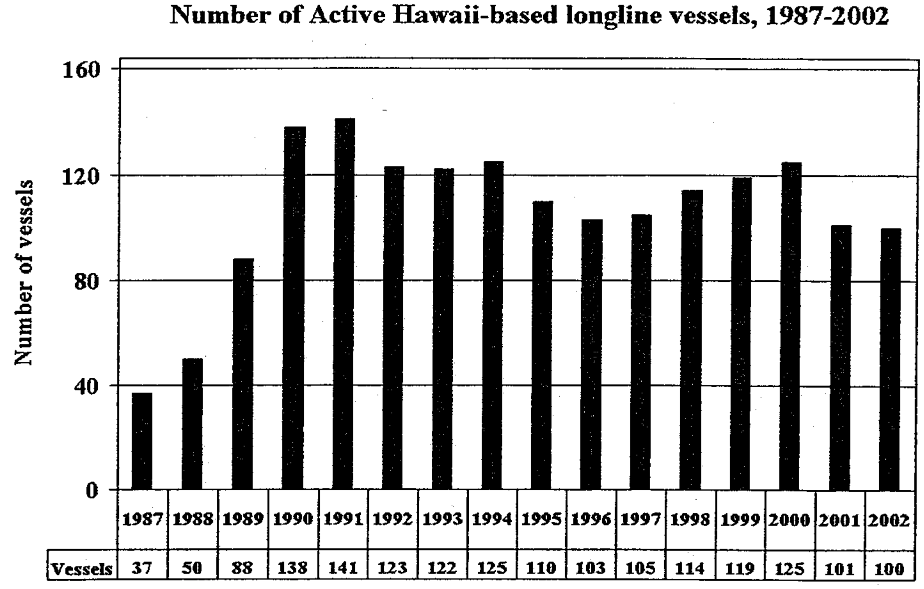


Figure 2.1. Number of active Hawaii-based longline vessels, 1987-2002. Source: NMFS unpublished data

The number of active vessels in the Hawaii-based longline fishery increased dramatically in the late 1980s and peaked at 141 vessels in 1991 (Figure 2.1). The number of vessels has since ranged from101 to125. The longline fishery operates year-round although vessel activity increases during the fall and is greatest during the winter and spring months.

The proposed action may result in an increase the number of fishery participants, but the likely number is unknown. Potentially important factors include the costs of relocating, the costs and risks associated with having to acquire a sufficient number of shallow-set certificates to enable full operations, and the regulatory environment in the west coast based fishery.

2.1.1.4. Number of Trips

In 2002, Hawaii-based longline vessels made 1,162 trips up by 128 trips from 2001. Trips were categorized on the basis of target species as tuna-, swordfish-, or mixed- (tuna and swordfish) target. In 2002, all trips were tuna-target trips.

The annual number of trips for the Hawaii-based longline fishery has remained relatively stable, but there has been a shift from mixed-target and swordfish-target trips to tuna-target trips from the early 1990s up to 2002 (Figure 2. 2).

The proposed action would allow 2,120 shallow sets to be made each year, which is equivalent to approximately 166 trips given the historical average of 13 sets per swordfish trip. It is predicted that there would be about 9, 759 annual deep sets targeting tuna, which is equivalent to approximately 813 tuna-target trips given the historical average of 12 sets per tuna trip.

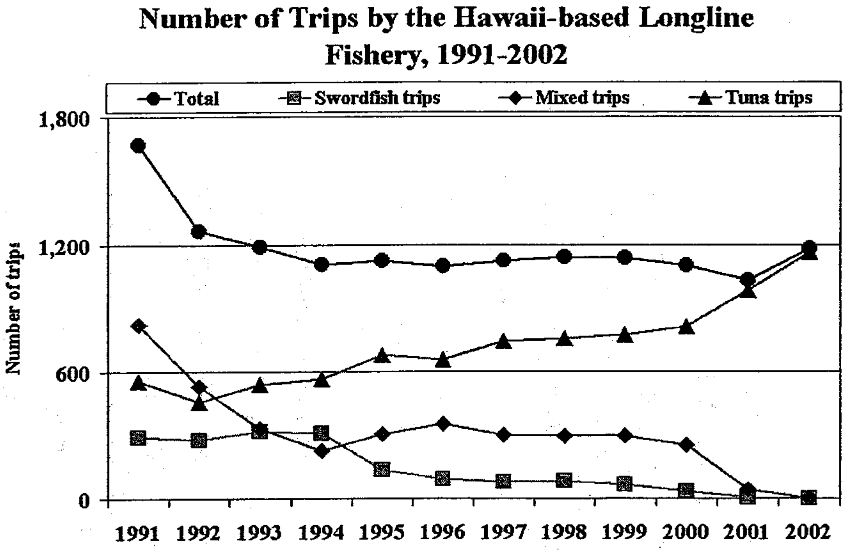
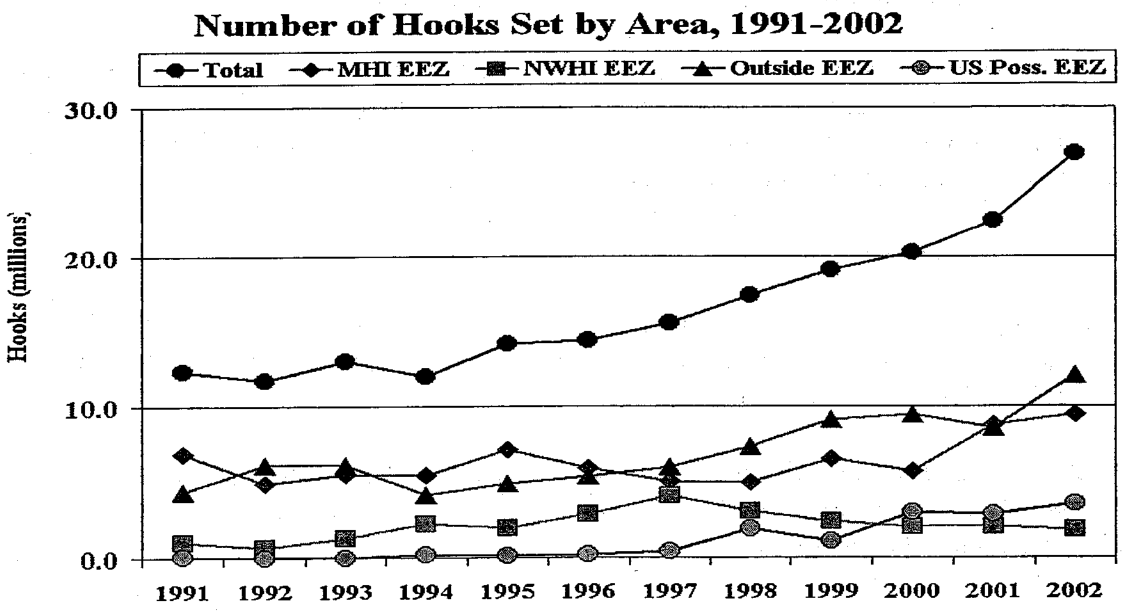


Figure 2.2. Number of trips in the Hawaii-based longline fishery, 1991-2002. Source: Ito and Machado, 2004, NMFS unpublished data.



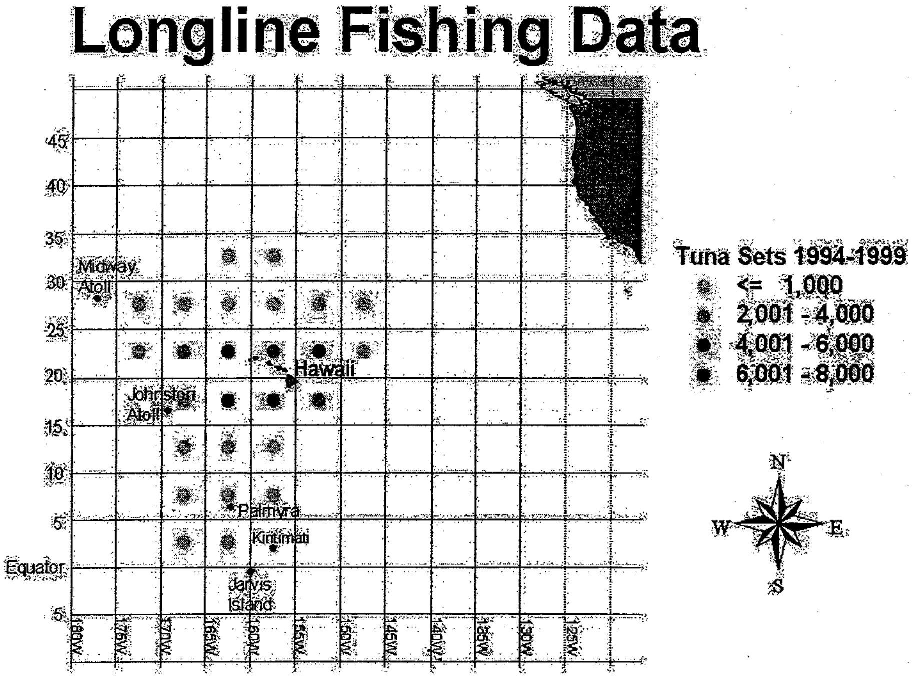


Figure 2.4. Fishing effort by the tuna component of the Hawaii-based longline fishery 1994-1999. Source: NMFS Annual Report

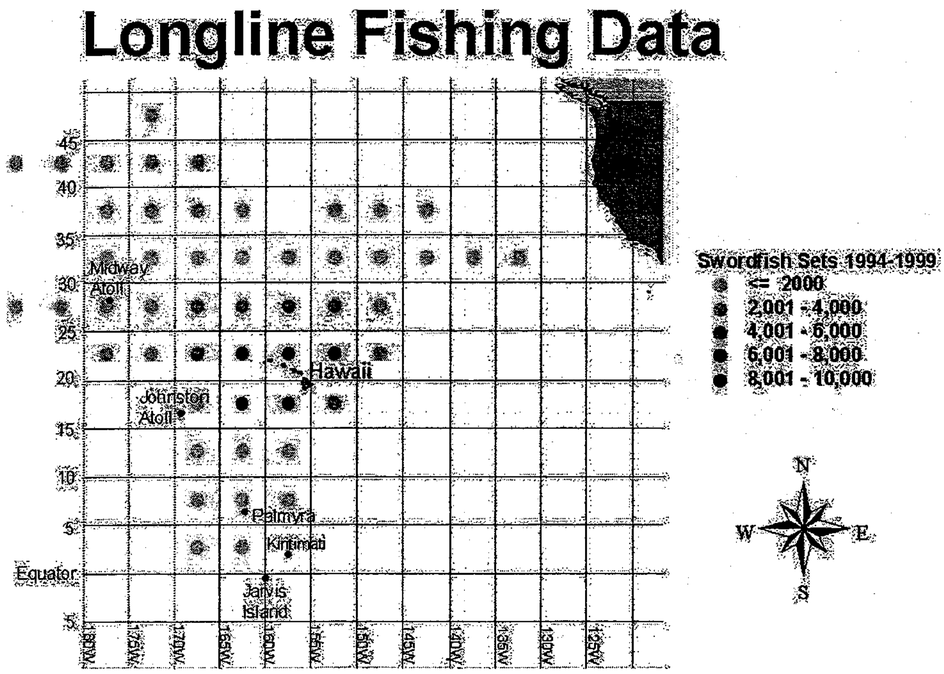


Figure 2.5. Fishing effort by the swordfish component of the Hawaii-based longline fishery 1994-1999. Source: NMFS Annual Report

2.1.1.5. *Number of Hooks Set*

A record number of 27 million hooks were set in 2002 (NMFS unpublished data, 2004) (Figure 2.3). This increase in number of hooks is a result of the shift in effort to tuna, which typically includes more than twice as many hooks per day fished than swordfish or mixed-target trips.

The spatial distribution of effort in the tuna and swordfish components of the fishery during the 1994-1999 period is shown in Figures 2.4 and 2.5, respectively.

As indicated in Figure 2.5, the swordfish component of the fishery operates well north of the Hawaiian Island chain, outside the EEZ. The proposed action, which would allow a limited amount of shallow-set effort, is therefore expected to result in an increase in longline effort in this northern area relative to 2001 and 2002. The proposed action will also eliminate the seasonal longline closure in waters south of the Hawaiian Islands. The closure currently applies to the waters from the equator to 15° N lat. and between 145° W long. and 180° W long. from April 1 to May 31. Tuna-directed effort is consequently expected to increase in those waters under the proposed action.

2.1.1.6. *Catches*

Between 1994 and 1999, before the imposition of measures to protect sea turtles, the fishery resulted in average annual catches of 6.5 million pounds of swordfish, 5.2 million pounds of bigeye, 2.5 million pounds of albacore, and 1.7 million pounds of yellowfin (Table 2.2). The 1994-1999 period provides a convenient baseline because the longline fishery was not yet subject to the sea turtle conservation measures. Data from 2002, which are included in Table 2.3, are indicative of fishery characteristics under the current management regime.

The proposed action is predicted to result in a decline of 45% in the swordfish catches relative to the 1994-1999 baseline. Catches are expected to increase by about 13% for bigeye tuna, 19% for albacore tuna, and 12% for yellowfin tuna. Ex-vessel revenues are expected to decrease by about 4.4% relative to the 1994-1999 period.

Table 2.2. Catch and revenues: estimates for 1994-1999 and 2002, and projections under action. Source: NOAA-Fisheries Pacific Islands Fisheries Science Center

	Swordfish catch (million lb)	Bigeye catch (million lb)	Albacore catch (million lb)	Yellowfin catch (million lb)	Ex-vessel revenue ($ million)
1994-1999 annual average	6.5	5.2	2.5	1.7	45.4
2002	0.5	5.2	2.8	2.2	33.1
Proposed action	3.3	5.9	3.0	1.9	43.5

2.1.1.7. *Observer program for the Hawaii-based longline fishery*

The NMFS observer program for the Hawaii-based longline fishery began in 1990, with the voluntary sampling of fishing operations because of unconfirmed reports of interactions between swordfish vessels and protected species, such as Hawaiian monk seals, sea turtles, and seabirds (Dollar, 1991).

Subsequently, a mandatory NMFS observer program was established in April 1994, to better characterize and understand the effects of the Hawaii-based longline fishery on the incidental take of sea turtle, sea bird, and marine mammal populations.

Initially, observers were placed aboard Hawaii-based longline vessels according to the Statistical Guidelines for a Pilot Observer Program to Estimate Turtle Takes in the Hawaii Longline Fishery (DiNardo, 1993). Using this approach, observer placements were distributed evenly across different strata based on target species and time. During each quarter, a pre-determined number of swordfish, tuna, mixed, or switcher vessels were randomly sampled by observers. Overall observer coverage between 1994 and 1996 was between 4.5 and 5.3%.

In April 1997, the observer placement strategy changed to reflect the guidelines established in the NOAA Administrative Report, "Recommendations for Scoping the Sea Turtle Observer Program for the Hawaii-Based Longline Fishery" (Skillman et al., 1996). These guidelines recommended that observer coverage be increased to at least 20% overall to obtain more reliable estimates of sea turtle take. However, because of a lack of funding, the Southwest Region began sampling approximately 5% of the overall fleet effort while focusing on the larger vessels, which were determined to account for 87% of the sea turtle takes. Monitoring a percentage of the small boat sector allowed the potential for detecting large changes in the turtle take rate in that portion of the fishery. The observer data are used to estimate the incidental take of sea turtles, marine mammals, and seabirds and to verify logbook data which are considered reliable for calculating fishing effort and target species, but not for estimating incidental take of sea turtles or other protected species. Observer coverage between 1997 and 1999 was 3.3 to 4.1%.

In August 2000, NMFS modified the observer program sampling design to comply with the terms and conditions of a Court Order. Vessel selections during this time period were based on observer availability and the percent observer coverage goals stipulated by the Court. In late 2000, observer services were contracted out on a permanent basis through a private contractor, Saltwater, Inc. Since January, 2001, 102 observers have been trained. An experienced corps of observers has emerged from this group enabling the NMFS observer program, administered by NMFS-PIRO, to maintain an observer staff ranging from 25 to 40 persons at a given time.

The sampling design for the program changed in 2001. Unstratified random sampling of vessels for observer placement was initiated when the entire fleet converted to targeting tuna in 2001. In May, 2002, a formal systematic sampling scheme, developed by the NMFS Honolulu Laboratory, was implemented to facilitate data analysis.

The observer program maintained observer coverage levels for the Hawaii-based longline fleet above 20% in 2001 and 2002. In the early part of 2002, coverage rates over 30% were attained when monies and personnel became available to the program. NMFS' practice is to maintain observer coverage rates slightly above 20% at any given time. The NMFS observer program completed four to five times the number of observed trips per year in 2001 and 2002 than in years prior to 2000 (see Table 2.3 for summary).

Table 2.3. Selected performance measures for the Hawaii-based longline vessel observer program, 1994-2003. (NMFS unpublished data)

Year	Number of trips[3]	Number of trips observed[4]	Percent Coverage[5] (%)
1994[6]	1031	55	5.3
1995[7]	937	42	4.5
1996	1062	52	4.9
1997	1123	40	3.6
1998	1180	48	4.1
1999	1136	38	3.3
2000	1134	118	10.4
2001	1035	233	22.5
2002	1129	278	24.6
2003 (9 months)[8]	875	187	21.4

For the purpose of monitoring with respect to the turtle interaction limits under the proposed action, NOAA-Fisheries will restructure the observer program by separating the shallow and deep set components of the fishery for the purpose of sampling design.

2.1.2. Hawaii-based Troll Fishery

The Hawaii troll fishery is a hook and-line fishery that typically uses rods and reels as well as hydraulic haulers, outriggers, and other gear to drag lures or baited hooks from moving vessels.

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[3] Based on dockside information obtained by NMFS

[4] Completed number of trips

[5] Observer coverage based on number of observed trips and dock-side information.

[6] Data from March 1994 through February 1995.

[7] Data from February through December 1995.

[8] Data from January through September 2003.

Up to six lines rigged with artificial lures or live bait may be trolled when outrigger poles are used to keep gear from tangling. When using live bait, trollers move at slower speeds to permit the bait to swim naturally (WPRFMC 1995). This fishery has three major sectors: commercial troll, charter, and recreational/subsistence. A detailed description of this fishery is presented in the March 2001 FEIS (Section 3.10.3.2, pages 257 to 287).

The Hawaii-based troll fishery operates mainly within the EEZ of the Main Hawaiian Islands (Table 2.4), usually well within the 50 nautical mile protected species zone closed to longliners. The fishery operates year round but activity is usually highest during the summer months. There were 1,490 active fishermen in the commercial Hawaii based troll fishery that made 25,372 trips and landed 2.1 million pounds of fish worth 3.0 million in 2002 (HDAR,2002 preliminary).

Under the current management regime, all vessels fishing with hook and line are required to employ sea turtle handling measures and to carry certain equipment to remove hooks and line from accidentally hooked or entangled sea turtles. The proposed action does not reinstate this requirement for the pelagic handline, troll, and pole-and-line vessels.

Table 2.4. Fishery Information on the Commercial Hawaii Troll Fishery, 2002. Source: Hawaii Division of Aquatic Resources, DLNR, State of Hawaii

Area Fished	Predominantly Main Hawaiian Island's EEZ
Total Landings (in pounds)	2,170,897
Target Species	Yellowfin tuna, Mahimahi, Blue Marlin, Ono, Skipjack Tuna
Catch Composition	31.7% Tuna, 24.3% Billfish, 27.5% mahimahi, 15.9% ono
Season	All year but highest during summer months
Active Vessels	1,490
Total Permits	NA
Total Trips	25,372
Total Ex-vessel Value	US$3,004,160

2.1.3. Hawaii-based Handline Fishery

The pelagic handline fishery is predominantly a tuna fishery conducted by small boats using relatively simple hook-and-line fishing methods. In Hawaii, three types of handline fishing methods are practiced: nighttime *ika-shibi* (squid-tuna) method, daytime *palu-ahi* (chum-tuna), and seamount fishing and weather buoy method (using both handline and troll methods). The Hawaii-based handline fishery operates within the EEZ of the Main Hawaiian Islands and outside the EEZ (Table 2.5). The fishery operates year round but activity is usually highest during the summer months. The Hawaii-based handline fishery made a total of 4,433 trips in 2002 (HDAR, 2002, preliminary). A detailed description of the Hawaii-based handline fishery is covered in the March 2001 FEIS (Section 3.10.3.3, pages 287 to 305).

Handline gear is set below the surface to catch relatively small quantities of large, deep-swimming tuna that are suitable for *sashimi* markets. The Hawaii handline fishery has nearshore and offshore components. The nearshore fishery targets large yellowfin and bigeye tunas. Nearshore areas have a public sector supported FAD system. The offshore fishery targets juvenile bigeye and yellowfin tuna around seamounts and weather buoys that are 50 to 320 km (35 to 200 nm) from shore (WPRFMC 1995). Some of the larger vessels are able to fish near seamounts and weather buoys located 100 to 200 nm from shore.

In the nighttime *ika-shibi* fishery, three to four handlines are set, each consisting of a long nylon rope connected to a dacron or polypropylene mainline attached to a monofilament nylon leader. The hooks are usually baited with mackerel scad, and the lines are lowered with lead weights. To attract baitfish and tuna, a low-wattage light bulb is placed in the water, and the surface is chummed with chopped squid and/or chopped anchovies (WPRFMC 1995).

The daytime *palu-ahi* technique adds a weighted, retrievable bag stuffed with chum that is opened at a depth of 120 to 140 meters (400 to 650 ft), releasing the bait to attract tuna to the baited hooks. When a fish is hooked, it is manually hauled in, gaffed and then killed with a bullet or wooden bat.

Under the current management regime, all vessels fishing with hook and line are required to employ sea turtle handling measures and to carry certain equipment to remove hooks and line from accidentally hooked or entangled sea turtles (65 FR 16346, March 28, 2000; 66 FR 67495 December 31, 2001; 67 FR 40232, June 12, 2002). The proposed action does not reinstate this requirement for the pelagic handline, troll, and pole-and-line vessels.

Table 2.5. Summary information on the Commercial Hawaii-based Handline Fishery, 2002. Source: Hawaii Division of Aquatic Resources, 2002, preliminary

Area Fished	**Predominantly EEZ Main Hawaiian Island and seamounts outside the EEZ**
Total Landings	2,102,150
Target Species	Yellowfin tuna, Bigeye tuna
Catch Composition	37.7% yellowfin tuna, 39.2% bigeye tuna, 16.7% albacore tuna
Season	Year round
Active Vessels	421
Total Permits	NA
Total Trips	
Total Ex-vessel Value	US$2,918,704

2.1.4. Hawaii-based Pole-and-Line Fishery

The Hawaii-based pole-and-line fishery is referred to as the *aku* (skipjack tuna) fishery. This fishery uses live-bait as chum to catch skipjack tuna and juvenile yellowfin tuna. A description of the pole-and- line fishery is included in the March 2001 FEIS (Section 3.10.3.4, pages 305 - 312). Hawaii's *aku* fishery began to decline in the mid-1970s prior to closure of the tuna cannery in Honolulu.

Skipjack tuna caught by this fishery are now sold to the local fresh fish market. The Hawaii-based pole-and-line fishery operates primarily within the EEZ around the Main Hawaiian Islands (Table 2.6). Seven pole-and-line vessels actively fished in 2002. These vessels operated year round but their activity was highest during the summer months. The Hawaii-based pole-and-line fishery made a total of 239 trips in 2002. The landings by the pole-and line fishery was 550 thousand pounds which consisted almost exclusively of skipjack tuna (HDAR, 2002, preliminary)

Under the current management regime, all vessels fishing with hook and line are required to employ sea turtle handling measures and to carry certain equipment to remove hooks and line from accidentally hooked or entangled sea turtles. The proposed action does not reinstate this requirement for the pelagic handline, troll, and pole-and-line vessels.

Table 2.6. Fishery Information on the Commercial Hawaii Pole-and Line Fishery, 2002. Source: Hawaii Division of Aquatic Resources, 2002, preliminary

Area Fished	Main Hawaiian Islands EEZ
Total Landings (in poiunds)	550,737
Target Species	Skipjack Tuna
Catch Composition	93% skipjack tuna
Season	All year
Active Vessels	7
Total Permits	NA
Total Trips	239
Total Ex-vessel Value	US$746,036

2.1.5. Pacific Remote Island Areas (PRIA) Pelagic Troll/Handline Fishery

The PRIA or "U.S. island possessions in the Pacific" include Howland Island, Baker Island, Jarvis Island, Wake Island, Kingman Reef, Johnston Atoll, Palmyra Atoll, and Midway Atoll. Midway Atoll, located in the Northwestern Hawaiian Islands, is not part of the State of Hawaii and is treated as one of the PRIA. A few years ago (1998-1999), there was interest in the potential development of a pelagic troll/handline fishery around Palmyra in the central Pacific. This was spurred by two or three Hawaii-based trollers journeying to Palmyra on fishing expeditions. Also there was indication that a charter troll fishery would expand at Midway Atoll as part of an ecotourism program administered by the U.S. Fish and Wildlife Service (USFWS) (March 2001 FEIS, Section 3.10.3.5, pages 312 - 313). Today, the fishery is dormant; there is no commercial troll/handline fishing activity in EEZ waters around the PRIA in the central Pacific, which is likely due to the lack of an infrastructure at Palmyra Atoll to support a fishery. At Midway Atoll, the U.S. Fish and Wildlife Service is currently engaged in securing a private contractor to reestablish an economically viable ecotourism program. At this time it is unclear if the program will have a troll charter fishing component as part of its program. Nonetheless, current Pelagics FMP management measures include permit and reporting requirements for any U.S. fishing vessel using troll/handline fishing gear to harvest PMUS in waters of the EEZ around the PRIA (67 FR 56500, September 4, 2002). The reporting requirements enable NMFS to monitor the fishery through the collection of catch and effort data , including fishery interactions with protected species.

2.2. Pelagic Fisheries in American Samoa

The American Samoa-based pelagic fleet includes the generally small twin-hulled *alia* longline vessels, a number of mid-sized and larger monohull longliners, and a relatively small number of vessels used for trolling. In addition, the U.S. distant-water tuna purse seine fleet delivers much of its catch to the canneries in Pago Pago, the capital of American Samoa, and U.S. distant-water albacore trollers occasionally do, as well. Substantial offloading at the canneries is done by foreign purse seine and longline vessels. A detailed description of the pelagic fisheries in American Samoa is presented in the March 2001 FEIS (Section 3.10.4, pages 313 - 335). The longline and troll fishing has been described earlier under the title *Pelagic Fisheries in Hawaii.*

American Samoa's domestic longline and troll fisheries are described further below. Relatively small amounts of pelagic species are also landed from methods not generally used to target pelagic species, including bottomfishing and spearfishing. For example, in 2001 these other methods resulted in about 6,000 pounds of pelagic species landings (WPRFMC 2002c).

2.2.1. American Samoa-based Pelagic Longline Fishery

Table 2.7 summarizes the recent status of the American Samoa-based longline fishery managed under the Pelagics FMP. In 2002, the longline fleet, composed of 60 active vessels, landed 423,046 fish (primarily tunas). Landings of the longline fleet have been dominated by albacore tuna, which comprised about 79% of the landings in 2002. The ex-vessel value of longline landings in 2001 was about $8 million.

Small-scale pelagic longlining was introduced into American Samoa in 1995 by fishermen from neighboring independent Samoa (former Western Samoa), where a longline fishery was already established. The longline fleet based on the island of Tutuila, American Samoa, has been, until recently, dominated by twin-hulled boats of aluminum or wood/fiberglass, called *alia*, most of which are about 30 feet long and powered by 40 horsepower outboard engines. These vessels, on which navigation is generally limited to visual methods, typically make only single-day trips, so most of their fishing effort occurs within 25 nautical miles of shore. The longline fishery grew fairly steadily through the late 1990s, but after 2000, it expanded rapidly with the entry of a number of large vessels. In 2002, the fleet was composed of about 27 of the relatively small (< 40 feet) *alia*, about five mid-sized (40-50 feet) monohull vessels, and about 28 large (> 50 feet) monohull vessels (WestPacFin, 2003). These large vessels, which have hydraulically powered reels and electronic navigation equipment and substantially greater gear and storage capacities than the small *alia*, tend to conduct multi-day fishing trips and can range throughout the EEZ (WPRFMC 2002b and WPRFMC 2002c). The rapid influx of the large domestic longliners during just the last three years has resulted in both a dramatic increase in longline fishing effort in the EEZ around American Samoa and a shift in the spatial distribution of longline effort towards waters more distant from shore. The large-vessel closed area that went into effect early in 2002, which prohibits vessels longer than 50 feet from fishing for PMUS within approximately 50 nautical miles of the islands of American Samoa, has presumably contributed to the shift.

Table 2.7. Summary information on the American Samoa-based Pelagic Longline Fishery, 2002. Source: HDAR, 2002, preliminary

Gear	Longline
Area Fished	Inshore and EEZ
Season	All year
Active Vessels	70
Total Permits	75 (open access)
Total Sets	6,861
Total Landings (total number of fish)	423,023
Catch Composition	79% Albacore, 11% Skipjack, 4% Yellowfin, <4% all others
Total ex-vessel value	US$13.7 million

The domestic longline fleet in American Samoa expanded with the development of the *alia* longline fleet in 1996. A four-fold increase in landings between 2000 and 2001 reflects the near-doubling of the longline fleet size in that one-year period, including the entry of a number of relatively large vessels.

According to the Daily Effort Census, a dockside monitoring program, 70 vessels were engaged in longline fishing in 2002 (Western Pacific Regional Fisheries Management Council 2003). Of these 60 vessels turned in longline logs for sets made during the year. The logbooks submitted by the 60 vessels indicated that they made 6,861 sets (Figure 2.6) and set about 13 million hooks. Even though the number of vessels turning in logbooks was down from the 67 of 2001, the number of sets increased by 43% and the number of hooks set increased by 125% due to the increasing proportion of large vessels in the fleet.

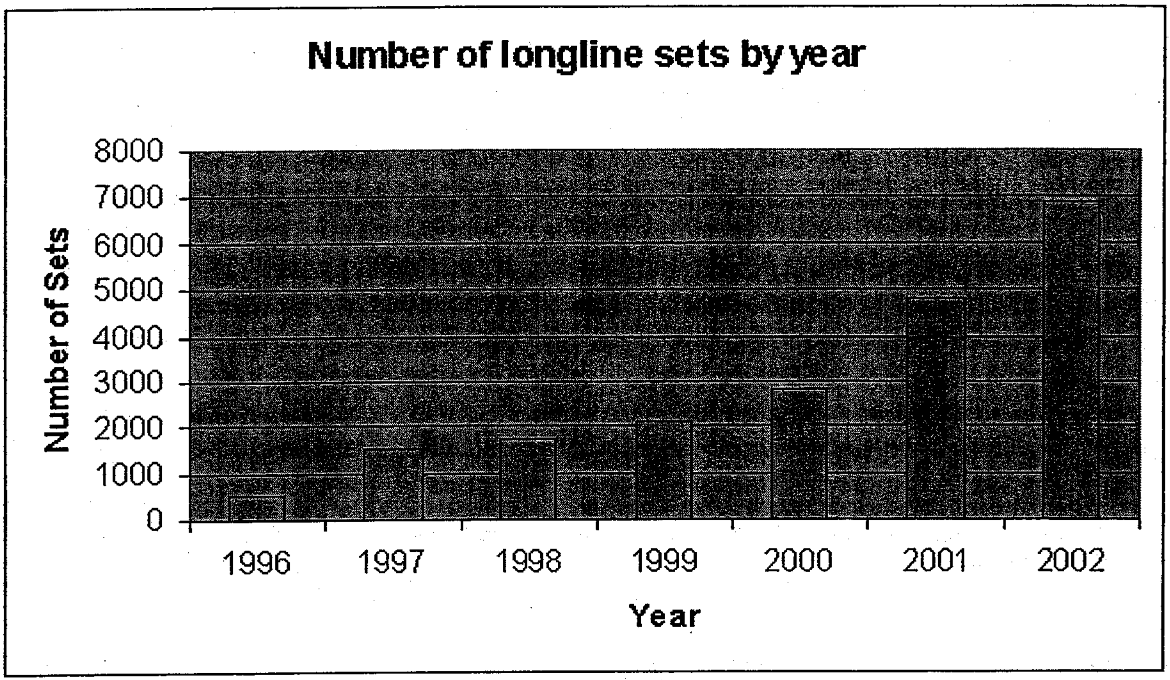

Figure 2.6. Number of sets in American Samoa longline fishery, 1996-2002. Source: WestPacFin, 2003

A record number of 478,855 fish were caught by the American Samoa longline fishery in 2002. Albacore continued to dominate the catch with 333,196 fish caught in 2002. This is a 77% increase over the record catch of 2001. Catches of skipjack tuna experienced almost a five-fold increase from 2001, while catches of yellowfin tuna increased by 107% and bigeye tuna increased by 119%.

The large increase in the number of longline vessels in 2001 and 2002 was due primarily to the entry of large (> 50 ft) vessels from outside American Samoa. O'Malley and Pooley (2002) reported that among recent new entrants in the fishery, three came from Hawaii, six came from the U.S. west coast, three came from the Gulf of Mexico, and four were U.S.-owned longliners that were foreign-built.

A Western Pacific general longline permit or Hawaii longline permit is required to longline in the EEZ around American Samoa, Guam, the Commonwealth of the Northern Mariana Islands, or the PRIA. With the exception of the troll/handline permit requirement for the PRIA (67 FR 56500, September 4, 2002), access by U.S. vessels to the pelagic fishery in these waters is not otherwise restricted. In addition to the permit requirement, longline operators must record certain information about their fishing activity, including catch by set and species, as well as interactions with protected species, in daily logbooks. Pelagics FMP management measures also prohibit large fishing vessels (greater than 50 feet in length), except as exempted, from fishing for PMUS within approximately 50 nm of the islands of American Samoa, including Tutuila, Manua, and Swains Islands, and Rose Atoll. Pelagics FMP sea turtle measures require operators of vessels that fish for pelagic fish with hook-and-line gear to carry line clippers and bolt or wire cutters and to employ specific sea turtle handling and resuscitation methods in the event that sea turtles are accidentally hooked or entangled during fishing operations. In addition, the operators of longline vessels must annually attend a protected species workshop conducted by NMFS. Under the proposed action, vessels based in American Samoa would continue to be prohibited to fish in the EEZ around Hawaii or land longline-caught fish in Hawaii. These vessels would be allowed to engage in shallow setting north of the Equator without any of the restrictions to which the Hawaii-based longline vessels would be subject. Although this represents new fishing opportunities for these vessels, the restrictions of fishing in the EEZ around Hawaii and on landing fish in Hawaii make it unlikely to be a cost-effective option, and it is unlikely to be taken advantage of by the vessels based out of Hawaii and American Samoa.

In response to the unrestricted expansion of the pelagic longline fishery in American Samoa, the Council developed an amendment to the Pelagics FMP (Amendment 11, incorporated by reference in this Opinion), which identifies nine alternatives to control longline effort around American Samoa (WPRFMC 2002b). The preferred alternative adopted by the Council would establish a limited access program in which eligibility to participate in the fishery is limited to owners of vessels that legally harvested PMUS with longline gear in the EEZ around American Samoa on or prior to March 21, 2002. Once the initial permits are issued to eligible participants, the number of available permits would be limited to that number. The limits would be broken down into each of four vessel size classes, although there would be limited opportunities for permit upgrades (by vessel size class) during the first four years of the program.

It has been estimated that a maximum of 138 individuals would likely to be eligible for initial permits, although not all eligible individuals would necessarily obtain a permit (WPRFMC 2002b). This is almost twice the number of currently permitted longline vessels, with most of the difference in the smallest of the four vessel size classes (≤ 40 ft), as indicated in Table 2.8.

Table 2.8. Current number of Permitted Longline Vessels based in American Samoa and likely number of permits available under the preferred alternative of proposed Amendment 11. Source: WPRFMC, 2002b.

Vessel Size	Current Number of Pemits	Potential Number of Pemits in 2003
< 40 ft.	40	93
40-50 ft.	5	9
50-70 ft.	15	15
> 70 ft.	15	21
All	75	138

The wide range of longline vessel types and associated fishing power in the American Samoa-based longline fleet is highlighted in data from two sources. Table 2.9 lists some of the typical characteristics of three vessel types used in American Samoa and neighboring Samoa, including the most common *alia* design (28 ft), the less common 40-foot *alia*, and the typical monohull longliner greater than 50 feet in length. Table 2.9 shows, for 2001, some of the same characteristics for three vessel types, as measured for the American Samoa fleet from logbook data. The estimates from the logbook data of sets per boat-year and hooks per boat-year are substantially less than the estimates for comparable vessel types given in Table 2.9. One possible explanation for the difference is that some of the vessels monitored in American Samoa in 2001 arrived in 2001 and did not fish the full year.

Table 2.9. Profiles of Longline Vessels based in American Samoa and Samoa (formerly Western Samoa). Source: Mulipola, 2000, pers. comm. cited in WPRFMC 2000.

Vessel Size and Type	28 ft. alia	40 ft. alia	50 + ft Monohull
Purchase Price (USD)	$25,000	$60,000	$250,000
Miles of mainline set	7-10	20-25	35-50
Sets per trip	1-2	up to 4	6-8
Hooks/set	250-350	500-900	1,200-1,600
Trips/year	100-200 (wester dependent)	50	40
Hooks/year	30,000-60,000	160,000	400,000

Amendment 11 includes an assessment of the likely effects of the management alternatives. In order to estimate the likely total fleet-wide fishing effort that would occur under the preferred alternative, certain assumptions were made about likely levels of participation and per-vessel fishing effort, by size class. Using the assumptions made in the first three rows of Table 2.10, the estimate of total likely effort in the EEZ around American Samoa was about 17 million hooks per year assuming no permit upgrades and about 25 million hooks per year assuming all available permit upgrades are taken. It was estimated that about 85 to 90 percent of total effort would occur beyond 50 nm from shore (WPRFMC 2002b). The estimates were based on the assumption that 75 vessels (the number permitted as of the control date, March 21, 2002) would obtain permits and actually fish. In comparison, the Hawaii-based longline fleet included 100 vessels in 2002 and set a record of 27 million hooks.

2.2.2. American Samoa-based Troll Fishery

Table 2.11 contains a summary of the recent status of the American Samoa-based troll fishery managed under the Pelagics FMP. In 2001, the troll fleet, composed of about 18 active vessels, landed about 24,000 pounds of pelagic species (Figure 2.9), dominated by skipjack and yellowfin tuna, with an ex-vessel value of about $24,000. In 2002, longlining constituted approximately 99.8 % of the total landings whereas trolling constituted 0.2% of landings recorded.

While the longline fishery has grown rapidly during the last few years, the troll fishery has diminished (although not as dramatically as the longline fishery has grown) due to troll fishermen switching to longlining. Trolling catches decreased fairly steadily from 1982 through 2001. This trend may reflect the increase in effort of longline fishing method used by local fishermen during this period. In 2002, landings by the troll fishery comprised only about 0.2% of total pelagic species landings in American Samoa by the locally based fleets. Recently there have been anecdotal reports of *alia* fishermen returning to trolling, but data are not available to indicate the extent of the shift.

Table 2.10. Estimates of Likely Fishing Effort in American Samoa's Longline Fishery under Amendment 11. Source: WPRFMC 2002b.

	< 40 ft.	40-50 ft.	50-70 ft.	>70 ft.	Total
Average sets/boat year	125	175	225	225	
Average hooks/boat year	43, 750	218,750	472,550	472,550	
Boats	40	5	15	15	75
Total sets/year	5,000	900	3,400	3,400	13,000
Total hooks/year	1,750,000	1,094,000	7,087,000	7,087,000	17,000,000

Table 2.11. Summary information on the American Samoa-based Troll Fishery, 2002. Source: NMFS unpublished data

Gear	**Troll**
Area Fished	Nearshore and EEZ
Total Landings	24,000 lb.
Landings Composition (by weight)	50% skipjack, 20% yellowfin, 9% pomfret
Season	All year
Active vessels	18
Total Permits	n/a
Total trips	343
Total ex-vessel value	$24,000

Values were estimated as the product of total landings and average prices by species group reported in WPRFMC 2002c.

2.3. Pelagic Fisheries in the Territory of Guam

U.S. domestic fishing vessels based in Guam that target PMUS are the distant-water tuna purse seiners, longliners, and smaller recreational trollers. The larger purse seiners fish outside the EEZ around Guam and transship their catch through Guam. The smaller recreational fishing vessels, which are either towed to boat launch sites or berthed in marinas, and domestic longliners, fish within the EEZ around Guam or the adjacent EEZ around the Northern Mariana Islands. There is no active domestic longline fishery in Guam at this time. Background information on the pelagic fisheries in Guam is contained in the March 2001 FEIS (Section 3.10.5, pages 335 - 352), which is incorporated in this Opinion by reference.

Guam's domestic pelagic fishery consists mostly of small trolling vessels that are recreational, subsistence and/or commercial (part-time). There is also a small, but significant charter troll fleet that are full-time commercial. Table 2.12 profiles the pelagic fishery as it existed in 2002. The number of troll fishing vessels active in the Guam pelagic fishery in 2002 remained constant with 375 active vessels (Figure 2.12). In 2002, the total pelagic landings were about 533,855 pounds, a decrease of 42% from 2001.

Table 2.12. Summary Information on the Pelagic Fishery in Guam, 2001. Source: NMFS unpublished data.

Gear	**Troll/Charter**
Area Fished	Nearshore and EEZ
Total Landings (in pounds)	533,855
Targets and Catch Composition	33% skipjack tuna, 32% mahimahi, 13% wahoo, 8% yellowfin tuna, 10% Pacific blue marlin
Season	All year
Active Vessels (est.)	375
Total Permits	NA
Total Trips	8,933
Total Ex-vessel (Commercial) Value*	US$486,946

*Data (inflation-unadjusted) are available for commercial value, and unavailable for noncommercial values.

2.4 Pelagic Fisheries in the Commonwealth of the Northern Mariana Islands

The CNMI-based pelagic fleet is composed primarily of vessels less than 24 feet in length that are used for trolling and that tend to range no more than about 20 miles from shore (WPRFMC 2002a). The charter component of the fleet includes larger vessels. Most of the fleet is based on Saipan, with smaller numbers of boats on Rota and Tinian. No longlining or purse seining currently occurs in the EEZ around the CNMI. Background information on the pelagic fisheries of the CNMI is presented in the March 2001 FEIS (Section 3.10.6, pages 352 - 362), which is incorporated by reference in this Opinion.

Table 2.13 contains a summary of the recent status of the CNMI-based troll fisheries managed under the Pelagics FMP. These estimates of landings, fishing effort, and revenues are derived from records of fish sales only on the island of Saipan, and not all fish sales on Saipan are recorded. It was estimated in WPRFMC (2002c:4-1) that "the commercial purchase database landings include more than 90% of all commercial landings on Saipan."

Table 2.13. Summary Information on the Pelagic Fisheries of the Commonwealth of the Northern Mariana Islands, 2002. Source: draft WPRFMC 2003

Gear	Troll/Charter
Area Fished	Nearshore and EEZ
Total Landings (in pounds)	253,274
Landings Composition (by weight)	70 % skipjack tuna, 7% mahimahi, 12% yellowfin tuna
Season	All year
Active Vessels	86
Total Permits	NA
Total Trips	1,803
Total Ex-vessel Value	US$499,730

Note: These estimates are only for commercial activity that resulted in fish sales on Saipan. The estimates of CNMI-based fishing activity that are provided in Table 2.13 and throughout this section have not been adjusted to account for the unreported components of the fisheries, so unless otherwise noted, the estimates provided here under-represent total pelagic fishing activity in the CNMI

Figure 2.13 shows the numbers of fishermen that were active in Saipan's commercial pelagic fisheries from 1984 through 2002. Each year's estimate is the number of individuals that were recorded as having sold any pelagic species during the year. Based on the WPRFMC draft report (draft WPRFMC,2003) about 55 vessels were identifed as involved in full time commercial fishing and 41 vessels were classified as part-time. No fishing and/or recreational usage included 312 vessels. Twenty-six vessels were registered as charter vessels for 2002.

3.0 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* (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)).

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' Pacific Islands Regional Office ensure that the proposed management regime of the pelagics fisheries of the western Pacific region is not likely to jeopardize the continued existence of threatened or endangered species. Because the proposed fisheries are not likely to adversely affect critical habitat that has been designated in the action area for this consultation, this Opinion will focus only on jeopardy analyses.

3.1 Method

After receiving a complete description of the proposed management regime for the fisheries from Pacific Islands 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, and) 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 mask their extinction risks over time so long-term projections 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. 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).

Finally, jeopardy analyses must look into the future to encompass any delays between the effects of an action and the population responses of threatened and endangered species. Some human activities appear to have "delayed" effects on plant and animal populations, which can occur for two primary reasons. First, a disease, toxic chemical, or other stressor may take time to accumulate and individuals animals may respond only after they receive particular threshold doses. Second, a human stressor may have immediate effects on individuals or populations, but the ecology of the species may mask our ability to detect the effect. In the previous example, the bald eagle populations had declined for several years before humans were able to detect it. This kind of "delayed" effect probably reflects limitations in our ability to detect effects below certain thresholds or our inability to identify abnormal population declines given background rates of population variability.

With sea turtles, we expect the second kind of "delayed" effect. Because of these delayed effects, assessments in the Services' biological opinions must look far enough into the future to (1) be certain of detecting a population's response to an effect, (2) be certain of detecting changes in a species' reproduction, numbers, and distribution, and (3) be certain of detecting changes in a species' likelihood of surviving and recovering in the wild (Crouse 1999). If we do not look far enough into the future, we increase the risk of failing to detect a population's response to a human activity and we are more likely to falsely conclude there was no effect when, in fact, an effect occurred (which, in the case of fisheries, means that adult, subadult, and juvenile turtles will have been captured and killed for a period of years). If we look too far into the future, the passage of time can mask short-term collapses in a population and, again, we increase our likelihood of falsely concluding there was no effect when, in fact, an effect occurred.

3.2 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 U.S. Pacific pelagics 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 NMFS 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 U.S. Pacific pelagics fisheries managed under the Western Pacific Pelagics Fisheries Management Plan 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.

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.

3.3 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 (NMFS and USFWS, 1995; USFWS, 1997); recovery plans for the eastern Pacific green turtle (NMFS and USFWS, 1998a), U.S. Pacific populations of hawksbill sea turtles (NMFS and USFWS, 1998b), loggerhead sea turtle (NMFS and USFWS, 1991), leatherback sea turtle (NMFS and USFWS, 1992), and U.S. Pacific populations of olive-ridley sea turtles (NMFS 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 National Research Council (1990) identified many of these limits and recommended research on a wide array of variables, including age at reproductive maturity, age-specific rates of survivorship and fecundity, distribution, and migration.

To conduct this assessment, we relied on three kinds of evidence: (1) empirical information, that is data, studies, other observations of species and populations that have become extinct; (2) quantitative analyses using data that had been gathered from other threatened, endangered, or extinct species or populations; and (3) the results of computer simulations and similar analyses. From the perspective of the evidence available for this consultation, we must distinguish between the data that are available, the various methods that are available to analyze those data, the theoretical foundations for our current understanding of population ecology, and computer simulations and similar analyses that are conducted to gain insights into population ecologies, but are often based on a series of assumptions rather than data collected through empirical study.

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.

Absent the robust, long-term demographic data required by the more complex computer simulation models, we have to rely on simpler analytical methods whose results require qualitative interpretation because of the uncertainty and assumptions underlying these methods. Bolten *et al.* (1996) and Heppell et al. (2003) concluded that developing analytical tools to support assessments like the one we must conduct in this Opinion requires much more information than is currently available. Pritchard (1996) concluded that we do not currently have enough life history data on sea turtles to construct models that can be used for predictive purposes. Until we have more robust demographic data, we must interpret the products of any "quantitative" analyses with sufficient caution and reason to blur any distinction between qualitative and quantitative analyses (Beissinger and Westphal 1998, Feiberg and Ellner 2000, Heppell et al. 2003, Ludwig 1998). To do otherwise with the limited data available would give the appearance of numerical precision without the reality of it Burgman et al. 1993, Caughley 1994, Cortes 1999, Morris and Doak 2002, Reed et al. 1998). As a result of these limits, we cannot quantify the effects of changes in abundance, reproductive success, and other vital rates on a sea turtle population's likelihood of surviving and recovering in the wild.

4.0 DESCRIPTION OF THE ACTION AREA

The action area is all the areas that will be affected directly or indirectly by the Western Pacific Pelagics Fisheries. These fisheries occur throughout the central, western, and northern Pacific Ocean, including inside the EEZ around U.S. islands in the Pacific. These are the islands of American Samoa (Tutuila, Rose Atoll, Swain's Island, and Manua group islands); Commonwealth of the Northern Mariana Islands (Saipan, Rota, Tinian, Farallon de Medinilla, Anatahan, Sarigan, Guguan, Alamagan, Pagan, Agrihan, Asuncion, Farallon de Pajaros); Hawaii (main and Northwestern Hawaiian Islands), Guam, and the largely uninhabited U.S. Pacific remote island areas comprised of Johnston Atoll, Kingman Reef, and Palmyra, Jarvis, Howland, Baker, Midway, and Wake Islands (see Figure II-15). Thus the action area, for purposes of this opinion, is the EEZs around the U.S. Pacific islands and the high sea waters where U.S. fishing vessels that target Pacific pelagic management unit species using longline, troll, and handline gear are managed under the Pelagics FMP.

4.1. Pelagic Fisheries in Hawaii

4.1.1. Hawaii Longline Fishery

The Hawaii longline fishery operates inside and outside the EEZ around the main Hawaiian islands and Northwestern Hawaiian Islands (NWHI). Longline fishing is prohibited inside the protected species zone surrounding the NWHI (50 nautical miles from the center geographical positions of Nihoa Island, Necker Island, French Frigate Shoals, Gardner Pinnacles, Maro Reef, Laysan Island, Lisianski Island, Pearl and Hermes Reef, Midway Island, and Kure Island) to protect monk seals (see Figure II-16). The area closed around the main Hawaiian Islands varies from 25 to 75 nautical miles seaward of the shore depending on the season, island, and direction of the facing shore. These closures are in place to alleviate potential gear conflicts among small boat handline/troll fishers, charter boat operators, recreational fishers, and longline fishers. From

Figure 4.1. Exclusive Economic Zones (EEZs) of the Pacific Islands. Western Pacific Regional Fishery Management Council managed areas are shaded. Source: NMFS, Pacific Islands Area Office.

February 1 through September 30 each year, longline fishing is prohibited up to 75 nautical miles around the main Hawaiian Islands in the portion of the EEZ seaward of Hawaii bounded by straight lines. From October 1 through the following January 31 each year, longline fishing is prohibited further inshore around the main Hawaiian Islands in the portion of the EEZ seaward of Hawaii (see Figure 4.3).

Hawaii-based longline vessels vary their fishing grounds depending on their target species. Most effort is to the north and south of the Hawaiian Islands between the equator and 40 ° N and longitudes 140 ° and 180 ° W.

4.1.2. Hawaii-based Troll Fishery

The Hawaii troll fishery, composed of commercial, troll, charter, and recreational/subsistence sectors, generally operates within the EEZ around the main Hawaiian Islands, between 4.9 miles to 53.5 miles offshore. Most of the trips occur within 25 miles from shore (NMFS, 2000). The charter trolling fleet typically operates about 7.5 miles from shore.

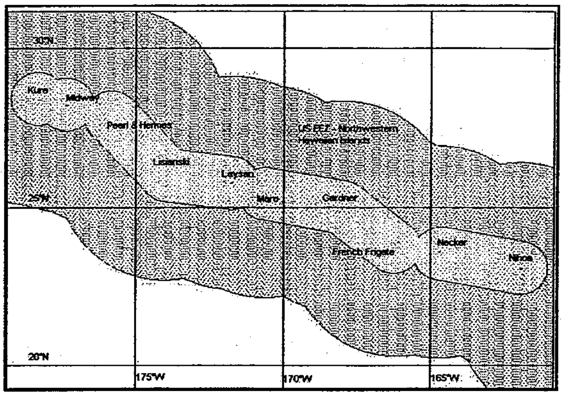

Figure 4-2. Protected species zone around the Northwestern Hawaiian Islands closed to longline fishing.

4.1.3. Hawaii-based Handline Fishery

The offshore handline fisheries occur between 35 to 200 nautical miles from shore, whereas the inshore handline fisheries occur between 5 and 14 miles from shore.

4.1.4. Hawaii-based Pole-and-line Fishery

The pole-and-line fishery operates within 25 miles offshore in the EEZ around the main Hawaiian Islands.

4.1.5. Troll/Handline fishery in the U.S. Pacific remote island areas (PRIA)

At present, there is no pelagic troll/handline fishery in the EEZ around the PRIA. It is expected that an active fishery would occur within 25 miles from shore.

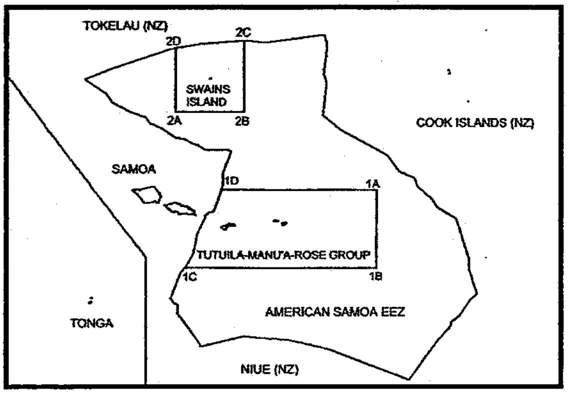

Figure 4.5. Areas around American Samoa closed to vessels greater than 50 feet in length.

4.2. Pelagic Fisheries in American Samoa

Most of the pelagic hook-and-line fishery based in American Samoa occurs within the EEZ, although vessels greater than 50 feet in length targeting PMUS are prohibited from fishing within approximately 50 nautical miles of the islands of American Samoa, including Tutuila, Manua group and Swains Islands, and Rose Atoll (see Figure II-19). The local, small (<40 ft in length) *alia* longline fleet and charter trolling vessels typically operate within 50 nm from the islands.

The American Samoa-based pelagic longline fleet, which until recently was comprised exclusively of *alia* less than 30 feet in length, generally fished within 25 nautical miles from shore. The recent entry of numerous large (>50 ft) longline vessels, most of which can range throughout the EEZ, has resulted in not only a dramatic increase in longline fishing effort but also a shift of fishing effort in waters between 50 and 200 nm from shore.

4.2.1. American Samoa Troll Fishery

The majority of the American Samoa troll fleet, which is composed of relatively small boats, fishes within 50 nautical miles from shore, although some vessels may fish as far out as 100 nautical miles.

4.3. Pelagic Fisheries in Guam

4.3.1. Guam-based troll (recreational, commercial, and charter) Fishery

The troll vessels in Guam are small, primarily recreational trolling boats that are either trailered to launch sites or berthed in marinas. The vessels generally fish inshore (within 25 nautical miles of

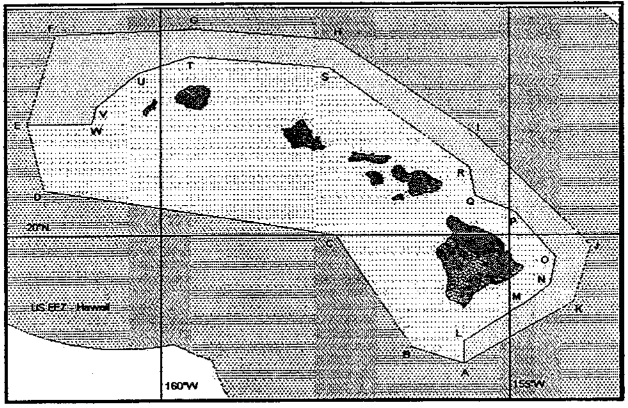

Figure 4.3. Closed areas to longline fishing around the main Hawaiian Islands. Distance from shore varies from 25 to 75 nautical miles. Vessels are required to fish further away from shore on the windward side (Northwest) of the islands from February 1 through September 30 each year.

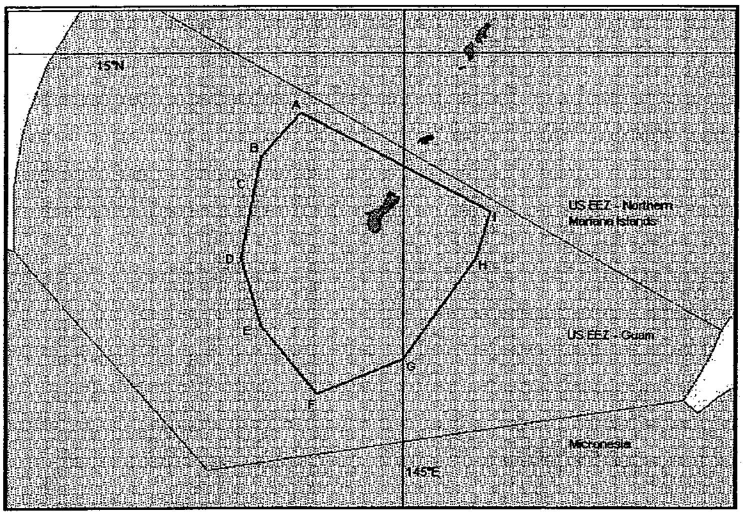

Figure 4.5. Closed area to longline fishing around the island of Guam.

shore) because of their small size. The larger vessels may fish further out within the EEZ.

4.3.2. Guam-based Longline Fishery

Figure II-20 depicts the 50-nm area around Guam that is closed to longline fishing. Vessels registered with longline general permits fish outside this closed area. There is no domestic longline fishery off Guam at this time.

4.4. Pelagic Fisheries of Commonwealth of Northern Mariana Islands

4.4.1. Commonwealth of Northern Mariana Islands Troll Fishery

The troll fishery occurs primarily between the island of Farallon de Medinilla and the island of Rota to the south. Most of the pelagic troll vessels are based on Saipan. They are small (generally <24 feet) and operate primarily within 20 nautical miles from shore. Larger vessels may fish further offshore within the EEZ. There is no domestic longline fishery in the Commonwealth of the Northern Mariana Islands at this time.

4.5. Western Pacific Tuna Purse Seine Fishery

The U.S. tuna purse seine fishery is managed under the Treaty on Fisheries Between the Governments of Certain Pacific Island States and the Government of the United States of America (the South Pacific Tuna Treaty). Although the fishery occurs in the action area and subject to management under the Magnuson-Stevens Fishery Conservation and Management Act, for the most part this fishery is not regulated under the Pelagics FMP. [Note: Pelagics FMP regulations prohibit domestic purse seine vessels from fishing within the 50-nm area closure around American Samoa].

5.0 STATUS OF LISTED RESOURCES 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
Hawaiian monk seal (*Monachus schauinslandi*)	Endangered
Humpback whale (*Megaptera novaeangliae*)	Endangered
Pacific right whale (*Eubalaena japonica*)	Endangered
Sei whale (*Balaenoptera borealis*)	Endangered
Sperm whale (*Physeter macrocephalus*)	Endangered
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

Except for the Hawaiian monk seal, no critical habitat has been designated for any of these threatened or endangered species in the Pacific Ocean. In May 1988, NMFS designated critical habitat for the Hawaiian monk seal out from shore to 20 fathoms in 10 areas of the Northwestern Hawaiian Islands. Critical habitat for these species includes "all beach areas, sand spits and islets, including all beach crest vegetation to its deepest extent inland, lagoon waters, inner reef waters, and ocean waters out to a depth of 20 fathoms around the following: Kure Atoll, Midway Islands, except Sand Island and its harbor, Lisianski Island, Laysan Island, Maro Reef, Gardner Pinnacles, French Frigate Shoals, Necker Island, and Nihoa Island" (50 CFR § 226.201). Some U.S. fisheries regulated under the Pelagics FMP fish in critical habitat areas of the Hawaiian monk seal (i.e., ocean waters out to 20 fathoms depth), although they do not adversely affect physical features identified as critical habitat. In addition, these fisheries do not target or incidentally catch prey species of the Hawaiian monk seals. Therefore, although the action area for the proposed fisheries includes the critical habitat for the endangered Hawaiian monk seals, the proposed action is not likely to adversely affect critical habitat that has been designated for the Hawaiian monk seal.

Although blue whales, fin whales, northern right whales, and sei whales are found within the action area and could potentially interact with the U.S. fisheries under the Pelagics FMP, there have been no reported or observed incidental takes of these species in these fisheries. Therefore, although the action area for the proposed fisheries includes the distribution of endangered blue whales, fin whales, Pacific right whales, and sei whales, the proposed action is not likely to adversely affect these species, which will not be considered further in this Opinion.

The endangered Hawaiian monk seal is currently found throughout the northwest Hawaiian Islands, specifically: Kure Atoll, Midway Islands, Pearl and Hermes Reef, Lisianki Island, Laysan Island, French Frigate Shoals, Gardner Pinnacles, Necker Island and Nihoa Island. These islands form a chain approximately 1,840 km long. Hawaiian monk seals are also occasionally found in the main Hawaiian Islands. The longline area closure around the northwest Hawaiian Islands that was instituted in 1991 (longline fishing prohibited within 50 nm of the northwest Hawaiian Islands and in 100 nm closed corridors connecting the non-contiguous closed circles) appears to have eliminated monk seal interactions with the Hawaii-based longline fleet, as there have been no observed or reported interactions with this fishery since then. In addition, there have been no reported interactions between Hawaiian monk seals and other fisheries under the Pelagics FMP. Therefore, although the action area for the proposed fisheries includes the distribution of the endangered Hawaiian monk seals, NOAA-Fisheries has determined that the proposed action is not likely to adversely affect the Hawaiian monk seal.

In 1991, one humpback was reported by an observer entangled in the mainline of a Hawaii-based longline vessel. The animal was released with trailing gear (Dollar 1991). The interaction occurred inside what is now the protected species zone (50 nautical miles) of the islands and atolls of the Northwestern Hawaiian Islands (Bob Harman, NMFS, personal communication, November, 2000). Another humpback whale was reported entangled in longline gear off Lanai (Nitta and Henderson 1993) and by whalewatch operators off Maui in 1993 (Hill and DeMaster 1999). Although the reports did not identify the gear as pelagic longline gear, both observations are believed to have been of the same whale.

Humpback whales favor waters less than 100 fathoms (183 meters) around the main Hawaiian Islands. The highest densities of humpback whales occur in the shallow-water, inter-island channels of the four-island region (Maui, Lāna'i, Moloka'i, and Kaho'olawe) and Penguin Bank (Hudnall 1978, Baker and Herman 1981, Mobley and Bauer 1991 *in* Mazzuca *et al.* 1998). Because humpback whales prefer shallower waters and the 1991 interaction occurred inside the 50 nautical mile area now closed to longline fishing, NOAA-Fisheries believes there is almost no likelihood of another interaction between the Hawaii-based longline fishery and a humpback whale.

NMFS has observed one sperm whale interaction by the Hawaii-based longline fishery. The event occurred in May 1999 inside the Northwestern Hawaiian Islands exclusive economic zone or EEZ (about 140 nautical miles north of Raita Bank) and the vessel was targeting swordfish (gear was set at night, lightsticks were used, and no line shooter was used). 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. This is the first reported interaction by the observer program since the Hawaii-based longline fleet has been monitored (1991). No other interactions between sperm whales and other fisheries have been reported under the Pelagics FMP.

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 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. NOAA-Fisheries acknowledges the possibility of interactions between the proposed fisheries (particularly the Hawaii-based longline fishery) and sperm whales but interprets the available data to conclude that future interactions are improbable. Therefore, NOAA-Fisheries does not anticipate another sperm whale interaction in the foreseeable future by the Hawaii-based longline fishery. As a result, although the action area for the proposed fisheries includes the distribution of endangered humpback whales and sperm whales, NOAA-Fisheries has determined that the proposed fisheries are not likely to adversely affect these whale species.

Based on observed and reported interactions between the Hawaii-based longline fishery and four species of sea turtles, NOAA-Fisheries has determined that the proposed action is likely to adversely affect green, leatherback, loggerhead, and olive ridley turtles. Therefore, formal consultation is required in order to analyze the effects of the proposed action on these listed species.

The following narratives summarize the current state of knowledge on the life history, distribution, and population trends of these sea turtle species and that NOAA-Fisheries expects may be incidentally taken as a result of the proposed action. These narratives 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 Pacific Basin and the factors affecting them in the action area are similar to those throughout their range in the Pacific Ocean.

5.1 Status of Listed Sea Turtles

For the purposes of this consultation, this Opinion focuses on the effects of the Pelagics FMP fisheries on sea turtle populations in the Pacific Ocean as distinct from their, as listed, global distribution. This approach is allowable based on interagency policy on the recognition of distinct vertebrate populations (Federal Register 61: 4722-4725). To address specific criteria outlined in that policy, sea turtle populations in the Pacific Ocean are geographically discrete from their populations in the Atlantic Ocean , for example, with limited genetic exchange (see NMFS and USFWS 1998a-e). 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, NMFS must make its final determination of the effect of the Pelagics 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 populations, which have been listed separately as endangered species, 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 over time; 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 stage to the next life-stage 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 Pelagics 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, hawksbill, 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 Pelagics 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 (NMFS and USFWS, 1998a-e) and are reviewed extensively in Eckert (1993).

Status Assessments 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, see Appendix A for details of these analyses) to assess the probable trend for the different populations. We had chosen 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. Absent robust, long-term data, we have to rely on simpler population models whose results require qualitative interpretation because of the uncertainty and assumptions underlying these models.

The "Dennis" model is a relatively simple model that relies on 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 (see Appendix A for details of these conversions). 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 (see Appendix A).

Table 3.1 lists the products of our analyses using the Dennis model, 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). In the narratives that follow, we provide a summary table of the results with supporting narrative when the available census data allowed us to conduct more detailed assessments. 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. Although the results of these analyses require qualitative interpretations, they provide important insights into the probable status and future trend of the different sea turtle species.

Table 3.1. 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 (σ^2)
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 a measure of a 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, $\mu > 0$, then most population trajectories will increase; if $\mu < 0$, then most population trajectories will decline (Morris and Doak 2002).

The variance in a population's mean log growth rate (σ^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.

These two statistics can be used to estimates other statistics that population biologists commonly use to characterize a population's status and trend: finite rates of population increase (or decrease) and intrinsic or continuous rates of increase (or decrease). A population's finite rate of increase (λ) captures a population's growth rate or the amount by which a population size multiplies from year to year. In the face of stable environmental conditions, this growth rate would be constant and a population would increase geometrically ($\lambda > 1$), decrease geometrically ($\lambda < 1$), or remain the same ($\lambda = 1$). The continuous or intrinsic rate of population increase is another statistic that is important to population biology; it is the natural logarithm of the finite rate of increase and represents rates of population growth over short intervals of time. 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 following narratives for the species.

Green Turtles

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.

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).

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* NMFS and USFWS, 1998a). Nesting females at the Bramble Cay rookery in Queensland, Australia averaged 105.9 cm CCL (Limpus *et al.*, 2001).

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 20°C 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 18°C. 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 20°C (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 (NMFS 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 (NMFS, 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).

Life Cycle and Population Dynamics

Figure 3.1 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 3.1). 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 3.1. Life-cycle graph of the green turtle

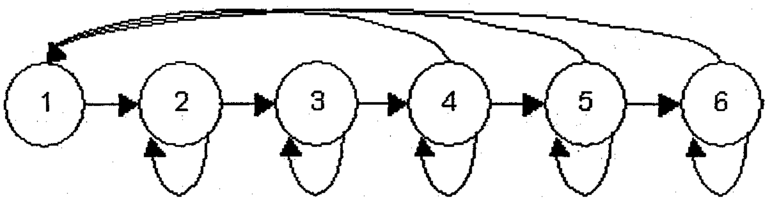

Table 3.1: Stage specific demographic information for the southern Great Barrier Reef green turtle (Chaloupka 2002)

Life Stage	Name of stage	Mean Stage Duration (# years)	Stable Stage Structure	Survival Probability (Px)	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 3.1 indicated that the southern Great Barrier Reef green turtle population has a finite population growth rate (λ) 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 λ, of this stage structure confirms the general relationships in this life cycle. Table 3.2 includes the elasticities of the vital rates of each life stage in the green turtle life cycle.

Table 3.2. Stage elasticities (Chaloupka, 2002)

Life Stage	Survival Rate	Transition Rates	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 (λ). Changes in the survival rates of the 3rd, 4th, and 5th life stages have the next highest proportional effect on λ, 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 Pelagics 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 (λ), 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 λs significantly lower than the southern Great Barrier Reef green turtle population, meaning that these populations would be declining.

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 (NMFS 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 be 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 NMFS 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; Craig et al. 2004).

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 15°N to 5°S along 90°W, and between the Galapagos Islands and Central American Coast (NMFS 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* NMFS and USFWS, 1998a; P. Dutton, NMFS, personal communication, March, 2002); however, the possibility exists that some are from Hawaii (P. Dutton, NMFS, 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).

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.

Pacific Ocean

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.

Australia

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 (NMFS 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 3.2). 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 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 (NMFS and USFWS, 1998a).

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, NMFS, 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 (NMFS 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.

Mexico

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 3.3)..

Table 3.11. 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 (σ^2)	0.584556
Upper 95% confidence interval	0.637932
Lower 95% confidence interval	0.342150

Our analyses of estimates of the number of female green turtles that nest at Colola Beach suggest that most trajectories of this population would be expected to decline slightly (μ = -0.261), although the population seems to be capable of growth (λ = 1.30, which is greater than 1) and may experience short periods of growth. 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

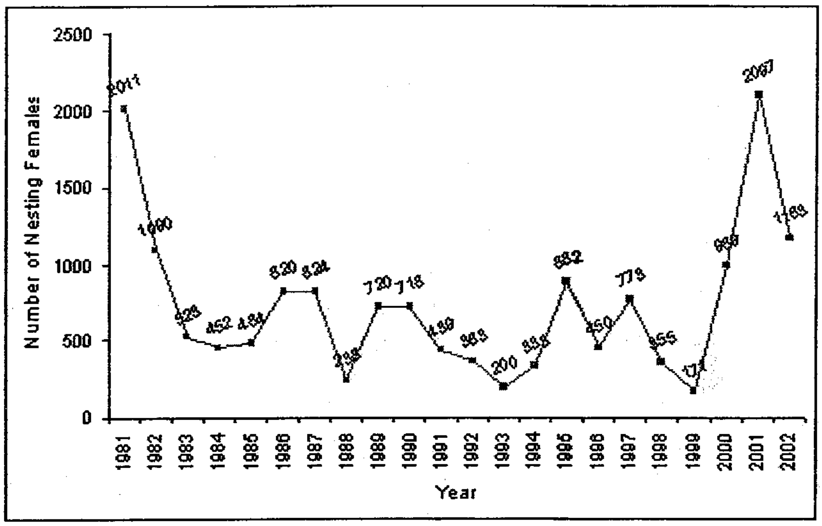

Figure 3.3. Number of Nesting Female Green Turtles at Colola beach, Michoacan, Mexico (from Alvarado-Diaz, personal communication, October, 2003).

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* NMFS 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, NMFS, 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 about 5% of the nests were poached at Colola during this season, about 50% of the nests at Maruata were poached (Delgado and Alvarado, 1999).

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 (NMFS 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 3.3). 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 3.3. 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* NMFS and USFWS, 1998a).

Green turtles encountered by U.S. vessels managed under the Pelagics 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 longline fishery based off the west coast. Genetic analyses revealed that the one green turtle observed taken in the CA/OR drift gillnet fishery originated from the eastern Pacific stock, most likely a Mexican nesting beach (P. Dutton, personal communication, January, 2000). Green turtles taken in the eastern tropical Pacific 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 on observer trips 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, NMFS, personal communication, December, 2003).

Hawksbill Turtles

Global Status

The hawksbill turtle is listed as endangered under the ESA. Under Appendix I of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), the hawksbill is identified as "most endangered." Anecdotal reports throughout the Pacific indicate that the current population is well below historical levels. In the Pacific, this species is rapidly approaching extinction primarily due to the harvesting of the species for its meat, eggs, and shell, as well as the destruction of nesting habitat by human occupation and disruption (NMFS, 2001).

Physical Description

The following characteristics distinguish the hawksbill from other sea turtles: two pairs of prefrontal scales; thick, posteriorly overlapping scutes on the carapace; four pairs of costal scutes; two claws on each flipper; and a beak-like mouth. The carapace is heart-shaped in very young turtles, and becomes more elongate or subovate with maturity. Its lateral and posterior margins are sharply serrated in all but very old individuals. The epidermal scutes that overlay the bones of the shell are the tortoiseshell of commerce. They are unusually thick, and overlap posteriorly on the carapace in all but hatchlings and very old individuals. Carpacial scutes are often richly patterned with irregularly radiating streaks of brown or black on an amber background. The scutes of the plastron of Atlantic hawksbills are usually clear yellow, with little or no dark pigmentation. The soft skin on the ventral side is cream or yellow, and may be pinkish-orange in mature individuals. The scales of the head and forelimbs are dark brown or black with sharply defined yellow borders. There are typically four pairs of inframarginal scutes. The head is elongate and tapers sharply to a point. The lower jaw is V-shaped.

The hawksbill is a small to medium-sized sea turtle. In the U.S. Caribbean, nesting females average about 62-94 cm in straight carapace length. Weight is typically to 80 kg in the wider Caribbean, with a record weight of 127 kg. Hatchlings average about 42 mm straight carapace length and range in weight from 13.5-19.5 g.

General Distribution

The hawksbill occurs in tropical and subtropical seas of the Atlantic, Pacific and Indian Oceans. The species is widely distributed in the Caribbean Sea and western Atlantic Ocean, with representatives of at least some life history stages regularly occurring in southern Florida and the northern Gulf of Mexico (especially Texas); in the Greater and Lesser Antilles; and along the Central American mainland south to Brazil. Within the United States, hawksbills are most common in Puerto Rico and its associated islands, and in the U.S. Virgin Islands. In the continental U.S., the species is recorded from all the gulf states and from along the eastern seaboard as far north as Massachusetts, with the exception of Connecticut, but sightings north of Florida are rare. Hawksbills are observed in Florida with some regularity on the reefs off Palm Beach County, where the warm Gulf Stream current passes close to shore, and in the Florida Keys. Texas is the only other state where hawksbills are sighted with any regularity. Most sightings involve post-hatchlings and juveniles. These small turtles are believed to originate from nesting beaches in Mexico.

Nesting within the Caribbean dependent areas of the United States occurs principally in Puerto Rico and the U.S. Virgin Islands, the most important sites being Mona Island and Buck Island. Nesting also occurs on other beaches of St. Croix, and on Culebra Island, Vieques Island, mainland Puerto Rico, St. John and St. Thomas. Within the continental United States, nesting is restricted to the southeast coast of Florida and Florida Keys.

In the U.S. Pacific Ocean, there have been no hawksbill sightings off the west coast. Hawksbills have been observed in the Gulf of California as far north as 29°N, throughout the northwestern states of Mexico, and south along the Central and South American coasts to Columbia and Ecuador. In the Hawaiian Islands, nesting occurs in the main islands, primarily on several small sand beaches on the Islands of Hawaii and Molokai. Two of these sites are at a remote location in the Hawaii Volcanos National Park.

Along the far western and southeastern Pacific, hawksbill turtles nest on the islands and mainland of southeast Asia, from China to Japan, and throughout the Philippines, Malaysia, Indonesia, Papua New Guinea, the Solomon Islands (McKeown, 1977) and Australia (Limpus, 1982). Along the eastern Pacific rim, hawksbill turtles were common to abundant in the 1930s (Cliffton *et al.*, 1982). By the 1990s, the hawksbill turtle was rare to absent in most localities where it was once abundant (Cliffton *et al.*, 1982; Cornelius, 1982).

Hawksbills utilize different habitats at different stages of their life cycle. Posthatchling hawksbills occupy the pelagic environment, taking shelter in weedlines that accumulate at convergence points. Hawksbills reenter coastal waters when they reach approximately 20-25 cm carapace length. Coral reefs are widely recognized as the resident foraging habitat of juveniles, subadults and adults. This habitat association is undoubtedly related to their diet of sponges, which need solid substrate for attachment. The ledges and caves of the reef provide shelter for resting both during the day and night. Hawksbills are also found around rocky outcrops and high energy shoals, which are also optimum sites for sponge growth. Hawksbills are also known to inhabit mangrove-fringed bays and estuaries, particularly along the eastern shore of continents where coral reefs are absent. In Texas, juvenile hawksbills are associated with stone jetties (Hidebrand 1987, Amos 1989).

Tag return data (Pritchard and Trebbau 1984) and recent genetic studies (Bowen *et al.*, 1996) suggest that individual foraging areas support hawksbills from distant breeding populations rather than just from nearby rookeries. They are found in all tropical seas between about 30°N and 30°S latitudes (NMFS and USFWS, 1998b). They are generally associated with coral reefs or other hard substrate structures close to shore where they feed on sponges and small crustaceans.

Life Cycle and Population Dynamics

The best estimate of age at sexual maturity for hawksbill turtles is about 20 to 40 years (Chaloupka and Limpus, 1997; Crouse, 1999a). Boulon (1994) estimated that juvenile hawksbills from the U.S. Virgin Islands would require between 16.5 and 19.3 additional years to reach maturity after entering nearshore habitats at several years of age at 21.4 cm straight carapace length.

Growth rates within benthic stage (juvenile turtles which have returned from pelagic developmental habitats) Australian hawksbill turtles are sex dependent with the female growing faster. Maximal growth rates for both males and females occurred at 60 cm curved carapace length (CCL) and then declined to minimal rates of growth as the turtles neared maturity at 80 cm CCL (Chaloupka and Limpus, 1997). The growth rates of Australian hawksbills appear to be less than those of Caribbean turtles, indicating geographic variation in growth.

Data on the transition rates between life stages are unavailable for hawksbill turtles. As a result, we were unable to analyze the stage structure of this species to estimate its finite population growth rate (λ) or the elasticities of the various life stages. The typical population structure for long-lived, late-maturing species like hawksbill turtles would be expected to have 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.

Biological Characteristics

There is little information available on the biology of hawksbills most likely because they are sparsely distributed throughout their range and they nest in very isolated locations (Eckert, 1993).

Diet
Hawksbills have a relatively unique diet of sponges (Meylan, 1985; 1988). While data are somewhat limited on diet in the Pacific, it is well documented in the Caribbean where hawksbill turtles are selective spongivores, preferring particular sponge species over others (Dam and Diez, 1997b).

Diving Behavior
Foraging dive durations are often a function of turtle size with larger turtles diving deeper and longer. At a study site also in the northern Caribbean, foraging dives were made only during the day and dive durations ranged from 19-26 minutes in duration at depths of 8-10 m. At night, resting dives ranged from 35-47 minutes in duration (Dam and Diez, 1997a).

Life History/Reproduction
As hawksbill turtle grow from juveniles to adults, data suggest that the turtle switches foraging behaviors from pelagic surface feeding to benthic reef feeding (Limpus, 1992). Within the Great Barrier Reef of Australia, hawksbills move from a pelagic existence to a "neritic" life on the reef at minimum CCL of 35 cm. The maturing turtle establishes foraging territory and will remain in this territory until it is displaced (Limpus, 1992). As with other sea turtles, hawksbills will make long reproductive migrations between foraging and nesting areas (Meylan, 1999), but otherwise they remain within coastal reef habitats. In Australia, juvenile turtles outnumber adults 100:1. These populations are also sex biased with females outnumbering males 2.57:1 (Limpus, 1992).

Although hawksbill nesting is broadly distributed, at no one place do hawksbills nest in large numbers, and many areas have experienced notable declines. Hawksbills utilize both low- and high-energy nesting beaches in tropical oceans of the world. Both insular and mainland nesting sites are known. Hawksbills will nest on small pocket beaches, and, because of their small body size and great agility, can traverse fringing reefs that limit access by other species. They exhibit a wide tolerance for nesting substrate type. Nests are typically placed under vegetation.

There is much variation in clutch size from site to site and among sizes of turtles, with the larger turtles laying the largest clutches. Known clutch size in the Pacific averages 130 eggs per clutch, around 3 clutches per year, and anecdotal reports indicate that hawksbill remigration intervals average around two years (Eckert, 1993; NMFS and USFWS, 1998b). Hawksbills nest throughout the insular tropical Pacific, though only in low density colonies. In the Campbell Island colony of northeastern Australia, nesting females average 83.2 cm CCL, weigh 51.6 kg and lay three clutches of eggs 14 days apart. Average clutch size was 132 eggs (Limpus *et al.*, 1983). In Samoa, hawksbill nests averaged 149.5 eggs.

Mrosovsky *et al.* (1995) evaluated the effect of incubation temperature on sex determination in hawksbill hatchlings. Incubation temperatures warmer than approximately 29.2°C produced females, while cooler temperatures produced males.

Migration
Like other sea turtles, hawksbills are highly migratory, although they may be less of a long-distant migrant. An adult female tagged in its foraging ground in the Torres Strait was observed nesting 322 days later in the Solomon Islands, a distance of over 1,650 km (Pritchard and Trebbau 1984). Another female traveled 1,400 km from the Solomon Islands to its foraging grounds in Papua New Guinea (Parmenter 1983).

Population Status and Trends

The hawksbill is a solitary nester, and thus, population trends or estimates are difficult to determine. There are no world population estimates for hawksbill turtles, but a minimum of 15,000 to 25,000 females are thought to nest annually in more than 60 geopolitical entities (Groombridge and Luxmoore 1989). Moderate population levels appear to persist around the Solomons, northern Australia, Palau, Persian Gule islands, Oman, and parts of the Seychelles (Groombridge 1982). In a more recent review, Groombridge and Luxmoore (1989) list Papua New Guinea, Queensland, and Western Australia as likely to host 500-1,000 nesting females per year, while Indonesia and the Seychelles may support >1,000 nesting females. The largest known nesting colony in the world is located on Milman Island, Queensland, Australia where Loop (1995) tagged 365 hawksbills nesting within an 11 week period. With the exception of Mexico, and possibly Cuba, nearly all Wider Caribbean countries are estimated to receive <100 nesting females per year (Meylan 1989).

Hawksbills appear to be declining throughout their range. By far the most serious problem hawksbill turtles face is the harvest by humans, while a less significant threat, but no less important, is loss of habitat due to expansion of resident human populations and/or increased tourism development. Dramatic reductions in the numbers of nesting and foraging hawksbills have occurred in Micronesia and the Mexican Pacific coast, probably due largely to technological advances in fishing gear, which facilitate legal and illegal harvest. In addition, the hawksbill tortoiseshell trade probably remains an important contributing factor in the decline of the hawksbill. Although the Japanese market was closed in 1994, southeast Asia and Indonesia markets remain lucrative (NMFS and USFWS, 1998b). In addition to the demand for the hawksbill's shell, there is a demand for other products including leather, oil, perfume, and cosmetics. Prior to being certified under the Pelly Amendment, Japan had been importing about 20 metric tons of hawksbill shell per year, representing approximately 19,000 turtles. A negotiated settlement was reached regarding this trade on June 19, 1992. The hawksbill shell commands high prices (currently $225/kilogram), a major factor preventing effective protection[9].

Distribution and Abundance of Nesting Females in the Pacific Ocean

Generally, in the Pacific, the largest nesting concentrations of hawksbills occur on remote oceanic islands off Australia (Torres Strait), while remote beaches in the Solomon Islands, Papua New Guinea, Indonesia, and Malaysia serve as less significant sites. Otherwise, hawksbill nesting does not occur in abundance in the Pacific. Throughout Micronesia, hawksbill nesting is in decline, with Palau representing the highest activity, with conceivably as few as 20 nesting females per year (NMFS and USFWS, 1998b). In Japan, nesting is very rare and is confined to the southern islands. Hawksbill nesting also occurs in Vietnam and China, although the status in these areas is unknown. Nesting is widespread throughout the Philippines, although the sites are relatively poorly known, and population abundance has not been quantified (Eckert, 1993).

Australia

In Queensland, Australia, there are two genetic stocks of hawksbills, with about 4,000 females nesting annually in the area. A study began at Milman Island to look at hawksbill nesting trends; researchers have documented a three percent per annum decline in the number of females nesting and a four percent decline in the clutches laid over the past ten years. Anthropogenic threats include hunting, ingestion and entanglement in marine debris, and disease (Dobbs, 2002).

Palau

The Palau nesting population of hawksbills is the largest in Oceania north of the equator; nesting is concentrated on small beaches of the Rock Islands between Koror and Peleliu islands (Maragos 1991). This population is severely stressed by chronic egg poaching and the hunting of turtles for jewelry and crafts (Maragos 1991). Residents are nearly unanimous in their opinion that nesting numbers are down significantly during their lifetimes. Maragos (1991) reported an average of 58 nests found per year (1982-1990), of which 76% were identified as "nests without eggs" or nests that were illegally poached. The annual number of nests in the Rock Islands might approach one hundred under the most favorable of circumstances. This would represent 20-25 nesting females per season, assuming 4-5 nests per turtle per season. If 40% of adult female hawksbills return to nest each year, given an average remigration interval of 2.5 years for the population, then approximately 50-60 adult females might remain in the Rock Island nesting population today.

Fiji

In Fiji, hawksbill nesting is widely dispersed and populations are small. In addition, the nesting populations at most sites have been reduced to 10-20% during the last five years (Batibasaga, 2002). Most nesting takes place on the eastern and southern parts of the Fiji Islands, particularly Namena Island, which used to have over 100 nests per season in the early 1970s. Currently, it is the main nesting site for hawksbills in Fiji, with approximately 40 nests per season between 1995 and 1999 (Batibasaga, 2002). The breeding population of hawksbills in Fiji is estimated to be

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[9] http://www.nmfs.noaa.gov/prot_res/species/turtles/hawksbill.html

approximately 120-150 turtles (*in* Rupeni *et al.*, 2002). While there was a moratorium on killing sea turtles instituted in Fiji in 1997, this ban ended after December, 2000, so harvesting of turtles remains a significant threat to hawksbills.

Malaysia

In Malaysia, hawksbills do not nest in large numbers, and likely decades of excessive egg collection and exploitation is the primary cause of the decimation of sea turtle populations in this country. Liew (2002) reports that there are approximately 500 hawksbill nests per year in the Sabah Turtle Islands and the trend appears to be stabilized. Basintal (2002), however, notes a decline in hawksbill nesting in this region since 1995, with less than 300 nests in 2001. At Gulisaan Island, where 87% of the hawksbill nesting takes place in the Turtle Islands, severe erosion is taking place, so this decreasing trend will likely continue. In Malacca, there are approximately 250 nests per year, and appears to be stable. In Terengganu, there has been a significant decrease in the number of hawksbill nestings in the last 20 years: from a high of 120 nests in 1986 to an average of around 20 nests per years from 1994 through 2000 (Liew, 2002).

Indonesia

In Indonesia, although hawksbill populations have been declining, they can still be found nesting throughout the country in significant numbers. Important hawksbill rookeries include the Anambas and Natuna-Riau islands; Lima Momperang; Pesemut-Belitung, Segamat Islands, Lampung; South of Ujungpandang; Bira-Birahan, and the Derawan Islands, East Kalimantan (*in* Dermawan, 2002). Nesting trends of hawksbills have been monitored since the early 1980s at three sites: Alas Purwo National Park, East Java, Jamursba-Medi beach, Papua; and Sukamade beach, Meru Betiri, East Java. At Meru Betiri, hawksbill nestings have declined significantly since 1980, with very low numbers nesting in the last decade, while the trends for the other two sites appear to fluctuate over the past two decades, but appear stable (Dermawan, 2002).

Thailand

In Thailand, hawksbills nest on the Khram Islands, in the Gulf of Thailand, and since their nesting areas have been protected and controlled since 1950, the population of nesting females has remained stable. Since 1973, there are approximately 100 hawksbill nests per year at the Khram Islands (Charuchinda *et al.*, 2002).

American Samoa

For American Samoa, based on interviews, Tuato'o-Bartley *et al.* (1993) estimated 50 nesting female hawksbills per year on Tutuila and 30 nesting females per year on the Manu'a island group of Ofu, Olosega and Ta'u, using an average 2.8 nesting turtles per active beach. A total of 120 nesting females were estimated throughout American Samoa. However, since local people almost always seem to underestimate individual fecundity (numbers of clutches per female), the actual number of turtles nesting at Tutuila and Manu'a could be significantly lower than Tuato'o-Bartley's estimates. Given an estimate of 120 nesting females, recent records indicate that there is a decline in the number of females nesting annually, based on confirmed nests and clutches of hatchlings. (Utzurrum, 2002).

CNMI

There are no reports of hawksbills nesting in the Commonwealth of the Northern Mariana Islands (CNMI) (Pritchard, 1982a). This is partly because there is a long history of occupation on the more southern islands of Saipan, Rota, and Tinian, and partly because almost no hawksbill nesting surveys of small pocket beaches have ever been done in remote areas of the CNMI. However, lack of evidences does not rule out the possibility of hawksbills nesting at low levels at unknown locations.

Hawaii

Within the State of Hawaii, hawksbill turtles are known to nest on the Hawaiian Islands of Maui, Molokai, and Hawaii. Two nesting sites are located in the Hawaii Volcanoes National Park (Balazs *et al.*, 1992; Katahira *et al.*, 1994). In surveys conducted between 1989 and 1993, 18 hawksbill turtles were tagged and 98 nests documented (NMFS and USFWS, 1998b). Although total population numbers and trends in abundance are not known for the Hawaiian population of hawksbill turtles, probably no more than 35 females nest annually on all beaches combined (J. Wetherall, NMFS, personal communication, 1999). The peak nesting occurs from late July to early September (Katahira *et al.*, 1994). Recent nesting activity has occurred at Kealia Beach on Maui. There are no records of nesting hawksbill turtles or reported observations of their occurrence near the Main Hawaiian Islands, although they may have occupied the region in the past. Hawksbill turtles appear to prefer nesting sites with steep beaches and coarse sand, and this may explain, in part, their presence in the main Hawaiian Islands.

Only the ETP purse seine fishery has observed the take of hawksbills. Unfortunately, turtles in this fishery are not sampled to determine nesting origin. Nonetheless, because of the location of fishing effort, hawksbills taken in this fishery likely originate from eastern Pacific nesting beaches.

Leatherback Turtles

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).

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 (NMFS 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* NMFS and USFWS, 1998c), while adult female leatherback turtles nesting in eastern Australia averaged 162 cm CCL (Limpus, *et al.*, 1984, *in* NMFS and USFWS, 1998c). Leatherbacks in Papua, Indonesia and Papua New Guinea averaged 161 cm CCL (Starbird and Suarez, 1994; Hirth *et al.*, 1993, respectively).

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 in all pelagic waters of the Pacific Ocean between 71°N and 47°S latitude and in all other major pelagic ocean habitats (NMFS 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 26°C, which should generally place them outside of areas in which longline swordfish fleets operate (Eckert, 1999b; Eckert, 2002).

Life Cycle and Population Dynamics

Figure 3.4 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 3.4). 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 3.4. Life-cycle graph of the leatherback turtle (based on Chaloupka 2001)

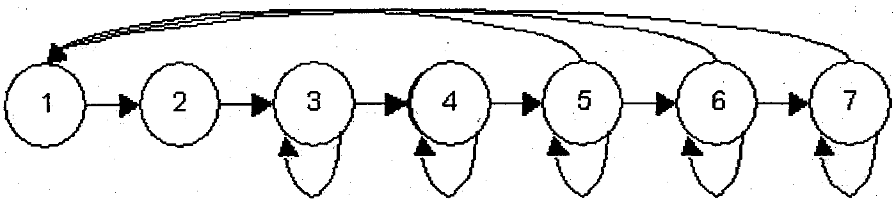

Table 3.4: Stage structure and vital rates for leatherback turtles (Chaloupka, 2001; 2002)

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 difficult. 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 (λ) 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 a final and irreversible.

The dynamics of most leatherback turtle populations today, however, 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 λs 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.

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 (NMFS 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[10] 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* NMFS 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%

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

of the time the leatherback turtles were at depths less than 80 meters (Peter Dutton, NOAA Fisheries, personal communication, January 2004).

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 (NMFS 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[11], 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

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[11] 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.

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 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 8°N, 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, NMFS, 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, NMFS, 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, NMFS, 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, NMFS, 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, NMFS, 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, NMFS, 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.

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.

Atlantic Ocean/Caribbean Sea

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 (NMFS 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 (NMFS 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* NMFS 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 (NMFS 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 NMFS 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 (NMFS 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; NMFS 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. 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

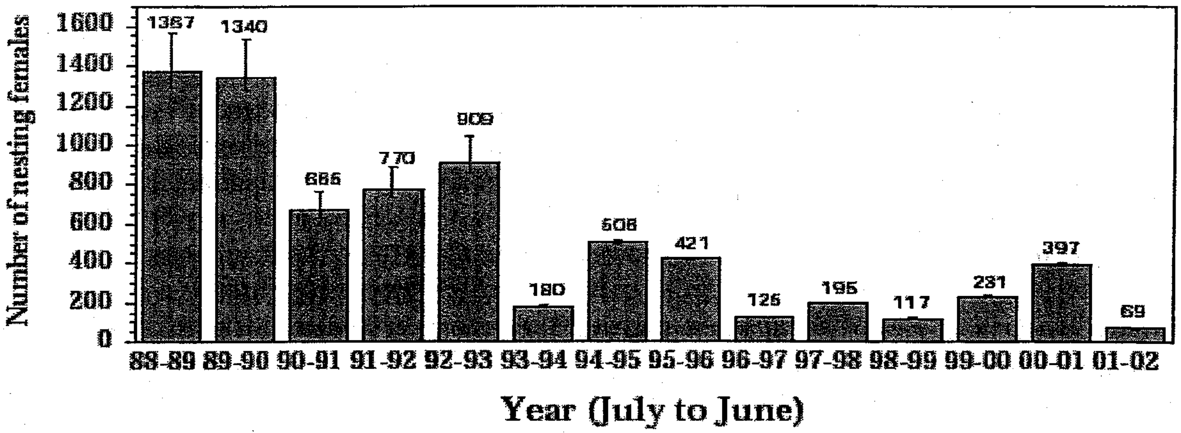

Figure 3.5. **Number of female leatherbacks nesting at Playa Grande (Las Baulas, Costa Rica) from 1988-2002. (Source: R. Reina, Drexel University, personal communication, September, 2003).**

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 III-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).

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 29.5°C produce female hatchlings, while below 29.5°C, 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 3.11. 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 (σ^2)	0.226610	0.268697
Upper 95% confidence interval	0.653270	0.774597
Lower 95% confidence interval	0.113718	0.134839

The results of our analyses of the lower and upper estimates of the number of female leatherback turtles that nest at Playa Grande suggest that most trajectories of this population would be expected to decline ($\mu = -0.229501$) although the population is capable of growth ($\lambda = 1.17$, which is greater than 1) and may grow in some years. 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.

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[12] 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:

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."

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[12] 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).

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 3.5.

Table 3.5. 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 3.6. 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.

Table 3.6. 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 yet	not available yet

[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).

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 3.7). 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. NMFS has committed funding for three years to help implement these objectives (Dutton *et al.*, 2002).

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).

Table 3.7. 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

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).

Estimates of the Total Abundance of Nesting Females in the 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:

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

Western Pacific Nesting Populations of Leatherbacks

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 3.8). 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 3.8. 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, see Appendix A for details of these analyses) to assess the probable trend for this population (see Table 3.9 for results).

Table 3.9. 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 (σ^2)	0.776462
Upper 95% confidence interval	2.115806
Lower 95% confidence interval	0.399266

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 ($\lambda = 0.97$, which is less than 1, and $\mu = -0.282579$), 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.

Indonesia

In Indonesia, leatherback turtles 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 3.9). 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 3.9 (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 3.9). 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.

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[13] 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[14] 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

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

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

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).

Table 3.9. **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.

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 3.10).

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, NMFS, personal communication, October 2002), and coastal patrols are currently being conducted to prevent disturbance and exploitation of the beach (Hitipeuw, 2003b).

Table 3.10. 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

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 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, see Appendix A for details of these analyses) to better assess the probable trend for this population (see Table 3.11 for results).

Table 3.11. 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 (σ^2)	0.226610	0.268697
Upper 95% confidence interval	0.653270	0.774597
Lower 95% confidence interval	0.113718	0.134839

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 most population trajectories are declining slightly ($\mu = -0.0484$). Projecting these results over 25-, 50-, and 100-year intervals suggest that this population has a low risk of extinction (declining to 1 or 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 ears 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[15]. 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, NMFS, 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).

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[15] 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.

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

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.

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 was found to originate from an eastern Pacific nesting beach, the north Pacific Ocean is 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. Leatherbacks have been observed taken by the ETP purse seine fishery; however, genetic data was not collected from any turtles in this fishery.

Loggerhead Turtles

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.

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).

General Distribution

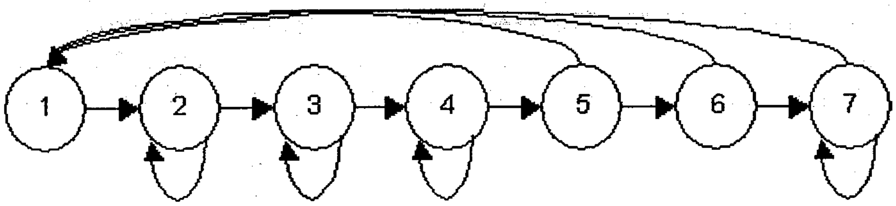
Loggerheads are circumglobal, inhabiting continental shelves, bays, estuaries, and lagoons in temperate, subtropical, and tropical waters. Major nesting grounds are generally located in temperate and subtropical regions, with scattered nesting in the tropics (*in* NMFS and USFWS, 1998d).

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, NMFS 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 (NMFS 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.

Life Cycle and Population Dynamics

Figure 3.6 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 3.11). 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 3.6. Life-cycle graph of the loggerhead turtle (Crouse *et. al.*, 1987)



Numerical analyses of the survival rates, transition rates, and fecundities in Table 3.11 indicated that the modeled loggerhead turtle population has a finite population growth rate (λ) 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[16] 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

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[16] 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).

structure is probably an appropriate surrogate of other declining loggerhead populations because of its atypical structure.

Table 3.11: 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

[1] Bjorndal, *et al.* (2001) indicate that the pelagic stage may last from 6.5 to 11.5 years.

Elasticity analyses for the stage matrix of this population support these general conclusions (Table 3.12). 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 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.

In contrast to the rates provided in Crouse, *et al.* (1987; Table 3.11), 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 3.13). 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 3.12. 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

Table 3.13. 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.

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 20°C at 28°N and another of 17°C at 32°N.

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* NMFS 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* NMFS 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, NMFS, 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).[17]

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 17° and 20°C 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 (17°C), surface chlorophyll and an eastward geostrophic current of about 4 centimeters/second (cm/sec). Three others were associated with a warmer front (20°C), 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 28°N and 40°N. During January through June the 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 15° and 25° 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

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[17] 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).

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.

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 29° N (approximately 7,500 nests in 1998); (2) a south Florida nesting subpopulation, occurring from 29° 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) (NMFS 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%) (NMFS SEFSC, 2001), there are confidence intervals about these estimates that include no growth[18]. 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%; NMFS SEFSC 2001).

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

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[18] Meta-analyses conducted by NMFS' 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.

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 (NMFS 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, dredge, and trap fisheries.

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 (NMFS 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.

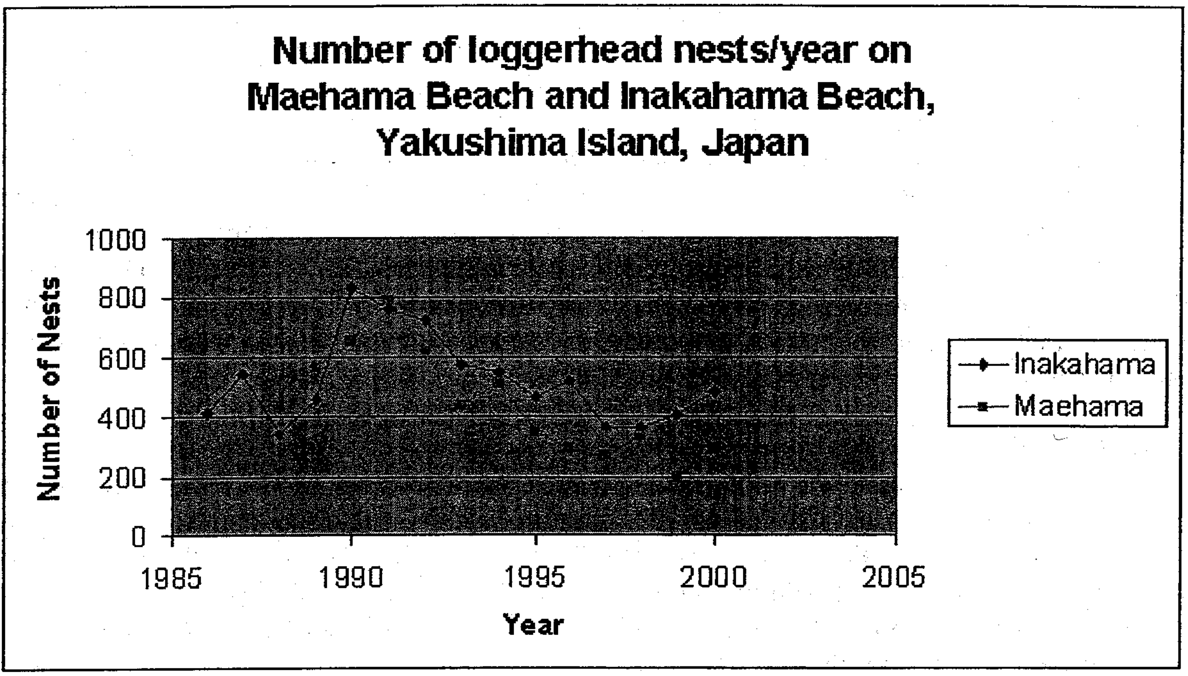
In Japan, loggerheads nest on beaches across 13 degrees of latitude (24°N to 37°N), from the mainland island of Honshu south to the Yaeyama Islands, which appear to be the southernmost extent of loggerhead nesting in the western North Pacific. Researchers have separated 42 beaches into five geographic 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 in Japan, 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 3.7 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).

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 Table 2 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 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 (27.6°S) northwards to Bustard Head (24.0°S) 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 (NMFS 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, NMFS, 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 (NMFS and USFWS, 1998d).

Loggerhead mortality from human activities in the Pacific Ocean is not well-documented except for estimates based on NMFS 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, NMFS, 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.

Olive Ridley Turtle

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.

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 (NMFS 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.

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.

Life Cycle and Population Dynamics

Figure 3.8 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 3.14 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 3.8. Life-cycle graph of the olive ridley sea turtle

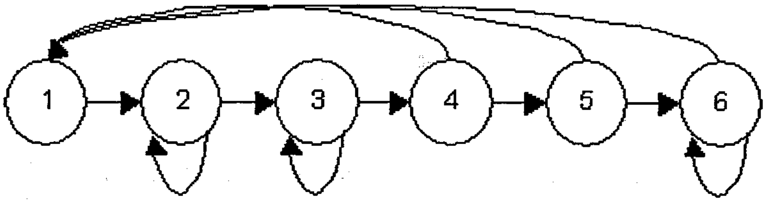


Table 3.14: Stage structure and vital rates for olive ridleys (based on Marquez (1981, 1994) data for Kemp's ridley turtles)

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 (λ) 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 λ and life stage elasticities of these populations are likely indicative of declining populations (λs less than 1, and changed proportional importance of different life stage elasticities on λ). 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 λ, some of these aggregations could slow their rate of increase, begin to decline, or increase the rate of their decline.

Biological Characteristics

Diet

Olive ridleys feed on tunicates, salps, crustaceans, other invertebrates and small fish. Montenegro *et al.* 1986 (*in* NMFS 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* NMFS and USFWS, 1998e).

Dive Behavior

Olive ridleys have been caught in trawls at depths of 80-110 meters (NMFS 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* NMFS 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 140°W. 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 31°N, with SSTs of 23° to 28°C (primarily in areas with SSTs of 24° or 27°C). 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 14° and 28°N, but not in the area between 20° and 24° 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.

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* NMFS 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. (NMFS 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.

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 arribada*s* 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 (NMFS 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* NMFS 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* NMFS 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 3.15 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 3.15. Consensus estimates for nesting populations in Gahirmatha, derived from multiple sources (Source: Shanker *et al.*, 2003b)

Year	Consensus estimate[1]	Comments	Year	Consensus estimate[1]	Comments
1975-1976	158,000		1989-1990	200,000	1 arribada
1976-1977	150,000		1990-1991	350,000	2 arribadas
1977-1978	150,000	1 arribada	1991-1992	320,000	2 arribadas
1978-1979	133,000	1 arribada	1992-1993	350,000	?
1979-1980	218,000	1 arribada	1993-1994	350,000	2 arribadas
1980-1981	191,000	1 arribada	1994-1995	340,000	?
1981-1982	0	no arribada	1995-1996	200,000	1 arribada
1982-1983	200,000	2 arribadas	1996-1997	0	no arribada
1983-1984	300,000	2 arribadas	1997-1998	0	no arribada
1984-1985	280,000	2 arribadas	1998-1999	180,000	1 arribada
1985-1986	50,000	1 arribada	1999-2000	?	1 arribada
1986-1987	386,000	2 arribadas	2000-2001	?	1 arribada
1987-1988	0	no arribada	2001-2002	0	no arribada
1988-1989	300,000	1 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* NMFS 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, NMFS, 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, NMFS 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 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, NMFS, personal communication, January, 2001), indicating the animals from both sides of the Pacific converge in the north Pacific pelagic environment.

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.

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 Pelagics 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 NMFS has no bycatch information for sea turtles.

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 3.16). Of these incidental entanglements, an estimated 1,009 turtles were killed (77 percent survival rate).

Table 3.16. 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.

Japan

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.

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 30°N and 40°N. 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, NMFS, 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.

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.

Taiwan

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 3.17, 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"[19] 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 3.17. 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

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[19] "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).

Source: Cheng and Chen, 1997

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).

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 3.18.

Table 3.18. Summary of 1994-95 sample survey of fishermen for incidental catch of sea turtles in 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).

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).

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 3.19 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 3.19. **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.

Foreign tuna fisheries in the western and central Pacific Ocean

As described above, the western and central Pacific Ocean (area west of 150°W longitude, and between 10°N and 45°S) contains the largest industrial tuna fishery in the world. Much of the effort takes place in the EEZs of Pacific-Island counties, in the western tropical Pacific area (10°N - 10°S). 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 fisheries (e.g. pole-and-line, troll, ring-net). About five thousand longliners operate throughout the western and central Pacific (45°N to 45°S), 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 fishery, 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 to encounter gear that is shallow-set (by an order of magnitude) verses deep-set (for longline fishery). When encountered on deep-set gear, sea turtles were likely to be taken on the shallowest hooks. From available observer data, the longline fishery 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 composition of 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 fishery 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 composition of 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).

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[20]. As shown in Table 3.20, the take of sea turtles by the artisanal driftnet fishery in the late 1980s appeared to be comprised primarily of leatherback turtles.

Table 3.20. Chile – turtle bycatch of artisanal driftnet fishermen, 1988-89.

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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[20] Based on all information from Chile and Peru, Eckert (1997) estimated that a <u>minimum</u> 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 3.21. 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 3.21). 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 3.21. 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 anticipated 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.

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 3.22 shows a breakdown of turtle bycatch by species.

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).

Ecuador

Currently, the artisanal 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).

Table 3.22. 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.

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 3.21). 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).

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.

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.

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 3.23 shows the estimated turtle catch by shrimp trawlers in Central America, by country, for 1993.

Table 3.23. 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%. Researchers estimate that approximately 15,000 sea turtles are captured by Costa Rican shrimpers per year without TEDs. (Arauz, *et al.*, 1998; López and Arauz, 2003).

Arauz (personal communication, December, 2003) reports that few countries in Central America use turtle excluder devices, despite Section 609 of Public Law 101-162. Section 609 states that the "importation of shrimp or products from shrimp which have been harvested with commercial fishing technology which may affect adversely such species of sea turtles shall be prohibited" unless the President certifies to the U.S. Congress by May 1 of each year, that any nation which intends to export shrimp to the U.S. conforms with certain conditions. With few exceptions, "certification" requires that shrimpers from other nations use Trawling Efficiency Devices (also known as "Turtle Excluder Devices," or "TEDs" Arauz, 2000). Costa Rica and Honduras are currently embargoed (R. Arauz, personal communication, December, 2003).

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 3.24. Immediate sea turtle mortality was 0%, and most of the hooks were removed prior to release (Arauz, 2001).

Table 3.24. Costa Rican longline fleet - observed number and condition of sea turtles taken on nine cruises, August, 1999 - February, 2000

Species and Condition	Number of Individuals
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[21]).

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

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

consumption by humans." Hooks were also found in the mouths and esophagus of sea turtles, primarily in olive ridleys (Vasquez and Morales, 2003).

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).

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 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; therefore 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 90% of sea turtles recorded dead were either green turtles or loggerheads (Table 3.25), 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 3.25. 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[22], estimating 15,600 to 31,200 sea turtles consumed annually (no differentiation between species).

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 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

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[22] "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.

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 3.26) 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 3.26. 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.

United States (Fisheries Other Than the Pelagic Fisheries of the Western Pacific Region)

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. The treaty was signed by the United States and 16 Pacific Island Parties belonging to the Forum Fisheries Agency, 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 155°W to 140°E 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 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 Fisheries Forum Agency 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).

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, NMFS, personal communication, October 2002).

Hawaii-based longline fishery (Since 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)[23]. These measures were promulgated as a regulatory amendment to the Pelagics FMP, under the authority of the Magnuson-Stevens Act. The final rule, approved by the Secretary of Commerce, was implemented by NOAA Fisheries on June 12, 2002 (67 FR 40232). 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.

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[23] 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).

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 3.27 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 3.27. Annual sea turtle capture and mortality estimates in the Hawaii-based longline fishery.

Species	Incidental Take	Incidental Mortalities
Green	8	7
Leatherback	8	3
Loggerhead	14	8
Olive Ridley	26	24

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

Direct harvest

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.

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).

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).

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 3.28) (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).

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).

Table 3.28. 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.

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).

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.

Australasia (Bali, Torres Strait)
Bali appears to have the largest trade in live green turtles. Reports from World Wildlife Fund/International Union for the Conservation of Nature (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).

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).

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.

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.

Hawksbill Tortoiseshell Trade
By far the most serious problem hawksbill turtles face is the harvest by humans (NMFS and USFWS, 1998b). Turtles have been harvested for centuries by native inhabitants of the Pacific region. Many adults are taken for the shell, which has a commercial value, rather than food. Hawksbill generally are considered to taste poor, and infrequently are toxic to humans (NMFS and USFWS, 1998b). Until recently, tens of thousands of hawksbills were sacrificed each year to meet the demand for jewelry, ornamentation, and whole stuffed turtles (Milliken and Tokunaga *1987 in* Eckert, 1993). In 1988, Japan's imports from Jamaica, Haiti and Cuba represented some 13,383 hawksbills: it is extremely unlikely that this volume could have originated solely from local waters (Greenpeace 1989 *in* Eckert, 1993). Japan ceased the importation of turtle shell in 1992. Today the illegal domestic harvest of eggs and turtles continues in the United States, especially in Caribbean and Pacific island territories. Law enforcement, as well as conservation and management efforts, are hindered by diffuse nesting distributions and the remoteness of some rookeries. It is not easy to determine whether remaining populations are stable, increasing, or declining (Eckert, 1993).

Scientific Research Permits

Scientific Research Permit #1277

Scientific Research Permit #1277 was issued under an ESA Section 10, to the Southwest Fisheries Science Center, NMFS, 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, NMFS-SWFSC, personal communication, January, 2004).

Other Risk Factors
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* NMFS and USFWS, 1998a-e).

5.3 Summary of the Status of Sea Turtles

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 (NMFS 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 to 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 this 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.

Efforts to Conserve Threatened and Endangered Sea Turtles in the Pacific

The Western Pacific Regional Fisheries Management Council, NOAA-Fisheries' Pacific Islands Regional Office and Southwest Fisheries Science Center are continuing to collaborate with regional and local governments around the Pacific rim, conservation and wildlife groups internationally, and the fishing industry both nationally and internationally. Over the past year, these parties have started to implement five specific projects to conserve leatherback and loggerhead sea turtles in the Pacific in cooperation with experienced non-governmental organizations (NGOs) such as World Wildlife Fund - Indonesia (WWF-Indo), Kamiali Integrated Conservation Development Group (KICDG) of Papua New Guinea, the Sea Turtle Association of Japan, and Wildcoast in Baja, Mexico.

Papua (formerly Irian Jaya), War-mon Beach:
Under this contract, World Wildlife Fund-Indonesia (WWF-Indo) would hire villagers to protect the War-mon nesting beach at Jamursba-Medi, Bird's Head Peninsula in Papua (formerly Irian Jaya). This effort is designed to monitor and protect about 33% of the known leatherback nesting beach habitat along the north coast of Papua and protect between 90% and 100% of the currently

unprotected War-mon beach at Jamursba-Medi (the largest known leatherback nesting site in Indonesia). This effort is expected to protect about 1,000 leatherback nests per year (TAC 2003, P. Dutton, NMFS SWFSC) from predation by feral pigs, beach erosion and egg collectors. Protection may be achieved through the use of an electric fence to keep pigs off beaches, by relocating eggs to more secure areas, and deter poachers through monitoring presence. In addition, through monitoring presence, measures are expected to conserve an additional 10 adult nesting females per year from poachers.

Western Papua coastal foraging grounds:
Under this contract, WWF-Indo would work with villagers in western Papua's Kei Kecil Islands to reduce or eliminate the harpooning of about 100 adult leatherback turtles per year in the coastal foraging grounds (TAC 2003, P. Dutton, NMFS SWFSC). In addition, effort will be made to explore and identify alternative food resources for these villagers.

Papua New Guinea nesting beaches:
Under this contract, the Kamiali Integrated Conservation Development Group (KICDG), would work with up to three villages of the Kamiali community in Papua New Guinea to eliminate egg harvesting and nest predation of leatherback eggs, and move those eggs laid in areas likely to be lost to beach erosion. Current practices have a two km section of beach marked off as a "no take" area. This effort is expected to protect about 90% of the nesting beach, and save about 1,000 to 1,500 nests per year (TAC 2003, P. Dutton, NMFS SWFSC).

In addition to establishing nesting beach management measures in Papua New Guinea, this effort will conduct aerial surveys of the coastal areas of northern Papua New Guinea, Solomon Islands and Vanuatu over the next four years to establish a comprehensive inventory of leatherback nesting beaches for which further conservation projects might be established. An initial feasibility study to conduct the initial surveys has been funded for late 2003 (WPRFMC in prep.).

Baja, Mexico halibut gillnet fishery:
Under this contract, Wildcoast would conduct mortality reduction workshops with fishermen and place observers on local boats to insure that all the live loggerheads that comprise the estimated 3,000 loggerhead juveniles per year caught in the halibut gillnets are returned to the ocean (TAC 2003, P. Dutton, NMFS SWFSC). Without observers, these loggerheads become part of the catch.

Japan nesting beaches:
As part of this effort, the Sea Turtle Association of Japan (STAJ) would moving loggerhead eggs from locations prone to washing out and provide shading to nests that experience extreme temperatures at two nesting beaches. A contract has been developed with STAJ for this work to begin during the May 2004 nesting season. The goal of this effort is to save 53 loggerhead nests (TAC 2003, G. Balazs, NMFS PIFSC).

In addition, a contract has been developed with the Ostional National Wildlife Refuge in Costa Rica to help refuge managers to convene workshops to reduce sea turtle mortalities in longline fisheries based in Costa Rica.

Although these conservation measures would benefit leatherback and loggerhead sea turtles, it is not clear whether and how they will improve the status of leatherback sea turtles in the Pacific. Based on our understanding of the measures and their intended benefits, NOAA-Fisheries hopes they are as effective as proposed because these projects should improve the status of leatherback and loggerhead sea turtles.

6.0 EFFECTS OF THE PROPOSED ACTION

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.

The ESA defines "species" to include any subspecies of fish or wildlife or plants, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature." This biological opinion assesses the effects of NOAA-Fisheries' proposal to modify the management regime for the Hawaii-based longline fisheries managed under the Pelagics FMP while continuing the management regime for the other fisheries managed under that FMP on threatened and endangered species and critical habitat that has been designated for these species. NOAA-Fisheries has previously concluded that the fisheries authorized under the Pelagics FMP are likely to adversely affect listed species through gear interactions that are known to injure or kill individuals of these species.

In the *Description of the Action* section of this Opinion, NOAA-Fisheries provided an overview of the fisheries, particularly the distribution of timing of fisheries that use gear that has been a problem for threatened and endangered species. In the *Status of the Species* (which is also the *Environmental Baseline* for this biological opinion) section of this Opinion, NOAA-Fisheries provided an overview of the threatened and endangered species that are likely to be adversely affected by fisheries authorized under the Pelagics FMP.

In this section of a biological opinion, NOAA-Fisheries assesses the probable direct and indirect effects of the fisheries authorized under the Pelagics FMP on threatened and endangered species and designated critical habitat. The purpose of this assessment is to determine if it is reasonable to expect that the fisheries can be expected to have direct or indirect effects on threatened and endangered species that appreciably reduce their likelihood of surviving and recovering in the wild or appreciably diminish the value of designated critical habitat for both the survival and recovery of threatened and endangered species in the wild. Before beginning our analyses, we will discuss our approach to the assessment, the evidence available for our assessment, and assumptions we had to make to overcome limits in our knowledge.

6.1 Effects of Fisheries Authorized Under the Pelagics FMP

As discussed in the Action Area, the fisheries authorized under the Pelagics FMP occur throughout the central, western, eastern and northern Pacific Ocean, including the western Pacific islands of the Northwestern Hawaiian Islands, the main Hawaiian Islands, American Samoa, Guam, Commonwealth of the Northern Mariana Islands (Saipan, Rota, and Tinian), and the U.S. possessions of Johnston Atoll, Kingman Reef, and the islands: Palmyra, Jarvis, Howland, Baker, Midway, and Wake.

The Hawaii-based longline fishery generally operates around the main and northwestern Hawaiian islands except for prohibited areas (a) that extend 50 nautical miles from the center geographical positions of Nihoa Island, Necker Island, French Frigate Shoals, Gardner Pinnacles, Maro Reef, Laysan Island, Lisianski Island, Pearl and Hermes Reef, Midway Island, and Kure Island, which were established to protect Hawaiian monk seals; (b) around the main Hawaiian Islands that seasonally and geographically range from 25 to 75 nautical miles seaward of the shore of particular islands that were established to reduce potential gear conflicts among small boat fishers; and (c) an area that can extend up to 75 nautical miles around the main Hawaiian Islands in the portion of the EEZ seaward of Hawaii; from February 1 through September 30 each year, longline fishing is prohibited up to 75 nautical miles from these islands; from October 1 through the following January 31 each year, longline fishing is prohibited further inshore.

Hawaii-based longline vessels vary their fishing grounds depending on their target species. Most of the effort is north and south of the Hawaiian Islands between latitudes 5° and 40° N and longitudes 140° and 180° W.

The other fisheries authorized under the Pelagics FMP generally occur closer to shore. Most of the vessels associated with the pelagic longline fishery based out of American Samoa occur with 25 nautical miles of shore, although newer, larger vessels are capable of fishing out to and beyond 50 nautical miles. Similarly, the pole-and-line fishery based in Hawaii; the recreational fisheries that target pelagic species around Hawaii; the Hawaiian charter boat fishery; and troll fisheries based out of American Samoa, Guam; Hawaii, and the Commonwealth of Northern Mariana Island all generally occur within 25 miles of shore (NMFS, 2000). In all of these fisheries some fishing vessels range as far as 100 nautical miles from land.

The primary effects of the fisheries on threatened and endangered species results from interactions with the fishing gear, resulting in the capture, injury, or death of the species. In addition to these direct effects of the fisheries, the Western Pacific Regional Fisheries Management Council and the Hawaii Longline Association considered the potential effects of closing the Hawaii-based longline fisheries, which they called "transferred effects." These analyses concluded that imports replaced the fresh swordfish that had been produced by the U.S. swordfish fleet. From that displacement, the analyses asserted that more sea turtles were killed in the fisheries of the foreign suppliers of fresh swordfish. That conclusion seemed predicated on an assumption that effort in those foreign fisheries actually increased to supply the U.S. market rather than that swordfish that would have been caught anyway was redirected from foreign markets to U.S. markets. From this chain of reasoning, the Council and Hawaii Longline Association suggest that prosecuting the Hawaii-based longline fisheries for swordfish would benefit threatened and endangered sea turtles. NOAA-Fisheries recognizes the possibility of "transferred effects" but cannot evaluate the potential adverse or beneficial indirect effects of these transfers on threatened and endangered sea turtles without additional information on the market relationships between shifts in swordfish landings in the Hawaii-based longline fisheries and changes in fishing effort of specific foreign fleets.

6.2 Exposure Analysis

This section of the Opinion evaluates the available information to determine the likelihood of a listed sea turtle co-occurring and interacting with one or more of the fisheries authorized by the Pelagics FMP. As part of our exposure analyses, we consider the probable duration, frequency, and severity of these interactions. This analysis assumes that sea turtles are not likely to be adversely affected by a fishery if they do not interact with the fishery; these analyses also assume that the potential effects of the fisheries would be proportional to the number of interactions between the fisheries and sea turtles.

The only source of information available for this analysis are reports of actual interactions between the fisheries and sea turtles that have been derived from observer programs and logbooks. These sources do not allow us to determine the abundance of sea turtles from different nesting aggregations that *could* interact with the Pelagic fisheries (that is, the number of turtles that are susceptible to interactions with the fisheries). As a result, we cannot analyze potential interactions or the probability of interactions that remain unreported. Therefore, this interaction analysis assumes that the spatial and temporal patterns derived from reported interactions between the fisheries and turtles represents the actual spatial and temporal distribution of the sea turtle populations in the action area. However, given the distribution of their nesting aggregations, turtles probably occur throughout the entire fishing area.

In addition to sea turtles, humpback whales, sperm whales, and Hawaiian monk seals have been exposed to have been known to be exposed to the pelagic fisheries of the western Pacific region. Humpback and sperm whales have been entangled in longline fishing gear and monk seals have been both entangled and hooked in longline fishing gear. On December 29, 2003, a small sperm whale surfaced near a longline vessel and, when it swam away, was reported to have been entangled in longline gear (hooks, line, and floats). Unfortunately, the information available is insufficient to determine whether the whale was harassed or harmed in the incident.

6.2.1 Exposure Probabilities By Gear Type

In general, five different fishing gear types are used under the western Pacific Pelagics Fishery Management Plan: troll, handline, pole-and-line, and longline gear. The type of fishing gear used and the area fished will affect the likelihood of an interaction with a sea turtle. The following section discusses the likelihood of interactions between these gear types and sea turtles.

6.2.1.1 Troll fishing gear

Trolling is conducted by towing lures or baited hooks from a moving vessel, using big-game-type rods and reels as well as hydraulic haulers, outriggers, and other gear. Up to six lines rigged with artificial lures or live bait may be trolled when outrigger poles are used to keep gear from tangling. When using live bait, trollers move at slower speeds to permit the bait to swim naturally (WPRFMC, 1995). Freshly caught small yellowfin tuna or skipjack tuna may be used as live bait to attract marlin. Once a fish is hooked, the gear is immediately retrieved.

Although the distribution of trolling overlaps with the distribution of sea turtles, there have been no reported interactions by vessel operators. In addition, sea turtles are not likely to captured by troll fishing gear because the gear is towed through the water faster than sea turtles may be traveling. Furthermore, sea turtles do not prey on the same type of prey as used by the troll fisheries. A small potential exists that the fishing gear may incidentally hook or entangle a sea turtle when the gear is towed through the water. However, this type of interaction has been extremely rare and NOAA-Fisheries expects interactions to remain rare n the future. Furthermore, because the gear in these fisheries are retrieved almost immediately, interactions are not likely to injure or killed turtles that are captured. Therefore, NOAA-Fisheries does not believe trolling gear is likely to adversely affect sea turtle populations.

6.2.1.2 Pole-and-line

A small pole-and-line fishery operates from Hawai'i that targets skipjack tuna. It is sometimes referred to as the aku (skipjack tuna) fishery or baitboat fishery. The pole-and-line fishery uses live bait thrown from a fishing vessel (ranging from 65 - 80 feet) to stimulate a surface tuna school into a feeding frenzy. The pole and line used are of equal length (3 meters). Fishing is conducted using a barbless hook with feather skirts slapped against the water until a fish strikes. The hooked fish is then yanked into the vessel in one motion. The fish unhooks when the line is slacked so that the process can be repeated. The bait most often used is anchovy.

Although the distribution of the pole-and-line fishery overlaps with the distribution of sea turtles, there is a very low likelihood of an interaction with a sea turtle because the turtle would need to be in the vicinity and the fisher would need to hook the turtle or the turtle would need to strike the hook. This type of an event is unlikely to occur because sea turtles are not likely to prey on anchovy, and the activity of the fish feeding frenzy would deter turtles from remaining in the area. For these reasons, NOAA-Fisheries concludes that the pole-and-line fishery is not likely to adversely affect sea turtle populations.

6.2.1.3 Handline fishery

Two types of pelagic handline fishing methods are practiced in Hawai'i, the *ika-shibi* method, and the *palu-ahi* method. The *ika-shibi* or night handline fishery developed from a squid (*ika*) fishery which switched to target the incidental catch of tuna (*shibi*). Lights and chum are used to attract small prey species and larger target tunas to handlines baited with squid. The vessels typically fish between 5 - 6.5 nm from shore. The night-time fishery is mostly conducted off Hilo and off Keahou, both of the island of Hawaii (Hamilton, 1996 *in* NOAA-Fisheries, 2000a).

The *palu-ahi* or day-handline fishery also targets tuna but fishing occurs during the day. A baited hook on the end of a handline is laid against a stone and the line wound around it. Additional pieces of chum are wound into the bundle which is then tied in a slip knot (Rizzuto, 1983 *in* NOAA-Fisheries, 2000a). The bundle is lowered to the preferred depth (commonly 20-30 meters) where the line is jerked to untie the knot so the baited hook and chum are released at the target depth. Fishing usually takes place by smaller vessels within 6.5 nm from shore and by larger vessels around fish aggregating device or around sea mounts and weather buoys (100 - 200 nm from shore). As soon as a fish is caught, the gear is brought back on board.

There have been no reported interactions between gear used in the handline fishery and sea turtles. Although there is the risk that sea turtles may become hooked or entangled in the fishing gear, any caught animal can be immediately dehooked or disentangled and released. Moreover, most turtles found in the area of the handline fisheries are not likely to prey on the baited hooks. For these reasons, NOAA-Fisheries concludes the handline fishery, as managed under the Pelagics FMP is not likely to adversely affect listed sea turtle populations.

6.2.1.4 Longline fisheries

Longline fishing is a passive fishing method that consists of suspending a monofilament line (main line) in the water column, by using floats, and attaching baited hooks along the line to attract fish. While the main line is deployed over the stern of the vessel, floats and hooks are attached to the main line using clips. Each float is attached to a float line and each hook is attached to a "branch line." The branch line is sometimes called a "gangion" or "dropper" line. For the most part, the branch lines are evenly spaced along the main line, except between floats where the placement of the float on the main line may lengthen the distance between the branch lines. The lengths of the branch lines and the float lines affect how deep the gear (hook) will fish and the type of species that might be caught. The depth that hooks actually fish is also determined by the vessel speed, drum speed, and shooter speed. The faster the main line is set (more line set in a shorter distance), the deeper the line will sink because of the line sag between the floats. In addition to the speed that the main line is set, the number of hooks and the size of the weight on each branch line can affect the depth and rate that the gear will sink. The type of species that are caught are also affected by the time of day the gear is set and the type of bait that is used.

American Samoa longline fishery.
Apart from a few larger (> 40 ft) inboards, longlining out of American Samoa generally takes place on *alias*, twin-hulled (wood with fiberglass or aluminum) boats about 30 feet long, and powered by small gasoline outboard engines. The gear is stored on deck attached to a hand crank reel which can hold as much as 10 miles of monofilament mainline. Participants set between 100 and 300 hooks on a typical eight-hour trip. The gear is set by spooling the mainline off the reel and retrieved by hand cranking back onto the reel. Currently most fishing is done within 25 miles of shore, but with better equipped vessels, fishing activity may extend further. Generally, gear setting begins in early morning; with retrieval in the mid-morning to afternoon.

6.2.1.5 Past sea turtle take in the American Samoa-based longline fishery

For the American Samoa-based longline fishery, the federal logbooks from 1992 to 1999 indicate a range of 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 five species of sea turtles reportedly caught by the fishery do occur in the fishing grounds of this longline fishery. In addition, as discussed immediately below, logbook data may not be a reliable method to measure sea turtle interaction in the fisheries. Therefore, for the purposes of this Opinion, from 1992-1999, the take of sea turtles by the American Samoa-based longline fishery included at least 4 hardshelled turtles (with 3 released alive, 1 mortality), 1 leatherback, and 1 unidentified sea turtle.

6.2.1.6 Hawaii-based longline fishery

SWORDFISH OR MIXED TARGET (SHALLOW SET) LONGLINE FISHERY. Pacific Ocean longline vessels targeting swordfish or a mixture of tuna and swordfish, typically deploy about 42 horizontal miles of main line in the water. Most branch lines are about 17 meters (56 feet) in length and float lines are about 8 meters (26 feet) in length. For bait, fishers use squid (either large or small) and a number 9 Mustad (J-shaped) hook or, more rarely, an offset J-shaped hook. In addition, fishers use lightsticks on almost half of the hooks (every other hook). A typical set uses about 820 hooks and 189 floats which means there are approximately 4 or 5 hooks between each float. Assuming the branch lines and the float lines are evenly spaced, the distance between them is approximately 67 meters (220 feet). On average, fishers try to set their gear at about 28 meters (92 feet) below the water surface. The gear is allowed to soak during the night and soak times of the gear typically last about 20 hours, including setting and hauling of gear. This type of set is referred to below as "shallow set."

TUNA (DEEP SET) LONGLINE FISHERY. Vessels targeting tuna in the Pacific Ocean 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. Instead of squid, 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 soak time typically lasts about 19 hours, including setting and hauling of gear. This type of set is referred to below as "deep set."

DIFFERENCES BETWEEN THE SHALLOW SET AND DEEP SET LONGLINE FISHERIES. Shallow set fishing effort rarely occurs below 20°N throughout the year (see Figures 1 through 5 in Appendix B), although there has been some concentration of effort between 18° and 20° N in the third quarter. Deep set fishing effort rarely occurs north of 25°N. Shallow set vessels fish shallower using fewer hooks per float than vessels targeting tuna. For practical purposes, vessels using fewer than 10 hooks per float are targeting swordfish, or a mix of swordfish and tuna, and vessels using more than 10 hooks per float are targeting tuna (D. Kobayashi, NOAA-Fisheries, personal communication, January, 2001). Shallow set vessels set about 42 horizontal miles of gear whereas deep set vessels set only 34 horizontal miles of gear. The shorter horizontal distance is because deep set vessels put more sag in their gear and the gear is fishing deeper even though the mainline itself may be the same length between the two types of sets. To increase the amount of sag in the main line on deep sets, fewer floats are used with more mainline between each float, which will increase the amount of sag. In addition, shallow sets are made during the night, while deep sets are soaked during the day.

6.2.2 Exposure Probabilities By Turtle Species

The narratives that follow describe the co-occurrence between the fisheries and sea turtles using three variables.

:

a. *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. gear can interact with individuals from all populations of threatened and endangered species or they can interact with particular populations.

b. *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.

c. *Spatial patterns of exposure.* The spatial patterns of interactions between fishing gear and marine species 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.

d. *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 that are incidentally captured in a fishery. To the extent that information was available, we described these patterns using all three dimensions.

6.2.2.1 Green Sea Turtle

DEMOGRAPHIC PATTERNS OF EXPOSURE. Green turtles are exposed to the Hawaii-based longline fisheries and have been reported being exposed to the longline fisheries based out of American Samoa. The turtles that interact with longline vessels based in American Samoa are likely to be individuals from western Pacific nesting beaches, although we cannot assign probabilities to any particular nesting aggregations.

Those green turtles that are captured by the Hawaii-based longline fisheries will be members of the endangered Mexican (Pacific coast) or threatened Hawaiian (French Frigate Shoals) nesting aggregations. Out of eight green turtles caught by the Hawaii-based longline fishery, genetic analyses concluded that four of the eight turtles (50%) represented nesting aggregations from the eastern Pacific (Mexico), one turtle (12.5%) represented the Hawaiian nesting aggregations, the remaining three turtles (37.5%) could have been from either of these two nesting aggregations (P. Dutton, NOAA-Fisheries, personal communication, January, 2001) but may also have represented other nesting aggregations in the Pacific Ocean.

Life history information collected by observers suggests that the Hawaii-based longline fisheries are likely to capture sub-adult and adult green turtles (straight carapace lengths ranged from 28.5 cm to 73.5 cm with an average of 51.5 cm).

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. The existing data on these interactions revealed clear spatial patterns between the Hawaii-based longline fisheries and green turtles. Green turtles have been caught in the area bounded by 155°W and approximately 180°E longitude and between 5°N and 30°N latitude. Ten out of the 18 turtles were caught in an area around the Hawaiian island chain between 155°W and 165°W longitude and between 15°N and 30°N latitude. Of the remaining turtles, four were captured far to the south of the Hawaiian islands and four were captured to the northwest of the main Hawaiian Islands.

Within the water column, more green turtles were captured in shallow sets compared to deep sets. Thirteen of 18 turtles caught were captured in sets with less than 10 hooks per float, indicative of shallow-set gear. The only mortalities (n = 4) observed were on deep sets; therefore, it is likely that the turtles died as a result of their inability to reach the surface. Because sub-adult green turtles reportedly perform routine dives of 20 meters (Brill, *et al.*, 1995, *in* Lutcavage and Lutz, 1997)routinely dive to depths of 20 meters (Brill, *et al.*, 1995, *in* Lutcavage and Lutz, 1997), they are more likely to encounter shallow-set longline gear than deep-set longline gear (which is often set at depths greater than 100 meters).

TEMPORAL PATTERNS OF EXPOSURE. The existing data also revealed that green turtles have been captured in all months of the year except January and September. From observer data, and using a model-based predictor, McCracken (2000) estimated that between 37 and 45 green turtles (mean = 40) were captured each year by the Hawaii-based longline fishery.

EXPOSURE TO THE PROPOSED FISHERIES. Assuming that patterns observed in the past represent future patterns, green sea turtles will be exposed primarily to the longline fisheries; the number of green turtles that have been captured by other fisheries included in the Pelagics FMP are expected to be minimal. One green turtle was reported to have been captured and killed by the longline fisheries based in American Samoa in 1999, but no interactions have been reported since then (it is important to note that these fisheries are not observed, so the absence of reports may only reflect the absence of sampling rather than the absence of interactions). With the proposed fishery management regime, about 7 (95% confidence interval = 2 - 17) green turtles are expected to be captured by the Hawaii-based longline fisheries each year. Unlike the past, most of these turtles (86%) would be exposed to deep-set gear: about 6 green sea turtles (95% confidence interval = 1 - 13) are expected to be captured by deep-set gear, while 1 turtle (95% confidence interval = 1 to 5) is expected to be captured by shallow-set gear.

Based on genetic sampling of the green sea turtles captured in these fisheries in the past, the turtles that are exposed to the Hawaii-based longline fleets will represent nesting beaches both the eastern Pacific Ocean or Hawaii, although most of the green turtles will be endangered turtles that will have migrated from the east (P. Dutton *et al.*, 2000; P. Dutton, NOAA-Fisheries, personal communication, January, 2001). If the longline fisheries affect green turtle populations proportional to their relative abundance in the action area, about 4 to 6 of the 7 green turtles that are expected to be captured in the Hawaii-based longline fisheries each year would represent endangered green turtles from the eastern tropical Pacific , while 1 or 2 of the 7 turtles would represent turtles from the Hawaiian nesting aggregations.

6.2.2.2 Hawksbill Sea Turtle

DEMOGRAPHIC PATTERNS OF EXPOSURE. Hawksbill turtles are exposed to the Hawaii-based longline fisheries and have been reported being exposed to the longline fisheries based out of American Samoa. The turtles that are captured by longline vessels based in American Samoa are likely to be individuals from western Pacific nesting beaches, although we cannot assign probabilities to any particular nesting aggregations.

SPATIAL AND TEMPORAL PATTERNS OF EXPOSURE. It is impossible to identify any patterns from the single report, particularly since that report had not been observed.

6.2.2.3 Leatherback Sea Turtle

DEMOGRAPHIC PATTERNS OF EXPOSURE. Leatherback turtles are exposed to the Hawaii-based longline fisheries and have been reported being exposed to the longline fisheries based out of American Samoa. Genetic analyses of leatherback turtles captured previously in the Hawaii-based longline fishery, identified 12 of the 14 leatherback turtles captured in the fishery from nesting aggregations in the southwestern Pacific; the remaining 2 turtles, which were captured in the southern range of the fishery, were from nesting aggregations in the eastern Pacific (P. Dutton, *et al.*, in press, and P. Dutton, NOAA-Fisheries, personal communication, May, 2000). Based on these data we assume that most of the leatherback turtles that are exposed to the Hawaii-based longline fisheries are from two nesting aggregations: the eastern Pacific region (Mexico and Costa Rica), and the western Pacific region (Indonesia, Malaysia, Papua New Guinea, Fiji, and the Solomon Islands). Leatherback turtles that are captured by the longline fisheries based out of American Samoa are most likely to represent one or more of the western Pacific nesting aggregations, although we cannot assign probabilities to any particular nesting aggregations.

Observers collected life history records for 34 leatherback turtles, but only five of the turtles captured in the fishery had been measured (the unmeasured turtles may have been too large to be safely brought on board; therefore they may have been adults). The straight carapace lengths for the five turtles were 71, 80, 87.5, 110, and 130 centimeters, the smallest four of these turtles were probably early pelagic juveniles (n = 1) and late pelagic sub-adults (n = 3) based on growth rates that have been assumed for Malaysian turtles (see Bolten, *et al.* 1996). If the larger (>130 cm) leatherback turtle was from the western Pacific, it would have been a sub-adult turtle, if it was from the eastern Pacific nesting aggregations, it could have been an adult (P. Dutton, NOAA-Fisheries, personal communication, January, 2001). In either case, we assume that the leatherback turtles that are exposed to the Hawaii-based longline fisheries are either sub-adult or adult turtles.

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 can be expected to continue trying to swim, expending valuable amounts of energy and oxygen. As the turtle's available oxygen diminishes, anaerobic glycolysis will take over, producing high levels of lactic acid in the blood. Unlike the hard-shelled sea turtles, leatherback turtles lack calcium, which helps to neutralize the lactic acid build-up by building up bicarbonate levels. In addition, leatherback turtles 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 Hawaii-based longline fisheries are more likely to interact with leatherback sea turtles from western Pacific nesting aggregations than those from the eastern Pacific. Within the western Pacific aggregations, turtles from the Indonesian beaches may be more likely to be exposed to these fisheries because of their relative abundance

SPATIAL PATTERNS OF EXPOSURE. There appears to be consistent spatial patterns to the existing data on the interactions between the Pelagics fisheries and leatherback turtles. Vessels fishing off Hawaii generally observe leatherback turtles beyond the 100-fathom curve but within sight of land. Two areas where observations have been reported are off the north coast of Oahu and the west coast of the Island of Hawaii, and in the area of the seamounts above the Northwestern Hawaiian Islands (Skillman and Balazs 1992). Leatherback turtles apparently use the pelagic zone surrounding the Hawaiian Islands as foraging habitat and migratory pathways. Further to the north of the Hawaiian Islands, leatherback turtles are known to aggregate at 35°N latitude, between 175°W and 180°W longitudes (NOAA-Fisheries 1991).

In the past, leatherback turtles have been captured in longline gear in the area bounded by 170°E and 133°W longitude and between 5°N and 41°N latitude. Leatherback turtles caught in sets above 20°N latitude (43 out of 52 leatherback turtles observed) were caught in sets with less than 10 hooks per float, indicative of shallow-set gear and also indicative of the general area in which shallow-set fishing methods are used. Leatherback turtles were primarily captured in these sets between 165°W and 130°W longitude and 20°N and 40°N latitude.

The remaining leatherback turtles captured in the fisheries (9 out of 52), were associated with in sets with more than 10 hooks per float, suggesting deep-set gear. These interactions occurred between 153°W and 167°W longitude and 5°N and 26°N latitude. Sea surface temperatures, latitude, and the distance to the approximate 17°C and 19°C isotherms were associated with these interactions, but these variables were highly correlated (McCracken, 2000). When McCracken

(2000) examined four latitude predictor categories for leatherback turtles[24], she found that the proportion of sets associated with leatherback captured was higher in the northernmost and southernmost categories, even though these areas had lower proportions of the observed sets than the middle two categories, which had high observed sets but fewer observed takes. These observations suggest that the risk of an interaction increases toward the northern and southern boundaries of the action area.

TEMPORAL PATTERNS OF EXPOSURE. There have been no obvious temporal patterns to interactions between leatherback turtles and longline gear: leatherback turtles have been captured in every month of the year, except August.

EXPOSURE TO THE PROPOSED FISHERIES. Assuming that patterns observed in the past represent future patterns, leatherback sea turtles will be exposed primarily to the Hawaii-based longline fisheries. The number of leatherback turtles that have been captured by other fisheries included in the Pelagics FMP are expected to be minimal: one leatherback turtles was reported to have been captured but not killed by the longline fisheries based in American Samoa in 1999, no interactions have been reported since then but it is important to note that these fisheries are not observed, so the absence of reports may only reflect the absence of sampling rather than the absence of interactions).

As proposed, the Hawaii-based longline fisheries would be modified to require longline vessels to use management measures that have proven effective at reducing the number and rate of interactions between leatherback and loggerhead sea turtles: the use of 18.0 10° offset circle hook with squid bait. Fishing experiments conducted in the North Atlantic Ocean, demonstrated that interaction rates between leatherback turtles and the fisheries were significantly reduced with the18/0 10° offset circle hook with squid bait compared with J-hooks with squid bait that have been used historically. On average, the interaction rate of leatherback turtles declined by 50% in these experiments. Interaction rates between leatherback turtle were also significantly reduced by using mackerel as bait rather than squid on J-hooks. On average, the interaction rate of leatherback turtles declined by 67% using mackerel bait.

Using mackerel bait on hooks appeared to have some influence on turtle interactions. About one third of the vessels used a "single hooking" technique, which involved passing the hook point a single time through the bait's eye, back or tail. The remaining vessels employed a threading technique which involved passing the entire hook through the bait multiple times starting through the eyes, back or tail. This technique was used to better secure the bait to the hook and minimize bait loss. Interactions with loggerhead turtles were 74 % greater with this threading technique as compared to the single hooked mackerel. This may be due to the ease with which single hooked baits are torn away from the hooks during the feeding process. The single hooked baits also had the highest catch rates of swordfish. However, the single hooked baits had a 107% higher leatherback interaction rate than the threaded bait. This is likely due to the shielding effect offered

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[24] These four categories were: less than 14.95°N, between 14.95°N and 24.84°N, between 24.84°N and 33.82°N, and greater than 33.82°N (McCracken 2000).

by the threading of the baits, as leatherback turtles are known to be frequently flipper hooked, presumably because they tend to run into the longline gear accidentally rather than biting it.

With the proposed management regime for the Hawaii-based longline fisheries, which include these gear modifications, about 34 (95% confidence interval = 18 - 60) leatherback turtles are expected to be captured by the Hawaii-based longline fisheries each year. Interactions between the different types of gear and leatherback turtles is split almost equally. About 16 leatherback turtles (95% confidence interval = 9 - 30) are expected to be captured by the shallow-set gear, while about 18 turtles (95% confidence interval = 9 to 30) are expected to be captured by the deep set gear.

Based on the limited genetic sampling from the action area, about 86% of the leatherback turtle sample (12 out of 14 genetic samples) originated from western Pacific nesting beaches (P. Dutton *et al.*, 2000; P. Dutton, NOAA-Fisheries, personal communication, January, 2001). If the longline fisheries affect leatherback turtle populations proportional to their relative abundance in the action area, about 29 of the 34 leatherback turtles that are expected to be captured in the Hawaii-based longline fisheries each year would come from nesting aggregations in the western Pacific Ocean. These turtles could represent individuals from Indonesia (Jamursba-Medi or War-Mon), Papua New Guinea (Kamiali), Malaysia (Terengganu), the Solomon Islands, or Fiji, although satellite tracks from leatherback turtles tagged in Papua New Guinea suggest that these turtles tend to migrate south instead of north, which would take them away from Hawaiian waters. Further, the abundance of the nesting aggregations in Indonesia relative to the small size of the other nesting aggregations suggests that the interactions between Indonesian leatherback turtles and the Hawaii-based longline fisheries are most likely.

The remaining 14 percent of the interactions, or about 5 leatherback turtles per year, would represent turtles from the eastern Pacific Ocean. These turtles could represent individuals from nesting aggregations along the coast of Mexico, Costa Rica, or Panama, although turtles from these nesting aggregations may only migrate into Hawaiian waters when oceanic phenomena like El Nino events prevent them from migrating south to the coasts of Peru and Chile. Several investigators who have followed leatherback turtles equipped with satellite tags have reported that leatherback turtles from the beaches of Mexico and Costa Rica migrate through the equatorial current towards the coasts of Peru and Chile (Chandler 1991, Eckert 1997, Marquez and Vellanueva 1993, Morreale et al. 1996). Eckert (1997) suggested that these turtles migrate toward the coast of South America where upwelling water masses provide an abundance of prey.

Although these data suggest that the Hawaii-based longline fisheries are more likely to capture leatherback turtles from Indonesia, over a period of several years, we would expect these fisheries to capture turtles from the other, smaller nesting aggregations.

6.2.2.4 Loggerhead Sea Turtle

DEMOGRAPHIC PATTERNS OF EXPOSURE. Most interactions between the Hawaii-based longline fishery and sea turtles involve loggerhead sea turtles. Based on genetic analyses of 124 loggerheads captured previously in this fishery, almost all of the loggerhead turtles that are exposed to the fishery are from the Japanese nesting aggregations (Dutton, *et al.*, 1998). The majority of these turtles represent the 40 different nesting beaches in southern Japan while a small percentage (about 5 percent of the turtles sampled) represent a rare genetic type that is unique to two nesting beaches on Yakushima Island off southern Japan (Kamezaki et al. 2003, P. Dutton, NOAA-Fisheries, personal communication, December, 2003).

The proposed fisheries would primarily capture or interact with loggerhead sea turtles in the oceanic juvenile[25] stage of development(Bolten 2003).

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 20°C at 28°N and another of 17°C at 32°N. 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 patterns that have been observed 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

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[25] In previous biological opinions on the Pelagic fisheries, , this stage had been termed "pelagic juvenile" stage.

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. From 1994 to 2003, observers recorded the capture of 147 loggerhead sea turtles by the Hawaii-based longline fisheries. The existing data on these interactions revealed clear spatial patterns between the Hawaii-based longline fisheries and loggerhead turtles. For example, McCracken (2000) reported that none of 1,263 sets that were observed south of 22°N captured loggerhead turtles. Kleiber (1998) also found latitude seemed to determine the chances of interactions between the fisheries and loggerhead turtles.

Statistical analyses of these captures to determine possible associations with several different variables like sea surface temperature, latitude, and the distance to the approximate 17°C and 19°C isotherms was highly correlated with these variables. Degree of latitude appeared to be a primary determinant of the probability of loggerhead captures in the fisheries. Of 55 trips in which loggerheads were captured, 29 had captured loggerheads in more than one set, suggesting that loggerhead turtles forage or migrate in groups, that longline vessels target swordfish and tuna in areas of high loggerhead concentration, or both.

Other data suggest that loggerhead turtles forage or migrate in groups. Off Baja California, thousands of loggerhead turtles have been observed feeding on pelagic crabs. 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). Recent satellite tracking by Polovina *et al.* (2000, 2003, 2004) indicates that all life states of loggerhead turtles actively migrate, swimming against weak geostrophic currents along two convergent fronts as they travel from east to west across the Pacific. Of nine juvenile loggerheads tracked in the central North Pacific, six associated with a front characterized by 17°C sea surface temperature (termed "cool group") and the other three associated with a front with a sea surface temperature of 20°C ("warm group"). Seasonally, these 17°C and 20°C isotherms move north and south over 10 degrees of latitude, and as the turtles moved westward, they also appeared to move north and south coincident with these isotherms. During the first quarter, the distribution of shallow longline sets is largely between the 17°C and 20°C sea surface temperature fronts used by loggerheads.

Swordfish are believed to move south through these fronts, perhaps following squid. For example, during the second quarter, the fishery tends to locate well south of the front formed along the 17°C isotherm but overlapping with the front formed along the 20°C isotherm. Sea turtles tracked during the first quarter of the years (1997 and 1998) occupied waters with a mean of 17°C sea surface temperature (SST), with considerable overlap with the SST associated with the fishery in the northern portion of the fishing grounds. As the fishery moves south in the second quarter, those "warm group" turtles following the 20°C front may be well within the fishing ground, while the "cool group" will likely be well north of the fishing ground (Polovina, *et al.*, 2000). Observer data shows that the interaction rate (or the number of turtles captured per longline set) is substantially greater at 17°C SST than at 20°C SST (P. Kleiber, NOAA-Fisheries, personal communication *in* Polovina, *et al.*, 2000).

Finally, all of the 147 loggerheads observed taken by the Hawaii-based longline fishery from 1994-1999 were captured by longline vessels using shallow sets (i.e. target depth less than 100 meters, using less than 10 hooks per float, fishing at night, using lightsticks). The mean dive depth for loggerhead turtles (post-nesting female and subadult) is between 9 and 22 meters; therefore, loggerhead turtles are more likely to be captured by shallow sets than deep sets (which generally target depths greater than 100 meters).

TEMPORAL PATTERS OF EXPOSURE. The interactions between the Hawaii-based longline fisheries and loggerhead sea turtles had a temporal pattern. There are no reports of loggerhead turtle being captured by the fishery during May or June. In the past, most loggerhead turtles were captured during the fall and winter months, especially January and February.

EXPOSURE TO THE PROPOSED FISHERIES. Assuming that patterns observed in the past represent future patterns, loggerhead sea turtles will be exposed to the Hawaii-based longline fisheries. As proposed, the Hawaii-based longline fisheries would be modified to require longline vessels to use management measures that have proven effective at reducing the number and rate of interactions between leatherback and loggerhead sea turtles: the use of 18/0 10° offset circle hook with squid bait. Like the leatherback turtles that were discussed previously, fishing experiments conducted in the North Atlantic Ocean demonstrated that interaction rates of loggerhead turtles were significantly reduced with the18/0 10° offset circle hook with squid bait compared with J-hooks with squid bait that have been used historically. On average, the interaction rate of loggerhead turtles declined by 85% in these experiments. Interaction rates between leatherback turtle were also significantly reduced by using mackerel as bait rather than squid on J-hooks. On average, the interaction rate of leatherback turtles declined by 75% using mackerel bait. The greatest reduction in loggerhead interactions was achieved using a combination of mackerel bait with an 18/0 circle hook with a 10° offset. On average, the interaction rate of loggerhead turtles declined by 92% with these modifications.

With the proposed management regime for the Hawaii-based longline fisheries, including these gear modifications, about 21 (95% confidence interval = 8 - 64) loggerhead turtles are expected to be captured by the Hawaii-based longline fisheries each year. As in the past, most of these turtles (81%) will be exposed to shallow-set gear: about 17 loggerhead turtles (95% confidence interval = 7 - 55) are expected to be captured by the shallow-set gear, while about 18 turtles (95% confidence interval = 4 to 9) are expected to be captured by deep set gear.

Based on genetic sampling of the loggerhead turtles captured in these fisheries in the past, these loggerhead turtles will represent nesting beaches from southern Japan (P. Dutton *et al.*, 2000; P. Dutton, NOAA-Fisheries, personal communication, January, 2001). If the longline fisheries affect leatherback turtle populations proportional to their relative abundance in the action area, about 1 of the 21 loggerhead turtles that are expected to be captured in the Hawaii-based longline fisheries each year would represent the turtles that nest on two beaches on Yakushima Islands in southern Japan, while the remaining 20 of the 21 loggerhead turtles would represent turtles that nest elsewhere in southern Japan.

6.2.2.5 Olive Ridley Sea Turtle

DEMOGRAPHIC PATTERNS OF EXPOSURE. Olive ridley sea turtles are exposed to the Hawaii-based longline fisheries and have been reported being exposed to the longline fisheries based out of American Samoa. The turtles that interact with longline vessels based in American Samoa are likely to be individuals from western Pacific nesting beaches, although we cannot assign probabilities to any particular nesting aggregations.

Genetic analyses of olive ridley sea turtles captured previously in the Hawaii-based longline fisheries identified olive ridley turtles from nesting aggregations in the eastern, western, and Indian Pacific Ocean. Of 20 olive ridleys captured by the Hawaii-based longline fishery, 8 (40 %) were from the Indian Ocean or western Pacific Ocean and 12 (60%) were from the eastern Pacific (P. Dutton, NOAA-Fisheries, personal communication, January, 2001). Based on these data we assume that the olive ridley turtles that are exposed to the Hawaii-based longline fisheries represent the threatened western Pacific population and the endangered eastern Pacific population. Most of these sea turtles will be sub-adults or adults.

BEHAVIORAL PATTERNS OF EXPOSURE. Recent observations of the diving behavior of olive ridley sea turtles in the Pacific by the NMFS Pacific Islands Science Center (Jeffery Polovina, NMFS PIFSC, pers. comm.) suggest that they spend over 90% of their time at depths of less than 100 m. Swimming at these depths in the Pacific places olive ridley sea turtles in the zone where shallow-set swordfish targeting longline gear, or the shallowest of hooks on deeper tuna targeting gear would be deployed by the Hawaii longline fleet This information suggests that the Hawaii-based longline fishing fleet targeting tuna can reduce the exposure of olive ridley turtles to their gear by setting the line below the dive depths of these turtles.

SPATIAL PATTERNS OF EXPOSURE. The existing data on these interactions revealed clear spatial patterns between the Pelagics fisheries and olive ridley turtles. In addition, the fishery interacted with olive ridleys throughout the fishing area, with captures reported from as far north as 33°N to as far south as 7°N latitude, and from longitudes 143°W, west to 175°W. Sea surface temperatures, latitude, and the distance to the approximate 17°C and 19°C isotherms were associated with the takes, but there was a high degree of collinearity between these variables. There was a clear distinction between the proportion of takes between the two categories of sea surface temperature, but over latitude, the pattern was less clear (McCracken, 2000).

TEMPORAL PATTERNS OF EXPOSURE. In the past, interactions between olive ridley sea turtles and longline vessels had a weak temporal pattern: olive ridley turtles have been captured in every month of the year, except February, with most of the captures occurring between May and August.

EXPOSURE TO THE PROPOSED FISHERIES. Assuming that patterns observed in the past represent future patterns, olive ridley sea turtles will be exposed to longline fisheries based out of Hawaii and American Samoa. The number of olive ridley turtles that have been captured by other fisheries included in the Pelagics FMP are expected to be minimal one olive ridley turtle was reported to have been captured but not killed by the longline fisheries based in American Samoa in 1999. No interactions have been reported since then, but it is important to note that these fisheries are not observed, so the absence of reports may only reflect the absence of sampling rather than the absence of interactions.

The degree to which the proposed management measures (circle hooks and mackerel style bait) will benefit olive ridley sea turtles remains unknown since they were not (and could not have been) included in the experiments that were conducted in the North Atlantic. However, because of similarities between the diving behavior and dive depths of leatherback turtles and olive ridley turtles, we would expect the measures would benefit olive ridley turtles that interact with shallow-set gear like they benefit leatherback turtles. With the proposed management regime for the Hawaii-based longline fisheries, about 42 (95% confidence interval = 23 - 76) olive ridley turtles are expected to be captured by the Hawaii-based longline fisheries each year. As in the past, and because of their diving habit, most of these turtles (60%) will be exposed to deep-set gear: about 37 olive ridley sea turtles (95% confidence interval = 21 - 60) are expected to be captured by deep-set gear, while about 5 turtles (95% confidence interval = 2 to 16) are expected to be captured by shallow-set gear.

Based on genetic sampling of the olive ridley sea turtles captured in these fisheries in the past, these turtles will represent nesting beaches both the eastern and western Pacific Ocean as well as the Indian Ocean (P. Dutton *et al.*, 2000; P. Dutton, NOAA-Fisheries, personal communication, January, 2001). If the longline fisheries affect olive ridley turtle populations proportional to their relative abundance in the action area, about 31 of the 41 olive ridley turtles that are expected to be captured in the Hawaii-based longline fisheries each year would represent endangered olive ridley turtles from the eastern tropical Pacific , while 11 of the 41 olive ridley turtles would represent turtles from the western Pacific or Indian Oceans.

6.2.3. Factors contributing to the likelihood of an interaction with the longline fishery

6.2.3.1 Gear

FLOATS. Sea turtles may be attracted to the floats used on longline gear. Sea turtles have been observed associating with manmade floating objects significantly more frequently than with natural objects, perhaps related to turtles' affinity for three-dimensional objects. Turtles also show a preference for objects floating horizontally and nearly submerged and are strongly attracted to brightly colored objects (Arenas and Hall, 1992). Floats typically used during swordfish-style sets are bright orange, bullet-shaped, and slightly submerged. Tuna-style 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).

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 bait on swordfish longline gear. The authors speculate that the lightsticks may initially have attracted the turtle, by simulating natural prey (Skillman and Balazs, 1992).

6.2.3.2 Environmental conditions

Environmental conditions may also play a large part in whether or not a sea turtle interacts with longline gear. Sea turtles in the open ocean are often found associated with oceanographic discontinuities such as fronts and driftlines, areas often indicating high productivity. In addition, sea turtles also appear to associate with particular sea surface temperatures . As mentioned in more detail later, species such as the loggerheads have been tracked moving along convergent ocean fronts, in waters with sea surface temperatures of 17° C and 20° C (Polovina, *et al.*, 2000). Swordfish are caught by longliners in association with frontal zones where ocean currents or water masses meet to create turbulence and sharp gradients of temperature and salinity. Swordfish also make vertical migrations through the water column, rising near to the surface at night from deep waters. Thus, while searching for concentrations of swordfish, longliners set their gear across these temperature gradients ("breaks") indicative of intersecting water masses, and when sea turtles are associated with these fronts, interactions are more likely.

6.3 Response of Sea Turtles Given Exposure

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 the pelagics 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.

6.3.1. 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).

6.3.2. 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).

6.3.3. 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 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.

6.3.4. 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, NMFS, personal communication, April, 2002). The researchers made no assumptions about the fate of the 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 hooking incidents — including forced submergence, hauling of the longline and subsequent capture on the vessel — turtles that had been released may not have had time to recover from their experience. As discussed earlier, 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 (NMFS, 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 between hooking categories was not statistically significant 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, NMFS, 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. However, as we will discuss below, NOAA-Fisheries recently convened a workshop to review post-hooking mortality estimates.

6.3.4.1. 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 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. NMFS 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_{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).

6.3.4.2. Survival of Sea Turtles that Interact With Longline Gear

Between 1994 and 1999 observers recorded data on 239 interactions between turtles and longline fisheries that provide insights into the relative effects of the different longline sets on the survival of sea turtles. During this period, none of the turtles that had been captured in deep-set longline gear were released alive or uninjured from deep sets, compared to 6 of 225 turtles taken by shallow sets (2.7%). Although the number of interactions between the turtles and shallow-set longline gear represent a small sample size, these data still suggest that sea turtles caught in the two different gear types experience different rates of “immediate” mortality.

Eight of the 14 turtles (57%), or 0.0056 turtles per set, caught by deep sets were dead upon retrieval of the gear compared to three turtles (1.33%, or up to 4.9% if all unknown condition turtles are treated as dead) dead out of 225 turtles , or 0.0017 turtles per set, caught in shallow sets. Differences in species-specific “immediate” mortality between gear types are also apparent. Based on past observer data for 1,440 tuna-style sets, 50% of the green turtles, 33% of the leatherback turtles, and 83% of the olive ridley turtles died per set. Loggerhead turtles were not observed captured in this segment of the fishery. In 1,811 shallow sets, 11.8% of the leatherback turtles, 1.36% of the loggerhead turtles, 3.85% of the olive ridley turtles, and 40% of the unidentified sea turtles died per set (assuming that the 4 out of 10 turtles captured in “unknown” condition were mortalities). This difference between “immediate” death rates could have several explanations, although one possible explanation is that the turtles captured in deep sets could not reach the surface to breathe or rest, but turtles caught in shallow sets may be able to reach the surface. For example, on a shallow set, the length of the branch line is 17 meters. This length is more than half the distance between the hook and the surface (average target depth is 28 meters). With a float line length of 8 meters, the main line sag between floats would be about 3 meters or about 11 meters below the water surface. This means that a hooked turtle could swim to the surface and breathe because the branch line length is greater than the depth of the main line from the surface.

Overall mortality rates, or the combined, immediate and delayed mortality rates, are also notably different between the two fishing styles. In deep sets, five turtles were lightly hooked and one was entangled. These injuries were assigned a 0% post-interaction mortality rate (McCracken, 2000). In shallow sets, 108 turtles were lightly hooked, 103 were deeply hooked, and two were entangled. Deeply hooked turtles were assigned a post-interaction mortality rate of 29% (McCracken, 2000).

Given the assigned mortality rates for dead and deeply hooked sea turtles, the deep sets had a sea turtle mortality rate per take of 57% and the shallow sets had an overall sea turtle mortality rate per take of 14.7% (30 turtles killed by deep hook injuries [29% of 103 deep hooked turtles] and 3 turtles dead upon gear retrieval [100% mortality] = 33 turtles/225 turtles = 14.7%). This appears to be a considerable difference in overall mortality rates between the two types of fishing, however when overall mortality rates are calculated per set, it becomes apparent that shallow sets kill more turtles per set (0.0182 turtles per set versus 0.0056 turtles per set in deep sets) due to higher interaction rates and the higher incidence of deep hooking shallow sets have with turtles compared to deep sets. Revision of the kills per set data using more recent information on post-hooking delayed mortality (discussed below) which assigns a 27% mortality rate to externally hooked turtles with minor or moderate injuries and a 42% mortality rate to turtles with more serious injuries, including deep hooks, still indicates that shallow sets kill more turtles per set with 0.042 turtles killed per set compared to deep sets which kill 0.0081 turtles per set.

6.3.4.3. Post-Hooking Survival and Mortality

In February 2001, NOAA-Fisheries established a policy and criteria for estimating sea turtle survival and mortalities following interactions with longline fishing gear (Table 6.5 in NMFS 2001b). 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 6-1. Sea turtle mortality rates based on level and type of interaction with longline fishing gear. Source: NMFS, 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 6.2 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[26] (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.

Mortality estimates for turtles taken by the Hawaii-based longline fishery were based on limited data from Aguilar, *et al.* (1995) and from information recorded by observers on the condition of the turtles when released (Kleiber, 1998). Aguilar, *et al.* (1995) estimated a 29% mortality rate for loggerheads ingesting a longline hook; therefore all turtles (hard-shelled and leatherback) that had been hooked internally were assigned a mortality rate of 29%. Turtles recorded as dead had a 100% mortality rate, and turtles recorded as okay (released uninjured) were assigned a 0% mortality rate. All species of turtles hooked externally were also assigned a 0% mortality rate (McCracken, 2000).

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[26] Subordinate part such as tongue, extraembryonic membranes

Table 6.2. 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[27]) 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[28]
Entangled Only	Released Entangled 50 (60)		Fully Disentangled 1 (2)
Comatose/resuscitated	n/a[29]	70 (80)	60 (70)

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[27] Subordinate part such as tongue, extraembryonic membranes

[28] Per veterinary recommendation hooks would not be removed if the insertion point of the hook is not visible when viewed through the open mouth.

[29] 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.

Observers occasionally could not gather all of the information necessary to use these criteria (for example, they could not identify the species of some of the sea turtles that had been captured or assess their condition accurately). Turtles in this category, which are considered "hooked in an unknown location," were assigned the average mortality of the turtles of their species with a known hook location. Turtles with an unknown condition (i.e. not recorded) were assigned the average within species of turtles with condition "okay," internally hooked, or externally hooked. For hardshell turtles that were hooked in an unknown location or unknown condition, the averaging was conducted over all turtles except leatherback turtles (Kleiber, 1998) and adjusted to consider the temperature or latitude of the interaction (McCracken, 2000). For example, there were 10 unidentified hardshell turtles observed taken from 1994 to 1999. The identity of these turtles was apportioned to loggerhead, olive ridley, or green turtle takes in the same proportion as observed takes of these species, and, except for green turtles, using the prediction models for each species. Based on the prediction models, the number of olive ridley turtles captured by the fishery were higher at temperatures greater than 24.22°C, whereas the number of loggerhead turtles captured by the fishery were higher at temperatures less than 23.77°C. If the sea surface temperature was not a clear indicator, the observed latitude was used to determine the species, since loggerhead takes were higher in the northern latitudes. In the two instances where the choice between the two species was completely unknown, the turtles were divided proportionally between the three hardshelled turtles (McCracken, 2000).

Because the abundance and distribution, migration and foraging patterns, and physiology vary so significantly between the four species of sea turtles that may be encountered by longliners fishing in the Pacific Ocean, their vulnerability to the Hawaii-based longline fishing operations also varies. The following sections review the past impacts that the Hawaii-based longline fishery has had on each of the sea turtle species.

6.3.5 Summary of Sea Turtle Responses to Interactions with the Fisheries

Green Sea Turtle. Assuming that patterns observed in the past represent future patterns, green sea turtles will be exposed primarily to longline fisheries based out of Hawaii and American Samoa. With the management regime proposed for the Hawaii-based fisheries, about 7 (95% confidence interval = 2 - 17) green turtles are expected to be captured by the fisheries each year. Unlike the past, most of these turtles (86%) would be captured by deep-set gear: about 6 green sea turtles (95% confidence interval = 1 - 13) are expected to be captured by deep-set gear, while 1 turtle (95% confidence interval = 1 to 5) is expected to be captured by shallow-set gear. We assume that another green turtle is likely to be captured by the longline fisheries based out of American Samoa in a single year or over any 5-year period.

Of the turtles that interact with the Hawaii-based longline fisheries, 5 (95% confidence interval = 1 - 11) are expected to die as a result of the exposure. Two to 4 of these 5 might be endangered green turtles from nesting beaches in Mexico while the remaining 1 or 2 green turtles killed in an interaction with longline gear would have originated in the Hawaiian nesting beaches. We assume that green turtles captured in the longline fisheries based out of American Samoan fisheries will be released unharmed.

Hawksbill Sea Turtle. Assuming that patterns observed in the past represent future patterns, we assume that individual hawksbill turtles will be exposed to the longline fisheries based out of American Samoa. The turtles that are captured by longline vessels based in American Samoa are likely to be individuals from western Pacific nesting beaches, although we cannot assign probabilities to any particular nesting aggregations.

Leatherback Sea Turtle. Assuming that patterns observed in the past represent future patterns, leatherback sea turtles will be exposed primarily to the Hawaii-based longline fisheries, although individual leatherback turtles are likely to be captured by longline fisheries based out of American Samoa. With the management regime proposed for the Hawaii-based longline fisheries, about 34 (95% confidence interval = 18 - 60) leatherback turtles are expected to be captured by the Hawaii-based longline fisheries each year. Interactions between the different types of gear and leatherback turtles is split almost equally. About 16 leatherback turtles (95% confidence interval = 9 - 30) are expected to be captured by the shallow-set gear, while about 18 turtles (95% confidence interval = 9 to 30) are expected to be captured by the deep set gear.

Of these turtles, 9 (95% confidence interval = 5 - 16) are expected to die as a result of the exposure. One of these 9 might be leatherback turtles from the eastern Tropical Pacific while the remaining 8 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 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. With the proposed fishery management regime, about 21 (95% confidence interval = 8 - 64) loggerhead turtles are expected to be captured by the Hawaii-based longline fisheries each year. As in the past, most of these turtles (81%) will be exposed to shallow-set gear: about 17 loggerhead turtles (95% confidence interval = 7 - 55) are expected to be captured by the shallow-set gear, while about 4 turtles (95% confidence interval = 4 to 9) are expected to be captured by deep set gear.

Of these turtles, 4 (95% confidence interval = 2 - 13) are expected to die as a result of the exposure. Most of these loggerhead turtles would be oceanic juveniles originated from nesting beaches in southern Japan while oceanic juveniles from the two nesting beaches on Yakushima Island have a low risk of being killed in an interaction with longline gear in any particular year, but a small risk of being killed in those interactions over several years.

Olive Ridley Sea Turtle. Assuming that patterns observed in the past represent future patterns, olive ridley sea turtles will be exposed primarily to the longline fisheries; the number of olive ridley turtles that have been captured by other fisheries included in the Pelagics FMP are expected to be minimal – if any exposure occurs at all. With the proposed fishery management regime, about 42 (95% confidence interval = 23 - 76) olive ridley turtles are expected to be captured by the Hawaii-based longline fisheries each year. As in the past, and because of their diving habit, most of these turtles (60%) will be exposed to deep-set gear: about 37 olive ridley sea turtles (95% confidence interval = 21 - 60) are expected to be captured by deep-set gear, while about 5 turtles (95% confidence interval = 2 to 16) are expected to be captured by shallow-set gear.

Of these turtles, 36 (95% confidence interval = 20 - 56) are expected to die as a result of that exposure. Twenty-seven of these 36 olive ridley turtles would be endangered turtles that have migrated from the eastern Tropical Pacific while the remaining 9 killed in an interaction with longline gear would have originated in the western Pacific or Indian Oceans.

6.4 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 fisheries 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 species to become threatened or endangered and continue to keep their populations suppressed. This section of the Opinion examines the physical, chemical, and biotic effects of the fisheries associated with the Pelagics 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, 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. Although leatherback sea turtles appear to be faring better in the Atlantic, the species remains at risk in the Atlantic. This assumption is reasonable based on the relationship between local and regional persistence in species (Gotelli, 2001). 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.

Table 6.3. Annual estimates of the number of different species of sea turtles captured and killed by the Hawaii-based longline fisheries

Species and Scenario	Exposure			Mortalities		
	Mean	Lower 95% CI	Upper 95% CI	Mean	Lower 95% CI	Upper 95% CI
Green Sea Turtles	**7.0005**	**2.1948**	**17.3292**	**4.9246**	**1.4492**	**11.3654**
Scenario 1						
Eastern Tropical Pacific	3.50025	1.0974	8.6646	2.4623	0.7246	5.6827
Hawaiian population	0.84006	0.263376	2.079504	0.590952	0.173904	1.363848
Mixture	2.66019	0.834024	6.585096	1.871348	0.550696	4.318852
Rounded Values*						
Eastern Tropical Pacific	4.0	1.0	9.0	2.0	1.0	6.0
Hawaiian population	1.0	0.0	2.0	1.0	0.0	1.0
Mixture	3.0	1.0	7.0	2.0	1.0	4.0
Leatherback Sea Turtle	**34.2201**	**18.3357**	**60.4231**	**9.4051**	**4.8839**	**16.0657**
Scenario 1						
Eastern Tropical Pacific	4.790814	2.566998	8.459234	1.316714	0.683746	2.249198
Western Pacific nesting aggregations	29.429286	15.768702	51.963866	8.088386	4.200154	13.816502
Rounded Values*						
Eastern Tropical Pacific	5.0	3.0	8.0	1.0	1.0	2.0
Western Pacific nesting aggregations	29.0	16.0	52.0	8.0	4.0	14.0
Loggerhead Sea Turtle	**20.6438**	**8.2839**	**63.6645**	**4.4639**	**1.6061**	**12.8839**
Scenario 1						
Yakushima nesting aggregation	1.03219	0.414195	3.183225	0.223195	0.080305	0.644195
Other Japanese nesting aggregations	19.61161	7.869705	60.481275	4.240705	1.525795	12.239705
Rounded Values*						
Yakushima nesting aggregation	1.0	0.0	3.0	0.0	0.0	1.0
Other Japanese nesting aggregations	20.0	8.0	60.0	4.0	2.0	12.0
Olive Ridley Sea Turtle	**41.9963**	**23.0448**	**76.0295**	**35.8665**	**20.1617**	**59.2033**
Scenario 1						
Eastern Tropical Pacific	31.077262	17.053152	56.26183	26.54121	14.919658	43.810442
Western Pacific and Indian Ocean	10.919038	5.991648	19.76767	9.32529	5.242042	15.392858
Rounded Values*						
Eastern Tropical Pacific	31.0	17.0	56.0	27.0	15.0	44.0
Western Pacific and Indian Ocean	11.0	6.0	20.0	9.0	5.0	15.0

* Because of rounding, these values may produce estimates that, when summed, are higher than the initial total

6.4.1 Humpback Whale

Based on the available information on interactions between humpback whales and the longline fisheries, humpback whales have been entangled in longline 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 Western Pacific Pelagic 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.

6.4.2 Hawaiian Monk Seal

Based on unconfirmed logbook data, monk seals may become entangled or hooked in longline fishing gear. However, no interactions between monk seals and the fisheries have been observed (by personnel in NMFS' Observer Program), which we interpret to indicate that the likelihood of interactions between the fisheries and monk seals is small. Further, the single report of an interaction indicated that the seal was entangled in a shallow set which are now prohibited under the Pelagics FMP. Based on these data, NMFS does not anticipate future interactions between monk seals and longline gear.

As a result, the Western Pacific Pelagic Fisheries are not likely to reduce the reproduction, numbers, or distribution of monk seals. As a result, they are not likely to reduce the monk seals' likelihood of surviving and recovering in the wild.

6.4.3 Sperm Whale

Based on the available information on interactions between sperm whales and the longline fisheries, sperm whales have been entangled in longline 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 sperm whales did not appear to have been injured or killed by the interaction. Because the Western Pacific Pelagic Fisheries have a small probability of interacting with endangered sperm 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 sperm whales. As a result, they are not likely to reduce the sperm whales' likelihood of surviving and recovering in the wild.

6.4.4 Green Turtles

Assuming that patterns observed in the past represent future patterns, green sea turtles will be exposed primarily to longline fisheries based out of Hawaii and American Samoa. With the management regime proposed for the Hawaii-based fisheries, about 7 (95% confidence interval =

2 - 17) green turtles are expected to be captured by the fisheries each year. Unlike the past, most of these turtles (86%) would be captured by deep-set gear: about 6 green sea turtles (95% confidence interval = 1 - 13) are expected to be captured by deep-set gear, while 1 turtle (95% confidence interval = 1 to 5) is expected to be captured by shallow-set gear. We assume that another green turtle is likely to be captured by the longline fisheries based out of American Samoa in a single year or over any 5-year period.

Of the turtles that interact with the Hawaii-based longline fisheries, 5 (95% confidence interval = 1 - 11) are expected to die as a result of the exposure. Two to 4 of these 5 might be endangered green turtles from nesting beaches in Mexico while the remaining 1 or 2 green turtles killed in an interaction with longline gear would have originated in the Hawaiian nesting beaches. We assume that green turtles captured in the longline fisheries based out of American Samoan fisheries will be released unharmed.

These green turtles that interact with the Hawaii-based longline fisheries will be members of the endangered Mexican (Pacific coast) or threatened Hawaiian (French Frigate Shoals) nesting aggregations. Out of eight green turtles caught by the Hawaii-based longline fishery, genetic analyses concluded that four of the eight turtles (50%) represented nesting aggregations from the eastern Pacific (Mexico), one turtle (12.5%) represented the Hawaiian nesting aggregations, the remaining three turtles (37.5%) could have been from either of these two nesting aggregations (P. Dutton, NOAA-Fisheries, personal communication, January, 2001) but may also have represented other nesting aggregations in the Pacific Ocean.

Life history information collected by observers suggests that the Hawaii-based longline fisheries are likely to capture sub-adult and adult green turtles (straight carapace lengths ranged from 28.5 cm to 73.5 cm with an average of 51.5 cm).

Historically, the longline fishery has been more likely to hook green turtles externally than to entangle them or hook them internally. The tendency to be hooked externally seemed to result from their diet: because green turtles primarily feed on benthic, marine algae, they seemed less likely to be attracted to the baited hooks used in the longline fishery. As a result they were less likely to swallow baited hooks, which would reduce their likelihood of being hooked internally. Further, because of their diet and foraging strategy (green turtles usually forage in water less than 10 meters deep), green turtles were more likely to interact with shallow-set gear than deep-set gear.

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 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 annual loss of about 5 adult or sub- adult green turtles) 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 (NMFS 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, NMFS, 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 (NMFS and USFWS, 1998a). More current estimates of the status and trend of this population are not available.

Clearly, the additional loss of between 1 and 3 adult or sub-adult, green turtles from these nesting aggregations each year 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 (NMFS 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 (NMFS and USFWS, 1998a).

Killing 1 to 3 of these green turtles each year 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 Hawaii-based longline 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; purse seine fisheries for tuna in the eastern tropical Pacific Ocean, and California/Oregon drift gillnet fisheries. 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. 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 Western Pacific Pelagic 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 a 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.

In the past, we have concluded that the additional mortalities caused by the Hawaii-based longline fishery appreciably contributed to the green turtle's risk of extinction. In the past, the Hawaii-based longline fishery interacted with an average of 40 green turtles each year; we estimated that as many as 23 of these turtles died as a result of these interactions (McCracken, 2000). In the past, most of those interactions and deaths were associated with the shallow-set component of the Hawaii-based longline fisheries, which as now been modified to reduce the number of sea turtles that are likely to be captured by the fisheries and to reduce the consequences of those interactions to the turtles that are captured.[30]

Nevertheless, we estimate that about 5 green turtles could still be killed in the proposed fisheries. Of these turtles, 2 to 4 (95% confidence interval = 1 - 6) are likely to be adult or sub-adult female green turtles from the eastern Pacific nesting aggregations and 1 to 3 (95% confidence interval = 0 - 5) are likely to be from the Hawaiian nesting aggregations (because our estimates for green sea turtles resulted in three categories — eastern tropical Pacific, Hawaiian, and a mixture of the two

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[30] As discussed previously, green 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.

— the ranges in these estimates overlap because the "mixture" could consist of turtles from the eastern tropical Pacific, Hawaii, or both. Because of this overlap, these numbers, as presented, do not sum to 5). 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 need to ask if the deaths associated with the proposed fisheries are likely to be exceeded by the number of younger turtles recruiting into the adult of 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. 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).

If the variance in the vital rates of green turtles in the eastern tropical Pacific Ocean or Hawaii are roughly the same order of magnitude as those of green turtles from the southern Great Barrier Reef, we would not be able to detect the effect of the remaining mortalities associated with the current fisheries on the survival rates of adult and sub-adult green turtles from the eastern tropical Pacific Ocean or Hawaiian (assuming that we had the data necessary to reliably estimate survival rates). We believe the number of green turtles that would be expected to interact with the current pelagic fisheries and die as a result of those interactions is so small that it would be masked by background variance, even considering the effects of the other sources of mortality that were discussed in the *Environmental Baseline*.

6.4.5 Hawksbill Sea Turtle

Hawksbill turtles occur in the water around the Hawaiian Islands (on Molokai, Maui and Hawaii), but they are not known to interact with the Hawaii-based longline fishery (there have been no reported or observed interactions between these pelagic longliners and hawksbill turtles). Based on an unconfirmed logbook report of an interaction between a hawksbill turtle and longline gear in American Samoa, there is a possibility of hawksbill turtles becoming incidentally entangled or hooked in longline fishing gear. However, no hawksbill turtles have been observed to interact with the longline fisheries, which indicates that the likelihood of an interaction is small. Moreover, the single report in vessel logbooks of an interaction indicated that the animal was taken in a shallow set and shallow sets are now prohibited under the Pelagics FMP. Based on the available data and the distribution of hawksbill turtles relative to the distribution of the pelagic fisheries, NMFS does not anticipate future interactions between hawksbill turtles and longline gear. Consequently, the Western Pacific Pelagic Fisheries are not likely to reduce the reproduction, numbers, or distribution of hawksbill turtles. As a result, they are not likely to reduce the turtles' likelihood of surviving and recovering in the wild.

6.4.6 Leatherback Turtles

Assuming that patterns observed in the past represent future patterns, leatherback sea turtles will be exposed primarily to the Hawaii-based longline fisheries. The number of leatherback turtles that have been captured by other fisheries included in the Pelagics FMP are expected to be minimal one leatherback turtle was reported to have been captured but not killed by the longline fisheries based in American Samoa in 1999, no interactions have been reported since then, but it is important to note that these fisheries are not observed, so the absence of reports may only reflect the absence of sampling rather than the absence of interactions).

As proposed, the Hawaii-based longline fisheries would be modified to require longline vessels to use management measures that have proven effective at reducing the number and rate of interactions between leatherback and loggerhead sea turtles: the use of 18.0 10° offset circle hook with squid bait. Under the proposed management regime for the Hawaii-based longline fisheries, which include these gear modifications, about 34 (95% confidence interval = 18 - 60) leatherback turtles are expected to be captured by the Hawaii-based longline fisheries each year. Interactions between the different types of gear and leatherback turtles is split almost equally. About 16 leatherback turtles (95% confidence interval = 9 - 30) are expected to be captured by the shallow-set gear, while about 18 turtles (95% confidence interval = 9 to 30) are expected to be captured by the deep set gear.

Based on the limited genetic sampling from the action area, about 86% of the leatherback turtle sample (12 out of 14 genetic samples) originated from western Pacific nesting beaches (P. Dutton *et al.*, 2000; P. Dutton, NOAA-Fisheries, personal communication, January, 2001). If the longline fisheries affect leatherback turtle populations proportional to their relative abundance in the action area, about 29 of the 34 leatherback turtles that are expected to be captured in the Hawaii-based longline fisheries each year would come from nesting aggregations in the western Pacific Ocean. These turtles could represent individuals from Indonesia (Jamursba-Medi or War-Mon), Papua New Guinea (Kamiali), Malaysia (Terengganu), the Solomon Islands, or Fiji, although satellite tracks from leatherback turtles tagged in Papua New Guinea suggest that these turtles tend to migrate south instead of north, which would take them away from Hawaiian waters. Further, the abundance of the nesting aggregations in Indonesia relative to the small size of the other nesting aggregations suggests that the interactions between Indonesian leatherback turtles and the Hawaii-based longline fisheries are most likely.

The remaining 14 percent of the interactions, or about 5 of the leatherback turtles captured in a year, would represent turtles from the eastern Pacific Ocean. These turtles could represent individuals from nesting aggregations along the coast of Mexico, Costa Rica, or Panama, although turtles from these nesting aggregations may only migrate into Hawaiian waters when oceanic phenomena like El Nino events prevent them from migrating south to the coasts of Peru and Chile. Several investigators who have followed leatherback turtles equipped with satellite tags have reported that leatherback turtles from the beaches of Mexico and Costa Rica migrate through the equatorial current towards the coasts of Peru and Chile (Chandler 1991, Eckert 1997, Marquez and Vellanueva 1993, Morreale et al. 1996). Eckert (1997) suggested that these turtles migrate toward the coast of South America where upwelling water masses provide an abundance of prey. Although these data suggest that the Hawaii-based longline fisheries are more likely to interact with leatherback turtles from Indonesia, over a period of several years, we would expect these fisheries to interact with turtles from the other, smaller nesting aggregations.

Published estimates of the abundance of nesting female leatherbacks in the Pacific Ocean have established that leatherback populations have collapsed or have been declining at all major Pacific basin nesting beaches over the past two decades (Spotila *et al.*, 1996; NMFS 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 Section III,. *Status of the Species and Environmental Baseline*).

Leatherback turtles, like green turtles discussed previously, are long-lived, have high adult survival rates, and delayed maturity; as a result, we assume that changes in the survival of adult and sub-adult stages of leatherback turtles can 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 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 annual loss of about 9 adult or sub-adult leatherback turtles) would be expected to appreciably reduce the leatherback sea turtle's likelihood of surviving and recovering in the wild. This conclusion is based on the number of leatherback 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.*

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 *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 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 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 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. Our assessment of this population also suggest that this population is stable or increasing, on average. However, the confidence intervals around our estimates suggest that the population may, in fact, be declining and that most population trajectories would show slight declines.

The western Pacific nesting aggregation consists of about 1,000 adult females (about 500 in Indonesia, about 150 in the Solomon Islands, about 400 in Papua New Guinea, and a handful in Malaysia). Killing three of these adult leatherback turtles each year would reduce the abundance of this nesting aggregation. If we assume that all of the 4 to 14 adult turtles that are killed in interactions with the Hawaii-based longline fisheries are females (an incorrect assumption since at least some of the turtles captured in the fishery have been male), then the fishery would reduce the reproduction of this nesting aggregation, although, the consequences of losing a female turtle on the dynamics of the turtle's population will vary depending on whether the adult female dies before or after she lays her eggs.

If we assume that, in most years, all of these turtles migrate into the action area from Indonesia or Papua New Guinea, then the higher mortality estimate would represent about 1.0 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. As with the 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, 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.

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 of 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. 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.

If the leatherback turtles originating in the western Pacific are a random mix of individuals from Indonesia or Papua New Guinea, we would expect their combined populations, given their size, to be able to withstand the small mortality levels associated with the fisheries without measurable effect on the population's extinction risks. If, the leatherback turtles killed in the fisheries were exclusively from Indonesia, the effect of these mortalities would be small and might appear to be trivial, but those mortalities might have longer-term consequences for this population because of accumulating effects. If the leatherback turtles killed in the fisheries were exclusively from Papua New Guinea, the effect of these mortalities would be small, but those mortalities are less likely to be trivial for this nesting aggregation in any particular year or over several years. Although stronger cohorts in this nesting aggregation might be able to withstand these mortalities, these mortalities would be more significant to weaker cohorts and could cause those cohorts to decline. If the leatherback turtles killed in the fisheries were exclusively or primarily from the Malaysian nesting aggregation, they would be certain to drive that nesting aggregation closer to extinction.

Nesting populations of leatherback turtles in the eastern Pacific Ocean are declining along the Pacific coast of Mexico and Costa Rica. According to reports from the late 1970s and early 1980s, three beaches located on the Pacific coast of Mexico support as many as half of all leatherback turtle nests. Since the early 1980s, the eastern Pacific Mexican population of adult female leatherback turtles has declined to slightly more than 200 during 1998-99 and 1999-2000 (Sarti *et al.* 2000). Spotila *et al.* (2000) reported the decline of the leatherback turtle population at Playa Grande, Costa Rica, which had been the fourth largest nesting colony in the world. Between 1988 and 1999, the nesting colony declined from 1,367 to 117 female leatherback turtles. Based on their models, Spotila *et al.* (2000) estimated that the colony could fall to less than 50 females by 2003-2004. Although these predictions have not proved true, our assessment suggests that this population has a high risk of extinction (declining to 1 or 0 females) in the one human generation (about 20 years) if its trajectory does not change.

In the past, NMFS has concluded that the additional mortalities caused by the Hawaii-based longline fisheries and other domestic fisheries appreciably contributed to the leatherback turtle's risk of extinction. Although the number of leatherback turtles killed in these fisheries were "small" relative to other fisheries in the Pacific Ocean, they were not "small" in terms of their impacts on the survival and recovery of leatherback turtles.

Common perceptions to the contrary, several, published studies have demonstrated that the death of "small" numbers of individuals can substantially-increase a species' risk of extinction. For example, Walters (1992) chronicled how the incremental loss of small numbers of individuals contributed to the extinction of the endangered dusky seaside sparrow (*Ammodramus maritimus nigrescens*). Spotila et al. (1996, 2001) used population models to demonstrate that leatherback sea turtles in the eastern tropical Pacific could not withstand low levels of adult mortalities. Fujiwara and Caswell (2001) used population models to demonstrate that preventing just two adult, female North Atlantic right whales (*Eubalaena glacialis*) would be sufficient to change the declining trend of this endangered species. Wiegand et al. (1998) used population models to demonstrate that annual anthropogenic mortalities ranging between 0 and 10 individuals per year over a 15-year interval increased the extinction risk of endangered brown bears (*Ursus arctos*) in Spain. Saether et al. (1998) used demographic models to reach similar conclusions for Scandinavian brown bears. Studies of species like the endangered Sonoran pronghorn antelope (*Antillocarpa americana sonoriensis*), Iberian lynx (*Lynx pardinus*), Mediterranean monk seal (*Monachus monachus*), Florida panther (*Felis concolor coryi*), Hawaiian crow (*Corvus hawaiiensis*), California condor (*Gymnogyps californicus*), Puerto Rican parrot (*Amazona vittata*), among others, have also demonstrated that small mortalities — a handful or individuals — would increase these species' risk of extinction.

At the same time, almost every species has evolved to withstand the loss of some of their numbers, even when they are experiencing declines; otherwise any species that experienced any decline would begin a decline to extinction with any additional death within its population. Species and populations persist because, above certain population levels, there usually isn't a one-to-one (or several-to-one) relationship between the species' risk of extinction and the death of individual plants or animals. That is, the death of each individual usually does not result in a corresponding increase in the species' risk of extinction. Species like the North Atlantic right whale and others we identified in the previous paragraph are endangered because they have declined to a point where we can draw a direct relationship between the loss of individual adults and increases in the species' risk of final extinction.

Other species are endangered because they appear likely to decline to the condition of these species in the foreseeable future. For these species, the consequences of the death of small numbers of individuals in different populations will usually depend on which populations those individuals represent and the population's size, growth rates over time (which reflect differences in the numbers of individuals that die in the population compared with the number that are born into the population over the same time interval), birth rates, gender ratios, age structure, and how these rates vary with time. These characteristics of populations will determine the relationship between the loss of individuals and the population's or species' extinction risk.

In the past, the U.S. Pacific pelagic fisheries interacted with an average of 112 (95% confidence interval 75-157) leatherback turtles and caused the death of between 24 to 49 of these turtles each year. In the past, we concluded that these mortality levels would be expected to appreciably reduce the leatherback turtle's chances of surviving and recovering in the wild. That is, we concluded that these mortality levels would combine to increase the species' extinction risk. The proposed management regime is expected to result in the death of about 8 (95% confidence interval = 4 to 14) adult or sub-adult leatherback turtles from the western Pacific nesting aggregations and 1 (95% confidence interval = 1 to 2) from the would represent a small proportional change in the survival rates of female turtles in those life history stages (0.003 percent).

Given the size of leatherback sea turtles populations in the western Pacific region, particularly the nesting aggregations in Indonesia and Papua-New Guinea these leatherback turtles probably represent and the growth rates of this population, we do not expect the death of about 8 adult or sub-adult sea turtles to measurably increase this population's extinction risk. Given the size of leatherback sea turtles populations in the eastern tropical Pacific, we do not expect the death of 1 adult or sub-adult sea turtles to measurably increase this population's extinction risk. We believe this statement remains true despite the declining trend of the leatherback turtle populations in both the eastern and western Pacific.

We also expect the variance in the survival and fecundity rates of the western Pacific leatherback sea turtle populations to make it more difficult to detect increases in the population's extinction risks from these small number of deaths. Chaloupka and Limpus (2002) reported survival rates for adult green turtles in the southern Great Barrier reef region of Australia averaged 0.875 percent (with 95% confidence interval 0.84-0.91). Doak et al (1994) and Wisdom et al (2000) reported that the vital rates of adult and sub-adult desert tortoises (*Gopherus agassizii*) varied by about 8 to 15 percent. Woolfenden and Fitzpatrick (1984) reported that the estimated annual survival rates of adult Florida scrub jays (a threatened species) varied by about 11 percent (mean of 0.820 ±0.091). If the variance in the vital rates of leatherback turtles in the Pacific Ocean are roughly the same order of magnitude as those of green turtles from the southern Great Barrier Reef, we would not be able to detect the effect of the remaining mortalities associated with the current fisheries on the survival rates of adult and sub-adult leatherback turtles in the western Pacific (assuming that we had the data necessary to reliably estimate survival rates).

Because of the size of leatherback turtle populations in the eastern and western Pacific, relative to the small number of individual leatherback sea turtles that are expected to be captured and killed in the proposed fisheries in any particular year, we do not expect these mortalities to appreciably reduce the population's likelihood of surviving and recovering in the wild. Because these mortalities are not likely to reduce the population's likelihood of surviving and recovering in the wild, we do not expect these mortalities to reduce the species' likelihood of surviving and recovering in the wild.

6.4.7 Loggerhead Turtles

With the proposed fishery management regime, about 21 (95% confidence interval = 8 - 64) loggerhead turtles are expected to be captured by the Hawaii-based longline fisheries each year. As in the past, most of these turtles (81%) will be exposed to shallow-set gear: about 17 loggerhead turtles (95% confidence interval = 7 - 55) are expected to be captured by the shallow-set gear, while about 4 turtles (95% confidence interval = 4 to 9) are expected to be captured by deep set gear.

Of these turtles, 4 (95% confidence interval = 2 - 13) are expected to die as a result of the exposure. Most of these loggerhead turtles would be oceanic juveniles originated from nesting beaches in southern Japan while oceanic juveniles from the two nesting beaches on Yakushima Island have a low risk of being killed in an interaction with longline gear in any particular year, but a small risk of being killed in those interactions over several years.

Historically, most of the loggerhead turtles that interact with the fishery were either hooked internally or externally. The tendency to be hooked internally probably resulted from their diet: Loggerhead turtles in north Pacific pelagic habitats are opportunistic, omnivorous predators of the surface layer, feeding both by swallowing floating prey whole, biting off prey items from larger floating objects, or both. Based on past patterns of the condition of sea turtles that have been observed in the fishery and expected mortality rates for turtles given their condition when they were observed, we have estimated that about 8 of the 14 loggerhead turtles would be expected to die each year as a result of their interactions with the fisheries.

However, it is important to place these numbers into perspective: between July 2001 and June 2002 (when the fishery was modified to eliminate the targeted swordfish fishery and the shallow sets associated with it), three loggerhead turtles were captured in the fishery, two of those three turtles were captured by vessels that are believed to have been illegally using shallow sets to target swordfish. All of the 175 loggerheads that interacted with the Hawaii-based longline fishery from 1994 through March, 2001, were captured by longliners using shallow sets (i.e. target depths less than 100 meters, using less than 10 hooks per float, fishing at night, using lightsticks). 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.*, in press). Although loggerhead turtles have been reported to dive to depths of 128 meters, they spend most of their time (90%) at the surface or at depths less than 40 meters; therefore, loggerheads were more likely to interact with shallow sets than deep sets, which generally target depths greater than 100 meters. Eliminating the targeted swordfish fishery is expected to virtually eliminate the likelihood of interactions between the current fishery and loggerhead turtles.

Placed in this perspective, the estimated number of interactions between the current fishery and loggerhead turtles was biased by vessels that were allegedly fishing illegally; the Council and NOAA-Fisheries have taken further action to reduce the likelihood of such illegal activity in the future. As a result, although we estimated that 14 loggerhead turtles would be captured by the fishery and 8 of those turtles would die as a result of their capture, those estimates may overestimate the risk by more than 66%. In July 2002, a loggerhead turtle was caught in a deep set fishing at 13%N latitude, an interaction than diverged from NOAA-Fisheries' past observations of the overlap between this species' pelagic distribution and behavior and the deep-set portion of the longline fishery; this recent observation is evidence that the current fishery poses a risk to loggerhead turtles even if that risk is much smaller than our current estimates.

Killing about 4 (95% confidence interval = 2 - 13) pelagic juvenile loggerhead turtles each year would reduce the numbers of individuals in the species, particularly since population estimates for this species are based on estimated numbers of adult, female turtles. 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.

Within the Pacific Ocean, loggerhead sea turtles 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 C; 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 *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; purse seine fisheries for tuna in the eastern tropical Pacific Ocean, and California/Oregon drift gillnet fisheries. 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 Pelagic FMP fisheries added to the current status and trend of the Japanese loggerhead population would increase the Japanese loggerhead population's rate of decline.

Although the mortalities associated with the Western Pacific Pelagic Fisheries would clearly 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. As we discussed in the previous turtle narratives, we generally identify reductions in a listed species' likelihood of surviving and recovering in the wild by quantitatively or qualitatively analyzing the probable effect of changes in a 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.

In the past, we have concluded that the additional mortalities caused by the Hawaii-based longline fishery appreciably contributed to the green turtle's risk of extinction. In the past, the Hawaii-based longline fishery interacted with an average of 418 loggerhead turtles each year; we estimated that as many as 73 of these turtles died as a result of these interactions (McCracken 2000). The proposed management measures will dramatically reduce the number of these interactions and the consequences of those interactions for loggerhead sea turtles. Nevertheless, we estimate that between about 4 pelagic juvenile loggerhead turtles from the 40 nesting aggregations in southern Japan and perhaps 1 loggerhead turtle from the 2 nesting aggregations on Yakushima Island could still be killed in the proposed fishery.

As with the green and leatherback 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, although these results may say more about the power of the models than they say about the effect of these mortalities on the different nesting aggregations.

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 of 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. 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.

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 fewer than 1,000 female loggerheads return to Japanese beaches per nesting season. Two of the most important beaches in Japan, 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, with about 300 to 400 nesters in 2000.

Given the size of loggerhead sea turtles populations in Japan, we do not expect the death of about 5 oceanic, juvenile loggerhead sea turtles to measurably increase the extinction risk of one or more of the Japanese nesting aggregations. We believe this statement remains true despite the declining trend of the leatherback turtle populations in both the eastern and western Pacific. We also expect the variance in the survival and fecundity rates of the Japanese loggerhead sea turtle populations to make it more difficult to detect increases in the population's extinction risks from these small number of deaths of juvenile turtles. Chaloupka and Limpus (2002) reported survival rates for adult green turtles in the southern Great Barrier reef region of Australia averaged 0.875 percent (with 95% confidence interval 0.84-0.91). Doak et al (1994) and Wisdom et al (2000) reported that the vital rates of adult and sub-adult desert tortoises (*Gopherus agassizii*) varied by about 8 to 15 percent. Woolfenden and Fitzpatrick (1984) reported that the estimated annual survival rates of adult Florida scrub jays (a threatened species) varied by about 11 percent (mean of 0.820 ±0.091). If the variance in the vital rates of leatherback turtles in the Pacific Ocean are roughly the same order of magnitude as those of green turtles from the southern Great Barrier Reef, we would not be able to detect the effect of the remaining mortalities associated with the current fisheries on the survival rates of adult and sub-adult leatherback turtles in the western Pacific (assuming that we had the data necessary to reliably estimate survival rates).

Because of the size of loggerhead turtle populations in the Pacific, relative to the small number of individual loggerhead sea turtles that are expected to be captured and killed in the proposed fisheries in any particular year, we do not expect these mortalities to appreciably reduce the likelihood of loggerhead sea turtle's surviving and recovering in the wild in the Pacific Ocean. Because these mortalities are not likely to reduce the turtles likelihood of surviving and recovering in the wild in the Pacific Ocean, we do not expect these mortalities to reduce the species' likelihood of surviving and recovering in the wild.

6.4.8 Olive Ridley Turtle

Assuming that patterns observed in the past represent future patterns, olive ridley sea turtles will be exposed to longline fisheries based out of Hawaii and American Samoa. The number of olive ridley turtles that have been captured by other fisheries included in the Pelagics FMP are expected to be minimal one: olive ridley turtle was reported to have been captured but not killed by the longline fisheries based in American Samoa in 1999. No interactions have been reported since then, but it is important to note that these fisheries are not observed, so the absence of reports may only reflect the absence of sampling rather than the absence of interactions.

With the proposed management regime for the Hawaii-based longline fisheries, about 42 (95% confidence interval = 23 - 76) olive ridley turtles are expected to interact with the Hawaii-based longline fisheries each year. As in the past, and because of their diving habit, most of these turtles (60%) will be exposed to deep-set gear: about 37 olive ridley sea turtles (95% confidence interval = 21 - 60) are expected to interact with deep-set gear, while about 5 turtles (95% confidence interval = 2 to 16) are expected to interact with shallow-set gear.

Based on genetic sampling of the olive ridley sea turtles captured in these fisheries in the past, these turtles will represent nesting beaches both the eastern and western Pacific Ocean as well as the Indian Ocean (P. Dutton *et al.*, 2000; P. Dutton, NOAA-Fisheries, personal communication, January, 2001). If the longline fisheries affect olive ridley turtle populations proportional to their relative abundance in the action area, about 31 of the 41 olive ridley turtles that are expected to be captured in the Hawaii-based longline fisheries each year would represent endangered olive ridley turtles from the eastern tropical Pacific , while 11 of the 41 olive ridley turtles would represent turtles from the western Pacific or Indian Oceans.

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* NMFS 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. (NMFS 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 year, the mortalities associated with the U.S. pelagics 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 Hawaii- based longline 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.

Nevertheless, 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 current fisheries without appreciable reductions in the olive ridley turtle's likelihood of the surviving and recovering in the wild.[31]

7.0 CUMULATIVE EFFECTS

Cumulative effects[32] 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 (50 CFR 402.02). Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

Most of the fisheries described as occurring within the action area (Section III. *Status of the Species* and *Environmental Baseline*), are expected to continue as described into the foreseeable future. Therefore, NMFS 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, NMFS 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, NMFS expects that the levels of take of sea turtles described for each of the fisheries, except the California longline fishery, which will take fewer sea turtles if the proposed regulatory regime is made final, and non-fisheries will continue at similar levels into the foreseeable future.

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[31] 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.

[32] "Cumulative effects," as defined for the purposes of the Endangered Species Act in 50 CFR 402.14, should not be confused with the term "cumulative impact" as defined for the purposes of the National Environmental Policy Act of 1969 (42 U.S.C. 4321). "Cumulative effects" are limited to the effects of future, non-federal actions in an action area while "cumulative impacts" are "the impacts on the environment which results from the incremental impact of the action when added to other past, present, and reasonably foreseeable future actions regardless of what agency or person undertakes such other actions. Cumulative impacts can result from individually minor but collectively significant actions taking place over a period of time." (40 CFR 1508.7).

8.0 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 pelagic fisheries in the Western Pacific Region under the Pelagics FMP is not likely to jeopardize the continued existence of green turtles.

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 pelagic fisheries in the Western Pacific Region under the Pelagics FMP is not likely to jeopardize the continued existence of leatherback 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 pelagic fisheries in the Western Pacific Region under the Pelagics FMP is not likely to jeopardize the continued existence of loggerhead 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 pelagic fisheries in the Western Pacific Region under the Pelagics FMP is not likely to jeopardize the continued existence of olive ridley turtles.

8.0 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and protective regulations pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or attempt to engage in any such conduct. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the reasonable and prudent measures and terms and conditions of the Incidental Take Statement.

Section 7(b)(4)(c) of the ESA specifies that in order to provide an incidental take statement for an endangered or threatened species of marine mammal, the taking must be authorized under section 101(a)(5) of the MMPA. Since no incidental take of listed marine mammals is expected or has been authorized under section 101(a)(5) of the MMPA, no statement on incidental take of endangered whales is provided and no take is authorized. Nevertheless, NOAA-Fisheries' Office of Sustainable Fisheries (F/SF) must immediately (within 24 hours, if communication is possible) notify the NOAA-Fisheries' Office of Protected Resources if an endangered marine mammal is "taken" in any of the fisheries authorized under the Pelagics FMP. Following issuance of such regulations or authorizations, NOAA-Fisheries may amend this biological opinion to include an incidental take statement for humpback whales, sperm whales, or Hawaiian monk seals, as appropriate.

8.1 Amount or extent of take

NOAA-Fisheries believes that the following levels of incidental take may be expected to occur associated with the proposed action. The reasonable and prudent measures, with their implementing terms and conditions, are designed to minimize the impact of incidental take that might otherwise result from the proposed action. If, during the course of the action, this level of incidental take is exceeded in any of the three groups of fisheries for which individual incidental take level are given in the tables that follow, the Pacific Islands Region's Sustainable Fisheries Division must immediately reinitiate formal consultation with the Office of Protected Resources pursuant to Criterion 2 of the section 7 regulations (50 CFR 402.16).

8.1.1. Pelagic Longline Fisheries for Swordfish, Tuna, and Shark

The tables that follow identify the annual number of sea turtles that are expected to be captured, injured, or killed by one or more of the proposed fisheries managed by the Fishery Management Plan for Pelagic Fisheries of the Western Pacific Region. These tables are based on the estimates discussed in the Exposure Analysis, Response Analysis, and Risk Analysis sections of this biological opinion and are organized as follows: Hawaii-based longline fisheries using shallow-set gear (Table 8.1), Hawaii-based longline fisheries using deep-set gear (Table 8.2), other fisheries managed under the Pelagics Fishery Management Plan (Table 8.3).[33]

Table 8.1 The annual number of turtles expected to be captured or killed in the shallow-set, longline fisheries based out of Hawaii

	number captured	number killed
green sea turtles	1	1
hawksbill sea turtles	0	0
leatherback sea turtles	16	2
loggerhead sea turtle	17	3
olive ridley sea turtles	5	1

As discussed elsewhere in this Opinion, the estimates contained in Table 8.1 (above) assume that the new management measures (circle hooks, mackerel-style bait, and de-hooking methods) will

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[33] The number of individual sea turtles that are anticipated to be "taken" incidental to the fisheries was derived from our prior experience with the fisheries (that is, past interactions). These numbers are anticipated (because they are expectations that have yet to occur) but they are also estimates (because these expectations are based on data we collected in the past).

be as effective in the Pacific Ocean as they have been in the Atlantic Ocean. Currently, these methods will only apply to the longline fisheries based out of Hawaii using gear configured for shallow-sets that operate north of the equator and not to the shallow-set longline fisheries operating south of the equator or longline fisheries based out of Hawaii using gear configured for deep-sets. These shallow-set longline fisheries will have exceeded this level of incidental take if and when estimates from observer reports exceed these estimates of the number of turtles (in Table 8.1) captured in the fisheries, the number of turtles killed in the fisheries, or both.

The annual estimates of the incidental take associated with those fisheries are described in the following table. These fisheries will have exceeded the levels of incidental take outlined in Table 8.2 if and when the fishery is estimated to have exceeded these estimated values:

Table 8.2 The annual number of turtles expected to be captured or killed in the deep-set, longline fisheries based out of Hawaii

	number captured	number killed
green sea turtles	6	5
hawksbill sea turtles	0	0
leatherback sea turtles	18	7
loggerhead sea turtle	4	2
olive ridley sea turtles	37	35

All five species of sea turtles may be taken in the other fisheries authorized by the Pelagics FMP. These fisheries include all of the handline fisheries, troll fisheries, pole and line fisheries managed under the Pelagics Fisheries Management Plan as well as the longline fisheries based out of American Samoa. The known level of effort and the selectivity of the gear used in most of these fisheries has led NOAA-Fisheries to conclude that few sea turtles, if any, are captured, injured, or killed by these fisheries. Because NOAA-Fisheries observers have not worked with these fisheries, most of the sea turtles that have been reported to have been captured in these fisheries have not been identified to species, therefore we identify the species as hardshell (green, hawksbill, loggerhead, and olive ridley sea turtles) or leatherback sea turtles. In the former case, any interaction with one of these species would be considered a "take" for the purposes of Table 8.3.

Table 8.3 The annual number of turtles expected to be captured or killed in the handline fisheries, troll fisheries, pole and line fisheries managed under the Pelagics Fisheries Management Plan as well as the longline fisheries based out of American Samoa

	number captured	number killed
hardshell sea turtle	6	1
leatherback sea turtle	1	0

8.2 Impact of the Take

In the accompanying Opinion, NOAA-Fisheries determined that these levels of anticipated take are not likely to result in jeopardy to the green turtle, hawksbill, humpback whale, leatherback turtle, loggerhead turtle, monk seal, olive ridley turtle, or sperm whale when the reasonable and prudent measures are implemented.

8.3 Reasonable and Prudent Measures

Section 7(b)(4) of the ESA requires that when an agency action is found to comply 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 specifying the impact of any incidental taking. It also states that reasonable and prudent measures necessary to minimize impacts, and terms and conditions to implement those measures be provided and must be followed to minimize those impacts. Only incidental taking by the federal agency or applicant that complies with the specified terms and conditions is authorized.

NOAA-Fisheries believes the following reasonable and prudent measures, as implemented by the terms and conditions, are necessary and appropriate to minimize impacts to sea turtles and monitor levels of incidental take. 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 of the Pelagic Fisheries to continue.

1. NOAA-Fisheries shall collect data on capture, injury and mortality of sea turtles in addition to life history information on longline fishing vessels.

2. NOAA-Fisheries shall develop a system that will enable NOAA-Fisheries to collect basic listed species bycatch data in the troll, handline, and purse seine fisheries under the Pelagics FMP and assign these interactions into the categories developed through the February 2004 post-hooking mortality guidelines (either as drafted or amended by NOAA-Fisheries).

3. Sea turtles captured alive shall be released from fishing gear in a manner that minimizes injury and the likelihood of further gear entanglement or entrapment.

4. Comatose and lethargic sea turtles shall be retained on board, handled, resuscitated, and released according to established procedures.

5. Sea turtles that are dead when brought aboard a vessel or that die while on-board shall be disposed of at sea unless NOAA-Fisheries requests retention of the carcass for sea turtle research.

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 shall continue the observer program aboard Hawaii-based limited access permit longline vessels to collect data on the incidental take of marine mammals, sea turtles, and other protected species. No vessel using shallow-set gear in the Hawaii-based fisheries shall be permitted to fish without observer coverage. Observer coverage in the deep-set longline fisheries generally shall be maintained at an annual average level of at least 20 percent.

    1B. NOAA-Fisheries shall establish an observer program, where feasible, aboard longline vessels fishing under a Pelagics FMP general permit or a limited access permit for the American Samoa-based longline fishery, should such a permit program be established. The purpose of the observer program is to provide NOAA-Fisheries with information necessary to determine if these fisheries interact with listed species and what level of impact the fisheries might have on those species. The feasibility of establishing an observer program aboard these vessels may depend upon the space available on the vessel to house and maintain an observer to carry out their duties.

    1C. Observer programs shall collect information regarding the incidental capture, injury, and mortality of sea turtles by species, gear and set information in which each interaction occurred, and life history information.

    1D. NOAA-Fisheries shall also collect life history information on sea turtles captured by longline fisheries, including species identification; measurements, including direct measure or visual estimates of tail length; condition; skin biopsy samples; and estimated length of gear left on the turtle at release. To the extent practicable, these data should allow NOAA-Fisheries to assign these interactions into the categories developed through the February 2004 post-hooking mortality guidelines (either as drafted or amended by NOAA-Fisheries).

    1E. NOAA-Fisheries observers shall record the presence or absence of tags on all sea turtles captured by longline fisheries.

    1F. Data collected by observers shall be made available on a quarterly basis. "Quarterly Status Reports" shall be sent to the Assistant Regional Directors of Protected Resources and Sustainable Fisheries in NOAA-Fisheries' Pacific Islands Regional

Office and distributed to NOAA-Fisheries' Sea Turtle Coordinators in Honolulu, Hawaii (when established); Long Beach, California, and Silver Spring, Maryland.

2. The following terms and conditions implement reasonable and prudent measure No. 2.

2A. NOAA-Fisheries shall coordinate with the Forum Fisheries Agency observer program to collect life history information on sea turtles, such as species identification, measurements, condition, skin biopsy samples, the presence or absence of tags, and the application of flipper tags if none are present.

2B NOAA-Fisheries, in collaboration with the Western Pacific Fishery Management Council, shall develop a system to collect basic listed species bycatch data associated with non-longline pelagic fishing vessels fishing with hook-and-line within EEZ waters of the western Pacific region.

3. The following term and condition implements reasonable and prudent measure No. 3.

3A. NOAA-Fisheries shall continue to conduct protected species workshops for skippers of vessels registered for use with longline fishing permits issued under the Pelagics FMP to facilitate proficiency on mitigation, handling, and release techniques for turtles, as outlined in 50 CFR 223.206(d)(1).

3B. NOAA-Fisheries shall include information on sea turtle biology and ways to avoid and minimize sea turtle impacts to promote sea turtle protection and conservation in the protected species workshops for skippers of longline vessels registered for use with permits issued under the Pelagics FMP.

3C. NOAA-Fisheries shall continue to include sea turtle resuscitation techniques and sea turtle biology information during observer training.

3D. All sea turtles shall be removed from fishing gear or brought on deck prior to continuing with gear retrieval.

3E. Personnel aboard a vessel registered for use with a longline permit issued under the Pelagics FMP must remove the hook from a turtle, if feasible, as quickly and carefully as possible to avoid injuring or killing the turtle. Each vessel must carry a line clipper. If a hook cannot be removed (e.g., the hook is deeply ingested or the animal is too large to bring aboard), the line clipper must be used to cut the line as close to the hook as practicable and remove as much line as possible prior to releasing the turtle.

3F. Each longline vessel registered for use with a longline permit issued for use under the Pelagics FMP must carry a sea turtle dip net to hoist a sea turtle onto the deck, if practicable, to facilitate the removal of the hook. If the vessel is too small to carry a dipnet, sea turtles must be eased onto the deck by grasping its carapace or flippers, if practicable, to facilitate the removal of the hook. Any sea turtle brought on board must not be dropped on to the deck

3G. Each longline vessel registered for use with a longline permit issued under the Pelagics FMP must have a wire or bolt cutter aboard the vessel capable of cutting through a hook that may be imbedded externally, including the head/beak area of a turtle.

3H. NOAA-Fisheries shall make available and disseminate information on sea turtle biology and ways to avoid and minimize sea turtle impacts for promoting sea turtle protection and conservation at appropriate Regional forums (such as the Heads of Fisheries Meetings of the Pacific Community) in the western Pacific region.

3I. In the event of an interaction with a sea turtle, an operator of a vessel not using longlines but using hooks (i.e, handline, troll, and pole-and-line vessels) to target Pacific pelagic management unit species in waters of U.S. western Pacific EEZ, must handle the sea turtle in a manner to minimize injury and promote post-hooking survival. 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, the fishing line must be severed and as much line removed prior to releasing the turtle.

4. The following term and condition implements reasonable and prudent measure No. 4.

4A. Operators of vessels registered for use with longline permits issued under the Pelagics FMP shall bring comatose sea turtles aboard, if feasible, and perform resuscitation techniques according to the procedures described at 50 CFR 223.206 (d)(1) and 660.32(b),(c), and (d).

4B. If an observer is aboard the vessel, the observer shall perform resuscitation techniques on comatose sea turtles.

4C. In the event of an interaction with a sea turtle, an operator of a vessel not using longlines but using hooks (i.e, handline, troll, and pole-and-line vessels) to target Pacific pelagic management unit species in waters of U.S. western Pacific EEZ, must handle the sea turtle in a manner to minimize injury and promote post-hooking survival as outlined in 50 CFR 660.32 (c) and (d).

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 NOAA-Fisheries requests the turtle be kept for further study.

9.0 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 research modifications to existing gear that (1) reduce the likelihood of interactions between sea turtles and longline fishing gear and (2) reduce the immediate or delayed mortality rates of captured turtles. In particular, NOAA-Fisheries should develop circle hooks that are suitable for use in deep-set longline gear. Any research funded or implemented by NOAA-Fisheries that is likely to increase the number of turtles captured or killed in these fisheries beyond the levels considered in this Opinion or that involve fishing in a manner not considered in this biological opinion must be covered by a research and enhancement permit pursuant to section 10(a)(1)(a) of the ESA. The goal of any research should be to develop a technology or method, through robust experimental designs, that would achieve these goals while remaining economically and technically feasible for fishermen to implement.

2. NOAA-Fisheries should research development or modifications of 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.

4. NOAA-Fisheries should continue efforts to gather international support for the Inter-American Convention for the Protection and Conservation of Sea Turtles.

5. NOAA-Fisheries should support the development of a trans-Pacific international agreement that would include Pacific island and Pacific rim nations for the protection and conservation of sea turtle populations.

6. NOAA-Fisheries should continue and expand on existing efforts to implement measures and management actions that protect nesting sea turtles in their ocean environments and increase hatchling production at nesting beaches in the eastern and western Pacific. NOAA-Fisheries should continue to work with the Western Pacific Regional Fisheries Management Council and the relevant non-governmental organizations (such as World Wildlife Fund - Indonesia, Kamiali Integrated Conservation Development Group of Papua

New Guinea, the Sea Turtle Association of Japan, and Wildcoast in Baja, Mexico) to develop and implement long-term conservation programs for sea turtles in the Pacific that (1) protect the War-mon nesting beach at Jamursba-Medi, Bird's Head Peninsula in the State of Papua, Indonesia; (2) work with villagers in western Papua's Kei Kecil Islands to limit subsistence harvests of leatherback turtles to levels that would be sustainable by the population of leatherback turtles that uses those coastal foraging grounds; (3) work with villages of the Kamiali community in Papua New Guinea to eliminate nest predation of leatherback eggs, relocate leatherback nests from areas that are likely to be lost to beach erosion, and conduct subsistence harvests of leatherback turtle eggs that sustainable by this nesting aggregation of leatherback turtles; (4) conduct mortality reduction workshops with fishermen along the coast of Baja Mexico and place observers on local boats to reduce or eliminate the number of loggerhead turtles captured and killed in these fisheries; (5) conduct programs to relocate loggerhead sea turtle nests in Japan that are likely to be lost to beach erosion and provide shading to nests that experience extreme temperatures.

7. NOAA-Fisheries should provide technical and financial assistance necessary to export advances in knowledge of techniques and gear modifications that reduce interactions with sea turtles and/or dramatically reduce the immediate and/or delayed mortality rates of captured turtles with other nations engaged in similar fishing practices to reduce fishery impacts to sea turtle populations worldwide.

8. NOAA Fisheries should continue to work with the Forum Fisheries Agency to ensure that Forum Fisheries Agency observers who are deployed on U.S. purse seine vessels continue to collect and summarize data on the number of sea turtles captured, injured, and killed associated with interactions with the fishery (as an alternative, NOAA-Fisheries should continue to work with key member countries of the Forum Fisheries Agency to achieve the same outcomes discussed in this recommendation). NOAA Fisheries should request that Forum Fisheries Agency work with the Pacific Community's Oceanic Fisheries Program to ensure that relevant sea turtle information is provided in fishery summaries disseminated at annual consultations on the Treaty on Fisheries Between the Governments of Certain Pacific Island States and the Government of the United States of America (recognizing that these data are considered highly confidential by the 16 Pacific Island member countries and the release of these data must be authorized by these countries before any data can be released). NOAA-Fisheries should request that the collection of sea turtle information becomes a high priority for Forum Fisheries Agency observers and that sea-going observers have enough time to collect essential sea turtle life history and disposition information on any captured sea turtles and that VMS data be provided to NOAA-Fisheries for use in estimating the effects of U.S. vessels on sea turtles. This bycatch and fleet operation information is needed in an effort to obtain more representative data on sea turtle interactions. NOAA-Fisheries should ask the Forum Fisheries Agency to provide sea turtle data collected by observers aboard U.S. tuna purse seine vessels fishing under the Treaty on Fisheries Between the Governments of Certain Pacific Island States and the Government of the United States of America on a quarterly and annual basis. The report should include, at a minimum, the number of sea turtles incidentally captured,

injured, and killed by species, type of set in which each interaction occurred, and life history information on the turtles (size, gender, condition). Whenever possible, photographs should be taken.

10.0 REINITIATION NOTICE

This concludes formal consultation on the proposed regulatory amendments to the Fisheries Management Plan for pelagic fisheries of the Western Pacific Region. 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. If the amount or extent of incidental take identified in the incidental take statement that is enclosed in this biological opinion is exceeded, NOAA-Fisheries' Pacific Islands Region's Sustainable Fisheries Division should immediately request initiation of formal consultation.

This biological opinion has been predicated on several assumptions, which were necessary to overcome gaps in our knowledge. First, the exposure analyses in this biological opinion assumed that different nesting aggregations of green, leatherback, and loggerhead sea turtles were likely to be exposed to these fisheries proportional to their representation in genetics data collected in the area fished by the Hawaii-based longline fisheries. If new data reveals that these assumptions are incorrect, particularly if the eastern tropical Pacific population of green sea turtles, the Malaysian population of leatherback sea turtles, populations of leatherback sea turtles from New Caledonia or the Solomon Islands, or loggerhead sea turtle nesting aggregations on Maehama Beach on Yakushima Island in Japan are exposed in greater proportions than assumed in this opinion, then this new information is likely to satisfy the second requirement for reinitiating consultation.

Second, the response analyses of this biological opinion made assumptions about acute and chronic (post-hooking) mortality rates that were based on the information available from sea turtle experts. If new data, including data collected through the observer program, reveals that those assumptions substantially underestimated the number of sea turtles the would die from acute or chronic exposure to the fisheries, then this new information is likely to satisfy the second requirement for reinitiating consultation.

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APPENDIX A: TECHNICAL DISCUSSION OF ASSESSMENT APPROACH

As summarized in the *Approach to the Assessment* section of this biological opinion, 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.

This appendix provides more detailed descriptions of the specific approaches the Office of Protected Resources used to conduct its assessment of the effects of the proposed Fisheries Management Plan for the pelagic fisheries of the Western Pacific Region.

A.1 Method

We conducted our assessment of the effects of the proposed fisheries and fishery management regime using four discrete steps: deconstruct the action, exposure analyses, response analyses, and risk analyses.

Deconstruct the Action

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, and) to allow us to distinguish the effects of different fisheries and different fishing strategies on listed resources.

Exposure Analyses

The second step of our assessment consisted of exposure analyses which identified 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. Our exposure analyses used the following scenarios for the different sea turtles:

Green sea turtles: Green sea turtles in the Action Area seem to represent a mix of the threatened Hawaiian turtles and endangered eastern Tropical Pacific turtles, although a majority of the green turtles captured by the Hawaii-based fisheries seem to be ETP. For these risk analyses, exposure to the proposed fisheries and the response of nesting aggregations to that exposure will be evaluated using the following scenarios:

a. green turtles occurring in the Action Area are dominated by the endangered species with individual turtles from Hawaiian nesting aggregations (50% from eastern tropical Pacific; 35% random combination of ETP and Hawaiian turtles; 12% Hawaiian turtles);

b. green turtles in the Action Area are a random mix of these nesting aggregations

Hawksbill sea turtles: hawksbill sea turtles in the Action Area all seem to originate from local nesting aggregations. Exposure to the proposed fisheries and the response of nesting aggregations to that exposure assumed that hawksbill turtles in the Action Area are a random mix of the Pacific nesting aggregations

Leatherback sea turtles: the affiliation between leatherback turtles exposed to the longline fisheries and the different nesting aggregations of leatherback turtles in the Pacific remains uncertain, although the evidence available suggests that at least turtles from the eastern Pacific, Indonesia, Malaysia, and Papua New Guinea are exposed to the Hawaii-based longline.fisheries. We do not know the proportional representation of turtles from these and other aggregations in the Action Area. We also remain uncertain about the size of these nesting aggregations, although the nesting aggregations in the eastern tropical Pacific and Malaysia have been represented by less than handfuls of nesting females for the past several years. Our exposure analyses considered the following scenarios:

a. leatherback turtles occur in the Action Area proportional to the limited genetics information available

b. leatherback turtles occurring in the Action Area are dominated by the Papua nesting aggregation with individual turtles from other nesting aggregations occurring periodically

Loggerhead sea turtles loggerhead sea turtles in the Action Area all seem to originate from the Japanese nesting aggregations. Our exposure analyses considered the following scenarios:

a. loggerhead turtles in the Action Area are a single nesting population

b. loggerhead turtles in the Action Area represent two groups: a random mix of all nesting aggregations except for the Yakushima Island nesting aggregations (which have a rare haplotype and represent about 5 percent of the sample taken in Hawaiian waters) and a random mix of turtles from the Yakushima Island nesting aggregations

Olive ridley sea turtles: olive sea turtles in the Action Area seem to represent a mix of turtles from the eastern tropical Pacific, western tropical Pacific, and Indian Oceans, although a majority of the olive ridley turtles captured by the Hawaii-based fisheries seem to be ETP. Our exposure analyses considered the following scenarios:

a. olive turtles occurring in the Action Area are dominated by the endangered species with individual turtles from ETP nesting aggregations (74% from eastern tropical Pacific; 26% from the western Pacific and Indian Oceans);

b. olive ridley turtles in the Action Area are a random mix of these nesting aggregations

Additional scenarios had appeared in the analysis plans we had presented earlier in the consultation we had developed; however, we merged several of those scenarios because they were mathematically identical to other scenarios or because we did not have sufficient data to support them.

Response Analyses
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.

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 A.1 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[1] (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.

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[1] Subordinate part such as tongue, extraembryonic membranes

Risk Analyses

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. This step of our assessment asked (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.

To assess the probability of regional extinction (for example, the probability of leatherback turtles becoming extinct in the Pacific Ocean), we considered regional 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.

A.2 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).

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.

To apply the procedures described by Dennis et al. (1991) and Morris and Doak (2002), we first had to estimate two population metrics: (a) population's mean logarithmic growth rate and (b) variance in a population's mean logarithmic growth rate

Table X. 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[2]) 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[3]
Entangled Only	Released Entangled 50 (60)		Fully Disentangled 1 (2)
Comatose/resuscitated	n/a[4]	70 (80)	60 (70)

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[2] Subordinate part such as tongue, extraembryonic membranes

[3] Per veterinary recommendation hooks would not be removed if the insertion point of the hook is not visible when viewed through the open mouth.

[4] 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.

Population's Mean Logarithmic Growth Rate

A population's mean logarithmic growth rate (or μ), which is equal to the natural logarithm of the population's geometric mean growth rate, measures the mean of the change in a population's growth rate and is a measure of a population's stochastic growth over time (Dennis et al. 1991, Lande and Orzack 1988, Morris and Doak 2002). Forecasts of a population's stochastic growth over time would produce some trajectories that increase, some that remain somewhat stable, while others decrease. The mean logarithmic growth rate is a measure of a 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).

To estimate population mean logarithmic growth rates, we used counts of sea turtle nests in a particular year (or time interval) or counts of nesting adult females in a particular year (or time interval) to estimate the number of adult females in a nesting aggregation or colony at a particular time (see *Conversions* section below). We then converted these estimates of the number of adult females in a nesting aggregation into their logarithmic equivalents, the converting the result using the equation

$$(\log N_{t+1} - \log N_t)/(t+1 - t)^{0.5} \qquad \text{equation A1.}$$

where $(t+1 - t)^{0.5}$ is a transformation of the length of the time intervals between censuses or counts. ~~Using these transformations as *x-variables* and the results of equation A.1 as *y*~~ *variables*, we then performed a linear regression of the *y*s against the *x*s while forcing the regression intercept to be zero.

The slope of the regression is an estimate of μ and the regression's error mean square is an estimate of σ^2 (see further discussion of this parameter in the next section). We used the 95% confidence limits for the regression slope to estimate 95% confidence intervals around our estimates of the population's mean log growth rate.

Variance in a Population's Mean Logarithmic Growth Rate

The variance in a population's mean log growth rate (σ^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.

As discussed in the previous section, we used the slope of the regression to estimate μ and the regression's error mean square to estimate σ^2. To calculate 95% confidence limits for our estimates of variance of population mean log growth rates, we obtained the 2.5th and 97.5th percentiles of the chi-square distribution with q-1 degrees of freedom (where q

is the number of transitions in a census) and substituting the result into the following equation

$$((q-1)\hat{\sigma}^2 / \chi^2_{0.025,q-1}, (q-1)\hat{\sigma}^2 / \chi^2_{0.097.5,q-1}) \qquad \text{Equation A.2}$$

Using these estimates of μ and σ^2, we first constructed a probability density function which is given by the inverse Gaussian distribution for hitting a quasi-extinction threshold in a small time interval

$$g(t \big| u, \sigma^2, d) = \frac{d}{\sqrt{2\pi\sigma^2 t^3}} \exp\left[\frac{-(d+\mu t)^2}{2\sigma^2 t}\right] \qquad \text{Equation A.3}$$

then a cumulative distribution function of extinction time using the equation:

$$G(T \mid d, \mu, \sigma^2) = \phi\left(\frac{-d-\mu T}{\sqrt{\sigma^2 T}}\right) + \exp(-2\mu d / \sigma^2)\phi\left(\frac{-d+\mu T}{\sqrt{\sigma^2 T}}\right) \qquad \text{Equation A.4}$$

Where $G(T|d, \mu, \sigma^2)$ = the cumulative probability of reaching a quasi-extinction threshold at time T, given d, μ, and σ^2

φ = 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

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

The cumulative distribution function integrates the inverse Gaussian distribution from t = 0 to t = T to calculate the probability of reaching a quasi-extinction threshold at any time between the present (t = 0) and a future time (t = T). 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).

A.2.2. Conversions and Scenarios

Leatherback Sea Turtles

We conducted our analyses by started with published counts of the number of leatherback turtle nesters or nests from Las Baulas, Costa Rica; Jamursba-Medi, Indonesia; and Terengganu, Malaysia. We converted these counts to estimates of the number of adult females in the two nesting aggregations by multiplying these estimates by 2.5 to estimate the number of nesting females in the aggregation (the same value used by Spotila et al. 1996) and is an average of re-migration intervals reported for several nesting beaches

Table A.1 Additional parameters, definitions of the parameters, and equations used in Dennis model calculations

Parameter	Definition and Equation
q	The number transitions in a data set (count of transitions)
t_q	Length of time population has been observed (in years)
$\pi(x_d, \mu, \sigma^2)$	The probability of reaching a threshold population size
	$= \begin{cases} 1, \mu \le 0 \\ \exp\left(-2\mu x_d / \sigma^2\right) \mu > 0 \end{cases}$
E[T]	mean time until threshold (x_e) is reached
	$\equiv \theta(x_d, \mu) = x_d/\lvert\mu\rvert$
Var(T)	variance in mean time to extinction
	$= x_d/\lvert\mu\rvert^3$
$t^*(x_d, \mu, \sigma^2)$	Maximum likelihood estimate of most likely (mode) time to extinction
	$= \frac{x_d}{\lvert\mu\rvert}\left[\left(1+\frac{9}{4v^2}\right)^{0.5} - \frac{3}{2v}\right]$, where $v = x_d\lvert\mu\rvert/\sigma^2$
r	Continuous rate of population increase (a revision of "intrinsic" rate of increase)
	$= \mu + (\sigma^2/2)$
Var (r) ~	$= (\sigma^2/t_q)+\{\sigma^4/[2(q-1)]\}$
θ	$\theta(x_d, \mu\text{-cap}) = x_d/\lvert\mu\text{-cap}\rvert$
Var(θ)	$x_d^2\sigma^2/(\mu^4 t_q)$
	Lambda or the finite rate of population increase
	= exp (r)

We then applied the procedures prescribed by Dennis et al. (1991) and Morris and Doak (2002) to these estimates of the adult population size to estimate several demographic variables for each of these nesting aggregations of leatherback sea turtles:

a. the population's mean log growth rate,
b. the variance in the population's mean log growth rate,
c. continuous rate of increase (r), and
d. finite rate of population increase (lambda).

Using these parameters, we then calculated the nesting aggregation's risk of quasi-extinction (probability of threshold), mean time to threshold, median time to threshold, and modal time to threshold. In all instances, we used quasi-extinction thresholds 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.

To estimate the risks the mortalities associated with the proposed fisheries might pose to leatherback sea turtles, we asked the question: how would these additional deaths of large adult or sub-adult females affect the demographic variables and extinction risks of a nesting aggregation of leatherback sea turtles? In asking that question, we were interested in incremental changes in demographic variables or extinction risks, rather than an absolute increase in risk.

To answer that question, we used the converted census estimates for the three nesting aggregations, treating the time series of these counts as a "simulated" population of leatherback sea turtles with appropriate levels of year-to-year variation. We then subtracted the number of adult females identified in our exposure analyses from this "population" (we considered using a stochastic mortality rate, but decided that a constant rate would bracket the upper boundary of any stochastic estimate) and re-calculated the demographic variables for the "population" that had been affected by these mortalities.

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 "populations" that had not experienced these mortalities.

Loggerhead Sea Turtles. Japanese investigators had provided us with counts of the number of loggerhead turtle nests on Inakahama and Maehama Beaches on Yakushima Island, Japan (data we had received from Japanese investigators). We converted these counts to estimates of the number of adult females in the two nesting aggregations using two methods: Method 1 assumed 4.1 nests per nesting female per year based on estimates from Heppell et al. (2002). Method 2 assumed 2.06 nests per nesting female per year based on estimated from Schroeder et al. (2003). Both methods assumed remigrations intervals of 2.59 years from Miller et al. (2002)

Method 1: (number of nests/4.1)*2.59
Method 2: (no. nests/2.06)*2.59

Once we had completed these conversion, we applied the procedures described in the preceding sections and by Dennis et al. (1991) and Morris and Doak (2002) to these estimates

To estimate the risks the mortalities associated with the proposed fisheries might pose to loggerhead sea turtles, we asked the question: how would these additional deaths (of large oceanic or small neritic juveniles) affect the demographic variables and extinction risks of a population of loggerhead sea turtles with demographic rates similar to those of Inakahama and Maehama nesting beaches? In asking that question, we were interested in incremental changes in demographic variables or extinction risks, rather than an absolute increase in risk.

To answer this question, we used the 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" (we considered using a stochastic mortality rate, but decided that a constant rate would bracket the upper boundary of any stochastic estimate) and re-calculated the demographic variables for the "population" that had been affected by these mortalities.

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.

To assess the risks these mortalities might pose to the remaining Japanese nesting aggregations, we assumed that these aggregations were equivalent in size to Yakushima and that, by assigning the mortalities to the total number of adult females, we were approximating the consequences of randomly distributing the mortalities among the various nesting aggregations (since we did not have counts for the other nesting aggregations, we used this approach to approximate the risks).

We identified life stages using the new life history pattern presented by Bolten[1]: oceanic immatures (5 - 45 cm CCL); small neritic juveniles (45 - 72 cm), large neritic juveniles (72-92 cm), and adults (>92 cm). These growth intervals had been developed using data on sea turtles in the Atlantic rather than the Pacific, in the absence of stage intervals for Pacific loggerheads, we used the Atlantic life stages without conversion. Since most of the loggerhead turtles that have been captured in interactions with the longline fisheries in the Pacific ranged from 41 to 83 cm (straight carapace lengths) with a mean of 58 cm. This would place them in the upper size range of oceanic or the mid-range of small neritic juveniles.

As a result, we assumed that most of the loggerhead turtles that would interact with the fisheries would be small, neritic juveniles. Using the survival values between this stage and the adult stage (from Heppell et al. 2003; once again for Atlantic loggerheads) we converted estimates of the number of small neritic juveniles sea turtles to adult sea turtles: using two steps: multiplying by 0.6758 (the survival rate from small neritic to large neritic) and then by 0.8 and 0.7425 (the survival rate from large neritic to adult).

Matrix Models

We used published stage-based population structures for loggerhead sea turtles and stage-based survival rates, transition rates, and fecundity rates to build population matrices for loggerhead sea turtles (Crouse et al. 1987, Heppell et al. 2003). Using these matrices and operating with the Poptools software plug-in for Microsoft Excel©, we calculated dominant eigenvalues to estimate finite rates of population increase, stable stage structure, stage-elasticities and stage-sensitivities using to procedures described in Caswell (2002) and Morris and Doak (2002)

$$\mathbf{Mn}_0 = n_1$$
$$\mathbf{Mn}_1 = n_2 \qquad \text{or generalized as } \mathbf{Mn}_{t-1} = n_t$$
$$\mathbf{Mn}_2 = n_3$$

Where

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[1] A.B. Bolten. 2003. Active swimmer-passive drifters: the oceanic juvenile stages of loggerheads in the Atlantic system, Pages 63-78, in A.B. Bolten and B.E. Witherington (eds.). Loggerhead sea turtles. Smithsonian Institution Press; Washington, D.C.

$$\begin{pmatrix} F_0 & F_1 & F_2 & F_3 \\ P_0 & 0 & 0 & 0 \\ 0 & P_1 & 0 & 0 \\ 0 & 0 & P_2 & 0 \end{pmatrix} = M \qquad \begin{pmatrix} n_{00} \\ n_{10} \\ n_{20} \\ n_{30} \end{pmatrix} = n_o \qquad \begin{pmatrix} n_{01} \\ n_{11} \\ n_{21} \\ n_{31} \end{pmatrix} = n_1$$

To assess the relative effect of loggerhead mortalities associated with the proposed fisheries on the trajectory of loggerhead sea turtles, we created a "population"-vector with stage abundances consistent with a stable-stage structure, increased the mortality rates of the oceanic juveniles and, separately, small neritic juveniles consistent with the mortalities associated with the proposed fisheries, and projected this "population" matrix for 25, 50, and 100-years. We ran both deterministic (fixed rates) and stochastic (variable rates) projections of these matrices. We compared the results of these projections against an identical "population" that had not been subjected to these stage-specific mortalities.

APPENDIX A: TECHNICAL DISCUSSION OF ASSESSMENT APPROACH

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Green sea turtles: Green sea turtles in the Action Area seem to represent a mix of the threatened Hawaiian turtles and endangered eastern Tropical Pacific turtles, although a majority of the green turtles captured by the Hawaii-based fisheries seem to be ETP. For these risk analyses, exposure to the proposed fisheries and the response of nesting aggregations to that exposure will be evaluated using the following scenarios:

a. green turtles occurring in the Action Area are dominated by the endangered species with individual turtles from Hawaiian nesting aggregations (50% from eastern tropical Pacific; 35% random combination of ETP and Hawaiian turtles; 12% Hawaiian turtles);

b. green turtles in the Action Area are a random mix of these nesting aggregations

Hawksbill sea turtles: hawksbill sea turtles in the Action Area all seem to originate from local nesting aggregations. Exposure to the proposed fisheries and the response of nesting aggregations to that exposure assumed that hawksbill turtles in the Action Area are a random mix of the Pacific nesting aggregations

Leatherback sea turtles: the affiliation between leatherback turtles exposed to the longline fisheries and the different nesting aggregations of leatherback turtles in the Pacific remains uncertain, although the evidence available suggests that at least turtles from the eastern Pacific, Indonesia, Malaysia, and Papua New Guinea are exposed to the Hawaii-based longline.fisheries. We do not know the proportional representation of turtles from these and other aggregations in the Action Area. We also remain uncertain about the size of these nesting aggregations, although the nesting aggregations in the eastern tropical Pacific and Malaysia have been represented by less than handfuls of nesting females for the past several years. Our exposure analyses considered the following scenarios:

a. leatherback turtles occur in the Action Area proportional to the limited genetics information available

b. leatherback turtles occurring in the Action Area are dominated by the Papua nesting aggregation with individual turtles from other nesting aggregations occurring periodically

Loggerhead sea turtles loggerhead sea turtles in the Action Area all seem to originate from the Japanese nesting aggregations. Our exposure analyses considered the following scenarios:

a. loggerhead turtles in the Action Area are a single nesting population

b. loggerhead turtles in the Action Area represent two groups: a random mix of all nesting aggregations except for the Yakushima Island nesting aggregations (which have a rare haplotype and represent about 5 percent of the sample taken in Hawaiian waters) and a random mix of turtles from the Yakushima Island nesting aggregations

Olive ridley sea turtles: olive sea turtles in the Action Area seem to represent a mix of turtles from the eastern tropical Pacific, western tropical Pacific, and Indian Oceans, although a majority of the olive ridley turtles captured by the Hawaii-based fisheries seem to be ETP. Our exposure analyses considered the following scenarios:

a. olive turtles occurring in the Action Area are dominated by the endangered species with individual turtles from ETP nesting aggregations (74% from eastern tropical Pacific; 26% from the western Pacific and Indian Oceans);

b. olive ridley turtles in the Action Area are a random mix of these nesting aggregations

Additional scenarios had appeared in the analysis plans we had presented earlier in the consultation we had developed; however, we merged several of those scenarios because they were mathematically identical to other scenarios or because we did not have sufficient data to support them.

Response Analyses
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.

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 A.1 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[1] (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.

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[1] Subordinate part such as tongue, extraembryonic membranes

Risk Analyses

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. This step of our assessment asked (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.

To assess the probability of regional extinction (for example, the probability of leatherback turtles becoming extinct in the Pacific Ocean), we considered regional 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.

A.2 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).

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.

To apply the procedures described by Dennis et al. (1991) and Morris and Doak (2002), we first had to estimate two population metrics: (a) population's mean logarithmic growth rate and (b) variance in a population's mean logarithmic growth rate

Table X. 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[2]) 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[3]
Entangled Only	Released Entangled 50 (60)		Fully Disentangled 1 (2)
Comatose/resuscitated	n/a[4]	70 (80)	60 (70)

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[2] Subordinate part such as tongue, extraembryonic membranes

[3] Per veterinary recommendation hooks would not be removed if the insertion point of the hook is not visible when viewed through the open mouth.

[4] 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.

Population's Mean Logarithmic Growth Rate

A population's mean logarithmic growth rate (or μ), which is equal to the natural logarithm of the population's geometric mean growth rate, measures the mean of the change in a population's growth rate and is a measure of a population's stochastic growth over time (Dennis et al. 1991, Lande and Orzack 1988, Morris and Doak 2002). Forecasts of a population's stochastic growth over time would produce some trajectories that increase, some that remain somewhat stable, while others decrease. The mean logarithmic growth rate is a measure of a 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).

To estimate population mean logarithmic growth rates, we used counts of sea turtle nests in a particular year (or time interval) or counts of nesting adult females in a particular year (or time interval) to estimate the number of adult females in a nesting aggregation or colony at a particular time (see *Conversions* section below). We then converted these estimates of the number of adult females in a nesting aggregation into their logarithmic equivalents, the converting the result using the equation

$$(\log N_{t+1} - \log N_t)/(t+1-t)^{0.5} \qquad \text{equation A1.}$$

where $(t+1-t)^{0.5}$ is a transformation of the length of the time intervals between censuses or counts. Using these transformations as *x-variables* and the results of equation A.1 as *y variables*, we then performed a linear regression of the *y*s against the *x*s while forcing the regression intercept to be zero.

The slope of the regression is an estimate of μ and the regression's error mean square is an estimate of σ^2 (see further discussion of this parameter in the next section). We used the 95% confidence limits for the regression slope to estimate 95% confidence intervals around our estimates of the population's mean log growth rate.

Variance in a Population's Mean Logarithmic Growth Rate

The variance in a population's mean log growth rate (σ^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.

As discussed in the previous section, we used the slope of the regression to estimate μ and the regression's error mean square to estimate σ^2. To calculate 95% confidence limits for our estimates of variance of population mean log growth rates, we obtained the 2.5th and 97.5th percentiles of the chi-square distribution with q-1 degrees of freedom (where q

is the number of transitions in a census) and substituting the result into the following equation

$$((q-1)\hat{\sigma}^2 / \chi^2_{0.025,q-1}, (q-1)\hat{\sigma}^2 / \chi^2_{0.097.5,q-1}) \qquad \text{Equation A.2}$$

Using these estimates of μ and σ^2, we first constructed a probability density function which is given by the inverse Gaussian distribution for hitting a quasi-extinction threshold in a small time interval

$$g(t \mid u, \sigma^2, d) = \frac{d}{\sqrt{2\pi\sigma^2 t^3}} \exp\left[\frac{-(d+\mu t)^2}{2\sigma^2 t}\right] \qquad \text{Equation A.3}$$

then a cumulative distribution function of extinction time using the equation:

$$G(T \mid d, \mu, \sigma^2) = \phi\left(\frac{-d-\mu T}{\sqrt{\sigma^2 T}}\right) + \exp(-2\mu d / \sigma^2)\phi\left(\frac{-d+\mu T}{\sqrt{\sigma^2 T}}\right) \qquad \text{Equation A.4}$$

Where $G(T|d, \mu, \sigma^2)$ = the cumulative probability of reaching a quasi-extinction threshold at time T, given d, μ, and σ^2

φ = 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

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

The cumulative distribution function integrates the inverse Gaussian distribution from t = 0 to t = T to calculate the probability of reaching a quasi-extinction threshold at any time between the present (t = 0) and a future time (t = T). 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).

A.2.2. Conversions and Scenarios

Leatherback Sea Turtles

We conducted our analyses by started with published counts of the number of leatherback turtle nesters or nests from Las Baulas, Costa Rica; Jamursba-Medi, Indonesia; and Terengganu, Malaysia. We converted these counts to estimates of the number of adult females in the two nesting aggregations by multiplying these estimates by 2.5 to estimate the number of nesting females in the aggregation (the same value used by Spotila et al. 1996) and is an average of re-migration intervals reported for several nesting beaches

Table A.1 Additional parameters, definitions of the parameters, and equations used in Dennis model calculations

Parameter	Definition and Equation
q	The number transitions in a data set (count of transitions)
t_q	Length of time population has been observed (in years)
$\pi(x_d, \mu, \sigma^2)$	The probability of reaching a threshold population size
	$= \begin{cases} 1, \mu \leq 0 \\ \exp\left(-2\mu x_d / \sigma^2\right) \mu > 0 \end{cases}$
$E[T]$	mean time until threshold (x_e) is reached
	$\equiv \theta(x_d, \mu) = x_d/\lvert\mu\rvert$
Var(T)	variance in mean time to extinction
	$= x_d/\lvert\mu\rvert^3$
$t^*(x_d, \mu, \sigma^2)$	Maximum likelihood estimate of most likely (mode) time to extinction
	$= \frac{x_d}{\lvert\mu\rvert}\left[\left(1+\frac{9}{4v^2}\right)^{0.5} - \frac{3}{2v}\right]$, where $v = x_d\lvert\mu\rvert/\sigma^2$
r	Continuous rate of population increase (a revision of "intrinsic" rate of increase)
	$= \mu + (\sigma^2/2)$
Var (r) ~	$= (\sigma^2/t_q)+\{\sigma^4/[2(q-1)]\}$
θ	$\theta(x_d, \mu\text{-cap}) = x_d/\lvert\mu\text{-cap}\rvert$
Var(θ)	$x_d^2\sigma^2/(\mu^4 t_q)$
	Lambda or the finite rate of population increase
	= exp (r)

We then applied the procedures prescribed by Dennis et al. (1991) and Morris and Doak (2002) to these estimates of the adult population size to estimate several demographic variables for each of these nesting aggregations of leatherback sea turtles:

a. the population's mean log growth rate,
b. the variance in the population's mean log growth rate,
c. continuous rate of increase (r), and
d. finite rate of population increase (lambda).

Using these parameters, we then calculated the nesting aggregation's risk of quasi-extinction (probability of threshold), mean time to threshold, median time to threshold, and modal time to threshold. In all instances, we used quasi-extinction thresholds 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.

To estimate the risks the mortalities associated with the proposed fisheries might pose to leatherback sea turtles, we asked the question: how would these additional deaths of large adult or sub-adult females affect the demographic variables and extinction risks of a nesting aggregation of leatherback sea turtles? In asking that question, we were interested in incremental changes in demographic variables or extinction risks, rather than an absolute increase in risk.

To answer that question, we used the converted census estimates for the three nesting aggregations, treating the time series of these counts as a "simulated" population of leatherback sea turtles with appropriate levels of year-to-year variation. We then subtracted the number of adult females identified in our exposure analyses from this "population" (we considered using a stochastic mortality rate, but decided that a constant rate would bracket the upper boundary of any stochastic estimate) and re-calculated the demographic variables for the "population" that had been affected by these mortalities.

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 "populations" that had not experienced these mortalities.

Loggerhead Sea Turtles. Japanese investigators had provided us with counts of the number of loggerhead turtle nests on Inakahama and Maehama Beaches on Yakushima Island, Japan (data we had received from Japanese investigators). We converted these counts to estimates of the number of adult females in the two nesting aggregations using two methods: Method 1 assumed 4.1 nests per nesting female per year based on estimates from Heppell et al. (2002). Method 2 assumed 2.06 nests per nesting female per year based on estimated from Schroeder et al. (2003). Both methods assumed remigrations intervals of 2.59 years from Miller et al. (2002)

Method 1: (number of nests/4.1)*2.59
Method 2: (no. nests/2.06)*2.59

Once we had completed these conversion, we applied the procedures described in the preceding sections and by Dennis et al. (1991) and Morris and Doak (2002) to these estimates

To estimate the risks the mortalities associated with the proposed fisheries might pose to loggerhead sea turtles, we asked the question: how would these additional deaths (of large oceanic or small neritic juveniles) affect the demographic variables and extinction risks of a population of loggerhead sea turtles with demographic rates similar to those of Inakahama and Maehama nesting beaches? In asking that question, we were interested in incremental changes in demographic variables or extinction risks, rather than an absolute increase in risk.

To answer this question, we used the 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" (we considered using a stochastic mortality rate, but decided that a constant rate would bracket the upper boundary of any stochastic estimate) and re-calculated the demographic variables for the "population" that had been affected by these mortalities.

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.

To assess the risks these mortalities might pose to the remaining Japanese nesting aggregations, we assumed that these aggregations were equivalent in size to Yakushima and that, by assigning the mortalities to the total number of adult females, we were approximating the consequences of randomly distributing the mortalities among the various nesting aggregations (since we did not have counts for the other nesting aggregations, we used this approach to approximate the risks).

We identified life stages using the new life history pattern presented by Bolten[1]: oceanic immatures (5 - 45 cm CCL); small neritic juveniles (45 - 72 cm), large neritic juveniles (72-92 cm), and adults (>92 cm). These growth intervals had been developed using data on sea turtles in the Atlantic rather than the Pacific, in the absence of stage intervals for Pacific loggerheads, we used the Atlantic life stages without conversion. Since most of the loggerhead turtles that have been captured in interactions with the longline fisheries in the Pacific ranged from 41 to 83 cm (straight carapace lengths) with a mean of 58 cm. This would place them in the upper size range of oceanic or the mid-range of small neritic juveniles.

As a result, we assumed that most of the loggerhead turtles that would interact with the fisheries would be small, neritic juveniles. Using the survival values between this stage and the adult stage (from Heppell et al. 2003; once again for Atlantic loggerheads) we converted estimates of the number of small neritic juveniles sea turtles to adult sea turtles: using two steps: multiplying by 0.6758 (the survival rate from small neritic to large neritic) and then by 0.8 and 0.7425 (the survival rate from large neritic to adult).

Matrix Models

We used published stage-based population structures for loggerhead sea turtles and stage-based survival rates, transition rates, and fecundity rates to build population matrices for loggerhead sea turtles (Crouse et al. 1987, Heppell et al. 2003). Using these matrices and operating with the Poptools software plug-in for Microsoft Excel©, we calculated dominant eigenvalues to estimate finite rates of population increase, stable stage structure, stage-elasticities and stage-sensitivities using to procedures described in Caswell (2002) and Morris and Doak (2002)

$$\mathbf{Mn}_0 = n_1$$
$$\mathbf{Mn}_1 = n_2 \qquad \text{or generalized as } \mathbf{Mn}_{t-1} = n_t$$
$$\mathbf{Mn}_2 = n_3$$

Where

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[1] A.B. Bolten. 2003. Active swimmer-passive drifters: the oceanic juvenile stages of loggerheads in the Atlantic system, Pages 63-78, in A.B. Bolten and B.E. Witherington (eds.). Loggerhead sea turtles. Smithsonian Institution Press; Washington, D.C.

$$\begin{pmatrix} F_0 & F_1 & F_2 & F_3 \\ P_0 & 0 & 0 & 0 \\ 0 & P_1 & 0 & 0 \\ 0 & 0 & P_2 & 0 \end{pmatrix} = M \qquad \begin{pmatrix} n_{00} \\ n_{10} \\ n_{20} \\ n_{30} \end{pmatrix} = n_o \qquad \begin{pmatrix} n_{01} \\ n_{11} \\ n_{21} \\ n_{31} \end{pmatrix} = n_1$$

To assess the relative effect of loggerhead mortalities associated with the proposed fisheries on the trajectory of loggerhead sea turtles, we created a "population"-vector with stage abundances consistent with a stable-stage structure, increased the mortality rates of the oceanic juveniles and, separately, small neritic juveniles consistent with the mortalities associated with the proposed fisheries, and projected this "population" matrix for 25, 50, and 100-years. We ran both deterministic (fixed rates) and stochastic (variable rates) projections of these matrices. We compared the results of these projections against an identical "population" that had not been subjected to these stage-specific mortalities.