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November 1, 2006

MEMORANDUM FOR: Charles Karnella
Coordinator
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FROM: Chris Yates 
Assistant Regional Administrator
Protected Resources Division

SUBJECT: Transmittal of Final Biological Opinion under section 7 of
the Endangered Species Act on the effects of the U.S. tuna purse seine
fishery in the western and central Pacific Ocean on listed sea turtles and
marine mammals

Attached to this memorandum is NMFS' Final Biological Opinion (Opinion) under section 7 of the Endangered Species Act on the effects of the U.S. purse seine fishery in the western and central Pacific Ocean (WCPO) on the olive ridley turtle (*Lepidochelys olivacea*), leatherback turtle (*Demochelys coriacea*), loggerhead turtle (*Caretta caretta*), green turtle (*Chelonia mydas*), humpback whale (*Megaptera novaeangliae*), sperm whale (*Physeter macrocephalus*), blue whale (*Balaenoptera musculus*), sei whale (*Balaenoptera borealis*), and fin whale (*Balaenoptera physalus*).

This Opinion examines the effects of the tuna purse seine fishery in the WCPO under the jurisdiction of the U.S. as authorized by the South Pacific Tuna Act of 1988 (16 U.S.C. Chapter 16C) and the High Seas Fishing Compliance Act of 1995 (16 U.S.C. 5501 et seq.). The Opinion concludes that continued authorization of the U.S. WCPO purse seine fishery, is not likely to jeopardize the continued existence of threatened and endangered olive ridley turtles, threatened loggerhead turtles, threatened and endangered green turtles, endangered leatherback turtles, endangered humpback whales, endangered sperm whales, endangered blue whales, endangered sei whales, or endangered fin whales.

An Incidental Take Statement (ITS) has been issued for all sea turtle species and non-discretionary and discretionary measures to minimize the impact of the take on sea turtle populations have been included. Incidental take of marine mammal species is not exempted under Sect. 101(a)(5)(e) of the Marine Mammal Protection Act, therefore, take under the ESA has not been authorized for this fishery.

This concludes formal section 7 consultation on the U.S. WCPO purse seine fishery. As stated in 50 CFR 402.16, the International Fisheries Program should reinitiate formal consultation if the amount or extent of take specified in the ITS is exceeded; if new information reveals effects of the action that may affect listed species in a manner or to an extent not previously considered; if the action is subsequently modified in a manner that causes an effect to the listed species or critical habitat that was not considered in the biological opinion; or if a new species is listed or critical habitat designated that may be affected by the proposed action.

Attachments:
Biological Opinion

cc: PIR – W. Robinson
GC-PIR – S. Grimes



Endangered Species Act – Section 7 Consultation

Biological Opinion and Incidental Take Statement

Action Agency: National Marine Fisheries Service, Pacific Islands Region,
International Fisheries Program

Activity: The U.S. Western and Central Pacific Purse Seine Fishery as
authorized by the South Pacific Tuna Act and the High Seas Fishing
Compliance Act

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

Approved By:



William L. Robinson
Regional Administrator, Pacific Islands Region

Date Issued:

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1 Introduction

Section 7(a)(2) of the Endangered Species Act of 1973 (ESA), as amended (16 U.S.C. 1539(a)(2)) requires each Federal agency to ensure that any action they authorize, fund, or carry out is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such species. When 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 United States Fish and Wildlife Service (USFWS), depending upon the protected species that may be affected. For the actions described in this document, the action agency is the International Fisheries Program (IFP) of NMFS, Pacific Islands Region (PIR). The consulting agency is the Protected Resources Division (PRD), also of NMFS PIR.

This document represents NMFS’ biological opinion (Opinion) of the effects of the continued authorization of the United States (U.S.) Western and Central Pacific (WCPO) purse seine fishery on species protected under the ESA. This Opinion is based on our review of the April 24, 2006 Biological Assessment (BA) prepared by the IFP, recovery plans for U.S. Pacific populations of listed sea turtles and humpback whales, the most current marine mammal stock assessment reports, published and unpublished scientific information on the biology and ecology of threatened and endangered whales and sea turtles in the action area, population dynamics modeling efforts, monitoring reports from prior fishing activity and research in the region, and biological opinions on similar actions.

2 Consultation History

The IFP requested initiation of formal section 7 consultation on April 24, 2006, for the U.S. purse seine fishery operating in the WCPO. The IFP provided PRD with a BA of the effects of the proposed fishery on species listed under the ESA. The BA concluded that the fishery was likely to adversely affect five species of listed whales (sperm, *Physeter macrocephalus*; blue, *Balaenoptera musculus*; humpback, *Megaptera novaeangliae*; sei, *Balaenoptera borealis*; and fin, *Balaenoptera physalus*) and five species of listed sea turtles (green, *Chelonia mydas*; hawksbill, *Eretmochelys imbricate*; leatherback, *Dermochelys coriacea*; loggerhead, *Caretta caretta*; and olive ridley, *Lepidochelys olivacea*). PRD reviewed the BA and determined that the information contained in the BA was sufficient to initiate formal consultation. Consultation was initiated via memorandum from PRD to IFP on April 26, 2006.

3 Description of the Action

NMFS IFP proposes to issue regulations for continued authorization of the U.S. purse seine fishery (fishery) operating in the WCPO. This fishery is governed by the “Treaty on Fisheries between the Governments of Certain Pacific Island States and the Government of the United States of America” also known as the South Pacific Tuna Treaty (Treaty), an international agreement to which the United States is a party. NMFS IFP implements the terms of the treaty by issuing regulations under the authority of the South Pacific Tuna Treaty Act of 1988 (SPTA) (16 U.S.C. Chapter 16C). The regulations considered as part of this action include both regulations currently in effect (50 CFR 300.30 et seq.) and new regulations developed by the IFP to implement Treaty amendments agreed to in the Third Extension of the Treaty in 2002 and technical modifications resulting from the Seventeenth Annual Formal Consultation of the

Parties in 2005. The action also includes regulation of the U.S. WCPO purse seine fishery under the High Seas Fishing Compliance Act of 1995 (HSFCA) (16 U.S.C. 5501 et seq.) and its implementing regulations (50 CFR 300, Subpart B). Accordingly, this consultation includes the effects of the continued authorization of all purse seine fishing subject to U.S. jurisdiction in the WCPO.

3.1 Description of the Treaty and Fishery Management Regime

This section describes the management regime for U.S. purse seine vessels operating in the WCPO.

In the early 1980s, some U.S. purse seine vessels were granted access to areas in the WCPO under multilateral industry-to-government agreements. As a result, several disagreements arose between the U.S. and some of the Pacific Island countries (PICs). In response to these conflicts, the Treaty was negotiated and ratified by the U. S. and a number of PICs, and implemented on June 15, 1988. The Treaty provides licensed U.S. vessels with predictable access to most of the exclusive waters of 16 member states of the Pacific Islands Forum Fisheries Agency (FFA),¹ which, together with the U.S., comprise the parties to the Treaty². The Treaty is organized into a main body (consisting of 12 articles) and two annexes.

The Treaty has been formally renegotiated and extended on two occasions: in 1993, at the end of its first (five-year) period of validity, which led to the Second Extension; and in 2002, at the conclusion of the Second Extension, which led to the Third Extension, and will continue until 2013. The parties also have the opportunity to amend the Treaty in the course of annual formal consultations of the parties. Prior to the Third Extension, most Treaty amendments were minor in terms of their potential effect on fishing operations (changes in the number of available licenses were among the most substantial changes, but the limit on available licenses has rarely been reached).

From a fisheries management perspective, the U.S. WCPO purse seine fishery operating under the Treaty is distinct from other U.S. fisheries operating in the WCPO, such as domestic fisheries managed under the Magnuson-Stevens Fishery Conservation and Management Act (MSA). The Treaty and the SPTA and implementing regulations provide the management framework for the U.S. tuna purse seine fishery within the Treaty Area. Most of the fishing effort (more than 80%) by U.S. purse seiners operating under the Treaty occurs in the Exclusive Economic Zones (EEZs) of the PICs that are party to the Treaty. Of the remaining fishing effort, most occurs on the high seas; only a small amount typically occurs in the U.S. EEZ.

¹ The FFA is comprised of PICs that meet regularly as the Forum Fisheries Committee (FFC) to set regional fisheries policy on sustainable management and development of tuna resources in the WCPO. The FFA Secretariat is led by a Director General and consists of technical and administrative staff who advise and assist member governments in the management, conservation, and utilization of the tuna resource in their EEZs and beyond through enhancing national capacity and strengthening regional solidarity. The FFA Secretariat activities are guided by the FFC.

² The PICs that are party to the Treaty are Australia, Cook Islands, Federated States of Micronesia, Fiji, Kiribati, Marshall Islands, Nauru, New Zealand, Niue, Palau, PNG, Samoa, Solomon Islands, Tonga, Tuvalu, and Vanuatu.

The SPTA authorizes the Secretary of Commerce, with the concurrence of the Secretary of State and after consultation with the Secretary of the department with oversight over the Coast Guard, to issue regulations as may be necessary to carry out the objectives of the Treaty and the SPTA. The Secretary of Commerce has delegated authority and responsibility to NMFS to discharge operational, administrative, and enforcement commitments under the Treaty. U.S. Regulations promulgated under the SPTA are published at 50 CFR 300.30 et seq.

In addition to management measures established under the SPTA, there is one management measure established under the MSA that is relevant to the purse seine fishery. The measure prohibits large vessels (greater than 50 feet in length), including U.S. purse seine vessels, from fishing in the U.S. EEZ within approximately 50 nautical miles (nmi) of the islands of American Samoa, which are located in the Treaty Area.

All U.S. fisheries that take place on the high seas, including the U.S. WCPO purse seine fishery, are subject to the HSFCA and its implementing regulations at 50 CFR 300.10 et seq. The HSFCA is the Federal legislation implementing the Agreement to Promote Compliance with International Conservation and Management Measures by Fishing Vessels on the High Seas, adopted by the Conference of the Food and Agriculture Organization of the United Nations on November 24, 1993. It requires that high seas fishing vessels operate under permits issued by the Secretary of Commerce, be identifiable as such, remain in compliance with international conservation and management measures, and that vessel operators report catch and effort.

The main fishery management measures established under the SPTA and HSFCA are summarized below. Third Extension treaty amendments and technical modifications made in the Seventeenth Annual Formal Consultation of the Parties are specifically noted. NMFS published a proposed rule on August, 10, 2006, to implement these changes.

3.1.1 Access and Licensing

Access to the Treaty Area is provided via a licensing system which permits access to a maximum of 45 U.S. purse seine vessels, 5 of which are reserved for vessels owned by joint ventures between U.S. and PIC interests. The license requirement is codified at 50 CFR 300.32. Vessels may be licensed at any time during the licensing year (June 15 - June 14). While the application process is facilitated by NMFS, licenses are issued by the FFA, which serves as the Treaty Administrator on behalf of the PICs.

The HSFCA regulations require that any U.S. fishing vessel used for the commercial exploitation of living marine resources on the high seas obtain and carry a high seas fishing permit issued by NMFS. Permits may include appropriate restrictions or conditions.

In summary, a U.S. purse seine vessel operating in the WCPO must have an FFA-issued license and, if fishing on the high seas, a NMFS-issued high seas fishing permit.

3.1.2 Area Restrictions

Within the Treaty Area there are several types of designated geographical areas, as described below.

Biological Opinion on the U.S. WCPO Purse Seine Fishery, November 1, 2006

- **Treaty Area:** The Treaty Area, which is about 10 million square miles in size, encompasses what can be characterized as the world's most productive tuna fishing area, from Palau eastward to the Line Islands of Kiribati (Figure 1).
- **Licensing Area:** Articles 1 and 3 of the Treaty identify the Licensing Area which consists of the areas where licensed vessels are permitted to fish. The Licensing Area includes all waters in the Treaty Area except for waters subject to the jurisdiction of the U.S. in accordance with international law; and waters closed to fishing by U.S. vessels ("Closed Areas") in accordance with Annex I of the Treaty.
- **Closed Areas:** Closed Areas are those in which U.S. purse seine vessels are not allowed to fish (entry is not prohibited; however, fishing gear must be stowed when in a Closed Area). These areas typically include territorial seas, internal or archipelagic waters, waters in proximity to or used by domestic-based tuna fisheries in the PIC, or waters proximal to named offshore banks and reefs. In the Treaty, each of the 16 PICs has declared a portion of its waters as a Closed Area.

The Third Extension of the Treaty and the Seventeenth Annual Formal Consultation of the Parties resulted in two changes to the Closed Areas: closure of the archipelagic waters of Papua New Guinea (PNG), which were previously open; and opening of the majority of the Solomon Islands EEZ, which, except for a small Limited Area, was previously closed.

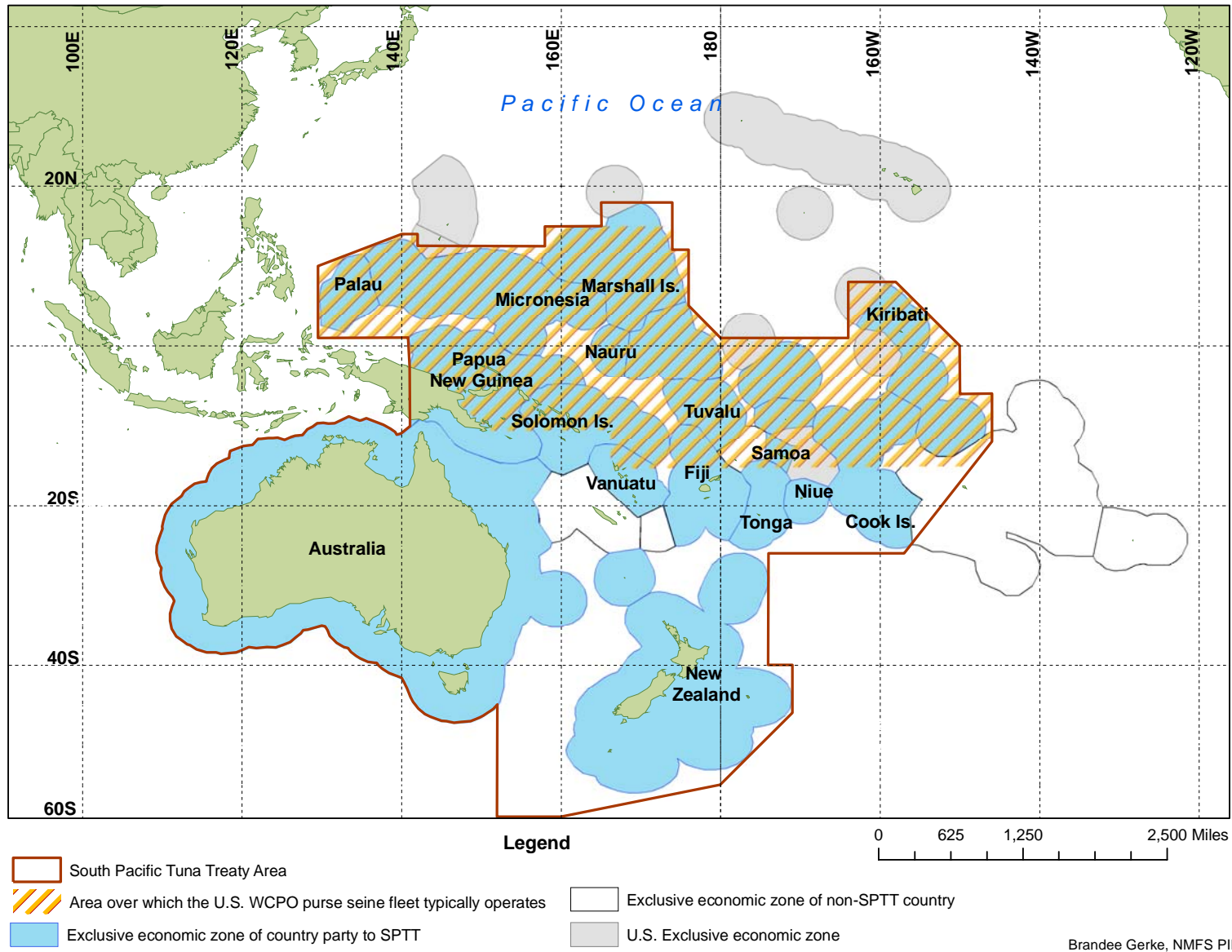


Figure 1. Treaty Area (bounded by red, solid line) and exclusive economic zones of Pacific Island Countries party to the SPTT. The dashed lines indicate the broad area of the Treaty Area over which the U.S. WCPO purse seine fleet typically operates.

3.1.3 Species Restrictions

U.S. purse seine vessels licensed to fish under the Treaty are only permitted to target tuna, though they are prohibited from targeting southern bluefin tuna (*Thunnus maccoyii*).

3.1.4 Vessel Reporting Requirements

Holders of vessel licenses issued under the Treaty are required to submit both written and electronic reports on their fishing activities in the Treaty Area. These include reports submitted to the FFA and a separate set of reports to the relevant PIC.³

Two main categories of reports are made to FFA: “telex reports” and logsheet reports (which nowadays are actually submitted by e-mail or fax, generally). Telex reports provide information on the position of the vessel and catch onboard. These reports are required⁴ before departure from port for the purpose of beginning a fishing trip in the Licensing Area; each Wednesday while within the Licensing Area or a Closed Area; and before entry into port for the purpose of unloading fish from any trip involving fishing in the Licensing Area.

The Treaty also requires two logsheet reports: the “Regional Purse Seine Logsheet” (RPL) completed daily by the vessel’s captain and submitted at the completion of a trip, describes and details the vessel’s daily estimated catch and other related activities; a second logsheet report provides details of offloading, including a quantitative summary of all catch offloaded.

In addition to reports to the FFA, seven of the PICs (Australia, Fiji, Kiribati, New Zealand, Solomon Islands, Tonga, and Tuvalu) have additional national reporting requirements, as identified in Treaty Annex I, Schedule 4, Part 3.⁵

The HSFCA regulations require that operators of vessels with high seas fishing permits report high seas catch and effort information to NMFS. Participants of the U.S. WCPO purse seine fishery are required to report their catch and effort information using the Treaty logsheet, which is the RPL described above (in other words, the SPTAs reporting requirements satisfy the HSFCA’s reporting requirements).

3.1.5 Vessel Monitoring System

Implementation of a vessel monitoring system (VMS) using satellite technology is a standard compliance measure throughout the world used to enhance data collection efforts while

³ To assist in the reporting and timely transmittal of the required information, NMFS acts as a conduit for the transmittal of these reports to the FFA (reporting requirements specifying that the completed logsheet forms are to be provided to the NMFS Regional Administrator are contained in 50 CFR 300.34).

⁴ Each PIC has its own domestic conservation and management measures, therefore telex reports must also be sent to each PIC in whose EEZ a U.S. vessel is about to enter, exit, or transship (the formulas of these reports are the same regardless of the national authority to which they are sent (Annex 1, Schedule 2, Part 2)).

⁵ These range from a requirement by Kiribati (among others) to report at least 24 hours before and immediately upon entry or departure into a Closed Area and at least 24 hours prior to and immediately after refueling from a tanker, to Tonga’s requirement to report daily position by radio or telex while in the Tonga EEZ.

monitoring fishing and other activities. VMS enables information such as a vessel's geographic position to be reported to a monitoring station on shore using an electronic unit known as a transponder. The VMS unit is placed aboard the vessel and information is then relayed via satellite to one or more land-based monitoring stations. All VMS position reports are automatically transmitted without any input or direction from the unit located on the vessel.

In 1992, the PICs recognized the potential value of vessel monitoring technology and agreed to language that was included in Annex 1, Part 8 of the Treaty.⁶ Under the Third Extension of the Treaty, U.S. purse seine vessels are required to carry and operate VMS units.

3.1.6 Enforcement Provisions

The Treaty requires the U.S., as a flag State, to enforce the provisions of the Treaty and the license conditions. The U.S. is also required to investigate specific incidents or alleged violations and take action against vessels that have not submitted to the jurisdiction of the PIC concerned (Article 4). A key provision of the Treaty eliminates the ability of (and need for) the U.S. to apply sanctions or restrictions on trade as a result of enforcement measures taken by a PIC, as long as those measures are consistent with the Treaty (Article 5.4).

Another provision of the Treaty requires a PIC to promptly release U.S. fishing vessels confiscated and any crew arrested for breach of the Treaty upon the posing of a reasonable bond or other security, and prohibits imprisonment or corporal punishment by the PIC for fishing violations under the Treaty (Article 5.3). Should a U.S. vessel be involved in an alleged infringement of the Treaty and not submit to the jurisdiction of the PIC, the U.S. is required to investigate. Any penalty assessed should be similar in amount to violations of U.S. law relating to foreign fishing vessels licensed to fish in the EEZ of the U. S., and not exceed the sum of \$250,000 (Article 4.6). The SPTA and HSFCA have their own respective enforcement provisions.

3.1.7 Vessel Observer Program

The Treaty provides for a vessel observer program for the U.S. fishery with a target coverage of 20% (in terms of trips), to be administered by the FFA. Under the Treaty and the SPTA regulations:

- The operator and each crew member of a vessel shall allow and assist any person identified as an observer under the Treaty by the Pacific Island Parties: full access to the bridge of the vessel; the vessel's records, including its logs and documentation for the purpose of records inspection and copying; catch on board; and areas which may be used to hold, process, weigh, and store fish. No operator or crew member of the vessel shall assault, obstruct, resist, delay, refuse boarding to, intimidate or interfere with an observer in the performance of his or her duties.

⁶ "It is understood that a region-wide vessel tracking system applicable to all vessels licensed to fish in the Treaty Area may be established. U.S. vessels with a license to fish under the Treaty shall participate in such a system and shall install and operate a transponder of a type and in such a manner as may be agreed by the parties. It is understood that data derived through the system shall be treated as confidential business information and that the terms and conditions for access to that information shall be a matter of discussions between the Parties."

- The operator shall provide the observer, while on board, with food, accommodation and medical care of a reasonable standard at no expense to the Pacific Island Parties.
- The vessel operator is required to meet the costs of travel, salary, and insurance for the observer.
- The observer program shall be conducted in accordance with this Treaty and provisions that may be agreed from time to time.

3.1.8 International Conservation and Management Measures

The HSFCA regulations require that vessels fishing on the high seas not be used in contravention of “international conservation and management measures”, which are measures for living marine resources that are recognized by the U.S. and that have been adopted and applied in accordance with international law. The U. S. implements such measures, where appropriate, via rulemaking.

3.1.9 Vessels Other Than Purse Seiners

Initial application of the Treaty was intended to apply only to activities of U.S. purse seine vessels. However, the Treaty provides flexibility for a PIC to permit other U.S. vessels to fish in the waters of that country, in which case the country must obtain U.S. concurrence to such an arrangement.⁷

3.2 Description of the Fishery

In this Opinion, we examine the effects of the U.S. WCPO purse seine fishery on listed species. However, the U.S. fleet is only one of many national fleets participating in the WCPO purse seine fishery. To put the U.S. fishery in perspective, the collective WCPO purse seine fishery of all nations is described first.

Purse seine vessels, which mainly target skipjack and yellowfin tuna, harvest about 60% of the world production of tuna (Joseph, 2003). The WCPO tuna purse seine fishery is the world’s largest tuna fishery, and produces 25-30% of the global tuna catch each year (Gillett and Lewis, 2003).

Williams (2003b) noted that purse seiners have accounted for approximately 55-60% of the WCPO total catch by weight since the early 1990s, with annual catches ranging between 790,000-1,200,000 metric tons (mt). The vast majority of the catch in the WCPO is taken in equatorial waters between 10° N and 10° S. Smaller seasonal fisheries exist in subtropical waters, such as off Japan and to a lesser extent off New Zealand.

3.2.1 Purse Seine Operations

The purse seine technique for catching tuna involves setting a net vertically in the water, with floats attached to the upper edge and chains for weight on the lower edge. A series of rings is

⁷ After such concurrence is received, the provisions of the Treaty relating to flag State responsibility and corresponding legal proceedings apply (Article 3).

attached to the lower edge of the net, and a pursing cable passes through the rings, enabling a winch on board the vessel to draw the net closed on the bottom. Purse seine nets can be up to 1,600 meters (m) or more in length and 220 m in depth. When the net is deployed from the purse seine vessel, a large skiff carrying the end of the net is then released from the stern of the fishing vessel. The purse seine vessel encloses the school of tuna, keeping it in visual contact if on the surface, or using sonar if below the surface, and then retrieves most of the net onto the vessel. The fish are confined in the “sack” portion of the net, which consists of finer mesh webbing that prohibits their escape. The catch is removed from the sack onto the vessel with large “scoops” holding 1 mt or more, and then is placed in brine tanks for freezing and later storage.

Purse seiners are one of the most complex classes of fishing vessels in terms of both technology and machinery. Hydraulic systems on large “super seiners” require more than 1,600 m of piping, and are equipped with at least four auxiliary engines in addition to the main propulsion engine (or engines). Purse seine gear configuration is shown in Figure 2.

Tuna purse seine vessels in the WCPO range in length from 50-115 m with the largest being able to hold up to 3,000 mt or more of frozen fish. Most tuna seiners are 70-80 m in length and can carry approximately 800-1,500 mt of frozen fish. Some vessels also carry helicopters that can improve their ability to find schools of fish and assist in keeping track of the school once the net is set.

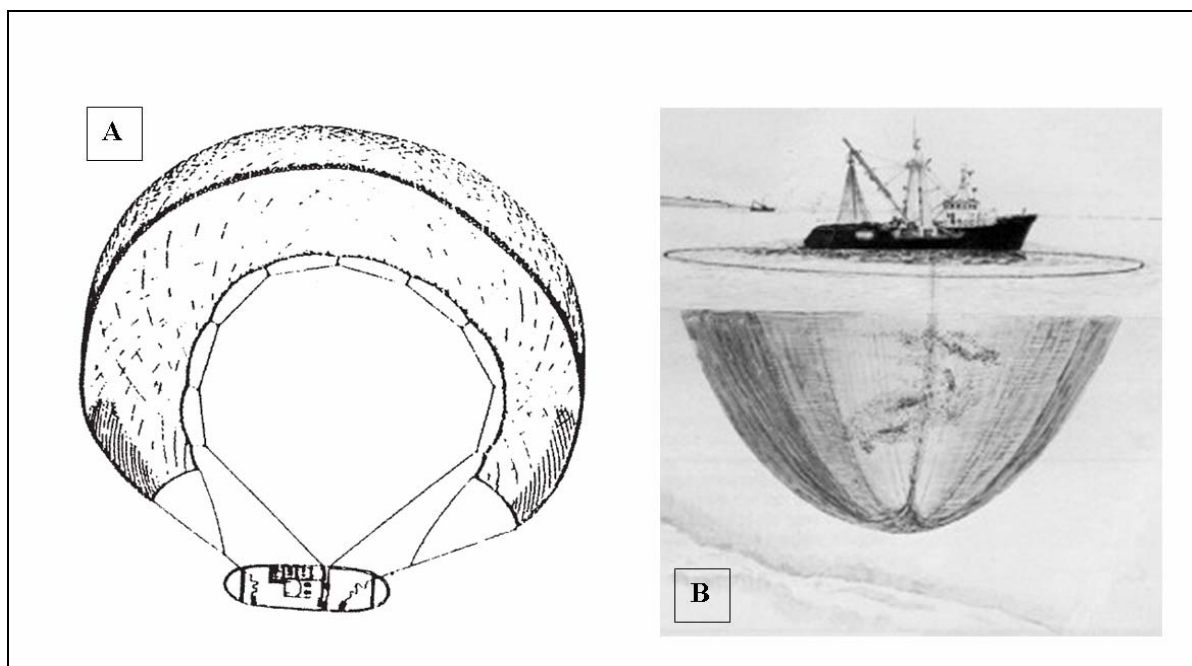


Figure 2. Examples of purse seine net deployment from above (A) and the side (B). (Source for A: *Commercial Fishing Methods: An introduction to vessels and gears*, 3rd ed. by John C. Sainsbury, published by Blackwell Science; Source for B: <http://www.cdli.ca/cod/purse.htm>).

3.2.2 Fishing Method by School Association

Purse seiners in general set on a variety of school types or ‘associations’, ranging from tuna schools associated with floating objects, such as logs and other naturally occurring debris, man-

made fish aggregating devices (FADs), and dead whales, to schools swimming with live animals such as whales and whale sharks. Tuna schools not associated with floating objects or other animals are also set upon. These schools are free-swimming or “unassociated” schools feeding on baitfish or schools associated with geographic features such as seamounts and islands, or with oceanographic features such as current interfaces and areas of upwelling. Such sets are collectively termed school sets (Bailey et al., 1996) or unassociated sets. These set types have been assigned to various categories for reporting purposes as the associations largely determine the catch composition and the quantity and kinds of by-catch and discards in the fishery.

Log Associations

Schools of tuna aggregate around logs and other floating debris which often concentrate along productive current or water mass interfaces throughout the WCPO. Tuna are associated with them for a variety of possible reasons (e.g., feeding, shelter, orientation) and a viable purse-seine fishery in the Western tropical Pacific (WTP) was initially based on seining tuna schools associated with drifting objects (Doulman, 1987). Logs can consist of sections of trunk, groups of branches or entire trees. Other debris includes almost any floating object that is washed or drifts out to sea or is jettisoned from ships, e.g., canoes and boats, drums, cable spools, polystyrene floats, discarded mooring lines, and wooden pallets. Most occurrences within this association type, however, involve logs. Log sets are usually made immediately before dawn, at a time when tuna are most vulnerable to purse seining as they are concentrated close to the log and cannot see and avoid the encircling net (Bailey et al., 1996).

FAD Associations

FADs in the WCPO operate much like logs in terms of fish aggregation, how the tuna behave in their vicinity, and the general strategies used by seiners to set on them. Two basic types of FAD association are recognized. The first involves FADs that are anchored in place, usually within a network of similar units, and the second occurs with FADs that have broken loose from their mooring lines and drifted away, or have been deliberately deployed without mooring lines. Within the second category, the Japanese appear to include associations with logs and debris that have been roped together (Tanaka, 1989). The Japanese are also known to anchor FADs near small islands and release them to drift after a suitable ‘ageing’ period has resulted in the accumulation of encrusting life and a population of baitfish. A large volume of literature exists on the types and designs of FADs in use in the Western Pacific Ocean (WPO) (e.g., Preston, 1982; Malig et al., 1991).

Animal Associations

Though prohibited in the U.S. WCPO purse seine fleet, another type of “associated” set includes “animal associations” which may consist of two distinct association types: tuna aggregating and feeding with sei whales and, to a lesser extent, minke whales (*Balaenoptera acutorostrata*) and schools associated with the slow-moving whale shark. Tuna schools found with live whales do not appear to form long-term associations with the whales; they seem only to come together to feed on pelagic baitfish schools and separate once the feeding activity is finished. In this sense, these schools are similar to the unassociated schools described below, and are set on in the same way. The seiner will, however, attempt to encircle the whale during the setting operation, as the tuna will tend to remain close to the whale, thus improving the chance for a successful set. Once pursed, the whale escapes by punching a hole through the net (Bailey et al., 1996).

Whale shark associations appear to be intermediate between live whales and logs in that the shark and tuna often come together to feed on anchovy but can maintain the association for some time in the absence of feeding behavior, much like tuna aggregating under a slow-moving log. Whale sharks are set on during the day, as it is impractical to mark them with buoys and therefore difficult to locate them in the dark. The amount of non-target fish, marine turtle, and marine mammal bycatch associated with these categories is typically low. In comparison, schools found associated with floating whale carcasses are similar to log associations, with large attendant schools of bait-fish species. Dead whales are rarely encountered but when so, are treated like logs, marked with radio and light buoys for tracking and set on before dawn.

Unassociated Schools

Unassociated schools are typically surface schools that range in activity from fast moving ‘breezers’ that appear like a breeze blowing across the sea surface to stationary ‘boilers’ and ‘foamers’ consisting of tuna churning the surface into a white froth while feeding on pelagic bait fish and other forage. The latter types of schools are most preferred for seining as the tuna are distracted by their feeding frenzy and easier to encircle with the seine. In comparison, breezing schools are more erratic in behavior and are often moving at speed, making them difficult to encircle and catch. School fishing in the WTP has required that nets be lengthened to effectively encircle the fast-moving schools and deepened to extend below the depth of the WTP thermocline. A typical U.S. net currently measures over 1,500 m long by 220 m deep. Along with these developments, there have been increases in mesh size and reductions in twine size to allow the net to sink faster with reduced water resistance during pursing and net retrieval, and increases in purse winch power allowing net pursing to be conducted in less than 15 minutes.

3.2.3 Participants in the WCPO Purse Seine Fishery

The Secretariat of the Pacific Community (SPC) (2002b) lists 18 countries whose fleets have participated in the WCPO purse seine fisheries at some time during the last 15 years. More than 70% of the catch is taken by four main distant-water fishing nations (DWFNs), Japan, Korea, Taiwan, and the United States, which together account for approximately 140 vessels. In addition to the 140 vessels, some Pacific Island domestic fleets have recently been contributing to the catch in the WCPO, particularly vessels flagged in PNG, Federated States of Micronesia (FSM) and the Republic of the Marshall Islands (RMI). The numbers of vessels participating in the fishery, by nation, during 1988, 1995, and 2003 are shown in Table 1. The estimated WCPO purse seine catches (mt) by the four major DWFN fleets from 2000-2002 are shown in Table 2.

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Table 1. Number of Active Vessels in the WCPO Purse Seine Fishery, by Flag State

Fleet	1988	1995	2003	Change Since 1988
Japan ^a	34+5	33	34 (1)	-5
United States	32	43	20 (6)	-12
Korea	23	30	27	+4
Taiwan ^a	16+2	42	38	+20
China	0	0	4	+4
Solomon Islands	4	3	1 (1)	-3
Papua New Guinea	0	3	7	+7
FSM	0	5	9	+9
Marshall Islands	0	0	6	+6
Kiribati	0	1	1	+1
Vanuatu	0	2	15 (3)	+15
New Zealand distant-water	0	0	4	+4
Australia distant-water	3	0	0	-3
Spain ^b	0	0	1 (7)	+1
Netherlands Antilles	0	0	1	+1
Panama	0	0	1	+1
U.S.S.R.	5	0	0	-5
Philippines distant-water	9	13	22	+13
Indonesia distant-water	3	0	0	-3
Total	136	175	191	+56

Source: Gillett and Lewis (2003)

Note: Only vessels for which catch records exist have been included in the number of active vessels in each National Fleet in the time periods covered by this table.

Numbers in parentheses indicate the number of additional vessels that appear on the Forum Fisheries Agency (FFA) Regional Register but were not licensed to fish under access arrangements in 2003 when the list was compiled.

^a The seven Japan and Taiwan vessels following plus signs (+) in 1988 are group-seining operations.

^b Includes the Spanish-owned vessels flagged in El Salvador and Guatemala.

Table 2. Estimated purse seine catches (mt) by the four major DWFN fleets, 2000–2002. Sources: SPC (2002b, 2003a).

Nation	2000	2001	2002
Japan	232,593	225,812	211,960
Korea	170,025	178,072	180,087
Taiwan	234,978	230,668	258,126
United States	125,351	115,524	119,158
Total	764,947	752,077	771,333

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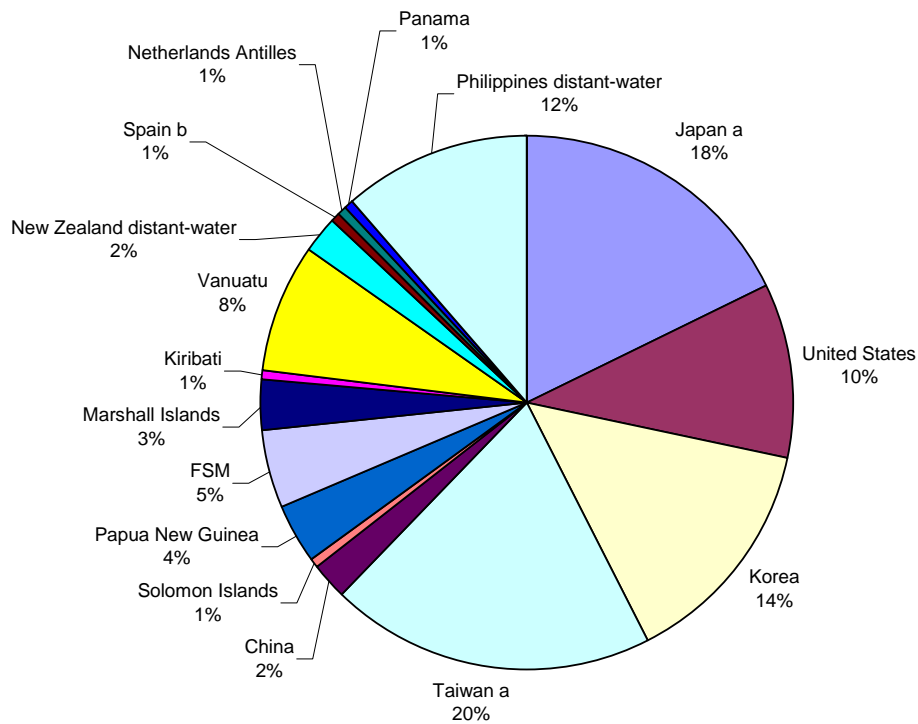


Figure 3. Percentage the number active vessels in the WCPO purse seine fishery in 2003, by country.

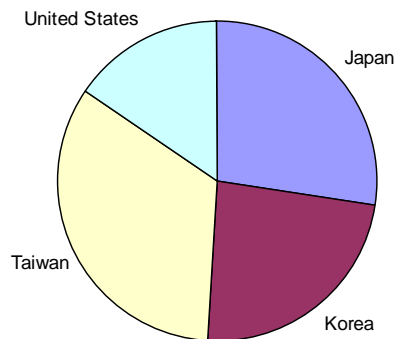


Figure 4. Chart of the estimated purse seine catch (mt) for the four major DWFN fleets for the year 2002 only.

Williams (2003a) states that the provisional 2002 purse seine catch by all fleets was 1,157,045 mt, the second largest catch on record. The largest recorded catch was in 1998, when more than 1,200,000 mt were landed.

In 2001, the majority of purse seine effort by vessels of the four DWFNs in the WCPO occurred between 150°E and 180°E, and between 5°N and 10°S. During 2000, the majority of effort by both Korean and Taiwanese fleets was considerably to the west of 150°E, while Japanese and U.S. fleets stayed to the east of that longitude. Taiwanese and Korean fleets appear to have expended approximately 10% or less of their fishing effort west of 160°E during 2001. The locations of effort by Taiwanese and Korean fleets coincided with Japanese and U.S. fleets eastward from 160°E (SPC, 2002b).

In the late 1990s there was a noticeable increase in the reliance of drifting FADs in the WCPO purse seine fishery. A notable exception is the Korean fleet, which continues to rely on fishing without using drifting FADs. Most fleets use drifting FADs as a means of aggregating and holding tuna to make them more vulnerable to purse seining. However, by 2001, the percentage of sets on drifting FADs dropped for all fleets. For the first time since 1998, the proportion of drifting FAD sets for the U.S. fleet was less than for unassociated free-swimming schools. Provisional 2003 data from the U.S. fleet indicate a continuing trend away from sets on drifting FADs (OFP, 2004).

Thailand is the single largest purchaser of WCPO purse seine-caught fish, although multiple destinations exist for the catch of the various purse seine fleets.⁸ Taiwanese and Korean fleets transship the majority of their catch in various ports in the western Pacific region. The Taiwanese sell their catch mainly to canneries in Thailand, while the Koreans split their catch between Thailand and Korean canneries. Fish caught by the Japanese purse seine fleet is returned to Japan. The U.S. fleet, which operates out of American Samoa, unloads more than 85% of its catch to the two canneries there, and the remaining catch goes to Fiji, Thailand, the Philippines, and more recently the RMI.

3.3 U.S. WCPO Purse Seine Fishery

The U.S. WCPO purse seine fleet originated in Southern California, or more generally, the Eastern Pacific Ocean (EPO), where a tuna fishery has operated since the 1920s. Several factors, including unilateral implementation of 200-mile EEZs extended jurisdictions by Latin American countries, expansion of fishing fleets in several of those countries, and U.S. domestic legislation protecting dolphins, motivated vessel owners to look elsewhere for productive fishing areas (Doulman, 1987).

The U.S. fishery in the equatorial WCPO began growing after “U.S. style” Japanese purse seiners successfully developed techniques to capture schools of tuna found in association with natural drifting objects, primarily in the waters between PNG and the FSM (Watanabe, 1983). Shortly thereafter, U.S. purse seiners adopted and refined these successful seining techniques. However, the U.S. fleet eventually returned to “traditional” pursing methods, and thus began to purse unassociated schools of large yellowfin and skipjack. By 1988, unassociated school sets accounted for approximately 80% of the fishing effort by the U.S. fleet.

⁸ Depending on supply and demand, purse seine-caught fish from the western Pacific can be sent to canneries in South America and as far away as Europe, in addition to canneries in Thailand and within the Pacific Island region.

A major turning point for the U.S. purse seine fleet came during an intense *El Niño* - Southern Oscillation (ENSO) event that occurred in 1982-1983, causing many of the U.S. eastern Pacific purse seiners to move fishing efforts to the WCPO. In subsequent years, vessel numbers in the Pacific decreased overall; however, more vessels were active in the western Pacific than in the eastern Pacific.

Fishing grounds for the U.S. purse seine fleet continued to expand eastward throughout the 1980s, eventually encompassing the Phoenix and Line Islands (Kiribati); the U.S. possessions of Howland, Baker, and Jarvis; Tokelau; and the high seas areas between these EEZs.

By the mid 1990s, only a few U.S. purse seine vessels operated in the eastern Pacific. Figure 2 depicts the numbers and general geographic fishing location of the U.S. purse seine fleet operating in the Pacific from 1976 to 2002.

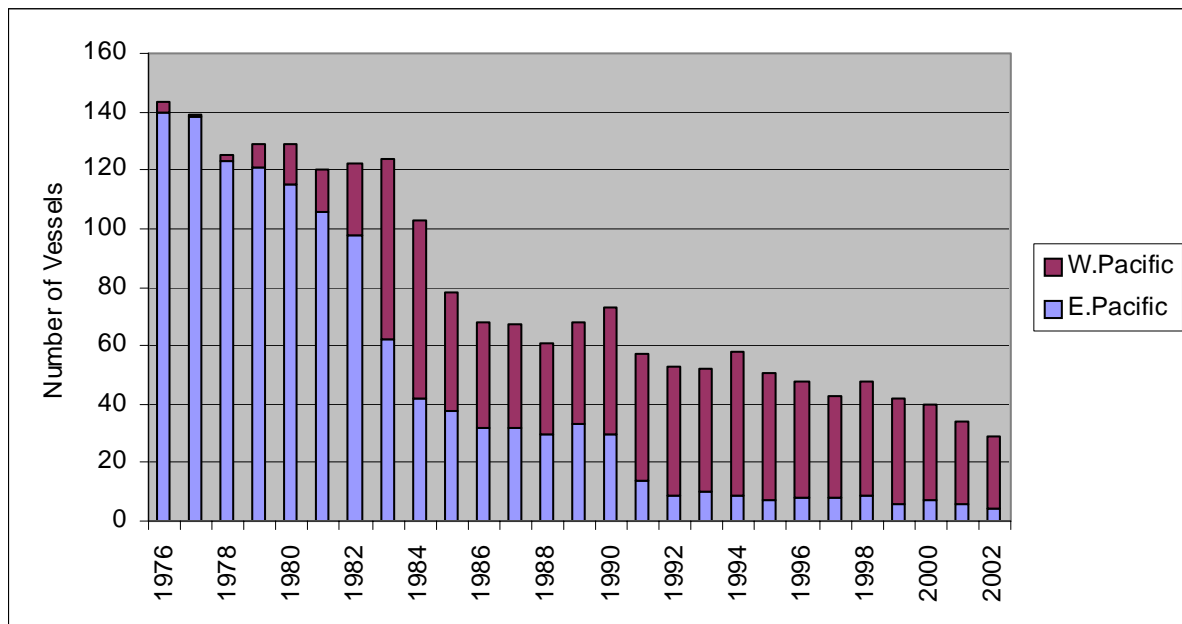


Figure 5. Numbers of Vessels in the U.S. Purse Seine Fleet for the Western Pacific and Eastern Pacific Oceans. Source: Gillett et al., (2002)

During 1995-1996, the fishing strategy of the U.S. fleet shifted again, to a higher reliance on “associated” setting and the utilization of drifting FADs, as shown in Figure 6. This allowed the U.S. fleet to operate in the eastern area of the fishery, where natural logs were scarce. As a result, these catches contained high proportions of smaller tunas (such as skipjack, and juvenile yellowfin and bigeye tuna) and bycatch species, thus eventually depressing ex-vessel value on a per-ton and per-trip basis (Coan and Itano, 2003). Since 2000, much of the fleet has reduced its reliance on drifting FADs and currently pursues unassociated schools of larger fish, as a result of depressed cannery prices for small skipjack, yellowfin, and bigeye.

Fishing operations for the U.S. WCPO fleet will likely continue as a combination of unassociated, log-associated, and drifting FAD-associated seining. This mix of fishing operations

is thought to occur as a result of various factors including: ex-vessel price of tuna, ENSO conditions, and inter-annual variability in availability of tropical tuna species.

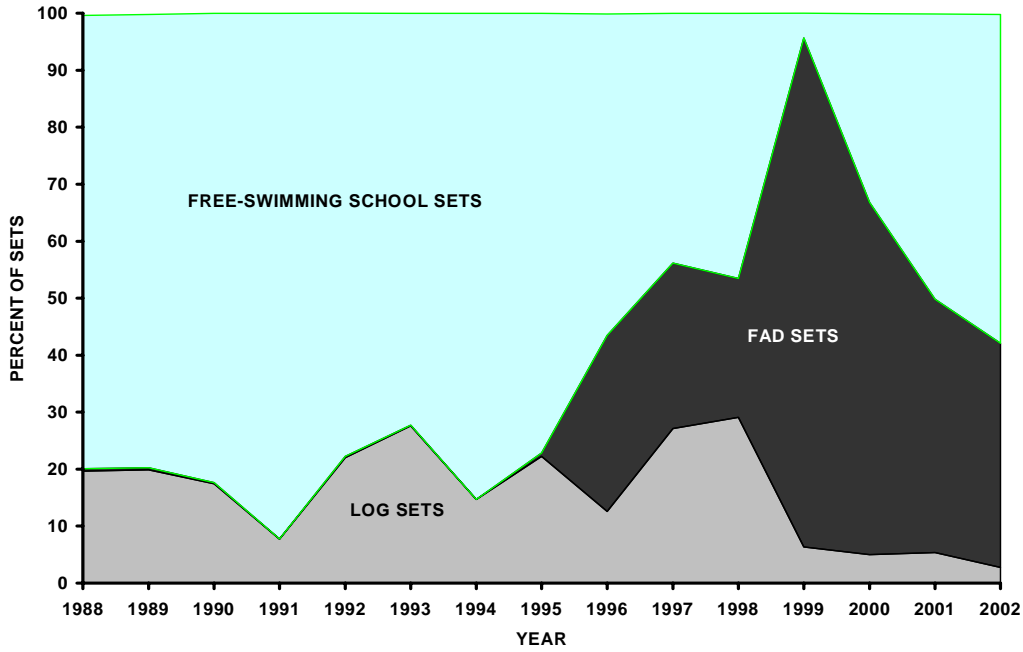


Figure 6. Proportion of Set Types for the U.S. WCPO Purse Seine Fleet, 1988–2002. Source: Coan and Itano (2003).

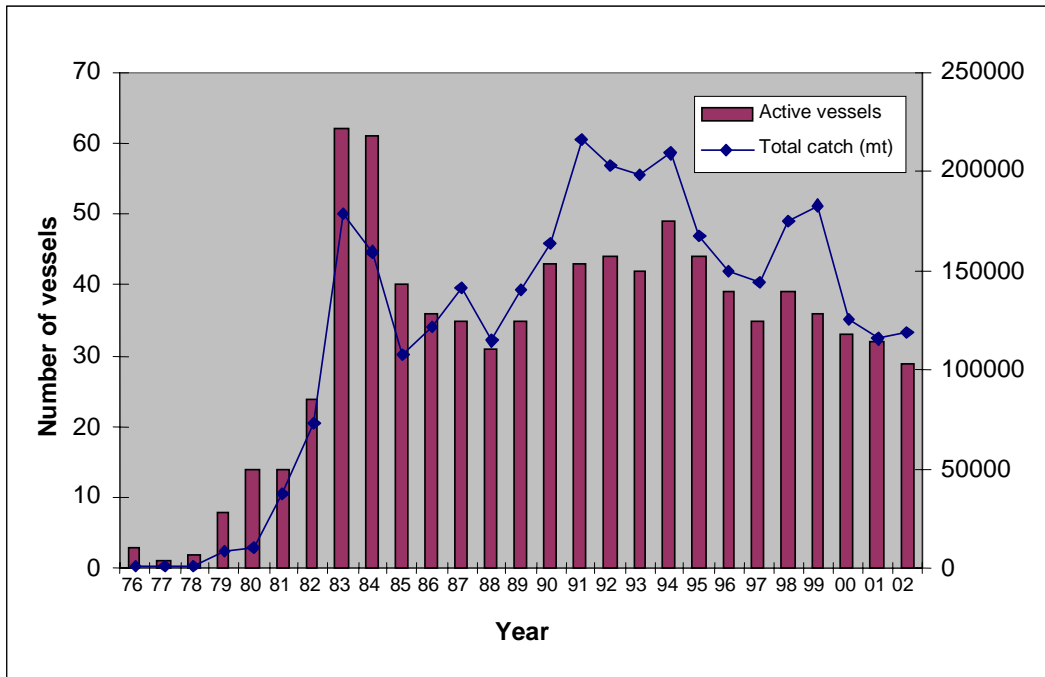


Figure 7. Number of U.S. Purse Seiners and Total Annual Catch (mt) in the WCPO for 1976-2002. Source: SPC (2003a).

Thirty-two U.S. vessels were fishing in the WCPO in 1987–88 (Figure 7). From 1990-1995, effort increased to more than 40 vessels per year, peaking at 49 active vessels in 1994. Vessel numbers have gradually decreased since the late 1990s. The standard “licensing year” under the Treaty is 15 June of one year to 14 June of the following year. As of May 2005, there were 17 purse seine vessels licensed to fish under the Treaty. In mid-June 2005, 14 U.S. purse seine vessels renewed their licenses to fish under the Treaty. Given the prevailing economic conditions, this decreasing trend is not anticipated to reverse itself in the near term. From a historical high catch of 216,000 mt in 1991, the catch decreased to less than 120,000 mt in 2002 (Gillett and Lewis, 2003; SPC, 2003b). Catch rates during the history of the fishery have not shown any clear trend. The greatest catch-per-unit-of-effort was recorded in 1999, at 34.1 mt per day (NMFS, 2004a).

3.3.1 Catch and Effort

The highest catch rates by U.S. purse seine vessels were achieved during or following strong ENSO events such as those observed in the 1983–84, 1991–92, and 1998–99 seasons. High catch rates during the 1998–99 period were also driven by the high percentage of drifting FAD sets that increased vessel efficiency. Total catch by the fleet peaked in 1991, at 216,000 mt taken by 43 vessels. This catch was driven by high catch rates of skipjack taken in unassociated sets.

In 1995, the average Gross Registered Tons (GRT) of the U.S. purse seine fleet was 1,181 with an overall average vessel length of 64.1 m. By 2003, the average GRT had risen to 1,241, and average vessel length to 73.2 m. Fish carrying capacity, an estimate of tonnage, varies as a result of the size and species of fish loaded onto the vessel. Fish carrying capacity was estimated to be approximately 31,600 mt for the U.S. fleet as a whole, with an average capacity of 1,264 mt (Gillett and Lewis, 2003).

The increase in overall capacity of the fleet can be explained partially by the physical size increase of existing vessels. Ten U.S. purse seiners were “stretched,” which involved cutting the ship aft of the deckhouse and adding hull and fish wells to increase vessel carrying capacity. These capacity increases can be very significant, with some vessels increasing their hold capacity by more than 50%. Currently, vessels in the U.S. fleet can carry approximately 1,000–1,770 mt, depending on the mix and sizes of species in the catch. Figure 8 indicates catch by species for the U.S. fleet (SPC, 2003b).

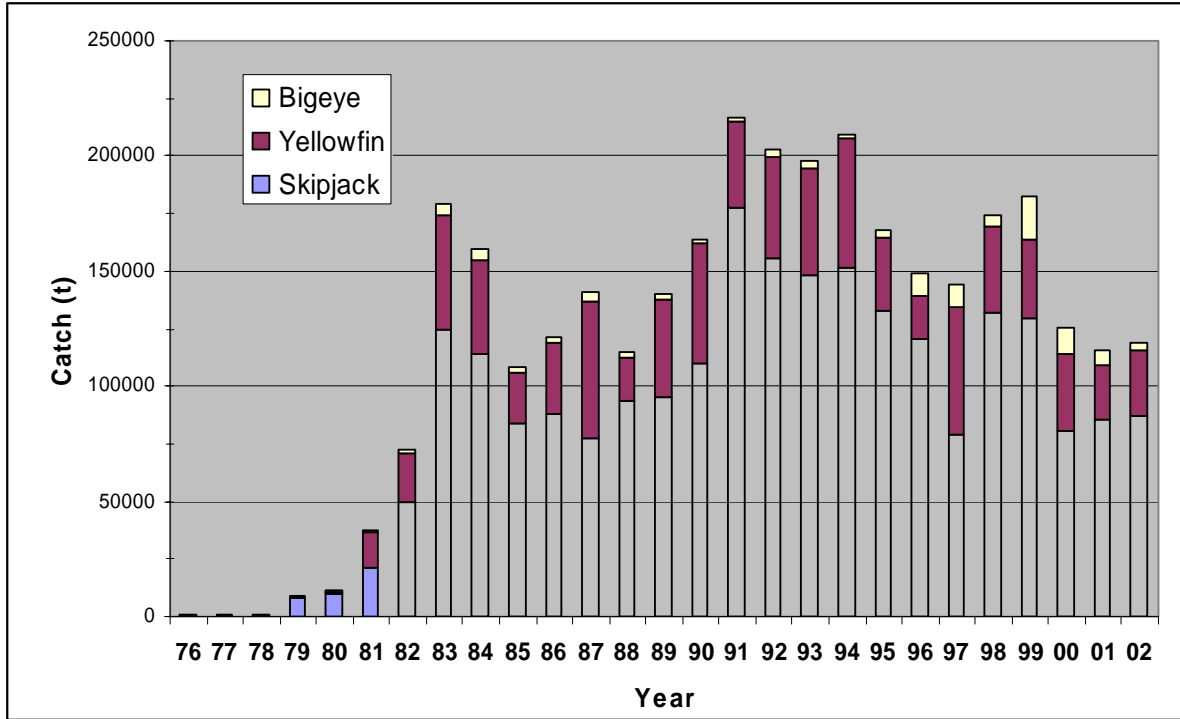


Figure 8. Total Catch (mt) by Species for the U.S. WCPO Purse Seine Fleet. Source: SPC (2003b)

Historically, the U.S. WCPO purse seine fishery has targeted mostly skipjack. The 27-year mean values (years 1976–2002) for species composition taken by the U.S. fleet are 73.3% skipjack, 23.8% yellowfin, and 2.9% bigeye. During this period, skipjack ranged from a low of 54.9% in 1987 (when high catches of yellowfin occurred) to 92.8% in 1979; and yellowfin ranged between 6.7% and 42.8% and usually were close to the long-term mean. Estimated landings of bigeye peaked in 1999 at 10.2% of the catch, which can generally be attributed to the high percentage of associated (FAD) sets that year.

Figure 9 illustrates U.S. purse seine effort during a transitional year between an *El Niño* and *La Niña* period in 2001, and an *El Niño* period in 2002. During strong *La Niña* conditions, purse seine effort usually shifts west of 160°E longitude (shown by the vertical line). However, in recent years, the U.S. fleet has been able to maintain high catch rates in the eastern region through the deployment of drifting FADs close to their homeport of Pago Pago, American Samoa (Gillett et al., 2002).

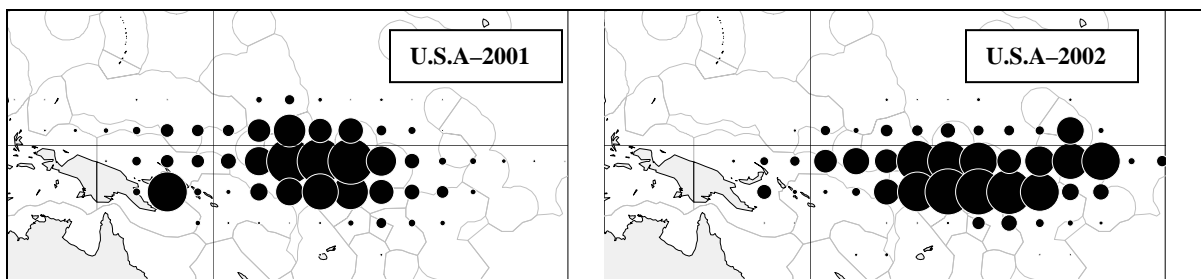


Figure 9. Distribution of U.S. Purse Seine Effort during 2001 and 2002. The largest circle size indicates ≥ 360 days fishing or searching. Source: Williams (2003c).

3.3.2 Landing Points and Disposition of Catch

Historically, the U.S. fleet unloaded the majority of its catch at the two canneries in Pago Pago, American Samoa, located slightly beyond the south-eastern limit of the purse seine fishery. Pago Pago continues to be the principal unloading port of the U.S. fleet. From 1980 to the early 1990s, a significant component of the fleet (14 vessels) transhipped fish from Guam and Tinian (located in the Northern Mariana Islands, also considered the western limit of the U.S. WCPO purse seine fishery). The company involved in those operations became insolvent in 1995, and U.S. firms along with other interests have since purchased these vessels. A few transshipments continue to take place in Kiribati and the RMI, particularly during *El Niño* periods. Sporadic offloading directly to processors has also occurred in the Philippines, Thailand, and Australia, and to a lesser extent in Fiji and the RMI.

3.3.3 Factors Affecting the Distribution of Purse Seine Effort and Catch

Purse seine fishing effort in the WCPO is not characterized by any marked seasonal patterns. The spatial distribution of fishing effort is, however, strongly influenced by the (irregular) cycles associated with ENSO events. Since the early 1980s it has been recognized that catch distribution of tunas in tropical areas of the WCPO is strongly influenced by ENSO events.⁹ Generally, purse seine fishing takes place further to the east during *El Niño* periods, and contracts westward during periods of *La Niña*. Fishing can move as far east as the Line Islands in Kiribati, approximately 150°W during a strong *El Niño* year. During the so-called “transitional” (or normal) periods, a greater distribution of effort occurs in the central and western portions of the region, sometimes centered at 160°E.

However, during *El Niño* periods, waters of the WCPO equatorial region experience consistent, westerly winds, that result in wind-generated, eastward-flowing currents. The wind and currents tend to move natural debris (logs and other flotsam) from large land masses such as PNG and eastern Indonesia further eastward than normal. This natural debris tends to aggregate tuna, and is generally most abundant in eastern areas during an *El Niño* year.

Figure 10 shows the distribution of purse seine effort for all fleets during 1997, a strong *El Niño* year. The contrast in the distribution of purse seine effort during a *La Niña* period is provided in Figure 11. In both figures, larger black circles represent where the largest amount of effort occurred, and the smaller black circles depict proportionally smaller amounts of effort.

⁹ ENSO events include the full range of variation observed between *El Niño* and *La Niña* events. *El Niño* is characterized by a large-scale weakening of the tradewinds and warming of the surface layers in the eastern and central equatorial Pacific. *El Niño* events occur irregularly at intervals of 2–7 years, although the average is about once every 3–4 years. These events typically last 12–18 months, and are accompanied by swings in the Southern Oscillation, an interannual “see-saw” in tropical sea level pressure between the eastern and western hemispheres. During *El Niño*, unusually high atmospheric sea level pressures develop in the western tropical Pacific and Indian Ocean regions, and unusually low sea level pressures develop in the southeastern tropical Pacific. Southern Oscillation tendencies for unusually low pressures west of the dateline and high pressures east of the dateline have also been linked to periods of anomalously cold equatorial Pacific sea surface temperatures sometimes referred to as *La Niña* (NOAA, 2004)

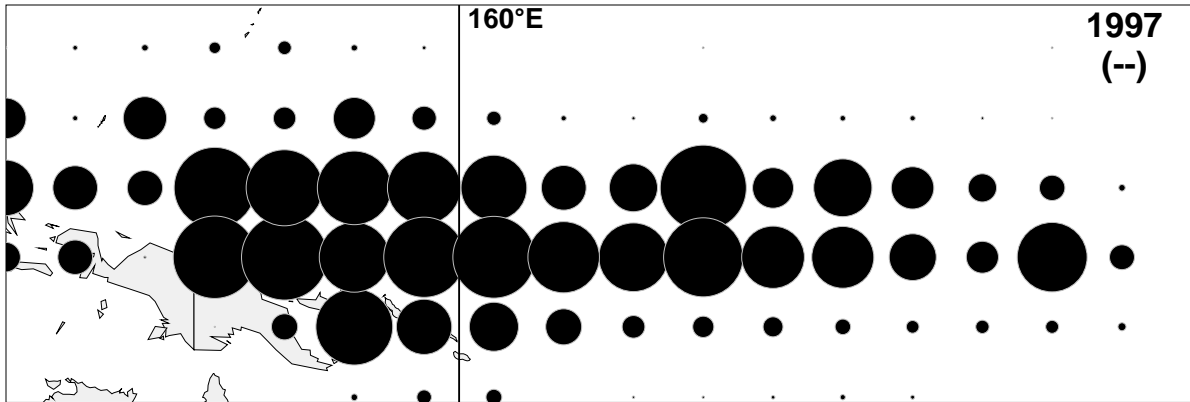


Figure 10. Distribution of Purse Seine Effort (All Fleets) in 1997, a Strong *El Niño* Year. Source: SPC (2002a).

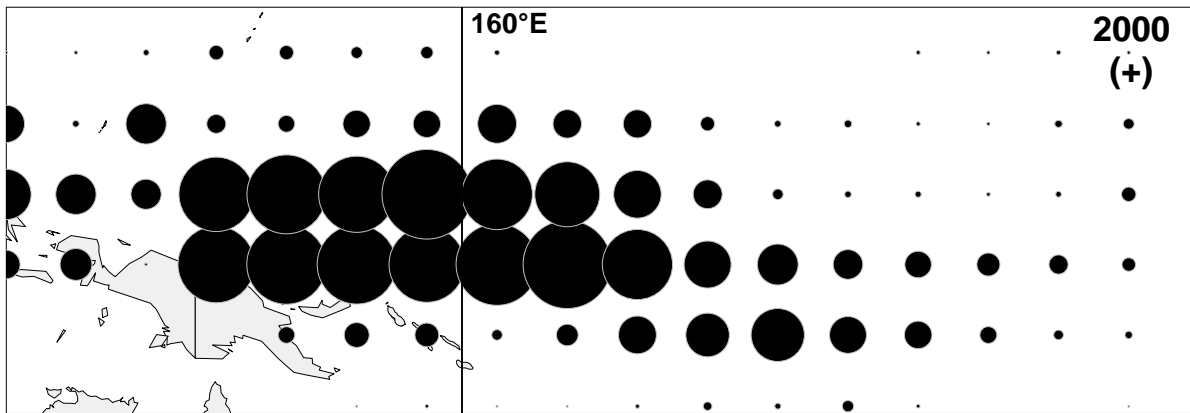


Figure 11. Distribution of Purse Seine Effort (All Fleets) in 2000, a *La Niña* Year. Source: SPC (2002a)

The WCPO experienced a “transitional” (or normal) period during 2001, for which purse seine effort is depicted in Figure 12. Fishing activity thus occurred more to the east during 2001 and 2002 than in 2000. In early 2003, the *El Niño* began to wane, and fishing effort once again moved to the west.

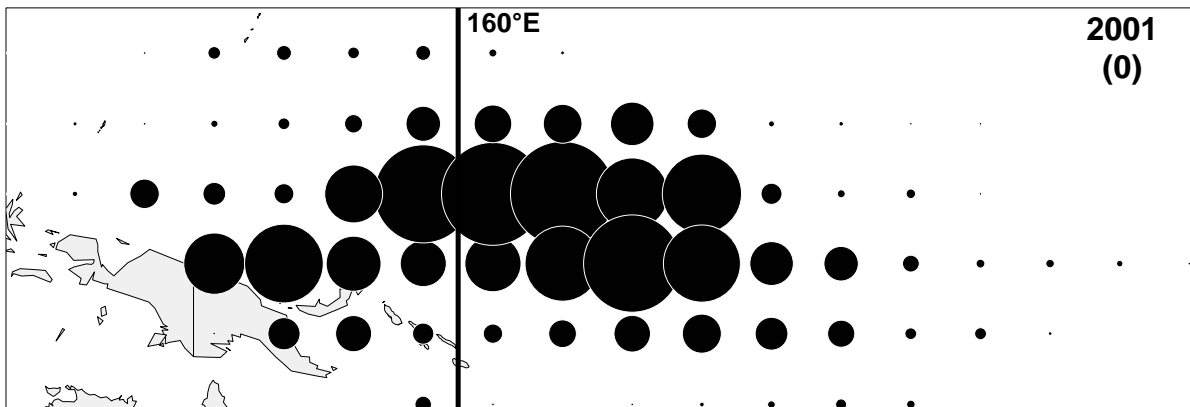


Figure 12. Distribution of Purse Seine Effort (All Fleets) in 2001, a Transitional Year. Source: SPC (2002a)

ENSO impacts on the U.S. purse seine fishery are still the subject of much study and are not completely understood. The relative strength of an ENSO event, coupled with other factors, such as a fleet's (other than the U.S.) ability to obtain fishery access to the EEZs of countries in the eastern portion of the WCPO, have an impact on the distribution of effort. Catch by purse seiners in some areas, notably the Bismarck Sea region of PNG, do not seem to be as greatly affected by ENSO events as compared to high seas regions that are large distances from large land masses in the WCPO.

4 Action Area

The Action Area is the area of the WCPO in which the Treaty-governed U.S. purse seine fishery operates. The Action Area for this consultation is the Treaty Area. Some portions of the Treaty Area are off-limits to fishing and in practice most fishing effort is concentrated in a fairly narrow latitudinal band of the tropics.

The Treaty Area includes the world's most productive tuna fishing area, from Palau eastward to the Line Islands of Kiribati. This area, which is defined in Article 1 of the Treaty, is depicted in Figure 1. Based on the historic fishing distribution of the U.S. purse seine fleet operating in the WCPO, the Action Area is the extent of the Treaty Area between approximately 15°N to 15°S and 125°E to 140°W (Figure 1).

5 Approach to the Assessment

NMFS approaches its section 7 analyses through a series of steps. The first step identifies those aspects of proposed actions that are likely to have direct and indirect effect on the physical, chemical, and biotic environment of an action area. As part of this step, we identify the spatial extent of these direct and indirect effects, including changes in that spatial extent over time. The results of this step represent the action area for the consultation. The second step of our analyses identifies the listed resources that are likely to co-occur with these effects in space and time and the nature of that co-occurrence (these represent our *exposure analyses*). In this step of our analyses, we try to identify the number, age (or life stage), and gender of the individuals that are likely to be exposed to an Action's effects and the populations or subpopulations those individuals represent. Once we identify which listed resources are likely to be exposed to an action's effects and the nature of that exposure, we examine the scientific and commercial data available to determine whether and how those listed resources are likely to respond given their exposure (these represent our *response analyses*).

The final steps of our analyses require establishing the risks those responses pose to listed resources (*risk analyses*). Our jeopardy determinations must be based on an action's effects on the continued existence of threatened or endangered species as those "species" have been listed, which can include true biological species, subspecies, or distinct population segments of vertebrate species. The continued existence of listed species depends on the fate of the populations that comprise them. Similarly, the continued existence of populations are determined by the fate of the individuals that comprise them; populations grow or decline as the individuals that comprise the population live, die, grow, mature, migrate, and reproduce (or fail to do so).

Our risk analyses reflect these relationships between listed species and the populations that comprise them, and the individuals that comprise those populations. Our risk analyses begin by

identifying the probable risks actions pose to listed individual that are likely to be exposed to an action’s effects. Our analyses then integrate those individuals risks to identify consequences to the populations those individuals represent. Our analyses conclude by determining the consequences of those population-level risks to the species those populations comprise.

We measure risks to listed individuals using the individual’s “fitness,” which are changes in an individual’s growth, survival, annual reproductive success, and lifetime reproductive success. In particular, we examine the scientific and commercial data available to determine if an individual’s probable responses to an Action’s effects on the environment (which we identify during our response analyses) are likely to have consequences for the individual’s fitness.

When individual, listed animals are expected to experience reductions in fitness, we would expect those reductions to also reduce the abundance, reproduction rates, or growth rates (or increase variance in one or more of these rates) of the populations those individual’s represent (see Stearns, 1992). Reductions in one or more of these variables (or one of the variables we derive from them) is a *necessary* condition for reductions in a population’s viability, which is itself a *necessary* condition for reductions in a species’ viability. On the other hand, when listed animals exposed to an Action’s effects are *not* expected to experience reductions in fitness, we would not expect the Action to have adverse consequences on the viability of the populations those individuals represent or the species those populations comprise (for example, see Anderson, 2000; Mills and Beatty, 1979; Stearns, 1992). If we conclude that listed animals are *not* likely to experience reductions in their fitness, we would conclude our assessment.

If, however, we conclude that listed animals are likely to experience reductions in their fitness, our assessment tries to determine if those fitness reductions are likely to be sufficient to reduce the viability of the populations those individual’s represent (measured using changes in the populations’ abundance, reproduction, spatial structure and connectivity, growth rates, or variance in these measures to make inferences about the population’s extinction risks). In this step of our analyses, we use the population’s base condition (established in the *Environmental Baseline* and *Species Status and Trends* sections of this opinion) as our point of reference. Finally, our assessment tries to determine if changes in population viability are likely to be sufficient to reduce the viability of the species those populations comprise. In this step of our analyses, we use the species’ status (established in the *Species Status and Trends* section of this opinion) as our point of reference.

6 Species Status and Trends

NMFS has determined that the actions considered in this biological opinion may affect the following species provided protection under the ESA of 1973 (16 U.S.C. 1531 et seq.; ESA):

Blue whale	<i>Balaenoptera musculus</i>	Endangered
Fin whale	<i>Balaenoptera physalus</i>	Endangered
Humpback whale	<i>Megaptera novaeangliae</i>	Endangered
Sei whale	<i>Balaenoptera borealis</i>	Endangered
Sperm whale	<i>Physeter macrocephalus</i>	Endangered
Green turtle	<i>Chelonia mydas</i>	Threatened and Endangered
Hawksbill turtle	<i>Eretmochelys imbricata</i>	Endangered

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Leatherback turtle	<i>Dermochelys coriacea</i>	Endangered
Loggerhead turtle	<i>Caretta caretta</i>	Threatened
Olive ridley turtle	<i>Lepidochelys olivacea</i>	Threatened and Endangered

Critical habitat has not been designated or proposed for any of the listed species likely to be affected by the WCPO purse seine fishery nor does critical habitat for any other species occur within the action area. No effects to designated critical habitat are anticipated by the proposed action. Therefore, critical habitat will not be considered further in this opinion.

In this section of the Opinion, we describe the status and trend of each species. We use the status and trend as a baseline to determine the resiliency of each population to additive effects likely to occur as a result of the proposed action. Summaries of the global status and trends of each species are presented to provide a foundation for the analysis.

6.1 Blue Whales

6.1.1 Species Description and Distribution

Blue whales are found in tropical to polar waters worldwide, and along the coastal shelves of North America and South America (Rice, 1974; Donovan, 1984; Clarke, 1980) in the Pacific Ocean. The International Whaling Commission's (IWC) Scientific Committee has formally recognized one blue whale population in the North Pacific (Donovan, 1991). However, there is mounting evidence that more than one population exists within this ocean basin (Gilpatrick et al., 1997; Barlow et al., 1994; Mizroch et al., 1984a; Ohsumi and Wada, 1974). One such tentative population designation is for concentrations of blue whales found during winter off Baja California and in the Gulf of California. Photo-identification studies have shown that individuals from these southern concentrations travel in the summer and fall to waters off California (Sears et al., 1987; Barlow et al., 1997; Calambokidis et al., 1990). Preliminary studies of these California/Mexico whales, based on length data from whaling records and aerial photogrammetry, have shown that they are morphologically distinct from blue whales of the Western and Central North Pacific (Gilpatrick et al., 1997).

Under the MMPA, NMFS recognizes two blue whale "stocks" or populations in U.S. waters of the North Pacific: an eastern North Pacific population (California/Oregon/Washington) which feeds primarily off California, and a Western North Pacific stock which includes whales found around the Hawaiian Islands during winter. The eastern North Pacific stock feeds in California waters in summer/fall (from June - November) and migrates south to productive areas off Mexico (Calambokidis et al., 1990) and as far south as the Costa Rica Dome (10° N) (Mate et al., 1999; Calambokidis, pers. comm.) in winter/spring. Blue whales are occasionally seen or heard off Oregon (McDonald et al., 1994; Stafford et al., 1998; VonSaunders and Barlow, 1999), but sightings there are rare. Reilly and Thayer (1990) speculate that blue whales found near the Costa Rica Dome from June - November are likely to be part of a southern hemisphere population or an isolated resident population. However, based on acoustic call similarities, Stafford et al. (1999) linked these animals to the population that feeds off California at the same time of year. Rice (1974) hypothesized that blue whales from Baja California migrated far offshore to feed in the eastern Aleutians or Gulf of Alaska and returned to feed in California waters. However, he has more recently concluded that the California population is separate from

the Gulf of Alaska population (Rice 1992). Recently, blue whale feeding aggregations have not been found in Alaska despite several surveys (Leatherwood et al., 1982; Stewart et al., 1987; Forney and Brownell, 1996).

Blue whales belonging to the Western Pacific stock appear to feed in summer southwest of Kamchatka, south of the Aleutians, and in the Gulf of Alaska (Stafford, 2003; Watkins et al., 2000), and in winter they migrate to lower latitudes in the western Pacific and less frequently in the central Pacific, including Hawaii (Stafford et al., 2001). The only published sighting record of blue whales near Hawaii is that of Berzin and Rovnin (1966). Two sightings have been made by observers on Hawaii-based longline vessels (NMFS/PIR, unpublished data). Additional evidence that blue whales occur in this area comes from acoustic recordings made off O`ahu and Midway Islands (Northrop et al., 1971; Thompson and Friedl, 1982), which included at least some within the U.S. EEZ. The recordings made off Hawaii showed bimodal peaks throughout the year (Stafford et al., 2001) with western Pacific call types heard during winter and eastern Pacific calls heard during summer.

The distribution of blue whales has been linked to their nutritional requirements. Reilly and Thayer (1990) and Palacios (1999) reported on blue whales associated with a plume of cool upwelling waters west and southwest of the Galapagos Islands during the austral winter and spring months. Palacios (1999) concluded that these whales feed on surface swarms of euphausiids and concluded that these whales may form a distinct feeding aggregation. Migration patterns are assumed for blue whales from known summer feeding areas in high latitudes to unknown, speculative winter breeding grounds (Perry et al., 1999).

6.1.2 Life History

Blue whale reproductive activities occur primarily in winter (Yochem and Leatherwood, 1985). Gestation takes 10-12 months, followed by a nursing period that continues for about 6-7 months. They reach sexual maturity at about five years of age (Yochem and Leatherwood, 1985). The age distribution of blue whales is unknown and limited information exists on natural sources of mortality (such as disease) and mortality rates. Killer whales are known to attack blue whales, but the rate of these attacks or their effect on blue whale populations is unknown.

Important foraging areas include the edges of continental shelves and ice edges in polar regions (Yochem and Leatherwood, 1985; Reilly and Thayer, 1990). Data indicate that some summer feeding takes place at low latitudes in upwelling-modified waters (Reilly and Thayer, 1990), and that some whales remain year-round at either low or high latitudes (Yochem and Leatherwood, 1985; Clark and Charif, 1998). The species *Thysanoëssa inermis*, *Thysanoëssa longipes*, *Thysanoëssa raschii*, and *Nematoscelis megalops* have been listed as prey of blue whales in the North Pacific (Kawamura, 1980; Yochem and Leatherwood, 1985).

6.1.3 Listing Status

The blue whale is listed as Endangered under the U.S. ESA and by the World Conservation Union (IUCN), and it is listed by Convention on International Trade in Endangered Species (CITES) as an Appendix I species. Blue whales are automatically protected under the Marine Mammal Protection Act of 1972 (MMPA). Three subspecies of blue whale are generally recognized. *B. musculus musculus* is found in the Northern Hemisphere; *B. musculus intermedia*

(the true blue whale) is an Antarctic species; and *B. musculus brevicauda* (the pygmy blue whale) inhabits the sub-Antarctic zone of the southern Indian Ocean and the southwestern Pacific Ocean (Perry et al., 1999; Sears, 2002). The IWC classified all blue whale stocks as “Protected Stocks”, recognizing that these stocks are 10% or more below their maximum sustainable yield levels.

6.1.4 Population Status and Trends

The global population of blue whales has been estimated to range from 11,200 to 13,000 animals (Maser et al., 1981; U. S. Department of Commerce, 1983) which is a fraction of pre-whaling population estimates of 200,000 animals. Uncertainty surrounds estimates of blue whale abundance in the North Pacific Ocean. Barlow (1994b) estimated the North Pacific population of blue whales between 1,400 and 1,900. Wade and Gerrodette (1993) and Barlow et al. (1997) estimated there were a minimum of 3,300 blue whales in the North Pacific Ocean in the 1990s.

From ship line-transect surveys, Wade and Gerrodette (1993) estimated 1,400 blue whales for the eastern tropical Pacific. A weighted average estimate of 1,744 blue whales is available for California, Oregon, and Washington, based on both shipboard line-transect surveys in 1996 and 2002 (Barlow, 2003) and photographic mark-recapture estimates (Calambokidis et al., 2003). Barlow (2003) estimated 1,736 (CV = 0.23) blue whales off California, Oregon, and Washington based on ship line-transect surveys in 1996 and 2002. Calambokidis et al. (2003) used photographic mark-recapture and estimated population sizes of 1,567 (CV = 0.32) based on 2000-2002 photographs of left sides and 1,953 (CV = 0.33) based on right sides. The average of the mark-recapture estimates of 1,760 (CV = 0.32) is very close to the line-transect estimate. The best estimate of blue whale abundance is the average of the line-transect and mark-recapture estimates, weighted by the inverse of their variances, or 1,744 (0.28).

No data are available to estimate population size for any other North Pacific blue whale population, including the putative Central stock that apparently summered along the Aleutians and wintered north of Hawai`i. No blue whale sightings were made during a summer 1994 shipboard survey south of the Aleutian Islands (Forney and Brownell, 1996), during 12 aerial surveys conducted between 1993-98 within approximately 25 nmi of the main Hawaiian Islands (Mobley et al., 2000), or during a summer/fall 2002 shipboard surveys of the entire Hawaiian Islands EEZ (Barlow, 2003). Therefore, no estimate of abundance is available for the Western Pacific blue whale stock. No data are available on current population trend. Potential biological removal (PBR), defined by the MMPA as the maximum number of animals, not including natural mortalities, that may be removed from a marine mammal stock while allowing that stock to reach or maintain its optimum sustainable population, cannot be calculated for this stock at this time.

There is some indication that blue whales have increased in abundance in California coastal waters between 1979-80 and 1991 (regression $p < 0.05$, Barlow, 1994) and between 1991 and 1996 (not significant, Barlow, 1997). Although this may be due to an increase in the stock as a whole, it could also be the result of an increased use of California as a feeding area. The size of the apparent increase abundance seen by Barlow (1994) is too large to be accounted for by population growth alone. Also, Larkman and Veit (1998) did not detect any increase along consistently surveyed tracklines in the Southern California Bight from 1987 - 1995.

Although the population in the North Pacific is expected to have grown since being given protected status in 1966, the possibility of continued unauthorized takes after blue whales were protected (Yablokov, 1994) and the existence of incidental ship strikes and gillnet mortality makes this uncertain. The PBR level for this Eastern North Pacific stock is calculated as the minimum population size (1,384) times one half the default maximum net growth rate for cetaceans ($\frac{1}{2}$ of 4%) times a F_R of 0.1 (for an endangered species which has a minimum abundance less than 1,500), resulting in a PBR of 2.8. Because this stock spends approximately half its time outside the U.S. EEZ, the PBR allocation for U.S. waters is half this total or 1.4 whales per year.

6.1.5 Impacts of Human Activity on Blue Whales

6.1.5.1 Whaling

The reported take of North Pacific blue whales by commercial whalers totaled 9,500 between 1910 and 1965 (Ohsumi and Wada, 1974). Approximately 2,000 were taken off the west coast of North America between 1919 and 1929 (Tønnessen and Johnsen, 1982). Partially overlapping with this is Rice's (1992) report of at least 1,378 taken by factory ships off California and Baja California between 1913 and 1937. Shore-based whaling stations in central California took 3 blue whales between 1919 and 1926 (Clapham et al., 1997) and 48 blue whales between 1958 and 1965 (Rice, 1974). Between 1947 and 1987, reported takes of blue whales in the North Pacific were approximately 2,400. Blue whales in the North Pacific were given protected status by the IWC in 1966.

Evidence of a population decline can be seen in the catch data from Japan. In 1912, 236 blue whales were caught, 58 whales in 1913, 123 whales in 1914, and from 1915 to 1965, the catch numbers declined continuously (Mizroch et al., 1984a). During the late 1950s and early 1960s, Japan caught 70 blue whales per year off the Aleutian Islands. The IWC banned commercial whaling in the North Pacific in 1966, and there have been no reported blue whale takes since then. Nevertheless, Soviet whaling probably continued after the ban so Soviet catch reports under-represent the number of blue whales killed by whalers (as cited in Forney and Brownell, 1996). Surveys conducted in these former whaling areas in the 1980s and 1990s failed to find any blue whales (Forney and Brownell, 1996).

6.1.5.2 Fisheries

There are no reports of fisheries-related mortality or serious injury in any of the blue whale populations (Perry et al., 1999). Blue whale interaction with fisheries may go undetected because the whales are not observed after they swim away with a portion of the net. However, fishers report that large blue and fin whales usually swim through their nets without entangling and with very little damage to the net (Barlow et al., 1997). Gillnets appear to capture marine mammals wherever they are used, and float lines from lobster traps and longlines can be expected to occasionally entangle whales (Perrin et al., 1994). Large whales have been entangled in longline gear off the Hawaiian Islands (Nitta and Henderson, 1993; Forney, 2004), but no interactions with blue whales were observed in the Hawai'i-based longline fishery between 1994 and 2002, with approximately 4-25% of all effort observed (Forney, 2004).

6.1.5.3 Ship Strikes

Ship strikes were implicated in the deaths of blue whales in 1980, 1986, 1987, 1993, and 2002 (J. Cordaro, Southwest Region, NMFS and J. Heyning, pers. comm.). During 1998-2002, there were an additional five injuries and two mortalities of unidentified large whales attributed to ship strikes. In addition, several photo-identified blue whales from California waters were observed with large scars on their dorsal areas that may have been caused by ship strikes. Several blue whales have been photographed in California with large gashes in their dorsal surface that appear to be from ship strikes (J. Calambokidis, personal communication). Additional mortality from ship strikes probably goes unreported because the whales do not strand or, if they do, they do not always have obvious signs of trauma. The average number of blue whale mortalities in California attributed to ship strikes was 0.2 per year for 1998-2002. It is estimated that between 9-25% of the whales in the Gulf of St. Lawrence have injuries or scars attributed to contact with ships. The St. Lawrence Seaway has heavy ship traffic during the time of year when blue whales are relatively abundant there.

Studies have shown that blue whales respond to approaching ships in a variety of ways, depending on the behavior of the animals at the time of approach, and speed and direction of the approaching vessel. While feeding, blue whales react less rapidly and with less obvious avoidance behavior than whales that are not feeding (Sears et al., 1987). The number of blue whales struck and killed by ships is unknown because the whales do not always strand, or because examinations of blue whales that have stranded were not identified with traumas that could have been caused by ship collisions.

6.1.5.4 Noise

Blue whales do not appear to be disturbed by noise. When noise pulses from air guns were produced off Oregon, blue whales continued vocalizing at the same rate as before the pulses, suggesting that at least their vocalization behavior was undisturbed by the noise (McDonald et al., 1995). In the presence of vessels approaching and the noise from vessel traffic, some blue whales are observed to react slower and with less obvious avoidance measures when they are feeding (Richardson et al., 1995). In the summer months, blue whales' reactions to commercial vessel traffic were more evident, especially when erratic approaches or sudden speed changes were made by the vessels (Edds and MacFarlane, 1987).

6.1.6 Distribution of Blue Whales in the Action Area

Blue whales are described with a worldwide distribution from the equator to the poles by the IWC. IWC International Decade of Cetacean Research (IDCR) surveys from the Southern Hemisphere indicated more than 500 blue whales, and the IWC lists the population of blue whales in the Southern Hemisphere from the years 1980-2000 to be 400-1,400 (CV = 0.4). There are six IWC stock areas for the Southern Hemisphere blue whales, consistent with the species' feeding locations (Donovan, 1991). There is evidence that individual blue whales reside in lower latitudes year-round (Perry et al., 1999). However, the location of these areas and wintering grounds remain speculative and unconfirmed.

Estimates of the Southern Hemisphere population range from 5,000-6,000 (Yochem and Leatherwood, 1985) with an average rate of increase of 4-5% per year. Young (2000) stated that blue whales are among the 12 cetacean species common in the Western Central Pacific and

southwestern Pacific Ocean. However, imprecise data did not allow for estimates of abundance, and there was no evidence that the blue whale numbers were increasing. Blue whales in the Southern Hemisphere generally reside south of 60°S and congregate near ice packs in the Antarctic (Branch et al., 2004). In this region, the population was estimated to include 460 blue whales, and this number was extrapolated to the region south of 30°S to include 610 blue whales (Butterworth et al., 1995).

The most recent study to detect whether Antarctic blue whales have increased since whaling included Bayesian models to fit data collected from three, long-term sightings studies from 1968 – 2001 (Branch et al., 2004). The plausible rates of increases were also obtained from blue whale biology, meta-analyses of other blue whale populations and formerly depleted populations of other baleen whale species. The results concluded that Antarctic blue whales have been increasing since the prohibition of illegal whaling at a rate of increase of 7.3% per annum, bringing the total population increase to 0.7% of their original abundance by 1996. There are no specific data on the occurrence of blue whales as far north as the proposed action area and their population estimates are unknown.

Reilly and Thayer (1990) argued that the whales seen along the equator are probably part of the southeast Pacific population which occupies the coastal shelf of South America and the Antarctic. The NOAA ship *Oscar Elton Sette* engaged as support for a Pacific Islands Fisheries Science Center (PIFSC), NMFS/NOAA, project from March, 5-28, 2006, for a total of 23 sea days conducting cetacean surveys in the waters of American Samoa, the central equatorial Pacific, Johnston Atoll, and surrounding areas. This survey did not include sightings of blue whales. Therefore, it seems that the best scientific data available include conflicting reports, and without unequivocal evidence that blue whales are not in the action area, it is reasonable to assume that blue whales may occur in the action area.

6.2 Fin Whales

6.2.1 Species Description and Distribution

Fin whales are widely distributed in the world's oceans. In the northern hemisphere, most migrate seasonally from Arctic feeding areas in summer to low latitude breeding and calving areas in winter. The population structure of fin whales remains unknown. The IWC recognized two management stocks in the North Pacific, seven stock areas in the North Atlantic, and six stock areas in the Southern Hemisphere, although the data in this region are insufficient (Perry et al., 1999).

NMFS provisionally recognizes three populations in the Pacific Ocean: (1) Alaska (northeast Pacific), (2) California/Oregon/Washington, and (3) Hawai'i (Barlow et al., 1997; Hill and DeMaster, 1998). Fin whales were reported as occurring offshore throughout the North Pacific from central Baja California to Japan and as far north as the Chukchi Sea (Rice, 1974), occurring in high densities in the northern Gulf of Alaska and southeastern Bering Sea from May - October, with some movement through the Aleutian passes into and out of the Bering Sea (Reeves et al., 1985). Fin whales of the northeast Pacific were observed and taken by Japanese and Soviet whalers off eastern Kamchatka and Cape Navarin, both north and south of the eastern Aleutians, and in the northern Bering and southern Chukchi seas (Berzin and Rovnin, 1966;

Nasu, 1974). In 1999, vessel surveys of the central Bering Sea reported 75 fin whale sightings (totaling 346 whales) clustered along the outer Bering Sea shelf break, primarily near the 200 m isobath (Moore et al., 2000). In the Gulf of Alaska, fin whales appear to congregate in the waters around Kodiak Island and south of Prince William Sound (Calkins, 1986). Recent surveys in the central-eastern and southeastern Bering Sea in 1999 and 2000 resulted in new information about the distribution and relative abundance of fin whales in these areas (Moore et al. 2000, 2002). Fin whale abundance estimates were nearly five times higher in the central-eastern Bering Sea than in the southeastern Bering Sea (Moore et al. 2002), and most sightings in the central-eastern Bering Sea occurred in a zone of particularly high productivity along the shelf break (Moore et al. 2000).

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 (Perry et al., 1999). Rice (1974) reported that several fin whales tagged from November - January off southern California were later killed by whalers in May - 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).

Fin whales are considered rare in Hawaiian waters. Balcomb (1987) observed 8-12 fin whales in a multispecies feeding assemblage on 20 May 1966 approximately 250 miles (mi) south of Honolulu. Additional sightings were reported north of O`ahu in May 1976 and in the Kaua`i Channel in February 1979 (Shallenberger, 1981). More recently, a single fin whale was observed north of Kaua`i in February 1994 (Mobley et al., 1996), and five sightings were made during a 2002 survey of waters within the U.S. EEZ of the Hawaiian Islands (Barlow, 2003). A single stranding has been reported on Maui (Shallenberger, 1981). Thompson and Friedl (1982) suggested that fin whales migrate into Hawaiian waters mainly in fall and winter, based on acoustic recordings off O`ahu and Midway Islands. Although the exact positions of the whales producing the sounds could not be determined, at least some of them were almost certainly within the U.S. EEZ. More recently, McDonald and Fox (1999) reported an average of 0.027 calling fin whales per 10,000 kilometers (km) (grouped by 8-hour periods) based on passive acoustic recordings within about 16 km of the north shore of O`ahu.

6.2.2 Life History

Fin whales are the second largest in the Balaenopteridae Family (Perry et al., 1999). With a maximum length between 20-27 m, fin whales become sexually mature between 6 to 10 years of age, depending on density-dependent factors (Gambell, 1985b). Reproductive activities for fin whales occur primarily in the winter. Gestation lasts about 12 months and nursing occurs for 6 to 11 months (Perry et al., 1999). The age distribution of fin whales in the North Pacific is unknown.

Natural sources and rates of mortality are largely unknown, but Aguilar and Lockyer (1987) suggest annual natural mortality rates may range from 0.04 to 0.06 (based on studies of northeast

Atlantic fin whales). The occurrence of the nematode *Crassicauda boopis* appears to increase the potential for kidney failure in fin whales and may be preventing some fin whale populations from recovering from whaling (Lambertsen, 1992, as cited in Perry et al., 1999). Killer whale or shark attacks may result in serious injury or death in very young and sick whales (Perry et al., 1999).

6.2.3 Listing Status

In the North Pacific, the IWC began management of commercial whaling for fin whales in 1969. Fin whales were fully protected from commercial whaling in 1976 (Allen, 1980). In international waters, the IWC classifies the North Pacific, Nova Scotia, West Norway/Faeroe Islands, and Southern Hemisphere stocks as Protected Stocks, indicating that these stocks are 10% or more below their maximum sustainable yield and whaling is prohibited. However, the East Greenland/Iceland stock is considered not more than 10% below and not more than 20% above its maximum sustainable yield, and is therefore classified as a Sustainable Management Stock (Perry et al., 1999). This designation permits whaling for this stock. Fin whales were listed as endangered under the ESA in 1973, and are automatically protected under the MMPA. Fin whales are listed as endangered on the IUCN Red List of Threatened Animals (Baillie and Groombridge, 1996). Critical habitat has not been designated for fin whales.

6.2.4 Population Status and Trends

Prior to exploitation by whaling vessels, the North Pacific population consisted of an estimated 42,000-45,000 fin whales (Ohsumi and Wada, 1974). In the early 1970s, the entire North Pacific population had been reduced to between 13,620 and 18,630 fin whales (Ohsumi and Wada, 1974). During the early 1970s, 8,520 - 10,970 fin whales were surveyed in the eastern half of the North Pacific (Braham, 1991). The current status and trend of the fin whale population in the North Pacific is largely unknown.

Reliable estimates of current and historical abundance for the entire Northeast Pacific fin whale stock are currently not available. A survey conducted in August 1994 covering 2,050 nmi of trackline south of the Aleutian Islands encountered only four fin whale groups (Forney and Brownell, 1996). However, this survey did not include all of the waters off Alaska where fin whale sightings have been reported, thus, no population estimate can be made. Aggregations of fin whales were often sighted in 1999 in areas where the ship's echosounder identified large aggregations of zooplankton, euphausiids, or fish (Moore et al., 2000). One aggregation of fin whales which occurred during an off-effort period involved greater than 100 animals and occurred in an area of dense fish echosign. Results of the surveys in 1999 and 2000 in the central-eastern Bering Sea and southeastern Bering Sea provided provisional estimates of 3,368 (CV = 0.29) and 683 (CV = 0.32), respectively (Moore et al., 2002). These estimates are considered provisional because they have not been corrected for animals missed on the trackline, animals submerged when the ship passed, and responsive movement. However, the provisional estimate for fin whales in each area is expected to be robust as previous studies have shown that only small correction factors are needed for this species. The Moore et al. (2002) estimate for 1999 is different than that of Moore et al. (2000) because it covers the southeastern Bering Sea as well as the central-eastern Bering Sea. Additionally, the region covered by Moore et al. (2000) did not have consistent effort and thus could be inaccurate. This estimate cannot be

used as an estimate of the entire Northeast Pacific stock of fin whales because it is based on a survey in only part of the stock's range.

Dedicated sighting cruises were conducted in coastal waters of western Alaska and the eastern and central Aleutian Islands in July-August 2001-2003 (Zerbini et al., in review). Over 9,053 km of tracklines were surveyed in coastal waters (as far as 85 km offshore) between the Kenai Peninsula (150°W) and Amchitka Pass (178°W). Fin whale sightings (n = 276) were observed from east of Kodiak Island to Samalga Pass, with high aggregations recorded near the Semidi Islands. Zerbini et al. (in review) estimated that 1652 (95% CI = 1142-2389) whales occurred in the area. Under the 1994 reauthorized MMPA, the PBR is defined using the F_R for this stock of 0.1, the recommended value for cetacean stocks which are listed as endangered (Wade and Angliss, 1997). Thus, the PBR level for this stock is 11.4 (5,703 x 0.02 x 0.1).

There is some indication that fin whales have increased in abundance in California coastal waters between 1979/80 and 1991 (Barlow, 1994) and between 1991 and 1996 (Barlow, 1997), but these trends are not significant. Although the population in the North Pacific is expected to have grown since receiving protected status in 1976, the possible effects of continued unauthorized take (Yablokov, 1994) and incidental ship strikes and gillnet mortality make this uncertain. The PBR level for the California/Oregon/Washington stock is calculated as the minimum population size (2,541) times one half the default maximum net growth rate for cetaceans ($\frac{1}{2}$ of 4%) times a F_R of 0.3 (for an endangered species, with $n_{min} > 1,500$ and $CV_{nmin} < 0.50$), resulting in a PBR of 15.

As part of the Marine Mammal Research Program of the Acoustic Thermometry of Ocean Climate (ATOC) study, a total of 12 aerial surveys were conducted within about 25 nmi of the main Hawaiian Islands in 1993-98 (Mobley et al., 2000). Only one sighting of a single fin whale was made (Mobley et al., 1996), and no abundance estimate was calculated. Using passive acoustic detections from a hydrophone north of O`ahu, MacDonald and Fox (1999) estimate an average density of 0.027 calling fin whales per 1,000 km² within about 16 km from shore. However, the relationship between the number of whales present and the number of calls detected is not known, and therefore this acoustic method does not provide an estimate of absolute abundance for fin whales. A 2002 shipboard line-transect survey of the entire Hawaiian Islands EEZ resulted in an abundance estimate of 174 (CV = 0.72) fin whales (Barlow, 2003). This is currently the best available abundance estimate for the Hawaii stock. The PBR level for this stock is calculated as the minimum population size (101) times one half the default maximum net growth rate for cetaceans ($\frac{1}{2}$ of 4%) times a F_R of 0.1 (the default value for an endangered species; Wade and Angliss, 1997), resulting in a PBR of 0.2 fin whales per year.

Based on the available information, it is feasible that the North Pacific population as a whole has failed to increase significantly over the past 20 years. The only contrary evidence comes from investigators conducting seabird surveys around the Pribilof Islands in 1975-1978 and 1987-1989. These investigators observed more fin whales in the second survey and suggested they were more abundant in the survey area (Baretta and Hunt, 1994). Moore et al. (2000) conducted surveys for whales in the central Bering Sea in 1999 and tentatively estimated the fin whale population was about 4,951 animals (95% C.I.: 2,833-8,653).

6.2.5 Impacts of Human Activity on Fin Whales

6.2.5.1 Whaling

As early as the mid-seventeenth century, the Japanese were capturing fin, blue, and other large whales using a fairly primitive open-water netting technique (Tønnessen and Johnsen, 1982; Cherfas, 1989). In 1864, explosive harpoons and steam-powered catcher boats were introduced in Norway, allowing the large-scale exploitation of previously unobtainable whale species. The North Pacific and Antarctic whaling operations soon added this modern equipment to their arsenal. Large numbers of fin whales were taken by commercial whalers throughout the North Pacific from the early 20th century until the 1970s (Tønnessen and Johnsen 1982).

Approximately 46,000 fin whales were taken from the North Pacific by commercial whalers between 1947 and 1987 (C. Allison, IWC, personal communication). Some of the whales taken may have been from a population or populations that migrate seasonally into the Hawaiian EEZ.

After blue whales were depleted in most areas, the smaller fin whale became the focus of whaling operations and more than 700,000 fin whales were landed in the twentieth century. The number of fin whales taken at three whaling stations in Canada from 1965-1971 totaled 3,528 whales (Mitchell, 1974). Reports of non-directed takes of fin whales are fewer over the last two decades than for other endangered large whales such as right and humpback whales. Between 1914 and 1975, over 26,040 fin whales were harvested throughout the North Pacific (Braham, 1991, as cited in Perry et al., 1999). Catches in the North Pacific and Bering Sea ranged from 1,000-1,500 fin whales annually during the 1950s and 1960s. However, not all Soviet catches were reported (Yablokov, 1994, as cited in Ferrero et al., 2000).

6.2.5.2 Fisheries

Prior to 1999, there were no observed or reported mortalities of fin whales incidental to commercial fishing operations within the range of the Northeast Pacific stock. However, in 1999, one fin whale was killed incidental to the Gulf of Alaska pollock trawl fishery. This single mortality results in an estimate of three mortalities in 1999, and an average 0.6 (95% CI = 0.20 - 1.55) mortalities over the five-year period from 1999-2003 for this stock. Although there have been a few strandings of Northeast Pacific stock fin whales recorded in recent years (2 and 1 in 1998 and 1999, respectively; (NMFS, unpublished data), none of these have been noted as having evidence of fishery interactions. Between 1994 and 2002, no interactions with the Hawaiian stock of fin whales were observed in the Hawaii-based longline fishery, with approximately 4-25% of all effort observed (Forney 2004).

The offshore drift gillnet fishery is the only fishery that is likely to take fin whales from the California/Oregon/Washington stock, and one fin whale death has been observed in 1999. After the 1997 implementation of a Take Reduction Plan, which included skipper education workshops and required the use of pingers and minimum 6-fathom extenders, overall cetacean entanglement rates in the drift gillnet fishery dropped considerably (Barlow and Cameron, 2003). Mean annual takes for this fishery are based on 1997-2001 data. This results in an average estimate of 1.0 fin whale taken annually. Some gillnet mortality of large whales may go unobserved because whales swim away with a portion of the net. However, fishermen report that large rorquals (blue and fin whales) usually swim through nets without entangling and with very little damage to the nets.

Drift gillnet fisheries for swordfish and sharks exist along the entire Pacific coast of Baja California, Mexico and may take animals from this population of the California/Oregon/Washington stock. Quantitative data are available only for the Mexican swordfish drift gillnet fishery, which uses vessels, gear, and operational procedures similar to those in the U.S. drift gillnet fishery, although nets may be up to 4.5 km long (Holts and Sosa-Nishizaki, 1998). The fleet increased from two vessels in 1986 to 31 vessels in 1993. The total number of sets in this fishery in 1992 can be estimated from data provided by these authors to be approximately 2700, with an observed rate of marine mammal bycatch of 0.13 animals per set (10 marine mammals in 77 observed sets; Sosa-Nishizaki et al., 1993). This overall mortality rate is similar to that observed in California driftnet fisheries during 1990-95 (0.14 marine mammals per set; Julian and Beeson, 1998), but species-specific information is not available for the Mexican fisheries. Previous efforts to convert the Mexican swordfish driftnet fishery to a longline fishery have resulted in a mixed-fishery, with 20 vessels alternately using longlines or driftnets, 23 using driftnets only, 22 using longlines only, and seven with unknown gear type (Berdegué 2002).

6.2.5.3 Ship Strikes

In 2000, a fin whale was struck by a vessel in Uyak Bay. Assuming this was the only ship strike which occurred during the five-year period from 1997-2001, the average number of ship strikes per year is 0.2 in the Northeast Pacific. Ship strikes were implicated in the deaths of one fin whale in 1997 and 2001 in the California/Oregon/Washington stock (J. Heyning and J. Cordaro, Southwest Region, NMFS, personal communication). During 1997-2001, there were an additional four injuries and two mortalities of unidentified large whales attributed to ship strikes for this region. Additional mortality from ship strikes probably goes unreported because the whales do not strand or, if they do, they do not always have obvious signs of trauma. The average observed annual mortality due to ship strikes is 0.4 fin whales per year for the period 1997-2001 for this stock (Carretta et al., 2004).

6.2.6 Distribution of Fin Whales in the Action Area

The IWC lists fin whales with a worldwide distribution, including a 1979 population estimate of 85,200 fin whales in the Southern Oceans, and notes that fin whales are less common in tropics. The NMFS Draft Recovery Plan (1998b) describes fin whales with an anti-tropical distribution. Young (2000) stated that fin whales are present in the Western Central Pacific, but with no estimate of abundance or distribution. Other studies list the species in latitudes higher than 20°N and 20°S, which is outside the action area (Perry et al., 1999; Leatherwood et al., 1982).

The most recent NMFS/PIFSC survey on the NOAA ship *Oscar Elton Sette* (March 5-28, 2006) in the waters of American Samoa, the central equatorial Pacific, Johnston Atoll, and surrounding areas did not include sightings of fin whales. Therefore, it seems that the best scientific data available include conflicting reports of both the species' presence and absence. Without unequivocal evidence that fin whales are not in the action area, it is reasonable to assume that fin whales would be found in the action area.

6.3 Humpback whales

6.3.1 Species Description and Distribution

The humpback whale is distributed worldwide in all ocean basins, from equator to subpolar waters (Perry et al., 1999). In winter, most humpback whales occur in the temperate and tropical waters of the North and South Hemispheres (from 10°-23°). Humpback whales generally occur over continental shelves, shelf breaks, and around some oceanic islands (Balcomb and Nichols, 1978; Whitehead, 1987). Humpback whales exhibit seasonal migrations between warmer temperate and tropical waters in winter and cooler waters of high prey productivity in summer.

Aerial, vessel, and photo-identification surveys and genetic analyses indicate that within the U.S. EEZ there are at least three relatively separate populations that migrate between their respective summer/fall feeding areas to winter/spring calving and mating areas (Calambokidis et al., 1997; Baker et al., 1998) winter/spring populations in coastal Central America and Mexico which migrate to the coast of California to southern British Columbia in summer/fall (Calambokidis et al., 1989; Steiger et al., 1991; Calambokidis et al., 1993) - referred to as the California/Oregon/Washington and Mexico or Eastern North Pacific stock; 2) winter/spring populations of the Hawaiian Islands which migrate to northern British Columbia/Southeast Alaska and Prince William Sound west to Unimak Pass (Baker et al., 1990; Perry et al., 1990; Calambokidis et al., 1997) - referred to as the Central North Pacific stock; and 3) winter/spring populations of Japan which, based on Discovery Mark information, probably migrate to waters west of the Kodiak Archipelago (the Bering Sea and Aleutian Islands) in summer/fall (Berzin and Rovnin, 1966; Nishiwaki, 1966; Darling, 1991) - referred to as the Western North Pacific stock.

Winter/spring populations of humpback whales also occur in Mexico's offshore islands. The migratory destination of those whales is not well known (Calambokidis et al., 1993; Calambokidis et al., 1997), although some whales from the Revillagigedo Archipelago have been matched to animals seen west of Kodiak, Alaska (Witteveen et al., 2004). Some recent exchange between winter/spring areas has been documented (Darling and McSweeney, 1985; Baker et al., 1986; Darling and Cerchio, 1993), as well as movement between Japan and British Columbia, and Japan and the Kodiak Archipelago (Darling et al., 1996; Calambokidis et al., 1997). Calambokidis et al. (2001) concludes that there are at least three subpopulations of humpback whales on the wintering grounds (Hawai'i, Japan, and Mexico), and possibly as many as six subpopulations, with subdivisions in Mexico, Japan, and Central America. Currently, there are insufficient data to apply the Dizon et al. (1992) phylogeographic approach to classify population structure in humpback whales. Until further information becomes available, three stocks of humpback whales are recognized within the U.S. EEZ of the North Pacific: one in the Eastern North Pacific (the California/Oregon/Washington - Mexico stock), one in the Central North Pacific, and one in the Western North Pacific.

Three feeding areas for the Central North Pacific stock have been studied using photo-identification techniques: southeastern Alaska, Prince William Sound, and Kodiak Island. There has been some exchange of individual whales between these locations. Calambokidis et al. (2001) reports interchange between Kodiak, Prince William Sound, and Southeast Alaska, although the number of individuals seen in multiple locations is small. Mizroch et al. (2004)

examined photographs from 1979-1996 and reported that less than 1% of the individual whales photographed in either Southeast Alaska or Prince William Sound moved between areas. Based on sightings across all Alaska feeding areas, fewer than 2% of the individuals were seen in more than one area (Mizroch et al., 2004). Fidelity to feeding areas is maternally directed; that is, whales return to the feeding areas where their mothers first brought them as calves (Martin et al., 1984; Baker et al., 1987).

6.3.2 Life History

Humpback whale reproductive activities occur primarily in winter. They become sexually mature at age four to six. Annual pregnancy rates have been estimated at about 0.40 to 0.42 (NMFS, unpublished; Nishiwaki, 1959) and female humpback whales are believed to become pregnant every two to three years. Cows will nurse their calves for up to 12 months. The age distribution of the humpback whale population is unknown, but the portion of calves in various populations has been estimated at about 4 to 12% (Chittleborough, 1965; Whitehead, 1982; Bauer, 1986; Herman et al., 1980; and Clapham and Mayo, 1987). The causes of natural mortality in humpback whales are generally unknown, but potential sources of mortality include parasites, disease, predation (killer whales, false killer whales, and sharks), biotoxins, and ice entrapment.

6.3.3 Listing Status

The IWC first protected humpback whales in the North Pacific in 1965. Humpback whales were listed as endangered under the ESA in 1973, and are automatically protected under the MMPA of 1972. They are also protected by CITES as an Appendix I species and are listed as Vulnerable by the IUCN (2003) and the MMPA. Critical habitat has not been designated for the species.

6.3.4 Population Status and Trends

Based on whaling statistics, the pre-1905 population of humpback whales in the North Pacific was estimated to be 15,000 (Rice, 1978), but this population was reduced by whaling to approximately 1,200 by 1966 (Johnson and Wolman, 1984). The North Pacific total now almost certainly exceeds 6,000 humpback whales (Calambokidis et al., 1997). The current abundance estimate of humpback whales in the North Pacific is based on data collected by nine independent research groups that conducted photo-identification studies of humpback whales in the three wintering areas (Mexico, Hawai'i, and Japan). Using Darroch's (1961) method, which uses only data from wintering areas, and averaging the 1991-92, 1992-93, and 1991-93 winter release-recovery information results in an abundance estimate of 4,005 (CV = 0.095) for the entire Central North Pacific humpback whale stock (Calambokidis et al., 1997).

6.3.4.1 Eastern North Pacific Stock

Estimates of the abundance of the Eastern Pacific stock of humpback whales were made by aerial survey (Dohl, 1983; Forney et al., 1995) and ship surveys (Barlow, 1995). More recent estimates are available from ship surveys and mark-recapture studies. Barlow (2003) estimated 1,314 (CV = 0.30) humpbacks in California, Oregon, and Washington waters based on summer/fall ship line-transect surveys in 1996 and 2001. Calambokidis et al. (2004) estimated humpback whale abundance in these feeding areas from 1991-2003 using Petersen mark-recapture estimates based on photo-identification collections in adjacent pairs of years. These data show a general upward trend in abundance followed by a large (but not statistically significant) drop in the 1999/2000 and 2000/2001 estimates. The 2002/2003 population estimate

(1,391, CV = 0.22) is higher than any previous estimates and may indicate that the apparent decline in the previous two estimates exaggerates any real decline that might have occurred (Calambokidis et al., 2003) or that a real decline was followed by an influx of new whales from another area (Calambokidis et al., 2004). This latter view is substantiated by the greater fraction of new whales seen for the first time in 2003 (Calambokidis et al., 2004).

Ship surveys provide some indication that humpback whales increased in abundance in California coastal waters between 1979/80 and 1991 (Barlow, 1994) and between 1991 and 1996 (Barlow, 1997). However population estimates have declined between 1996 and 2001 (Barlow, 2003). Mark-recapture population estimates increased steadily from 1988/90 to 1997/98 at about 8% per year (Calambokidis et al., 1999). The apparent dip in the 1999/2000 and 2000/2001 estimates may indicate that population growth is slowing, but the subsequent increases in 2001/2002 and 2002/2003 casts some doubt on this explanation. Population estimates for the entire North Pacific have also increased substantially from 1,200 in 1966 to 6,000-8,000 circa 1992. Although these estimates are based on different methods and the earlier estimate is extremely uncertain, the growth rate implied by these estimates (6-7%) is consistent with the recently observed growth rate of the Eastern North Pacific stock.

The PBR level for this stock is calculated as the minimum population size (1,158) times one half the estimated population growth rate for this stock of humpback whales ($\frac{1}{2}$ of 8%) times a F_R of 0.1 (for an endangered species with a total population size of less than 1,500), resulting in a PBR of 4.6. Because this stock spends approximately half its time outside the U.S. EEZ, the PBR allocation for U.S. waters is 2.3 whales per year.

6.3.4.2 Western North Pacific Stock

Available information about feeding areas in U.S. waters for the Western stock of humpback whales indicates that there is considerable overlap between the Western North Pacific and Central North Pacific stocks in the Gulf of Alaska between Kodiak Island and the Shumagin Islands. Over three years, Waite et al. (1999) collected photographs of 127 individuals located near Kodiak Island, 22 individuals located near the Shumagin Islands, 8 individuals located offshore to the southeast of the Shumagin Islands, and 7 individuals located near Akutan Island in the eastern Aleutian Islands. Only seven of these individuals have been documented in Prince William Sound or Southeast Alaska. Witteveen (2004) conducted a photo-identification study in Marmot and Chiniak Bays (on the northeast side of Kodiak Island), documented 103 individual animals, and estimated that the number of humpback whales in that area totaled 157 (95% CI: 114, 241). Witteveen et al. (2004) report matches between whales photographed at the Shumagin Islands between 1999 and 2002 and whales photographed in Hawai'i, offshore Mexico Islands, coastal Mexico waters, and Japan. In addition, individuals identified off Japan have been resighted in the Eastern North Pacific (Darling et al., 1996; Calambokidis et al., 1997).

New information from a variety of sources indicates that humpback whales from the Western and Central North Pacific stocks mix on summer feeding grounds in the central Gulf of Alaska and perhaps the Bering Sea. A major research effort was initiated in 2002 in order to better delineate stock structure of humpback whales in the North Pacific using a variety of techniques, and it is expected that this effort will assist in resolving stock structure within a few years. The F_R for this stock is 0.1, the value for cetacean stocks listed as endangered under the Endangered

Species Act (Wade and Angliss, 1997). Thus, for the Western North Pacific stock of humpback whale, PBR = 1.3 animals.

6.3.4.3 Central North Pacific Stock

Different studies have used different approaches to estimate the abundance of animals in Southeast Alaska. Baker et al. (1992) estimated an abundance of 547 (95% CI: 504-590) using data collected from 1979-1986. Straley (1994) recalculated the estimate using a different analytical approach (Jolly-Seber open model for capture-recapture data) and obtained a mean population estimate of 393 animals (95% CI: 331-455) using the same 1979-1986 data set. Using data from 1986-1992 and the Jolly-Seber approach, Straley et al. (1995) estimated that the annual abundance of humpback whales in southeastern Alaska was 404 animals (95% CI: 350-458). Straley et al. (2002) examined data for the northern portion of Southeast Alaska from 1994-2000 and provided an updated abundance estimate of 961 (95% CI: 657-1,076). The sum of the available estimates for the known feeding areas is 2,036 (149 in Prince William Sound, 651 in Kodiak, 961 in Southeast, and 275 in British Columbia), which is well below the Calambokidis et al. (1997) estimate of 4,005 based on data collected from 1991-1993. However, the estimate for Southeast Alaska is known to be a minimum estimate because there are few to no photo-identification effort in the lower half of Southeast Alaska (south of Frederick Sound). In addition, many humpback whales feed seasonally near the Shumagin Islands, where photo-identification studies have only recently been initiated, and humpbacks are seen pelagically in the Gulf of Alaska. Also, Moore et al. (2002) have documented humpback whales in the Bering Sea, and it is not known whether these animals belong to the Central or Western North Pacific humpback whale stock.

Mobley et al. (2001) conducted aerial surveys throughout the main Hawaiian Islands during 1993, 1995, 1998, and 2000. Abundance estimates resulting from these surveys was 2,754 (95% CI 2,044-3,468), 3,776 (95% CI: 2,925-4627), 4,358 (95% CI: 3,261-5,454), and 4,491 (95% CI 3,146-5,836). These estimates, which are based on line transect methods, are slightly more conservative than the estimates determined using mark-recapture techniques, perhaps due to computational problems associated with the assumption of a heterogeneous sighting probability across different regions of Hawai'i. Mobley et al. (2001) estimated an annual increase of 7% for 1993-2000 using data from aerial surveys that were conducted in a consistent manner for several years across the main Hawaiian Islands and were developed specifically to estimate a trend for the Central stock.

The estimated number of animals in the Southeast Alaska portion of this stock has increased. The 2000 estimate of 961 (Straley et al., 2002) is substantially higher than estimates from the early and mid-1980s. A trend for the Southeast Alaska portion of this stock cannot be estimated from the data, however, because of differences in methods and areas covered. Mobley et al. (2001) conducted annual surveys of the humpback whale breeding grounds in Hawai'i and estimated a rate of increase of 7% for the period 1993-2000. Furthermore, it is clear that the abundance has increased in Southeast Alaska in recent years. The best available estimate of current rate of increase is 7%, and while it may or may not be the same as the stock's maximum net productivity rate, it seems reasonable to use 7% as a conservative estimate of the maximum net productivity rate.

Comparison of the estimate for the entire stock provided by Calambokidis et al. (1997) with the 1981 estimate of 1,407 (95% CI: 1,113-1,701) from Baker and Herman (1987) suggests that the stock increased in abundance between the early 1980s and early 1990s. However, the robustness of the Baker and Herman (1987) estimate is questionable due to the small sample size and opportunistic nature of the survey. Mizroch et al. (2004) calculate an annual population rate of increase of 10%. This is within the range of 8.8-14.4% reported by Best (1993) for humpback whales off South Africa, and is identical to the 10% value reported by Bannister and Hedley (2001) for humpback whales off western Australia.

The F_R for the Central North Pacific stock is 0.1, the recommended value for cetacean stocks listed as endangered under the ESA (Wade and Angliss, 1997). The default value of 0.04 for the maximum net productivity rate will be replaced by 0.07, which is the best estimate of the current rate of increase and is considered a conservative estimate of the maximum net productivity rate. Thus, for the entire Central North Pacific stock of humpback whale, $PBR = 12.9$ animals ($3,698 + 0.035 + 0.1$). The PBR level for the Southeast Alaska portion of this stock, $PBR = 3.0$ animals ($868 + 0.035 + 0.1$), and the PBR level for the northern portion of the stock is 9.9 animals ($12.9 - 3.0$).

6.3.5 Habitat Concerns

The Hawaiian Islands Humpback Whale National Marine Sanctuary (HIHWNMS) protects the winter breeding, calving and nursing range of the largest Pacific population of the endangered humpback whale. Congress designated the HIHWNMS on November 4, 1992. The Hawaiian Islands National Marine Sanctuary Act designated the Sanctuary for the primary purpose of protecting humpback whales and their habitat within the Hawaiian Islands marine environment. The humpback whale habitat has been defined for purposes of Sanctuary management, as: “those areas in the waters around Hawai‘i that provide space for individual and population growth and normal behavior of humpback whales, and include sites used for reproductive activities, including breeding, calving and nursing.” It is the only National Marine Sanctuary dedicated to whales and their habitat.

The Sanctuary works collaboratively to conserve, enhance and protect humpback whales and their habitat by promoting and coordinating research, enhance public awareness, and fostering traditional uses by Native Hawaiians. The sanctuary is jointly managed by the sanctuary manager, the state of Hawai‘i co-manager, and other field staff via a cooperative Federal-state partnership. The Sanctuary is actually a series of five noncontiguous marine protected areas distributed across the main Hawaiian Islands. The total area of the Sanctuary is 1,370 square miles. Encompassing about half of the total Sanctuary area, the largest contiguous portion of the Sanctuary is delineated around Maui, Lana‘i, and Moloka‘i. The four smaller portions are located off the north shore of Kaua‘i, off Hawai‘i’s Kona coast, and off the north and southeast coasts of O‘ahu.

6.3.6 Impacts of Human Activity on Humpback Whales

6.3.6.1 Whaling

From 1900 - 1965, nearly 30,000 whales were taken in modern whaling operations of the Pacific Ocean. Prior to that, an unknown number of humpback whales were taken (Perry et al., 1999). In 1965, the IWC banned commercial hunting of humpback whales in the Pacific Ocean. The

reported take of North Pacific humpback whales by commercial whalers totaled approximately 7,700 between 1947 and 1987 (C. Allison, IWC unpubl. data). In addition, approximately 7,300 were taken along the west coast of North America from 1919-1929 (Tonnessen and Johnsen, 1982). From 1910-1965 total catch from the California-Washington stock includes at least 2,000 whales taken in Oregon and Washington, 3,400 taken in California, and 2,800 taken in Baja California (Rice, 1978). Shore-based whaling apparently depleted the humpback whale stock off California twice: once prior to 1925 (Clapham et al., 1997) and again between 1956 and 1965 (Rice, 1974). There has been a prohibition on taking humpback whales since 1966.

6.3.6.2 Fisheries

Eastern North Pacific Stock

After the 1997 implementation of a Take Reduction Plan, which included skipper education workshops and required the use of pingers and minimum six-fathom extenders, overall cetacean entanglement rates in the drift gillnet fishery dropped considerably (Barlow and Cameron, 2003). Some gillnet mortality of large whales may go unobserved because whales swim away with a portion of the net. The deaths of two humpback whales that stranded in the Southern California Bight have been attributed to entanglement in fishing gear (Heyning and Lewis, 1990), and a humpback whale was observed off Ventura, CA in 1993 with a 20 foot (ft) section of netting wrapped around and trailing behind. During the period 1999-2003, a humpback cow-calf pair was seen entangled in a net off Big Sur, California (1999) and another lone humpback was seen entangled in line and fishing buoys off Grover City (2000), but the fate of these animals is not known (J. Cordero, NMFS unpubl. data). One humpback whale was entangled and released alive in the swordfish/thresher shark drift gillnet fishery in November of 1999.

Other unobserved fisheries may also result in injuries or deaths of humpback whales. In 2001, a humpback whale with “pot gear” wrapped around its flukes was seen free-swimming eight miles offshore of Point Bonita, California (NMFS, Southwest Region, unpublished data). In 2003, there were five separate reports of humpback whales entangled in crab pot and/or polypropylene lines (J. Cordero, NMFS, unpubl. data). In March 2003, an adult female with a calf was seen off Monterey with crab pot line wrapped around its flukes. An adult humpback was seen in May 2003 in the Santa Barbara Channel with 100 feet of yellow polypropylene line wrapped around its pectoral fins and caudal peduncle. Another adult female with a calf was seen in August 2003 west of the Farallon Islands with crab pot line with floats wrapped around its caudal peduncle and fluke lobe; the adult was reported to be ‘diving awkwardly’. In November 2003, there were two reports within four days near Crescent City and south of Humboldt Bay of single humpback whales with crab pot line wrapped around their ‘torso’. These two reports may represent the same whale. The final status of all these whales is unknown.

Drift gillnet fisheries for swordfish and sharks exist along the entire Pacific coast of Baja California, Mexico and may take animals from the same population. Quantitative data are available only for the Mexican swordfish drift gillnet fishery, which uses vessels, gear, and operational procedures similar to those in the U.S. drift gillnet fishery, although nets may be up to 4.5 km long (Holts and Sosa-Nishizaki, 1998). The fleet increased from two vessels in 1986 to 31 vessels in 1993 (Holts and Sosa-Nishizaki, 1998). The total number of sets in this fishery in 1992 can be estimated from data provided by these authors to be approximately 2,700, with an observed rate of marine mammal bycatch of 0.13 animals per set (10 marine mammals in 77

observed sets; Sosa-Nishizaki et al., 1993). This overall mortality rate is similar to that observed in California driftnet fisheries during 1990-1995 (0.14 marine mammals per set; Julian and Beeson, 1998), but species-specific information is not available for the Mexican fisheries. Previous efforts to convert the Mexican swordfish driftnet fishery to a longline fishery have resulted in a mixed-fishery, with 20 vessels alternately using longlines or driftnets, 23 using driftnets only, 22 using longlines only, and seven with unknown gear type (Berdegué, 2002).

Western North Pacific Stock

Between 1999 and 2003, there were incidental serious injuries and mortalities of Western and Central North Pacific (CNP) humpback whales in the following observed fisheries in Alaska: Bering Sea/Aleutian Islands pollock trawl and Bering Sea/Aleutian Islands sablefish pot. Average annual mortality from observed fisheries was 0.49 humpbacks from this stock. Note, however, that the stock identification is uncertain and the mortality may have been attributable to the CNP stock of humpback whales. Thus, this mortality is assigned to both the Central and Western stocks.

The only fishery-related humpback stranding in an area thought to be occupied by animals from this stock was reported by a U.S. Coast Guard vessel in late June 1997 operating near the Bering Strait. The whale was found floating dead entangled in netting and trailing orange buoys (National Marine Mammal Laboratory, Platforms of Opportunity Program, unpubl. data, 7600 Sand Point Way NE, Seattle, WA 98115). With the given data it is not possible to determine which fishery (or even which country) caused the mortality. Note, that this mortality has been attributed the Western North Pacific stock, but without a tissue sample (for genetic analysis) or a photograph (for matching to known Japanese animals) it is not possible to be for certain (i.e., it may have belonged to the CNP stock). No strandings or sightings of entangled humpback whales of this stock were reported between 1999 and 2003. However, effort in western Alaska is low.

The estimated annual mortality rate incidental to commercial fisheries is 0.49 whales per year from the CNP stock. However, this estimate is considered a minimum because there are no data concerning fishery-related mortalities in Japanese, Russian, or international waters. In addition, there is a small probability that fishery interactions discussed in the assessment for the CNP stock may have involved animals from this stock because the only known matches to feeding areas come from areas typically used by the CNP stock. Finally, much information on fishery interaction with the CNP stock is based on information reported to the Alaska Region as stranding data. However, very few stranding reports are received from areas west of Kodiak.

Brownell et al. (2000) compiled records of bycatch in Japanese and Korean commercial fisheries between 1993 and 2000. During the period 1995-1999, there were six humpback whales indicated as "bycatch". In addition, two strandings were reported during this period. Furthermore, analysis of four samples from meat found in markets indicated that humpback whales are being sold. At this time, it is not known whether any or all strandings were caused by incidental interactions with commercial fisheries; similarly, it is not known whether the humpback whales identified in market samples were killed as a result of incidental interactions with commercial fisheries. It is also not known which fishery may be responsible for the bycatch. Regardless, these data indicate a minimum mortality level of 1.1/year (using bycatch data only) to 2.4/year (using bycatch, stranding, and market data) in the waters of Japan and Korea.

Central North Pacific Stock

In 1994, the incidental take of a humpback whale was reported in the Southeast Alaska salmon purse seine fishery. Another humpback whale is known to have been taken incidentally in this fishery in 1989. In 1996, a humpback whale was reported entangled and trailing gear as a result of interacting with the Southeast Alaska drift gillnet fishery. This whale is presumed to have died. Together, these two mortalities result in an annual mortality rate of 0.4 (0.2 + 0.2) humpback whales based on self-reported fisheries information for this stock. This is considered to be a minimum estimate because logbook records (fisher self-reports required during 1990-1994) are most likely negatively biased (Credle et al., 1994).

The primary effects of the Hawai`i-based pelagic, deep-set longline fishery on humpback whales result from direct interactions with the fishing gear. Humpback whales are present as they migrate to and from and occur in waters surrounding the Hawaiian Islands during the winter months. The deep-set longline fishery generally occurs at locations where humpback whales are uncommon. Thus, interactions between the Hawai`i-based deep-set longline fishery and humpback whales are rare and unpredictable events. Since 2001, there have been only four observed interactions between the species and the entire Hawai`i-based longline fleet. During this same time period the CNP stock of humpback whales has been steadily increasing in abundance. One interaction per year with adult humpback whales was observed in the deep-set longline fishery in 2001, 2002 and 2004 (Table 3). Two of these interactions occurred outside of the U.S. EEZ. The other two interactions occurred within the U.S. EEZ around Hawai`i. According to NMFS observer characterizations of these events, the whales were or may have become entangled in a main longline. In each instance, efforts were taken to disentangle the whale, and all whales were either released or able to break free from the gear without noticeable impairment the animals' ability to swim or feed. NMFS review also determined that any injuries to the animals as a result of these interactions were not likely to result in mortality under the MMPA serious injury guidelines (Angliss and DeMaster, 1998).

Table 3. Summary of observed interactions between humpback whales and the Hawai`i-based longline fleet from 1994-2006. Seriousness of injuries was assessed under MMPA serious injury guidelines (Angliss and Demaster, 1998).

DATE	EEZ	NMFS' DETERMINED INJURY SEVERITY
2/11/2001	HAWAI`I	NOT SERIOUS
10/12/2002	OUTSIDE	NOT SERIOUS
2/16/2004	OUTSIDE	NOT SERIOUS
2/19/2006	HAWAI`I	NOT YET DETERMINED

Further analyses of these interactions determined that these events resulted in non-serious injuries, indicating that the animals were hooked in a region other than the head, were released with no or minimal gear attached, and the interactions were not expected to result in mortality.

Such interactions are extremely rare events when viewed in relation to the amount of fishing effort that has occurred in the deep-set fishery during this period of time. Humpback whale interactions are likely rare events in this fishery because the fishery occurs largely in areas where humpback whales are unlikely to occur. Observed humpback interactions in the deep-set fishery were not extrapolated to the entire fishery due to the rare and sporadic occurrence of interactions, the fact that humpbacks occur in the action area only in the winter months, and the lack of a uniform occurrence of the species across spatial distribution of the deep-set longline fishery.

Observations of the Hawai'i-based shallow-set longline fishery between 1994 and 2005 recorded no interactions with CNP humpback whales. However, there has been one recent interaction during which a humpback whale was entangled in the main line of a shallow-set longline swordfish boat on February 19, 2006. There are no information available documenting interactions between CNP humpback whales and the Hawai'i based troll, handline and pole and line fisheries or the PRIA, American Samoan, Guam or CNMI based fisheries, although these fisheries are not observed. Given level of effort, selectivity of gear, and location of fishing effort relative to CNP humpback stock, NMFS expects that interactions between CNP humpbacks and these fisheries would be rare.

In Alaska, humpback whales are killed incidentally in Federal groundfish and longline fisheries and State managed-commercial salmon fisheries. Four commercial fisheries within the range of the CNP humpback whale population have been observed for incidental mortality of humpback whales between 1990 and 2003: Bering Sea/Aleutian Island (BSAI) groundfish trawl, Gulf of Alaska (GOA) groundfish trawl, longline, and pot fisheries. Average annual mortality from the observed fisheries during this time was 1.5 (CV = 0.47) humpback whales (National Marine Fisheries Service, 2005).

An additional source of information on the number of humpback whales killed or injured incidental to fisheries come from stranding reports. Under the MMPA, vessel operators are required to 'self-report' fishery information on the number of humpback whales killed or injured incidental to commercial fishery operations. There were no fisher self-reports of humpback whale injuries or mortalities from interactions with commercial fishing gear in any Alaska fishery within the range of the CNP humpback whale stock from 1990 and 1993 (Angliss and Outlaw, 2005). Logbook data are partially available from 1989-94. In 1994 incidental mortality reporting requirements were modified, logbook requirements were retracted and replaced with self-reporting requirements. Data for the 1994-95 phase-in period are fragmentary. After 1995, the overall level of reporting dropped dramatically, such that the records are considered incomplete and estimates of mortality are assumed to minimum estimates.

6.3.6.3 Entanglement

Central North Pacific Stock

In recent years, an increasing number of entangled humpback whales have been reported in Alaska. Fifty-two humpbacks were reported entangled in Alaska from 1997-2004 and 40 of these involved southeast Alaska humpbacks (Neilson et al., 2005). In 2005, 22 entangled humpback whales were reported to the NMFS Alaska stranding program. Twelve of these were reported in southeast Alaska, and nine in southcentral Alaska in the Kodiak, Homer, and Seward regions.

To understand more about the prevalence of these entanglement incidents, a study in 2003 and 2004 documented entanglement scarring in the humpback population in northern southeast Alaska. Using methodology developed in the Gulf of Maine to investigate scarring in Atlantic large whales, Neilson et al. (2005) photographed the caudal peduncle of individual humpbacks as they dove and examined them for scars indicative of previous entanglement. Their results indicate that, based on caudal peduncle scarring, 71% (95% CI = 62%-78%) of the humpback whales in northern southeast Alaska have been entangled at least once. The study also found that 8% of the whales photographed in Icy Strait/Glacier Bay acquired new entanglement scars between the two years that they were sampled. Calves were less likely to have entanglement scars than older whales, and there was no significant difference in scarring percentages between males and females. Overall, the percentage of whales with entanglement scars in northern southeast Alaska is comparable to Gulf of Maine humpback whales (48%-65% entanglement percentage). Based on similar scarring investigations carried out in Hawai`i, 14% of the humpbacks there appear to have been entangled (Robbins and Mattila, 2004).

From these sources, between 1997 and 2004, 51 reports of humpback whale entanglements in Alaska were submitted to NMFS. Much of the gear involved in these interactions has originated from pot, long line, seine, and gill net fisheries, while other gear is of unknown origin. In 2005, NMFS received 19 reports of humpback whale entanglements in Alaska, although it is not clear whether all are distinct records or some are re-sighted entanglements. Additionally, it is difficult to quantify these impacts relative to a specific fishery and to the whales themselves because of insufficient information obtained on these entanglements. For entanglements that do not result in immediate or discernable mortality, it is difficult to determine the extent of impact to the animal. Most entangled whales reported to the marine mammal stranding network in Alaska are not re-sighted. Without further information, it is unclear which types of entanglements are ultimately life-threatening. Data such as that collected by Neilson et al. (2005), however, leads to the conclusion that many humpback whales survive their entanglements. Some, it would appear, survive multiple entanglement incidents.

Reports of entangled humpback whales found swimming, floating, or stranded with fishing gear attached occur in both Alaskan and Hawaiian waters. There were 30 reports of human-related mortalities or injuries from 1999-2001. Of these, there were 21 incidents which involved commercial fishing gear, and 13 of those incidents involved serious injuries or mortalities. An additional seven incidents of human-related mortality or injury involved ship strikes and will be discussed in a forthcoming section. This estimate is considered a minimum because not all entangled animals strand and not all stranded animals are found, reported, or cause of death determined.

The number of confirmed reports of entangled whales in Hawaiian waters has increased in recent years (Table 4). Many of the whales reported entangled in Hawaiian waters most likely brought the gear with them from higher latitude feeding grounds. While the whales are not typically at risk from drowning or immediate death, they are at increased risk of starvation, infection, physical trauma from the gear, and ship strikes as a result of the entanglement. Since 2002, the Hawaiian Islands Humpback Whale National Marine Sanctuary and NMFS have worked together to improve outreach, response capabilities, and creating an emergency hotline number.

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Table 4. List of confirmed and unconfirmed (*) entanglements of humpback whales from 2001-2006. Data compiled by Hawaiian Island Humpback Whale National Marine Sanctuary.

Date	Location/region	Description of entanglement	Response
2001		Line and buoy from Alaska fishery	Successful release
1/8/2003	Hawai'i (SE)	Line wrapped around tail; trailing 20 ft with 2 plastic mooring balls.	Event not confirmed
2/24/2003	Auiau Channel (W.Maui)	Line wrapped pec fins; trailing 100-120 ft.	Successful release
3/2/2003*	Pailolo Channel, Moloka'i	Animal trailing large orange buoy.	No response mounted.
3/4/2003*	Kamalapau Harbor, Lana'i	Animal trailing buoy 30 ft.	Unsuccessful/ Animal not found
2/2/2004	Auiau Channel (W.Maui)		Unsuccessful disentanglement
2/13/2004	Kaua'i Channel	Animal towing 50 yards (yds) of line/ rope.	Event not confirmed
1/6/2005*	Port Allen, Kaua'i	Line trailing from forward with ball of blue/green net 20-30 ft behind.	Unsuccessful/ Animal not found
1/24/2005	O'ahu (E)	Gillnetting over the head, rope across jaw, and debris wrapped around pec fin.	Unsuccessful/ Unable to respond
2/4/2005*	Hapuna Beach, Big Island	Blue rope with 2 orange buoys running along flank near tail.	Unsuccessful/ Animal not found
2/9/2005	O'ahu (N)	Buoyline of local fish trap gear around tail with a 50 pound (lb) anchor, 2 round, and 1 bullet buoy.	Unsuccessful/ Animal not found
2/11/2005	Auiau Channel (W.Maui)	Line around pec and entering mouth trailing 150 ft.	Assessed/ Not in need of assistance/ disentanglement
2/28/2005	Auiau Channel (W.Maui)	At least one, perhaps two lines in mouth; line under the body between left and right flippers with gear 6-8 ft from fluke.	Partially successful disentanglement
3/2/2005*	O'ahu (W)		Unsuccessful/ Animal not found
12/27/2005	Kaua'i (E)	Rope with float trails 10-15 ft.	Assessed/ Not in need of assistance/ disentanglement
1/29/2006	Kawaihae Bay, Big Island	Line wrapped around tail; pair of lines trail 20-25 ft with ball of gear.	Unsuccessful/ Animal not found
2/9/2006	Kawaihae Bay, Big Island	Large red polyball at dorsal fin; lines trail to fluke with another polyball.	
2/12/2006	Auiau Channel (W.Maui)		Partially successful disentanglement
2/16/2006	Kawaihae Bay, Big Island	2 buoys trailing 35 ft on the tail and fluke was seen free of gear.	Unsuccessful/ Animal not found
2/18/2006*	O'ahu (N)	Animal may be entangled in gear with buoy near tail.	Unsuccessful/Animal not found
2/23/2006*	Waikiki Beach, O'ahu	Animal towing buoy 30 ft.	No response mounted
3/1/2006	North Pacific	Caught by entanglement in the main line and cut free, but not all the gear was removed.	Partially successful disentanglement
3/2/2006*	Kona Coast, Big Island	Animal has line around tail and trailing gear.	Unsuccessful/Animal not found
3/5/2006	Auiau Channel (western Maui)	Over 100 lbs/357 ft of line around the fluke and tail and trailed 20 ft with a ball of line.	Successful disentanglement

Humpback whales have been injured or killed elsewhere along the mainland U.S. and Hawai`i (Barlow et al., 1997). In 1991, a humpback whale was observed entangled in longline gear, disentangled, and released alive (Hill et al., 1997). In 1995, a humpback whale in Maui waters was found trailing numerous lines (not fishery-related) and entangled in mooring lines. The whale was successfully released, but subsequently became entrapped and was attacked and killed by tiger sharks in the surf zone. In 1996, a humpback whale calf was found stranded on O`ahu with evidence of vessel collision (propeller cuts; NMFS unpublished data). Also in 1996, a vessel from Pacific Missile Range Facility in Hawai`i rescued an entangled humpback, removing two crabpot floats from the whale; the gear was traced to a recreational fisherman in southeast Alaska (R. Inouye, personal communication). No information is available on the number of humpback whales that have been killed or seriously injured by interactions with fishing fleets outside of U.S. waters in the North Pacific Ocean.

The effects of trailing fishing gear on large whale species are largely unknown. NMFS sponsored a workshop to discuss methods for differentiating serious and non-serious injury of marine mammals taken in commercial fishing operations. Results of this workshop indicate that some but not all entanglements may result in serious injury or mortality (Angliss and DeMaster, 1997). Available evidence from entangled north Atlantic right whales indicates that while it is not possible to predict whether an animal will free itself of gear, a high proportion are believed to lose or extricate themselves based on scarring observed among apparently healthy animals. At the workshop, predicting the survivability of individual animals that are entangled was determined to be unreliable. Some whales have been observed to carry gear for over five years. The workgroup was in general agreement that entanglement that impedes locomotion or feeding, and entanglement of young whales, should be considered a serious injury (Angliss and Demaster, 1997).

The overall fishery-related minimum mortality and serious injury rate for the CNP stock is 3.39 humpback whales per year, based on observer data from Alaska (0.49), self reports from Alaska (0.4), stranding records from Alaska (2.25), and stranding records from Hawai`i (0.25). The estimated fishery-related minimum mortality and serious injury rate incidental to commercial fisheries for the northern portion of the stock is 1.74 humpback whales per year, based on observer data from Alaska (0.49), stranding records from Alaska (1.0), and stranding data from Hawai`i (0.25). Note that, because it is unknown whether the stranding reports for Hawai`i involve animals from the central or northern portion of the CNP stock, the level of serious injury/mortality is assessed as if it came from either stock. However, the 0.25 animals per year reported via stranding reports for Hawai`i is included once for the entire stock. As mentioned previously, these estimates of serious injury/mortality levels should be considered a minimum.

6.3.6.4 Ship Strikes

Many humpback whales are killed by ship strikes along both coasts of the U.S. On the Pacific coast, a humpback whale is killed about every other year by ship strikes (Barlow et al., 1997). Humpback whales, especially calves and juveniles, are highly vulnerable to ship strikes and other interactions with non-fishing vessels. Younger whales spend more time at the surface, are less visible and closer to shore (Herman et al., 1980; Mobley, et al., 1999), thereby making them more susceptible to collisions.

Eastern North Pacific Stock

Ship strikes were implicated in the deaths of at least two humpback whales from the Eastern North Pacific stock in 1993, one in 1995, and one in 2000 (J. Cordaro, NMFS unpubl. data). During 1999-2003, there were an additional five injuries and two mortalities of unidentified large whales attributed to ship strikes. Additional mortality from ship strikes probably goes unreported because the whales do not strand or, if they do, they do not have obvious signs of trauma. Several humpback whales have been photographed in California with large gashes in their dorsal surface that appear to be from ship strikes (J. Calambokidis, personal communication). The average number of humpback whale deaths by ship strikes for 1999-2003 is at least 0.2 per year.

Central North Pacific Stock

There appears to be an increased frequency at which collisions with humpback whales and vessels are occurring in Hawaiian waters (Table 5), especially in the shallow waters (less than 100 fathoms) of the four-island region of Maui county and Penguin Banks, the preferred habitat by the whales wintering in Hawai'i (Lammers et al., 2003). Three types of collisions reports were documented: collisions with little/no forewarning; collisions resulting from effort to avoid whales; circumstantial collisions not reported but evidence of trauma known. The majority of the collisions are with boats from 19-80 ft in length, including both slow and fast moving vessels. Also, the highest incidents of collisions were documented from the island of Maui, and the lowest number documented was from the island of Kaua'i.

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Table 5. List of humpback whale collisions with vessels from 2001-2006 compiled from (Jensen and Silber 2003), Lammers et al. 2003, and local print media (Honolulu Advertiser), and NMFS.

Date	Location	Description of Collision
02/08/01	Lana`i	Injured whale
02/13/01	Olowalu, Maui	Injured whale (5-6 m)
02/15/01	Port Allen, Kaua`i	Juvenile whale breached on a 40-ft whale-watching catamaran. Whale unhurt.
03/15/02	Ma`alaea, Maui	Whale hit 65-ft. catamaran; no apparent injuries
03/27/02	Lahaina, Maui	Claf reported with scars on back; struck by undisclosed vessel
04/04/02	Ma`alaea, Maui	No sign of injury
02/10/03	Ma`alaea, Maui	Subadult whale hit a boat; no visible injuries.
03/07/03	Ma`alaea, Maui	Whale struck whale-watching boat; no reported injuries
-/-/03		Whale collision with cargo ship at night
01/05/04	Maui	Whale struck by local fisherman
02/08/04	Lahaina, Maui	Calf unjured by speeding boat
02/28/05	Lahaina, Maui	Calfs observed with boat propeller injuries along back
02/06/05	Lana`i	Calf collision with passenger ferry
01/04/06	Maui	vessel collision with whale-watching boat
01/07/06	Kauai	whale watching tour struck adult whale; whale appeared healthy and no boat damage
01/17/06	Kauai	vessel collision
02/13/06	Maalea Bay, Maui	adult humpback was struck by a U.S. Coast Guard vessel
03/09/06	Maalea Bay, Maui	Head injury to mom and calf whale when surfaced near whale-watching boat
03/15/06	Maui	U.S.Coast Guard whale-watch tour reported collision; right pectoral fin injured
03/25/06	Lahaina, Maui	Mom and calf pair collided with whale-watching boat

The increasing rate of whale and vessel collisions may have a number of contributing factors, the most important of which may be that the population of humpback whales in Hawai`i is increasing (Lammers et al., 2003). In addition, there is a corresponding rise in the number of vessels in the preferred habitat for humpback whales, a direct result of the growing popularity of eco-tourism in Maui and the surrounding areas. Efforts to reduce these interactions include improved technological research into mapping models and radar and sonar detection systems, state regulations prohibiting parasailing and personal watercrafts in Maui waters during whale season (December 15 – May 15), and a NOAA hot line to report humpback whale interactions.

Although there is no official reporting system for ship strikes, numerous incidents of vessel collisions have been documented in Alaska. Forty-eight reports from 1986 to 2005 representing confirmed, unconfirmed and suspected ship strikes with humpback whales exist in the NMFS stranding database. This is a minimum estimate, as not all whales struck are reported and not all whales struck are identified to species or cause of mortality. The fate of struck animals is also not always determined unless the whale dies immediately upon impact or is discovered as a carcass on the bow of a ship and it can be determined that the strike was the cause of death.

Humpback whale distribution overlaps significantly with the transit routes of large commercial vessels that ply the waters off Alaska. The larger vessels are cruise ships, large tug and barge transport vessels, and oil transport tankers. Cruise ships frequent the inside waters of southeast Alaska, passing through areas used by humpback whales for feeding, such as Glacier Bay National Park and Preserve, Point Adolphus and, adjacent to the action area, the waters of Lynn Canal en route to Skagway and Haines. Tug and barge transport follows much of the traffic pattern of the cruise ships, as they frequent the same coastal communities. Oil transport tankers are generally operating farther offshore where there are presumably fewer concentrations of humpback whales, except for transit through Prince William Sound. Collisions in Alaska can generally occur throughout the region, peaking during the summer season.

Records of vessel collisions with large whales in Alaska indicate that strikes have involved cruise ships, recreational cruisers, whale watching catamarans, fishing vessels, and skiffs. Vessel lengths associated with these records ranged from approximately 20 feet to over 250 feet, indicating that all types and sizes of watercraft pose a threat of collision for whales (Jensen and Silber, 2003). Cruise ships are of particular concern, as they operate at considerably high speeds and frequent the inside waters of southeast Alaska with routes passing through areas of humpback whale abundance such as Glacier Bay National Park and Preserve, Point Adolphus and, adjacent to the action area, the waters of Lynn Canal. In addition to large ships, which are most likely to cause significant injury or death to humpback whales, smaller tour, charter and private vessels also significantly overlap with inshore humpback whale distribution in Alaska waters. Smaller ships also have the potential to cause disturbance, serious injury, and possibly mortality.

Several incidents of vessel interactions with humpback whales in Glacier Bay have been documented in recent years. In 2001, a dead and pregnant humpback whale was discovered in Park waters. A necropsy determined the whale likely had been killed by blunt trauma, possibly from a large vessel collision. In 2002, one mortality occurred inside Park waters and several additional collisions were documented (Doherty and Gabriele, 2002). Other interactions included close approaches and possible harassment by several vessels of different vessel classes including a kayak, a cruise ship and a floatplane. Researchers also documented an injury to the dorsal fin of a whale that could have been caused by a vessel collision/interaction. In 2003, a humpback whale was necropsied that had been first seen at Pt. Manby, Yakutat Bay. The results of that necropsy also indicated that the whale had been killed by blunt trauma as a result of large vessel collision. In 2004, a humpback whale calf in Glacier Bay was necropsied on Strawberry Island. Severe dislocation of six ribs caused massive bleeding and tissue damage; blunt trauma indicated injury consistent with vessel collision. A second incident in 2004 involved a humpback (nursing calf) necropsied on the south end of Douglas Island outside of Juneau. Results of this necropsy showed a severe scapular fracture and again indicated likely collision with a vessel based on blunt trauma to the animal.

Between 2001 and 2005, opportunistic reports of vessel collisions with humpback whales indicate an average of five humpback whales struck per year in Alaska. During this time, approximately one vessel strike per year has resulted in a known mortality to a humpback whale in southeast Alaska. In 2005, 12 humpback whale ship strikes were reported, a significant increase over previous years. It is unclear whether this reflects an increase in the incidence of

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collisions, or a greater awareness of about reporting such events. The higher number of whale and vessel collisions in 2005 may be a result of the increasing abundance of humpback whales foraging in Alaska, as well as the growing presence of marine-based tourism in Alaska’s coastal waters. Given these factors, it is likely that injury and mortality of humpback whales will continue into the future as a result of vessel strike.

To minimize the possibility of collision and the potential for harassment, NMFS implemented regulations on July 2, 2001, that imposed vessel restrictions on approaching humpback whales closer than 100 yards. Operating at a “slow, safe speed” when near humpback whales is also required. The National Park Service has implemented even greater minimum approach distances in Glacier Bay National Park (1/4 mile in all Park waters) for humpback whales, which likely reduces the whales’ underwater noise exposure and potential for behavioral disturbance. In addition, the Park has passed new vessel management measures that allow speed restrictions of 13 knots to be imposed by Park management on an as-warranted basis in the bay.

Table 6. Alaska Strandings (NMFS database) of collisions between humpbacks and vessels, 2001-2005. This table reflects opportunistic data collection, with the level of confidence varying from thoroughly investigated to unconfirmed reports involving animals positively identified as humpback whales to animals likely to have been humpback whales.

Year	Area	Vessel			Details
		Type	Length (ft)	Speed (knots)	
2001	Anchorage	Container ship	D7 class	12-19 knots	Dead on ship's bulbous bow
2001	Dixon Entrance	U.S. Coast Guard	110' LOA	12 knots	Fate unknown
2001	Glacier Bay	-	-	-	Necropsy: Injury consistent with strike
2001	Pacific Ocean (Southeast Alaska)	Cruise ship	963' LOA	-	Fate unknown (possible humpback)
2002	Fern Harbor	Charter	62'	Neutral Coasting	Apparently healthy, fate unknown
2003	Auke Bay	-	-	-	Fate unknown (possible humpback)
2003	Baranof Island	Cruise ship	780' LOA	19 knot (avg.)	Fate unknown (suspected collision, possible humpback)
2003	Bering Sea open water	-	-	-	Fate unknown (possible humpback)
2003	Icy Bay	-	-	-	Necropsy: Injury consistent with strike
2003	Sitka Sound	-	-	-	Fate unknown
2003	Wrangell	Cruise ship	754' LOA	Entering harbor	Fate unknown (suspected collision)
2004	Benjamin Island	-	-	Drifting	Fate unknown
2004	Glacier Bay	-	-	-	Necropsy: Injury consistent with strike
2004	Douglas Island	-	-	-	Necropsy: Injury consistent with strike
2005	George Inlet, Ketchikan	Whalewatch	48'		Fate unknown
2005	Glacier Bay	Cruise ship			Fate unknown
2005	Kachemak Bay	Charter boat	28'		Blood in water, whale swam away; fate unknown (possible humpback)
2005	Sitka Sound	Cruise ship	936'	10 knots	Fate unknown

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2005	Prince William Sound	Recreational vessel	26'	19 knots	Whale surfaced and swam away; fate unknown (vessel sank)
2005	Icy Strait	Whalewatch charter	26'		Fate unknown (humpback calf)
2005	Juneau area	Tour vessel	143'		Whale swam away after strike; fate unknown
2005	Kake area Frederick Sound	skiff	28'	25 knots	Whale dove after strike; fate unknown
2005	Stephens Passage south of Taku Inlet	Small tour vessel			Fate unknown
2005	Stephens Passage	Tour vessel		10 knots	Fate unknown
2005	Brothers Islands Frederick Sound	Cruise ship	294'	20 knots	Fate unknown
2005	Peril Strait	unknown			Necropsy: Blunt trauma consistent with strike

6.3.6.5 Whale Watching

The CNP stock is the focus of a large whale watching industry in its wintering grounds (Hawai'i) and a growing whale watching industry in its summering grounds (Alaska). The growth of the whale watching industry is a concern for humpback whales since harassment may occur, preferred habitats may be abandoned, and fitness or survivability may be compromised if disturbance levels are too high. Regulations concerning minimum distance (100 yards or 90 m when on the water; 1,000 feet or 300 m when operating an aircraft) to keep from whales and how to operate vessels when in the vicinity of whales have been developed for Hawai'i waters in an attempt to minimize the impact of whale watching (50 CFR 224.104). In 2001, NMFS issued regulations to prohibit most approaches to humpback whales in Alaska within 100 yards (50 CFR 224.104). The growth of the whale watching industry, however, is a concern as preferred habitats may be abandoned if disturbance levels are too high. Likewise, in Alaska, the number of cruise ships entering Glacier Bay has been limited to reduce possible disturbance (Baker et al., 1988) and vessel approach limits are set at 183 m (200 yd).

6.3.6.6 Noise

Humpback whales seem to respond to moving sound sources, such as whale-watching vessels, fishing vessels, recreational vessels, and low-flying aircraft (Anon, 1987; Beach and Weinrich, 1989; Clapham et al., 1993; Tinney, 1988; Atkins and Swartz, 1989; Green and Green, 1990). Their responses to noise are variable and have been correlated with the size, composition, and behavior of the whales when the noises occurred (Herman et al., 1980; Watkins et al., 1981; Krieger and Wing, 1986; Glockner-Ferrari and Ferrari, 1985; Glockner-Ferrari, 1990). Several investigators have suggested that noise may have caused humpback whales to avoid or leave feeding or nursery areas (Jurasz and Jurasz, 1979; Dean et al., 1985; Glockner-Ferrari and Ferrari, 1985; Glockner-Ferrari, 1990; Salden, 1988), while others have suggested that humpback whales may become habituated to vessel traffic and its associated noise (Watkins, 1986; Belt et al., 1989). Still other researchers suggest that humpback whales may become more vulnerable to vessel strikes once they habituate to vessel traffic (Swingle et al., 1993; Wiley et al., 1995).

In Hawai'i, a 1996 study measured the acoustic noise of different whale-watching boats (Au and Green, 2000) and determined that the sound levels were unlikely to produce grave effects on the

humpback whale auditory system. The background chorusing of the humpback whales were high enough at times to contaminate the boat sounds recorded. Noise from the ATOC program, the U.S. Navy's Low Frequency Active (LFA) sonar program, and other anthropogenic sources (i.e., shipping and whale watching) in Hawai'i waters may be of concern for this stock. Results from experiments in 1996 off Hawai'i indicated only subtle responses of humpback whales to ATOC-like transmissions (Frankel and Clark, 1998). Frankel and Clark (2002) indicated that there were also slight shifts in humpback whale distribution in response to ATOC. It was later confirmed (Mobley, 2005) that the numbers and patterns of humpback whales returning to winter in the waters off Kaua'i did not change after four years of exposure to the transmissions of ATOC (which recommenced in 2002 as a part of the North Pacific Acoustic Laboratory program [NPAL]). Efforts are underway to evaluate the relative contribution of noise (e.g., experiments with LFA sound sources) to Hawai'i's marine environment, although reports summarizing the results of recent research are not available.

6.3.7 Distribution of Humpback Whales in the Action Area

The worldwide population of humpback whales is divided into northern and southern ocean populations, but genetic analyses suggest some gene flow (either past or present) between the North and South Pacific (e.g., Baker et al., 1993; Caballero et al., 2001). Although considered to be mainly a coastal species, humpback whales often traverse deep pelagic areas while migrating. Most migratory paths for southern humpback whales are unknown (Perry et al., 1999). The Southern Hemisphere population that can be found south of 60°S in the summer feeding season has a population estimate of 10,000 individuals.

Humpback whales were observed 342 times during 20 years of the IWC/IDCR-Southern Ocean Whale and Ecosystem Research (SOWER) Antarctic summer sightings surveys (Branch and Butterworth, 2001). Fifty-seven of those sightings occurred in Area VI (120°W to 170°W) during the three summers that Area VI was surveyed. Those surveys provided abundance estimates of 7,100–9,300 humpback whales for the entire Antarctic population. Butterworth et al. (1994) calculated an uncorrected density estimate of 2.67/1000 nmi of survey effort in Antarctic Area VI (south of 60°S) for one of the IWC/IDCR summer sighting surveys. During the 1965–66 to 1987–88 summer whaling seasons, Japanese scouting vessels reported no sightings of humpback whales in Area VI (between the latitudes of 50°S and 40°S) during 14,695 nmi of survey effort, and no sightings between the latitudes of 40°S and 30°S during 122 nmi of survey effort (Butterworth et al., 1995).

Humpback whales spend winter on low-latitude breeding grounds (Clapham, 2002). Off the Cook Islands, 8°S of the action area, humpback whales have been sighted from July to October (Hauser et al., 2000). Genetic evidence suggests several discrete breeding grounds in the South Pacific, including distinction between the Cook Islands and French Polynesia (Olavarría et al., 2003). However, photo-identification work suggests some movement between those two areas and between the Cook Islands and Tonga (Garrigue et al., 2002). The southern Cook Islands appear to be a winter calving ground for humpback whales, presumably from Antarctic Area VI (Hauser et al., 2000).

In New Caledonia, humpback whales were estimated at 314 in the year 2000 (Garrigue et al., 2001), and 770 whales were estimated for Tonga (Baker et al., 2001). It is clear that these

numbers are well below the levels that supported commercial whaling in the past. Humpback whales were hunted consistently from New Zealand, but only very little whaling was conducted in Oceania of the South Pacific. Tongan whaling, though small in scale, recorded annual catches prior to 1960 at 30-40 whales.

Humpback whale wintering grounds also include all four archipelagos of French Polynesia (the Society, Marquesas, Tuamotu, and Australes Islands groups) as suggested by the presence of singing males (Gannier et al., 2003). Humpback whales were sighted 35 times during >4600 km of inshore survey effort and >550 km of offshore survey effort in the Society Islands during three years of fall and spring shipboard surveys (Gannier, 2000a). All sightings occurred during September–November. They were not sighted during November–January 1999 sighting surveys in the Marquesas Islands (Gannier, 2002a).

The most recent NMFS/PIFSC survey on the NOAA ship *Oscar Elton Sette* (March 5-28, 2006) in the waters of American Samoa, the central equatorial Pacific, Johnston Atoll, and surrounding areas did not include sightings of humpback whales. However, this survey was conducted during the late spring months when humpback whales are likely to be leaving their winter breeding grounds for higher latitude feeding areas. Young (2000) recognized the humpback whale as one of the 12 most common cetaceans in the western South Pacific, but no population estimates were provided. There are no specific data on the occurrence of humpback whales in the proposed action area. Therefore, it seems that based on the best scientific data available, it is likely that humpback whales would be found in the action area, especially during the winter breeding months.

6.4 Sei Whales

6.4.1 Species Description and Distribution

Sei whales are distributed in all of the world's oceans, except the Arctic Ocean. The IWC's Scientific Committee groups all of the sei whales in the entire North Pacific Ocean into one population (Donovan, 1991). However, some mark-recapture, catch distribution, and morphological research indicated that more than one population exists: one between 175°W and 155°W longitude, and another east of 155° W longitude (Masaki, 1976; 1977). During the winter, sei whales are found from 20° - 23° N and during the summer from 35° - 50° N (Masaki, 1976; 1977). Horwood (1987) reported that 75-85% of the total North Pacific population of sei whales resides east of 180° longitude.

In the North Pacific Ocean, sei whales have been reported primarily south of the Aleutian Islands, in Shelikof Strait and waters surrounding Kodiak Island, in the Gulf of Alaska, and inside waters of southeast Alaska (Nasu, 1974; Leatherwood et al., 1988). Sei whales have been occasionally reported from the Bering Sea and in low numbers on the central Bering Sea shelf (Hill and DeMaster, 1998). Masaki (1977) reported sei whales concentrating in the northern and western Bering Sea from July through September, although other researchers question these observations because no other surveys have ever reported sei whales in the northern and western Bering Sea. Horwood (1987) evaluated the Japanese sighting data and concluded that sei whales rarely occur in the Bering Sea.

Sei whales are distributed far out to sea in temperate regions of the world and do not appear to be associated with coastal features. There is still insufficient information to accurately determine population structure, but from a conservation perspective it may be risky to assume panmixia in the entire North Pacific. Four sightings of sei whales were recently made during a summer/fall 2002 shipboard survey of waters within the U.S. EEZ of the Hawaiian Islands (Barlow, 2003). For the MMPA stock assessment reports, sei whales within the Pacific U.S. EEZ are divided into three discrete, non-contiguous areas: 1) waters around Hawai'i (this report), 2) California, Oregon, and Washington waters, and 3) Alaskan waters.

6.4.2 Life History

Reproductive activities for sei whales occur primarily in winter. Gestation is about 12.7 months and the calving interval is about 3 years (Rice, 1977). Sei whales become sexually mature at about age 10 (Rice, 1977). The age structure of the sei whale population is unknown. Rice (1977) estimated total annual mortality for adult females as 0.088 and adult males as 0.103. Andrews (1916) suggested that killer whales attacked sei whales less frequently than fin and blue whales in the same areas. Sei whales in the North Pacific feed on euphausiids and copepods, which make up about 95% of their diets (Calkins, 1986). The balance of their diet consists of squid and schooling fish, including smelt, sand lance, Arctic cod, rockfish, pollock, capelin, and Atka mackerel (Nemoto and Kawamura, 1977). Rice (1977) suggested that the diverse diet of sei whales may allow them greater opportunity to take advantage of variable prey resources, but may also increase their potential for competition with commercial fisheries. Endoparasitic helminths are commonly found in sei whales and can result in pathogenic effects when infestations occur in the liver and kidneys (Rice, 1977).

6.4.3 Listing Status

In the North Pacific, the IWC began management of commercial harvest of sei whales in 1970, and sei whales were given full protection in 1976 (Allen, 1980). Sei whales were listed as endangered under the ESA in 1973 and by IUCN. They are also protected under CITES as an Appendix 1 species, and are automatically protected under the MMPA. They are listed as endangered under the IUCN Red List of Threatened Animals. Critical habitat has not been designated for sei whales.

6.4.4 Population Status and Trends

Ohsumi and Wada (1974) estimate the pre-whaling abundance of sei whales to be 58,000-62,000 in the North Pacific. Later, Tillman (1977) used a variety of different methods to estimate the abundance of sei whales in the North Pacific and revised this pre-whaling estimate to 42,000. His estimates for the year 1974 ranged from 7,260-12,620. All methods depend on using the history of catches and trends in catch per unit of effort or sighting rates. Japanese and Soviet catches of sei whales in the North Pacific and Bering Sea increased from 260 whales in 1962 to over 4,500 in 1968 and 1969, after which the sei whale population declined rapidly (Mizroch et al., 1984b). When commercial whaling for sei whales ended in 1974, the population of sei whales in the North Pacific had been reduced to between 7,260 and 12,620 animals (Tillman, 1977). Current abundance or trends are not known for sei whales in the North Pacific. There have been no direct estimates of sei whale abundance in the entire (or eastern) North Pacific based on sighting surveys.

Only two confirmed sightings of sei whales and five possible sightings (identified as sei or Bryde's whales) were made in California, Oregon, and Washington waters during extensive ship and aerial surveys in 1991, 1992, 1993, 1996, and 2001 (Hill and Barlow, 1992; Carretta and Forney, 1993; Mangels and Gerrodette, 1994; VonSaunders and Barlow, 1999; Barlow, 2003). Green et al. (1992) did not report any sightings of sei whales in aerial surveys of Oregon and Washington. Sei whales were not sighted in ten 5° latitude × 5° longitude survey blocks in the southwestern portion of the ETP during 1986-1996 summer and fall research vessel surveys (Ferguson and Barlow, 2001). Whales identified as either Bryde's or sei whales were sighted 12 times in nine 5° × 5° survey blocks in the surveys. Densities were 0.1–1.1/1000 km².

The abundance estimate for California, Oregon, and Washington waters out to 300 nmi, based on 1996 and 2001 shipboard surveys, is 56 (CV = 0.61) whales (Barlow, 2003). There are no estimates of the growth rate of sei whale populations in the North Pacific (Best, 1993). The PBR level for Northeast Pacific stock is calculated as the minimum population size (35) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a F_R of 0.1 (for an endangered species), resulting in a PBR of 0.1

As part of the Marine Mammal Research Program of the ATOC study, a total of 12 aerial surveys were conducted within about 25 nmi of the main Hawaiian Islands in 1993-1998 (Mobley et al., 2000), but no sightings of sei whales were made. A 2002 shipboard line-transect survey of the entire Hawaiian Islands EEZ resulted in a summer/fall abundance estimate of 77 (CV = 1.06) sei whales (Barlow, 2003). This is currently the best available abundance estimate for this stock, but the majority of sei whales would be expected to be at higher latitudes in their feeding grounds at this time of year. The PBR level for this Hawai'i stock is calculated as the minimum population size (37) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a F_R of 0.1 (the default value for an endangered species; Wade and Angliss, 1997), resulting in a PBR of 0.1 sei whales per year.

6.4.5 Impacts of Human Activity on Sei Whales

6.4.5.1 Whaling

From 1910-1975, approximately 74,215 sei whales were caught in the entire North Pacific Ocean (Horwood, 1987; Perry et al., 1999). From the early 1900s, Japanese whaling operations consisted of a large proportion of sei whales, with 300-600 sei whales killed per year from 1911-1955. The sei whale catch peaked in 1959, when 1,340 sei whales were killed. In 1971, after a decade of high sei whale catch numbers, sei whales were scarce in Japanese waters. In the eastern North Pacific, the sei whale population appeared to number about 40,000 animals until whaling began in 1963. By 1974, the sei whale population had been reduced to about 8,000 animals (Tilman, 1977). No recent reports indicate sei whales are being killed or seriously injured as a result of fishing activities in any eastern North Pacific fishery (Perry et al., 1999).

From 1958-1965, 384 were taken by shore-based whaling stations in central California (Rice, 1974), an additional 26 were taken off central and northern California from 1919-1926 (Clapham et al., 1997). There has been an IWC prohibition on taking sei whales since 1976, and commercial whaling in the U.S. has been prohibited since 1972. The offshore drift gillnet fishery is the only fishery that is likely to take sei whales from this stock, but no fishery mortalities or serious injuries have been observed. After the 1997 implementation of a Take Reduction Plan,

which included skipper education workshops and required the use of pingers and minimum six-fathom extenders, overall cetacean entanglement rates in the drift gillnet fishery dropped considerably (Barlow and Cameron, 2003). Mean annual takes for this fishery are based on 1997-2001 data. This results in an average estimate of zero sei whales taken annually. However, some gillnet mortality of large whales may go unobserved because whales swim away with a portion of the net.

6.4.5.2 Fisheries

There have been no reported entanglements or other interactions between sei whales and commercial fishing activities. Between 1994 and 2002, no interactions with sei whales were observed in the Hawai'i-based longline fishery, with approximately 4-25% of all effort observed (Forney, 2004). Ship strikes may occasionally kill sei whales as they have been shown to kill their larger relatives: blue and fin whales. However, no ship strikes have been reported for this species in this area. During 1997-2001, there were four injuries and two mortalities of unidentified large whales attributed to ship strikes.

6.4.5.3 Distribution of Sei Whales in the Action Area

Sei whales are listed by the IWC with a worldwide distribution from subtropical or tropical waters to high latitudes of the sub-Arctic and sub-Antarctic. An estimated 63,100 sei whales are recorded for all of the Southern Oceans by the IWC. The winter breeding grounds remain unknown with no conclusive data, but there is evidence that this species is restricted to more temperate waters (Perry et al., 1999). These whales are found in deeper waters associated with the continental shelf. Young (2000) listed sei whales among the 12 common cetacean species of the western Central Pacific, and the IWC describes the distribution of sei whales as worldwide from subtropical or tropical waters to high latitudes of the sub-Arctic and sub-Antarctic.

Sei whales were observed 31 times during 20 years of the IWC/IDCR-SOWER Antarctic summer sightings surveys (Branch and Butterworth, 2001). Seven of those sightings occurred in Area VI during the three summers that Area VI was surveyed. Butterworth et al. (1995) calculated an uncorrected density estimate of 0.268/1000 nmi of survey effort in Antarctic Area VI (south of 60°S) for one of the IWC/IDCR summer sighting surveys. During the 1965-66 to 1987-88 summer whaling seasons, Japanese scouting vessels reported sighting 532 sei whales in Area VI (120°W to 170°W) between the latitudes of 50°S and 40°S during 14,695 nmi of survey effort, and no sei whales between the latitudes of 40°S and 30°S during 122 nmi of survey effort (Butterworth et al., 1994).

The most recent NMFS/PIFSC survey on the NOAA ship *Oscar Elton Sette* (March 5-28, 2006) in the waters of American Samoa, the central equatorial Pacific, Johnston Atoll, and surrounding areas did not include sightings of sei whales. Therefore, it seems that the best scientific data available include inconclusive reports of the species' distribution with no reliable estimates for the WPCO. However, without unequivocal evidence that sei whales are not in the action area, it is reasonable to assume that sei whales would be found in the action area.

6.5 Sperm Whales

6.5.1 Species Description and Distribution

Sperm whales are distributed in all of the world's oceans, and are common along the equator especially in the Pacific. Their distribution is linked to their social structure, with mixed groups of adult females and juvenile animals of both sexes generally occurring in tropical and subtropical waters, whereas adult males are commonly found alone or in same-sex aggregations, often occurring in higher latitudes outside the breeding season (Best, 1979; Watkins and Moore, 1982; Arnborn and Whitehead, 1989; Whitehead and Waters, 1990). Mean group sizes are 20–30 animals (Whitehead 2003), and typical social unit sizes range from 3 to 24 (Christal et al., 1998). Mature, female, and immature sperm whales of both sexes are found in more temperate and tropical waters from the equator to around 45°N throughout the year. These groups of adult females and immature sperm whales are rarely found at latitudes higher than 50°N and 50°S (Reeves and Whitehead, 1987). Sexually mature males join these groups throughout the winter. Mature male sperm whales migrate to warmer waters to breed when they are in their late twenties (Best, 1979). They spend periods of at least two months on the breeding grounds, moving between mixed schools and spending only hours with each group (Whitehead, 1993, 2003).

Sperm whales are generally distributed over large areas that have high secondary productivity and steep underwater topography (Jaquet and Whitehead, 1996), and their distribution and relative abundance can vary in response to prey availability (Jaquet and Gendron, 2002). They routinely dive to depths of hundreds of meters, and may occasionally dive as deep as 3,000 m (Rice, 1989). Presumed feeding events have been shown to occur at depths greater than 1,200 m (Wahlberg, 2002). Sperm whales are capable of remaining submerged for longer than two hours, but most dives probably last a half hour or less (Rice, 1989). In the Galápagos Islands, sperm whales typically forage at depths of approximately 400 m, where they feed on squid (Papastavrou et al., 1989; Whitehead, 1989; Smith and Whitehead, 2000). Whales typically dove for approximately 40 min and then spent 10 minutes (min) at the surface (Papastavrou et al., 1989). Sperm whales are rarely found in waters less than 300 m in depth. They are often concentrated around oceanic islands in areas of upwelling, and along the outer continental shelf and mid-ocean waters.

Sperm whales are widely distributed across the entire North Pacific and into the southern Bering Sea in summer with the majority thought to be south of 40°N in winter (Rice, 1974; Goshō et al., 1984; Miyashita et al., 1996). During the summer, mature male sperm whales are thought to move north into the Aleutian Islands, Gulf of Alaska, and the Bering Sea. Because they inhabit deeper pelagic waters, these whales generally remain offshore in the eastern Aleutian Islands and Gulf of Alaska. Several authors have recommended three or more populations of sperm whales in the North Pacific for management purposes (Kasuya, 1991; Bannister and Mitchell, 1980). However, the IWC's Scientific Committee designated two sperm whale populations in the North Pacific: a western and eastern population (Donovan, 1991). The line separating these populations has been debated since their acceptance by the IWC's Scientific Committee. For population assessment purposes, NMFS recognizes three discrete population centers of sperm whales in the Pacific: (1) Alaska, (2) California/ Oregon/Washington, and (3) Hawai'i.

Summer/fall surveys in the eastern tropical Pacific (Wade and Gerrodette, 1993) show that although sperm whales are widely distributed in the tropics, their relative abundance tapers off markedly westward towards the middle of the tropical Pacific (near the IWC stock boundary at 150°W) and tapers off northward towards the tip of Baja California. The Hawaiian Islands marked the center of a major nineteenth century whaling ground for sperm whales (Gilmore, 1959; Townsend, 1935). Sperm whales have also been sighted around several of the Northwestern Hawaiian Islands (Rice, 1960; Barlow, 2003), off the main island of Hawai`i (Lee, 1993; Mobley et al., 2000) in the Kaua`i Channel and in the Alenuihaha Channel between Maui and the island of Hawai`i (Shallenberger, 1981). In addition, the sounds of sperm whales have been recorded throughout the year off O`ahu (Thompson and Friedl, 1982). A summer/fall 2002 shipboard survey of waters within the U.S. EEZ of the Hawaiian Islands resulted in 43 sperm whale sightings throughout the study area (Barlow, 2003).

6.5.2 Life History

Sperm whales are the largest of the toothed whales, with an extensive worldwide distribution (Rice, 1989). Female sperm whales become sexually mature at about nine years of age (Kasuya, 1991). Male sperm whales take 9 - 20 years to become sexually mature, but will require another 10 years to become large enough to successfully compete for breeding rights (Kasuya, 1991). Adult females give birth after about 15 months gestation and nurse their calves for 2 - 3 years. The calving interval is estimated to be about four to six years (Kasuya, 1991). The age distribution of the sperm whale population is unknown, but sperm whales are believed to live at least 60 years (Rice, 1978). Estimated annual mortality rates of sperm whales are thought to vary by age, but previous estimates of mortality rate for juveniles and adults are now considered unreliable (IWC, 1980). Potential sources of natural mortality in sperm whales include killer whales (Rice, 1989; Whitehead, 1995) and papilloma virus (Lambertson et al., 1987).

6.5.3 Listing Status

Sperm whales have been protected from commercial harvest by the IWC since 1981, although the Japanese continued to harvest sperm whales in the North Pacific until 1988 (Reeves and Whitehead, 1997). Sperm whales were listed as endangered under the ESA in 1973. On a worldwide basis, sperm whales are abundant and not biologically endangered. They are also protected under CITES as an Appendix I species, and sperm whales are automatically protected under the MMPA. Critical habitat has not been designated for sperm whales.

6.5.4 Population Status and Trends

Barlow and Taylor (2001) estimated 1,407 (CV = 0.39) sperm whales along the coasts of California, Oregon, and Washington during summer/fall based on ship line transect surveys in 1993 and 1996. Forney et al. (1995) estimated 892 (CV = 0.99) sperm whales off California during winter/spring based on aerial line-transect surveys in 1991-92, but this estimate does not adjust for diving whales that were missed and is now more than eight years out of date. The most recent abundance estimate is based on summer/autumn shipboard surveys conducted within 300 nmi of the coasts of California, Oregon, and Washington in 1996 (Barlow, 1997) and 2001 (Barlow, 2003). The combined weighted estimate for the 1996 and 2001 surveys is 1,233 (CV = 0.41) sperm whales (Barlow, 2003). Green et al. (1992) report that sperm whales were the third most abundant large whale (after gray and humpback whales) in aerial surveys off Oregon and Washington, but they did not estimate population size for that area.

Recently, a combined visual and acoustic line-transect survey conducted in the eastern temperate North Pacific in spring 1997 resulted in estimates of 24,000 (CV = 0.46) sperm whales based on visual sightings, and 39,200 (CV = 0.60) based on acoustic detections and visual group size estimates (Barlow and Taylor, 1998). The PBR level for the California portion of this stock is calculated as the minimum population size (885) times one half the default maximum net growth rate for cetaceans ($\frac{1}{2}$ of 4%) times a F_R of 0.1 (the default value for an endangered species), resulting in a PBR of 1.8.

In the eastern tropical Pacific, the abundance of sperm whales has been estimated as 22,700 (95% C.I. = 14,800-34,600; Wade and Gerrodette, 1993). However, it is not known whether any or all of these animals routinely enter the U.S. EEZ of the Hawaiian Islands. As part of the Marine Mammal Research Program of the ATOC study, a total of 12 aerial surveys were conducted within about 25 nmi of the main Hawaiian Islands in 1993, 1995 and 1998. An average abundance estimate of 66 (CV = 0.56) sperm whales was calculated from the combined survey data (Mobley et al., 2000). This study underestimated the total number of sperm whales within the U.S. EEZ off Hawai'i, because areas around the Northwestern Hawaiian Islands (NWHI) and beyond 25 nmi from the main islands were not surveyed. Furthermore, this species is known to spend a large proportion of time diving, causing additional downward bias in the abundance estimate. The data on which this estimate was based are now over six years old. A 2002 shipboard line-transect survey of the entire Hawaiian Islands EEZ resulted in an abundance estimate of 7,082 (CV = 0.30) sperm whales (Barlow, 2003), including a correction factor for missed diving animals. This is currently the best available abundance estimate for this stock. The PBR level for this stock is calculated as the minimum population size (5,531) times one half the default maximum net growth rate for cetaceans ($\frac{1}{2}$ of 4%) times a F_R of 0.1 (the default value for an endangered species; Wade and Angliss, 1997), resulting in a PBR of 11 sperm whales per year.

Although Kato and Miyashita (1998) believe their estimate to be upwardly biased, preliminary analysis indicates 102,112 (CV = 0.155) sperm whales in the Western North Pacific. The number of sperm whales of the North Pacific occurring within Alaska waters is unknown. As the data used in estimating the abundance of sperm whales in the entire North Pacific are well over six years old at this time and there are no available estimates for numbers of sperm whales in Alaska waters, a reliable estimate of abundance for the North Pacific stock is not available. The F_R for this stock is 0.1, the value for cetacean stocks which are classified as endangered (Wade and Angliss, 1997). However, because a reliable estimate of minimum abundance N_{MIN} is currently not available, the PBR for this stock is unknown.

6.5.5 Impacts of Human Activity on Sperm Whales

6.5.5.1 Whaling

Between 1800 and 1909, about 60,842 sperm whales were estimated taken in the North Pacific (Best, 1976). Thirteen sperm whales were taken by shore whaling stations in California between 1919 and 1926 (Clapham et al., 1997). The reported take of North Pacific sperm whales by commercial whalers between 1947 and 1987 totaled 258,000 (C. Allison, personal communication). Ohsumi (1980) lists an additional 28,198 sperm whales taken mainly in coastal whaling operations from 1910 - 1946. Based on the massive under-reporting of Soviet catches,

Brownell et al. (1998) estimate that about 89,000 whales were additionally taken by the Soviet pelagic whaling fleet between 1949 and 1979. The Japanese coastal operations also under-reported catches by an unknown amount (Kasuya, 1998). Thus, a total of at least 436,000 sperm whales were taken between 1800 and the end of commercial whaling for this species in 1987. Of this grand total, an estimated 33,842 were taken by Soviet and Japanese pelagic whaling operations in the eastern North Pacific from the longitude of Hawai`i to the U.S. West coast, between 1961 and 1976 (Allen, 1980, IWC statistical Areas II and III), and 965 were reported taken in land-based U.S. West coast whaling operations between 1947 and 1971 (Ohsumi, 1980). There has been a prohibition on taking sperm whales in the North Pacific since 1988, but large scale pelagic whaling stopped earlier, in 1980.

In 2000, the Japanese Whaling Association announced that it proposed to kill 10 sperm whales in the Pacific Ocean for research purposes, which would be the first time sperm whales would be taken since the international ban on commercial whaling took effect in 1987. Despite protests from the U.S. government and members of the IWC, the Japanese government plans to conduct this research. The implications of this action for the status and trend of sperm whales are uncertain.

6.5.5.2 Fisheries

In U.S. waters in the Pacific, sperm whales are known to have been incidentally taken only in drift gillnet operations, which killed or seriously injured an average of nine sperm whales per year from 1991 (Barlow et al., 1997). Of the eight sperm whales observed taken by the California/Oregon drift gillnet fishery, three were released alive and uninjured (37.5%), one was released injured (12.5%), and four were killed (50%) (NMFS, 2000). Therefore, approximately 63 % 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 4 out of the last 10 years (NMFS, 2000). During the three years the Pacific Coast Take Reduction Plan has been in place, a sperm whale was observed taken only once (in a set that did not comply with the Take Reduction Plan; NMFS, 2000).

Interactions between longline fisheries and sperm whales in the Gulf of Alaska have been reported over the past decade (Rice, 1989; Hill and DeMaster, 1999). Observers aboard Alaskan sablefish and halibut longline vessels have documented sperm whales feeding on longline-caught fish in the Gulf of Alaska (Hill and Mitchell 1998) and in the South Atlantic (Ashford and Martin, 1996). During 1997, the first entanglement of a sperm whale in Alaska's longline fishery was recorded, although the animal was not seriously injured (Hill and DeMaster, 1998). The available evidence does not indicate sperm whales are being killed or seriously injured as a result of these interactions, although the nature and extent of interactions between sperm whales and long-line gear is not yet clear. Ashford and Martin (1996) suggested that sperm whales pluck, rather than bite, the fish from the long-line.

Between 1994 and 2002, one sperm whale was observed entangled within the Hawaiian Islands EEZ in the Hawai`i-based longline fishery, with approximately 4-25% of all effort observed (Forney, 2004). During the 905 observed trips with 11,014 sets, the average interaction rate of sperm whales was one animal per 905 fishing trips, or one animal per 11,014 sets. The caught

animal was apparently able to free itself and was not considered seriously injured (Forney, 2004).

6.5.6 Distribution of Sperm Whales in the Action Area

Sperm whales are found in all ocean basins from the equator to Polar Regions (Perry et al., 1999). The Southern Hemisphere is recognized as one biogeographical area by the IWC, and this circumpolar region is divided into nine sperm whale “Divisions” (Donovan, 1991). In the South Pacific, males range into the Antarctic (65–70°S) in the summer, whereas mature females and immature whales are rarely found in the latitudes higher than 50°N and 50°S. In the Southern Hemisphere, mating occurs from July to March, with a peak from September to December, and most calves are born between November and March (Rice, 1989). Sperm whales migrate closer to equatorial waters in both hemispheres (Perry et al., 1999). Both solitary males and mixed groups of sperm whales are likely to occur in the action area during the breeding season.

Recent sightings have occurred in French Polynesia and the Cook Islands (South Pacific Whale Research Consortium (SPWRC), 2004). One sighting of a solitary sperm whale was made in 3500-m deep water between the Windward (Tahiti, Moorea, Maiao) and Leeward (Bora Bora, Maupiti, Tahaa, Huahine, Raiatea) Islands during > 550 km of offshore (water depths > 3000 m) survey effort during three years of spring and fall dedicated cetacean surveys off the Society Islands (Gannier, 2000a). Sperm whales were not sited during > 4600 km of inshore (≤ 10 km from shore) survey effort during that study. Gannier (2000) also reported encountering a group of 16 - 20 sperm whales in offshore waters of the Windward Islands. No sperm whales were seen during dedicated cetacean surveys in November - January 1999 off the Marquesas Islands during > 500 km of offshore (water depths > 2000 m) survey effort or during > 1000 km of inshore survey effort (Gannier, 2002a). Sperm whales were also not detected acoustically during 501 listening stations in that survey.

Sperm whale distribution is linked to their social structure. Mixed groups of adult females and juvenile animals of both sexes generally occur in tropical and subtropical waters, whereas adult males are commonly found alone or in same-sex aggregations, often occurring in higher latitudes outside the breeding season (Best, 1979; Watkins and Moore, 1982; Arnbom and Whitehead, 1989; Whitehead and Waters, 1990). Females and immature whales (juveniles and calves) remain above 50°S in warmer waters while males remain in high latitudes along Antarctic from December to March (Gosho, 1984); Mean group sizes are 20–30 animals (Whitehead, 2003), and typical social unit sizes range from 3-24 (Christal et al., 1998). Mature male sperm whales migrate to warmer waters to breed when they are in their late twenties (Best, 1979). They spend periods of at least months on the breeding grounds, moving between mixed schools and spending only hours with each group (Whitehead, 1993, 2003). In the Southern Hemisphere, mating occurs from July to March, with a peak from September to December, and most calves are born between November and March (Rice, 1989). In the South Pacific, males range into the Antarctic (65–70°S) in the summer, whereas females are rarely seen south of 40°S.

Sperm whales were sighted 804 times during 20 years (1978–79 to 1997–98) of the IWC/IDCR-SOWER summer sighting surveys in the Antarctic (Branch and Butterworth, 2001). Fifty-three of those sightings occurred in Antarctic Area VI (120–170°W and 60°S) during the three summers of surveys in that region. Population estimates from those surveys ranged from 5,400 to

10,000 for the entire Antarctic (Branch and Butterworth, 2001). Butterworth et al. (1995) calculated an uncorrected density estimate of 0.545/1000 nmi of survey effort in Antarctic Area VI (south of 60°S) for one of the IWC/IDCR summer sighting surveys. Those estimates did not consider animals missed because they were not at the surface when the survey vessel passed and are, therefore, biased downward.

Data from the most recent NMFS/PIFSC survey on the NOAA ship *Oscar Elton Sette* (March 5-28, 2006) in the waters of American Samoa, the central equatorial Pacific, Johnston Atoll, and surrounding areas included two sightings of sperm whales. Therefore, it seems that the best scientific data available do not include reliable estimates for the WCPO. However, it is reasonable to assume that sperm whales would be found in the action area.

6.6 Sea Turtles

Five species of listed sea turtles occur in the Action Area and may be exposed to interactions with the U.S. WCPO purse seine fishery. These species were listed under the ESA in the mid to late 1970s when very little was known about their population structure, migrations, or distribution. Turtle species considered in the Opinion: green, hawksbill, loggerhead, olive ridley, and leatherback, were each listed as a single entity throughout their entire range. Since the time of listing, it is recognized that populations in the Pacific Ocean are distinct from populations in the Atlantic Ocean, physically separated by continental land masses. Separate recovery plans were prepared for Atlantic and Pacific populations of each species, yet the species remain listed as a global entity (with the exception of olive ridley and green turtles for which certain nesting populations have been listed as endangered with the remainder of the species' nesting populations being listed as threatened).

This Opinion focuses first on the effects of the U.S. WCPO purse seine fishery on sea turtle populations in the Pacific Ocean as distinct from their listed distributions. Sea turtle populations in the Pacific are biologically significant, whereby the loss of populations from the Pacific would result in a significant gap in the distribution of each turtle species. Finally, the loss of these turtle species in the Pacific Ocean would dramatically reduce the distribution and abundance of these species and would, by itself, appreciably reduce the entire species' likelihood of surviving and recovering in the wild. Conversely, if effects from the proposed action are deemed not likely to reduce appreciably, the survival and recovery of Pacific sea turtle populations' in the wild, there would be no logical connection to state that the continued existence of the entire species would be jeopardized by the proposed action.

Defining the geographic range of a population of sea turtles in the Pacific Ocean is difficult. Sea turtles are highly migratory, and the life histories of all species exhibit complex movements and migrations through geographically disparate habitats. Until recently, virtually all of what was known about sea turtle biology and ecology was based on the accessible component of the population, nesting females and eggs. Discoveries using mitochondrial DNA and remote sensing technologies have begun to reveal answers to questions about sea turtle population structure and distribution. The level of understanding about the different species and populations varies considerably depending on the extent to which they have been researched and the size of the geographic range they occupy.

In this section of the Opinion, we describe the status and trend of each species. We use the status and trend as a baseline to determine the resiliency of each population to additive effects likely to occur as a result of the proposed action. Sea turtle populations in the Pacific are comprised of multiple, discrete nesting subpopulations. Nesting sea turtles exhibit natal philopatry and will not likely recolonize nesting populations that become extirpated. Therefore, as a first step in the analysis, we attempt to determine the resiliency of individual populations and assess the impacts of the proposed action at the population level and then expand those determinations to the species' level.

Given what is known about sea turtle life histories, we recognize that the most instructive approach for assessing impacts of the proposed action is at the subpopulation level, however, we have a very limited understanding of nesting population structure and abundance for most species in the Pacific. For other species, genetic information and/or long time series of nester abundance are available for only a few nesting populations. Moreover, there is currently no genetic or tag information available from turtles that have interacted with the fishery which would give us a clue as to their population origin. This lack of information limits the resolution of our analysis as do limitations in our understanding of the species status and trend for most populations. The following subsections focus on sea turtle populations likely to be affected by the proposed action in the WCPO given the best available information on the species.

6.7 Sea Turtle Life History

All sea turtles share a similar life history. Hatchlings of all species have an oceanic maturation phase of poorly known duration (again the level of information varies by population). This phase was once termed the "lost years" as post-hatchlings were not seen by researchers in the marine environment. Today scientists know that these years, which may endure for a decade in some species, are spent in the ocean adrift in currents where the young turtles feed mainly along convergence zones which concentrate food and provide cover from predators (Gulko and Eckert, 2003). During the oceanic phase, turtles grow into the juvenile phase which is commonly encountered in nearshore (or neritic) foraging habitats. Turtles reside in these coastal habitats until they reach sexual maturity which may take an additional 10 to 30 years depending on the species. When the turtles reach sexual maturity as adults, they undertake long migrations, sometimes crossing entire ocean basins, back to the region of their natal nesting beach. After the breeding season, adult turtles return to their coastal foraging grounds where they remain for approximately two to five years before returning to mate and nest again at their natal rookery.

Sea turtles are iteroparous and highly fecund. Turtles deposit an average of 6 to 12 clutches each nesting season and lay many eggs per clutch. Sea turtles lay an unusually high number of eggs for a reptile presumably to offset high and unpredictable hatchling and juvenile mortality (Van Buskirk and Crowder, 1994). This is a broad overview for all species; each of these parameters varies by some extent by species and region.

6.8 Green Turtles

6.8.1 General Distribution

Green turtles are found throughout the world, occurring primarily in tropical, and to a lesser extent, subtropical waters. The species occurs in five major regions: the Pacific Ocean, Atlantic

Ocean, Indian Ocean, Caribbean Sea, and Mediterranean Sea. These regions can be further divided into nesting aggregations within the eastern, central, and western Pacific Ocean; the western, northern, and eastern Indian Ocean; Mediterranean Sea; and eastern, southern, and western Atlantic Ocean, including the Caribbean Sea. Green turtles are highly migratory and undertake complex migrations through geographically disparate habitats. Nesting occurs in more than 80 countries worldwide (Hirth, 1997). Their movements within the marine environment are less understood but it is believed that green turtles inhabit coastal waters of over 140 countries (Groombridge and Luxmoore, 1989).

6.8.2 Global Status

Green turtles were listed as threatened throughout their range under the ESA in 1978, except for breeding populations in Florida and the Pacific coast of Mexico, which were listed as endangered. The results of a precautionary analysis of 32 index nesting sites, indicate that globally, the number of nesting green turtles has declined by 48% to 67% over the last three generations (approximately 150 yr)(Seminoff, 2004). Causes for this decline include harvest of eggs, subadults and adults; incidental capture by fisheries; loss of habitat; and disease. The degree of population change varies among index nesting beaches or among regions. Some nesting populations are stable or increasing due to effective conservation efforts and some populations remain to be threatened with extinction (Seminoff, 2004; Broderick et al., 2006). The global green turtle population is estimated to be in excess of 2.2 million individuals (Broderick et al., 2006) some populations appear to have recovered beyond the carrying capacity of their habitat, while others are at low numbers of abundance and decreasing, facing imminent extirpation unless effective conservation strategies can be identified and implemented. Therefore, the extinction risk of green turtle populations should be assessed on an individual subpopulation basis first (Seminoff, 2004a and Broderick et al., 2006) and then put back in the context of the species as listed.

6.8.3 Biological Characteristics

6.8.3.1 Diet

Although most green turtles appear to have a nearly exclusive herbivorous diet, consisting primarily of sea grass and algae (Wetherall et al., 1993; Hirth, 1997), those along some areas of the east Pacific coast seem to have a more carnivorous diet. Analysis of stomach contents of green turtles found off Peru revealed a large percentage of molluscs and polychaetes, while fish and fish eggs, and jellyfish and commensal amphipods comprised a lesser percentage (Bjorndal, 1997). In the Hawaiian Islands, green turtles are site-specific and consistently feed in the same areas on preferred substrates, which vary by location and between islands (*in* Landsberg et al., 1999).

6.8.3.2 Diving Behavior

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

6.8.3.3 Life History/ Reproduction

Compared to all other sea turtles, green turtles exhibit particularly slow growth rate, and age to maturity appears to be the longest. Based on age-specific growth rates, green turtles are estimated to attain sexual maturity beginning at age 25 to 50 years (Limpus and Chaloupka, 1997, Bjorndal et al., 2000, Balazs and Chaloupka, 2004; Seminoff, 2002, and Zug et al., 2002). Dobbs (2002) estimated the age at first breeding of green turtles in Australia to be 46 years of age. The length of reproductivity has been estimated to range from 17 to 23 years (Carr et al., 1978, Fitzsimmons et al., 1995 *in* Seminoff, 2002).

In Hawaii, green turtles lay up to six clutches of eggs per year (mean of 3.7), and clutches consist of about 100 eggs each. Females migrate to breed only once every two or possibly many more years. On the Hawaiian Archipelago, females nest every 3 to 4 years (Balazs and Chaloupka, 2004). Eastern Pacific green turtles have reported nesting between two and six times during a season, laying a mean of between 65 and 86 eggs per clutch, depending on the area studied (Michoacán, Mexico and Playa Naranjo, Costa Rica) (*in* Eckert, 1993 and NMFS and USFWS, 1998a). Mean observed and estimated clutch frequency for green turtles nesting at Colola beach (Michoacan, Mexico) was 2.5 and 3.2, respectively (Alvarado-Diaz et al., 2003). At the Bramble Cay rookery in Queensland, Australia, green turtles laid an estimated 6.2 clutches per season, with an average clutch containing 102.2 eggs. The renesting interval was 12.4 days (Limpus et al., 2001).

6.8.3.4 Migration

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

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

The northernmost reported resident population of green turtles occurs in San Diego Bay, where about 50-60 mature and immature turtles concentrate in the warm water effluent discharged by a

power plant (McDonald, *et al.*, 1994). These turtles appear to have originated from east Pacific nesting beaches and the Revillagigedo Islands (west of Baja California), based on morphology, genetic analyses, and tagging data (*in* NMFS and USFWS, 1998a; P. Dutton, NMFS, personal communication, March, 2002). In order to reach nesting beaches in late fall and winter, sea turtles in this area must depart these feeding areas by late summer, returning to the area again in early spring (Nichols, 2003).

6.8.4 Population Status and Trends

Green turtle nesting sites are distributed all throughout the WCPO and Southeast Asia. Recent studies have defined green turtle management units (or subpopulations) within the region based on mtDNA similarities among rookeries (Moritz *et al.*, 2002). To date, stock assessments have not been conducted using these management units. We have population abundance estimates for only a small number of the extant green turtle subpopulations in the Pacific and all are limited to the number of females estimated to nest annually. Heppell *et al.* (2003) discuss the implications of using nesting female census data to estimate trends for entire sea turtle populations and caution that such trend estimates do not account for males or other life stages in the population. However, census data on other life stages of sea turtle populations are extremely rare and many peer-reviewed studies rely on nesting female census data to estimate population trends and extinction risk; for example, Chaloupka (2004) and Balazs and Chaloupka (2004). Currently, nesting female census data comprise the best available information for sea turtle population status and trends; therefore, we assume that the trend in the number of nesting females is indicative of the trend of the entire subpopulation.

Dispersal and movement patterns of oceanic post-hatchlings and juveniles are unknown for all green turtle subpopulations; yet we know the turtles drift with ocean gyre systems for many years before recruiting to coastal foraging grounds, and may cross entire ocean basins during this phase. A summary compiled from published and unpublished data in Broderick *et al.*, (2006) show several linkages between feeding and nesting areas for green turtles in Southeast Asia and the western Pacific, which indicate extensive migrations across the Action Area. Some turtles foraging in Indonesia, Philippines and Malaysia migrate to breed at the Sulu Sea Turtle Islands. Some individuals comprising the foraging population around New Caledonia, Vanuatu, Fiji, Solomon Islands, PNG, and eastern Indonesia migrate to breed at the Great Barrier Reef rookeries along with turtles that live in Australian waters (Broderick *et al.*, 2006). Some nesting occurs in proximity to these foraging grounds as well. Currently, there are no data from the WCPO purse seine fishery (*i.e.*, genetics or tagging) to indicate which subpopulations of Pacific and Southeast Asia populations are encountered in the fishery. Therefore, we assume that green turtles originating from any of these subpopulations may interact with the fishery.

Table 7, based on Seminoff (2004), provides a summary of green turtle population abundance and trend estimates for populations with available index counts which may be affected by the U.S. WCPO purse seine fishery. In addition to these index sites, smaller nesting populations occur in Guam; the Commonwealth of the Northern Mariana Islands; RMI; American Samoa (*e.g.*, Rose Atoll and Tutuila); the Republic of Palau (*e.g.*, Merir); PNG (*e.g.*, Long Island); Kiribati; Tuvalu; Tokelau; Micronesia (*e.g.*, Elato, Ngulu, and Yap); New Caledonia, islands in the south Pacific (*e.g.*, Solomon Islands; Scilly Atoll, French Polynesia and Fiji); Vietnam; Thailand; and Japan

(e.g., Ogaswara Islands) (Seminoff, 2004 and NMFS and USFWS, 1998a). Most of these non-index sites are believed to be in general decline (Seminoff, 2004).

Table 7. Recent estimates of mean annual number of nesting green turtles at index sites in the Pacific Ocean based on data from Seminoff (2004). In some cases, values derived from linear and exponential functions are provided in Seminoff (2004). In this table we have included numbers derived from the linear function only (please consult Seminoff (2004) for a discussion of the methods and uncertainty regarding the various estimation methods). The range of years used to derive historic abundance estimates varies considerably among sites in Seminoff (2004) depending on the available data. ‘Present’ estimates are current through 2001. [GBR = Great Barrier Reef].

Region	Index Nesting Site	Past	Present	Percent Change	Trend
Eastern Pacific Ocean	Colola, Michoacán, Mexico	19,564	851	-96%	declining
	Galapagos Islands, Ecuador	1,400	1,400	0%	stable
Central Pacific Ocean	French Frigate Shoals, Hawaii, U.S.	378	583	54%	increasing
Western Pacific Ocean	Heron Is., sGBR, Australia	400	573	43%	increasing
	Raine Is., nGBR, Australia	11,538	18,000	56%	increasing
Southeast Asia	Sabah, Malaysia	3,800	3,620	-5%	declining
	Sarawak, Malaysia	13,556	763	-94%	declining
	Berau Islands, Indonesia	40,295	2,015	-95%	declining
	Penisular, Malaysia	4,841	603	-88%	declining
	Philippines	5,929	2,404	-59%	declining
Pacific Overall¹		101,701	30,812	-70%	declining

¹ Sum of past and present abundance estimates from Pacific index sites based on data from Seminoff (2004).

Despite efforts to protect green turtles in all areas of the world, threats to their survival continue. Primary threats include long-term harvest of eggs and adults at nesting beaches and capture of juveniles and adults at feeding areas. Secondary threats include incidental capture in marine fisheries, habitat loss at nesting and foraging areas, and disease (Seminoff, 2004).

6.9 Hawksbill Turtles

6.9.1 General Distribution

The hawksbill occurs in tropical and subtropical seas of the Atlantic, Pacific and Indian Oceans. The species is widely distributed in the Caribbean Sea and western Atlantic Ocean. Hawksbill turtles reside in the waters of at least 82 countries and nest on the beaches of 60 of those countries (Spotila, 2004). The largest populations occur in the Caribbean Sea, the Seychelles, Indonesia, and Australia. Hawksbill population estimates are plagued by a lack of data, however,

Spotila (2004) estimates that the worldwide population probably consists of 60,000 – 78,000 adult female hawksbills. The largest nesting populations appear to be in Australia where 6,000 – 8,000 turtles nest on the Great Barrier Reef. Indonesia hosts about 2,000 nesting hawksbills each year (Spotila, 2004).

Within the U.S., hawksbills are most common in Puerto Rico and its associated islands and in the U.S. Virgin Islands where approximately 210 and 120-200 hawksbills are estimated to nest, respectively. Within the continental U.S., nesting is restricted to the southeast coast of Florida and Florida Keys where one to two nesting hawksbills remain (Spotila, 2004). In the Hawaiian Islands, nesting occurs in the main islands, primarily on several small sand beaches on the Islands of Hawaii and Molokai. Very limited nesting (perhaps by as few as six females) occurs on the island of Maui.

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

6.9.2 Global Status

The hawksbill turtle is listed as endangered under the ESA. Under Appendix I of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), the hawksbill is identified as “most endangered.” Hawksbill populations have declined by over 90% over the past century (Spotila, 2004). This species is likely rapidly approaching extinction primarily due to the harvesting of the species for its meat, eggs, and shell, as well as the destruction of nesting habitat by human occupation and disruption (NMFS, 2001).

6.9.3 Biological Characteristics

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

6.9.3.1 Diet

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

6.9.3.2 Diving Behavior

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

6.9.3.3 Life History/Reproduction

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

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

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

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

6.9.3.4 Migration

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

6.9.4 Population Status and Trends

The hawksbill is a solitary nester, and thus, population trends or estimates are difficult to determine. This section summarizes available status data for hawksbill populations occurring in the vicinity of the U.S. WCPO purse seine fishery. As stated above, the largest nesting concentrations of hawksbills in the Pacific occur on remote oceanic islands off Australia, with smaller nesting populations occurring at remote beaches in the Solomon Islands, PNG, Indonesia, Malaysia and the Philippines (Table 8). Aside from these sites, hawksbill nesting is not known to occur in abundance in the Pacific. A small number of hawksbills are known to nest

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in the Main Hawaiian Islands and Palau and even fewer turtles are believed to nest in the Marshall Islands and possibly American Samoa (NMFS and USFWS, 1998c). In Japan, nesting is very rare and is confined to the southern islands. Hawksbill nesting also occurs in Vietnam and China, although the status in these areas is unknown (Eckert, 1993).

While trend estimates are not available for these populations, they are likely a very small fraction of their historical numbers due to the heavy exploitation of hawksbills. For example, from 1970 to 1986, Indonesia alone exported more than 700,000 juvenile and adult hawksbill turtles (Spotila, 2004).

Currently, the best available information does not allow us to determine the nesting stock origin of hawksbill turtles that may interact with the U.S. WCPO purse seine fishery. We assume that interactions in the fishery may involve hawksbills from any of the populations. Due to the expansive Action Area, we may be able to eliminate the possibility of some rookeries depending on the location of the interaction, however the available data do not include locations of interactions in the U.S. fleet.

Table 8. Number of females estimated to nest annually at sites with major hawksbill colonies in the Pacific.
Data source: Spotila, 2004.

Location	Number Females Nesting Per Year
GBR, Australia	6,000 - 8,000
Indonesia	800 - 2,000
Malaysia	100 - 500
Palau	20 - 50
Papua New Guinea	< 100
Philippines	100 - 500
Solomon Islands	< 500
Thailand	<100

By far the most serious threat to persistence of hawksbill populations is human consumption of hawksbills and their eggs. Loss of habitat due to expansion of resident human populations and/or increased tourism development is another significant threat to hawksbills. Dramatic reductions in the numbers of nesting and foraging hawksbills have occurred in Micronesia and the Mexican Pacific coast, probably due largely to technological advances in fishing gear, which facilitate legal and illegal harvest. In addition, the hawksbill tortoiseshell trade probably remains an important contributing factor in the decline of the hawksbill. Although the Japanese market was closed in 1994, southeast Asia and Indonesia markets remain lucrative (NMFS and USFWS, 1998b). In addition to the demand for the hawksbill's shell, there is a demand for other products

including leather, oil, perfume, and cosmetics. Prior to being certified under the Pelly Amendment, Japan had been importing about 20 metric tons (approximately 19,000 turtles) of hawksbill shell per year. A negotiated settlement was reached regarding this trade on June 19, 1992. The hawksbill shell commands high prices (currently \$225/kilogram), a major factor preventing effective protection¹⁰.

Most nations have enacted laws to protect hawksbills, however, turtles are still being killed in Indonesia and most eggs are taken in Malaysia (Spotila, 2004). Spotila (2004) notes that while the situation is dire, the future for this species looks better than it has in decades and heralds a continued ban on international trade, more ecotourism, and increased education and enforcement as means by which the decline of hawksbill populations can be reversed.

6.10 Olive Ridley Turtles

6.10.1 General Distribution

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

6.10.2 Global Status

The olive ridley turtle is listed as threatened in the Pacific, except for the Mexican nesting population, which is classified as endangered under the ESA. This latter classification was based on the extensive over-harvesting of olive ridley turtles in Mexico, which caused a severe population decline. Since the ban on the harvest of turtles in Mexico, the primary threat to the Mexican nesting population has been reduced and the population appears to be increasing. Olive ridley turtles are considered the most abundant sea turtle in the world (NMFS and USFWS, 1998e).

In the Atlantic, there has been a decline in abundance of olive ridley turtles since they were listed in 1978. Since 1967, the western North Atlantic (Surinam and adjacent areas) nesting population has declined more than 80%. In general, anthropogenic activities have negatively affected each life stage of the olive ridley turtle populations, resulting in the observed declines in abundance of some olive ridley turtle nesting aggregations. Other aggregations, such as those in the eastern Pacific, have experienced significant increases in abundance in recent years, often as a result of decreased adult and egg harvest pressure, indicating populations in which the birth rates are now exceeding death rates.

¹⁰http://www.nmfs.noaa.gov/prot_res/species/turtles/hawksbill.html

6.10.3 Biological Characteristics

6.10.3.1 Diet

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

6.10.3.2 Dive Behavior

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

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

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

6.10.3.3 Life History/Reproduction

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

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

6.10.3.4 Migration

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

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

In the eastern Pacific Ocean, adult olive ridleys are found in warm, tropical waters, bounded on the north by the California Current and on the south by the Humboldt Current. There are few

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

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

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

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

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

Olive ridley turtles from east and west Pacific stocks have different habitat associations. Western Pacific olive ridley turtles appear to be associated with major ocean currents, such as the southern edge of the KEC, the North Equatorial Current (NEC) and the Equatorial Countercurrent (ECC). Olive ridley turtles from the eastern Pacific were not associated with strong current systems, most of these turtles remained within the center of the Subtropical Gyre.

These waters are warm, vertically stratified with deep thermoclines, and do not have strong surface temperature or chlorophyll gradients. Olive ridley turtles of either nesting aggregation origin were not associated with strong surface chlorophyll fronts. However, olive ridleys from the western Pacific were found in habitat characterized by wind-induced upwelling and shoaling of the thermocline, which may allow olive ridley turtles to forage more shallowly in these areas. Polovina et al. (2004) theorize that these conditions may provide an energetic advantage to turtles migrating across the Pacific to nesting beaches.

6.10.4 Population Status and Trend

As with the other species, the available data do not allow us to determine the nesting beach origin of the olive ridley turtles that have interacted with U.S. WCPO purse seine fishery and the proportion of turtles from various nesting beaches likely to interact with the fishery in the future. Because olive ridley turtles have such extensive migrations and have a highly pelagic, oceanic life history strategy, we assume that olive ridley turtles likely to be exposed to the U.S. WCPO purse seine fishery may originate from any of the following nesting populations.

6.10.4.1 Eastern Pacific Ocean

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

Mexico

The nationwide ban on commercial harvest of sea turtles in Mexico, enacted in 1990, has improved the situation for the olive ridley. Surveys of important olive ridley nesting beaches in Mexico indicate increasing numbers of nesting females in recent years (Marquez et al., 1995; Arenas et al., 2000). Annual nesting at the principal beach, Escobilla Beach, Oaxaca, Mexico, averaged 138,000 nests prior to the ban, and since the ban on harvest in 1990, annual nesting has increased to an average of 525,000 nests (Salazar et al., 1998). At a smaller olive ridley nesting beach in central Mexico, Playon de Mismalayo, nest and egg protection efforts have resulted in more hatchlings, but the population is still “seriously decremented and is threatened with extinction” (Silva-Batiz et al., 1996). There is discussion in Mexico that the species should be considered recovered (Arenas et al., 2000).

Costa Rica

In Costa Rica, 25,000 to 50,000 olive ridleys nest at Playa Nancite and 450,000 to 600,000 turtles nest at Playa Ostional each year (NMFS and USFWS, 1998e). In an 11-year review of the nesting at Playa Ostional, (Ballesteros et al., 2000) report that the data on numbers of nests deposited is too limited for a statistically valid determination of a trend; however, there does appear to be a six-year decrease in the number of nesting turtles. Under a management plan, the community of Ostional is allowed to harvest a portion of eggs. Between 1988 and 1997, the average egg harvest from January to May ranged between 6.7 and 36%, and from June through

December, the average harvest ranged from 5.4 to 20.9% (Ballesterio et al., 2000). At Playa Nancite, concern has been raised about the vulnerability of offshore aggregations of reproductive individuals to “trawlers, longliners, turtle fishermen, collisions with boats, and the rapidly developing tourist industry” (Kalb et al., 1996).

The greatest single cause of olive ridley egg loss comes from the nesting activity of conspecifics on *arribada* beaches, where nesting turtles destroy eggs by inadvertently digging up previously laid nests or causing them to become contaminated by bacteria and other pathogens from rotting nests nearby. At a nesting site in Costa Rica, an estimated 0.2 % of 11.5 million eggs laid during a single *arribada* produced hatchlings (NMFS and USFWS, 1998e). Hatching success at both *arribada* beaches (Playa Ostional and Playa Nancite) is very low. Hatching success rates were estimated to be approximately 8% per year for Playa Ostional (Arauz and Mo, 1994) and as low as 1-4% at Playa Nancite (Cornelius and Robinson, 1985). Low natural hatching success rates were used persuasively to permit a limited, legal egg harvest at Ostional (Cambell, 1998).

Some female olive ridleys nesting in Costa Rica have also been found afflicted with the fibropapilloma disease (Aguirre et al., 1999).

Guatemala

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

Nicaragua

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

Indian Ocean

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

India

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

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

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

and unsuitable beach conditions, including reduction in beach width due to erosion (Pandav and Choudhury, 1999).

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

6.10.4.2 Western Pacific Ocean

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

Indonesia

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

Malaysia

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

Thailand

In Thailand, olive ridleys occur along the southwest coast, on the Surin and Similan islands, and in the Andaman Sea. On Phra Thong Island, on the west coast of Thailand, the number of nesting turtles have declined markedly from 1979 to 1990. During a 1996-97 survey, only six olive ridley nests were recorded, and of these, half were poached, and one was predated by feral dogs. During the 1997-98 survey, only three nests were recorded. The main threats to turtles in

Thailand include egg poaching, harvest and subsequent consumption or trade of adults or their parts (i.e., carapace), indirect capture in fishing gear, and loss of nesting beaches through development (Aureggi et al., 1999).

6.10.4.3 Central Pacific Ocean

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

Trends for the primary nesting beach of olive ridleys in the eastern Pacific are very promising and the conservation efforts that have resulted in the dramatic increases are commendable (Marquez et al., 1996). Probabilities of extinction risks indicate negligible risks over the next several decades given that current conservation practices are continued (Snover, 2005). As with all population of marine turtles, these trends can change quickly with changes in conservation efforts.

6.11 Loggerhead Turtles

6.11.1 General Distribution

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

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

6.11.2 Global Status

The loggerhead turtle was listed as a threatened species throughout its global range on July 28, 1978. It was listed because of direct take, incidental capture in various fisheries, and the alteration and destruction of its nesting habitat. Loggerhead sea turtles inhabit the Atlantic Ocean, Pacific Ocean, Indian Ocean, Caribbean Sea and Mediterranean Sea.

The number of nesting females per year for all primary rookeries in the Atlantic Ocean, combined is slightly lower than 25,000 and at 1,200 nesting females annually in the Pacific (Spotila, 2004). Estimates for all known rookeries in the Indian Ocean and the Mediterranean are 14,930 and 2,510 nesting females, respectively (Spotila, 2004).

NMFS recognizes five subpopulations of loggerhead sea turtles in the western north Atlantic based on genetic studies. There are no detectable nesting trends for the two largest western Atlantic subpopulations: the South Florida subpopulation and the northern subpopulation. Because of its size (estimated at 19,000 nesters per year in Spotila (2004)), the South Florida subpopulation may be critical to the survival of the species in the Atlantic Ocean. In the past, this nesting aggregation was considered second in size only to the nesting aggregation on islands in the Arabian Sea off Oman (Ehrhart, 1989, NMFS and USFWS, 1991). However, the Oman colony is located in an area of the world where it is highly vulnerable to disruptive events such as political upheavals, wars, catastrophic oil spills, and lack of strong protections for sea turtles (Meylan et al., 1995) and the South Florida nesting population has replaced Oman as the largest remaining loggerhead colony (Spotila, 2004).

6.11.3 Biological Characteristics

6.11.3.1 Diet

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

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

6.11.3.2 Diving Behavior

Studies of loggerhead diving behavior indicate varying mean depths and surface intervals, depending on whether they were located in shallow coastal areas (short surface intervals) or in deeper, offshore areas (longer surface intervals). The maximum recorded dive depth for a post-nesting female was 211-233 m, while mean dive depths for both a post-nesting female and a subadult were 9-22 m. Routine dive times for a post-nesting female were between 15 and 30 min, and for a subadult, between 19 and 30 min (Sakamoto et al., 1990 *in* Lutcavage and Lutz, 1997). Two loggerheads tagged by Hawaii-based longline observers in the North Pacific and attached with satellite-linked dive recorders were tracked for about 5 months. Analysis of the dive data indicate that most of the dives were very shallow - 70% of the dives were no deeper than 5 m. In addition, the loggerheads spent approximately 40% of their time in the top meter and nearly all of their time in waters shallower than 100 m. For only 5% of the days, the turtles dove deeper than 100 m; the deepest daily dive recorded was 178 m (Polovina et al., 2003).

A recent study (Polovina et al., 2004) found that tagged turtles spent 40 % of their time at the surface and 90 percent of their time at depths shallower than 40 m. On only five percent of recorded dive days loggerheads dove to depths greater than 100 m at least once. In the areas that the loggerheads were diving, there was a shallow thermocline at 50 m. There were also several strong surface temperature fronts the turtles were associated with, one of 20°C at 28°N and another of 17°C at 32°N.

6.11.3.3 Life History/Reproduction

For loggerheads, the transition from hatchling to young juvenile occurs in the open sea, and evidence from genetic analyses and tracking studies show that this part of the loggerhead life cycle involves trans-Pacific developmental migration. The size structure of loggerheads in coastal and nearshore waters of the eastern and western Pacific suggest that Pacific loggerheads have a pelagic stage similar to the Atlantic. The high seas driftnet fishery, which operated in the CNP in the 1980s and early 1990s, incidentally caught juvenile loggerheads (mostly 40-70 cm in length) (Wetherall, et al., 1993). Large aggregations (numbering in the thousands) of mainly juveniles and subadult loggerheads are found off the southwestern coast of Baja California, over 10,000 km from the nearest significant nesting beaches (Pitman, 1990; Nichols, et al., 2000). Genetic studies (Bowen et al., 1995) and recent satellite tracking information (which can be viewed at www.seaturtle.org) have shown these loggerheads from Japan nesting subpopulations migrate to foraging grounds in Baja Mexico and return to nest in Japan at sexual maturity. Recent studies demonstrate that larger loggerheads forage in the offshore oceanic environment than previously believed (Hawkes et al., 2006). Loggerheads off Australia occur in open ocean pelagic habitat until at least 10 to 15 years of age, or approximately 78 cm in carapace length (Dobbs, 2002).

Based on skeletochronological and mark-recapture studies, mean age at sexual maturity for loggerheads ranges between 25 to 37 years of age, depending on the subpopulation (*in* Chaloupka and Musick, 1997, and Frazer et al., 1994). Upon reaching maturity, adult female loggerheads migrate long distances from resident foraging grounds to their preferred nesting beaches. Clutch size averages 110 to 130 eggs, and one to six clutches of eggs are deposited during the nesting season (Dodd, 1988). The mean number of clutches deposited are 1.1 for females at Miyazaki, Japan, 2.06 for females at Yakushima Island, Japan (Schroeder et al.,

2003), and 3.4 clutches per season estimated for loggerheads in eastern Australia (Limpus and Limpus, 2003). The average re-nesting interval for eastern Australian loggerheads is 14 days (Limpus and Limpus, 2003). The average re-migration interval is between 2.6 and 3.5 years (*in* NMFS and USFWS, 1998d) (average 3.8 years for eastern Australian loggerheads (Limpus and Limpus, 2003)), and adults can breed up to 28 years (Dobbs, 2002). Nesting is preceded by offshore courting, and individuals return faithfully to the same nesting area over many years.

6.11.3.4 Migration

Loggerhead hatchlings on nesting beaches in Japan undertake developmental migrations in the North Pacific, using the Kuroshio and North Pacific Currents. Tagging programs to study migration and movement of sea turtles and genetic analyses provide evidence that loggerhead turtles undergo trans-Pacific migrations and forage off Baja California. Genetic analyses of all loggerheads caught and sampled in the Hawaii-based and the west coast-based longline fishery all originated from the Japan nesting stock (NMFS, 2005). Most loggerheads taken in the Hawaii-based longline fishery are non-adults, suggesting that loggerheads in the Pacific are pelagic until they become sexually mature, returning to nesting beaches (Parker et al., 2003). As adults, loggerheads were believed to be forage in resident coastal environments; however recent studies from the Atlantic indicate a more oceanic distribution for this life stage than previously believed (Hawkes et al., 2006).

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

Loggerheads originating from south Pacific nesting stocks have been documented foraging in the waters off southern Peru and northern Chile. Genetic analyses conducted on three specimens incidentally taken by Peruvian artisanal fisheries confirmed them to be loggerheads originating from Australian nesting stocks (Alfaro-Shigueto, et al., 2004). In eastern Australia, nesting females have been documented migrating to feeding areas spread over a 2,600 kilometer radius throughout eastern and northern Australia, eastern Indonesia, PNG, the Solomon Islands, and New Caledonia (Limpus and Limpus, 2003).

In the north Pacific Ocean, satellite telemetry studies show that loggerhead turtles from Japan beaches tend to follow 17° and 20°C sea surface isotherms north of the Hawaiian Islands (Polovina et al., 2004). Relationships between other turtle species and sea surface temperatures have also been demonstrated, with most species preferring distinct thermal regimes (Stinson, 1984). After capture in the Hawaii-based longline fishery, six satellite transmitter-equipped loggerheads traveled westward along two convergent oceanic fronts, against prevailing currents and associated with a “cool” front characterized by sea surface temperature (17°C), surface chlorophyll and an eastward geostrophic current of about 4 centimeters/second (cm/sec). Three others were associated with a warmer front (20°C), lower chlorophyll levels, and an eastward geostrophic flow of about 7 cm/sec. This study supports a theory that fronts are important juvenile habitat (Polovina et al., 2000).

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

Recent telemetry studies have described the oceanic habitat of loggerheads in more detail. Polovina et al. (2004) tagged 26 loggerheads captured in Hawaii-based longline fishery. All of these turtles came from Japan nesting beaches. Three of the 26 loggerhead turtles tagged may have been sexually mature based on straight carapace lengths, the remainder with immature turtles. These turtles tended to migrate west following interactions. The turtles also shifted seasonally north and south between 28°N and 40°N. During January through June the loggerheads were found in the southern portion of this range, shifting to the northern end during July through December. The turtles also associated with areas with sea surface temperatures (SSTs) between 15° and 25° C. The loggerhead turtles were found in cooler waters during winter and spring, warmer waters in summertime.

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

Principle foraging areas for eastern Australian loggerhead turtles are eastern Australia, the Coral Sea, southern and eastern PNG, and the Gulf of Carpentaria (Schroeder et al., 2003). Foraging areas for eastern Australia loggerheads may be located relatively near the nesting beach or thousands of kilometers distant as with the Japan populations. Loggerhead turtles tagged from eastern Australian beaches were discovered to migrate distances from 11 to 2,620 km, highlighting the fact that female loggerhead turtles may undertake a variety of migratory strategies (Limpus and Reimer, 1992). There are no data available from the U.S. WCPO purse seine fishery to indicate the extent to which loggerheads may interact with the fishery, nor the nesting population from which these turtles may originate. Available satellite tracking data for the Japan population indicate that loggerheads migrating across the Pacific tend to stay north of the Action Area (www.seaturtle.org), however sub-adult and adult loggerhead turtles from Australian rookeries may occur in the Action Area.

6.11.4 Population Status and Trends

In the Pacific Ocean, loggerhead turtles are represented by a northwestern Pacific nesting aggregation (located in Japan) and a smaller southwestern nesting aggregation that occurs in eastern Australia (Great Barrier Reef and Queensland) and New Caledonia (Spotila, 2004). There are no reported loggerhead nesting sites in the eastern or central Pacific Ocean basin.

6.11.4.1 Japan

In Japan, loggerheads nest on beaches across 13 degrees of latitude (24°N to 37°N), from the mainland island of Honshu south to the Yaeyama Islands, which appear to be the southernmost

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

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

Latest estimates of number of nests on almost all of the rookeries were provided by the Sea Turtle Association of Japan (Table 9). Less than 1,000 female loggerheads are estimated to return to Japan beaches per nesting season, (Kamezaki et al., 2003). Numbers in Spotila (2004) estimate the number of annual nesting females to be 700 for all of Japan.

In general, nesting populations in Japan have declined by an estimated 50-90%. Recent genetic analyses on female loggerheads nesting in Japan suggest that this subpopulation” is comprised of genetically distinct nesting aggregations (Hatase et al., 2002) with precise natal homing of individual females. As a result, Hatase et al. (2002) indicate that loss of one of these aggregations would decrease the genetic diversity of Japan loggerheads; recolonization of the site would not be expected on an ecological time scale.

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

Many conservation projects aimed at protecting nesting loggerhead turtles, nests, and hatchlings have been initiated in the past few years due to the high rate of interactions between loggerhead turtles from Japan nesting beaches in domestic fisheries and the precarious state of this population (see NMFS, 2005). Projects are also underway in Baja, Mexico to educate fishers about reducing interactions and increasing survival of foraging loggerheads from Japan nesting beaches. These projects appear to be affective at raising awareness and are likely having beneficial impacts on the species persistence. Conservation projects are described in more detail in NMFS (2005).

Though interactions of loggerhead turtles of Japan nesting beach origin are common in fishery interactions in the north Pacific, satellite tracking data (www.seaturtle.org and <http://www.umigame.org/>) indicate that the trans-Pacific migrations of sub-adult and adult turtles likely occurs north of the Action Area.

Table 9. Total nests observed from 1998-2003 at all nesting beaches in southern Japan. (Source: Sea Turtle Association of Japan).

Year	Loggerhead Nests
1998	2,479
1999	2,255
2000	2,589
2001	3,122
2002	4,035
2003	4,519

6.11.4.2 Australia

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

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

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

6.11.4.3 New Caledonia

Although loggerheads are the most common nesting sea turtle in the Île de Pins area of southern New Caledonia, there is no quantitative information available, and surveys in the late 1990s failed to locate regular nesting. However, anecdotal information from locals indicate that there may be more substantial loggerhead nesting occurring on peripheral small coral cays offshore of

the main island. The annual nesting population in the Île de Pins area may be in the tens or the low hundreds (Limpus and Limpus, 2003).

6.11.4.4 Other Countries

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

6.11.5 Threats

The life history strategy of loggerhead turtles makes them susceptible to many natural and human impacts, including impacts while they are on land and in the ocean, including both the benthic and the pelagic environment. Hurricanes and predators are particularly destructive to sea turtle nests. Sand accretion and rainfall that result from these storms as well as wave action can appreciably reduce hatchling success. Other sources of natural mortality include cold stunning and biotoxin exposure.

Anthropogenic factors that impact hatchlings and adult female turtles on land, or the success of nesting and hatching include: beach erosion, beach armoring and nourishment; artificial lighting; beach cleaning; increased human presence; recreational beach equipment; beach driving; coastal construction and fishing piers; exotic dune and beach vegetation; poaching. An increased human presence at some nesting beaches or close to nesting beaches has led to secondary threats such as the introduction of exotic fire ants, feral hogs, dogs, and an increased presence of native species (*e.g.*, raccoons, armadillos, and opossums), which raid and feed on turtle eggs. Although sea turtle nesting beaches are protected along large expanses of the northwest Atlantic coast, other areas along these coasts have limited or no protection. Sea turtle nesting and hatching success on unprotected high density east Florida nesting beaches from Indian River to Broward County are affected by all of the above threats (NMFS SEFSC, 2001).

Loggerhead turtles are affected by a completely different set of anthropogenic threats in the marine environment. These include oil and gas exploration, coastal development, and transportation; marine pollution; underwater explosions; hopper dredging, offshore artificial lighting; power plant entrainment and/or impingement; entanglement in debris; ingestion of marine debris; marina and dock construction and operation; boat collisions; poaching, and fishery interactions. In the pelagic environment, loggerheads are exposed to a series of longline fisheries. In the coastal environment in waters off the U.S. and Baja, Mexico, loggerheads are exposed to a suite of fisheries in Federal and state waters including trawl, purse seine, hook and line, gillnet, pound net, longline, dredge, and trap fisheries. A study in 2004, estimated that thousands of loggerhead turtles likely die in Pacific fisheries (Lewison et al., 2004).

6.12 Leatherback Turtles

6.12.1 General Distribution

Leatherback turtles are the largest living turtles and range farther than any other sea turtle species. Leatherback turtles are widely distributed throughout the oceans of the world. The species is found in four main regions of the world: the Pacific, Atlantic, and Indian Oceans, and

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

Their large size and tolerance of relatively low temperatures allow them to occur in northern waters such as off Labrador and in the Barents Sea (NMFS and USFWS, 1998e). Adult leatherbacks forage in temperate and subpolar regions from 71° N to 47° S latitude in all oceans and undergo extensive migrations to and from their tropical nesting beaches.

Leatherback turtles lead a completely pelagic existence, foraging widely in temperate waters except during the nesting season, when gravid females return to tropical beaches to lay eggs. Males are rarely observed near nesting areas, and it has been proposed that mating most likely takes place outside of the tropical waters, before females move to their nesting beaches (Eckert and Eckert, 1988). Leatherbacks are highly migratory, exploiting convergence zones and upwelling areas in the open ocean, along continental margins, and in archipelagic waters (Morreale et al., 1994; Eckert, 1998; Eckert, 1999a). In a single year, a leatherback may swim more than 10,000 km (Eckert, 1998). Leatherback turtles use the entire Pacific Ocean, foraging on one side and breeding on the other.

6.12.2 Global Status

The leatherback turtle was listed as endangered throughout its global range on June 2, 1970. In 1980, the leatherback population was estimated at approximately 115,000 adult females globally¹²(Pritchard, 1982). By 1995, the global population of adult females had declined to 34,500¹³ (Spotila et al., 1996). In 2004, the number of nesting female leatherbacks, worldwide was estimated to be 35,860 (Spotila, 2004). That estimate does not account for any other segments of the population except for nesting females. The vast majority of these turtles occur in the Atlantic and Caribbean. In 2006, the first “Status of the World’s Sea Turtles” (SWoT) report was released and documented the location and most recent estimate of the number of nests at each known leatherback nesting beach, providing what the author’s describe as ‘the world’s first glimpse of leatherback nesting’ (Mast et al., 2006). In the SWoT report, leatherback populations in the Pacific are listed as the number one burning issue for global sea turtle conservation (Mast and Pritchard, 2006) as populations have experienced severe declines (upwards of 90% or more) at most of the major rookeries from low levels of population abundance.

¹²). That number, however, is probably an overestimation as it was based on a particularly good nesting year in 1980 (Pritchard, 1996).

¹³ Pritchard (1996) suggested that the population estimates from Spotila et al. (1996) likely under-estimated the actual population size as the data modeled in the time series ended with a particularly bad nesting year (1994) while excluding nesting data from 1995, which was a good nesting year.

The total Atlantic population size is undoubtedly larger than in the Pacific, but overall population trends are unclear. Spotila (2004) estimates the number of nesting females in the eastern and western Atlantic combined to be 23,690 and only 4,910 in the eastern and western Pacific combined. In 1996, the entire western Atlantic population was characterized as stable at best with numbers of nesting females reported to be on the order of 18,800 (Spotila et al., 1996). A subsequent analysis by Spotila (personal communication) indicated that by 2000, the western Atlantic nesting population had decreased to about 15,000 nesting females and a summary by Spotila in 2004, revises that number to only 13,800 total nesting females in the western Atlantic.

According to NMFS' Southeast Fishery Science Center (2001) the nesting aggregation in French Guiana has been declining at about 15% per year since 1987. However, from 1979-1986, the number of nests was increasing at about 15% annually which could mean that the current 15% decline could be part of a nesting cycle which coincides with the erosion cycle of Guiana beaches described by Schultz (1975). In Suriname, leatherback nest numbers have shown large recent increases (with more than 10,000 nests per year since 1999 and a peak of 30,000 nests in 2001), and the long-term trend for the overall Suriname and French Guiana population may show an increase (Girondot, 2002 in Hilberman and Goverse, 2003). The number of nests in Florida and the U.S. Caribbean has been increasing at about 10.3% and 7.5%, respectively, per year since the early 1980s but the magnitude of nesting is much smaller than that along the French Guiana coast (NMFS SEFSC, 2001). Also, because leatherback females can lay 10 nests per season, the recent increases to 400 nests per year in Florida may only represent as few as 40 individual female nesters per year. The increase in nests observed in Florida can be explained by increases in nesting survey effort in recent years, as well as a real increase in documented nests.

In summary, the paucity of information regarding the status of Atlantic leatherbacks makes it difficult to characterize the current status. 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.

6.12.3 Biological Characteristics

6.12.3.1 Diet

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

Surface feeding by leatherbacks has been reported in U.S. waters, especially off the west coast (Eisenberg and Frazier, 1983), but foraging may also occur at depth. Based on offshore studies of diving by adult females nesting on St. Croix, U.S. Virgin Islands, Eckert et al. (1989) proposed

that observed interesting¹⁴ dive behavior reflected nocturnal feeding within the deep scattering layer (strata comprised primarily of vertically migrating zooplankton, chiefly siphonophore and salp colonies, as well as medusae). Hartog (1980, *in* NMFS and USFWS, 1998c) also speculated that foraging may occur at depth, when nematocysts from deep water siphonophores were found in leatherback stomach samples. Davenport (1988, *in* Davenport and Balazs, 1991) speculated that leatherback turtles may locate pyrosomas at night due to their bioluminescence; however direct evidence is lacking.

6.12.3.2 Diving Behavior

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

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

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

6.12.3.3 Life History/Reproduction

Using a small sample size of leatherback sclerotic ossicles, analysis by Zug and Parham (1996) suggested that mean age at sexual maturity for leatherback turtles is around 13 to 14 years, giving them the highest juvenile growth rate of all sea turtle species. Zug and Parham (1996) concluded that for conservation and management purposes, 9 years is a likely minimum age for maturity of leatherback turtles, based on the youngest adult in their sample. The natural longevity

¹⁴Interesting – time spent between laying clutches of eggs during a single nesting season.

of leatherback turtles has not been determined (NMFS and USFWS, 1998c), although there are recorded documentations of post-maturation survival on the order of about 20 years (Pritchard, 1996).

On the Pacific coast of Mexico, female leatherback turtles lay an average of 4 clutches per season, with clutch size averaging 64 yolked eggs per clutch (García and Sarti, 2000) (each clutch contains a complement of yolckless eggs¹⁵, sometimes comprising as much as 50% of total clutch size, a unique phenomenon among leatherback turtles and some hawksbills (Hirth and Ogren, 1987)). Each clutch is laid within a 9.3 day interval (García and Sarti, 2000). In Las Baulas, Costa Rica, the average clutch size is also 64.7 ± 1.4 yolcked eggs. Reproductive output ranged from 4.3 ± 0.2 to 7.9 ± 0.3 clutches per female per nesting season (Reina et al., 2002). Clutch sizes in Terengganu, Malaysia, and in Pacific Australia were larger, averaging around 85-95 yolcked eggs and 83 yolcked eggs, respectively (*in* Eckert, 1993).

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

6.12.3.4 Migration

Migratory routes of leatherback turtles originating from eastern and western Pacific nesting beaches are not entirely known. However, satellite tracking of post-nesting females and genetic analyses of leatherback turtles caught in U.S. Pacific fisheries or stranded on the west coast of the U.S. present some strong insight into at least a portion of their routes and the importance of particular foraging areas. Aerial surveys conducted during the late summer and fall months of 1990-2001 reveal that leatherbacks forage off central California, generally at the end of the summer, when upwelling relaxes and sea surface temperatures increase. Leatherbacks were most often spotted off Point Reyes, south of Point Arena, in the Gulf of the Farallones, and in Monterey Bay. These areas are upwelling “shadows,” regions where larval fish, crabs, and jellyfish are retained in the upper water column during relaxation of upwelling. Researchers estimated an average of 170 leatherbacks (95% CI = 130-222) were present between the coast and roughly the 50 fathom isobath off California. Abundance over the study period was variable between years, ranging from an estimated 20 leatherbacks (1995) to 366 leatherbacks (1990) (Benson et al., 2003).

¹⁵Bell et al. (2003) note that “yolckless eggs” is an incorrect nomenclature, since they do not contain a 1 N nucleous with an associated yolk that together make up a gamete or oöcyte.

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

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

Morreale et al., (1994) demonstrated that satellite tagged, post-nesting leatherback turtles leaving Costa Rica followed precisely defined, long-distance migratory pathways after nesting. Despite differences in dates of departure from the nesting areas, nesting cohorts followed along nearly identical pathways. All 6 leatherback turtles' (from the Pacific and Caribbean coasts of Costa Rica) movements paralleled deepwater bathymetric contours ranging from 200-3,500 m. When a turtle's path intersected an abyssal plain, it veered along the outer slope, and when an abyssal plain was unavoidable, the turtle crossed it at its narrowest point. These studies underscore the importance of this offshore habitat and migratory routes and the likelihood that sea turtles are present on fishing grounds, particularly for large commercial fishing fleets south of the equator (Eckert, 1997). Eckert, (1999a) speculates that leatherback turtles leaving the nesting areas of Mexico and Costa Rica may be resource-stressed by a long reproductive season with limited food and the high energetic requirements brought about by the demands of reproduction, elevated water temperatures, or both. When they leave, their greatest need is to replenish energy stores (e.g. fat) and they must move to areas where food is concentrated (e.g. upwelling areas). Most of these eastern Pacific nesting stocks migrate south, although one genetic sample from a leatherback turtle caught south of the main Hawaiian Islands by the Hawaii-based longline fishery indicated representation from eastern Pacific nesting beaches (P. Dutton, NMFS, personal communication, October 2002).

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

Observations of tracked leatherbacks captured and tagged off the west coast of the United States have revealed an important migratory route from central California, to south of the Hawaiian Islands, leading to western Pacific nesting beaches. In September 2000, researchers captured their first two leatherbacks off Monterey, California. Of two females, one was of a size normally associated with the western Pacific nesting stock, which are, on average, 10-20 cm larger than eastern Pacific nesting stocks (Zug and Parham, 1996). Both headed on a southwest migratory path, appearing to be heading to the western Pacific nesting beaches (Dutton and Eckert, 2005). In 2001, a male and female leatherback were captured and tagged. The male headed north of the migratory route taken by the two females the year before and stopped transmitting on 12/17/01, while the female traveled north to the Farallon Islands and then headed west, where transmissions stopped on 10/11/01 (D. Parker and P. Dutton, NMFS, personal communication, June 2002). Genetic analysis confirmed that all four of these leatherbacks tagged and outfitted with transmitters were from the western Pacific stock (P. Dutton, NMFS, personal communication, October 2002). Since then, eight leatherbacks (six females, two males) were captured in 2002, and six (five females, one male) were captured in 2003. All were outfitted with satellite tags and tracked. Most followed the southwest migratory route, heading towards western Pacific nesting beaches. Two that have been tracked for an extended period of time did not arrive on the nesting beaches, instead heading north and east, back towards the northern part of Hawaii. One leatherback did not follow a southwest track out of Monterey and instead headed southeast, along Baja California, Mexico, and into the Gulf of California. All leatherbacks captured off central California have been found to originate from western Pacific nesting beaches (P. Dutton, NMFS, personal communication, December 2003).

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

6.12.4 Population Status and Trends

Though the stock origin of leatherback turtles susceptible to interactions with the U.S. WCPO purse seine fishery is not known, genetic samples from leatherback turtles taken in the Hawaii-based longline fishery show that 94% of the leatherback turtles sampled (17 of 18 genetic samples) originated from western Pacific nesting beaches and 6% (only 1 of 18 samples) originated from eastern Pacific nesting beaches (NMFS, 2004b; P. Dutton, NMFS, personal communication, April, 2005). New satellite tracking data are revealing insights into the migratory patterns of leatherback turtles. Many leatherback turtles have been equipped with satellite transmitters and their tracks can be viewed on the internet at www.toppcensus.org (accessed August 2, 2006). The available data confirm that nesting leatherbacks tagged on western Pacific nesting beaches migrate across the Pacific toward California and Oregon. Leatherback turtles tagged at eastern Pacific nesting beaches appear to migrate south and are not likely to occur in the vicinity of the U.S. WCPO purse seine fishery with the same frequency and abundance as turtles from western Pacific nesting beaches. Because much remains unknown

about distribution of these turtles over various life stages, we assume that leatherback turtles from the eastern Pacific may occur in the Action Area to some extent, though with a likely lower frequency and abundance than leatherback turtles of western Pacific origin.

Turtles from western nesting beaches could represent individuals from Indonesia (Jamursba-Medi or War-Mon), PNG (Kamiali or other areas of the Huon Gulf), Malaysia (Terrenganu), the Solomon Islands, or Fiji, although satellite tracks from leatherback turtles tagged in PNG suggest that these turtles tend to migrate south instead of north, which would take them away from the action area. Further, the abundance of the nesting aggregations in Indonesia relative to the small size of the other nesting aggregations suggests that the interactions between Indonesian leatherback turtles and the U.S. WCPO purse seine fisheries are most likely.

Leatherback turtles of eastern Pacific origin could represent individuals from nesting aggregations along the coast of Mexico, Costa Rica, or Panama; although turtles from these nesting aggregations may only migrate into the Action Area when oceanic phenomena like El Niño events prevent them from migrating south to the coasts of Peru and Chile. Several investigators who have followed leatherback turtles equipped with satellite tags have reported that leatherback turtles from the beaches of Mexico and Costa Rica migrate through the equatorial current towards the coasts of Peru and Chile (Eckert, 1997; Marquez and Villanueva, 1993; Morreale et al., 1994). These turtles migrate toward the coast of South America where upwelling water masses provide an abundance of prey (Eckert, 1997).

6.12.4.1 Western Pacific

Leatherback turtles originating from the western Pacific are threatened by poaching of eggs, killing of nesting females, human encroachment (development, beach armoring, beachfront lighting, etc.) on nesting beaches, incidental capture in fishing gear, beach erosion, and egg predation by animals. We are only beginning to learn about the status of the western Pacific leatherback nesting populations though once major leatherback nesting assemblages are declining along the coasts of Malaysia and Indonesia, and anecdotal information suggest that population declines have also occurred in PNG, the Solomon Islands, and Vanuatu. Low density and scattered nesting of leatherback turtles occurs in Fiji, Thailand, and Australia (primarily western and to a lesser extent, eastern).

Research has been conducted in the last several years to more thoroughly identify leatherback nesting beaches and estimate numbers of nesting animals in the western Pacific (Papua Indonesia, PNG, Solomon Islands, and Vanuatu). At the Cooperative Workshop sponsored by the Western Pacific Fishery Management Council (Council) from May 17-21, 2004, a total of 25 leatherback nesting sites were identified for the western Pacific region, of which 19 were previously unknown or poorly documented (Dutton et al., in review). Annual nesting among these 25 sites is estimated to be at least 2,000 females. Prior to identification of these additional nesting beaches, Spotila et al. (2000) estimated the number of nesting females in the western Pacific at 1,800. With the inclusion of recently reported nesting sites this estimate has been revised to approximately 2,000 - 5,000 nesting females in the western Pacific. There are still indications of a long term decline in leatherback nesting in the western Pacific. Hitipeuw et al. (in review) note that due to the remoteness and lack of consistent monitoring, the status of most

leatherback populations in the Pacific is unclear. Dutton et al., (in review) highlight the need to conduct beach monitoring and protection work at key nesting sites in the western Pacific.

Malaysia

The decline of leatherback turtles is severe at one of the most significant nesting sites in the western Pacific region - Terengganu, Malaysia, with current nesting representing less than 2 percent of the levels recorded in the 1950s. The nesting population at this location has declined from an estimated 3,103 females nesting in 1968 to 2 nesting females in 1994 (Chan and Liew 1996). Data provided in the SWoT report note that five leatherback nests were laid in 2004, likely representative of one nesting female (Turtle and Marine Ecosystem Center *in* Mast et al., 2006). With one or two females reportedly nesting each year, this population has essentially been eradicated.

Indonesia

The northwest coast of the province of Papua in Indonesia is thought to support the largest remaining leatherback nesting population in the Pacific (Hitipeuw et al., in review). In the state of Papua, leatherback nesting generally takes place on two major beaches: Jamursba-Medi (18 km long) and War-Mon beach (4.5 km long) (Starbird and Suarez, 1994). Approximately 30 km of coastline separates the two nesting sites. Nesting activity was monitored at Jamursba-Medi from 2001 to 2005 and at War-Mon from 2002 to 2005 (Figure 7, Table 10 and Table 11). Approximately 500 to 1,500 females nest annually at Jamursba-Medi (Hitipeuw et al., in review). Although this population has not been monitored consistently, it appears there has been a long term decline since the 1970s. Hitipeuw et al. (in review) reanalyzed previous sporadic records of nesting activity at these beaches from 1981-2001 and found that while there are indications of a long term decline, the Papua, Indonesia population has not yet reached the severely depleted levels evident at other rookeries in the Pacific (Hitipeuw et al., in review).

Using lessons learned from a decade of field activities at Jamursba Medi and technical support from NMFS' Southwest Fishery Science Center and funding support from the Western Pacific Fishery Management Council (Council), WWF-Indo implemented a conservation and monitoring project at War-mon Beach, Papua as part of a larger framework to conserve critically endangered Pacific leatherback turtles in Indonesia. The primary goals of this project are to quantify nesting population dynamics and maximize leatherback hatchling production by reducing predation and human induced impacts at this previously unmonitored and unmanaged leatherback nesting beach. Prior to implementation of this project, egg harvest and predation were considerable threats at War-mon (Irene Kinan, Council, personal communication, July 5, 2005; Starbird and Suarez, 1994; Suarez et al., 2000). As documented by Starbird and Suarez (1994), poaching at unprotected War-mon Beach exceeded 60% and pig predation impacted the remaining 40%. With the establishment of a year-round monitoring project in 2003/04, coastal patrols are currently being conducted to prevent disturbance and exploitation of the beach (Hitipeuw, 2003; Hitipeuw, in review). During the 2003/04 nesting season, a major reduction in impacts was realized. Of the, 2,881 nests laid, only 18% were predated upon and none were poached by humans. These population level benefits continue in 2006.

Population estimates for Papua must be treated with caution given the recent discovery of the large nesting aggregation at War-Mon Beach, Papua. It remains to be determined whether

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Jamursba-Media and War-Mon are two distinct nesting stocks (Dutton et al., in review). Information on leatherback nesting is lacking for a large area of coastline stretching from War-Mon and Jamursaba-Medi to the border with PNG (Dutton et al., in review). Leatherback turtles have been protected since 1978 in Indonesia. Low density nesting also occurs along western Sumatra (200 females nesting annually) and in southeastern Java (50 females nesting annually), although the last known information for these beaches is from the early 1980s (*in* Suarez and Starbird 1996; Dermawan 2002).

The Jamursba-Medi nesting population is probably in long-term decline (Peter Dutton, NMFS, personal communication, August 14, 2006). Though a trend analysis of this nesting beach indicates that it is and has been relatively stable for the past decade; however the numbers of nesting females do not show increasing numbers indicating that they are recovering to historical levels (Snover, 2005). Assuming the Jamursba-Medi nesting population is stable, increases in adult mortality or decreases in recruitment into the adult population (as from poor hatchling production) can cause the nest numbers to decline and the extinction risks presented here to change rapidly (Snover, 2005).

Table 10. Estimated numbers of female leatherback turtles nesting on Jamursba-Medi Beach, along the north coast of the State of Papua (Summarized by Hitipeuw and Maturbongs, 2002 and Hitipeuw, 2003b; Hitipeuw et al. in review; T. Hitipeuw, WWF, personal communication, 2006).

Survey Period	# of Nests	Adjusted # Nests	Estimated # of Females ³
September, 1981	4,000+	7,143 ¹	1,232 - 1,623
April - Oct. 1984	13,360	13,360	2,303 - 3,036
April - Oct. 1985	3,000	3,000	658 - 731
June - Sept. 1993	3,247	4,091 ²	705 - 930
June - Sept. 1994	3,298	4,155 ²	716 - 944
June - Sept. 1995	3,382	4,228 ²	729 - 961
June - Sept., 1996	5,058	6,373 ²	1,099 -- 1,448
May - Aug., 1997	4,001	4,481 ⁴	773 -- 1,018
May - Sept. 1999	2,983	3,251	560 – 739
April - Dec., 2000	2,264	No	390 – 514
March - Oct., 2001	3,056	No	527 – 695
March - Aug., 2002	1,865	1,921	331 – 437
March – Nov., 2003	3,601	2,904	621 – 818
March – Aug., 2004	3,183	3,871	667 – 879
April – Sept., 2005	2,666	2,562	441 - 582

¹The 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.

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

³Based on Bhaskar's tagging data, an average number of nests laid by leatherback turtles on Jamursba-Medi in 1985 was 4.4 nests per female. This is consistent with estimates for the average number of nests by leatherback turtles during a season on beaches in Pacific Mexico, which

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range from 4.4 to 5.8 nests per female. The range of the number of females is estimated using these data.

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

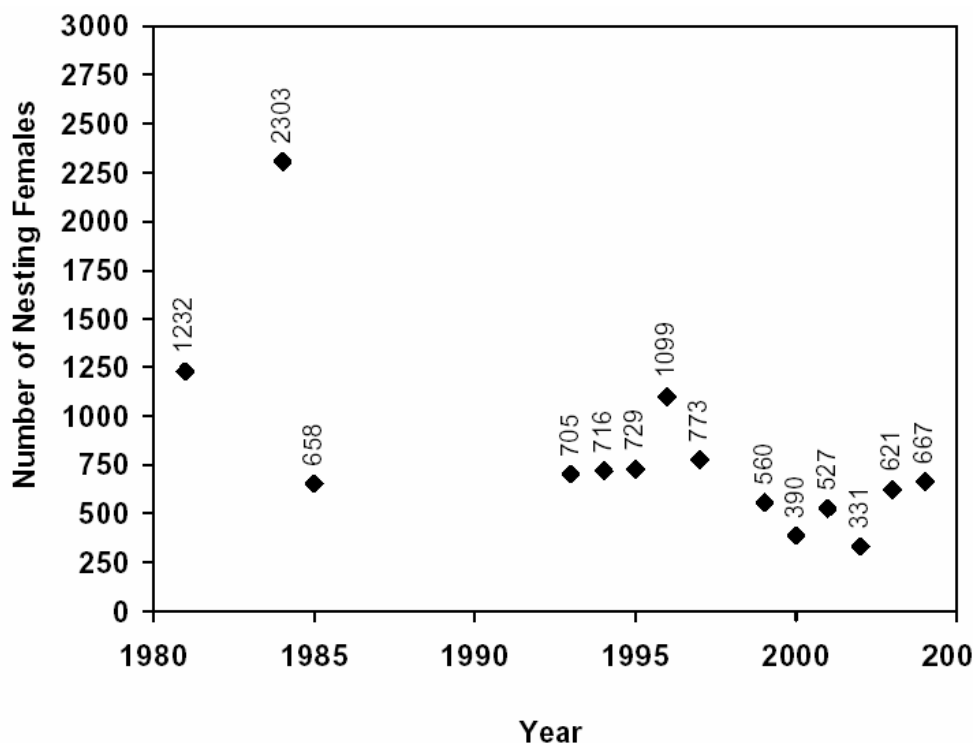


Figure 13. Estimated number of nesting leatherback turtles at Jamursba-Medi, Papua (Hitipeuw et al. in review). These data represent the lower number of nesting females estimated from nest counts. No data were reported for 1998, thus the intermediate value between 1997 and 1999 was interpolated to estimate 1998 nesting abundance. (Figure Source: Snover 2005).

Table 11. Numbers of leatherback nests on Warmon Beach, Papua, Indonesia.

<u>Monitoring Period</u>	<u># nests</u>	<u>Source</u>
Nov. 23-Dec. 20, 1984 and Jan. 1-24, 1985	1,012	Starbird and Suárez, 1994; Suárez <i>et al.</i> , 2000
Dec. 6-22, 1993	406	Starbird and Suárez, 1994; Suárez <i>et al.</i> , 2000
Nov., 2002 - June, 2003	1,442	Hitipeuw, 2003b
Nov., 2003 – Sept., 2004	2,881	Thebu and Hitipeuw, 2005
Oct. 2004 – Sept. 2005	1,980	Hitipeuw, WWF, pers. comm., 2006

Papua New Guinea

The number of leatherback turtles nesting on the north coast of PNG remains unknown but is likely much lower than in War-Mon and Jamursba-Medi, Indonesia (Benson, 2005). In PNG,

leatherbacks nest primarily along the coast of the Huon Gulf in the Morobe Province. The Kamiali nesting beach (located in the Morobe Province and within the Kamiali Wildlife Management Area) is approximately 11 km long and is an important nesting area for leatherbacks. For the periods 2000-2001 and 2003-2004 a total of 41 and 71 nesting females were recorded, respectively (Benson, 2005).

Due to increasing awareness and concern about the local declines in nesting leatherbacks, the Kamiali community agreed to a 100 meter no-take zone in 1999, increased to a 1 km no-take zone in 2000, and 0.5 km was added in 2001 (1.5 km total). The no-take zone is effective from December to February (nesting season). The Council sponsored a community meeting in Kamiali in October, 2003. At this meeting, the Kamiali community maintained this moratorium and expanded it by another 0.5 km (total of 2 km) effectively banning villagers and outsiders from harvesting eggs and meat for the entire 2003/04 nesting season. As of October 2004, the area was expanded to encompass the entire 10km stretch of beach at Kamiali Wildlife Management Area (Karol Kisokau, Kamiali Integrated Conservation Development Group, personal communication, May 19-21, 2004). To date, the Kamiali community implements a community-based nesting beach monitoring program (supported by the Council) and nests laid at Kamiali are conserved *in situ*.

In January 2004 aerial surveys of 2,800 km of coastline in north PNG and New Britain Island were completed. A total of 415 nests were located, of which 71% were found within the Huon Gulf region. Within the Huon Gulf region only 29% of nests were located in areas other than the two nesting beaches of Kamiali and Maus Bang (also known as Baung Buassi). After applying a correction factor based on missed nests identified from beach walk surveys, the total estimate for nest numbers was 559 (Benson, 2005).

Solomon Islands

In the Solomon Islands, the rookery size is estimated to be on the order of 100s of females nesting per year (Dutton et al., in review). Past studies have identified four important nesting beaches in Isabel Province: Sasakolo, Lithoghahira, Lilika, and Katova. Egg harvest by humans has been reported in the past. In addition, lizards and iguanas have been documented preying on leatherback eggs (Rahomia et al., 2001).

Fiji

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

Australia

In Australia, leatherback nesting is sporadic, less than five per year, generally outside of GBR in southeast Queensland. Human related threats include incidental capture in fisheries and ingestion and entanglement in marine debris (Dobbs, 2002).

6.12.4.2 Eastern Pacific

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

Costa Rica

During the 1980s researchers realized that the beaches of Playa Grande, Playa Ventanas and Playa Langosta collectively hosted the largest remaining Pacific leatherback populations in Costa Rica. Since 1988, leatherback turtles have been studied at Playa Grande (in Las Baulas), the fourth largest leatherback nesting colony in the world. During the 1988-89 season (July-June), 1,367 leatherback turtles nested on this beach, and by the 1998-99 season, only 117 leatherback turtles nested (Figure 14) (Spotila et al., 2000). The 2003/2004 nesting season showed an increase in nesting abundance from the previous two seasons. An estimated 159 females nested at Playa Grande in 2003/2004 up from 69 and 55 in 2001/2002 and 2002/2003. Scientists speculate that the low turnout during 2002-03 may have been due to the “better than expected season in 2000-01 (397 nesting females) which temporarily depleted the reproductive pool of adult females in reproductive condition following the El Niño/La Niña transition” (R. Reina, Drexel University, personal communication, September 2003).

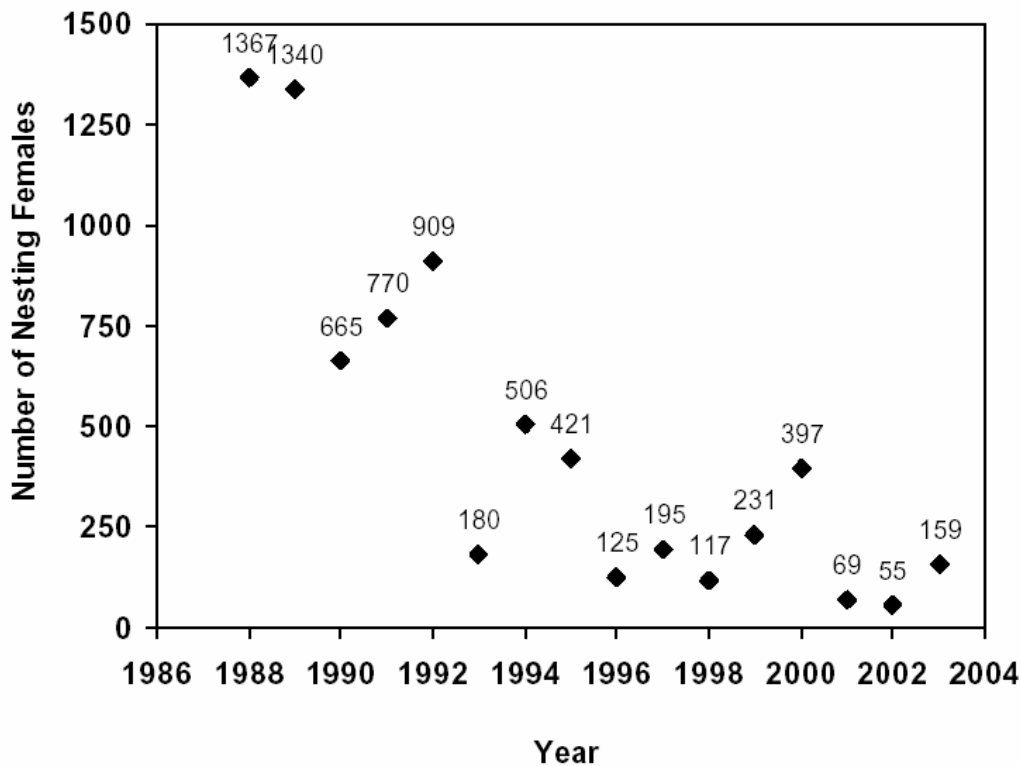


Figure 14. Estimated number of nesting female leatherback turtles at Playa Grande, Costa Rica (Spotila et al. 2000; Reina et al. 2002; numbers for the 2002/2003 and 2003/2004 seasons came from <http://www.leatherback.org/pages/project/report/report0304.htm> and were confirmed by personal communication from R. Reina to C. Fahy). The nesting season occurs over the winter months and hence over

two calendar years. Therefore, the year on the x-axis is the earlier calendar year of the census and the season would be denoted year/yaer+1. (Figure Source: Snover, 2005).

Researchers began tagging females at Playa Grande in 1994. Since then, tagged leatherbacks have had a low return rate - 16% and 25% in the five or six years following tagging. Spotila et al., (2000) calculated a mean annual mortality rate of 35% for leatherbacks nesting at Las Baulas. At St. Croix, US Virgin Islands nesting grounds, female leatherbacks returned approximately 60% over the same period (McDonald and Dutton, 1996 *in* Reina et al., 2002) indicative of mean annual mortality rates from 4-10% (Dutton et al., 1999 *in* Reina et al., 2002). Thus, comparatively few leatherback turtles are returning to nest on east Pacific nesting beaches and it is likely that eastern Pacific leatherback turtles are experiencing abnormally high mortalities during non-nesting years. Since 1993, environmental education and conservation efforts through active law enforcement have greatly reduced egg poaching in Costa Rica (Chaves et al., 1996). During the 1993-94 nesting season, poaching accounted for a loss of only 1.3% of nests on Playa Grande. Other losses were due to predation, tidal effects and failure in egg development or infestation by maggots (Schwandt et al., 1996). Bell et al., (2003) found that while leatherbacks at Playa Grande had a high rate of fertility (mean = 93.3% \pm 2.5%), embryonic death was the main cause of low hatchling success in this population. Researchers at Playa Grande have also found that temperature of the sand surrounding the egg will determine the sex of the hatchlings during a critical phase of their embryonic development. At this beach, temperatures above 29.5°C produce female hatchlings, while below 29.5°C, the hatchlings are male (Bell et al., 2003).

As evidenced by trends in the nesting beach census data, there is a high probability of quasi- and ultimate extinction of the Playa Grande, Costa Rica population of leatherbacks (Snover, 2005), consistent with Spotila et al., (2000). A trend analysis conducted by Snover, (2005) indicates with near certainty, that the population will reach quasi-extinction thresholds (defined as 50 nesting females) within the next 20-25 yr and a high probability of ultimate extinction of the population over a 50-100 yr time period.

In 2000, Spotila et al. estimated that there were 1,690 adult female leatherbacks in the eastern Pacific. The estimated number of nesting females in the eastern Pacific decreased to 910 in 2004 (Spotila, 2004)¹⁶.

Mexico

The decline of leatherback subpopulations is even more dramatic off the Pacific coast of Mexico. Surveys indicate that the eastern Pacific Mexican population of adult female leatherback turtles has declined from 70,000¹⁷ in 1980 (Pritchard, 1982b, *in* Spotila et al., 1996) to approximately 60

¹⁶ Spotila (2004) is an anecdotal compilation of data presented in a book and not the results of an original scientific study. Sources for the numbers are not provided in Spotila (2004).

¹⁷ This estimate of 70,000 adult female leatherback turtles comes from a brief aerial survey of beaches by Pritchard (1982), who has commented: "I probably chanced to hit an unusually good nesting year during my 1980 flight along the Mexican Pacific coast, the population estimates derived from which (Pritchard, 1982b) have possibly been used as baseline data for subsequent estimates to a greater degree than the quality of the data would justify" (Pritchard 1996).

nesting females during the 2002/03 nesting season, the lowest seen in 20 years (L. Sarti, UNAM, personal communication, June 2003).

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

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

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

Censuses of four index beaches in Mexico during the 2000-2001 nesting season showed a slight increase in the numbers of females nesting compared to the all-time lows observed from 1996 through 1999 (Sarti et al., in prep). However, the number of nests during the 2001/2002 and 2002/2003 were the lowest ever recorded, as shown in Table 12.

Table 12. Annual number of leatherback nests from 2000-2003 on primary and secondary nesting beaches.

Index Beach	2000-2001	2001-2002¹	2002-2003²
Primary Nesting Beach (40-50% of total nesting activity)			
Mexiquillo	624	20	36
Tierra Colorada	535	49	8
Cahuitan	539	52	73
Barra de la Cruz	146	67	3
Secondary Nesting Beaches			
Aqua Blanca	113	No data	No data
Total – all index beaches	1,957	188	120
Total – Mexican Pacific	4,513	658	Not yet available

¹ Sarti, personal communication, March, 2002 – index beaches; Sarti et al., 2002 for totals.

² Source: Sarti, personal communication, December, 2003 – index beaches, totals

A summary of total leatherback nests counted and total females estimated to have nested along the Mexican coast from 1995 through 2003 is shown in Table 13. During the 1980s, 30% of the nesting females per season were remigrants, but since the mid-1990s, there has been very little evidence of remigration (Sarti et al., 2000). During the 1999-2000 and 2000-01 nesting seasons, only a small increment in the number of remigrant turtles was observed (Sarti, 2002).

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

Table 13. Total leatherback nests counted and total number of females estimated to nest along the Mexican Pacific coast per season. (Source: Sarti et al., 2000 (1995-1999 data), Sarti et al., 2002 (2001-02 data), Sarti, personal communication, June 2003 (2002-03 data).

Season	Nests	Females
1995-1996	5,354	1,093
1996-1997	1,097	236
1997-1998	1,596	250
1998-1999 ¹	799 ¹	67 ²
1999-2000	1,125	225
2000-2001	4,513	991
2001-2002	658	109-120

¹ Value corrected for E1 (error due to track and bodypit aging) and E2 (error due to difficulty of observation from the air) only.

² Number of females only includes tagged females at the key beaches.

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) estimate that up to 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 14).

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

Table 14. Nest protection at index beaches on the Pacific coast of Mexico (Source: Sarti et al., personal communication, December 2003)

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

Nicaragua

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

Conclusions on the status of leatherbacks in the Pacific

Although quantitative data on human-caused mortality are scarce, the available information suggests that leatherback mortality on many nesting beaches remains at unsustainable levels (Tillman, 2000). Published assessments of the extinction risks of leatherback turtles in the Pacific Ocean have concluded that these turtles have a very high risk of disappearing from the Pacific Ocean within one or two human generations (Spotila et al., 1996, 2000). Based on our review of the available information, eastern Pacific leatherback populations appear to be at much lower levels of abundance than western Pacific leatherback populations and the status of leatherbacks in the Pacific is worse than the status of Atlantic populations. Recent information (Dutton et al., in review) reveals that the status of nesting female leatherback populations in the south western Pacific region appears to be better than previously stated in Spotila, (2000) or NMFS, (2004). Though greater numbers of nesting female leatherbacks have been discovered in the western Pacific region, trend information is not available for these newly described nesting sites (Dutton et al., in review) thus, no statements can be made describing the anticipated outlook for these populations for which we have no trend data. Different nesting aggregations of sea turtles are effectively isolated from one another; female leatherback turtles from other nesting

beaches will not re-colonize beaches where nesting activity has become extinct. Therefore, if a nesting aggregation becomes extinct, it will remain extinct.

6.12.5 Sea Turtle Conservation Projects in the Western Pacific Region

Over the past several years, the NMFS Pacific Islands Fishery Science Center (PIFSC), NMFS Pacific Islands Regional Office (PIR), NMFS Southwest Fishery Science Center (SWFSC), and the Council have laid the groundwork for significant turtle conservation efforts in the Western Pacific Region. These conservation efforts are aimed at increasing the capacity for the continued survival and recovery of Pacific sea turtle populations in the wild.

Priorities for the region's sea turtle conservation program are directed towards the following five areas of concentration and function in coordination with all relevant regional organizations: data management to fill information gaps; conservation measures to reduce direct harvest of sea turtles and protect nesting beach habitat; education and outreach about sea turtle conservation; international management and networking; and fishery mitigation through research and transfer of gear technologies designed to reduce bycatch of sea turtles to foreign fisheries.

Currently, approximately 70 initiatives have been implemented through support from these organizations, which were selected based on their priority for filling an information gap, building local capacity and awareness for sea turtle conservation, and mitigating threats to the continued existence of sea turtle populations. These projects can be categorized as education and outreach; supporting and convening meetings and forums to promote information exchange and maintain momentum for continued sea turtle research, conservation, and management; protecting nesting beaches, nests, and nesting females; conducting aerial and in-water surveys to determine distribution and abundance; conducting studies to fill knowledge gaps about sea turtle biology, ecology, and life history; studies to determine gear modifications and best management practices likely to result in reduced injury and mortality to sea turtles from fisheries; and research into policy for sustainable land based sea turtle conservation activities in the WCPO. These efforts were developed and initiated with the overall goal of increasing the capacity for sea turtle recovery in the Pacific and are anticipated to result in beneficial effects for sea turtle populations in the Pacific Ocean.

7 Environmental Baseline

By regulation, environmental baselines for biological opinions include the past and present impacts of all state, Federal or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process (50 CFR 402.02). The environmental baseline for this biological opinion includes the effects of several activities that affect the survival and recovery of ESA-listed large whale and sea turtle species in the action area.

Although some information on the presence of species within the action area have been collected, the information is too limited to determine patterns of distribution or abundance of marine mammals and sea turtles in the Western Central Pacific. The following narratives summarize available information on natural and anthropogenic phenomena that are known to or

are suspected to influence the distribution, abundance, status, and trends of these species in the WCPO.

7.1 Factors Affecting Species within the Action Area

This section identifies and describes all known human-induced sources of impact to the listed species in the Action Area, except those caused by the action (i.e., the U.S. purse seine fishery in the WCPO). Although the sources described in this section are limited to those in the Action Area, it should be noted that additional sources outside the Action Area impact the same individuals and populations that are impacted in the Action Area. These sources include pelagic fisheries, nearshore fisheries, and for sea turtles, directed harvest (of turtles and eggs) and various sources of nesting beach degradation.

7.1.1 Marine Mammals

7.1.1.1 Whaling

The majority of large whale species are listed as endangered species under the ESA because their populations were severely depleted by whalers in the nineteenth and twentieth centuries. In 1986, the IWC set a zero quota for commercial whaling, and three countries first objected this moratorium: Norway, the Soviet Union, and Japan. In 1987, Japan began its first phase of scientific whaling in the Antarctic, expanding to the North Pacific in 1994 originally targeting minke whales. Japan's catches soon expanded to include other whale species such as sei and sperm whales. Although commercial whaling, research, and subsistence whaling have been a problem in the past, stringent regulations have significantly decreased the numbers of marine mammals taken by these methods.

Blue whales

Antarctic whaling began in 1905, with a peak season in 1931 with 29,409 whales caught (Branch et al., 2004). During this time, blue whales provided 75% of the world's whale oil production. However, by 1963, the Antarctic population of blue whales had reached a level of "serious danger of extermination," and in 1964, the IWC began protection for this species. Despite this, Soviet whaling continued and killed 852 Antarctic blue whales by 1973, when illegal whaling ended.

Fin Whales

In the Southern Hemisphere, 703,693 fin whales were caught in Antarctic from 1904 - 1975 (Perry et al., 1999). When humpbacks became rare in the Southern Hemisphere around 1913, the catch of fin and blue whales rose. Between 1911 and 1924, 2,000 – 5,000 fin whales were taken per year. Factory whaling ships of 1925 increased the numbers taken, and by 1962 fin whales were becoming scarce while the sei whale catches increased. In 1974, less than 1,000 fin whales were taken, and the IWC prohibited whaling in the Southern Hemisphere in 1976. Japan's scientific whaling proposed to take about 820 fin whales from the Southern Ocean in 2005, in addition to catches in the Western Pacific Ocean (Gales et al., 2005).

Humpback Whales

In the Southern Hemisphere, about 28,000 humpback whales were taken in the pelagic waters of the Antarctic from 1917-1938 (Perry et al., 1999). The IWC first regulated the catches in this

region in 1938. Humpback whales were hunted consistently from Australian and New Zealand waters post-1938, during which time approximately 22,000 whales were taken. Only very little whaling was conducted in the South Pacific. Tongan whaling, though small in scale, recorded annual catches prior to 1960 at 30 - 40 whales, and an aboriginal catch of 12 whales was recorded in Tonga in 1978. Worldwide protection for humpback whales from whaling operations began in 1966. Despite this, Japan proposed in 2005 to take about 800 humpback whales from the Southern Pacific Ocean for scientific purposes (Gales et al., 2005).

Sei Whales

From 1910 – 1975, 152,233 sei whales were caught in the Southern Hemisphere (Perry et al., 1999). Sei whales became a target species for whaling operations in the late 1950s when blue and fin whale catches became rare. A peak of over 20,000 sei whales caught was recorded in 1964, dropping to less than 2,000 whales taken by 1976. Sei whales received IWC protection in 1977. However, Japan continues to hunt sei whales for scientific purposes. Since 1987, Japan has taken approximately 140 sei whales (Gales et al., 2005). In 2004, 100 sei whales were caught by Japan in the North Pacific Ocean, the same amount in their current 2006 permit as listed on the IWC website.

Sperm Whales

Whaling areas for sperm whales in the South Pacific occurred south of 40°N, located around the Hawaiian Islands, the Philippines, and along the equator around the Society Islands, the Marquesas Islands, Fiji, and Samoa toward South America (Gosho et al., 1984). Based on historical whaling records, Hawai`i became an important whaling center with more than 600 ships by 1846. The number of U.S. whaling ships began to decline from 315 vessels in 1844, to 134 vessels in 1875. From 1883 - 1924, less than 1,000 sperm whales were estimated for an annual worldwide catch. An estimated 96,200 sperm whales were in the Central Pacific waters for the year 1946 (Gosho et al., 1984). Following this, over 20,000 sperm whales were caught in the Southern Hemisphere from 1956 – 1976. The IWC finally banned the killing of sperm whales in 1988.

Since 1987, an estimated 38 sperm whales were taken by Japan's scientific whaling (Gales et al., 2005). In 2004, Japan caught 10 sperm whales in the Southern Hemisphere and another 5 in the North Pacific Ocean, also for research purposes. The same number of sperm whales was included in Japan's scientific permit for 2006. The implications of this action for the status and trend on the population of sperm whales are uncertain.

7.1.1.2 WCPO Longline Fisheries

Longline fisheries that occur in the Action Area include U.S. longline fisheries, particularly the American Samoa-based longline fishery (the range of the Hawai`i-based longline fishery overlaps only slightly with the Action Area), and the longline fisheries of other nations.

The most common types of interactions between marine mammals and longline fisheries are depredation (the removal of, or damage to, hooked fish from longlines¹⁸), bait removal, and

¹⁸ Some authors choose to include the removal of baitfish from the longline in this definition; however, because different species of cetaceans tend to undertake these two behaviors, they are treated here as separate types of interactions.

entanglement (including becoming hooked on the lines). The reporting of incidences of depredation and bait removal appears to have increased recently in the WCPO region (Donaghue et al., 2002). However, it is unknown if this reflects an increase in the occurrence of depredation, an increase in effort in the tuna longline fisheries, changes in cetacean populations, or increased reporting. Because the specificity of available data with respect to the identification of marine mammal species is often poor, this section is written about marine mammals or cetaceans generally.

7.1.1.3 Depredation

Depredation is not a new phenomenon to the longline tuna industry. It has been recognized as a worldwide problem for the industry since the early 1950s. The extent and nature of depredation varies by target species, region, gear deployment, and gear hauling methods (Donaghue et al., 2002).

Although several species of cetaceans are believed to depredate hooked fish in the WCPO, there is limited reliable information regarding the actual species of marine mammals involved, in part because of fishermen's limited ability to accurately identify marine mammals to the species level. The most common cetacean species known to be involved in depredation are the killer whale, false killer whale, and pilot whale, all of which are small toothed whales, and none of which are listed species. The only listed marine mammal species in the Action Area that is toothed and that could be involved in depredation is the sperm whale. The specificity of the available data with respect to species identification is too poor to determine the extent to which sperm whales might be involved in depredation.

The major source of information on interactions of the longline tuna fishery with cetaceans in the Treaty Area is the limited data collected by the FFA observers deployed on longline vessels or those opportunistically deployed on vessels of other flags over the past 10 years by the SPC. Additionally, limited amounts of information come from the few PICs that have observer programs.²⁰ Observer data from 1995 to 2002 found a similar geographic distribution of depredation by sharks and small toothed whales (sperm whales were not specifically documented, but they cannot be ruled out), where shark damage (2.1%) was greater than whale damage (0.8%). It was also observed that depredation by whales was more frequent on yellowfin and bigeye tuna, whereas damage by sharks was less discriminate and spread across various species (wahoo, yellowfin tuna, skipjack tuna, blue marlin, striped marlin, spearfish, and swordfish). It is believed that no significant difference exists in damage (caused by depredation) that occurs in tropical and subtropical areas of the WCPO (Lawson, 2001b).

7.1.1.4 Bait Removal

Removal of baitfish from the hooks on the longline usually occurs during the line-setting procedure. Bait removal is typically observed to be practiced by small (non-listed) cetaceans (eight species of dolphins have been specifically documented to be in the vicinity of longline sets in the South Pacific: bottlenose, common, Fraser's, pan-tropical spotted, Risso's, rough-toothed,

²⁰ In general, the longline fleets operating in the WCPO (except the Hawaii-based fleet) have very low observer coverage rates, and it is believed that in areas where observers do exist, they are not well-trained in the identification of specific marine mammal species.

spinner, and striped dolphin, but it is uncertain whether all of these species participate in actual bait removal). The only listed marine mammal species in the Action Area that is toothed and that could be involved in bait removal is the sperm whale. The specificity of the available data with respect to species identification is too poor to determine the extent to which sperm whales might be involved in bait removal. As with depredation, bait removal is known to occur in the WCPO tuna longline fishery, but its extent is currently unclear.

Interactions between sperm whales and longline fisheries have been well-documented in other parts of the Southern hemisphere, in particular off South Georgia, the Kerguelen Islands, and Southern Chile. Such interactions included entanglement in gear (Anonymous, 1994; Ashford et al., 1996), following vessels for periods of days (Ashford et al., 1996; Capdeville, 1997), and observed feeding off gear (Crespo et al., 1997; Anonymous, 1994). This evidence, combined with anecdotal reports, suggests that interactions between sperm whales and longline operations may be widespread in Southern Ocean waters.

In Alaska waters, aside from scattered anecdotal reports (Dahlheim, 1988; Rice, 1989), few data are available regarding sperm whale behavior in relation to commercial longline operations. In 1997 and 1998, sperm whale depredation of longline-caught fish was recorded in the Gulf of Alaska (Hill et al., 1999). Within the Gulf of Alaska, the sperm whale/longline interaction pilot program demonstrated a high percentage of sets during which sperm whales were observed. Whales were present during 28.5% of the 562 sets monitored over the two years. Depredation was recorded during 46.2% of the 160 sets in which sperm whales were present over that same period.

Sperm whales clearly affect the longline fishery, although at this time it is not possible to assess the level of impact. There was no evidence that mortality or serious injury to sperm whales was occurring as a result of this interaction. However, in longline fisheries off South America, entanglements of sperm whales in longline gear were recorded. The first and only documented sperm whale entanglement in Alaska's longline fishery occurred in 1997 (the animal was not considered seriously injured according to the NMFS definition; Angliss and DeMaster, 1998), providing further impetus to continue research on the nature and magnitude of this interaction.

7.1.1.5 Entanglement

Entanglement of small cetaceans in longlines often results in the death of the individuals, as they are unable to surface and therefore cannot breathe. Larger cetaceans, because of their size, are able to break the line and escape, although the entangled line may remain wrapped around the individual and impair their abilities to move and feed. This entanglement with gear may eventually result in the death of larger cetaceans, depending on the location of the entangled lines.

Cetaceans may become entangled in the longline while removing the hooked catch, removing the baitfish or preying on the free-swimming fish in the vicinity of the longline. They are typically hooked only during the process of removing bait from the longline hooks.

There are about 6,000 longline vessels operating in the WCPO. The WCPO longline fishery as a whole was observed at rates of less than 1% from the early 1990s through 2004 (Molony, 2005).

Excluding observations of the Hawai'i-based longline fleet and sets made south of 31° South latitude, Molony (2005) found that the available WCPO longline observer data for 1995-2004 contained 22 records of marine mammal interactions. The majority, 19, were not identified to species. Two were recorded as unidentified toothed whales, which may or may not be sperm whales, and more than likely may have been false killer whales or pilot whales. The fate and condition of 19 were recorded: 14 were alive at the time of capture and 5 were dead. Eleven were in healthy condition at the time of release. After raising the observed rates of capture and mortality according to the level of observer coverage, Molony (2005) estimated that up to 2,200 marine mammal captures occurred each year in the WCPO longline fisheries, with mortality rates less than 30% in most years.

7.1.1.6 Summary of WCPO Longline Fishery

Longline depredation and bait removal involve toothed whales, only one of which, the sperm whale, is listed under the ESA and occurs in the Action Area. There is no evidence to indicate whether or to what extent sperm whales are involved in longline depredation or bait removal in the Action Area, and consequently, no information as to whether depredation or bait removal adversely impacts sperm whales.

Entanglement in longline gear can conceivably occur with any marine mammal species, including listed species. Observer data for the 1995 - 2004 period indicate that up to 2,200 marine mammals are captured annually in the WCPO longline fisheries (Molony, 2005), some of which could be listed species, but there are no records affirmatively indicating entanglement, capture, or mortality of any listed marine mammals in the longline fisheries in the Action Area.

7.1.1.7 Non-U.S. WPCO Purse Seine Fisheries

The tuna purse seine fleets of a number of nations operate in the Action Area. There are about 400 purse seine vessels operating in the WCPO. The WCPO purse seine fisheries as a whole (all nations) was observed at rates between 2-11% from 1994 - 2004 (Molony, 2005). Most observed marine mammal interactions have not been identified to species, or even to the level of whales versus other marine mammals, so it is difficult to gauge the impact of the WCPO purse seine fisheries on listed marine mammals. Because of those shortcomings, and because observations of U.S. fleet make up the bulk of the pool of observer data (the database also includes observations of the fleets of the FSM, Marshall Islands, Kiribati, and PNG operating under the FSM Arrangement and limited data from Taiwanese and Korean purse seine vessels; SPC, 2001), the data for the U.S. fleet are included in the following review of the WCPO purse seine fisheries' observer data for the 1994 - 2004 period.

Molony (2005) reported that in the 27,644 purse seine sets observed in the WCPO between 1994 and 2004, a total of 687 marine mammals in 137 sets were reported by observers as captured (Table 15). The majority of the observed captures were not identified as to species. Most (581 from 110 sets) were identified as "unidentified marine mammals." "Unidentified dolphins and porpoises" numbered 33 from 11 sets; "unidentified toothed whales" numbered 19 from one set; and "unidentified whales" numbered 5 from 2 sets. Forty-nine were identified to species, none of which were listed species (24 common dolphins from 8 sets, 18 bottlenose dolphins from 3 sets, 4 spinner dolphins from 1 set, 2 short-finned pilot whales from 2 sets, and 1 pygmy killer whale from 1 set).

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Table 15. Summary of observed WCPO purse seine sets with marine mammal interactions by species from 1994-2004. Source: Molony, 2005.

Year	Observed Sets	Sets with marine mammals	Sets with unidentified marine mammals	Sets with unidentified dolphins & porpoises	Sets with unidentified toothed whales	Sets with unidentified whales
1994-2004	27,644	137	110 (581 animals)	11 (33 animals)	1 (19 animals)	2 (5 animals)

The condition of most of the captured marine mammals (92%) was not recorded. Of the remaining observations, 42 were reported as dead and 16 as alive. Marine mammals were also recorded as incidentally captured, with 652 animals discarded, 29 animals escaping from the purse seine net, and 6 animals retained for unknown reasons (Table 16). Of these six retained, only one animal was identified as a common dolphin and the remaining five were of unknown species. It is unknown which animals came from which set (i.e., all animals incidentally captured at once or separately). After raising the observed capture rates according to the rate of observer coverage, Molony (2005) estimated that about 900 marine mammals were captured (95% confidence interval of about 16,000) and about 50 were killed (confidence interval of 2,800).

Table 16. Summary of the fate of marine mammals incidentally captured in WCPO purse seine fishery interactions from 1994-2004. Source: Molony, 2005.

Year	Total no. marine mammals	Marine mammals discarded	Marine mammals escaped	Marine mammals retained
1994-2004	687	652	29	6

The incidental capture of marine mammals also varied if the set was made with an associated object. There were a total of 116 observed sets with associated floating objects, which comprises 84.7% of the total 137 observed sets that captured marine mammals (Molony, 2005). This included 56 sets on anchored FADs, 35 sets on logs, 13 sets on drifting FADs, 11 sets on live whales, and one set on whale sharks. A total of 16 sets that were made with unassociated schools were recorded, of which 7 were sets with unassociated schools and 9 were sets with baitfish associated schools. Although sets with associated schools have a low, overall impact on marine mammals, those set on anchored FADs and logs had a higher rate of marine mammal interaction, possibly due to the diverse marine community retained by floating objects.

This comprises the largest interactions between the purse-seine fishery and marine mammals, those sets that were deliberately set on whales associated with tuna schools. Of all the baleen whales, sei whales are the most common to be encircled in the purse-seine net on baitfish associated sets (Bailey et al., 1996). These animals generally will escape on their own by punching through the net when close to the surface, or the corkline will be submerged to aid in the whales' release. It has been suggested that this experience does not present a negative impact

based on the few recorded cases of whales observed to return to feeding after encircled by the purse-seine net.

7.1.1.8 U.S. Eastern Tropical Pacific Tuna Purse Seine Fishery

The effects on listed marine mammal species of the U.S. tuna purse seine fishery in the ETPO were previously considered in a 1999 biological opinion (NMFS, 1999) and reviewed in the biological opinion for the Fishery Management Plan for the West Coast Fisheries for Highly Migratory Species (NMFS, 2004a). From these documents, between 1979 and 1990, only five sets out of 21,554 sets made by the U.S. tuna purse seine fishery (large vessels only (>400 st)) in the ETPO resulted in the accidental encirclement (net is not “pursed” yet) of a large whale (R. Rasmussen, NMFS, personal communication). Out of these five sets, only two large whales were captured (net is “pursed”) and none were reported as resulting in mortality. Based on these data, the baleen whale “encirclement rate” in the U.S. ETPO tuna purse seine fishery is estimated to be 0.000231 whales per set. In other words, for every 10,000 sets, approximately 2 large whales may be accidentally encircled in the U.S. ETPO tuna purse seine fishery, and the likelihood that a large whale may be captured is even lower. The data indicate that encircled and captured whales escape the net, or are released uninjured from the net circle. Therefore, interactions are uncommon for the U.S. purse seine fisheries in the ETPO.

7.1.1.9 Other Fisheries

There is limited information indicating the extent to which listed marine mammals interact with other fisheries in the Action Area. Examining humpback whales specifically, NMFS (2005b) found that there was no information available documenting interactions between CNP humpback whales and the Pacific Remote Islands Areas (PRIA) or American Samoa-based troll fisheries (or with the Hawai`i-, Guam- or CNMI-based troll, handline or pole-and-line fisheries, which occur outside the Action Area). However, these fisheries are not observed. Given recent levels of effort, the selectivity of the gear, and the location of fishing effort relative to the CNP humpback stock, NMFS expects that interactions between CNP humpbacks and these U.S. fisheries would be rare (2005b). Interactions between listed marine mammals and the similar small-scale pelagic fisheries of the Pacific Island countries within the Action Area are also expected to be rare.

7.1.1.10 Sources Other Than Fisheries

In addition to fisheries, other factors that may affect listed marine mammals in the Action Area include noise (acoustic), vessel strikes, and marine pollution and debris.

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 (NMFS, 2000). Increases in the number of motorized vessels throughout the world’s oceans means that the level of man-made noise in the ocean has increased. The possible impacts and long-term effects on cetaceans of increased levels of man-made noise in the action area, or even in the world’s oceans as a whole, are currently unknown. Cetaceans are highly acoustic animals and many species rely on audible feedback in order to locate prey and navigate within their environment (Dawson et al., 1998).

The rate of accidental boat collisions with cetaceans in tuna fisheries is thought to be very low in the action area, as there are no reported incidences of such collisions. If they did occur, they

would most likely affect large whales, which are more prone to this type of interaction due to their slower movements and large size.

All large whales are vulnerable to the effects of marine pollution. Marine pollution from sewage outfalls, dumping at sea, bilge cleaning, discarded trash, or fishing gear, etc., could adversely impact the habitat of listed marine mammal species by having a negative effect on their prey, causing entanglement or disrupting the digestive system through ingestion of foreign materials (e.g., occlusion of the digestive tract) (NMFS, 2004a).

The individual and cumulative effects of these sources of noise, disturbance, and pollution on marine mammals are unknown (NMFS, 2000).

7.1.2 Sea Turtles

7.1.2.1 Fishery Impacts

U.S. Fisheries Managed under the Pelagics Fisheries Management Plan (FMP)

Fisheries managed under the Pelagics FMP that occur in the Action Area include the deep-set component of the Hawai`i-based longline fishery (but its overlap with the Action Area is small), the PRIA pelagic troll and handline fisheries, and the America Samoa-based pelagic longline and troll fisheries (fisheries managed under the Pelagics FMP that occur outside the Action Area include the shallow-set component of the Hawai`i longline fishery and the troll, handline, and pole-and-line fisheries in Hawai`i, the Territory of Guam, and the Commonwealth of the Northern Mariana Islands). These fisheries are described and analyzed in detail in a 2005 biological opinion on the deep-set component of the Hawai`i longline fishery (NMFS, 2005), a 2004 biological opinion on all the fisheries managed under the Pelagics FMP (“2004 Opinion”) (NMFS, 2004), and a 2001 environmental impact statement on the Pelagics FMP (NMFS, 2001) and its 2004 supplement (NMFS, 2004a).

All five species of sea turtles may be taken in the fisheries managed under the Pelagics FMP. The known level of effort and the selectivity of the gear used in all the fisheries but the Hawai`i longline fishery have led NMFS to conclude that few sea turtles, if any, are captured, injured, or killed in these fisheries (NMFS, 2005b). These fisheries are not observed and most of the sea turtles that have been reported to have been captured in these fisheries have not been identified to species. Estimated interaction rates in these fisheries have consequently been described only in terms of hardshell turtles (i.e., green, hawksbill, loggerhead, and olive ridley) and leatherback turtles.

The numbers of sea turtle interactions expected to occur incidentally in both the deep-set and shallow-set component of the Hawai`i longline fishery and the other Pelagics FMP-managed fisheries are shown in Table 17, Table 18 and, Table 19, respectively.

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Table 17. The number of turtles expected to be captured or killed incidentally over a three-year period in the deep-set component of the Hawai'i-based longline fishery. Source: NMFS (2005).

Sea Turtle Species	Number Captured	Number Killed
Green	21	18
Leatherback	39	18
Loggerhead	18	9
Olive Ridley	123	117

Table 18. The annual number of turtles expected to be captured or killed incidental to the Hawai'i-based pelagic, shallow-set longline fishery (Source: NMFS, 2004).

Sea Turtle Species	Number Captured	Number Killed
Green	1	1
Leatherback	16	2
Loggerhead	17	3
Olive Ridley	5	1

Table 19. The number of turtles expected to be captured or killed incidentally over a one-year period in the American Samoa-based longline fishery and the handline, troll, and pole-and-line fisheries managed under the Pelagics FMP.

Sea Turtle Species	Number Captured	Number Killed
Hardshell sea turtles	6	1 ^a
Leatherback sea turtles	1	0

Source: NMFS, (2004).

^a The number provided in the table is from NMFS, 2004; however, in the American Soma longline fishery in 2006, two green turtles were observed to have been killed incidental to the fishery to date. The 2006 longline fishery is still ongoing.

Note that this table includes the non-longline pelagic fisheries in Hawai'i and the pelagic fisheries in Guam and the CNMI, which occur outside the Action Area.

Foreign WCPO Longline Fisheries

There are roughly 6,000 longline vessels operating in the WCPO. The WCPO longline fishery as a whole was observed at rates of less than 1% from the early 1990s through 2004 (Molony, 2005). In contrast, the U.S. fleet has, since the mid-1990s, received about 20% coverage. The relatively low rates for foreign fleets, along with generally poor resolution in terms of taxonomic identification of the observed turtles, mean that interaction and mortality estimates for the WCPO longline fishery as a whole are much less certain than for the U.S. longline fishery, as described above. For the purpose of giving an indication of the impacts of the foreign WCPO

longline fisheries, the data for the U.S. fleet are included in the following review of the WCPO longline observer data.

Molony (2005) found that longline observer data from 1980 through 2004 contained records of 481 sea turtle interactions, but only 159 of those records were from the area between 31° South and 15° North latitude, which better represents the Action Area. Of the 481 observed turtles, 180 were loggerheads, 104 were olive ridleys, 76 were unidentified, 65 were leatherbacks, 44 were greens, and 12 were hawksbills. After raising the observed capture and mortality rates according to the level of observer coverage, Molony (2005) estimated that between 4,000 and 15,000 turtles were captured and between 500 and 3,000 turtles were killed each year.

Non-U.S. WCPO Purse Seine Fisheries

Turtle interactions in purse seine operations occur when turtles – which are apparently attracted to the diverse prey items in the vicinity of logs and other floating debris that are set on (SPC, 2001) – are pursed within the net.¹⁹ If they become entangled in the net, turtles may drown. In most cases observed in the WCPO, turtles are encountered alive in the net, at which point they are scooped up and released by the fishermen. In very rare instances, they may be crushed during the process of loading the net on board the vessel after a set has been completed (SPC, 2001).

There are about 400 purse seine vessels operating in the WCPO. The WCPO purse seine fishery as a whole (all nations) was observed at rates between 2-11% from 1994 through 2004 (Molony, 2005; Table 20). The observer database contains information from vessels operating under the FSM Arrangement which includes fleets from the FSM, Marshall Islands, Kiribati, and PNG; limited information from Taiwanese and Korean vessels; and information from U.S. vessels (SPC, 2001).

Molony (2005) analyzed the WCPO purse seine observer database (which includes U.S. and non-U.S. fishery information) and reported that in 27,644 sets observed in the WCPO between 1994 and 2004, a total of 104 sea turtles were captured in 99 sets (5 sets captured 2 turtles each). The majority of the observed captures (80) were not identified as to species. Twenty-four were identified to species: 10 olive ridleys, 8 hawksbills, 5 greens, and 1 leatherback. The condition of most of the captured sea turtles (75) at the time of capture was not recorded. Twenty-five were alive at the time of capture, 24 of which were classified as healthy. Four were dead at the time of capture. After raising the observed capture and mortality rates according to the level of observer coverage, Molony (2005) estimated that about 200 sea turtles (with a 95% confidence interval of about 5,600) were taken each year between 1990 and 2004, with fewer than 20 mortalities (confidence interval of about 1,600) per year.

¹⁹ Sea turtles have also been observed associating with manmade floating objects more frequently than with natural objects. This behavior may be related to turtles' affinity for three-dimensional objects (NMFS, 2004b). Turtles also tend to exhibit a preference for objects floating horizontally and almost fully submerged, and are strongly attracted to brightly colored objects (Arenas and Hall, 1992).

Table 20. Observer coverage in the WCPO purse seine fisheries. Source: Molony (2005).

Year	Sets observed	Estimated total sets made	Estimated observer coverage
1994	1,174	67,952	0.02
1995	1,341	70,208	0.02
1996	2,215	73,110	0.03
1997	2,211	75,959	0.03
1998	2,685	99,779	0.03
1999	1,837	90,164	0.02
2000	2,127	51,012	0.04
2001	2,364	46,163	0.05
2002	3,560	53,962	0.07
2003	3,631	58,682	0.06
2004	3,368	29,611	0.11

Note: See Molony (2005) for precise description of area covered.

Molony (2005) found a significant difference in turtle capture rates among set types. Fewer than expected captures occurred on unassociated sets, baitfish-associated sets, and drifting FAD sets. More than expected captures occurred on log sets, anchored FAD sets, and whale sets. The exception was for green turtles, for which more than 60% of captures occurred in unassociated sets.

SPC (2001) offered three hypotheses to explain the relatively low interaction rates associated with drifting FADs: 1) drifting FADs have been in the ocean for relatively brief periods and thus have had relatively little time to attract turtles and other species; 2) natural debris (logs and animals) are more influenced by currents and are aggregated at current lines, where turtles may be relatively abundant; and 3) the area where drifting FADs are more prevalent – that is, in the eastern areas of the WCPO – may have a lower abundance of turtles than other areas in the WCPO.

Almost half of the interactions in the observer data examined in SPC (2001) occurred in the months of June and July, which may be related to the turtle populations' migratory patterns (but the data are insufficient to draw any conclusions about seasonality). ENSO events, which affect the extent of the warm pool and the distribution of natural debris, may also be important factors with respect to the distribution of turtles and turtle interaction rates (SPC, 2001).

Since January 1993, turtle bycatch data have also been collected by Micronesian Maritime Authority (MMA) observers active on purse seine vessels operating in the WTP. Of the 493 sets observed during the period January 1993 - April 1994, 10 turtles (5 hawksbills, 2 olive ridley turtles, 1 leatherback, 2 unidentified) were taken; at least 6 of these were alive when released.

Most of this bycatch was taken in log sets (seven) and the remainder (i.e., one olive ridley, one hawksbill and one unidentified turtle) were accidentally taken in separate school sets (Heberer, 1994a in Bailey et al., 1996).

7.1.2.2 Eastern Tropical Pacific Tuna Purse Seine Fishery

The international fleet represents the majority of the fishing effort and carrying capacity in the ETP tuna fishery, with much of the total capacity consisting of purse seiners greater than 400 short tons (st). The latest information from the IATTC shows that the number of active purse seiners of all sizes is 242 vessels, with Mexico and Ecuador comprising the majority of the fleet (77 and 87 vessels, respectively) (Source: IATTC, 2005 (www.iattc.org)).

Data from observers on both U.S. and foreign tuna purse seine vessels have been gathered collectively by the IATTC since the early 1990s. In 2005, 1,350 sets in the ETP purse seine fishery involved 1,965 sea turtles (IATTC, 2006).

From 1993-2003, between approximately 9 and 55 sea turtles were killed per year by vessels over 400 st (363 mt) in the ETP purse seine fishery. The primary species taken were olive ridleys (Table 21; M. Hall, IATTC, personal communication, 2005), likely because they are proportionately more abundant than any other sea turtle species in the ETP and they have been observed to have an affinity for floating objects (Arenas and Hall 1992). The mortality estimates contain fractions because while the IATTC has a known number of sets and turtle mortality from their observer database, they only have a known number of sets (not turtle mortality) from the national observer programs. Therefore, the mortality is pro-rated to make up for the sets for which the IATTC has no known turtle mortality data. The majority of sea turtles were taken in sets on floating objects (Table 21).

Since 1999, seminars have been given by the IATTC to skippers and their crews to educate them on, among other issues, status of sea turtles, and handling and recovery of turtles taken by purse seiners in the ETP. In addition, during the meeting held in Lima, Peru from June 14-18, 2004, the IATTC passed Consolidated Resolution C-04-05. Under the resolution, purse seine fishermen are required to promptly release unharmed, to the extent practicable, all sea turtles. In addition, crews are required to be trained in techniques for handling turtles to improve survival after release. Vessels should be encouraged to release sea turtles entangled in FADs and recover FADs when they are not being used in the fishery. Specific to the purse seine fishery operation, whenever a sea turtle is sighted in the net, all reasonable efforts should be made to rescue the turtle before it becomes entangled, including, if necessary, the deployment of a speedboat. If a sea turtle is entangled in the net, net roll should stop as the turtle comes out of the water and should not start again until the turtle has been disentangled and released. If a turtle is brought aboard the vessel, all appropriate efforts to assist in the recovery of the turtle should be made before returning it to sea (IATTC Resolution C-04-05, Action #4).

Seven turtle mortalities were observed in the 2005 ETP tuna purse seine fishery; one turtle died due to failure to assist the turtle when on-board (IATTC, 2006). Four turtles were also documented as passing through the power block in 2005; the fate of two of the turtles was documented as severely injured and the fate of two others was unknown (IATTC, 2006).

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Table 21. Estimated sea turtle mortality by species for the ETP tuna purse seine fishery (including US) from 1993 to 2003. Includes only large (364 metric ton capacity and greater) vessels. Source: M. Hall, IATTC, 2005.

Name	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003
Green	15.0	16.1	13.0	12.0	13.0	9.0	10.9	6.1	7.8	2.1	0.0
Hawksbill	0.0	1.8	0.0	1.0	0.0	3.0	2.0	1.0	1.3	0.0	0.0
Leatherback	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Loggerhead	3.6	1.8	2.0	0.0	4.6	1.0	4.0	1.8	1.3	0.0	0.0
Olive Ridley	77.8	80.1	91.3	65.8	93.8	107.6	109.1	92.1	74.2	30.7	17.1
Unidentified	21.0	45.3	34.0	37.6	42.0	41.0	46.2	29.4	55.3	13.8	9.1
Total	117.4	146.3	140.3	116.4	153.4	161.6	172.2	130.4	139.9	46.6	26.2

The data contained in Table 22 indicate that some sea turtles killed by the entire ETP 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, 86% would be olive ridleys, 10.8% would be green turtles, 2.1% would be loggerheads, 1% would be a hawksbill, and 0.1% would be leatherbacks.

Table 22. Number of sea turtles killed (or had sustained injuries judged likely to lead to death) by all ETP purse seine fisheries (including US), by set type, from 1998-2003.

Year/type of set	Dolphin sets	Floating object sets	Unassociated (tuna school) sets	Total
1998	28	103	31	162
1999	17	128	27	172
2000	17	72	41	130
2001	16	88	33	137
2002	11	26	9	46
2003	7	17	2	26

As mentioned, the U.S. ETP fleet (large vessels only) has 100% observer coverage; therefore, the fate of every sea turtle taken is documented. Because the U.S. fleet does not set on dolphins, sea turtles are taken in school sets and log/FAD sets. Therefore, the fate of sea turtles that interact with the U.S. purse seine fleet during such sets may only be comparable to the non-U.S. fleet that sets on logs/FADs and tuna schools. Table 23 documents sea turtle interactions with the US purse seine fleet from 1998 through 2003. Similar to the entire purse seine fleet (Table 23), the majority of the sea turtles taken by the fishery are olive ridleys, and as shown in Table 23 most sea turtles are released unharmed.

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Table 23. Sea turtle interactions with the US tuna purse seine fleet (large (>363 mt (400 st)) vessels only) in the ETP, 1998-2003. [Source: M. Hall, IATTC, 2005]

Name	Fate	1998	1999	2000	2001	2002	2003
Green	Released unharmed	3	5	2	2	1	5
Hawksbill	Released unharmed	0	0	0	1	1	0
Loggerhead	Released unharmed	0	1	5	0	0	0
Olive Ridley	Released unharmed	38	27	3	16	10	34
	Escaped/evaded net	0	0	1	0	0	0
	Light injuries*	4	6	2	0	0	7
	Grave injuries**	1	0	0	3	0	0
	Killed	0	0	0	0	0	1
Unidentified	Released unharmed	2	0	3	6	1	10
	Escaped/evaded net	2	1	1	0	0	0
	Light injuries*	0	0	0	1	0	0
	Other/Unknown	1	0	0	0	0	1
Total		51	40	17	29	13	58

*Light injuries are considered to be non-lethal injuries.

**Grave injuries are considered to be eventually lethal to the turtle.

7.1.2.3 Sources Other Than Fisheries

Other factors that may affect listed sea turtles in the Action Area include vessel strikes and marine pollution and debris, such as plastic, which is ingested by turtles, but the extent and consequences of these sources are not known.

7.1.2.4 Resolutions for Sea Turtle Conservation

Fishery Resolutions

In recognition of the precarious state of most sea turtle populations in the Pacific Ocean and/or the ecological and cultural significance of sea turtles in the Pacific, and the threat that fisheries pose to the viability of those populations, many governmental organizations and regional fishery management organizations (RFMOs) have passed resolutions aimed at reducing the frequency and severity of sea turtle/fishery interactions. In 2004, the Food and Agriculture Organization of the United Nations (FAO) convened a Sea Turtles Conservation and Fisheries Technical Consultation in Bangkok, Thailand from November 29 to December 2, 2004. The technical consultation addressed major issues regarding sea turtle conservation and fisheries. The technical consultation, attended by 28 members of the FAO and observers from intergovernmental and international non-governmental organizations, agreed on recommendations for FAO, RFMOs, and for member states related to future work on sea turtle conservation and reduction of sea turtle mortality in fishing operations.

The full suite of recommendations includes measures for the FAO, RFMOs, FAO member countries, and for “all” (FAO, 2005). Measures for RFMOs such as the recently established Western and Central Pacific Fisheries Commission (WCPFC) include:

- pay urgent attention to interactions between fisheries and sea turtles, especially in regard to the collection of statistics on bycatch and fisheries interactions and the adoption of mitigation measures;

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- develop networks with a view to sharing information on mitigation measures adopted and experiences undertaken at national, regional, and global levels.

Recommendations for FAO member countries include (among others):

- pay urgent attention to the sea turtle stocks and areas identified of greatest threat
- cooperate to broaden the mandates of RFMOs to reduce the impacts of fishing on sea turtle populations
- collect and make available, data and information on trends in sea turtle/fishery interactions, including trends in fishery related mortalities
- support the initiatives with respect to sea turtle conservation and fisheries interactions that FAO will develop and, in particular those member countries in a position to do so, mobilize the necessary funding for their implementation.

The IATTC, established by international convention in 1950, is responsible for the conservation and management of fisheries for tuna and other species taken by tuna-fishing vessels in the eastern Pacific Ocean. The IATTC convention area extends from the west coast of north and central America to 150°W. The IATTC convention area overlaps the action area for the WCPO purse seine fishery from approximately 146°W to 150°W. As stated above, at its 72nd Meeting, June, 2004, the IATTC passed the *Consolidated Resolution of Bycatch* which includes a requirement for purse-seine vessels in the EPO to release unharmed, to the extent practicable, non-target species, with special requirements for releasing sea turtles.

In December, 2005, the WCPFC passed a similar resolution to reduce the frequency and severity of interactions with sea turtles in WCPO purse seine fisheries and urge Commission Members, Cooperating non-Members, and participating Territories (CCMs) to require purse seine vessels flying their flags to collect and provide information on sea turtle interactions and to take measures to reduce entanglement and injury and to increase survival of turtles captured incidental to purse seine activities. The measures contained in the WCPFC resolution were designed to be consistent with requirements of the IATTC sea turtle resolution and to align with the FAO Guidelines to Reduce Sea Turtle Mortality in Fishing Operations.

Memoranda of Understanding for Sea Turtle Conservation

Among the many initiatives, treaties and resolutions for sea turtle conservation in recent years, the Memorandum of Understanding (MoU) on the Conservation and Management of Marine Turtles and their Habitats of the Indian Ocean and South-East Asia (IOSEA) is relevant to this consultation. The IOSEA MoU puts a framework in place through which States of the Indian Ocean and South-East Asian region, as well as other concerned States, can work together to conserve and replenish depleted sea turtle populations for which they share responsibility. Collective implementation of an associated Conservation and Management Plan will result in the achievement of this conservation objective. The MoU applies to the waters and coastal States of the Indian Ocean and South-East Asia and adjacent seas, extending eastwards to the Torres Strait. For implementation purposes, the area is divided into four sub-regions: South-East Asia and Australia, Northern Indian Ocean, Northwestern Indian Ocean, and Western Indian Ocean. The species of marine turtles covered by the MoU are the loggerhead, olive ridley, green, hawksbill, leatherback, and flatback (*Natator depressus*). The IOSEA MoU's Conservation and Management Plan contains 24 programmes and 105 specific activities which focus on reducing

threats, conserving critical habitat, exchanging scientific data, increasing public awareness and participation, promoting regional cooperation, and seeking resources for implementation.

The MoU and associated Conservation and Management Plan were developed over a series of intergovernmental negotiation sessions held in Perth, Australia (October 1999), Kuantan, Malaysia (July 2000), and Manila, Philippines (June 2001). The MoU came into effect on September 1, 2001, and the Signatory States held their first meeting in Bangkok in January 2003.

Various international agreements and national regulations strive to conserve and protect marine turtles from excessive exploitation. However, the success of these initiatives depends on effective implementation of measures by a wide range of actors: governmental (at all levels), non-governmental (NGOs, civil society) and intergovernmental. In the face of other pressing development priorities, many countries lack the capacity and resources to undertake conservation measures for these species. This makes it all the more important to offer support, assistance and encouragement to build capacity among those who are the real custodians of these natural resources. The IOSEA MoU is developing a well-coordinated network of interested stakeholders, delivering a comprehensive programme of necessary interventions, and providing an inclusive forum for regular review of implementation progress.

8 Effects of the Action

Our assessment of the effects of the U.S. WCPO purse seine fishery on listed species focuses on aspects of the fishery which have been identified to have potential adverse effects on listed species occurring in the Action Area: vessel traffic; gear deployment and retrieval; entanglement in FADs; and removal of fish biomass from the pelagic ecosystem. In our assessment we assume that there is no risk to a listed species due to the proposed action if it is not exposed to one of these stressors. We first evaluate the available evidence to determine the likelihood of a listed species being exposed to direct effects of vessel strikes and interactions with the fishing gear and indirect effects from removal of pelagic biomass. After we determine exposure, we review the available scientific and commercial data to determine how the listed species is likely to respond given exposure to analyze the potential risk of the fishery on listed species.

8.1 Effects Analysis

The sections below describe how we considered various sources of information to determine the exposure, response, and risk of listed species to interactions with the fishery. Section 8.2 describes the information needed for a rigorous effects analysis. The subsequent sections explain the use of fishery observer data as a proxy for the data needed for a rigorous assessment and the assumptions required to interpret the observer data for this analysis. The general methodology in the following sections was to use observer data collected aboard U.S. purse seine vessels fishing in the WCPO to determine the rate and severity of interactions with threatened and endangered species. The sections below note the limitations and uncertainty with the U.S. observer data with regard to interactions with ESA listed species. The approach taken was to estimate all parameters based on the U.S. observer data. The U.S. data were deemed the most representative source of information regarding interactions for the U.S. fleet (NMFS, 2006). When there were “unknowns” in the U.S. data (e.g., documented interactions with no species identification), we relied on surrogates by reviewing published and unpublished literature. In all cases, we extracted the greatest amount of information and inferences possible from the U.S. WCPO observer data

before relying on surrogate sources. However, when there were unknowns in the U.S. data it was necessary to rely on surrogates.

8.1.1 Fishery Data Sources

The BA describes two data sources collected during the course of fishing activity that may be useful for estimating potential levels of listed species' exposure to the U.S. WCPO purse seine fishery: logsheets maintained by vessel operators and data recorded by on-board observers (NMFS, 2006).

8.1.1.1 Vessel Logbooks

Vessel operators are required to record daily catch and effort on FFA-provided logsheets, in the manner specified in the Treaty (RPL). RPLs provide the primary means of estimating effort in the fishery (in terms of both days-at-sea and sets), including the types of sets made (e.g., whether or not made on floating objects). The RPLs are also used to estimate catch-by-species. Although discards are also recorded on the RPLs, the data are typically not useful for estimating rates or consequences of interactions with marine mammals or sea turtles.

Once collected, the RPLs are transmitted to the FFA, which in turn transmits them to the Oceanic Fisheries Programme of the SPC. Under the terms and conditions of the Treaty, once the RPLs or any other data related to the Treaty are transmitted to the FFA, the U.S. no longer has the authority to release this information, absent enforcement intervention. However, NMFS is generally able to obtain particular data by way of case-by-case requests to the SPC, via the FFA.

8.1.1.2 Observer Program

The SPTT contains provisions to maintain compliance by targeting 20% of trips by the U.S. fleet with observer coverage. Each vessel is targeted to carry an observer on one trip per year. Within the constraints of those targets and with the aim of providing appropriate advance notice to vessel operators, the FFA places observers on vessels opportunistically and at its own discretion.

Although the observer program is operated by the FFA Secretariat on behalf of the PICs, the U.S. tuna industry is responsible for meeting the full costs of the program, including fixed and observer training costs. Beyond the basic objectives for the observer program and the target coverage rate, as established in the Treaty, the United States does not have any control over the operation of the observer program. The United States has an opportunity to review the performance of the program through a consultative process established under the Treaty, particularly the informal consultations among the parties that take place annually. NMFS, particularly through its field office in American Samoa, provides logistical, training, and other support to the FFA observer program, and also facilitates the placement of observers.

The FFA observer program has a significant compliance aspect to it. As such, collection of scientific information, especially information related to species of special interest (e.g., species that are listed under the U.S. ESA), has been an ancillary duty of the observers. Data collected by observers includes catch composition by species, fish sizes, fishing effort, location, environmental conditions, gear type, and information on bycatch and discards. The completeness and quality of the data with respect to interactions with species of special interest has changed over the course of the program.

Once collected by FFA-deployed observers, the observer data are transmitted to, and held at, the Oceanic Fisheries Programme of the SPC. Under the Treaty, the U.S. does not have any particular rights to the data, but NMFS obtains particular sets of data or data summaries by way of case-by-case requests to the SPC, via the FFA.

The vessel observers record information on various forms contained in an observer workbook, which is developed jointly by SPC and FFA and deployed as standard reporting by FFA (SPC and FFA, no date). FFA-deployed observers collect a range of information on the forms, as well as provide a narrative log of events and activities. A daily log form records time and position of the vessel, including fish school sightings and whether the school was unassociated (free swimming or feeding on baitfish) or associated with floating objects (FADs, logs, flotsam, or dead animals) or other unspecified objects. Details for each set are recorded on a separate form, including information on interactions with, capture, and fate of protected species such as marine mammals, sea turtles, and birds. Observers are instructed to complete an additional form for all sightings, interactions and/or capture of all species of special interest, which includes marine mammals, turtles, birds, and whale sharks. Observers also keep a vessel trip compliance record, which includes a section for recording whether the vessel deliberately attempted to make a set on marine mammals or other species of special interest. A written report describing the fate and condition of the animal(s) and the vessels' attitude concerning the animal(s) is also recorded.

8.1.2 Use of Data in the Effects Analysis

The BA provides the available data on observed interactions between the fishery and marine mammals and sea turtles from 1997 through 2004 (NMFS, 2006). The level of fishing effort (in sets) and observer coverage in the U.S. WCPO purse seine fishery from 1997 through 2004 is shown in Table 24.

Table 24. Total effort and observed effort in the U.S. WCPO purse seine fishery, 1997-2004. Sources: ¹FFA unpublished data; ²NMFS (2004a); ³SPC (2005).

Year	Total effort (sets)	Observed effort (sets)	Observer coverage (percent; by sets)
Jan 1997 – Jun 2002	24,166 ²	6,058 ²	25.1
2003	3,204 ¹	698 ^{1,3}	21.8
2004	2,656 ¹	801 ^{1,3}	30.2

We evaluated these data for the effects analysis and identified several limitations in the available observer data for estimating potential levels of exposure between listed species and the fishery. Here we provide a summary of the available observer data and discuss how the data were used in the analysis and where limitations exist with regard to drawing inferences about future patterns of interactions between listed species and the fishery.

There have been some reports compiled on the impacts of the WCPO purse seine fishery on non-target species such as turtles, whales, and sharks (Molony, 2005; SPC, 2001, and Bailey et al., 1996). These reports as well as a report by Lawson and Williams (2005), which extrapolate the WCPO purse seine observer data to the entire fleet, discuss the uses and limitations of the observer data. In these reports the data are aggregated among vessels from all flag areas, therefore these limitations are in reference to the observer data in general and not the U.S. fleet

in particular; however, we have found these limitations to be highly relevant when examining the available data from the U.S. fleet as well. One distinguishing condition is the percent of observer coverage which has been significantly higher in the U.S. fleet (typically greater than 20%) compared to the aggregated fleet (1994-2004 fleet-wide average of approximately 4%) (Molony, 2005).

A common concern raised in the bycatch summary reports is that the purpose of the observer program is to document operational compliance and record tuna catch composition data. Information on protected species interactions, such as turtle and whale species listed under the U.S. ESA have been collected ancillary to other objectives of the program. In recent years, increasing emphasis has been placed on documenting interactions with marine mammals, turtles and sharks, however, the reliability and accuracy of the data have not been assessed. Lawson and Williams (2005) raise several issues about the reliability and accuracy of the observer data in a comparison of catch composition estimates from SPC housed landings data and the observer data. Two conclusions drawn by Lawson and Williams (2005) are the following:

1. The observer data should be considered accurate and reliable because the observers are well-trained technicians. However, the basic level of skill is quite low compared to observers in other areas, such as the Eastern Pacific, and many observers in the WCPO have not undergone significant debriefing to improve their skills.
2. The observer data have been evaluated and screened for data quality, so they should be considered accurate and reliable. However, certain elements of the evaluation of data quality are highly subjective.

Whereas we recognize the high level of observer coverage in the U.S. fleet relative to other areas and the rest of the WCPO purse seine fleet, we are concerned about the veracity and thoroughness of the data due to cautionary statements such as the ones above and in Molony (2005) and Bailey et al. (1996) with respect to the degree that protected species interactions are either under-reported or inadequately reported (e.g., condition information absent; not high priority of observer program; etc.). Nonetheless, there has been a relatively high level of coverage in the fishery and it is recognized that the importance of documenting interactions between the U.S. fishery and ESA listed species has become a greater priority in recent years and training has been bolstered on these topics. Therefore, we assume that the available U.S. observer data are accurate and reliable.

The observer data provided for this analysis lack key variables and details to confidently extrapolate the observed interaction rates to the fleet-wide total. Of the various reports that have examined observer data from the WCPO purse seine fishery, all have indicated the importance of set-type as the most important factor by which catch data (for fish and protected species) should be stratified (Lawson and Williams, 2005; Molony, 2005; SPC, 2001; and Bailey et al., 1996). Marine mammal and sea turtle interactions in purse seine fisheries are disproportionate among set types (Molony, 2005; SPC, 2001; and Bailey et al., 1996). Interactions with marine mammals and sea turtles are disproportionately higher in sets associated with floating objects (associated sets) than in free swimming schools (unassociated sets). These reports have also indicated that the portion of set types observed is highly variable among years. To extrapolate the observed proportion to the fleet-wide total it is important to know the interaction rate by set type and the

proportion of observed set types. These data were not available for this analysis. For this analysis we assume that the proportion of observed sets by type in the U.S. fleet is representative of future fleet activity.²⁰

Moreover, the spatial distribution of the observer coverage in relation to the fleet is an important consideration when extrapolating the observer data. The majority of interactions between both marine mammals and turtles occur in the western tropical WCPO (Molony, 2005). Molony (2005) notes the disproportionately higher number of sea turtle and marine mammal interactions west of 170° W. Most turtle interactions reported in the WCPO occur within and around the EEZs of PNG, the Federate States of Micronesia, the Solomon Islands, and Nauru. Data on the spatial distribution of the interactions or of observed sets in relation to the annual fleet-wide fishing effort were not available for this analysis. However, the distribution of purse seine effort in the WCPO is broadly similar to the distribution of turtles captured by the purse seine fleet (Molony, 2005). For this analysis we assume that spatial patterns implicit in the observer data for the U.S. fleet, while not explicitly explained in the BA, are representative of the fleet and the distribution of future interactions.

Another significant limitation of the observer data is the resolution with which marine mammal and sea turtle interactions have been documented. Since 1998, only 3 of 10 sea turtles interacting with the purse seine fishery have been identified to species. The frequency with which individual marine mammal species interact with the fishery is even more uncertain. Of 13 sets with documented marine mammal interactions from 1997-2004, only 1 contains information on the species involved. To supplement the observer data on species which may be potentially involved in interactions with the fishery, we reviewed the available literature to determine the distribution of each listed species identified in the BA (NMFS, 2006) and reviewed information on marine mammal and turtle interactions in the entire WCPO fleet (Molony, 2005; Bailey et al., 1996).

Interaction rates between fisheries and marine mammals in a particular area will vary significantly by season. Spatial/temporal information was not available for observed sets with marine mammal interactions in the U.S. WCPO purse seine fishery.

Despite the limitations mentioned above, the observer data from the U.S. WCPO fleet comprise the best available data on marine mammal and sea turtle interactions in this fishery (NMFS, 2006). These limitations were highlighted to denote that the aspect of the analysis pertaining to species identification relies on data that are highly uncertain and in the case of marine mammals, incomplete. We draw inferences from the available information to estimate exposure, yet note that the effects analysis weights the response of the animal with greater influence than the number of animals exposed by life stage and population. Modifications to the observer program are needed for collection of detailed data on interactions between the WCPO purse seine fishery and marine mammals to further assess fishery impacts on various populations and species (Molony, 2005; Bailey et al., 1996).

²⁰ The assumption is deemed to be precautionary as the reliance on unassociated-sets, which were found to have the lowest bycatch rate of all purse seine set-types (SPC, 2001), is expected to continue to increase in the U.S. fleet (see Section 3.3).

8.2 Exposure Analysis

As discussed in the Approach to the Assessment section of this Opinion, exposure analyses are designed to identify the listed species that are likely to co-occur with these effects in space and time and the nature of that co-occurrence. In this step of our analyses, we try to identify the number and age (or life stage) of the individuals that are likely to be exposed to an Action's effects and the populations or subpopulations those individuals represent. We assume that the consequences of exposure to the stressors of the purse seine fishery are in part a function of the intensity, duration, and frequency of exposure as well as a function of the species response to the exposure.

A rigorous assessment of exposure requires information on the spatial and temporal distribution of the fishery and listed species to determine the extent of overlap. Information on the listed species spatial/temporal distribution by life stage, behavior (diving, foraging, and associations with pelagic features and oceanographic fronts), and spatial/temporal distribution of fishing activities are needed to thoroughly assess overlap and estimate exposure by species and life stage. This information is not usually available for large scale fishing actions such as the one evaluated in this opinion and exposure estimates are often based on monitoring data collected by observers during the course of fishing activity.

Table 25 lists the types of information needed for a robust exposure analysis and the information sources we relied upon in the absence of this complete information. For this analysis we first examined the available U.S. observer data (summarized by the IFP in the BA, NMFS, 2006). When we encountered roadblocks due to uncertainty we reviewed fishery technical reports (Molony, 2005; Bailey et al., 1996); published and unpublished scientific information on listed species distribution and abundance, habitat associations, and behavior; and other biological opinions (e.g., NMFS, 1999, etc.) to estimate probable levels of exposure of listed species to the U.S. WCPO purse seine fishery.

Table 25. Information needed to conduct a robust exposure analysis and the type of information used as a proxy in the absence of complete data for this analysis.

Data needed for Rigorous Exposure Assessment		Proxy
Listed Species Distribution and Abundance Surveys	Season	<ul style="list-style-type: none"> • Literature on distribution, habitat associations, and behavior • Surveys conducted for other studies • Other fishery monitoring data
	Life-Stage	
Fishery Monitoring Data	Full documentation of the interaction ¹	<ul style="list-style-type: none"> • Available observer data documenting previous occurrences of interactions • Anecdotal data • Inferences
	Random and representative sampling	

¹ Including identification to species level; purse seine set type; number of interactions by species; the location of the interaction; and the disposition of the animal.

8.2.1 Exposure of Marine Mammals to Encirclement/Capture in the Fishery

As shown in Table 26, 25% of U.S. purse seine sets were observed from 1997 - 2002 and 20-30% of the U.S. WCPO purse seine sets were observed in 2003 and 2004. The available data indicate that only 0.18% (or one in 551 sets) of the sets from 1997 - 2002 resulted in interactions with marine mammals. More important than the rate of interactions is the number of animals likely to be affected by the fishery. The number of individuals likely to be impacted by the fishery is the base unit for our effects analysis.

Over 7,557 purse seine sets were observed in the U.S. WCPO fleet from 1997-2004 and only 13 sets were documented as having interactions with marine mammals (Table 26). Because six of these sets were recorded as dolphin interactions, and no dolphin species are listed under the ESA, we assume that at most, only seven observed sets from 1997 - 2004 may have involved interactions with listed marine mammals. Of these seven sets, four were listed as interactions with whales and three were just identified as marine mammals. Therefore, giving the benefit of doubt to the species, we surmise that between four and seven of the 7,557 sets may have involved listed whales according to the available observer data.

Of the sets with marine mammal interactions, four involved interactions with more than one animal. We do not know how many animals were involved, which marine mammal species were affected, nor if these were the sets recorded as dolphin sets. Due to the large size of most of the listed whale species, it is unlikely that any multiple large whales were involved in any one of these sets. Data from the entire fleet indicate that the bulk of marine mammal interactions identified to species were either common or bottlenose dolphins. There have been no documented interactions between listed marine mammals and the U.S. WCPO purse seine fleet,

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or in the entire fleet (Molony, 2005). However there have been interactions between unidentified marine mammals and unidentified whales and Bailey et al. (1996) note that baleen whales, most commonly sei whales, are occasionally encircled during purse seine operations on tuna schools that are usually feeding on pelagic baitfish. To be precautionary, we assume that the unidentified interactions were all interactions with listed marine mammals to evaluate the worst case scenario of the fishery on the populations of listed whales.

The available observer data on marine mammal interactions in the U.S. WCPO purse seine fishery were provided in the BA (NMFS, 2006) and are summarized Table 26. Table 26 highlights the limitations to extrapolating the existing observer data.

Despite these limitations, the available observer data were used to estimate the number of sets in the U.S. WCPO purse seine fleet likely to result in interactions with marine mammals in the future based on interaction rates from 1997-2004; 2003; 2004; and 2003 and 2004 combined; based on the projected number of sets likely to occur in the fishery in the future based on NMFS (2006). These values were rounded up to the nearest integer and are shown in Table 27. Table 27 shows that based on the rudimentary procedure used to expand the data to the entire fleet, the number of sets with expected interactions with listed marine mammals in the U.S. WCPO purse seine fleet is variable from year to year. To be precautionary, we used the interaction rate from 2003, which results in an estimated three sets with listed marine mammal interactions in the U.S. WCPO purse seine fleet in future years (Table 27). Data are not available to indicate how many listed individuals may be involved in these interactions or to indicate which species may be involved. However, given the size of the listed whales considered in this Opinion, a purse seine net is not likely to encircle more than one or two listed whales in any set. To be precautionary, we assume that these interactions may involve any of the listed whale species considered in this Opinion (sperm, blue, sei, humpback or fin whales).

Table 26. Summary of observed marine mammal interactions in the U.S. WCPO purse seine fishery from 1997-2004 (Data source: NMFS, 2006).

Year(s)	n Observed Sets	Total Sets	Observed Sets with Interactions (s _i)	Identification ^A					Number of Animals/Event ^A			Potential ESA Listed MM Events Only ^{A,B}		
				Marine Mammal	Whale	Unidentified	Dolphin*	Risso's Dolphins*	0	1	>1	Potential Mortality	Interaction Rate/1,000 Sets (r) ^C	
1997 - 2002	6,058	24,166	11	2	4		5			1	6	4	4	0.99
2003	698	3,204	1			1						1	0	1.43
2004	801	2,656	1					1				1	0	0.00
2003 and 2004	1,499	5,860	2			1		1				2	0	0.67

* Not listed under the ESA

^A Refers to number of sets (events) not number of individuals

^B Includes only interactions documented as marine mammals, whales, or unidentified

^C $r = \left[\sum_{i=1}^n s_i \div n \right] \cdot 1000$; where r = the interaction rate per 1,000 sets; s_i = the number of observed sets with interactions; and n = the number of observed sets

Table 27. Number of U.S. WCPO purse seine sets expected to interact with listed marine mammals each year based on past, observed fishery interaction rates and future effort projected at 2,400 sets per year based on NMFS (2006). Note these numbers are regarding the number of sets with interactions, not the number of individuals likely to interact with the fishery. Number of interactions has been rounded up to the nearest whole number.

Year(s)	Interaction Rate	Projected Interactions/Year
1997-2002	0.99	2
2003	1.43	3
2004	0.00	0
2003 and 2004	0.67	2

8.2.2 Exposure of Sea Turtles to Encirclement/Capture in the Fishery

To assess exposure of sea turtle species to the U.S. WCPO purse seine fishery, we rely on the vessel observer data and published and unpublished accounts of sea turtle distribution by species. The first interaction between the U.S. purse seine fishery and sea turtles was documented in 1998. Ten sea turtle interactions have been documented by observers from 1998 - 2004 (Table 28). Interactions identified to the species level have been between hawksbill and green turtles only. However, as noted in the BA, previously, only three reporting codes were recorded in the FFA/SPC database: unidentified to species level; hawksbill; and green. Further complicating estimation of species composition of turtles encountered in the fishery are logistical difficulties noted in SPC (2001). Though observers are usually in a good position to observe the early stages of the set and see whether turtles have been released, it is difficult for the observer to identify sea turtles when they are encircled in the net but released prior to brailing (SPC, 2001).

As mentioned in Section 8.1, we extracted all of the information from the U.S. WCPO purse seine observer data to determine patterns of listed species exposure to the fishery; when there were gaps in the U.S. observer data (e.g., interactions not identified to species); we relied on surrogates to surmise probable values. To determine the probable species comprising the “unidentified” interactions, we reviewed distribution and migration information for the various sea turtles species and observer data from the entire WCPO purse seine fleet (Molony, 2005). Of 104 interactions between the entire WCPO purse seine fleet between 1994 and 2004, 80 were unidentified to species. Of the identified species, ten were olive ridleys, eight were hawksbills, five were greens, and one was a leatherback. Given the spatial extent of the fishery, it is expected that the fishery may interact with these species given their migrations and the proximity of their nesting grounds. Sea turtle interactions with the purse seine fishery may occur during the period when young turtles are in the open ocean, drifting with or without debris and prior to association with inshore feeding grounds, as non-breeding adults, and sexually mature adults migrating to and from nesting/breeding sites.

We reviewed available literature to determine the probability of a loggerhead turtle interacting with the U.S. WCPO purse seine fishery. The action area for the WCPO purse seine fishery is noted as being just south of the migration route for loggerheads originating from Japan nesting

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beaches and just north of the migration route for loggerheads of Australian nesting beach origin (Limpus and Limpus, 2003). Loggerhead turtles may occur in the Action Area and thus may be exposed to interactions with the purse seine fishery; however, the frequency of their occurrence in the action area is probably much lower than the other species with documented interactions in the fishery. Loggerheads may nest at Tokelau or Vanuatu which would make them more susceptible to the fishery than is currently anticipated (Limpus and Limpus, 2003). We reviewed available satellite tracking data for loggerheads, available on internet websites such as www.seaturtle.org, and found that none of the migration tracks overlapped with the action area. The life-stages of loggerheads that may occur in the Action Area would likely be sub-adults or adults migrating to and from their nesting/breeding grounds.

Table 28. Sea turtle interactions reported by FFA deployed observer on U.S. purse seine vessels operating under the South Pacific Tuna Treaty, January 1998 through December 2004. A1 = released alive and healthy. Source: Secretariat of the Pacific Community Oceanic Fisheries Programme, unpublished data.

Date	Time	Species	Condition
16-Sep-98	1130	Unidentified	A1
28-Jun-01	1100	Unidentified	A1
21-Apr-02	1309	Unidentified	A1
15-May-02	0542	Unidentified	A1
26-Aug-02	1229	Hawksbill	A1
24-Apr-03	0547	Unidentified	A1
11-Jun-03	0505	Green	A1
06-Oct-03	0503	Unidentified	A1
26-Dec-03	0443	Unidentified	A1
24-Dec-03	0434	Hawksbill	A1

The available observer data indicate that interactions between the U.S. WCPO purse seine fleet and sea turtles are rare and sporadic occurrences (Figure 15). On average, roughly 25% of the U.S. fleet has been observed from 1998 through 2004 and only 10 interactions with sea turtles have been observed. We know for sure that these interactions involved at least two species (green and hawksbill), and may have involved all five. Additionally, there may be unobserved interactions occurring with all species, as roughly 75% of the sets were unobserved.

The numerous, varied methods to expand the observer data to estimate the proportion of interactions that may have involved the various species and the total number of interactions from the unobserved sets, are all associated with a high amount of uncertainty. Estimates about the total number of interactions in the fleet and the total number of each turtle species likely to be exposed to the fishery are subject to sampling and process error. Moreover, as shown in Figure 15, the number of interactions observed in the fishery is highly variable from year to year. Five of the ten observed interactions from 1998 - 2004 were observed in 2003. In 2004, the year with highest percentage of observer coverage to date, there were no observed sea turtle interactions. As stated above, sea turtle interactions appear to be rare, patchy events in this fishery.

Many factors likely influence the degree to which the fishery and sea turtles overlap. Factors affecting the distribution of fishing effort (e.g., ENSO events) and the distribution and abundance of sea turtles (currents, fronts, re-migration intervals, etc.) would all contribute to variable

degrees of overlap between the two. Moreover, the distribution of observer coverage in space, time, and by purse seine set types may also have a large influence on the number of observed turtle interactions. As stated above, the information to tease out these relationships and draw conclusions about these sources of variability are not available. However, the occurrence of observed sea turtle interactions in the fishery is so rare that perhaps, even if the data were available it may be difficult to determine the relationships and decrease the uncertainty and sources of error about the estimates.

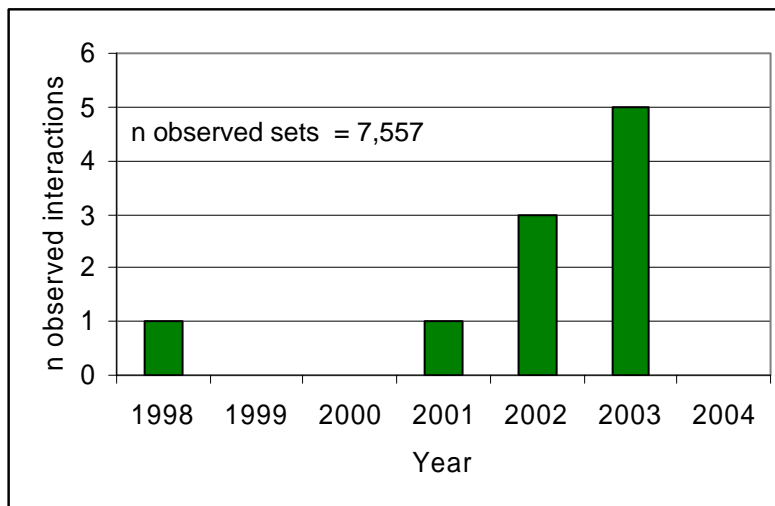


Figure 15. Number of sea turtle interactions observed in the U.S. WCPO purse seine fishery from 1998-2004. These numbers have been aggregated across all species. From 1997-2002, 25.1% of the effort was observed. In 2003 and 2004 there was 21.8% and 30.2% observer coverage respectively.

Given the caveats stated above, we evaluated rudimentary extrapolations of the observer data to the total expected effort in the fishery based on observed sea turtle/fishery interaction rates from 2003 and 2004 to estimate the anticipated number of sea turtles that may be exposed to the U.S. WCPO purse seine fishery (Table 28). Observer data from 2003 and 2004 are the most reliable for evaluating sea turtle interaction rates based on recent improvements in the observer program for collecting information on sea turtle interactions (NMFS, 2006).

The number of observed sea turtle interactions in the U.S. WCPO purse seine fishery varies among years (Table 29). There were more sea turtle interactions in 2003 than in any other year in the time series of available data. To be conservative, interaction rates from 2003 were used to estimate future exposure of each turtle species to the U.S. WCPO purse seine fishery. A fishery interaction rate was calculated for each turtle species based on observations from 2003. In 2003, a total of five sea turtles were observed in the U.S. fishery and were recorded as one green, one hawksbill, and three which were not identified. We assumed the worst case scenario for each species and estimate that four of the interactions may have been with green or hawksbill turtles and a maximum of the observed interactions may have been with three leatherback, olive ridley, and loggerhead turtles according to the available data (Table 29). There was only one turtle in each of the documented interactions. Because only a fraction of the purse seine sets were observed, we calculated an interaction rate based on the number of interactions divided by the number of observed sets. The binomial variance function was used to calculate confidence

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intervals for the proportion of purse seine sets with interactions for each species according to the methods in Thompson (1992).

Table 29. Summary of U.S. WCPO purse seine fishery observed sea turtle interactions in 2003 and 2004. Observer data source: Secretariat of the Pacific Community Oceanic Fisheries Program, unpublished data.

Year	Observed Sets	Total Sets	Observed Turtle Interactions			
			Unidentified	Hawksbill	Green	Total
2003	698	3,204	3	1	1	5
2004	801	2,656	0	0	0	0
2003 and 2004 Combined	1,499	5,860	3	1	1	5

The future number of interactions likely to occur in the fishery based on the 2003 estimated interaction rate was multiplied by the future number of sets anticipated in NMFS (2006) to estimate the number of each species predicted be exposed to the U.S. fleet in subsequent years (Table 30). A 95% prediction interval was calculated for each future estimate and is shown in Table 30.

Table 30. Results from the sea turtle exposure analysis. Number of interactions predicted to occur between each species and the U.S. WCPO purse seine fishery based on the observed interaction rate from 2003.

Species	Interactions ^a	Interaction Rate (per 100 sets) ^b	Predicted Future Interactions ^c	95% Prediction Interval	
				Lower	Upper
Green	4	0.57	14	1	28
Hawksbill	4	0.57	14	1	28
Olive Ridley	3	0.43	11	0	22
Leatherback	3	0.43	11	0	22
Loggerhead	3	0.43	11	0	22

^a Plausible, estimated number of fishery interactions in 2003 based on five observed interactions comprised of one green, one hawksbill and three unidentified turtles.

^b Interaction rate based on 2003 observer data; given the plausible, estimated number of interactions for each species.

^c Number of interactions predicted in future years in the fishery based on the annual observed interaction rate multiplied by the number of projected sets (n = 2,400) (NMFS, 2006) rounded up to the nearest integer.

In conclusion, we note that there have been documented interactions between sea turtles and the U.S. WCPO purse seine fleet. With relatively high observer coverage, these interactions appear to be rare and sporadic occurrences. Interactions have been documented with green and hawksbill turtles and with unidentified turtles which may be leatherbacks, loggerheads or olive ridley turtles. Olive ridley and leatherback turtles have been documented as interacting with the foreign WCPO purse seine fleet. These species' adult migration routes appear to directly overlap the Action Area. While loggerhead turtles equipped with satellite tags have not been observed in the Action Area, loggerhead turtles from eastern Australia nesting beaches may occur in the Action Area and may occasionally interact with the U.S. WCPO purse seine fishery.

Because the species identification could not be determined for 3 of the 5 interactions documented in 2003, we analyzed a worse case scenario and estimated that annually, 14 green or hawksbills and 11 olive ridleys, leatherbacks, or loggerheads may interact with the U.S. WCPO purse seine fishery. These estimates are based on the highest number of interactions observed in the fishery in the past eight years and interactions not identified to species have been attributed to all five species. It is unlikely that the number of interactions estimated for each species would occur in all subsequent years and it is even more unlikely that the total of the estimates for all species would occur in a single year. However, using the highest point estimate based on existing data seems to be a reasonable approach to estimating a precautionary number of annual interactions between the fishery and sea turtles.

8.3 Response Analysis

As discussed in the Assessment Approach, once we have identified which listed species are likely to be exposed to the U.S. WCPO purse seine fishery, we conduct response analyses to identify how listed species are likely to respond once exposed to the fishery.

The U.S. WCPO purse seine fishery poses direct impacts on whales and sea turtles through vessel collisions and encirclement and/or capture in the purse seine during deployment and retrieval. We also considered potential indirect effects to listed species resulting from changes in the trophic structure of the system due to removal of fish biomass.

8.3.1 Response of Whales Given Exposure

8.3.1.1 Response of Whales to Vessel Collisions

The WCPO purse seine fishery vessel activity poses direct impacts on whales through potential collisions. Injuries and deaths resulting from ship collisions with whales remain a significant threat worldwide (Jensen and Silber, 2003). A review of ship strike records worldwide from 1975 to 2002 documents a total of 292 records of confirmed or possible ship strikes to large whales (Jensen and Silber, 2003). Of the 11 confirmed species that were victims of ship strikes worldwide, five of the species are found in the WCPO Action Area: blue, fin, humpback, sei, and sperm whales (Jensen and Silber, 2003). Worldwide, fin whales are the most often reported species hit (75 records of strike), followed by humpback (44 records), and sperm whales (17 records) (Jensen and Silber, 2003). Far fewer reports exist of strikes to blue (eight records), and sei whales (three records).

Of these 292 ship strike records, large whale ship strikes were recorded in waters nearby to the Action Area including Japan, Australia, and New Zealand; no strikes, however, were specifically recorded in the Action Area (Jensen and Silber, 2003). However, this is not unexpected given the low likelihood of a collision being reported.

In addition to no recorded ship strikes specifically in the Action Area, recent U.S. WCPO observer data show that interactions with large whales, including listed whales, are relatively uncommon in both the Action Area and throughout the Pacific Ocean in general. Despite the low probability of potential interactions in the Action Area, a collision between a vessel and whale cannot be ruled out. Of the total 292 worldwide large whale ship strike reports, 48 (16.4%) resulted in injury to the animal and 198 (68.0%) were fatal (Jensen and Silber, 2003). Of the 134 cases of known vessel type, fishing vessels were responsible for four records (3.0%) of strike (Jensen and Silber, 2003).

We have determined that the probability of collisions with all five listed whale species (blue, fin, humpback, sei, and sperm) is low in the Action Area, however, should a collision occur, direct impacts are likely to be beyond the level of a behavioral disturbance. Impacts from a vessel collision are likely to be either a sub-lethal response of stranding, or a lethal response of death.

8.3.1.2 Response of Whales to Net Deployment and Retrieval

Purse seine net deployment and retrieval poses a direct impact to whales, including listed whales, resulting in possible encirclement or capture. Larger cetaceans, because of their strength and size, are able to break through the net and escape, an ability indicated in U.S. WCPO tuna purse seine fishery observer data. According to Bailey et al. (1996), baleen whales, most commonly the sei whale, were occasionally encircled during purse seine operations on tuna schools that are usually feeding on pelagic baitfish. When this occurred, it was observed that the whales generally punched through the net, usually close to the surface, or were aided in their release by submerging a portion of the corkline. It has been documented in some cases that whales returned to feeding after being set on (D.G. Itano personal observation), and this was interpreted by Bailey et al. (1996) that the whales' encounters with the purse seine operation were not overly traumatic.

There is also evidence from the ETPO that encircled and captured whales escape tuna purse seine nets (NMFS, 1999). NOAA Fisheries Stock Assessment Reports document that fishermen have reported that large whales tend to swim through their nets without entangling and causing little damage to nets (Barlow et al. 1997). In 1997, the ETPO 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 had 100% 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 is considered to be extremely low.

Based on the available data, stressors resulting from accidental encirclement in the U.S. WCPO purse seine fishery on large whales are expected to result in short-term behavioral alterations in the form of displays, avoidance, or abandonment based on observed cases of large whales returning to normal feeding post purse seine net interaction (Bailey et al., 1996). Potential effects

from encirclement are likely to be short in duration and not likely to result in alteration of the species vital rates such as decreased survival, delayed reproduction, or reduced growth rates, or long-term abandonment of territory. Short-term interruptions to feeding may occur, however, the low frequency and intensity and short duration of these interactions are not likely to result in measurable changes to the species energy budgets (e.g., time spent feeding, resting, searching for food, etc.).

8.3.1.3 Response of Whales to Removal of Fish Biomass

Although direct competition between fisheries and marine mammals for prey appears to be limited, there may be considerable indirect competition for primary production (Trites et al., 1997). Trites et al. (1997) stated that $\geq 60\%$ of the food caught by marine mammals consisted of deep sea squids and very small deep sea fishes not harvestable by humans, thus limiting the extent of direct competition between fisheries and marine mammals. Young (2000) stated that differences in feeding behavior and migration patterns largely preclude direct competition between whales and pelagic fisheries in the South Pacific Ocean, which includes the Action Area. This is due, in particular, to the fact that many of the whales in the region obtain an estimated 90% of their annual consumption of food from Antarctic waters in the summer months (Young, 2000). Large whales that migrate to the South Pacific Ocean, including the Action Area, are there to breed and conceive their young (Lockyer, 1984). For those species that do not migrate from tropical/subtropical South Pacific waters, such as female sperm whales, consumption of food occurs at depths greater than that usually occupied by species that are the targets of commercial fisheries, such as the tunas (Young, 2000).

In essence, despite high biomass consumption from whales, dietary overlap with commercially fished species is relatively low since much of their feeding is in waters that are not exploited by fisheries (Young, 2000). In a simplified trophic model created by Trites et al. (1997), marine mammals and fisheries are depicted as generally occupying different trophic niches; the groups overlap both spatially and temporally, but because of their different feeding strategies they rarely compete for the same food (Trites et al., 1997). It should be noted, however, that there is a lack of qualitative and quantitative data on the specific trophic interaction with food competition between whale species and fisheries. Based on the best available information, NMFS concludes that the U.S. WCPO purse seine fishery is not likely to impact the energy budget of large whales to the degree that vital rates would be altered through competition by removal of fish biomass.

8.3.2 Response of Sea Turtles Given Exposure

8.3.2.1 Response of Sea Turtles to Vessel Collisions

Sea turtles can be injured or killed when struck by a vessel, especially an engaged propeller, and thus vessel collisions pose a direct impact. However, according to recovery plans for each of the U.S. Pacific populations of sea turtles, relative to other threats, vessel collisions are not considered a current problem. The extent of the threat of vessel collisions in the Pacific is unknown, but is presumably inconsequential on an individual level for all five listed species. Exceptions include the green and hawksbill turtle in Hawai'i where serious injuries and mortalities have been documented, and for green turtles possibly in the Republic of Palau where high speed skiffs travel frequently throughout the lagoon south of the main islands (Balazs et al., 1994). The frequency of vessel collisions with turtles is presumed to be low; however the

intensity of a vessel collision is expected to be quite high, likely resulting in reduced fitness and potentially death.

8.3.2.2 Response of Sea Turtles to Net Deployment and Retrieval

Interactions with sea turtles have been documented in WCPO purse seine fisheries, including the U.S. fishery. Sea turtle encounters in purse seine fisheries occur when turtles are within the pursed net after the operation of encircling a school of tuna. Records of turtle interactions observed in the U.S. WCPO purse seine fishery indicate that 100% of the turtles that were encircled in the purse seine net were released alive with no gear attached. These data indicate that the threat of mortality or physical injury resulting from the U.S. purse seine fishery is low. The intensity of the interactions is deemed to be low as turtles captured in purse seine nets are not entangled in the net and do not spend a lot of time encircled before being released. Captured and released turtles might suffer sub-lethal adverse effects. A qualitative description of such possible effects follows.

Some turtles entangled in purse seine nets may be forcibly submerged but emerge before drowning. Studies have shown that sea turtles that are allowed time to stabilize after being forcibly submerged have a higher survival rate. However, this depends on the physiological condition of the turtle (e.g., overall health, age, size), time of last breath, time of submergence, environmental conditions (e.g., sea surface temperature, wave action, etc.), and the nature of any sustained injuries at the time of submergence (NMFS, 2004b).

Sea turtles that are 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, indicating that turtles are probably more susceptible to lethal metabolic acidosis if they experience multiple captures in a short period of time, because they would not have had time to process lactic acid loads (NMFS, 2004b).

Respiratory and metabolic stress due to forcible submergence is also correlated with additional factors such as size and activity of the sea turtle, water temperature, and biological and behavioral differences between species and will therefore also affect the survivability. For example, large sea turtles are capable of longer voluntary dives than small turtles; therefore juveniles may be more vulnerable to the stress of forced submergence than adults. During the warmer months, regular metabolic rates are higher, so the impacts of the stress due to entanglement may also be magnified (e.g., Gregory et al., 1996, in NMFS, 2004b).

The condition of most sea turtles encircled in other fleets in the WCPO purse seine fishery is recorded as “unknown.” Of 104 observed turtle interactions in the WCPO fleet from 1995-2004, the condition was recorded for 29 turtles. Of these 29, 25 were recorded as alive and in good condition and 4 were recorded as dead upon time of capture; one was recorded as “gear damaged” (Molony, 2005). SPC (2001) cautions that it is not possible to draw conclusions about the condition of documented turtle interactions in WCPO purse seine fisheries due to the high percentage of turtles documented as released in “unknown” condition (69% of the observations). However, the condition was recorded for all turtles captured in the U.S. fleet from 1998-2004 and 100% of the turtles were documented as being released alive and healthy (NMFS, 2006).

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The SPC (2001) acknowledges that in most cases, turtles are encountered alive in the net and are subsequently scooped up and released over the side (SPC, 2001). There is some motivation by the crew to identify and then release turtles found in the net before the net is hauled through the power blocks, as this can cause damage to the gear.

Mortalities, when they occur, are likely due to drowning as a result of entanglement in the net or, in rare instances, to being crushed during the process of loading the net on-board (SPC, 2001). Drowning may be either “wet” or “dry”. In the case of dry drowning, a reflex spasm seals the lungs from both air and water. With wet drowning, water enters the lungs, causing damage to the organs and/or causing asphyxiation, eventually leading to death. Sea turtles may become comatose or unconscious before dying as a result of drowning.

Evidence from the EPO suggests that sea turtles also exhibit dynamic endocrine responses to stress. In male vertebrates, androgen and glucocorticoid hormones (cortcosterone (CORT) in reptiles) can mediate physiological and behavioral responses to various stimuli that influence both the success and costs of reproduction. Typically, the glucocorticoid hormones increase in response to a stressor in the environment, including interaction with fishing gear. “During reproduction, elevated circulating CORT levels in response to a stressor can inhibit synthesis of testosterone or other hormones mediating reproduction, thus leading to a disruption in the physiology or behavior underlying male reproductive success” (NMFS, 2004b).

Female green turtles have also been studied to evaluate their stress response to capture/restraint. Studies showed that during the breeding season, female green turtles exhibited a limited adrenocortical stress response when exposed to ecological stressors and when captured and restrained. Researchers speculate that the apparent adrenocortical modulation could function as a hormonal tactic to maximize maternal investment in reproductive behavior such as breeding and nesting (NMFS, 2004).

In the history of the entire WCPO purse seine fishery there have been documented mortalities of four olive ridley turtles in the international fleet (Molony, 2005). These mortalities were observed in 1995. The condition of most sea turtles observed in the international fleet was not documented; however, the condition of all turtles observed in the U.S. fleet has been recorded. All sea turtles captured in the U.S. WCPO purse seine fleet have been released alive with no gear attached (the observer code states that all turtles have been released “alive and healthy”).

There have been no documented turtle mortalities in the WCPO purse seine fishery since 1995. The SPC (2001) attributes increased crew awareness and sea turtle handling in recent years to a decrease in sea turtle mortality in WCPO purse seine fisheries. In the past several years the overall awareness about the biology and status of sea turtles has increased significantly and reducing the impact of fisheries on sea turtles has become a focused priority of RFMOs.

Based on reported interactions between the U.S. purse seine fishery and sea turtles, the intensity and duration of stressors expected to result to sea turtles from encirclement in U.S. purse seine nets is expected to be low, such that vital rates are not likely to be impacted to a degree that would result in reduced fitness to the individual. Based on data from the entire WCPO purse seine fleet, NMFS recognizes that there is the potential for more severe responses, such as

mortality, however to date; with a high level of observer coverage; no sea turtle mortality has been documented in the U.S. WCPO purse seine fishery.

8.3.2.3 Response of Sea Turtles to Entanglement in FADs

New information as of 2006, notes that sea turtles can become entangled in webbing discarded at sea or webbing hung under FADs (IATTC, 2006). In 2005, 28 turtles were documented as being entangled in FADs in the ETP tuna purse seine fishery (IATTC, 2006). These turtles were all alive at the time they were observed: 15 of the 28 entangled turtles were released alive, unharmed or with slight injuries, 2 were released with severe injuries, 2 were left entangled/alive, and the condition of 9 of the entangled turtles was unknown. It is likely that turtles die as a result of becoming entangled if they are not released from the webbing (IATTC, 2006). There are no documented incidences of sea turtles observed entangled in webbing under FADs in the U.S. WCPO purse seine fishery. In the event that sea turtles become entangled in webbing in the fishery, they will likely survive if they are released alive, however, if they are not assisted in dis-entanglement they will likely die.

8.3.2.4 Response of Sea Turtles to Removal of Fish Biomass

The removal of fish biomass from the ecosystem trophic structure poses no effect to sea turtles. Sea turtles typically prey on benthic invertebrates in hard bottom habitats, but depending on the species, the diet range includes algae, sea grasses, sponges, cnidarians, tunicates, and benthic and pelagic crustaceans, among others. However, due to their omnivorous and opportunistic habits, sea turtles are known to scavenge in the bounty of sea life discarded by trawlers and other indiscriminate fishing industries. Foraging has also been reported at sea, far from coastal hard bottom habitats. With no direct or indirect trophic interaction for competition between sea turtles and tuna species, alteration to the species vital rates and fitness are not expected.

8.4 Risk Analysis

To assess risk posed by the U.S. WCPO purse seine fishery on listed species, we integrate the probable exposure of the listed species to potential stressors with the species' response to the stressor. We first evaluate the risk posed to individuals in terms of the likely magnitude of effects which includes numbers of individuals and impacts to their reproductive potential and survival. After assessing the impacts to individuals, we evaluate how the impacts to the populations those individuals comprise and lastly on the species those populations comprise.

8.4.1 Marine Mammals

As stated in the Exposure Analysis, there are many uncertainties with respect to the number and species of listed marine mammals that may interact with the U.S. WCPO purse seine fishery. Our review of the literature does not allow us to rule out the potential for interactions between any of the listed marine mammals considered in this opinion and the fishery. There is evidence to suggest that interactions between the fishery and listed cetaceans are most likely to involve sei whales (Bailey et al., 1996). Interactions between purse seine fisheries and dolphinids are much more common than between baleen whales and some of the unidentified marine mammals in the observer database are likely dolphin species. Nonetheless, we cannot rule out the potential for interactions between listed baleen whales and the fishery. If these interactions occur they are likely to be relatively rare. A projection of future interactions based on a worst-case-scenario

with the available data, predict that three sets per year may involve interactions with listed marine mammal species Table 27.

The available data from the WCPO purse seine fishery indicate that large baleen whales are likely to escape from the purse seine gear, virtually unharmed. As indicated in the response analysis, we do not anticipate changes in the individual's energy budget or fitness as a result of interactions with the purse seine fishery such that the individual's reproductive fitness would be reduced. Moreover, because there have been no documented injuries or mortalities to large baleen whales in the U.S. WCPO purse seine fishery, exposure to the fishery is not likely to result in a reduction in the individuals' survival.

No reduction in the reproductive fitness or survival is expected at the individual level as result of interacting with the U.S. WCPO purse seine fishery. Therefore, we do not expect a reduction in the numbers, distribution, or reproduction in the populations these individuals comprise and thus do not expect the species these populations comprise to be compromised by the effects of the U.S. WCPO purse seine fishery.

8.4.2 Sea Turtles

As with marine mammals, the available data preclude an in depth assessment of the rate at which individual sea turtle species are likely exposed to the stressors of the U.S. WCPO purse seine fishery. Despite these limitations, the available data indicate that turtle interactions are rare²¹, patchy events and that the condition of all the sea turtles in observed interactions were released alive with no gear attached. Due to the mechanics of the gear involved in purse seine fisheries, sea turtles are not likely to experience physical injuries such as those that may occur in other gear types such as longlines and trawls. Sea turtles may be encircled in the net but are able to surface and breathe and do not sustain injuries such as those imposed by hooks. It is to the benefit of the crew to remove the turtle before the gear is retrieved to prevent equipment failure and operational slow downs. A worst-case-scenario was used to estimate the number of individuals of each species which may be exposed to the fishery, yet there is much uncertainty surrounding these estimates. Despite the uncertainty in the numbers, we are fairly confident that interactions with the U.S. WCPO purse seine fishery are not likely to reduce the fitness or survival of individual sea turtles. Because the fishery is not likely to reduce the reproductive fitness or survival of individual turtles, there is no casual link to suggest that the populations and species these individuals comprise will be compromised by the U.S. WCPO purse seine fishery.

9 Cumulative Effects

Cumulative effects²² include the effects of future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area considered in this Assessment (50 CFR 402.02). Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

²¹ Based on a worst-case-scenario an estimated 14 green or hawksbill or 11 olive ridley, leatherback, or loggerhead turtles may be exposed to the fishery annually.

²² "Cumulative effects", as defined for the purposes of the ESA, are limited to the effects of future, non-Federal actions in the Action Area.

Most of the fisheries described as occurring within the Action Area are expected to continue into the foreseeable future. One initiative that is likely to affect the way fisheries in the Action Area operate is the recently established WCPFC. Participation of the U.S. as a cooperating non-member in the WCPFC and domestic implementation of the Commission's decisions are Federal actions and are thus not relevant with respect to cumulative effects. The actions of the other members of the Commission (e.g., in response to the decisions of the Commission), however, could result in cumulative effects on the listed species considered in this Assessment. Although NMFS cannot predict such future actions or their effects, NMFS expects the other members of the Commission to be relatively conservative with respect to listed sea turtles and marine mammals, in accord with the objectives and principles established in the WCPFC. In short, NMFS is not aware of any specific proposed or anticipated changes in the fisheries in the Action Area that would result in any adverse cumulative effects on the sea turtles and marine mammals covered by this Assessment.

As far as non-fishery cumulative effects, NMFS is not aware of any proposed or anticipated changes in other human-related actions (e.g., poaching, habitat degradation) or natural conditions (e.g., over-abundance of land or sea predators, changes in oceanic conditions, etc.) that would substantially change the impacts that each threat has on the sea turtles or marine mammals covered by this Opinion.

10 Conclusion

After reviewing the current status of the endangered blue whales, fin whales, humpback whales, sei whales, sperm whales, green sea turtles, hawksbill sea turtles, leatherback sea turtles, loggerhead sea turtles, and olive ridley sea turtles, the environmental baseline for the Action Area, the effects of the U.S. WCPO purse seine fishery, and the cumulative effects, it is NMFS' biological opinion that the continued authorization of this fishery according to existing and proposed regulations (16 W.S.C. 973 et seq. and 50 CFR 300.30 et seq.) is not likely to jeopardize the continued existence of blue whales, fin whales, humpback whales, sei whales, sperm whales, green turtles, hawksbill turtles, leatherback turtles, loggerhead turtles or olive ridley turtles. Critical habitat has not been designated in the proposed action area, so no critical habitat would be affected by the proposed action.

11 Incidental Take Statement

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

The measures described below are nondiscretionary, and must be undertaken by NMFS for the exemption in section 7(o)(2) to apply. NMFS IFP has a continuing duty to regulate the activity covered by this incidental take statement. If NMFS IFP 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 IFP 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)).

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 incidental taking of listed marine mammals must be authorized under section 101(a)(5)(E) of the MMPA, before incidental take of listed marine mammals may be exempt from the taking prohibition of section 9(a), pursuant to section 7(o) of the ESA. As MMPA 101(a)(5)(e) authorization has not been completed, incidental take of listed marine mammals is not authorized for the proposed action at this time.

11.1 Amount or Extent of Take

NMFS expects that 14 green, 14 hawksbill, 11 leatherback, 11 loggerhead, and 11 olive ridley turtles per year may be incidentally taken as a result of the U.S. WCPO purse seine fishery. The nature of the take from encirclement and/or capture in the fishery may result in harassment and temporary harm. The best available data do not indicate that take in the form of mortality is likely to result to any sea turtle species due to interactions with the U.S. WCPO purse seine fishery.

11.2 Impact of the Take

In the accompanying Opinion, NMFS determined that this level of anticipated take is not likely to result in jeopardy to the green turtle, leatherback turtle, loggerhead turtle, olive ridley turtle, or hawksbill turtle.

11.3 Reasonable and Prudent Measures

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

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

1. NMFS shall ensure collection of data on capture, injury, and mortality of sea turtles and marine mammals on purse seine fishing vessels.

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2. NMFS shall reinitiate formal consultation under section 7 of the ESA if the amount of incidental take specified in this statement is exceeded, or if sea turtle or marine mammal mortality results incidental to the fishery.
3. NMFS shall require that sea turtles captured alive be released from fishing gear in a manner that minimizes injury and the likelihood of further gear entanglement or entrapment.
4. NMFS shall require that comatose or lethargic sea turtles be retained on board, handled, resuscitated, and released according to the established procedures.
5. NMFS shall require sea turtles that are dead when brought on board a vessel or that do not resuscitate be disposed of at sea unless NMFS requests retention of the carcass for sea turtle research.
6. NMFS shall require vessel operators to take all reasonable efforts to rescue and release sea turtles encircled or entangled in the purse-seine net or entangled in the FAD.

11.3.1 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. Observer Program: NMFS shall continue to assist the FFA observer program such that observers aboard U.S. WCPO permitted purse seine vessels collect data on the incidental take of sea turtles and other protected species and that observer coverage in the U.S. WCPO purse seine fishery is maintained at a level of 20 percent (in terms of trips on a calendar year basis).
 - 1B. Data Collection: NMFS shall work with FFA to ensure that observers collect standardized information regarding the incidental capture, injury, and mortality of sea turtles including: species, gear and set information for each interaction that occurs.
2. The following term and condition implements reasonable and prudent measure No. 2.
 - 2A. NMFS' IFP shall monitor and assess incidental take of sea turtles on an annual (calendar-year) basis (subject to the availability of data from the Treaty Administrator) and shall immediately request reinitiation of formal consultation under section 7 of the ESA if the amount of take authorized by this ITS has been exceeded in any given calendar year.
3. The following terms and conditions implement reasonable and prudent measure No. 3.

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- 3A. NMFS shall provide information to the vessel operators and crew on sea turtle biology and ways to avoid and minimize sea turtle impacts to promote sea turtle protection and conservation.
- 3B. NMFS shall work with the FFA to ensure that observer training by the FFA continue to include sea turtle handling and resuscitation techniques and sea turtle biology information.
- 3C. NMFS shall require that operators of vessels licensed to fish in the U.S. WCPO purse seine fishery make all reasonable efforts to remove any encircled or entangled sea turtle from the net before retrieving the purse seine net.
- 3D. NMFS shall work with vessel owners and operators to ensure that any sea turtles brought on board are not be dropped on the deck.
- 4. The following term and condition implements reasonable and prudent measure No. 4.
 - 4A. NMFS shall require that operators of vessels licensed to fish with purse seine gear in the U.S. WCPO fishery bring comatose sea turtles aboard, if feasible, and perform resuscitation techniques according to the procedures described at 50 CFR 223.206. 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. NMS shall work with vessel owners and operators to ensure that dead sea turtles are not consumed, sold, landed, offloaded, transshipped or kept below deck and are returned to the ocean after identification unless NMFS requests the turtle be kept for further study.
- 6. The following term and condition implements reasonable and prudent measure No. 6.
 - 6A. NMFS shall work with vessel owners and operators to ensure that whenever a sea turtle is sighted in the net, all reasonable efforts are made to release the turtle before it becomes entangled in the net, including, if necessary, the deployment of a speedboat.
 - 6B. NMFS shall work with vessel owners and operators to ensure that if a turtle is entangled in a net, net roll is stopped as soon as the turtle comes out of the water and does not start again until the turtle has been disentangled and released.
 - 6C. NMFS shall work with vessel owners and operators to ensure that if a turtle is brought aboard the vessel, all reasonable efforts to assist in the recovery of the turtle are made before returning it to the water.

- 6D. NMFS shall work with vessel owners and operators to ensure that if a sea turtle is sighted entangled in a FAD, all reasonable efforts should be made to rescue the turtle.

12 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 support and encourage the FFA Observer Program to increase FFA observers' ability to accurately identify sea turtles and marine mammals interacting with WCPO purse seine fisheries to the level of species and increase the rate of observers reporting condition and fate of captured animals. Observers should record standardized information on the incidental capture, injury, and mortality of sea turtles and marine mammals by species, gear and set information in which each interaction occurred. Observers should collect basic life history information for sea turtles and marine mammals, including estimated animal length, animal condition, and number of animals caught per set. Observers should also collect information on the location of the interaction.
2. FFA observers deployed on purse seine vessels in the WCPO should record the presence or absence of tags on all sea turtles captured by the purse seine fishery.
3. Whenever a marine mammal is sighted in the net, all reasonable efforts should be made to rescue the marine mammal before it becomes entangled in the net, including, if necessary, the deployment of a speedboat.
4. In the unlikely event of a sea turtle mortality, a tissue sample should be obtained, preserved in salt, and provided to NMFS to determine genetic information.
5. Purse seine sets on anchored-FADs and other floating-object-associated sets should be minimized and sets on unassociated schools maximized in WCPO purse seine fleets to minimize interactions with sea turtles and marine mammals.
6. NMFS should research development or modifications of existing technologies, to detect and alert fishers if sea turtles or marine mammals become entangled in their gear.
7. NMFS should continue efforts to gather international support to implement the FAO Guidelines to Reduce Sea Turtle Mortality in Fishing Operations and the WCPFC

Resolution to Mitigate the Impact of Fishing for Highly Migratory Fish Species on Sea Turtles (WCPFC Resolution-2005-04).

8. In order to promote sea turtle protection and conservation, NMFS should make available and disseminate information on sea turtle biology and ways to avoid and minimize adverse impacts on sea at appropriate regional fora (such as the Heads of Fisheries Meetings of the Pacific Community) in the western Pacific region.
9. NMFS should continue and expand on existing efforts to implement measures and management actions that protect sea turtles in their ocean environments and increase hatchling production at nesting beaches in the eastern and western Pacific.
10. NMFS should continue to provide technical and financial assistance necessary to export advances in knowledge of techniques and gear modifications that reduce interactions with sea turtles and marine mammals and/or dramatically reduce the immediate and/or delayed mortality rates of captured turtles with other nations engaged in similar fishing practices to reduce fishery impacts to sea turtle populations worldwide.

13 Reinitiation Notice

This concludes formal consultation on the continued authorization of the U.S. WCPO purse seine fishery according to the provisions of the SPTA and its implementing regulations. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of the incidental take is exceeded; (2) new information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not considered in this opinion; (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in this opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action. If the amount or extent of incidental take identified in the incidental take statement that is enclosed in this biological opinion is exceeded, NMFS IFP should immediately request initiation of formal consultation.

This Opinion has been predicated on several assumptions, which were necessary to overcome gaps in our knowledge. First, the exposure analyses in this biological opinion assumed that interaction rates between listed marine mammal and sea turtle species observed in recent years in the U.S. WCPO purse seine fishery are likely to reflect future rates of interaction. If new data reveal that this assumption is incorrect then this new information is likely to satisfy the second requirement for reinitiating consultation.

Second, the response analyses of this Opinion made assumptions about acute and chronic mortality rates based on information available from the fishery. If new data, including data collected through the observer program, reveals that those assumptions underestimated the effects, and listed sea turtles or marine mammals die from acute or chronic exposure to the fishery, then this new information is likely to satisfy the second requirement for reinitiating consultation.

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