APPENDIX G

STATUS OF PROTECTED SPECIES AND CRITICAL HABITAT IN THE AREA OF HMS FISHERIES

The following endangered and threatened species occur in the action area or adjacent areas and may be affected by the HMS fisheries, as they would operate under the regulations to implement this FMP:

SPECIES	**STATUS**
Fish	
Chum salmon (Hood Canal summer, Columbia River) (*Oncorhynchus keta)*	Threatened
Coho salmon (Central California)*(Oncorhynchus kisutch)*	Threatened
Coho salmon (S. Oregon/N. Calif. Coast)(*Oncorhynchus kisutch)*	Threatened
Steelhead (Upper Columbia River, Southern California)*(Oncorhynchus mykiss* ssp.)	Endangered
Steelhead (Snake River Basin)*(Oncorhynchus mykiss* ssp.)	Threatened
Steelhead (Upper Willamette River)*(Oncorhynchus mykiss* ssp.)	Threatened
Steelhead (Columbia River)(*Oncorhynchus mykiss* ssp.)	Threatened
Steelhead (South-Central California,Central Valley, Northern California) *(Oncorhynchus mykiss* ssp)	Threatened
Sockeye salmon (Snake River)*(Oncorhynchus nerka)*	Endangered
Sockeye salmon (Ozette Lake)(*Oncorhynchus nerka*)	Threatened
Chinook salmon (Lower Columbia River)(*Oncorhynchus tshawytscha)*	Threatened
Chinook salmon (Upper Willamette River)(*Oncorhynchus tshawytscha*)	Threatened
Chinook salmon (Snake River Spring/Summer/Fall runs)*(Oncorhynchus tshawytscha)*	Threatened
Chinook salmon (Sacramento River winter, Upper Columbia Spring) (*Oncorhynchus tshawytscha*)	Endangered
Chinook salmon (Central Valley Spring, California Coastal) *(Oncorhynchus tshawytscha*)	Threatened
Tidewater goby (*Eucyclogobius newberryi*)	Endangered
Marine Mammals	
Blue whale (*Balaenoptera musculus*)	Endangered
Fin whale (*Balaenoptera physalus*)	Endangered
Guadalupe fur seal (*Arctocephalus townsendii*)	Threatened
Humpback whale (*Megaptera novaeangliae*)	Endangered
Right whale (*Eubalaena glacialis*)	Endangered
Sei whale (*Balaenoptera borealis*)	Endangered
Sperm whale (*Physeter macrocephalus*)	Endangered
Steller sea lion - eastern population (*Eumetopias jubatus*)	Threatened
Southern Sea Otter (*Enhydra lutris nereis*)	Threatened
Sea turtles	
Green turtle (*Chelonia mydas*)	Endangered/Threatened
Leatherback turtle (*Dermochelys coriacea*)	Endangered
Loggerhead turtle (*Caretta caretta*)	Threatened
Olive ridley turtle (*Lepidochelys olivacea*)	Endangered/Threatened
Seabirds	
Short-tailed albatross (*Phoebaotria albatrus*)	Endangered
Brown pelican (*Pelecanus occidentalis*)	Endangered

California least tern (*Sterna antillarum browni)*	Endangered
Marbled murrelet (*Brachyramphus marmoratus)*	Threatened
Bald eagle (*Haliaeetus leucocephalus*)	Threatened
California clapper rail (*Rallus longirostris obsoletus)*	Endangered

The following subsections are synopses of the current state of knowledge on the life history, distribution, and population trends of marine mammal, sea turtle and seabird species listed above, some of which have been observed incidentally taken by the CA/OR drift gillnet or longline fisheries since those fisheries were first observed, beginning in 1990, and that NMFS believes may be taken as a result of the future operations of the fisheries pursuant to this FMP.

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

1. FISH SPECIES STATUS

a. Salmon

The following table lists the salmon evolutionarily significant units (ESUs) that are designated as threatened or endangered and incorporates by reference the status information contained in the specified *Federal Register* notice announcing the respective designations.

Table 1. Salmon and Steelhead ESUs listed under the Endangered Species Act

Species	Evolutionarily Significant Unit	Status	Federal Register Notice
Chinook Salmon (O. tshawytscha)	Sacramento River Winter	Endangered	54 FR 32085 8/1/89
	Snake River Fall	Threatened	57 FR 14653 4/22/92
	Snake River Spring/Summer	Threatened	57 FR 14653 4/22/92
	Puget Sound	Threatened	64 FR 14308 3/24/99
	Lower Columbia River	Threatened	64 FR 14308 3/24/99
	Upper Willamette River	Threatened	64 FR 14308 3/24/99
	Upper Columbia River Spring	Endangered	64 FR 14308 3/24/99
	Central Valley Spring	Threatened	64FR 50394 9/16/99
	California Coastal	Threatened	64FR 50394 9/16/99
Chum Salmon (O. keta)	Hood Canal Summer-Run	Threatened	64 FR 14508 3/25/99
	Columbia River	Threatened	64 FR 14508 3/25/99
Coho Salmon (O. kisutch)	Central California Coastal	Threatened	61 FR 56138 10/31/96
	S. Oregon/ N. California Coastal	Threatened	62 FR 24588 5/6/97
	Oregon Coastal	Threatened	63 FR 42587 8/10/98
Sockeye Salmon (O. nerka)	Snake River	Endangered	56 FR 58619 11/20/91
	Ozette Lake	Threatened	64 FR 14528 3/25/99

Species	Evolutionarily Significant Unit	Status	Federal Register Notice
Steelhead (O. mykiss)	Southern California	Endangered	62 FR 43937 8/18/97
	South-Central California	Threatened	62 FR 43937 8/18/97
	Central California Coast	Threatened	62 FR 43937 8/18/97
	Upper Columbia River	Endangered	62 FR 43937 8/18/97
	Snake River Basin	Threatened	62 FR 43937 8/18/97
	Lower Columbia River	Threatened	63 FR 13347 3/19/98
	California Central Valley	Threatened	63 FR 13347 3/19/98
	Upper Willamette River	Threatened	64 FR 14517 3/25/99
	Middle Columbia River	Threatened	64 FR 14517 3/25/99
	Northern California	Threatened	65 FR 36074 6/7/00

b. Tidewater Goby

The tidewater goby was listed as endangered on March 7, 1994 (59 FR 5494). Critical habitat was designated on November 20, 2000 (65 FR 69693). A recovery plan has not been published.

Description. The tidewater goby is a small, elongate, gray-brown fish with dusky fins not exceeding 2 inches standard length.

Habitat Affinities. The tidewater goby, which is endemic to California, is typically found in coastal lagoons, estuaries, and marshes with relatively low salinities (approximately ten parts per thousand (ppt)). Its habitat is characterized by brackish shallow lagoons and lower stream reaches where the water is fairly still but not stagnant. However, tidewater gobies can withstand a range of habitat conditions: they have been documented in waters with salinity levels from 0 to 42 ppt, temperatures from 8 to 25 degrees Celsius, depths from 10 to 80 inches, and dissolved oxygen levels of less than one milligram per liter.

Tidewater gobies may range upstream into fresh water, up to 1.2 miles from the estuary. In San Antonio Creek and the Santa Ynez River, Santa Barbara County, tidewater gobies are often collected 2-3 miles upstream of the tidal or lagoonal areas, sometimes in beaver impounded sections of streams. Conversely, tidewater gobies enter marine environments if sandbars are breached during storm events. The species' tolerance of high salinities (up to 60 ppt for shorter time periods) likely enable it to withstand the marine environment, allowing it to colonize or re-establish in lagoons and estuaries following flood events.

The primary constituent elements for tidewater goby critical habitat include those that meet the needs of foraging, nest construction, spawning, sheltering, and dispersal. Such elements are provided by coastal lagoons and estuaries supported by a relatively natural hydrological regime and an environment with no exotic fishes, or so few, that the goby is unaffected.

Life History. The tidewater goby is primarily an annual species in central and southern California, although some variation has been observed. If reproductive output during a single season fails, few, if any, tidewater gobies survive into the next year. For this reason, populations are exceedingly sensitive to short-term adverse environmental conditions. In one notable case, a population estimated at between 10,000 and 30,000 individuals was extirpated after a single construction project. However, recent research suggests that tidewater gobies have adapted to climatically dynamic conditions and are adept at re-colonizing sites from which they have been extirpated.

Reproduction peaks from late April or May to July and can continue into November or December depending on seasonal temperatures and rainfall. Males begin the breeding ritual by digging burrows (3 to 4 inches deep) in clean coarse sand. Females then deposit eggs into the burrows at an average of 400 eggs per spawning effort. Males remain in the burrows to guard the eggs and frequently forgo feeding during this period, possibly contributing to the mid-summer mortality noted in some populations. Within 9 to 10 days larvae emerge at approximately 0.2-0.3 inches standard length. The larvae live in vegetated areas within the lagoon until they are 0.6-0.7 inches, when they become substrate oriented, spending the majority of time on the bottom rather than in the water column. Both males and females can breed more than once in a season, with a lifetime reproductive potential of 3 to 12 spawning events.

Tidewater gobies feed on small invertebrates, usually mysids, amphipods, ostracods, snails, and aquatic insect larvae, particularly dipterans. Small tidewater gobies (0.16-0.32 inches) probably feed on unicellular phytoplankton or zooplankton similar to many other early stage larval fishes.

Historic and Current Range. Historically, the tidewater goby occurred in at least 110 California coastal lagoons from Tillas Slough near the Oregon border to Agua Hedionda Lagoon in northern San Diego County. The southern extent of its distribution has been reduced by approximately 8 miles. The species is currently known to occur in about 85 locations, although the number of sites fluctuates with climatic conditions. Today, the most stable populations are in lagoons and estuaries of intermediate sizes (5 to 124 acres) that have remained relatively unaffected by human activities. These populations have probably provided colonists for the smaller ephemeral sites.

Rangewide Trends and Current Threats. The decline of the tidewater goby can be attributed primarily to urban, agricultural, and industrial development in and surrounding the coastal wetlands and alteration of habitats from seasonally closed lagoons to tidal bays and harbors. Some extirpations are believed to be related to pollution, upstream water diversions, and the introduction of exotic fish species (most notably sunfishes and black basses [Centrarchidae]). These threats continue to affect some of the remaining populations of tidewater gobies. Tidewater gobies have been extirpated from several water bodies that are impaired by degraded water quality (*e.g.*, Mugu Lagoon, Ventura County), but still occur in others (*e.g.*, Santa Clara River, Ventura County).

There is no evidence that there have ever been any interactions between HMS fisheries that would be operating under this FMP and the tidewater goby. The HMS fisheries generally operate far from shore and well beyond the range of the tidewater goby. The HMS fisheries operating under this FMP are not likely to directly or indirectly affect the tidewater goby.

2. MARINE MAMMAL SPECIES STATUS[1]

a. *Fin Whale*

Fin whales are widely distributed in the world's oceans and are the second largest member of the family Balaenopteridae, reaching lengths of between 20 and 29 meters at adulthood. Fin whales are dark gray dorsally and white underneath, with a long, slender body and a prominent dorsal fin about two-thirds of the way back on their body. Like other baleen whales, fin whales have fringed baleen plates and ventral grooves, which expand during feeding. In the north Pacific Ocean, fin whales prefer to feed on euphausiids and large copepods (mainly *Calanus cristatus*), followed by schooling fish such as herring, walleye pollock, and capelin. Euphausiids may be a basic food of fin whales, but they also may take advantage of fish when sufficiently concentrated, particularly in the pre-spawning, spawning, and post-spawning adult fish stages on the continental shelf and in coastal waters. They have been known to associate with steep contours, either because tidal and current mixing along such gradients drives high biological production, or because changes in depth aid their navigation. The local distribution of fin whales during much of the year is probably governed by prey availability. Although there has been considerable discussion of interspecific competition among mysticete whales for prey, there has been no conclusive evidence to demonstrate that it occurs. Critical habitat for the fin whale has not been designated or proposed within the action area.

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[1] The references cited for information in this section are found in the National Marine Fisheries Service's U.S. Pacific Marine Mammal Stock Assessments:2002 (in press)

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

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

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

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

Threats to fin whales. Because little evidence of ship strikes and entanglement in fishing gear exists, and large whales such as the fin whale may often die later and drift far enough not to strand on land after such incidents, it is difficult to estimate the numbers of fin whales killed and injured by ship strikes or gear entanglement. However, the evidence that has been gathered demonstrates that such events are rare occurrences (Barlow, *et al*., 1997). Since 1993, the eastern Pacific tropical tuna purse seine fishery has had 100 percent observer coverage, and in over 100,000 sets, only one baleen whale has been killed. Therefore, the likelihood of this fishery taking a large listed baleen whale, such as a fin whale, is considered to be extremely low. There have been no observations of a fin whale interaction with the longline fishery. The HMS fisheries operating under this FMP are not likely to directly or indirectly affect the fin whale. In addition, no major habitat concerns have been identified for the fin whale, and there is no evidence that levels of organochlorines, organotins or heavy metals in baleen whales generally (including the fin whale) are high enough to cause toxic or other damaging effects. There is a growing concern that the increasing levels of anthropogenic noise in the ocean may be a habitat concern for whales, particularly for whales that use low frequency sound to communicate, such as baleen whales (Forney *et al.*, 2000) but this FMP cannot address that concern.

b. *Humpback Whale*

The humpback whale, also a member of the family Balaenopteridae, is distributed worldwide in all ocean basins. Most humpback whales winter in shallow, nearshore temperate and tropical waters, whereas in summer, most are in areas of high biological productivity, usually in the higher latitudes. They probably mate and give birth while in the wintering areas, but reproductive events may take place during migration. Following reproduction and birthing, most humpback whales migrate considerable distances to the higher latitude summering areas, where they feed intensively on locally abundant prey (NMFS, 1991). Humpback whales are often found alone or in groups of two or three, but throughout their breeding and feeding ranges, they may congregate in groups of up to fifteen animals. Their distribution in general is over shallow banks and in shelf waters. Critical habitat for the humpback whale has not been designated or proposed within the action area.

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

Humpback whales calve between the months of January and March. Age at sexual maturity has been estimated to range from four to nine years in females, but there is no reliability associated with those estimates, since age estimates used in the past have been questioned, as have the reliability of the data. The length at sexual maturity for females is estimated to be between 11.4 and 12.0 meters, and for males, between 11.1 and 11.4 meters. The calving interval is also variable: a range of 2-3 years has been given; however, there is some evidence of calving by females in consecutive years. Gestation averages around 12 months, and lactation lasts nearly a year. The majority of calves are weaned at 1 year, but the specific timing of separation is still unknown. In the north Pacific, annual reproductive rates have been estimated from information collected in wintering and summering areas: the least biased estimate came from southeastern Alaska, where the calving rate[2] was estimated to be 0.37. Thus, on average, a mature female gives birth only once every 2.7 years (inverse of calving rate) to a calf that survives its first 6 months of life and its first migratory transit.

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

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

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

Calambokidis, *et al*. (1996) estimated the total north Pacific population of humpback whales to exceed 4,000; however, without knowing where some of the Mexican breeding stocks migrate, the current estimate is lower than this. The most precise and least biased population estimate for the CA/OR/WA - Mexico stock feeding group is 905, with a minimum estimate of 861 animals. Mark-recapture population estimates have increased from 1988-90 to 1997-98 at about 8% per year (Forney, *et al*., 2000). Based on photographic identification of individual animals, the estimated population size of the Mexican coastal stock was about 1,800 animals and the abundance of the Revillagigedo stock was estimated to be about 914 animals in the late 1990s. Based on the results of photo-identification studies of humpback whales in their wintering areas, the current population estimate for the Central North Pacific stock is 4,005 (CV=0.095), with a minimum estimate of 3,698 whales. Using these data, a recent abundance estimate for the Western North Pacific stock of humpback whales is about 400 animals. Combining all three stocks yields a total abundance estimate of 5,300 humpback whales in the north Pacific. This estimate does not include the Mexican breeding stock abundance estimates, because most of these animals are included in the estimates of the CA/OR/WA - Mexico feeding stock. Furthermore, population estimates for the entire north Pacific have increased substantially from 1966 to the early 1990s, at 6-7% per year (Forney, *et al*, 2000). Ship surveys conducted from 1986 through 1990 in the eastern tropical Pacific Ocean yielded sightings of humpback whales in the California and Peru currents, in the Gulf of Panama, and along the coast of Guatemala; however, there was not enough information to provide abundance estimates.

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

Based on observer data from six different Alaskan commercial fisheries from 1990-98, and self-reported fisheries information from 1990-98, there was one humpback whale, probably from the Western North Pacific stock[3], observed dead and entangled in the Bering Sea/Aleutian Islands groundfish trawl fishery in the period,

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

yielding an average mortality for this stock of 0.2 whales per year. In addition, one humpback was reported floating dead, entangled in netting and trailing buoys in 1997, although it is unclear which fishery (or even which country) was responsible. Nevertheless, averaging this mortality over a five-year period (1994-98) yields an average annual mortality of 0.2 humpback whales, bringing the total estimated annual mortality rate incidental to commercial fisheries for this stock to be 0.4 whales per year (Ferrero, *et al*., 2000).

Of the Central North Pacific stock of humpback whales, one animal was observed entangled and expected to die due to interaction with a Hawaiian longliner from 1990-1999; however, due to the low level of coverage during that year (1991), a mortality estimate was not given. The one humpback mortality in the Bering Sea/Aleutian Islands groundfish trawl fishery (described above) brings the estimated mean annual mortality rate from 1994-98 to 0.2 per year for this stock. In addition, during this time period, humpback whales were reported killed by the southeastern Alaska salmon drift gillnet fishery (one mortality reported by self reports, two mortalities from stranding data), the salmon purse seine fishery (one animal reported by self reports), and by unknown fisheries in Alaska and Hawaii (estimated 2 per year). The mean annual mortality of the Central North Pacific stock of humpback whales due to fisheries-related interactions is estimated to be 2.8 whales per year (Ferrero, *et al*, 2000). The eastern Pacific tropical tuna fishery accidentally killed one unidentified baleen whale in 1997, but it could not be determined whether the whale was a listed species. However, since 1993, this fishery has had 100 percent observer coverage, and in over 100,000 sets, only one baleen whale has been killed. Therefore, the likelihood of this fishery taking a large listed baleen whale, such as a humpback, is considered to be extremely low.

Drift gillnet fisheries for swordfish and sharks exist along the entire Pacific coast of Baja California and may take animals from the CA/OR/WA-Mexican stock of humpback whales. Since 1986, the Mexican fleet has increased from 2 vessels to 31 in 1993, and in 1992, the observed bycatch of marine mammals was 0.13 animals (10 animals in 77 observed sets, with approximately 2,700 total sets for that year). Unfortunately, species-specific information is not available. In addition, there are currently efforts underway to convert the Mexican swordfish driftnet fishery to a longline fishery, which would considerably reduce the incidental take of marine mammals. There are no records of takes of humpbacks in the California/Oregon drift gillnet fishery, which has had 20 percent observer coverage for many years. There also have been no observations of a humpback whale interaction with the longline fishery or any other HMS gear. The HMS fisheries operating under this FMP are therefore not likely to directly or indirectly affect humpback whales.

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

c. *Sperm Whale*

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

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

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

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

Killer whales (*Orcinus orca*) have been observed attacking sperm whales, and serological studies have indicated that sperm whales are carriers of and are infected by calciviruses and papilomavirus. Estimated natural mortality rates for sperm whales age zero to two years old is nine9 percent, while older (age two and above) sperm whales have an estimated mortality rate of five percent; however, the lack of information on the causes of natural mortality have rendered these estimates statistically unreliable.

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

A study conducted in 1997 to estimate the breeding season abundance of sperm whales in the eastern temperate north Pacific (between 20° – 45° N latitude, and west to 165° W longitude) used passive acoustic listening devices to detect numbers of sperm whales, coupled by visual surveys. Sperm whales appear to be uniformly distributed in the study area, with no north to south density gradient. MtDNA and microsatellite DNA of animals sampled in the California Current is significantly different from animals sampled further offshore, although the line of delineation is unknown. It is likely somewhere between the North American coast and halfway to Hawaii. Genetic differences appear to be greater in an east-west direction than in a north-south direction. This is confirmed by tagging studies of three whales tagged in San Francisco and later caught by whalers as far north as British Columbia. Based on differences in gene samples between sperm whales in the Gulf of California, and coastal California, the California-Mexico border is probably near the southern limit of the U.S. West Coast stock; however, scientists cannot state with certainty how far west or north the stock may range.

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

A combined visual and acoustic survey conducted by NMFS in 1997 estimated the population of sperm whales in the survey area to be between 24,000 (cv=0.46, based on visual surveys) and 39,200 (cv=0.60, based on acoustic detections and visual group size estimates). However, it is not known how many of these animals enter the U.S. EEZ. The border to the west and north is less clearly defined, although sperm whales are known to exist thousands of miles from the California/Oregon coastline. Therefore, the abundance estimates contained in the most recent stock assessment report are probably much lower than actual abundance. The Pacific Scientific Review Group (PSRG)[4] has also concluded that sperm whale group size is underestimated and largely a function of the time spent sighting (minutes of PSRG meeting, 5-6 December, 1999), especially since sperm whales can stay submerged for over 60 minutes. Since little is known about the western and northern boundaries of the sperm whale stock, the best estimates of abundance within the EEZ, off California, Oregon and Washington, is 1,191 sperm whales, with a minimum estimate of 992 (Barlow (1997)). Furthermore, although sperm whale abundance appears to have been rather variable off California between 1979-80 and 1996, and the eastern north Pacific population is expected to have grown since whaling stopped in 1980, it does not show any obvious trends (Forney *et al*., 2000).

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

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

Drift gillnet fisheries for swordfish and sharks exist along the entire Pacific coast of Baja California and may take sperm whales. In 1992, observers in Mexico documented the rate of marine mammal bycatch to be 0.13 animals per set, although species-specific information is not available for this Mexican fleet (*in* Forney, *et al*., 2000). There are no observations of sperm whale interactions in the California/Oregon drift gillnet fishery (which has had 20 percent observer coverage for many years), longline fishing in the eastern Pacific (with limited observer coverage the past few years), or any other HMS fisheries. The likelihood of interactions between sperm whales and the fisheries being managed under this FMP is believed to be very low.

d. *Steller Sea Lion*

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

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

Suckling. On May 5, 1997, the western U.S. stock was reclassified as endangered, while the eastern stock remained on the threatened species list (62 FR 24345).

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

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

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

Relatively little is known about the life history of sea lions during the juvenile years between weaning and maturity. Males reach sexual maturity between the age of 3 and 7 years of age and physical maturity by age 10. Males and females are expected to live approximately 20 and 30 years, respectively. The probability of pupping appears to be low (about 10%) for animals 4 years of age or younger. Maturation of 100% of a cohort of females occurs over a prolonged period which may be as long as 4 years. The reported constancy of fecundity for females extending from age 6 to 30 indicates that either senescence has no effect on fecundity, or information on fecundity rates is not sufficiently detailed to allow confident estimation of age-specific rates for animals older than age 6. Given the limited evidence available, the latter is a more likely explanation for such constancy.

Steller sea lions are not known to migrate, but they disperse widely during the breeding season. Males breeding in California appear to spend the non-breeding months (September - April) in Alaska and British Columbia, whereas animals marked at rookeries in Alaska have traveled to British Columbia. There appears to be limited exchange between rookeries by breeding adult females and males (other than between adjoining rookeries). They are opportunistic feeders, foraging mostly near the shore and over the continental shelf for predominantly demersal and off-bottom schooling fish, including walleye pollock, herring, capelin, mackerel, rockfish, and salmon, and cephalopods such as squid and octopus. They are believed to be capable of diving as deep as 100 fathoms (600 feet), and often reach depths of 60 to 80 fathoms (360 to 480 feet).

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

Threats to Steller sea lions. Steller sea lions have been observed or reported incidentally taken in the following Alaskan fisheries: drift gillnet, set gillnet, salmon troll, groundfish and halibut longline/ set line, and the groundfish trawl. Based on observer data, strandings, self reports, and permit reports, information on known incidental mortality of the eastern stock of Steller sea lions from 1990-1998 include the following: one animal was observed killed (7 estimated for the year) in the northern Washington marine set gillnet fishery, five Stellers were observed killed in the southeastern Alaska salmon drift gillnet fishery, one Steller sea lion was observed killed in the Alaska salmon troll fishery, and 84 animals were killed due to the British Columbia aquaculture predator control program (1991-1997). The minimum annual average of incidental mortality due to all of these fisheries combined was approximately 14 Steller sea lions per year. There are no records of interactions between Steller sea lions and any of the West Coast-based HMS fisheries.

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

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

e. Southern Sea Otter

The southern sea otter was listed as threatened in 1977 (42 FR 2965); critical habitat was not designated. A revised draft recovery plan was published in January 2000 (USFWS 2000b). The factors leading to the listing included increased tanker traffic and the potential for oil spills, municipal pollution, and increased harassment caused by increased use of nearshore areas for a variety of human activities.

Description. The sea otter is the second largest member of the family Mustelidae, weighing up to 88 pounds and attaining lengths of 55 inches. Males are larger than females (Riedman and Estes 1990). Their pelage consists of dense water-resistant brown fur.

Habitat Affinities. Southern sea otters generally forage in both rocky and soft-sediment communities in water depths of 82 feet or less, although individuals occasionally will move into deeper water. Most southern sea otters remain within 1.2 miles of shore and are most often found in association with kelp forests. Rocky habitats that are topographically heterogeneous and support kelp forests are likely to support the greatest diversity and abundance of food resources. The density of southern sea otters within most of the population's range is most likely related to substrate type; rocky bottom habitats support an average density of 5 individuals per square kilometer; while areas with sandy bottoms support an average of 0.8 individuals per square kilometer. Southern sea otter seldom come to the mainland except when sick or during extreme storm events.

Life History. Unlike most other marine mammals, sea otters have little subcutaneous fat; they depend on their clean, dense, water-resistant fur for insulation against the cold. Sea otters also maintain a high level of internal heat production to compensate for the lack of blubber. Consequently, their energetic requirements are high and they are estimated to consume an amount of food equivalent to 23 to 33 percent of their body weight per day. Southern sea otter prey items include abalone, rock crabs, sea urchins, kelp crabs, clams, turban snails, mussels, octopus, barnacles, scallops, sea stars, and chitons. Their mobility, forelimb dexterity, and ability to crush large invertebrates, either with their teeth or rocks, enable sea otters to prey on virtually any invertebrate of any size. Because of their ability to eat large quantities of marine invertebrates, sea otters play an extremely important role in the nearshore marine community.

The patterns in which southern sea otters move throughout the year are complicated and vary between males and females. Generally, the home ranges of southern sea otters consist of several heavily used areas with travel corridors between them. Animals often remain in an area for a long period of time and then suddenly move long distances; these movements can occur at any time of the year. Male southern sea otters have larger home ranges and are less sedentary than females. Juvenile males move further from natal groups than juvenile females, likely due to territorial and aggressive behavior exhibited towards juvenile males by older males. Most male southern sea otters leave the central portion of the range and travel to its southern end during the pupping season, which occurs in the winter and spring (Riedman and Estes 1990).

Southern sea otters mate and pup throughout the year. The northern and southern portions of the population seem to exhibit different mating peaks. A peak period of pupping occurs from January to March; a secondary pupping season occurs in late summer and early fall. Parental care is provided solely by the female.

Historic and Current Range. The southern sea otter once ranged from the central coast of Baja California north to at least northern California, although they may have ranged as far north as Prince William Sound in Alaska. Prior to being protected from hunting for its pelt in 1911, the southern sea otter was reduced to only a remnant colony near Bixby Creek along the Big Sur coast. Since 1911, southern sea otters have expanded north and south from the Bixby Creek colony. By the late 1980s, the expansion of southern sea otters to the north had essentially stopped at Point Ano Nuevo. By 1995, southern sea otters were commonly observed as far south as Point Arguello at Vandenberg Air Force Base in Santa Barbara. A translocation program begun in 1987 established a population of southern sea otters around San Nicolas Island. Southern sea otters currently inhabit shallow waters along the coast of California in San Mateo, Santa Cruz, Monterey, San Luis Obispo, and Santa Barbara counties and at San Nicolas Island in Ventura County.

Rangewide Trends and Current Threats. Historically, the number of southern sea otters was estimated at 14,000; they were known to be abundant at San Nicolas Island. By the end of the 19th century, the sea otter had been hunted nearly to extinction throughout its range, which extended from Japan, along the northern Pacific rim, to central Baja California; about a dozen colonies remained by 1911 when it was afforded protection from further hunting. Southern sea otters along the central coast of California experienced a general recovering trend, increasing to an estimated 1,789 in 1976. However, the estimated number of southern sea otters declined to 1,443 in 1979 and 1,277 in 1983. The decline was likely caused primarily by entanglement in coastal setnets. Escalating closures and limitations of the setnet fishery by the CDFG between 1982 and 1991 contributed to population increases up to 2,377 individuals in 1995. However, the population of southern sea otters declined over the next 4 or 5 years, to a low of 2,090 in 1999. The reason(s) for the decline cannot be fully explained to date. During the spring of 2000, 2,317 southern sea otters were counted.

The effects of infectious disease on southern sea otters is receiving increased attention as a cause of decline. Acanthocephalan parasites, protozoal encephalitis caused by *Toxoplasma gondii*, *coccidioidomycosis*, and various bacterial infections have been documented. The typical infectious stage of the encephalitis is usually from cat feces; the mechanism by which *Toxoplasma gondii* is transmitted to the southern sea otter is unknown but the protozoan may be entering the ocean in runoff from beach soils or sewage effluent.

Threats to the California population of the southern sea otter include habitat degradation from oil spills and other environmental contaminants, prey depletion by human exploitation, shooting, and entanglement in fishing gear. Oil spills that could strike at any time, threaten catastrophic decimation or localized extinction. Southern sea otters are vulnerable to dietary exposure of contaminants such as mercury, which is prevalent in coastal waters. The incidence of infection by acanthocephalan parasites appears to be increasing and southern sea otters are contaminated with potentially harmful levels of environmental contaminants.

There is no evidence to suggest or indicate that there have ever been or would be interactions between the HMS fisheries being managed under this FMP and southern sea otter. The HMS fisheries generally occur far out at sea and well beyond the range of the southern sea otter. Thus the HMS fisheries should not directly or indirectly affect the status of the southern sea otter.

f. Other Listed Marine Mammal Species

There is no information to demonstrate or suggest that the HMS fisheries as they are expected to operate under this FMP will in any way affect other listed marine mammal species (blue whale, right whale, sei whale or Guadalupe fur seal).

3. SEA TURTLE SPECIES STATUS

All stocks/populations of sea turtles incidentally taken in the HMS fisheries are in decline. Impacts on sea turtles in the Pacific Ocean are primarily due to the composite effect of human activities which include: the legal harvest and illegal poaching of adults, immatures, and eggs; incidental capture in fisheries (coastal and high-seas); and loss and degradation of nesting and foraging habitat as a result of coastal development, including predation by domestic dogs and pigs foraging on nesting beaches (associated with human settlement). Increased environmental contaminants (e.g. sewage, industrial discharge) and marine debris, which adversely impact nearshore ecosystems that turtles depend on for food and shelter, including sea grass and coral reef communities, also contribute to the overall decline. While it is generally accepted by turtle biologists and others that these factors are the primary cause of turtle population declines, in many cases there is a paucity of quantitative data on the magnitude of human-caused mortality. In addition to anthropogenic factors, natural threats to the nesting beaches and pelagic-phase turtles such as coastal erosion, seasonal storms, predators, temperature variations, and phenomena such as El Niño also affect the survival and recovery of sea turtle populations. More information on the status of these species along with an assessment of overall impacts are found in this section as well as the Pacific Sea Turtle Recovery Plans (NMFS and USFWS, 1998a-d) and are reviewed extensively in Eckert (1993). Critical habitat for the green, leatherback, loggerhead, and olive ridley turtles has not been designated or proposed within the action area.

a. *Green Turtle*

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

Green turtles are distinguished from other sea turtles by their smooth carapace with four pairs of lateral scutes, a single pair of prefrontal scutes, and a lower jaw-edge that is coarsely serrated. Adult green turtles have a light to dark brown carapace, sometimes shaded with olive, and can exceed 1 meter in carapace length and 100 kilograms (kg) in body mass. Females nesting in Hawaii averaged 92 cm in straight carapace length (SCL), while at the Olimarao Atoll in Yap, females averaged 104 cm in curved carapace length (CCL) and approximately 140 kg. In the rookeries of Michoacán, Mexico females averaged 82 cm in CCL, while males averaged 77 cm CCL (*in* NMFS and USFWS, 1998a). Based on growth rates observed in wild green turtles, skeletochronological studies, and capture-recapture studies, all in Hawaii, it is estimated that green sea turtles attain sexual maturity at an average age of at least 25 years (*in* Eckert, 1993).

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

Green turtles prefer waters that usually remain about 20° C in the coldest month; for example, during warm spells (e.g. El Niño), green turtles may be found considerably north of their normal distribution. Based on the behavior of post-hatchlings and juveniles raised in captivity, it is presumed that those in pelagic habitats live and feed at or near the ocean surface, and that their dives do not normally exceed several meters in depth (NMFS and USFWS, 1998a). The maximum recorded dive depth for an adult green turtle was 110 meters (Berkson, 1967, *in* Lutcavage and Lutz, 1996), while subadults routinely dive 20 meters for 9-23 minutes, with a maximum recorded dive of 66 minutes (Brill, *et al*., 1995, *in* Lutcavage and Lutz, 1996). Additionally, it is presumed that drift lines or surface current convergences are preferential zones due to increased densities of likely food items. In the western Atlantic, drift lines commonly contain floating *Sargassum* capable of providing small turtles with shelter and sufficient buoyancy to raft upon (NMFS and USFWS, 1998a). Although most green turtles appear to have a nearly exclusive herbivorous diet, consisting primarily of sea grass and algae (Wetherall *et al*., 1993), those along the east Pacific coast seem to have a more carnivorous diet. Analysis of stomach contents of green turtles found off Peru revealed a large percentage of molluscs and polychaetes, while fish and fish eggs, and jellyfish and commensal amphipods comprised a lesser percentage (Bjorndal, 1997). The non-breeding range of green turtles is generally tropical, and can extend approximately 500-800 miles from shore in certain regions (Eckert, 1993).

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

The primary green turtle nesting grounds in the eastern Pacific are located in Michoacán, Mexico, and the Galapagos Islands, Ecuador (NMFS and USFWS, 1998a). Here, green turtles were widespread and abundant prior to commercial exploitation and uncontrolled subsistence harvest of nesters and eggs. More than 165,000 turtles were harvested from 1965 to 1977 in the Mexican Pacific. In the early 1970s nearly 100,000 eggs per night were collected from these nesting beaches (*in* NMFS and USFWS, 1998a). The nesting population at the two main nesting beaches in Michoacán (Colola, responsible for 70% of total green turtle nesting in Michoacán (Delgado and Alverado, 1999) and Maruata decreased from 5,585 females in 1982 to 940 in 1984. Despite long-term protection of females and their eggs at these sites since 1990, the population continues to decline, and it is believed that adverse impacts (including incidental take in various coastal fisheries as well as illegal directed take at forage areas) continue to prevent recovery of endangered populations (P. Dutton, NMFS, personal communication, 1999). Although the poaching of adult green turtles is now nearly negligible, the black market for sea turtle eggs in Mexico has remained as brisk as before the ban (Delgado and Alvarado, 1999). On Colola, an estimated 500-1,000 females nested nightly in the late 1960s. In the 1990s, that number dropped to 60-100 per night, or about 800-1,000 turtles per year. During the 1998-99 season, based on a comparison of nest counts and egg collection data, an estimated 600 greens nested at Colola. Although only about 5% of the nests were poached at Colola during this season, approximately 50% of the nests at Maruata were poached, primarily because of difficulties in providing protections as a result of political infighting (Delgado and Alvarado, 1999).

There are no historical records of abundance of green turtles from the Galapagos - only residents are allowed to harvest turtles for subsistence, and egg poaching occurs only occasionally. An annual average of 1,400 nesting females was estimated for the period 1976-1982 in the Galapagos Islands (NMFS and USFWS, 1998a).

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

Tag returns of eastern Pacific green turtles establish that these turtles travel long distances between foraging and nesting grounds. In fact, 75 percent of tag recoveries from 1982-90 were from turtles that had traveled more than 1,000 kilometers from Michoacán, Mexico. Even though these turtles were found in coastal waters, the species is not confined to these areas, as indicated by 1990 sightings records from a NOAA research ship. Observers documented green turtles 1,000-2,000 statute miles from shore (Eckert, 1993). The east Pacific green is also the second-most sighted turtle in the east Pacific during tuna cruises; they are frequent along a north-south band from 15° N to 5° S latitude along 90° W longitude, and between the Galapagos Islands and Central American Coast (NMFS and USFWS, 1998a). In a review of sea turtle sighting records from northern Baja California to Alaska, Stinson (1984, *in* NMFS and USFWS, 1998a) determined that the green turtle was the most commonly observed sea turtle on the U.S. Pacific Coast, with 62% reported in a band from southern California and southward. The northernmost reported resident population of green turtles occurs in San Diego Bay, where about 50-60 mature and immature turtles concentrate in the warm water effluent discharged by a power plant. These turtles appear to have originated from east Pacific nesting beaches, based on morphology and preliminary genetic analysis (*in* NMFS and USFWS, 1998a). California stranding reports from 1990-99 indicate that the green turtle is the second most commonly found stranded sea turtle (48 total, averaging 4.8 annually) (J. Cordaro, NMFS, personal communication, April, 2000).

Green turtles encountered during drift gillnet fishing off California and Oregon may originate from a number of known proximal, or even distant, breeding colonies in the region. However the most likely candidates would include those from Hawaii (French Frigate Shoals) and the Pacific coast of Mexico population. This is based on limited genetic sampling conducted within the NMFS observer program for the CA/OR drift gillnet fishery (1 turtle genetically analyzed was found to originate from eastern Pacific stock - most likely Mexican nesting beach) (P. Dutton, NMFS, personal communication, January, 2000).

No green turtles have been observed interacting with HMS fisheries off the West Coast, and the HMS fisheries operating under this FMP are not expected to directly or indirectly affect green sea turtles.

b. *Leatherback Turtle*

The leatherback turtle is listed as endangered throughout its global range. Increases in the number of nesting females have been noted at some sites in the Atlantic, but these are far outweighed by local extinctions, especially of island populations, and the demise of once large populations throughout the Pacific, such as in Malaysia and Mexico. The most recent estimate of the world population of leatherbacks is currently only 25,000 to 42,000 turtles (Spotila *et al.*, 1996).

Leatherbacks are the largest of the marine turtles, with a CCL often exceeding 150 cm and front flippers that are proportionately larger than in other sea turtles and may span 270 cm in an adult (NMFS and USFWS, 1998b). In view of its unusual ecology, the leatherback is not surprisingly morphologically and physiologically distinct from other sea turtles. Its streamlined body, with a smooth, dermis-sheathed carapace and dorso-longitudinal ridges may improve laminar flow of this highly pelagic species. Adult females nesting in Michoacán, Mexico averaged 145 cm CCL (L. Sarti, Universidad Naçional Autonoma de Mexico, unpublished data, *in* NMFS and USFWS, 1998b), while adult female leatherbacks nesting in eastern Australia averaged 162 cm CCL (Limpus, *et al*., 1984, *in* NMFS and USFWS, 1998b).

Leatherbacks have the most extensive range of any living reptile and have been reported circumglobally from 71°N to 42°S latitude in the Pacific and in all other major oceans (NMFS and USFWS, 1998b). For this reason, however, studies of their abundance, life history and ecology, and pelagic distribution are difficult. Similar to the olive ridley turtle, they lead a completely pelagic existence, foraging widely in temperate waters except during the nesting season, when gravid females return to tropical beaches to lay eggs. They are highly migratory, exploiting convergence zones and upwelling areas in the open ocean, along continental margins, and in archipelagic waters.

Recent satellite telemetry studies indicate that adult leatherbacks follow bathymetric contours over their long pelagic migrations and typically feed on cnidarians (jellyfish and siphonophores) and tunicates, and their commensals, parasites and prey (NMFS and USFWS, 1998b). Because of the low nutritive value of jellyfish and tunicates, it has been estimated that an adult leatherback would need to eat about 50 large jellyfish (equivalent to approximately 200 liters) per day to maintain its nutritional needs (Duron, 1978, *in* Bjorndal, 1997). Surface feeding has been reported in U.S. waters, especially off the west coast, but foraging may also occur at depth. Based on offshore studies of diving by adult females nesting on St. Croix, U.S. Virgin Islands, Eckert *et al*. (1989) proposed that observed internesting[5] 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, 1998b) also speculated that foraging may occur at depth, when nematocysts from deep water siphonophores were found in leatherback stomach samples.

Leatherbacks also appear to spend almost the entire portion of each dive traveling to and from maximum depth, suggesting that maximum exploitation of the water column is of paramount importance to the leatherback (Eckert, *et al*., 1989). Maximum dive depths for post-nesting females in the Carribean have been recorded at 475 meters and over 1,000 meters, with routine dives recorded at between 50 and 84 meters. The maximum dive length recorded for such female leatherbacks was 37.4 minutes, while routine dives ranged from 4-14.5 minutes (*in* Lutcavage and Lutz, 1997). A total of six adult female leatherbacks from Playa Grande, Costa Rica were monitored at sea during their internesting intervals and during the 1995 through 1998 nesting seasons. The turtles dived continuously for the majority of their time at sea, spending 57-68% of their time submerged. Mean dive depth was 19±1 meters and the mean dive duration was 7.4±0.6 minutes (Southwood, *et al*., 1999). Migrating leatherbacks also spend a majority of time at sea submerged, and they display a pattern of continual diving (Standora, *et al*., 1984, *in* Southwood, *et al*., 1999). Eckert (1999) placed transmitters on nine leatherback females nesting at Mexiquillo Beach and recorded dive behavior during the nesting season. The majority of the dives were less than 150 meters depth, although maximum depths ranged from 132 meters to over 750 meters. Although the dive durations varied between individuals, the majority of them made a large proportion of very short dives (less than two minutes), although Eckert speculates that the short duration dives most likely represent surfacing activity after each dives. Excluding these short dives, five of the turtles preferred dive durations greater than 24 minutes, while three others preferred dives durations between 12-16 minutes. On the Pacific coast of Mexico, female leatherback turtles lay 1-11 clutches per year (mean=5.7), with clutch size averaging 64 yolked eggs (each clutch contains a complement of yolkless eggs, sometimes comprising as much as 50 percent of total clutch size, a unique phenomenon among leatherbacks and some hawksbills (Hirth and Ogren, 1987)). Clutch sizes in Terengganu, Malaysia, and in Pacific Australia were larger, averaging around 85-95 yolked eggs and 83 yolked eggs, respectively (*in* Eckert, 1993). Females are believed to migrate long distances between foraging and breeding grounds, at intervals of typically two or three years. Spotila *et al*. (2000), found the mean re-nesting interval of females on Playa Grande, Costa Rica to be 3.7 years. Using skeletochronological analysis of a small sample size of leatherback sclerotic ossicles, Zug and Parham (1996) suggested that mean age at sexual maturity for leatherbacks is around 13 to 14 years, giving them the highest juvenile growth rate of all sea turtle species, although this data is speculative (*in* Chaloupka and Musick, 1997). Zug and Parham (1996) concluded that for conservation and management purposes, 9 years is a likely minimum age for maturity of leatherbacks, based on the youngest adult in their sample. The natural longevity of leatherback turtles have not been determined (NMFS and USFWS, 1998b).

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

For nesting females from Mexiquillo Beach, Mexico, the eastern Pacific region has been shown to be a critical

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

migratory route for female leatherbacks. Nine females outfitted with satellite transmitters in 1997 traveled along almost identical pathways away from the nesting beach. These individuals moved south and, upon encountering the North Equatorial Current at about 8° N latitude, diverted west for approximately 800 km and then moved east/southeast towards the waters off Peru and Chile (Eckert, 1999). Satellite tagged, post-nesting leatherbacks leaving Costa Rica moved south after nesting. These studies underscore the importance of this offshore habitat and the likelihood that sea turtles are present on fishing grounds, particularly for large commercial fishing fleets south of the equator. Eckert (1999) speculates that leatherbacks leaving the nesting areas of Mexico and Costa Rica may be resource-stressed by a long reproductive season with limited food and the high energetic requirements brought about by the demands of reproduction, elevated water temperatures, or both. When they leave, their greatest need is to replenish energy stores (e.g. fat) and they must move to areas where food is concentrated (e.g. upwelling areas). These eastern Pacific nesting stocks may also move northwest, as genetic samples from two leatherbacks caught by the Hawaiian longline fishery indicated representation from eastern Pacific nesting beaches (Dutton *et al*., in press, and unpublished). NMFS and USFWS (1998b) and Eckert (1999) speculate that the high density of leatherback sightings in and around Monterey, peaking in August (Starbird, *et al*., 1993), and the October to January nesting period on the Pacific coast of Mexico suggests that the turtles may migrate southward along the U.S. coastline to Mexican nesting beaches. However, genetic analyses of leatherbacks that have stranded and been taken in fisheries off Oregon and California have indicated representation from the western Pacific nesting beaches (P. Dutton, *et al*., in press, and P. Dutton, NMFS, personal communication, May, 2000).

Migratory corridors of leatherbacks originating from western Pacific nesting beaches most likely exist along the eastern seaboards of Australia, Asia and the former Soviet Union (NMFS and USFWS, 1998b). Genetic markers in 12 of 14 leatherbacks sampled to date from the central north Pacific (captured in the Hawaii longline fishery) have identified those turtles as originating from nesting populations in the southwestern Pacific; the other 2 specimens, taken in the southern range of the Hawaii fishery, were from nesting beaches in the eastern Pacific (P. Dutton, *et al*., in press, and P. Dutton, NMFS, personal communication, May, 2000). Stranding records from 1990-99 indicate that the leatherback is the most commonly stranded sea turtle off California (50 total, averaging 5 annually, J. Cordaro, NMFS, personal communication, April, 2000). In the U.S., leatherbacks have been sighted and reported stranded as far north as Alaska (60° N latitude) and as far south as San Diego, California (NMFS and USFWS, 1998b). Of the stranded leatherbacks that have been sampled to date, all have been of western Pacific nesting stock origin (Dutton *et al*., in press). Genetic analysis of samples from two leatherback turtles taken off California and Oregon by the CA/OR drift gillnet fishery revealed that they both originated from western Pacific nesting beaches (i.e. Indonesia/Solomon Islands/Malaysia) (P. Dutton, NMFS, personal communication, March, 2000). Lastly, two leatherbacks were recently captured and tagged in Monterey Bay on September 7-8, 2000 and fitted with transmitters. By September 21,2000, both were on a southwest migratory path, most likely headed to the western Pacific nesting beaches. One of these individuals was of a size normally associated with the western Pacific nesting stock, which are, on average, 10-20 centimeters larger than eastern Pacific nesting stocks (Zug and Parham, 1996).

Based on published estimates of nesting female abundance, leatherback populations are declining at all major Pacific basin nesting beaches, particularly in the last two decades (Spotila *et al*., 1996; NMFS and USFWS, 1998b; Spotila, *et al*., 2000). Declines in nesting populations have been documented through systematic beach counts or surveys in Malaysia (Rantau Abang, Terengganu), Mexico and Costa Rica. In other leatherback nesting areas, such as Irian Jaya and the Solomon Islands, systematic nesting surveys are just beginning or have been ongoing for several years. In all areas where leatherback nesting has been documented, however, current nesting populations are reported by scientists, government officials, and local observers to be well below abundance levels of several decades ago. The collapse of these nesting populations was most likely precipitated by a tremendous overharvest of eggs coupled with incidental mortality from fishing, specifically the advent of the high seas driftnet fishery in the 1980s (Sarti *et al*., 1996).

Eastern Pacific Nesting Populations of Leatherback Turtles

Leatherback nesting populations are declining along the Pacific coast of Mexico and Costa Rica. At Las Baulas National Park, Costa Rica, the number of nesting leatherback has declined from 1,500 in 1988-1989 to 193 in 1993-1994 (Steyermark *et al.*, 1996). Leatherback have been studied at Playa Grande (in Las

Baulas), the fourth largest leatherback nesting colony in the world, since 1988. During the 1988-89 season (July-June), 1,367 leatherback nested on this beach, and by the 1998-99 season, only 117 leatherback nested. Furthermore, during 3 recent nesting seasons (1996 through 1999), an average of only 25% of the turtles were remigrants (turtles returning to nest that were observed nesting in previous nesting seasons). Less than 20% of the turtles tagged in 1993 through 1995 returned to nest in the next five years (Spotila, *et al*., 2000). Remigration intervals for leatherbacks at nesting beaches in South Africa and the U.S. Caribbean have been documented as over 91% returning within 5 years or less (Boulon, *et al*. 1996 *in* Spotila, *et al*., 2000). Comparatively few leatherbacks are returning to nest on east Pacific nesting beaches and it is likely that leatherbacks are experiencing abnormally high mortalities during non-nesting years. Since 1993, environmental education and conservation efforts through active law enforcement has greatly reduced egg poaching in Costa Rica (Chaves, *et al*., 1996). If current estimates of age to maturity are correct, the effects of such efforts should be observed beginning sometime this decade (Spotila and Steyermark, 1998), barring any increase in current levels of juvenile and adult mortality.

The decline of leatherback subpopulations is even more dramatic off Mexico. According to reports from the late 1970s and early 1980s, three beaches located on the Pacific coast of Mexico sustained a large portion of all global nesting of leatherbacks, perhaps as much as one-half. Since the early 1980s, the eastern Pacific Mexican population of adult female leatherbacks has declined from 70,000[6] in 1982 (Spotila *et al*., 1996) to less than 1,000 in 1999-2000 (Sarti *et al*., personal communication, 2000). Monitoring of the nesting assemblage at Mexiquillo, Mexico has been continuous since 1983-84. According to Sarti *et al.* (1996), nesting declined at this location at an annual rate of over 22 percent for the last 12 years. Sarti *et al*. (1998) reports:

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

Although the causes of the decline in the nesting populations are not entirely clear, Sarti *et al*. (1998) surmises that the decline could be a result of intensive egg poaching in the nesting areas, incidental capture of adults or juveniles in high seas fisheries, and natural fluctuations due to changing environmental conditions. Leatherbacks are not captured for meat or their skin in Mexico, but the eggs are highly desirable. In addition, there is little information on incidental capture of adults due to coastal fisheries off Mexico, but entanglement in longlines and driftnets probably account for some mortality of leatherback. The swordfish gillnet fisheries in Peru and Chile may have contributed to the decline of the leatherback in the eastern Pacific. The decline in the nesting population at Mexiquillo, Mexico occurred at the same time that effort doubled in the Chilean driftnet fishery.

Most conservation programs aimed at protecting nesting sea turtles in Mexico have continued since the early 1980s, and there is little information on the degree of poaching prior to the establishment of these programs. However, Sarti *et al.* (1998) estimates that as much as 100% of the clutches were taken from the Mexican beaches. Since protective measures have been in place, particularly emergency measures recommended by a joint U.S./Mexico leatherback working group meeting in 1999, there has been greater nest protection and nest success (Table 2). Mexican marines were present during the 1999-2000 season at three of the primary nesting beaches in Mexico (Llano Grande, Mexiquillo, and Tierra Colorado), responsible for approximately 34% of all nesting activity in Mexico. Of 1,294 nests documented, 736 were protected (57%), resulting in a

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

total of 25,802 hatchlings. Monitoring and protection measures at two secondary nesting beaches resulted in the protection of 67% and 10% at Barra de la Cruz and Playa Ventura, respectively. Currently, the primary management objective is to protect over 95% of nests laid at the three index beaches (includes protecting nesting females, eliminating illegal egg harvest, and relocating nests to protected hatcheries) and to maximize protection of all the secondary nesting beaches over the next three years. NMFS has committed funding for the next three years to help implement these objectives (minutes from joint U.S./Mexico Leatherback Working Group meeting, 23-24 May, 2000).

Table 2. Nest protection at index beaches on the Pacific coast of Mexico (Source: Sarti *et al*., personal communication, 2000)

Season	Number of clutches laid	Number of clutches protected	Percentage of clutches protected
1996-97	445	86	19.3%
1997-98	508	101	19.9%
1998-99	442	150	33.9%
1999-00	1590	943	58.7%

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

Western Pacific Populations of Leatherback Turtles

Similar to their eastern Pacific counterparts, leatherbacks originating from the western Pacific are also threatened by poaching of eggs, killing of nesting females, human encroachment on nesting beaches, incidental capture in fishing gear, beach erosion, and egg predation by animals. Little is known about the status of the western Pacific leatherback nesting populations but once major leatherback nesting assemblages are declining along the coasts of Malaysia, Indonesia and the Solomon Islands. Low density and scattered nesting of leatherback occurs in Fiji, Thailand, Australia, and Papua New Guinea. In Indonesia, low density nesting occurs along western Sumatra (200 females nesting annually) and in southeastern Java (50 females nesting annually), although the last known information is from the early 1980s (*in* Suarez and Starbird, 1996). The largest extant leatherback rookery in the Indo-Pacific lies on the north Vogelkop coast of Irian Jaya, with over 1,000 females nesting during the 1996 season (Suarez *et al.*, in press).

As with the eastern Pacific nesting populations, the decline of leatherback is severe at one of the most significant nesting sites in the region - Terengganu, Malaysia, with current nesting representing less than 2 percent of the levels recorded in the 1950s, and the decline is continuing. The nesting population at this location has declined from 3,103 females estimated nesting in 1968 to 2 nesting females in 1994 (Chan and Liew, 1996) (Table 3). With one or two females reportedly nesting each year, this population has essentially been eradicated (P. Dutton Declaration, June 9, 2000). Years of excessive egg harvest, egg poaching, the direct harvest of adults in this area as well as incidental capture in various fisheries in territorial and international waters have impacted the Malaysian population of leatherbacks. There were two periods in which there were sharp declines in nesting leatherbacks at this location: 1972-74 and 1978-80. Between 1972 and 1974, the number of females nesting declined 21% and coincided with a period of rapid development in the fishing industry, particularly trawling, in Terengganu (Chan *et al*., 1988 *in* Chan and Liew, 1996). Between 1978 and 1980, nestings dropped an average of 31% annually, and coincided directly with the introduction of the Japanese high seas squid fishery of the north Pacific in 1978 (Yatsu *et al*., 1991, *in* Chan and Liew, 1996). Because tagged individuals from Rantau Abang have been recovered from as far away as Taiwan, Japan and Hawaii, this fishery, as well as fisheries operating within the South China Sea, are presumed to have impacted the Malaysian leatherback population. After 1980, rates of decline averaged 16% annually, suggesting continuing threats from fisheries (Chan and Liew, 1996).

Table 3. Number of nesting females per year in Terengganu, Malaysia (summarized in Spotilla, *et al*., 1996)

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

Similarly, the nesting populations of leatherbacks in Irian Jaya, Indonesia are reported to be declining. Leatherback nesting generally takes place on two major beaches on the north Vogelkop coast of Irian Jaya, Jamursba-Medi and War-Mon beach. As shown in Table 4, Suarez, *et al*. (in press) has compiled, re-analyzed, and standardized data collected from leatherback nesting surveys in the 1980s and 1990s. In addition, Suarez *et al*. (in press) has included information on the estimated number of nests lost due to both natural and anthropogenic causes. For example, during 1984 and 1985, on Jamursba-Medi, 40-60% of nests were lost to inundation and erosion, while 90% of those nests not taken by poachers[7] or by the sea were destroyed by feral pigs (*Sus scrofa*). Eggs from poached nests were commercially harvested for sale in the Sarong markets until 1993, when the beaches first received protection by the Indonesian government (J. Bakarbessy, personal communication, *in* Suarez and Starbird, 1996). During the 1993-96 seasons, environmental education activities in nearby villages and protection measures on this same beach were put into place, with unreported results. Approximately 90% of those nests not taken by poachers or the sea[8] were destroyed by pigs (Suarez *et al.* in press). War-Mon beach supports a lower percentage of nesting females, yet egg poaching for subsistence accounted for over 60% of total nest loss during 1993-94, and loss of nests due to pig predation was 40% (because there are more people in this region, there is more pig hunting; hence less pig predation of leatherback eggs).

Table 4. Estimated numbers of female leatherbacks nesting along the north coast of Irian Jaya (Summarized by Suarez, *et al*., in press.)

Survey Period	**# of Nests**	**Adjusted # Nests**	**Estimated # of Females**
Jamursba-Medi Beach:			
September, 1981	4,000+	7,173[1]	1,232-1,623
April - Oct. 1984	13,360	13,360	2,303-3036
April - Oct. 1985	3,000	3,000	[(658)-731]
June - Sept. 1993	3,247	4,329[2]	746-984
June - Sept. 1994	3,298	4,397[2]	758-999

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

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

June - Sept. 1995	3,382	4,509[2]	777-1025
June - Sept., 1996	5,058	6,744[2]	1,163-1,533
War-Mon Beach:			
Nov. 1984 - Jan. 1985	1,012	N/A	175-230
Dec. 1993	406	653	128 - 169

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

[2] The total number of nests have been adjusted based on data from Bhaskar's surveys from 1984-85 from which it was determined that 25% of the total number of nests laid during the season (4/1-10/1) are laid between April and May.

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

In the Kai Islands, located approximately 1,000 kilometers southwest of the Irian Jaya nesting beaches, adult leatherback are traditionally hunted and captured at sea by local people. Villagers hunt leatherback only for ritual and subsistence purposes, and, according to their beliefs (known as *adat*), they are forbidden to sell or trade the meat. Based on a study conducted during October-November, 1994, Suarez and Starbird (1996) estimated that approximately 87 leatherback were taken annually by villagers in the Kai Islands, and this estimate did not include incidental take by local gill and shark nets. Locals report that sea turtle populations in the area have declined dramatically. Overall, approximately 200 leatherback are killed per year in traditional fisheries in Maluku, Indonesia (*in* Chan and Liew, 1996) (the Kai Islands take is assumed included in this estimate), and this take level is most likely continuing (C. Starbird, personal communication, 1998, *in* Clever Magazine, Issue No. 6).

As shown in Table 4, since the early-to-mid 1980s, the number of female leatherbacks nesting annually on the two primary beaches of Irian Jaya appear to be stable. However, given the current, serious threats to all life stages of the Indonesian leatherback populations, this trend may not be sustained and this population could collapse, similar to what occurred in Terrengganu, Malaysia. As human populations in Indonesia increase, the need for meat and competition between the expanding human population and turtles for space increases, all leading to more direct takes of leatherbacks or incidental take by local fisheries. There is no evidence to indicate that the preceding threats are not continuing today, as problems with nest destruction by feral pigs, beach erosion, and harvest of adults in local waters have been reported (Suarez et al., unpublished report). In addition, local Indonesian villagers indicate dramatic declines in local sea turtle populations; without adequate protection of nesting beaches, emerging hatchlings, and adults, this population will continue to decline.

Regarding the status of the Irian Jaya population of nesting leatherbacks, Suarez *et al*. (in press) comment: "Given the high nest loss which has occurred along this coast for over thirty years it is not unlikely that this population may also suddenly collapse. Nesting activity must also continue to be monitored along this coast, and nest mortality must be minimized in order to prevent this population of leatherbacks from declining in the future."

Conclusion on Status of Eastern and Western Pacific Leatherback Turtles

Although quantitative data on human-caused mortality are scarce, available information suggests that leatherback mortality on many nesting beaches remains at unsustainable levels (Tillman, 2000). In addition, except for elimination of fishing mortality in the now-defunct high-seas driftnet fisheries in the north and south Pacific, and reductions of effort in a few other fisheries, risks of mortality in fisheries generally have not been reduced.

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

There is substantial information on the nature and frequency of interactions between HMS fishing gear and leatherback turtles. The Hawaii-based longline fishery has been prohibited from making sets directed at swordfish to reduce takes of leatherback turtles (and loggerhead turtles as well); observer records indicated that, without this control, the fishery’s takes would result in jeopardy to the leatherback turtle. Similarly, the California/Oregon drift gillnet fishery is controlled in part to minimize the risk of excessive takes of leatherback sea turtles. At the same time, the West Coast-based longline fishery has been observed to take leatherback, and even at the low rate observed, it may be that allowing swordfish targeting, although limited to waters east of 150˚ West longitude, may result in estimates of leatherback takes that could result in a jeopardy finding under the ESA. This will be addressed in a consultation on the FMP. Estimates of takes are provided in Chapter 9.

c. *Loggerhead Turtle*

The loggerhead turtle is listed as threatened under the ESA throughout its range, primarily due to exploitation, incidental capture in various fisheries, and the alteration and destruction of its habitat. The loggerhead is categorized as Endangered, by the International Union for Conservation of Nature and Natural Resources where taxa so classified are considered to be facing a very high risk of extinction in the wild in the near future. Loggerheads are a cosmopolitan species, found in temperate and subtropical waters and inhabiting pelagic waters, continental shelves, bays, estuaries and lagoons . In the Pacific Ocean, major nesting grounds are generally located in temperate and subtropical regions, with scattered nesting in the tropics (*in* NMFS and USFWS, 1998c).

The loggerhead is characterized by a reddish brown, bony carapace, with a comparatively large head, up to 25 cm wide in some adults. Adults typically weigh between 80 and 150 kg, with average CCL measurements for adult females worldwide between 95-100 cm CCL (*in* Dodd, 1988) and adult males in Australia averaging around 97 cm CCL (Limpus, 1985, *in* Eckert, 1993). Juveniles found off California and Mexico measured between 20 and 80 cm (average 60 cm) in length (Bartlett, 1989, *in* Eckert, 1993).

Nesting of loggerheads in the Pacific Basin is restricted to the western and southern region (Japan and Australia, primarily); there are no reported loggerhead nesting sites in the eastern or central Pacific. Upon reaching maturity, adult females migrate long distances from resident foraging grounds to their preferred nesting beaches. The average re-migration interval is between 2.6 and 3.5 years, in Queensland, Australia (*in* NMFS and USFWS, 1998c). Nesting is preceded by offshore courting, and individuals return faithfully to the same nesting area over many years. Clutch size averages 110 to 130 eggs, and one to six clutches of eggs are deposited during the nesting season (Dodd, 1988). Based on skeletochronological and mark-recapture studies, mean age at sexual maturity for loggerheads ranges between 25 to 35 years of age, depending on the stock (*in* Chaloupka and Musick, 1997), although Frazer *et al*. (1994 *in* NMFS and USFWS, 1998c) determined that maturity of loggerheads in Australia occurs between 34.3 and 37.4 years of age.

The transition from hatchling to young juvenile may occur in the open sea and evidence is accumulating that this part of the loggerhead life cycle may involve trans-Pacific movement. Although the distribution of loggerheads in foraging areas is not well known for any population, it has been suggested that juvenile Pacific loggerheads follow a migration similar to loggerheads in the Atlantic. Hatchlings from the southeastern United States enter driftlines composed of *Sargassum* and other flotsam and are passively transported by currents in the north Atlantic gyre, perhaps one or more times, before taking up residence in developmental habitats in coastal waters of the eastern seaboard (Carr, 1987, *in* NMFS and USFWS, 1998c). The size structure of loggerheads in coastal and nearshore waters of the eastern and western Pacific suggest that Pacific loggerheads have a pelagic stage similar to the Atlantic. This is supported by the fact that the high seas driftnet fishery, which operated in the central north Pacific in the 1980s and early 1990s, incidentally caught juvenile loggerheads (mostly 40-70 cm in length) (Wetherall, *et al*., 1993). In addition, large aggregations of mainly juveniles and subadult loggerheads, numbering in the thousands, are found off the southwestern coast of Baja California, over 10,000 km from the nearest significant nesting beaches (Pitman, 1990). Genetic studies have shown these animals originate from Japanese nesting stock (Bowen *et al*., 1995), and their presence reflects a migration pattern probably related to their feeding habits (Cruz, *et al*., 1991, *in* Eckert, 1993). These loggerheads are primarily juveniles, although carapace length measurements indicate that some of them are 10 years old or older.

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

For their first years of life, loggerheads forage in open ocean pelagic habitats. Both juvenile and subadult loggerheads feed on pelagic crustaceans, mollusks, fish, and algae. The large aggregations of juveniles off Baja California have been observed foraging on dense concentrations of the pelagic red crab, *Pleuronocodes planipes* (Pitman, 1990), and preliminary data from stomach samples collected from turtles captured in north Pacific driftnets indicate a diet of gooseneck barnacles (*Lepas* sp.), pelagic purple snails (*Ianthina* sp.), and medusae (*Vellela* sp.) (G. Balazs, personal communication, *in* NMFS and USFWS, 1998c). As they age, loggerheads begin to move into shallower waters, where, as adults, they forage over a variety of benthic hard- and soft-bottom habitats (reviewed *in* Dodd, 1988). Most subadults and adults are found in nearshore benthic habitats around southern Japan, in the East China Sea and the South China Sea (e.g. Philippines, Taiwan, and Viet Nam).

Studies of loggerhead diving behavior indicate varying mean depths and surface intervals, depending on whether they were located in shallow coastal areas (short surface intervals) or in deeper, offshore areas (longer surface intervals). Loggerheads appear to spend a longer portion of their dive time on the bottom (or suspended at depth), which may be related to foraging and refuge. Unlike the leatherback, to the loggerhead foraging in the benthos, bottom time may be more important than absolute depth (Eckert, *et al*., 1989). The maximum recorded dive depth for a post-nesting female was 211-233 meters, while mean dive depths for both a post-nesting female and a subadult were 9-22 meters. Routine dive times for a post-nesting female were between 15 and 30 minutes, and for a subadult, between 19 and 30 minutes (Sakamoto, *et al*., 1990 *in* Lutcavage and Lutz, 1997).

In the western Pacific the only major nesting beaches are in the southern part of Japan (Dodd, 1988). An early estimate was that 2,000 to 3,000 female loggerheads may nest annually in all of Japan; however, more recent data suggest that only approximately 1,000 female loggerhead turtles may nest there (Bolten *et al*. 1996). Nesting beach monitoring at Gamoda (Tokushima Prefecture) has been ongoing since 1954. Surveys at this site showed a marked decline in the number of nests between 1960 and the mid-1970s. Since then, the number of nests has fluctuated, but has been downward since 1985 (Bolten *et al*., 1996). Monitoring on several other nesting beaches, surveyed since the mid-1970s, revealed increased nesting during the 1980s before declining during the early 1990s.

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

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

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

Loggerhead mortality from most human activities is not well-documented, except for estimates based on NMFS observer data in the Hawaii longline fishery and the CA/OR drift gillnet fishery. A high mortality in the north Pacific high-seas driftnet fisheries of Japan, Republic of Korea, and Taiwan was estimated in the 1990s, but those fisheries no longer operate. Mortality of loggerheads in the East China Sea and other benthic habitats of this population are a concern and thought to be "high," but have not been quantified (Kamezaki, personal communication, *in* Tillman, 2000).

As with leatherback turtles, there is substantial information about interactions between U.S. fisheries and loggerhead turtles. The Hawaii longline fishery has been prohibited from targeting on swordfish in part to reduce takes of loggerheads; the projected levels of take in the absence of this control were deemed to result in jeopardy. Similarly, the California/Oregon drift gillnet fishery is subject to time-area controls in El Niño years to prevent excessive takes of loggerheads. The West Coast-based longline fishery (as well as the Hawaii longline fishery) has been documented as having a significant level of take of loggerhead east of 150˚ W longitude. While the rate of interaction appears to decrease as fishing gets closer to the coast, it appears that the level of loggerhead turtle takes will still be significant. In the process of considering action on this FMP, the NMFS will conduct a Section 7 consultation under the ESA to determine if the fisheries as they would operate under the FMP will have adverse effects on loggerheads and possibly result in a jeopardy finding. If so, other corrective action will be needed. Estimates of take are presented in Chapter 9.

d. *Olive Ridley Turtle*

The olive ridley populations on the Pacific coast of Mexico are listed as endangered under the ESA; all other populations are listed as threatened. The olive ridley is categorized as endangered, by the International Union for Conservation of Nature and Natural Resources where taxa so classified are considered to be facing a very high risk of extinction in the wild in the near future. They are the smallest living sea turtle, with an adult carapace length between 60 and 70 cm, and rarely weighing over 50 kg. They are olive or grayish green above, with a greenish white underpart, and adults are moderately sexually dimorphic (NMFS and USFWS, 1998d).

Like leatherback turtles, most olive ridley turtles lead a primarily pelagic existence (Plotkin *et al*., 1993), migrating throughout the Pacific, from their nesting grounds in Mexico and Central America to the north Pacific. Surprisingly little is known of their oceanic distribution and critical foraging areas, despite being the most populous of north Pacific sea turtles. The species appears to forage throughout the eastern tropical Pacific Ocean, often in large groups, or flotillas, and are occasionally found entangled in scraps of net or other floating debris. In a three year study of communities associated with floating objects in the eastern tropical Pacific, Arenas and Hall (1992, *in* Eckert, 1993) found sea turtles, present in 15 percent of observations and suggested that flotsam may provide the turtles with food, shelter, and/or orientation cues in an otherwise featureless landscape. Olive ridleys comprised the vast majority (75%) of these sea turtle sightings. Small crabs, barnacles and other marine life often reside on the debris and likely serve as food attractants to turtles. Thus, it is possible that young turtles move offshore and occupy areas of surface current convergences to find food and shelter among aggregated floating objects until they are large enough to recruit to benthic feeding grounds of the adults, similar to the juvenile loggerheads mentioned previously. Olive ridleys feed on tunicates, salps, crustaceans, other invertebrates and small fish. Although they are generally thought to be surface feeders, olive ridleys have been caught in trawls at depths of 80-110 meters (NMFS and USFWS, 1998d), and a post-nesting female reportedly dove to a maximum depth of 290 meters. The average dive length for an adult female and adult male is reported to be 54.3 and 28.5 minutes, respectively (Plotkin, 1994, *in* Lutcavage and Lutz, 1997).

Olive ridley turtles are the most abundant sea turtle in the Pacific basin. Turtles begin to aggregate near the nesting beach two months before the nesting season, and most mating is generally assumed to occur in the vicinity of the nesting beaches, although copulating pairs have been reported over 100 km from the nearest nesting beach. The mean clutch size for females nesting on Mexican beaches is 105.3 eggs, in Costa Rica, clutch size averages between 100 and 107 eggs (*in* NMFS and USFWS, 1998d). Females generally lay two clutches of eggs per season in Costa Rica (Eckert, 1993). Data on the remigration intervals of olive ridleys are scarce.

In the eastern Pacific, nesting occurs all along the Mexico and Central American coast, with large nesting aggregations occurring at a few select beaches located in Mexico and Costa Rica. Where population densities are high enough, nesting takes place in synchronized aggregations known as *arribadas*. The largest known *arribadas* in the eastern Pacific are off the coast of Costa Rica (~475,000 - 650,000 females estimated nesting annually) and in southern Mexico (~600,000+ nests/year (Eckert, 1993; NMFS and USFWS, 1998d; Salazar *et al.*, in press). Historically, it was estimated that over 10 million olive ridleys inhabited the waters in the eastern Pacific off Mexico. However, human-induced mortality led to declines in this population. Beginning in the 1960's an enormous number of adult olive ridleys were harvested for commercial trade with Europe and Japan, several million olive ridleys were landed during the period 1960-1975. (NMFS and USFWS, 1998d). The nationwide ban on commercial harvest of sea turtles in Mexico, enacted in 1990, appears to have improved the situation for the olive ridley. Surveys of important olive ridley nesting beaches in Mexico indicate increasing numbers of nesting females in recent years (Marquez, *et al*., 1995). Annual nesting at the principal beach, Escobilla Beach, Oaxaca, Mexico, averaged 138,000 nests prior to the ban, and since the ban on harvest in 1990, annual nesting has increased to an average of 525,000 nests (Salazar, *et al*., in press). The greatest single cause of olive ridley egg loss comes from the nesting activity of conspecifics on *arribada* beaches, where nesting turtles destroy eggs by inadvertently digging up previously laid nests or causing them to become contaminated by bacteria and other pathogens from rotting nests nearby.

In the western Pacific, olive ridley nesting is known to occur on the eastern and western coasts of Malaysia; however, nesting has declined rapidly in the past decade. The highest density of nesting was reported to be in Terengganu, Malaysia, and at one time yielded 240,000 eggs (2,400 nests, with approximately 100 eggs per nest) (Siow and Moll, 1982, *in* Eckert, 1993)), while only 187 nests were reported from the area in 1990 (Eckert, 1993).

While olive ridleys generally have a tropical to subtropical range, individuals do occasionally venture north, some as far as the Gulf of Alaska. The post-nesting migration routes of olive ridleys, tracked via satellite from Costa Rica, traversed thousands of kilometers of deep oceanic waters ranging from Mexico to Peru and more than 3,000 kilometers out into the central Pacific (Plotkin *et al*. 1993). Stranding records from 1990-99 indicate that olive ridleys are rarely found off the coast of California, averaging 1.3 strandings annually (J. Cordaro, NMFS, personal communication, April, 2000).

Recent genetic information analyzed from 15 olive ridleys taken in the Hawaii-based longline fishery indicate that 9 of the turtles originated from the eastern Pacific and 6 of the turtles were from the southwest or Indo-Pacific (i.e. Malaysia) (P. Dutton, NMFS, personal communication, 1999). An olive ridley taken in the CA/OR drift gillnet fishery originated from an eastern Pacific stock (i.e. Costa Rica or Mexico) (P.Dutton, NMFS, personal communication, January, 2000).

Olive ridleys have been taken with some frequency by the Hawaii longline fishery but no takes have been observed in the West Coast-based longline fishery east of 150˚ W longitude. It is likely that the West Coast-based vessels would remain east of east of 150˚ W longitude under this FMP because of the availability of swordfish targeting if the FMP is approved. There would be a very low probability of olive ridley turtle takes in these waters.

4. SEABIRD SPECIES STATUS

a. Short-tailed Albatross

George Steller provided the first record of the short-tailed albatross in the 1740s. The type specimen for the species was collected offshore of Kamchatka, Russia, and was described in 1769 by P.S. Pallas in Specilegia Zoologica (AOU 1998). In the order of tubenose marine birds, Procellariiformes, the short-tailed albatross is classified within the family Diomedeidae. Until recently, it was assigned to the genus *Diomedea*. Following results of the genetic studies, the family Diomedeidae was arranged in four genera. The genus *Phoebastria*, north Pacific albatrosses, now includes the short-tailed albatross, the Laysan albatross (*P. immutabilis*), the black-footed albatross (*P. nigripes*), and the waved albatross (*P. irrorata*) (AOU 1998).

The short-tailed albatross is a large pelagic bird with long narrow wings adapted for soaring just above the water surface. The bill is disproportionately large compared to other northern hemisphere albatrosses; it is pink and hooked with a bluish tip, has external tubular nostrils, and has a thin but conspicuous black line extending around the base. Adult short-tailed albatrosses are the only northern Pacific albatross with an entirely white back. The white head develops a yellow-gold crown and nape over several years. Newly fledged birds are dark brown-black, but soon obtain pale bills and legs that distinguish them from black-footed albatross (Tuck 1978, Robertson 1980). Subadult birds have mixed white and brown-black areas of plumage, gradually getting more white feathers at each molt until reaching fully mature plumage.

Life History. Available evidence from historical accounts and from current breeding sites indicates that short-tailed albatross nesting habitat is characterized by flat or sloped sites with sparse or full vegetation on isolated windswept offshore islands with restricted human access (Arnoff 1960, Sherburne 1993, DeGange 1981). Current nesting habitat on Torishima Island is steep sites on soil containing loose volcanic ash; the island is dominated by a grass, *Miscanthus sinensis* var. *condensatus*, but a composite, *Chrysanthemum pacificum*, and a nettle, *Boehmeria biloba*, are also present (Hasegawa 1977). The grass probably stabilizes the soil, provides protection from weather, and minimizes mutual interference between nesting pairs while allowing for safe, open take-offs and landings (Hasegawa 1978). The nest is a grass or moss-lined concave scoop about 2 ft (0.75 m) in diameter (Tickell 1975).

Short-tailed albatrosses are long-lived and slow to mature; the average age at first breeding is about six years (USFWS 1999). As many as 25 percent of breeding age adults may not return to the colony in a given year (USFWS 1999; Cochrane and Starfield, in press.). Females lay a single egg each year, which is not replaced if destroyed (Austin 1949). Adult and juvenile survival rates are high (96 percent), and an average of 0.24 chicks per adult bird in the colony survive to fledge at six months of age (Cochrane and Starfield, in press.). However, chick survival can be reduced severely in years when catastrophic volcanic or weather events occur during the breeding season.

At Torishima, birds arrive at the breeding colony in October and begin nest building. Egg-laying begins in late October and continues through late November. The female lays a single egg; incubation involves both parents and lasts for 64-65 days. Eggs hatch in late December and January, and by late May or early June the chicks are almost fully grown and the adults begin abandoning their nests (USFWS 1999; Hasegawa and DeGange 1982). The only known currently active breeding colonies of short-tailed albatross are on Torishima and Minami-kojima islands, Japan. The chicks fledge soon after the adults leave the colony, and by mid-July, the colony is deserted (Austin 1949). Non-breeders and failed breeders disperse from the breeding colony in late winter through spring (Hasegawa and DeGange 1982). There is no detailed information on phenology on Minami-kojima, but it is believed to be similar to that on Torishima.

Short-tailed albatrosses are monogamous and highly philopatric to breeding sites. Chicks hatched at Torishima return there to breed. However, young birds may occasionally disperse from their natal colonies to breed, as evidenced by the appearance of adult birds displaying courtship behavior on Midway Atoll that were banded as chicks on Torishima (USFWS 1999, Richardson 1994).

The diet of short-tailed albatrosses includes squid, fish, flying fish eggs, shrimp and other crustaceans (USFWS 1999). There is currently no information on variation of diet by season, habitat, or environmental condition.

<u>Population Status and Distribution</u>. The species once ranged throughout most of the north Pacific Ocean and Bering Sea, with known nesting colonies on numerous western Pacific Islands in Japan and Taiwan (Hasegawa 1979, King 1981). Though other undocumented nesting colonies may have existed, there is no conclusive proof that short-tailed albatross once nested at locations beyond the colonies in Japan and Taiwan. Short-tailed albatross courtship behavior and reproductive activities have been observed at Midway Atoll NWR. The question of the future potential of Midway Atoll NWR to serve as a successful nesting colony, through either natural colonization or propagation efforts, remains unknown (USFWS 1999).

At the beginning of the 20th century, the species declined in population numbers to near extinction, primarily as a result of hunting at breeding colonies in Japan. Albatross were killed for their feathers and various other body parts. The feathers were used for writing quills, their bodies were processed for fertilizer, their fat was rendered, and their eggs were collected for food (Austin 1949). Hattori (in Austin 1949) commented that short-tailed albatrosses were "...killed by striking them on the head with a club, and it is not difficult for a man to kill between 100 and 200 birds daily." He also noted that the birds were "very rich in fat, each bird yielding over a pint."

Pre-exploration worldwide population estimates of short-tailed albatrosses are not known; the total number of birds harvested may provide the best estimate, as the harvest drove the species nearly to extinction. Between approximately 1885 and 1903, an estimated 5 million short-tailed albatrosses were harvested from the breeding colony on Torishima (Yamashina in Austin 1949), and harvest continued until the early 1930s, except for a few years following the 1903 volcanic eruption. One of the residents on the island, a schoolteacher, reported 3,000 albatrosses killed in December 1932 and January 1933. Yamashina (in Austin 1949) stated that "This last great slaughter was undoubtedly perpetrated by the inhabitants in anticipation of the island's soon becoming a bird sanctuary." By 1949, there were no short-tailed albatrosses breeding at any of the historically known breeding sites, including Torishima, and the species was thought to be extinct (Austin 1949).

In 1950, the chief of the weather station at Torishima, M. Yamamoto, reported nesting of the short-tailed albatross (Tickell 1973, 1975), and by 1954 there were 25 birds and at least 6 breeding pairs present on Torishima (Ono 1955). These were presumably juvenile birds that had been wandering the north Pacific during the final several years of slaughter. Since then, as a result of habitat management projects, stringent protection, and the absence of any significant volcanic eruption events, the population has gradually increased. Breeding-age population estimates come primarily from egg counts and breeding bird observations. The average growth of the colony on Torishima Island (the colony is called "Tsubamesaki") between 1950 and 1977 was 2.5 adults per year; between 1978 and 1991 the average population growth was 11 adults per year. An average annual population growth of at least 6 percent per year (Hasegawa 1982; Cochrane and Starfield, in press) resulted in a continuing increase in the breeding population to an estimated 440 breeding birds on Torishima in 1999 (USFWS, 1999). The population at Torishima is estimated to be growing at a rate of between 6.5 and 8.0% per year (USFWS , 1999). During the 2002 nesting season, there were 267 nesting pairs at Torishima, and the estimated total population at Torishima is 1,400 individuals, including an estimated 670 mature adults and 740 pre-breeders (Hasegawa, email to K. Rivera, NMFS, December 2002). With respect to other breeding sites, in 1971, 12 adult short-tailed albatrosses were discovered on Minami-kojima in the Senkaku Islands, one of the former breeding colony sites (Hasegawa 1984). Aerial surveys in 1979 and 1980 resulted in observations of between 16 and 35 adults. In April 1988, the first confirmed chicks on Minami-kojima were observed, and in March 1991, 10 chicks were observed. In 1991, the estimate for the population on Minami-kojima was 75 birds, including 15 breeding pairs (Hasegawa 1991). The most recent population estimate on Minami-kojima is 200-250 individuals (K. Rivera, NMFS, email, December 2002). The estimated world population of short-tailed albatrosses, calculated by combining the Torishima and Minami-kojima populations, is therefore about 1,600-1,650 birds. No measures of uncertainty are available for this estimate. There is no evidence that the breeding population on Torishima is nest site-limited at this point; therefore, ongoing management efforts focus on maintaining high rates of breeding success.

Torishima Island is under Japanese government ownership and management and is managed for the conservation of wildlife. Two management projects have been undertaken to enhance breeding success on Torishima. First, erosion control efforts at the Tsubamesaki colony have improved nesting success. Second, there are continuing attempts to establish a second breeding colony on Torishima by luring breeding birds to the opposite side of the island from the Tsubamesaki colony through the use of decoys and recorded colony sounds. Preliminary results of this experiment are promising; the first chick was fledged from this site in 1997. The expectation is that, absent a volcanic eruption or some other catastrophic event, the population on Torishima will continue to grow, and it will be many years before the breeding sites are limited (U.S. Fish and Wildlife Service 1999).

At-sea sightings since the 1940s indicate that the short-tailed albatross, while very small in number today, is distributed widely throughout its historical foraging range of the temperate and subarctic north Pacific Ocean (Sanger 1972; USFWS unpublished data) and has been observed close to the U.S. West Coast. In summer (i.e., non-breeding season), individuals appear to disperse widely throughout the historical range of the temperate and subarctic north Pacific Ocean (Sanger 1972), with observations concentrated in the northern Gulf of Alaska, Aleutian Islands, and Bering Sea (McDermond and Morgan 1993; Sherburne 1993; USFWS unpublished data). Individuals have been recorded along the west coast of North America as far south as the Baja Peninsula, Mexico.

Between the 1950s and 1970, there were few records of the species away from the breeding grounds, according to the AOU Handbooks of North American Birds (Vol. 1, 1962) and the Red Data Book (Vol. 2, Aves, International Union for the Conservation of Nature, Morges, Switzerland, 1966) (Tramontano 1970). In the northern Pacific, however, there were 12 reported marine sightings in the 1970s, 55 sightings in the 1980s, and over 250 sightings reported in the 1990s to date (Sanger 1972; Hasegawa and DeGange 1982, unpublished data). This observed increase in opportunistic sightings should be interpreted cautiously, however, because of the potential temporal, spatial, and numerical biases introduced by opportunistic shipboard observations. Observation effort, total number of vessels present, and location of vessels may have affected the number of observations independent of an increase in total numbers of birds present. On the other hand, the numbers are consistent with the apparent growth in the population in recent years.

Listing Status. Prior to its current listing as endangered throughout its range, the short-tailed albatross was listed as endangered under the Act, throughout its range, except in the U.S. During this period, the U.S. Fish and Wildlife Service (Service) considered the short-tailed albatross to be afforded protection under the Act in all portions of its range farther than 3 nautical miles (5.6 km) from U.S. shores, and included those waters of the EEZ (3-200 mi [5.6-370 km] from shore).

The exclusion of the U.S. from the range in which the species was listed resulted from an oversight in administrative procedures, rather than from any biological evaluation of the species' status within the U.S. The species was originally listed as endangered in accordance with the Endangered Species Conservation Act of 1969 (ESCA). Pursuant to the ESCA, two separate lists of endangered wildlife were maintained, one for foreign species and one for species native to the United States. The short-tailed albatross appeared only on the List of Endangered Foreign Wildlife (35 Federal Register [FR] 8495; June 2, 1970). When the current Act became effective on December 28, 1973, it superseded the ESCA. The native and foreign lists were combined to create one list of endangered and threatened species (38 FR 1171; January 4, 1974). When the lists were combined, prior notice of the action was not given to the governors of the affected states (Alaska, California, Hawaii, Oregon and Washington) as required by the Act, because available data were interpreted as not supporting resident status for the species. Thus, native individuals of this species were not formally proposed for listing pursuant to the criteria and procedures of the Act.

On July 25, 1979, the USFWS published a notice (44 FR 43705) stating that, through an oversight in the listing of the short-tailed albatross and six other endangered species, individuals occurring in the U.S. were not protected by the Act. The notice stated that it was always the intent of the USFWS that all populations and individuals of the seven species should be listed as endangered wherever they occurred. Therefore, the notice stated that the USFWS intended to take action as quickly as possible to propose endangered status for individuals occurring in the U.S.

On July 25, 1980, the USFWS published a proposed rule (45 FR 49844; July 25, 1980) to list, in the U.S., the short-tailed albatross and four of the six other species identified in the 1979 notice. No final action was taken on the July 25, 1980, proposal. The USFWS then designated the species as a candidate for listing in the U.S. (62 FR 49398; September 19, 1997). The USFWS published a proposal to list the short-tailed albatross as endangered in the U.S. (63 FR 58692) on November 2, 1998. A final rule was published on July 31, 2000 (65 FR 46643), listing the species as endangered throughout its range.

The Japanese government designated the short-tailed albatross as a protected species in 1958, as a Special National Monument in 1962 (Hasegawa and DeGange 1982), and as a Special Bird for Protection in 1972 (King 1981). Torishima was declared a National Monument in 1965 (King 1981). These designations have resulted in tight restrictions on human activities and disturbance on Torishima (USFWS 1999). In 1992, the species was classified as "endangered" under the then-newly implemented "Species Preservation Act" in Japan, which makes federal funds available for conservation programs and requires that a 10-year plan be in place, which sets forth conservation goals for the species. The current Japanese "Short-tailed Albatross Conservation and Management Master Plan" outlines general goals for continuing management and monitoring of the species, and future conservation needs (Environment Agency 1996). The principal management practices used on Torishima are legal protection, habitat enhancement, and population monitoring. Since 1976, Hasegawa has systematically monitored the breeding success and population numbers of short-tailed albatrosses breeding on Torishima.

There is no evidence that the West Coast-based longline fishery has had any interactions with short-tailed albatross in the past, though the potential for such interactions would presumably increase as the population of the species increases. The longline fishery in the western Pacific, and especially the fishery based in Hawaii, is subject to fishery conservation and management measures intended to minimize the risk of takes of short-tailed albatross in that fishery. These measures are described in Chapter 8 and would be applied to West Coast-based vessels wherever they fish if the FMP is approved. This should minimize the risk of adverse impacts on short-tailed albatross (and other albatrosses) from the West Coast-based fishery.

b. Bald Eagle

Habitat affinities. While not strictly a seabird, bald eagles nest in the tops of large trees and are strongly associated with freshwater and marine habitats, rarely nesting in Oregon further than one mile from water and their primary prey of fish (USFWS 1995). Bald eagles have been observed on marine shorelines in the northwest and a bald eagle was observed picking a dead or injured pollock out of marine waters off Juneau, Alaska (S. Fougner, pers. comm.). Bald eagle nests can be up to nine feet across and three feet deep, although in the Pacific Northwest nests are typically only five feet across (USFWS 1986). Foraging bald eagles require perch trees with an unobstructed view that are adjacent to the water, a dependence that makes bald eagles specifically vulnerable to aquatic-associated disturbance (USFWS 1995).

The bald eagle was listed as a threatened species in Oregon and Washington under the ESA. Listing was due to population declines resulting from, among other factors, habitat loss, shooting, electrocution, poisoning, and the adverse effects of the pesticide DDT (Washington Department of Fish and Wildlife 1990). A recovery plan for the bald eagle in the Pacific states was issued in 1986 in accordance with Section 4(f)(1) of the Act. The Pacific States Bald Eagle Recovery Plan established recovery zones for a seven-state Pacific Recovery Region. It outlined the following criteria for delisting the bald eagle in the Recovery Region (USFWS 1986):

1. There should be a minimum of 800 pairs nesting pairs in the Recovery Region.

2. These pairs should be producing an annual average of at least 1.0 fledged young per pair, with an average success rate per occupied territory of not less than 65 percent over a five-year period.

3. To ensure an acceptable distribution of nesting pairs, population recovery goals must be met in at least 80 percent of the management zones (*e.g* . 38 out of 47 Recovery Zones) identified in the Recovery Plan.

4. Wintering populations should be stable or increasing.

Currently available information indicates increasing bald eagle populations range wide. In the Pacific States Recovery Region, the number of occupied territories has consistently increased since 1986 to 1,482 pairs in 1998, thereby exceeding the 800 pair goal for 5 years beginning in 1990 when 861 territories were reported. However, distribution goals and nesting targets in several Recovery Zones have not been met. Productivity objectives have been met and averaged about 1.03 young per occupied territory since 1990. Currently, a proposal to delist the species in the lower 48 states has been under consideration by the Service since July 6, 1999 (USFWS 1999)

The Recovery Plan (USFWS 1986) guidelines recommend restricting human disturbance at bald eagle use areas by excluding activities that are withing 400 meters of nests and roosts during periods of eagle use. Disturbances could include timber harvest, blasting, firearms use, heavy machinery operation, camping or picnicking, etc. Further, the Recovery Plan recommends that these activities should also be regulated up to 800 meters from nests and roosts where eagles have line-of-sight vision. Critical nesting periods will vary, but generally fall between 1 January and 31 August.

There is no evidence that bald eagles have occurred in the range of the West Coast-based HMS fisheries. The specification of seabird conservation and management measures to limit takes of short-tailed albatross in the longline fishery would likely contribute to prevention of any interactions with bald eagles in the unlikely event that a bald eagle would range far enough from shore to potentially interact with longline gear. The likelihood of interaction with any other HMS gear is extremely low.

c. Brown Pelican

The brown pelican was federally listed as endangered in 1970 (35 FR 16047). The recovery plan describes the biology, reasons for decline, and actions needed for recovery of the brown pelican.

Description. The California brown pelican is one of six recognized subspecies of the brown pelican. The brown pelican is a large bird recognized by the long, pouched bill that is used to catch surface-schooling fishes. Unlike other brown pelican subspecies, the California brown pelican typically has a bright red gular pouch during courtship and egg-laying period. The California brown pelican weighs up to ten pounds and has a wingspan of up to eight feet. It can be distinguished from the eastern brown pelican by having a larger size and darker hindneck while in breeding plumage.

Habitat Affinities. The brown pelican is a conspicuous presence along the coasts of California and Baja California. Brown pelicans nest in colonies and on small coastal islands that are free of mammalian predators and human disturbance. They are associated with an adequate and consistent food supply and areas with appropriate roosting sites for both resident and migrant pelicans (USFWS 1983). During the non-breeding season, brown pelicans roost communally in areas that are near adequate food supplies, have some type of physical barrier to predation and disturbance, and that provide some protection from environmental stresses such as wind and high surf. Offshore rocks, breakwaters, and jetties are often used for roosting.

Life History. Brown pelicans are gregarious birds; they nest, forage, and roost communally. Brown pelicans forage for fish from the air, diving from heights of up to sixty feet. The offshore zone within 20-30 miles of colonies are critical for food supplies, especially when young are being fed. Coastal pelagic species, including anchovies and sardines, are the principal prey of brown pelicans.

Nest sites are generally on steep, rocky slopes. Nests are constructed on the ground or in brush of whatever materials are available, including grasses, sticks, feathers, and seaweed. Brown pelican nesting season typically begins in February and lasts until August. A brown pelican pair attends a clutch of two or three eggs, which are incubated alternatively by both parents. Young are born altricial and are fed by both parents. Chicks take about thirteen weeks to fledge, at which time they weigh about 20 percent more than adults. Brown pelicans reach breeding age in about three to five years.

Communal roost sites are essential habitat for brown pelicans at all times of year, throughout their range (USFWS 1983). Brown pelicans are unlike many seabirds in that they can have wet plumage and will become heavy and hypothermic in cold water if they do not come ashore regularly to dry and restore their plumage. Pelicans spread out to a larger number of roosts by day and gather into a smaller number of highest quality roosts at night.

Historic and Current Range. The breeding distribution of the California brown pelican ranges from the Channel Islands of southern California southward to the islands off Nayarit, Mexico. Prior to 1959, intermittent nesting was observed as far north as Point Lobos in Monterey County, California. Breeding populations can be differentiated into geographically separate entities that are isolated from each other by long stretches of uninhabited coastline. Some genetic exchange occurs among colonies by the recruitment of new breeders. When not breeding, pelicans may range as far north as Vancouver Island, British Columbia, Canada, and south to Colima, Mexico.

The maximum breeding population of the California brown pelican throughout its range may number about 55,000 to 60,000 pairs. The largest breeding group is located on the Gulf of California, comprising approximately 68 percent of the total breeding population. Only two breeding colonies exist in the U.S.; these are on Anacapa and Santa Barbara Islands. However, the majority of brown pelicans seen foraging along coastal California likely come from Mexico, as those pelicans tend to be more mobile. In the past, breeding occurred as far north as Point Lobos near Monterey.

Rangewide Trends and Current Threats. Brown pelican numbers and breeding success fluctuate greatly with the available food supply. Brown pelicans experienced widespread reproductive failures in the 1960s and early 1970s. Extremely low productivity in the early 1970s was attributed to eggshell thinning caused by high concentrations of DDE, a metabolite of DDT. Other factors implicated in the decline of this subspecies include human disturbance at nesting colonies and food shortages. As brown pelicans are well-studied on Anacapa Island, this account will use breeding success data from Anacapa as a surrogate for breeding trends rangewide. At Anacapa Island, only 4 young were fledged from 750 nesting attempts in 1969; the following year, only 1 young was fledged. With the prohibition on use of DDT in the U.S., brown pelicans numbers increased substantially. Productivity increased dramatically in 1974 and 1975 (305 and 256 young fledged on Anacapa Island), attributed to a increase in mean eggshell thickness and an increase in northern anchovy abundance. In 1977 and 1978, breeding productivity was low and nest abandonment was high; this was attributed to a declining anchovy population. Since 1979, productivity has ranged from 0.25 to 1.24 on Anacapa Island, with low numbers typically being a result of low anchovy numbers. Brown pelicans have not reached productivity targets identified for recovery; this is thought to be due to the increasing effect of human disturbance and its effect on the breeding success of colonies.

Current threats to the brown pelican include residual DDT in southern California, habitat degradation and mortality from oil spills, and human disturbance at roosting and breeding sites. The large commercial and sport fisheries around the Channel Islands may constitute a large disturbance to breeding colonies on Anacapa and Santa Barbara islands. However, little HMS fishing occurs around these islands or in other coastal areas where brown pelicans are most likely to be found. Recreational fishing for HMS also occurs relatively far from shore and beyond the normal range of brown pelicans. Thus it is not likely that the fisheries operating under this FMP will have any direct or indirect impacts on brown pelicans.

d. California Least Tern

The California least tern was federally listed as endangered in 1970 (35 FR 8495). Critical habitat has not been designated; however, a recovery plan for the species has been published (USFWS 1980).

Description. The California least tern is one of 12 recognized subspecies of the least tern, three of which inhabit the United States. Least terns are the smallest members of the subfamily Sterninae, measuring about 9 inches long with a 20-inch wingspan. Sexes look alike, being characterized by a black cap, gray wings with black wingtips, orange legs, and a black-tipped yellow bill. The California least tern cannot be reliably differentiated from other races of least tern on the basis of plumage characteristics alone (Burleigh and Lowery 1942).

Habitat Affinities. California least terns traditionally nest on open, sandy, ocean-fronting beaches that are often at the mouths of estuaries. For successful nesting, they require well-protected, undisturbed sites and an adequate food supply. Basic ecological requirements include a relatively flat, open area; a sandy or dried mud substrate; relative seclusion from disturbance and predation; and proximity to a lagoon or estuary with a dependable food supply. The creation of artificial breeding sites meeting these criteria has often been successful in attracting birds. California least terns have been able to find suitable conditions on airfields, landfills, salt evaporation pond dikes, saltflats, and vacant lots, as well as on the few remaining natural areas (*e.g.*, beaches) along the coast.

Life History. The California least tern is a migratory species which arrives in California by late April to breed and begins to depart to unknown southerly locations by August. California least terns nest on coastal, sandy, open areas, usually around bays, estuaries, and creek and river mouths. Nests are simply scrapes or depressions in the sand that the birds often adorn with small fragments of shell or pebbles. One to four eggs are laid, with two and three-egg clutches being most common. During the 21-day incubation period, the nest is tended continually. Both adults of a mated pair take turns tending the nest. Adults tend the flightless, but quite mobile chicks for approximately three weeks after hatching. After fledging, the young terns do not become fully proficient at capturing fish until after they migrate from the breeding grounds. Adults and fledglings usually leave the breeding colony within about 10 days of fledging.

California least terns obtain most of their food from shallow estuaries and lagoons. The California least tern has not been observed to feed on anything but fish. With the exception of spiny fish (*e.g.*, sculpins), least terns will catch any fish of suitable size. General size characteristics of suitable fish prey include a maximum body depth of about 0.5 inch and a maximum body length of about 3 inches. Much smaller fish, often fry, must be taken by adults to feed very small chicks. At least 50 species of forage fish have been identified from fish dropped at colony sites. The main food species taken varies among colonies, but usually includes northern anchovy (*Engraulis mordax*) and topsmelt (*Atherinops affinis*). Other locally important species include shiner surfperch (*Cymatogaster aggregata*), several goby species (notably the yellowfin goby (*Acanthogobius flavimanus*)), the longjaw mudsucker (*Gillichthys mirabilis*), California killifish (*Fundulus parvipinnis*), jacksmelt (*Atherinops californiensis*), and mosquitofish (*Gambusia affinis*).

Historic and Current Range. The historical breeding range of the California least tern has usually been described as extending along the Pacific Coast from Moss Landing, Monterey County, California, to San Jose del Cabo, Baja California Sur, Mexico. Since 1970, nesting sites have been documented in California from the San Francisco Bay area to the Tijuana River at the Mexican border. They have also been documented in Mexico, within the Gulf of California and on the western coast of Baja California, from Ensenada to San José del Cabo at the tip of the peninsula. Two nesting colonies in Baja California are identified in the early literature: Laguna Ojo de Liebre and San José del Cabo. In 1991 and 1992, a survey of the entire west coast of the peninsula documented 13 breeding colonies at 5 different locations from Ensenada through Bahía Magdalena, with 1 to 6 sites at each location.

The nesting range in California is widely discontinuous, with the majority of birds nesting in southern California from southern Santa Barbara County south through San Diego County. The breeding population has been distributed in five clusters along the coast: San Francisco Bay, San Luis Obispo/Santa Barbara County, Ventura County, Los Angeles/Orange County, and San Diego. The heaviest concentration of breeding California least terns occur in Los Angeles, Orange, and San Diego counties. The Santa Margarita River mouth in San Diego County generally has supported the largest numbers of terns in recent years. Between Ventura County and the San Francisco Bay area, Ormand Beach, Purisima Point, and Mussel Rock Dunes are the primary breeding areas.

Rangewide Trends and Current Threats. Although no reliable estimates are available relating to the historic, total population size of the California least tern, the species apparently was once abundant and well-distributed on barrier beaches and beach strand along the southern California coast. "Good-sized" colonies formerly were located in Los Angeles County. The species was noted as seriously declining in its range before the 1930's. From uncountable thousands of breeding pairs, the population was reduced to several hundred by the time of its listing as an Endangered Species in 1970. The decline of the California least tern is attributed to prolonged and widespread destruction and degradation of nesting and foraging habitats, and increasing human disturbance to breeding colonies.

The breeding population in California averaged approximately 600 pairs in 1973 through 1985, 664 pairs in 1976, and 775 pairs in 1977. The population more than doubled in the next seven years to 1,264 pairs. The major El Niño event of 1982/1983 led to a dramatic 25 percent decline to a low of 952 pairs. By 1994, the population increased to approximately 2,800 pairs, a population that was over four times the 1973 level. This trend continued, with 4,000 pairs being estimated in 1999. The demographic pattern of the California least tern’s range has not substantially changed; the increase in numbers generally has been distributed proportionally among the five breeding clusters. Currently, approximately 90 percent of the breeding pairs nest in southern California.

Conflicting uses of southern and central California beaches during the California least tern nesting season have led to isolated colony sites that are extremely vulnerable to predation from native, feral, and exotic species, overwash by high tides, and vandalism and harassment by beach users. Since its classification as a federal and state endangered species, considerable effort has been expended on annual population surveys, protection and enhancement of existing nesting colonies, and the establishment of new nesting locations. Control of predators, such as coyotes, foxes, and avian predators constitutes one of the most crucial management responsibilities at California least tern nesting sites. Predators on eggs include spotted skunk (*Spilogale putorius*), American crow (*Corvus brachyrhynchos*), northern raven (*Corvus corax*), coyote (*Canis latrans*), red fox (*Vulpes fulva*), rats (*Rattus* sp.), Beechey ground squirrel (*Spermophilus beecheyi*), and feral cat (*Felis domesticus*). Predators on chicks include American kestrel (*Falco tinnunculus*) , northern harrier (*Circus cyaneus*), loggerhead shrike (*Lanius ludovicianus*), gull-billed tern (*Sterna nilotica*), red fox, and feral cat. Fledglings and adults have been preyed upon by the American kestrel, peregrine falcon (*Falco peregrinus*), great horned owl (*Bubo virginianus*), burrowing owl (*Speotyto cunicularia*), and feral cat.

There is no evidence that California least terns have ever had any contacts with fisheries for HMS. Those fisheries generally occur far from shore and far from the feeding range of the least tern. The HMS fisheries operating under this FMP are therefore not expected to directly or indirectly affect the California least tern.

e. Western Snowy Plover

The Pacific coast population of the western snowy plover was federally listed as threatened on March 5, 1993 (58 FR 12864) and critical habitat was designated on December 7, 1999 (64 FR 68508). A recovery plan is currently being prepared.

Description. The western snowy plover is one of 12 subspecies of the snowy plover (*Charadrius alexandrinus*) in the family Charadriidae. The western snowy plover is a small, pale-colored shorebird with dark patches on either side of the upper breast. The bill and legs are blackish. Males have black head and breast markings in breeding plumage; in females, these markings are usually dark brown.

Habitat Affinities. Western snowy plovers prefer coastal beaches that are relatively free from human disturbance and predation. This habitat is unstable due to unconsolidated soils, high winds, storms, wave action, and colonization by plants. Sand spits, dune-backed beaches, beaches at creek and river mouths, and salt pans at lagoons and estuaries are the preferred habitats for nesting. The attributes considered essential to the conservation of the coastal population of the western snowy plover can be found in the final ruling for the designation of critical habitat (64 FR 68508). The physical and biological features and primary constituent elements of critical habitat are provided by intertidal beaches (between mean low water and mean high tide), associated dune systems, and river estuaries. Important components of the beach/dune/estuarine ecosystem include surf-cast kelp, sparsely vegetated foredunes, interdunal flats, spits, washover areas, blowouts, intertidal flats, salt flats, and flat rocky outcrops. Several of these components (sparse vegetation, salt flats) are mimicked in artificial habitat types used less commonly by western snowy plovers (*i.e.*, dredge spoil sites and salt ponds and adjoining levees).

Life History. The breeding season for western snowy plovers extends from March to late September, with birds at more southerly locations breeding earlier. Most nesting occurs on unvegetated or moderately vegetated, dune-backed beaches and sand spits. Other less common nesting habitats include salt pans, dredge spoils, and salt pond levees. Nest site fidelity is common, and mated birds from the previous breeding season frequently reunite. During courtship, males defend territories and usually make multiple scrapes. Females choose which scrape becomes the nest site by laying eggs (typically 3 but up to 6) in them. Both sexes incubate eggs, with the female tending to incubate during the day and the male at night. Snowy plovers often renest if eggs are lost. Hatching lasts from early April through mid-August, with chicks fledging approximately one month after hatching. Plover chicks are precocial, feeding on their own within hours of hatching. Adult plovers tend chicks while feeding, often using distraction displays to lure predators and people away from chicks. Females generally desert mates and broods by the sixth day after hatching, and thereafter the chicks are typically accompanied by only the male. While males rear broods, females obtain new mates and initiate new nests.

Western snowy plovers tend to be gregarious in winter. Western snowy plovers are primarily visual foragers, feeding on invertebrates in the wet sand and surf-cast kelp within the intertidal zone, in dry, sandy areas above the high tide, on salt pans, on spoil sites, and along the edges of salt marshes, salt ponds, and lagoons.

Historic and Current Range. The Pacific coast population of the western snowy plover breeds primarily on coastal beaches from southern Washington to southern Baja California, Mexico. Historically, western snowy plovers bred or wintered at 157 locations on the Pacific coast, including 5 sites in Washington, 19 sites in Oregon, and 133 sites in California. In Baja, western snowy plovers breed in coastal wetland complexes as far south as Bahia Magdalena. At present, fewer than 40 adults are believed to be nesting in Washington, slightly more than 100 in coastal Oregon, and fewer than 100 in California north of the Golden Gate. Larger numbers are found in southern and central California, in Monterey Bay (estimated 200 to 250 breeding adults), Morro Bay (estimated 85 to 93 breeding adults), Pismo Beach to Point Sal (estimated 130 to 246 breeding adults), Vandenberg Air Force Base (estimated 130 to 240 breeding adults), and the Oxnard Lowland (estimated 69 to105 breeding adults). In California, western snowy plovers also breed on San Nicolas and Santa Rosa islands, Bolsa Chica in Orange County, along the coast of San Diego County. Probably as many snowy plovers nest along the west coast of Baja California as along the U. S. Pacific Coast.

During the non-breeding season western snowy plovers may remain at breeding sites or may migrate to other locations. Most winter south of Bodega Bay, California. Many birds from the interior population winter on the central and southern coast of California.

Range-wide Trends and Current Threats. Historical records indicate that nesting western snowy plovers were once more widely distributed in coastal Washington, Oregon and California that they are currently. Only 1,200 to 1,900 adult western snowy plovers remain on the Pacific coast of the United States. In 1995, approximately 1,000 western snowy plovers occurred in coastal California. Historically, western snowy plovers bred at 53 coastal locations in California prior to 1970. Between 1970 and 1981, western snowy plovers stopped breeding in parts of San Diego, Ventura, and Santa Barbara counties, most of Orange County, and all of Los Angeles County. In 1995, 974 adult western snowy plovers were counted mid-breeding season in California coastal areas. Only eight sites continue to support 78 percent of the remaining California coastal breeding population. These are San Francisco Bay, Monterey Bay, Morro Bay, the Callendar-Mussel Rock dunes area, the Point Sal to Point Conception area (Vandenberg Air Force Base), the Oxnard lowland, Santa Rosa Island and San Nicolas Island.

The Pacific coast population of the western snowy plover has experienced widespread loss of nesting habitat and reduced reproductive success at many nesting locations due to urban development and the encroachment of European beachgrass (*Ammophila arenaria*). Human activities such as walking, jogging, unleashed pets, horseback riding, and off-road vehicles can destroy the western snowy plover's cryptic nests and chicks. These activities can also hinder foraging behavior, cause separation of adults and their chicks, and flush adults off nests and away from chicks, thereby interfering with essential incubation and chick-rearing behaviors. Predation by coyotes, foxes, skunks, ravens, gulls and raptors has been identified as a major factor limiting western snowy plover reproductive success at many Pacific coast sites.

There is no evidence that the western snowy plovers have ever had any contacts with fisheries for HMS. Those fisheries generally occur far from shore and far from the feeding range of the western snowy plover. The HMS fisheries operating under this FMP are not expected to directly or indirectly affect the western snowy plover.

f. Marbled Murrelet

The murrelet was federally listed as a threatened species in Washington, Oregon, and northern California effective September 28, 1992 (USDI 1992). Excessive harvest of late-successional and old-growth forests, the habitat preferred for nesting by murrelets, was the primary reason for the listing. Other factors include high predation rates, mortality in gillnets, and oil spill mortality. For a detailed discussion of the biology and status of the murrelet, refer to the *Federal Register* notice of determination of threatened status for the murrelet, final rule (USDI 1992); the Marbled Murrelet Recovery Plan (USDI 1997); Ecology and Conservation of the Marbled Murrelet (Ralph *et al.* 1995a); and the *Federal Register* notice of designation of critical habitat for the murrelet, final rule (USDI 1996).

Population Size and Distribution. In the past, the size of the listed population of the marbled murrelet in Washington, Oregon and California was estimated at 18,550-32,000 (Ralph *et al.* 1995b). The large range in the population estimate was the result of two widely divergent population estimates in Oregon. Population trend monitoring for the marbled murrelet, as part of effectiveness monitoring for the Northwest Forest Plan, began in 2000 (Bentivoglio *et al.* 2002). In addition, a separate population monitoring effort was conducted in Zone 6, which is not part of the Northwest Forest Plan area. Current population estimates and 95% confidence intervals (CI) by Marbled Murrelet Conservation Zone are: Zone 1 (5,635: 95% CI 3,198-8,453); Zone 2 (769: 95% CI 500-1,100); Zone 3 (6,738: 95% CI 3,940-11,707); Zone 4 (4,876: 95% CI 4,135-8,100); Zone 5 (79: 95% CI 13-168). Per Peery et *al.* (2002), the estimate for Zone 6 was 474 birds with a 95% CI of 337-668. Unfortunately, data from Marbled Murrelet Conservation Zone 2 was analyzed differently than the other Marbled Murrelet Conservation Zones due to an unanticipated complication in data collection. It is assumed the estimate is likely to be an underestimate because the data were not corrected for detectability, however, the magnitude of the underestimate is unknown (J. Baldwin, pers. comm. 2002). It is important to note that although these data cannot be directly compared to previous estimates, they can be used to illustrate how new information on population baseline numbers reveals impacts the Service may not have previously considered in biological opinions.

Preliminary population estimates for 2001, by Marbled Murrelet Conservation Zone are: Zone 1 (8,421: 95% CI 5,506-11,882); Zone 2 (1,963: 95% CI 573-3,725); Zone 3 (6,879: 95% CI 5,389-9,243); Zone 4 (3,888: 95% CI 2,901-6,567); Zone 5 (117: 95% CI 30-276) (Jodice *et al.* 2002); Zone 6 (615: 95% CI 515-733) (Peery *et al.* 2002).

Given that it is premature to try and detect biologically meaningful trends in population size with only 2 years of monitoring, the best available information on range-wide population trends remains summarized in the murrelet recovery plan (USDI 1997). Both Ralph *et al.* (1995b) and the Marbled Murrelet Recovery Team (1994) have concluded that the listed population appears to be in a long-term downward trend. Best estimates show that the population may be declining at a rate of 4 to 7 percent per year, and perhaps as much as 12 percent per year (Beissinger 1995). A downward trend of that magnitude means that in 20 years the population could be less than one-half to one-twelfth its current size.

The Recovery Plan states that four of the six Marbled Murrelet Conservation Zones must be functional in order to effectively recover the murrelet both in the short- and long-term, that is, maintain viable populations that are well-distributed. However, based on the new population estimates and recent radio telemetry work, it appears only three of the Marbled Murrelet Conservation Zones contain relatively robust numbers of murrelets (Marbled Murrelet Conservation Zones 1, 3, and 4). Two of these Marbled Murrelet Conservation Zones, however, have experienced oil spills within the last five years, resulting in significant marbled murrelet mortality (Zones 3 and 4).

Zone 1

The 2001 population point estimate for Marbled Murrelet Conservation Zone 1 is much larger than the 2000 estimate and the 95 percent confidence intervals are both quite large. It is difficult to interpret or compare these numbers, but relative to Marbled Murrelet Conservation Zones 2, 5 and 6, Marbled Murrelet Conservation Zones 1 and 3 appear to contain larger, more robust populations.

Zone 2

The 2000 population point estimate for Marbled Murrelet Conservation Zone 2 (769 birds) is much lower than the 2,400 reported in Ralph *et al.* (1995a), the previous best estimate, and the number available for use as a baseline for biological opinions written between 1995 and 2001. Taking into consideration that the 2000 population estimate for Marbled Murrelet Conservation Zone 2 may be low, a conservative look at the high end of the 95 percent confidence interval (1,100 birds) reveals a total number of birds that is less than half the previous best estimate. The 2001 population point estimate for Marbled Murrelet Conservation Zone 2 (1,963: 95% CI 573-3,725) is much higher than the 2000 estimate (769: 95% CI 500-1,100), falling outside the upper end of the 95 percent confidence interval for 2000. However, the lower end of the 95 percent confidence intervals for 2000 and 2001 are similar. Given differences in methodologies, and the high degree of variation in the point estimates and their associated wide confidence intervals, it is difficult to interpret or compare these numbers. Until a third year of data has been collected and analyzed, concern for the potentially small population size in Marbled Murrelet Conservation Zone 2 is warranted.

Zone 3

In February and March of 1999, the New Carissa oil spill occurred in Marbled Murrelet Conservation Zone 3 near Coos Bay, Oregon; an estimated 262 Marbled murrelets were killed impacting approximately 4 percent of the population in Marbled Murrelet Conservation Zone 3 (Ford *et al.* 2001a). The 2000 and 2001 population point estimates for Marbled Murrelet Conservation Zone 3 are very similar and the 95 percent confidence intervals are both quite large.

Zone 4

The M/V Kure oil spill occurred in Humboldt Bay in November 1997. The bodies of 10 marbled murrelets were recovered during cleanup and recovery efforts. It was assumed that mortality from the Kure spill was “probably several times higher” than the documented mortality of nine murrelets. The new estimate is approximately 15 times higher than the known mortality of 10 murrelets (Ford *et al.* 2001b). The Stuyvesant oil spill occurred in September 1999 at the entrance to Humboldt Bay. The bodies of 24 murrelets were recovered during the cleanup and recovery efforts. Preliminary modeling efforts have been completed and the estimated mortality is approximately fifteen times the known mortality of 24 murrelets. In total, the M/V Kure and the Stuyvesant oil spills are estimated to have killed 10 percent of the population in Marbled Murrelet Conservation Zone 4 based on the year 2000 population estimate of more than 4,876 birds (Bentivoglio *et al.* 2002). These impact estimates are for direct mortality only; oil can have a number of adverse effects on seabirds other than direct mortality, but these effects have not yet been quantified for either of the spills. Juvenile-to-adult ratios for Marbled Murrelet Conservation Zone 4 based on surveys conducted from 1993 through 2001 (Table 2) continue to be too low to support a stable or increasing population (mean 0.042) (Ralph *et al.* 2001).

Zone 5
The population in Marbled Murrelet Conservation Zone 5 (2000 point estimate of 78 birds; 2001 point estimate of 117 birds) is extremely low, though the Zone was not expected to substantially contribute to recovery.

Zone 6
Marbled Murrelet Conservation Zone 6 is not covered by monitoring from the Northwest Forest Plan, but independent research conducted in Marbled Murrelet Conservation Zone 6 provides reliable population estimates. Additional new information from a radio telemetry study in this Zone indicates a general failure to successfully breed, and that the population in Marbled Murrelet Conservation Zone 6 is highly endangered. Juvenile-to-adult ratios derived from surveys at sea are alarmingly low (0.02) and further indicate a general failure of successful reproduction (Peery *et al.* 2002). Although the recovery plan acknowledged the vulnerability of Zone 6, it assumed Marbled Murrelet Conservation Zone 6 would contribute to recovery in the short-term (50-100 years). Recent evidence does not support this assumption.

Life history and productivity. Juvenile-to-adult ratios obtained in the marine environment are currently the only parameter available to evaluate productivity. Beissinger (1995) estimated that the rangewide juvenile-to-adult ratio for murrelets is about 5 percent. This is a low level of productivity. One reason for such a low productivity level is the fact that murrelets experience high rates of predation, particularly of their eggs and young. In fact, predation is the number one cause of nest failure (Nelson and Hamer 1995). Predation rates are influenced mainly by habitat stand size, habitat quality, nest placement (on edge of stand vs. interior of stand), and proximity of the stand to humans. As stands of habitat get smaller, their quality decreases as the amount of forest edge increases in relation to the amount of interior forest. Nests placed near the edge of a stand are more likely to be subject to predation (Ralph *et al.* 1995b). If a stand of habitat has more edge than interior, the likelihood increases that a murrelet will place its nest near the edge. Forest stands near (<1 kilometer) to human activity centers can experience increased nest predation because supplemental feeding by humans attracts corvids (Marzluff *et al.* 2000).

Distribution. The distribution of the marbled murrelet population has been significantly reduced as population numbers have declined, and suitable nesting habitat for murrelets has virtually disappeared at some locations. Several areas of concern have been identified where only small numbers of murrelets persist (USDI 1997) and maintenance of the species' distribution is at risk. These include portions of central California, northwestern Oregon, and southwestern Washington, where very little suitable habitat remains, and what does remain is in small, fragmented patches.

Habitat affinities. The loss of nesting habitat (old-growth/mature forest) has generally been identified as the primary cause of the marbled murrelet population decline and disappearance across portions of its range (Ralph *et al.* 1995b). The loss of habitat has been due primarily to extensive timber cutting, particularly over the past 150 years.

Marbled murrelets are dependent upon old-growth forests, or forests with an older tree component, for nesting habitat (Ralph *et al.* 1995b). Sites occupied by marbled murrelets tend to have a higher proportion of mature forest classes than do non-occupied sites. To date, more than 82 to 87 percent of the old-growth forests that existed in western Washington and Oregon prior to the 1840s are now gone (Booth 1991; Teensma *et al.* 1991; Ripple 1994).

Washington has approximately 977,811 acres, Oregon has approximately 565,185 acres, and California has approximately 819,472 acres, for a total of 2,362,469 acres of marbled murrelet habitat. These estimates are largely based upon interpretations of satellite imagery and have not been thoroughly ground-truthed. They also refer to quantity of potential habitat, not quality. Murrelet habitat quality depends on proximity to the coast, landscape context, and stand size. Quality habitat must meet basic nesting requirements, provide refuge from predators, and be relatively stable against catastrophic disturbances. It is not possible at this time to estimate with confidence the proportion of remaining habitat that could be considered of high enough quality to allow long-term nesting success.

Based on past analyses by the USDA and USDI (1994) analyses, the Service concludes that the actual amount of quality nesting habitat available to murrelets in Washington, Oregon, and California is less than 2,362,469 acres. The actual amount of suitable nesting habitat could be significantly less, but the Service currently does not have the information to quantify this figure.

There is no evidence that murrelets have ever had any interactions with fisheries for HMS. Those fisheries generally occur far from shore and far from the feeding range of the murrelet. The HMS fisheries are not expected to directly or indirectly affect the marbled murrelet.

g. California Clapper Rail

The clapper rail was federally listed as endangered in 1970 (35 FR 1604). A detailed account of the taxonomy, ecology, and biology of the clapper rail is presented in the approved Recovery Plan for this species (Service 1984). The clapper rail was listed as endangered primarily as a result of habitat loss. The factors described above have contributed to the more recent population reduction, which has occurred since the mid-1980s. Although many factors are at work, predation by native and non-native predators, in conjunction with extensive habitat loss and fragmentation, are the current primary threats. With historic populations at Humboldt Bay, Elkhorn Slough, and Morro Bay now extinct, San Francisco Bay represents the last stronghold and breeding population of this subspecies.

Distribution. Of the 193,800 acres of tidal marsh that bordered the Bay in 1850, about 30,100 acres currently remain. This represents an 84 percent reduction from historical conditions. Furthermore, a number of factors influencing remaining tidal marshes limit their habitat values for clapper rails. Much of the East Bay shoreline from San Leandro to Calaveras Point is rapidly eroding, and many marshes along this shoreline could lose their clapper rail populations in the future, if they have not already. In addition, an estimated 600 acres of former salt marsh along Coyote Creek, Alviso Slough, and Guadalupe Slough, has been converted to fresh- and brackish-water vegetation due to freshwater discharge from South Bay wastewater facilities and is of lower quality for clapper rails. This conversion has at least temporarily stabilized as a result of the drought since the early 1990s.

Habitat affinities. The suitability of many marshes for clapper rails is further limited, and in some cases precluded, by their small size, fragmentation, and lack of tidal channel systems and other micro-habitat features. These limitations render much of the remaining tidal marsh acreage unsuitable or of low value for the species. In addition, tidal amplitudes are much greater in the South Bay than in San Pablo or Suisun bays. Consequently, many tidal marshes are completely submerged during high tides and lack sufficient escape habitat, likely resulting in nesting failures and high rates of predation. The reductions in carrying capacity in existing marshes necessitate the restoration of larger tracts of habitat to maintain stable populations.

Threats. Throughout the Bay, the remaining clapper rail population is besieged by a suite of mammalian and avian predators. At least 12 native and 3 non-native predator species are known to prey on various life stages of the clapper rail. Artificially high local populations of native predators, especially raccoons, result as development occurs in the habitat of these predators around the Bay margins. Encroaching development not only displaces lower order predators from their natural habitat, but also adversely affects higher order predators, such as coyotes, which would normally limit population levels of lower order native and non-native predators, especially red foxes. Hunting intensity and efficiency by raptors on clapper rails also is increased by electric power transmission lines, which criss-cross tidal marshes and provide otherwise-limited hunting perches. Non-native Norway rats (*Rattus norvegicus*) long have been known to be effective predators of clapper rail nests. Placement of shoreline riprap favors rat populations, which results in greater predation pressure on clapper rails in certain marshes. These predation impacts are exacerbated by a reduction in high marsh and natural high tide cover in marshes.

The proliferation of non-native red foxes into tidal marshes of the South Bay since 1986 has had a profound effect on clapper rail populations. As a result of the rapid decline and almost complete elimination of rail populations in certain marshes, the San Francisco Bay National Wildlife Refuge (Refuge) implemented a predator management plan in 1991 (Foerster and Takekawa 1991) with an ultimate goal of increasing rail population levels and nesting success through management of red fox predation. This program has proven successful in increasing the overall South Bay populations from an all-time low (see below); however, it has been difficult to effectively conduct predator management over such a large area as the South Bay, especially with the many constraints associated with conducting the work in urban environments.

Predator management for clapper rails is not being regularly practiced in the North Bay, and clapper rail populations in this area remain susceptible to red fox predation. Red fox activity has been documented west of the Petaluma River and along Dutchman Slough at Cullinan Ranch. Along Wildcat Creek near Richmond, where recent red fox activity has been observed, the rail population level in one tidal marsh area has declined considerably since 1987, even though limited red fox management was performed in 1992 and 1993.

Mercury accumulation in eggs is perhaps the most significant contaminant problem affecting clapper rails in San Francisco Bay, with the South Bay containing the highest mercury levels. Mercury is extremely toxic to embryos and has a long biological half-life. The Service collected data from 1991 and 1992 on mercury concentrations in rail eggs in the southern portion of the Estuary and found that the current accumulation of mercury in rail eggs occurs at potentially harmful levels. The percentage of non-viable eggs ranged from 24 to 38 percent (mean = 29 percent).

Research in a North Bay marsh suggests that the clapper rail breeding season, including pair bonding and nest construction, may begin as early as February. Field observations in South Bay marshes suggest that pair formation also occurs in February in some areas. The end of the breeding season is typically defined as the end of August, which corresponds with the time when eggs laid during re-nesting attempts have hatched and young are mobile.

There is no evidence that clapper rail have ever had any interactions with HMS fisheries. Those fisheries generally occur far from shore and far from the feeding range of the clapper rail. The HMS fisheries are therefore not expected to directly or indirectly affect the clapper rail.

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