BIOLOGICAL OPINION

AGENCY: Marine Mammal Division, Office of Protected Species, National Marine Fisheries Service

ACTIVITY: Section 7 Consultation on Authorization to take Listed Marine Mammals Incidental to Commercial Fishing Operations under Section 101(a)(5)(E) of the Marine Mammal Protection Act for the California/Oregon Drift Gillnet Fishery

CONDUCTED BY: Endangered Species Division, Office of Protected Resources, NMFS

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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 (NMFS) or the U.S. Fish and Wildlife Service, depending upon the protected species that may bc affected. For the actions described in this document, the action agency is the Marine Mammal Division, Office of Protected Resources of NMFS. The consulting agency is the Endangered Species Division, Office of Protected Resources, also of NMFS. Section 7(b) of the Act requires that the consultation be summarized in a biological opinion detailing how the action may affect protected species.

This document represents the NMFS' biological opinion (Opinion) based on our review of the California/Oregon (CA/OR) drift gillnet fishery and authorization to take listed marine mammals incidental to commercial fishing operations under section 101 (a)(5)(E) of the Marine Mammal Protection Act (MMPA) and the effects of this action on humpback whales (*Megaptera novaeangliae*), fin whales (*Balaenoptera physalus*), sperm whales (*Physeter macrocephalus*), Steller sea lions (*Eumetopias jubatus*), green turtles (*Chelonia mydas*), lcatherback turtles (*Dermochelys coriacea*) loggerhead turtles (*Caretta caretta*), and olive ridley turtles (*Lepidochelys olivacea*), in accordance with section 7 of the ESA.

This Opinion is based on information provided in the draft and final recovery plans for the fin whale, the humpback whale, and the Steller sea lion, the most current marine mammal stock humpback whale, and the Steller sea lion, the most current marine mammal stock assessment reports, sea turtle recovery plans, past and current research, and biological opinions for this and other relevant fisheries. A complete administrative record of this consultation is on file at the NMFS, Southwest Regional Office, Long Beach, California.

I. CONSULTATION HISTORY

NMFS issued a biological opinion on September 30, 1997, to evaluate the effect of the final regulations to implement the Pacific Offshore Cetacean Take Reduction Plan (PCTRP) for the CA/OR drift gillnet fishery on listed sea turtle and marine mammal populations (NMFS, 1997a). NMFS concluded in this opinion that establishing a minimum extender length requirement of 6 fathoms (36 feet), conducting skippers workshops, and using pingers on the nets would most likely reduce the incidental catch of listed marine mammals and sea turtles. Based on analyses of the final regulations, NMFS concluded that the continued operation of the CA/OR drift gillnet fishery under the PCTRP was not likely to jeopardize the continued existence of the humpback whale, sperm whale, leatherback turtle, or loggerhead turtle.

Using observer data collected from 1991-95, and anticipating that average take levels would decrease with the use of extenders by drift gillnet vessels, NMFS authorized the annual incidental entanglement and mortality level of 18 and 3 loggerhead turtles, respectively, and the annual incidental entanglement and mortality level of 30 and 19 leatherback turtles, respectively. Because NMFS was unable to determine that the mortality and serious injury of sperm whales or humpback whales incidental to the CA/OR drift gillnet fishery was having a negligible impact on the species or stock (see section 101(a)(5)(E) of the MMPA), no takes of sperm whales or humpback whales were authorized (NMFS, 1997a).

On December 22, 1999, the Marine Mammal Division, Protected Resources Program, Southwest Regional Office, NMFS, requested reinitiation of formal section 7 consultation on the continued implementation of the PCTRP regulations for the CA/OR drift gillnet fishery with the Endangered Species Division, Protected Resources Program, NMFS. Consultation was initiated on February 16, 2000. Reinitiation of consultation was required because the incidental take of loggerhead sea turtles anticipated in 1997 had been exceeded, and 1999 observer data indicated that three listed species, previously thought unaffected by the fishery, have interacted with the fishery.

Since the implementation of the PCTRP, one loggerhead mortality was observed, in 1998, resulting in an estimated mortality of six loggerheads (1 loggerhead observed killed/17.5% observed sets translates to about 5.7 loggerheads estimated killed). Because this estimated mortality exceeded the anticipated annual mortality of three loggerheads, reinitiation of section 7 consultation is required. In addition, since the effective date of the PCTRP, the CA/OR drift gillnet fishery has interacted with three new listed species previously considered not affected by the fishery. One fin whale and one green turtle were observed killed in 1999, and one olive ridley turtle was observed entangled (released alive) during that same year. Therefore, section 7 consultation must also be reinitiated so that the possible causes of new species interactions can be analyzed and measures to further reduce or avoid these takes in the future can be developed.

Based on a review of incidental take rates of listed species before and after implementation of the PCTRP and the likely effects of the continuation of PCTRP regulations on listed species, NMFS determined that the continued implementation of PCTRP regulations for the CA/OR drift gillnet fishery was likely to affect, but was not likely to adversely affect listed species under its jurisdiction. Any increases in incidental take rates or takes of previously unaffected species do not appear to be attributable to PCTRP regulations. Based on these data, which were unavailable during the 1997 consultation, NMFS determined that an informal consultation was appropriate for the PCTRP and, accordingly, changed ins policy regarding consultations on this TRP. The informal consultation is on file at the Southwest Regional Office, NMFS.

At the same time, NMFS proposes to issue a permit, pursuant to section 101(a)(5)(E) of the Marine Mammal Protection Act (MMPA), for the incidental taking of four stocks of threatened or endangered marine mammals by the CA/OR drift gillnet fishery (65 FR 35904, June 6, 2000). This permit constitutes a Federal action for the purposes of section 7(a)(2) of the ESA.

II. DESCRIPTION OF THE PROPOSED ACTION

The National Marine Fisheries Service's Marine Mammal Division proposes to authorize the take of marine mammals incidental for a three-year period incidental to the CA/OR drift gillnet fishery to incidentally take marine mammals under section 101(a)(5)(E) of the MMPA. Also, a federal fishery management plan (FMP) is currently being developed for those fisheries originating from the West Coast of the U.S. that fish for highly migratory species. The CA/OR drift gillnet fishery will be part of this plan, which should be in effect, along with implementing regulations, in approximately two years. At that time, consultation will be required for the FMP and its implementing regulations. For the reasons listed above, the duration of the proposed action and this Opinion is limited to three years.

Under Section 101(a)(5)(E) of the Marine Mammal Protection Act (MMPA), during any period of up to three consecutive years, NMFS shall allow the incidental, but not intentional, taking of listed marine mammals by a fishery if: 1) the incidental mortality and serious injury from commercial fisheries will have a negligible impact on such species or stock; 2) a recovery plan has been developed or is being developed for such species or stock pursuant to the Endangered Species Act; 3) a monitoring plan is established; 4) vessels are registered in accordance with Section 118 of the MMPA; and 5) a take reduction plan has been developed or is being developed for such species or stock under Section 118. Conditions 2 through 5 have been met, and NMFS has prepared a draft negligible impact determination (May 4, 2000). On June 6, 2000, NMFS published in the Federal Register a request for comments on its proposal to issue a permit to authorize the incidental taking of four stocks of threatened or endangered marine mammals by the CA/OR drift gillnet fishery (65 FR 35904).

The CA/OR drift gillnet fishery is known to entangle and, in some cases, kill several listed species of marine mammals and sea turtles. For the purpose of determining the effects of the proposed permit issuance on listed species, the CA/OR drift gillnet fishery is described below and analyzed in the subsequent Effects of the Action section.

A. History and Description of the CA/OR Drift Gillnet Fishery

During the late 1970s and early 1980s, the use of entangling nets in coastal California waters to harvest a number of oceanic and near-shore species expanded rapidly. The modern drift gillnet fishery developed during the late 1970s in the waters between Point Arguello, surrounding the Channel Islands, and San Diego, off southern California. Since then, the fishery developed rapidly and extensively along the west coast as far north as Oregon, and seaward beyond 200 miles. Initially, the targeted species were pelagic sharks, primarily common thresher (*Alopias vulpinus*) and shortfin mako (*Isurus oxyrinchus*), known locally as bonito shark (Hanan *et al*., 1993). Swordfish (*Xiphias gladius*) has overtaken shark in both the quantity and the value of drift gillnet landings, and the shortfin mako shark and two additional fish (opah (*Lampris guttatus*) and louvar (*Luvarus imperialis*)) have become important components of the catch (Herrick and Hanan, 1988). Currently, the drift gillnet fishery operates primarily in southern and central California, between San Diego and Cape Mendocino, and swordfish constitutes the majority of the catch (NMFS, 1997b). The majority of the total effort is concentrated in the Southern California Bight (Hanan, *et al*., 1993; NMFS, 1997b).

Vessels fishing in the CA/OR drift gillnet fishery range from 30 to 82 feet (9 to 25 meters), with more than 60 percent of the vessels greater than 50 feet. Over time, the average size of the vessels has increased, especially for those fishing farther offshore and northward. The nets are constructed of 3-strand twisted nylon, tied to form 16 to 22 inch stretched mesh (14 inch minimum). Large fish, such as swordfish, get entangled, while smaller fish pass through the mesh. The net length ranges from 750 to 1000 fathoms (4,500 to 6,000 feet) horizontally and is stretched vertically by a float line and a weighted lead line. The depth of the net ranges from 100 to 150 meshes. Prior to the PCTRP, the float line was set usually 18 to 26 feet below the surface of the water to allow small boats to pass over the net (referred to as the “buoy line depth” or “extender length”). Under the PCTRP, the float line is set at a minimum depth of 36 feet

Oceanic conditions and long-term climatic trends significantly affect swordfish populations and their distribution. Water temperature appears to be one of the most significant factors affecting distribution, although thermal fronts may be a more important determinant rather than absolute temperatures. Other phenomenon include upwelling and thermocline depths, both of which involve changes in temperature. Bottom topography may also affect swordfish behavior, although less is known of this association. Swordfish primarily occur in temperatures from 13°-24°C and appear to be most abundant in areas with sharp temperature gradients, such as regions of upwelling, zones where various water masses converge, or along pronounced ocean currents (Weidner and Serrano, 1997). Drift gillnet fishermen take advantage of these swordfish associations with ocean phenomena, and nets are often set perpendicular to currents or across gradients such as temperature, salinity, or turbidity fronts. Swordfish monitored with acoustic telemetry have been found to remain near the bottom waters during the daytime, where they may feed on demersal fish, and at night, they stayed close to the surface, where they are believed to have been feeding on squid and other fauna concentrated in the upper layer of the water column (*in* Joseph, *et al*., 1994).

Fishermen primarily set their nets in the evening, in varying depths of water (250-2,250 fathoms), soak them overnight, then retrieve them in the morning. Soak duration is typically 12-14 hours, depending on the length of the night (Barlow *et al*., 1997). The vessel remains attached to one end of the net during this soak period, drifting with the net. During retrieval, the net is pulled over the stern by a hydraulic net reel. As the net is pulled, anything caught in the net can usually be seen coming to the surface, at which point the reel is slowed and stopped if the catch is too large. The catch is either pulled aboard in the net, or if too large, tied with a line so as not to be lost and winched aboard. Once onboard, entangled fish are removed from the net using routine procedures. Marine mammals and sea turtles tend to roll up in the net when caught; therefore a few strands of the net usually have to be cut to remove them. Those marine mammals which are still alive are released at the water line when possible or, if necessary for human health and safety concerns, killed, removed from the net, and discarded. A marine mammal or sea turtle that is too large for the hydraulic equipment to pull aboard is typically cut free at the water line (Hanan *et al*., 1993).

The observed distribution of swordfish catch and effort by drift gillnetters from 1990 to 1998 extended from the U.S.-Mexican border to the Columbia River (Oregon-Washington border), with swordfish being the primary target in more than 95% of the observed sets. Effort initially concentrates in the southern portion of the fishing grounds, expanding to its full range by October, and finally retreating back to the south to avoid the winter storms and because the ocean waters in this area are usually the last to cool down. The highest catch of swordfish occurs 10 to 100 miles off the California coast, and in the higher latitudes, swordfish catch and effort tends to be further offshore. Fishing effort within 10 miles of the coast or near the Channel Islands usually targets pelagic sharks (Rasmussen and Holts, 1999).

1. California Drift Gillnet Fishery

The California Department of Fish and Game (CDFG) manages the California drift gillnet fishery, which is regulated by laws passed by the state legislature. The fishery became a limited entry fishery in 1980, setting a maximum number of 150 permits and allowing those already involved to continue fishing (Hanan *et al*, 1993). Thus, drift gillnet fishermen are required to possess a valid drift gillnet permit in order to fish. In addition, fishermen are required to maintain and submit a logbook detailing their fishing activities (NMFS, 1997b). CDFG does not issue new permits except to those who applied prior to 1986. General gillnet permits must be renewed annually and are only transferable under limited conditions; the permittee must have held the permit for at least 20 years and have made landings for 15 of those years.

Since the early 1980s, in response to concerns for the bycatch of other fish species and marine mammals (mainly seals and sea lions) and due to conflicts with recreational or harpoon fisheries, the California drift gillnet fishing season has become shorter, and area restrictions have increased. Currently, the drift gillnet season is closed from February 1 through April 30, although fishers can fish past 200 nautical miles (nm) from shore. From May 1 through August 14, drift gillnets cannot be used to take shark or swordfish in open waters within 75 nm from the mainland coastline between the westerly extension of the CA/OR boundary down to the U.S. Mexican border. However, a permittee may land swordfish or thresher shark if the fish were taken in waters more than 75 nm from the mainland shore. Swordfish can be taken within 75 nm from August 15th through January 31st, pursuant to additional area restrictions contained in the CDFG code. The majority of fishing effort takes place from October through December (NMFS, 1997b).

Overall, fishing effort in the California drift gillnet fishery has declined since the mid-1980s, mainly due to time-area closures. In the 1986-87 season, there were 11,000 sets (equivalent to days fished), while during the 1994-95 and 1995-96 season there were 3,689 and 3,755 sets, respectively (NMFS, 1997b), and in 1997 and 1998, approximately 3,039 and 2,907 sets were made (Cameron and Forney, 1999). The decrease in effort coincides with increasing regulations and laws, and a decrease in the number of active permittees. Legislation passed in the early 1980s established the fishery as a limited entry fishery with a maximum of 150 permits. Since the actual number of permittees at that time exceeded 150, new entrants were not allowed. However, an additional 35 permits, referred to as experimental swordfish permits, were established in 1984 for taking swordfish north of Point Arguello. In the 1986-87 season, there were over 210 active permittees (those that caught and landed fish) participating in the fishery, while in the 1994-95 seasons, there were 124 active permittees, with 31% making three or less landings. Recently, the 35 experimental swordfish permits were combined with the 150 permits. However, not all available permits have been re-issued, due to attrition, retirement, death, etc. (NMFS, 1997b). Thus, the number of eligible permit holders in California from 1994-1998 (calendar year) was 162, 185, 167, 120 and 147, respectively, while the number of vessels actively fishing during 1995, 1997 and 1998 were 130, 115 and 123 (3 from Oregon), respectively (Forney, *et al*., 2000). Data for 1994 and 1996 were not available.

California drift gillnet landings for swordfish, common thresher shark, and mako shark vary from season to season. As described previously, swordfish comprise the majority of the catch and demand the highest price per pound. From 1990-95, the California drift gillnet fishery averaged $7.2 million in ex-vessel value from landings of shark and swordfish. Based on multipliers of ex-vessel values, including shipyard, fuel docks, insurance companies, wholesale and retail fish markets, restaurants, etc., the total economic value of the fishery was estimated to be in excess of $36 million dollars per year (NMFS, 1997b). Due to declining effort in recent years (average 4,503 sets made annually from 1990-95 verses an average of 3,033 sets made annually from 1996-99), the economic value of this fishery has most likely decreased.

2. Oregon Drift Gillnet Fishery

Prior to 1995, even though drift gillnet vessels originating from California ports fished for swordfish off the coast of Oregon (outside 3 nm since 1987), no swordfish could be landed in the state. Therefore, the State of Oregon did not benefit economically from the fishery. In 1995, the Oregon state legislature enacted a new developmental fishing program, which essentially allowed the Oregon Department of Fish and Wildlife (ODFW) to implement a developmental gillnet fishing program. Consequently, the ODFW issued (by lottery) ten "unlimited" landing permits, which allowed swordfish to be landed in Oregon ports by drift gillnet vessels. From 1995-98, ten permits were issued, and in 1999, only 6 were issued, although landings have only been made by between 3 and 6 vessels each year. From 1995 to 1999, the total number of annual landings into Oregon ports were 2, 8, 6, 21, and 8, respectively (J. McCrae, ODFW, personal communication, May, 2000).

The number of developmental fishing permits that could be issued by ODFW is currently unlimited. However, ODFW's current policy is that only ten permits with "unlimited" landing ability will be issued each year. In this biological opinion, we analyzed the fishery based on effort in the recent past (including the low number of permits for Oregon). Based on past effort in Oregon, see landings info above, it doesn't seem likely that effort will suddenly increase. Developmental fishing gillnet permits are not transferable, and ODFW has stipulated that federal regulations apply to Oregon drift gillnet vessels fishing for swordfish (e.g. requiring the use of pingers, and minimum length of extenders) (J. McCrae, ODFW, personal communication, May, 2000).

3. Observer Coverage

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

Since 1990, fishing effort has been observed from the waters off San Diego to the waters off Oregon, and out beyond 200 miles from shore. Observers record bycatch by taxon for fish, marine mammals, and sea turtles, collect specimens, and record data on environmental conditions and over 10 different net characteristics (NMFS, 1997b). From 1990-99, the percentage of observer coverage was 4.0%, 9.9%, 13.2%, 13.5%, 18.0%, 15.6%, 13.0%, 22.8%, 17.5% and 20.0%, for an annual average of approximately 16% from 1991-99 (full year) (CDFG unpublished data). Between July, 1990 and January 31, 2000, NMFS has observed 5,580 sets. Observer coverage is distributed equally along the coast based on expected effort. The observer coverage is representative of the effort occurring off the west coast. Vessels are selected on an opportunistic basis. A vessel is required to carry an observer about 20 percent of the time. Therefore, if a boat just had an observer, they are not required to carry another observer until it would approach their 20 percent requirement. Vessels are notified of this obligation when they report their arrival or departure information - or at the docks, by an observer monitoring vessel activity

4. Take Reduction Plan and Implementing Regulations for the CA/OR drift gillnet fishery

Section 118(f) of the MMPA requires that NMFS develop and implement take reduction plans (TRPs) to assist in the recovery, or prevent the depletion of, strategic marine mammal stock(s) which interact with Category I or II fisheries. A strategic stock is (1) a marine mammal species that is listed as endangered or threatened under the ESA; (2) a marine mammal stock for which the human-caused mortality exceeds the potential biological removal (PBR) level; or (3) a marine mammal stock which is declining and likely to become listed as a threatened species under the ESA. The PBR level is the maximum number of animals, not including natural mortalities, that may be annually removed from a marine mammal stock while allowing that stock to reach or maintain its optimal population level.

Because the CA/OR drift gillnet fishery for thresher shark and swordfish, classified as a Category I fishery under the MMPA, incidentally took several marine mammal stocks at levels that were estimated to be above their PBR levels, NMFS convened the Pacific Offshore Cetacean Take Reduction Team on February 12, 1996 (61 FR 5385). The team was charged to provide a draft PCTRP to NMFS by August 1996. After a series of meetings to formulate, draft, and discuss the plan, the team submitted the final take reduction plan to NMFS on July 18, 1997.

Since the implementation of the PCTRP on October 30, 1997, the CA/OR drift gillnet fishery for swordfish and thresher shark has incidentally taken the following species of marine mammals and sea turtles: green turtle, leatherback turtle, loggerhead turtle, and olive ridley turtle, fin whale, humpback whale, and sperm whale.

PCTRP Regulations
On October 3, 1997, NMFS published regulations to implement the PCTRP (62 FR 51813), which became effective on October 30, 1997. An interim final rule, published on January 22, 1999 (64 FR 3431) allowed pingers to be deployed further away from the net (see Pingers, below). The regulations apply to all U.S. drift gillnet vessels operating in waters seaward of the coast of California or Oregon, including adjacent high seas waters. The fishing season is defined as beginning May 1 and ending on January 31 of the following year. The current regulations are:

Extenders – all CA/OR drift gillnet vessels must adhere to the minimum depth-of fishing requirement of 6 fathoms (36 feet). Thus, all vessels in this fishery must use extenders (lines attaching the float line to buoys/floats on the sea surface, suspending the net in the water at a particular depth) of at least 6 fathoms (36 feet) for all sets.

Pingers – pingers must be used on all sets beginning October 30, 1997. Pingers are acoustic deterrent devices which, when immersed in water, broadcasts a 10 kHz (±2 kHz) sound at 132 dB (±4 dB) re 1 microPascal at 1 meter, lasting 300 milliseconds (+15 milliseconds) and repeating every 4 seconds (+0.2 seconds), and remain operational to a water depth of at least 100 fathoms (600 feet). Pingers alert animals that are acoustically sensitive to this frequency to the existence of the net by acoustically "illuminating" it, allowing the mammal to "see" and avoid the mesh. Pingers must be attached within 30 feet of the floatline and within 36 feet of the leadline, and spaced no more than 300 feet apart. Pingers attached within 30 feet of the floatline and within 36 feet of the leadline must be staggered, such that the horizontal distance between them is no more than 150 feet.

Skipper education workshops – vessel owners must attend a skipper education workshop before commencing fishing each fishing season.

B. Description of the Action Area

The proposed action is the authorization for the CA/OR drift gillnet fishery to incidentally take marine mammals under section 101(a)(5)(E) of the MMPA. Fishing effort for swordfish by the CA/OR drift gillnet fishery primarily occurs in waters off San Diego, north to San Francisco, and within 300 miles of shore. Small numbers of swordfish are also caught between San Francisco and the California-Oregon border and within 125 miles of shore, and very few swordfish catches are made north of Oregon. Fishing effort for swordfish usually peaks in October and November and tapers off in December and January (Holts and Sosa-Nishizaki, 1998). Thresher shark are mainly targeted within 9 miles (8 nm) of the coast or near the Channel Islands, where mean water depth is approximately 400 fathoms. Thus the action area, for the purposes of this Opinion, is the body of water delineated by the California-Mexico border to the south (30°N latitude), the Oregon-Washington border to the north (45°N), extending as far west as 129°W (Julian and Beeson, 1998).

III. STATUS OF AFFECTED SPECIES

The following endangered and threatened species occur in the action area and may be affected by the continued operation of the CA/OR drift gillnet fishery, as regulated under the PCTRP, and the issuance of section 101(a)(5)(e) permits for this fishery to take marine mammals:

Marine Mammals	**Status**
Blue whale (*Balaenoptera musculus*)	Endangered
Fin whale (*Balaenoptera physalus*)	Endangered
Guadalupe fur seal (*Arctocephalus townsendii*)	Threatened
Humpback whale (*Megaptera novaeangliae*)	Endangered
Right whale (*Eubalaena glacialis*)	Endangered
Sei whale (*Balaenoptera borealis*)	Endangered
Sperm whale (*Physeter macrocephalus*)	Endangered
Steller sea lion - eastern population (*Eumetopias jubatus*)	Threatened

Sea turtles	**Status**
Green turtle (*Chelonia mydas*)	Endangered/Threatened
Leatherback turtle (*Dermochelys coriacea*)	Endangered
Loggerhead turtle (*Caretta caretta*)	Threatened
Olive ridley turtle (*Lepidochelys olivacea*)	Endangered/Threatened

Although blue whales (*Balaenoptera musculus*), right whales (*Eubalaena glacialis*), sei whales (*Balaenoptera borealis*), and Guadalupe fur seals (*Arctocephalus townsendi*) are found within the action area and could potentially interact with the CA/OR drift gillnet fishery, there have been no reported or observed incidental takes of these species in the drift gillnet fishery since the fishery was first observed by NMFS, in 1990. Consequently, NMFS has determined that the CA/OR drift gillnet fishery does not require authorization pursuant to section 101(a)(5)(E) of the MMPA for these species. The proposed action is not likely to adversely affect blue whales, northern right whales, sei whales, or Guadalupe fur seals, and these species will not be considered further in this Opinion.

In addition, all listed species of Pacific salmon (*Oncorhynchus spp.*) may occur within the action area during the pelagic stage of their life history. These species have never been reported as captured during CA/OR drift gillnet fishery operations; therefore, NMFS has determined that these species are not likely to be adversely affected by the proposed action and will not be further considered in this opinion.

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

Critical habitat was established for the Steller sea lion in 1993 (58 FR 45269). In 1997, the Steller sea lions separated into two distinct population segments; eastern and western populations, although critical habitat had been designated for both populations. All major rookeries for Steller sea lion in the action area, which are contained in the eastern population of Steller sea lions, and associated air and aquatic zones were designated as critical habitat (Oregon: Rogue Reef/Pyramid Rock, Orford Reef/Long Brown Rock, and Seal Rock; California: Ano Nuevo Island, Southeast Farallon Islands, Sugarloaf Island/Cape Mendocino). The air zone extends 3,000 feet (0.9 km) above areas historically occupied by Steller sea lions at each major rookery in California and Oregon, measured vertically from sea level. The aquatic zone extends 3,000 feet (0.9 km) seaward in state and federally managed waters from the baseline or basepoint of each major rookery in California and Oregon.

The proposed action and the associated operation of the CA/OR drift gillnet fishery should not occur within or near areas of Steller sea lion critical habitat off the coasts of Oregon or California due to state regulations which prohibit operation of the fishery within 3 miles of the coast. Therefore, designated critical habitat will not be considered in this Opinion.

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

The following subsections are synopses of the current state of knowledge on the life history, distribution, and population trends of marine mammal and sea turtle species that have been observed incidentally taken by the CA/OR drift gillnet fishery since the fishery was first observed, beginning in 1990, to the present, and that NMFS expects may be taken as a result of the issuance of marine mammal permits under section 101(a)(5)(E) of the MMPA and the ongoing operations of the fishery, as amended by PCTRP regulations.

A. Status of Marine Mammals

Most large whales are listed as endangered species under the ESA because their populations were depleted by whalers in the nineteenth and twentieth centuries. Currently, ship strikes and incidental take in commercial fishing operations (domestic and international) are most likely the greatest threat to the recovery of large cetaceans. The factors that have caused the decline in Steller sea lion abundance are poorly known; however, concerns have been raised regarding reduced prey availability due to increased commercial fishing in critical foraging areas. Furthermore, vessels such as tankers, freighters, military vessels, commercial fishing vessels, whale watching and recreational boats all create disturbance and underwater noise that is potentially harmful to marine mammals. The individual and cumulative effects of these sources of noise and disturbance on marine mammals is unknown.

Under the 1994 amendments to the MMPA, NMFS was required to produce stock assessment reports (SARs) for all marine mammal stocks that occur in U.S. waters. These reports include information on the status and trends of marine mammals and assessments of all human-caused mortality and serious injury of the listed marine mammal stocks. Information on fin whales, humpback whales, sperm whales and the Steller sea lion was obtained from both final and draft SARs and is presented below, along with other relevant information (sources identified therein).

1. Individual Marine Mammal Species and Factors Affecting Them in the Pacific Ocean

a. *Fin whale*

Fin whales are widely distributed in the world's oceans and are the second largest member of the family Balaenopteridae, reaching lengths of between 20 and 29 meters at adulthood (Aguilar and Lockyer, 1987). Fin whales are dark gray dorsally and white underneath, with a long, slender body and a prominent dorsal fin about two-thirds of the way back on their body (Agler *et al.*, 1990, *in* Reeves, *et* *al*., 1998). Like other baleen whales, fin whales have fringed baleen plates and ventral grooves, which expand during feeding. In the North Pacific Ocean, fin whales prefer to feed on euphausiids and large copepods (mainly *Calanus cristatus*), followed by schooling fish such as herring, walleye pollock, and capelin (Reeves, *et al*., 1998). Sergent (1977, *in* Reeves, *et al*, 1998) suggested that euphasiids were the basic food of fin whales, but that they took advantage of fish when sufficiently concentrated, particularly in the pre-spawning, spawning, and post-spawning adult stages on the continental shelf and in coastal waters. They have been known to associate with steep contours, either because tidal and current mixing along such gradients drives high biological production, or because changes in depth aid their navigation. The local distribution of fin whales during much of the year is probably governed by prey availability. Although there has been considerable discussion of interspecific competition among mysticete whales for prey, there has been no conclusive evidence to demonstrate that it occurs (Clapham and Brownell, 1996, *in* Reeves, *et al*., 1998).

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

Fin whales have a complex migratory behavior that appears to depend on their age or reproductive state as well as their “stock” affinity. Movements can be either inshore-offshore or north-south. Fin whales have been observed year-round off central and southern California, with peak numbers in the summer and fall. Peak numbers of fin whales have also been seen during the summer off Oregon and in summer and fall in the Gulf of Alaska and southeastern Bering Sea (*in* Perry, *et al*, 1999). Rice (1974) reported that several fin whales tagged from November to January off southern California were later killed by whalers in May to July off central California, Oregon, and British Columbia and in the Gulf of Alaska, suggesting possible southern California wintering areas and summering areas further north. Although fin whale abundance is lower in winter/spring off California, and higher in the Gulf of California, further research and surveys need to be conducted in order to determine whether fin whales found off southern and central California migrate to the Gulf of California for the winter (Forney, *et al*., 2000).

Prior to whaling, the total north Pacific fin whale population was estimated to be between 42,000 and 45,000, based on catch data and a population model (Ohsumi and Wada, 1974, *in* Perry, *et al*., 1999). Of this, the "American population" (i.e. the component of the population centered in waters east of 180° longitude) was estimated to be 25,000-27,000. Fin whales were hunted, often intensely, in all the world's oceans for the first three-quarters of a century, until they were given full protection from commercial whaling in the Pacific Ocean in 1976 (Reeves, *et al*., 1998). The fin whale was listed as endangered in 1970, under the Endangered Species Conservation Act of 1969, the predecessor to the current ESA.

Based on a "conservative management approach," NMFS recognizes three stocks of fin whales in U.S. Pacific waters: Alaska, California/Washington/Oregon, and Hawaii (Barlow *et al*., 1997; Reeves, *et al.*, 1998). Shipboard sighting surveys in the summer and autumn of 1991, 1993 and 1996 produced an estimate of 1,236 (coefficient of variation (CV)=0.20) fin whales comprising the California, Oregon and Washington "stock," with a minimum estimate of 1,044 animals (Forney, *et al*., 2000). An increasing trend between 1979-80 and 1993 is suggested by the available survey data, but it is not statistically significant (Barlow, 1997). No data are available on the estimated abundance of the Hawaiian stock or the Northeast Pacific (Alaska) stock of fin whales (Forney, *et al*., 2000; Hill and DeMaster, 1999). Only one fin whale was seen on vessel cruises in the eastern tropical Pacific Ocean from 1986 through 1990; therefore, no abundance estimates were available for this region (Wade and Gerrodette, 1993).

Threats to fin whales. Because little evidence of ship strikes and entanglement in fishing gear exists, and large whales such as the fin whale may often die later and drift far enough not to strand on land after such incidents, it is difficult to estimate the numbers of fin whales killed and injured by ship strikes or gear entanglement. However, the evidence that has been gathered demonstrates that such events are rare occurrences (Heyning and Lewis, 1990; Barlow, *et al*., 1997). In 1997, the eastern tropical Pacific tuna purse seine fishery accidentally killed "one unidentified baleen whale," although there is no information available to determine whether the whale was a listed species (IATTC, 1999). However, since 1993, the fishery has had 100 percent observer coverage, and in over 100,000 sets, only one baleen whale has been killed. Therefore, the likelihood of this fishery taking a large listed baleen whale, such as a fin whale, is considered to be extremely low. In addition, no major habitat concerns have been identified for the fin whale, and there is no evidence that levels of organochlorines, organotins or heavy metals in baleen whales generally (including the fin whale) are high enough to cause toxic or other damaging effects (O'Shea and Brownell, 1995, *in* Reeves, *et al.*, 1998). However, there is a growing concern that the increasing levels of anthropogenic noise in the ocean may be a habitat concern for whales, particularly for whales that use low frequency sound to communicate, such as baleen whales (Forney *et al.*, 2000).

b. *Humpback whale*

The humpback whale, also a member of the family Balaenopteridae, is distributed worldwide in all ocean basins. Most humpback whales winter in shallow, nearshore temperate and tropical waters, whereas in summer, most are in areas of high biological productivity, usually in the higher latitudes (Nitta and Naughton, 1989). They probably mate and give birth while in the wintering areas, but reproductive events may take place during migration. Following reproduction and birthing, most humpback whales migrate considerable distances to the higher latitude summering areas, where they feed intensively on locally abundant prey (NMFS, 1991). Humpback whales are often found alone or in groups of two or three, but throughout their breeding and feeding ranges, they may congregate in groups of up to fifteen animals. Their distribution in general is over shallow banks and in shelf waters (Leatherwood and Reeves, 1983).

The humpback whale is of medium size relative to other large whales, with females reaching an average length of around 14 meters, while males average 1 meter shorter (Nitta and Naughton, 1989) and a weight of about 34 metric tons at maturity (Johnson and Wolman, 1984 *in* Perry *et al.*, 1999). They are characterized by wing-like pectoral flippers that are from one-fourth to one-third of their total body length and their heads are covered in tubercles. Humpback whales have a varied diet, preying on krill (euphausiids), copepods, juvenile salmonids (*Oncorhynchus* spp.), Arctic cod (*Boreogadus saida*), walleye pollock (*Theragra virens*), pteropods and some copepods (Johnson and Wolman, 1984, *in* Perry, *et al*., 1999). Humpback whales observed in the Gulf of the Farallones and adjacent waters off California from 1988-90 fed primarily on euphasiids, and only occasionally fed on small schooling fish (Kieckhefer, 1992). Humpback whales use a wide variety of fishing techniques, at times involving more than one individual and resembling a form of cooperative participation. The two most observable techniques are lob-tail feeding and bubble-cloud feeding. Recently, there has also been documentation of bottom-feeding by humpback whales in the Atlantic (*in* Perry, *et al*., 1999). Whether humpback whales in the Pacific feed in this manner is currently unknown; however it is assumed that baleen whales do not dive beyond 300 meters in depth (Nemoto, 1963, *in* Kieckhefer, 1992). A study of dive behaviors of humpback whales in Alaska found that 66 percent of the dives were made to depths of between 0 and 20 meters (~65 feet), while only 15 percent of the dives extended beyond 60 meters (Dolphin, 1986).

Humpback whales calve between the months of January and March. Age at sexual maturity has been estimated to range from 4 to 9 years in females, but there is no reliability associated with those estimates, since age estimates used in the past have been questioned, as have the reliability of the data (Clapham and Mayo, 1987). Nishiwaki (1965, *in* Nitta and Naughton, 1989) reported the length at sexual maturity for females to be between 11.4 and 12.0 meters, and for males, between 11.1 and 11.4 meters. The calving interval is also variable: a range of 2-3 years has been given; however, there is some evidence of calving by females in consecutive years. Gestation averages around 12 months, and lactation lasts nearly a year. The majority of calves are weaned at 1 year, but the specific timing of separation is still unknown (*in* Perry, *et al*., 1999). In the North Pacific, annual reproductive rates have been estimated from information collected in wintering and summering areas: the least biased estimate came from southeastern Alaska, where the calving rate[1] was estimated to be 0.37. Thus, on average, a mature female gives birth only once every 2.7 years (inverse of calving rate) to a calf that survives its first six months of life and its first migratory transit (Baker, *et al*., 1987).

Prior to 1905, there were an estimated 15,000 humpback whales in the entire North Pacific (Rice, 1978). Following heavy exploitation, the population was estimated to be between 1,000 (Rice, 1978) and 1,200 (Johnson and Wohlman, 1984, *in* Perry, *et al.,* 1999) animals in 1967, when it was given protective status by the International Whaling Commission, although it is not clear whether these estimates represent the entire North Pacific or only the eastern North Pacific (Perry *et al*., 1999). The humpback whale was listed as endangered under the Endangered Species Conservation Act of 1969 throughout its range on June 2, 1970.

Currently, there are no statistically reliable estimates of humpback whale population abundance for the entire North Pacific Ocean. Based on aerial, vessel, and photo-identification surveys, and genetic analyses, within the Exclusive Economic Zone (EEZ), there are at least three relatively separate populations that migrate between their respective summer/fall feeding areas and their winter/spring calving and mating areas: 1) winter/spring populations in coastal Central America and the Pacific coast of Mexico which migrate to the coast of California and north to southern British Columbia in the summer/fall, referred to as the California/Oregon/Washington - Mexico stock; 2) winter/spring populations off the Hawaiian Islands which migrate to northern British Columbia/Southeast Alaska, and Prince William Sound west to Kodiak, referred to as the Central North Pacific stock; and 3) winter/spring populations of Japan which probably migrate to waters west of the Kodiak Archipelago (Bering Sea and Aleutian Islands), referred to as the Western North Pacific Stock. Winter/spring populations of humpback whales also occur in Mexico's offshore islands (i.e. Revillagigedo Archipelago), but the migratory destination of these whales is not well known (Forney, *et al*., 2000). Medrano-González, *et al*. (1995) reports of resightings of a few of these offshore Mexican island breeding whales in Vancouver and in the western Gulf of Alaska. Connections between humpback whales in the Hawaiian and Mexican breeding areas and the North Pacific feeding areas have been observed (Darling and Jurasz, 1983; Baker *et al*., 1990; Calambokidis, *et al*., 1997), although fewer genetic differences were found between the two breeding areas than the two feeding areas (California and Alaska) (Baker, 1992). In addition, the genetic exchange rate between California and Alaska is estimated to be less than one female per generation (Baker, 1992), and only 2 out of 81 humpback whales photographed in British Columbia have matched with whales photographed in California (Calambokidis, *et al*., 1996). Therefore, the U.S./Canadian border is estimated to be the northern boundary of the California/ Oregon/ Washington - Mexico stock. Humpback whale stocks that may interact with the CA/OR drift gillnet fishery most likely include those that range from the western coast of Costa Rica to southern British Columbia, but are most common in coastal waters off California (in

[1]Calving rate - the proportion of individually identified females, assumed to be sexually mature, accompanied by calves in a given year or summed across years and expressed on a per-year basis. The calving rate of an individual female is equal to the inverse of her calving interval (Baker *et al*., 1987).

summer/fall) and Mexico (in winter/spring).

Calambokidis, *et al*. (1997) estimated the total North Pacific population of humpback whales to exceed 4,000; however, without knowing where some of the Mexican breeding stocks migrate, the current estimate is lower than this. The most precise and least biased population estimate for the CA/OR/WA - Mexico stock feeding group is 905, with a minimum estimate of 861 animals. Mark-recapture population estimates have increased from 1988-90 to 1997-98 at about 8% per year (Forney, *et al*., 2000). Based on photographic identification of individual animals, Urbán, *et al*. (1999) estimates the population size of the Mexican coastal stock to be 1,813 and the abundance of the Revillagigedo stock to be 914. Based on the results of photo-identification studies of humpback whales in their wintering areas, the current population estimate for the Central North Pacific stock is 4,005 (CV=0.095), with a minimum estimate of 3,698 whales. Using this same data, the most recent abundance estimate for the Western North Pacific stock of humpback whales is 394 (CV=0.084) animals, with a minimum estimate of 367 (Hill and DeMaster, 1999). Combining all three stocks yields a total abundance estimate of 5,304 (minimum 4,926) humpback whales in the North Pacific. This estimate does not include the Mexican breeding stock abundance estimates, because most of these animals are included in the estimates of the CA/OR/WA - Mexico feeding stock. Furthermore, population estimates for the entire North Pacific have increased substantially from 1966 to the early 1990s, at 6-7% per year (Forney, *et al*, 2000). Ship surveys conducted from 1986 through 1990 in the eastern tropical Pacific Ocean yielded sightings of humpback whales in the California and Peru currents, in the Gulf of Panama, and along the coast of Guatemala; however, there was not enough information to provide abundance estimates (Wade and Gerrodette, 1993).

Threats to Humpback Whales. Humpback whales are rarely taken in commercial fishery operations, although any estimates are probably much lower than actual, as observer coverage for some fisheries (e.g. Hawaiian longline) has been low, and in recent years, the numbers of humpback whales reported with trailing fishing gear have increased (Mazzuca *et al*., 1998).

Based on observer data from six different Alaskan commercial fisheries from 1990-98, and self-reported fisheries information from 1990-98, there was one humpback whale, probably from the Western North Pacific stock[2], observed dead and entangled in the Bering Sea/Aleutian Islands groundfish trawl fishery in 1998, yielding an average mortality for this stock of 0.2 whales per year. In addition, one humpback was reported floating dead, entangled in netting and trailing buoys in 1997, although it is unclear which fishery (or even which country) was responsible. Nevertheless, averaging this mortality over a five-year period (1994-98) yields an average annual mortality of 0.2 humpback whales, bringing the total estimated annual mortality rate incidental to commercial fisheries for this stock to be 0.4 whales per year (Ferrero, *et al*., 2000).

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[2]Because the stock identification is uncertain, and mortality may have been attributable to the Central North Pacific Stock, this mortality is assigned to both Central and Western stocks.

Of the Central North Pacific stock of humpback whales, one animal was observed entangled and expected to die due to interaction with a Hawaiian longliner from 1990-1999; however, due to the low level of coverage during that year (1991), a mortality estimate was not given. The one humpback mortality in the Bering Sea/Aleutian Islands groundfish trawl fishery (described above) brings the estimated mean annual mortality rate from 1994-98 to 0.2 per year for this stock. In addition, during this time period, humpback whales were reported killed by the southeastern Alaska salmon drift gillnet fishery (one mortality reported by self reports, two mortalities from stranding data), the salmon purse seine fishery (one animal reported by self reports), and by unknown fisheries in Alaska and Hawaii (estimated 2 per year). The mean annual mortality of the Central North Pacific stock of humpback whales due to fisheries-related interactions is estimated to be 2.8 whales per year (Ferrero, *et al*, 2000). Lastly, in 1997, the eastern tropical Pacific tuna fishery accidentally killed "one unidentified baleen whale," although there is no information available to determine whether the whale was a listed species (IATTC, 1999). However, since 1993, the fishery has had 100 percent coverage, and in over 100,000 sets, only one baleen whale has been killed. Therefore, the likelihood of this fishery taking a large listed baleen whale, such as a humpback, is considered to be extremely low.

Drift gillnet fisheries for swordfish and sharks exist along the entire Pacific coast of Baja California and may take animals from the CA/OR/WA-Mexican stock of humpback whales. Since 1986, the Mexican fleet has increased from two vessels to 31 in 1993, and in 1992, the observed bycatch of marine mammals was 0.13 animals (10 animals in 77 observed sets, with approximately 2,700 total sets for that year). Unfortunately, species-specific information is not available (Holts and Sosa-Nishizaki, 1998). In addition, there are currently efforts underway to convert the Mexican swordfish driftnet fishery to a longline fishery (P. Ulloa, National Institute for Fisheries, Mexico City, personal communication, May, 2000), which would considerably reduce the incidental take of marine mammals.

In addition to mortality from commercial fishing interactions, humpback whales have been killed by ship strikes and interactions with vessels unrelated to fisheries. The average annual mortality due to ship strikes and entanglement in non-fisheries gear for the Central North Pacific stock is 0.6 whales per year, and none reported for the Western Pacific stock (Ferrero, *et al*., 2000). Lastly, there is a growing concern that the increasing levels of anthropogenic noise in the ocean may be a habitat concern for whales, particularly for whales that use low frequency sound to communicate, such as baleen whales (Forney *et al.*, 2000).

c. *Sperm whale*

The sperm whale, a member of the family Physeteridae, is the largest of the toothed whales, and is one of the most widely distributed of marine mammals in all oceans of the world, between 60°N and 70°S (Leatherwood and Reeves, 1983). The sperm whale is distinguished by its huge boxlike head (up to 40 percent of its body length), a dark grayish brown body, with a rounded or triangular hump followed by knuckles along its spine. Its blunt snout houses a large reservoir of spermaceti, a high-quality oil. Sperm whales are generally found in waters deeper than 180 meters (Leatherwood and Reeves, 1983), and have been recorded diving deeper than 2,000 m (Watkins *et al*., 1993). They feed primarily on squid, including the giant squid, *Architeuthis* sp. but may occasionally eat octopus and a variety of fish, including salmon, rockfish, lingcod and skates (Leatherwood and Reeves, 1983; Perry *et al*., 1999). How sperm whales find and catch their prey can only be inferred, because it has never been possible to observe them feeding. Feeding probably takes place at night, and at great depth, so that vision would be of little use to them, except for locating luminous prey species (luminous species of squid comprised 0-97% of the sperm whale's diet in different areas (Clarke, 1980, *in* Rice, 1989). In total darkness, potential prey could not see an approaching whale, so that active random tactile searching, perhaps with the jaw lowered, is one possible method, and may explain why whales have been found entangled in deep-sea cables (Heezen, 1957), and in drift gillnet fishing gear. Although Matsushita (1955, *in* Rice, 1989) claimed that sperm whale feeding was more frequent at dusk and dawn, few studies have found evidence of a daily feeding cycle.

Due to the under-reporting of sperm whale catches to the International Whaling Commission (IWC) by large-scale pelagic whalers in the North Pacific Ocean, the recorded sperm whale catch numbers are most likely significantly under-estimated. Nevertheless, prior to World War II, commercial whalers killed approximately 24,000 sperm whales (includes western and eastern North Pacific), while from 1947-1987, whalers killed an estimated 258,000 sperm whales. By the late 1970s, whalers found few whales, and the IWC banned the killing of all sperm whales in 1988 (*in* Perry, *et al*., 1999). The sperm whale was listed as endangered throughout its range under the Endangered Species Conservation Act of 1969 on June 2, 1970.

Female sperm whales of all ages and juvenile males associate and migrate in groupings called breeding schools, while young males which have approached physiological sexual maturity and have left the breeding schools congregate in bachelor schools. As males grow older (around 30 years old), they become less gregarious and tend to become solitary, only joining the breeding school during the mating season (Gosho, *et al*., 1984).

Females reach sexual maturity at a mean age of 9 years (average 9 meters), after which they generally produce calves every 3-6 years. The gestation period is approximately 15 months, and lactation lasts 1-2 years. Male sperm whales have a long puberty; they begin maturing sexually at around 9 years of age (9 meters body length) and are considered completely sexually mature when the testes are fully spermatogenic at around 20 years of age (12 meters body length) (*in* Perry, *et al*., 1999). The young are conceived and born in the areas of the breeding schools, concentrated between 40°N and 40°S latitude, and off the California coast, the breeding season extends from April to August (Caldwell, *et al*., 1966).

Killer whales (*Orcinus orca*) have been observed attacking sperm whales (Pitman and Chivers, 1999), and serological studies have indicated that sperm whales are carriers of and are infected by calciviruses and papilomavirus (*in* Perry, *et al*., 1999). Estimated natural mortality rates for sperm whales age zero to two years old is 9 percent, while older (age 2 and above) sperm whales have an estimated mortality rate of 5 percent (IWC, 1971, *in* Perry, *et al*., 1999); however, the lack of information on the causes of natural mortality have rendered these estimates statistically unreliable.

In the eastern North Pacific, sperm whales are widely distributed. Females and younger sperm whales tend to remain in tropical and temperate waters year-round, while in the summer, adult males move north to feed in the Gulf of Alaska, Bering Sea and in the waters around the Aleutian Islands. During the winter, sperm whales are generally distributed south of 40°N (Small and DeMaster, 1995). Off California, sperm whales are found year-round, with peak abundances from April through mid-June and from the end of August through mid-November (*in* Forney *et al*., 2000), which suggests a northward migration in the spring and a southward migration in the fall (Gosho *et al*., 1984).

A study conducted in 1997 to estimate the breeding season abundance of sperm whales in the eastern temperate North Pacific (between 20°N-45°N, and west to 165°W) used passive acoustic listening devices to detect numbers of sperm whales, coupled by visual surveys. Barlow and Taylor (1998) found sperm whales to be uniformly distributed in the study area, with no north to south density gradient. Mesnick *et al*. (1999) recently analyzed the genetic relationships of animals in the eastern Pacific and found that the mtDNA and microsatellite DNA of animals sampled in the California Current is significantly different from animals sampled further offshore, although the line of delineation is unknown. It is likely somewhere between the North American coast and halfway to Hawaii (B. Taylor, NMFS - SWFSC, personal communication, March, 2000). Mesnick *et al*. (1999) also found that genetic differences appeared larger in an east-west direction than in a north-south direction. This is confirmed by tagging studies conducted by Rice (1974), who documented three whales tagged in San Francisco and later caught by whalers as far north as British Columbia. Based on differences in gene samples between sperm whales in the Gulf of California, and coastal California, the California-Mexico border is probably near the southern limit of the U.S. west coast stock; however, scientists cannot state with certainty how far west or north the stock may range (B. Taylor, NMFS-SWFSC, March, 2000).

Because of the long dive times and complex social behaviors of sperm whales, it is difficult to estimate their population size, particularly in the eastern Pacific, where scientists are hindered by a lack of data. Nevertheless, sperm whales of the eastern North Pacific have been divided into three separate stocks as dictated by the U.S. waters in which they are found: 1) Alaska (North Pacific stock); 2) California/ Oregon/ Washington; and 3) Hawaii.

A combined visual and acoustic survey conducted by NMFS in 1997 estimated the population of sperm whales in the survey area to be between 24,000 (cv=0.46, based on visual surveys) and 39,200 (cv=0.60, based on acoustic detections and visual group size estimates) (Barlow and Taylor, 1998). However, it is not known how many of these animals enter the U.S. EEZ, where most of the CA/OR drift gillnet fishery takes place. The border to the west and north is less clearly defined, although sperm whales are known to exist thousands of miles from the California/Oregon coastline. Therefore, the abundance estimates contained in the most recent stock assessment report are probably much lower than actual abundance. The Pacific Scientific Review Group (PSRG)[3] has also concluded that sperm whale group size is underestimated and largely a function of the time spent sighting (minutes of PSRG meeting, 5-6 December, 1999; also noted by Barlow and Taylor (1998)), especially since sperm whales can stay submerged for over 60 minutes (Watkins, *et al*., 1993). Since little is known about the western and northern boundaries of the sperm whale stock, the best estimates of abundance within the EEZ, off California, Oregon and Washington, is 1,191 sperm whales, with a minimum estimate of 992 (Barlow (1997)). Furthermore, although sperm whale abundance appears to have been rather variable off California between 1979-80 and 1996, and the eastern North Pacific population is expected to have grown since whaling stopped in 1980, it does not show any obvious trends (Forney *et al*., 2000).

The number of sperm whales occurring along Alaska are unknown; therefore, there are no abundance estimates for the North Pacific stock (Hill and DeMaster, 1999). The abundance of the Hawaiian stock of sperm whales has been estimated to be 66 whales (minimum 43). This number is underestimated, however, because areas around the Northwest Hawaiian Islands and beyond 25 nm from the main islands were not surveyed (Forney, *et al*., 2000). Sperm whales were found throughout the eastern tropical Pacific Ocean on vessel cruises from 1986-90, but appeared to be most abundant in the Gulf of Panama, one of the primary sperm whaling grounds in the eastern Pacific. Abundance estimates of sperm whales in this area were 22,666 animals (95% confidence interval) (Wade and Gerrodette, 1993). It is not known whether any or all of these animals routinely enter the U.S. EEZ of Hawaii (Forney, *et al*., 2000).

Threats to sperm whales. There have been no reported injuries or mortalities of sperm whales in any of the fisheries of Alaska or Hawaii (Hill and DeMaster, 1999; Forney, *et al*., 2000). However, because gillnets and longlines are used in both areas and do take marine mammals, there is the potential that sperm whales could be incidentally captured. In addition, sperm whale interactions with longline fisheries operating in the Gulf of Alaska have been increasing. The first entanglement (uninjured) of a sperm whale was documented in June, 1997 (*in* Forney, *et al*., 2000).

Drift gillnet fisheries for swordfish and sharks exist along the entire Pacific coast of Baja California and may take sperm whales. In 1992, observers documented the rate of marine mammal bycatch to be 0.13 animals per set, although species-specific information is not available for this Mexican fleet (*in* Forney, *et al*., 2000). In addition, the driftnet fleet is currently making an effort to convert to a longline fishery (P. Ulloa, National Institute for Fisheries, Mexico City, personal communication, May, 2000), which would considerably reduce the incidental take of marine mammals.

d. *Steller sea lion*

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[3]The 1994 Amendments to the MMPA required that NMFS establish independent regional scientific review groups in order to advise NMFS on stock assessment reports, research needs, and other appropriate issues. The PSRG was formed in June, 1994.

Steller sea lions range along the North Pacific Ocean rim, from northern Japan, to a centered abundance and distribution in the Gulf of Alaska and the Aleutian Islands, south to California, with the southernmost rookery being Año Nuevo Island (37°N) (*in* NMFS, 1992). Because of a rapid decline (approximately 64%) in Steller sea lion numbers occurring throughout its range, for the three previous decades, NMFS published a 1990 emergency rule listing the Steller sea lion as a threatened species under the ESA (55 FR 49204). In 1997, NMFS reclassified Steller sea lions into two separate stocks within U.S. waters based on distributional data, population response data, and genotypic data: an eastern U.S. stock, which includes animals east of Cape Suckling, Alaska (144°W), and a western U.S. stock, which includes animals at and west of Cape Suckling. On May 5, 1997, the western U.S. stock was reclassified as endangered, while the eastern stock remained on the threatened species list (62 FR 24345).

Steller sea lions are the largest of the family Otariidae, and show marked sexual dimorphism, males averaging 282 cm, 566 kg, and females averaging 228 cm and 263 kg. They have a light buff to reddish brown pelage, and the adult males have long coarse hair on their upper body and back and a massive chest and neck (*in* NMFS, 1992).

The Steller sea lion breeding season is from mid-May to mid-July, and individuals appear to have strong fidelity for their breeding rookery. Pregnant females arrive at the rookery about 3 days before they give birth, and copulation occurs approximately 10-14 days postpartum. Females reach sexual maturity between the age of 3 and 6 and may breed and produce young up into their early 20s. Most adult females breed annually, giving birth to pups after an 8.5 month gestation period (after a 3-4 month delayed implantation of the embryo) (*in* NMFS, 1992). The sex ratio of pups at birth is assumed to be approximately 1:1 (e.g. York, 1994) or biased toward slightly greater production of males (NMFS, 1992). The female-pup bond usually lasts a year; however, 1- to 3-year old animals have been seen still suckling (Pitcher and Calkins, 1981).

Relatively little is known about the life history of sea lions during the juvenile years between weaning and maturity. Males reach sexual maturity between the age of 3 and 7 years of age and physical maturity by age 10. Males and females are expected to live approximately 20 and 30 years, respectively (*in* NMFS, 1992). York (1994) derived age-specific fecundity rates based on data from Calkins and Pitcher (1982). Those rates illustrate a number of important points and assumptions. First, the probability of pupping is rare (about 10%) for animals 4 years of age or younger. Second, maturation of 100% of a cohort of females occurs over a prolonged period which may be as long as 4 years. Third, the reported constancy of fecundity for females extending from age 6 to 30 indicates that either senescence has no effect on fecundity, or information on fecundity rates is not sufficiently detailed to allow confident estimation of age-specific rates for animals older than age 6. Given the small size of the sample taken, the latter is a more likely explanation for such constancy.

Steller sea lions are not known to migrate, but they disperse widely during the breeding season. Males breeding in California appear to spend the non-breeding months (September - April) in Alaska and

British Columbia, whereas animals marked at rookeries in Alaska have traveled to British Columbia (NMFS, 1992). There appears to be limited exchange between rookeries by breeding adult females and males (other than between adjoining rookeries (*in* Ferrero, *et al*., 2000). They are opportunistic feeders, foraging mostly near the shore and over the continental shelf for predominantly demersal and off-bottom schooling fish, including walleye pollock, herring, capelin, mackerel, rockfish, and salmon, and cephalopods such as squid and octopus (NMFS, 1992). They are believed to be capable of diving as deep as 100 fathoms (600 feet), and often reach depths of 60 to 80 fathoms (360 to 480 feet) (Kenyon, 1952).

The most recent abundance estimate of the eastern stock of Steller sea lion is based on: 1) 1996 aerial surveys in Southeast Alaska (14,571 animals); 2) 1996 aerial and ground survey counts of California, Oregon, and Washington rookeries and major haulout sites (6,555 animals) and 3) 1994 aerial surveys of rookeries and haulouts in British Columbia (9,277 animals). Combining the total count for the three regions results in a minimum estimated abundance of 30,403 Steller sea lions in this eastern stock (Ferrero, *et al*., 2000). Trends in Steller sea lion abundance for the three regions has been slightly variable over the past 2 decades. Steller sea lion numbers in California, especially southern and central California, have declined significantly, from 5,000-7,000 non-pups from 1927-1947, to 1,500-2,000 non-pups between 1980-1998. While overall counts of nonpups in northern California and Oregon have been relatively stable since the 1980s, counts of nonpups in Southeast Alaska and British Columbia have increased by an average of 5.9% (1979-97) and 2.8% (1971-98), respectively. Overall, counts of non-pups at haulout trend sites (data from British Columbia include all sites) have increased from approximately 15,000 to over 20,000 eastern stock Steller sea lions from 1982-98 (*in* Ferrero, *et al*., 2000).

Threats to Steller sea lions. Steller sea lions have been observed or reported incidentally taken in the following Alaskan fisheries: drift gillnet, set gillnet, salmon troll, groundfish and halibut longline/ set line, and the groundfish trawl (*in* Ferrero, *et al.*, 2000). Based on observer data, strandings, self reports, and permit reports, information on known incidental mortality of the eastern stock of Steller sea lions from 1990-1998 include the following: one animal was observed killed (7 estimated for the year) in the northern Washington marine set gillnet fishery, five Stellers were observed killed in the southeastern Alaska salmon drift gillnet fishery, one Steller sea lion was observed killed in the Alaska salmon troll fishery, and 84 animals were killed due to the British Columbia aquaculture predator control program (1991-1997). The minimum annual average of incidental mortality due to all of these fisheries combined was approximately 14 Steller sea lions per year (Hill and DeMaster, 1999).

In addition to the incidental take by commercial fisheries, Steller sea lions occasionally are shot illegally (approximately 3 per year), entangled in marine debris, and from 1992-96, there was a subsistence harvest by Alaska natives (approximately 2 per year). Because the stock has been declining in the southern end of its range (California), there has been concern regarding reduced prey availability, contaminants and disease (*in* Ferrero, *et al*., 2000).

In addition to anthropogenic threats to Steller sea lions, there may also be several factors which affect the population parameters in California and which may help to explain the declining trends at central California rookeries. First, a general warming trend of the Pacific Ocean may have reduced prey availability by affecting the characteristics of the California Current food web. Secondly, the expanded California sea lion (*Zalophus californianus*) population may be competing with Stellers for prey. Third, evidence exists that possible synergistic interactions between contaminants and disease in Stellers may be influencing the population (e.g premature births accounted for 20-60% of pup mortality in the South Farallon Islands between 1973-83, and organochlorine and trace metal contaminant levels are still elevated in central California Stellers). Lastly, unpredictable variability in demographic characteristics such as low birth rates, etc., may influence the population (*in* Sydeman and Allen, 1999).

2. Status of Marine Mammal Species Summary

Most large whale stocks, including the baleen whales and sperm whales, were severely depleted by modern whaling, and despite moratoriums on hunting by the International Whaling Commission, human-related interactions continue to pose the largest threat to these species. Collisions with vessels, entanglement in fishing gear, habitat degradation, and disturbance from low frequency noise are the most obvious potential threats. Because many of these incidents may occur far offshore and thus are unreported, the overall anthropogenic impact to large whales is most likely underestimated. In addition, because cetaceans spend their entire life in the ocean, and often well underwater, stock abundance estimates and life history information are very difficult to obtain, especially when sightings of particular species are rare, like the fin whale. Although available survey data suggest that humpback whales, fin whales and sperm whales have remained steady or have increased in abundance, the trends are not statistically significant, or the sample size is too small to accurately assess the rate of increase. Nevertheless, despite these uncertainties, it is clear that fisheries and non-fisheries related impacts continue to pose a threat to the recovery of these large whales.

The factors that have caused the decline in the eastern stock of Steller sea lions are poorly known; however, concerns have been raised, particularly in Alaska, regarding reduced prey availability due to increased commercial fishing in critical foraging areas. Although Steller sea lion subpopulations are increasing, or at least steady, in rookeries north of California, the central California subpopulation is declining, especially when compared to population estimates made prior to the 1980s. As mentioned in the “threats to Steller sea lion” section, in addition to occasional takes by fisheries, or interactions with humans, the reasons for the decline in this subpopulation may more be due to environmental factors such as competition with other species and warming sea surface temperatures.

B. Status of Sea Turtles

All stocks/populations of sea turtles incidentally taken in the CA/OR drift gillnet fishery are in decline. Impacts to sea turtles in the Pacific Ocean are primarily due to the composite effect of human activities which include: the legal harvest 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). 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 it is generally accepted by turtle biologists and others that these factors are the primary cause of turtle population declines, in many cases there is a paucity of quantitative data on the magnitude of human-caused mortality. In addition to anthropogenic factors, natural threats to the nesting beaches and pelagic-phase turtles 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-d) and are reviewed extensively in Eckert (1993).

a. *Green Turtle*

Green turtles are listed as threatened, except for breeding populations found in Florida and the Pacific coast of Mexico, which are listed as endangered. The genus *Chelonia* is generally regarded as comprising two distinct subspecies, the eastern Pacific (so-called "black turtle," *C. m. agassizii*), which ranges from Baja California south to Peru and west to the Galapagos Islands, and the nominate *C. m. mydas* in the rest of the range. Since both subspecies can be found in the eastern Pacific, and are generally referred to as green or black turtles, for the purposes of this Opinion, NMFS will treat them as one species.

Green turtles are distinguished from other sea turtles by their smooth carapace with four pairs of lateral scutes, a single pair of prefrontal scutes, and a lower jaw-edge that is coarsely serrated. Adult green turtles have a light to dark brown carapace, sometimes shaded with olive, and can exceed one meter in carapace length and 100 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. In the rookeries of Michoacán, Mexico females averaged 82 cm in CCL, while males averaged 77 cm CCL (*in* NMFS and USFWS, 1998a). Based on growth rates observed in wild green turtles, skeletochronological studies, and capture-recapture studies, all in Hawaii, it is estimated that green sea turtles attain sexual maturity at an average age of at least 25 years (*in* Eckert, 1993).

Green turtles are declining virtually throughout the Pacific Ocean, with the possible exception of Hawaii, as a direct consequence of an historical combination of overexploitation and habitat loss (Eckert, 1993). They are a circumglobal and highly migratory species, nesting mainly in tropical and subtropical regions. In Hawaii, green turtles lay up to six clutches of eggs per year (mean of 1.8), and clutches consist of about 100 eggs each. Females migrate to breed only once every two or possibly many more years, although the common remigration intervals reported for several rookeries worldwide are two and three years (Eckert, 1993; NMFS and USFWS, 1998a).

Green turtles prefer waters that usually remain about 20°C in the coldest month; for example, during warm spells (e.g. El Niño), green turtles may be found considerably north of their normal distribution. Based on the behavior of post-hatchlings and juveniles 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, 1996), 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, 1996). 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). Although most green turtles appear to have a nearly exclusive herbivorous diet, consisting primarily of sea grass and algae (Wetherall *et al*., 1993), those along 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). The non-breeding range of green turtles is generally tropical, and can extend approximately 500-800 miles from shore in certain regions (Eckert, 1993).

In the western Pacific, the only major (> 2,000 nesting females) populations of green turtles occur in Australia and Malaysia. Smaller colonies occur in the insular Pacific islands of Polynesia, Micronesia, and Melanesia (Wetherall *et al*., 1993) and on six small sand islands at French Frigate Shoals, a long atoll situated in the middle of the Hawaiian Archipelago (Balazs, 1995).

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. More than 165,000 turtles were harvested from 1965 to 1977 in the Mexican Pacific. In the early 1970s nearly 100,000 eggs per night were collected from these nesting beaches (*in* NMFS and USFWS, 1998a). The nesting population at the two main nesting beaches in Michoacán (Colola, responsible for 70% of total green turtle nesting in Michoacán (Delgado and Alverado, 1999) and Maruata) decreased from 5,585 females in 1982 to 940 in 1984. 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). Although the poaching of adult green turtles is now nearly negligible, the black market for sea turtle eggs in Mexico has remained as brisk as before the ban (Delgado and Alvarado, 1999). 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. During the 1998-99 season, based on a comparison of nest counts and egg collection data, an estimated 600 greens nested at Colola. Although only about 5% of the nests were poached at Colola during this season, approximately 50% of the nests at Maruata were poached, primarily because of difficulties in providing protections as a result of political infighting (Delgado and Alvarado, 1999).

There are no 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).

The nesting population of green turtles in Hawaii appears to have increased over the last 17 years. However, this encouraging trend is tempered by poaching and incidental capture in nearshore gillnets and longline gear. Also, the green turtle population in this area is afflicted with a tumor disease, fibropapilloma, which is of an unknown etiology and usually fatal. Ninety percent of nesting in Hawaii occurs at the French Frigate Shoals, where 200-700 females are estimated to nest annually (NMFS and USFWS, 1998a).

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 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, *in* NMFS and USFWS, 1998a) 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 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, based on morphology and preliminary genetic analysis (*in* NMFS and USFWS, 1998a). California stranding reports from 1990-99 indicate that the green turtle is the second most commonly found stranded sea turtle (48 total, averaging 4.8 annually) (J. Cordaro, NMFS, personal communication, April, 2000).

Green turtles encountered during drift gillnet fishing off California and Oregon may originate from a number of known proximal, or even distant, breeding colonies in the region. However the most likely candidates would include those from Hawaii (French Frigate Shoals) and the Pacific coast of Mexico population. This is based on limited genetic sampling conducted within the NMFS observer program for the CA/OR drift gillnet fishery (1 turtle genetically analyzed was found to originate from eastern

Pacific stock - most likely Mexican nesting beach) (P. Dutton, NMFS, personal communication, January, 2000).

b. *Leatherback Turtle*

The leatherback turtle is listed as endangered 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. The most recent estimate of the world population of leatherbacks is currently only 25,000 to 42,000 turtles (Spotila *et al*., 1996).

Leatherbacks 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, 1998b). In view of its unusual ecology, the leatherback is not surprisingly 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. Adult females nesting in Michoacán, Mexico averaged 145 cm CCL (L. Sarti, Universidad Naçional Autonoma de Mexico, unpublished data, *in* NMFS and USFWS, 1998b), while adult female leatherbacks nesting in eastern Australia averaged 162 cm CCL (Limpus, *et al*., 1984, *in* NMFS and USFWS, 1998b).

Leatherbacks have the most extensive range of any living reptile and have been reported circumglobally from 71°N to 42°S latitude in the Pacific and in all other major oceans (NMFS and USFWS, 1998b). For this reason, however, studies of their abundance, life history and ecology, and pelagic distribution are difficult. Similar to the olive ridley turtle, they 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. They are highly migratory, exploiting convergence zones and upwelling areas in the open ocean, along continental margins, and in archipelagic waters.

Recent satellite telemetry studies indicate that adult leatherbacks follow bathymetric contours over their long pelagic migrations and typically feed on cnidarians (jellyfish and siphonophores) and tunicates, and their commensals, parasites and prey (NMFS and USFWS, 1998b). 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). Surface feeding has been reported in U.S. waters, especially off the west coast (Eisenberg and Frazier, 1983), but foraging may also occur at depth. Based on offshore studies of diving by adult females nesting on St. Croix, U.S. Virgin Islands, Eckert *et al*. (1989) proposed that observed internesting[4] dive behavior reflected nocturnal feeding within the deep

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

scattering layer (strata comprised primarily of vertically migrating zooplankton, chiefly siphonophore and salp colonies, as well as medusae). Hartog (1980, *in* NMFS and USFWS, 1998b) also speculated that foraging may occur at depth, when nematocysts from deep water siphonophores were found in leatherback stomach samples.

Leatherbacks 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). Maximum dive depths for post-nesting females 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 leatherbacks was 37.4 minutes, while routine dives ranged from 4-14.5 minutes (*in* Lutcavage and Lutz, 1997). A total of six adult female leatherbacks 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). Migrating leatherbacks 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). 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 speculates that the short duration dives most likely represent surfacing activity after each dives. Excluding these short dives, five of the turtles preferred dive durations greater than 24 minutes, while three others preferred dives durations between 12-16 minutes.

On the Pacific coast of Mexico, female leatherback turtles lay 1-11 clutches per year (mean=5.7), with clutch size averaging 64 yolked eggs (each clutch contains a complement of yolkless eggs, sometimes comprising as much as 50 percent of total clutch size, a unique phenomenon among leatherbacks and some hawksbills (Hirth and Ogren, 1987)). 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 three years. Spotila *et al*. (2000), found the mean re-nesting interval of females on Playa Grande, Costa Rica to be 3.7 years. Using skeletochronological analysis of a small sample size of leatherback sclerotic ossicles, Zug and Parham (1996) suggested that mean age at sexual maturity for leatherbacks is around 13 to 14 years, giving them the highest juvenile growth rate of all sea turtle species, although this data is speculative (*in* Chaloupka and Musick, 1997). Zug and Parham (1996) concluded that for conservation and management purposes, 9 years is a likely minimum age for maturity of leatherbacks, based on the youngest adult in their sample. The natural longevity of leatherback turtles have not been determined (NMFS and USFWS, 1998b).

Migratory routes of leatherbacks originating from eastern and western Pacific nesting beaches are not entirely known. However, satellite tracking of post-nesting females and genetic analyses of leatherbacks caught in U.S. Pacific fisheries or stranded on the west coast of the U.S. present some strong insight into at least a portion of their routes and the importance of particular foraging areas. Current data from genetic research suggest that Pacific leatherback stock structure (natal origins) may vary by region. Because leatherbacks are highly migratory and stocks mix in high seas foraging areas, leatherbacks inhabiting the west coast of California are likely comprised of individuals originating from nesting assemblages located south of the equator in Indonesia and in the eastern Pacific along the Americas (e.g., Mexico, Costa Rica).

For nesting females from Mexiquillo Beach, Mexico, the eastern Pacific region has been shown to be a critical migratory route for female leatherbacks. 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). Morreale *et al*. (1996, in Eckert, 1997) demonstrated that satellite tagged, post-nesting leatherbacks leaving Costa Rica moved south after nesting. These studies underscore the importance of this offshore habitat 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 leatherbacks 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). These eastern Pacific nesting stocks may also move northwest, as genetic samples from two leatherbacks caught by the Hawaiian longline fishery indicated representation from eastern Pacific nesting beaches (Dutton *et al*., in press, and unpublished). NMFS and USFWS (1998b) and Eckert (1999a) speculate that the high density of leatherback sightings in and around Monterey, peaking in August (Starbird, *et al*., 1993), and the October to January nesting period on the Pacific coast of Mexico suggests that the turtles may migrate southward along the U.S. coastline to Mexican nesting beaches. However, genetic analyses of leatherbacks that have stranded and been taken in fisheries off Oregon and California have indicated representation from the western Pacific nesting beaches (P. Dutton, *et al*., in press, and P. Dutton, NMFS, personal communication, May, 2000).

Migratory corridors of leatherbacks originating from western Pacific nesting beaches most likely exist along the eastern seaboards of Australia, Asia and the former Soviet Union (NMFS and USFWS, 1998b). Genetic markers in 12 of 14 leatherbacks sampled to date from the central North Pacific (captured in the Hawaii longline fishery) have identified those turtles as originating from nesting populations in the southwestern Pacific; the other 2 specimens, taken in the southern range of the Hawaii fishery, were from nesting beaches in the eastern Pacific (P. Dutton, *et al*., in press, and P. Dutton, NMFS, personal communication, May, 2000). Stranding records from 1990-99 indicate that the leatherback is the most commonly stranded sea turtle off California (50 total, averaging 5 annually, J. Cordaro, NMFS, personal communication, April, 2000). In the U.S., leatherbacks have been sighted and reported stranded as far north as Alaska (60°N) and as far south as San Diego, California (NMFS and USFWS, 1998b). Of the stranded leatherbacks that have been sampled to date, all have been of western Pacific nesting stock origin (Dutton *et al*., in press). Genetic analysis of samples from two leatherback turtles taken off California and Oregon by the CA/OR drift gillnet fishery revealed that they both originated from western Pacific nesting beaches (i.e. Indonesia/Solomon Islands/Malaysia) (P. Dutton, NMFS, personal communication, March, 2000). Lastly, two leatherbacks were recently captured and tagged in Monterey Bay on September 7-8,2000 and fitted with transmitters. As of 9/21/00, both are on a southwest migratory path, most likely headed to the western Pacific nesting beaches (SWFSC, personal communication, September 2000). One of these individuals 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).

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, 1998b; 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 Irian Jaya and the Solomon Islands, systematic nesting surveys are just beginning or have been ongoing for several years. 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 (Eckert, 1997), specifically the advent of the high seas driftnet fishery in the 1980s (Sarti *et al*., 1996).

Eastern Pacific nesting populations of leatherbacks

Leatherback nesting populations are declining along the Pacific coast of Mexico and Costa Rica (Appendix B, Table 1). At Las Baulas National Park, Costa Rica, the number of nesting leatherbacks has declined from 1,500 in 1988-1989 to 193 in 1993-1994 (Steyermark *et al.*, 1996). Leatherbacks have been studied at Playa Grande (in Las Baulas), the fourth largest leatherback nesting colony in the world, since 1988. During the 1988-89 season (July-June), 1,367 leatherbacks nested on this beach, and by the 1998-99 season, only 117 leatherbacks nested. Furthermore, during the last three nesting seasons (1996 through 1999), an average of only 25% of the turtles were remigrants (turtles returning to nest that were observed nesting in previous nesting seasons). Less than 20% of the turtles tagged in 1993 through 1995 returned to nest in the next five years (Spotila, *et al*., 2000). Remigration intervals for leatherbacks at nesting beaches in South Africa and the U.S. Caribbean have been documented as over 91% returning within 5 years or less (Hughes, 1996; Boulon, *et al*. 1996 *in* Spotila, *et al*., 2000). Comparatively few leatherbacks are returning to nest on east Pacific nesting beaches and it is likely that leatherbacks 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). If current estimates of age to maturity are correct, the effects of such efforts should be observed beginning sometime this decade (Spotila and Steyermark, 1998), barring any increase in current levels of juvenile and adult mortality.

The decline of leatherback subpopulations is even more dramatic off Mexico. According to reports from the late 1970s and early 1980s, three beaches located on the Pacific coast of Mexico sustained a large portion of all global nesting of leatherbacks, perhaps as much as one-half. Since the early 1980s, the eastern Pacific Mexican population of adult female leatherbacks has declined from 70,000[5] in 1982 (Spotila *et al*., 1996) to less than 1,000 in 1999-2000 (Sarti *et al*., personal communication, 2000). Monitoring of the nesting assemblage at Mexiquillo, Mexico has been continuous since 1983-84. According to Sarti *et al.* (1996), nesting declined at this location at an annual rate of over 22 percent for the last 12 years. 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."

Although the causes of the decline in the nesting populations are not entirely clear, Sarti *et al*. (1998) surmises that the decline could be a result of intensive egg poaching in the nesting areas, incidental capture of adults or juveniles in high seas fisheries, and natural fluctuations due to changing environmental conditions. Leatherbacks are not captured for meat or their skin in Mexico, but the eggs are highly desirable. 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 leatherbacks. Eckert (1997) speculates that the swordfish gillnet fisheries in Peru and Chile have contributed to the decline of the leatherback in the eastern Pacific. The decline in the nesting population at Mexiquillo, Mexico occurred at the same time that effort doubled in the Chilean driftnet fishery.

Most conservation programs aimed at protecting nesting sea turtles in Mexico have continued since the early 1980s, and there is little information on the degree of poaching prior to the establishment of these programs. However, Sarti *et al.* (1998) estimates that as much as 100% of the clutches were taken from the Mexican beaches. Since protective measures have been in place, particularly emergency measures recommended by a joint U.S./Mexico leatherback working group meeting in 1999, there has been greater nest protection and nest success (Table 1). Mexican marines were present during the 1999-2000 season at three of the primary nesting beaches in Mexico (Llano Grande, Mexiquillo, and

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[5]This estimate of 70,000 adult female leatherbacks 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, 1982) have possibly been used as baseline data for subsequent estimates to a greater degree than the quality of the data would justify."

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. Currently, the primary management objective is 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 the next three years to help implement these objectives (minutes from joint U.S./Mexico Leatherback Working Group meeting, 23-24 May, 2000).

Table 1. Nest protection at index beaches on the Pacific coast of Mexico (Source: Sarti *et al*., personal communication, 2000)

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%

Spotila *et al.* (2000) have estimated that there are currently 687 adult females and 518 subadults comprising the entire eastern Pacific Central American population of leatherbacks. With an estimated Mexican population of 1,000 adults and 750 subadults, the entire east Pacific leatherback population has been estimated by Spotila *et al.* 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).

Western Pacific Populations of Leatherback Turtles

Similar to their eastern Pacific counterparts, leatherbacks 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 leatherbacks occurs in Fiji, Thailand, Australia, and Papua New Guinea. In Indonesia, 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, 1996). The largest extant leatherback rookery in the Indo-Pacific lies on the north Vogelkop coast of Irian Jaya, with over 1,000 females nesting during the 1996 season (Suarez *et al.*, in press) (see Table 3).

As with the eastern Pacific nesting populations, the decline of leatherbacks is severe at one of the most significant nesting sites in the 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 2). With one or two females reportedly nesting each year, this population has essentially been eradicated (P. Dutton Declaration, June 9, 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 leatherbacks. There were two periods in which there were sharp declines in nesting leatherbacks 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, are presumed to have impacted the Malaysian leatherback population. After 1980, rates of decline averaged 16% annually, suggesting continuing threats from fisheries (Chan and Liew, 1996).

Table 2. 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

Similarly, the nesting populations of leatherbacks in Irian Jaya, Indonesia are reported to be declining. Leatherback nesting generally takes place on two major beaches on the north Vogelkop coast of Irian Jaya, Jamursba-Medi and War-Mon beach. As shown in Table 3, Suarez, *et al*. (in press) has compiled, re-analyzed, and standardized data collected from leatherback nesting surveys in the 1980s and 1990s. In addition, Suarez *et al*. (in press) has included 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[6] 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, 1996). 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. Approximately

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

90% of those nests not taken by poachers or the sea[7] were destroyed by pigs (Suarez *et al.* in press). War-Mon beach supports a lower percentage of nesting females, yet egg poaching for subsistence accounted for over 60% of total nest loss during 1993-94, and 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).

Table 3. **Estimated numbers of female leatherbacks nesting along the north coast of Irian Jaya (Summarized by Suarez, *et al*., in press.)**

Survey Period	# of Nests	Adjusted # Nests	Estimated # of Females
Jamursba-Medi Beach:			
September, 1981	4,000+	7,173[1]	1,232-1,623
April - Oct. 1984	13,360	13,360	2,303-3036
April - Oct. 1985	3,000	3,000	[(658)-731]
June - Sept. 1993	3,247	4,329[2]	746-984
June - Sept. 1994	3,298	4,397[2]	758-999
June - Sept. 1995	3,382	4,509[2]	777-1025
June - Sept., 1996	5,058	6,744[2]	1,163-1,533
War-Mon Beach:			
Nov. 1984 - Jan. 1985	1,012	N/A	175-230
Dec. 1993	406	653	128 - 169

[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 25% 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 leatherbacks on Jamursba-Medi in 1985 was 4.4 nests per female. This is consistent with estimates for the average number of nests by leatherbacks 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.

In the Kai Islands, located approximately 1,000 kilometers southwest of the Irian Jaya nesting beaches, adult leatherbacks are traditionally hunted and captured at sea by local people. Villagers hunt leatherbacks only for ritual and subsistence purposes, and, according to their beliefs (known as *adat*),

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

they are forbidden to sell or trade the meat. Based on a study conducted during October-November, 1994, Suarez and Starbird (1996) estimated that approximately 87 leatherbacks 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 leatherbacks are killed per year in traditional fisheries in Maluku, Indonesia (*in* Chan and Liew, 1996) (the Kai Islands take is assumed included in this estimate), and this take level is most likely continuing (C. Starbird, personal communication, 1998, *in* Clever Magazine, Issue No. 6).

As shown in Table 3, since the early-to-mid 1980s, the number of female leatherbacks nesting annually on the two primary beaches of Irian Jaya appear to be stable. However, given the current, serious threats to all life stages of the Indonesian leatherback populations, this trend may not be sustained and this population could collapse, similar to what occurred in Terrengganu, Malaysia. 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 leatherbacks or incidental take by local fisheries. There is no evidence to indicate that the preceding threats 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). In addition, local Indonesian villagers report dramatic declines in local sea turtle populations (Suarez, 1999); without adequate protection of nesting beaches, emerging hatchlings, and adults, this population will continue to decline.

Regarding the status of the Irian Jaya population of nesting leatherbacks, Suarez *et al*. (in press) comment: "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 leatherbacks from declining in the future."

Conclusion on status of eastern and western Pacific leatherbacks
Although quantitative data on human-caused mortality are scarce available information suggests that leatherback mortality on many nesting beaches remains at unsustainable levels (Tillman, 2000). In addition, 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, risks of mortality in fisheries generally have not been reduced.

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 Irian Jaya, 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, incidental take in fisheries, and a fractured demographic structure. 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 effective population as older individuals die or are removed.

c. *Loggerhead Turtle*

The loggerhead turtle is listed as threatened under the ESA throughout its range, primarily due to exploitation, incidental capture in various fisheries, and the alteration and destruction of its habitat. The loggerhead is categorized as Endangered, by the International Union for Conservation of Nature and Natural Resources where taxa so classified are considered to be facing a very high risk of extinction in the wild in the near future (IUCN Red List, 2000 - not sure of the exact citation but will get it). Loggerheads are a cosmopolitan species, found in temperate and subtropical waters and inhabiting pelagic waters, continental shelves, bays, estuaries and lagoons . In the Pacific Ocean, major nesting grounds are generally located in temperate and subtropical regions, with scattered nesting in the tropics (*in* NMFS and USFWS, 1998c).

The loggerhead is characterized by a reddish brown, bony carapace, with a comparatively large head, up to 25 cm wide in some adults. 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).

Nesting of loggerheads in the Pacific Basin are restricted to the western and southern region (Japan and Australia, primarily); there are no reported loggerhead nesting sites in the eastern or central Pacific. Upon reaching maturity, adult females migrate long distances from resident foraging grounds to their preferred nesting beaches. The average re-migration interval is between 2.6 and 3.5 years, in Queensland, Australia (*in* NMFS and USFWS, 1998c). Nesting is preceded by offshore courting, and individuals return faithfully to the same nesting area over many years. Clutch size averages 110 to 130 eggs, and one to six clutches of eggs are deposited during the nesting season (Dodd, 1988). 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 stock (*in* Chaloupka and Musick, 1997), although Frazer *et al*. (1994 *in* NMFS and USFWS, 1998c) determined that maturity of loggerheads in Australia occurs between 34.3 and 37.4 years of age.

The transition from hatchling to young juvenile may occur in the open sea and evidence is accumulating that this part of the loggerhead life cycle may involve trans-Pacific movement. Although the distribution of loggerheads in foraging areas is not well known for any population, it has been suggested that juvenile Pacific loggerheads follow a migration similar to loggerheads in the Atlantic. Hatchlings from the southeastern United States enter driftlines composed of *Sargassum* and other flotsam and are passively transported by currents in the North Atlantic gyre, perhaps one or more times, before taking up residence in developmental habitats in coastal waters of the eastern seaboard (Carr, 1987, *in* NMFS and USFWS, 1998c). 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 of mainly juveniles and subadult loggerheads, numbering in the thousands, are found off the southwestern coast of Baja California, over 10,000 km from the nearest significant nesting beaches (Pitman, 1990). Genetic studies have shown these animals originate from Japanese nesting stock (Bowen *et al*., 1995), and their presence reflects a migration pattern probably related to their feeding habits (Cruz, *et al*., 1991, *in* Eckert, 1993). These loggerheads are primarily juveniles, although carapace length measurements indicate that some of them are 10 years old or older.

Recent satellite telemetry data from pelagic juvenile loggerheads tagged after being captured in the Hawaiian longline fishery indicate movements westward against prevailing currents and along the southern margin of the North Pacific Transition Zone, associating strongly with oceanic fronts in the subtropical frontal zone (Polovina *et al*., (in press)). Genetic analyses of 24 loggerheads caught in the Hawaiian longline fishery indicated that the majority (95 percent) originated from Japanese nesting stock (Dutton, *et al*., 1998). Loggerheads are not commonly documented in U.S. Pacific waters. Stranding data from 1990-99 for California indicate that an average of 2.1 loggerheads strand per year (21 total in ten years) (J. Cordaro, NMFS, personal communication, April, 2000). Genetic analyses on four loggerheads taken in the CA/OR drift gillnet fishery indicate they originated on Japanese nesting beaches (P. Dutton, NMFS, personal communication, March, 2000). Loggerhead occurrence in the CA/OR drift gillnet fishery is probably associated with the northward extension of Transition Zone waters along the North American coast during El Niño years.

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), and preliminary data from stomach samples collected from turtles captured in North Pacific driftnets indicate a diet of gooseneck barnacles (*Lepas* sp.), pelagic purple snails (*Ianthina* sp.), and medusae (*Vellela* sp.) (G. Balazs, personal communication, *in* NMFS and USFWS, 1998c). As they age, 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). Most subadults and adults are found in nearshore benthic habitats around southern Japan, in the East China Sea and the South China Sea (e.g. Philippines, Taiwan, and Viet Nam).

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). Loggerheads appear to spend a longer portion of their dive time on the bottom (or suspended at depth), which may be related to foraging and refuge. Unlike the leatherback, to the loggerhead foraging in the benthos, bottom time may be more important than absolute depth (Eckert, *et al*., 1989). 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).

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 may nest annually in all of Japan; however, more recent data suggest that only approximately 1,000 female loggerhead turtles may nest there (Bolten *et al*. 1996). Nesting beach monitoring at Gamoda (Tokushima Prefecture) has been ongoing since 1954. Surveys at this site showed a marked decline in the number of nests between 1960 and the mid-1970s. Since then, the number of nests has fluctuated, but has been downward since 1985 (Bolten *et al*., 1996). Monitoring on several other nesting beaches, surveyed since the mid-1970s, revealed increased nesting during the 1980s before declining during the early 1990s.

Quantitative data on nesting levels since 1995 are unavailable, but are reported to show a continuing decline (Tillman, 2000). Nesting of loggerheads may also occur along the south China Sea, but it is a rare occurrence (Marquez, 1990, *in* Eckert, 1993).

In the south Pacific, Limpus (1982) reported an estimated 3,000 loggerheads nesting annually in Queensland, Australia. Long-term trend data from Queensland indicate a decline in nesting which is corroborated by studies of breeding females at adjacent feeding grounds (Limpus and Reimer, 1994). By 1997, the number of females nesting annually in Queensland was thought to be as low as 300 (1998 Draft Recovery Plan for Marine Turtles in Australia). Survey data are not available for other nesting assemblages in the south Pacific. Scattered nesting has also been reported on Papua New Guinea, New Zealand, Indonesia, and New Caledonia; however, population sizes on these islands have not been ascertained (NMFS and USFWS, 1998c).

As mentioned, aggregations of juvenile loggerheads off Baja California Mexico have been reported, although their status with regard to increasing or declining abundance has not been determined. NMFS and USFWS (1998c) report "foraging populations ... range from 'thousands, if not tens of thousands' (Pitman, 1990) to 'at least 300,000 turtles' (Bartlett 1989). Extrapolating from 1988 offshore census data, Ramirez-Cruz et al. (1991) estimated approximately 4,000 turtles in March, with a maximum in July of nearly 10,000 turtles."

Loggerhead mortality from human activities is not well-documented, except for estimates based on NMFS observer data in the Hawaii longline fishery and the CA/OR drift gillnet fishery. A high mortality in the North Pacific high-seas driftnet fisheries of Japan, Republic of Korea, and Taiwan was estimated in the 1990s, but those fisheries no longer operate. 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).

d. *Olive Ridley Turtle*

The olive ridley populations on the Pacific coast of Mexico are listed as endangered under the ESA; all other populations are listed as threatened. The olive ridley is categorized as endangered, by the International Union for Conservation of Nature and Natural Resources where taxa so classified are considered to be facing a very high risk of extinction in the wild in the near future (IUCN Red List, 2000 - not sure of the exact citation but will get it). They 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, 1998d).

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. Surprisingly little is known of their oceanic distribution and critical foraging areas, despite being the most populous of north Pacific sea turtles. The species appears 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, *in* Eckert, 1993) 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. Thus, it is possible that young turtles 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, similar to the juvenile loggerheads mentioned previously. Olive ridleys feed on tunicates, salps, crustaceans, other invertebrates and small fish. Although they are generally thought to be surface feeders, olive ridleys have been caught in trawls at depths of 80-110 meters (NMFS and USFWS, 1998d), 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).

Olive ridley turtles are the most abundant sea turtle in the Pacific basin. 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. 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, 1998d). Females generally lay two clutches of eggs per season in Costa Rica (Eckert, 1993). Data on the remigration intervals of olive ridleys are scarce.

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. Where population densities are high enough, nesting takes place in synchronized aggregations known as *arribadas*. The largest known *arribadas* in the eastern Pacific are off the coast of Costa Rica (~475,000 - 650,000 females estimated nesting annually) and in southern Mexico (~600,000+ nests/year (Eckert, 1993; NMFS and USFWS, 1998d; Salazar *et al.*, in press). Historically, it was estimated that over 10 million olive ridleys inhabited the waters in the eastern Pacific off Mexico. However, human-induced mortality led to declines in this population. Beginning in the 1960's an enormous number of adult olive ridleys were harvested for commercial trade with Europe and Japan, several million olive ridleys were landed during the period 1960-1975. (NMFS and USFWS, 1998d). 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). 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). 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.

In the western Pacific, 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).

While olive ridleys generally have a tropical to subtropical range, individuals do occasionally venture north, some as far as the Gulf of Alaska. 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). Stranding records from 1990-99 indicate that olive ridleys are rarely found off the coast of California, averaging 1.3 strandings annually (J. Cordaro, NMFS, personal communication, April, 2000).

Recent genetic information analyzed from 15 olive ridleys taken in the Hawaii-based longline fishery indicate that 9 of the turtles originated from the eastern Pacific and 6 of the turtles were from the southwest or Indo-Pacific (i.e. Malaysia) (P. Dutton, NMFS, personal communication, 1999). 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, January, 2000).

2. 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 fisheries and non-fisheries-related threats to all sea turtles in the Pacific Ocean. A description of the factors affecting each marine mammal species (i.e. fin, humpback, sperm whale and Steller sea lion) was summarized individually by species, in section IIIA.

a. *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. Such estimates are hampered by a lack of data on pelagic distribution of sea turtles, especially leatherbacks and olive ridleys, that spend the majority of their non-breeding life history in the open ocean.

This section will summarize known fisheries that have been observed or reported to incidentally or intentionally take sea turtles. 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.

(i). North Pacific Driftnet Fisheries (pre-12/92)

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 cessation of high-seas driftnet fishing should have reduced the incidental take of listed species. However, nations involved in driftnet fishing may have shifted to longline fishing worldwide, or to coastal gillnet operations within their respective EEZs, which may increase (e.g. more coastal driftnets may impact post-pelagic stage loggerheads), maintain, or decrease (i.e. longlines, in general) the take of sea turtles. Tables 1 and 2 in Appendix A provide a summary of the number of active Japanese, Korean, and Taiwanese vessels fishing mainly for tuna in the Central Western Pacific Ocean from 1990-99.

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 4,373 turtles, mostly loggerheads and leatherbacks, were entangled by the combined fleets of Japan, Korea and Taiwan during June, 1990 through May, 1991, when all fleets were monitored (Table 4). Of these incidental entanglements, an estimated 1,011 turtles were killed (77 percent survival rate).

Table 4. 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
(hawksbill)	7	2
TOTAL	4,373	1,011

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 leatherbacks were immature, although the size of leatherbacks 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 impacts of the driftnet fishery on sea turtles, and a full assessment of impacts would consider the turtle mortality generated by the driftnet fleets over their entire range. Unfortunately, comprehensive data are lacking, but the observer data does indicate the possible magnitude of turtle mortality given the best information available. Wetherall *et al*. (1993) speculate that the actual mortality of sea turtles may have been between 2,500 and 9,000 per year, with most of the mortalities being loggerheads taken in the Japanese and Taiwanese large-mesh fisheries.

While a comprehensive, quantitative assessment of the 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.

(ii). Japanese tuna longliners in the Western Pacific Ocean and South China Sea

Nishimura and Nakahigashi (1990) estimated that 21,200 turtles, including greens, leatherbacks, 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. These estimates were based on turtle sightings and capture rates reported in a survey of fisheries research and training vessels and extrapolated to total longline fleet effort. Using commercial logbooks, research-vessel data and questionnaires from longliners from 1988, Nishimura estimated that for every 10,000 hooks in the north 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, hawksbill - 10.3 percent, olive ridley - 1.7 percent, leatherback - 13.7 percent, and unknown - 19 percent. Because the data collected by Nishimura and Nakahigashi were based on observations by training and research vessels, logbooks and a questionnaire, and assumed that turtles were distributed homogeneously, such estimates may be biased. NMFS is unaware of any follow-up studies since 1990 (J.Wetherall, NMFS, personal communication, 1999).

As shown in Appendix A, Table 1, the number of active coastal and distant water Japanese tuna longliners has remained relatively constant from 1990-99 (averaging 740 and 660 vessels, respectively, per year), while the number of active offshore tuna longliners in 1997-99 have declined by nearly one half (from approximately 360 vessels to 180 vessels) since 1990[8]. Two other countries comprising a large amount of longline effort in the central western Pacific include Korea and Taiwan. Taiwan's offshore fleet is particularly large, composed of an average of 1,500 active vessels per year (based on data from 1990-99), while the number of vessels included in its distant water fleet ranged from 52 to 88 vessels over the past ten years. The number of vessels included in Korea's longline fleet has remained relatively constant from 1990-99, averaging 168 active vessels per year (Appendix A, Table 1). Although the data and analysis 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. Unfortunately, current bycatch information is not available for these fisheries. Future investigations into the level of sea turtle bycatch in these fisheries would allow a more complete assessment of cumulative effects on pelagic sea turtles in the Pacific Ocean.

(iii). South American fisheries

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

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[8] In reference to the Japanese tuna longline fleet, "offshore" refers to vessels that fish outside Japan's EEZ but closer to Japan, while "distant water" refers to vessels which fish in other areas throughout the Pacific Ocean (A. Coan, NMFS, personal communication, August, 2000).

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 leatherbacks 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 leatherbacks 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.[9] As shown in Table 5, the take of sea turtles by the artisanal driftnet fishery in the late 1980s appeared to be comprised primarily of leatherbacks.

Table 5. 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

Effort by the artisanal driftnet fishery for swordfish appears to be relatively constant through 1996, as shown in Table 6. 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

[9]Based on all information from Chile and Peru, Eckert (1997) estimated that a minimum of 2,000 leatherbacks are killed annually by Peruvian and Chilean swordfish operations, representing a major source of mortality for leatherbacks 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.

Table 6). 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 of from 3-13 (*in* Weidner and Serrano, 1997). Assuming the current artisanal driftnet fishing effort is equivalent to 1996, this fishery would currently take an estimated 39 greens, 76 leatherbacks, 4 loggerheads, and 29 olive ridleys annually, assuming the proportion of species taken is equivalent to data collected from the 1988-89 fishing season.

Table 6. Chile - artisanal (driftnet) swordfish effort, by year, from 1989-1996

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 there has been no collection or collaboration 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) 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. 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 massive 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.

Columbia

A description of known Columbian 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 Columbia. 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.

Ecuador

Appendix A contains a description of known current commercial and artisanal fisheries in Ecuador. 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 7). 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).

Peru

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. 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).

Peru conducted directed commercial turtle harvests throughout the 1980s, and, as recently as 1990, over 100 metric tons of turtles were taken (Table 7) (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 leatherbacks 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). The Ministerio de Pesqueria (MIPE), which is the Peruvian agency responsible for fisheries, prohibited the taking of all leatherbacks and green turtles less or equal to 80cm in length through a resolution in January, 1977, although observers report that regulations are rarely enforced. Other species were not protected and were still unprotected as of 1989, although catches appear to have declined to negligible levels (Weidner and Serrano, 1997), although specific take levels remain unknown.

Table 7. 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.

(iv). Fisheries in the Federated States of Micronesia

Incidental capture of sea turtles was reported by observers aboard tuna purse seine and longline vessels licensed to fish in the EEZ of the Federated States of Micronesia for the years 1980-1993. Seven of the thirteen turtles reported taken by longliners were unidentified and released alive and unharmed. The remainder included 1 hawksbill, 2 leatherbacks, and 3 olive ridleys. Only one turtle, an olive ridley, was reported as killed; the rest were released alive and unharmed. A total of 38 out of a possible 54 observer trips were reviewed, and out of 280,110 hooks monitored, seven turtles were observed caught, giving a rough estimate of 0.025 turtles caught for every 1,000 hooks (or for every 40,000 hooks set, one sea turtle is incidentally caught). For purse seiners fishing in Micronesia, 7 sea turtles were reported incidentally captured - 2 hawksbills, 2 olive ridleys, and 3 recorded as unidentified. Of these, one olive ridley, one hawksbill and two unidentified turtles were released alive and unharmed, one hawksbill was reported as dead/discarded, one olive ridley was injured in the power block, and the condition of one unidentified turtle was unknown (Thoulag, 1993).

The number of active longline vessels in Micronesia has tripled, from 7 in 1993 to 21 in 1998 and 1999. In contrast, the number of Micronesian purse seine vessels has been reduced from 6-7 in the early 1990s to 3 in 1998 and 1999 (Appendix A, Table 1). Because such a small number of trips have been monitored in these fisheries, the estimated take of turtles, based on observer data, is small compared to the current longline and purse seine effort in Micronesia's EEZ. However, without knowing the percentage observer coverage, annual take estimates by these fisheries will only take into account the reported take of turtles from 1980-93; therefore, these are *minimum* estimates. Apportioning the unidentified turtles to observed known species taken, and using only the observed take of sea turtles, the estimated annual take of species included in this Opinion by longliners is: 0.3 leatherbacks taken (0 mortalities) and 0.5 olive ridleys taken (0.2 mortalities). The estimated annual take of sea turtles included in this Opinion by Micronesian purse seiners, based on observer data collected from 1980-93, is: 0.3 olive ridleys (0.2 mortalities).

(v). Western Pacific U.S. tuna purse seine fishery

Commercial fishing for tropical tunas in the western Pacific Ocean by U.S. registered purse seiners has been managed according to requirements of the South Pacific Regional Tuna Treaty since June, 1988. The treaty was signed by the United States and 16 Pacific island countries, 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. The agreement ends in 2003 (Coan, *et al*., 1997). In 1998, most of the U.S. fleet, which consisted of 39 vessels, fished between 165°W and 155°E longitude and between 10°N and 10°S latitude (Coan, *et al*., 1999). Because there is not the characteristic tuna-dolphin association in the western Pacific as there is in the ETP, U.S. fishermen set on floating objects (logs and fish aggregating devices) and schools to catch tuna. The U.S. fleet is required to take observers on a minimum of 20 percent of their fishing trips. In 1998, observers recorded one loggerhead turtle taken, although it is unclear as to whether the turtle was released unharmed, injured or killed (Coan, *et al*., 1999). From June, 1997 to June, 1998, observers anecdotally (recorded in their logbooks) observed one green turtle taken and released unharmed, and one unidentified turtle taken and released unharmed (Forum Fisheries Agency, 1998). Extrapolating this information based on percentage of observer coverage, the entire U.S. western Pacific fleet may capture 5 loggerheads, 5 greens, and 5 unidentified sea turtles each year, assumed to be either loggerheads or greens, based on observer data.

(vi). Hawaii-based longline fishery

The Hawaii-based longline fishery ranges over 2,000 nautical miles (nm) of latitude from waters well south of the Hawaiian Archipelago to waters north of the islands in the North Pacific Transition Zone (Wetherall, 1993). At present, there are 164 limited entry permits in this fishery. Limited quantitative data exist on the number of sea turtles caught by this fishery and the immediate or consequent injury and mortality that takes place. Information on the likelihood of fishery interactions with each species has been collected by scientific observers deployed by NMFS since February, 1994. Data from the NMFS observer program collected from 1994 through 1999 and associated longline logbook statistics were used to estimate the turtle take and mortality by species for each year. Table 8 shows the estimated total incidental takes and mortalities of sea turtles in the Hawaiian longline fishery from 1994-1999. Estimates of takes and mortalities for the years 1994-1997 were calculated differently than those computed and reported in 1998 and later (these estimates may underestimate the number of sea turtles killed in the fishery because some turtles that were lethargic when they were released, which were considered "alive" when they were released, probably died from their injuries subsequent to their release). The revised estimates (according to calculations conducted after 1997) are based on a larger accumulation of observer statistics and different prediction models. Further details of the analysis are described in SWFSC Administrative Report H-00-06 (in preparation). In addition, Table 8 indicates for each species the estimated probability that the annual take and mortality exceeded the "anticipated incidental take" or "anticipated incidental mortality," as specified in NMFS' most recent biological opinion (NMFS, 1998a).

Table 8. **Estimated turtle takes by species, 1994-1999, in the Hawaiian longline fishery**

Species	Anticipated incidental take	Estimated takes (and mortalities) by the Hawaii-based longline fleet and probabilities that the take exceeded the "anticipated incidental take."					
		1994	1995	1996	1997	1998	1999
Green	1994-1997 119(18)	37(5)	38(5)	40(5)	38(5)	42(5)	45(6)
	1998-1999 52(15)	.00(.01)	.00(.02)	.00(.02)	.00(.02)	.28(.06)	.34(.06)
Leatherback	1994-1997: 271(23)	109(9)	99(8)	106(9)	88(7)	139(12)	132(11)
	1998-1999 244(19)	.00(.03)	.00(.02)	.00(.02)	.00(.00)	.00(.23)	.00(.20)
Loggerhead	1994-1997: 305(46)	501(88)	412(72)	445(78)	371(65)	407(71)	369(64)
	1998-1999: 489(103)	.98(.95)	.90(.89)	.96(.92)	.82(.84)	.09(.06)	.01(.03)
Olive Ridley	1994-1997: 152(41)	107(36)	143(47)	153(51)	154(51)	157(52)	164(55)
	1998-1999: 168(46)	.04(.36)	.39(.63)	.53(.70)	.54(.70)	.38(.62)	.46(.66)

In its November 3, 1998, biological opinion on the impacts of the fishery management plan for the Hawaii-based longline fishery on listed species, NMFS estimated the maximum annual incidental takes and mortalities of sea turtles for 1998-2001: greens - 52 taken, 15 killed; leatherbacks - 244 taken, 19 killed; loggerheads - 489 taken, 103 killed; olive ridleys - 168 taken, 46 killed (NMFS, 1998a).

On June 7, 2000, NMFS reinitiated consultation on the Hawaii-based longline fishery because the anticipated incidental take of olive ridley sea turtles was exceeded. The forthcoming biological opinion will include the entire fishery governed by the fishery management plan for the western Pacific pelagic fishery. Currently the Hawaii-based longline fishery is operating under a court-ordered plan until NMFS completes a National Environmental Policy Act (NEPA) analysis of the fishery. The court order established four fishing areas with ranges of fishing effort from total closure to limited numbers of sets allowed and ranges in observer coverage requirements from 20 percent to 100 percent. Estimates of the expected level of take of listed species that may occur as a result of this order are not available, however it is expected that takes of most, if not all, species should decrease as a result of restricted fishing effort. NMFS is using the pre-court order take estimates to assess the maximum effects of the proposed action in context with all other factors affecting the species.

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

The number of large (>400 short tons (st) carrying capacity) ETP tuna purse seine vessels has remained steady since 1992, varying between 5 and 7 vessels, and the number of smaller (#400 st) vessels has also remained steady, averaging 18 vessels between 1993 and 1997 (NMFS, 1998b). Although all large tuna purse seine vessels fishing in the ETP for tuna have been required to carry observers since 1989 (100 percent coverage), smaller purse seine vessels are not required to carry observers. Thus, no data are available on sea turtle interactions with the small tuna purse seine vessels in the ETP. Most smaller tuna vessels fishing off southern California fish on tuna schools because the vessels are old, slow, and lack the resources (e.g. helicopters) needed to place and find floating objects (B. Jacobson, NMFS, personal communication, 1999). Based on observer data from the large vessels, the chances of incidentally capturing a sea turtle during a school set are much less than incidentally capturing a sea turtle during floating object sets. NMFS believes that the capture of sea turtles in the small vessel fleet is rare. In addition to collecting tuna life history and marine mammal and bycatch data during a set, observers on large U.S. purse seiners in the ETP complete a sea turtle life history form when a sea turtle is taken in a set (i.e., sea turtle was captured or at any time entangled in the net).

Table 9 shows sea turtle interactions in the large U.S. tuna purse seine fleet from 1990 to 1997. Data for 1998 and most of 1999 has not been entered into a database and is therefore currently unavailable. The 1990-1997 data include 174 turtles taken in the fishery that were not identified to species, although only 1 of these unidentified turtles is listed as accidentally killed (as discussed earlier, these estimates may underestimate the number of sea turtles killed in the fishery because some turtles that were lethargic when they were released, which were considered "alive" when they were released, probably died from their injuries subsequent to their release). Most of unidentified sea turtles probably never came on board, but escaped after being encircled or captured, and the observer was not close enough to identify the turtle as it swam away. Assuming that these unidentified turtle interactions occurred in the same proportions as the identified sea turtle interactions, these 174 turtles would most likely be comprised of 143 olive ridleys, 28 green turtles, and 1 to 3 leatherback, hawksbill or loggerhead turtles, in unknown proportion. It is likely that most of these 174 unidentified turtles were uninjured by their capture or encirclement if they did release themselves from the net and swim away.

Table 9. Sea turtle interactions by U.S. tuna purse seine fleet (1990 - 1997) - large vessels only [Note: there is some discrepancy between the numbers in the two parts of the table because previously dead turtles were not included in species estimates and hawksbill turtles were not included in the top part of the table and not accounted for it in the lower part]

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

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

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

(viii). Foreign tuna purse seine fishery in the ETP

The international fleet represents the majority of the fishing effort and carrying capacity in the ETP tuna fishery, with most of the total capacity consisting of purse seiners greater than 400 st. These large vessels comprised about 87 percent of the total fishing capacity operating in the ETP in 1996 (IATTC, 1998). An average of 107 foreign vessels with a carrying capacity greater than 400 st fished in the ETP during 1993 to 1997. In addition to these larger vessels, the foreign fleet contains smaller vessels less than 400 st that target tuna in the ETP. From 1993 to 1997, an average of 63 foreign vessels ranging from 45 to 400 st carrying capacity fished in the ETP each year.

Data from observers on both U.S. and foreign tuna purse seine vessels have been gathered collectively by the IATTC since the early 1990s (Table 10; data are in addition to Table 9). The most recent data from the IATTC indicate that an average of 172 sea turtles per year were killed by vessels over 400 st in the entire ETP purse seine fishery (U.S. included) from 1993-97 (IATTC, 1999).

Table 10. Estimated sea turtle mortality by species for the entire ETP tuna purse seine fishery (U.S. and foreign) from 1993-1997[1]

Species/Year	1993	1994	1995	1996	1997
Olive ridley	197	103	94	83	99
Loggerhead	5	10	2	3	7
Green/black	39	8	12	7	19
Leatherback	0	0	0	1	0
Unidentified	46	36	32	29	25
TOTAL	287	157	140	123	150

[1] (M. Hall, IATTC, personal communication, 1999)

The 1993-1997 data indicate that 168 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, these 168 turtles would be 140 olive ridleys, 20 green turtles, 7 loggerhead turtles and one would be either a leatherback or hawksbill.

(ix). Mexican (Baja California) fisheries and direct harvest

Based on a combination of analyses of stranding data, tag-recapture studies and extensive interviews, all carried out between 1994 and 1999, Nichols (personal communication, October 2000) has conservative estimates of the annual take of green turtles and loggerhead turtles by various fisheries and through direct harvest in the Baja California, Mexico region. Nichols and his affiliates estimated the annual mortality of green turtles in this region to be *greater* than 7,800 turtles, impacting both immature and adult turtles. 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.

b. *Other impacts*

Threats to sea turtles vary among the species, depending on their distribution and behavior. The value of their meat, eggs, shell or other parts plays an important role in the extent of directed harvest. All sea turtle life stages are vulnerable to human-induced mortality. On nesting beaches, direct exploitation of turtles for meat, eggs, skin or shell, and other products takes place for both commercial markets and local utilization, and to a much lesser degree for traditional ceremonies. Nesting beach and in-water habitat degradation and destruction have occurred due to many factors, including coastal development, dredging, vessel traffic, erosion control, sand mining, vehicular traffic on beaches, and artificial lighting, which repels the adults and disorients the hatchlings. Human alteration of terrestrial habitats can also change the feeding patterns of natural predators, thereby increasing predation on marine turtle nests and eggs.

Petroleum and other forms of chemical pollution affect turtles throughout their marine and terrestrial habitats. Direct 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. Both the entanglement in, and ingestion of, this synthetic debris have been documented (*in* NMFS and USFWS, 1998a-d).

3. Status of the Sea Turtle Species Summary

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 modification or degradation of the turtle's terrestrial and aquatic habitat. 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. The most significant anthropogenic impacts in the marine environment is the incidental capture and mortality in various commercial fisheries. Mortality resulting from the effects of marine pollution are important but much less significant. Increased mortality at the egg and early life history stages has impacted the species' ability to maintain or increase its numbers by limiting the number of individuals that survive to sexual maturity. In addition, the 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, every 2-4 years, minimally. The potential for an egg to develop into a hatchling, into a juvenile, and finally into a sexually mature adult sea turtle will vary among species, populations, and 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 enable a full understanding of the precise effects of this loss of reproductive potential, it is intuitively clear that sea turtle populations cannot withstand abnormally high levels of mortality.

Given the continuing declines of most populations of listed sea turtle species in the Pacific Ocean, it is likely that individuals of the population are not currently able to replace themselves.

IV. ENVIRONMENTAL BASELINE

This section is an analysis of the effects of past and ongoing human and natural factors leading to the current status of the species, their habitat, and ecosystem within the action area.

A. Marine Mammals

1. Fisheries Impacts

a. *Halibut and angel shark set gillnet fishery*

The set gillnet fishery for California halibut (*Paralichthys californicus*) and Pacific angel shark (*Squatina californica*) was observed from July, 1990 to December, 1993 and in the Monterey Bay area from April, 1999 until the present time. Set gillnet fishermen generally use a monofilament net, with a mean length of approximately 470 meters, mean net depth of 24 meshes, and a mean mesh size of 21.2 centimeters, and generally make 2 to 4 sets per day. From 1990-93, estimated effort in this fishery was fairly stable with heavy effort along the southern California coast up to Pt. Conception. There was also effort in the Morro Bay and Monterey Bay areas, as well as some areas around the Channel Islands. Effort decreased sharply in 1994 because gillnet fishing was banned inside 3 nm of shore from Pt. Arguello south to the U.S.-Mexico border (Julian and Beeson, 1998).

Because the most frequently killed mammals in the set gillnet fishery are the California sea lion (*Zalophus californianus*), harbor seal (*Phoca vitulina*) and the northern elephant seal (*Mirounga angustirostris*), there is a possibility that Steller sea lions have also been entangled in this fishery. Effort in this fishery coincides substantially with pinniped habitat. However, there have been no observations of Steller sea lion entanglement or mortality in this fishery since it was observed in 1990 (Julian and Beeson, 1998; D. Petersen, NMFS, personal communication, April, 2000). From 1995-98, one Steller sea lion death was attributed to interaction with fishing gear, most likely a net fishery, but no further information was available (J. Cordaro, NMFS, personal communication, May, 2000). In addition, because Steller sea lions are larger than other species of pinnipeds, they are more likely to survive entanglement in gillnets (Angliss and DeMaster, 1998).

The only cetaceans observed to have been taken (entanglement and mortality) in this fishery are the small harbor porpoise (*Phocoena phocoena*) and the common dolphin (*Delphinus spp.*) (Julian and Beeson, 1998). The likelihood that this fishery has taken any large whales is extremely low because the nets are generally set in shallow waters (~30 fathoms) and are set along the bottom with a height of 6 feet off the bottom. However, a humpback whale was observed off Ventura, California in 1993 with a 20 foot section of netting wrapped around it and trailing behind it, but the source of the gear was unknown (i.e. what fishery it originated from) (J. Cordaro, NMFS, personal communication, *in* Forney, *et al*., 2000). Due to concerns for high incidental catch of seabirds, the fishery was recently closed September 13, 2000 for 120 days within 60 fathoms of the coastline from Point Sal to Point Arguello and between Point Reyes and Yankee Point. This closure may result in interactions with marine mammal species not previously encountered by the fishery, although the anticipated impact of this closure is unknown.

b. *U.S. Albacore Troll Fishery*

Vessels off California troll for Pacific salmon species, generally from Point Conception to the California-Oregon border out to the continental shelf (30 to 40 miles offshore). Observers have obtained catch and bycatch information, including marine mammal interactions with gear, from dockside interviews for the past five years, and starting in June, 2000, observers began gathering data on board the trollers. Although marine mammals interact with trolling gear, there have been no reports by fishermen of injuries or death as a result of these interactions, based on 5 years of dockside sampling (A. Grover, CDFG, personal communication, May, 2000). In 1997, however, one humpback whale was snagged by a salmon troller off central California, and the animal swam away with the hook and many feet of trailing monofilament (*in* Forney, *et al*., 2000). Potential impacts to marine mammals by U.S. commercial trollers primarily involve interactions of pinnipeds with the gear and catch. Seals and sea lions may be injured while taking bait or catch, either by being shot by fishermen trying to protect their catch or by being snagged by the gear. Stranding data from 1995-98 indicate that 2 Steller sea lions were shot and killed (also reported in "Other Impacts"), and in 1996, a Steller died due to a flasher lodged in its throat, which could have been attributed to commercial or recreational gear (J. Cordaro, NMFS, personal communication, May, 2000). Biologists have on occasion observed Steller sea lions on Año Nuevo with flashers embedded in their mouths (P. Thorson, SRS Technologies, personal communication, April, 2000).

U.S. troll vessels have fished for Pacific albacore (*Thunnus alalunga*) since the early 1900s, with collections of logbook and length-frequency data since 1951. Observer data was collected from 1990-97, for a total of only 27 trips. Vessel captains and NMFS observers have noted bycatch as anecdotal comments in logbooks and observer notes, and prior to 1999, these comments were not computerized. Beginning in 1999, logbooks include a space for entry of bycatch and are computerized annually.

Preliminary indications are that bycatch is relatively low (A. Coan, NMFS, personal communication, April, 2000).

c. *California Longline Fishery*

Longliners that fish in waters outside the 200 nm EEZ and unload their catch and reprovision in California ports are required to have a license from California and are subject to state regulations. Since 1993, the number of vessels in this fishery have increased. From 1991 to 1993, only three high-seas longline vessels fished in waters beyond the EEZ for swordfish and tunas and landed their catch in southern California. In late August, 1993, longline vessels from the Gulf of Mexico began arriving in southern California, and by 1994, a total of 31 vessels landed swordfish and tuna taken beyond the EEZ (Vojkovich and Barsky, 1998). Currently, approximately 40-50 longline vessels unload in California, and this number is expected to increase as more vessels from Hawaii fish further east as a result of closures around the Hawaiian Islands due to a court injunction. In fact, since December, 1999, 40 longline boats that originated in Hawaii have unloaded their catch in California ports (D. Petersen, NMFS, personal communication, April, 2000).

Typically, longline vessels fish 24-72 km of mainline, rigged with 22 m gangions at approximately 60 m intervals. Anywhere from 800 to 1,300 hooks are deployed in a set, with large squid (*Illex* sp.) used for bait, typically. Variously colored lightsticks are used, for fishing takes place primarily during the night, when more swordfish are available in surface waters. The mainline is deployed in 4-7 hours and left to drift unattached for 7-10 hours. Retrieval typically takes about 7-10 hours. No observer program exists for this longline fishery; therefore, bycatch has not been documented. However, captains and crew members have reported to dockside samplers that unmarketable species have been caught, and a video has been seen showing striped marlin, bird, marine mammal, and sea turtle bycatch (Vojkovich and Barsky, 1998).

Preliminary catch data has been compiled from skipper logbooks for this fishery from August 1, 1995 through December 31, 1999. None of the listed marine mammals currently being analyzed in this Opinion were reported taken by the California-based longline fishery. However, because this fishery has not been observed, and skippers typically do not report every species they incidentally capture, the effects of this fishery on humpback whales, fin whales, or sperm whales is currently unknown. Interactions between the Hawaii longline fishery and humpback whales have been reported, as have interactions between the Gulf of Alaska longline fishery and sperm whales. Therefore, it is not unlikely that this fishery may take large whales. Because Steller sea lions are not likely to be found beyond 200 miles from the coast, they most likely do not interact with the California-based longline fishery.

2. Other impacts

Strandings of listed marine mammals are rare occurrences off the California coast, and often the cause for the stranding is unknown, especially whether or not they were precipitated by human-related interactions . The following reported strandings occurred off California from 1990-99: 6 fin whales (0.6 annual), 23 humpback whales (2.3 annual), 12 sperm whales (1.2 annual), 3 unidentified baleen whales, and 73 Steller sea lions (7.3 annual). In addition, there have been reports of ship strikes with marine mammals. From 1995-98, one humpback and two fin whales died as a result of collisions with vessels, and two unidentified whales were reportedly struck by vessels (one mortality, one unknown fate). In addition, pinnipeds are shot and killed on occasion, particularly those interacting with commercial and/or recreational fishing gear. From 1995-98, two Steller sea lions were reported shot and killed (J. Cordaro, NMFS, personal communication, May, 2000).

B. Sea Turtles

Many of the impacts described in the previous section (Factors affecting Sea Turtles in the Pacific Ocean) also affect sea turtle populations off the U.S. west coast. Most sea turtle species migrate very long distances between nesting sites and foraging areas, as indicated by tag and recapture studies, so fisheries or other activities located off California and Oregon may impact sea turtles that originated from as far away as Indonesia or Japan.

1. Fisheries impacts

Fisheries other than the U.S. drift gillnet fishery off California and Oregon incidentally take sea turtles. U.S. fisheries include the California set gillnet fisheries, California-based longline fishery, and U.S. albacore troll fishery.

a. *Halibut and angel shark set gillnet fishery*

The California set gillnet fishery for halibut and angel shark has been observed to take sea turtles. In July, 1990, NMFS implemented an observer program for this fishery in order to monitor marine mammal bycatch. NMFS observer coverage ranged from 0% to 15.4% between July, 1990 and July, 1994. The observer program for the set gillnet fishery was terminated in July, 1994 because of a significant decrease in fishing effort in that fishery (due to regulations that restricted areas open to gillnet fishing) (Julian and Beeson, 1998). In April, 1999, the set gillnet fishery off Monterey was again monitored, but no sea turtle interactions have yet been reported (D. Petersen, NMFS, personal communication, April, 2000). Table 11 provides a summary of observed and estimated sea turtle mortalities by species in this fishery from 1990 to 1994. Four of the observed mortalities occurred offshore of Ventura, California. In addition to mortalities, two unidentified sea turtles were observed entangled and released alive in 1993 (estimated total take=13). Five unidentified turtles were estimated (no observer coverage) to have been entangled in 1995. The 1995 estimates were based on stratified rates from 1993 results (Julian and Beeson, 1998) and will therefore not be used in the integration and synthesis of effects. Due to concerns for high incidental catch of seabirds, the fishery was recently closed September 13, 2000 for 120 days within 60 fathoms of the coastline from Point Sal to Point Arguello and between Point Reyes and Yankee Point. This closure may result in interactions with sea turtle species not previously encountered by the fishery, although the anticipated impact of this closure is unknown.

Table 11. **Observed and estimated (in parenthesis) sea turtle *mortalities* in the California set gillnet fishery for halibut and angel shark from 1990-95[1].**

Species/Year	**1990**	**1991**	**1992**	**1993**	**1994**	**1995[2]**
green turtle	0 (0)	0 (0)	1 (8)	1 (6)	0 (0)	(2)
loggerhead	0 (0)	0 (0)	1 (8)	0 (0)	0 (0)	(0)
leatherback	0 (0)	0 (0)	0 (0)	0 (0)	1 (8)	(0)
unidentified	0 (0)	0 (0)	0 (0)	1 (6)	0 (0)	(2)

[1]From Julian and Beeson (1998).
[2]Estimates for 1995 were based on stratified rates from 1993 results (Julian and Beeson, 1998)

b. *U.S. Albacore Troll Fishery*

Anecdotal information indicates that there are rare occurrences of sea turtles taken in the U.S. albacore troll fishery. We cannot determine how many of these turtles are killed or seriously injured based on the data available. Data on bycatch of species other than finfish have not been compiled (J. Wetherall, NMFS, personal communication, 1999).

c. *California longline fishery*

As described in the previous section (section IV.A.3.), longliners that fish for swordfish and tunas in waters outside the 200 nm EEZ and unload their catch in California ports are required to have a California license and are subject to state regulations. Since 1993, the number of vessels in this fishery have increased, from 3 to the current estimate of 40-50 high-seas longline vessels unloading their catch in southern California. This increase in vessels initially resulted from the movement of vessels based in the Gulf of Mexico into southern California in the summer of 1993, and more recently from increased effort eastward by vessels originating in Hawaii, responding to a court injunction closing fishing areas around the Hawaiian islands. Currently, approximately 40-50 longline vessels unload in California, and of these, 40 boats originating from Hawaii (and also have Hawaii longline limited entry permits) have unloaded their catch in California ports since December, 1999 (D. Petersen, NMFS, personal communication, April, 2000).

Typically, vessels fish 24-72 km of mainline, rigged with 22 m gangions at approximately 60 m intervals. Anywhere from 800 to 1,300 hooks are deployed in a set, with large squid (*Illex* sp.) used for bait, typically. Variously colored lightsticks are used, for fishing takes place primarily during the night, when more swordfish are available in surface waters. The mainline is deployed in 4-7 hours and left to drift unattached for 7-10 hours. Retrieval typically takes about 7-10 hours. No observer program exists for this longline fishery; therefore, bycatch has not been documented. However, captains and crew members have reported to dockside samplers that unmarketable species have been caught, and a video has been seen showing striped marlin, bird, marine mammal, and sea turtle bycatch (Vojkovich and Barsky, 1998). In addition, preliminary catch data from skipper logbooks have recently been compiled for this fishery, although the data has not been verified or standardized for effort, seasonality, size, etc. From August 1, 1995 through December 31, 1999, thirty different vessels fished a total of 2,090 days, deploying a total of 7,071,745 hooks (CDFG unpublished data). Table 12 shows the total number of turtles reported taken by this fishery during this time period and whether they were released alive, injured, or dead.

Table 12. Logbook reports of Sea turtles reported taken in the California longline fishery from August 1, 1995-December 31, 1999

Species	Alive	Injured	Dead
Green turtle	12	0	0
Leatherback	33	2	0
Loggerhead	21	0	0
Olive ridley	19	0	0

Source: unedited data from high-seas longline logbooks submitted to CDFG, and reported by M. Vojkovich (CDFG) on 9/29/00.

2. *Other impacts*

Sea turtles are occasionally entrained in power plants off the coast of California, which depend on ocean water to cool the steam that powers the energy-generating turbines. Generally the animals either get pulled into the inlet, located a few thousand feet off-shore, or enter looking for food. In addition, dead animals could get inadvertently sucked into the intake pipes. The majority of animals survive, however, because of technology in place in most plants that allows animals to be safely returned to the ocean, uninjured. From 1983-91, 12 green turtles were entrained, and of these, 9 were released alive and the remaining three were dead, but decomposed, indicating that the turtles were probably already dead when pulled into the intake pipes. During this same time period, two loggerheads were entrained, and both of these were released alive (J. Cordaro, NMFS, personal communication, April, 2000).

Summary of the Environmental Baseline on Sea Turtles

Since sea turtles are wide-ranging species, the status of the green, leatherback, loggerhead, and olive ridley sea turtles are generally the same within the action area as throughout their entire range. None of the factors within the action area described above appear to improve an individual of the species' ability to replace itself, or improve the survival rates of individuals of the species.

V. EFFECTS OF THE PROPOSED ACTION

Pursuant to Section 7(a)(2) of the ESA (16 USC §1536), federal agencies are directed to ensure that their activities are not likely to jeopardize the continued existence of any listed species or result in the destruction or adverse modification of critical habitat. During this consultation, NMFS has analyzed the effects of the action on the listed species to determine whether the action is likely to jeopardize the continued existence of that species. This analysis is done after a careful review of the listed species' status and the factors that affect the survival and recovery of that species, as described above.

The proposed action is the authorization, valid for three years after issuance, for the CA/OR drift gillnet fishery to incidentally take marine mammals under section 101(a)(5)(E) of the MMPA. Prior to issuance of the 101(a)(5)(E) authorization for the fishery, NMFS must ensure that the effects of this issuance are not likely to jeopardize listed species. Therefore, in order to assess the likely effects to listed species, NMFS has prepared the following analysis of the expected effects of the fishery covered by the 101(a)(5)(E) permit on listed marine mammals and sea turtles.

Expected fishing effort by the CA/OR drift gillnet fishery

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

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

Existing drift gillnet shark and swordfish permits may only be transferred when: 1) the permittee has held the permit for 3 years; or 2) the permittee is injured or has a serious illness and hardship if the permit cannot be transferred; or 3) a marriage is dissolved and the permit is held as community property; or 4) the permittee has died and the surviving family wishes to transfer the permit. Permits may only be transferred to a person who holds a commercial fishing license and a general gill and trammel net permit[10]. Permits can be revoked or suspended by the director upon conviction for willful violation of CDFG code. Currently, permits can only be renewed by individuals who possess a valid general gillnet and trammel net permit and a valid drift gillnet permit. In addition, during one of the two immediately preceding seasons, the permittee must land at least 2,500 pounds of swordfish, or 1,000 pounds of shark, or landed shark or swordfish for which the permittee was paid $1,000 (CDFG code, §8561.5).

The overall fishing effort trend has continued to decline during the last 13 years with the lowest fishing effort occurring in 1999 with only 2,634 total sets (Figure 1). Despite the slight increase in fishing effort during the 1998 fishing season, the overall fishing effort trend has been downward. Based on this downward trend, NMFS anticipates the fishing effort for the next three years will continue to decline. In addition, NMFS does not expect that the overall fishing effort for any of the next three calendar years will exceed 3,000 sets. This annual estimate is supported by the fact that the fishing effort average for calendar years 1997 - 1999, is equal to approximately 3,000 sets per year. Furthermore, the number of vessels that have obtained Marine Mammal Authorization Certificates during the past three years have decreased from 126 vessels in 1997 to 109 vessels in 1999 (D. Petersen, NMFS, personal communication, May 2000), and the number of vessels actually making landings has dropped from 115 vessels to 96 vessels, respectively (R. Read, CDFG, personal communication, June 2000). This reduction in the number of fishing vessels during the mid-1990s can be attributed partly to the larger vessels (greater than 50 feet) switching from fishing swordfish using a drift gillnet to fishing squid using a purse seine net and other vessels switching to longline gear. In addition, the number of fishing days was further reduced during the mid-1990s when many of the larger vessels began targeting albacore tuna during the summer months and into late September rather than target swordfish using drift gillnet gear. This reduction in the number of fishing vessels participating in the fishery, the reduced fishing days by vessels targeting albacore, and the number of permits lapsing because of retirement, illness, injury, and death is expected to keep the overall fishing effort by the CA/OR drift gillnet fishery to below 3,000 sets for each subsequent calendar year.

[10]General gill and trammel net permits must be renewed annually and are only transferable if a person has: 1) previous experience as a crewmember of a vessel using gillnet or trammel nets; or 2) successfully passed a proficiency test administered by CDFG; or 3) met the landing requirements specified under Title 14 (California Code of Regulations, Title 14, §174).

A

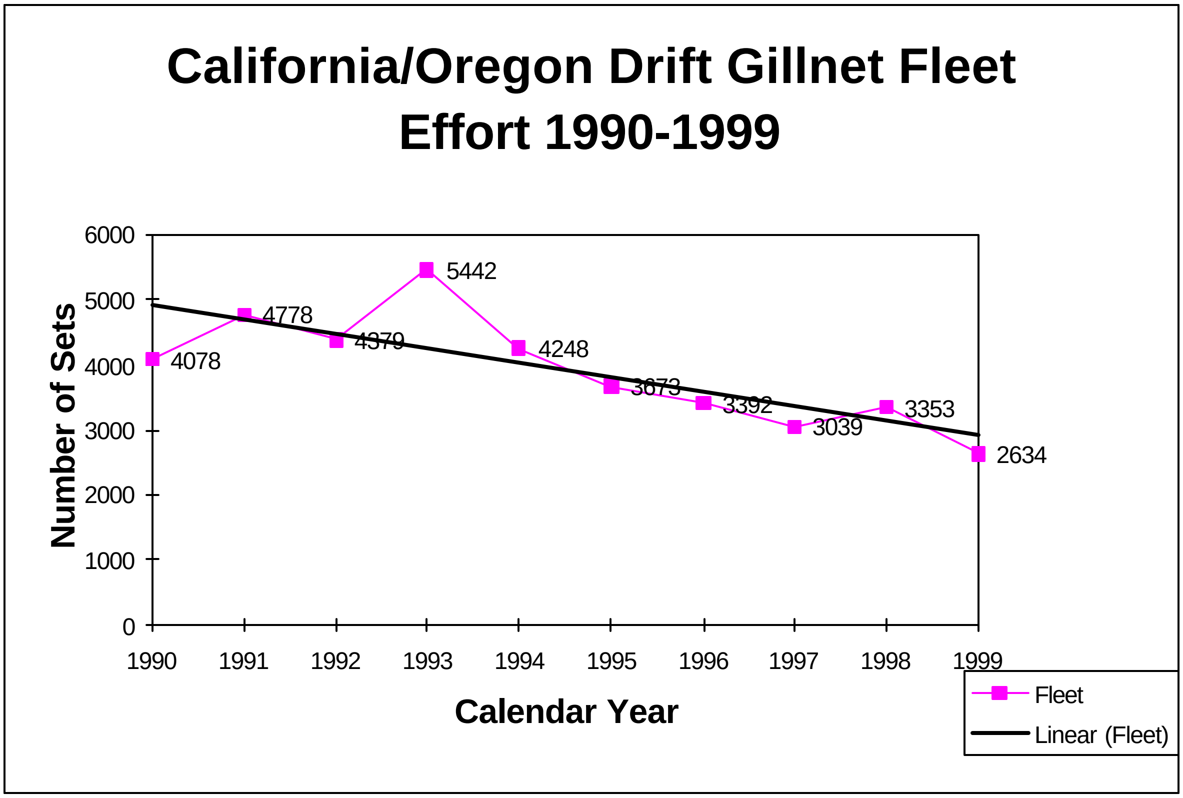

Figure 1 Calendar year fishing effort estimates for the California/Oregon drift gillnet fishery 1990 through 1999. The overall fishing effort trend indicates a decrease in fishing effort to levels below 3,000 sets per year.

. Marine Mammals

1. General impacts to marine mammals

In the CA/OR drift gillnet fishery, a wide variety of marine mammals are killed, which is most likely attributable to the large geographic range of many of the species, nonselectivity of gear, and the amount and location of fishing effort. For example, cetacean bycatch in the CA/OR drift gillnet fishery is greater and more diverse than for the California set net fishery because the area of driftnet effort contains more diverse habitat than the area of the set net fishery.

The probability that a marine mammal will initially survive an entanglement in fishing gear depends largely on the species and age of marine mammal involved. For instance, larger animals such as balaenopterids, sperm whales and Steller sea lions may become entangled in gillnet but often survive the initial contact with the gear. Such entanglement may cause considerable damage to the gear, as the large whales "punch" through and continue swimming. Such damage may be related to the type of net used, however, for fishermen do report that large blue and fin whales usually break through drift gillnets without entangling, and that very little damage is done to the net (Barlow, *et al*., 1997). Marine mammals may also swim away with a portion of the gillnet wrapped around a pectoral fin, the tail stock, the neck or the mouth. For large whales, there are generally three areas of entanglement in a net: 1) the gape of the mouth, 2) around the flippers, and 3) around the tail stock (although this area is often difficult to view, as most balaenopterids do not fluke frequently). Documented cases have indicated that entangled animals may travel for extended periods of time and over long distances before either freeing themselves, being disentangled by an outside network, or dying as a direct result of the entanglement (Angliss and DeMaster, 1998). In most cases, it is unknown whether the injury is serious enough or debilitating enough to lead to death. If the debris fragments are heavy, the animal will most likely drown. Less heavy fragments may lead the animal to exhaustion, depletion and starvation due to the increased drag (Wallace, 1985). In addition, if an animal's appendage or head (in the case of a pinniped) is caught in a mesh, the debris can debilitate the animal, especially if it is constricting, causes lacerations, or impairs swimming or feeding ability (Scordino, 1985). Younger animals are particularly at risk if the entangling gear is tightly wrapped, for as they continue to grow, the gear will likely become more constricting. The majority of large cetaceans that become entangled are juveniles (Angliss and DeMaster, 1998). Marine mammals that die as a result of entanglement in drift gillnets usually drown. With a typical soak time of 12-14 hours, the animal is unable to survive without oxygen, especially if it is entangled at the beginning of the set, or deep in the net.

Marine mammals may also be indirectly affected as a result of being captured in a drift gillnet. An entanglement may compromise the animal by causing cuts or impeding mobility or feeding, which may make the animal more susceptible to disease or predation. In addition, although marine mammals have evolved to handle a wide variety of stressors, including a saline environment, predation, food shortages, etc, only healthy animals have an optimal healing response. Cetaceans in particular have developed a very unique healing process, which requires salt and water to kill several cell layers to block penetration of additional salt water. After this process is completed, healing from within can begin. A sustained stress response, such as repeated or prolonged entanglement in gear, makes marine mammals less able to fight infection or disease. Pinnipeds have a physiologically different response to stress than cetaceans. Chronic exposure to stress causes an imbalance of numerous hormones or enzymes which can lead to metabolic anomalies, such as increased sodium concentration in the blood, tissue necrosis, and hypoxia. Such symptoms may not manifest for several days after entanglement, and in severe cases, death could be the result, even though superficially an animal might appear healthy (Angliss and DeMaster, 1998).

In the CA/OR drift gillnet fishery observers record detailed information on marine mammals entangled in the net. Animals that are released alive from the net with netting attached are classified as "injured." Animals that release themselves or are released from the net by fishermen and can swim normally are recorded as "alive." Marine mammals that have been entangled and are released alive usually only have minor abrasions as a result of interaction with the net. However, as discussed above, effects from the stress of capture may cause temporary and/or long-term effects that may not be visible upon release.

Because no long-term stress studies have been conducted on the impacts of capture by a fishery on marine mammals, NMFS is only able to make assumptions on the condition of marine mammals that have been released "unharmed" from a drift gillnet. Although marine mammals released "unharmed" do not have visible injuries, they may have been stressed from being caught or entangled in a net. This stress may cause an interruption in essential feeding behaviors or migration patterns; however, NMFS believes this effect, if experienced, is likely to be temporary and short-term. For these reasons, NMFS will assume that most of the marine mammals released and reported as "unharmed," or uninjured, have not been harmed or harassed by their capture in a drift gillnet, and that latent effects are limited to short-term physiological stress or interruption of normal behavioral patterns.

All marine mammal species that forage or migrate by diving or swimming at depth in areas of concentrated fishing effort are vulnerable to drift gillnets. Susceptibility to capture largely depends on a species' physical characteristics and behavior. Not surprisingly, survival rate likely varies among marine mammal species incidentally taken by the CA/OR drift gillnet fishery. This is due in part to variations in size and diving and foraging behavior, as well as location in the net and time of capture. With few observed marine mammal captures in the CA/OR drift gillnet fishery, it is difficult to speculate as to the survival rate of the four listed species observed taken in the fishery from 1990-2000. However, because the baleen whales (humpback and fin) and the sperm whale differ so greatly in the nature of their food and foraging behavior (e.g. the sperm whale is capable of diving to much greater depths than the baleen whales in order to find their preferred prey of squid, depending largely on oxygen storage and metabolism, while the baleen whales rely less on diving, if possible, and tend to skim and gulp for euphausiids at the surface or below) and their physiology, survival rates following gillnet entanglement most likely vary greatly as well. Of the 8 sperm whales entangled in CA/OR drift gillnet gear, 3 survived uninjured (37.5 percent), 1 was released injured (12.5 percent), and 4 were killed (50 percent). Of the 3 baleen whales entangled in drift gillnets from this fishery, 2 were released alive (both humpback whales) (66 percent), and one was killed (fin whale) (33 percent). Both Steller sea lions observed entangled in this fishery were killed, giving them a survival rate of 0 percent.

a. *Fin whale impacts*

The incidental take of fin whale in the CA/OR drift gillnet fishery is extremely rare. From July, 1990 until January, 2000, observers recorded the entanglement and mortality of only one fin whale by the fishery, in 1999, off southern California. The net had a full complement of pingers (40), and had 36 foot extenders, as required by the PCTRP.

The fin whale taken in 1999 was entangled southwest of San Clemente Island, in an area characterized by a generally counterclockwise current flow or gyre centered in the Gulf of Catalina. About the center of the current gyre, sea surface temperatures tend to be higher than temperatures found to the north or south of the Gulf of Catalina. These warmer temperatures attract subtropical species such as striped marlin and swordfish, as well as large whales, such as the fin whale. In addition, coastal upwelling areas are prime foraging areas for fish and marine mammals, attracted to the high primary productivity. The local distribution of fin whales during much of the year is probably governed by prey availability. Like swordfish, fin whales have been known to associate with steep bottom contours, most likely because tidal and current mixing along such gradients drives high biological production. During the year immediately following the 1997-98 El Niño event, zooplankton production was exceptionally high, primarily because this period saw a transition from the warm-water conditions associated with the El Niño event to cool water conditions which were still prevalent in coastal southern California in October, 1999. Because euphausiids are a fin whale's prey of choice, this fin whale was most likely taking advantage of the locally high biological productivity, either by surface feeding, or foraging by diving. From November, 1999 through January, 2000, an anomalously high upwelling event occurred off southern California, which most likely increased primary productivity and attracted large whales to the area. Observers also recorded the incidental take of one humpback and one minke whale (*Balaenoptera acutorostrata*), two other baleen whale species rarely taken by the fishery, on the same day (11/29/99) and in the same general area that the fin whale was taken, further indicating that high forage density may have played a role in the fin whale interaction.

Fin whales are very rarely taken in the CA/OR drift gillnet fishery. Based on a worst-case scenario, NMFS estimates that a maximum of 6 fin whales ((1 fin whale observed entangled and killed in 1999/526 sets observed in 1999) x 3,000 maximum expected sets per year) in a given year could be captured by the CA/OR drift gillnet fleet and killed. Based on anecdotal reports from fishermen, who have evidence of large whales punching through their nets, fin whales have likely interacted with the CA/OR drift gillnet fishery before. However, because of their size and strength, fin whales likely punch through the net, and entanglement is a rare event. Entanglement, and any associated mortality, of fin whales is not anticipated to occur every year. Based on past fishery performance, fin whales were observed taken once in ten years, or once during the three years the PCTRP has been in place. Erring conservatively for the species, takes of fin whales could occur this frequently again (once in three years), resulting in a total expected impact to fin whale populations of 6 whales entangled and killed during the three year period of the proposed action.

b. *Humpback whale impacts*

From July, 1990 to October 29, 1997, the day before the effective date of the PCTRP, observers recorded the incidental entanglement of one humpback by the CA/OR drift gillnet fishery, in 1994, off southern California. This animal was released alive and uninjured. Following the implementation of the PCTRP, only one humpback was observed entangled, in 1999, off southern California; this animal was also released alive and uninjured. The net had a full complement of pingers (41) and 36 foot extenders.

Both humpback whales caught by this fishery were caught south of Point Conception during years immediately following El Niño events (1992-93 and 1997-98), and during the months (August and November) when humpback whales typically are found north of their breeding grounds, taking advantage of coastal upwelling events. Humpback whales feed both at the surface and at depths. Surface feeding is characterized by fast, short-duration dives, and rapid surface swim speeds compared with deep diving. Humpback whales observed off the California continental shelf from 1988-90 primarily fed on euphausiids; however their foraging behavior changed as environmental conditions changed. The whales fed at the surface 56% of the time in 1988 and 32% of the time in 1990, using primary lateral lunges to capture swarms of euphausiids. In 1989, however, no surface feeding was observed; instead, deep, long-duration dives were followed by extended surface intervals with many respirations. These 1989 observations coincided with increased prey depth as indicated by depth sounder records of diving whales and prey scattering layer. The increased prey depth and associated feeding behaviors were strongly associated with unusually high sea surface temperatures, calm seas, and changes in water circulation (Kieckhefer, 1992).

The humpback observed entangled in 1994 was taken in an area and during a time of the year (August) when the average monthly sea surface temperature was approximately 20°C, and about 0.5-1.0°C above normal (Coastwatch El Niño watch). Although there was coastal upwelling in the area, which could have brought food to the surface for the whale, the animal may have had to forage at depth, causing it to interact with the driftnet gear. The humpback observed entangled in 1999 was taken in an area and at a time (November) when the fishery was observed to capture a higher number of large whales and sea turtles than normal. The waters off southern California during this time period were characterized by an extremely strong and anomalous upwelling event. Marine mammals, sea turtles, and other pelagic species that feed on zooplankton and small fish were likely attracted to this concentrated food source, and because drift gillnet fishery effort in that area and during that time period is normally high, the concurrence of fishing effort and foraging animals caused more entanglements than normal.

Humpback whales are rarely taken by the CA/OR drift gillnet fishery, and of the two whales taken in ten years, both have survived uninjured. Based on a worst-case scenario, NMFS estimates that a maximum of 6 humpback whales ((1 humpback observed taken in 1999/526 sets observed in 1999) x 3,000 maximum expected sets per year) in a given year could be captured by the CA/OR drift gillnet fleet. Fishermen have reported anecdotally evidence of large whales punching through their nets; therefore, humpback whales likely interact with the CA/OR drift gillnet fishery. However, because of their size and strength, humpback whales likely punch through the net, and entanglement is a rare event. Entanglement, and any associated mortality, of humpback whales is not anticipated to occur every year. Based on past fishery performance, humpback whales were observed taken twice in ten years, or once during the three years the PCTRP has been in place. Therefore, NMFS anticipates that humpback whale entanglement could occur once during the three year period of the proposed action, resulting in a total expected impact to humpback whale populations of 6 whales entangled during the three year period of the proposed action, with no anticipated mortalities.

c. *Sperm whale impacts*

Prior to the implementation of the PCTRP on October 30, 1997, the CA/OR drift gillnet fishery was observed to incidentally take seven sperm whales; of these whales, three were dead (43%), three were released alive and uninjured (43%), and one was released injured and was not expected to survive (14%). In 1992 the CA/OR drift gillnet fishery was observed taking 3 sperm whales in one set off central California; two were alive and released uninjured, and one was dead. The net was suspended 36 feet below the surface. In 1993, 2 sperm whales were entangled in one set off southern California; one was alive and released uninjured, and one was dead. The extender length of the net was 60 feet. Also in 1993, one sperm whale was observed entangled and died in a drift gillnet off central California, with a net that was using 36 feet extenders. In 1996, one sperm whale was observed entangled and released injured (trailing gear, and wounded from ramming the vessel) off central California. The net was configured with 33 pingers, and was suspended 36 feet below the surface. Since the implementation of the PCTRP, only one sperm whale was observed incidentally taken in 1998. This animal died in a net off central California which did not have the full complement of pingers.

There is speculation that sperm whales tend to feed at nighttime, and because they often forage by diving to great depths, possibly with an open jaw, they may be more vulnerable to a drift gillnet than perhaps other large whales. In addition, because sperm whales often prey on luminous squid, they may be attracted to light sticks occasionally used by drift gillnetters, which may explain why the CA/OR drift gillnet fishery has been observed taking over twice as many sperm whales (eight) as it has fin and humpback whales combined (three).

All of the sperm whales incidentally taken in the CA/OR drift gillnet fishery were caught between October and December, in waters with an average sea surface temperature of between 13 and 18°C. Sperm whales are found in peak abundance off California from the end of August to mid-November, during the same time period when effort in the fishery increases. All but two (caught in the same net) of the sperm whales were taken in a concentrated area 50-75 miles west of Monterey Bay, California. Three of the sperm whales caught in this area were entangled in the same set, and based on their estimated length (12, 14, and 20 feet), they were likely subadults from a breeding school, beginning their south-bound migration down the coast. In addition, most (6/8) of the sperm whales taken were caught during the 1992-93 El Niño, when a lack of upwelling and unusually high sea surface temperatures resulted in animals having to forage at depth for longer periods of time for food, making them increasingly vulnerable to a drift gillnet. Sperm whales appear to be vulnerable to becoming entangled in uncomplicated gear, and this may be due to their foraging behavior, curiosity, or something unexplainable. Heezen (1957) documented 14 instances where sperm whales were entangled in deep sea cables, some as deep as 3,000 meters, along the ocean floor.

Of the eight sperm whales observed taken by the CA/OR drift gillnet fishery, three were released alive and uninjured (37.5 percent), one was released injured (12.5 percent), and four were killed (50 percent). Therefore, approximately 63 percent of captured sperm whales could be killed accidentally or injured (based on the mortality and injury rate of sperm whales observed taken by the U.S. fleet from 1990-2000). Based on past fishery performance, sperm whales are not observed taken in every year; they were observed taken in four out of the last ten years. During the three years the PCTRP has been in place, a sperm whales was observed taken only once (in a non-PCTRP compliant set).

Therefore, NMFS conservatively anticipates that sperm whale entanglement could occur once during the three year period of the proposed action, resulting in a total expected impact to sperm whale populations of 6 whales entangled and of these, 4 whales would be killed during the three year period of the proposed action.

d. *Steller sea lion impacts*
Steller sea lions are rarely taken in the CA/OR drift gillnet fishery. In the 10 years that NMFS observers have been collecting data, Steller sea lions have been observed entangled and killed in two instances in the CA/OR drift gillnet fishery, one in 1992, off central California (net extenders were 20 feet), and one in 1994, off the California/Oregon border (net extenders were 30 feet). No Stellers have been caught since the implementation of the PCTRP, in October, 1997.

The two Steller sea lions observed taken by the fishery were caught in waters with sea surface temperature of approximately 17°C in areas where upwelling was occurring. The Steller taken in June, 1992, off Catalina Island, was found considerably further south than its southern-most rookery (Año Nuevo), and since the breeding season generally extends from late May to early July, this animal was likely a juvenile, foraging for dwindling food sources characteristic of an El Niño year. The Steller observed taken in September, 1994 was found just north of the California-Oregon border, so it may have originated from the northernmost California rookery (Point St. George) or the southernmost Oregon rookery, Orford Reef, both equidistant from the capture location.

Only one of the entangled Stellers was measured and sexed, and at 270 cm, the animal was an adult female. In general, there appears to be no detectable environmental anomaly or pattern of fishing strategy that would explain the incidental take of Stellers. Therefore, because the Steller sea lion and the CA/OR drift gillnet fishery are known to co-occur in areas off the California and Oregon coast, and the implementation of the PCTRP appears to have reduced the incidental take of pinnipeds, NMFS expects the entanglement of Stellers in this fishery to be a rare event.

Of the two Steller sea lions observed taken by the CA/OR drift gillnet fishery, both were killed (100 percent mortality). The incidental take of Steller sea lions is a rare event, and the use of mid-frequency pingers and longer extenders appears to reduce the likelihood of interactions between pinnipeds and the CA/OR drift gillnet fishery (i.e. no Stellers have been observed taken by this fishery since the implementation of the PCTRP (see Informal Consultation)). Even though Steller sea lions have not been observed taken during the three years the PCTRP has been in place, NMFS conservatively anticipates that Steller sea lion entanglement could be observed once during the three year period of the proposed action. Based on a worst-case scenario, NMFS estimates that a maximum of 5 Steller sea lions ((1 Steller sea lion observed entangled and killed in 1992/596 sets observed in 1992) x 3,000 maximum expected sets per year) could be incidentally entangled and killed by the CA/OR drift gillnet fleet over the next three years.

B. Sea Turtles

1. General impacts to sea turtles

Determining the scope and magnitude of impacts of the CA/OR drift gillnet fishery on sea turtle populations is complicated by the fact that all species lead an oceanic existence during most of their life history. There are broad gaps in our knowledge of sea turtles in the marine environment due to the difficulties in studying them away from their nesting beaches. Recent technological developments in satellite telemetry are rapidly expanding our knowledge on the movements and habits of sea turtles in the marine environment, but much remains unknown. In contrast, at certain nesting beaches, reasonably good ecological data exist for the breeding phase when adult females, eggs, and hatchlings are accessible. The leatherbacks and olive ridleys are the most pelagic species, living well offshore from the time they leave the beach as hatchlings until they return to breed as adults. Others, such as the green and the loggerhead, inhabit coastal waters as adults, but spend varying segments of their immature life in the open ocean. Even then, the adults regularly undertake breeding migrations, over deep water.

It is apparent that sea turtles are prone to entanglement as a result of their body configuration and behavior (Balazs, 1985). Records of stranded or entangled sea turtles reveal that fishing debris can wrap around the neck or flipper, or body of a sea turtle and severely restrict swimming or feeding. Over time, if the sea turtle is entangled when young, the fishing line will become tighter and more constricting as the sea turtle grows, cutting off blood flow and/or causing deep gashes. Sea turtles have also been found trailing gear that has been snagged on the bottom, thus causing them to be anchored in place (Balazs, 1985).

As summarized earlier, “take” refers to any capture or entanglement in the net and subsequent release, injury, or mortality of a sea turtle. Potential impacts from the CA/OR drift gillnet fishery on sea turtles will generally be related to injury or mortality, although the entanglement episode, whether or not it develops into an injury or mortality, may also impact sea turtles. Injury or mortality of turtles entangled in a long-soaking drift gillnet may result from drowning due to forced submergence, and/or impairment or wounds suffered as a result of net entanglement.

While most voluntary dives by sea turtles appear to be aerobic, showing little if any increases in blood lactate and only minor changes in acid-base status (pH level of the blood), sea turtles that are stressed as a result of being forcibly submerged rapidly consume oxygen stores, triggering an activation of anaerobic glycolysis, and subsequently disturbing the acid-base balance, sometimes to lethal levels. It is likely that the rapidity and extent of the physiological changes that occur during forced submergence are functions of the intensity of struggling as well as the length of submergence (Lutcavage and Lutz, 1997). In a field study examining the effects of shrimp trawl tow times and sea turtle deaths, there was a strong positive correlation between the length of time of the tow and sea turtle deaths (Henwood and Stuntz, 1987, *in* Lutcavage and Lutz, 1997). Sea turtles forcibly submerged for extended periods of time show marked, even severe, metabolic acidosis as a result of high blood lactate levels. With such increased lactate levels, lactate recovery times are long (even as much as 20 hours), indicating that turtles are probably more susceptible to lethal metabolic acidosis if they experience multiple captures, because they would not have had time to process lactic acid loads (*in* Lutcavage and Lutz, 1997). Presumably, however, a sea turtle recovering from a forced submergence would most likely remain resting on the surface, which would reduce the likelihood of being recaptured in a drift gillnet submerged over 30 feet. Recapture would also depend on the condition of the turtle and the intensity of fishing pressure in the area. NMFS has no information on the likelihood of recapture of sea turtles by the CA/OR driftnet fishery or other fisheries.

Additional factors such as size, activity, water temperature, and biological and behavioral differences between species also bear directly on metabolic rates and aerobic dive limits and will therefore also influence survivability in a gillnet. For example, larger sea turtles are capable of longer voluntary dives than small turtles, so juveniles may be more vulnerable to the stress of enforced submergence than adults. During the warmer months, routine metabolic rates are higher, so the impacts of the stress due to entanglement may be magnified. In addition, disease factors and hormonal status may also play a role in anoxic survival during forced submergence. Any disease that causes a reduction in the blood oxygen transport capacity could severely reduce a sea turtle's endurance in a net, and since thyroid hormones appear to have a role in setting metabolic rate, they may also play a role in increasing or reducing the survival rate of an entangled sea turtle (*in* Lutz and Lutcavage, 1997). As discussed further in the upcoming leatherback and loggerhead subsections, some sea turtle species are better equipped to deal with forced submergence.

No stress studies have been conducted on sea turtles that have been released alive after being caught in a drift gillnet. Survivability studies have been conducted on the Hawaii longline fishery and the Atlantic shrimp trawl fishery. Sea turtles captured in the Hawaii longline fishery may suffer stress and injury from entanglement and from internal or external hooking injuries and continued submergence. Sea turtles in the Atlantic shrimp trawl fishery are forcibly submerged by the trawls and kept submerged for long periods, often resulting in high mortalities. Similar to the Atlantic shrimp trawl fishery, turtles entangled in a long-soaking drift gillnet may drown due to forced submergence, or may suffer injuries from net entanglement. Thus, NMFS is only able to make assumptions on the condition of turtles that have been released "unharmed" from a drift gillnet. Although turtles released "unharmed" do not have visible injuries, they may have been stressed from being caught or entangled in a net. This stress may cause an interruption in essential feeding behaviors or migration patterns; however, NMFS believes this effect, if experienced, is likely to be temporary and short-term. For these reasons, NMFS will assume that any turtle released and reported as "unharmed," or uninjured, has not been harmed or harassed by its capture in the gillnet and that latent effects are limited to short-term physiological stress or interruption of normal behavior patterns.

Caution is warranted in making this determination because it is based on two important assumptions. The first assumption is that the "unharmed" turtle will not be subsequently caught in fishing gear. Turtles that are involuntarily submerged experience an imbalance in blood homeostasis and require time to recover to normal ph, CO_2 , and lactate levels. If this recovery time is interrupted by additional forced submergence, the turtle may die as a result. The second assumption is that the "unharmed" turtle is able to recover. A loggerhead recovered from a shrimp trawl net was initially reported as normal, and subsequently became limp. The turtle was kept onboard and went through several periods of activity and lethargy. The turtle was transported to a laboratory facility and continued to exhibit periods of activity with alternate "limp periods" and was finally determined to have died (Stender, unpublished report, 2000). Thus, an apparent normal, active turtle that is returned to the water may subsequently drown.

Mortalities of sea turtles as a result of the proposed action may have long-term effects on the affected population. Other than the obvious impact of a loss of an individual turtle, mortalities also result in the loss of the reproductive potential of that turtle. National Research Council (NRC) (1990) estimates that the reproductive value of an adult loggerhead is 584 times that of an egg or hatchling, because so few eggs or hatchlings survive to maturity. Sea turtles are long-lived and delay sexual maturity for several decades. Loggerheads and green turtles may reach sexual maturity as early as 22 or 30 years of age, or as late as 30 to 60 years of age, respectively. Females of each species lay approximately 100 eggs per clutch in 2 or 3 clutches every 2 to 4 years. Thus, the death of adult or juvenile females could potentially preclude the production of hundreds of hatchling turtles, though most of these would not survive to sexual maturity. NMFS is not aware of a disproportionate mortality of adult female turtles in the CA/OR drift gillnet fishery. Mortalities of adult or large juvenile males would preclude their contribution to future generations, although it is difficult to quantify this impact given the minimal data on male sea turtles, including their abundance.

Three unidentified turtles were observed taken in 1993 off southern California, all in the same trip, but in different sets. Only one of these sea turtles was measured, and at 43 centimeters, the average length of measured loggerheads captured incidentally in this fishery from 1990-2000, this turtle was most likely a loggerhead. In addition, all three turtles were caught in the same concentrated area that all loggerheads in the past 10 years have been caught by this fishery. They were also caught during an El Niño, which is the only time that loggerheads have been caught in this fishery since July, 1990, when the fishery was first observed by NMFS. Only one leatherback out of 23 observed taken was found this far south, and as a species, leatherbacks are very easy to identify and distinguish from the hard-shelled turtles. The only green turtle observed taken by this fishery was caught north of Point Conception. Although the one olive ridley observed taken by this fishery was found in the same general area as where all loggerheads were caught, and field researchers and observers have historically had difficulty distinguishing olive ridleys from loggerheads, only one olive ridley has been observed taken by this fishery since 1990, and it was taken in 1999. For these reasons, and because loggerheads off Baja California have been observed feeding in large concentrated groups, NMFS is assuming that the three unidentified turtles caught were loggerheads. Of these three turtles, two were caught and released alive and one was killed.

Survival rates appear to be greater for hard-shelled turtles than for leatherbacks when forcibly submerged (see further discussion in leatherback and loggerhead impacts section) . For the purposes of this Opinion, the survival rates for the hard-shelled turtles (green, loggerhead and olive ridley) will be combined and the survival rate for the leatherback turtle will be calculated separately. Both survival rates are based on incidental capture data from July, 1990 to January, 2000 by the CA/OR drift gillnet fleet. Leatherbacks caught in this fishery had a survival rate of 39 percent (9 released unharmed/23 total captured), while the hard-shelled turtles had a combined survival rate of 68 percent (13 released unharmed/19 total captured). The total survival rate for all species combined is approximately 52 percent (22 released unharmed/42 total captured), 2.5 percent were released injured (1 injured/42 total), and 43 percent were killed accidentally (18 killed/42 total). The rest were unknown (1).

Because the abundance, distribution, and the migration and foraging patterns vary so significantly between the sea turtle species that may be encountered by drift gillnetters off the west coast of the United States, their vulnerability to the CA/OR drift gillnet fishing operations will also vary. The following sections review the possible impacts of the proposed action on each of the sea turtle species.

a. *Green turtle impacts*

The incidental take of green turtles in the CA/OR drift gillnet fishery is extremely rare. In the ten years that NMFS has been collecting data on the CA/OR drift gillnet fishery, observers have recorded the incidental catch and mortality of only one green turtle, in 1999, off south-central California. Because of the regulations imposed by the PCTRP, effective in October, 1997, the net had a full complement of pingers (41), and 36 foot extenders.

The one green turtle caught in November, 1999 had a CCL of 74.5 centimeters; therefore, it was an immature animal, and genetic analysis indicated that it was an eastern Pacific stock, most likely originating from a nesting beach in Mexico. It is not known whether green turtles regularly migrate from breeding grounds in Mexico to specific areas along the North American coast, or whether reported sightings and strandings of these species are vagrants that occasionally stray into more northern waters, perhaps moving with El Niño currents. Stinson (1984, *in* Eckert, 1993) reviewed sighting records from northern Baja California, Mexico to the Gulf of Alaska and concluded that the green turtle was the most commonly observed hard-shelled sea turtle on the western coast of the United States. Furthermore, the green turtle was the second most commonly stranded sea turtle along the California coast from 1990-99, averaging around 5 strandings per year (J. Cordaro, NMFS, personal communication, May, 2000). In addition, a resident population of green turtles does occur in San Diego Bay, where approximately 50 - 60 mature and immature turtles concentrate in the warm water effluent of a power plant (McDonald, *et al*., 1994). Although temperatures were fairly normal in the fall of 1999 (based on 1950-79 data), there was a sea surface temperature warming trend (2-3°C) from October to November, 1999 off the west coast of the United States, perhaps attracting more warm water species, such as the green turtle. Closer in towards shore, sea surface temperatures were colder than normal from November to January, 2000, due primarily to a very strong upwelling event. As mentioned previously, this upwelling event probably increased the primary production in southern California, attracting large whales and sea turtles who prefer to feed on zooplankton, and increasing their vulnerability to becoming entangled by the CA/OR drift gillnet fishery, which target swordfish in the same area during that time period. During the month of November, 1999, observers recorded the incidental catch by the fishery of three new listed species, all in the southern California Bight, that had never been observed taken in the fishery in the past ten years, further indicating that high forage density may have played a role in the green turtle interaction.

Green turtles are rarely taken in the CA/OR drift gillnet fishery. Based on a worst-case scenario, NMFS estimates that a maximum of 6 green turtles ((1 green turtle observed taken in 1999/526 sets observed in 1999) x 3,000 maximum expected sets per year) in a given year could be incidentally taken by the CA/OR drift gillnet fleet. Assuming that 32 percent of these captured green turtles would be killed accidentally or injured (based on the survival rate of hard-shelled turtles caught by the CA/OR drift gillnet fleet from 1990-2000), NMFS estimates that no more than 2 green turtles would be killed by the CA/OR drift gillnet fleet. The only observed take, in 1999, appears to be related to unusual environmental conditions. Therefore, NMFS expects the capture of green turtles to be a rare event –entanglement, and any associated mortality, of green turtles is not anticipated to occur every year. Based on past fishery performance, green turtles were observed taken once in ten years, or once during the three years the PCTRP has been in place. Erring conservatively for the species, takes of green turtles could occur this frequently again (once in three years), resulting in a total expected impact to green turtle populations of 6 turtles entangled, including 2 killed, during the three year period of the proposed action.

b. *Leatherback Impacts*

Prior to the implementation of the PCTRP, from July, 1990 to October 30, 1997, observers recorded the incidental take of 21 leatherbacks. Of these turtles, 13 were killed (62%), 7 were released alive and uninjured (33%), and the fate of one was recorded as “unknown” (5%), assumed to be a mortality. Since the implementation of the PCTRP, only 2 leatherbacks have been incidentally captured, and of those, both were released alive. Both nets had pingers, one with 41 and one with 36, and 36 foot extenders. Therefore, from June, 1990 to January, 2000, a total of 23 leatherbacks have been taken, with 14 killed or had an unknown fate, assumed to be a mortality (61%), and 9 were released alive (39%).

Leatherbacks are vulnerable sea turtle to fishing gear. Their long pectoral flippers and their extremely active behavior make them particularly vulnerable to any ocean debris. Observed leatherbacks entanglements have primarily involved the front flippers and/or the neck and head region. Studies of daily swimming patterns over time yielded a very small percentage (0-7%) of time in which the leatherback was not swimming (S. Eckert, in prep. May, 2000). Leatherback hatchlings studied in captivity for almost 2 years swam persistently without ever recognizing the tank sides as a barrier (Deraniyagala, 1939, *in* Wyneken, 1997). A leatherback entangled in a net will most likely continue trying to swim, expending valuable amounts of energy and oxygen. As available oxygen diminishes, anaerobic glycolysis takes over, producing high levels of lactic acid in the blood. Unlike the shelled turtles, leatherbacks lack calcium, which helps to neutralize the lactic acid build-up by building up bicarbonate levels. In addition, leatherbacks store an enormous amount of oxygen in their tissues, similar to marine mammals, and have comparatively high hematocrits, which is efficient for such a deep-diving turtle but means that they have relatively less oxygen available for submergence. Maximum dive duration for the species is substantially less than half that of other turtles. The disadvantage of this is that they are not able to hold their breath as long and are probably more vulnerable to drowning in long, drift gillnet sets.

All of the leatherbacks observed taken by the CA/OR drift gillnet fishery, except for one, were located north of Point Conception, and all were observed taken from September to January, with approximately 60% of the captures occurring in October. The leatherbacks were found in waters with an average monthly sea surface temperature of between 10 to 17.5°C, the majority of them were found in areas of coastal upwelling and some were found on distinct temperature breaks. Only five of the turtles were measured, all between 132 to 160 cm (sub-adults and adults). The rest were most likely too large to be brought on board and measured; therefore, they were probably adults. In addition, based on data compiled from a variety of sources, including published reports, stranding networks, etc., the distribution of leatherbacks with less than 100 cm curved carapace length seem limited to regions warmer than 26°C (Eckert, 1999b).

Leatherbacks caught in the drift gillnet fishery off central and northern California most probably originated from offshore portions of 13-15°C isotherms pushed in-shore in the late summer (Stinson, 1984, *in* Eckert, 1993). The highest density of leatherback sightings on the U.S. West Coast is in and around Monterey Bay, with a peak in sightings in August (Starbird, *et al.*, 1993). In this area, north of Point Conception, major upwelling begins in the spring, when the inverted bottom water is often 3° to 5°C colder than the sun-warmed surface water it replaces. By summertime, seawater temperatures are relatively cold compared to other areas in the same latitude and coastal upwelling generates high productivity, attracting species such as the leatherback, which can tolerate and may favor the highly productive cool coastal waters.

Genetic analyses on a limited number of leatherbacks that have stranded off California and have been incidentally taken by the CA/OR drift gillnet fishery indicate that these turtles originated from western Pacific nesting beaches. Samples from only two of the 23 leatherbacks taken in the drift gillnet fishery were genetically analyzed and found to be representative of nesting turtles from western Pacific beaches (i.e. Malaysia, Indonesia, Solomon Islands). Similarly, all samples taken from stranded leatherbacks on the California coast have indicated representation from western Pacific nesting beaches (Dutton, *et al*., in press, and P. Dutton, personal communication, March, 2000). Lastly, two leatherbacks tagged off of Monterey, California in early September, 2000 appear to be headed towards western Pacific nesting beaches. However, because there has been speculation that leatherbacks caught in the north Pacific high seas drift net fishery in the 1980s and 1990s may have originated from eastern Pacific nesting beaches and that peak sightings of leatherbacks off Monterey in August may correspond to a southern movement to Mexican and Costa Rican breeding grounds (NMFS and USFWS, 1998b; Eckert, 1999a), the possibility that leatherbacks taken in the drift gillnet fishery could have eastern Pacific origins should not be discounted.

The annual average number of leatherbacks taken by the CA/OR drift gillnet fishery has fluctuated during the period from 1990-2000. NMFS can detect no pattern, either environmentally, or fishery-related, to explain these fluctuations in take levels. Therefore, based on a worst-case scenario, NMFS estimates that a maximum of 27 leatherback turtles (5 leatherbacks observed taken in 1995/572 sets observed in = 0.009 turtles per set; 3,000 sets expected per year) in a given year could be incidentally taken by the CA/OR drift gillnet fleet. Assuming that 61 percent of these captured leatherback turtles would be killed accidentally or injured (based on the survival rate (39 percent) of leatherback turtles caught by the CA/OR drift gillnet fleet from 1990-2000), NMFS estimates that as many as 17 leatherbacks could be killed by the CA/OR drift gillnet fleet annually. Based on overall fishery performance observed from July, 1990 to January, 2000, NMFS estimates that, on average, 13 leatherbacks ((23 leatherbacks observed taken from 1990-2000/5,580 sets observed from 1990-2000) x 3,000 maximum expected sets per year) would be captured annually by the CA/OR drift gillnet fleet. These leatherbacks would be expected to have a survival rate of 39 percent (5 would live, 8 would die), based on a calculation of past captures and resultant mortalities (9 leatherbacks survived of 23 leatherbacks captured between July 1990 and January 2000).

c. *Loggerhead Impacts*

The observed incidental take of loggerhead turtles by the CA/OR drift gillnet fishery is infrequent, although they were the second most common sea turtle species caught since the fishery was observed by NMFS in 1990. This may be due in part because loggerheads are rarely seen in the eastern Pacific north of Baja California, Mexico. Prior to the implementation of the PCTRP, from July, 1990 to October 30, 1997, observers recorded the incidental take of 13 loggerheads (3 unidentified, assumed to be loggerheads). Of these 13 turtles, 3 were killed (23%), and the rest (10) were released alive (77%). Since the implementation of the PCTRP, 4 loggerheads have been incidentally taken, with 1 killed (25%), 1 injured (25%), and 2 caught and released alive, uninjured (50%). Therefore, from June, 1990 to January, 2000, a total of 17 loggerheads have been taken, with 12 released alive (70%), 1 injured (6%), and 4 killed (24%).

Loggerheads do appear to have a higher survival rate when caught, compared to leatherbacks. This may be explained by both their physiology and their behavior. First, loggerheads routinely perform shallower dives and tend to remain longer at depth than leatherbacks. In addition, they have the extraordinary ability to survive many hours of anoxia (inadequate supply of oxygen to the brain), the ultimate determinant of dive endurance. In essence, the anoxic turtle brain is able to maintain adenosine triphosphate levels, essential for energy, and ionic homeostasis by severely reducing its metabolic demands to a level that can be fully met by anaerobic glycolysis (*in* Lutcavage and Lutz, 1997). The stress of trying to escape a net causes oxygen stores to be used up, and anaerobic glycolysis is activated. Leatherback dive behavior is one of more continuous aerobic activity, especially when caught, so their oxygen stores are more likely to be used up more quickly. In addition, loggerheads tend to store more oxygen in their lungs compared to leatherbacks (which store large quantities of oxygen in their blood), allowing for an energetically less expensive transport of oxygen.

Genetic information on loggerheads caught in the Hawaiian longline fishery and in the CA/OR drift gillnet fishery indicate that a majority (at least 95 percent, and 100 percent, respectively) of the turtles originated from nesting areas in Japan (Dutton *et al*., in press; P. Dutton, NMFS, personal communication, March, 2000). In addition, studies of large aggregations of mainly subadult and juvenile loggerheads feeding off the west coast of Baja California have shown these animals to originate from the Japanese nesting stock (Bowen, *et al*., 1995). As mentioned above, it has been postulated that loggerhead developmental migrations in the Pacific may be analogous to what scientists suspect is going on in the Atlantic Ocean: breeding adults appear on one side of the ocean (e.g. Japan); hatchlings disappear after departing from nesting beaches there and, perhaps are transported on the Kuroshio and North Pacific Currents (Bowen, *et al*., 1995), show up as juveniles on the other side of the ocean (e.g. Mexico); they then migrate back to their originating side of the ocean as subadults to complete the cycle (Carr, 1986 *in* Pitman, 1990). Juveniles and subadults prefer pelagic crustaceans and fish to the benthic invertebrates that adult loggerheads prefer, and those loggerheads off Baja most likely feed on the vast swarms of pelagic red crab, which are so abundant at times that they turn the ocean red.

Loggerheads caught by the CA/OR drift gillnet fishery ranged in curved carapace length from 32 to 59 centimeters (average 43 centimeters), with the majority (12/15 measured) under 50 centimeters in length. Therefore, the loggerhead turtles caught in these drift gillnets are most likely early and late pelagic stage juveniles and subadults, which most likely originate from Japan. In addition, since 1990, all of the loggerheads incidentally taken in this fishery were located in a concentrated area south of San Clemente Island, and the majority (9/14) of them were caught in the summertime, when sea surface temperatures are highest. All but three loggerheads were captured in waters with reported average monthly sea surface temperatures of from 18°C to 21°C. Three loggerheads were caught in January, in waters with an average monthly temperature of around 15°C, where CoastWatch in January, 1993 reported "relatively high incidence of red crabs (a southern species) throughout the southern California Bight." More importantly, however, all of the loggerheads were caught during El Niño years (1992-93, and 1997-98), when unusually warm sea surface temperatures and northward flowing equatorial currents bring hundreds of thousands of pelagic red crabs from Baja California north up the coast of California. These planktonic crustaceans were reported in abundance off southern California during the two El Niños in the 1990s (Los Angeles Times, March 15, 1998). Loggerheads taken by the fishery had most likely migrated north from Baja California, Mexico, following their primary food source. No loggerheads were observed taken by the CA/OR drift gillnet fishery in non-El Niño years.

Because loggerheads were only observed taken by the CA/OR drift gillnet fishery during El Niño years (1992-93 and 1997-98), NMFS expects the incidental take of loggerheads by the fishery to occur only during an El Niño year. Based on a worst-case scenario (maximum observed taken in a year from 1990-2000), during an El Niño, NMFS estimates that a maximum of 33 loggerheads ((8 loggerheads observed taken in 1993/728 sets observed in 1993) x 3,000 maximum expected sets per year) in a given year could be incidentally taken by the CA/OR drift gillnet fleet. This assumes that the three unidentified turtles incidentally taken in 1993 were loggerheads. Assuming that 32 percent of these captured loggerhead turtles would be killed accidentally or injured (based on the survival rate of hard-shelled turtles caught by the CA/OR drift gillnet fleet from 1990-2000), NMFS estimates that no more than 11 loggerheads would be killed by the CA/OR drift gillnet fleet. Again, during non-El Niño years, NMFS does not anticipate any take of loggerheads by the CA/OR drift gillnet fishery.

d. *Olive Ridley Impacts*

Although the olive ridley is widely regarded as the most abundant sea turtle in the world, they are very rarely caught in the CA/OR drift gillnet fishery, probably because the olive ridley prefers tropical and warm temperate waters. Of all sea turtle strandings in California from 1990-99, the olive ridley was the sea turtle most rarely found (J. Cordaro, NMFS, personal communication, May, 2000). In the past ten years (1990 - 1999), observers recorded the incidental take of only one olive ridley by the CA/OR drift gillnet fishery, in 1999, off southern California in a fully compliant net (41 pingers, extender length of 36 feet). The turtle was released alive and uninjured.

Olive ridleys caught in the Hawaiian longline fishery were genetically sampled and were found to originate from nesting beaches in the eastern Pacific and the southwest or Indo-Pacific. The olive ridley caught in the CA/OR drift gillnet fishery was found to originate from an eastern Pacific stock, and at 67.5 CCL, it was most likely an adult. Although data are inconclusive as to whether olive ridleys actively migrate or passively drift with surface currents, there is evidence to suggest that many olive ridleys undergo a regular migration within the eastern Pacific between breeding grounds in the north and feeding grounds in the south. In addition, satellite monitoring of post nesting movements showed migration routes traversing thousands of kilometers over deep (>1000 m) oceanic water, distributed over a very broad range, suggesting that olive ridleys are nomadic and exploit multiple feeding areas, rather than migrate to one specific foraging area (*in* NMFS and USFWS, 1998d). Because olive ridleys off western Baja California feed almost entirely on pelagic red crabs (Márquez, 1990, *in* NMFS and USFWS, 1998d), the olive ridley observed entangled off southern California could likely have been feeding on an abundant source of prey. The sea surface temperature in the area on the day the olive ridley was caught was between 17° and 18°C, which is the preferable temperature for these species, and normal for November. Warmer than normal waters south of the olive ridley entanglement could have brought the turtle further north, or there may have been a large abundance of prey due to anomalously strong upwelling in the area at that time, as mentioned previously. Both the fin whale and the green turtle were also observed taken by the CA/OR drift gillnet fishery in November, 1999, and in ten years had never been observed caught by the fishery, further indicating that unusually high forage density during this time period could explain the interaction with the olive ridley sea turtle.

Olive ridleys are rarely taken by the CA/OR drift gillnet fishery. Based on a worst-case scenario, NMFS estimates that a maximum of 6 olive ridleys ((1 olive ridley observed taken in 1999/526 sets observed in 1999) x 3,000 maximum expected sets per year) in a given year could be incidentally taken by the CA/OR drift gillnet fleet. Assuming that 32 percent of these captured olive ridleys would be killed accidentally or injured (based on the survival rate of hard-shelled turtles caught by the CA/OR drift gillnet fleet from 1990-2000), NMFS estimates that no more than 2 olive ridleys would be killed by the CA/OR drift gillnet fleet annually. The only observed take, in 1999, appears to be related to unusual environmental conditions. Therefore, NMFS expects the capture of olive ridley turtles to be a rare event –entanglement, and any associated mortality, of olive ridley turtles is not anticipated to occur every year. Based on past fishery performance, olive ridley turtles were observed taken once in ten years, or once during the three years the PCTRP has been in place. Erring conservatively for the species, takes of olive ridley turtles could occur this frequently again (once in three years), resulting in a total expected impact to olive ridley turtle populations of 6 turtles entangled, including 2 killed, during the three year period of the proposed action.

VI. CUMULATIVE EFFECTS

Cumulative effects include the effects of future State, tribal, local, or private actions that are reasonably certain to occur in the action area considered in this Opinion. Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

Most of the fisheries described as occurring within the action area (section IV. Environmental Baseline), are expected to continue as described into the foreseeable future. Therefore, NMFS is not aware of any proposed or anticipated changes in these fisheries that would substantially change the impacts each fishery has on the marine mammals and sea turtles covered by this Opinion. Vessels participating in the California longline fishery, however, appear to be increasing due to the influx of Hawaii-based longliners targeting swordfish in waters 200 nm off the California coast as a result of a recent court injunction. Therefore, interactions between listed species commonly found in this area and the California longline fishery may increase. Because this fishery is not observed, the current level of incidental take of listed marine mammals and sea turtles is unknown.

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 marine mammals and sea turtles covered by this Opinion. Therefore, NMFS expects that the levels of incidental take of marine mammals and sea turtles described for each of the fisheries including the CA/OR driftnet beyond the 3 year permit issuance. See earlier comment in baseline (except the California longline fishery) and non-fisheries will continue into the foreseeable future.

VII. INTEGRATION AND SYNTHESIS OF EFFECTS

This section provides a summary of the anticipated impacts marine mammals and sea turtles will face in the future. It is based on information provided in the *Status of the Species, Environmental Baseline*, and *Effects of the Action* sections of this Opinion. The intent of this discussion is to provide context for the impacts of the continuing California/Oregon drift gillnet fishery and an analysis of whether the proposed action will appreciably reduce the likelihood of survival and recovery of the affected marine mammal and sea turtle species.

As defined by the ESA, an action is likely to jeopardize a listed species if that action 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). Several terms within this definition require further explanation. The term 'appreciably' is not defined in the statute or NMFS' implementing regulations (16 USC § 1531). However, NMFS is directed to provide the "benefit of the doubt" to the listed species (House of Representatives Conference Report No. 697). This direction, often termed the "precautionary principle" requires NMFS to base its determinations on the most conservative approach for listed species populations. Definitions of the terms "survival" and "recovery" are also helpful to this discussion. "Survival" is defined by the U.S. Fish and Wildlife Service's and NMFS' joint Consultation Handbook, *Procedures for Conducting Consultation and Conference Activities Under Section 7 of the Endangered Species Act*, as the species' persistence as listed or as a recovery unit, beyond the conditions leading to its endangerment, with sufficient resilience to allow for the potential recovery from endangerment. "Recovery" is defined at 50 CFR § 402.02 as an improvement in the status of listed species to the point at which listing is no longer appropriate under the criteria set out in Section 4(a)(1) of the ESA.

Conservation Considerations

Over the short-term, the survival of listed populations of marine mammals and sea turtles will largely depend on their ability to retain sufficient abundances that enable the populations to persist in the face of random events that could drive them to extinction. Chance events operate at several levels that affect the likelihood of extinction, including demographic, environmental, and genetic stochasticities. Listed species populations, because they are defined as either in danger of becoming extinct (endangered) or likely to become endangered in the foreseeable future (threatened), are typically very small populations.

When populations become small, there is concern that changes in population dynamics can take place which make the populations more susceptible to extinction and less able to recover. One example is a decline in the reproductive success due to a decrease in population size, which is variously known as depensation, an Allee effect, and inverse density dependence. Average productivity may decline due to a skewed sex ratio, or from decreasing spatial and temporal overlap between males and females. Such depensatory dynamics in a population where abundance has been severely reduced may preclude the population from recovering, even when mortality is reduced.

Genetic risks include the loss of genetic variation in a population, which results in decreased fitness through random genetic drift (Primack 1993). A population remains viable when it maintains sufficient genetic variation for evolutionary adaptation to a changing environment. The genetically effective population size[11] conveys information about expected rates of inbreeding and genetic drift, which can affect fitness and adaptive potential (Hedrick and Miller 1992 *in* Meffe and Carroll 1997).

Several minimum effective population sizes have been proposed as general recommendations for species to maintain population numbers and genetic variation (Franklin 1980; Lande and Barrowclough 1987). An effective population size (including males and females) of 50 has been prescribed to prevent inbreeding depression (Franklin 1980). An effective population size of 500 has been recommended to avoid long-term genetic losses (Franklin 1980; Lande and Barrowclough 1987), which is considered the primary threat to the loss of genetic variation essential to continuing adaptation. While these are merely 'rules of thumb' and the necessary sizes will vary from species to species, it has been strongly recommended that effective population sizes of at least hundreds of individuals be maintained to preserve evolutionarily important amounts of genetic variation (Lande and Barrowclough 1987). The effective population size of a population is often substantially less than the total number of individuals in the population (Primack 1993).

Primack (1993) further wrote:

> "The smaller a population becomes, the more vulnerable it is to demographic variation, environmental variation, and genetic factors that tend to reduce population size even more and drive the population to extinction. This tendency of small populations to decline towards extinction has been likened to a vortex effect (Gilpin and Soule 1986). For example, a natural catastrophe, environmental variation, or human disturbance could reduce a large population to a small size. This small population could then suffer from inbreeding depression, with an associated lower juvenile survival rate. This increased death rate could result in an even lower population size and even more inbreeding. Similarly, demographic variation will often reduce population size, resulting in even greater demographic fluctuations and a greater probability of extinction. These three factors–environmental variation, demographic variation, and loss of genetic viability–act together so that a decline in population size caused by one factor will increase the vulnerability of the population to the other factors."

Long lived marine species may be particularly vulnerable to human perturbations which increase mortalities at all life stages. Annual survival rates of some stages, particularly large juveniles and adults, may be extremely critical to population maintenance and recovery. Species with delayed maturity, such

[11]Genetically effective population size is the functional size of a population, in a genetic sense, based on the numbers of actual breeding individuals and the distribution of offspring among families.

as fin whales, male sperm whales, and sea turtles, are vulnerable to increases in mortality of juveniles (sub-adults) and adults – those life stages with the highest reproductive value.

Crouse (1999) found that delayed maturity comes with certain risks. For example, annual survival of juveniles (sub-adults) and adults must be relatively high. Population growth can be very sensitive to changes in survival rates of these individuals which have a higher reproductive value to the population than early life stages. For example, in loggerhead populations where large juveniles outnumber the reproductive adult population, changes in the survival of the large juveniles has long-term implications for population abundance and growth as these age classes succeed older, reproductive age classes. In other words, even if there appears to be a lot of adults in a population, if there are very few juveniles to replace them as the adults die out, the population can collapse. Juvenile and adult survival rates must be sufficiently high to ensure enough juveniles survive to and through their reproductive years to maintain stable populations (Crouse 1999). Even seemingly small numbers of takes, especially of certain life stages, may have negative effects on the population health of long-lived species.

Ultimately, because of the importance of large animals to the populations, protection of breeding grounds or nesting beaches might not be sufficient to stop the decline of these populations. Also, as Congdon *et al.* (1993 *in* Crouse, 1999) note, the same traits that make long-lived species so vulnerable to reduced survival rates also make them slow to recover once depleted, leaving them vulnerable to other threats. For example, while directed harvests and high seas gill nets have been largely mitigated in recent years, depleted leatherback populations are now highly vulnerable to other takes, even if they might otherwise have been able to accommodate such losses (Crouse, 1999)..

A. Marine Mammals

One of the primary goals of the MMPA is to ensure that each marine mammal stock does not have a level of human-caused mortality and injury that is likely to cause the stock to be reduced below the level which results in the maximum net productivity of the population or the species. The 1994 amendments required NMFS to prepare assessment reports for each stock of marine mammal that occurs in waters under its jurisdiction. Such stock assessment reports contain items such as (1) a description of the stock, including its geographic range; (2) a minimum population estimate, maximum net productivity rate, and a description of the current population trend; (3) an estimate of the annual human-caused mortality and serious injury of the stock, and, for a strategic stock (a stock which has a level of human-caused mortality that is likely to cause the stock to be reduced or kept below its optimum sustainable population), other factors that may be causing a decline or impeding recovery of the stock. Such information is used to estimate the number of animals that can be removed from the population without impeding recovery or sustainability.

The Potential Biological Removal (PBR) level is the maximum number of animals, not including natural mortalities, that may be annually removed from a marine mammal stock while allowing that stock to reach or maintain its optimal sustainable population level (OSP). Optimum sustainable population means the number of animals which will result in the maximum productivity of the population or species. The PBR level is the product of the following factors: 1) The minimum population estimate of the stock (N_{MIN}); 2) One-half the maximum theoretical or estimated net productivity rate of the stock at a small population size, where net productivity is the annual per capita rate of increase in a stock resulting from additions due to reproduction, less losses due to mortality ($0.5R_{MAX}$); and 3) A recovery factor (R_F) or "safety factor" of between 0.1 and 1.0 to hasten the recovery of depleted populations and to account for additional uncertainties. The use of PBR as a management scheme is a conservative approach that will allow populations to recover to or remain above OSP.

NMFS does not anticipate that the PCTRP or state regulations governing the operation of the fishery will change in the near future, nor does it anticipate fishermen changing their methods of fishing. Thus, assuming all fishermen fish in compliance with the PCTRP and state regulations, NMFS expects the level of bycatch in the fishery to remain the same since October 30, 1997. The following sections describe possible long-term impacts to each of the affected marine mammal species due to continued operation of the CA/OR drift gillnet fleet under the PCTRP, with unaltered fishing methods, in conjunction with other fishery and non-fishery sources of impact and mortality.

General Effects on Marine Mammals

Because whales are long-lived species, it is often difficult to determine the effects of removing animals from particular life stages or stocks. Large whales may not reach sexual maturity for ten or twenty years, and once of age, females may produce one calf every 2-3 years; therefore, removing one juvenile female may have large repercussions for the entire population, but these effects may not be seen for decades. Furthermore, as numbers of animals decline, the loss of individuals becomes more and more significant. As a long-lived species that produces only a few, often fairly large progeny, large whales have evolved to live within the carrying capacity of their environments, and when that environment changes, they are affected. In addition, from genetic and photographic data collected from different stocks of large whales (e.g. humpback and sperm whale), it appears that there are significant differences between stocks; e.g. a breeding stock off the coast of Mexico will not breed with a stock in Hawaii. Therefore, analyzing the effects of removing individuals from a particular stock that is declining may have far-reaching implications for that stock and may be more conservative from a management perspective than analyzing the removal of an individual from the entire population.

Because Steller sea lions spend part of the breeding season on land and often haul out for extended periods of time, information on abundance, life history, and population structure has been easier for scientists to obtain. What has been difficult, however, is determining the causes for decline in the population over the past three decades. For the eastern stock, the numbers of Steller sea lions in California have declined from historic numbers, and at this time, the causes are unknown. However, decreased prey availability due to fisheries or environmental factors may play a large role, similar to what has been postulated for the western stock. In addition, interactions with fisheries (and fishermen) may also play a role in their decline. Unlike the large whales, which forage over great distances and depths, often in the open ocean, Steller sea lions forage out to the edge of the continental shelf and are therefore not as vulnerable to distant fisheries. Because Stellers need to haul out in order to give birth or to rest, habitat degradation of a rookery or traditional haulout may also impact a population, especially since females frequently return to the same pupping site year after year and often to the same or near the same site of the female's birth. Since Steller sea lions reach sexual maturity at an earlier age (between 3 and 6 years for females) and most breed annually, they may not be as vulnerable to removal of individuals on a population level as the large whales.

In summary, marine mammals in general appear to use variances of one strategy in order to maximize their populations. Females produce only one calf or pup per pregnancy (in most cases), and exhibit strong maternal attendance to ensure the maximum survival rate for their young. Large whales in general have a calving interval of between 2 to 6 years, while Steller sea lions give birth in consecutive years. Both Stellers and the large whales have long lactation periods, which may be a disadvantage, for if mothers and their young are separated for any reason during this critical period (e.g. the mother is taken by a fishery), the survival rate of the abandoned young is greatly reduced, and no new young are produced, intensifying the effect of the removal of one adult female from the population.

1. Fin whale effects

The only known incidental take and mortality of fin whales in the Pacific Ocean by various known commercial fisheries is the CA/OR drift gillnet fishery, based on available data. In addition to fisheries-related mortalities, ship strikes and interactions with non-fishery vessels account for approximately 0.5 fin whale mortalities per year, or one fin whale every two years (J. Cordaro, NMFS, personal communication, May, 2000).

There are currently an estimated 1,236 fin whales in the Pacific Ocean, with a minimum estimate of 1,044 animals. This estimate only includes the California/ Oregon/ Washington stock, because estimates are not available for the Alaska or Hawaiian stocks, or from any eastern tropical Pacific cruises. For this reason, and because fin whales are a cosmopolitan species, migrating over great distances of the ocean, and are not often seen during ship surveys, the abundance of fin whales in the Pacific Ocean may be underestimated. In addition, an increasing trend has been suggested by survey data, although it is not statistically significant.

The fin whale entangled by the CA/OR drift gillnet fishery in 1999 was approximately 20 meters long; therefore, it was probably an adult. Although there did not appear to be any unusual circumstances surrounding the entanglement event, aside from anomalously strong upwelling in the area which may have attracted more marine mammals and sea turtles than normal, capture of fin whales by the CA/OR drift gillnet fishery is rare. Fin whales are commonly found year-round off central and southern California, with a peak in summer and fall, when fishing effort increases by drift gillnetters targeting swordfish. Therefore, although fin whales and the driftnet fishery co-occur, large whales are rarely taken by this fishery, so maximum annual take estimates of fin whales by the fishery are based on estimated worst case impacts during the past ten years.

Based on a worst-case scenario, NMFS estimates that a maximum of 6 fin whales could be captured by the CA/OR drift gillnet fleet and killed over the next three years. With a minimum population estimate of 1,044 fin whales in the CA/OR/WA stock, the CA/OR drift gillnet fleet may accidentally kill 6 fin whales (or approximately 0.57 percent of the affected stock) sometime in the next three years. However, following the cessation of whaling operations, this stock appears to be at least stable and is expected to continue to recover. NMFS expects that the loss of 6 individuals sometime within the next three years will not have an appreciable impact on the numbers or reproduction of this stock. In a separate analysis prepared for the proposed 101(a)(5)(E) permit, NMFS has determined that this level of effect to the stock is negligible (NMFS, 2000). This determination is based on NMFS' use of a conservative recovery factor for this stock which is ten percent of the number of individuals that could be removed from a stock while still allowing it to achieve its optimum sustainable population level. Therefore, this loss is not anticipated to result in detectable effects to the numbers, distribution, or reproduction of the stock, and is not expected to appreciably reduce the likelihood of survival and recovery of the species.

2. Humpback whale effects

Table 13 provides a summary of the estimated rates of annual incidental take and mortality of humpback whales in the Pacific Ocean by various known commercial fisheries, based on available data.

In addition to fisheries-related mortalities, ship strikes and interactions with non-fishing vessels account for approximately 0.8 humpback mortalities per year (J, Cordaro, NMFS, personal communication, May, 2000; Ferrero, *et al*., 2000). Therefore, not including the effects of take by the CA/OR drift gillnet fishery, approximately 4 humpback whales in the Pacific Ocean are killed each year as a result of human interactions.

Table 13. Estimated rates of annual incidental take and mortality for humpback whales due to fisheries-related interactions based on available or extrapolated data.

Fishery	Incidental Take	
Bering Sea/Aleutian Islands groundfish trawl	n/a	0.2
Bering Sea unknown fishery	n/a	$0.2
SE Alaska drift gillnet and purse seine fisheries	n/a	$0.6
Hawaiian longline fishery	n/a	0.1[2]

Alaska and Hawaii fishery-related strandings	n/a	2.0
California salmon troll fishery	>0.2	n/a
CA/OR drift gillnet fishery	6[3]	0[3]

[1]Mortality is a subset of total incidental take
[2]Observed mortality, not estimated because coverage was <1%.
[3]Based on a worst-case scenario. NMFS expects this incidental take and mortality to occur in only one of the next three years.

With a current population estimate of 5,304 humpback whales (4,926 minimum estimate) in the Pacific Ocean and a growth rate of approximately 6-8% per year, humpback whales appear to be recovering from the effects of the whaling era, despite ongoing fishery impacts. Because they tend to associate in more coastal waters, and generally require shallow water for successful calving, humpback whales may be more vulnerable to coastal fisheries and ship traffic than other large whales.

The two humpback whales entangled by the CA/OR drift gillnet fishery were released alive; therefore, length estimates and sex determination were not available. In addition, there did not appear to be any unusual circumstances surrounding the entanglement. Both were caught in areas of localized upwelling, in a year following an El Niño event. Humpback whales are generally seen off California in the summer and fall, having migrated from their winter/spring breeding area in coastal Central America and Mexico. Coastal areas off southern and central California are highly productive due to upwelling, converging currents, and other physical oceanographic factors. Therefore, although the entanglement event was rare, and few large whales are taken by the fishery, humpback whales are found foraging in areas and times of CA/OR drift gillnet fishing effort. Maximum annual take estimates of humpback whales by the fishery are shown in Table 13 and are based on estimated worst case impacts during the past ten years.

Based on a worst-case scenario, NMFS estimates that a maximum of 6 humpback whales could be captured by the CA/OR drift gillnet fleet sometime in the next three years. Based on past fishery performance, all 6 humpback whales are expected to survive. Therefore, the capture and release of 6 humpback whales over the next three years is not anticipated to result in detectable effects to the numbers, distribution, or reproduction of the stock, and is not expected to appreciably reduce the likelihood of survival and recovery of the species.

3. Sperm whale effects

Table 14 provides a summary of the estimated rates of annual incidental take and mortality of sperm whales in the Pacific Ocean by various known commercial fisheries, based on available data.

Table 14. **Estimated rates of annual incidental take and mortality of sperm whales due to fisheries- related interactions based on available or extrapolated data.**

Fishery	Incidental Take	Mortality[1]
Gulf of Alaska longline fishery	0.125[2]	0.0
CA/OR drift gillnet fishery	6[3]	4[3]

[1]Mortality is a subset of total incidental take

[2]From observer data, not extrapolated

[3]Based on a worst-case scenario. NMFS expects this incidental take and mortality to occur in only one of the next three years.

Sperm whales occasionally strand off California (approximately once a year); however, the causes for stranding are usually unknown. Mortality from ship strikes probably goes unreported because sperm whales are generally found far offshore in deeper water and therefore do not strand, or if they do, they do not always have obvious signs of trauma.

Clearly, historic whaling operations were primarily responsible for reducing sperm whales to very low levels. Furthermore, it is unknown whether the population is growing, declining, or stationary. The current population of sperm whales in the CA/OR/WA stock (the stock affected by the CA/OR drift gillnet fishery) is estimated to be 1,191, with a minimum estimate of 992 whales. This is probably an underestimate, since abundance estimates from an eastern tropical Pacific survey (22,666 animals (Wade and Gerrodette, 1993)) and an eastern temperate Pacific survey (39,200 animals based on acoustic detections and visual group size estimates (Barlow and Taylor, 1998)) are not included because it is not known how many or if these whales travel to the higher latitudes, or west, and are therefore part of the California/ Oregon/ Washington stock.

Females attain sexual maturity by around 9 years old, while the males are delayed sexually until they reach 20 years old. Females produce a calf every 3-6 years, and the breeding season off California takes place from April to August. With a longer calving interval than the large baleen whales, female sperm whales may have fewer calves over their lifetime and therefore sperm whale populations may take slightly longer to recover from exploitation than baleen whales. They are uniformly and widely distributed, found year-round off California with peak abundance from April to mid-June and then from the end of August to mid-November, when fishing effort by the CA/OR drift gillnet fishery increases. Because sperm whales forage at great depths for their preferred prey, squid, they may be more vulnerable than the large baleen whales to a suspended driftnet which could hang as deep as 100 meters below the surface of the water.

Because no data on sex determinations were collected on the eight sperm whales caught by the CA/OR drift gillnet fishery, NMFS cannot speculate as to whether the fishery is interacting with a particular sex. However, based on estimated length data (visual) of a few of the sperm whales observed taken, the fishery appears to interact with both subadults and adults. As mentioned, three sperm whales were entangled in one set in 1992, and based on their length estimates, all appeared to be subadults. Two sperm whales (one adult, and one unknown age-class) were also entangled in the same set in 1993, and because adult males tend to be solitary outside of the breeding season, these animals may have been part of a breeding school (adult females and juveniles) or a bachelor school (subadult males). These multiple entanglements also occurred during an El Niño event, when a lack of upwelling forced foraging animals to dive deeper and for a longer period of time to find food, making them more vulnerable to capture by a net suspended below the sea surface. The 6 sperm whales taken outside of Monterey Bay, California, were most likely foraging on the abundant squid, as this is a prime fishing area for squid fishermen, and an area of coastal upwelling. In general, there do not appear to be any unusual circumstances surrounding the entanglements of sperm whales by the CA/OR drift gillnet fishery. Maximum annual take estimates of sperm whales by the fishery are shown in Table 14 and are based on observed performance over the past three years.

Based on past fishery performance, sperm whales were observed taken in four out of the last ten years, but only once during the three years the PCTRP has been in place. NMFS anticipates that sperm whale entanglement could occur once during the three year period of the proposed action, resulting in a total expected impact to the CA/OR/WA sperm whale stock of 6 whales entangled, including 4 whales killed, during the three year period of the proposed action. NMFS believes that the sperm whales captured and released unharmed from the drift gillnets will survive unimpaired. Therefore, the capture and release of 2 sperm whales over the next three years is not expected to affect the status of the affected stock. With a minimum population estimate of 992 sperm whales in this stock[12], the CA/OR drift gillnet fleet may accidentally kill or seriously injure 4 sperm whales (or approximately 0.4 percent of the affected stock) sometime in the next three years. However, following the cessation of whaling operations, this stock appears to be at least stable and is expected to continue to recover. NMFS expects that the loss of 4 individuals sometime within the next three years will not have an appreciable impact on the numbers or reproduction of the stock. In a separate analysis prepared for the proposed 101(a)(5)(E) permit, NMFS has determined that this level of effect to the stock is negligible (NMFS, 2000). This determination is based on NMFS' use of a conservative recovery factor for this stock which is ten percent of the number of individuals that could be removed from a stock while still allowing it to achieve its optimum sustainable population level. Therefore, this loss is not anticipated to result in detectable effects to the numbers, distribution, or reproduction of the stock and is not expected to appreciably reduce the likelihood of survival and recovery of the species.

4. Steller sea lion effects

Table 15 provides a summary of the estimated rates of annual incidental take and mortality of Steller sea lions by various known fisheries, based on available data.

[12] The abundance estimates contained in the most recent stock assessment reports are probably much lower than actual abundance due to minimal time spent sighting animals and consequently missing submerged animals (minutes of PSRG meeting, 5-6 December 1999).

Table 15. **Estimated rates of annual incidental take and mortality for Steller sea lions due to fisheries-related interactions based on available or extrapolated data.**

Fishery	Incidental Take	Mortality[1]
SE Alaska salmon drift gillnet	n/a	$1.25[2]
Alaska salmon troll	n/a	$0.2[2]
BC aquaculture predator control program	n/a	12.4
Northern WA marine set gillnet (tribal fishery)	n/a	0.2
WA/OR/CA groundfish trawl fishery for Pacific whiting	n/a	0.6
CA/OR drift gillnet fishery	5[3]	5[3]

[1]Mortality is a subset of total incidental take

[2]Because these are from fisher self-reports, these values are most likely negatively biased

[3]Based on a worst-case scenario. NMFS expects this incidental take and mortality to occur in only one of the next three years.

In addition, subsistence harvest of Steller sea lions was estimated for 1992-96 and averaged 2 mortalities per year (Ferrero, *et al*., 2000). Even though intentionally killing a marine mammal is illegal, Steller sea lions have been killed by gun-shot wounds, usually because of interactions with fishing gear and/or catch. Stranding records from Oregon, Washington and Alaska from 1990 to 1997 and from California for the period 1995-98 resulted in an estimated annual mortality of 3.3 Steller sea lions due to gunshot wounds (Ferrero, *et al*., 2000; J. Cordaro, NMFS, personal communication, May, 2000). This is considered a minimum since information from British Columbia was not available, and some data from Alaska were not included because it was unclear if the gunshot wounds were attributable to subsistence harvest by Alaska natives. Therefore, not including the effects of capture by the CA/OR drift gillnet fishery, at least 20 eastern stock Steller sea lions are killed annually due to human-related interactions. In addition, changes in environmental conditions such as warmer sea surface temperatures, competition with expanding populations of California sea lions, and bioaccumulation of organochlorides may be correlated with declines in Steller sea lion populations, especially at the southern end of its range.

Females reach sexual maturity between the age of 3 to 6, and most breed annually, producing a pup nearly every year until their early 20s. Therefore, a healthy female could potentially produce nearly 15 offspring during the course of her lifetime. Although the eastern stock of Stellers appears to be increasing in the northern end of its range, and relatively stable in Oregon and northern California, ground counts of pups and non-pups in central California (Año Nuevo) have decreased significantly over the past 20-30 years. Current population estimates for the southeastern Alaskan stock are 14,571 animals, for the British Columbia stock, 9,277 animals, and for the California/Oregon/Washington stock, 6,555 animals, resulting in a total estimate of 30,403 eastern stock Steller sea lions. This is considered a minimum estimate (Ferrero, *et al*., 2000).

The two Steller sea lions entangled by the CA/OR drift gillnet fishery were killed; however, biological data on only one animal were collected. This animal was an adult female. Both animals were entangled in very different areas (offshore southern California and right off the Oregon coast) and at different times of the year (June, 1992 and September, 1994). In addition, there did not appear to be any unusual circumstances surrounding the entanglements. The Steller caught off southern California may have been there due to El Niño conditions, as it was during the breeding season, when it would normally be found near or on the rookery. Although entanglements of Stellers by the CA/OR drift gillnet fishery are rare events, Steller sea lions are found foraging in areas and times of CA/OR drift gillnet fishing effort.

Based on past fishery performance, and despite the fact that the fishery under the PCTRP has not interacted with any Steller sea lions, NMFS conservatively estimates that a maximum of 5 Steller sea lions could be incidentally entangled and killed by the CA/OR drift gillnet fleet during the next three years. The affected sea lions likely come from the California, Oregon, or Washington rookeries, which have a combined minimum population estimate of 6,555 Steller sea lions. Therefore, the CA/OR drift gillnet fleet may accidentally kill less than 0.08 percent of this population over three years. This sub-population appears to be stable overall, and NMFS expects that the loss of 5 individuals will not have an appreciable impact on the numbers or reproduction of this sub-population. In a separate analysis prepared for the proposed 101(a)(5)(E) permit, NMFS has determined that this level of effect to the stock is negligible (NMFS, 2000). This determination is based on NMFS' use of a conservative recovery factor for this stock which is 0.75 percent of the number of individuals that could be removed from a stock while still allowing it to achieve its optimum sustainable population level. Therefore, this loss is not anticipated to result in detectable effects to the numbers, distribution, or reproduction of the species and is not expected to appreciably reduce the likelihood of survival and recovery of this species.

B. Sea Turtles

Long-term analysis of the impacts of the CA/OR drift gillnet fishery on sea turtles, in addition to the impacts of other sources of mortality, is complicated by the lack of information needed for population viability analysis. For example, data on age structure and sex composition of the mortality incidental to the CA/OR drift gillnet fishery and many other fisheries are limited. There is generally little information on survival rate of various age classes of turtles, and the population structure of sea turtles on the fishing grounds is uncertain. However, given the current declines of most sea turtle populations, NMFS assumes that the current survival rate throughout a turtle’s life cycle is not high. In the absence of the population information necessary for a robust analysis of the impact of the proposed action on a sea turtle species, NMFS instead relies upon making conservative assumptions and findings for that species in order to provide the necessary benefit of the doubt to that species and to ensure that we do not erroneously conclude that an action does not jeopardizes a listed species when, in fact, it does.

NMFS does not anticipate that the PCTRP or state regulations governing the operation of the fishery will change in the near future, nor does NMFS anticipate fishermen changing their methods of fishing. Thus, assuming all fishermen fish in compliance with the PCTRP and state regulations, NMFS expects the level of bycatch in the fishery to remain the same since October 30, 1997. The following sections describe possible long-term impacts to each of the affected sea turtle species from operations of the CA/OR drift gillnet fleet, with unaltered fishing methods, in conjunction with other fishery and non-fishery sources of impact and mortality.

1. Green turtle effects

Table 16 provides a summary of the estimated rates of annual incidental take and mortality of green turtles by various known fisheries, based on available data. Although the high-seas driftnet fisheries no longer operate, they may have had relatively high levels of incidental mortality prior to the 1992 moratorium, and since sea turtles are long-lived species, the effects of this fishery may still be felt by the population. The western Pacific and South China Sea longline fisheries capture and kill green turtles; however, the extent of these impacts on the population is unknown. There was a harvest of sea turtles in 1995 in Ecuador and Peru; however, it is unknown if this harvest continues to occur. Therefore, this table does not contain estimates of take, including mortality, from other sources such as non-observed fisheries, habitat degradation, poaching, or direct harvest.

Over the next three years, NMFS has estimated that about 24,237 green turtles (8,079 per year) may be captured, entangled, or hooked by fisheries other than the CA/OR drift gillnet fleet. This estimate could be higher or lower due to unknown capture rates and changes in effort in other fisheries, such as in the Japanese, Korean, and Taiwanese longline fisheries, which may take large numbers of sea turtles, and other fisheries described in Appendix A. Of the 837 green turtles captured, approximately 222 (or 74 per year) may be killed over the next three years.

In addition, an unknown number of green turtles may be injured or killed from non-fishery related impacts such as disease, direct harvest, egg poaching, vessel collisions, or ingestion of or entanglement in debris. Habitat impacts, including loss of nesting sites or degradation of nesting or foraging areas are also expected to continue. Quantitative data on the extent of these impacts to green turtle populations are lacking. Many green turtles in Hawaii are also afflicted with fabropapilloma, which is often fatal.

Table 16. **Estimated rates of annual incidental take and mortality for green turtles based on available or extrapolated data from fisheries known to take green turtles.**

Fishery	Incidental Take	
North Pacific driftnet	378	
Foreign ETP purse seine fleet[2]	150	15.0
Chilean artisanal driftnet fleet[3]	39	39

Mexican (Baja CA) fisheries and direct harvest	no data	>7,800
U.S. ETP purse seine fleet[4]	35	2
Hawaiian longline[5]	52	15
California set gillnet	no data	2.7
California longline	2.7	0
CA/OR drift gillnet fishery	6[6]	2[6]

[1]Mortality is a subset of total incidental take

[2]Based on 1994-98 data for the entire fleet with the mortality estimates for the U.S. fleet subtracted. Incidental Take is back-calculated from mortality assuming a 10 percent mortality rate.

[3]Based on projected take estimates for 1996 effort data, and assuming all takes were mortalities.

[4]Based on 1992-97 data, expressed as an annual rate of mortality.

[5]This a maximum estimate based on past fishery performance. Future takes may be lower due to expected changes in the fishery.

[6]Based on a worst-case scenario. NMFS expects this incidental take and mortality to occur in only one of the next three years.

= pre-12/92

Green turtles that could be entangled by the CA/OR drift gillnet fishery most likely originate from Hawaii or the Pacific coast of Mexico. A small (50-60) group of green turtles residing in San Diego Bay, California, show characteristics of both the Mexican and the Hawaiian breeding population (NMFS and USFWS, 1998a). The one green turtle that has been observed taken by this fishery originated from Mexico; therefore, it was an endangered eastern Pacific green. Population estimates for the entire species are not available. The last reported estimate for one of the major nesting areas in North America, Michoacán, Mexico at Colola Beach, was 600 nesting females in 1998-1999 (down from over 5,000 at the two major beaches in the early 1980s). Current estimates are not available for Maruata Beach; however, Colola Beach has been responsible for approximately 70 percent of total green nesting in Michoacán. At the French Frigate Shoals in Hawaii, nesting populations of threatened green turtles are estimated at 200-700 females annually. The population here has tripled since enactment of the Endangered Species Act in 1973.

NMFS expects the capture of green turtles by the CA/OR drift gillnet fishery to be a rare event – entanglement, and any associated mortality, of green turtles is not anticipated to occur every year. The CA/OR drift gillnet fleet is anticipated to capture/entangle an additional 6 green turtles off the California/Oregon coast over the next three years. Of these 6 captured green turtles, 2 may be killed over the next three years. NMFS assumes that the turtles captured and released unharmed from the drift gillnets will survive unimpaired. Therefore, the capture and release of 4 green turtles over the next three years is not expected to affect the status of the green turtle populations.
The CA/OR drift gillnet fleet may accidentally kill no more than 2 green turtles sometime over the next three years. Removal of 2 green turtles from the Mexican population represents a loss of less than 0.3

percent of the population, while taking 2 greens from the Hawaiian population represents a 0.29 to 1 percent loss to the population. These are conservative estimates, however, because the take of green turtles in the fishery is likely not limited to adult females, the only segment of the population for which NMFS has population numbers. Even if both turtles were reproductive females, this loss is not anticipated to have a detectable effect on the numbers or reproduction of the affected sub-populations, and therefore is not expected to appreciably reduce the likelihood of survival and recovery of the species.

2. Leatherback turtle effects

Table 17 provides a summary of the estimated rates of annual incidental take and mortality of leatherbacks by various known fisheries, based on available data. Although the high-seas driftnet fisheries no longer operate, they evidently had relatively high levels of incidental mortality prior to the 1992 moratorium, especially the Taiwanese large-mesh driftnet fleet. The full effects of the high-seas driftnet fishery on leatherbacks are unknown although they are believed to be significant (Wetherall, *et al*., 1993). The western Pacific and South China sea longline fisheries may also impact leatherbacks, as there are reported sightings of them in the area; however, the current and past extent of these fisheries is unknown. Mexican driftnet and longline fisheries also likely take leatherbacks, but NMFS has no data on the extent of captures or mortalities of leatherbacks in these fisheries. There was a harvest of sea turtles in 1995 in Ecuador and Peru; however, it is unknown if this harvest continues to occur. Lastly, the Chilean and Peruvian swordfish fisheries may take more leatherbacks than has been estimated in the past; however, the extent of this take is currently unknown. Therefore, this table does not contain estimates of take, including mortality, from other sources such as habitat degradation, poaching, or direct harvest.

Over the next three years, NMFS has estimated that at least 2,866 leatherbacks may be captured, entangled, or hooked by fisheries other than the CA/OR drift gillnet fleet (955 per year). This estimate could be higher or lower due to unknown capture rates and changes in effort in other fisheries, such as in the Japanese, Korean, and Taiwanese longline fisheries, which may take large numbers of sea turtles, and other fisheries described in Appendix A. Of the 2,866 leatherbacks captured, approximately 2,163 (or 721 per year) may be killed over the next three years. In addition, an unknown number of leatherbacks may be injured or killed from non-fishery related impacts such as direct harvest, vessel collisions, or ingestion of debris. Habitat impacts, including loss of nesting sites or degradation of nesting or foraging areas are also expected to continue. Quantitative data on the extent of these impacts to leatherback turtle populations are lacking.

Table 17. **Estimated rates of annual incidental take and mortality for leatherbacks by known fisheries based on available or extrapolated data.**

Fishery	Incidental Take

Chilean and Peruvian fisheries[2]	500	500
Maluku, Indonesia - directed take	200	200
Micronesian longliners	0.3	0
Foreign ETP purse seine fleet[3]	1	0.10
U.S. ETP purse seine fleet[4]	2	0.10
Hawaiian longline fishery[5]	244	19
California set gillnet	no data	1.3
California longline	8	0.5
CA/OR drift gillnet fishery	13 (27)[6]	8 (17)[6]

[1]Mortality is a subset of total incidental take

[2] Source (Montero, personal communication, 1997, *in* Eckert, 1997)

[3]Based on 1994-98 data for the entire fleet with the mortality estimates for the U.S. fleet subtracted. Incidental take is back-calculated from mortality assuming a 10 percent mortality rate.

[4]Based on 1992-97 data, expressed as an annual rate of mortality.

[5]This a maximum estimate based on past fishery performance. Future takes may be lower due to expected changes in the fishery.

[6]Based on CA/OR drift gillnet fishery performance from 1990-2000. Number in parentheses is a worst-case based on the year with the highest capture rate (1995).

= pre-12/92

Two out of the 23 leatherbacks taken by the CA/OR drift gillnet fishery were genetically tested and found to originate from western Pacific nesting beaches (i.e. probably Indonesia, Solomon Islands, or Malaysia). Leatherbacks taken by the fishery off California and Oregon may also originate from the nesting beaches of the Pacific coast of Mexico, based on speculations by NMFS and USFWS (1998a) and Eckert (1999a). In addition, five leatherbacks taken were measured and found to be subadults and adults. Populations of leatherbacks are declining at all major Pacific basin rookeries. A once major nesting beach on Irian Jaya (Jamursba-Medi) has declined from at least 2,300 nesting females in 1984 to at least less than 1,200 nesting females in 1996 (Suarez, *et al*., in press). Similarly, in Terengganu, Malaysia, only 2 nesting females were observed in 1994, down from over 3,000 in the late 1960s (Chan and Liew, 1996). There are no numbers available from the Solomon Islands. The decline of leatherback is equally as dramatic off Mexico and Costa Rica. Since the early 1980s, the eastern Mexican population of adult female leatherbacks has declined from 70,000 in 1982 (Spotila *et al*., 1996) to less than 1,000 in 1999-2000 (Sarti *et al*., personal communication, 2000). At Las Baulas National Park, Costa Rica, the number of nesting leatherbacks has declined from 1,500 in 1988-1989 to 193 in 1993-1994 (Steyermark *et al.*, 1996). During the 1988-89 season (July-June), 1,367 leatherbacks nested on Playa Grande (in Las Baulas), and by the 1998-99 season, only 117 leatherbacks nested. Clearly all leatherback populations are facing threats throughout all age classes, from illegal harvests and egg poaching while on land, to fisheries interactions while migrating.

Leatherbacks appear to reach sexual maturity between 13 and 14 years of age: an earlier age than the hard-shelled turtles . Females lay 1-11 clutches of 65-85 yolked eggs, depending on the nesting location, generally every 2 to 4 years. It is not known how long leatherbacks live, but the earlier age of recruitment into adulthood may be an advantage to a declining species such as the leatherback.

Based on fishery performance observed from July, 1990 to January, 2000, the CA/OR drift gillnet fleet is anticipated to capture/entangle an average of 13 leatherbacks per year off the California/Oregon coast. Of these 13 captured leatherbacks, 8 may be killed or seriously injured per year. In 1992, 1995 and 1997, however, the CA/OR drift gillnet fishery was estimated to take more than 13 leatherbacks per year. For example, the fishery was estimated to take 33 leatherbacks in 1995 (5 leatherbacks observed taken/572 total sets observed in 1995 x 3,673 total sets). There does not appear to be any environmental anomaly or unusual circumstance that can explain these higher take levels. Therefore, in a worst-case scenario, NMFS anticipates that the CA/OR drift gillnet fishery could capture up to 81 leatherbacks (27/year, based on a maximum capture rate of 0.009 turtles per set x 3,000 expected sets per year) over the next three years. Of these 81 leatherbacks, 51 may be killed or seriously injured over the next three years (17 per year). NMFS assumes that the turtles captured and released unharmed from the drift gillnet will survive unimpaired, which is not expected to affect the status of leatherback populations.

Pacific leatherback populations are declining at all Pacific Ocean nesting beaches, most likely as a result of a combination of environmental variation (including natural and anthropogenic factors), demographic variation, and a loss of genetic viability. Most of the leatherbacks taken by the CA/OR drift gillnet fishery likely originate from the western Pacific stock. The populations at two major nesting beaches for this population are currently estimated at approximately 1,163 -1,533 females at Jamursba-Medi Beach and 128-169 females at War-Mon Beach in Irian Jaya. These populations lack adequate protection of nesting beaches, emerging hatchlings, and adults, and therefore these populations are expected to continue their decline – even seemingly small additional impacts to this population, particularly to large juvenile and adult life stages, could have negative effects. Annual survival rates of some stages, especially large juveniles and adults may be critical to population persistence and recovery. Species with delayed maturity are vulnerable to increases in mortality of juveniles and adults – those life stages with the highest reproductive value (National Research Council, 1990).

Therefore, any additional impacts to the western Pacific leatherback stocks are likely to maintain or exacerbate the decline in these populations. This would further hinder population persistence or attempts at recovery as long as mortalities exceed any possible population growth, which appears to be the current case, appreciably reducing the likelihood that western Pacific leatherback populations will persist. Additional reductions in the likelihood of persistence of western Pacific stocks are likely to affect the overall persistence of the entire Pacific Ocean leatherback population by reducing genetic diversity and viability, representation of critical life stages, total population abundance, and metapopulation resilience as small sub-populations are extirpated. These effects would be expected to appreciably reduce the likelihood of both the survival and recovery of the Pacific Ocean population of the leatherback sea turtle.

3. Loggerhead turtle effects

Table 18 provides a summary of the estimated rates of annual incidental take and mortality of loggerheads by various known fisheries, based on available data. Although the high-seas driftnet fisheries no longer operate, they had relatively high levels of incidental mortality prior to the 1992 moratorium, especially the Taiwanese large-mesh driftnet fleet. The full effects of the high-seas driftnet fishery on loggerheads are unknown, although they are believed to be significant (Wetherall, *et al.*, 1993). In addition, the current and past extent of the effects of the western Pacific and South China Sea longline and bottom trawl fisheries on loggerheads is unknown, although they have been reported captured and killed in these fisheries. Lastly, very little is known about the incidental take of loggerheads by the Mexican swordfish fisheries (gillnetters and longliners), although if fishing effort occurs in the waters off Baja California where large aggregations of juveniles have been reported, their impact on the loggerhead population could be significant. There was a harvest of sea turtles in 1995 in Ecuador and Peru; however, it is unknown if this harvest continues to occur. Therefore, this table does not contain estimates of take, including mortality, from other sources such as non-observed fisheries, habitat degradation, poaching, or direct harvest.

Over the next three years, NMFS has estimated that at least 7,512 loggerheads (or 2,504 per year) may be captured, entangled, or hooked by fisheries other than the CA/OR drift gillnet fleet. This estimate could be higher or lower due to unknown capture rates and changes in effort in other fisheries, such as in the Japanese, Korean, and Taiwanese longline fisheries, which may take large numbers of sea turtles, and other fisheries described in Appendix A. Of the 1,662 loggerheads captured, approximately 342 (or 114 per year) may be killed over the next three years. In addition, an unknown number of loggerheads may be injured or killed from non-fishery related impacts such as direct harvest, vessel collisions, or entanglement or ingestion of debris. Habitat impacts, including loss of nesting sites or degradation of nesting or foraging areas are also expected to continue. Quantitative data on the extent of these impacts to loggerhead turtle populations are lacking, and there is little information on ongoing recovery efforts for this species.

Genetic data from loggerheads caught by the CA/OR drift gillnet fishery (100% of samples - four loggerheads) as well as by the Hawaii longline fleet (95%) and the now defunct North Pacific high-seas driftnet fisheries indicate that these turtles originated from Japanese nesting populations. However, loggerheads reported from the insular Pacific, including states and territories under U.S. jurisdiction probably also derive from populations genetically affiliated with Indonesia or eastern Australia, in addition to Japan (NMFS and USFWS, 1998c). Population estimates for the entire species are not available. It has been recently estimated that approximately 1,000 females nest annually in Japan, while only 300 females were thought to nest annually in Queensland, Australia (1998 Draft Recovery Plan for

Marine Turtles in Australia). There is no current information on the numbers of loggerheads nesting in Indonesia.

Table 18. **Estimated rates of annual incidental take and mortality for loggerheads based on available or extrapolated data.**

Fishery	Incidental Take	Mortality[1]
North Pacific driftnet	2,986	805
Chilean artisanal driftnet fleet[2]	4	4
Mexican (Baja CA) fisheries and direct harvest	no data	1,950
Foreign ETP purse seine fleet[3]	52.5	5.25
U.S. ETP purse seine fleet[4]	3	0.15
Hawaiian longline[5]	489	103
California set gillnet	no data	1.3
California longline	4.8	0
CA/OR drift gillnet	10 (33)[6]	3 (11)[6]

[1]Mortality is a subset of total incidental take.
[2]Based on projected take estimates for 1996 effort data, and assuming all takes were mortalities
[3]Based on 1994-98 data for the entire fleet with the mortality estimates for the U.S. fleet subtracted. Incidental take is back-calculated from mortality, assuming a 10 percent mortality rate.
[4]Based on 1992-97 data, expressed as an annual rate of mortality.
[5]This a maximum estimate based on past fishery performance. Future takes may be lower due to expected changes in the fishery.
[6]Based on CA/OR drift gillnet fishery performance from 1990-2000. Number in parentheses is a worst-case based on the year with the highest capture rate (1993).

= pre-12/92

Because El Niño recurs at irregular intervals, ranging from two to seven years, and the last El Niño ended in the late fall of 1998, the Pacific Ocean could potentially experience an El Niño event in the next three years. As of October, 2000, the statistical model forecast of the Climate Prediction Center, of the National Weather Service predicts near-normal conditions remaining through October, 2001. Therefore, in a worst case scenario, over the next three years, the Pacific Ocean could experience two years of El Niño conditions. Therefore, the CA/OR drift gillnet fishery could incidentally take up to 66 loggerheads over the next three years (0 loggerheads taken in year one, 33 loggerheads taken each following year, given years two and three are El Niño years). Of these 66 captured loggerheads, 44 would be released alive, and 22 loggerheads would be killed. NMFS expects that the 44 loggerheads released alive will survive unimpaired.

Loggerheads reach sexual maturity between approximately 25 and 38 years of age. Once mature, females re-migrate to the nesting areas every 2 to 3 or more years, and deposit 1 to 6 clutches (average

2 in Japan) of 110 to 130 eggs per clutch. Like other long-lived sea turtles, loggerheads delay maturity to allow individuals to grow larger and produce more offspring. Evidence of this fecundity are the "tens of thousands" of loggerheads off Baja California, most likely the offspring of females nesting in Japan or Australia, representing three life history stages, small juveniles, large juveniles, and subadults, all of whom have not yet reached reproductive age. More offspring may compensate for high natural mortality in the early life stages; i.e. mortality rates of eggs and hatchling are generally high and decrease with age and growth. The risks of delayed maturity are that annual survival of the later life stages must be high in order for the population to grow. Crouse *et al*. (1987 *in* Crouse, 1999) developed a stage-based matrix projection population model for loggerheads in the southeastern U.S. (which have similar reproduction, although females appear to reach sexual maturity at an earlier age than Pacific loggerheads (Frazer, 1983, *in* NRC, 1990)) which showed that population growth was highly sensitive to changes in annual survival of the juvenile and adult stages. Crouse (1999) reports, "Not only have large juveniles already survived many mortality factors and have a high reproductive value[13], but there are more large juveniles than adults in the population. Therefore, relatively small changes in the annual survival rate impact a large segment of the population, magnifying the effect."

Pacific loggerhead populations are most likely declining at all Pacific Ocean nesting beaches, probably as a result of a combination of natural and anthropogenic factors, demographic variation, and a loss of genetic viability. Most of the loggerheads taken by the CA/OR drift gillnet fishery may originate from the Japanese nesting population. This population is currently estimated at approximately 1,000 females and one trend site has showed a steady decline since the mid-1980s, indicating that even seemingly small additional impacts to this population, particularly to juvenile life stages, could have costly effects. Although every life stage and age of a loggerhead has value, population modeling of this species has shown that protection of large juveniles has the greatest effect on increasing the future growth of the population.

NMFS anticipates that the CA/OR drift gillnet fishery will not take any loggerheads during non-El Niño years. Loggerheads have not been observed taken in non-El Niño years (based on observer data from 1990-2000). However, if El Niño conditions arise, NMFS anticipates that the CA/OR drift gillnet fishery could take 33 loggerheads annually. This is based on a worst case scenario, when, during 1993, the fishery took an estimated maximum number of loggerheads. Of these 33 captured loggerheads, 22 may be released alive, and 11 may be seriously injured or killed. NMFS assumes that the turtles captured and released unharmed from the drift gillnets will survive unimpaired. The capture and release of 22 loggerheads in an El Niño year is not expected to affect the status of loggerhead populations.

[13]The value of an individual of a particular age or life stage can be stated according to its expected production of offspring, or "reproductive value." Reproductive value thus is the relative contribution of an individual of a given age to the growth rate of the population.

The additional loss of 22 juvenile or sub-adult loggerheads from the Japanese stock over the next three years (if El Niño conditions occur over two of the next three years, 11 loggerheads per year is likely to maintain or exacerbate ongoing declines in this population. This would further hinder population persistence or attempts at recovery as long as mortalities exceed any possible population growth, which appears to be the current case, appreciably reducing the likelihood that western Pacific loggerhead populations will persist. Additional reductions in the likelihood of persistence of western Pacific stocks are likely to affect the overall persistence of the entire Pacific Ocean loggerhead population by reducing genetic diversity and viability, representation of critical life stages, total population abundance, and metapopulation resilience as small sub-populations are extirpated. These effects would be expected to appreciably reduce the likelihood of both the survival and recovery of the Pacific Ocean population of the loggerhead sea turtle.

4. Olive ridley impacts

Table 19 provides a summary of the estimated rates of annual incidental take and mortality of olive ridley turtles by various known fisheries, based on available data. The South China Sea, western Pacific longliners and bottom trawlers, and Mexican fisheries may incidentally take olive ridleys; however, the extent of the take is unknown. There was a harvest of sea turtles in 1995 in Ecuador and Peru; however, it is unknown if this harvest continues to occur. Therefore, this table does not contain estimates of take, including mortality, from other sources such as non-observed fisheries, habitat degradation, poaching, or direct harvest.

Over the next three years, NMFS has estimated that at least 4,251 olive ridleys (or 1,417 per year) may be captured, entangled, or hooked by fisheries other than the CA/OR drift gillnet fleet. This estimate could be higher or lower due to unknown capture rates and changes in effort in other fisheries, such as in the Japanese, Korean, and Taiwanese longline fisheries, which may take large numbers of sea turtles, and other fisheries described in Appendix A. Of the 4,251 olive ridleys captured, approximately 573 (or 191 per year) may be killed. In addition, an unknown number of olive ridleys may be injured or killed from non-fishery related impacts, such as direct harvest or ingestion of debris. Habitat impacts, including loss of nesting sites or degradation of nesting or foraging areas are also expected to continue. Quantitative data on the extent of these impacts to olive ridley turtle populations are lacking.

Like the green turtle, the olive ridley turtle is the turtle least likely to interact with the CA/OR drift gillnet fishery, based on their distribution and habits. Olive ridleys are generally found in tropical and warm temperate waters, tend to bask at the surface and regularly associate with floating objects. Based on genetic data obtained from the one olive ridley observed taken in this fishery, this turtle originated from an eastern Pacific stock, most likely from Mexico or Costa Rica, where the largest nesting concentrations occur. No unusual circumstances surrounded this entanglement, although it was taken in an area and during a time of high upwelling, where large whales and sea turtles were probably attracted to the high nutrient level in the area, and species which had not been observed taken in the fishery in ten years were observed entangled.

Table 19. **Estimated rates of annual incidental take and mortality for olive ridleys based on available or extrapolated data. This table does not contain estimates of take, including mortality, from other sources such as habitat degradation, poaching, or direct harvest.**

Fishery	Incidental Take	Mortality[1]
Micronesian longliners	0.5	0.2
Micronesian purse seiners	0.3	0.2
Chilean artisanal driftnet fleet[2]	29	29
Foreign ETP purse seine fleet[3]	1,082	108.2
U.S. ETP purse seine fleet[4]	133	7
Hawaiian longline[5]	168	46
California longline	4.3	0
CA/OR drift gillnet fishery	6[6]	2[6]

[1]Mortality is a subset of total incidental take
[2]Based on projected take estimates for 1996 effort data, and assuming all takes were mortalities
[3]Based on 1994-98 data for the entire fleet with the mortality estimates for the U.S. fleet subtracted. Incidental take is back-calculated from mortality assuming a 10 percent mortality rate.
[4]Based on 1992-97 data, expressed as an annual rate of mortality.
[5]This a maximum estimate based on past fishery performance. Future takes may be lower due to expected changes in the fishery.
[6]Based on a worst-case scenario. NMFS expects this incidental take and mortality to occur in only one of the next three years.

The eastern Pacific olive ridley nesting populations have increased in the last few years, most likely as a result of the ban on direct harvest in Mexico in 1990. The most recent data indicate that over 160,000 females nest annually along the west coast of Mexico (NMFS and USFWS, 1998d), while annual nesting at the principal beach, Playa Escobilla, has increased from an annual average of 138,000 nests prior to the ban, to 525,000 nests currently (Salazar, *et al*., in press). Many of these nesting areas support large, seasonal *arribadas*, where egg and hatchling mortality is high. Costa Rica supports the largest nesting aggregations of olive ridleys in the eastern Pacific. The nesting population at Playa Nancite appears to be declining with 25,000-50,000 turtles per year, and no tend data are available at Playa Ostional where 450,000-600,000 turtles nest annually. Average annual hatch success ranges from 3.0 to 22 percent. Olive ridleys also nest in Central America, in smaller numbers (*in* NMFS and USFWS, 1998d).

Age at sexual maturity for the olive ridley is unknown; however, the average length of turtles nesting in Costa Rica was 63.3 cm, with the smallest being 54.0 cm (*in* NMFS and USFWS, 1998d). At 67.5 cm, the olive ridley taken in the CA/OR drift gillnet fishery was probably an adult. Females generally deposit two clutches of eggs per year, each producing 100-110 eggs on each occasion. This genus is also unique in that females commonly nest each year, without intervening non-breeding seasons, as shown by the leatherback and the other hard-shelled turtles. In addition, while the tremendous reproductive output of *arribadas* may help to recover the olive ridley population, the excessive egg loss and subsequent decline in reproductive output suggests that the *arribada* beaches may fall short of their reproductive potential and should not be held primarily responsible for maintaining the population (*in* NMFS and USFWS, 1998d).

NMFS expects the capture of olive ridley turtles to be a rare event –entanglement, and any associated mortality, of olive ridley turtles is not anticipated to occur every year. Based on past fishery performance, olive ridley turtles were observed taken once in ten years, or once during the three years the PCTRP has been in place. Erring conservatively for the species, takes of olive ridley turtles could occur this frequently again (once in three years), resulting in a total expected impact to olive ridley turtle populations of 6 turtles entangled, including 2 killed, during the three year period of the proposed action. NMFS believes that the turtles captured and released unharmed from the drift gillnets will survive unimpaired. Therefore, the capture and release of 4 olive ridleys over the next three years is not expected to affect the status of olive ridley populations. The mortality of olive ridley turtles incidental to the CA/OR drift gillnet fishery, estimated to be 2 olive ridley turtles every 3 years, represents approximately 0.0013 percent of the Mexican nesting population (2/160,000), and 0.0004 percent of the two major nesting beaches in Costa Rica (2/475,000). This is a conservative estimate because the take of olive ridleys in the CA/OR drift gillnet fishery is not restricted to adult females, the only segment of the population for which NMFS has population numbers. The possible loss of 2 olive ridleys over the next three years from eastern Pacific olive ridley turtle populations is not anticipated to have a detectable effect on the numbers or reproduction of the species, and therefore is not expected to appreciably reduce the likelihood of survival and recovery of the species.

VIII. CONCLUSION

After reviewing the available scientific and commercial data, current status of listed marine mammals, the environmental baseline for the action area, the effects of the proposed action and the cumulative effects, it is the NMFS' biological opinion that the issuance of authorization pursuant to section 101(a)(5)(E) of the MMPA for the incidental taking of listed marine mammals associated with the CA/OR drift gillnet fishery is not likely to jeopardize the continued existence of fin whales, sperm whales, humpback whales, and Steller's sea lion.

After reviewing the available scientific and commercial data, current status of olive ridley and green sea turtles, the environmental baseline for the action area, the effects of the proposed action and the cumulative effects, it is the NMFS' biological opinion that the issuance of authorization pursuant to section 101(a)(5)(E) of the MMPA for the incidental taking of listed marine mammals associated with the CA/OR drift gillnet fishery and the associated continued operation of the CA/OR drift gillnet fishery, as regulated under the PCTRP is not likely to jeopardize the continued existence of olive ridley and green sea turtles.

However, after reviewing the available scientific and commercial data, current status of Pacific leatherback and loggerhead sea turtles, the environmental baseline for the action area, the effects of the proposed action and the cumulative effects, it is the NMFS' biological opinion that the issuance of authorization pursuant to section 101(a)(5)(E) of the MMPA for the incidental taking of listed marine mammals associated with the CA/OR drift gillnet fishery and the associated continued operation of the CA/OR drift gillnet fishery, as regulated under the PCTRP is likely to jeopardize the continued existence of Pacific leatherback and loggerhead sea turtles.

Given the current status of Pacific leatherback populations as judged by trends in the abundance of females nesting annually at well-monitored nesting beaches, the cumulative human-caused mortality of leatherbacks known or reasonably surmised to occur appears to be more than the populations can sustain. This is likely to be true even without the incidental mortality estimated to be caused by the CA/OR drift gillnet fishery. Thus, given the total mortality from other human activities, and assuming such mortality rates persist, additional leatherback mortalities caused by the CA/OR drift gillnet fishery are probably not sustainable. Unless the cumulative human-caused mortality of leatherbacks is reduced, the populations probably will continue to decline.

Lacking current data on nesting levels and cumulative mortalities of Japanese loggerheads, NMFS cannot assess the status of that population. The documented declines of nesting levels in the early 1990's may reverse as a result of the elimination of the North Pacific high-seas driftnet fisheries in 1993 and the high mortality associated with those fisheries. However, without a knowledge of recent nesting levels or current mortality caused by other human activities (in addition to the CA/OR drift gillnet fishery), it is uncertain whether current mortality rates are sustainable.

IX. REASONABLE AND PRUDENT ALTERNATIVES

Regulations (50 CFR §402.02) implementing section 7 of the ESA define reasonable and prudent alternatives as alternative actions, identified during formal consultation, that (1) can be implemented in a manner consistent with the intended purpose of the action; (2) can be implemented consistent with the scope of the action agency's legal authority and jurisdiction; (3) are economically and technologically feasible; and (4) would, NMFS believes, avoid the likelihood of jeopardizing the continued existence of listed species or resulting in the destruction or adverse modification of critical habitat.

This Opinion has concluded that the issuance of authorization to take listed marine mammals under section 101(a)(5)(E) of the MMPA and the associated continued operation of the CA/OR drift gillnet fishery, as regulated under the PCTRP, is likely to jeopardize the continued existence of the leatherback sea turtle and the loggerhead sea turtle. The clause "jeopardize the continued existence of" means "to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (CFR §402.02).

A. Principles of the reasonable and prudent alternative

On October 12, 2000, the United States ratified the Inter-American Convention for the Protection and Conservation of Sea Turtles. This treaty, which has been signed by 12 countries and ratified by six countries (including the U.S.), promotes the use of appropriate measures for the protection and conservation of sea turtles throughout their range in the Americas, as well as their habitats. The Convention directs each Party to undertake these measures in the land territory and maritime areas over which it exercises sovereignty and with respect to vessels on the high seas which are authorized to fly its flag. These measures shall include the 1) restriction of human activities, to the extent practicable, that could seriously affect sea turtles; 2) protection, conservation, and restoration of sea turtle habitats, as well as the necessary restrictions on the use of such habitats; 3) promotion of research efforts aimed at understanding and enhancing sea turtle populations, education programs for the protection and conservation of sea turtles; and 4) reduction, to the greatest extent practicable, of the incidental capture, retention, harm or mortality of sea turtles in the course of fishing activities.

As discussed in the preceding biological opinion, the CA/OR drift gillnet fishery threatens leatherback and loggerhead sea turtles primarily by entangling them in the drift gillnet, frequently causing injury and mortality through drowning (approximately 61 percent mortality rate for leatherbacks, 32 percent mortality rate for loggerheads, based on observer data from 1990-2000). Fishery management measures and other conservation measures, as necessary, must be undertaken to reduce a) the number of leatherbacks and loggerheads that are incidentally captured, injured, or killed by the CA/OR drift gillnet fishery and, b) the expected impacts to Pacific leatherback populations, to such an extent that the likelihood of jeopardy is avoided, and permits pursuant to section 101(a)(5)(E) can be issued. The following reasonable and prudent alternative has three elements, two that relate to leatherback turtles and one that relates to loggerhead turtles:

1. Leatherback Sea Turtles

a. Time and Area Closure
By August 1, 2001, NMFS, or the states of California and Oregon, must implement regulations to close an area to drift gillnets from Point Conception, California (34°27'N), north to 45°N, and west to 129°W, from August 15th to October 31st for a period of three years (2001 - 2003). Based on observer data from July, 1990 through January, 2000, 78 percent of the leatherback entanglements occurred in this area and during this time period. In addition, the highest density of leatherback sightings on the US West Coast is in and around Monterey Bay, with a peak in sightings in August, with sightings decreasing significantly in September and October (e.g. Stinson (1984 *in* NMFS and USFWS, 1998b; Starbird, *et al*., 1993). Off Point Conception, and around the Channel Islands, leatherbacks are reportedly common during the months of July, August and September and in years

when water temperatures are above normal, while their abundance is "far lower" during October through May (Department of the Navy, 2000). According to studies by Starbird, et al, 1993 and observations by Stinson (1984, *in* leatherback recovery plan), observations of leatherbacks correspond with the movement of the 16°C isotherm along the west coast of North America. This isotherm remains offshore during much of the year, but during early spring and summer, the coast of CA is inundated by warmer offshore waters, and the 16°C isotherm is found closer to shore where most leatherbacks are sighted.

These waters meet the coast south of San Diego and move north reaching central CA during July and August. As these waters move northward, leatherbacks are observed progressively further north and most often where this 16°C isotherm is encountered. Starbird's results showed that sea surface temperatures influenced the number of leatherbacks sighted. Based on observer data collected on entanglement of leatherbacks by the CA/OR DGF and leatherback sighting data, NMFS expects that this time/area closure will result in approximately a 78 percent reduction in the number of leatherback turtles that would be entangled in the fishery with a corresponding reduction in the number of leatherback turtles that

Based on NMFS' knowledge of this fishery, it is expected that the closure outlined above will result in some shift of drift gillnet fishing effort to the areas south of Point Conception during the closure period, although at this point it is speculative to predict how much effort in the southern area will increase. Based on the 1997 - 1999 average, 16 percent of yearly fishery effort occurs in this area August through October. The closure could reduce Steller sea lion and sperm whale interactions as these species have primarily been captured north of Point Conception as well, although most sperm whale interactions have occurred in November and December. The time/area closure is not expected to change the effects of the proposed action on the fin whale, green turtle, or olive ridley turtle because these species are so rarely captured by the fishery (one observed capture, each, in ten years). This shift in effort is also not expected to increase interactions or mortalities of loggerhead sea turtles, which have been predominantly captured in this area during El Niño years, because during such events there will be a time/area closure that will be in effect to protect that species.

b. Conservation, Education, and Protection of Adults and Nesting Beaches in the Western Pacific

We expect the first management measure to reduce the number of leatherback turtles captured in gear associated with these fisheries by approximately a 78 percent, with an analogous reduction in the number of leatherback turtles that would be seriously injured or killed by the fisheries. Despite this management measure, we would expect about 9 leatherback turtles to be entangled in gear associated with this fishery over the 3-year life of the proposed Marine Mammal permit; about 6 of these leatherback turtles would die over three years. These numbers are averages: in some years, the previous measure may prevent any leatherback turtles from being killed in through entanglements in this fishery; in other years, some leatherback turtles were captured and killed in this fishery, although we expect those numbers to remain very small.

In an effort to eliminate the probability of any leatherback turtles from being taken in the CA/OR drift gillnet fishery, NMFS examined numerous measures that could modify the fishery or the proposed Marine Mammal Permit (including additional time/area closures and gear modifications). NMFS also examined numerous measures that could modify other fisheries it manages that capture, injure, or kill leatherback turtles (including the U.S. tuna purse seine fishery in the eastern tropical Pacific Ocean and the California longline fishery), but could identify no measures that would further reduce the numbers of leatherback turtles taken in those fisheries.

After exhausting all options that could further modify the CA/OR drift gillnet fishery to prevent any leatherback turtles from being captured, exhausting all options that could minimize any harm to leatherback turtles that could not be avoided, the next management measure is designed to counteract the harm the CA/OR drift gillnet fishery could still pose to leatherback turtles.

Therefore, NMFS shall continue to fund and implement measures to protect and conserve leatherback turtle populations in the eastern Pacific Ocean and expand these measures to the western Pacific Ocean. For four years, NMFS has funded and implemented a program to protect and conserve leatherback turtles by working cooperatively with the government of Mexico and Costa Rica to protect eastern Pacific leatherback populations. Because of this program, there has been greater clutch protection and hatchling success. Currently, the primary management objective is 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 the next three years to help implement these objectives (minutes from leatherback working group meeting, 23-24 May, 2000). These continuing efforts should help to offset the remaining portion of the effects of the proposed action borne by the eastern Pacific leatherback population.[14]

Beginning with Fiscal Year 2001, NMFS shall fund and implement a similar program in the western Pacific Ocean (contingent on the ability of U.S. nationals to safely conduct such a program in different countries in the region) using measures patterned after measures that have been used in the eastern Pacific Ocean, which have protected female leatherback turtles from being killed while nesting, protecting leatherback turtle nests, and reducing the number of leatherback turtles captured, injured, and killed in fisheries in this region. These conservation efforts shall be designed to protect sufficient numbers of nests and hatchlings to compensate for the loss at least 3 adult leatherback turtles per year

[14]Genetic testing done on 2 of the 23 leatherbacks caught to date in the CA/OR drift gillnet fishery indicate that these turtles originate from western Pacific nesting beaches. However, based on available information about the migratory behaviors of eastern Pacific leatherbacks, some leatherbacks caught in the CA/OR drift gillnet fishery could originate from eastern Pacific beaches.

(the actual number shall be calculated prior to beginning this conservation effort using elasticity analyses for leatherback turtles). Due to the wide range of this species, NMFS recognizes that intensive cooperative efforts with other countries whose activities affect leatherback populations are necessary to address the ongoing factors which are contributing to the depressed status of this species. The recent ratification of the Inter-American Convention for the Conservation and Protection of Sea Turtles supports this recognition.

The expected result of the leatherback portion of the RPA is that the reduction in impacts to the migrating leatherbacks by the proposed action *coupled with* conservation and protection measures taken overseas to address impacts to the nesting leatherbacks and their eggs and hatchlings will nullify the action causing jeopardy.

2. Loggerhead Sea Turtles

NMFS has analyzed the patterns of loggerhead sea turtle captures and mortalities in the CA/OR drift gillnet fishery. Based on this assessment, it appears that all of the observed takes occurred south of Point Conception, California, (32°45'N) during the months of June through January, during El Niño years (14.6 percent of fishing effort occurs during the months of August and January, south of Point Conception).

Similar to the review of the fishery and observed interactions with leatherback sea turtles described above, NMFS has found no apparent correlation between variations in fishing strategy and loggerhead sea turtle interactions. Loggerhead sea turtles have been observed captured by the CA/OR drift gillnet fishery only during El Niño events. Therefore, it appears that modifications in gear or gear deployment are not likely to achieve significant or measurable reductions in the capture and mortality rate of these turtles. NMFS has determined that the most effective method of avoiding loggerhead interactions and mortality is a time/area closure during El Niño years. The most reasonable time/area closure, based on past observed interactions, will avoid a significant portion of the expected captures (approximately 92 percent based on past performance) and mortalities. Furthermore, additional conservation measures to reduce impacts or enhance the status of affected populations may be necessary to nullify the remaining possible impacts to loggerhead sea turtles.

B. Reasonable and prudent alternative

The following RPA, which contains two management measures designed to avoid the likelihood of jeopardizing leatherback turtles and one management measure designed to avoid the likelihood of jeopardizing loggerhead turtles, is based on expected CA/OR drift gillnet fishery operations in compliance with current state and federal regulations. Therefore, time/area closures have been tailored to fit in with pre-existing closures of the fishery under state regulation.

1. Leatherback Sea Turtles

a. Time and Area Closure

By August 1, 2001, NMFS or the states of California and Oregon, must implement regulations to close an area to drift gillnets from Point Conception, California (34°27'N), north to 45°N, and west to 129°W, from August 15th to October 31st for a period of three years (2001 - 2003). Based on observer data from July, 1990 through January, 2000, 78 percent of the leatherback turtle entanglements occurred in this area and during this time period. In addition, the highest density of leatherback turtle sightings on the U.S. Pacific Coast is in and around Monterey Bay; sightings peaked in August then decreased significantly in September and October (e.g. Stinson 1984 [cited *in* NMFS and USFWS, 1998b]; Starbird, *et al*., 1993). Off Point Conception and around the Channel Islands, leatherback turtles are reportedly common during the months of July, August, and September; in years when water temperatures are above normal, they occur from October through May, although their abundance is "far lower" (Department of the Navy, 2000).

According to studies by Starbird et al. (1993) and observations by Stinson (1984), observations of leatherbacks correspond with the movement of the 16°C isotherm along the west coast of North America. This isotherm remains offshore during much of the year, but during early spring and summer, the coast of California is inundated by warmer, offshore waters and the 16°C isotherm is found closer to shore where most leatherbacks are sighted. These waters meet the coast south of San Diego and move north reaching central California during July and August. As these waters move northward, leatherbacks are observed progressively further north and most often where this 16°C isotherm is encountered. Starbird et al. (1993) showed that sea surface temperatures influenced the number of leatherbacks sighted.

Based on observer data collected on entanglement of leatherbacks by the CA/OR drift gillnet fishery and leatherback sighting data, NMFS expects this time/area closure to reduce the number of leatherback turtles captured in gear associated with these fisheries by approximately a 78 percent, with an analogous reduction in the number of leatherback turtles that would be seriously injured or killed by the fisheries. Through this management measure, we would expect about 9 leatherback turtles to be entangled in gear associated with this fishery over the 3-year life of the proposed Marine Mammal permit; we would expect about 6 of these leatherback turtles to die over three years.

Based on NMFS' knowledge of this fishery, we expect some gillnet fishing effort to shift to the areas south of Point Conception in response to the time/area closure we have prescribed above, although at this point it is speculative to predict how much effort in the southern area will increase. Based on the 1997 - 1999 average, 16 percent of annual fishery effort occurs in this area August through October. The closure could reduce Steller sea lion and sperm whale interactions as these species have primarily been captured north of Point Conception as well, although most sperm whale interactions have occurred in November and December. The time/area closure is not expected to change the effects of the proposed action on the fin whale, green turtle, or olive ridley turtle because these species are so rarely captured by the fishery (one observed capture, each, in ten years). This shift in effort is also not expected to increase interactions or mortalities of loggerhead sea turtles, which have been predominantly captured in this area during El Niño events, because during such events there will be a time/area closure that will be in effect to protect that species .

b. Conservation, Education, and Protection in the Western Pacific

We expect the first management measure to reduce the number of leatherback turtles captured in gear associated with these fisheries by approximately a 78 percent, with an analogous reduction in the number of leatherback turtles that would be seriously injured or killed by the fisheries. Despite this management measure, we would expect about 9 leatherback turtles to be entangled in gear associated with this fishery over the 3-year life of the proposed Marine Mammal permit; about 6 of these leatherback turtles would die over three years. These numbers are averages: in some years, the previous measure may prevent any leatherback turtles from being killed through entanglements in this fishery; in other years, some leatherback turtles would be captured and killed in this fishery, although we expect those numbers to remain very small.

In an effort to eliminate the probability of any leatherback turtles from being taken in the CA/OR drift gillnet fishery, NMFS examined numerous measures that could modify the fishery or the proposed Marine Mammal Permit (including additional time/area closures and gear modifications). NMFS also examined numerous measures that could modify other fisheries it manages that capture, injure, or kill leatherback turtles (including the U.S. tuna purse seine fishery in the eastern tropical Pacific Ocean and the California longline fishery), but could identify no measures that would further reduce the numbers of leatherback turtles taken in those fisheries.

After exhausting all options that could further modify the CA/OR drift gillnet fishery to prevent any leatherback turtles from being captured, exhausting all options that could minimize any harm to leatherback turtles that could not be avoided, the next management measure is designed to counteract the harm the CA/OR drift gillnet fishery could still pose to leatherback turtles.

NMFS shall continue to fund and implement measures to protect and conserve leatherback turtle populations in the eastern Pacific Ocean and expand these measures to the western Pacific Ocean. For four years, NMFS has funded and implemented a program to protect and conserve leatherback turtles by working cooperatively with the government of Mexico and Costa Rica to protect eastern Pacific leatherback populations. Because of this program, there has been greater clutch protection and hatchling success. Currently, the primary management objective is 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 the next three years to help implement these objectives (minutes from leatherback working group meeting, 23-24 May, 2000).

These continuing efforts should help to offset the remaining portion of the effects of the proposed action borne by the eastern Pacific leatherback population.[15]

Beginning with Fiscal Year 2001, NMFS shall fund and implement a similar program in the western Pacific Ocean (contingent on the ability of U.S. nationals to safely conduct such a program in different countries in the region) using measures patterned after measures that have been used in the eastern Pacific Ocean, which have protected female leatherback turtles from being killed while nesting, protecting leatherback turtle nests, and reducing the number of leatherback turtles captured, injured, and killed in fisheries in this region. Due to the wide range of this species, NMFS recognizes that intensive cooperative efforts with other countries whose activities affect leatherback populations are necessary to address the ongoing factors which are contributing to the depressed status of this species. The recent ratification of the Inter-American Convention for the Conservation and Protection of Sea Turtles supports this recognition.

The expected result of the leatherback portion of the RPA is that the reduction in impacts to the migrating leatherbacks by the proposed action *coupled with* conservation and protection measures taken overseas to address impacts to the nesting leatherbacks and their eggs and hatchlings will nullify the action causing jeopardy.

2. Loggerhead Sea Turtles

a. Time and Area Closure

By August 1, 2001, NMFS, or the states of California and Oregon, must implement regulations to close an area to drift gillnets south of Point Conception, California (34°27'N), and west to 120°W, from August 15th to August 31st and again from January 1st through January 31st, *during a forecasted, or occurring, El Niño event*. Based on observer data from July, 1990 through January, 2000, 92 percent of the loggerhead entanglements occurred in this area and during this time period (11 observed takes out of 12 observed takes that would occur under current fishery regulations that open the fishery within 75 nautical miles from August 15th to January 31st) . Since 1990, all of the loggerheads observed incidentally taken in this fishery were located in a concentrated area south of San Clemente Island, and the majority (9/14) of them were caught in the summertime (June through August), when sea surface temperatures are highest. All but three loggerheads were observed captured in waters with reported average monthly sea surface temperatures of from 18°C to 21°C. Three loggerheads were caught in January, 1993 in waters with an average monthly temperature of around 15°C, and where CoastWatch reported a "relatively high incidence of red crabs (a southern species) throughout the southern California Bight." Unusually warm sea surface temperatures and northward flowing equatorial

[15]Genetic testing done on 2 of the 23 leatherbacks caught to date in the CA/OR drift gillnet fishery indicate that these turtles originate from western Pacific nesting beaches. However, based on available information about the migratory behaviors of eastern Pacific leatherbacks, some leatherbacks caught in the CA/OR drift gillnet fishery could originate from eastern Pacific beaches.

currents during El Niño events bring hundreds of thousands of pelagic red crabs from Baja California north up the coast of California. Loggerheads taken by the fishery most likely migrate north from Baja California, Mexico, following their primary food source. Therefore, based on observer data collected on entanglement of loggerheads by the CA/OR drift gillnet fishery and loggerhead sighting data, NMFS expects that this time/area closure will result in approximately a 92 percent reduction in loggerhead entanglement and associated mortalities.

Based on NMFS' knowledge of this fishery, it is possible that the closure outlined above will result in a reduction or complete cessation of fishing effort during August and January. Fifteen percent of yearly fishery effort occurs in this area during August and January, based on the 1997 - 1999 average. Some shift of drift gillnet fishing effort to the areas west of the El Niño closure may occur, although at this point it is speculative to predict how much effort west of the closure will increase. The El Niño time/area closure is not expected to change the effects of the proposed action on the fin whale, green turtle, or olive ridley turtle because these species are so rarely captured by the fishery (one observed capture, each, in ten years). Based on the past observed interaction patterns the drift gillnet fishery has had with these species, NMFS also does not anticipate any increase in entanglements of sperm whales, humpback whales, or Steller sea lions.

X. Incidental Take Statement

Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct. NMFS further defines "harm" as an act which actually kills or injures fish or wildlife. Such an act may include significant habitat modification or degradation where it actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns including breeding, spawning, rearing, migrating, feeding or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of section 7(b)(4) and 7(o)(2), taking that is incidental to and not intended as part of the proposed action is not considered to be prohibited taking under the Act provided that such taking is in compliance with this Incidental Take Statement.

The measures described below are nondiscretionary, and must be undertaken by NMFS for the exemption in section 7(o)(2) to apply. The NMFS has a continuing duty to regulate the activity covered by this incidental take statement. If NMFS (1) fails to assume and implement the terms and conditions the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, NMFS must monitor the progress of the action and its impact on the species as specified in the incidental take statement. (50 CFR §402.14(I)(3))

Section 7(b)(4) of the Endangered Species Act (ESA) requires that when a proposed agency action is found to be consistent with section 7(a)(2) of the ESA and the proposed action may incidentally take individuals of listed species, NMFS will issue a statement that specifies the impact of any incidental taking of endangered or threatened species. It also states that reasonable and prudent measures, and terms and conditions to implement the measures, be provided that are necessary to minimize such impacts. Only incidental take resulting from the agency action and any specified reasonable and prudent measures and terms and conditions identified in the incidental take statement are exempt from the taking prohibition of section 9(a), pursuant to section 7(o) of the ESA.

A marine mammal species or population stock which is listed as threatened or endangered under the ESA is, by definition, also considered depleted under the MMPA. The ESA allows takings of threatened and endangered marine mammals only if authorized by section 101(a)(5) of the MMPA. The proposed action is the issuance of authorization for the incidental taking of marine mammals under section 101(a)(5)(E) of the MMPA, therefore the incidental takes of marine mammals described below are also exempt from the taking prohibition of section 9(a), pursuant to section 7(o) of the ESA.

AMOUNT OR EXTENT OF TAKE

Mortality and entanglement rates of marine mammals and sea turtles have been calculated by multiplying the number of individual animals observed entangled or killed during a given calendar year by the inverse percentage of the fishing effort observed during that year (e.g. one sperm whale observed entangled during a year in which 20% of the fishery is observed results in an estimated entanglement rate of five sperm whales). Mortality and entanglement rates vary from year to year, with some species observed killed every year, and others observed killed only every few, or even, ten years.

The NMFS has developed the following incidental take statement based on the premise that the reasonable and prudent alternatives will be implemented. The incidental take of leatherbacks and loggerheads has been calculated based on the estimated impact that remains following the implementation of the time/area closures. The following table describes the anticipated amounts of estimated and observed entanglements and/or mortalities that will occur over the next three years, while the 101(a)(5)(E) permit is in effect:

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

[1]Total expected observations of an entanglement or mortality over the three year permit period.

Based on the method NMFS currently uses to estimate incidental entanglement in the fishery, one observed entanglement results in an estimate of five entanglements. Therefore, for all of the above species except loggerheads, if more than one entanglement is observed during the three year period this Opinion covers, NMFS is likely to determine that incidental take has been exceeded. In the past NMFS has also applied the same calculation to estimated mortalities – if one individual of a species is entangled and killed, then 5 individuals have been entangled and killed. However, this likely overestimates the mortal take because, based on past observations, most of the above species have an expected survival rate per entanglement. For example, hard shelled turtles are estimated to have a 68 percent survival rate. Of the 6 green turtles anticipated to be taken in the drift gillnet fishery over the next three years, 4 are expected to survive their entanglement and 2 are expected to die. However, based on the random sampling of the species incidentally caught in the fishery by observed vessels, the one mortality observed is likely one of the instances of expected mortal incidental take and not grounds for reinitiation of the consultation. This is applicable to all species described in this Opinion except for the humpback whale. If a mortality of a humpback whale is observed, this would require reinitiation of this consultation because no humpback mortalities are expected.

EFFECT OF THE TAKE

In the accompanying biological opinion, NMFS determined that this level of anticipated take is not likely to result in jeopardy to the fin whale, humpback whale, sperm whale, Steller sea lion, green turtle, leatherback turtle, loggerhead turtle, or olive ridley turtle when the reasonable and prudent alternative is implemented.

Reasonable and Prudent Measures

NMFS believes the following reasonable and prudent measures, as implemented by the terms and conditions, are necessary and appropriate to minimize impacts to minimize impacts to sea turtles. NMFS has determined that the requirements of the PCTRP are adequate and appropriate to minimize the impact of the take on marine mammals and therefore no additional measures are necessary. The measures described below are non-discretionary, and must be undertaken by NMFS for the exemption in section 7(o)(2) to apply. If NMFS fails to adhere to the terms and conditions of the incidental take statement, the protective coverage of section 7(o)(2) may lapse. Thus, the following reasonable and prudent measures must be implemented to allow activities by the CA/OR drift gillnet fishery to continue.

1. CA/OR drift gillnet vessel operators and observers shall be educated on sea turtle biology and on methods that will reduce injury or mortality during fishing operations.

2. Live captured sea turtles shall be released uninjured from the net in a manner that minimizes the likelihood of further gear entanglement or entrapment.

3. NMFS shall continue to collect data on capture, injury and mortality of sea turtles in addition to life history information.

4. Comatose and lethargic sea turtles shall be retained on board, handled, resuscitated, and released according to the procedures outlined in the 50 CFR 223.206(d)(1).

5. Sea turtle mortalities shall be disposed of at sea unless an observer 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, NMFS 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. NMFS will incorporate into the Pacific Offshore Cetacean Take Reduction Plan skipper education workshops a module on sea turtle resuscitation requirements, as outlined in 50 CFR §223.206(d)(1).

    1B. NMFS will include in the skipper education workshops a module of information on sea turtle biology and ways to avoid and minimize sea turtle impacts.

    1C. NMFS will encourage skipper workshop participants to suggest additional strategies or techniques that might minimize impacts of fishing gear or practices on sea turtles.

    1D. NMFS will include sea turtle resuscitation techniques and sea turtle biology information during observer training.

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

    2A. Removing turtles from the net takes priority over transferring catch from the net to the vessel.

    2B. Turtles must be untangled as quickly and carefully as possible to avoid injury or mortality. The sea turtles must not be dropped on to the deck.

3. The following terms and conditions implement reasonable and prudent measure No. 3.

3A. NMFS shall continue to maintain the observer program to collect data on the incidental take of marine mammals, sea turtles, and other protected species.

3B. NMFS shall continue to collect information aboard CA/OR drift gillnet vessels, including, at a minimum, the incidental capture, injury, and mortality of sea turtles by species, gear and set information in which each interaction occurred, and life history information.

3C. NMFS shall continue to collect life history information on sea turtles, such as species identification, measurements, condition, skin biopsy samples, the presence or absence of tags.

3D. NMFS observers shall directly measure or visually estimate tail length on all sea turtles captured by the CA/OR drift gillnet fishery.

3E. NMFS collected data and other available information shall be evaluated on an annual basis to determine whether estimated annual incidental injuries or mortalities of sea turtles has exceeded allowable removal levels. The report will be sent to the Sea Turtle Coordinator in Silver Spring, Maryland.

4. The following terms and conditions implement reasonable and prudent measure No. 4.

4A. Drift gillnet vessel operators shall bring comatose sea turtles aboard, if feasible, and perform resuscitation techniques according to the procedures described at 50 CFR §223.206(d)(1).

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

5. The following term and condition implements reasonable and prudent measure No. 5.

5A. Dead sea turtles may not be consumed, sold, landed, offloaded, transhipped or kept below deck, but must be returned to the ocean after identification unless the observer requests the turtle to be kept for further study.

XI. 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. NMFS should explore the possibility of developing or modifying existing gear to reduce the likelihood of gear interactions (visual or acoustic cues).

2. NMFS should explore the possibility of developing or modifying existing technologies, such as sonar, to detect and alert fishers if sea turtles or marine mammals become entangled in their gear.

3. NMFS should explore the feasibility of developing a system for fishermen to collect life history information on sea turtles.

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

5. NMFS 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.

XII. REINITIATION NOTICE

This concludes formal consultation on the action outlined above. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of the incidental take is exceeded; (2) new information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not considered in this opinion; (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in this opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action. In instances where the amount or extent of incidental take is exceeded, the Marine Mammal Division, Office of Protected Resources, NMFS, should immediately request initiation of formal consultation.

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Appendix A

Fishing Effort in the Pacific Ocean

Despite the ban on high-seas driftnet fishing in the north Pacific Ocean in the early 1990s, fishing effort by longliners, purse seiners, trollers, and coastal driftnetters continues throughout the Pacific Ocean. The number of vessels per country varies greatly, from less than 10 for some of the small island nations (e.g. Papua New Guinea, Tonga, and Vanuatu), to over 600 for the more economically powerful countries, such as Japan. For most of these fishing fleets, little or no data exists regarding the incidental bycatch of marine mammal and sea turtle populations, particularly for those species in danger of extinction. Without such information, it is difficult to assess the impacts of these fisheries on species included in this Opinion. Nevertheless, bycatch information, including survival rates following entanglements, collected by observers and through fisher self reporting does exist for some fisheries in the eastern and western Pacific Ocean. Given such data, coupled with distribution and abundance records for the various species, one can at least gain a sense of the possible impacts of those fisheries for which no information exists. The following sections summarize past and current fishing effort in the eastern and central western Pacific Ocean.

A. Fishing effort in the Central Western Pacific Ocean

Although high-seas driftnet vessels no longer operate in the North Pacific, longliners, purse seiners, trollers, and pole-and-liners continue to fish, mainly for tuna species, in the Central Western Pacific Ocean, typically west of the 150°W longitude and north and south of the equator.

Table 1 provides a summary of the known number of active longline vessels, by country, by year, from 1990-1999 in the Central Western Pacific Ocean. Preliminary estimates are represented in parenthesis ( ). Where known, the number of total hooks deployed during a particular year was included (e.g. Korea and Taiwan-distant water). Okamoto *et al*. (1999) estimated the number of hooks deployed by Japan's offshore and distant water longline fleet in the Western Pacific Ocean: 1990 - 192,000,000 hooks; 1991 - 170,000,000 hooks; 1992 - 154,000,000 hooks; 1993 - 164,000,000 hooks, 1994 - 158,000,000 hooks; 1995 - 141,000,000 hooks; 1996 - 127,000,000 hooks; and 1997 - 131,000,000 hooks. While many of the small Pacific island countries have relatively small longline fleets, Japan and Taiwan clearly dominate this fishery, fishing coastally, in distant water, and offshore. The number of active vessels per country over the past 10 years appears relatively steady, with American Samoa and Vanuatu entering the fishery in 1995, an increase by China in the mid-1990s, and increases since the early and mid-1990s by Australia, Fiji, French Polynesia, Micronesia, New Zealand, and Samoa.

Table 1. Number of active longline vessels fishing, by country, by year, in the Central Western Pacific Ocean

Country/Year	1990	1991	1992	1993	1994	1995	1996	1997	1998	
American Samoa	–	–	–	–	–	6	13	22	27	(27)
Australia - domestic	117	111	124	109	110	109	115	137	156	174
Australia-Japan JV	14	29	56	66	52	21	–	–	–	–
China	23	39	72	310	456	422	323	140	116	(116)
Cook Islands	–	–	–	–	2	2	3	–	–	–
Fed States Micronesia	–	2	7	7	9	11	9	15	21	(21)
Fiji	6	9	18	22	37	48	42	34	39	(39)
French Polynesia	2	8	25	50	66	65	58	59	54	57
Indonesia	151	145	141	309	293	(293)	(293)	(293)	(293)	(293)
Japan - coastal	685	768	793	790	819	738	711	698	(698)	(698)
Japan- distant water	791	790	768	767	749	744	703	695	(695)	(695)
Japan - offshore	362	332	302	272	255	222	200	180	(180)	(180)
Kiribati	–	–	–	–	–	1	1	–	–	–
Korea (# vessels and total # hooks)[1]	182	220	166	148	160	154	156	148	169	(169)
	73,216	53,452	62,125	56,190	76,380	81,831	73,420	68,241	66,193	–
Marshall Islands	–	–	2	5	2	4	–	–	–	–
New Caledonia	7	6	4	4	5	8	8	9	11	13
New Zealand	17	21	30	40	56	96	84	56	(56)	(56)
Papua New Guinea	–	–	–	2	4	11	7	8	8	(8)
Philippines	26	12	10	10	10	10	10	(10)	(10)	(10)
Samoa	–	–	–	17	25	45	90	150	150	151
Solomon Islands	–	–	–	–	–	20	36	31	22	14
Taiwan - distant water (# vessels and # hooks)[1]	52	74	88	72	67	62	56	53	64	65
	37,681	58,783	35,089	28,440	41,083	52,615	31,394	–	–	--
Taiwan - offshore Taiwan	1,139	800	1,898	1,791	1,753	1,603	1,274	1,877	1,712	1,696
Taiwan - offshore Micronesia	–	–	–	254	132	92	123	217	208	(208)
Tonga	1	1	1	6	5	7	7	(7)	(7)	(7)
United States	138	143	131	129	141	127	115	110	118	(118)

Vanuatu	–	–	–	–	–	2	3	1	1	(1)
TOTAL	**3,713**	**3,510**	**4,686**	**5,180**	**5,208**	**4,923**	**4,440**	**4,950**	**4,815**	**4,816**

[1]Number of hooks is in thousands (000s)

Source: Southwest Fisheries Science Center (SWFSC) unpublished data from Secretariat of the Pacific Community based in Noumea, New Caledonia.

Table 2 provides a summary of the known number of active purse seine vessels, by country, by year, from 1990-1999 in the Central Western Pacific Ocean.

Table 2. Number of active purse seine vessels fishing, by country, by year, in the Central Western Pacific Ocean

Country/Year	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999
Australia - domestic	1	6	13	7	4	2	4	5	4	7
Australia - distant water	8	6	2	1	–	–	–	–	–	–
Fed. States of Micronesia	–	6	7	7	8	6	4	4	3	(3)
Japan - coastal	43	38	31	27	23	20	21	20	(20)	(20)
Japan - offshore and distant water	35	35	38	36	33	31	32	35	35	35
Kiribati	–	–	–	–	1	1	1	1	1	(1)
Korea	39	36	36	34	32	30	28	27	26	26
New Zealand	3	6	7	5	6	6	6	6	6	6
Papua New Guinea	–	–	–	–	2	3	4	10	13	(13)
Philippines - domestic (purse seine & ring)	549	546	407	399	(399)	(399)	(399)	(399)	(399)	(399)
Philippines - distant water	13	15	12	12	11	13	12	12	12	(12)
Russia	5	4	3	8	4	–	–	–	–	–
Solomon Islands	4	3	3	3	3	3	3	4	4	4
Taiwan	32	39	45	43	43	42	42	42	42	42
United States	43	43	44	42	49	44	40	35	39	36
Vanuatu	–	–	–	–	1	2	2	5	5	(5)
Total	**775**	**783**	**648**	**624**	**619**	**603**	**598**	**605**	**609**	**609**

Source: Southwest Fisheries Science Center (SWFSC) unpublished data from Secretariat of the Pacific

Community based in Noumea, New Caledonia.

Table 3 provides a summary of the known number of active troll vessels, by country, by year, from 1990-1999 in the Central Western Pacific Ocean.

Table 3. Number of active troll vessels fishing, by country, by year, in the Central Western Pacific Ocean

Country/year	89-90	90-91	91-92	92-93	93-94	94-95	95-96	96-97	97-98	98-99
Canada	–	–	–	–	–	–	–	–	–	2
French Polynesia	3	4	2	4	–	4	4	1	–	–
New Zealand	125	229	247	425	500	478	429	268	268	268
United States	38	58	55	47	14	21	50	28	35	–
Total	**166**	**291**	**304**	**476**	**514**	**503**	**483**	**297**	**303**	**270**

Source: Southwest Fisheries Science Center (SWFSC) unpublished data from Secretariat of the Pacific Community based in Noumea, New Caledonia.

B. Fishing effort in the Eastern Pacific Ocean

Chile

Despite significant reverses since swordfish catch peaked in 1991, Chile continues to conduct the largest gillnet and longline swordfish fishery in Latin America. This fishery is economically significant to both commercial and artisanal fishermen. Since 1980, this fishery grew by nearly an order of magnitude as fishermen developed more efficient methods and foreign demand for swordfish expanded. From 1987 the Chilean swordfish driftnet fishery expanded rapidly with many hundreds of boats concentrated primarily in four ports - Chañaral, Valparaiso, San Antonio, and Concepcion. Most of these vessels were small (14-15 meters) and switched from a harpoon fishery to a driftnet fishery. The artisanal swordfish fleet alone expanded from 4,777 days-at-sea in 1987 to 40,692 days-at-sea in 1993 (Weidner and Serrano, 1997).

Table 4. Chile – Fishing fleet, 1993-96. ( ) indicates # of artisanal swordfish vessels.

Year	Seiners	Trawlers	Long-liners	Driftnets	Traps	Harpoon	Comm. Total	Artisanal	
1993	410	72	115	40	2	1	640	8,904 (350)	9,544
1994	383	72	88	32	3	--	578	10,864 (275)	11,442
1995	370	70	74	28	3	--	545	12,045 (286)	12,590
1996	385	73	45*	19	2	--	524	12,619	13,143

*Primarily targeting groundfish
Source: *in* Weidner and Serrano (1997).

Columbia

Columbia's commercial fleet is dominated by the coastal shrimp fleet (nearly half), but there are also a large number of tuna vessels, primarily purse seiners. As shown in the following table, foreign vessels also operate out of Columbia, including Japanese longliners targeting tuna. In 1997, there were eight foreign longliners operating out of Columbia, with over half of Japanese nationality. Most artisanal fishermen out of Columbia operate very close to shore, targeting mainly pelagics. Little is known about the number of artisanal vessels operating off the Pacific coast of Columbia; however, they are known to deploy small longlines, driftnets and purse seines (Weidner and Serrano, 1997).

Table 5. Columbia - commercial fishing fleet 1992-95.

Year	Domestic Vessels	Foreign vessels*	Total
1992	252	215	467
1993	n/a	150	n/a
1994	156	174	330
1995	192	168	360

*Foreign vessels licensed and working in association with Columbia companies.
Source: Weidner and Serrano, 1997.

Ecuador

Ecuadoran commercial fishing operations, consisting mainly of seiners, are conducted nearly entirely within their 200 mile (Exclusive Economic Zone) EEZ; however, sever larger longliners have operated over 1,000 km off the coast, west of the Galapagos Islands. The fleet continues to expand, and it is likely that operations to the west of the Galapagos will increase, given the profitability of swordfish and expanding technical capabilities. The artisanal fisheries of Ecuador are generally limited to inshore coastal waters, although some longline fishermen have begun to fish around the Galapagos. In 1996, the bulk of the Ecuadoran fishing fleet was comprised of artisanal vessels (93%), and although estimates vary, there may have been as many as 6,000-8,000 active fishing vessels in Ecuador (3,971 estimated vessels in 1996). Foreign longliners have also been operating off Ecuador for many years, and most are from Japan. In 1996, there were an estimated 15-20 foreign longliners, all Japanese, except for one (Weidner and Serrano, 1997).

Peru

Unlike Chile and Ecuador, Peru has not developed a substantial longline or driftnet fleet, and there is little information on the number of vessels, both commercial and artisanal, that are currently operating in various fisheries off Peru. Artisanal longliners generally deploy in shallow water, no deeper than 100 meters, and generally close to shore (normally no more than 50 km from the mainland), while drift gillnetters rarely fish more than 20-30 miles offshore. As of 1997 there were only four domestically built longliners, and fishing generally occurs about 50 km from the coast. Between 1980-97, there were 17 Peruvian-owned commercial lon6gliners. There is limited information on foreign fleet operations off Peru, but vessels, mostly Japanese and Spanish, do fish primarily off the southern coast, both inside and outside the 200 mile limit. Between 1980-97, there were at least 9 foreign leased longliners (Japan) and 11 Spanish longliners (Weidner and Serrano, 1997).

Mexico

In 1983, Mexico established a 50 nautical mile (NM) sportfishing-only zone along its coast to protect billfish, swordfish, tuna, and other popular species and to manage them for the recreation and tourist industries. Commercial fisheries for swordfish outside this exclusion zone included longliners until 1990 and drift gillnetters. Limited longline permits were issued in 1987, allowing only about 15 vessels to fish within Mexico's EEZ. Operating under these permits, the Japanese/Mexico joint venture fleet increased fishing effort to 2.3 million hooks between 1986-88. Due to the reduction in longline permits, a small (2 vessels) drift gillnet fleet appeared in northern Baja California in 1986, growing to 20 by 1990 and to 31 by 1993. Fishing effort increased from 15 days/month in 1989 to 20-30+ days/month in 1993. By 1994, the number of vessels had declined to 16, primarily due to low catch. Most of these vessels are home-based out on Ensenada and are similar in appearance and operation to the California/Oregon drift gillnet fishery, although they use nets up to twice as long as those used on U.S. vessels. The swordfish fishery begins in the fall for the Mexican drift gillnetters, off Ensenada, moving south to central Baja, California (between 25/N and 27/N) during December and January (Holts and Sosa-Nishizaki, 1998).

REFERENCES

Any references used in this Appendix are listed in the Biological Opinion.

Appendix B

Trends in Eastern Pacific Nesting Populations of Leatherbacks

Table 1. Estimated abundance of nesting female leatherbacks in the Eastern Pacific. [ ] = number of nests

Year/Beach	80-81	83-84	86-87	87-88	88-89	89-90	90-91	91-92	92-93	93-94	94-95	95-96	96-97	97-98	98-99	
Mexico																
Tierra Colorada	10,000[1]								1,000-2,000[1]		50-100[1]					[402][7]
Bahía Chacahua	2,000[1]								50-100[1]	50-100[1]						
Mexiquillo	3,000-5,000[1]		959[1]	240[1]						16[1]		[1,280][5]	[60][5]	[123][5]	[53][5]	[463[5]-469[7]]
Barra de la Cruz										299[1]						[296][7]
Other Mex. Beaches																[922][7]
Pacific Coast of Mexico												700-900[1] 1,093[3] [5,354][5]	236[3] [981[3] - 1,093[5]]	250-329[3] [1,596][5]	[1,117][5]	[4,317][5]
El Salvador																
Guatamala																

Total (Mex. and Central America, w/o Costa Rica)																~1000[7]
Year/Beach	**80-81**	**83-84**	**86-87**	**87-88**	**88-89**	**89-90**	**90-91**	**91-92**	**92-93**	**93-94**	**94-95**	**95-96**	**96-97**	**97-98**	**98-99**	**99-00**
Costa Rica																
Playa Grande					1,367[4] - 1,646[2]	1,340[4] - 1,643[2]	665[4]-830[2]	770[4]-932[2]	909[4]-1,078[2]	180[4]-202[2]	469[2]-506[4]	421[4] 800-1000[1,6]	125[4]	195[4]	117[4]	
Playa Langosta							229[8]				239[2]	(800-1000)[1,6]				
Playa Naranjo		62[2]				93[2]	242[2]					30-240[1]				

[1]Summarized in Spotilla, *et al*. (1996), using an estimated clutch frequency of 5.
[2]Summarized in Steyermark, *et al*. (1996), using an estimated clutch frequency of 5.
[3]Summarized in Sarti *et al*. (1998), using annual average clutch frequency (observed and estimated).
[4]Summarized in Spotilla, *et al*. (2000).
[5]Sarti,L., personal communication, 2000.
[6]800-1,000 estimated nesting females on both Playa Grande and Playa Langosta
[7]Preliminary results presented by L. Sarti, Leatherback Working Group Meeting, 23-24 May, 2000. Other beaches include Llano Grande, Playa Ventura, Agua Blanca.
[8]Chaves, *et al*., 1996.