LOGGERHEAD SEA TURTLE *(CARETTA CARETTA)*
2009 STATUS REVIEW
UNDER THE U.S. ENDANGERED SPECIES ACT

LOGGERHEAD BIOLOGICAL REVIEW TEAM

Therese A. Conant, Peter H. Dutton, Tomoharu Eguchi, Sheryan P. Epperly,
Christina C. Fahy, Matthew H. Godfrey, Sandra L. MacPherson, Earl E. Possardt,
Barbara A. Schroeder, Jeffrey A. Seminoff, Melissa L. Snover,
Carrie M. Upite, and Blair E. Witherington

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LOGGERHEAD BIOLOGICAL REVIEW TEAM

Therese A. Conant
National Marine Fisheries Service
Office of Protected Resources

Peter H. Dutton
National Marine Fisheries Service
Southwest Fisheries Science Center

Tomoharu Eguchi
National Marine Fisheries Service
Southwest Fisheries Science Center

Sheryan P. Epperly
National Marine Fisheries Service
Southeast Fisheries Science Center

Christina C. Fahy
National Marine Fisheries Service
Southwest Regional Office

Matthew H. Godfrey
North Carolina Wildlife Resources Commission

Sandra L. MacPherson
U.S. Fish and Wildlife Service

Earl E. Possardt
U.S. Fish and Wildlife Service

Barbara A. Schroeder
National Marine Fisheries Service
Office of Protected Resources

Jeffrey A. Seminoff
National Marine Fisheries Service
Southwest Fisheries Science Center

Melissa L. Snover
National Marine Fisheries Service
Pacific Islands Fisheries Science Center

Carrie M. Upite
National Marine Fisheries Service
Northeast Regional Office

Blair E. Witherington
Florida Fish and Wildlife Conservation
Commission
Fish and Wildlife Research Institute

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EXECUTIVE SUMMARY

The loggerhead sea turtle (*Caretta caretta*) was listed as threatened on July 28, 1978. The threatened listing was applied to wherever the species occurs. In 2007, the National Marine Fisheries Service (NMFS) and the U.S. Fish and Wildlife Service (FWS) completed a 5-year review for the loggerhead. A 5-year review is an assessment of a listed species to determine whether its status has changed since the time of its listing such that it should be delisted or classified differently than its current status. The agencies concluded that new information available since the completion of the previous 5-year review in 1995 indicated a possible separation of populations by ocean basins but that a more in-depth analysis was needed to determine the application of the distinct population segment (DPS) policy. Based on the new information and the need for further analysis under the DPS policy, NMFS and FWS recommended that no change in listing status was warranted. However, they committed to fully assemble and analyze all relevant information in accordance with the DPS policy.

On July 16, 2007, NMFS and FWS received a petition from the Center for Biological Diversity and Turtle Island Restoration Network requesting that loggerhead turtles in the North Pacific be reclassified as a DPS with endangered status and that critical habitat be designated. On November 16, 2007, NMFS and FWS received a petition from the Center for Biological Diversity and Oceana requesting that loggerhead turtles in the Northwest Atlantic Ocean be reclassified as a DPS with endangered status and that critical habitat be designated. NMFS and FWS determined that the July 16, 2007, North Pacific petition and the November 16, 2007, Northwest Atlantic petition both presented substantial information that the petitioned actions may be warranted. In the published 90-day petition findings, NMFS and FWS committed to assess the loggerhead listing status on a global basis, which is consistent with the recommendation in the 2007 5-year review.

NMFS and FWS convened a biological review team (BRT) in February 2008 to review the best available scientific information, determine whether DPSs exist, and assess the extinction risk for each potential DPS. The BRT organized their evaluation by ocean basin: Pacific Ocean, Atlantic Ocean (including the Mediterranean Sea), and Indian Ocean. This approach was consistent with the 2007 5-year review for the loggerhead and was chosen to facilitate data assembly and evaluation. It was not meant to preclude identification of DPSs on a broader or finer scale.

The NMFS and FWS 1996 DPS policy defines a population to be a "distinct population segment" if it is both **discrete** and **significant** relative to its taxon. The BRT evaluated genetic evidence, tagging (flipper and PIT tags) and satellite telemetry data, demographics information, oceanographic features, and geographic barriers, and determined that there are at least nine **discrete** population segments for loggerhead sea turtles globally. These discrete population segments are markedly separated from each other as a consequence of ecological, behavioral, and oceanographic factors, and based on genetic evidence. Therefore, the BRT unanimously concluded that two discrete population segments exist in the Pacific Ocean, three in the Indian Ocean, and four in the Atlantic Ocean/Mediterranean Sea.

The BRT then considered whether each of the nine identified discrete population segments is **significant** relative to its taxon. The BRT determined that each of the nine discrete population segments were biologically and ecologically significant. They each represent a large portion of the species range, sometimes encompassing an entire hemispheric ocean basin. The range of each discrete population segment represents a unique ecosystem, influenced by local ecological and physical factors. The loss of any single discrete population segment would result in a significant gap in the loggerhead's range. Each discrete population segment is genetically unique; the loss of any one discrete population segment would represent a significant loss of genetic diversity. Therefore, the BRT concluded that the nine identified population segments are both **discrete** from other conspecific population segments and **significant** to the species to which they belong, *Caretta caretta*.

The BRT has identified the following nine loggerhead DPSs distributed globally:
(1) North Pacific Ocean DPS,
(2) South Pacific Ocean DPS,
(3) North Indian Ocean DPS,
(4) Southeast Indo-Pacific Ocean DPS,
(5) Southwest Indian Ocean DPS,
(6) Northwest Atlantic Ocean DPS,
(7) Northeast Atlantic Ocean DPS,
(8) Mediterranean Sea DPS, and
(9) South Atlantic Ocean DPS.

The BRT next conducted two independent analyses to assess extinction risks of the identified DPSs. The first analysis used the diffusion approximation approach based on a time series of counts of nesting females or nests to provide a metric, susceptibility to quasi-extinction (SQE). SQE is an increasing function of quasi-extinction threshold (QET). The second analysis focused on determining the effects of known anthropogenic mortality on each potential DPS with respect to the vital rates of the species. This approach focused on how additional mortalities may affect the future growth and status of each DPS.

For three of five DPSs with sufficient data to conduct the SQE analysis (Northwest Atlantic Ocean, South Pacific Ocean, and North Pacific Ocean), the threshold of SQE = 0.3 was reached at QET < 0.3, indicating high likelihood of quasi-extinction over a wide range of QET values. There were not enough data to conduct the SQE analysis for the North Indian Ocean, Southeast Indo-Pacific Ocean, Northeast Atlantic Ocean, and Mediterranean Sea DPSs.

According to the analysis using experts' opinions in the matrix model framework, all loggerhead turtle DPSs have the potential to decline in the future. Although some DPSs are indicating increasing trends at nesting beaches (Southwest Indian Ocean and South Atlantic Ocean), available information about anthropogenic threats to juveniles and adults in neritic and oceanic environments indicate possible unsustainable additional mortalities. According to the threat matrix analysis, the potential for future decline is greatest for the North Indian Ocean, Northwest Atlantic Ocean, Northeast Atlantic Ocean, Mediterranean Sea, and South Atlantic Ocean DPSs.

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LIST OF ACRONYMS AND ABBREVIATIONS

The following are standard abbreviations for acronyms and terms found throughout this document:

AFR	Age at First Reproduction
ASEAN	Association of South East Asian Nations
BRT	Biological Review Team
CBD	Convention on Biological Diversity
CCL	Curved Carapace Length
CI	Confidence Interval
CITES	Convention on the International Trade in Endangered Species of Wild Fauna and Flora
CMS	Convention on the Conservation of Migratory Species of Wild Animals
COFI	FAO Committee on Fisheries
CPPS	Commision Permanente del Pacifico Sur
CV	Coefficient of Variation
DNA	Deoxyribonucleic Acid
DPS	Distinct Population Segment
ENSO	El Niño Southern Oscillation
ESA	Endangered Species Act of 1973, as amended
FAO	Food and Agriculture Organization of the United Nations
FWS	U.S. Fish and Wildlife Service
GBR	Great Barrier Reef
GCRU	Greater Caribbean Recovery Unit
GIWA	Global International Waters Assessment
IAC	Inter-American Convention for the Conservation and Protection of Marine Turtles
ICCAT	International Commission for the Conservation of Atlantic Tunas
IOSEA	Indian Ocean – South-East Asian Marine Turtle Memorandum of Understanding
IPCC	Intergovernmental Panel on Climate Change
LME	Large Marine Ecosystem
MARPOL	International Convention for the Prevention of Pollution from Ships
MOU	Memorandum of Understanding
MSA	U.S. Magnuson-Stevens Fishery Conservation and Management Act
mtDNA	Mitochondrial Deoxyribonucleic Acid
NBSD	Negative Binomial Stage Distribution
NGMRU	Northern Gulf of Mexico Recovery Unit
NMFS	National Marine Fisheries Service
NPF	Northern Australian Prawn Fishery
NRU	Northern Recovery Unit
PIT	Passive Integrated Transponder
PFRU	Peninsular Florida Recovery Unit
PVA	Population Viability Analysis
QET	Quasi-extinction Threshold
RFMO	Regional Fishery Management Organizations
SCL	Straight Carapace Length
SD	Standard Deviation

SEAFO	South-East Atlantic Fisheries Organization
SPAW	Protocol Concerning Specially Protected Areas and Wildlife
SPOIR	Significant Portion of its Range
SPREP	South Pacific Regional Environment Programme
SQE	Susceptibility to Quasi-extinction
STAJ	Sea Turtle Association of Japan
TED	Turtle Excluder Device
UNCLOS	United Nations Convention on the Law of the Sea
UNEP	United Nations Environment Programme

SECTION 1—INTRODUCTION AND BACKGROUND

1.1. ESA Overview

1.1.1. Purpose

The purpose of the Endangered Species Act of 1973, as amended (ESA) (16 U.S.C. 1531 *et seq*.) is to provide a means to conserve ecosystems upon which endangered and threatened species depend, to provide a program for the conservation of endangered and threatened species, and to take appropriate steps to recover endangered and threatened species. The National Marine Fisheries Service (NMFS) and U.S. Fish and Wildlife Service (FWS) share responsibility for administering the ESA. NMFS and FWS are responsible for determining whether species, subspecies, or distinct population segments of vertebrate species are threatened or endangered under the ESA. FWS typically has the lead for terrestrial and freshwater species, and NMFS typically has the lead for marine, estuarine, and anadromous species. NMFS and FWS share jurisdiction for recovering sea turtles – NMFS is responsible for sea turtles in their marine environment and FWS is responsible for sea turtles in their terrestrial habitat.

1.1.2. Definitions

Species: includes any subspecies of fish or wildlife or plants and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature.

Endangered Species: any species which is in danger of extinction throughout all or a significant portion of its range.

Threatened Species: any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range.

1.1.3. Listing

Section 4 of the ESA specifies a process upon which a species may be determined to be listed as threatened or endangered with extinction, changed in status, or removed from the list. The determination is based solely on the best available scientific and commercial data after reviewing the status of the species and taking into account conservation efforts. NMFS and FWS must determine whether any species is an endangered species or a threatened species because of any of the following factors (section 4(a)(1)(A)-(E)):

- (A) the present or threatened, destruction, modification, or curtailment of its habitat or range;
- (B) overutilization for commercial, recreational, scientific, or educational purposes;
- (C) disease or predation;
- (D) the inadequacy of existing regulatory mechanisms; and
- (E) other natural or manmade factors affecting its continued existence.

NMFS and FWS can begin the review for listing determinations, or any interested person may petition for a listing determination under section 553(e) of U.S.C. title 5.

1.1.4. Distinct Population Segment

In 1996, NMFS and FWS published a policy to define the phrase "distinct population segment" (DPS) (FWS and NMFS 1996, 61 FR 4722). The policy defines a population to be a DPS if it is both **discrete** and **significant** relative to its taxon. A population may be considered discrete if it satisfies either one of the following conditions:

a. It is markedly separated from other populations of the same taxon as a consequence of physical, physiological, ecological, or behavioral factors. Quantitative measures of genetic or morphological discontinuity may provide evidence of this separation.
b. It is delimited by international governmental boundaries within which differences in control of exploitation, management of habitat, conservation status, or regulatory mechanisms exist that are significant in light of section 4(a)(1)(D) of the ESA.

If a population segment is considered discrete, NMFS and/or FWS must then consider whether the discrete segment is significant relative to its taxon. Criteria that can be used to determine whether the discrete population segment is significant include, but are not limited to, the following:

a. Persistence of the discrete population segment in an ecological setting unusual or unique for the taxon,
b. Evidence that loss of the discrete population segment would result in a significant gap in the range of the taxon,
c. Evidence that the discrete population segment represents the only surviving natural occurrence of a taxon that may be more abundant elsewhere as an introduced population outside its historic range, or
d. Evidence that the discrete population segment differs markedly from other populations of the species in its genetic characteristics.

1.2. History of ESA Listing, Status Reviews, and Petitions

1.2.1. ESA Listing

The loggerhead sea turtle (*Caretta caretta*) was listed as threatened on July 28, 1978 (FWS and NMFS 1978, 43 FR 32800). The threatened listing was applied to wherever the species occurs. The major factors contributing to its status included human encroachment and associated activities on nesting beaches; commercial harvest of eggs, subadults, and adults; predation; lack of comprehensive and consistent protective regulations; and incidental take in fisheries. Critical habitat has not been designated for the loggerhead sea turtle.

1.2.2. ESA 5-year Reviews

Under the ESA, FWS and NMFS are required to conduct a review of listed species under their jurisdiction at least once every 5 years. A 5-year review is an assessment of a listed species to determine whether its status has changed since the time of its listing such that it should be delisted or classified differently than its current status. The purpose of a 5-year review is to ensure that a listed species has the appropriate level of protection under the ESA.

NMFS conducted the first 5-year review for the loggerhead in 1985 (Mager 1985). Data on population trends were limited and were based largely on the number of nests and nesting females. Of 52 nesting populations examined throughout the Atlantic, Pacific, and Indian Oceans, 33 were thought to be declining, 18 were unknown, and only one – the southeast U.S. Atlantic – was thought to be increasing. Although commercial harvest of eggs had decreased and the U.S. had implemented protective regulations, many threats continued both domestically and abroad. NMFS determined that, since the 1978 listing, information was insufficient to assess whether a listing change was warranted.

FWS also conducted a 5-year review for the loggerhead in 1991 (FWS 1991, 56 FR 56882). In this review, the status of many species was simultaneously evaluated with no in-depth assessment of the section 4(a)(1) factors as they pertained to the individual species. The notice stated that FWS was seeking any new or additional information reflecting the necessity of a change in the status of the species under review. The notice indicated that if significant data were available warranting a change in a species' classification, FWS would propose a rule to modify the species' status. No change in the loggerhead threatened listing classification was recommended following this 5-year review.

In 1995, NMFS and FWS conducted a joint 5-year review (Plotkin 1995). New information on population structure indicated the nesting assemblage along the southeast U.S. coast may consist of two separate populations – Florida and those nesting in Georgia and northward. Nesting in Georgia, South Carolina, and North Carolina was declining. The Florida nesting population was stable, but it was determined that human presence in areas of heavy nesting could impact the population in the future. Although no change in the loggerhead's listing classification was recommended, a need for further study of U.S. loggerhead population structure was identified. The 5-year review further stated that should additional research confirm the existence of two separate nesting populations, and if the decline of the northern nesting population continued, careful consideration should be given to reclassification during the next 5-year review.

The last 5-year review was completed in 2007 (NMFS and FWS 2007). Many technological advances and a diversity of research occurred since the 1995 review. Molecular markers (i.e., mitochondrial DNA and microsatellites) helped define the genetic structuring within and among ocean basins, both at the nesting beaches and at foraging grounds. New information existed on demographic parameters such as age at first reproduction and survival rates and the biology of loggerheads, especially away from the nesting beach. These data indicated a possible separation of populations by ocean basins; however, a more in-depth analysis was needed to determine the application of the DPS policy. Based on the new information and the need for further analysis under the DPS policy, NMFS and FWS recommended that no change in listing status was warranted. They committed to fully assemble and analyze all relevant information in accordance with the DPS policy.

1.2.3. Petitions

Past Petitions

In 2002, NMFS and FWS received a petition to reclassify the Northern (northeast Florida through North Carolina) and Florida Panhandle subpopulations as DPSs, change their status to endangered, and designate critical habitat. NMFS found that the petition presented substantial scientific and commercial information indicating that the petitioned reclassification may be warranted (a “positive 90-day finding”) (NMFS and FWS 2002, 67 FR 38459). Based on further analysis of the best available data, NMFS and FWS found that although there was some degree of discreteness between the nesting assemblages, the separation was not highly rigid and did not qualify as DPSs (a “12-month determination of not warranted”) (FWS and NMFS 2003, 68 FR 53947). Therefore, the species’ listing status was not changed.

Current Petitions
On July 16, 2007, NMFS and FWS received a petition from the Center for Biological Diversity and Turtle Island Restoration Network requesting that loggerhead turtles in the North Pacific be reclassified as a DPS with endangered status and that critical habitat be designated. On November 16, 2007, NMFS and FWS received a petition from the Center for Biological Diversity and Oceana requesting that loggerhead turtles in the Northwest Atlantic Ocean be reclassified as a DPS with endangered status and that critical habitat be designated.

NMFS and FWS determined that the July 16, 2007, North Pacific petition and the November 16, 2007, Northwest Atlantic petition both presented substantial information that the petitioned actions may be warranted (NMFS 2007b, 72 FR 64585; NMFS 2008b, 73 FR 12941). In the published 90-day petition findings, NMFS and FWS committed to assess the loggerhead listing status on a global basis, which is consistent with the recommendation in the 2007 5-year review (see Section 1.2.2.).

1.3. Approach to the Status Review

NMFS and FWS convened a biological review team (BRT) in February 2008 to review the best available scientific information, determine whether DPSs exist, and assess the extinction risk for each potential DPS. The BRT organized their evaluation by ocean basin: Pacific Ocean, Atlantic Ocean (including the Mediterranean Sea), and Indian Ocean. This approach was consistent with the 2007 5-year review for the loggerhead and was chosen to facilitate data assembly and evaluation. It was not meant to preclude identification of DPSs on a broader or finer scale. The BRT primarily reviewed genetic, tagging (flipper and PIT tags), and satellite telemetry datasets and examined oceanographic features and geographic barriers to determine discreteness and significance (Section 3). The BRT conducted two independent analyses to assess extinction risks of potential DPSs (Section 4). The first analysis used the diffusion approximation approach based on a time series of counts of nesting females or nests to provide a metric, susceptibility to quasi-extinction. The second analysis focused on determining the effects of known anthropogenic mortality on each potential DPS with respect to the vital rates of the species. This approach focused on how additional mortalities may affect the future growth and status of potential DPSs.

SECTION 2—SPECIES OVERVIEW

2.1. Taxonomy

Kingdom: Animalia
Phylum: Chordata
Class: Reptilia
Order: Testudines
Family: Cheloniidae
Genus: *Caretta*
Species: *caretta*
Common name: Loggerhead sea turtle

The loggerhead was first described by Linnaeus in 1758 and named *Testudo caretta*. Over the next two centuries more than 35 names were applied to the species (Dodd 1988), but there is now agreement on *Caretta caretta* as the valid name. While Deraniyagala described an Indo-Pacific form as *C. gigas* in 1933, he revised that view in 1939 to hold that *gigas* was a subspecies of *C. caretta* (Deraniyagala 1933, 1939). The genus has been regarded as monotypic since that time. The subspecific designation of *gigas* has likewise been challenged persuasively (Brongersma 1961, Pritchard 1979). Thorough synonymies and taxonomic reviews of this form are given most recently by Pritchard and Trebbau (1984) and Dodd (1988). Subspecies assignments are not supported based on genetic evidence (Bowen 2003) or other diagnostic characters (Dodd 1998).

2.2. Physical Appearance

The carapace of adult and juvenile loggerheads is reddish-brown. The dorsal and lateral head scales and the dorsal scales of the flippers are also reddish-brown, but with light to medium yellow margins. The unscaled areas of the integument (neck, shoulders, limb bases, inguinal area) are dull brown dorsally and light to medium yellow laterally and ventrally. The plastron is medium to light yellow, and the thick, bony carapace is covered by non-overlapping scutes that meet along seam lines. There usually are 11 or 12 pairs of marginal scutes, five pairs of costals, five vertebrals, and a nuchal (precentral) scute that is in contact with the first pair of costal scutes. The plastron is composed of paired gular, humeral, pectoral, abdominal, femoral, and anal scutes and connected to the carapace by three pairs of poreless inframarginal scutes. Hatchlings vary from light to dark brown to dark gray dorsally and lack the reddish-brown coloration of adults and juveniles. Flippers are dark gray to brown above with distinct white margins. The ventral coloration of the plastron and other areas of the integument are generally yellowish to tan. The carapace has three keels and the plastron has two keels. At emergence, hatchlings average 45 mm in SCL and weigh approximately 20 g (Dodd 1988).

2.3. Distribution and Habitat

The loggerhead occurs throughout the temperate and tropical regions of the Atlantic, Pacific, and Indian Oceans (Dodd 1988). However, the majority of loggerhead nesting is at the western rims of the Atlantic and Indian Oceans. The most recent reviews show that only two loggerhead nesting aggregations have greater than 10,000 females nesting per year: Peninsular Florida, United States and Masirah Island, Oman (Baldwin *et al*. 2003, Ehrhart *et al*. 2003, Kamezaki *et al*. 2003, Limpus and Limpus 2003b, Margaritoulis *et al*. 2003). Nesting aggregations with 1,000 to 9,999 females nesting annually are Georgia through North Carolina (U.S.), Quintana Roo and Yucatán (Mexico), Brazil, Cape Verde Islands (Cape Verde), Western Australia (Australia), and Japan. Smaller nesting aggregations with 100 to 999 nesting females annually occur in the Northern Gulf of Mexico (U.S.), Dry Tortugas (U.S.), Cay Sal Bank (The Bahamas), Tongaland (South Africa), Mozambique, Arabian Sea Coast (Oman), Halaniyat Islands (Oman), Cyprus, Peloponnesus (Greece), Zakynthos (Greece), Crete (Greece), Turkey, and Queensland (Australia). In contrast to determining population size on nesting beaches, determining population size in the marine environment has been very localized (Bjorndal and Bolten 2000). At present, there are no data on population size in the oceanic habitat. Detailed information on distribution and habitat by ocean basin follows.

Pacific Ocean
Loggerheads can be found throughout tropical to temperate waters in the Pacific; however, their breeding grounds include a restricted number of sites in the North Pacific and South Pacific. Within the North Pacific, loggerhead nesting has been documented only in Japan (Kamezaki *et al*. 2003), although low level nesting may occur outside of Japan in areas surrounding the South China Sea (Chan *et al*. 2007). In the South Pacific, nesting beaches are restricted to eastern Australia and New Caledonia and, to a much lesser extent, Vanuatu and Tokelau (Limpus and Limpus 2003b).

Important loggerhead nesting locations in Japan include Yakushima Island, and Miyazaki, Minabe, and Atsumi beaches on the mainland. Approximately 40% of all loggerhead nesting in Japan occurs at three primary nesting beaches on Yakushima Island (Kamezaki *et al*. 2003). Based on tag-recapture studies, the East China Sea has been identified as the major habitat for post-nesting adult females (Iwamoto *et al*. 1985, Kamezaki *et al*. 1997, Balazs 2006), while satellite tracking of juvenile loggerheads indicates the Kuroshio Extension Bifurcation Region to be an important pelagic foraging area for juveniles (Polovina *et al*. 2006). Other important juvenile foraging areas have been identified off the coast of Baja California Sur, Mexico (Pitman 1990, Peckham and Nichols 2006).

Nesting occurs along the mainland of Australia from South Stradbroke Island to Bustard Head, and on the islands of the Capricorn Bunker Group and Swain Reefs, and on Bushy Island (Limpus and Limpus 2003b, Limpus 2009). Within this area, five rookeries account for 70% of nests in eastern Australia: (1) Mon Repos, (2) Wreck Rock, (3) mainland and Wreck Island, (4) Erskine Island, and (5) Tryon Island (Limpus and Reimer 1994). Nesting females tagged on the coast of eastern Australia have been recorded foraging in New Caledonia; Queensland, New South Wales, and Northern Territory, Australia; Solomon Islands; Papua New Guinea; and

Indonesia (Limpus and Limpus 2003b). Foraging Pacific loggerheads originating from nesting beaches in Australia are known to migrate to Chile and Peru (Alfaro-Shigueto *et al*. 2004, 2008a; Donoso and Dutton 2006; Boyle *et al*. 2009).

Although nesting in the South Pacific is concentrated in eastern Australia, nesting has also been reported in New Caledonia, Vanuatu, and Tokelau (Limpus and Limpus 2003b). Nesting may occur in other areas of the South Pacific, but it remains unsubstantiated. In New Caledonia, the most substantial loggerhead nesting has been reported on peripheral small coral cays offshore from the main island of Île des Pins (Beloff personal communication cited in Limpus and Limpus 2003b). The population in the Île des Pins area has been estimated at 10-100 females nesting annually (Limpus and Limpus 2003b). Based on aerial surveys of New Caledonia nesting beaches conducted between December 2006 and January 2008, it was estimated that the nesting female population for this nesting season was approximately 200 individuals (World Wildlife Fund 2008). In Vanuatu, low density nesting was reported at Malekula in 1993 (Atuary 1994 cited in Limpus and Limpus 2003b); however, the status of loggerhead nesting is uncertain because most of Vanuatu has been poorly surveyed. In 1981, nesting was reported in Tokelau, a territory of New Zealand that comprises three coral atolls in the South Pacific, but is believed to be uncommon (Balazs 1983).

Indian Ocean

In the Southwest Indian Ocean, loggerhead nesting occurs on the southeastern coast of Africa, from the Paradise Islands in Mozambique southward to St. Lucia in South Africa, and on the south and southwestern coasts of Madagascar (Baldwin *et al*. 2003). Foraging habitats are only known for post-nesting females from Tongaland, South Africa; tagging data show these loggerheads migrating eastward to Madagascar, northward to Mozambique, Tanzania, and Kenya, and southward to Cape Agulhas at the southernmost point of Africa and into the Atlantic Ocean (Baldwin *et al*. 2003, Luschi *et al*. 2006).

In the North Indian Ocean, Oman hosts the vast majority of loggerhead nesting. The majority of the nesting in Oman occurs on Masirah Island, on the Al Halaniyat Islands, and on mainland beaches south of Masirah Island all the way to the Oman-Yemen border (IUCN - The World Conservation Union 1989a, 1989b; Salm 1991; Salm and Salm 1991). In addition, nesting probably occurs on the mainland of Yemen on the Arabian Sea coast, and nesting has been confirmed on Socotra, an island off the coast of Yemen (Pilcher and Saad 2000). Limited information exists on the foraging habitats of North Indian Ocean loggerheads; however, foraging individuals have been reported off the southern coastline of Oman (Salm *et al*. 1993). Satellite telemetry studies conducted in Oman have revealed new information about post-nesting migrations of loggerheads nesting on Masirah Island (Environment Society of Oman and Ministry of Environment and Climate Change, Oman, unpublished data). Results reveal extensive use of the waters off the Arabian Peninsula, with the majority of telemetered turtles (15 of 20) traveling southwest, following the shoreline of southern Oman and Yemen, and circling well offshore in nearby oceanic waters. A minority traveled north as far as the western Persian (Arabian) Gulf (3 of 20) or followed the shoreline of southern Oman and Yemen as far west as the Gulf of Aden and the Bab-el-Mandab (2 of 20). These preliminary data suggest that post-nesting migrations and adult female foraging areas may be centered within the region

(Environment Society of Oman and Ministry of Environment and Climate Change, Oman, unpublished data).

The only verified nesting beaches for loggerheads on the Indian subcontinent are found in Sri Lanka. A small number of nesting females use the beaches of Sri Lanka every year (Deraniyagala 1939, Kar and Bhaskar 1982, Dodd 1988); however, there are no records indicating that Sri Lanka has ever been a major nesting area for loggerheads (Kapurusinghe 2006). No confirmed nesting occurs on the mainland of India despite historical papers suggesting loggerhead sightings on mainland beaches (Tripathy 2005, Kapurusinghe 2006). This discrepancy may be attributed to inaccurate identification of nesting species, as loggerheads are sometimes confused with olive ridleys in the Indian Ocean (Tripathy 2005). In addition, the Gulf of Mannar provides foraging habitat for juveniles and post-nesting adults (Tripathy 2005, Kapurusinghe 2006). Although loggerheads have been reported nesting in low numbers in Myanmar, these data may not be reliable because of misidentification of species (Thorbjarnarson *et al*. 2000).

In the East Indian Ocean, western Australia hosts all known loggerhead nesting (Dodd 1988). Nesting distributions in western Australia span from the Shark Bay World Heritage Area northward through the Ningaloo Marine Park coast to the North West Cape and to the nearby Muiron Islands (Baldwin *et al*. 2003). Nesting individuals from Dirk Hartog Island have been recorded foraging within Shark Bay and Exmouth Gulf, while other adults range much farther (Baldwin *et al*. 2003).

Atlantic Ocean

In the Northwest Atlantic, the majority of loggerhead nesting is concentrated along the coasts of the United States from southern Virginia through Alabama. Additional nesting beaches are found along the northern and western Gulf of Mexico, eastern Yucatán Peninsula, at Cay Sal Bank in the eastern Bahamas (Addison and Morford 1996, Addison 1997), on the southwestern coast of Cuba (F. Moncada-Gavilán, personal communication, cited in Ehrhart *et al*. 2003), and along the coasts of Central America, Colombia, Venezuela, and the eastern Caribbean Islands. In the Southwest Atlantic, loggerheads nest in significant numbers only in Brazil. In the eastern Atlantic, the largest nesting population of loggerheads is in the Cape Verde Islands (L.F. López-Jurado, personal communication, cited in Ehrhart *et al*. 2003), and some nesting occurs along the West African coast (Fretey 2001).

As post-hatchlings, Northwest Atlantic loggerheads hatched on U.S. beaches migrate offshore and become associated with *Sargassum* habitats, driftlines, and other convergence zones (Carr 1986, Witherington 2002). These Northwest Atlantic oceanic juveniles use the North Atlantic gyre and enter Northeast Atlantic waters (Carr 1987); they are also found in the Mediterranean Sea (Carreras *et al*. 2006, Eckert *et al*. 2008). In these areas, they overlap with animals originating from the Northeast Atlantic and the Mediterranean Sea (Laurent *et al*. 1993, 1998; Bolten *et al*. 1998; LaCasella *et al*. 2005; Carreras *et al*. 2006, Monzón-Argüello *et al*. 2006; Revelles *et al*. 2007; Eckert *et al*. 2008). The oceanic juvenile stage in the North Atlantic has been primarily studied in the waters around the Azores and Madeira (Bolten 2003). In Azorean waters, satellite telemetry data and flipper tag returns suggest a long period of residency (Bolten 2003), whereas turtles appear to be moving through Madeiran waters (Dellinger and Freitas

2000). Preliminary genetic analyses indicate that juvenile and subadult loggerheads found in Moroccan waters are of western Atlantic origin (M. Tiwari, NMFS, and A. Bolten, University of Florida, unpublished data). Other concentrations of oceanic juveniles exist in the Atlantic (e.g., in the region of the Grand Banks off Newfoundland). Genetic information indicates the Grand Banks off Newfoundland are foraging grounds for a mixture of loggerheads from all the North Atlantic rookeries (LaCasella *et al*. 2005, Bowen *et al*. 2005), and a large size range is represented (Watson *et al*. 2004, 2005).

After departing the oceanic zone, neritic juvenile loggerheads in the Northwest Atlantic inhabit continental shelf waters from Cape Cod Bay, Massachusetts, south through Florida, The Bahamas, Cuba, and the Gulf of Mexico (neritic refers to the inshore marine environment from the surface to the sea floor where water depths do not exceed 200 meters). In the U.S., estuarine waters, including areas such as Long Island Sound, Chesapeake Bay, Pamlico and Core Sounds, Mosquito and Indian River Lagoons, Biscayne Bay, Florida Bay, and numerous embayments fringing the Gulf of Mexico, comprise important inshore habitat. Along the Atlantic and Gulf of Mexico shoreline, essentially all shelf waters are inhabited by loggerheads.

Habitat preferences of Northwest Atlantic non-nesting adult loggerheads in the neritic zone differ from the juvenile stage in that relatively enclosed, shallow water estuarine habitats with limited ocean access are less frequently used. Areas such as Pamlico Sound and the Indian River Lagoon in the U.S., regularly used by juveniles, are only rarely frequented by adult loggerheads. In comparison, estuarine areas with more open ocean access, such as Chesapeake Bay in the U.S. mid-Atlantic, are also regularly used by juveniles, as well as by adults primarily during warmer seasons. Shallow water habitats with large expanses of open ocean access, such as Florida Bay, provide year-round resident foraging areas for significant numbers of male and female adult loggerheads. Offshore, adults primarily inhabit continental shelf waters, from New York south through Florida, The Bahamas, Cuba, and the Gulf of Mexico. Seasonal use of mid-Atlantic shelf waters, especially offshore New Jersey, Delaware, and Virginia during summer months, and offshore shelf waters, such as Onslow Bay (off the North Carolina coast), during winter months has been documented (Hawkes *et al*. 2007a; Georgia Department of Natural Resources, unpublished data; South Carolina Department of Natural Resources, unpublished data). Shelf waters along the west Florida coast, The Bahamas, Cuba, and the Yucatán Peninsula have been identified, using satellite telemetry, as important resident areas for adult female loggerheads that nest in Florida (Foley *et al*. 2008; M. Lamont, Florida Cooperative Fish and Wildlife Research Unit, personal communication, 2009; M. Nicholas, National Park Service, personal communication, 2009). The southern edge of the Grand Bahama Bank is important habitat for loggerheads nesting on the Cay Sal Bank in The Bahamas, but nesting females are also resident in the bights of Eleuthera, Long Island, and Ragged Islands as well as Florida Bay in the U.S., and the north coast of Cuba (A. Bolten and K. Bjorndal, University of Florida, unpublished data). Moncada *et al*. (2009) report the recapture in Cuban waters of five adult female loggerheads originally flipper tagged in Quintana Roo, Mexico, indicating that Cuban shelf waters likely also provide foraging habitat for adult females that nest in Mexico.

In the Northeast Atlantic, satellite telemetry studies of post-nesting females from Cape Verde identified two distinct dispersal patterns; larger individuals migrated to benthic foraging areas off the northwest Africa coast and smaller individuals foraged primarily oceanically off the northwest Africa coast (Hawkes *et al*. 2006). Monzon-Arguello *et al*. (2009) conducted a mixed stock analysis of juvenile loggerheads sampled from foraging areas in the Canary Islands, Madeira, Azores, and Andalusia and concluded that while juveniles from the Cape Verde population were distributed among these four sites, a large proportion of Cape Verde juveniles appear to inhabit as yet unidentified foraging areas.

In the South Atlantic, relatively little is known about the at-sea behavior of loggerheads originating from nesting beaches in Brazil. Recaptures of tagged juveniles and nesting females have shown movement of animals up and down the coast of South America (Almeida *et al*. 2000, Marcovaldi *et al*. 2000, Laporta and Lopez 2003, Almeida *et al*. 2007). Juvenile loggerheads, presumably of Brazilian origin, have also been captured on the high seas of the South Atlantic (Kotas *et al*. 2004, Pinedo and Polacheck 2004) and off the coast of Atlantic Africa (Bal *et al*. 2007, Petersen 2005, Petersen *et al*. 2007) suggesting that loggerheads of the South Atlantic may undertake transoceanic developmental migrations (Bolten *et al*. 1998, Peckham *et al*. 2007).

Mediterranean Sea

Loggerhead turtles are widely distributed in the Mediterranean Sea. However, nesting is almost entirely confined to the eastern Mediterranean basin, with the main nesting concentrations in Cyprus, Greece, and Turkey (Margaritoulis *et al*. 2003). Preliminary surveys in Libya suggested nesting activity comparable to Greece and Turkey, although a better quantification is needed (Laurent *et al*. 1999). Minimal to moderate nesting also occurs in other countries throughout the Mediterranean including Egypt, Israel, Italy (southern coasts and islands), Lebanon, Syria, and Tunisia (Margaritoulis *et al*. 2003). Recently, isolated nesting events have been recorded in the western Mediterranean basin, namely in Spain, Corsica (France), and in the Tyrrhenian Sea (Italy) (Bentivegna *et al*. 2005, Delaugerre and Cesarini 2004, Tomás *et al*. 2002).

In Cyprus, nesting occurs mainly on beaches of the western coast and Chrysochou Bay (Demetropoulos and Hadjichristophorou 1989), as well as along the northern coast (Broderick and Godley 1996). Nesting occurs along the western and southern coasts of Greece and on the island of Crete, with the vast majority of nesting occurring on the island of Zakynthos (Margaritoulis 1987, 1998, 2005; Margaritoulis *et al*. 1995, 2003). Seventeen important loggerhead nesting sites have been identified on Turkey's beaches (Margaritoulis *et al*. 2003). Nesting activity in Libya is spread throughout the entire coast, but may be less dense in western areas (Laurent *et al*. 1999).

Important neritic marine habitats have been suggested for the large continental shelves of: (1) Tunisia-Libya, (2) northern Adriatic Sea, (3) Egypt, and (4) Spain (Margaritoulis 1988, Argano *et al*. 1992, Laurent and Lescure 1994, Lazar *et al*. 2000, Gomez de Segura *et al*. 2006, Broderick *et al*. 2007, Casale *et al*. 2007b, Nada and Casale 2008). At least the first three constitute shallow benthic habitats for adults (including post-nesting females). Some other neritic foraging areas include Amvrakikos Bay in western Greece, Lakonikos Bay in southern Greece, and southern Turkey. Oceanic foraging areas for small juveniles have been identified in the south Adriatic Sea (Casale *et al*. 2005b), Ionian Sea (Deflorio *et al*. 2005), Sicily Strait (Casale *et al*. 2007b), and western Mediterranean (Spain) (e.g., Caminas *et al*. 2006). In addition, tagged juveniles have been recorded crossing the Mediterranean from the eastern to the western basin and vice versa, as well as in the Eastern Atlantic (Argano *et al*. 1992, Casale *et al*. 2007b).

Reproductive migrations have been confirmed by flipper tagging and satellite telemetry. Female loggerheads, after nesting in Greece, migrate primarily to the Gulf of Gabès and the northern Adriatic (Margaritoulis 1988, Margaritoulis *et al*. 2003, Lazar *et al*. 2004, Zbinden *et al*. 2008). Loggerheads nesting in Cyprus migrate to Egypt and Libya, exhibiting fidelity in following the same migration route during subsequent nesting seasons (Broderick *et al*. 2007). In addition, directed movements of juvenile loggerheads have been confirmed through flipper tagging (Argano *et al*. 1992, Casale *et al*. 2007b) and satellite tracking (Rees and Margaritoulis 2009).

2.4. Biological Characteristics

Loggerheads nest on ocean beaches and occasionally on estuarine shorelines. Although specific characteristics vary between rookeries, loggerhead nesting beaches tend to be wide, sandy beaches backed by low dunes and fronted by a flat, sandy approach from the water (Miller *et al*. 2003). Nests are typically laid between the high tide line and the dune front (Routa 1968, Witherington 1986, Hailman and Elowson 1992).

Sea turtle eggs require a high-humidity substrate that allows for sufficient gas exchange and temperatures conducive to egg development (Miller 1997, Miller *et al*. 2003). Mean clutch size varies greatly between populations, but on average is approximately 100-130 eggs per clutch (Dodd 1988). Loggerhead nests incubate for variable periods of time. The length of the incubation period is inversely related to nest temperature, such that between 26°C and 32°C, a change of 1°C adds or subtracts approximately 5 days (Mrosovsky 1980). The warmer the sand surrounding the egg chamber, the faster the embryos develop (Mrosovsky and Yntema 1980). Sand temperatures prevailing during the middle third of the incubation period also determine the sex of hatchlings (Mrosovsky and Yntema 1980). Incubation temperatures near the upper end of the tolerable range produce only female hatchlings while incubation temperatures near the lower end of the tolerable range produce only male hatchlings. The pivotal temperature (i.e., the incubation temperature that produces equal numbers of males and females) in loggerheads is approximately 29°C (Limpus *et al*. 1983, Mrosovsky 1988, Marcovaldi *et al*. 1997). Moisture conditions in the nest influence incubation period, hatching success, and hatchling size (McGehee 1990, Carthy *et al*. 2003).

Loggerhead hatchlings pip and escape from their eggs over a 1- to 3-day interval and move upward and out of the nest over a 2- to 4-day interval (Christens 1990). The time from pipping to emergence ranges from 4 to 7 days with an average of 4.1 days (Godfrey and Mrosovsky 1997). Hatchlings emerge from their nests en masse almost exclusively at night, and presumably using decreasing sand temperature as a cue (Hendrickson 1958, Mrosovsky 1968, Witherington *et al*. 1990). Moran *et al*. (1999) concluded that a lowering of sand temperatures below a critical threshold, which most typically occurs after nightfall, is the most probable trigger for hatchling emergence from a nest. After an initial emergence, there may be secondary emergences on subsequent nights (Carr and Ogren 1960, Witherington 1986, Ernest and Martin 1993, Houghton and Hays 2001).

Hatchlings use a progression of orientation cues to guide their movement from the nest to the marine environments where they spend their early years (Lohmann and Lohmann 2003). Hatchlings first use light cues to find the ocean. On naturally lighted beaches without artificial lighting, ambient light from the open sky creates a relatively bright horizon compared to the dark silhouette of the dune and vegetation landward of the nest. This contrast guides the hatchlings to the ocean (Daniel and Smith 1947, Limpus 1971, Salmon *et al*. 1992, Witherington 1997, Witherington and Martin 1996, Stewart and Wyneken 2004).

Immediately after hatchlings emerge from the nest, they begin a period of frenzied activity. During this active period, hatchlings move from their nest to the surf, swim, and are swept through the surf zone (Carr and Ogren 1960; Carr 1962, 1982; Wyneken and Salmon 1992; Witherington 1995). Orientation cues used by hatchlings as they crawl, swim through the surf, and migrate offshore are discussed in detail by Lohmann and Lohmann (2003).

Neonate loggerheads that have migrated away from land differ from swim frenzy stage hatchlings in that they are largely inactive and only exhibit infrequent low-energy swimming, and they have begun to feed, no longer relying on their retained yolk (Witherington 2002). As post-hatchlings, loggerheads are pelagic and are best known from neritic waters along the continental shelf. This neritic post-hatchling stage is weeks or months long (Witherington 2002) and may be a transition to the oceanic stage that loggerheads enter as they grow and are carried within ocean currents (Bolten 2003).

In the northwest Atlantic, post-hatchling loggerheads inhabit areas where surface waters converge to form local downwellings (Witherington 2002). These areas are characterized by linear accumulations of floating material, especially *Sargassum*, and are common between the Gulf Stream and the southeast U.S. coast, and between the Loop Current and the Florida coast in the Gulf of Mexico. Post-hatchlings within this habitat are observed to be low-energy float-and-wait foragers that feed on a wide variety of floating items (Witherington 2002). Witherington (2002) found that small animals commonly associated with the *Sargassum* community, such as hydroids and copepods, were most commonly found in esophageal lavage samples. Post-hatchling loggerheads from southeast U.S. nesting beaches may linger for months in waters just off the nesting beach or become transported by ocean currents within the Gulf of Mexico and North Atlantic.

The oceanic juvenile stage begins when loggerheads first enter the oceanic zone (Bolten 2003). Juvenile loggerheads originating from nesting beaches in the Northwest Atlantic, West Indian, and West Pacific Oceans appear to use oceanic developmental habitats and move with the predominant ocean gyres for several years before returning to their neritic foraging and nesting habitats (Bolten 2003, Bowen *et al*. 1995, Hughes 1974a, Musick and Limpus 1997, Pitman 1990, Zug *et al*. 1995). However, the actual duration of the oceanic juvenile stage varies with loggerheads leaving the oceanic zone over a wide size range (Bjorndal *et al*. 2000). In the Atlantic, Bjorndal and colleagues (Bjorndal *et al*. 2000, 2003a) estimated the duration of the oceanic juvenile stage to be between 7 and 11.5 years, with juveniles recruiting to neritic habitats in the western Atlantic over a size range of 46-64 cm curved carapace length (CCL) (Bolten *et al*. 1993, Turtle Expert Working Group 2009). However Snover (2002) suggests a much longer oceanic juvenile stage duration with a range of 9-24 years and a mean 14.8 years over similar size classes. However, in Japan and Australia, juvenile loggerheads do not disperse to neritic habitats until larger than around 60 cm SCL (Y. Matsuzawa and Sea Turtle Association of Japan, unpublished data) and 70 cm CCL or larger (Limpus *et al*. 1994), respectively, while in the Mediterranean they do so at around 25 cm CCL (Casale *et al*. 2008a).

The neritic juvenile stage begins when loggerheads exit the oceanic zone and enter the neritic zone (Bolten 2003). After migrating to the neritic zone, juvenile loggerheads continue maturing until they reach adulthood. Some juveniles may periodically move between the neritic and oceanic zones (Witzell 2002, Bolten 2003, Morreale and Standora 2005, McClellan and Read 2007, Mansfield 2006, Eckert *et al*. 2008). The neritic zone also provides important foraging habitat, inter-nesting habitat, and migratory habitat for adult loggerheads. Some adults may also periodically move between the neritic and oceanic zones (Harrison and Bjorndal 2006). See Schroeder *et al*. (2003) and Limpus and Limpus (2003a) for reviews of this life stage for the Atlantic and Pacific, respectively.

The duration of the adult stage can be reasonably estimated for females from tag return data at nesting beaches. For the Northwest Atlantic nesting assemblages, data from Little Cumberland Island, Georgia, show reproductive longevity, and hence duration of adult female stage, as long as 25 years (Dahlen *et al*. 2000). This is likely an underestimate of the average reproductive life span given tag loss and incomplete surveys of nesting beaches at night. Comparable data for adult males do not exist.

Based on stable isotope analyses and satellite telemetry, Hatase *et al*. (2002a) demonstrated that some adult female loggerheads nesting in Japan inhabit oceanic habitats rather than neritic habitats. Satellite tagged adult loggerheads in western Africa have also been demonstrated to use oceanic foraging areas (Hawkes *et al*. 2006). Preliminary results from stable isotope analyses suggest that some loggerheads nesting in Florida also may inhabit oceanic habitats (Reich *et al*. 2007). In both Japan and Florida, the females inhabiting oceanic habitats were significantly smaller than those in neritic habitats. The extent to which adult loggerheads occupy oceanic habitats needs to be evaluated, and effects on survival probabilities and reproductive output should be assessed.

In both the oceanic and neritic zones, loggerheads are primarily carnivorous, although they do consume some plant matter as well (see Bjorndal 1997 and Dodd 1988 for reviews). Loggerheads are able to exist on a wide variety of food items with ontogenetic and regional differences in diet. Loggerhead diets have been described from just a few coastal regions, and very little information is available about differences or similarities in diet at various life stages. Very little is known of the diet of oceanic juveniles.

SECTION 3—DETERMINATION OF DPS

3.1. Overview of Information Used to Determine DPS

The BRT considered a vast array of information in assessing whether there are any loggerhead population segments that satisfy the Distinct Population Segment criteria of being both discrete and significant.

First, the BRT discussed whether there were any loggerhead population segments that were discrete. As noted previously, joint NMFS/FWS policy defines a population to be a DPS if it is both discrete and significant relative to the taxon to which it belongs (FWS and NMFS 1996, 61 FR 4722). Under the policy, a population may be considered discrete if it satisfies either one of the following conditions: (1) it is markedly separated from other populations of the same taxon as a consequence of physical, physiological, ecological, or behavioral factors; or (2) it is delimited by international governmental boundaries within which differences in control of exploitation, management of habitat, conservation status, or regulatory mechanisms exist that are significant in light of Section 4(a)(1)(D) of the ESA.

Data relevant to the discreteness question include physical, physiological, ecological, behavioral, and genetic data. Upon looking at the global loggerhead population, the physical separation of ocean basins by continents was first considered. The result was an evaluation of the data for each ocean basin (Pacific Ocean, Atlantic Ocean, and Indian Ocean). This was not to preclude any larger or smaller DPS delineation, but to aid in data organization and assessment. The BRT then evaluated genetic information by ocean basin. The genetic data consisted of results from studies using maternally inherited mitochondrial DNA (mtDNA) and biparentally inherited nuclear DNA microsatellite markers. Next, tagging data (both flipper and PIT tags) and telemetry data were reviewed. Additional information, such as potential differences in morphology, was also evaluated. Finally, the BRT considered whether the available information on loggerhead population segments was bounded by any oceanographic features (e.g., current systems) or geographic features (e.g., land masses).

In accordance with the DPS policy, the BRT also reviewed whether the population segments identified in the discreteness analysis were significant. If a population segment is considered discrete, its biological and ecological significance must then be considered. NMFS/FWS must consider available scientific evidence of the discrete segment's importance to the taxon to which it belongs. Data relevant to the significance question include the morphological, ecological, behavioral, and genetic data, as described above. The BRT considered the following factors, listed in the DPS policy, in determining whether the discrete population segments were significant:

a) persistence of the discrete segment in an ecological setting unusual or unique for the taxon;
b) evidence that loss of the discrete segment would result in a significant gap in the range of the taxon;
c) evidence that the discrete segment represents the only surviving natural occurrence of a taxon that may be more abundant elsewhere as an introduced population outside its historical range; and

d) evidence that the discrete segment differs markedly from other populations of the species in its genetic characteristics.

A discrete population segment needs to satisfy only one of these criteria to be considered significant. The NMFS/FWS policy also allows for consideration of other factors if they are appropriate to the biology or ecology of the species. As will be described in subsequent sections, the BRT evaluated the information and considered items (a), (b) and (d), as noted above, to be most applicable to loggerheads.

3.1.1. Discreteness Determination

The loggerhead sea turtle (*Caretta caretta*) is present in all tropical and temperate ocean basins, and has a life history that involves nesting on coastal beaches and foraging in neritic and oceanic habitats, as well as long-distance migrations between and within these areas. As with other globally distributed marine species, today's global loggerhead population has been shaped by a sequence of isolation events created by tectonic and oceanographic shifts over geologic time scales, the result of which is population substructuring in many areas (Bowen *et al*. 1994, Bowen 2003). Globally, loggerhead turtles comprise a mosaic of populations, each with unique nesting sites and in many cases possessing disparate demographic features (e.g., mean body size, age at first reproduction) (Dodd 1988). However, despite these differences, loggerheads from different populations often mix in common foraging grounds (Bolten and Witherington 2003), thus creating unique challenges when attempting to delineate distinct population segments for management or listing purposes.

Examining the phylogeography of loggerheads across their global distribution through mtDNA sequence diversity, Bowen *et al*. (1994) found it to be similar to green turtles (*Chelonia mydas*), with a separation of loggerheads in the Atlantic-Mediterranean basins from those in the Indo-Pacific basins since the Pleistocene period. The divergence between these two primary lineages corresponds to approximately three million years, based on a molecular clock for control region sequences assessed originally for green turtles (2% per million years; Dutton *et al*. 1996, Encalada *et al*. 1996). Geography and climate appear to have shaped the evolution of these two matriarchal lineages with the onset of glacial cycles, the appearance of the Panama Ithsmus creating a land barrier between the Atlantic and eastern Pacific, and upwelling of cold water off southern Africa creating an oceanographic barrier between the Atlantic and Indian Ocean (Bowen 2003). Recent warm temperatures during interglacial periods allowed bi-directional invasion by the temperate-adapted loggerheads into the respective basins (Bowen *et al*. 1994; J.S. Reece, Washington University, personal communication, 2008). Today, it appears that loggerheads within a basin are effectively isolated from populations in the other basin, but some dispersal from the Tongaland rookery in the Indian Ocean into feeding and developmental habitat in the South Atlantic is possible via the Agulhas current (G.R. Hughes, unpublished data, cited in Bowen *et al*. 1994). In the Pacific, extensive mtDNA studies show that the northern loggerhead populations are isolated from the southern Pacific populations, and that juveniles from these distinct genetic populations do not disperse across the equator (Hatase *et al*. 2002a; Dutton 2007, unpublished data).

Mitochondrial DNA data indicate that regional turtle rookeries within an ocean basin have been strongly isolated from one another over ecological timescales (Bowen *et al*. 1994, Bowen and Karl 2007). These same data indicate strong female natal homing and suggest that each regional nesting population is an independent demographic unit (Bowen and Karl 2007). It is difficult to determine the precise boundaries of these demographically independent populations in regions, such as the eastern U.S. coast, where rookeries are close to each other and range along large areas of a continental coastline. There appears to be varying levels of connectivity between proximate rookeries facilitated by imprecise natal homing and male mediated gene flow (Pearce 2001, Bowen 2003, Bowen *et al.* 2005). Regional genetic populations often are characterized by allelic frequency differences rather than fixed genetic differences. There is concern that current analytical tools are unable to identify discrete or demographically independent populations based on genetic data when the allelic frequency differences are slight, and there is a need for new approaches to identifying the appropriate units to conserve (Taylor and Dizon 1999). For example, incorporating a metapopulation approach to regional population structure analysis may be informative for future work addressing connectivity and discreteness of marine turtle populations (Crooks and Sanjayan 2006, Kritzer and Sale 2006, Dutton *et al*. 2007).

Through the evaluation of genetic data, tagging data, telemetry, and demography, the BRT determined that there are at least nine discrete population segments for loggerhead sea turtles globally. These discrete population segments are markedly separated from each other as a consequence of ecological, behavioral, and oceanographic factors, and given the genetic evidence, the BRT has unanimously concluded that each regional population identified is discrete from other populations of loggerheads. Information considered by the BRT in its delineation of discrete population segments is presented below by ocean basin.

Pacific Ocean

Perhaps the most distinct and easily recognized of all loggerhead populations is that from the North Pacific Ocean. The primary nesting areas for this population are found along the southern Japanese coastline and Ryukyu Archipelago (Kamezaki *et al*. 2003). Loggerhead turtles hatching on Japanese beaches undertake extensive developmental migrations utilizing the Kuroshio and North Pacific Currents (Balazs 2006, Kobayashi *et al*. 2008), and some turtles reach the vicinity of Baja California in the eastern Pacific (Uchida and Teruya 1988, Bowen *et al*. 1995, Peckham *et al*. 2007). After spending years foraging in the central and eastern Pacific, loggerheads return to their natal beaches for reproduction (Resendiz *et al*. 1998, Nichols *et al*. 2000) and remain in the western Pacific for the remainder of their life cycle (Iwamoto *et al.* 1985, Kamezaki *et al*. 1997, Sakamoto *et al*. 1997, Hatase *et al*. 2002c).

Despite the long-distance developmental movements of loggerheads in the North Pacific, current scientific evidence, based on genetic analysis, flipper tag recoveries, and satellite telemetry, indicates that individuals originating from Japan remain in the North Pacific for their entire life cycle, never crossing the equator or mixing with individuals from the South Pacific (Hatase *et al*. 2002a; Dutton 2007, unpublished data; LeRoux and Dutton 2006). Indeed, this apparent complete separation of two adjacent populations is unique and most likely results from: (1) the presence of two distinct Northern and Southern Gyre (current flow) systems in the Pacific (Briggs 1974), (2) near-passive movements of post-hatchlings in these gyres that initially move them farther away from areas of potential mixing among the two populations along the equator, and (3) the nest-site fidelity of adult turtles that prevents turtles from returning to non-natal nesting areas.

Pacific loggerheads are further partitioned evolutionarily from other loggerheads throughout the world based on additional analyses of mtDNA. The haplotypes from both North and South Pacific loggerheads are distinguished by a minimum genetic distance (d) equal to 0.017 from other conspecifics, which indicates isolation of approximately one million years (Bowen 2003).

Within the Pacific, Bowen *et al*. (1995) used mtDNA to identify two genetically distinct nesting populations in the Pacific – a northern hemisphere population nesting in Japan and a southern hemisphere population nesting primarily in Australia. This study also suggested that some loggerheads sampled as bycatch in the North Pacific might be from the Australian nesting population (Bowen *et al*. 1995). However, more extensive mtDNA rookery data from Japan (Hatase *et al*. 2002a) taken together with preliminary results from microsatellite (nuclear) analysis confirms that loggerheads inhabiting the North Pacific actually originate from nesting beaches in Japan (P. Dutton, NMFS, unpublished data). LeRoux *et al*. (2008) report additional genetic variation in North Pacific loggerheads based on analyses using new mtDNA primers designed to target longer mtDNA sequences, and suggest finer scale population structure in North Pacific loggerheads may be present.

Although these studies indicate genetic distinctness between loggerheads nesting in Japan versus those nesting in Australia, Bowen *et al*. (1995) did identify individuals with the common Australian haplotype at foraging areas in the North Pacific, based on a few individuals sampled as bycatch in the North Pacific. More recently, Hatase *et al*. (2002a) detected this common haplotype at very low frequency at Japanese nesting beaches. However, the presence of the common Australian haplotype does not preclude the genetic distinctiveness of Japanese and Australian nesting populations, and is likely the result of rare gene flow events occurring over geologic time scales.

The distinct status of loggerheads in the North Pacific is further supported by the results from flipper tagging in the North Pacific. Flipper tagging of loggerheads has been widespread throughout this region, occurring on adults nesting in Japan and incidentally bycaught in the coastal pound net fishery (Y. Matsuzawa, Sea Turtle Association of Japan, personal communication, 2006), juveniles reared and released in Japan (Uchida and Teruya 1988, Hatase *et al*. 2002a), juveniles foraging near Baja California, Mexico (Nichols 2003, Seminoff *et al*. 2004), and loggerheads captured in and tagged from commercial fisheries platforms in the North Pacific high seas (NMFS, unpublished data). To date, there have been at least three transPacific tag recoveries showing east-west and west-east movements (Uchida and Teruya 1988; Resendiz *et al*. 1998; W.J. Nichols, Ocean Conservancy, and H. Peckham, Pro Peninsula, unpublished data) and several recoveries of adults in the western Pacific (Iwamoto *et al*.1985, Kamezaki *et al*. 1997). However, despite the more than 30,000 marked individuals, not a single tag recovery has been reported outside the North Pacific.

A lack of movements by loggerheads south across the equator has also been supported by extensive satellite telemetry. As with flipper tagging, satellite telemetry has been a tool used widely in the North Pacific, with satellite transmitters being placed on adult turtles departing nesting beaches (Sakamoto *et al*. 1997; Japan Fisheries Resource Conservation Association 1999; Hatase *et al*. 2002b, 2002c), on adults and sub-adults bycaught in pound nets off the coast of Japan (Sea Turtle Association of Japan, unpublished data), on headstarted juveniles released in Japan (Balazs 2006), on juvenile, subadult and adult turtles bycaught in the eastern and central North Pacific (e.g., Kobayashi *et al*. 2008), and on juvenile and subadult turtles foraging in the eastern Pacific (Nichols 2003; Peckham *et al*. 2007; J. Seminoff, NMFS, unpublished data). Of the nearly 200 transmitters deployed on loggerheads in the North Pacific, none have moved south of the equator. These studies have demonstrated the strong association loggerheads show with oceanographic mesoscale features such as the Transition Zone Chlorophyll Front or the Kuroshio Current Bifurcation Region (Polovina *et al*. 2000, 2001, 2004, 2006; Etnoyer *et al*. 2006; Kobayashi *et al*. 2008). Kobayashi *et al*. (2008) demonstrated that loggerheads strongly track these zones even as they shift in location, thus suggesting that strong habitat specificity during the oceanic stage also contributes to the lack of mixing. Telemetry studies in foraging areas of the eastern Pacific, near Baja California, Mexico (Nichols 2003; Peckham *et al*. 2007; H. Peckham, Pro Peninsula, unpublished data) and Peru (J. Mangel, Pro Delphinus, unpublished data) similarly show a complete lack of long distance north or south movements.

The North Pacific population of loggerheads appears to occupy an ecological setting distinct from other loggerheads, including those of the South Pacific population. In general, this is the *only* population of loggerheads to be found north of the equator in the Pacific Ocean, foraging in the eastern Pacific as far south as Baja California Sur, Mexico (Seminoff *et al*. 2004, Peckham *et al*. 2007) and in the western Pacific as far south as the Philippines (Limpus 2009) and the mouth of Mekong River, Vietnam (Sadoyama *et al.* 1996). Pelagic juveniles have been found to spend much of their time foraging in the central and eastern North Pacific Ocean. The Kuroshio Extension Current, lying west of the international date line, serves as the dominant physical and biological habitat in the North Pacific and contains high productivity, likely due to unique features such as eddies and meanders that concentrate prey and allow food webs to develop. Juvenile loggerheads originating from nesting beaches in Japan were found to exhibit high site fidelity to an area referred to as the Kuroshio Extension Bifurcation Region, an area with extensive meanders and mesoscale eddies (Polovina *et al*. 2006). Juveniles also were found to correlate strongly with areas of surface chlorophyll a levels in an area known as the Transition Zone Chlorophyll Front, an area concentrating surface prey for loggerheads (Polovina *et al*. 2001, Parker *et al*. 2005, Kobayashi *et al*. 2008). Another area found ecologically unique to the North Pacific population of loggerheads, likely because of the high density of pelagic red crabs (*Pleuronocodes planipes*), is located off the Pacific coast of the Baja California Peninsula, Mexico, where researchers have documented a foraging area for juveniles based on aerial surveys and satellite telemetry (Seminoff *et al*. 2006, Peckham *et al*. 2007). Tag returns show post-nesting females migrating into the East China Sea off South Korea, China, and the Philippines, and the nearby coastal waters of Japan (Iwamoto *et al.* 1985; Kamezaki *et al*. 1997, 2003). Clearly, the North Pacific population of loggerheads is uniquely adapted to the ecological setting of the North Pacific Ocean and throughout its long life history serves as an important part of the ecosystem it inhabits.

Loggerheads inhabiting the North Pacific Ocean are derived from Japanese beaches, with the possible exception of rare waifs over evolutionary time scales. Furthermore, nesting colonies of Japanese loggerheads are found to be genetically distinct based on mtDNA analyses, and when compared to much larger and more genetically diverse loggerhead populations in the Atlantic and Mediterranean, Pacific loggerheads have likely experienced critical bottlenecks (in Hatase *et al*. 2002a), underscoring the importance of management and protection in retaining this genetic population.

In the South Pacific Ocean, loggerhead turtles nest primarily in Queensland, Australia, and, to a lesser extent, New Caledonia and Vanuatu (Limpus and Limpus 2003b, Limpus *et al*. 2006, Limpus 2009). Loggerheads from these rookeries undertake an oceanic developmental migration, traveling to habitats in the central and southeastern Pacific Ocean where they may reside for several years prior to returning to the western Pacific for reproduction. Loggerheads in this early life history stage differ markedly from those originating from western Australia beaches in that they undertake long west-to-east migrations, likely using specific areas of the pelagic environment of the South Pacific Ocean. An unknown portion of these loggerheads forage off Chile and Peru, and preliminary genetic information from foraging areas in the southeastern Pacific confirms that the haplotype frequencies among immature turtles in these areas closely match those found at nesting beaches in eastern Australia (Alfaro-Shigueto *et al*. 2004; Donoso and Dutton 2006, 2007; Boyle *et al*. 2009). Large immature and adult loggerheads generally remain in the western South Pacific, inhabiting neritic and oceanic foraging sites during non-nesting periods (Limpus *et al*. 1994, Limpus 2009).

Loggerheads from Australia and New Caledonia apparently do not travel north of the equator. Flipper tag recoveries from nesting females have been found throughout the western Pacific, including sites north of Australia, the Torres Straight, and the Gulf of Carpentaria (Limpus 2009). Of approximately 1,000 (adult and immature; male and female) loggerheads that have been tagged in eastern Australian feeding areas, only two have been recorded migrating to breed outside of Australia; both traveled to New Caledonia (Limpus 2009). Flipper tagging programs in Peru and Chile tagged approximately 500 loggerheads from 1999-2006, none of which have been reported from outside of the southeastern Pacific (Alfaro-Shigueto *et al*. 2008a; S. Kelez, Duke University Marine Laboratory, unpublished data; M. Donoso, ONG Pacifico Laud - Chile, unpublished data). Limited satellite telemetry data (12 tags) in the area show a similar trend (J. Mangel, Pro Delphinus, unpublished data).

The spatial separation between the North Pacific and South Pacific loggerhead populations has contributed to substantial differences in the genetic profiles of the nesting populations in these two regions. Whereas the dominant mtDNA haplotypes among loggerheads nesting in Japan are CCP2 and CCP3 (equivalent to B and C respectively in Bowen *et al*. 1995 and Hatase *et al*. 2002a; Leroux *et al*. 2008; P. Dutton, NMFS, unpublished data), loggerheads nesting in eastern Australia have a third haplotype (CCP1, previously A) which is dominant (98% of nesting females) (Bowen *et al*. 1994, FitzSimmons *et al*. 1996, Boyle *et al*. 2009). Further, preliminary genetic analysis using microsatellite markers (nuclear DNA) indicates genetic distinctiveness between nesting populations in the North versus South Pacific (P. Dutton, NMFS, personal communication, 2008).

The separateness between nesting populations in eastern and western Australia is less clear, although these too are considered to be genetically distinct from one another (Limpus 2009). For example, mtDNA haplotype CCP1, which is the overwhelmingly dominant haplotype among eastern Australia nesting females (98%), is also found in western Australia, although at much lower frequency (33%) (FitzSimmons *et al*. 1996, 2003). The remaining haplotype for both regions was the CCP5 haplotype. Further, FitzSimmons (University of Canberra, unpublished data) found significant differences in nuclear DNA microsatellite loci from females nesting in these two regions. Estimates of gene flow between eastern and western Australian populations was an order of magnitude less than gene flow within regions. Although some level of male-mediated gene flow may occur within ecological time frames, presumably during mixing at foraging areas near the Torres Straight and in the Gulf of Carpentaria, the result may also indicate homoplasy within the genetic markers in the two regions leading to a false impression of similarity. Male-mediated gene flow between eastern and western Australia, therefore, may be insignificant, which, when considered in light of the substantial disparity in mtDNA haplotype frequencies between these two regions, provides further evidence of population separation between these two regions.

At present, there is no indication from genetic studies that the loggerhead turtles nesting in eastern Australia are distinct from those nesting in New Caledonia. Of 27 turtles sequenced from New Caledonia, 93% carried the CCP1 haplotype and the remaining had the CCP5 haplotype; similar to eastern Australia (Boyle *et al.* 2009). There remains a need to analyze these populations using nuclear DNA microsatellites.

The South Pacific population of loggerheads occupies an ecological setting distinct from other loggerheads, including the North Pacific population. Much less is known regarding the ecosystem upon which oceanic juveniles and oceanic adults depend. Sea surface temperature and chlorophyll frontal zones in the South Pacific have been shown to dramatically impact the movements of green turtles, *Chelonia mydas* (Seminoff *et al*. 2008) and leatherback turtles, *Dermochelys coriacea* (Shillinger *et al*. 2008), and it is likely that loggerheads similarly benefit from interactions with these mesoscale oceanographic features.

Loggerheads in the South Pacific are substantially impacted by periodic environmental perturbations such as the El Niño Southern Oscillation (ENSO). This 3- to 6-year cycle within the coupled ocean-atmosphere system of the tropical Pacific brings increased surface water temperatures and lower primary productivity, both of which have profound biological consequences (Chavez *et al*. 1999). Loggerheads are presumably adversely impacted by the lower food availability that often results from ENSO events, although data on this subject are lacking. Although ENSO may last for only short periods and thus not have a long-term effect on loggerheads in the region, recent studies by Chaloupka *et al*. (2008) suggest that long-term increases in sea surface temperature within the South Pacific may influence the ability of the Australian nesting population to recover from historic population declines.

Loggerheads originating from nesting beaches in the western South Pacific are the only population of loggerheads to be found south of the equator in the Pacific Ocean. As post-hatchlings, they are generally swept south by the East Australian Current (Limpus *et al*. 1994), spend a large portion of time foraging in the oceanic South Pacific Ocean, and a fraction migrate to the southeastern Pacific Ocean off the coasts of Peru and Chile as juveniles (Alfaro-Shigueto *et al*. 2004, Donoso *et al*. 2000, Boyle *et al*. 2009). As large immatures and adults, these loggerheads' foraging range encompasses the eastern Arafura Sea, Gulf of Carpentaria, Torres

Strait, Gulf of Papua, Coral Sea, and western Tasman Sea to southern New South Wales including the Great Barrier Reef, Hervey Bay, and Moreton Bay. The outer extent of this range includes the coastal waters off eastern Indonesia northeastern Papua New Guinea, northeastern Solomon Islands, and New Caledonia (in Limpus 2009).

All loggerheads inhabiting the South Pacific Ocean are derived from beaches in eastern Australia and a lesser known number of beaches in southern New Caledonia, Vanuatu, and Tokelau (Limpus and Limpus 2003b, Limpus 2009). Furthermore, nesting colonies of the South Pacific population of loggerheads are found to be genetically distinct from loggerheads in the North Pacific and Indian Ocean.

Given the information presented above, the BRT has unanimously concluded that two discrete population segments exist in the Pacific Ocean as a consequence of ecological, behavioral, and oceanographic factors, and based on genetic evidence: (1) North Pacific Ocean and (2) South Pacific Ocean.

Indian Ocean

Loggerhead sea turtles in the Indian Ocean have a life history that involves nesting on coastal beaches, foraging in neritic and oceanic habitats, and long-distance migrations between and within these areas. The distribution of loggerheads in the Indian Ocean is limited by the Asian landmass to waters south of 30^{o}N latitude. In comparison to potential loggerhead distributions in southern waters of the Atlantic and Pacific, Indian Ocean distributions east and west are not restricted by landmasses south of approximately 38^{o}S latitude.

Historical accounts of loggerhead turtles in the Indian Ocean were given by Smith (1849), who described the species in South Africa, and Deraniyagala (1933, 1939) who described Indian Ocean loggerheads within the subspecies *C. c. gigas*. Hughes (1974a) argued that there was little justification for this separation. This work by Deraniyagala provided evidence for the significant historical distribution of loggerheads around Sri Lanka.

Loggerhead nesting in the Southwest Indian Ocean includes the southeastern coast of Africa from the Paradise Islands in Mozambique southward to St. Lucia in South Africa, and on the south and southwestern coasts of Madagascar (Baldwin *et al*. 2003). Foraging habitats are only known for the Tongaland, South Africa adult female loggerheads. Returns of flipper tags describe a range that extends eastward to Madagascar, northward to Mozambique, Tanzania, and Kenya, and southward to Cape Agulhas at the southernmost point of Africa (Baldwin *et al*. 2003). Four post-nesting loggerheads satellite tracked by Luschi *et al*. (2006) migrated northward, hugging the Mozambique coast and remained in shallow shelf waters off Mozambique for more than 2 months. Only one post-nesting female from the Southwest Indian Ocean population (South Africa) has been documented migrating north of the equator (to southern Somalia) (Hughes and Bartholomew 1996).

In the North Indian Ocean, Oman hosts the vast majority of loggerhead nests. The largest nesting assemblage is at Masirah Island, Oman, in the northern tropics at 21^{o}N (Baldwin *et al*. 2003). Other key assemblages occur on the Al Halaniyat Islands, and on mainland beaches south of Masirah Island to the Oman-Yemen border (IUCN - The World Conservation Union 1989a, 1989b; Salm 1991; Salm and Salm 1991). In addition, nesting probably occurs on the mainland of Yemen on the Arabian Sea coast, and nesting has been confirmed on Socotra, an island off the coast of Yemen (Pilcher and Saad 2000). Baldwin *et al*. (2003) list other major nesting assemblages (> approximately 400 nesting females/year) at Oman's Arabian Sea Coast (17-20^{o}S) and Al Halaniyat Islands, Oman (17^{o}S).

Outside of Oman, loggerhead nesting is rare in the North Indian Ocean. The only verified nesting beaches for loggerheads on the Indian subcontinent are found in Sri Lanka (Deraniyagala 1939, Kar and Bhaskar 1982, Dodd 1988, Kapurusinghe 2006). Reports of regular loggerhead nesting on the Indian mainland are likely to be from misidentifications of olive ridleys (*Lepidochelys olivacea*) (Tripathy 2005, Kapurusinghe 2006). Although loggerheads have been reported nesting in low numbers in Myanmar, these data may not be reliable because of misidentification of species (Thorbjarnarson *et al*. 2000).

Limited information exists on foraging locations of North Indian Ocean loggerheads. Foraging individuals have been reported off the southern coastline of Oman (Salm *et al*. 1993) and in the Gulf of Mannar, between Sri Lanka and India (Tripathy 2005, Kapurusinghe 2006). Satellite telemetry studies conducted in Oman have revealed new information on post-nesting migrations of loggerheads nesting on Masirah Island (Environment Society of Oman and Ministry of Environment and Climate Change, Oman, unpublished data). Results reveal extensive use of the waters off the Arabian Peninsula, with the majority of telemetered turtles (15 of 20) traveling southwest, following the shoreline of southern Oman and Yemen, and circling well offshore in nearby oceanic waters. A minority traveled north as far as the western Persian (Arabian) Gulf (3 of 20) or followed the shoreline of southern Oman and Yemen as far west as the Gulf of Aden and the Bab-el-Mandab (2 of 20). These preliminary data suggest that post-nesting migrations and adult female foraging areas may be centered within the region (Environment Society of Oman and Ministry of Environment and Climate Change, Oman, unpublished data).
No tag returns or satellite tracks indicated that loggerheads nesting in Oman traveled south of the equator.

In the East Indian Ocean, western Australia hosts all known loggerhead nesting (Dodd 1988). Nesting distributions in western Australia span from the Shark Bay World Heritage Area northward through the Ningaloo Marine Park coast to the North West Cape and to the nearby Muiron Islands (Baldwin *et al*. 2003). Nesting individuals from Dirk Hartog Island have been recorded foraging within Shark Bay and Exmouth Gulf, while other adults range into the Gulf of Carpentaria (Baldwin *et al*. 2003). At the eastern extent of this apparent range, there is possible overlap with loggerheads that nest on Australia's Pacific coast (Limpus 2009). However, despite extensive tagging at principal nesting beaches on Australia's Indian Ocean and Pacific coasts, no exchange of females between nesting beaches has been observed (Limpus 2009).

The available genetic information relates to connectivity and broad evolutionary relationships between ocean basins. There is a lack of genetic information on population structure among rookeries within the Indian Ocean. Bowen *et al*. (1994) described mtDNA sequence diversity among eight loggerhead nesting assemblages and found one of two principal branches in the Indo-Pacific basins. Using additional published and unpublished data, Bowen (2003) estimated divergence between these two lineages to be approximately three million years. Bowen points out evidence for more recent colonizations (12,000-250,000 years ago) between the Indian Ocean and the Atlantic-Mediterranean. For example, the sole mtDNA haplotype (among eight samples) observed by Bowen *et al*. (1994) at Masirah Island, Oman, is known from the Atlantic and suggests some exchange between oceans some 250,000 years ago. The other principal Indian Ocean haplotype reported by Bowen *et al*. (1994) was seen in all loggerheads sampled (n=15) from Natal, South Africa. Encalada *et al*. (1998) reported that this haplotype was common throughout the North Atlantic and Mediterranean, thus suggesting a similar exchange between Atlantic and Indian oceans as recently as 12,000 years ago (Bowen *et al*. 1994). Bowen (2003) speculated that Indian-Atlantic Ocean exchanges took place via the temperate waters south of South Africa and became rare as the ocean shifted to cold temperate conditions in this region.

In estimates of loggerhead gene flow in and out of the Indian Ocean, J.S. Reece (Washington University, personal communication, 2008) factored 100 samples from Masirah Island, 249 from Atlantic rookeries (from Encalada *et al*. 1998), and 311 from Pacific rookeries (from Hatase *et al*. 2002a and Bowen *et al*. 1995). Reece used lineage coalescence methods to estimate that gene flow, expressed as number of effective migrants, or exchanges of breeding females between Indian Ocean rookeries and those from the Atlantic or Pacific occurred at the rate of less than 0.1 migrant per generation. Reece estimated gene flow based on coalescence of combined mtDNA and nuclear DNA data to be approximately 0.5 migrants per generation. These unpublished results, while somewhat theoretical, may indicate that there is restricted gene flow into and out of the Indian Ocean. The low level of gene flow most likely reflects the historical connectivity over geological timescales, rather than any contemporary migration, and is consistent with Bowen's hypothesis that exchange occurred most recently over 12,000-3,000,000 years ago, and has been restricted over recent ecological timescales.

The discrete status of three loggerhead groups in the Indian Ocean is primarily supported by observations of tag returns and satellite telemetry. The limited genetic information currently available based on mtDNA sequences does not allow for a comprehensive analysis of genetic population structure analysis for Indian Ocean rookeries, although Bowen *et al*. 1994 indicate the Oman and South African rookeries are genetically distinct, and once sequencing studies are completed for these rookeries, it is likely that they will also be distinct from the rookeries in western Australia. Based on multiple lines of evidence, discrete status is supported for the North Indian Ocean, Southwest Indian Ocean, and Southeast Indo-Pacific Ocean. Although there is not a sufficiently clear picture of gene flow between these regions, we propose that significant vicariant barriers exist between these three Indian Ocean groups that would prevent migration of individuals on a time scale relative to management and conservation efforts. These vicariant barriers are the oceanographic phenomena associated with Indian Ocean equatorial waters, and the large expanse between continents in the South Indian Ocean without suitable benthic foraging habitat.

Given the information presented above, the BRT has unanimously concluded that three discrete population segments exist in the Indian Ocean as a consequence of ecological, behavioral, and oceanographic factors, and given the genetic evidence: (1) North Indian Ocean, (2) Southwest Indian Ocean, and (3) Southeast Indo-Pacific Ocean.

Atlantic Ocean and Mediterranean Sea
Within the Atlantic Ocean, loss and re-colonization of nesting beaches over evolutionary time scales has been determined by climate, natal homing, and rare dispersal events (Encalada *et al*. 1998, Bowen and Karl 2007). At times, temperate beaches were too cool to incubate eggs and nesting could have continued only on tropical beaches. Thus, the contemporary distribution of nesting is the product of colonization events from the tropical refugia during the last 12,000 years. Apparently, turtles from the Northwest Atlantic colonized the Mediterranean and at least two matrilines were involved (Schroth *et al*. 1996); these rookeries became isolated from the Atlantic populations in the last 10,000 years (Encalada *et al*. 1998). A similar colonization event appears to have populated the Northeast Atlantic (C. Monzón-Argüello, Instituto Canario de Ciencias Marinas - Spain, personal communication, 2008).

Nesting in the western South Atlantic occurs primarily along the mainland coast of Brazil from Sergipe south to Rio de Janeiro, with peak concentrations in northern Bahia, Espírito Santo, and northern Rio de Janeiro (Marcovaldi and Chaloupka 2007). In the eastern South Atlantic, diffuse nesting may occur along the mainland coast of Africa (Fretey 2001), with more than 200 loggerhead nests reported for Rio Longa beach in central Angola in 2005 (Brian 2007). However, other researchers have been unable to confirm nesting by loggerheads in the last decade anywhere along the south Atlantic coast of Africa, including Angola (Fretey 2001, Weir *et al*. 2007). There is the possibility that reports of nesting loggerheads from Angola and Namibia (Márquez M. 1990, Brian 2007) may have arisen from misidentified olive ridley turtles (Brongersma 1982, Fretey 2001). At the current time, it is not possible to confirm that regular, if any, nesting of loggerheads occurs along the Atlantic coast of Africa, south of the equator.

Genetic surveys of loggerheads have revealed that the Brazilian rookeries have a unique mtDNA haplotype (Encalada *et al*. 1998, Pearce 2001). The Brazilian mtDNA haplotype, relative to North Atlantic haplotypes, indicates isolation of South Atlantic loggerheads from North Atlantic loggerheads on a scale of 0.25-0.5 million years ago, and microsatellite DNA results show divergence on the same time scale (Bowen 2003). Brazil's unique haplotype has been found only in low numbers in foraging populations of juvenile loggerheads of the North Atlantic (Bass *et al*. 2004). Other lines of evidence support a deep division between loggerheads from the South Atlantic and from the North Atlantic, including: (1) a nesting season in Brazil that peaks in the austral summer around December/January (Marcovaldi and Laurent 1996), as opposed to the May-August nesting season in the southeast U.S. in the northern hemisphere (Witherington *et al*. 2009); and (2) no observations of tagged loggerheads moving across the equator in the Atlantic, except a single case of a captive-reared animal that was released as a juvenile from Espírito Santo and was recaptured 3 years later in the Azores (Bolten *et al*. 1990). Post-nesting females from Espírito Santo moved either north or south along the coast, but remained between 10°S and 30°S (Projeto TAMAR, unpublished data).

Relatively little is known about the at-sea behavior of loggerheads originating from nesting beaches in Brazil. Recaptures of tagged juveniles and nesting females have shown movement of animals up and down the coast of South America (Almeida *et al*. 2000, Marcovaldi *et al*. 2000, Laporta and Lopez 2003, Almeida *et al*. 2007). Juvenile loggerheads, presumably of Brazilian origin, have also been captured on the high seas of the South Atlantic (Kotas *et al*. 2004, Pinedo and Polacheck 2004) and off the coast of Atlantic Africa (Petersen 2005, Petersen *et al*. 2007, Weir *et al*. 2007) suggesting that, like their North Pacific and Northwest Atlantic counterparts, loggerheads of the South Atlantic may undertake transoceanic developmental migrations (Bolten *et al*. 1998, Peckham *et al*. 2007). There is also the possibility that some loggerhead individuals from the Southwest Indian Ocean population segment are swept around the Cape of Good Hope by the Benguela Current (Baldwin *et al*. 2003), into the Southwest Atlantic Ocean, where they overlap with loggerheads from the South Atlantic Ocean population segment.

The mean size of reproductive female loggerheads in Brazil is 92.9 cm SCL, which is comparable to the size of nesters in the Northwest Atlantic, but larger than nesters of the Northeast Atlantic and Mediterranean (Tiwari and Bjorndal 2000, Margaritoulis *et al*. 2003, Varo Cruz *et al*. 2007). Egg size and mass of Brazilian loggerheads are smaller than those from the Northwest Atlantic, but larger than those of the Mediterranean (Tiwari and Bjorndal 2000).

Within the Northwest Atlantic, the majority of nesting activity occurs from April through September, with a peak in June and July (Williams-Walls *et al*. 1983, Dodd 1988, Weishampel *et al*. 2006). Nesting occurs within the Northwest Atlantic along the coasts of North America, Central America, northern South America, the Antilles, and The Bahamas, but is concentrated in the southeastern U.S. and on the Yucatán Peninsula in Mexico (Sternberg 1981, Ehrhart 1989, Ehrhart *et al*. 2003, NMFS and FWS 2008). Many nesting beaches within the Northwest Atlantic have yet to be sampled for genetic analysis. Five recovery units (subpopulations) have been identified based on genetic differences and a combination of geographic distribution of nesting densities and geographic separation. These recovery units are: Northern Recovery Unit (Florida/Georgia border through southern Virginia), Peninsular Florida Recovery Unit (Florida/Georgia border through Pinellas County, Florida), Northern Gulf of Mexico Recovery Unit (Franklin County, Florida, through Texas), Greater Caribbean Recovery Unit (Mexico through French Guiana, The Bahamas, Lesser Antilles, and Greater Antilles), and Dry Tortugas Recovery Unit (islands located west of Key West, Florida) (NMFS and FWS 2008). There is limited exchange of nesting females among these recovery units (Encalada *et al*. 1998, Foote *et al*. 2000, J. Richardson personal communication cited in NMFS 2001, Hawkes *et al*. 2005). Based on the number of haplotypes, the highest level of loggerhead mtDNA genetic diversity in the Atlantic has been observed in females of the Greater Caribbean Recovery Unit that nest at Quintana Roo, Mexico (Encalada *et al*. 1999, Nielsen *et al*. in press). However, genetic diversity should be evaluated further using haplotype and nucleotide diversity calculated similarly for each recovery unit. Genetic data are not available for all the nesting assemblages in the region, including a key nesting assemblage in Cuba. New genetic markers have recently been developed, including primers that produce additional mtDNA sequence data (Abreu-Grobois *et al*. 2006, LeRoux *et al*. 2008), and an array of microsatellite markers (Shamblin *et al*. 2008) that will enable finer resolution of population boundaries.

Loggerheads in the Northwest Atlantic display complex population structure based on life history stages. Based on mtDNA, oceanic juveniles show no structure, neritic juveniles show moderate structure, and nesting colonies show strong structure (Bowen *et al*. 2005); however, a recent study suggests some structure may exist among oceanic juveniles, at least in the Northeast Atlantic oceanic foraging grounds (Monzón-Argüello *et al*. 2009), and is related to the latitudinal distributions of the Northwest Atlantic rookeries. In contrast, a survey using microsatellite (nuclear) markers showed no significant population structure among nesting populations (Bowen *et al*. 2005), indicating that while females exhibit strong philopatry, males may provide an avenue of gene flow between nesting colonies in this region. However, the power to detect structure with the nuclear markers used in this study may have been limited due to the few markers used and small sample sizes. Additional work with larger sample sizes and better nuclear markers is underway (B. Shamblin, University of Georgia, personal communication, 2008). Nevertheless, Bowen *et al*. (2005) argued that male-mediated gene flow within the Northwest Atlantic does not detract from the classification of breeding areas as independent populations (e.g., recovery units) because the production of progeny depends on female nesting success. All Northwest Atlantic recovery units are reproductively isolated from populations within the Northeast Atlantic, South Atlantic, and Mediterranean Sea.

As oceanic juveniles, loggerheads from the Northwest Atlantic use the North Atlantic gyre and often are associated with *Sargassum* communities (Carr 1987); they also are found in the Mediterranean Sea. In these areas, they overlap with animals originating from the Northeast Atlantic and the Mediterranean Sea (Laurent *et al*. 1993, 1998; Bolten *et al*. 1998; Bowen *et al*. 2005; LaCasella *et al*. 2005; Carreras *et al*. 2006, Monzón-Argüello *et al*. 2006; Revelles *et al*. 2007). In the western Mediterranean, they tend to be associated with the waters off the northern African coast and the northeastern Balearic archipelago, areas generally not inhabited by turtles of Mediterranean origin (Carreras *et al*. 2006, Revelles *et al*. 2007, Eckert *et al*. 2008). As larger, neritic juveniles, they show more structure and tend to inhabit areas closer to their natal origins (Bowen *et al*. 2004), but some do move to and from oceanic foraging grounds throughout this life stage (Mansfield 2006, McClellan and Read 2007), and some continue to use the Mediterranean Sea (Casale *et al*. 2008b, Eckert *et al*. 2008). Adult populations are highly structured with no overlap in distribution among adult loggerheads from the Northwest Atlantic, Northeast Atlantic, South Atlantic, and Mediterranean. Carapace epibionts suggest the adult females of different subpopulations use different foraging habitats (Caine 1986). In the Northwest Atlantic, based on satellite telemetry studies and flipper tag returns, non-nesting adult females from the Northern Recovery Unit reside primarily off the east coast of the U.S.; movement into the Bahamas or the Gulf of Mexico is rare (Bell and Richardson 1978, Williams and Frick 2001, Mansfield 2006, Turtle Expert Working Group 2009). Adult females of the Peninsular Florida Recovery Unit are distributed throughout eastern Florida, the Bahamas, Greater Antilles, Cuba, the Yucatán Peninsula, and the Gulf of Mexico, as well as along the Atlantic seaboard of the U.S. (Meylan 1982, Meylan *et al*. 1983, Foley *et al*. 2008; Turtle Expert Working Group 2009). Adult females from the Northern Gulf of Mexico Recovery Unit remained in the Gulf of Mexico, including off the Yucatán Peninsula, based on satellite telemetry and flipper tag returns (Foley *et al*. 2008; Turtle Expert Working Group 2009; M. Lamont, Florida Cooperative Fish and Wildlife Research Unit, personal communication, 2009; M. Nicholas, National Park Service, personal communication, 2009).

Nesting in the Northeast Atlantic is concentrated in the Cape Verde Archipelago, with some nesting occurring on most of the islands, and the highest concentration on the beaches of Boa Vista Island (López-Jurado *et al*. 2000, Varo Cruz *et al*. 2007, Loureiro 2008). On mainland Africa, there is minor nesting on the coasts of Mauritania to Senegal (Brongersma 1982, Arvy *et al*. 2000, Fretey 2001). Earlier reports of loggerhead nesting in Morocco (Pasteur and Bons 1960) have not been confirmed in recent years (Tiwari *et al*. 2001). Nesting has not been reported from Macaronesia (Azores, Madeira Archipelago, The Selvagens Islands, and the

Canary Islands), other than in the Cape Verde Archipelago (Brongersma 1982). In Cape Verde, nesting begins in mid June and extends into October (Cejudo *et al*. 2000), which is somewhat later than when nesting occurs in the Northwest Atlantic.

Based on an analysis of mtDNA of 196 nesting females from Boa Vista Island, the Cape Verde nesting assemblage is genetically distinct from other studied rookeries (C. Monzón-Argüello, Instituto Canario de Ciencias Marinas - Spain, personal communication, 2008; Monzón-Argüello *et al*. 2009). The results also indicate that despite the close proximity of the Mediterranean, the Boa Vista rookery is most closely related to the rookeries of the Northwest Atlantic.

The distribution of juveniles from the Northeast Atlantic is largely unknown but they have been found on the oceanic foraging grounds of the North Atlantic (A. Bolten, University of Florida, personal communication, 2008, based on Bolten *et al*. 1998 and LaCasella *et al*. 2005; Monzón-Argüello *et al*. 2009; M. Tiwari, NMFS, and A. Bolten, University of Florida, unpublished data) and in the western and central Mediterranean (A. Bolten, University of Florida, personal communication, 2008, based on Carreras *et al*. 2006), along with small juveniles from the Northwest Atlantic. Loggerheads of both juvenile and adult size also have been reported captured incidental to fishing operations off the coasts of Europe and Africa and have been reported stranded and in the markets (Maigret 1983; Tiwari *et al*. 2001; Benhardouze 2004; Benhardouze *et al*. 2004; Duguy *et al*. 2004, 2005; Witt *et al*. 2007). The size of nesting females in the Northeast Atlantic is comparable to those in the Mediterranean (average 72-80 cm SCL; Margaritoulis *et al*. 2003) and smaller than those in the Northwest Atlantic or the South Atlantic; 91% of the nesting turtles are < 86.5 cm CCL (Hawkes *et al*. 2006) and nesting females average 77.1 cm SCL (Cejudo *et al*. 2000). Post-nesting females demonstrated two behaviors, depending on size (Hawkes *et al*. 2006). The larger turtles (>93 cm) foraged in coastal waters along northwest Africa and the smaller animals (<87 cm) foraged oceanically, mostly between Cape Verde and the African shelf from Mauritania to Guinea Bissau. Loureiro (2008) only observed the smaller size class of reproductive female loggerheads nesting on Santiago Island, Cape Verde.

Nesting occurs throughout the central and eastern Mediterranean on the shores of Italy, Greece, Cyprus, Turkey, Syria, Lebanon, Israel, the Sinai, Egypt, Libya, and Tunisia (Sternberg 1981, Margaritoulis *et al*. 2003, SWOT 2007). Sporadic nesting also has been reported in the western Mediterranean on Corsica (Delaugerre and Cesarini 2004), southwestern Italy (Bentivegna *et al*. 2005), and on the Spanish Mediterranean coast (Tomás *et al*. 2003, 2008). Nesting in the Mediterranean is concentrated between June and early August (Margaritoulis *et al*. 2003).

Within the Mediterranean, a recent study of mitochondrial and nuclear DNA in nesting assemblages from Greece to Israel indicated genetic structuring, philopatry by both females and males, and limited gene flow between assemblages (Carreras *et al*. 2007). Genetic differentiation based on mtDNA indicated that there are at least four independent nesting subpopulations within the Mediterranean and usually they are characterized by a single haplotype: (1) mainland Greece and the adjoining Ionian Islands, (2) eastern Turkey, (3) Israel, and (4) Cyprus. There is no evidence of adult female exchange among these four subpopulations (Carreras *et al*. 2006). In studies of the foraging grounds in the western and central Mediterranean, seven of the 17 distinct haplotypes detected had not yet been described, indicating that nesting beach data to describe the natal origins of juveniles exploiting the western Mediterranean Sea are incomplete (Carreras *et al*. 2006, Casale *et al*. 2008b). Gene flow among the Mediterranean rookeries estimated from nuclear DNA was significantly higher than that calculated from mtDNA, consistent with the scenario of female philopatry maintaining isolation between rookeries, offset by male-mediated gene flow. Nevertheless, the nuclear data show there was a higher degree of substructuring among Mediterranean rookeries compared to those in the Northwest Atlantic (Bowen *et al*. 2005; Carreras *et al*. 2007).

Small oceanic juveniles from the Mediterranean Sea use the eastern basin (defined as inclusive of the central Mediterranean, Ionian, Adriatic, and Aegean Seas) and the western basin (defined as inclusive of the Tyrrhenian Sea) along the European coast (Laurent *et al*. 1998, Margaritoulis *et al*. 2003, Carreras *et al*. 2006, Revelles *et al*. 2007). Larger juveniles also use the eastern Atlantic and the eastern Mediterranean, especially the Tunisia-Libya shelf and the Adriatic Sea (Laurent *et al*. 1993, Margaritoulis *et al*. 2003, Monzón-Argüllo *et al*. 2006, Revelles *et al*. 2007). Adults appear to forage closer to the nesting beaches in the eastern basin; most tag recoveries from females nesting in Greece have occurred in the Adriatic Sea and off Tunisia (Margaritoulis *et al*. 2003, Lazar *et al*. 2004).

Straight carapace lengths of loggerheads nesting in the Mediterranean were significantly smaller than the lengths of loggerheads nesting in the Northwest Atlantic and the South Atlantic, and within the Mediterranean showed some variability among subpopulations. Carapace lengths ranged from 58 to 95 cm SCL (Margaritoulis *et al*. 2003). Greece's loggerheads averaged 77-80 cm SCL (Tiwari and Bjorndal 2000, Margaritoulis *et al*. 2003), whereas Turkey's loggerheads averaged 72-73 cm SCL (Margaritoulis *et al*. 2003). The Greece turtles also produced larger clutches (relative to body size) than those produced by Florida or Brazil nesters (Tiwari and Bjorndal 2000). The authors suggested that sea turtles in the Mediterranean encounter environmental conditions significantly different from those experienced by populations elsewhere in the Atlantic Ocean basin.

Given the information presented above, the BRT has unanimously concluded that four discrete population segments exist in the Atlantic Ocean/Mediterranean Sea as a consequence of ecological, behavioral, and oceanographic factors, and based on genetic evidence: (1) Northwest Atlantic Ocean, (2) Northeast Atlantic Ocean, (3) South Atlantic Ocean, and (4) Mediterranean Sea. These population segments are genetically distinct (Bowen *et al*. 1994; Encalada *et al*. 1998; Pearce 2001; Carerras *et al*. 2007; C. Monzón-Argüello, Instituto Canario de Ciencias Marinas - Spain, personal communication, 2008; Monzon-Arguello *et al*. 2009) and, although they may comingle on oceanic foraging grounds as juveniles, adults apparently are isolated from each other; they also differ demographically.

3.1.2. Significance Determination

In Section 3.1.1., the BRT identified nine discrete population segments. As described below by ocean basin, each of the nine discrete population segments is biologically and ecologically significant. They each represent a large portion of the species range, sometimes encompassing an entire hemispheric ocean basin. The range of each discrete population segment represents a unique ecosystem, influenced by local ecological and physical factors. The loss of any single discrete population segment would result in a significant gap in the loggerhead's range. Each discrete population segment is genetically unique, often identified by unique mtDNA haplotypes, and the BRT believes that these unique haplotypes could represent adaptive differences; the loss of any one discrete population segment would represent a significant loss of genetic diversity. Therefore, the BRT concludes that these nine population segments are both discrete from other conspecific population segments and significant to the species to which they belong, *Caretta caretta*.

The geographic delineations given below for each discrete population segment were determined primarily based on nesting beach locations, genetic evidence, oceanographic features, thermal tolerance, fishery bycatch data, and information on loggerhead distribution and migrations from satellite telemetry and flipper tagging studies. With rare exception, adults from discrete population segments remain within the delineated boundaries. In some cases, juveniles from two or more discrete population segments may mix on foraging areas and therefore, their distribution and migrations may extend beyond the geographic boundaries delineated below for each discrete population segment (e.g., juveniles from the Northwest Atlantic Ocean, Northeast Atlantic Ocean, and Mediterranean Sea discrete population segments share foraging habitat in the western Mediterranean Sea).

Pacific Ocean
The BRT considers 60ºN latitude and the equator as the north-south boundaries of the North Pacific population segment based on oceanographic features, loggerhead sightings, thermal tolerance, fishery bycatch data, and information on loggerhead distribution from satellite telemetry and flipper tagging studies. The loss of the North Pacific population segment would result in a significant gap in the range of the taxon. There is no evidence or reason to believe that female loggerheads from South Pacific nesting beaches would repopulate the North Pacific nesting beaches should those nesting assemblages be lost. Tagging studies show that the vast majority of nesting females return to the same nesting area. As summarized by Hatase *et al.* (2002a), of 2,219 tagged nesting females from Japan, only five females relocated their nesting sites. In addition, flipper tag and satellite telemetry research, as described in detail in Section 3.1.1., shows no evidence of north-south movement of loggerheads across the equator. The BRT concludes that the North Pacific population segment is both discrete from other conspecific population segments and significant to the taxon to which it belongs, and therefore, it satisfies the DPS policy criteria.

The BRT considers the equator and 60ºS latitude as the north-south boundaries and 67ºW longitude and 139ºE longitude as the east-west boundaries of the South Pacific population segment based on oceanographic features, loggerhead sightings, thermal tolerance, fishery bycatch data, and information on loggerhead distribution from satellite telemetry and flipper tagging studies. The loss of the South Pacific population segment would result in a significant gap in the range of the taxon. The South Pacific population is the only population of loggerheads found south of the equator in the Pacific Ocean and there is no evidence or reason to believe that female loggerheads from North Pacific nesting beaches would repopulate the South Pacific nesting beaches should those nesting assemblages be lost. In addition, flipper tag and satellite telemetry research, as described in detail in Section 3.1.1., shows no evidence of north-south movement of loggerheads across the equator. We also do not expect that recolonization from

Indian Ocean loggerheads would occur in eastern Australia within ecological time frames. Despite evidence of foraging in the Gulf of Carpentaria by adult loggerheads from the nesting populations in eastern Australia (South Pacific population segment) and western Australia (Southeast Indo-Pacific Ocean population segment), the nesting females from these two regions are considered to be genetically distinct from one another (Limpus 2009). In addition to a substantial disparity in mtDNA haplotype frequencies between these two populations, FitzSimmons (University of Canberra, unpublished data) found significant differences in nuclear DNA microsatellite loci between females nesting in these two regions, indicating separation between the South Pacific Ocean and the Southeast Indo-Pacific Ocean population segments. Long term studies show a high degree of site fidelity by adult females in the South Pacific, with most females returning to the same beach within a nesting season and in successive nesting seasons (Limpus 1985, 2009; Limpus *et al.* 1994). This has been documented as characteristic of loggerheads from various rookeries throughout the world (Schroeder *et al.* 2003). The BRT concludes that the South Pacific population segment is both discrete from other conspecific population segments and significant to the taxon to which it belongs, and therefore, it satisfies the DPS policy criteria.

Indian Ocean

The BRT considers 30ºN latitude and the equator as the north-south boundary of the North Indian Ocean population segment based on oceanographic features, loggerhead sightings, thermal tolerance, fishery bycatch data, and information on loggerhead distribution from satellite telemetry and flipper tagging studies. The loss of the North Indian Ocean population segment would result in a significant gap in the range of the taxon. Genetic studies have shown that adult populations are highly structured with no overlap in distribution among adult loggerheads in the North Indian Ocean, Southwest Indian Ocean, and Southeast Indo-Pacific. There is no evidence or reason to believe that female loggerheads from the Southwest Indian Ocean or Southeast Indo-Pacific would repopulate the North Indian Ocean nesting beaches should those populations be lost. The BRT concludes that the North Indian Ocean population segment is both discrete from other conspecific population segments and significant to the taxon to which it belongs, and therefore, it satisfies the DPS policy criteria.

The BRT considers the equator and 60ºS latitude as the north-south boundaries and 20ºE longitude at Cape Agulhas on the southern tip of Africa and 80ºE longitude as the east-west boundaries of the Southwest Indian Ocean population segment based on oceanographic features, thermal tolerance, fishery bycatch data, and information on loggerhead distribution from satellite telemetry and flipper tagging studies. The loss of the Southwest Indian Ocean population segment would result in a significant gap in the range of the taxon. Genetic studies have shown that adult populations are highly structured in the North Indian Ocean, Southwest Indian Ocean, and Southeast Indo-Pacific. Only one post-nesting female from the Southwest Indian Ocean population (South Africa) has been documented migrating north of the equator (to southern Somalia) (Hughes and Bartholomew 1996). There is no evidence or reason to believe that female loggerheads from the North Indian Ocean or Southeast Indo-Pacific would repopulate the Southwest Indian Ocean nesting beaches should those populations be lost. There is also no evidence of movement of adult Southwest Indian Ocean loggerheads west of 20ºE longitude at Cape Agulhas, the southernmost point on the African continent or east of 80ºE longitude within the Indian Ocean. The BRT concludes that the Southwest Indian Ocean population segment is both discrete from other conspecific population segments and significant to the taxon to which it belongs, and therefore, it satisfies the DPS policy criteria.

The BRT considers the equator and 60ºS latitude as the north-south boundaries and 139ºE longitude and 80ºE longitude as the east-west boundaries of the Southeast Indo-Pacific Ocean population segment based on oceanographic features, thermal tolerance, fishery bycatch data, and information on loggerhead distribution from satellite telemetry and flipper tagging studies. The loss of the Southeast Indo-Pacific Ocean population segment would result in a significant gap in the range of the taxon. Genetic studies have shown that adult populations are highly structured with no overlap in distribution among adult loggerheads in the North Indian Ocean, Southwest Indian Ocean, and Southeast Indo-Pacific. There is no evidence or reason to believe that female loggerheads from the North Indian Ocean or Southwest Indian Ocean would repopulate the Southeast Indo-Pacific nesting beaches should those populations be lost. There is also no evidence of movement of adult Southeast Indo-Pacific Ocean loggerheads west of 80ºE longitude within the Indian Ocean. Despite evidence of foraging in the Gulf of Carpentaria by adult loggerheads from the nesting populations in eastern Australia (South Pacific population segment) and western Australia (Southeast Indo-Pacific Ocean population segment), the nesting females from these two regions are considered to be genetically distinct from one another (Limpus 2009). In addition to a substantial disparity in mtDNA haplotype frequencies between these two regions, FitzSimmons (University of Canberra, unpublished data) found significant differences in nuclear DNA microsatellite loci from females nesting in these two regions, indicating separation between the South Pacific population segment and the Southeast Indo-Pacific Ocean population segment. The BRT concludes that the Southeast Indo-Pacific Ocean population segment is both discrete from other conspecific population segments and significant to the taxon to which it belongs, and therefore, it satisfies the DPS policy criteria.

Atlantic Ocean and Mediterranean Sea

The BRT considers 60ºN latitude and the equator as the north-south boundaries and 40ºW longitude as the east boundary of the Northwest Atlantic Ocean population segment based on oceanographic features, loggerhead sightings, thermal tolerance, fishery bycatch data, and information on loggerhead distribution from satellite telemetry and flipper tagging studies. The loss of the Northwest Atlantic population segment would result in a significant gap in the range of the taxon. Genetic studies have shown that adult populations are highly structured with no overlap in distribution among adult loggerheads from the Northwest Atlantic, Northeast Atlantic, South Atlantic, and Mediterranean. There is no evidence or reason to believe that female loggerheads from the Northeast Atlantic, Mediterranean Sea, or South Atlantic nesting beaches would repopulate the Northwest Atlantic nesting beaches should these populations be lost. Data from satellite telemetry studies and flipper tag returns show that the vast majority of nesting females from the Northwest Atlantic return to the same nesting area; they reveal no evidence of movement of adults south of the equator or east of 40ºW longitude. The BRT concludes that the Northwest Atlantic population segment is both discrete from other conspecific population segments and significant to the taxon to which it belongs, and therefore, it satisfies the DPS policy criteria.

The BRT considers 60ºN latitude and the equator as the north-south boundaries and 40ºW longitude as the west boundary of the Northeast Atlantic Ocean population segment. The BRT considers the boundary between the Northeast Atlantic Ocean and Mediterranean Sea population segments as 5º 36’W longitude (Strait of Gibraltar). These boundaries are based on oceanographic features, loggerhead sightings, thermal tolerance, fishery bycatch data, and information on loggerhead distribution from satellite telemetry and flipper tagging studies. The loss of the Northeast Atlantic population segment would result in a significant gap in the range of the taxon. Genetic studies have shown that adult populations are highly structured with no overlap in distribution among adult loggerheads from the Northwest Atlantic, Northeast Atlantic, South Atlantic, and Mediterranean. There is no evidence or reason to believe that female loggerheads from the Northwest Atlantic, Mediterranean Sea, or South Atlantic nesting beaches would repopulate the Northeast Atlantic nesting beaches should these populations be lost. There is also no evidence of movement of Northeast Atlantic adults west of 40ºW longitude or east of the Strait of Gibraltar (5º 36’W longitude). The BRT concludes that the Northeast Atlantic population segment is both discrete from other conspecific population segments and significant to the taxon to which it belongs, and therefore, it satisfies the DPS policy criteria.

The BRT considers the Mediterranean Sea west to 5º 36’W longitude (Strait of Gibraltar) as the boundary of the Mediterranean Sea population segment based on oceanographic features, loggerhead sightings, thermal tolerance, fishery bycatch data, and information on loggerhead distribution from satellite telemetry and flipper tagging studies. The loss of the Mediterranean Sea population segment would result in a significant gap in the range of the taxon. Genetic studies show that adult populations are highly structured with no overlap in distribution among adult loggerheads from the Northwest Atlantic, Northeast Atlantic, South Atlantic, and Mediterranean. There is no evidence or reason to believe that female loggerheads from the Northwest Atlantic, Northeast Atlantic, or South Atlantic nesting beaches would repopulate the Mediterranean Sea nesting beaches should these populations be lost. As previously described, adults from the Mediterranean Sea population segment appear to forage closer to the nesting beaches in the eastern basin, and most flipper tag recoveries from females nesting in Greece have occurred in the Adriatic Sea and off Tunisia (Margaritoulis *et al*. 2003, Lazar *et al*. 2004). There is no evidence of movement of adult Mediterranean Sea loggerheads west of the Strait of Gibraltar (5º 36’W longitude). The BRT concludes that the Mediterranean Sea population segment is both discrete from other conspecific population segments and significant to the taxon to which it belongs, and therefore, it satisfies the DPS policy criteria.

The BRT considers the equator and 60ºS latitude as the north-south boundaries and 20ºE longitude at Cape Agulhas on the southern tip of Africa and 67ºW longitude as the east-west boundaries of the South Atlantic Ocean population segment based on oceanographic features, loggerhead sightings, thermal tolerance, fishery bycatch data, and information on loggerhead distribution from satellite telemetry and flipper tagging studies. The loss of the South Atlantic population segment would result in a significant gap in the range of the taxon. Genetic studies show that adult populations are highly structured with no overlap in distribution among adult loggerheads from the Northwest Atlantic, Northeast Atlantic, South Atlantic, and Mediterranean. There is no evidence or reason to believe that female loggerheads from the Northwest Atlantic, Northeast Atlantic, or Mediterranean Sea nesting beaches would repopulate the South Atlantic nesting beaches should these populations be lost. The BRT concludes that the South Atlantic population segment is both discrete from other conspecific population segments and significant to the taxon to which it belongs, and therefore, it satisfies the DPS policy criteria.

In summary, the BRT has identified nine loggerhead DPSs distributed globally: (1) North Pacific Ocean DPS, (2) South Pacific Ocean DPS, (3) North Indian Ocean DPS, (4) Southeast Indo-Pacific Ocean DPS, (5) Southwest Indian Ocean DPS, (6) Northwest Atlantic Ocean DPS, (7) Northeast Atlantic Ocean DPS, (8) Mediterranean Sea DPS, and (9) South Atlantic Ocean DPS.

3.2. Significant Portion of its Range (SPOIR) Assessment

The BRT determined that the range of each DPS contributes meaningfully to the conservation of the DPS and that populations that may contribute more or less to the conservation of each DPS throughout a portion of its range cannot be identified due to the highly migratory nature of the listed entity.

The loggerhead sea turtle is highly migratory and crosses multiple domestic and international geopolitical boundaries. Depending on the life stage, they may occur in oceanic waters or along the continental shelf of landmasses, or transit back and forth between oceanic and neritic habitats. Protection and management of both the terrestrial and marine environments is essential to recovering the listed entity. Management measures implemented by any State, foreign nation, or political subdivision likely would only affect individual sea turtles during certain stages and seasons of the life cycle. Management measures implemented by any State, foreign nation, or political subdivision may also affect individuals from multiple DPSs because juveniles from disparate DPSs can overlap on foraging grounds or migratory corridors (e.g., Northwest Atlantic, Northeast Atlantic, and Mediterranean Sea DPSs). The “significant” term in SPOIR refers to the contribution of the population(s) in a portion of the range to the conservation of the listable entity being considered. The BRT was unable to subdivide the geographic range of loggerhead populations below the DPS level in which threats and conservation efforts specific to any portion of its range can be identified and analyzed for listing status without disregarding the impacts of these threats and conservation efforts to the listed entity throughout all of its range.

SECTION 4—ASSESSMENT OF EXTINCTION RISK

4.1. Description of Extinction Risk Assessment Approaches

To assess extinction risks of loggerhead DPSs, we conducted two independent analyses. The first analysis used the diffusion approximation approach based on time series of counts of nesting females (Lande and Orzack 1988, Dennis *et al*. 1991, Holmes 2001, Snover and Heppell 2009). This analysis provided a metric (susceptibility to quasi-extinction or SQE) to determine if the probability of a population's risk of quasi-extinction is high enough to warrant a particular status listing (Snover and Heppell 2009). This approach is based on stochastic projections of observed trends and variability in the numbers of mature females at various nesting beaches. The second approach used a deterministic stage-based population model that focused on determining the effects of known anthropogenic mortalities on each DPS with respect to the vital rates of the species. Anthropogenic mortalities were added to natural mortalities and possible ranges of population growth rates were computed as another metric of population health. Because this approach is based on matrix models, we call it a threat matrix analysis. This approach focused on how additional mortalities may affect the future growth and recovery of a loggerhead turtle DPS. The first approach (SQE) was solely based on the available time-series data on the numbers of nests at nesting beaches, whereas the second approach (threat matrix analysis) was based on the known biology of the species and anthropogenic mortalities independent of observed nesting beach data.

4.2. Computation of Susceptibility to Quasi-extinction (SQE)

4.2.1. Methods

Estimates of quasi-extinction risk are known to have high degrees of uncertainty due to the stochastic nature of populations and their environments and the error involved in data collection and subsequent parameter estimation (Holmes *et al*. 2007). However, there are ongoing needs for management to classify populations in terms of their status when only limited data, often with high observation error rates, are available. To address this need, Snover and Heppell (2009) presented a quasi-extinction risk index called susceptibility to quasi-extinction (SQE) that can be used to classify sea turtle nesting populations based on relative risks using only nesting beach census data. This index integrates parameter uncertainty and stochasticity in extinction risk forecasting while allowing managers to balance the risk of making Type I (considering a population to be not at risk when it is) and Type II (considering a population to be at risk when it is not) errors when making decisions. While they showed that the method is robust in assessing actual risk (in terms of a binary metric of "at risk" or "not at risk") using population simulations, they clarify that SQE values are not indicative of a true probability of quasi-extinction because they assume constant distributions of trend and variance over long time frames (three generations) and ignore density dependence. Rather, the index serves as a tool for classifying populations by relative status.

The technique involves standard methods of diffusion approximation (Lande and Orzack 1988, Dennis *et al*. 1991). These methods are based on a density-independent exponential model in a randomly varying environment:

$$N_{t+1} = N_t \exp(\mu + \varepsilon) , \qquad (1)$$

where $\varepsilon \sim N(0,\sigma^2)$, N is the population size, and t is time (Dennis *et al.* 1991, Holmes 2001). Assuming that the lognormal distribution can be used to compute the probability that the population will be of a certain size, two parameters estimated by this method are μ, the arithmetic mean of the log population growth rate, and σ^2, the variance of the log population growth rate, which accounts for sources of variability, including environmental and demographic stochasticity, and observation error (Holmes 2001, Morris and Doak 2002 - Chapter 5). These parameters were estimated using the regression analysis, whereas the confidence intervals were estimated using the method of Dennis *et al.* (1991) and Morris and Doak (2002). While not as statistically robust as the methods presented by Holmes (2001), the methods of Dennis *et al.* (1991) and Morris and Doak (2002) are simpler and more tractable for managers. For the lengths of time series typically available for sea turtle populations (i.e., <20 years), Snover and Heppell (2009) demonstrated that both methods performed equally well in determining SQE. These estimated parameters were used to make inferences on total population growth rates and quasi-extinction probability.

The data available for this analysis were time series of annual counts of nests or nesting females. When the data were in units of nests per yr, we converted these numbers to estimates of nesting females (see Section 4.2.2 for details). For each nesting beach or region, we followed the recommendations in Snover and Heppell (2009), and used a running-sum of 3 yr. This data smoothing provides a more accurate reflection of population change by reducing year-to-year fluctuations in the number of nests that affect our translation of the number of nests to the number of adult females and actual population processes. Current adult female population size, n_0, was estimated as the sum of the last 3 yr of data. We used the parametric bootstrap estimation procedure from Morris and Doak (2002) to compute the μ and σ^2 distributions required to calculate susceptibility to quasi-extinction (SQE) and a wide range of quasi-extinction thresholds (QETs). We used a range from 2.5 to 97.5% of the current abundance of nesting females as potential QETs. Because loggerhead turtles are likely to mature at > 30 yr (Snover 2002), we used the time period of 100 yr to compute QETs, which is consistent with the IUCN criteria (3 generations or 100 years, whichever is shorter). To incorporate the uncertainty of parameter estimates in determining SQE, we used 95% confidence limits of $\hat{\mu}$ and $\hat{\sigma}^2$, using the parametric bootstrapping method of Morris and Doak (2002).

The following steps were used to obtain SQEs:

1. μ_i and σ_i^2 were drawn randomly and independently from respective distributions, $\{\tilde{\mu}_i, \tilde{\sigma}_i^2\}$.
2. Each pair $\{\tilde{\mu}_i, \tilde{\sigma}_i^2\}$ was used to compute the probability of reaching a QET within the next 100 years.
3. Steps 1 and 2 were repeated 5000 times ($i = 1, \ldots, 5000$) to create a distribution of probabilities of reaching the QET.
4. The SQE metric for the QET is the proportion of the probabilities that are >0.9 (cut-off probability).

5. Steps 1 through 4 were repeated for a range of QETs between 2.5% and 97.5% of the current abundance of nesting females.

Using simulations, Snover and Heppell (2009) demonstrated that SQE values greater than 0.4 indicated a population has >0.9 probability of quasi-extinction. At this critical value (SQE = 0.40), Type I and Type II errors are minimized simultaneously at approximately 10%. Reducing the critical value to 0.3 lessens the ‘Type I’ error rate but increases the ‘Type II’ error rate (Snover and Heppell 2009). The choice of 0.9 as the cut-off probability was arbitrary and values other than 0.9 could be used. However, new critical values other than 0.4 needed to be established for different values of the cut-off probability. Qualitatively, the results would not differ if a value other than 0.9 was used (Snover and Heppell 2009). In this assessment, we used the cut-off probability of 0.9 as in Snover and Heppell (2009) and a critical value for the SQE of 0.30, which reduced the ‘Type I’ error (a DPS is considered to be not at risk when in fact it is). SQE values greater than 0.30, therefore, indicate the DPS is at risk.

4.2.2. Count Data

The following datasets were used for the computations of SQEs. Ideally, the SQE analysis is conducted on the numbers of nesting females over time, which represent the temporal change in the number of reproductive females. The statistic, however, is difficult to determine because of the required effort to mark and identify individual turtles. Consequently, data are often collected for the number of nests per nesting season. To estimate the number of females from the observed number of nests, we use the average number of nests per female per nesting season for that region. Because the average is treated as a constant over time for each DPS (Table 1) and SQE is calculated as probability of a proportional reduction in population size, the SQE analysis is unaffected by the choice of unit (i.e., the number of females or nests). In the following, we use the number of females as the unit of analysis. Snover and Heppell (2009) suggest that time series at least 15 years in length are needed for diffusion approximation analysis, which limits the populations that have adequate data for applying this approach. We did include one additional nesting recovery unit in the Northwest Atlantic Ocean DPS, the Northern Gulf of Mexico Recovery Unit, with only 12 years of data in order to characterize as much of that region as possible. Results from that recovery unit should be interpreted with caution, although the general trends observed are consistent with region as a whole.

Table 1. Life history parameters used for the nine DPSs. Juvenile survival rates (oceanic and neritic) were computed by using the negative binomial stage duration model and fixing survival rates of adults and the first year and fecundity. See text for details. Also please consult the excel file posted on the NMFS Office of Protected Resources website [http://www.nmfs.noaa.gov/pr/species/statusreviews.htm] to access information used in identifying the life history parameters.

	North Pacific	South Pacific	Indian*	NW Atlantic	NE Atlantic	Med	South Atlantic
Fecundity							
Remigration interval (yrs)	2.7	3.8	3	3	3	2.6	3
Clutch freq. (yr^{-1})	3	3.4	5	5	5	2	5
Clutch size	112	127	101-118	115	85	95	123
Sex ratio (% female)	50	50	50	50	50	50	50
Emergence success	0.61	0.71	0.54	0.54	0.319	0.67	0.67
Habitat use							
Juveniles							
Prop. neritic	0.14	0.50	0.655	0.655	0.655	0.50	0.655
Yrs oceanic	27.0	14.5	10	10	14.5	14.5	10
Yrs neritic	2.0	14.5	19	19	14.5	14.5	19
Mean AFR (yrs)	30	30	30	30	30	30	30
SD AFR	5	5	5	5	5	5	5
Adults							
Prop. neritic (non-nesting)	0.82	0.85	0.58	0.95	0.30	0.66	0.95
Prop. neritic (nesting)	0.82	0.85	0.73	0.95	0.30	0.66	0.95
Survival rates							
Aquatic hatchlings	0.400	0.400	0.400	0.400	0.400	0.400	0.400
Oceanic juvenile							
$\lambda_0 = 1.05$	0.858	0.828	0.796	0.794	0.858	0.856	0.787
$\lambda_0 = 1.10$	0.914	0.894	0.877	0.875	0.933	0.933	0.863
Neritic juvenile							
$\lambda_0 = 1.05$	0.928	0.911	0.895	0.893	0.922	0.921	0.883
$\lambda_0 = 1.10$	0.955	0.949	0.933	0.932	0.956	0.954	0.923
Oceanic adult	0.950	0.950	0.950	0.950	0.950	0.950	0.950
Neritic adult	0.950	0.950	0.950	0.950	0.950	0.950	0.950

*All parameters are identical for three DPSs in the Indian Ocean.

4.2.2.1. North Pacific Ocean DPS

All loggerhead nesting in the North Pacific occurs in Japan. Nesting data for this region are from two sources, the Sea Turtle Association of Japan (STAJ; unpublished data provided to the Western Pacific Fisheries Management Council) and Kamezaki *et al*. (2002). Data from the STAJ represent total counts for Japan from 1998-2007. Snover (2008) combined these datasets to achieve a time series from Japan from 1990-2007 (Figure 1). Due to the nature of the STAJ data, the data from Japan are represented by a single time series, rather than by nesting beaches.

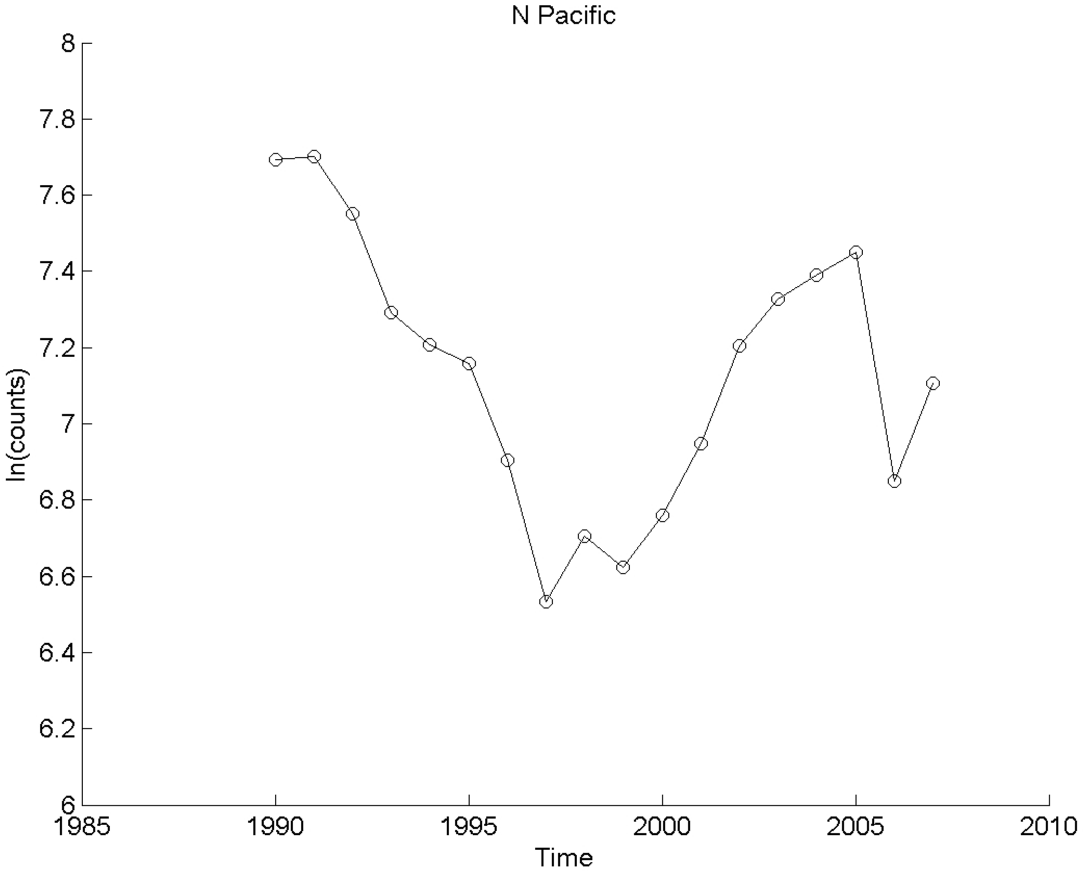


Figure 1. Change in the number of nesting females at nesting beaches for the North Pacific Ocean DPS. The number of nesting females was computed from the observed number of nests divided by the mean clutch frequency (3 yr^{-1}; Table 1).

4.2.2.2. South Pacific Ocean DPS

We used nesting census data for index beaches in eastern Australia (Limpus 2009; Figure 2). These include mainland beaches, Wreck Rock beaches, Great Barrier Reef (GBR) Coral Cays, and Wreck and Tyron Islands. Each beach was analyzed separately.

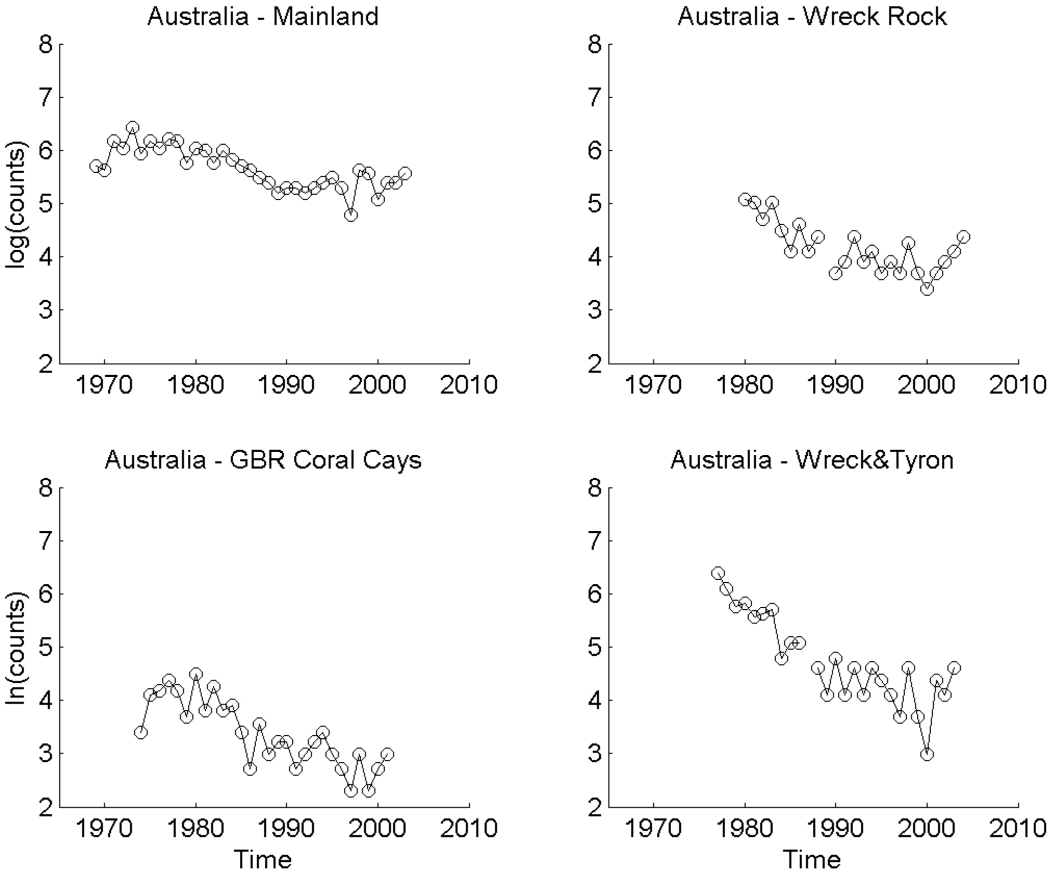


Figure 2. The change in the numbers of nesting females at four nesting beaches for the South Pacific Ocean DPS.

4.2.2.3. North Indian Ocean DPS

No adequate time series of nesting beach data was available for this DPS.

4.2.2.4. Southeast Indo-Pacific Ocean DPS

No adequate time series of nesting beach data was available for this DPS.

4.2.2.5. Southwest Indian Ocean DPS

Count data for the Southwest Indian Ocean DPS were obtained from Baldwin *et al*. (2003; Figure 3).

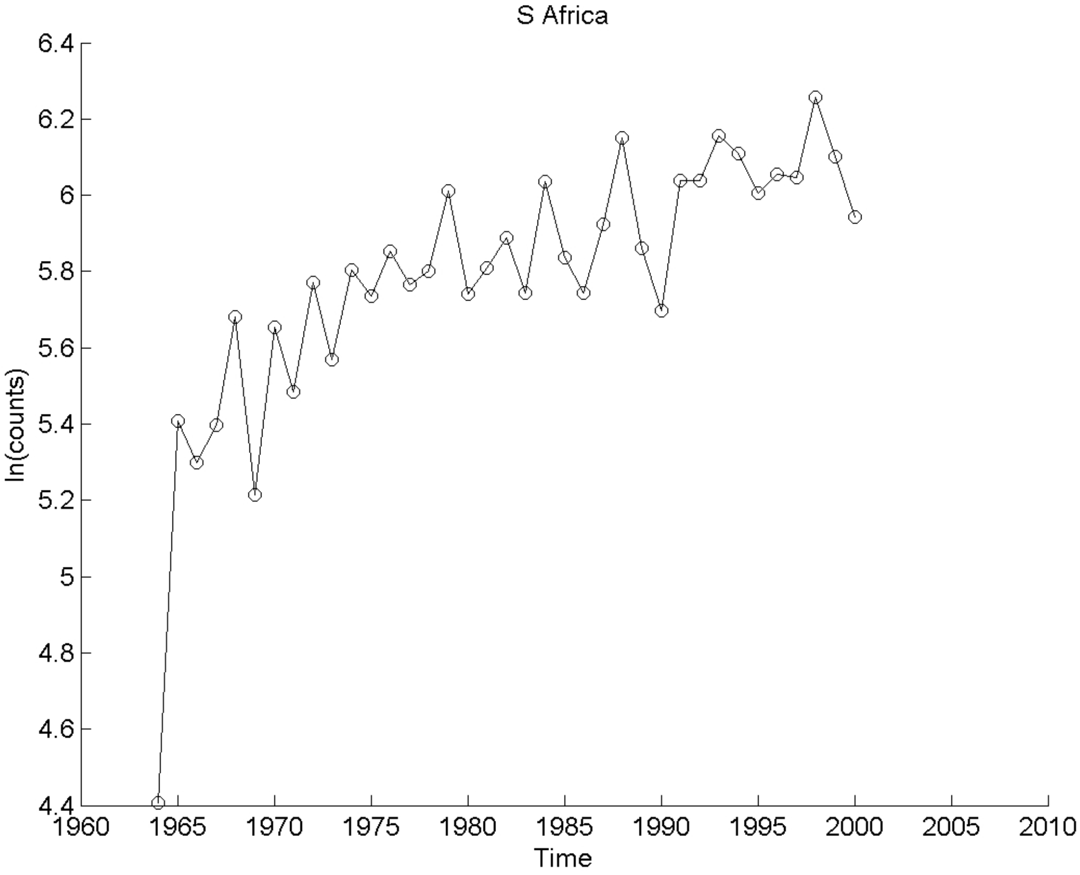


Figure 3. Change in the number of nesting females for the Southwest Indian Ocean DPS.

4.2.2.6. Northwest Atlantic Ocean DPS

The Recovery Plan for the Northwest Atlantic Population of the Loggerhead Sea Turtle (NMFS and FWS 2008) recognized five recovery units (subpopulations) of loggerhead turtles within the Northwest Atlantic:

1. Northern Recovery Unit (southern VA through FL/GA border)
2. Peninsular Florida Recovery Unit (FL/GA border through Pinellas County, FL)
3. Dry Tortugas Recovery Unit (islands located west of Key West, FL)
4. Northern Gulf of Mexico Recovery Unit (Franklin County, FL, through TX)
5. Greater Caribbean Recovery Unit (Mexico through French Guiana, The Bahamas, Lesser Antilles, and Greater Antilles)

Of these recovery units, four have adequate time series data for applying this analysis (Figure 4). Data for the Northern Recovery Unit and the Peninsular Florida Recovery Unit comprise sums of numerous individual beaches. For the Northern Gulf of Mexico Recovery Unit data, we used a time series from the Florida Panhandle. We used nesting beach data from the Yucatan Peninsula (J. Zurita, personal communication, 2008) to represent the Greater Caribbean Recovery Unit.

4.2.2.7. Northeast Atlantic Ocean DPS

Although nesting data are available for the Cape Verde Archipelago, the time series is not long enough for the diffusion approximation analysis.

4.2.2.8. Mediterranean Sea DPS

Representative nesting beach data were not available for this DPS. Although nesting data are available for Greece and Turkey, the BRT does not think these data are sufficiently representative of the entire region given the potential differences in trends among nesting areas.

4.2.2.9. South Atlantic Ocean DPS

Data for the South Atlantic Ocean DPS were obtained from Marcovaldi and Chaloupka (2007; Figure 5).

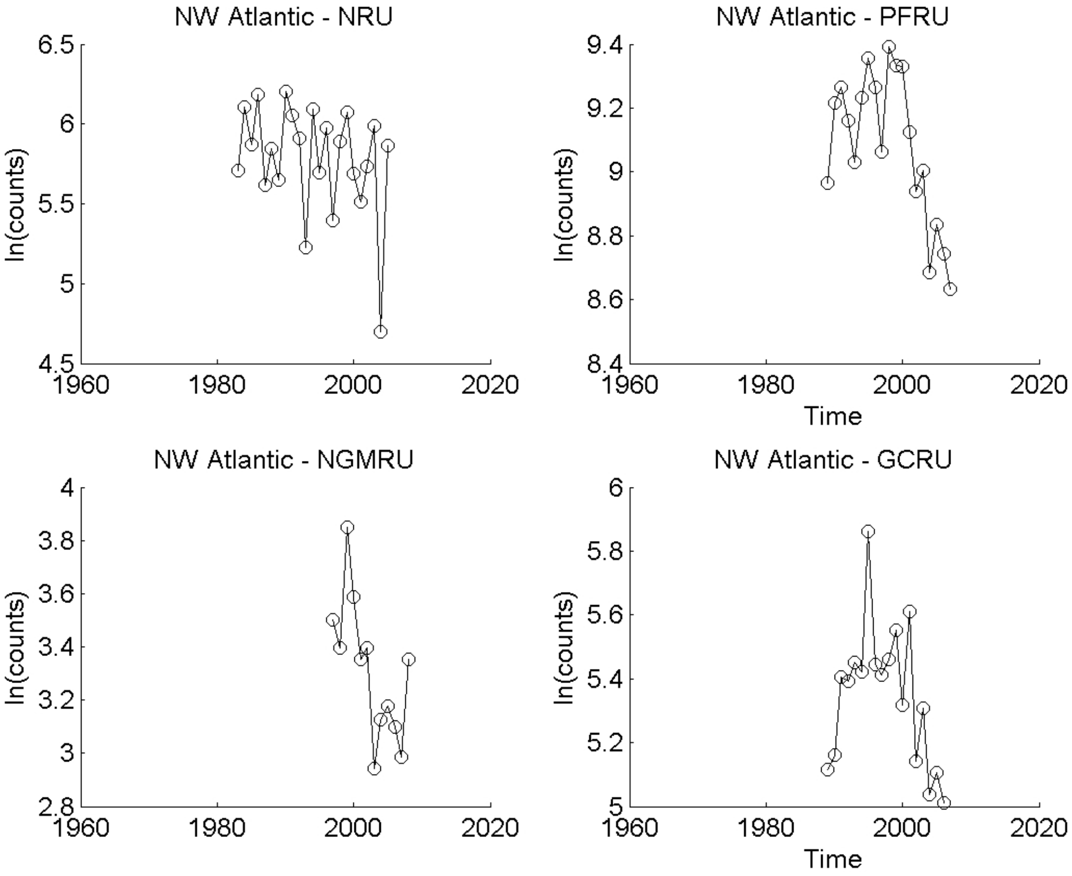


Figure 4. Changes in the numbers of nesting females at nesting beaches for the Northwest Atlantic Ocean DPS. The number of nesting females was computed from the observed number of nests divided by the mean clutch frequency (5 yr^{-1}; Table 1). NRU = Northern Recovery Unit, PFRU = Peninsular Florida Recovery Unit, NGMRU = Northern Gulf of Mexico Recovery Unit, and GCRU = Greater Caribbean Recovery Unit.

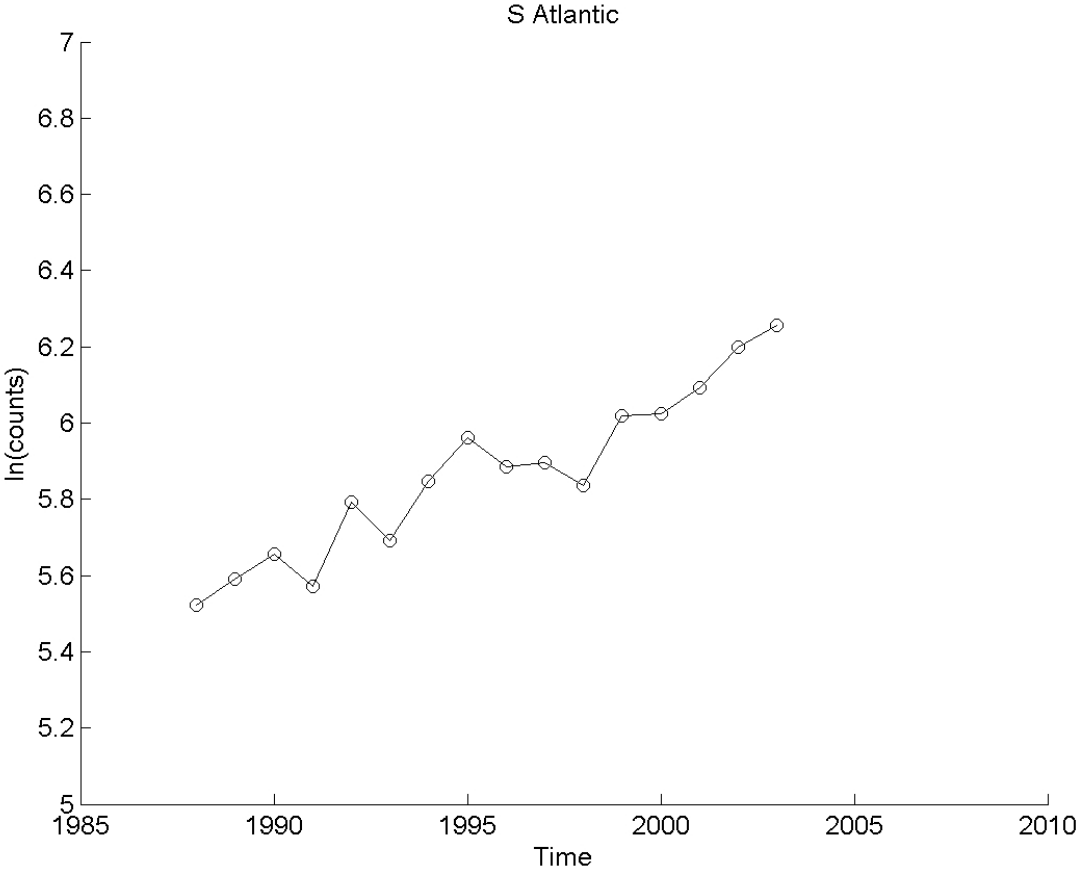


Figure 5. Change in the number of nesting females at nesting beaches for the South Atlantic Ocean DPS. The number of nesting females was computed from the observed number of nests divided by the mean clutch frequency (5 yr^{-1}; Table 1).

4.2.3. Results

The metric (susceptibility to quasi-extinction or SQE) is an increasing function of quasi-extinction threshold (QET). Unless a DPS is increasing, the likelihood of the population reaching some level of QET, as measured in the proportion of current abundance, increases with QET. For example, if 95% of the current abundance is used as the QET, the likelihood of a declining population reaching the QET is high. For severely declining populations, the QET needs to be set very low to reach the defined SQE value of 0.3. Only the Southwest Indian Ocean and South Atlantic Ocean DPSs indicated increasing trends ($\mu > 0$; Table 2). Large variability in the count data resulted in wide confidence intervals; all but three confidence intervals included zero (Table 2).

For the North Pacific Ocean DPS, SQE = 0.3 was reached at approximately 3% of the current female abundance, indicating the high likelihood of quasi-extinction for almost all levels of QET (Figure 6). This was caused by the recent decline of nesting females at the majority of nesting beaches in Japan (Snover 2008).

All monitored nesting beaches for the South Pacific Ocean DPS indicated high likelihood of SQE (Figure 7). As it was expected from observed counts (Figure 2), nesting beaches on Mainland Australia indicated a better chance of persisting than other sites (Figure 7). For Wreck Rock, Great Barrier Reef (GBR), and Wreck and Tyron nesting beaches, quasi-extinction was certain for all values of QET.

For the recovery units with sufficient data within the Northwest Atlantic Ocean DPS, the likelihood of quasi-extinction was highest for the Northern Gulf of Mexico Recovery Unit (NGMRU), where SQE was greater than 0.3 for all values of QET (Figure 8). For the other three recovery units, SQE = 0.3 was reached at QET < 0.3 (Figure 8).

Two DPSs indicated low likelihoods of SQE; the Southwest Indian Ocean (data from South Africa) and South Atlantic Ocean DPSs (Figures 9 and 10). Because of the observed increases in the nesting females in both time series (Baldwin *et al*. 2003, Figure 3; Marcovaldi and Chaloupka 2007, Figure 5), the likelihood of quasi-extinctions are negligible for these DPSs using the SQE analysis.

Table 2. Results of the diffusion approximation analysis for each nesting beach or region. The parameter μ is the arithmetic mean of the log population growth rate, and σ^2 is the variance of the log population growth rate. CI is confidence interval. PFRU = Peninsular Florida Recovery Unit, NRU = Northern Recovery Unit, NGMRU = Northern Gulf of Mexico Recovery Unit, and GCRU = Greater Caribbean Recovery Unit.

DPS	μ	95% CI μ	σ^2	95% CI σ^2
North Pacific				
All Japan	-0.032	[-0.111, 0.046]	0.020	[0.011, 0.048]
South Pacific				
Mainland Australia	-0.013	[-0.047, 0.021]	0.009	[0.006, 0.016]
Wreck Rock, Australia	-0.038	[-0.107, 0.032]	0.023	[0.014, 0.048]
Great Barrier Reef, Australia	-0.050	[-0.116, 0.017]	0.026	[0.016, 0.049]
Wreck and Tyron Islands, Australia	-0.075	[-0.150, -0.001]	0.030	[0.018, 0.058]
Southwest Indian	0.029	[0.003, 0.054]	0.005	[0.004, 0.009]
Northwest Atlantic				
PFRU	-0.026	[-0.065, 0.013]	0.005	[0.003, 0.013]
NRU	-0.012	[-0.079, 0.055]	0.021	[0.012, 0.043]
NGMRU	-0.049	[-0.121, 0.022]	0.009	[0.004, 0.029]
GCRU	-0.012	[-0.068, 0.043]	0.010	[0.006, 0.025]
South Atlantic				
Bahia and Espirito Santa, Brazil	0.046	[0.023, 0.068]	0.001	[0.001, 0.004]

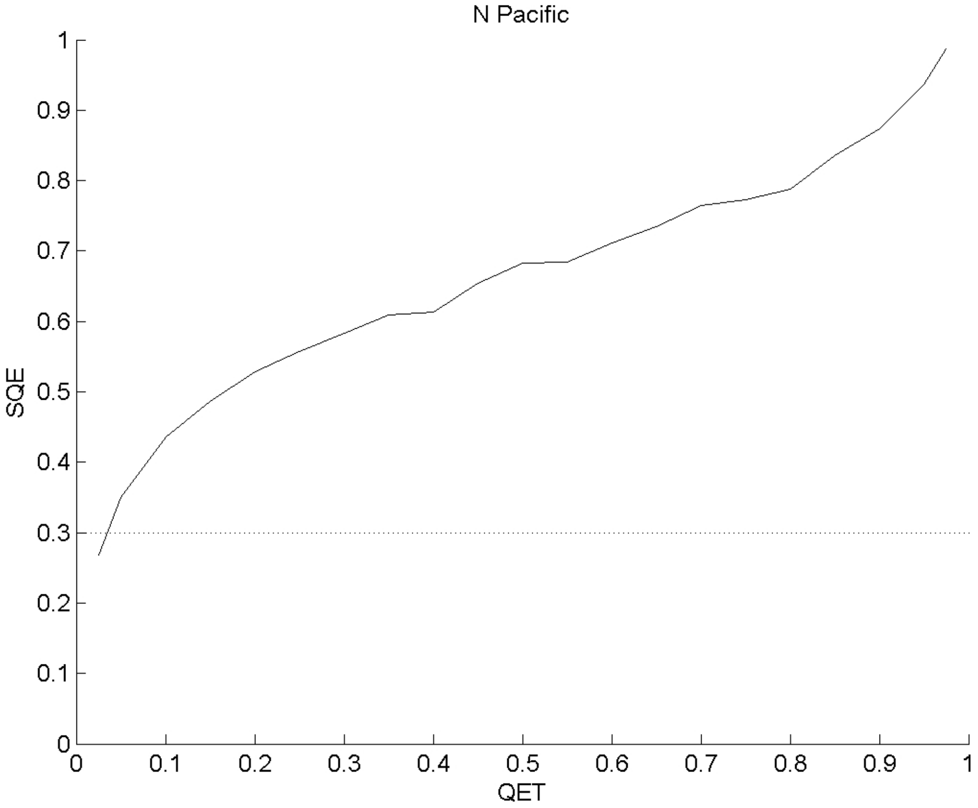


Figure 6. Susceptibility to quasi-extinction (SQE) as a function of quasi-extinction threshold (QET) for the North Pacific Ocean DPS. QET is defined as the proportion of the current female abundance. The dotted line indicates SQE = 0.3, which was adapted as the threshold for the analysis.

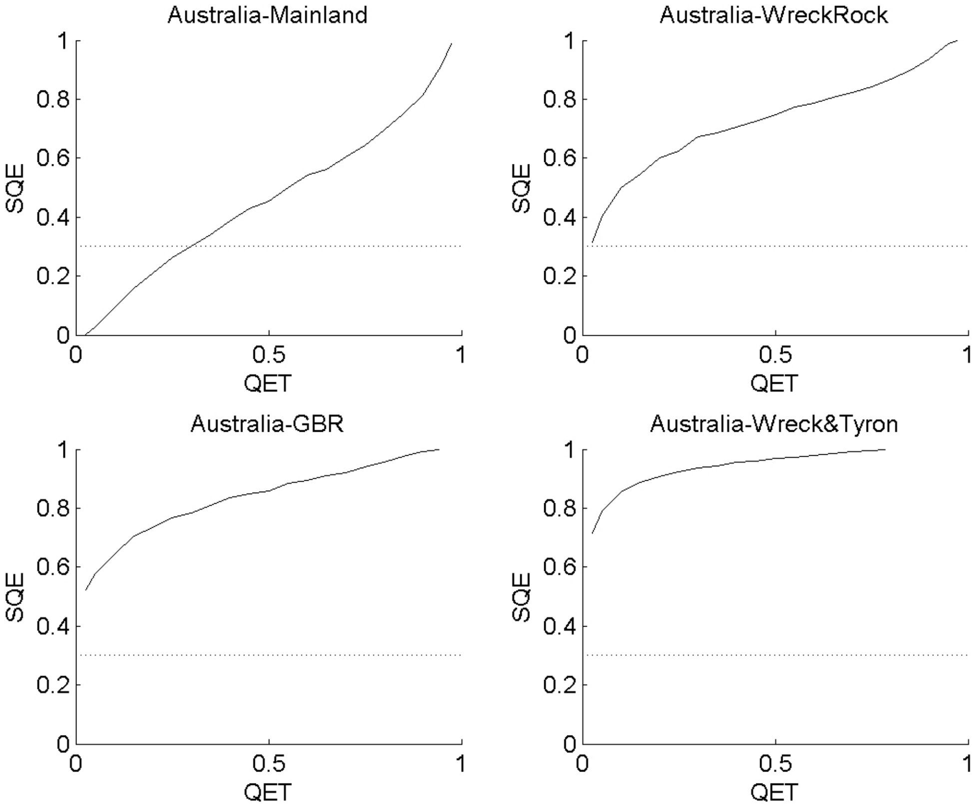


Figure 7. Susceptibility to quasi-extinction (SQE) as a function of quasi-extinction threshold (QET) for the South Pacific Ocean DPS. QET is defined as the proportion of the current female abundance. Dotted lines indicate SQE = 0.3, which was adapted as the threshold for the analysis.

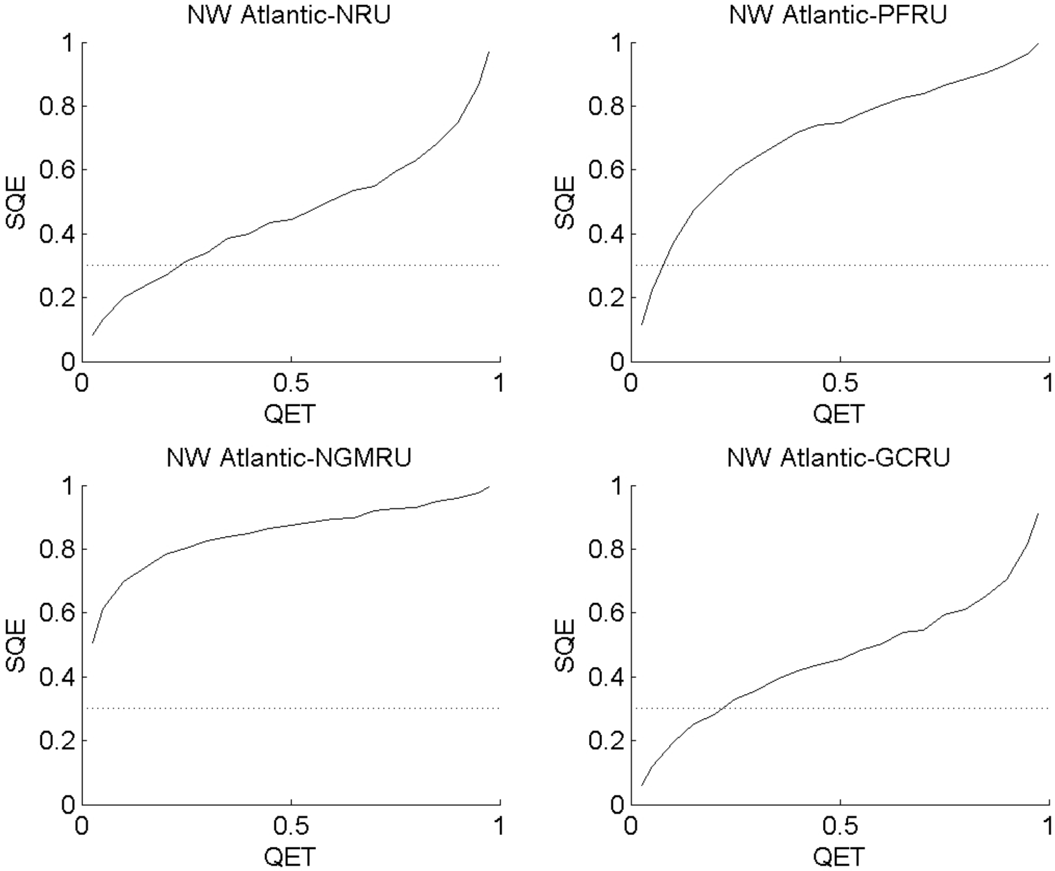


Figure 8. Susceptibility to quasi-extinction (SQE) as a function of quasi-extinction threshold (QET) for the Northwest Atlantic Ocean DPS. QET is defined as the proportion of the current female abundance. Dotted lines indicate SQE = 0.3, which was adapted as the threshold for the analysis. NRU = Northern Recovery Unit, PFRU = Peninsular Florida Recovery Unit, NGMRU = Northern Gulf of Mexico Recovery Unit, and GCRU = Greater Caribbean Recovery Unit.

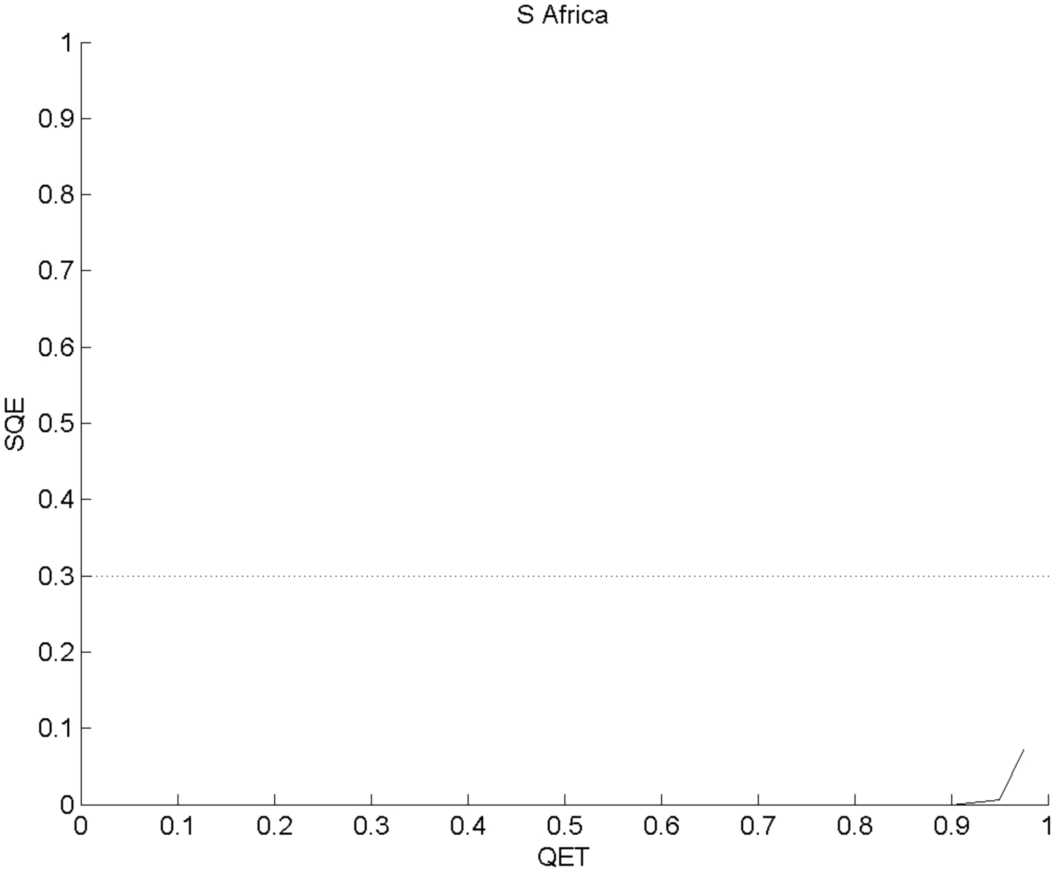


Figure 9. Susceptibility to quasi-extinction (SQE) as a function of quasi-extinction threshold (QET) for the Southwest Indian Ocean DPS. Data are for South Africa nesting beaches. QET is defined as the proportion of the current female abundance. Dotted lines indicate SQE = 0.3, which was adapted as the threshold for the analysis.

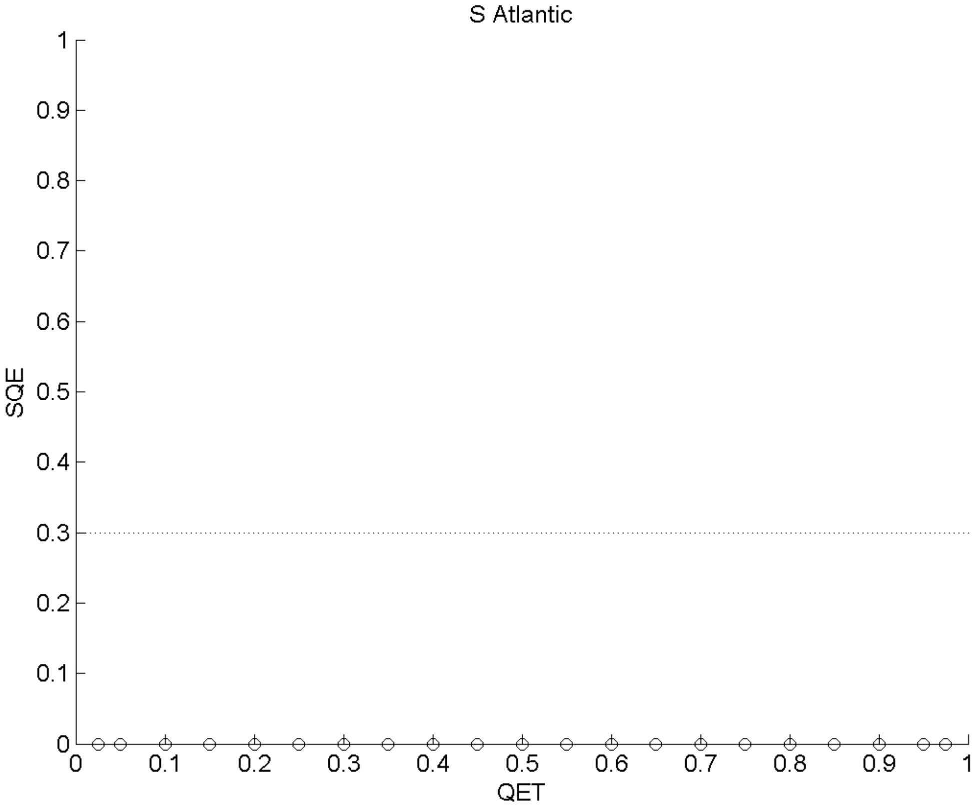


Figure 10. Susceptibility to quasi-extinction (SQE) as a function of quasi-extinction threshold (QET) for the South Atlantic Ocean DPS. QET is defined as the proportion of the current female abundance. The dotted line indicates SQE = 0.3, which was adapted as the threshold for the analysis.

4.3. Threat Matrix Analysis

4.3.1. General Modeling Approach

The second approach to our risk analysis was based on a metric that indicates whether or not known threats may be sufficient to keep a DPS from recovering. Using as much information on the biology of loggerhead turtles as possible, a discrete-time stage-structured population model was constructed for each DPS. First, the model was parameterized to represent the maximum plausible vital rates and population growth rate, which may be attained for a recovering DPS (base population model hereafter). We did not incorporate demographic and environmental stochasticities in the base matrix. Known anthropogenic threats to each life stage of a DPS, measured as additional annual mortality, were quantified using available data and experts' opinions, where the stage-specific additional annual mortality was summarized in a matrix format (threat matrix). The base population model and threat matrix then were multiplied to determine effects of the additional mortalities on the base population model. We use the dominant eigenvalue (λ) of the product of the two matrices as an index of DPS status, where $\lambda > 1$ indicates a healthy population, whereas $\lambda < 1$ indicates possible decline of the population in the future. Because the base matrix of a DPS was developed on hypothesized dominant eigenvalues, the actual value of λ should not be considered as the population growth rate of a DPS. We also provide the proportion of $\lambda > 1$. This proportion increases as the additional mortalities decrease.

Briefly, the risk analysis for each DPS consisted of four stages. First, a base population model was constructed. Second, anthropogenic threats were quantified. Third, effects of the threats were calculated via the dominant eigenvalue of a linear time-invariant model. Finally, a range of dominant eigenvalues was determined according to the ranges of threat levels. Details of each step are described in the following sections. Note that this analysis is similar to a demographic population viability analysis (PVA). However, we did not consider environmental or demographic stochasticity, density dependence, autocorrelations in vital rates, or sampling variations. Consequently, this analysis should not be considered as a complete PVA. The two metrics (the dominant eigenvalue and the proportion of $\lambda > 1$) are indices of potential population decline of a DPS, considering the existing information about the biology of the species and experts' opinions on known anthropogenic threats to the DPS. The analysis does not provide estimates for the likelihood or probability of extinction.

4.3.2. Survival Rates and Population Growth Rate for the Base Models

Using knowledge of life history of loggerhead turtles and following the previous work of others (Crouse *et al*. 1987, Crowder *et al*. 1994, Heppell *et al*. 2003), we constructed a stage-based population model for each DPS. The model consisted of four stages; first year, oceanic juveniles, neritic juveniles, and adults. The durations of juvenile stages were modeled using the negative binomial stage distribution model (NBSD model; Caswell 2001, pp. 164-165).

The NBSD model does not rely on the assumption of stable age distribution within a stage to compute the transition probability, as other methods would. Detailed descriptions are available in Caswell (2001). Briefly, for a series of k identical pseudo-stages within a juvenile stage, we assign the transition probability (p) of moving from one pseudo-stage to the next pseudo-stage.

Because the total time required to step through all *k* pseudo-stages is equal to the time required for the k^{th} success in a series of identical Bernoulli trials with probability *p*, the total time (*T*) can be computed with a negative binomial distribution:

$$\Pr(T = t) = \binom{t-1}{k-1} p^k (1-p)^{t-k}, \tag{1}$$

where $t \geq k$, $k = 1, 2, \cdots, t$, and $0 < p < 1$. The mean (E[T]) and variance (var[T]) of this distribution are:

$$E[T] = \frac{k}{p} \text{ and} \tag{2}$$

$$\text{var}[T] = \frac{k(1-p)}{p^2}. \tag{3}$$

These equations can be solved for *p* and *k* by rearranging the above expressions:

$$p = \frac{E[T]}{\text{var}[T] + E[T]} \text{ and} \tag{4}$$

$$k = \frac{(E[T])^2}{\text{var}[T] + E[T]}. \tag{5}$$

Survival rates and transition probabilities among pseudo-stages are assumed identical. Therefore, each element of a pseudo-stage is multiplied by the stage specific survival rate. In other words, the *k* diagonal elements for a juvenile stage are $(1-p)\varphi$, whereas the sub-diagonal elements are $p\varphi$, where φ is the stage specific survival rate. For the same mean stage duration, a larger variance implies fewer pseudo stages (*k*) with smaller *p*, and vice versa. This method allowed us to construct projection matrices without assuming the stable age distribution within each juvenile stage or a fixed stage duration, while acknowledging the insufficient data to construct age-based models with variable growth rates.

The NBSD model requires the mean and variance of durations of stages. The age at first reproduction (AFR), however, has not been estimated directly for loggerhead turtles. Skeletochronological studies have indicated that loggerhead turtles in the western Atlantic may reach their first reproduction at approximately 30 years (Snover 2002). The experts of the team agreed on a standard deviation for the AFR of 5 years. Assuming a negative binomial distribution with a mean 30 years and a standard deviation of 5 years, 95% of AFRs are between 21 years and 41 years. These values were deemed reasonable by the experts. For computing the mean and variance for each stage of juveniles, i.e., oceanic and neritic, we used the same coefficient of variation (CV = 5/30 = 0.17). For example, if juveniles of a DPS spend 15 years in oceanic and 14 years in neritic habitat, standard deviations for these stages are 2.6 and 2.4, respectively. We conducted a sensitivity analysis to evaluate the effects of these parameters on our conclusions.

The stage-based model requires estimates for stage-specific fertility, mean and variance of durations of juvenile stages, and survival rates. Information from various sources provided data for fertility and hatching success of eggs (Table 1). The post-breeding census model was used for computing the fertility.

The DPS-specific matrices (**A**) have the following basic format:

$$\begin{bmatrix} 0 & 0 & 0 & \cdots & 0 & p_2\varphi_2 f & \varphi_A f \\ \varphi_0 & (1-p_1)\varphi_1 & 0 & \cdots & \vdots & 0 & 0 \\ 0 & p_1\varphi_1 & \ddots & \cdots & \vdots & \vdots & \vdots \\ 0 & 0 & \ddots & (1-p_2)\varphi_2 & 0 & \vdots & \vdots \\ \vdots & \vdots & 0 & p_2\varphi_2 & \ddots & 0 & \vdots \\ \vdots & \vdots & \vdots & 0 & \ddots & (1-p_2)\varphi_2 & 0 \\ \vdots & \vdots & \vdots & \cdots & 0 & p_2\varphi_2 & \varphi_A \end{bmatrix},$$

where φ is survival rate, p is transition probability, f is the fertility (i.e., the number of female hatchlings that emerge per adult female), and subscripts indicate the first year (0), oceanic juveniles (1), neritic juveniles, (2), and adults (A). The numbers of sub-stages for juveniles vary according to the NBSD model described previously.

For survival rates, the vast majority of information comes from studies at nesting beaches, where hatching success (egg survival) is evaluated routinely. However, the incubation period consists of only approximately 6.5 weeks of the first year (Miller *et al.* 2003). Consequently, the total survival rate during the first year of their life is still unavailable. In the past, 0.4 was used for the Kemp's ridley turtle (*Lepidochelys kempii*), based on a model fit to the observed numbers of nests and hatchling production for that species (Heppell *et al.* 2005). Because of the morphological and behavioral similarities between the two species during the first year, we use the same survival rate for loggerhead turtles. Adult survival rates have been estimated for some loggerhead DPSs. Studies have indicated that the survival rates of adult loggerhead turtles are generally greater than 0.8/yr (0.81; Frazer 1983, 0.88; Chaloupka and Limpus 2002, 0.85; Hedges 2007). These estimates, however, likely include anthropogenic mortalities. Consequently, the available estimates are negatively biased as the natural survival rate. Similarly for juveniles, available estimates for juvenile survival rates include anthropogenic mortality rates (Bjorndal *et al.* 2003b, Braun-McNeill *et al.* 2007b, Sasso and Epperly 2007).

The relationship between possible ranges of juvenile survival rates and maximum population growth rates was evaluated graphically. The asymptotic population growth rate of a stage-based matrix model is a function of fertility, survival rates, and transition probabilities. Consequently, the relationship among the average juvenile survival rates, first year survival rates, and dominant eigenvalues can be plotted. Similar approaches have been used to evaluate the relationships among juvenile survival rates, adult survival rates, fecundity, and average age at sexual maturity for the Blanding's turtle (*Emydoidea blandingii*; Congdon *et al.* 1993). We considered adult survival rates from 0.80 to 0.99 per year, juvenile survival rates from 0.70 to 0.95 per year, and the first year survival from 0.01 to 0.50 per year. To constrain the parameter space, we made an assumption that the average juvenile survival rate was less than the average adult survival rate. Examinations of relationships among these parameters allowed us to find the plausible maximum population growth rate (λ).

To compute the juvenile survival rates, a numerical minimization method was used. By fixing the dominant eigenvalue (λ_0), the first year survival rate, adult survival rate, and fertility, the following equation was solved for two juvenile survival rates with the constraint that the oceanic juvenile survival rate (φ_1) is less than the neritic juvenile survival rate (φ_2):

$$f(\varphi_1, \varphi_2) = \left|\lambda_0 - eig(\mathbf{A})\right|,$$

where *eig*() is the characteristic equation to find the dominant eigenvalue of a matrix **A** and | | is the absolute value. The general form of the matrix **A** was shown previously.

In this analysis, several life history parameters were determined from experts' knowledge or based on estimates from small sample sizes (SD of AFR, AFR, proportion of time spent in particular habitat). To determine how these parameters would affect the result of the analysis, sensitivity analyses were conducted. Results of these sensitivity analyses are provided as supplemental material (Section 4.6).

4.3.3. Quantifying Known Threats

We quantified experts' knowledge about the existing anthropogenic threats on loggerhead turtles, which subsequently were combined with the projection model described in the previous section (see Tables 3-11 in Sections 4.4.2.1.-4.4.2.9. for the results of the experts' threats assessments; also consult the excel file posted on the NMFS Office of Protected Resources website [http://www.nmfs.noaa.gov/pr/species/statusreviews.htm] to access information used to derive the estimated threats levels). Because the levels of the existing anthropogenic threats differ among habitats (e.g., high seas vs. coastal oceanic shelf), juveniles and adults were further separated into neritic and oceanic stages. Threats, therefore, were determined for the following six stages: (1) eggs/hatchlings, (2) neritic juveniles, (3) oceanic juveniles, (4) neritic adults, (5) oceanic adults, and (6) nesting females. For each of the following four factors, experts were asked to categorize known threats for each stage in high, medium, low, or very low, according to the level of additional annual mortality (m), where high = $0.20 < m \leq 0.25$, medium = $0.10 < m \leq 0.20$, low = $0.01 < m \leq 0.10$, and very low = $0 \leq m \leq 0.01$. The four factors are:

- The present or threatened, destruction, modification, or curtailment of its habitat or range (habitat).
- Overutilization for commercial, recreational, scientific, or educational purposes (overuse).
- Disease or predation.
- Other natural or manmade factors affecting its continued existence (other).

The estimated threat levels were based on the best information available. Justifications and references for each threat are provided in Section 4.4.2. and in the online threats matrix spreadsheets.

These four factors corresponded to four of the five factors in section 4(a)(1) of the ESA. Another factor (inadequacy of existing regulatory mechanisms) was excluded from this analysis because the effects of the factor were not considered to be reducing the survival rates of loggerhead turtles directly.

For two DPSs (South Pacific and North Pacific), experts were not able to distinguish anthropogenic mortalities from the natural mortalities. In other words, the provided mortalities included natural and anthropogenic mortalities. Consequently, anthropogenic mortalities were computed using the assumed base matrix model for each DPS, which was described in the previous section. Let m_n = annual natural mortality, m_a = annual anthropogenic mortality, and m = annual total mortality. In the discrete time scale, the total mortality is:

$$m = 1 - (1 - m_a)(1 - m_n)\,. \tag{6}$$

Consequently, the annual anthropogenic mortality can be computed from the total and natural mortalities:

$$m_a = 1 + \frac{m - 1}{m_n - 1} \tag{7}$$

To use these threat levels in calculations, we used the range limits of mortality values. For example, when the effect of a factor to a stage is categorized as ‘Low,’ 0.01 and 0.10 were used in the calculations as the lower and upper limits, respectively. To compute the total mortality, mortalities from four factors were summed, which bounded the upper limit of “High” at 0.25 so that the sum would not exceed one. The use of lower and upper limits provided the best and worst case scenarios. The best case scenario was given by the sum of lower limits, whereas the upper limits were used to compute the worst case scenario.

For the three adult stages, threats for three habitats (neritic, oceanic, and nesting) were pooled to conform to the projection model. Because adult loggerhead turtles may use multiple habitats within a year, where they are exposed to different threats, we computed the total annual anthropogenic mortality rate from a threat table using the following two equations. During a nesting year, the total anthropogenic mortality of nesting females is:

$$m_{adult,nesting} = 1 - (1 - m_{adult}^{neritic})^{T_{N,nesting}} (1 - m_{adult}^{oceanic})^{T_{O,nesting}} (1 - m_{adult}^{terrestrial})\,. \tag{8}$$

where $m_{adult,nesting}$ = annual anthropogenic mortality rate for adult females during the nesting year, m_{adult}^{h} = annual adult anthropogenic mortality in the habitat h from a threat table, where h is either oceanic, neritic, or nesting, and $T_{N,j}$, and $T_{O,j}$ are the average durations in years adult loggerhead turtles spend their time annually in neritic and oceanic habitats in years, respectively, and j is either nesting or non-nesting. The total anthropogenic mortality rate during a non-nesting year is:

$$m_{adult,non-nesting} = 1 - (1 - m_{adult}^{neritic})^{T_{N,non-nesting}} (1 - m_{adult}^{oceanic})^{T_{O,non-nesting}}\,, \tag{9}$$

Finally, with an estimated mean remigration interval (R), we calculated the average annual anthropogenic mortality rate of adults by:

$$m_{adult} = 1 - \sqrt[(R+1)]{\left(1 - m_{adult,non-nesting}\right)^R \left(1 - m_{adult,nesting}\right)}. \tag{10}$$

Using the pre-defined mortality thresholds for each level of threats (very low, low, medium, and high), the total anthropogenic mortalities were computed for each of six stages. Although these thresholds may not represent the actual mortalities and may overestimate the anthropogenic mortalities, comparisons of mortalities among life stages within a DPS, as well as comparisons of mortalities among DPSs of a life stage, were conducted to provide relative severity of anthropogenic mortalities among life stages. Finally, to evaluate the effects of these additional anthropogenic mortalities on a DPS, these additional mortality rates were combined with the base matrix model and annual mortality rates for all stage classes computed.

4.3.4. Combining Additional Mortalities and the Base Population Model

Anthropogenic mortalities, quantified by the methods described in the previous section, were combined with the base population model (**A**) from the first section via the "harvest" model described in Caswell (2001; Chapter 18). In the harvest model, the proportion of each stage group that survives the harvest is used to construct a "harvest matrix." Because many of these anthropogenic threats to loggerhead turtle DPSs are not considered as harvest for human consumptions, we call them "threat survival matrices." A threat survival matrix and the base population model for each DPS are multiplied together to construct a projection model with anthropogenic mortalities (Caswell 2001).

A threat survival matrix is a diagonal matrix: $\mathbf{S} = diag(s_1, \ldots, s_c)$, where s_i is the proportion of stage i surviving the threats ($s_i = 1 - \mu_i$) and c is the number of stages. We assume a linear model:

$$\mathbf{n}(t+1) = \mathbf{SAn}(t), \tag{11}$$

where t indicates time and **n** is a vector of stage-specific abundances (Caswell 2001). The dominant eigenvalue of **SA** (λ) is the asymptotic growth rate of the population with additional anthropogenic mortalities. Using the ranges of anthropogenic mortalities for each threat matrix, we computed the best (λ_B) and worst (λ_W) case scenarios of the dominant eigenvalues for each DPS, using the lower and upper limits of threat levels, respectively.

Finally, we provide the proportion of $\lambda > 1.0$ in percent (P_λ), which is computed by $P_\lambda = 100\times(\lambda_B - 1.0)/(\lambda_B - \lambda_W)$, when $\lambda_B > 1.0$, and $P_\lambda = 0.0$, when $\lambda_B \leq 1.0$. This metric (P_λ) is a function of several components; the assumed maximum population growth rate, assumed life history parameters, thresholds of threat levels, and state of knowledge about precision of the existing threats. Assumed life history parameters and the maximum population growth rate define how much additional mortalities can be tolerated before the dominant eigenvalue becomes < 1.0. Assumed natural survival rates of various life stages can affect the computed proportion; stages with high natural survival rates and low reproductive values can tolerate greater additional mortalities than those with low natural survival rates and high reproductive values. Changes in threshold values of threat matrices also can change the results. If the thresholds for each threat level were decreased, e.g., 0.01 to 0.05 for low, rather than from 0.01 to 0.10 and so forth for other levels, the difference between λ_B and λ_W would decrease and the proportion would increase. Decreasing the uncertainty in additional mortalities would also decrease the difference (and increase the proportion). Consequently, P_λ can be considered as an index of the future population health, where it ranges from 0 to 100, conditional on the specific assumptions. The management goal for each DPS is to maximize P_λ, which may be attained by reducing additional mortalities and increasing the precisions of estimated life history parameters and additional mortalities.

To determine how a "very low" level of threats may affect the computations of the population growth rate, an analysis was conducted with all threat categories set to "very low." This analysis provided the baseline with which the other results can be compared. Because experts' opinions on mortalities were provided either with natural mortality (South Pacific and North Pacific) or without natural mortality, the effect of this difference also was determined with analysis using this hypothetical threat matrix.

4.4. Results

4.4.1. Results—Base Population Model

For plausible ranges of adult survival rates (0.80-0.99/yr), first year survival rates (0.01-0.50/yr), and juvenile survival rates (0.70-0.95/yr), the maximum population growth rates appeared to be less than 10%/yr for all DPSs. We found many combinations of the parameters to be unlikely because of the necessarily high average juvenile survival rates (Figures 11-17).

Except for the Mediterranean Sea DPS (Figure 16), differences among DPSs were negligible. This exception was due to the differences in fertility parameters, where the Mediterranean Sea DPS had low average eggs per clutch (95 eggs per clutch) and low number of clutches per female (2, Table 1).

Parameters that defined habitat use were similar among all DPSs, except the North Pacific Ocean DPS (Table 1). For the North Pacific Ocean DPS, the proportion of time juveniles spend in the neritic habitat (off the coast of Baja Peninsula, Mexico, and in coastal areas of the western Pacific) was less (14%) than for other DPSs (50-65.5%). The difference for this DPS comes from the experts' opinions that a large proportion of the juvenile loggerhead turtles of this DPS remain in the pelagic habitat. Further, those juveniles along the coast of Baja Peninsula are thought to remain in the area for many years.

As expected, the sensitivity analysis indicated the change in the mean AFR can affect the relationship among the survival rates and asymptotic population growth rates (Supplemental Figures S1-S7). For the same value of CV, older age at first reproduction resulted in higher average survival rates of juveniles and adults to sustain an asymptotic population growth rate. Effects of CVs were less than those of the mean AFR. For the following analyses, we use the mean AFR = 30 and CV = 0.17 (or SD of AFR = 5 years).

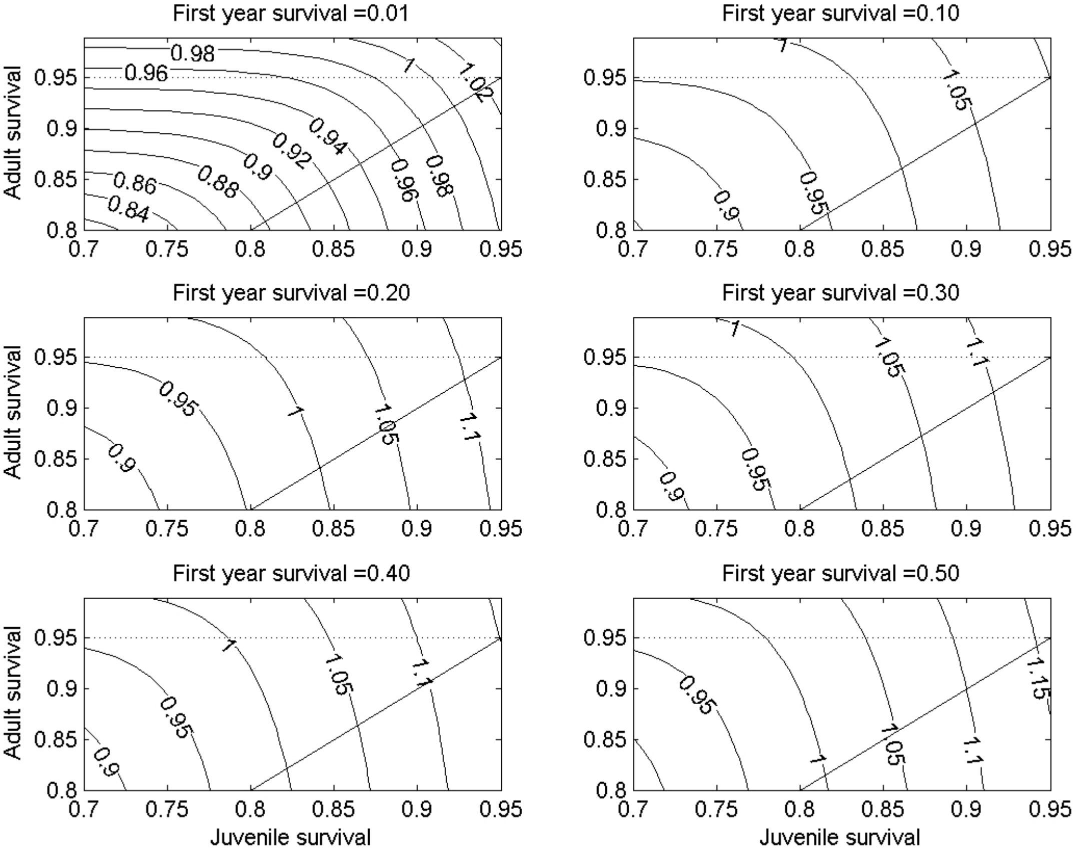


Figure 11. Relationships among adult survival rate, juvenile survival rate, first year survival rate, and the dominant eigenvalue for the North Pacific Ocean DPS. Contour lines indicate the dominant eigenvalues, whereas the diagonal lines indicate equal annual survival rates between juveniles and adults. Dashed horizontal lines indicate the assumed adult survival rate for the base matrix model. The mean age at first reproduction was assumed at 30 and its standard deviation 5.

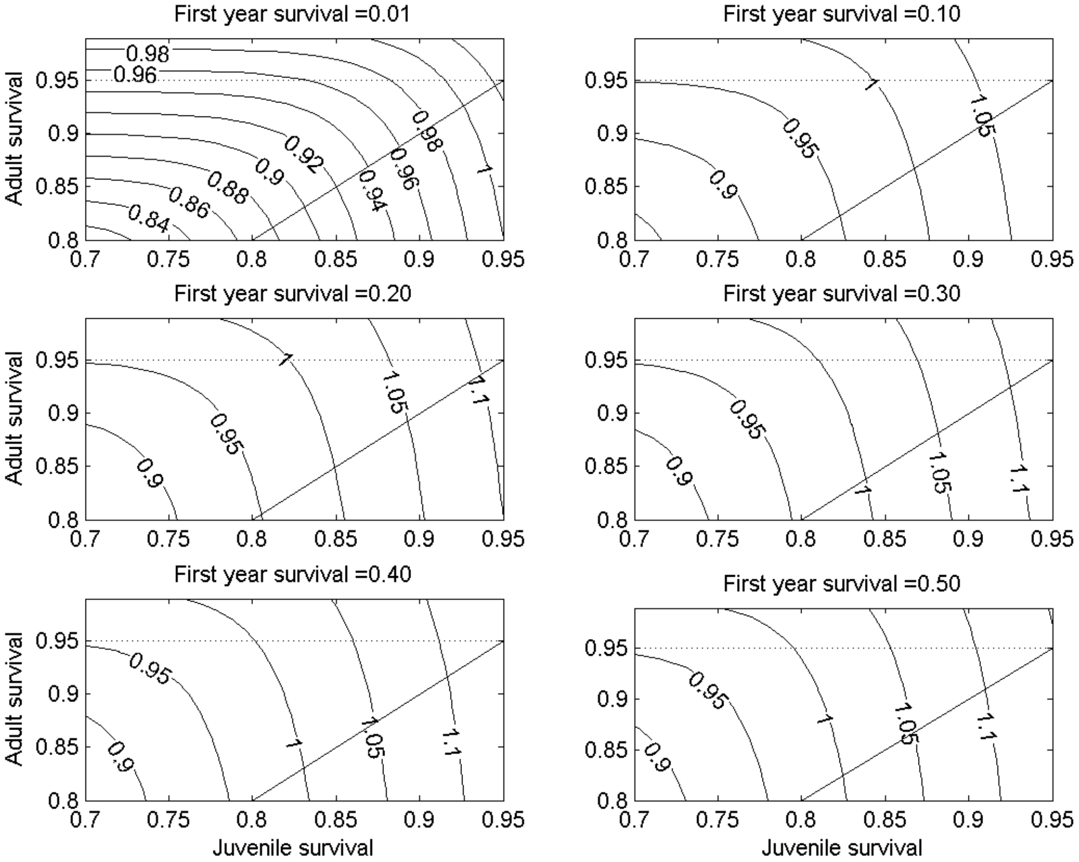
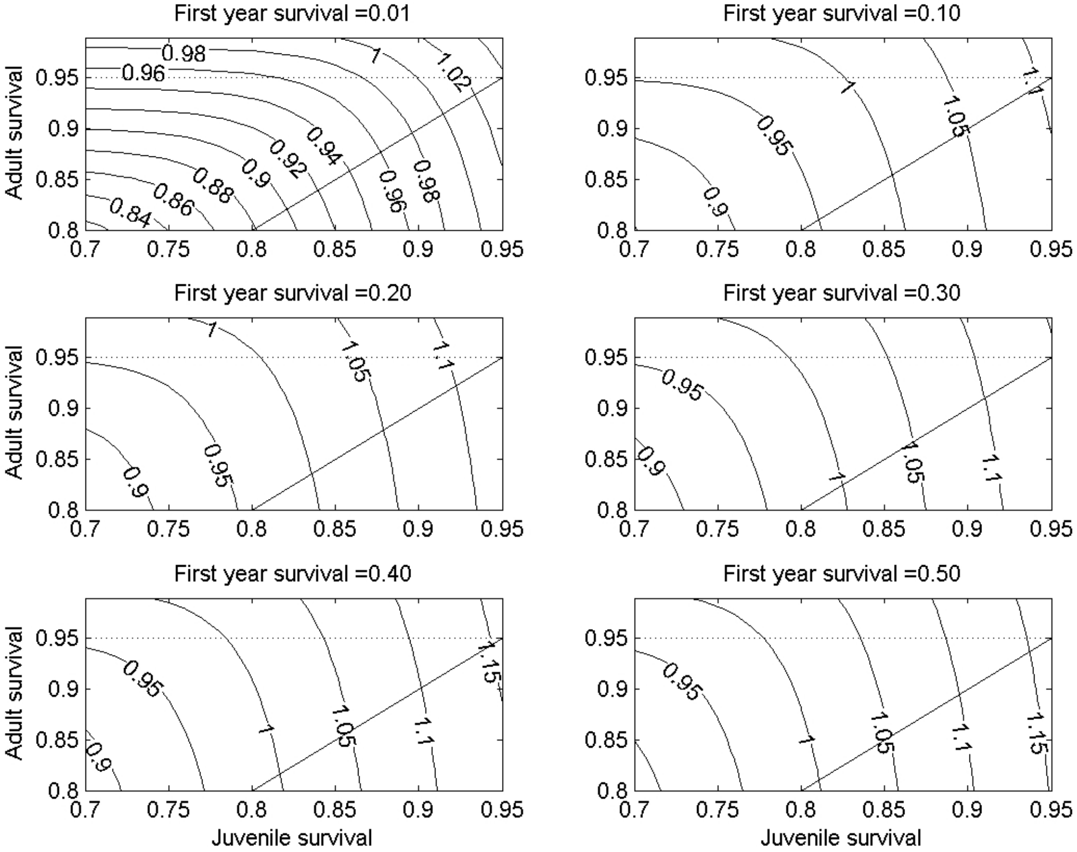


Figure 12. Relationships among adult survival rate, juvenile survival rate, first year survival rate, and the dominant eigenvalue for the South Pacific Ocean DPS. Contour lines indicate the dominant eigenvalues, whereas the diagonal lines indicate equal annual survival rates between juveniles and adults. Dashed horizontal lines indicate the assumed adult survival rate for the base matrix model. The mean age at first reproduction was assumed at 30 and its standard deviation 5.


First year survival =0.01

First year survival =0.10

First year survival =0.20

First year survival =0.30

First year survival =0.40

First year survival =0.50

Figure 13. Relationships among adult survival rate, juvenile survival rate, first year survival rate, and the dominant eigenvalue for the North Indian Ocean, Southeast Indo-Pacific Ocean, and Southwest Indian Ocean DPSs. Contour lines indicate the dominant eigenvalues, whereas the diagonal lines indicate equal annual survival rates between juveniles and adults. Dashed horizontal lines indicate the assumed adult survival rate for the base matrix model. The mean age at first reproduction was assumed at 30 and its standard deviation 5.

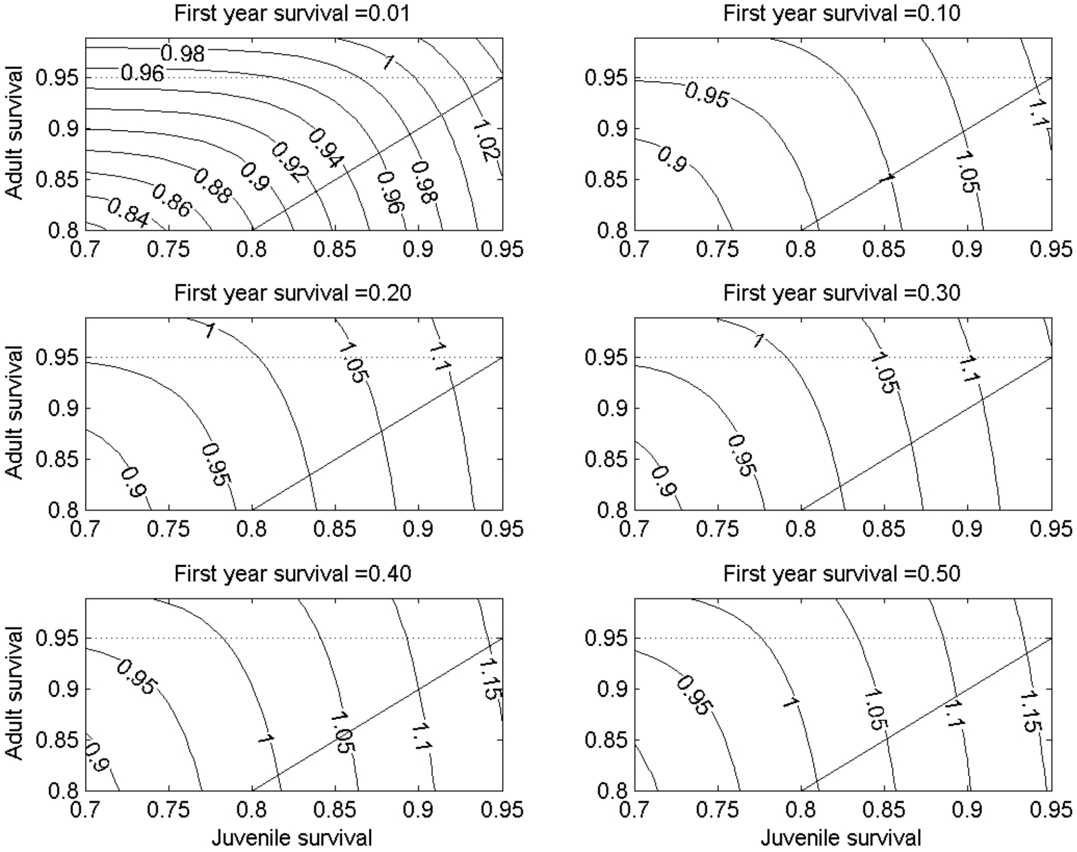


Figure 14. Relationships among adult survival rate, juvenile survival rate, first year survival rate, and the dominant eigenvalue for the Northwest Atlantic Ocean DPS. Contour lines indicate the dominant eigenvalues, whereas the diagonal lines indicate equal annual survival rates between juveniles and adults. Dashed horizontal lines indicate the assumed adult survival rate for the base matrix model. The mean age at first reproduction was assumed at 30 and its standard deviation 5.

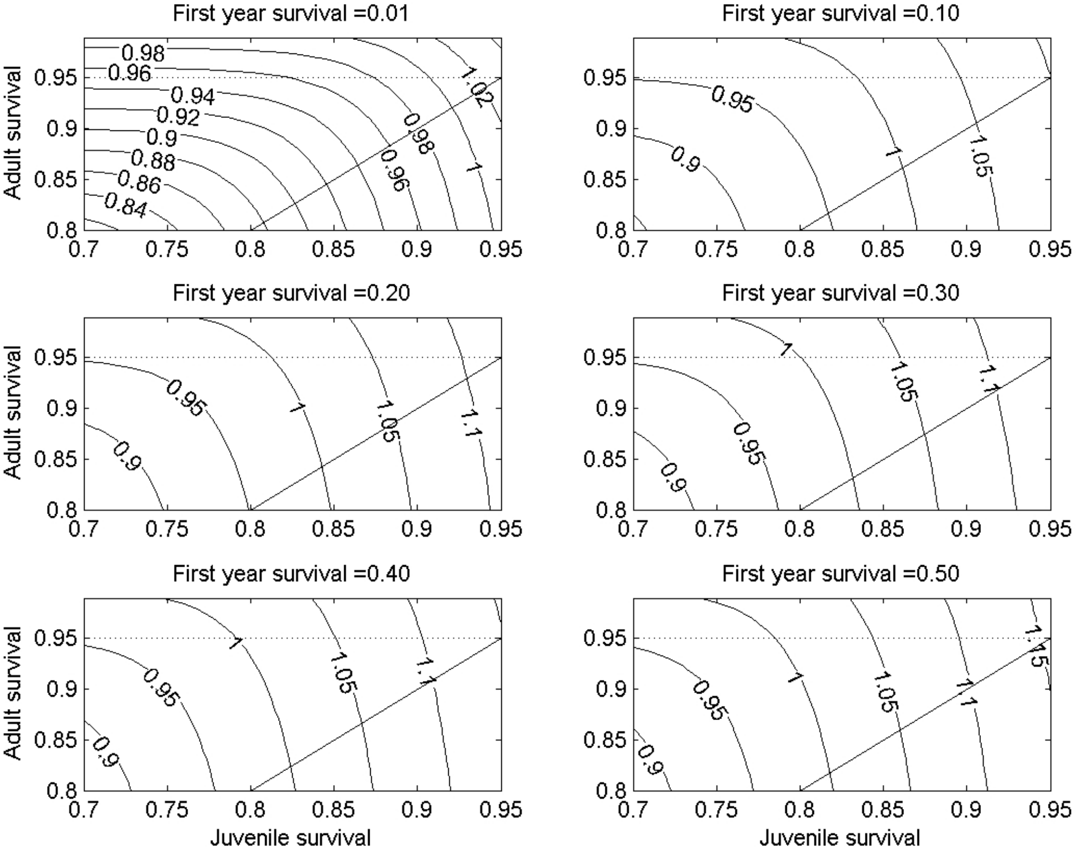


Figure 15. Relationships among adult survival rate, juvenile survival rate, total first year survival rate, and the dominant eigenvalue for the Northeast Atlantic Ocean DPS. Contour lines indicate the dominant eigenvalues, whereas the diagonal lines indicate equal annual survival rates between juveniles and adults. Dashed horizontal lines indicate the assumed adult survival rate for the base matrix model. The mean age at first reproduction was assumed at 30 and its standard deviation 5.

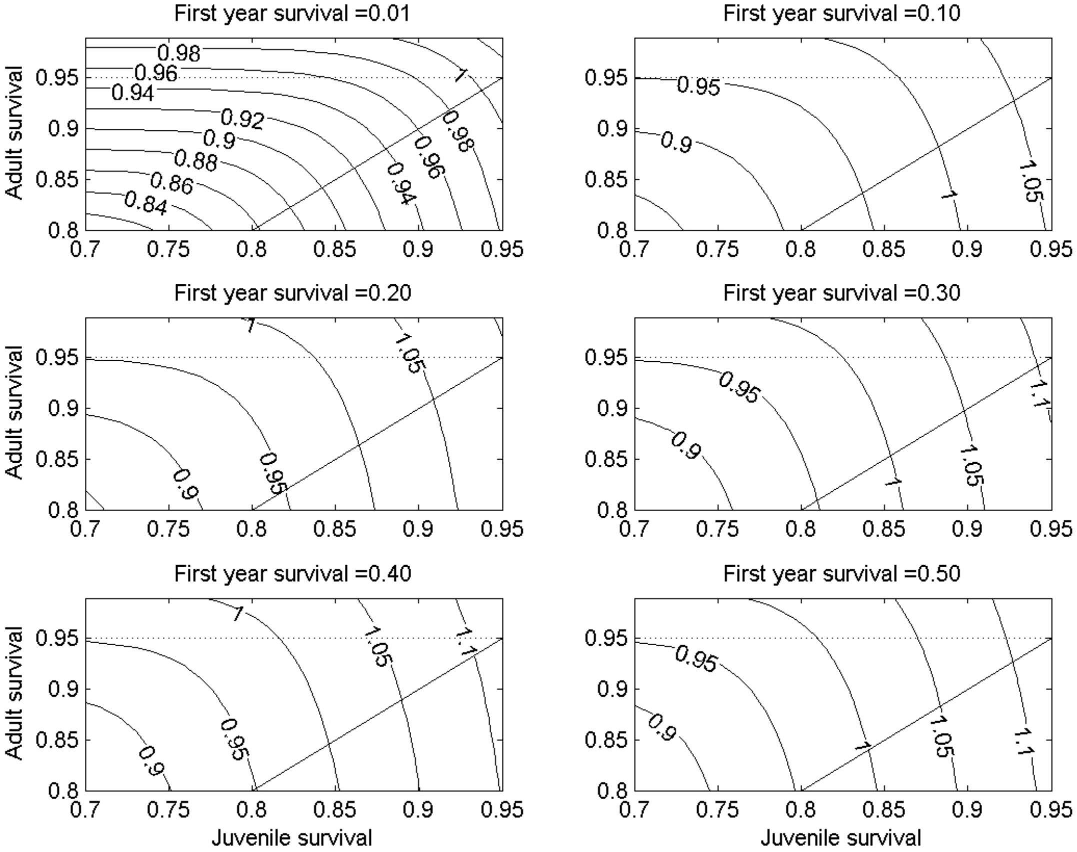


Figure 16. Relationships among adult survival rate, juvenile survival rate, first year survival rate, and the dominant eigenvalue for the Mediterranean Sea DPS. Contour lines indicate the dominant eigenvalues, whereas the diagonal lines indicate equal annual survival rates between juveniles and adults. Dashed horizontal lines indicate the assumed adult survival rate for the base matrix model. The mean age at first reproduction was assumed at 30 and its standard deviation 5.

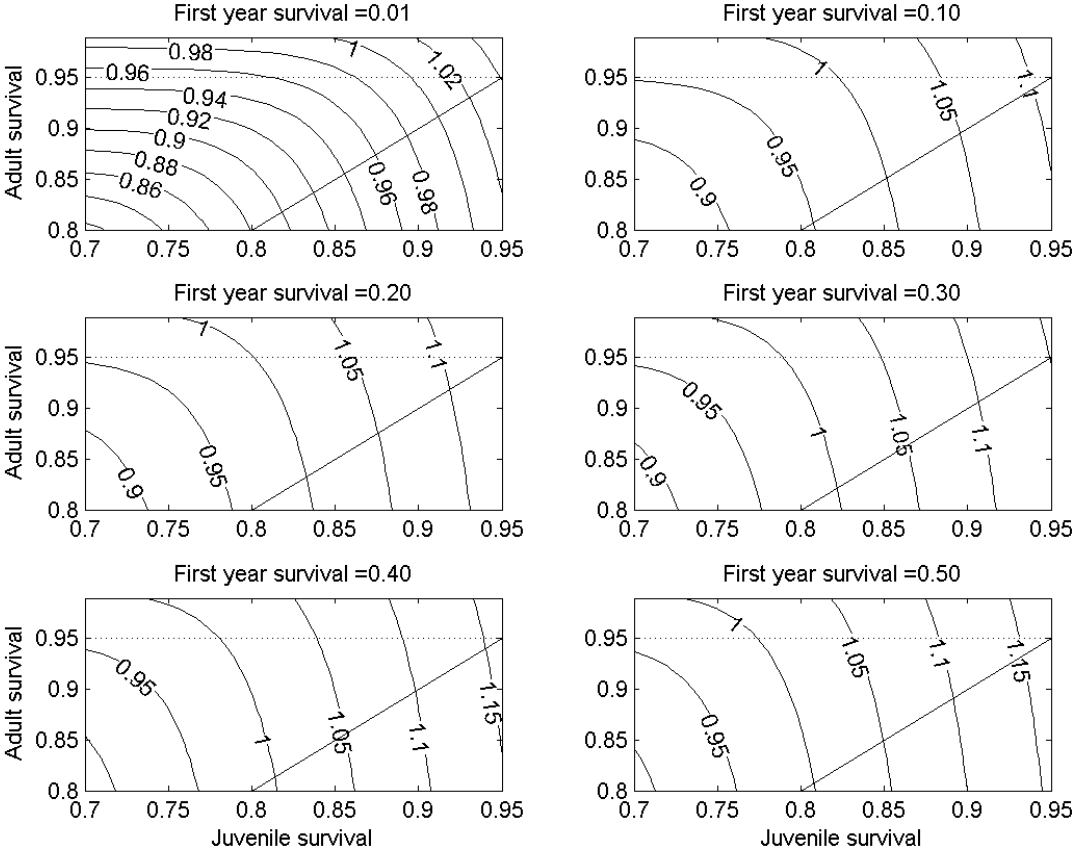


Figure 17. Relationships among adult survival rate, juvenile survival rate, first year survival rate, and the dominant eigenvalue for the South Atlantic Ocean DPS. Contour lines indicate the dominant eigenvalues, whereas the diagonal lines indicate equal annual survival rates between juveniles and adults. Dashed horizontal lines indicate the assumed adult survival rate for the base matrix model. The mean age at first reproduction was assumed at 30 and its standard deviation 5.

4.4.2. Results—Threat Matrix

This analysis indicated potential overestimations by the experts of anthropogenic mortalities (Tables 3-11). The threat matrices indicated that many DPSs might be in critical condition with high mortalities for all stages. One reviewer suggested using a different set of mortality thresholds for building the threat matrices. The suggested change, however, was arbitrary and expresses a false sense of certainty, although our matrices may provide a false sense of urgency. Because our threat levels were determined from available information on anthropogenic mortalities and abundance of loggerhead turtles, we retained the thresholds. The potential implications of the differences in opinions will be discussed later. In the following nine subsections, we provide threat levels for six life stages of each DPS based on available information and experts' opinions. We also provide evidence for each threat. Further, using the threshold values presented previously, we provide numerical values as sums of all threats for each life stage under the four categories.

4.4.2.1. North Pacific Ocean DPS

For the North Pacific Ocean DPS, the greatest anthropogenic threats exist for the neritic juveniles and neritic adults, where the coastal fisheries in Baja California, Mexico, and Japan kill significant number of turtles (Table 3) (Kamezaki *et al*. 2003, Peckham *et al*. 2007). A large number of loggerhead turtles (~1,000 annually) die in Baja bottom-set gillnet/longline fisheries (Koch *et al*. 2006, Peckham *et al*. 2007, Peckham *et al*. 2008). Fishery bycatch also is present in West Pacific neritic foraging areas (Cheng and Chen 1997). Recent investigations show a major threat off coastal Japan due to pound net fisheries (T. Ishihara, Sea Turtle Association of Japan, personal communication, 2007). Coastal development, such as the building of seawalls, erosion control structures, beachfront lighting, vehicular and pedestrian traffic, sand extraction, beach pollution, removal of native vegetation, and planting of non-native vegetation, has resulted in reduction of available habitat for the eggs and hatchlings of this DPS (Suganuma 2002, Kamezaki *et al*. 2003, Kudo *et al*. 2003). Beach debris also contributes to deaths of hatchlings and nesting adults (H. Peckham, ProPeninsula, personal communication, 2009). Although no quantitative studies have been conducted, low hatching success has been documented at important nesting beaches in recent years. Threat levels for this DPS include anthropogenic and natural mortalities.

Table 3. Estimated levels and quantification of anthropogenic threats affecting loggerhead turtles in the North Pacific Ocean DPS. Threat levels were estimated by life stages and habitats, and were grouped into four categories based on four of the five factors in section 4(a)(1) of the ESA. Another factor (inadequacy of existing regulatory mechanisms) was excluded from this analysis because the effects of the factor were not considered to be reducing the survival rates directly. The column with "Best" indicates the total mortality using the lowest threshold values for all factors, whereas the column with "Worst" indicates the total mortality using the highest threshold value for all factors.

	Habitat	Overuse	Disease/ Predation	Other	Best	Worst
Eggs/hatchlings	M	VL	L	M	0.21	0.51
Neritic juveniles	L	L	L	M	0.13	0.50
Oceanic juveniles	VL	L	VL	L	0.02	0.22
Neritic adults	L	L	VL	M	0.12	0.41
Oceanic adults	VL	VL	L	L	0.02	0.22
Nesting females	VL	VL	VL	VL	0.00	0.04

4.4.2.2. South Pacific Ocean DPS

For the South Pacific Ocean DPS, the greatest threats come from bycatch in the oceanic environment for juveniles and adults (Table 4). Threats from numerous fisheries have been reported (Limpus and Reimer 1994; Poiner and Harris 1996; Robins *et al*. 2002a, b; Kelez *et al*. 2003; Alfaro-Shigueto *et al*. 2006; Donoso and Dutton 2006; Alfaro-Shigueto *et al*. 2008b; Limpus 2009). At nesting beaches, an average of 40 adult females are taken annually in aboriginal harvest (Limpus 2009). Threat levels for this DPS include anthropogenic and natural mortalities.

Table 4. Estimated levels and quantification of anthropogenic threats affecting loggerhead turtles in the South Pacific Ocean DPS. Threat levels were estimated by life stages and habitats, and were grouped into four categories based on four of the five factors in section 4(a)(1) of the ESA. Another factor (inadequacy of existing regulatory mechanisms) was excluded from this analysis because the effects of the factor were not considered to be reducing the survival rates directly. The column with "Best" indicates the total mortality using the lowest threshold values for all factors, whereas the column with "Worst" indicates the total mortality using the highest threshold value for all factors.

	Habitat	Overuse	Disease/ Predation	Other	Best	Worst
Eggs/hatchlings	L	L	L	L	0.04	0.40
Neritic juveniles	L	L	L	L	0.04	0.40
Oceanic juveniles	VL	VL	VL	M	0.10	0.23
Neritic adults	L	L	L	L	0.04	0.40
Oceanic adults	VL	VL	L	M	0.11	0.32
Nesting females	VL	M	VL	VL	0.10	0.23

4.4.2.3. North Indian Ocean DPS

Threat levels for the North Indian Ocean DPS were “very low” to “low” for all factors except for the fourth factor (“other”) in the neritic environment (Table 5). The relatively high mortalities in the neritic environment result from fishery bycatch. Consistent low threats across all factors for eggs and hatchlings may affect a significant portion of this stage. These threats include coastal development and light pollution, beach habitat degradation from overgrazing and beach driving, and bycatch from coastal fisheries.

Table 5. Estimated levels and quantification of anthropogenic threats affecting loggerhead turtles in the North Indian Ocean DPS. Threat levels were estimated by life stages and habitats, and were grouped into four categories based on four of the five factors in section 4(a)(1) of the ESA. Another factor (inadequacy of existing regulatory mechanisms) was excluded from this analysis because the effects of the factor were not considered to be reducing the survival rates directly. The column with “Best” indicates the total mortality using the lowest threshold values for all factors, whereas the column with “Worst” indicates the total mortality using the highest threshold value for all factors.

	Habitat	Overuse	Disease/ Predation	Other	Best	Worst
Eggs/hatchlings	L	L	L	L	0.04	0.40
Neritic juveniles	L	VL/L	L	M	0.12	0.41
Oceanic juveniles	VL	VL	VL	L	0.01	0.13
Neritic adults	L	VL/L	L	M	0.12	0.41
Oceanic adults	VL	VL	VL	L	0.01	0.13
Nesting females	L	VL	VL	L	0.02	0.22

4.4.2.4. Southeast Indo-Pacific Ocean DPS

Anthropogenic effects are very low to low at all life stages for the Southeast Indo-Pacific Ocean DPS (Table 6). Cumulatively, however, a large proportion of eggs and hatchlings (~30%) may be affected by human activities.

Table 6. Estimated levels and quantification of anthropogenic threats affecting loggerhead turtles in the Southeast Indo-Pacific Ocean DPS. Threat levels were estimated by life stages and habitats, and were grouped into four categories based on four of the five factors in section 4(a)(1) of the ESA. Another factor (inadequacy of existing regulatory mechanisms) was excluded from this analysis because the effects of the factor were not considered to be reducing the survival rates directly. The column with "Best" indicates the total mortality using the lowest threshold values for all factors, whereas the column with "Worst" indicates the total mortality using the highest threshold value for all factors.

	Habitat	Overuse	Disease/ Predation	Other	Best	Worst
Eggs/hatchlings	VL/L	VL	L	L	0.02	0.31
Neritic juveniles	L	VL	VL	L	0.02	0.22
Oceanic juveniles	VL	VL	VL	L	0.01	0.13
Neritic adults	L	VL	VL	L	0.02	0.22
Oceanic adults	VL	VL	VL	L	0.01	0.13
Nesting females	VL	VL	VL	VL	0.00	0.04

4.4.2.5. Southwest Indian Ocean DPS

For the Southwest Indian Ocean DPS, wide ranges of total mortalities were found for eggs/hatchlings, neritic juveniles, and neritic adults (Table 7). These wide ranges result from conservative estimation based on limited information on anthropogenic mortalities.

Table 7. Estimated levels and quantification of anthropogenic threats affecting loggerhead turtles in the Southwest Indian Ocean DPS. Threat levels were estimated by life stages and habitats, and were grouped into four categories based on four of the five factors in section 4(a)(1) of the ESA. Another factor (inadequacy of existing regulatory mechanisms) was excluded from this analysis because the effects of the factor were not considered to be reducing the survival rates directly. The column with "Best" indicates the total mortality using the lowest threshold values for all factors, whereas the column with "Worst" indicates the total mortality using the highest threshold value for all factors.

	Habitat	Overuse	Disease/ Predation	Other	Best	Worst
Eggs/hatchlings	VL/L	L	L	L	0.03	0.40
Neritic juveniles	L	VL/L	VL	L/M	0.02	0.40
Oceanic juveniles	VL	VL	VL	L	0.01	0.13
Neritic adults	L	VL/L	VL	L/M	0.02	0.41
Oceanic adults	VL	VL	VL	L	0.01	0.13
Nesting females	VL	VL	VL	VL	0.00	0.04

4.4.2.6. Northwest Atlantic Ocean DPS

High cumulative threat levels were estimated for neritic and oceanic juveniles and neritic and oceanic adults (Table 8). These mortalities primarily result from fishery bycatch. Significant mortality occurs in longline fisheries, bottom and mid-water trawl fisheries, dredge fisheries, gillnet fisheries, and pot/trap fisheries. Although total mortality from all fisheries has not been estimated, the combined mortalities are likely significant. Entanglement in marine debris is an additional threat. Further, boat strikes are another growing anthropogenic source of mortality in neritic waters (NMFS, unpublished data; Florida Fish and Wildlife Conservation Commission, unpublished data).

Table 8. Estimated levels and quantification of anthropogenic threats affecting loggerhead turtles in the Northwest Atlantic Ocean DPS. Threat levels were estimated by life stages and habitats, and were grouped into four categories based on four of the five factors in section 4(a)(1) of the ESA. Another factor (inadequacy of existing regulatory mechanisms) was excluded from this analysis because the effects of the factor were not considered to be reducing the survival rates directly. The column with “Best” indicates the total mortality using the lowest threshold values for all factors, whereas the column with “Worst” indicates the total mortality using the highest threshold value for all factors.

	Habitat	Overuse	Disease/ Predation	Other	Best	Worst
Eggs/hatchlings	L	VL	L	L	0.03	0.31
Neritic juveniles	L	L	L	M	0.13	0.50
Oceanic juveniles	VL	VL	VL	M/H	0.10	0.28
Neritic adults	L	L	L	M	0.13	0.50
Oceanic adults	VL	VL	VL	M/H	0.10	0.28
Nesting females	VL	VL	VL	VL	0.00	0.04

4.4.2.7. Northeast Atlantic Ocean DPS

For the Northeast Atlantic Ocean DPS, relatively high threats are reported for terrestrial, neritic, and oceanic habitats (Table 9). The largest source of anthropogenic mortality is deliberate hunting of nesting females. Other threats include harvest of eggs and hatchlings, incidental capture, particularly in fisheries, propeller and boat strikes, exposure to hydrocarbons, marine debris ingestion, and bioaccumulation of metals (see Section 5.2.7.). Sand mining has impacted the terrestrial habitats in some areas and changes in trophic structure likely have occurred in both the neritic and oceanic environments of the region. Shifts in ocean currents, and ecosystem shifts in prey distribution and abundance, could occur with climate change.

Table 9. Estimated levels and quantification of anthropogenic threats affecting loggerhead turtles in the Northeast Atlantic Ocean DPS. Threat levels were estimated by life stages and habitats, and were grouped into four categories based on four of the five factors in section 4(a)(1) of the ESA. Another factor (inadequacy of existing regulatory mechanisms) was excluded from this analysis because the effects of the factor were not considered to be reducing the survival rates directly. The column with “Best” indicates the total mortality using the lowest threshold values for all factors, whereas the column with “Worst” indicates the total mortality using the highest threshold value for all factors.

	Habitat	Overuse	Disease/ Predation	Other	Best	Worst
Eggs/hatchlings	L	L	VL	L	0.03	0.31
Neritic juveniles	L	L	VL	M	0.12	0.41
Oceanic juveniles	VL	VL	VL	M/H	0.10	0.28
Neritic adults	L	L	VL	M	0.12	0.41
Oceanic adults	VL	VL	VL	M/H	0.10	0.28
Nesting females	VL	H	VL	L	0.21	0.37

4.4.2.8. Mediterranean Sea DPS

Fishery bycatch was determined to be the greatest threat for the Mediterranean Sea DPS (Table 10). For neritic and oceanic juveniles and adults, these threats were considered to be “medium” or “high.” Bycatch of juveniles and adults in trawl, pelagic and bottom longline, set net (gillnet and trammel nets), and driftnet fisheries is considered significant (Laurent 1991, Argano *et al*. 1992, Di Natale 1995, Lazar and Tvrtkovic 1995, Laurent *et al*. 1996, Camiñas 1997, Gerosa and Casale 1999, Silvani *et al*. 1999, Lazar *et al*. 2000, Laurent *et al*. 2001, Camiñas 2004, Carreras *et al*. 2004, Casale *et al*. 2004, Tudela *et al*. 2005, Casale *et al*. 2007a, Jribi *et al*. 2007, Casale 2008). See Section 5.2.8.5. for capture levels/citations. Other notable concerns are pollution/debris and vessel collisions. The other significant threat comes from habitat destruction that affects eggs and hatchlings at nesting beaches.

Table 10. Estimated levels and quantification of anthropogenic threats affecting loggerhead turtles in the Mediterranean Sea DPS. Threat levels were estimated by life stages and habitats, and were grouped into four categories based on four of the five factors in section 4(a)(1) of the ESA. Another factor (inadequacy of existing regulatory mechanisms) was excluded from this analysis because the effects of the factor were not considered to be reducing the survival rates directly. The column with “Best” indicates the total mortality using the lowest threshold values for all factors, whereas the column with “Worst” indicates the total mortality using the highest threshold value for all factors.

	Habitat	Overuse	Disease/ Predation	Other	Best	Worst
Eggs/hatchlings	M	L	L	L	0.13	0.50
Neritic juveniles	L	L	VL	M/H	0.12	0.46
Oceanic juveniles	VL	L	VL	M/H	0.11	0.37
Neritic adults	L	L	L	M/H	0.13	0.55
Oceanic adults	VL	L	VL	M/H	0.11	0.37
Nesting females	VL	L	VL	VL	0.01	0.13

4.4.2.9. South Atlantic Ocean DPS

Significant anthropogenic threats exist for juvenile stages of the South Atlantic Ocean DPS (Table 11). For these stages, the most significant factor is fishing bycatch, where coastal and oceanic fishery operations interact with loggerheads in the area (Bugoni *et al*. 2008). Further, bycatch in longline fisheries appear to have significant effects on juvenile loggerhead turtles in the high seas (Pinedo and Polacheck 2004, Kotas *et al*. 2004).

Table 11. Estimated levels and quantification of anthropogenic threats affecting loggerhead turtles in the South Atlantic Ocean DPS. Threat levels were estimated by life stages and habitats, and were grouped into four categories based on four of the five factors in section 4(a)(1) of the ESA. Another factor (inadequacy of existing regulatory mechanisms) was excluded from this analysis because the effects of the factor were not considered to be reducing the survival rates directly. The column with "Best" indicates the total mortality using the lowest threshold values for all factors, whereas the column with "Worst" indicates the total mortality using the highest threshold value for all factors.

	Habitat	Overuse	Disease/ Predation	Other	Best	Worst
Eggs/hatchlings	L	L	VL	L	0.03	0.31
Neritic juveniles	L	VL	L	M	0.12	0.41
Oceanic juveniles	VL	VL	L	H	0.21	0.37
Neritic adults	L	VL	L	L	0.03	0.31
Oceanic adults	VL	VL	L	L	0.02	0.22
Nesting females	VL	VL	VL	VL	0.00	0.04

Although the experts were asked to provide qualitative threat levels (very low, low, medium, or high) based on threshold values for each level (see Section 4.3.3 for details), it was sometimes a difficult task to assign an anthropogenic mortality level to a particular stage class when few or no quantitative data existed. Because precise estimates of anthropogenic mortalities do not exist, we used the precautionary principle for characterizing the threat level based on best available information and/or expert opinion.

Even though threats were expressed in a qualitative manner, every entry was derived from available data and experts' knowledge. The compilation of known threats for all DPSs provided an opportunity to evaluate the potential negative impacts of these threats on each DPS in the future. Regardless of the threshold values for each threat level (per suggestion of a reviewer), comparisons among life stages within a DPS, as well as comparisons of a life stage among DPSs, are meaningful. For example, the North Pacific Ocean and Mediterranean Sea DPSs were shown to have high anthropogenic mortalities for the eggs and hatchling stage, relative to other DPSs. Within the North Pacific Ocean DPS, for another example, neritic habitats (along the west coast of Baja California, Mexico, and the nearshore waters of Japan) indicated greater anthropogenic mortalities than other life stages. Finally, the indices of DPS health (λ and P_{λ}) also provided metrics that combined the status and uncertainty of threats levels for each DPS. Interpretations of these indices in terms of the DPS status are explained in the following section.

4.4.3. DPS Status

The analysis with a hypothetical threat matrix with all threat categories set to "very low" indicated the obvious effects of combining anthropogenic and natural mortalities in a threat matrix. For those threat matrices of DPSs that included natural mortalities (North Pacific and South Pacific), computed anthropogenic mortalities were necessarily less than the assumed thresholds. Consequently, ranges of possible λ values for those DPSs were less than those DPSs that included only anthropogenic mortalities in their threat matrices (Figures 18 and 19). Even when all threat levels were set at "very low," P_{λ} was not 100% for the DPSs that included only anthropogenic mortalities in the threat matrices and the assumed maximum population growth rate was 5% annum (Figure 18). "Very low," therefore, may not have been an appropriate term for this category. As it was stated in the method section, management decisions should focus on maximizing P_{λ}.

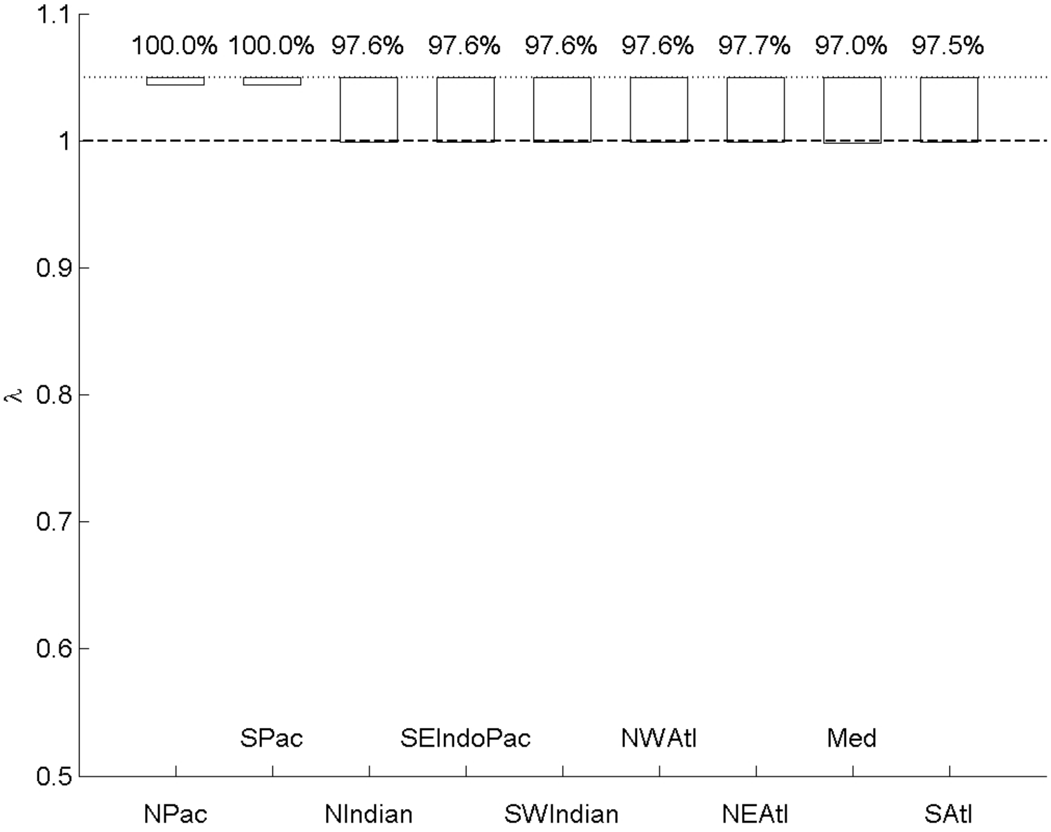


Figure 18. Ranges of dominant eigenvalues for nine DPSs when anthropogenic mortalities were considered to be "very low" for all factors and the maximum natural population growth rates were assumed to be 5% per year ($\lambda_0 = 1.05$). The mean age at first reproduction (AFR) was assumed to be 30 years and its standard deviation (SD) 5. The values above the dotted line indicates the proportion of the bars that are above $\lambda = 1.0$ (P_λ)

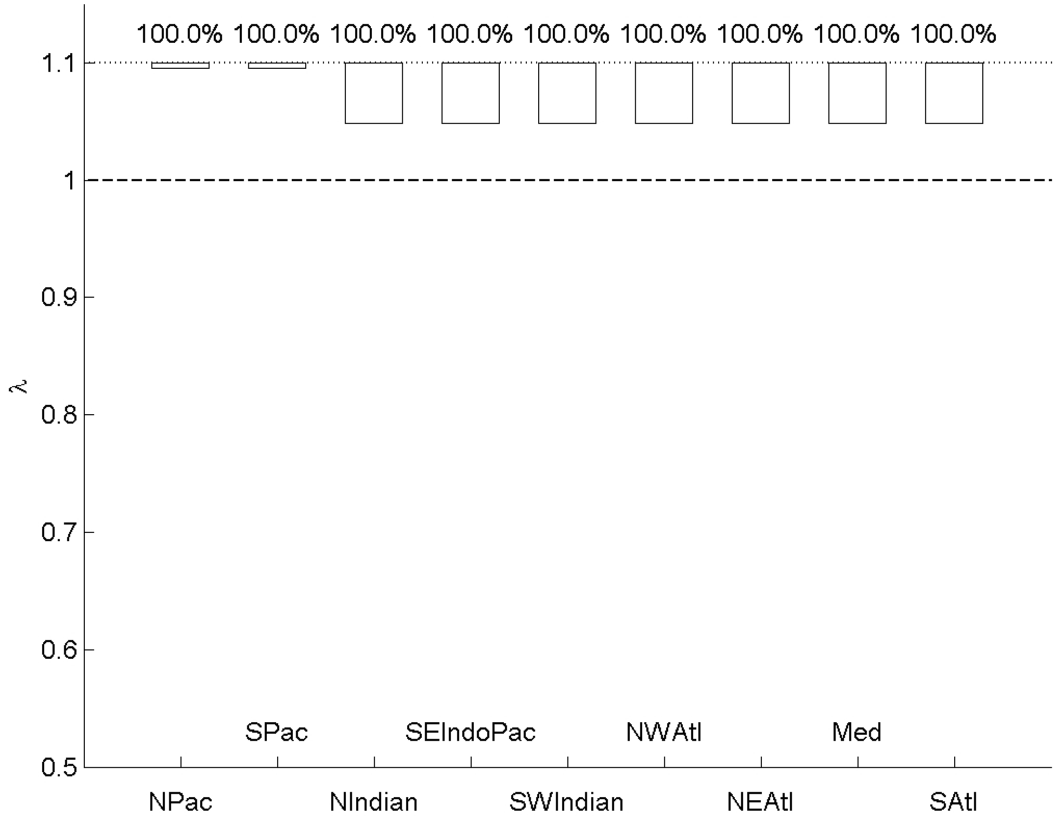


Figure 19. Ranges of dominant eigenvalues for nine DPSs when anthropogenic mortalities were considered to be "very low" for all factors and the maximum natural population growth rates were assumed to be 10% per year ($\lambda_0 = 1.10$). The mean age at first reproduction (AFR) was assumed to be 30 years and its standard deviation (SD) 5. The values above the dotted line indicates the proportion of the bars that are above λ = 1.0 (P_λ).

For computing P_λ, the BRT did not consider functional forms of threat levels. In other words, all values between the best and worst case scenarios were considered equally likely. This assumption was fundamental for computing the proportion of $\lambda > 1.0$. With more information on threat levels, it may be possible to assign particular distributions to these threat levels, thereby increasing the precision and accuracy of P_λ.

Among the nine DPSs in this analysis, an index of future population health (P_λ = the proportion of $\lambda > 1.0$) was 0 for North Indian Ocean, Northwest Atlantic Ocean, Northeast Atlantic Ocean, Mediterranean Sea, and South Atlantic Ocean DPSs, when the maximum population growth rate was assumed to be 5% per year (Figure 20). Even if the maximum population growth rate was 10% per year, P_λ for the Northwest Atlantic Ocean, Northeast Atlantic Ocean, Mediterranean Sea, and South Atlantic Ocean DPSs was zero (Figure 21). When the maximum population growth rate was assumed to be 5% per year (Figure 20), P_λ ranged from 0.0% to 16.9% (North Pacific Ocean DPS). The large value for the North Pacific Ocean DPS is caused by precise estimates of threats for the DPS, where the length of the bar is less than for the other DPSs. For those DPSs that had $P_\lambda = 0$ when the maximum population growth rate was assumed to be 5% per year, only the North Indian Ocean DPS resulted in $P_\lambda > 0$ when the maximum population growth rate was assumed to be 10% per year. P_λ ranged from 0.0% to 40.4% (Southeast Indo-Pacific Ocean DPS), when the maximum population growth rate was assumed to be 10% per year (Figure 21). According to P_λ, all DPSs are faced with a potential decline in the future due to the existing additional mortalities, relative to the assumed natural survival rates and fertility.

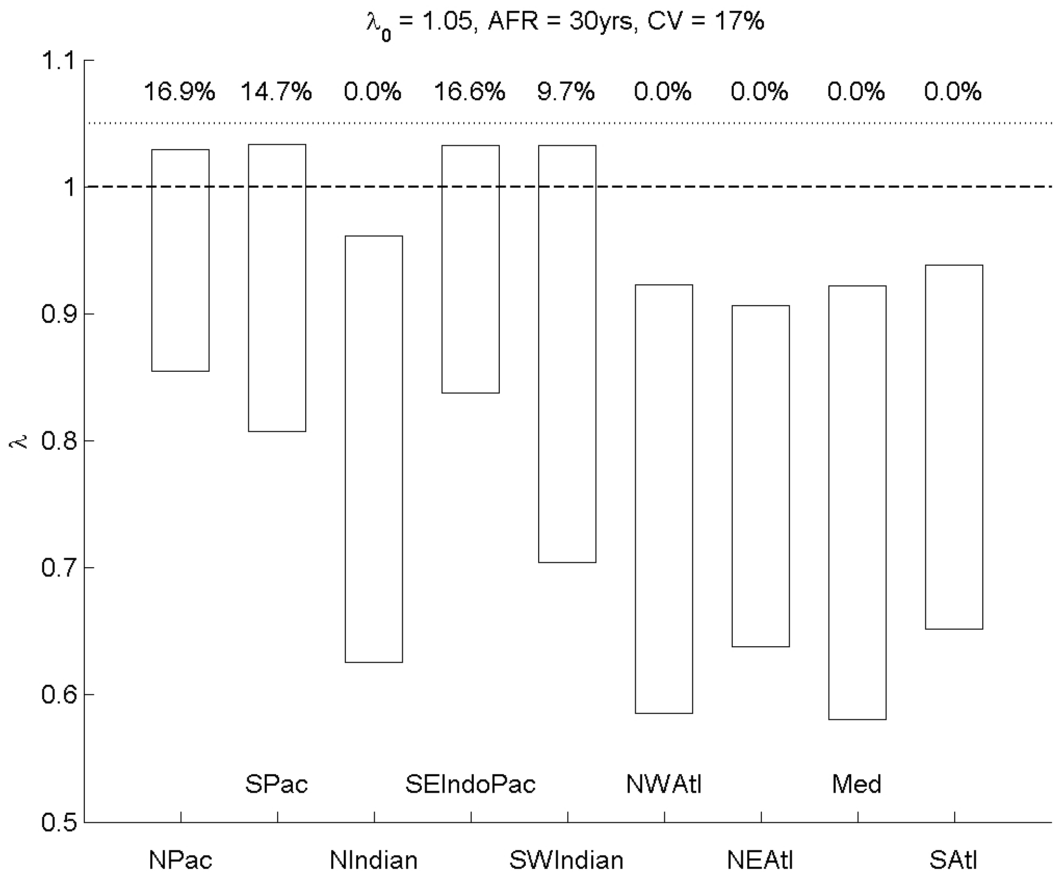


Figure 20. Ranges of dominant eigenvalues for nine DPSs with anthropogenic mortalities that were provided by the experts. For this figure, the maximum population growth rates for all DPSs were assumed to be 5% per year (λ_0 = 1.05). The mean age at first reproduction (AFR) was assumed to be 30 years and its standard deviation (SD) 5. The values above the dotted line indicates the proportion of the bars that are above λ = 1.0 (P_λ).

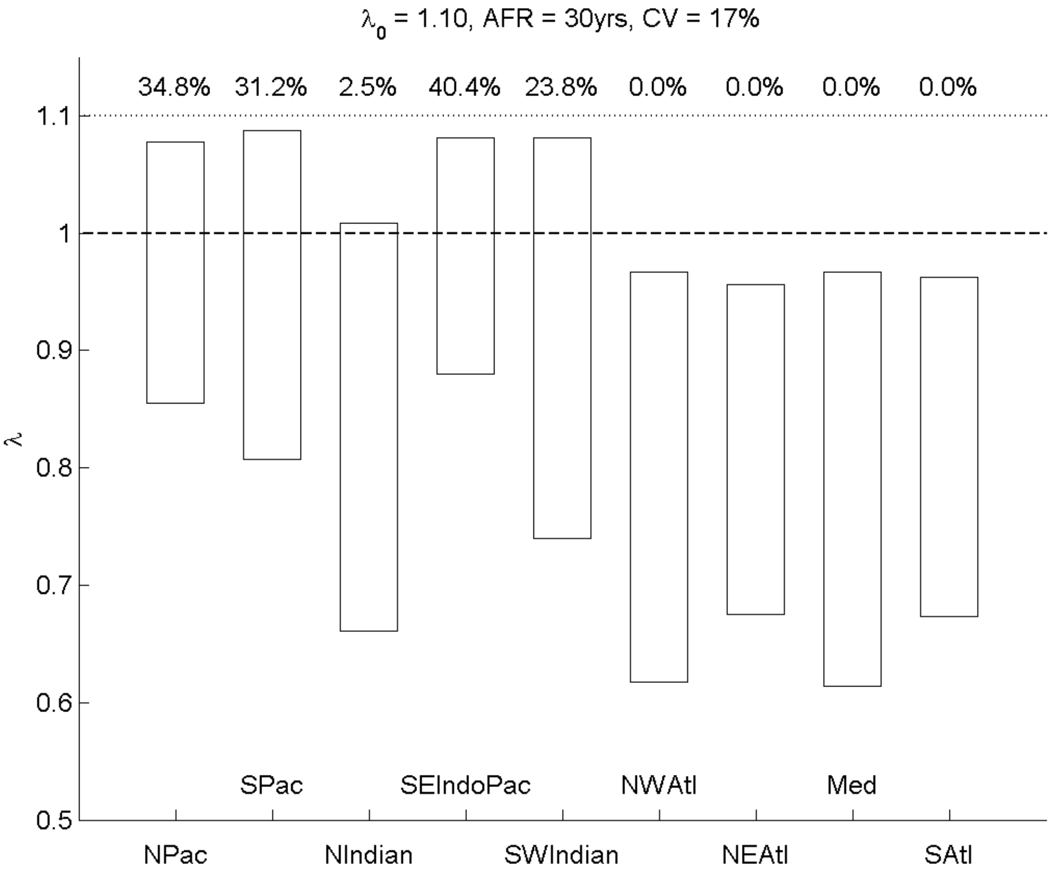


Figure 21. Ranges of dominant eigenvalues for nine DPSs with anthropogenic mortalities that were provided by the experts. For this figure, the maximum population growth rates for all DPSs were assumed to be 10% per year ($\lambda_0 = 1.10$). The mean age at first reproduction (AFR) was assumed to be 30 years and its standard deviation (SD) 5. The values above the dotted line indicates the proportion of the bars that are above λ = 1.0 (P_λ).

4.5. Synthesis

For three of five DPSs with sufficient data to conduct the SQE analysis (Northwest Atlantic Ocean, South Pacific Ocean, and North Pacific Ocean), the threshold of SQE = 0.3 was reached at QET < 0.3, indicating high likelihood of quasi-extinction over a wide range of QET values. Similarly, the threat matrix analysis indicated that all other DPSs have the potential for a severe decline in the future (Figures 20 and 21). There were not enough data to conduct the SQE analysis for the Northeast Atlantic Ocean, Mediterranean Sea, North Indian Ocean, and Southeast Indo-Pacific Ocean DPSs.

It should be noted that the SQE analysis is based on nesting beach trend data. Trends at nesting beaches are often used to assess the status of marine turtle populations, because nesting beach studies are usually more easily implemented than in-water studies and, in many cases, have been ongoing for many years. However, because of the protracted age to maturity in marine turtles, mortality that is acting on younger life stages will not yet be reflected in nest or nesting female trends, and may take many years to be detected at the nesting beach. Conversely, mortality acting on older life stages, including adults, will be reflected more rapidly in nest or nesting female trends at nesting beaches. The SQE approach does not incorporate different mortalities among life stages. The threat matrix approach was used to determine effects of known anthropogenic mortalities on each DPS.

According to the threat matrix analysis, using experts' opinions in the matrix model framework, all loggerhead turtle DPSs have the potential to decline in the future. Although some DPSs are indicating increasing trends at nesting beaches (Southwest Indian Ocean and South Atlantic Ocean), available information about anthropogenic threats to juveniles and adults in neritic and oceanic environments indicate possible unsustainable additional mortalities (Figures 20 and 21). According to the threat matrix analysis, the potential for future decline is greatest for the North Indian Ocean, Northwest Atlantic Ocean, Northeast Atlantic Ocean, Mediterranean Sea, and South Atlantic Ocean DPSs. We emphasize, however, that these indices were used to measure the negative effects of known anthropogenic mortalities on the overall health of each DPS and not to estimate the actual population growth rates of these DPSs.

Our approach to the risk analysis presented several important points. First, the lack of precise estimates of age at first reproduction hindered precise assessment of the status of any DPS. Within the range of possible ages at first reproduction of the species, however, some DPSs could decline rapidly regardless of the exact age at first reproduction because of high anthropogenic mortality.

Second, the lack of precise estimates of anthropogenic mortalities resulted in a wide range of possible status. For the best case scenario, a DPS may be healthy, whereas the same DPS may be considered as declining rapidly for the worst case scenario (Figures 20 and 21). The precise prognosis of each DPS relies on obtaining precise estimate of anthropogenic mortality and vital rates.

Third, the assessment of a population without the information on natural and anthropogenic mortalities is difficult. Because of the longevity of the species, loggerhead turtles require high survival rates throughout their life to maintain a population. Anthropogenic mortality on the species occurs at every stage of their life, where the exact magnitude of the mortality is often unknown. As described in this document, the upper end of natural mortality can be computed from available information. In the future, as more comprehensive and precise information on anthropogenic threats is gathered and available, improvements in model outcomes will result.

Nesting beach count data for the North Pacific Ocean DPS have indicated a decline of loggerhead turtle nesting in the last 20 years (Figure 1). The SQE approach reflected the observed decline (Figure 6). However, in the threat matrix analysis, the range of λ values contains 1.0, indicating the large uncertainty about the future of the DPS. As stated elsewhere, fishery bycatch along the coast of the Baja Peninsula and the nearshore waters of Japan are the main known sources of mortalities (Table 3). Mortalities in the high-seas, where a large number of juvenile loggerhead turtles reside (Kobayashi *et al*. 2008), from fishery bycatch are still unknown.

The SQE approach indicated that, based on nest count data for the past three decades, the South Pacific Ocean DPS is at risk and thus likely to decline in the foreseeable future. These results were based on recently published nesting census data for loggerhead turtles at index beaches in eastern Australia (Limpus 2009). The threat matrix approach provided uncertain results: in the case of the lowest anthropogenic threats, the South Pacific Ocean DPS may recover, but in the worst-case scenario, the DPS may substantially decline in the foreseeable future. These results are largely driven by the ongoing threats to juvenile and adult loggerheads from fishery bycatch that occur throughout the South Pacific Ocean and the uncertainty in estimated mortalities (Table 4).

For the North Indian Ocean DPS, there were no nesting beach data available to conduct the SQE analysis. The threat matrix analysis indicated a decline of the DPS in the foreseeable future, primarily as a result of fishery bycatch in neritic habitats. Cumulatively, substantial threats may exist for eggs/hatchlings (Table 5). Because of the lack of precise estimates of bycatch, however, the range of possible λ values was large (Figures 20 and 21).

Similar to the North Indian Ocean DPS, no nesting beach data were available for the Southeast Indo-Pacific Ocean DPS. The level of anthropogenic mortalities is low for the Southeast Indo-Pacific Ocean DPS, based on the best available information, resulting in relatively large P_λ values and a narrow range (Figures 20 and 21). The greatest threats for the Southeast Indo-Pacific Ocean DPS exist for the first year of the life stages (eggs and hatchlings; Table 6). For the Southwest Indian Ocean DPS, the SQE approach, based on a 37-year time series of nesting female counts at Tongaland, South Africa (1963-1999), indicated this segment of the population, while small, has increased, and the likelihood of quasi-extinction is negligible. The threat matrix approach, on the other hand, provided a wide range of results: in the best case scenario, the DPS would grow slowly, whereas in the worst case scenario, the DPS would decline in the foreseeable future. The results of the threat matrix approach were driven by uncertainty in anthropogenic mortalities in the neritic environment and the eggs/hatchling stage (Table 7, Figures 20 and 21).

The SQE analysis indicated differences in SQEs among recovery units within the Northwest Atlantic Ocean DPS (Figure 8). Although the Northern Gulf of Mexico Recovery Unit indicated the worst result among the four recovery units under consideration (Figure 8), the length of the time series was shortest (12 data points; Figure 4). Other recovery units, however, appeared to show similar declining trends, which were also indicated through the SQE approach. The threat matrix analysis indicated a likely decline of the DPS in the foreseeable future. The greatest threats to the DPS result from cumulative fishery bycatch in neritic and oceanic habitats (Table 8).

The high likelihood of the predicted decline of the Northeast Atlantic Ocean DPS from the threat matrix analysis is largely driven by the ongoing harvest of nesting females, low hatchling and emergence success, and mortality of juveniles and adults from fishery bycatch throughout the Northeast Atlantic Ocean (Table 9). The threat matrix analysis indicated a consistently pessimistic future for the DPS (Figures 20 and 21).

Nesting beach data for the Mediterranean Sea DPS were not available to conduct the SQE analysis. The threat matrix analysis indicated the DPS is likely to decline in the foreseeable future (Figures 20 and 21). The primary threats are fishery bycatch in neritic and oceanic habitats (Table 10).

The two approaches for determining risks to the South Atlantic Ocean DPS provided different, although not incompatible, results. The SQE approach indicated that, based on nest count data for the past two decades, the population was unlikely to decline in the foreseeable future (Figure 10). These results were based on recently published nesting beach trend analyses by Marcovaldi and Chaloupka (2007) and this QET analysis was consistent with their conclusions. However, the SQE approach was based on past performance of the DPS, specifically only nesting beach data, and did not address ongoing or future threats to segments of the DPS that might not have been nor might not yet be reflected by nest count data. The threat matrix approach indicated that the South Atlantic Ocean DPS is likely to decline in the foreseeable future (Figures 20 and 21). These results were largely driven by the ongoing mortality threats to juveniles from fishery bycatch that occurs throughout the South Atlantic Ocean (Table 11). Although conservation efforts by national and international groups in the South Atlantic are currently working toward mitigating bycatch in the South Atlantic, it is unlikely that this source of mortality can be greatly reduced in the near future, largely due to inadequate funding and knowledge gaps that together inhibit implementation of large-scale management actions (Domingo *et al*. 2006).

4.6. Supplemental Material—Sensitivity Analyses

Because our threat matrix analysis was based on a deterministic model, and a few life history parameter values were determined from little empirical data, we conducted sensitivity analyses to evaluate the effects of these life history parameters on the results of the threat matrix analysis.

4.6.1. Effects of AFR and CV of AFR on the Base Population Model

Figures S1-S7 indicate relationships among AFR, CV of AFR, average adult survival rates, average juvenile survival rates, and the dominant eigenvalues. Effects of AFR on the dominant eigenvalues were greater than those of CV of AFR (differences are greater among columns than rows in Figures S1-S7). Differences among DPSs were small as was expected from the small differences in vital rates among the DPSs.

For a deterministic age or stage based matrix model, the asymptotic population growth rate (or the dominant eigenvalue) is a function of survival rates, transition probabilities, and fertility. Consequently, for reasonable ranges of survival rates, the upper bound of the dominant eigenvalue can be found. Fertility parameters of the base matrices were obtained from averages (clutch size, remigration intervals, and nests per female). Consequently, if we were to construct a stochastic demographic model, our base matrix would be treated as the average matrix. The dominant eigenvalue of the average matrix is greater than or equal to the average dominant eigenvalue from stochastic matrices (Jensen's inequality). Consequently, the dominant eigenvalue from our matrix, based on the average life history parameters used in this analysis, may be considered as the maximum possible.

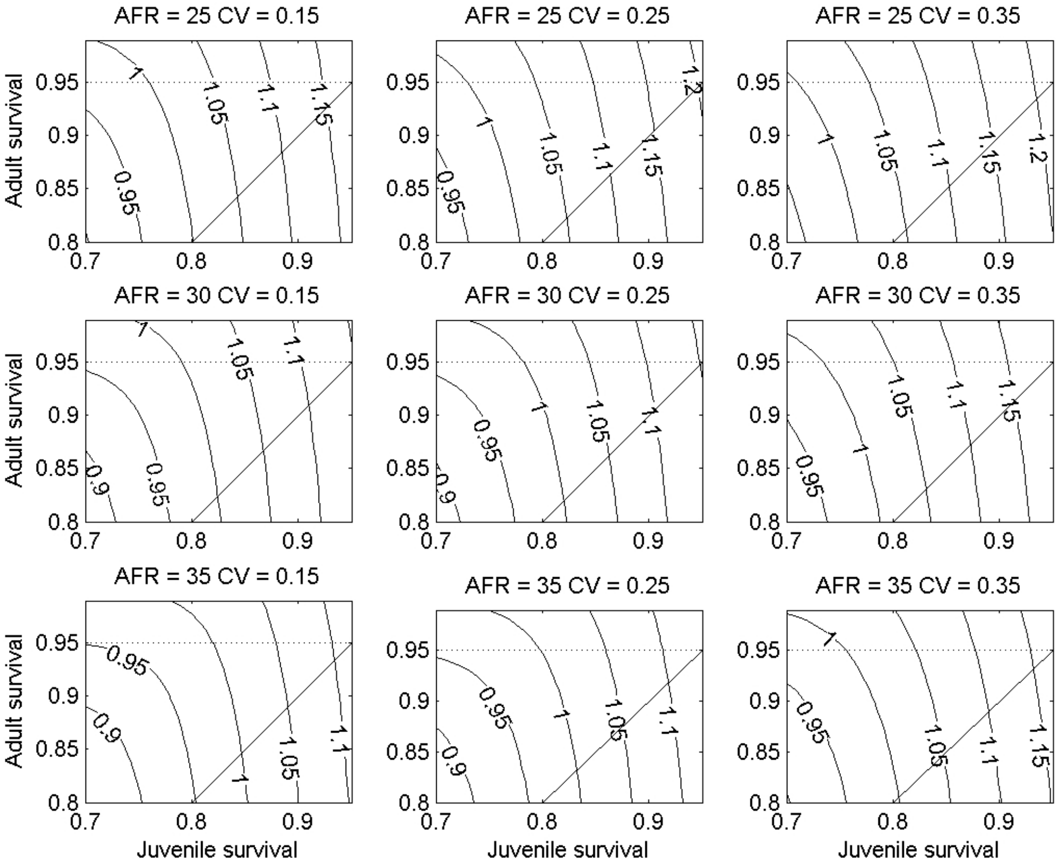
4.6.2. Effects of AFR on the Threat Matrix Analysis

Figures S8-S11 indicate changes in results of the threat matrix analysis due to the assumed AFR. For S8 and S9, AFR was assumed to be 25 years, whereas the CV of AFR was fixed at 15%. Because of the small difference caused by CV of AFR from the previous analysis, results of the sensitivity analysis for CV of AFR are not shown. Greater proportions of dominant eigenvalues are above 1 as the AFR decreases.

4.6.3. Proportion of Neritic Juvenile Stage Duration

To incorporate different levels of anthropogenic mortalities between neritic and oceanic environment in the threat matrix analysis, the matrix model required the durations of neritic and oceanic juvenile stages. Because these estimates are based on a combination of published data, extrapolations, and experts' opinions, and because empirical data were not available for several DPSs, we used the Northwest Atlantic Ocean DPS (19 years in neritic habitat) as a proxy for five DPSs (Table 1). To determine the sensitivity of the default value to the results, the threat matrix analysis was conducted while the proportion of time juveniles spend in neritic habitat was changed from 10% to 90%.

Lower bounds of the dominant eigenvalues decreased as juveniles spent more time in the neritic habitat (Figures S12-S13). This result was caused by the higher anthropogenic mortalities in neritic habitats for all DPSs than in oceanic habitats. The upper bounds of the dominant eigenvalues did not change as much as the lower bound, except the South Atlantic Ocean DPS.


AFR = 25 CV = 0.15

AFR = 25 CV = 0.25

AFR = 25 CV = 0.35

AFR = 30 CV = 0.15

AFR = 30 CV = 0.25

AFR = 30 CV = 0.35

AFR = 35 CV = 0.15

AFR = 35 CV = 0.25

AFR = 35 CV = 0.35

Figure S1. Relationships among adult survival rate, juvenile survival rate, age at first reproduction (AFR), coefficient of variation (CV) of AFR, and the dominant eigenvalue for the North Pacific Ocean DPS. Contour lines indicate the dominant eigenvalues, whereas the diagonal lines indicate equal survival rates between juveniles and adults. Dashed horizontal lines indicate the assumed adult survival rate for the base matrix model. The oceanic survival rate during the first year was fixed at 0.4/year.

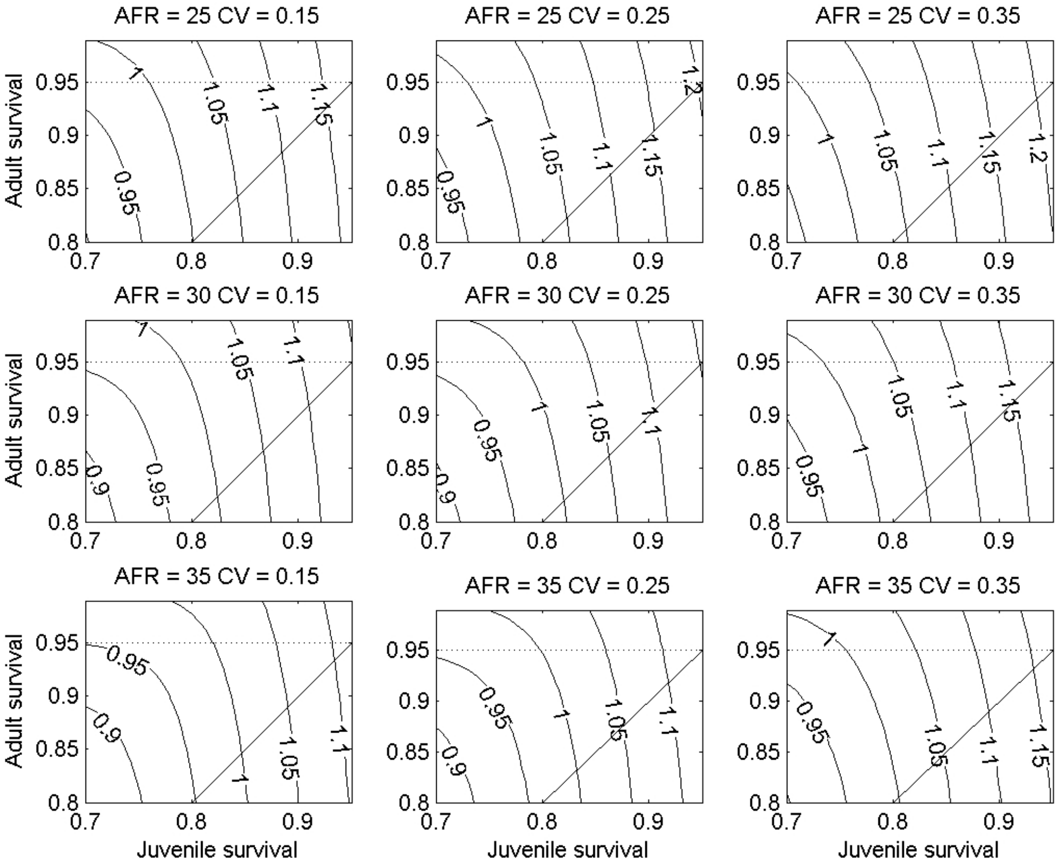


Figure S2. Relationships among adult survival rate, juvenile survival rate, age at first reproduction (AFR), coefficient of variation (CV) of AFR, and the dominant eigenvalue for the South Pacific Ocean DPS. Contour lines indicate the dominant eigenvalues, whereas the diagonal lines indicate equal survival rates between juveniles and adults. Dashed horizontal lines indicate the assumed adult survival rate for the base matrix model. The oceanic survival rate during the first year was fixed at 0.4/year.

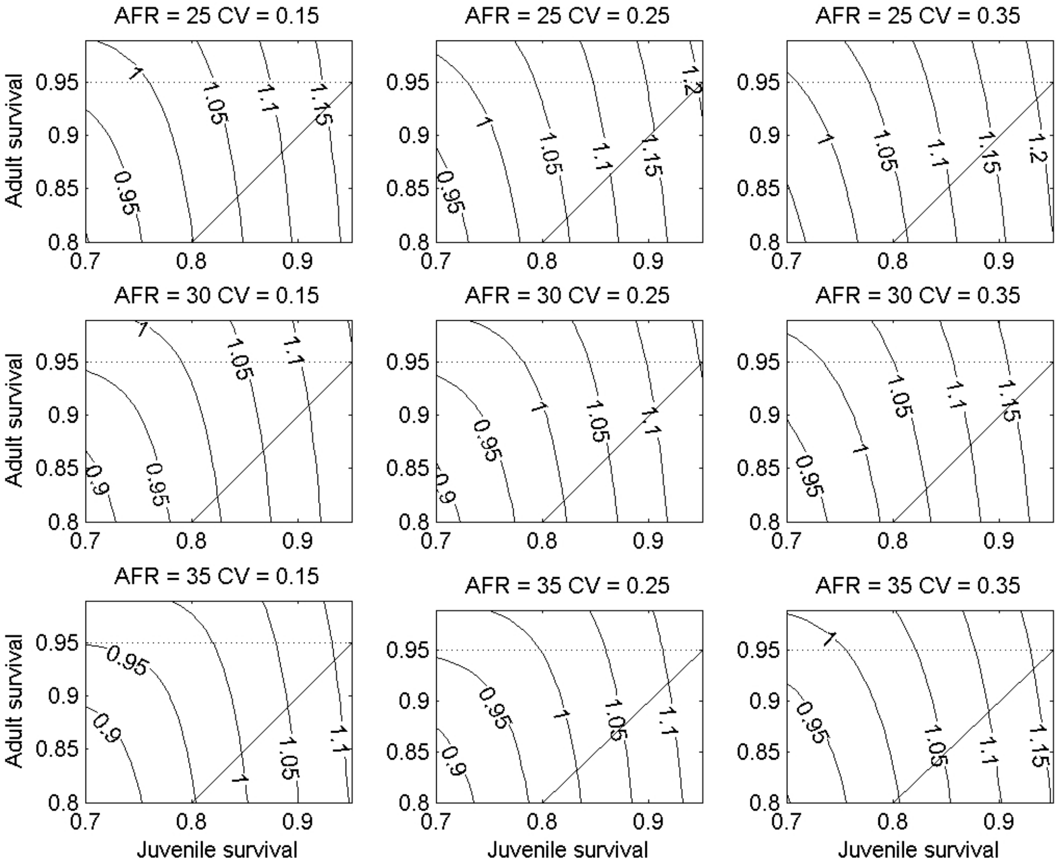


Figure S3. Relationships among adult survival rate, juvenile survival rate, age at first reproduction (AFR), coefficient of variation (CV) of AFR, and the dominant eigenvalue for the North Indian Ocean, Southeast Indo-Pacific Ocean, and Southwest Indian Ocean DPSs. Contour lines indicate the dominant eigenvalues, whereas the diagonal lines indicate equal survival rates between juveniles and adults. Dashed horizontal lines indicate the assumed adult survival rate for the base matrix model. The oceanic survival rate during the first year was fixed at 0.4/year.

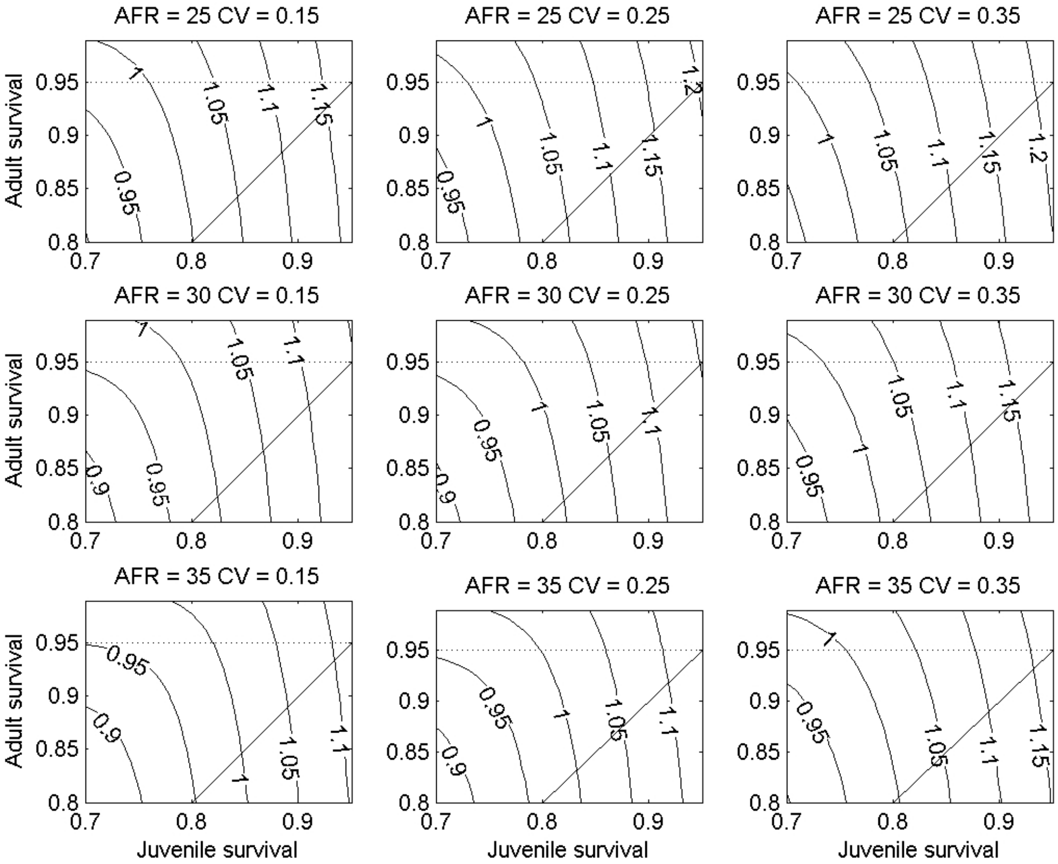


Figure S4. Relationships among adult survival rate, juvenile survival rate, age at first reproduction (AFR), coefficient of variation (CV) of AFR, and the dominant eigenvalue for the Northwest Atlantic Ocean DPS. Contour lines indicate the dominant eigenvalues, whereas the diagonal lines indicate equal survival rates between juveniles and adults. Dashed horizontal lines indicate the assumed adult survival rate for the base matrix model. The oceanic survival rate during the first year was fixed at 0.4/year.

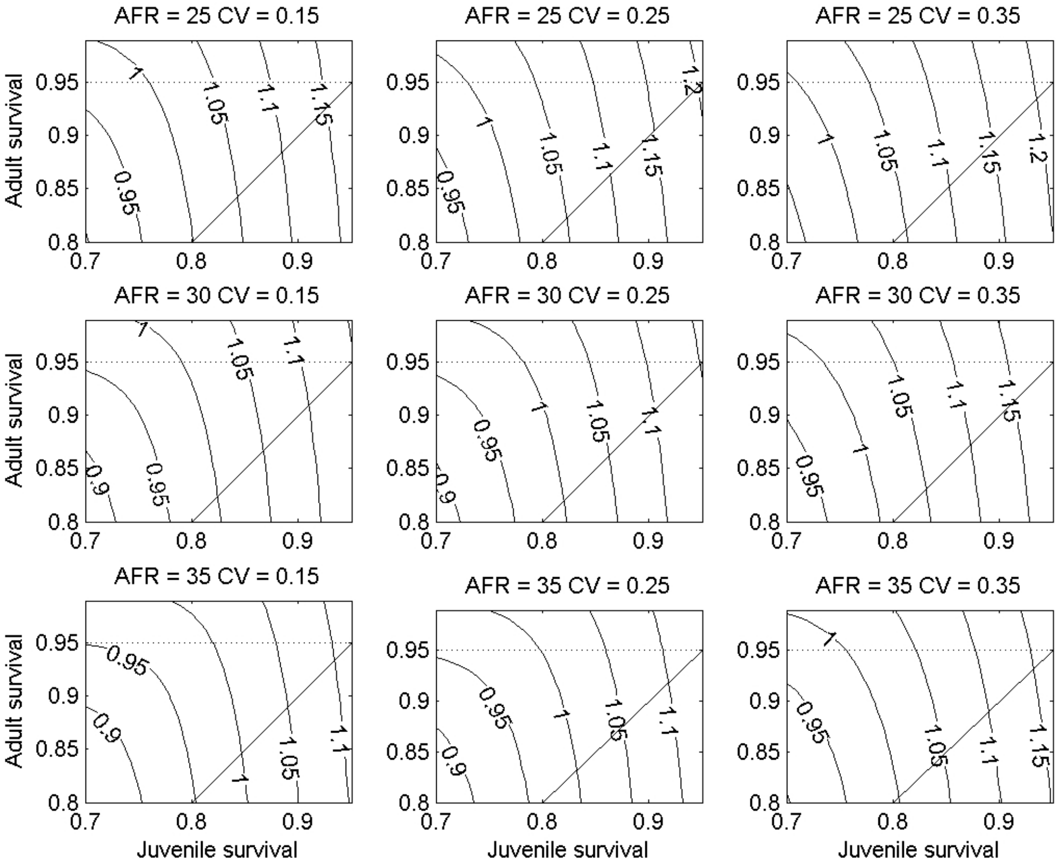


Figure S5. Relationships among adult survival rate, juvenile survival rate, age at first reproduction (AFR), coefficient of variation (CV) of AFR, and the dominant eigenvalue for the Northeast Atlantic Ocean DPS. Contour lines indicate the dominant eigenvalues, whereas the diagonal lines indicate equal survival rates between juveniles and adults. Dashed horizontal lines indicate the assumed adult survival rate for the base matrix model. The oceanic survival rate during the first year was fixed at 0.4/year.

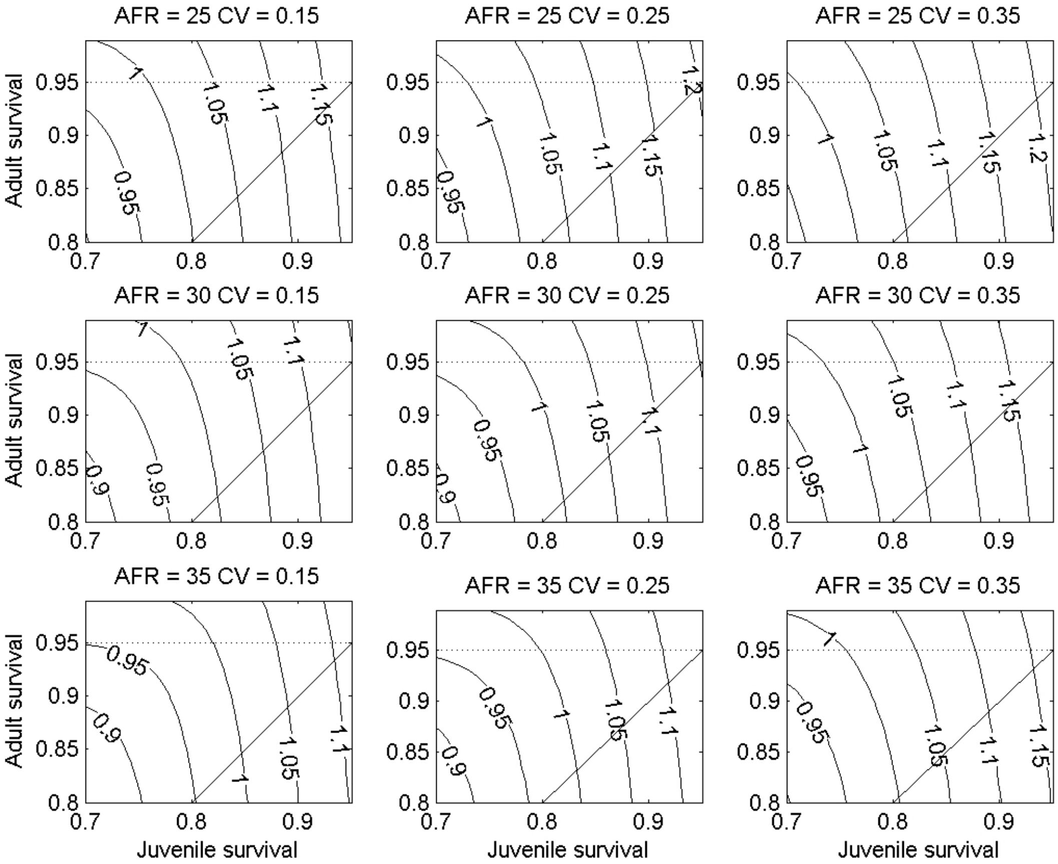


Figure S6. Relationships among adult survival rate, juvenile survival rate, age at first reproduction (AFR), coefficient of variation (CV) of AFR, and the dominant eigenvalue for the Mediterranean Sea DPS. Contour lines indicate the dominant eigenvalues, whereas the diagonal lines indicate equal survival rates between juveniles and adults. Dashed horizontal lines indicate the assumed adult survival rate for the base matrix model. The oceanic survival rate during the first year was fixed at 0.4/year.

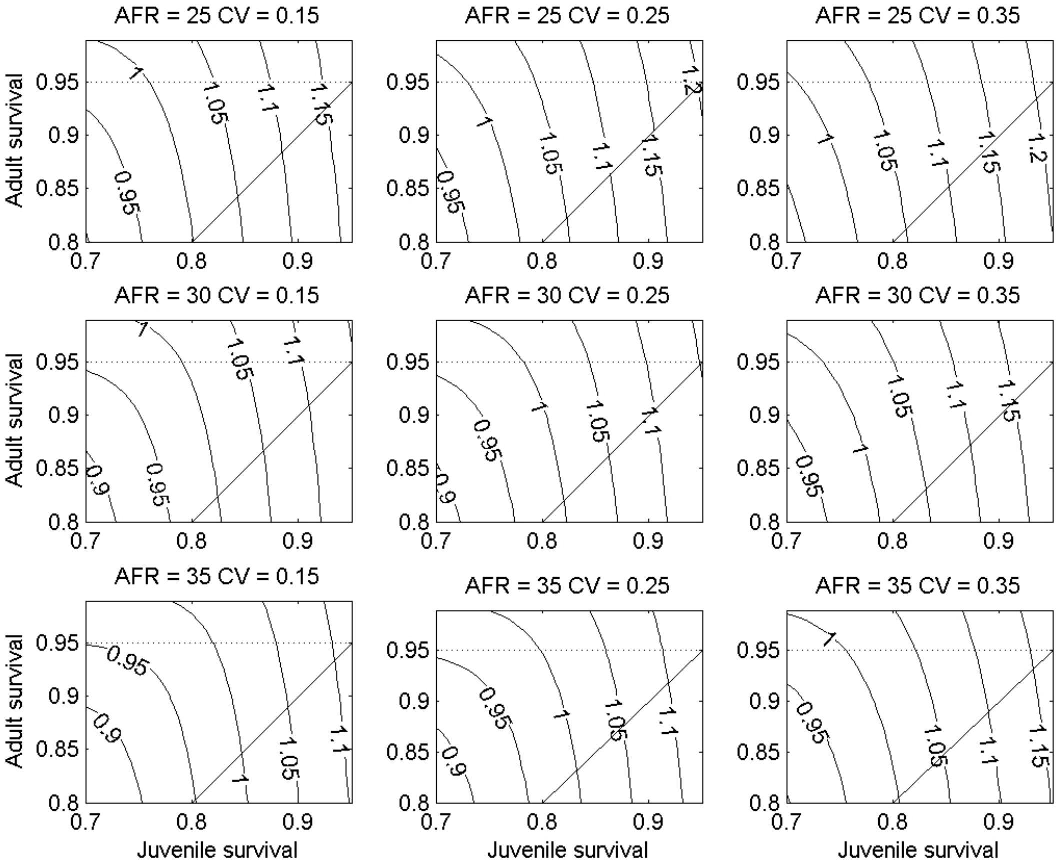


Figure S7. Relationships among adult survival rate, juvenile survival rate, age at first reproduction (AFR), coefficient of variation (CV) of AFR, and the dominant eigenvalue for the South Atlantic Ocean DPS. Contour lines indicate the dominant eigenvalues, whereas the diagonal lines indicate equal survival rates between juveniles and adults. Dashed horizontal lines indicate the assumed adult survival rate for the base matrix model. The oceanic survival rate during the first year was fixed at 0.4/year.

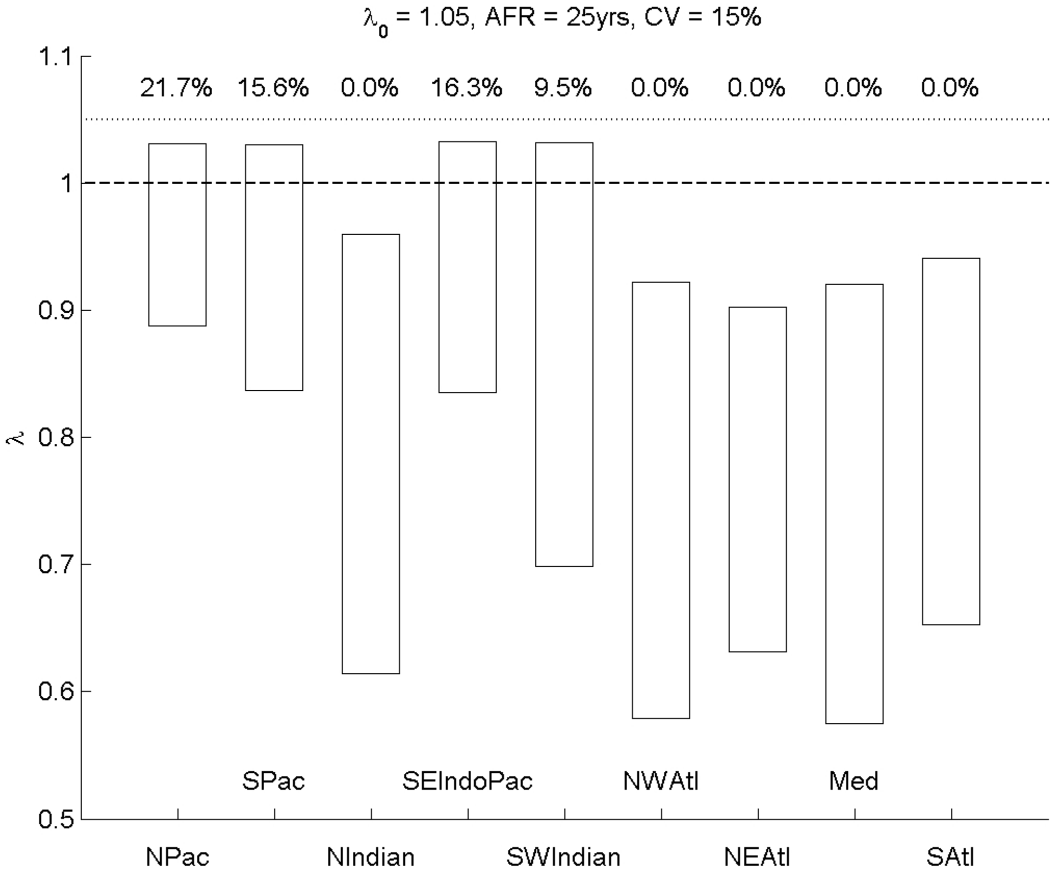


Figure S8. Ranges of dominant eigenvalues for nine DPSs with anthropogenic mortalities when the maximum population growth rates are assumed to be 5% per year. The mean age at first reproduction was assumed to be 25 years and its standard deviation 3.75 (CV=15%). The values above the dotted line indicates the proportion of the bars that are above λ = 1.0.

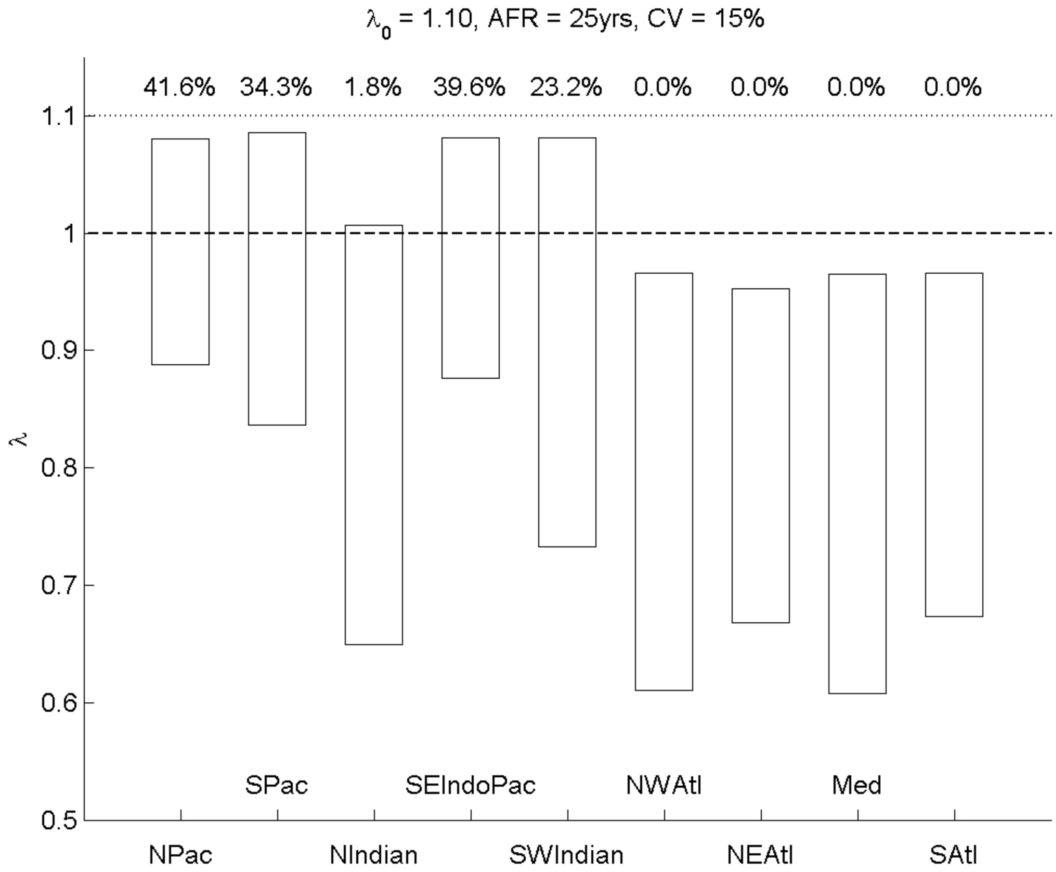


Figure S9. Ranges of dominant eigenvalues for nine DPSs with anthropogenic mortalities when the maximum population growth rates are assumed to be 10% per year. The mean age at first reproduction was assumed to be 25 years and its standard deviation 3.75 (CV = 15%). The values above the dotted line indicates the proportion of the bars that are above $\lambda = 1.0$.

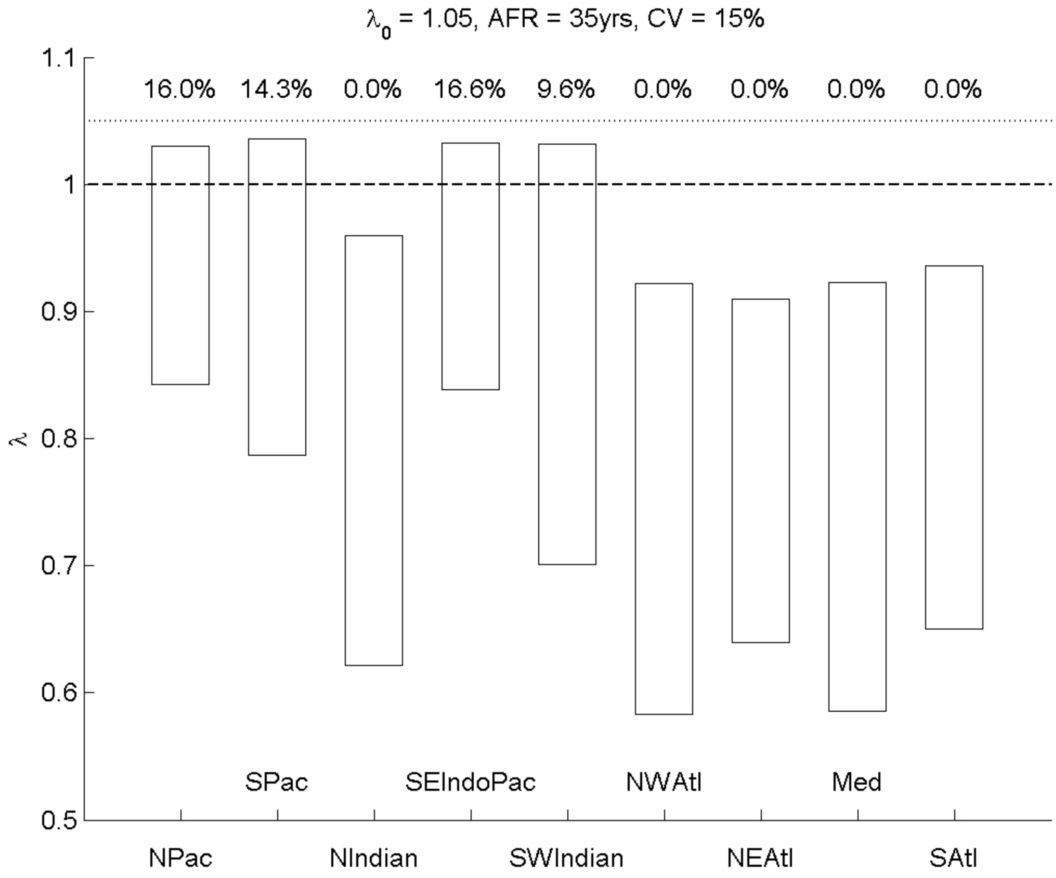


Figure S10. Ranges of dominant eigenvalues for nine DPSs with anthropogenic mortalities when the maximum population growth rates are assumed to be 5% per year. The mean age at first reproduction was assumed to be 35 years and its standard deviation 5.25 (CV = 15%). The values above the dotted line indicates the proportion of the bars that are above λ = 1.0.

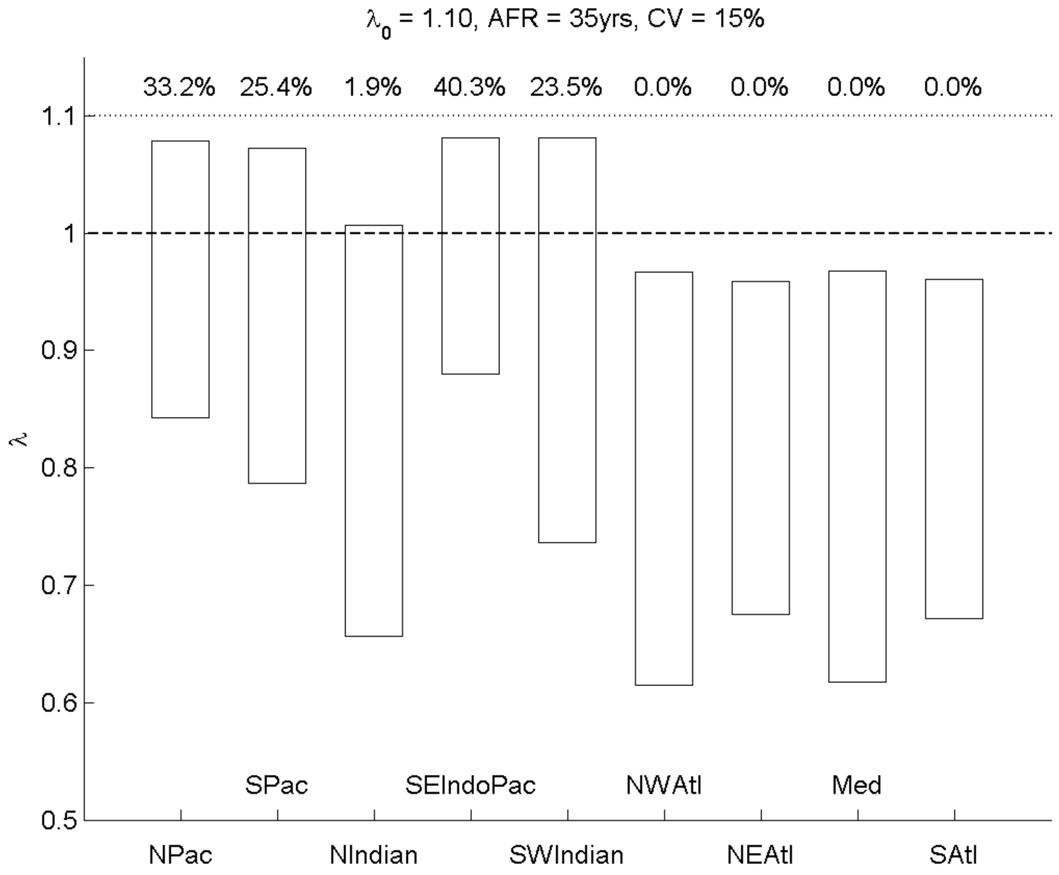


Figure S11. Ranges of dominant eigenvalues for nine DPSs with anthropogenic mortalities when the maximum population growth rates are assumed to be 10% per year. The mean age at first reproduction was assumed to be 35 years and its standard deviation 5.25 (CV = 15%). The values above the dotted line indicates the proportion of the bars that are above λ = 1.0.

λ_0 = 1.05, AFR = 30yrs, CV = 17%, $J_{neritic}$ = 10:10:90%

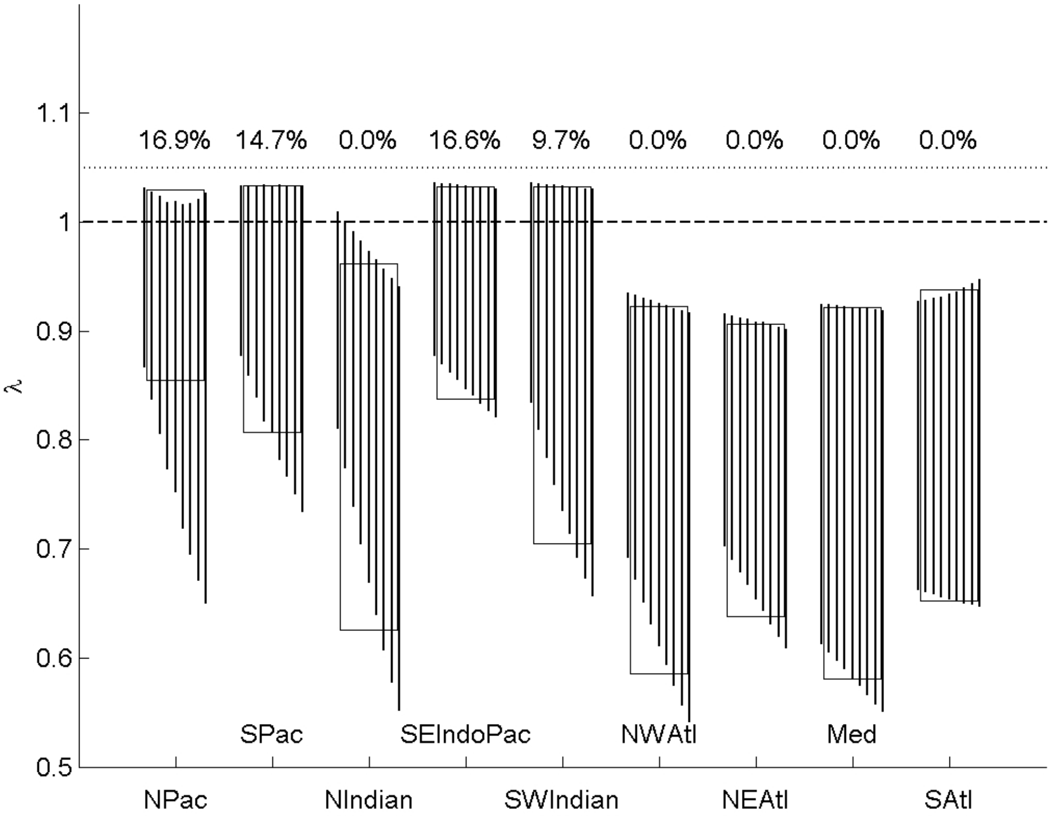


Figure S12. Effects of the changes in the proportion of time juveniles spend in neritic habitat on results of the threat matrix analysis. For each DPS, vertical lines indicate the range of the dominant eigenvalues for the best and worst threat scenarios. Each vertical line is for a proportion of neritic juvenile stage, where the proportion is from 10% to 90% with 10% increments (from left to right). Empty boxes are based on the values in Table 1. The mean age at first reproduction was assumed to be 30 years and its standard deviation 5 (CV=17%). The asymptotic population growth rate was assumed to be 5% per year.

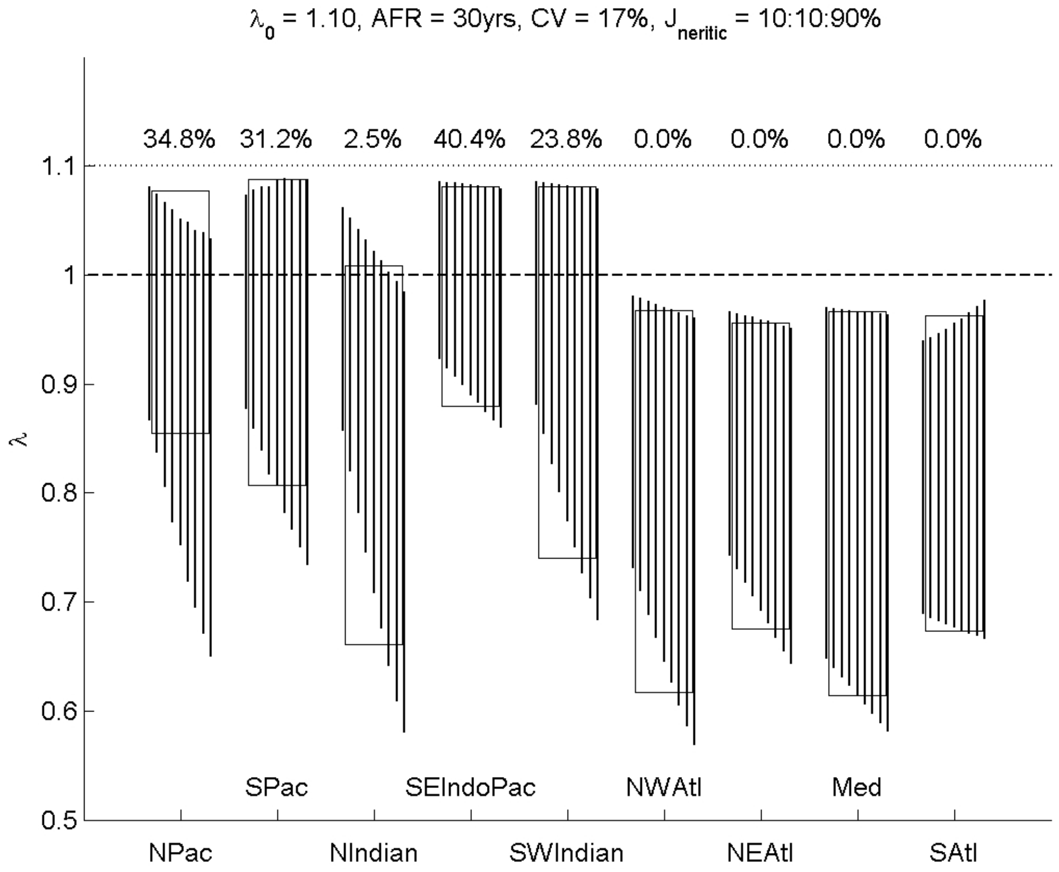


Figure S13. Effects of the proportion of time juveniles spend in neritic habitat on results of the risk analysis. For each DPS, vertical lines indicate the range of the dominant eigenvalues for the best and worst threat scenarios. Each vertical line is for a proportion of neritic juvenile stage, where the proportion is from 10% to 90% with 10% increments (from left to right). Empty boxes are based on the values in Table 1. The mean age at first reproduction was assumed to be 30 years and its standard deviation 5 (CV=17%). The asymptotic population growth rate was assumed to be 10% per year.

SECTION 5—ANALYSIS OF FACTORS LISTED UNDER ESA SECTION 4(a)(1)

Under Section 4(a)(1) of the ESA, NMFS and FWS are required to determine whether any species is an endangered or threatened species because of any of the following factors. These factors are:

(A) the present or threatened destruction, modification, or curtailment of its habitat or range;
(B) overutilization for commercial, recreational, scientific, or educational purposes;
(C) disease or predation;
(D) the inadequacy of existing regulatory mechanisms; or
(E) other natural or manmade factors affecting its continued existence.

The following is an analysis of these factors as they relate to the current status of the nine loggerhead DPSs:

1. North Pacific Ocean
2. South Pacific Ocean
3. North Indian Ocean
4. Southeast Indo-Pacific Ocean
5. Southwest Indian Ocean
6. Northwest Atlantic Ocean
7. Northeast Atlantic Ocean
8. Mediterranean Sea
9. South Atlantic Ocean

In Section 5.1., a general description of the threats that occur for all DPSs is presented under the relevant section 4(a)(1) factor to avoid duplication. In Section 5.2, threats that are specific to a particular DPS are presented by DPS under each section 4(a)(1) factor.

5.1. General Description of Threats

5.1.1. Present or Threatened Destruction, Modification, or Curtailment of Its Habitat or Range

Terrestrial Zone
Destruction and modification of loggerhead nesting habitats are occurring worldwide throughout the species range. The main anthropogenic threats impacting loggerhead nesting habitat include coastal development/construction, placement of erosion control structures and other barriers to nesting, beachfront lighting, vehicular and pedestrian traffic, sand extraction, beach erosion, beach sand placement, beach pollution, removal of native vegetation, and planting of non-native vegetation (Baldwin 1992; NMFS and FWS 1998, 2008; Margaritoulis *et al*. 2003).

Worldwide, development near coastal areas continues and is a major problem as more and more people are moving to or visiting coastal areas. Coastal development may include, but is not limited to, the construction of roads, highways, public infrastructure, hotels, condominiums, houses, harbors, nearshore shoreline stabilization structures (e.g., groins, jetties, breakwaters), and seawalls and other forms of coastal armoring. All of these various forms of coastal construction alter nesting habitat to one degree or another, typically making it less suitable for nesting females, egg incubation, and hatchling emergence. This development can also lead to deleterious effects of animals at-sea through increased fishing activity near important foraging areas (Peckham *et al*. 2007), along migratory corridors, and coastally, near the nesting beaches (e.g., Kamezaki *et al*. 2003).

Erosion of nesting beaches can occur as a result of coastal development when native dune vegetation, which enhances beach stability and acts as an integral buffer zone between land and sea, is degraded or destroyed. This in turn often leaves insufficient nesting opportunities above the high tide line, and nests may be washed out. In contrast, the planting or invasion of less stabilizing, non-native plants can lead to increased erosion and degradation of suitable nesting habitat (Schmelz and Mezich 1988). In addition, taller plants and structures can increase shading and alter the natural sex ratios of hatchlings (Mrosovsky *et al*. 1995). Non-native vegetation also may form impenetrable root mats that can prevent proper nest cavity excavation, as well as trap hatchlings attempting to emerge from nests.

Beach nourishment also affects the incubation environment and nest success. Although the placement of sand on beaches may provide a greater quantity of nesting habitat, the quality of that habitat may be less suitable than pre-existing natural beaches. Constructed beaches tend to differ from natural beaches in several important ways. They are typically wider, flatter, more compact, and the sediments are more moist than those on natural beaches (Nelson *et al*. 1987, Ackerman *et al*. 1991, Ernest and Martin 1999). On severely eroded sections of beach, where little or no suitable nesting habitat previously existed, sand placement can result in increased nesting (Ernest and Martin 1999). However, on most beaches, nesting success typically declines for the first year or two following construction, even though more nesting habitat is available for turtles (Trindell *et al*. 1998, Ernest and Martin 1999, Herren 1999). Reduced nesting success on constructed beaches has been attributed to increased sand compaction, escarpment formation, and changes in beach profile (Nelson *et al*. 1987, Crain *et al*. 1995, Lutcavage *et al*. 1997, Steinitz *et al*. 1998, Ernest and Martin 1999, Rumbold *et al*. 2001). Compaction can inhibit nest construction or increase the amount of time it takes for turtles to construct nests, while escarpments often cause female turtles to return to the ocean without nesting or to deposit their nests seaward of the escarpment where they are more susceptible to frequent and prolonged tidal inundation. In short, sub-optimal nesting habitat may cause decreased nesting success, place an increased energy burden on nesting females, result in abnormal nest construction (Carthy 1996), and reduce the survivorship of eggs and hatchlings. In addition, sand used to nourish beaches may have a different composition than the original beach; thus introducing lighter or darker sand, consequently affecting the relative nest temperatures (Ackerman 1997, Milton *et al*. 1997). Crain *et al*. (1995) provides a review of the potential effects of beach nourishment on sea turtles.

Groins and jetties, shore-perpendicular structures designed to trap sand that would otherwise be transported by longshore currents, interfere with nesting turtle access to the beach, result in a change in beach profile and width (e.g., downdrift erosion, loss of sandy berms, and escarpment formation) (Komar 1983, Pilkey *et al*. 1984, National Research Council 1987), trap hatchlings, and concentrate predatory fishes, resulting in hatchling predation.

The construction of beachfront armoring (i.e., rigid structures placed parallel to the shoreline on the upper beach to prevent both landward retreat of the shoreline and inundation or loss of upland property by flooding and wave action) includes bulkheads, seawalls, soil retaining walls, rock revetments, sandbags, and geotextile tubes. These structures can greatly impact nesting opportunities and hatching success of loggerhead turtles. Mosier (1998) reported that fewer loggerheads made nesting attempts on beaches fronted by seawalls than on adjacent beaches where armoring structures were absent. Mosier (1998) and Mosier and Witherington (2002) found that when turtles did emerge in the presence of armoring structures, more returned to the water without nesting than those on non-armored beaches. Additionally, Mosier (1998) found that turtles on armored sections of beach tended to wander greater distances than those that emerged on adjacent natural beaches. Armoring structures can effectively eliminate a turtle's access to upper regions of the beach/dune system. Consequently, nests on armored beaches were generally found at lower elevations than those on non-walled beaches. Nests laid at lower elevations are subject to a greater risk of repeated tidal inundation and erosion, which can potentially alter thermal regimes, and thus sex ratios (Mrosovsky and Provancha 1992, Mrosovsky 1994, Ackerman 1997). The demand for both nourishment and the placement of hardened structures on the beach as management options for beach erosion are likely to increase in the future in the face of projected sea level rise and more intense storm activity associated with global climate change.

Developments, such as roadways, high rise hotels, and condominiums, also contribute to habitat degradation by increasing noise and light pollution. Studies have shown that light pollution disorients hatchlings, causing them to move inland away from the ocean (Witherington 1997). Hatchlings unable to find the ocean, or delayed in reaching it, incur high mortality from dehydration, exhaustion, or predation (Ehrhart and Witherington 1987, Witherington and Martin 1996). Hatchlings lured into lighted parking lots or toward streetlights are often crushed by passing vehicles (Witherington and Martin 1996). In addition, light pollution deters nesting females from emerging onto the beach to nest (Witherington 1992).

Burgeoning numbers of visitors to beaches may cause sand compaction and nest trampling (Kudo *et al*. 2003). In addition, the placement of recreational beach equipment (e.g., lounge chairs, cabanas, umbrellas, catamarans) degrades the suitability of beaches as nesting habitat by hampering or deterring nesting turtles from accessing the upper beach (Sobel 2002; Margaritoulis *et al*. 2007; Florida Fish and Wildlife Conservation Commission, unpublished data), thus limiting the potential area for nesting. Debris on nesting beaches is also a threat to hatchlings and nesting females. Hatchlings often must navigate through a variety of obstacles before reaching the ocean. These include natural and human-made debris. Debris on the beach may interfere with a hatchling's progress toward the ocean. Research has shown that travel times of hatchlings from the nest to the water may be extended when traversing areas of heavy foot traffic or vehicular ruts (Hosier *et al*. 1981); the same is true of debris on the beach. Hatchlings may be upended and spend both time and energy in righting themselves. Some beach debris may have the potential to trap hatchlings and prevent them from successfully reaching the ocean. In addition, debris over the tops of nests may impede or prevent hatchling emergence. Nesting females also may be impeded by beach debris in their movement up or down the beach, and this may influence selection of nest sites. In rarer circumstances, large debris items may entrap nesting females.

Operating public vehicles on nesting beaches for recreational purposes or beach access also degrades nesting habitat. The ruts left by vehicles in the sand prevent or impede hatchlings from reaching the ocean following emergence from the nest (Mann 1977, Hosier *et al*. 1981, Cox *et al*. 1994, Hughes and Caine 1994). Hatchlings impeded by vehicle ruts are at greater risk of death from predation, fatigue, desiccation, and being crushed by additional vehicle traffic. Light pollution from vehicle lights on the beach after dark may deter females from nesting and disorient hatchlings. Sand compaction due to vehicles on the beach may hinder nest construction and hatchling emergence from nests. Driving directly above incubating egg clutches can cause sand compaction, which may decrease hatching and emerging success and directly kill pre-emergent hatchlings (Mann 1977). Additionally, vehicle traffic on nesting beaches may contribute to erosion, especially during high tides or on narrow beaches where driving is concentrated on the high beach and foredune.

Neritic/Oceanic Zone

Habitat degradation and destruction in the neritic and oceanic zones occur less conspicuously than they do in the terrestrial zone. Direct impacts to bottom habitats occur from activities including bottom trawl and dredge fishing, channel dredging, and sand extraction. Indirect effects can result from both point and non-point source pollution (e.g., upland runoff, direct sewage discharge) associated with coastal development. The impacts of climate change may also result in trophic level alterations, and therefore may affect loggerhead prey abundance and/or distribution. There are currently no data available to quantify impacts from these potential trophic level alterations.

Other fishing methods such as drift and set gillnets, longlines, and pots/traps affect both neritic and oceanic zones by not only incidentally capturing loggerheads but also depleting fish populations, and thus altering ecosystem dynamics. In many cases, loggerhead foraging areas coincide with fishing zones. Selective and usually intense harvest of species in fisheries will result in changes in neritic and oceanic ecosystems (e.g., predator-prey interactions, trophic dynamics and food webs; see Bjorndal 2003). Seney and Musick (2007) reported changes in loggerhead diet resulting from alterations of prey availability in the Chesapeake Bay, Virginia, USA, an important foraging habitat. Perhaps the most destructive fishing methods in neritic ecosystems are bottom trawling (Auster 1998, Brailovskaya 1998, Engel and Kvitek 1998, Kaiser 1998, Pilskaln *et al*. 1998, Schwinghamer *et al*. 1998, Watling and Norse 1998) and dredging (Murray 2004, 2005, 2007). Benthic habitat alteration by mobile fishing gear, especially trawls and dredges, constitutes a globally significant physical disturbance to the marine environment and has significant effects on marine biodiversity (Watling and Norse 1998). The ecological effects of trawling and dredging on the marine environment have been likened to the terrestrial ecological effects of clearcutting forests (Watling and Norse 1998). Mobile fishing gear has been shown to result in short and long-term changes in benthic community composition, including species groups on which loggerheads forage (Gordon *et al*. 1998). The National Research Council (1994) found that habitat alteration by fishing activities is perhaps the least understood of the important environmental effects of fishing. Comprehensive data are lacking to fully understand and quantify these impacts.

Marine pollution can affect loggerhead habitats in both the neritic and oceanic zones. These impacts include contamination from herbicides, pesticides, oil spills, and other chemicals, as well as impacts on water quality (e.g., increases in water column sediments) resulting from structural degradation from excessive boat anchoring, dredging, and marine explosives (Francour *et al*. 1999, Lee Long *et al*. 2000, Margaritoulis *et al*. 2003, Waycott *et al*. 2005).

5.1.2. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes

Deliberate hunting of loggerheads for their meat, shells, and eggs is reduced from previous exploitation levels, but still exists. The level of harvest varies significantly by region. Despite national laws in various countries, in many areas the poaching of eggs and hunting of adults and juveniles is still a problem. Details on these threats are described by DPS in Section 5.2.

5.1.3. Disease or Predation

The population level effects of diseases in loggerheads are not known. At least two bacterial diseases have been described in wild loggerhead populations, including bacterial encephalitis and ulcerative stomatitis/obstructive rhinitis/pneumonia (George 1997). There are few reports of fungal infections in wild loggerhead populations (Homer *et al*. 2000). Both bacterial and fungal infections are common in captive sea turtles (Herbst and Jacobson 1995, George 1997).

Viral diseases have not been documented in free-ranging loggerheads, with the possible exception of sea turtle fibropapillomatosis, which may have a viral etiology (Herbst and Jacobson 1995, George 1997). Fibropapillomatosis is a disease that is characterized by the presence of internal and/or external tumors (fibropapillomas). External tumors can interfere with swimming, vision, and feeding. Although fibropapillomatosis reaches epidemic proportions in some wild green turtle populations, the prevalence of this disease in most loggerhead populations is thought to be small. Mortality levels associated with the disease are still unknown.

A variety of endoparasites, including trematodes, tapeworms, and nematodes have been described in loggerheads (Herbst and Jacobson 1995). Heavy infestations of endoparasites may cause or contribute to debilitation or mortality in sea turtles.

Depredation of sea turtle eggs and hatchlings by native and introduced species occurs on a large number of nesting beaches, as described by DPS in Section 5.2. Planting of non-native plants can have a detrimental effect in the form of roots invading eggs (e.g., tamarisk tree roots invading eggs in Zakynthos, Greece) (Margaritoulis *et al*. 2007).

Harmful algal blooms, such as a red tide, also impact loggerheads. *Karenia brevis*, a type of microalgae known as a dinoflagellate that produces a toxin, has been known to cause red tides (Florida Marine Research Institute 2003). Other types of microorganisms cause different kinds of harmful algal blooms in other parts of the world as well (Florida Marine Research Institute 2003, Gilbert *et al*. 2005).

5.1.4. Inadequacy of Existing Regulatory Mechanisms

Loggerheads are highly migratory, which makes them a shared resource among many nations. Therefore, conservation efforts for loggerhead populations in one country may be jeopardized by activities in another. Many countries lack regulations or have inadequate regulations in place to address the impacts of a wide range of anthropogenic activities that directly injure and kill loggerheads, disrupt necessary behaviors, and alter terrestrial and marine habitats used by the species. In particular, improved regulations of fisheries that incidentally capture loggerheads are needed to reduce mortality. Improved fishery observer coverage is also needed to provide more basic information on loggerhead bycatch. Government regulations and community programs need to be initiated or strengthened to address the impacts of turtle hunting and egg poaching. Overall, increased efforts are needed to assist many foreign countries with the enactment and enforcement of national regulations to protect loggerheads.

The conservation and recovery of sea turtles is facilitated by a number of regulatory instruments at international, regional, national, and local levels. As a result of these designations and agreements, many of the intentional impacts directed at sea turtles have been lessened: harvest of eggs and adults has been slowed at several nesting areas through nesting beach conservation efforts and an increasing number of community-based initiatives are in place to slow the take of turtles in foraging areas. Moreover, there is now a more internationally concerted effort to reduce sea turtle interactions and mortality in artisanal and industrial fishing practices.

Despite these advances, human impacts continue throughout the world. The lack of comprehensive and effective monitoring and bycatch reduction efforts in many pelagic and near-shore fisheries operations still allows substantial direct and indirect mortality, and the uncontrolled development of coastal and marine habitats threatens to destroy the supporting ecosystems of long-lived sea turtles. Although several international agreements provide legal protection for sea turtles, additional multi-lateral efforts are needed to ensure they are sufficiently implemented and/or strengthened, and key non-signatory parties need to be encouraged to accede.

Considering the worldwide distribution of loggerhead turtles, the majority of legal instruments that target or impact sea turtles cover loggerhead turtles. A summary of the main regulatory instruments from throughout the world that relate to loggerhead turtle management is provided below. Cross-regional regulatory instruments will be highlighted here. If an agreement or act is specific to a DPS's geographic area, that action will be discussed under the appropriate DPS in Section 5.2. The pros and cons of many of these were evaluated by Hykle (2002), and a summary of his findings is given when appropriate.

United States Magnuson-Stevens Fishery Conservation and Management Act

The recently-amended U.S. Magnuson-Stevens Fishery Conservation and Management Act (MSA), implemented by NMFS, mandates environmentally responsible fishing practices within U.S. fisheries and has provisions that extend to fishing activities in waters beyond U.S. jurisdiction. Section 610 calls on the U.S. Secretary of Commerce to identify nations with fishing vessels that are engaged or have been engaged in fishing activities in waters beyond any national jurisdiction or in areas beyond the Exclusive Economic Zone of the United States. This section authorizes NMFS to conduct bilateral consultations with such nations to protect living marine resources. If a nation continues to conduct fishing activities that result in bycatch of protected living marine resources, they can be certified to the United States Congress. A result of this certification could be suspension in the trade of fisheries products. Finally, the Act specifically encourages NMFS to conduct international cooperation and assistance with foreign nations that are identified so that bycatch of protected living marine resources can be reduced.

FAO Technical Consultation on Sea Turtle-Fishery Interactions
While not a true international instrument for conservation, the Food and Agriculture Organization of the United Nations' (FAO) technical consultation on sea turtle-fishery interactions was groundbreaking in that it solidified the commitment of this international body to reduce sea turtle bycatch in marine fisheries operations. Recommendations from the technical consultation were endorsed by the FAO Committee on Fisheries (COFI) and called for the immediate implementation by member nations and Regional Fishery Management Organizations (RFMOs) of guidelines to reduce sea turtle mortality in fishing operations, developed as part of the technical consultation. Compliance with these guidelines is voluntary.

Convention on the Conservation of Migratory Species of Wild Animals
This Convention, also known as the Bonn Convention or CMS, is an international treaty that focuses on the conservation of migratory species and their habitats. As of March 2008, the Convention had 108 member states, including parties from Africa, Central and South America, Asia, Europe, and Oceania. While the Convention has successfully brought together about half the countries of the world with a direct interest in sea turtles, it has yet to realize its full potential (Hykle 2002). Its membership does not include a number of key countries, including Brazil, Canada, China, Indonesia, Japan, Mexico, Oman, and the United States. Additional information is available at http://www.cms.int.

Convention on Biological Diversity (CBD)
The primary objectives of this international treaty are: (1) the conservation of biological diversity, (2) the sustainable use of its components, and (3) the fair and equitable sharing of the benefits arising out of the utilization of genetic resources. This Convention has been in force since 1993 and had 191 Parties as of June 2008. While the Convention provides a framework within which broad conservation objectives may be pursued, it does not specifically address sea turtle conservation (Hykle 2002). Additional information is available at http://www.cbd.int.

Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES)
Known as CITES, this Convention was designed to regulate international trade in a wide range of wild animals and plants. CITES was implemented in 1975 and currently includes 173 Parties. Although CITES has been effective at minimizing the international trade of sea turtle products, it does not limit legal harvest within countries, nor does it regulate intra-country commerce of sea turtle products (Hykle 2002). Additional information is available at http://www.cites.org.

United Nations Convention on the Law of the Sea (UNCLOS)
To date, 155 countries, including most mainland countries lining the western Pacific, and the European Community have joined in the convention. The United States has signed the treaty, but the Senate has not ratified it. Aside from its provisions defining ocean boundaries, the convention establishes general obligations for safeguarding the marine environment through mandating sustainable fishing practices and protecting freedom of scientific research on the high seas. Additional information is available at http://www.un.org/Depts/los/index.htm.

United Nations Resolution 44/225 on Large-Scale Pelagic Driftnet Fishing
In 1989, the United Nations called, in an unanimous resolution, for the elimination of all high seas driftnets by 1992. Additional information is available at http://www.intfish.net/treaties/ga44_225.htm.

The International Convention for the Prevention of Pollution from Ships (MARPOL)
The MARPOL Convention is a combination of two treaties adopted in 1973 and 1978 to prevent pollution of the marine environment by ships from operational or accidental causes. The 1973 treaty covered pollution by oil, chemicals, harmful substances in packaged form, sewage and garbage. The 1978 MARPOL Protocol was adopted at a Conference on Tanker Safety and Pollution Prevention which included standards for tanker design and operation. The 1978 Protocol incorporated the 1973 Convention as it had not yet been in force and is known as the International Convention for the Prevention of Marine Pollution from Ships, 1973, as modified by the Protocol of 1978 relating thereto (MARPOL 73/78). The 1978 Convention went into force in 1983 (Annexes I and II). The Convention includes regulations aimed at preventing and minimizing accidental and routine operations pollution from ships. Amendments passed since have updated the convention. To date there are six Annexes with Annexes I and II being mandatory for State Parties and the others being voluntary.

Annex I Regulations for the Prevention of Pollution by Oil
Annex II Regulations for the Control of Pollution by Noxious Liquid Substances in Bulk
Annex III Prevention of Pollution by Harmful Substances Carried by Sea in Packaged Form
Annex IV Prevention of Pollution by Sewage from Ships
Annex V Prevention of Pollution by Garbage from Ships
Annex VI Prevention of Air Pollution from Ships

The Convention Concerning the Protection of the World Cultural and Natural Heritage (World Heritage Convention)
The World Heritage Convention was signed in 1972 and, as of November 2007, 185 states were parties to the Convention. The instrument requires parties to take effective and active measures to protect and conserve habitat of threatened species of animals and plants of scientific or aesthetic value. The World Heritage Convention currently includes 31 marine sites, including important marine turtle habitat such as the Belize Barrier Reef Reserve System, Belize. Additional information is available at http://whc.unesco.org/en/conventiontext.

Ramsar Convention on Wetlands
The Convention on Wetlands, signed in Ramsar, Iran, in 1971, is an intergovernmental treaty, which provides the framework for national action and international cooperation for the conservation and wise use of wetlands and their resources. Currently, there are 158 parties to the convention, with 1,752 wetland sites, including important marine turtle habitat such as the Turtle Beaches and Coral Reefs of Tongaland, South Africa. Additional information is available at http://www.ramsar.org.

5.1.5. Other Natural or Manmade Factors Affecting Its Continued Existence

Incidental Bycatch in Fishing Gear

The most significant manmade factor affecting the conservation and recovery of the loggerhead is incidental capture in commercial and artisanal fisheries. Incidental capture (bycatch) of loggerheads occurs in various fisheries throughout the range of the species. Fishing gear types that may affect loggerheads include pelagic and demersal longlines, drift and set nets (e.g., gillnets, trammel nets), bottom and mid-water trawling, fishing dredges, pound nets and weirs, haul and purse seines, pots and traps, and hook and line gear. While significant progress has been made to reduce bycatch in some fisheries in certain parts of the loggerhead's range, serious bycatch problems remain unaddressed. Specific information on bycatch rates or the number of takes is presented by DPS in Section 5.2.

Other Manmade and Natural Impacts

In addition to fishery bycatch, climate change is another factor that has the potential to greatly impact loggerhead turtles. Impacts from global climate change induced by human activities are likely to become more apparent in future years (Intergovernmental Panel on Climate Change (IPCC) 2007a). The global mean temperature has risen 0.76ºC over the last 150 years, and the linear trend over the last 50 years is nearly twice that for the last 100 years (IPCC 2007a). There is a high confidence, based on substantial new evidence, that observed changes in marine systems are associated with rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels, and circulation. Ocean acidification resulting from massive amounts of carbon dioxide and pollutants released into the air can have major adverse impacts on the calcium balance in the oceans and, thus, on sea turtles. Changes to the marine ecosystem due to climate change include shifts in ranges and changes in algal, plankton, and fish abundance (IPCC 2007b), which could affect loggerhead prey distribution and abundance.

One of the most certain consequences of climate change is rising sea levels (Titus and Narayanan 1995), which will result in increased erosion rates along nesting beaches. This could particularly impact areas with low-lying beaches where sand depth is a limiting factor, as the sea will inundate nesting sites and decrease available nesting habitat (Daniels *et al*. 1993, Fish *et al*. 2005, Baker *et al*. 2006). The loss of habitat as a result of climate change could be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion (Antonelis *et al*. 2006, Baker *et al*. 2006). On some undeveloped beaches, shoreline migration will have limited effects on the suitability of nesting habitat. Bruun (1962) hypothesized that during a sea level rise a typical beach profile will maintain its configuration but will be translated landward and upward. However, along developed coastlines, and especially in areas where erosion control structures have been constructed to limit shoreline movement, rising sea levels will cause severe effects on nesting females and their eggs. Erosion control structures can result in the permanent loss of dry nesting beach or deter nesting females from reaching suitable nesting sites (National Research Council 1990). Nesting females may deposit eggs seaward of the erosion control structures potentially subjecting them to repeated tidal inundation.

Climate change also may affect loggerhead sex ratios. Loggerhead turtles exhibit temperature-dependent sex determination. Rapidly increasing global temperatures may result in warmer incubation temperatures and highly female-biased sex ratios (e.g., Glen and Mrosovsky 2004, Hawkes *et al*. 2009).

Loggerhead sea turtles may be killed or injured from collisions with boat hulls and propellers. These interactions are becoming more common. Capture and mortality of loggerheads during periodic dredging of sediments from navigational channels also have been documented. Power plants have also been found to impinge and/or entrain loggerheads on/in their intake screens/structures.

Marine pollution, including marine debris, oil spills, and bioaccumulative chemicals, is one of the main anthropogenic threats to sea turtles (Tomás *et al*. 2002). Because of their habitat and feeding behavior, loggerheads appear to be one of two sea turtle species that ingest more debris in all of its life stages (Lutcavage *et al*. 1997). Direct or indirect disposal of anthropogenic waste introduces potentially lethal materials into loggerhead foraging habitats or into convergence zones, affecting oceanic juveniles. Loggerheads will ingest plastic pieces, styrofoam pieces, and other marine debris. Ingestion occurs when debris is mistaken for or associated with prey items. Effects may be lethal or non-lethal, resulting in varying side effects that may increase the probability of death (Balazs 1985, Carr 1987, McCauley and Bjorndal 1999, Witherington 2002). For example, nutrient dilution occurs when non-nutritive debris displaces food in the gut, which then may decrease somatic growth and reproduction (McCauley and Bjorndal 1999). Sea turtles at all life stages appear to be highly sensitive to oil spills, perhaps due to certain aspects of their biology and behavior, including a lack of avoidance behavior, indiscriminate feeding in convergence zones, and large pre-dive inhalations (Milton *et al*. 2003). Milton *et al*. (2003) state that the oil effects on turtles include increased egg mortality and developmental defects, direct mortality due to oiling in hatchlings, juveniles and adults, and impacts to the skin, blood, salt glands, and digestive and immune systems. In addition, loggerheads can become entangled in marine debris, such as discarded fishing gear or other entangling materials.

All loggerhead life history stages are vulnerable to the harmful effects of oil through direct contact, degradation of food resources, and loss of habitat (Vargo *et al*. 1986, Minerals Management Service 2000). Vargo *et al*. (1986) reported that sea turtles would be at substantial risk if they encountered an oil spill or large amounts of tar in the environment. In a review of available information on debris ingestion, Balazs (1985) reported that tar balls were the second most prevalent type of debris ingested by sea turtles. Physiological experiments showed that sea turtles exposed to petroleum products may suffer inflammatory dermatitis, ventilatory disturbance, salt gland dysfunction or failure, red blood cell disturbances, immune response, and digestive disorders (Vargo *et al*. 1986, Lutz and Lutcavage 1989, Lutcavage *et al*. 1995). Operational discharge of produced waters, drill muds, and drill cuttings are routinely discharged in marine waters as a result of petroleum production activities (Minerals Management Service 2000). Loggerheads may bioaccumulate heavy metals found in drill muds resulting in debilitation or death. The effects of large-scale oil spills on loggerheads could be locally catastrophic. The impacts of offshore lighted oil production platforms on loggerheads are unknown. Lighted platforms may attract hatchlings, making them more susceptible to predation (de Silva 1982). Neritic juveniles and adults may be attracted by high prey concentrations around the structures, making them more susceptible to ingestion of petroleum products.

Natural factors that have the potential to affect loggerhead recovery include the effects of aperiodic hurricanes, seasonal typhoons, and catastrophic environmental events such as tsunamis. In general, these events are episodic and, although they may affect loggerhead hatchling production, the results are generally localized and they rarely result in whole-scale losses over multiple nesting seasons. The negative effects of these types of events on low-lying and/or developed shorelines may be longer-lasting and a greater threat overall.

Loggerheads are also susceptible to cold stunning, a phenomenon in which turtles become incapacitated as a result of rapidly dropping water temperatures (Witherington and Ehrhart 1989, Morreale *et al*. 1992). As temperatures fall below 8-10°C, turtles may lose their ability to swim and dive, often floating to the surface. It appears to be the rate of cooling that precipitates cold stunning rather than the water temperature itself (Milton and Lutz 2003). Sea turtles that overwinter in inshore waters are most susceptible to cold stunning, because temperature changes are most rapid in shallow water (Witherington and Ehrhart 1989).

5.2. Threats Specific to DPS

5.2.1. North Pacific Ocean DPS

5.2.1.1. Present or Threatened Destruction, Modification, or Curtailment of Its Habitat or Range

Terrestrial Zone
As described in Section 5.1.1., destruction and modification of loggerhead nesting habitat in the North Pacific result from coastal development and construction, placement of erosion control structures and other barriers to nesting, beachfront lighting, vehicular and pedestrian traffic, sand extraction, beach erosion, beach sand placement, beach pollution, removal of native vegetation, and planting of non-native vegetation (NMFS and FWS 1998). Beaches in Japan where loggerheads nest are extensively eroded due to dredging and dams constructed upstream, and are obstructed by seawalls as well. Unfortunately, no quantitative studies have been conducted to determine the impact to the loggerhead nesting populations (Kamezaki *et al*. 2003). However, it is clear that loggerhead nesting habitat has been impacted by erosion and extensive beach use by tourists, both of which have contributed to unusually high mortality of eggs and pre-emergent hatchlings at many Japanese rookeries (Matsuzawa 2006).

Maehama Beach and Inakahama Beach on Yakushima in Kagoshima Prefecture account for approximately 30% of loggerhead nesting in Japan (Kamezaki *et al*. 2003), making Yakushima an important area for nesting beach protection. However, the beaches suffer from beach erosion and light pollution, especially from passing cars, as well as from tourists encroaching on the nesting beaches (Matsuzawa 2006). Burgeoning numbers of visitors to beaches may cause sand compaction and nest trampling. Egg and pre-emergent hatchling mortality in Yakushima has been shown to be higher in areas where public access is not restricted and is mostly attributed to human foot traffic (“stomping”) on nests (Kudo *et al*. 2003). Fences have been constructed around areas where the highest densities of nests are laid; however, there are still lower survival rates of eggs and pre-emergent hatchlings due to excessive foot traffic (Ohmuta 2006). Recently, the Ministry of Environment has supported the local non-governmental organization conducting turtle surveys and conservation on Yakushima in establishing guidelines for surveys and minimizing impacts by humans encroaching on the nesting beaches. As of the 2009 nesting season, humans accessing Inakahama, Maehama, and Yotsuse beaches at night must comply with the established rules (Y. Matsuzawa, Sea Turtle Association of Japan, personal communication, 2009).

Loggerhead nesting habitat also has been lost at important rookeries in Miyazaki due in part to port construction that involved development of a groin of one kilometer from the coast into the sea, a yacht harbor with breakwaters and artificial beach, and an airport, causing erosion of beaches on both sides of the construction zone. This once excellent breeding habitat for loggerheads is now seriously threatened by erosion (Takeshita 2006).

Minabe-Senri beach, Wakayama Prefecture is a "submajor" nesting beach (in Kamezaki *et al.* 2003), but is one of the most important rookeries on the main island of Japan (Honshu). Based on unpublished data, Matsuzawa (2006) reported hatching success of unwashed-out clutches at Minabe-Senri beach to be 24% in 1996, 50% in 1997, 53% in 1998, 48% in 1999, 62% in 2000, 41% in 2001, and 34% in 2002.

Neritic/Oceanic Zone
As described in Section 5.1.1., threats to habitat in the loggerhead neritic and/or oceanic zones include fishing practices, channel dredging, sand extraction, marine pollution, and climate change. These threats also occur in the North Pacific Ocean. Fishing methods not only incidentally capture loggerheads (Section 5.1.5.), but also deplete invertebrate and fish populations and thus alter ecosystem dynamics. In many cases loggerhead foraging areas coincide with fishing zones. For example, using aerial surveys and satellite telemetry, juvenile foraging hotspots have recently been identified off the coast of Baja California, Mexico; these hotspots overlap with intensive small-scale fisheries (Peckham and Nichols 2006; Peckham *et al.* 2007, 2008). Comprehensive data currently are unavailable to fully understand how intense harvesting of fish resources changes neritic and oceanic ecosystems. Climate change also may result in future trophic changes, thus impacting loggerhead prey abundance and/or distribution.

5.2.1.2. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes

In Japan, the use of loggerhead meat for food is not popular except historically in local communities such as Kochi and Wakayama prefectures. In addition, egg collection was common in the coastal areas during times of hunger and later by those who valued loggerhead eggs as revitalizers or aphrodisiacs and acquired them on the black market (in Kamezaki *et al.* 2003, Takeshita 2006). Currently, due in large part to research and conservation efforts throughout the country, egg harvesting no longer represents a problem in Japan (Kamezaki *et al.* 2003, Ohmuta 2006, Takeshita 2006). Laws were enacted in 1973 to prohibit egg collection on Yakushima, and in 1988, the laws were extended to the entire Kagoshima Prefecture, where two of the most important loggerhead nesting beaches are protected (Matsuzawa 2006).

Despite national laws, in many other countries where loggerheads are found migrating through or foraging, the hunting of adults and juveniles is still a problem, as seen in Baja California Sur, Mexico (Koch *et al*. 2006). Sea turtles have been protected in Mexico since 1990, when a federal law decreed the prohibition of the "extraction, capture and pursuit of all species of sea turtle in federal waters or from beaches within national territory ... [and a requirement that] ... any species of sea turtle incidentally captured during the operations of any commercial fishery shall be returned to the sea, independently of its physical state, dead or alive" (in Garcia-Martinez and Nichols 2000). Despite the ban, studies have shown that sea turtles continue to be caught, both indirectly in fisheries and by a directed harvest of juveniles and subadults. Turtles are principally hunted using nets, longlines, and harpoons. While some are killed immediately, others are kept alive in pens and transported to market. The market for sea turtles consists of two types: the local market (consumed locally) and the export market (sold to restaurants in Mexico cities such as Tijuana, Ensenada, and Mexicali, and U.S. cities such as San Diego and Tucson). Consumption is highest during holidays such as Easter and Christmas (Wildcoast/Grupo Tortuguero de las Californias 2003).

Based on a combination of analyses of stranding data, beach and sea surveys, tag-recapture studies, and extensive interviews, all carried out between June 1994 and January 1999, Nichols (2003) conservatively estimated the annual take of sea turtles by various fisheries and through direct harvest in the Baja California, Mexico, region. Sea turtle mortality data collected between 1994 and 1999 indicate that over 90% of sea turtles recorded dead were either green turtles (30% of total) or loggerheads (61% of total), and signs of human consumption were evident in over half of the specimens. These studies resulted in an estimated 1,950 loggerheads killed annually, affecting primarily immature size classes. The primary causes for mortality were the incidental take in a variety of fishing gears and direct harvest for consumption and [illegal] trade (Nichols 2003).

From April 2000 to July 2003 throughout the Bahia Magdalena region (including local beaches and towns), researchers found 1,945 sea turtle carcasses, 44.1% of which were loggerheads. Of the sea turtle carcasses found, slaughter for human consumption was the primary cause of death for all species (63% for loggerheads). Over 90% of all turtles found were juveniles or subadults (Koch *et al*. 2006). As the population of green turtles has declined in Baja California Sur waters, poachers have switched to loggerheads (H. Peckham, Pro Peninsula, personal communication, 2006).

5.2.1.3. Disease or Predation

The potential effects of diseases and endoparasites, as described in Section 5.1.3., also exist for loggerheads found in the North Pacific Ocean. As in other nesting locations, egg predation also exists in Japan, particularly by raccoon dogs (*Nyctereutes procyonoides*) and weasels (*Mustela itatsi*); however, quantitative data do not exist to evaluate the impact on loggerhead populations (Kamezaki *et al*. 2003). Loggerheads in the North Pacific Ocean also may be impacted by harmful algal blooms.

5.2.1.4. Inadequacy of Existing Regulatory Mechanisms

Unless identified to a specific geographic area, the regulatory mechanisms described in Section 5.1.4. also apply to sea turtles found in the North Pacific. The following additional instruments also apply to loggerheads found in this area.

United States Magnuson-Stevens Fishery Conservation and Management Act
The recently-amended U.S. Magnuson-Stevens Fishery Conservation and Management Act (MSA), implemented by NMFS, mandates environmentally responsible fishing practices within U.S. fisheries. Section 301 of the MSA establishes National Standards to be addressed in management plans. Any regulations promulgated to implement such plans, including conservation and management measures, shall, to the extent practicable, (A) minimize bycatch and (B) to the extent bycatch cannot be avoided, minimize the mortality of such bycatch. Section 301 by itself does not require specific measures. However, mandatory bycatch reduction measures can be incorporated into management plans for specific fisheries, as has happened with the U.S. pelagic longline fisheries in the Atlantic and Pacific oceans. Section 316 requires the establishment of a bycatch reduction engineering program to develop "technological devices and other conservation engineering changes designed to minimize bycatch, seabird interactions, bycatch mortality, and post-release mortality in Federally managed fisheries."

Indian Ocean – South-East Asian Marine Turtle Memorandum of Understanding (IOSEA)
This Memorandum of Understanding (MOU) puts in place a framework through which States of the Indian Ocean and South-East Asian region, as well as other concerned States, can work together to conserve and replenish depleted marine turtle populations for which they share responsibility. This collaboration is achieved through the collective implementation of an associated Conservation and Management Plan. Currently, there are 27 signatory states. The United States became a signatory in 2001. Numerous accomplishments have been made under the auspices of this MOU. For detailed information, visit the IOSEA website at http://www.ioseaturtles.org.

Memorandum of Understanding on ASEAN Sea Turtle Conservation and Protection
The objectives of this MOU, initiated by the Association of South East Asian Nations (ASEAN), are to promote the protection, conservation, replenishing, and recovery of sea turtles and their habitats based on the best available scientific evidence, taking into account the environmental, socio-economic and cultural characteristics of the Parties. It currently has 10 signatory states in the South East Asian Region: Brunei Darussalam, Cambodia, Indonesia, Laos, Malaysia, Myanmar, Philippines, Singapore, Thailand, and Vietnam. Additional information is available at http://www.aseansec.org/6185.htm.

Inter-American Convention for the Protection and Conservation of Sea Turtles (IAC)
This Convention is one of only a handful of international treaties dedicated exclusively to sea turtles, setting standards for the conservation of these endangered animals and their habitats with a large emphasis on bycatch reduction. It is the only binding multi-national agreement for sea turtles and is open to all countries in North, Central, and South America, and the Caribbean. It currently has 12 Contracting Parties, with the United States being a signatory in 1999. Additional information is available at http://www.iacseaturtle.org.

Convention for the Protection of the Marine Environment and Coastal Area of the South-East Pacific (Lima Convention)
This Convention's signatories include all countries along the Pacific Rim of South America from Panama to Chile. Among other resource management components, this Convention established a protocol for the conservation and management of protected marine resources. Stemming from this Convention is the Commision Permanente del Pacifico Sur (CPPS) that has developed a Marine Turtle Action Plan for the Southeast Pacific that outlines a strategy for protecting and recovering marine turtles in this region.

Convention for the Protection of Natural Resources and Environment of the South Pacific Region (Noumea Convention)
In force since 1990, this Convention includes 19 Parties from throughout the Southwest Pacific Ocean: Australia, Cook Islands, Federated States of Micronesia, Fiji, France, Kiribati, Marshall Islands, Nauru, New Zealand, Niue, Palau, Papua New Guinea, Samoa, Solomon Islands, Tonga, Tuvalu, United Kingdom, United States, and Vanuatu. This Convention is relevant only for the Exclusive Economic Zones of Party nations, and does not have jurisdiction in international waters. Relevant to marine turtles are the associated Protocol for the Prevention of Pollution of the South Pacific Region by Dumping (reduction of marine debris), and the Action Plan for Managing the Natural Resources and Environment of the South Pacific Region, which occurs under the auspices of the South Pacific Regional Environment Programme (SPREP). Additional information is available at http://sedac.ciesin.org/entri/texts/natural.resources.south.pacific.1986.html.

5.2.1.5. Other Natural or Manmade Factors Affecting Its Continued Existence

Incidental Bycatch in Fishing Gear
Incidental capture in artisanal and commercial fisheries is a significant threat to the survival of loggerheads in the North Pacific. Sea turtles may be caught in pelagic and demersal longlines, drift and set gillnets, bottom and mid-water trawling, fishing dredges, pound nets and weirs, haul and purse seines, pots and traps, and hook and line gear.

Based on turtle sightings and capture rates reported in a survey of fisheries research and training vessels and extrapolated to total longline fleet effort by the Japanese fleet in 1978, Nishemura and Nakahigashi (1990) estimated that 21,200 turtles, including greens, leatherbacks, loggerheads, olive ridleys, and hawksbills, were captured annually by Japanese tuna longliners in the western Pacific and South China Sea, with a reported mortality of approximately 12,300 turtles per year. Using commercial tuna longline logbooks, research vessel data, and questionnaires, Nishemura and Nakahigashi (1990) estimated that for every 10,000 hooks in the western Pacific and South China Sea, one turtle is captured, with a mortality rate of 42%. Although species-specific information on the bycatch is not available, vessels reported that 36% of the sightings of turtles in locations that overlap with these commercial fishing grounds were loggerheads.

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

Foreign high-seas driftnet fishing in the North Pacific Ocean for squid, tuna, and billfish ended with a United Nations moratorium in December 1992. Except for observer data collected in 1990-1991, there is virtually no information on the incidental take of sea turtle species by the driftnet fisheries prior to the moratorium. The high-seas squid driftnet fishery in the North Pacific was observed in Japan, Korea, and Taiwan, while the large-mesh fisheries targeting tuna and billfish were observed in the Japanese fleet (1990-1991) and the Taiwanese fleet (1990). A combination of observer data and fleet effort statistics indicate that 2,986 loggerhead turtles were entangled by the combined fleets of Japan, Korea, and Taiwan during June 1990 through May 1991, when all fleets were monitored. Of these incidental entanglements, an estimated 805 loggerheads were killed (27% mortality rate) (Wetherall 1997). Data on size composition of the turtles caught in the high-seas driftnet fisheries also were collected by observers. The majority of loggerheads measured by observers were immature (Wetherall 1997). The cessation of high-seas driftnet fishing in 1992 should have reduced the incidental take of marine turtles. However, nations involved in driftnet fishing may have shifted to other gear types (e.g., pelagic or demersal longlines, coastal gillnets); this shift in gear types could have resulted in either similar or increased turtle bycatch and associated mortality.

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

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

Extensive ongoing studies regarding loggerhead mortality and bycatch have been administered off the coast of Baja California Sur, Mexico. The location and timing of loggerhead strandings documented in 2003-2005 along a 43-kilometer beach (Playa San Lázaro) indicated bycatch in local small-scale fisheries. In order to corroborate this, in 2005, researchers observed two small-scale fleets operating closest to an area identified as a high-use area for loggerheads. One fleet, based out of Puerto López Mateos, fished primarily for halibut using bottom set gillnets, soaking from 20 to 48 hours. This fleet consisted of up to 75 boats in 2005, and, on a given day, 9 to 40 vessels fished the deep area (32-45 meter depths). During a two-month period, 11 loggerheads were observed taken in 73 gillnet day-trips, with eight of those loggerheads landed dead (observed mortality rate – 73%). The other fleet, based in Santa Rosa, fished primarily for demersal sharks using bottom-set longlines baited with tuna or mackerel and left to soak for 20 to 48 hours. In 2005, the fleet numbered only five to six vessels. During the seven daylong bottom-set longline trips observed, 26 loggerheads were taken, with 24 of them landed dead (observed mortality rate – 92%). Based on these observations, researchers estimated that in 2005 at least 299 loggerheads died in the bottom-set gillnet fishery and at least 680 loggerheads died in the bottom-set longline fishery. This annual bycatch estimate of approximately 1,000 loggerheads is considered a minimum and is also supported by shoreline mortality surveys and informal interviews (Peckham *et al*. 2007).

These results suggest that incidental capture at Baja California Sur is one of the most significant sources of mortality identified for the North Pacific loggerhead population and underscores the importance of reducing bycatch in small-scale fisheries.

In the U.S. Pacific, longline fisheries targeting swordfish and tuna and drift gillnet fisheries targeting swordfish have been identified as the primary fisheries of concern for loggerheads. Bycatch of loggerhead turtles in these fisheries has been significantly reduced as a result of time-area closures, required gear modifications, and hard caps imposed on turtle bycatch, with 100% observer coverage in certain areas.

The California/Oregon (CA/OR) drift gillnet fishery targets swordfish and thresher shark off the west coast of the United States. The fishery has been observed by NMFS since July 1990, and currently averages 20%. From July 1990 to January 2000, the CA/OR drift gillnet fishery was observed to incidentally capture 17 loggerheads (12 released alive, 1 injured, and 4 killed). Based on a worst-case scenario, NMFS estimated that a maximum of 33 loggerheads in a given year could be incidentally taken by the CA/OR drift gillnet fleet. Sea turtle mortality rates for hard-shelled species were estimated to be 32% (NMFS 2000).

In 2000, analyses conducted under the mandates of the ESA showed that the CA/OR drift gillnet fishery was taking excessive numbers of sea turtles, such that the fishery "jeopardized the continued existence of" loggerheads and leatherbacks. In this case, the consulting agency (NMFS) was required to provide a reasonable and prudent alternative to the action (i.e., the fishery). In order to reduce the likelihood of interactions with loggerhead sea turtles, NMFS has regulations in place to close areas to drift gillnet fishing off southern California during forecasted or occurring El Niño events from June 1 through August 31, when loggerheads are likely to move into the area from the Pacific coast of Baja California following a preferred prey species, pelagic red crabs.

Prior to 2000, the Hawaii-based longline fishery targeted highly migratory species north of Hawaii using gear largely used by fleets around the world. From 1994-1999, the fishery was estimated to take between 369 and 501 loggerheads per year, with between 64 and 88 mortalities per year (NMFS 2000). Currently, the Hawaii-based shallow longline fishery targeting swordfish is strictly regulated such that an annual take of 17 loggerheads is authorized for the fishery, beginning in 2004, when the fishery was re-opened after being closed for several years. In 2004 and 2005, the fishing year was completed without reaching the turtle take levels (1 and 10 loggerheads were captured, respectively, with fleets operating with 100% observer coverage). However, in 2006, 17 loggerheads were taken, forcing the fishery to be shut down early. In 2007, 15 loggerheads were taken by the fishery. Most loggerheads were released alive (NMFS-Pacific Islands Regional Office, Observer Database Public Website, 2008).

Recent investigations off the coast of Japan, particularly focused off the main islands of Honshu, Shikoku, and Kyushu, have revealed a major threat to the more mature stage classes of loggerheads (approximately 70-80 cm SCL) due to pound net fisheries set offshore of the nesting beaches and in the coastal foraging areas. While pound nets constitute the third largest fishery in terms of metric tons of fish caught in Japan, they account for the majority of loggerhead bycatch by Japanese fisheries. Open-type pound nets studied in an area off Shikoku were shown to take loggerheads as the most prevalent sea turtle species caught but had lower mortality rates (less than 15%), primarily because turtles could reach the surface to breathe. Middle layer and bottom-type pound nets in particular have high rates of mortality (nearly 100%), because the nets are submerged and sea turtles are unable to reach the surface. Estimates of loggerhead mortality in one area studied between April 2006 and September 2007 were on the order of 100 individuals. While the fishing industry has an interest in changing its gear to open-type, it is very expensive, and the support from the Japanese government is limited (T. Ishihara, Sea Turtle Association of Japan, personal communication, 2007). Nonetheless, the BRT recognizes that coastal pound net fisheries off Japan may pose a significant threat to the North Pacific population of loggerheads.

Quantifying the magnitude of the threat of fisheries in the North Pacific Ocean on loggerhead sea turtles is very difficult given the low level of observer coverage or investigations into bycatch conducted by countries that have large fishing fleets. Efforts have been made to quantify the effect of pelagic longline fishing on loggerheads, and annual estimates of bycatch were on the order of over 10,000 sea turtles, with as many as 2,600 individual loggerheads killed annually through immediate or delayed mortality as a result of interacting with the gear (Lewison *et al.* 2004).

Other Manmade and Natural Impacts

Similar to other areas of the world (see Section 5.1.5.), climate change and sea level rise have the potential to impact loggerheads in the North Pacific Ocean. For example, Matsuzawa *et al.* (2002) found heat-related mortality of pre-emergent hatchlings in Minabe Senri Beach and concluded that this population is vulnerable to even small temperature increases resulting from global warming because sand temperatures already exceed the optimal thermal range for incubation. Recently, Chaloupka *et al.* (2008) used generalized additive regression modeling and autoregressive-prewhitened cross-correlation analysis to consider whether changes in regional ocean temperatures affect long-term nesting population dynamics for Pacific loggerheads from primary nesting assemblages in Japan and Australia. Researchers chose four nesting sites with a generally long time series to model, two in Japan (Kamouda rookery, declining population, and Yakushima rookery, generally increasing in the last 20 years), and two nesting sites in Australia (Woongarra rookery, generally declining through early 1990s and beginning to recover, and Wreck Island rookery, which is generally declining). Analysis of 51 years of mean annual sea surface time series around two core foraging areas off Japan and eastern Australia, showed a general warming of the oceans in these regions. In general, nesting abundance for all four rookeries was inversely related to sea surface temperatures; that is, higher sea surface temperatures during the previous year in the core foraging area resulted in lower summer season nesting at all rookeries. Given that cooler ocean temperatures are generally associated with increased productivity and that female sea turtles generally require at least one year to acquire sufficient fat stores for vitellogenesis to occur in the foraging grounds, as well as the necessary energy required for migration, any lag in productivity due to warmer temperatures has physiological basis. Over the long-term, warming ocean temperatures could therefore lead to lower productivity and prey abundance, and thus reduced nesting and recruitment by Pacific loggerheads (Chaloupka *et al*. 2008).

As described in Section 5.1.5., other anthropogenic impacts, such as boat strikes and ingestion or entanglement in marine debris, as well as entrainment in coastal power plants, also apply to loggerheads in the North Pacific.

Natural environmental events, such as cyclones and hurricanes, may affect loggerheads in the North Pacific Ocean. Typhoons have also been shown to cause severe beach erosion and negatively affect hatching success at many loggerhead nesting beaches in Japan, especially in areas already prone to erosion. For example, during the 2004 season, the Japanese archipelago suffered a record number of typhoons and many nests were drowned or washed out. Extreme sand temperatures at nesting beaches also create highly skewed female sex ratios of hatchlings or threaten the health of hatchlings. Without human intervention to protect clutches against some of these natural threats, many of these nests would be lost (Matsuzawa 2006).

5.2.2. South Pacific Ocean DPS

The following information on threats pertains to all loggerhead turtles that may be found within the boundaries of the South Pacific Ocean DPS. Because foraging loggerheads from this DPS also are found within the boundaries of the Southeast-Indo Pacific Ocean DPS (specifically in the Gulf of Carpentaria west of 142.5ºE), the narrative for that region should also be consulted.

5.2.2.1. Present or Threatened Destruction, Modification, or Curtailment of Its Habitat or Range

Terrestrial Zone
As described in Section 5.1.1., destruction and modification of loggerhead nesting habitat in the South Pacific result from coastal development and construction, placement of erosion control structures and other barriers to nesting, beachfront lighting, vehicular traffic, beach erosion, beach pollution, removal of native vegetation, and planting of non-native vegetation (NMFS and FWS 1998, Limpus 2009).

Removal or destruction of native dune vegetation, which enhances beach stability and acts as an integral buffer zone between land and sea, results in erosion of nesting habitat. Preliminary studies on nesting beaches in New Caledonia include local oral histories that attribute the decrease in loggerhead nesting to the removal of vegetation for construction purposes and subsequent beach erosion (Limpus *et al*. 2006).

Beach armoring presents a barrier to nesting in the South Pacific. On the primary nesting beach in New Caledonia, a rock wall was constructed to prevent coastal erosion, and sea turtle nesting attempts have been unsuccessful. Local residents are seeking authorization to extend the wall further down the beach (Limpus *et al*. 2006).

Neritic/Oceanic Zone
As described in Section 5.1.1., threats to habitat in the loggerhead neritic and/or oceanic zones include fishing practices, channel dredging, sand extraction, marine pollution, and climate change. These threats also occur in the South Pacific Ocean. Climate change, for instance, may result in future trophic changes, thus impacting loggerhead prey abundance and/or distribution.

5.2.2.2. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes

Legislation in Australia outlaws the harvesting of loggerheads by indigenous peoples (Limpus *et al*. 2006). Despite national laws, in many areas the poaching of eggs and hunting of adults and juveniles is still a problem, and Limpus (2009) suggests that the harvest rate of loggerheads by indigenous hunters, both within Australia and in neighboring countries, is on the order of 40 turtles per year. Preliminary studies suggest that local harvesting in New Caledonia constitutes about 5% of the nesting population (Limpus *et al*. 2006). Loggerheads also are consumed after being captured incidentally in high-seas fisheries of the southeastern Pacific (Alfaro-Shigueto *et al*. 2006), and occasionally may be the product of illegal trade throughout the region.

5.2.2.3. Disease or Predation

The potential effects of diseases and endoparasites, as described in Section 5.1.3., also exist for loggerheads found in the South Pacific. While the prevalence of fibropapillomatosis in most loggerhead populations is thought to be small, an exception is in Moreton Bay, Australia, where 4.4% of the 320 loggerheads captured exhibited the disease during 1990-1992 (Limpus *et al*. 1994). A subsequent study also found a high prevalence of fibropapillomatosis in the area (Quackenbush *et al*. 2000).

Predation on nests and hatchlings by terrestrial vertebrates is a major problem at loggerhead rookeries in the South Pacific. At mainland rookeries in eastern Australia, for example, the introduced fox (*Vulpes vulpes*) has been the most significant predator on loggerhead eggs (Limpus 1985, 2009). Although this has been minimized in recent years (to <5%; Limpus 2009), researchers believe the earlier egg loss will greatly impact recruitment to this nesting population in the early 21st century (Limpus and Reimer 1994). Predation on hatchlings by crabs and diurnal birds is also a threat (Limpus 2009). In New Caledonia, feral dogs pose a predation threat to nesting loggerheads, and thus far no management has been implemented (Limpus *et al*. 2006).

5.2.2.4. Inadequacy of Existing Regulatory Mechanisms

Unless identified to a specific geographic area, the regulatory mechanisms described in Section 5.1.4. also apply to sea turtles found in the South Pacific Ocean. The following additional instruments also apply to loggerheads found in this area.

Indian Ocean – South-East Asian Marine Turtle Memorandum of Understanding (IOSEA)
For information about this instrument, see section 5.2.1.4. for the North Pacific Ocean DPS.

Memorandum of Understanding on ASEAN Sea Turtle Conservation and Protection
For information about this instrument, see section 5.2.1.4. for the North Pacific Ocean DPS.

Inter-American Convention for the Protection and Conservation of Sea Turtles (IAC)
For information about this instrument, see section 5.2.1.4. for the North Pacific Ocean DPS.

Convention for the Protection of the Marine Environment and Coastal Area of the South-East Pacific (Lima Convention)
For information about this instrument, see section 5.2.1.4. for the North Pacific Ocean DPS.

Convention for the Protection of Natural Resources and Environment of the South Pacific Region (Noumea Convention)
For information about this instrument, see section 5.2.1.4. for the North Pacific Ocean DPS.

5.2.2.5. Other Natural or Manmade Factors Affecting Its Continued Existence

Incidental Bycatch in Fishing Gear
Incidental capture in artisanal and commercial fisheries is a significant threat to the survival of loggerheads throughout the South Pacific. The primary gear types involved in these interactions include longlines, driftnets, set nets, and trawl fisheries. These are employed by both artisanal and industrial fleets, and target a wide variety of species including tunas, sharks, sardines, swordfish, and mahi mahi (see below).

In the southwestern Pacific, bottom trawling gear has been a contributing factor to the decline in the eastern Australian loggerhead population (Limpus and Reimer 1994). The northern Australian prawn fishery (NPF) is made up of both a banana prawn fishery and a tiger prawn fishery, and extends from Cape York, Queensland (142ºE) to Cape Londonberry, Western Australia (127ºE). The fishery is one of the most valuable in all of Australia and in 2000 comprised 121 vessels fishing approximately 16,000 fishing days (Robins *et al*. 2002a). In 2000, the use of turtle excluder devices (TEDs) in the NPF was made mandatory, due in part to several factors: (1) objectives of the Draft Australian Recovery Plan for Marine Turtles, (2) requirement of the Australian Environment Protection and Biodiversity Conservation Act for Commonwealth fisheries to become ecologically sustainable, and (3) the 1996 U.S. import embargo on wild-caught prawns taken in a fishery without adequate turtle bycatch management practices (Robins *et al*. 2002a). Data primarily were collected by volunteer fishers who were trained extensively in the collection of scientific data on sea turtles caught as bycatch in their fishery. Prior to the use of TEDs in this fishery, the NPF annually took between 5,000 and 6,000 sea turtles as bycatch, with a mortality rate of an estimated 40% due to drowning, injuries, or being returned to the water comatose (Poiner and Harris 1996). Since the mandatory use of TEDs has been in effect, the annual bycatch of sea turtles in the NPF has dropped to less than 200 sea turtles per year, with a mortality rate of approximately 22% (based on recent years). This lower mortality rate also may be based on better sea turtle handling techniques adopted by the fleet. In general, loggerheads were the third most common sea turtle taken in this fishery.

Loggerheads are also taken by longline fisheries operating out of Australia (Limpus 2009). For example, Robins *et al*. (2002b) estimate that approximately 400 turtles are killed annually in Australian pelagic longline fishery operations. Of this annual estimate, leatherbacks accounted for over 60% of this total, while unidentified hardshelled turtles accounted for the remaining species. Therefore, the effect of this longline fishery on loggerheads is unknown.

Loggerheads also have been the most common turtle species captured in shark control programs in Australia (Kidston *et al*. 1992, Limpus 2009). From 1998-2002, a total of 232 loggerheads were captured with 195 taken on drum lines and 37 taken in nets, both with a low level of direct mortality (Limpus 2009).

In the southeastern Pacific, significant bycatch has been reported in artisanal gillnet and longline shark and mahi mahi fisheries operating out of Peru (Kelez *et al*. 2003, Alfaro-Shigueto *et al*. 2006) and to a lesser extent, Chile (Donoso and Dutton 2006). The fishing industry in Peru is the second largest economic activity in the country, and, over the past few years, the longline fishery has rapidly increased. Currently, nearly 600 longline vessels fish in the winter and over 1,300 vessels fish in the summer. During an observer program in 2003/2004, 588 sets were observed during 60 trips, and 154 sea turtles were taken as bycatch. Loggerheads were the species most often caught (73.4%). Of the loggerheads taken, 68% were entangled and 32% were hooked. Of the two fisheries, sea turtle bycatch was highest during the mahi mahi season, with 0.597 turtles/1,000 hooks, while the shark fishery caught 0.356 turtles/1,000 hooks (Alfaro-Shigueto *et al*. 2008b). A separate study by Kelez *et al*. (2003) reported that approximately 30% of all turtles bycaught in Peru were loggerheads. In many cases, loggerheads are kept on board for human consumption; therefore, the mortality rate in this artisanal longline fishery is likely high because sea turtles are retained for future consumption or sale.

Data on loggerhead bycatch in Chile are limited to the industrial swordfish fleet. Since 1990, fleet size has ranged from 7 to 23 vessels with a mean of approximately 14 vessels per year. These vessels fish up to and over 1,000 nautical miles along the Chilean coast with mechanized sets numbering approximately 1,200 hooks (M. Donoso, ONG Pacifico Laud - Chile, personal communication, 2007). Loggerhead bycatch is present in Chilean fleets; however, the catch rate is substantially lower than that reported for Peru (P. Dutton, NMFS, and M. Donoso, ONG Pacifico Laud - Chile, unpublished data).

Other Manmade and Natural Impacts

Other threats such as debris ingestion, boat strikes, and port dredging also impact loggerheads in the South Pacific, although these threats have been minimized in recent years due to a variety of legislative actions (Limpus 2009). Loggerhead mortality resulting from dredging of channels in Queensland is a persistent, albeit minor problem. From 1999-2002, the average annual reported mortality was 1.7 turtles per year (range = 1-3) from port dredging operations (Limpus 2009). Climate change and sea level rise have the potential to impact loggerheads in the South Pacific Ocean, yet the impact of these threats has not been quantified.

Natural environmental events, such as cyclones or hurricanes, may affect loggerheads in the South Pacific Ocean. These types of events may disrupt loggerhead nesting activity, albeit on a temporary scale. As described in more detail in Section 5.2.1.5., Chaloupka *et al*. (2008) demonstrated that nesting abundance of loggerheads in Australia was inversely related to sea surface temperatures, and suggested that a long term warming trend in the South Pacific may be adversely impacting the recovery potential of this population.

5.2.3. North Indian Ocean DPS

5.2.3.1. Present or Threatened Destruction, Modification, or Curtailment of Its Habitat or Range

Terrestrial Zone
As described in Section 5.1.1., destruction and modification of loggerhead nesting habitat in the North Indian Ocean result from coastal development and construction, beachfront lighting, vehicular and pedestrian traffic, beach pollution, removal of native vegetation, and planting of non-native vegetation (NMFS and FWS 1998).

The primary loggerhead nesting beaches of this DPS are at Masirah Island, Oman, and are still relatively undeveloped but now facing increasing development pressures. Newly paved roads closely paralleling most of the Masirah Island coast are bringing newly constructed highway lights (E. Possardt, FWS, personal observation, 2008) and greater access to nesting beaches by the public. Light pollution from the military installation at Masirah Island also is evident at the most densely nested northern end of the island and is a likely cause of hatchling misorientation and nesting female disturbance (E. Possardt, FWS, personal observation, 2008). Beach driving occurs on most of the major beaches outside the military installation. This vehicular traffic creates ruts that obstruct hatchling movements (Mann 1977, Hosier *et al*. 1981, Cox *et al*. 1994, Baldwin 1992), tramples nests, and destroys vegetation and dune formation processes, which exacerbates light pollution effects. Free ranging camels, sheep, and goats overgraze beach vegetation, which impedes natural dune formation (E. Possardt, FWS, personal observation, 2008). Development of a new hotel on a major loggerhead nesting beach at Masirah Island is near completion and, although not yet approved, there are plans for a major resort at an important loggerhead nesting beach on one of the Halaniyat Islands. Armoring structures common to many developed beaches throughout the world are not yet evident on the major loggerhead nesting beaches of this DPS.

Neritic/Oceanic Zone
As described in Section 5.1.1., threats to habitat in the loggerhead neritic and/or oceanic zones include fishing practices, channel dredging, sand extraction, marine pollution, and climate change. These threats also occur in the North Indian Ocean.

Fishing methods not only incidentally capture loggerheads (Section 5.1.5.), but also deplete invertebrate and fish populations and thus alter ecosystem dynamics. In many cases loggerhead foraging areas coincide with fishing zones. There has been an apparent growth in artisanal and commercial fisheries in waters surrounding Masirah Island (Baldwin 1992). Climate change also may result in future trophic changes, thus impacting loggerhead prey abundance and/or distribution.

5.2.3.2. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes

The use of loggerhead meat for food in Oman is not legal or popular. However, routine egg collection on Masirah Island does occur (Baldwin 1992). The extent of egg collection as estimated by Masirah rangers and local residents is approximately 2,000 clutches per year (<10%).

5.2.3.3. Disease or Predation

The potential effects of diseases and endoparasites, as described in Section 5.1.3., also exist for loggerheads found in the North Indian Ocean. Natural egg predation on Oman loggerhead nesting beaches undoubtedly occurs, but is not well documented or believed to be significant. Predation on hatchlings by Arabian red fox (*Vulpes vulpes arabica*), ghost crabs (*Ocypode saratan*), night herons (*Nycticorax nycticorax*), and gulls (*Larus* spp.) likely occurs. While quantitative data do not exist to evaluate these impacts on the North Indian Ocean loggerhead population, they are not likely to be significant.

5.2.3.4. Inadequacy of Existing Regulatory Mechanisms

Unless identified to a specific geographic area, the regulatory mechanisms described in Section 5.1.4. also apply to sea turtles found in the North Indian Ocean. There are two additional instruments that apply to loggerheads found in this area.

Indian Ocean – South-East Asian Marine Turtle Memorandum of Understanding (IOSEA)
For information about this instrument, see section 5.2.1.4. for the North Pacific Ocean DPS.

Nairobi Convention for the Protection, Management and Development of the Marine and Coastal Environment of the Eastern African Region
The Nairobi Convention was signed in 1985 and came into force in 1996. The Convention covers 10 States, including five island States in the Western Indian Ocean. The Contracting Parties are Comoros, France (La Reunion), Kenya, Madagascar, Mauritius, Mozambique, Seychelles, Somalia, Tanzania and the Republic of South Africa. This instrument "provides a mechanism for regional cooperation, coordination and collaborative actions, and enables the Contracting Parties to harness resources and expertise from a wide range of stakeholders and interest groups towards solving interlinked problems of the coastal and marine environment." Additional information is available at http://www.unep.org/NairobiConvention.

5.2.3.5. Other Natural or Manmade Factors Affecting Its Continued Existence

Incidental Bycatch in Fishing Gear
The magnitude of the threat of incidental capture of sea turtles in artisanal and commercial fisheries in the North Indian Ocean is difficult to assess. A bycatch survey administered off the coast of Sri Lanka between September 1999 and November 2000 reported 5,241 total turtle entanglements, of which 1,310 were loggerheads, between Kalpitiya and Kirinda (Kapurusinghe and Saman 2001, Kapurusinghe and Cooray 2002). Sea turtle bycatch has been reported in driftnet and set gillnets, longlines, trawls, and hook and line gear (Kapurusinghe and Saman 2001, Kapurusinghe and Cooray 2002, Lewison *et al.* 2004).

Quantifying the magnitude of the threat of fisheries on loggerheads in the North Indian Ocean is difficult given the low level of observer coverage or investigations into bycatch conducted by countries that have large fishing fleets. Efforts have been made to quantify the effects of pelagic longline fishing on loggerheads globally (Lewison *et al*. 2004). While there were no turtle bycatch data available from the North Indian Ocean to use in their assessment, extrapolations that considered bycatch data for the Pacific and Atlantic basins gave a conservative estimate of 6,000 loggerheads captured in the Indian Ocean in the year 2000. Interviews with rangers at Masirah Island reveal that shark gillnets capture many loggerheads off nesting beaches during the nesting season. As many as 60 boats are involved in this fishery with up to 6 km of gillnets being fished daily from June through October along the Masirah Island coast. Rangers reported one example of 17 loggerheads in one net (E. Possardt, FWS, personal communication, 2008).

Other Manmade and Natural Impacts
As described in Section 5.1.5., other anthropogenic impacts, such as boat strikes and ingestion or entanglement in marine debris, as well as entrainment in coastal power plants, likely apply to loggerheads in the North Indian Ocean. Similar to other areas of the world (see Section 5.1.5.), climate change and sea level rise have the potential to impact loggerheads in the North Indian Ocean. This includes beach erosion and loss from rising sea levels, skewed hatchling sex ratios from rising beach incubation temperatures, and abrupt disruption of ocean currents used for natural dispersal during the complex life cycle. Climate change impacts could have profound long term impacts on nesting populations in the North Indian Ocean, but it is not possible to quantify the potential impacts at this point in time.

Natural environmental events, such as cyclones, tsunamis, and hurricanes, affect loggerheads in the North Indian Ocean. For example, during the 2007 season, Oman suffered a rare typhoon. In general, however, severe storm events are episodic and, although they may affect loggerhead hatchling production, the results are generally localized and they rarely result in whole-scale losses over multiple nesting seasons.

5.2.4. Southeast Indo-Pacific Ocean DPS

The following information on threats pertains to all loggerhead turtles that may be found within the boundaries of the Southeast Indo-Pacific Ocean DPS. Because foraging loggerheads from this DPS also are found within the boundaries of the South Pacific Ocean DPS, the narrative for that region should also be consulted.

5.2.4.1. Present or Threatened Destruction, Modification, or Curtailment of Its Habitat or Range

Terrestrial Zone
The primary loggerhead nesting beaches for this DPS occur in Australia on Dirk Hartog Island and Murion Islands (Baldwin *et al*. 2003), which are undeveloped. Dirk Hartog Island is soon to become part of the National Park System.

Neritic/Oceanic Zone
As described in Section 5.1.1., threats to habitat in the loggerhead neritic and/or oceanic zones include fishing practices, channel dredging, sand extraction, marine pollution, and climate change. These threats also occur in the Southeast Indo-Pacific Ocean. Fishing methods not only incidentally capture loggerheads (Section 5.1.5.), but also deplete invertebrate and fish populations and thus alter ecosystem dynamics. In many cases, loggerhead foraging areas coincide with fishing zones. Climate change also may result in future trophic changes, thus impacting loggerhead prey abundance and/or distribution.

5.2.4.2. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes

Legislation in Australia outlaws the harvesting of loggerheads by indigenous peoples (Limpus *et al*. 2006). Dirk Hartog Island and Murion Islands are largely uninhabited, and poaching of eggs and turtles is likely negligible.

5.2.4.3. Disease or Predation

The potential effects of diseases and endoparasites, as described in Section 5.1.3., also exist for loggerheads found in the Southeast Indo-Pacific Ocean. On the North West Cape and the beaches of the Ningaloo coast of mainland Australia, a long established feral European red fox (*Vulpes vulpes*) population preyed heavily on eggs and is thought to be responsible for the lower numbers of nesting turtles on the mainland beaches (Baldwin *et al*. 2003). The fox populations have been eradicated on Dirk Hartog Island and Murion Islands (Baldwin *et al*. 2003).

5.2.4.4. Inadequacy of Existing Regulatory Mechanisms

Unless identified to a specific geographic area, the regulatory mechanisms described in Section 5.1.4. also apply to sea turtles found in the Southeast Indo-Pacific Ocean. The following additional instrument also applies to loggerheads found in this area.

Indian Ocean – South-East Asian Marine Turtle Memorandum of Understanding (IOSEA)
For information about this instrument, see section 5.2.1.4. for the North Pacific Ocean DPS.

5.2.4.5. Other Natural or Manmade Factors Affecting Its Continued Existence

Incidental Bycatch in Fishing Gear
The extent of the threat of incidental capture of sea turtles in artisanal and commercial fisheries in the Southeast Indo-Pacific Ocean is unknown. Sea turtles are caught in pelagic and demersal

longlines, gillnets, trawls, seines, and pots and traps (Environment Australia 2003). There is evidence of significant historic bycatch from prawn fisheries, which may have depleted nesting populations long before nesting surveys were initiated in the 1990s (Baldwin *et al*. 2003).

Quantifying the magnitude of the threat of fisheries on loggerheads in the Southeast Indo-Pacific Ocean is very difficult given the low level of observer coverage or investigations into bycatch conducted by countries that have large fishing fleets. Efforts have been made to quantify the effects of pelagic longline fishing on loggerheads globally (Lewison *et al*. 2004). While there were no turtle bycatch data available from the Southeast Indo-Pacific Ocean to use in their assessment, extrapolations that considered bycatch data for the Pacific and Atlantic basins gave a conservative estimate of 6,000 loggerheads captured in the Indian Ocean in the year 2000. Loggerheads are known to be taken by Japanese longline fisheries operating off of Western Australia (Limpus 2009). The effect of the longline fishery on loggerheads in the Indian Ocean is largely unknown (Lewison *et al*. 2004).

The northern Australian prawn fishery (NPF) is made up of both a banana prawn fishery and a tiger prawn fishery, and extends from Cape York, Queensland (142ºE) to Cape Londonberry, Western Australia (127ºE). The fishery is one of the most valuable in all of Australia and in 2000 comprised 121 vessels fishing approximately 16,000 fishing days (Robins *et al*. 2002a). In 2000, the use of turtle excluder devices in the NPF was made mandatory, due in part to several factors: (1) objectives of the Draft Australian Recovery Plan for Marine Turtles, (2) requirement of the Australian Environment Protection and Biodiversity Conservation Act for Commonwealth fisheries to become ecologically sustainable, and (3) the 1996 U.S. import embargo on wild-caught prawns taken in a fishery without adequate turtle bycatch management practices (Robins *et al*. 2002a). Data primarily were collected by volunteer fishers who were trained extensively in the collection of scientific data on sea turtles caught as bycatch in their fishery. Prior to the use of TEDs in this fishery, the NPF annually took between 5,000 and 6,000 sea turtles as bycatch, with a mortality rate of an estimated 40%, due to drowning, injuries, or being returned to the water comatose (Poiner and Harris 1996). Since the mandatory use of TEDs has been in effect, the annual bycatch of sea turtles in the NPF has dropped to less than 200 sea turtles per year, with a mortality rate of approximately 22% (based on recent years). This lower mortality rate also may be based on better sea turtle handling techniques adopted by the fleet. In general, loggerheads were the third most common sea turtle taken in this fishery.

Loggerheads also have been the most common turtle species captured in shark control programs in Pacific Australia (Kidston *et al*. 1992, Limpus 2009); however, the Western Australian demersal longline fishery for sharks has no recorded interaction with loggerheads. From 1998-2002, a total of 232 loggerheads were captured with 195 taken on drum lines and 37 taken in nets, both with a low level of direct mortality (Limpus 2009).

Other Manmade and Natural Impacts

As described in Section 5.1.5., other anthropogenic impacts, such as boat strikes and ingestion or entanglement in marine debris, likely apply to loggerheads in the Southeast Indo-Pacific Ocean. Similar to other areas of the world (see Section 5.1.5.), climate change and sea level rise have the potential to impact loggerheads in the Southeast Indo-Pacific Ocean. This includes beach erosion and loss from rising sea levels, skewed hatchling sex ratios from rising beach incubation temperatures, and abrupt disruption of ocean currents used for natural dispersal during the complex life cycle. Climate change impacts could have profound long term impacts on nesting populations in the Southeast Indo-Pacific Ocean, but it is not possible to quantify the potential impacts at this point in time.

Natural environmental events, such as cyclones and hurricanes, may affect loggerheads in the Southeast Indo-Pacific Ocean. In general, however, severe storm events are episodic and, although they may affect loggerhead hatchling production, the results are generally localized and they rarely result in whole-scale losses over multiple nesting seasons.

5.2.5 Southwest Indian Ocean DPS

5.2.5.1. Present or Threatened Destruction, Modification, or Curtailment of Its Habitat or Range

Terrestrial Zone
All nesting beaches within South Africa are within protected areas (Baldwin *et al*. 2003). In Mozambique, nesting beaches in the Maputo Special Reserve (approximately 60 km of nesting beach) and in the Paradise Islands are within protected areas (Baldwin *et al*. 2003, Costa *et al*. 2007). There are no protected areas for loggerheads in Madagascar (Baldwin *et al*. 2003).

Neritic/Oceanic Zone
As described in Section 5.1.1., threats to habitat in the loggerhead neritic and/or oceanic zones include fishing practices, channel dredging, sand extraction, marine pollution, and climate change. These threats also occur in the Southwest Indian Ocean. Fishing methods not only incidentally capture loggerheads (Section 5.1.5.), but also deplete invertebrate and fish populations and thus alter ecosystem dynamics. In many cases, loggerhead foraging areas coincide with fishing zones. Climate change also may result in future trophic changes, thus impacting loggerhead prey abundance and/or distribution.

5.2.5.2. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes

In the Southwest Indian Ocean, on the east coast of Africa, subsistence hunting by local people is a continued threat to loggerheads (Baldwin *et al*. 2003). Illegal hunting of marine turtles and egg harvesting remains a threat in Mozambique as well (Louro *et al*. 2006).

5.2.5.3. Disease or Predation

The potential effects of diseases and endoparasites, as described in Section 5.1.3., also exist for loggerheads found in the Southwest Indian Ocean. Side striped jackals (*Canis adustus)* and honey badgers (*Melivora capensis*) are known to depredate nests (Baldwin *et al*. 2003).

5.2.5.4. Inadequacy of Existing Regulatory Mechanisms

Unless identified to a specific geographic area, the regulatory mechanisms described in Section 5.1.4. also apply to sea turtles found in the Southwest Indian Ocean. The following additional instrument also applies to loggerheads found in this area.

Indian Ocean – South-East Asian Marine Turtle Memorandum of Understanding (IOSEA)
For information about this instrument, see section 5.2.1.4. for the North Pacific Ocean DPS.

Nairobi Convention for the Protection, Management and Development of the Marine and Coastal Environment of the Eastern African Region
For information about this instrument, see section 5.2.3.4. for the North Indian Ocean DPS.

5.2.5.5. Other Natural or Manmade Factors Affecting Its Continued Existence

Incidental Bycatch in Fishing Gear
The full extent of the threat of incidental capture of sea turtles in artisanal and commercial fisheries in the Southwest Indian Ocean is unknown. Sea turtles are caught in demersal and pelagic longlines, trawls, gillnets, and seines (Petersen 2005; Louro *et al*. 2006; Petersen *et al.* 2006, 2009; Costa *et al*. 2007; Fennessy and Isaksen 2007). There is evidence of significant historic bycatch from prawn fisheries, which may have depleted nesting populations long before nesting surveys were initiated in the 1990s (Baldwin *et al*. 2003).

Quantifying the magnitude of the threat of fisheries on loggerheads in the Southwest Indian Ocean is very difficult given the low level of observer coverage or investigations into bycatch conducted by countries that have large fishing fleets. Efforts have been made to quantify the effects of pelagic longline fishing on loggerheads globally (Lewison *et al*. 2004). While there were no turtle bycatch data available from the Southwest Indian Ocean to use in their assessment, extrapolations that considered bycatch data for the Pacific and Atlantic basins gave a conservative estimate of 6,000 loggerheads captured in the Indian Ocean in the year 2000. The effect of the longline fishery on loggerheads in the Indian Ocean is largely unknown (Lewison *et al*. 2004).

Other Manmade and Natural Impacts
As described in Section 5.1.5., other anthropogenic impacts, such as boat strikes and ingestion or entanglement in marine debris, likely apply to loggerheads in the Southwest Indian Ocean. Similar to other areas of the world (see Section 5.1.5.), climate change and sea level rise have the potential to impact loggerheads in the Southwest Indian Ocean. This includes beach erosion and loss from rising sea levels, skewed hatchling sex ratios from rising beach incubation temperatures, and abrupt disruption of ocean currents used for natural dispersal during the complex life cycle. Climate change impacts could have profound long term impacts on nesting populations in the Southwest Indian Ocean, but it is not possible to quantify the potential impacts at this point in time.

Natural environmental events, such as cyclones, tsunamis and hurricanes, may affect loggerheads in the Southwest Indian Ocean. In general, however, severe storm events are episodic and, although they may affect loggerhead hatchling production, the results are generally localized and they rarely result in whole-scale losses over multiple nesting seasons.

5.2.6. Northwest Atlantic Ocean DPS

The following information on threats pertains to all loggerhead turtles that may be found in the Northwest Atlantic. Because it is likely that oceanic juveniles from this DPS also are found in the Northeast Atlantic Ocean and in the Mediterranean Sea, the narratives for those regions should also be consulted.

5.2.6.1. Present or Threatened Destruction, Modification, or Curtailment of Its Habitat or Range

Terrestrial Zone
As described in Section 5.1.1., destruction and modification of loggerhead nesting habitat in the Northwest Atlantic results from coastal development and construction, placement of erosion control structures and other barriers to nesting, placement of nearshore shoreline stabilization structures, beachfront lighting, vehicular and pedestrian traffic, beach erosion, beach sand placement, removal of native vegetation, and planting of non-native vegetation (NMFS and FWS 2008).

In the southeastern U.S., numerous erosion control structures that create barriers to nesting have been constructed. The proportion of coastline that is armored is approximately 18% (239 km) in Florida (Clark 1992, Schroeder and Mosier 2000, Witherington *et al*. 2006), 9% (14 km) in Georgia (M. Dodd, GDNR, personal communication, 2000), 12% (29 km) in South Carolina (D. Griffin, SCDNR, personal communication, 2009), and 3% (9 km) in North Carolina (M. Godfrey, North Carolina Wildlife Resources Commission, 2009). These assessments of armoring extent do not include structures that are a barrier to sea turtle nesting but that do not fit the definition of armoring, such as dune crossovers, cabanas, sand fences, and recreational equipment.

In the Northwest Atlantic, jetties have been placed at many ocean inlets to keep transported sand from closing the inlet channel. Witherington *et al*. (2005) found a significant negative relationship between loggerhead nesting density and distance from the nearest of 17 ocean inlets on the Atlantic coast of Florida. The effect of inlets in lowering nesting density was observed both updrift and downdrift of the inlets, leading researchers to propose that beach instability from both erosion and accretion may discourage loggerhead nesting.

Numerous beaches in the southeastern U.S. are eroding due to both natural (e.g., storms, sea level changes, waves, shoreline geology) and anthropogenic (e.g., construction of armoring structures, groins, and jetties; coastal development; inlet dredging) factors. Such shoreline erosion leads to a loss of nesting habitat for sea turtles. As a result, beach nourishment is a frequent activity, and many beaches are on a periodic nourishment schedule. On severely eroded sections of beach, where little or no suitable nesting habitat previously existed, beach nourishment has been found to result in increased nesting (Ernest and Martin 1999). However, on most beaches in the southeastern U.S., nesting success typically declines for the first year or two following construction, even though more nesting habitat is available for turtles (Trindell *et al*. 1998, Ernest and Martin 1999, Herren 1999). Reduced nesting success on constructed beaches has been attributed to increased sand compaction, escarpment formation, and changes in beach profile (Nelson *et al*. 1987, Crain *et al*. 1995, Lutcavage *et al*. 1997, Steinitz *et al*. 1998, Ernest and Martin 1999, Rumbold *et al*. 2001).

Stormwater and other water source runoff from beachfront parking lots, building rooftops, roads, decks, and draining swimming pools adjacent to the beach is frequently discharged directly onto Northwest Atlantic beaches and dunes either by sheet flow, through stormwater collection system outfalls, or through small diameter pipes. These outfalls create localized erosion channels, prevent natural dune establishment, and wash out sea turtle nests (Florida Fish and Wildlife Conservation Commission, unpublished data). Contaminants contained in stormwater, such as oils, grease, antifreeze, gasoline, metals, pesticides, chlorine, and nutrients, are also discharged onto the beach.

In the U.S., vehicular driving is allowed on certain beaches in northeast Florida (Nassau, Duval, St. Johns, and Volusia counties), northwest Florida (Walton and Gulf counties), Georgia (Cumberland, Little Cumberland, and Sapelo Islands), North Carolina (Fort Fisher State Recreation Area, Carolina Beach, Freeman Park, Onslow Beach, Emerald Isle, Indian Beach/Salter Path, Pine Knoll Shores, Atlantic Beach, Cape Lookout National Seashore, Cape Hatteras National Seashore, Nag's Head, Kill Devil Hills, Town of Duck, and Currituck Banks), Virginia (Chincoteague NWR and Wallops Island), and Texas (the majority of beaches except for a highly developed section of South Padre Island and Padre Island National Seashore, San Jose Island, Matagorda Island, and Matagorda Peninsula where driving is not allowed or is limited to agency personnel, land owners, and/or researchers). Beach driving has been found to reduce the quality of loggerhead nesting habitat in several ways. In the southeastern U.S., vehicle ruts on the beach have been found to prevent or impede hatchlings from reaching the ocean following emergence from the nest (Mann 1977, Hosier *et al*. 1981, Cox *et al*. 1994, Hughes and Caine 1994). Sand compaction by vehicles has been found to hinder nest construction and hatchling emergence from nests (Mann 1977). Vehicle lights and vehicle movement on the beach after dark results in reduced habitat suitability, which can deter females from nesting and disorient hatchlings. Additionally, vehicle traffic on nesting beaches contributes to erosion, especially during high tides or on narrow beaches where driving is concentrated on the high beach and foredune.

As described in Section 5.1.1., coastal development contributes to habitat degradation by increasing light pollution. Reports of hatchling disorientation events in Florida alone describe several hundred nests each year and are likely to involve tens of thousands of hatchlings (Nelson *et al*. 2002). However, this number calculated from disorientation reports is likely a vast underestimate. Independent of these reports, Witherington *et al*. (1996) surveyed hatchling orientation at nests located at 23 representative beaches in six counties around Florida in 1993 and 1994 and found that, by county, approximately 10 to 30% of nests showed evidence of hatchlings disoriented by lighting. From this survey and from measures of hatchling production (Florida Fish and Wildlife Conservation Commission, unpublished data), the number of hatchlings disoriented by lighting in Florida is calculated in the range of hundreds of thousands per year.

Neritic/Oceanic Zone
As described in Section 5.1.1., threats to habitat in the loggerhead neritic and/or oceanic zones include fishing practices, channel dredging, sand extraction, oil exploration and development, marine pollution, and climate change. These threats also occur in the Northwest Atlantic Ocean. Fishing methods not only incidentally capture loggerheads (Section 5.1.5.), but also deplete invertebrate and fish populations and thus alter ecosystem dynamics. Although anthropogenic disruptions of natural ecological interactions have been difficult to discern, a few studies have been focused on the effects of these disruptions on loggerheads. For instance, Youngkin (2001) analyzed gut contents from hundreds of loggerheads stranded in Georgia over a 20-year period. His findings point to the probability of major effects on loggerhead diet from activities such as shrimp trawling and dredging. Lutcavage and Musick (1985) found that horseshoe crabs strongly dominated the diet of loggerheads in Chesapeake Bay in 1980-1981. Subsequently, fishermen began to harvest horseshoe crabs, primarily for use as bait in the eel and whelk pot fisheries, using several gear types. Atlantic coast horseshoe crab landings increased by an order of magnitude (0.5 to 6.0 million pounds) between 1980 and 1997, and in 1998 the Atlantic States Marine Fisheries Commission implemented a horseshoe crab fishery management plan to curtail catches (Atlantic States Marine Fisheries Commission 1998). The decline in horseshoe crab availability has apparently caused a diet shift in juvenile loggerheads, from predominantly horseshoe crabs in the early to mid-1980s to blue crabs in the late 1980s and early 1990s, to mostly finfish in the late 1990s and early 2000s (Seney 2003, Seney and Musick 2007). These data suggest that turtles are foraging in greater numbers in or around fishing gears and on discarded bycatch (Seney 2003).

Periodic dredging of sediments from navigational channels is carried out at large ports to provide for the passage of large commercial and military vessels. In addition, sand mining (dredging) for beach renourishment and construction projects occurs in the Northwest Atlantic along the U.S., Mexico, Central American, Colombia, and Venezuela coasts. Although directed studies have not been conducted, dredging activities, which occur regularly in the Northwest Atlantic, have the potential to destroy or degrade benthic habitats used by loggerheads. Channelization of inshore and nearshore habitat and the subsequent disposal of dredged material in the marine environment can destroy or disrupt resting or foraging grounds (including grass beds and coral reefs) and may affect nesting distribution by altering physical features in the marine environment (Hopkins and Murphy 1980). Oil exploration and development on live bottom areas may disrupt foraging grounds by smothering benthic organisms with sediments and drilling muds (Coston-Clements and Hoss 1983). The effects of benthic habitat alteration on loggerhead prey abundance and distribution, and the effects of these potential changes on loggerhead populations, have not been determined but are of concern. Climate change also may result in trophic changes, thus impacting loggerhead prey abundance and/or distribution.

5.2.6.2. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes

Deliberate hunting of loggerheads for their meat, shells, and eggs is reduced from previous exploitation levels, but still exists. In the Caribbean, 13 of 29 (45%) countries/territories allow the harvest of loggerheads (NMFS and FWS 2008, see Appendix 3). The loggerhead harvest in the Caribbean is generally restricted to the non-nesting season with the exception of St. Kitts and Nevis, where turtle harvest is allowed annually from March 1 through September 30, and the

Turks and Caicos Islands, where turtle harvest is allowed year-round. Most countries/territories that allow harvest have regulations that favor the harvest of large juveniles and adults, the most reproductively valuable members of the population. Exceptions include the Cayman Islands, which mandates maximum size limits, and Haiti and Trinidad and Tobago, which have no size restrictions. All North, Central, and South American countries in the Northwest Atlantic legislate complete protection of loggerheads from harvest in their territorial waters with the exception of Guyana. Despite national laws, in many countries the poaching of eggs and hunting of adults and juveniles still occurs at varying levels (NMFS and FWS 2008, see Appendix 3).

5.2.6.3. Disease or Predation

The potential effects of diseases and endoparasites, as described in Section 5.1.3., also exist for loggerheads found in the Northwest Atlantic. Viral diseases have not been documented in free-ranging loggerheads, with the possible exception of sea turtle fibropapillomatosis, which may have a viral etiology (Herbst and Jacobson 1995, George 1997). Although fibropapillomatosis reaches epidemic proportions in some wild green turtle populations, the prevalence of this disease in most loggerhead populations is thought to be small. An exception is Florida Bay where approximately 9.5% of the loggerheads captured exhibit fibropapilloma-like external lesions (B. Schroeder, NMFS, personal communication, 2006). Mortality levels and population-level effects associated with the disease are still unknown. Heavy infestations of endoparasites may cause or contribute to debilitation or mortality in sea turtles. Trematode eggs and adult trematodes were recorded in a variety of tissues including the spinal cord and brain of debilitated loggerheads during an epizootic in South Florida, USA, during late 2000 and early 2001. These endoparasites were implicated as a possible cause of the epizootic (Jacobson *et al*. 2006). Although many health problems have been described in wild populations through the necropsy of stranded turtles, the significance of diseases on the ecology of wild loggerhead populations is not known (Herbst and Jacobson 1995).

Predation of eggs and hatchlings by native and introduced species occurs on almost all nesting beaches throughout the Northwest Atlantic. The most common predators at the primary nesting beaches in the southeastern United States are ghost crabs (*Ocypode quadrata*), raccoons (*Procyon lotor*), feral hogs (*Sus scrofa*), foxes (*Urocyon cinereoargenteus* and *Vulpes vulpes*), coyotes (*Canis latrans*), armadillos (*Dasypus novemcinctus*), and red fire ants (*Solenopsis invicta*) (Stancyk 1982, Dodd 1988). In the absence of well managed nest protection programs, predators may take significant numbers of eggs; however, nest protection programs are in place at most of the major nesting beaches in the Northwest Atlantic.

Non-native vegetation has invaded many coastal areas and often outcompetes native plant species. Exotic vegetation may form impenetrable root mats that can invade and desiccate eggs, as well as trap hatchlings. The Australian pine (*Casuarina equisetifolia*) is particularly harmful to sea turtles. Dense stands have taken over many coastal areas throughout central and south Florida. Australian pines cause excessive shading of the beach that would not otherwise occur. Studies in Florida suggest that nests laid in shaded areas are subjected to lower incubation temperatures, which may alter the natural hatchling sex ratio (Marcus and Maley 1987, Schmelz and Mezich 1988, Hanson *et al*. 1998). Fallen Australian pines limit access to suitable nest sites and can entrap nesting females (Austin 1978, Reardon and Mansfield 1997). The shallow root network of these pines can interfere with nest construction (Schmelz and Mezich 1988). Davis and Whiting (1977) reported that nesting activity declined in Everglades National Park where dense stands of Australian pine took over native dune vegetation on a remote nesting beach. Beach vitex (*Vitex rotundifolia*) was introduced to the horticulture trade in the mid-1980s and is often sold as a "dune stabilizer." The plant is native to countries in the western Pacific and was introduced into the southeastern U.S. in the mid-1980s. Its presence on North Carolina and South Carolina beaches have a negative effect on sea turtle nesting as its dense mats interfere with sea turtle nesting and hatchling emergence from nests (Brabson 2006). This exotic plant is crowding out the native species, such as sea oats and bitter panicum, and can colonize large areas in just a few years. Sisal, or century plant, (*Agave americana*) is native to arid regions of Mexico. The plant was widely grown in sandy soils around Florida in order to provide fiber for cordage. It has escaped cultivation in Florida and has been purposely planted on dunes. Although the effects of sisal on sea turtle nesting are uncertain, thickets with impenetrable sharp spines are occasionally found on developed beaches.

Harmful algal blooms, such as a red tide, also impact loggerheads in the Northwest Atlantic. In Florida, USA, the species that causes most red tides is *Karenia brevis*, a dinoflagellate that produces a toxin (Florida Marine Research Institute 2003). During four red tide events along the west coast of Florida, sea turtle stranding trends indicated that these events were acting as a mortality factor (Redlow *et al*. 2003). Sea turtles that washed ashore alive during these red tide events displayed symptoms that were consistent with acute brevitoxicosis (e.g., uncoordinated and lethargic but otherwise robust and healthy in appearance) and completely recovered within days of being removed from the area of the red tide. The population level effects of these events are not yet known.

5.2.6.4. Inadequacy of Existing Regulatory Mechanisms

Unless identified to a specific geographic area, the regulatory mechanisms described in Section 5.1.4. also apply to sea turtles found in the Northwest Atlantic Ocean. The following additional instruments also apply to loggerheads found in this area.

United States Magnuson-Stevens Fishery Conservation and Management Act
For information about this instrument, see section 5.2.1.4. for the North Pacific Ocean DPS.

Inter-American Convention for the Protection and Conservation of Sea Turtles (IAC)
For information about this instrument, see section 5.2.1.4. for the North Pacific Ocean DPS.

Convention for the Protection and Development of the Marine Environment of the Wider Caribbean Region
Also called the Cartagena Convention, this instrument has been in place since 1986 and currently has 21 signatory states. Under this Convention, the component that may relate to loggerhead turtles is the Protocol Concerning Specially Protected Areas and Wildlife (SPAW) that has been in place since 2000. The goals of this protocol are to encourage Parties "to take all appropriate measures to protect and preserve rare or fragile ecosystems, as well as the habitat of depleted, threatened or endangered species, in the Convention area." All six sea turtle species in the Wider Caribbean are listed in Annex II of the protocol, which prohibits (a) the taking, possession or

killing (including, to the extent possible, the incidental taking, possession or killing) or commercial trade in such species, their eggs, parts or products, and (b) to the extent possible, the disturbance of such species, particularly during breeding, incubation, estivation, migration, and other periods of biological stress. Hykle (2002) believes that in view of the limited participation of Caribbean States in the aforementioned Convention on the Conservation of Migratory Species of Wild Animals, the provisions of the SPAW Protocol provide the legal support for domestic conservation measures that might otherwise not have been afforded. Additional information is available at http://www.cep.unep.org/cartagena-convention.

Convention for the Protection of the Marine Environment of the North-East Atlantic
This Convention is included under the Northwest Atlantic Ocean DPS because it is likely that oceanic juveniles from this DPS also are found in the Northeast Atlantic Ocean. For information about this instrument, see section 5.2.7.4. for the Northeast Atlantic Ocean DPS.

5.2.6.5. Other Natural or Manmade Factors Affecting Its Continued Existence

Incidental Bycatch in Fishing Gear
Bycatch of loggerheads in commercial and recreational fisheries in the Northwest Atlantic is a significant threat facing the species in this region. A variety of fishing gears that incidentally capture loggerhead turtles are employed including gillnets, trawls, hook and line, longlines, seines, dredges, pound nets, and various types of pots/traps. Among these, gillnets, longlines, and trawl gear contribute to the vast majority of bycatch mortality of loggerheads annually throughout their range in the Atlantic Ocean and Gulf of Mexico (Epperly *et al*. 1995; NMFS 2002a, 2004, 2007a, 2008a; Lewison *et al*. 2003, 2004; Richards 2007; NMFS, unpublished data). Considerable effort has been expended since the 1980s to document and address fishery bycatch, especially in the United States and Mexico. Observer programs have been implemented in some fisheries to collect turtle bycatch data, and efforts to reduce bycatch and mortality of loggerheads in certain fishing operations have been undertaken and implemented or partially implemented. These efforts include developing gear solutions to prevent or reduce captures or to allow turtles to escape without harm (e.g., TEDs, circle hooks and bait combinations), implementing time and area closures to prevent interactions from occurring (e.g., prohibitions on gillnet fishing along the mid-Atlantic coast during the critical time of northward migration of loggerheads, implementation of careful release protocols (e.g., requirements for careful release of turtles captured in longline fisheries), prohibitions of gillnetting in some U.S. state waters), and/or modifying gear (e.g., requirements to reduce mesh size in the leaders of pound nets in certain U.S. coastal waters to prevent entanglement).

The primary bycatch reduction focus in the Northwest Atlantic, since the 1978 ESA listing of the loggerhead, has been on bycatch reduction in shrimp trawls. The development of turtle excluder devices (TEDs) in the 1970s and the refinement of these devices over the past three decades has been a primary focus of loggerhead bycatch reduction efforts. The U.S. has required the use of TEDs throughout the year since the mid-1990s, with modifications required and implemented as necessary (NMFS 1987, 52 FR 24244; NMFS 1992, 57 FR 57348). Most notably, in 2003, NMFS implemented new requirements for TEDs in the shrimp trawl fishery to ensure that large loggerheads could escape through TED openings (NMFS 2003, 68 FR 8456). Significant effort has been expended to transfer this technology to other shrimping fleets in the Northwest Atlantic;

however, not all nations where loggerheads occur require the device be used. Enforcement of TED regulations is difficult and compliance is not believed to be complete. Because TEDs are not 100% effective, a significant number of loggerheads are estimated to still be killed annually in shrimp trawls throughout the Northwest Atlantic. In the U.S. Southeast food shrimp trawl fishery, NMFS estimated the annual mortality of loggerheads in the Gulf of Mexico and Southeast U.S. Atlantic Ocean as 3,948 individuals (95% CI, 1,221-8,498) (NMFS 2002a). Shrimping effort in the Southeast U.S. has reportedly declined, a revised estimate of annual loggerhead mortality for the Gulf of Mexico segment of the Southeast food shrimp trawl fishery is 647 individuals (NMFS, unpublished data).

Other trawl fisheries operating in Northwest Atlantic waters that are known to capture sea turtles include, but are not limited to, summer flounder, calico scallop, sea scallop, blue crab, whelk, cannonball jellyfish, horseshoe crab, and mid-Atlantic directed finfish trawl fisheries and the *Sargassum* fishery. In the U.S., the summer flounder fishery is the only trawl fishery (other than the shrimp fishery) with federally mandated TED use (in certain areas). Loggerhead annual bycatch estimates in 2004 and 2005 in U.S. mid-Atlantic scallop trawl gear ranged from 81 to 191 turtles, depending on the estimation methodology used (Murray 2007). Estimated average annual bycatch of loggerheads in other mid-Atlantic federally managed bottom otter trawl fisheries during 1996-2004 was 616 turtles (Murray 2006). The harvest of *Sargassum* by trawlers can result in incidental capture of post-hatchlings and habitat destruction (Schwartz 1988, Witherington 2002); this fishery is not currently active.

Dredge fishing gear is the predominant gear used to harvest sea scallops off the mid- and northeastern U.S. Atlantic coast. Sea scallop dredges are composed of a heavy steel frame and cutting bar located on the bottom part of the frame and a bag, made of metal rings and mesh twine, attached to the frame. The gear is fished along the bottom and weighs from 500-1,000 pounds (National Research Council 2002). Turtles can be struck and injured or killed by the dredge frame and/or captured in the bag where they may drown or be further injured or killed when the catch and heavy gear are dumped on the vessel deck. Total estimated bycatch of loggerhead turtles in the U.S. sea scallop dredge fishery operating in the mid-Atlantic region (New York to North Carolina) from June through November is on the order of several hundred turtles per year (Murray 2004, 2005, 2007). The impact of the sea scallop dredge fishery on loggerheads in U.S. waters of the Northwest Atlantic remains a serious concern.

Incidental take of oceanic-stage loggerheads in pelagic longline fisheries has recently received significant attention (Balazs and Pooley 1994; Bolten *et al*. 1994, 2000; Aguilar *et al*. 1995; Laurent *et al*. 1998; Long and Schroeder 2004; Watson *et al*. 2005). Large-scale commercial longline fisheries operate throughout the pelagic range of the Northwest Atlantic loggerhead, including the western Mediterranean. The largest size classes in the oceanic stage are the size classes impacted by the swordfish longline fishery in the Azores (Bolten 2003) and on the Grand Banks (Watson *et al*. 2005), and this is likely the case for other nation's fleets operating in the region, including but not limited to, the European Union, United States, Japan, and Taiwan. The demographic consequences relative to population recovery of the increased mortality of these size classes have been discussed (Crouse *et al*. 1987; see also Heppell *et al*. 2003 and Chaloupka 2003). Estimates derived from data recorded by the international observer program (IOP) suggest that thousands of mostly immature loggerheads have been captured in the Canadian pelagic longline fishery in the western North Atlantic since 1999 (Brazner and McMillan 2008). NMFS (2004a) estimates that 635 loggerheads (143 lethal) will be taken annually in the U.S. pelagic longline fishery.

Incidental capture of neritic-stage loggerheads in demersal longline fishing gear has also been documented. Richards (2007) estimated total annual bycatch of loggerheads in the Southeast U.S. Atlantic and U.S. Gulf of Mexico commercial directed shark bottom longline fishery from 2003-2005 as follows: 2003: 302 – 1,620 (CV 0.45); 2004: 95 – 591 (CV 0.49); 2005: 139-778 (CV 0.46). NMFS (2009) estimated the total number of captures of hardshell turtles in the U.S. Gulf of Mexico reef fish fishery (demersal longline fishery) from July 2006 – December 2008 as 861 turtles (95% CI 383 – 1934). These estimates are not comprehensive across this gear type (i.e., pelagic and demersal longline) throughout the Northwest Atlantic Ocean. Cumulatively, the bycatch and mortality of Northwest Atlantic loggerheads in longline fisheries is significant.

Gillnet fisheries may be the most ubiquitous of fisheries operating in the neritic range of the Northwest Atlantic loggerhead. Comprehensive estimates of bycatch in gillnet fisheries do not yet exist and, while this precludes a quantitative analysis of their impacts on loggerhead populations, the cumulative mortality of loggerheads in gillnet fisheries is likely high. In the U.S. mid-Atlantic, the average annual estimated bycatch of loggerheads from 1995 – 2006 was 350 turtles (CV= 0.20., 95% CI over the 12-year period: 234 to 504) (Murray in press). In the U.S., some states (e.g., South Carolina, Georgia, Florida, Louisiana, and Texas) have prohibited gillnets in their waters, but there remain active gillnet fisheries in other U.S. states, in U.S. federal waters, Mexican waters, Central and South America, and the Northeast Atlantic.

Pound nets are fixed gear composed of a series of poles driven into the bottom upon which netting is suspended. Pound nets basically operate like a trap with the pound constructed of a series of funnels leading to a bag that is open at the top, and a long leader of netting that extends from shallow to deeper water where the pound is located. In some configurations, the leader is suspended from the surface by a series of stringers or vertical lines. Sea turtles incidentally captured in the open top pound, which is composed of small mesh webbing, are usually safe from injury and may be released easily when the fishermen pull the nets (Mansfield *et al*. 2002). However, sea turtle mortalities have been documented in the leader of certain pound nets. Large mesh leaders (greater than 12-inch stretched mesh) may act as a gillnet, entangling sea turtles by the head or foreflippers (Bellmund *et al*. 1987) or may act as a barrier against which turtles may be impinged (NMFS, unpublished data). Nets with small mesh leaders (less than 8 inches stretched mesh) usually do not present a mortality threat to loggerheads, but some mortalities have been reported (Morreale and Standora 1998; Epperly *et al*. 2000, 2007; Mansfield *et al*. 2002). In 2002, the U.S. prohibited, in certain areas within the Chesapeake Bay and at certain times, pound net leaders having mesh greater than or equal to 12 inches and leaders with stringers (NMFS 2002b, 67 FR 41196). Subsequent regulations have further restricted the use of certain pound net leaders in certain geographic areas and established pound net leader gear modifications (NMFS 2004b, 69 FR 24997; NMFS 2006, 71 FR 36024).

Pots/traps are commonly used to target crabs, lobsters, whelk, and reef fishes. These traps vary in size and configuration, but all are attached to a surface float by means of a vertical line leading to the trap. Entanglement and mortality of loggerheads has been documented in various pot/trap fisheries in the U.S. Atlantic and Gulf of Mexico. Data from the U.S. Sea Turtle Stranding and Salvage Network indicate that 82 loggerheads (dead and rescued alive) were documented by the stranding network in various pot/trap gear from 1996-2005, of these approximately 30-40% were adults and the remainder juveniles (NMFS, unpublished data). Without intervention it is likely that the majority of the live, entangled turtles would die. Additionally, documented strandings represent only a portion of total interactions and mortality. Recently, a small number of loggerhead entanglements also have been recorded in whelk pot bridles in the U.S. Mid-Atlantic (Meredith Fagan, Virginia Institute of Marine Science, personal communication, 2008). However, no dedicated observer programs exist to provide estimates of take and mortality from pot/trap fisheries; therefore, comprehensive estimates of loggerhead interactions with pot/trap gear are not available, but the gear is widely used throughout the range of the DPS, and poses a continuing threat.

Other Manmade and Natural Impacts

Similar to other areas of the world (see Section 5.1.5.), climate change and sea level rise have the potential to impact loggerheads in the Northwest Atlantic Ocean.

Propeller and collision injuries from boats and ships are becoming more common in sea turtles. In the U.S. Atlantic, from 1997 to 2005, 14.9% of all stranded loggerheads were documented as having sustained some type of propeller or collision injuries (NMFS, unpublished data). The incidence of propeller wounds observed in sea turtles stranded in the U.S. has risen from approximately 10% in the late 1980s to a record high of 20.5% in 2004 (NMFS, unpublished data). In the U.S., propeller wounds are greatest in southeast Florida; during some years, as many as 60% of the loggerhead strandings found in these areas had propeller wounds (Florida Fish and Wildlife Conservation Commission, unpublished data). As the number of vessels increases, in concert with increased coastal development, especially in nearshore waters, propeller and vessel collision injuries are also expected to rise.

Several activities associated with offshore oil and gas production, including oil spills, water quality (operational discharge), seismic surveys, explosive platform removal, platform lighting, and noise from drillships and production activities, are known to impact loggerheads (National Research Council 1996; MMS 2000; Gregg Gitschlag, NMFS, personal communication, 2007; Viada *et al*. 2008). Currently, there are 3,443 federally regulated offshore platforms in the Gulf of Mexico dedicated to natural gas and oil production. Additional state-regulated platforms are located in state waters (Texas and Louisiana). There are currently no active leases off the Atlantic coast.

Oil spills also threaten loggerheads in the Northwest Atlantic. Two oil spills that occurred near loggerhead nesting beaches in Florida were observed to affect eggs, hatchlings, and nesting females. Approximately 350,000 gallons of fuel oil spilled in Tampa Bay in August 1993 and was carried onto nesting beaches in Pinellas County. Observed mortalities included 31 hatchlings and 176 oil covered nests; an additional 2,177 eggs and hatchlings were either exposed to oil or disturbed by response activities (FDEP *et al*. 1997). Another spill near the beaches of Broward County in August 2000 involved approximately 15,000 gallons of oil and tar (NOAA and FDEP 2002). Models estimated that approximately 1,500 to 2,000 hatchlings and 0 to 1 adults were injured or killed. Annually about 1% of all sea turtle strandings along the U.S.

east coast have been associated with oil, but higher rates of 3 to 6% have been observed in South Florida and Texas (Teas 1994, Rabalais and Rabalais 1980, Plotkin and Amos 1990).

In addition to the destruction or degradation of habitat described in Section 5.2.6.1., periodic dredging of sediments from navigational channels can also result in incidental mortality of sea turtles. Direct injury or mortality of loggerheads by dredges has been well documented in the southeastern and mid-Atlantic U.S. (National Research Council 1990). Solutions, including modification of dredges and time/area closures, have been successfully implemented to reduce mortalities and injuries in the U.S. (NMFS 1991, 1995, 1997; Nelson and Shafer 1996).

The entrainment and entrapment of loggerheads in saltwater cooling intake systems of coastal power plants has been documented in New Jersey, North Carolina, Florida, and Texas (Eggers 1989; National Research Council 1990; Carolina Power and Light Company 2003; FPL and Quantum Resources, Inc. 2005; Progress Energy Florida, Inc. 2003). Average annual incidental capture rates for most coastal plants from which captures have been reported amount to several turtles per plant per year. One notable exception is the St. Lucie Nuclear Power Plant located on Hutchinson Island, Florida. During the first 15 years of operation (1977-1991), an average of 128 loggerheads per year was captured in the intake canal with a mortality rate of 6.4%. During 1991-2005, loggerhead captures more than doubled (average of 308 per year), while mortality rates decreased to 0.3% per year (FPL and Quantum Resources, Inc. 2005).

Although not a major source of mortality, cold stunning of loggerheads has been reported at several locations in U.S., including Cape Cod Bay, Massachusetts (Still *et al*. 2000); Long Island Sound, New York (Meylan and Sadove 1986, Morreale *et al*. 1992); the Indian River system, Florida (Mendonça and Ehrhart 1982, Witherington and Ehrhart 1989); and Texas inshore waters (Hildebrand 1982, Shaver 1990).

Another natural factor that has the potential to affect recovery of loggerhead turtles is aperiodic hurricanes. In general, these events are episodic and, although they may affect loggerhead hatchling production, the results are generally localized and they rarely result in whole-scale losses over multiple nesting seasons. The negative effects of hurricanes on low-lying and/or developed shorelines may be longer-lasting and a greater threat overall.

Similar to other areas of the world (see Section 5.1.5.), climate change and sea level rise have the potential to impact loggerheads in the Northwest Atlantic. This includes beach erosion and loss from rising sea levels, repeated inundation of nests, skewed hatchling sex ratios from rising beach incubation temperatures, and abrupt disruption of ocean currents used for natural dispersal during the complex life cycle.

5.2.7. Northeast Atlantic Ocean DPS

The following information on threats pertains to all loggerhead turtles that may be found in the Northeast Atlantic. Because it is likely that oceanic juveniles from this DPS also are found in the Northwest Atlantic Ocean and in the Mediterranean Sea, the narratives for those regions should also be consulted.

5.2.7.1. Present or Threatened Destruction, Modification, or Curtailment of Its Habitat or Range

Terrestrial Zone
As described in Section 5.1.1., destruction and modification of loggerhead nesting habitat in the Northeast Atlantic result from coastal development and construction, placement of erosion control structures and other barriers to nesting, beachfront lighting, vehicular and pedestrian traffic, sand extraction, beach erosion, and beach pollution (NMFS and FWS 1998, Formia *et al*. 2003).

In the Northeast Atlantic, the only loggerhead nesting of note occurs in the Cape Verde Islands. The Cape Verde government's plans to develop Boa Vista Island, the location of the main nesting beaches, could increase the terrestrial threats to loggerheads (van Bogaert 2006). Sand extraction on Santiago Island, Cape Verde, may be responsible for the apparent decrease in nesting there (Loureiro 2008). Scattered and infrequent nesting occurs in western Africa, where much industrialization is located on the coast and population growth rates fluctuate between 0.8% (Cape Verde) and 3.8% (Côte D'Ivoire) (Abe *et al*. 2004, Tayaa *et al*. 2005). Land mines on some of the beaches of mainland Africa, within the reported historical range of nesting by loggerheads (e.g., the Western Sahara region) would be detrimental to nesters and are an impediment to scientific surveys of the region (Tiwari *et al*. 2001). Tiwari *et al*. (2001) noted a high level of human use of many of the beaches in Morocco – enough that any evidence of nesting activity would be quickly erased. Garbage litters many developed beaches (Formia *et al*. 2003). Erosion is a problem along the long stretches of high energy ocean shoreline of Africa and is further exacerbated by sand mining and harbor building (Formia *et al*. 2003); crumbling buildings claimed by the sea may present obstructions to nesting females.

Neritic/Oceanic Zone
As described in Section 5.1.1., threats to habitat in the loggerhead neritic and/or oceanic zones include fishing practices, marine pollution and climate change. These threats also occur in the Northeast Atlantic Ocean. Ecosystem alterations have occurred due to the tremendous human pressure on the environment in the region. Turtles, including loggerheads, usually are included in ecosystem models of the region (see Palomares and Pauly 2004). In the Canary Current Large Marine Ecosystem (LME), the area is characterized by the Global International Waters Assessment (GIWA) as severely impacted in the area of modification or loss of ecosystems or ecotones and health impacts, but these impacts are decreasing (http://www.lme.noaa.gov). The Celtic-Biscay Shelf LME is affected by alterations to the seabed, agriculture, and sewage (Valdés and Lavin 2002). The Gulf of Guinea has been characterized as severely impacted in the area of solid wastes by the GIWA; this and other pollution indicators are increasing (http://www.lme.noaa.gov). Marine pollution, such as oil and debris, has been shown to negatively impact loggerheads and represent a degradation of the habitat (Orós *et al*. 2005, 2009; Calabuig Miranda and Liria Loza 2007). Climate change also may result in future trophic changes, thus impacting loggerhead prey abundance and/or distribution.

Additionally, fishing is a major source of ecosystem alteration of the neritic and oceanic habitats of loggerhead turtles in the region. Fishing effort off the western African coast is increasing and record low biomass has been recorded for exploited resources, representing a 13X decline in biomass since 1960 (see Palomares and Pauly 2004). Throughout the North Atlantic, fishery landings fell by 90% during the 20th century, foreboding a trophic cascade and a change in food-web competition (Pauly *et al*. 1998, Christensen *et al*. 2003). For a description of the exploited marine resources in the region, see Lamboeuf (1997). The Celtic-Biscay Shelf LME, the Iberian Coastal Ecosystem LME, the Canary Current LME, and the Guinea Current LME all are severely overfished, and effort now is turning to a focus on pelagic fisheries, whereas historically there were demersal fisheries. The impacts continue to increase in the Guinea Current LME despite efforts throughout the region to reduce fishing pressure (http://www.lme.noaa.gov).

The threats to bottom habitat for loggerheads previously described in this document include modification of the habitat through bottom trawling. Trawling occurs off the European coast and the area off Northwest Africa is one of the most intensively trawled areas in the world (Zeeberg *et al*. 2006). Trawling has been banned in the Azores, Madeira, and Canary Islands to protect cold-water corals (Lutter 2005). Although illegal, trawling also occurs in the Cape Verde Islands (López-Jurado *et al*. 2003). The use of destructive fishing practices, such as explosives and toxic chemicals, has been reported in the Canary Current area, causing serious damage to both the resources and the habitat (Tayaa *et al*. 2005).

5.2.7.2. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes

Deliberate hunting of loggerheads for their meat, shells, and eggs still exists and remains the most serious threat facing nesting turtles in the Northeast Atlantic. Historical records indicate turtles were harvested throughout Macaronesia (see López-Jurado 2007). Intensive exploitation has been cited for the extirpation of the loggerhead nesting colony in the Canary Islands (López-Jurado 2007), and heavy human predation on nesting and foraging animals occurred on Santiago Island, Cape Verde, the first in the Archipelago to be settled (Loureiro 2008), as well as on Sal and Sao Vicente islands (López-Jurado 2007). Nesting loggerheads and eggs are still harvested at Boa Vista, Cape Verde (Cabrera *et al*. 2000, López-Jurado *et al*. 2003). In 2007, over 1,100 (36%) of the nesting turtles were hunted, which is about 15% of the estimated adult female population (Marco *et al.* in press). In 2008, the military protected one of the major nesting beaches on Boa Vista where in 2007 55% of the mortality had occurred; with the additional protection, only 17% of the turtles on that beach were slaughtered (Roder *et al.* in press). On Sal Island, 11.5% of the emergences on unprotected beaches ended with mortality, whereas mortality was 3% of the emergences on protected beaches (Cozens *et al.* in press). The slaughter of nesting turtles is a problem wherever turtles nest in the Cape Verde Islands and may approach 100% in some places (C. Roder, Turtle Foundation, Münsing, Germany, personal communication, 2009; Cozens in press). The meat and eggs are consumed locally as well as traded among the archipelago (C. Roder, Turtle Foundation, Münsing, Germany, personal communication, 2009). Hatchlings are collected on Sal Island, but this activity appears to be rare on other islands of the archipelago (J. Cozens, SOS Tartarugas Cabo Verde, Santa Maria, Sal Island, Cape Verde, personal communication, 2009). Additionally, free divers target turtles for consumption of meat, often selectively taking large males (López-Jurado *et al*. 2003). Turtles are harvested along the African coast and, in some areas, are considered a significant source of food and income due to the poverty of many residents along the African coast (Formia *et al*. 2003). Loggerhead carapaces are sold in markets in Morocco and Western Sahara (Fretey 2001, Tiwari *et al*. 2001, Benhardouze *et al*. 2004).

5.2.7.3. Disease or Predation

The potential effects of diseases and endoparasites, as described in Section 5.1.3., also exist for loggerheads found in the Northeast Atlantic Ocean. Spontaneous diseases documented in the Northeast Atlantic include pneumonia, hepatitis, meningitis, septicemic processes, and neoplasia (Orós *et al*. 2005). Pneumonia could result from the aspiration of water from forced submergence in fishing gear. The authors also reported nephritis, esophagitis, nematode infestation, and eye lesions. Fibropapillomatosis does not appear to be an issue in the Northeast Atlantic.

Nest depredation by ghost crabs (*Ocypode cursor*) occurs in Cape Verde (López-Jurado *et al*. 2000). The ghost crabs feed on both eggs and hatchlings. Arvy *et al*. (2000) reported predation of loggerhead eggs in two nests in Mauritania by golden jackals (*Canis aureus*); a loggerhead turtle creating a third nest also had been killed, with meat and eggs eaten, but the predator was not identified.

Loggerheads in the Northeast Atlantic also may be impacted by harmful algal blooms, which have been reported infrequently in the Canary Islands and the Iberian Coastal LME (Ramos *et al*. 2005, Akin-Oriola *et al*. 2006, Amorim and Dale 2006, Moita *et al*. 2006; http://www.lme.noaa.gov).

5.2.7.4. Inadequacy of Existing Regulatory Mechanisms

Unless identified to a specific geographic area, the regulatory mechanisms described in Section 5.1.4. also apply to sea turtles found in the Northeast Atlantic Ocean, although not all nations in the region are signatory to them all (e.g., Western Sahara is not signatory to the CMS/Bonn Convention). The following additional instruments and regulation also apply to loggerheads found in this area.

Convention for the Protection of the Marine Environment of the North-East Atlantic
Also called the OSPAR Convention, this 1992 instrument combines and updates the 1972 Oslo Convention against dumping waste in the marine environment and the 1974 Paris Convention addressing marine pollution stemming from land-based sources. The convention is managed by the OSPAR Commission, which is comprised of representatives from 15 signatory nations (Belgium, Denmark, Finland, France, Germany, Iceland, Ireland, Luxembourg, The Netherlands, Norway, Portugal, Spain, Sweden, Switzerland, and United Kingdom), as well as the European Commission, representing the European Community. The mission of the OSPAR Convention "…is to conserve marine ecosystems and safeguard human health in the North-East Atlantic by preventing and eliminating pollution; by protecting the marine environment from the adverse effects of human activities; and by contributing to the sustainable use of the seas." Loggerheads are included on the OSPAR List of Threatened and/or Declining Species and Habitats, which is used by the OSPAR Commission for setting priorities for work on the conservation and protection of marine biodiversity. Additional information is available at http://www.ospar.org.

Convention on the Conservation of European Wildlife and Natural Habitats
Also known as the Bern Convention, the goals of this instrument are to conserve wild flora and fauna and their natural habitats, especially those species and habitats whose conservation requires the cooperation of several States, and to promote such cooperation. The Convention was enacted in 1982 and currently includes 48 European and African States and the European Union. Sea turtles are included on the "strictly protected" list. According to Hykle (2002), while the Convention's "innovative approach to holding States to account for their implementation of the Convention is laudable, and has certainly drawn attention to issues of species and habitat protection, its efficacy in relation to particular marine turtle cases that have been deliberated for many years is debatable." Additional information is available at http://conventions.coe.int/Treaty/EN/Treaties/Html/104.htm.

African Convention on the Conservation of Nature and Natural Resources (Algiers Convention)
Adopted in September 1968, the contracted states were "to undertake to adopt the measures necessary to ensure conservation, utilization and development of soil, water, floral and faunal resources in accordance with scientific principles and with due regard to the best interests of the people". It was followed by the 1972 Stockholm Conference on the Human Environment and led to the establishment of environmental ministries in African nations and the creation of the United Nations Environment Programme (UNEP) headquartered in Nairobi. The Algiers Convention recently has undergone revision (not yet in force) and its objectives are to enhance environmental protection, foster conservation and sustainable use of natural resources, and harmonize and coordinate policies in these fields with a view to achieving ecologically rational, economically sound, and socially acceptable development policies and programs. Additional information is available at http://www.unep.ch/regionalseas/legal/afr.htm.

Convention on the Conservation of Migratory Species of Wild Animals, Memorandum of Understanding Concerning Conservation Measures for Marine Turtles of the Atlantic Coast of Africa (Abidjan Memorandum)
This MOU was concluded under the auspices of the Convention on the Conservation of Migratory Species of Wild Animals (CMS) and became effective in 1999. The MOU area covers 26 Range States along the Atlantic coast of Africa extending approximately 14,000 km from Morocco to South Africa. The goal of this MOU is to improve the conservation status of marine turtles along the Atlantic Coast of Africa. It aims at safeguarding six marine turtle species – including the loggerhead turtle – that are estimated to have rapidly declined in numbers during recent years due to excessive exploitation (both direct and incidental) and the degradation of essential habitats. This includes the projection of hatchlings through adults with particular attention paid to the impacts of fishery bycatch and the need to include local communities in the development and implementation of conservation activities. However, despite this agreement, killing of adult turtles and harvesting of eggs remains rampant in many areas along the Atlantic African coast. Additional information is available at http://www.cms.int/species/africa_turtle/AFRICAturtle_bkgd.htm.

The Convention for the Co-operation in the Protection and Development of the Marine and Coastal Environment of the West and Central African Region (Abidjan Convention)
The Abidjan Convention covers the marine environment, coastal zones, and related inland waters from Mauritania to Namibia. The Abidjan Convention countries are Angola, Benin, Cameroon,

Cape Verde, Congo, Cote d'Ivoire, Democratic Republic of Congo, Equatorial Guinea, Gabon, Gambia, Ghana, Guinea, Guinea-Bissau, Liberia, Mauritania, Namibia, Nigeria, Sao Tome and Principe, Senegal, Sierra Leone, and Togo. The Abidjan Convention is an agreement for the protection and management of the marine and coastal areas that highlights sources of pollution, including pollution from ships, dumping, land-based sources, exploration and exploitation of the sea-bed, and pollution from or through the atmosphere. The Convention also identifies where co-operative environmental management efforts are needed. These areas of concern include coastal erosion, specially protected areas, combating pollution in cases of emergency, and environmental impact assessment. The Action Plan and the Abidjan Convention were adopted by the Governments in 1981; the Convention entered into force in 1984. Western Sahara and Morocco are not signatories of the Abidjan Convention.

Accra Declaration of the Ministerial Committee of the Gulf of Guinea Large Marine Ecosystem (GOG-LME)-1998 Abuja Declaration of the Guinea Current Large Marine Ecosystem Project-2006
In 1998, the environmental ministers of Cote d'Ivoire, Ghana, Togo, Benin, Nigeria, and Cameroon signed the Accra Declaration to strengthen regional capacity to prevent and correct pollution in the LME and prevent and correct degradation of critical habitats. The ministers identified the living resources and management problems in the area. The countries decided on a detailed survey of industries, defined regional effluent standards, instituted community based mangrove restoration activities, and created a campaign for the reduction, recovery, recycling, and re-use of industrial wastes. In 2006, the Guinea Current LME Project expanded the project scope to 10 neighboring countries (Guinea-Bissau, Guinea, Sierra Leone, Liberia, Sao Tome and Principe, Equatorial Guinea, Gabon, Congo Brazzaville, Congo-Kinshasa, and Angola).

Council Regulation (EC) No. 1239/98 of 8 June 1998 Amending Regulation (EC) No. 894/97 Laying Down Certain Technical Measures for the Conservation of Fishery Measures (Council of the European Union)
This measure banned the use of driftnets by 1 January 2002 for European fleets. Fleets from other nations fishing in international waters can still use driftnets.

5.2.7.5. Other Natural or Manmade Factors Affecting Its Continued Existence

Incidental Bycatch in Fishing Gear
Loggerhead turtles strand throughout the Northeast Atlantic (Fretey 2001; Tiwari *et al*. 2001; Duguy *et al*. 2004, 2005; Witt *et al*. 2007), and there are indications that the turtles become entangled in nets and monofilament and swallow hooks in the region (Orós *et al*. 2005, Calabuig Miranda and Liria Loza 2007). On the European coasts, most stranded loggerheads are small (mean < 30 cm SCL), but a few are greater than 60 cm SCL (Witt *et al*. 2007). Similarly, Tiwari *et al*. (2001) and Benhardouze *et al*. (2004) indicated that the animals they viewed in Morocco and Western Sahara were small juveniles and preliminary genetic analyses of stranded turtles indicate that they are of western Atlantic origin (M. Tiwari, NMFS, and A. Bolten, University of Florida, unpublished data), whereas Fretey (2001) reported that loggerheads captured and stranded in Mauritania were both juvenile and adult-sized animals.

Incidental capture of sea turtles in artisanal and commercial fisheries is a threat to the survival of loggerheads in the Northeast Atlantic. Sea turtles may be caught in a multitude of gears deployed in the region: pelagic and demersal longlines, drift and set gillnets, bottom and mid-water trawling, weirs, haul and purse seines, pots and traps, cast nets, and hook and line gear (see Pascoe and Gréboval 2003, Bayliff *et al*. 2005, Tayaa *et al*. 2005, Dossa *et al*. 2007). Fishing effort off the western African coast has been increasing (see Palomares and Pauly 2004). Impacts continue to increase in the Guinea Current LME, but, in contrast, the impacts are reported to be decreasing in the Canary Current LME (http://www.lme.noaa.gov). Throughout the region, fish stocks are depleted and management authorities are striving to reduce the fishing pressure.

In the Northeast Atlantic, loggerheads, particularly the largest size classes in the oceanic environment (most of which are small juveniles), are captured in surface longline fisheries targeting swordfish (*Ziphias gladius*) and tuna (*Thunnus spp*.) (Ferreira *et al*. 2001, Bolten 2003). Bottom longlines in Madeira Island targeting black-scabbard (*Aphanopus carbo*) capture and kill small juvenile loggerhead turtles as the fishing depth does not allow hooked turtles to surface (Dellinger and Encarnaçâo 2000, Delgado *et al*. in press).

In United Kingdom and Irish waters, loggerhead bycatch is uncommon but has been noted in pelagic driftnet fisheries (Pierpoint 2000, Rogan and Mackey 2007). Loggerheads have not been captured in pelagic trawls, demersal trawls, or gillnets in United Kingdom and Irish waters (Pierpoint 2000), but have been captured in nets off France (Duguy 2004, 2005).

International fleets of trawl fisheries operate in Mauritania and have been documented to capture sea turtles, including loggerheads (Zeeberg *et al*. 2006). Despite being illegal, trawling occurs in the Cape Verde Islands and has the potential to capture and kill loggerhead turtles; one piece of abandoned trawl net washed shore with eight live and two dead loggerheads (López-Jurado *et al*. 2003). Longlines, seines, and hook and line have been documented to capture loggerheads 35-73 cm SCL off the northwestern Moroccan coast (Benhardouze 2004).

Other Manmade and Natural Impacts

As described in Section 5.1.5., other anthropogenic impacts, such as boat strikes and ingestion or entanglement in marine debris, also apply to loggerheads in the Northeast Atlantic. Propeller and boat strike injuries have been documented in the Northeast Atlantic (Orós *et al*. 2005, Calabuig Miranda and Liria Loza 2007). Exposure to crude oil is also of concern. Loggerhead strandings in the Canary Islands have shown evidence of hydrocarbon exposure as well as ingestion of marine debris, such as plastic and monofilament (Orós *et al*. 2005, Calabuig Miranda and Liria Loza 2007), and in the Azores and elsewhere plastic debris is found both on the beaches and floating in the waters (Barrerios and Barcelos 2001, Tiwari *et al*. 2001). Pollution from heavy metals is a concern for the seas around the Iberian Peninsula (European Environmental Agency 1998) and in the Guinea Current LME (Abe *et al*. 2004). Bioaccumulation of metals in loggerheads has been measured in the Canary Islands and along the French Atlantic Coast (Caurant *et al*. 1999, Torrent *et al*. 2004). However, the consequences of long term exposure to heavy metals are unknown (Torrent *et al*. 2004). The effects of pollution on the ecosystem are addressed in Section 5.2.7.1.

Natural environmental events, such as climate change, could impact affect loggerheads in the Northeast Atlantic. Similar to other areas of the world (see Section 5.1.5.), climate change and sea level rise have the potential to impact loggerheads in the Northeast Atlantic, and the changes may be further exacerbated by the burning of fossil fuels and deforestation. These effects range from flooding of nesting beaches, shifts in ocean currents, ecosystem shifts in prey distribution and abundance, and a shift in the sex ratio of the population if rookeries do not migrate concurrently (e.g., northward in the case of global warming) or if nesting phenology does not change (see Doody *et al*. 2006). Tropical and sub-tropical storms occasionally strike the area and could have a negative impact on nesting, although such an impact would be of limited duration.

5.2.8. Mediterranean Sea DPS

The following information on threats pertains to all loggerhead turtles that may be found in the Mediterranean Sea. Because it is likely that oceanic juveniles from this DPS also are found in the Northeast Atlantic Ocean, the narrative for that region should also be consulted.

5.2.8.1. Present or Threatened Destruction, Modification, or Curtailment of Its Habitat or Range

Terrestrial Zone
In the Mediterranean, some areas known to host nesting activity in the past have been lost to turtles (e.g., Malta) or severely degraded (e.g., Israel) (Margaritoulis *et al*. 2003). As described in Section 5.1.1., destruction and modification of loggerhead nesting habitat in the Mediterranean result from coastal development and construction, placement of erosion control structures and other barriers to nesting, beachfront lighting, vehicular and pedestrian traffic, sand extraction, beach erosion, beach sand placement, beach pollution, removal of native vegetation, and planting of non-native vegetation (Baldwin 1992, Margaritoulis *et al*. 2003). These activities may directly impact the nesting success of loggerheads and survivability of eggs and hatchlings. Nesting in the Mediterranean almost exclusively occurs in the Eastern basin, with the main concentrations found in Cyprus, Greece, Turkey, and Libya (Margaritoulis *et al*. 2003, Laurent *et al*. 1999); therefore, the following threats to the nesting habitat are concentrated in these areas.

The Mediterranean experiences a large influx of tourists during the summer months, coinciding with the nesting season. Margaritoulis *et al*. (2003) state that extensive urbanization of the coastline, largely a result of tourism and recreation, is likely the most serious threat to loggerhead nesting areas. The large numbers of tourists that use Mediterranean beaches result in an increase in umbrellas, chairs, garbage, and towels, as well as related hotels, restaurants, and stationary (e.g., street lights, hotels) and moving (e.g., cars) lighting, all which can impact sea turtle nesting success (Demetropoulos 2000). Further, the eastern Mediterranean is exposed to high levels of pollution and marine debris, in particular the nesting beaches of Cyprus, Turkey, and Egypt (Camiñas 2004).

Construction and infrastructure development also have the potential to alter nesting beaches and subsequently impact nesting success. The construction of new buildings on or near nesting beaches has been a problem in Greece and Turkey (Camiñas 2004). The construction of a jetty and waterworks around Mersin, Turkey, has contributed significantly to the continuous loss of adjacent beach (Camiñas 2004).

Beach erosion and sand extraction also pose a problem for sea turtle nesting sites. The noted decline of the nesting population at Rethymno, Island of Crete, Greece, is partly attributed to beach erosion caused by construction on the high beach and at sea (e.g., groins) (Margaritoulis *et al*. 2009). A 2001 survey of Lebanese nesting beaches found severe erosion on beaches where previous nesting had been reported, and in some cases the beaches had disappeared completely (Venizelos *et al*. 2005). Definitive causes of this erosion were found to be sand extraction, offshore sand dredging, and sediment removal from river beds for construction and military purposes. Beach erosion also may occur from natural changes, with the same deleterious effects to loggerhead nesting. On Patara, Turkey, beach erosion and subsequent inundation by waves and shifting sand dunes are responsible for about half of all loggerhead nest losses (Camiñas 2004). Erosion can further be exacerbated when native dune vegetation, which enhances beach stability and acts as an integral buffer zone between land and sea, is degraded or destroyed. This in turn often leaves insufficient nesting opportunities above the high tide line, and nests may be washed out. In contrast, the planting or invasion of less stabilizing, non-native plants can lead to increased erosion and degradation of suitable nesting habitat. Finally, sand extraction has been a serious problem on Mediterranean nesting beaches, especially in Turkey (Türkozan and Baran 1996), Cyprus (Godley *et al*. 1996, Demetropoulos and Hadjichristophorou 1989), and Israel (Levy 2003).

While the most obvious effect of nesting beach destruction and modification may be to the existence of the actual nests, hatchlings are also threatened by habitat alteration. In the Mediterranean, disorientation of hatchlings due to artificial lighting has been recorded mainly in Greece (Rees 2005, Margaritoulis *et al*. 2007, Margaritoulis *et al*. 2009), Turkey (Türkozan and Baran 1996), and Lebanon (Newbury *et al*. 2002). Additionally, vehicle traffic on nesting beaches may disrupt the natural beach environment and contribute to erosion, especially during high tides or on narrow beaches where driving is concentrated on the high beach and foredune. On Zakynthos Island in Greece, Venizelos *et al*. (2006) reported that vehicles drove along the beach and sand dunes throughout the tourist season on East Laganas and Kalamaki beaches, leaving deep ruts in the sand, disturbing sea turtles trying to nest, and impacting hatchlings trying to reach the sea.

Neritic/Oceanic Zone

As described in Section 5.1.1., threats to habitat in the loggerhead neritic and/or oceanic zones include fishing practices, channel dredging, sand extraction, marine pollution, and climate change. These threats also occur in the Mediterranean.

Trawling occurs throughout the Mediterranean, most notably in areas off Albania, Algeria, Croatia, Egypt, France, Greece, Italy, Libya, Morocco, Slovenia, Spain, Tunisia, and Turkey (Gerosa and Casale 1999, Camiñas 2004, Casale 2008). This fishing practice has the potential to destroy bottom habitat in these areas. Fishing methods affect neritic zones by not only impacting bottom habitat and incidentally capturing loggerheads but also depleting fish populations, and thus altering ecosystem dynamics. For example, depleted fish stocks in Zakynthos, Greece, likely contributed to predation of adult loggerheads by monk seals (*Monachus monachus*)

(Margaritoulis *et al*. 1996). Further, by depleting fish populations, the trophic dynamics will be altered, which may then in turn affect the ability of loggerheads to find prey resources. If loggerheads are not able to forage on the necessary prey resources, their long term survivability may be impacted. Climate change also may result in future trophic changes, thus impacting loggerhead prey abundance and/or distribution.

Marine pollution, including direct contamination and structural habitat degradation, can affect loggerhead neritic and oceanic habitat. As the Mediterranean is an enclosed sea, organic and inorganic wastes, toxic effluents, and other pollutants rapidly affect the ecosystem (Camiñas 2004). The Mediterranean has been declared a "special area" by the MARPOL Convention, in which deliberate petroleum discharges from vessels are banned, but numerous repeated offenses are still thought to occur (Pavlakis *et al*. 1996). Some estimates of the amount of oil released into the region are as high as 1,200,000 metric tons (Alpers 1993). Direct oil spill events also occur as happened in Lebanon in 2006 when 10,000 to 15,000 tons of heavy fuel oil spilled into the eastern Mediterranean (United Nations Environment Programme 2007).

Destruction and modification of loggerhead habitat also may occur as a result of other activities. For example, underwater explosives have been identified as a key threat to loggerhead habitat in internesting areas in the Mediterranean (Margaritoulis *et al*. 2003). Further, the Mediterranean is a site of intense tourist activity, and corresponding boat anchoring also may impact loggerhead habitat in the neritic environment.

5.2.8.2. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes

Mediterranean turtle populations were subject to severe exploitation until the mid-1960s (Margaritoulis *et al*. 2003). Deliberate hunting of loggerheads for their meat, shells, and eggs is reduced from previous exploitation levels, but still exists. For example, Nada and Casale (2008) found that egg collection (for individual consumption) still occurs in Egypt. In some areas of the Mediterranean, like on the Greek Island of Zakynthos, nesting beaches are protected (Panagopoulou *et al*. 2008), so egg harvest by humans in those areas is likely negligible.

Exploitation of juveniles and adults still occurs in some Mediterranean areas. In Tunisia, clandestine trade for local consumption is still recorded, despite prohibition of the sale of turtles in fish markets in 1989 (Laurent *et al*. 1996). In Egypt, turtles are sold in fish markets despite prohibitive laws; of 71 turtles observed at fish markets in 1995 and 1996, 68% were loggerheads (Laurent *et al*. 1996). Nada (2001) reported 135 turtles (of which 85% were loggerheads) slaughtered at the fish market of Alexandria in six months (December 1998-May 1999). Based on observed sea turtle slaughters in 1995 and 1996, Laurent *et al*. (1996) estimated that several thousand sea turtles were probably killed each year in Egypt. More recently, a study found that the open selling of sea turtles in Egypt generally has been curtailed due to enforcement efforts, but a high level of intentional killing for the black market or for direct personal consumption still exists (Nada and Casale 2008). Given the high numbers of turtles caught in this area, several hundred turtles are currently estimated to be slaughtered each year in Egypt (Nada and Casale 2008). This estimate likely includes both juvenile and adult loggerheads, as Egyptian fish markets have been documented selling different sized sea turtles. While the mean sea turtle size was 65.7 cm CCL (range 38-86.3 cm CCL; n=48), 37.5% of observed loggerhead samples were greater than 70 cm CCL (Laurent *et al*. 1996).

5.2.8.3. Disease or Predation

The potential effects of diseases and endoparasites, as described in Section 5.1.3., also exist for loggerheads found in the Mediterranean. Endoparasites in loggerheads have been studied in the western Mediterranean. While the composition of the gastrointestinal community of sea turtles is expected to include digeneans, nematodes, and aspidogastreans, loggerheads in the Mediterranean were found to harbor only four digenean species typical of marine turtles (Aznar *et al*. 1998). There have been no records of fibropapillomatosis in the Mediterranean. While there is the potential for disease in this area, information on the prevalence of such disease is lacking.

In the Mediterranean Sea, loggerhead hatchlings and eggs are subject to depredation by wild canids (i.e., foxes (*Vulpes vulpes*), golden jackals (*Canis aureus*)), feral/domestic dogs, and ghost crabs (*Ocypode cursor*) (Margaritoulis *et al*. 2003). Predators have caused the loss of 48.4% of loggerhead clutches at Kyparissia Bay, Greece (Margaritoulis 1988), 70-80% at Dalyan Beach, Turkey (Erk'akan 1993), 36% (includes green turtle clutches) in Cyprus (Broderick and Godley 1996), and 44.8% in Libya (Laurent *et al*. 1995). A survey of the Syrian coast in 1999 found 100% nest predation, mostly due to stray dogs and humans (Venizelos *et al*. 2005). Loggerhead eggs are also depredated by insect larvae in Cyprus (McGowan *et al*. 2001), Turkey (Özdemir *et al*. 2004), and Greece (Lazou and Rees 2006). Ghost crabs have been reported preying on loggerhead hatchlings in northern Cyprus and Egypt, suggesting 66% of emerging hatchlings succumb to this mortality source (Simms *et al*. 2002). Predation also has been influenced by anthropogenic sources. On Zakynthos, Greece, a landfill site next to loggerhead nesting beaches has resulted in an artificially high level of seagulls (*Larus* spp.), which results in increased predation pressure on hatchlings (Panagopoulou *et al*. 2008). Planting of non-native plants also can have a detrimental effect on nests in the form of roots invading eggs (e.g., tamarisk tree (*Tamarix* spp.) roots invading eggs in Zakynthos, Greece; Margaritoulis *et al*. 2007).

Predation on adult and juvenile loggerheads has also been documented in the Mediterranean. Predation of nesting loggerheads by golden jackals has been recorded in Turkey (Peters *et al*. 1994). During a 1995 survey of loggerhead nesting in Libya, two nesting females were found killed by carnivores, probably jackals (Laurent *et al*. 1997). Off the sea turtle nesting beach of Zakynthos, Greece, adult loggerheads were found being predated upon by Mediterranean monk seals (*Monachus monachus*). Of the eight predated turtles observed or reported, 62.5% were adult males (Margaritoulis *et al*. 1996). Further, stomach contents were examined from 24 Mediterranean white sharks (*Carcharodon carcharias*), and 17% contained remains of marine turtles, including two loggerheads, one green, and one unidentifiable turtle (Fergusson *et al*. 2000). One of the loggerhead turtles ingested was a juvenile with a carapace length of approximately 60 cm (length not reported as either SCL or CCL). Fergusson *et al*. (2000) report that white shark interactions with sea turtles are likely rare east of the Ionian Sea, and while the impact of shark predation on turtle populations is unknown, it is probably small compared to other sources of mortality.

The Mediterranean is a low-productivity body of water, with high water clarity as a result. However, harmful algal blooms do occur in this area (e.g., off Algeria in 2002), and the problem is particularly acute in enclosed ocean basins such as the Mediterranean. In the northern Adriatic Sea, fish kills have occurred as a result of noxious phytoplankton blooms and anoxic conditions (Mediterranean Sea LME). While fish may be more susceptible to these harmful algal blooms, loggerheads in the Mediterranean also may be impacted by such noxious or toxic phytoplankton to some extent.

5.2.8.4. Inadequacy of Existing Regulatory Mechanisms

Unless identified to a specific geographic area, the regulatory mechanisms described in Section 5.1.4. also apply to sea turtles found in the Mediterranean Sea. The following additional instruments and regulation also apply to loggerheads found in this area.

Protocol Concerning Specially Protected Areas and Biological Diversity in the Mediterranean
This Protocol is under the auspices of the Barcelona Convention of 1976 for the Protection of the Mediterranean Sea against Pollution (amended in 1995). The Protocol has been in force since 1999 and includes general provisions to protect sea turtles and their habitats within the Mediterranean Sea. The Protocol requires Parties to protect, preserve, and manage threatened or endangered species, establish protected areas, and coordinate bilateral or multilateral conservation efforts (Hykle 2002). In the framework of the Barcelona Convention, to which all Mediterranean countries are parties, the Action Plan for the Conservation of Mediterranean Marine Turtles has been in effect since 1989. The main objectives of the Action Plan are to enhance the population of marine turtles, protect their critical habitats (including nesting, feeding, wintering, and migrating areas), and improve scientific knowledge by research and monitoring. In particular, the Action Plan has focused on promulgating education and training (especially among fishermen), establishing and improving rescue centers, promoting legislation guidelines, and improving research and monitoring of sea turtles (Demetropoulos 2007). Unfortunately, little research on or implementation of fishing gear bycatch techniques to reduce sea turtle incidental captures has occurred. Additional information is available at http://www.rac-spa.org.

Convention on the Conservation of European Wildlife and Natural Habitats
For information about this instrument, see section 5.2.7.4. for the Northeast Atlantic Ocean DPS.

Council Directive 92/43/EEC on the Conservation of Natural Habitats and of Wild Fauna and Flora (EC Habitats Directive)
The EC Habitats Directive was adopted by the European Community in May 1992, as the means by which the Community meets its obligations as a signatory of the Bern Convention. It aims to protect approximately 200 habitats and 1,000 animal and plant species listed in the Directive's Annexes, of which loggerhead sea turtles are included. The provisions in the Directive require Member States to introduce a range of measures including the protection of species listed in the Annexes, undertake surveillance of habitats and species, and produce a report every six years on the implementation of the Directive. The first complete set of country data was reported in 2007. The Directive led to the establishment of a network of Special Areas of Conservation that, together with the existing Special Protection Areas classified under the separate EC Birds

Directive, form a network of protected areas known as Natura 2000. Additional information is available at http://ec.europa.eu/environment/nature/legislation/habitatsdirective/index_en.htm.

Council Regulation (EC) No. 1239/98 of 8 June 1998 Amending Regulation (EC) No. 894/97 Laying Down Certain Technical Measures for the Conservation of Fishery Measures (Council of the European Union)
For information about this instrument, see section 5.2.7.4. for the Northeast Atlantic Ocean DPS.

Convention for the Protection of the Marine Environment of the North-East Atlantic
This Convention is included under the Mediterranean Sea DPS because it is likely that oceanic juveniles from this DPS also are found in the Northeast Atlantic Ocean. For information about this instrument, see section 5.2.7.4. for the Northeast Atlantic Ocean DPS.

In addition to these international mechanisms, most Mediterranean countries have developed legislation to protect sea turtles and/or nesting habitats (Margaritoulis 2007). For example, in 1999, a National Marine Park was established on Zakynthos in western Greece, with the primary aim to provide protection to loggerhead nesting areas (Dimopoulos 2001). However, while improved over the last several years, it appears that management and protection at this Park is lacking (Robinson *et al*. 2007, Panagopoulou *et al.* 2008). In Turkey, five nesting beaches were designated Specially Protected Area status, and in Cyprus, two nesting beaches have been afforded protection through the Fisheries Regulation, with a maritime zone extending to the 20 meter isobath (Margaritoulis 2007). In Italy, a reserve to protect nesting on Lampedusa was established in 1984 (Margaritoulis *et al*. 2003). The overall effectiveness of this country specific legislation is unknown at this time.

5.2.8.5. Other Natural or Manmade Factors Affecting Its Continued Existence

As noted in Section 5.1.5., other anthropogenic and natural factors affecting loggerhead survival include incidental bycatch in fisheries, vessel collisions, marine pollution, climate change, and cyclonic storm events. Fishing practices alone have been estimated to result in over 150,000 sea turtle captures per year, with approximately 50,000 mortalities (Casale 2008).

The only estimation of loggerhead survival probabilities in the Mediterranean was calculated by using capture-mark-recapture techniques from 1981-2003 (Casale *et al*. 2007c). Of the 3,254 loggerheads tagged, 134 were recaptured at different sites throughout the Mediterranean. Most recaptured animals were juveniles (mean 54.4 cm CCL; range 25-88 cm CCL), but the study did not delineate between juvenile life stages. This research estimated a loggerhead annual survival probability of 0.73 (95% confidence intervals; 0.67-0.78), recognizing that there are methodological limitations of the technique used. Nonetheless, Casale *et al*. (2007a) stated that assuming a natural survivorship no higher than 0.95 and a tag loss rate of 0.1, a range of 0.1-0.2 appears reasonable for the additional human induced mortality (from all sources).

Incidental Bycatch in Fishing Gear
Incidental capture of sea turtles in artisanal and commercial fisheries is a significant threat to the survivability of loggerheads in the Mediterranean. Sea turtles may be caught in pelagic and demersal longlines, drift gillnets, set gillnets and trammel nets, bottom and mid-water trawls,

seines, dredges, traps/pots, and hook and line gear. In a 2004 FAO Fisheries Report, Camiñas (2004) stated that the main fisheries affecting sea turtles in the Mediterranean Sea (at that time) were Spanish and Italian longline, North Adriatic Italian, Tunisian, and Turkish trawl, and Moroccan and Italian driftnet. Available information on sea turtle bycatch by gear type is discussed below. There is growing evidence that artisanal/small vessel fisheries (set gillnet, bottom longline, and part of the pelagic longline fishery) may be responsible for a comparable or higher number of captures with higher mortality rates than the commercial/large vessel fisheries (Casale 2008) as previously suggested by indirect clues (Casale *et al*. 2005a).

Mediterranean fish landings have increased steadily since the 1950s, but the FAO 10-year capture trend from 1990-1999 shows stable landings (Mediterranean LME, http://www.lme.noaa.gov). However, stable fish landings may result from stable fishing effort at the same catch rates, or higher fishing effort at lower catch rates. As fish stocks in the Mediterranean are being depleted (P. Casale, MTSG-IUCN Italy, personal communication, 2009), fishing effort in some areas may be increasing to catch the available fish. This trend has not yet been verified throughout the Mediterranean, but fishing pressures may be increasing even though landings appear stable.

Longline fisheries

In the Mediterranean, pelagic longline fisheries targeting swordfish (*Ziphias gladius*) and albacore (*Thunnus alalunga*) may be the primary source of loggerhead bycatch. It appears that most of the incidental captures occur in the western and central portions of the area (Demetropoulos and Hadjichristophorou 1995). The most severe bycatch in the Mediterranean occurs around the Balearic Islands where 1,950-35,000 juveniles are caught annually in the surface longline fishery (Mayol and Castelló Mas 1983; Camiñas 1988, 1997; Aguilar *et al*. 1995). Specifically, the following regions have reported annual estimates of total turtle bycatch from pelagic longlines: Spain – 17,000 to 35,000 turtles (Aguilar *et al*. 1995, Camiñas *et al*. 2003); Italy (Ionian Sea) – 1,084 to 4,447 turtles (Deflorio *et al*. 2005); Morocco – 3,000 turtles (Laurent 1990); Greece – 280 to 3,310 turtles (Panou *et al*. 1999, Kapantagakis and Lioudakis 2006); Italy (Lampedusa) – 2,100 turtles (Casale *et al*. 2007a); Malta – 1,500 to 2,500 turtles (Gramentz 1989); South Tunisia (Gulf of Gabès) – 486 turtles (Jribi *et al*. 2008); and Algeria – 300 turtles (Laurent 1990).

For the entire Mediterranean pelagic longline fishery, an extrapolation resulted in a bycatch estimate of 60,000 to 80,000 loggerheads in 2000 (Lewison *et al*. 2004). Further, a more recent paper used the best available information to estimate that Spain, Morocco, and Italy have the highest level of sea turtle bycatch, with over 10,000 turtle captures per year for each country, and Greece, Malta, Libya, and Tunisia each catch 1,000 to 3,000 turtles per year (Casale 2008). Available data suggest the annual number of loggerhead sea turtle captures by all Mediterranean pelagic longline fisheries may be greater than 50,000 (Casale 2008). Note that these are not necessarily individual turtles, as the same sea turtle can be captured more than once.

Mortality estimates in the pelagic longline fishery at gear retrieval appear to be lower than in some other types of gear (e.g., set gillnet). Although limited to observations of direct mortality at gear retrieval, Carreras *et al*. (2004) found mortality to be low (0-7.7%) in the longline fishery off the Balearic Islands, and Jribi *et al*. (2008) reported 0% direct mortality in the southern

Tunisia surface longline fishery. These estimates are consistent with those found in other areas; direct mortality was estimated at 4.3% in Greece (n=23), 0% in Italy (n=214), and 2.6% in Spain (n=676) (Laurent *et al*. 2001). However, considering injured turtles and those released with hooks, the potential for mortality is likely much higher. Based upon observations of hooked loggerhead turtles in captivity, Aguilar *et al*. (1995) estimated 20-30% of animals caught in longline gear may eventually die. More recently, Casale *et al*. (2008c) found, given variations in hook position affecting survivability, the mortality rate of turtles caught by pelagic longlines may be higher than 30%, which is greater than previously thought (17-42%; Lewison *et al*. 2004). Considering direct and post-release mortality, Casale (2008) used a conservative approach to arrive at 40% for the average mortality from Mediterranean pelagic longlines. The result is an estimated 20,000 turtles killed per year by pelagic longlines (Casale 2008).

These incidental capture levels may impact the Mediterranean Sea DPS. Of loggerheads caught in the summer pelagic longline fishery from the western and eastern Mediterranean, more than half originated from the Mediterranean nesting population (as opposed to all animals in the western Mediterranean originating from the Atlantic as previously believed; Laurent *et al*. 1998). In general, most of the turtles captured in the Mediterranean surface longline fisheries are juvenile animals (Aguilar *et al*. 1995, Panou *et al*. 1999, Camiñas *et al*. 2003, Casale *et al*. 2007a, Jribi *et al*. 2008), but some adult loggerhead bycatch is also reported. Considering data from many Mediterranean areas and research studies, the average size of turtles caught by pelagic longlines was 48.9 cm CCL (range 20.5-79.2 cm CCL; n=1868) (Casale 2008). Specifically, in the Spanish surface longline fishery, 13% of estimated carapace sizes (n=455) ranged from 75.36 to 107 cm CCL, considered to be adult animals (Camiñas *et al*. 2003), and in the Ionian Sea, 15% of a total 157 loggerhead turtles captured in swordfish longlines were adult animals (estimated size ≥75 cm) (Panou *et al*. 1999).

Bottom longlines are also fished in the Mediterranean, but specific capture rates for loggerheads are largely unknown for many areas. The countries with the highest number of documented captures (in the thousands per year) are Tunisia, Libya, Greece, Turkey, Egypt, Morocco, and Italy (Casale 2008). Available data suggest the annual number of loggerhead sea turtle captures (not necessarily individual turtles) by all Mediterranean demersal longliners may be greater than 35,000 (Casale 2008).

Mortality from bottom longlines occurs if the hooked animal cannot reach the surface to breathe or if the ingested hooks and branch lines result in injuries and post-release mortality. For example, in the Gulf of Gabès off Tunisia, the estimated total capture from bottom longlines was 733 loggerheads, with a direct and potential mortality (assuming all comatose turtles would die) of 12.5 to 33%, respectively (Jribi *et al*. 2008). Given available information and using a conservative approach, mortality from bottom longlines may be at least equal to pelagic longline mortality (40%; Casale 2008). The result is an estimated 14,000 turtles killed per year in Mediterranean bottom longlines (Casale 2008). It is likely that these animals represent mostly juvenile, but some adult, loggerheads. Casale (2008) reported an average turtle size of 51.8 cm CCL (n=35) in bottom longlines based on available data throughout the Mediterranean.

Artisanal longline fisheries also have the potential to take sea turtles. A survey of 54 small boat (4-10 meter length) artisanal fishermen in Cyprus and Turkey resulted in an estimated minimum bycatch of over 2,000 turtles per year, with an estimated 10% mortality rate (Godley *et al.* 1998a). These small boats fished with a combination of longlines and trammel/gillnets. However, note that it is likely that a proportion (perhaps a large proportion) of the turtle bycatch estimated in this study are green turtles.

Set net (gillnet) fisheries

As in other areas, sea turtles have the potential to interact with set nets (gillnets or trammel nets) in the Mediterranean. Mediterranean set nets refer to gillnets (a single layer of net) and trammel nets, which consist of three layers of net with different mesh size. Casale (2008) estimated that the countries with the highest number of loggerhead captures (in the thousands per year) are Tunisia, Libya, Greece, Turkey, Cyprus, and Croatia. Italy, Morocco, Egypt, and France likely have high capture rates as well. Available information suggests the annual number of loggerhead captures by Mediterranean set nets may be greater than 30,000 (Casale 2008).

Due to the nature of the gear and fishing practices (e.g., relatively long soak times), incidental capture in gillnets is among the highest source of direct sea turtle mortality. An evaluation of turtles tagged then recaptured in gillnets along the Italian coast found 14 of 19 loggerheads (73.7%) to be dead (Argano *et al.* 1992). Gillnets off France were observed to capture six loggerheads with a 50% mortality rate (Laurent 1991). Six loggerheads were recovered in gillnets off Croatia between 1993 and 1996; 83% were found dead (Lazar *et al.* 2000). Off the Balearic Islands, 196 sea turtles were estimated to be captured in lobster trammel nets in 2001, with a CPUE of 0.17 turtles per vessel (Carreras *et al.* 2004). Mortality estimates for this artisanal lobster trammel net fishery ranged from 78 to 100%. Given this mortality rate and the number of turtles reported in lobster trammel nets, Carreras *et al.* (2004) estimate that a few thousand loggerhead turtles are killed annually by lobster trammel nets in the whole western Mediterranean. Considering data throughout the entire Mediterranean, as well as a conservative approach, Casale (2008) considered mortality by set nets to be 60%, with a resulting estimate of 16,000 turtles killed per year. Most of these animals are likely juveniles; Casale (2008) evaluated available set net catch data throughout the Mediterranean and found an average size of 45.4 cm CCL (n=74).

As noted above, artisanal set net fisheries also may capture numerous sea turtles, as observed off Cyprus and Turkey (Godley *et al.* 1998a).

Driftnet fisheries

Historically, driftnet fishing in the Mediterranean caught large numbers of sea turtles. An estimated 16,000 turtles were captured annually in the Ionian Sea driftnet fishery in the 1980s (De Metrio and Megalofonou 1988). The United Nations established a worldwide moratorium on driftnet fishing effective in 1992, but unregulated driftnetting continued to occur in the Mediterranean. For instance, a bycatch estimate of 236 loggerhead turtles was developed for the Spanish swordfish driftnet fishery in 1994 (Silvani *et al.* 1999). While the Spanish fleet curtailed activity in 1994, the Moroccan, Turkish, French, and Italian driftnet fleets continued to operate. Tudela *et al.* (2005) presented bycatch rates for driftnet fisheries in the Alboran Sea and off Italy. The Moroccan Alboran Sea driftnet fleet bycatch rate ranged from 0.21 to 0.78 loggerheads per haul, whereas the Italian driftnet fleet had a lower bycatch rate of 0.046 to 0.057 loggerheads per haul (Di Natale 1995, Camiñas 1997, Silvani *et al.* 1999). The use of driftnets in the

Mediterranean continues to be illegal: the General Fisheries Commission for the Mediterranean prohibited driftnet fishing in 1997; a total ban on driftnet fishing by the European Union fleet in the Mediterranean went into effect in 2002; and the International Commission for the Conservation of Atlantic Tunas (ICCAT) banned driftnets in 2003. Nevertheless, there are an estimated 600 illegal driftnet vessels operating in the Mediterranean, including fleets based in Algeria, France, Italy, Morocco, and Turkey (Environmental Justice Foundation 2007). In particular, the Moroccan fleet, operating in the Alboran Sea and Straits of Gibraltar, comprises the bulk of Mediterranean driftnetting, and has been found responsible for high bycatch, including loggerhead turtles (Environmental Justice Foundation 2007, Aksissou *et al*. in press). Driftnet fishing in the Mediterranean, and accompanying threats to loggerhead turtles, continues to occur.

Trawl fisheries

Sea turtles are known to be incidentally captured in trawls in Albania, Algeria, Croatia, Egypt, France, Greece, Italy, Libya, Morocco, Slovenia, Spain, Tunisia, and Turkey (Gerosa and Casale 1999, Camiñas 2004, Casale 2008). Laurent *et al*. (1996) estimated that approximately 10,000 to 15,000 sea turtles (most of which are loggerheads) are captured by bottom trawling in the entire Mediterranean. More recently, Casale (2008) compiled available trawl bycatch data throughout the Mediterranean and reported that Italy and Tunisia have the highest level of sea turtle bycatch, potentially over 20,000 captures per year combined, and Croatia, Greece, Turkey, Egypt, and Libya each catch more than 2,000 turtles per year. Further, Spain and Albania may each capture a few hundred sea turtles per year (Casale 2008). Available data suggest the annual number of sea turtle captures by all Mediterranean trawlers may be greater than 40,000 (Casale 2008). Note that these are capture events and not necessarily individual turtles.

Information from specific study areas of the Mediterranean provides additional insight into individual bycatch rates and details of the captures. Casale *et al*. (2004) estimated that 4,273 loggerheads per year are caught by the Italian bottom trawl fishery in the western part of the northern Adriatic Sea, but as catch rates in the eastern part of the Adriatic Sea are higher than in the western, it is likely that actual total catch in the entire North Adriatic Sea is much higher. In southern Italy, off Lampedusa, Casale *et al*. (2007a) estimated an annual total catch of 1,016 for the vessels in the study port. However, the catch estimate increased to approximately 4,000 to 11,000 turtles if trawlers from other ports fishing in the study area were considered. In the Gulf of Gabès, Tunisia, 5,458 loggerhead turtles per year were estimated to be captured by the trawl fishery (Jribi *et al*. 2007), while in Croatia, an annual incidental catch of 2,500 turtles was previously estimated based on fishermen interviews (Lazar and Tvrtkovic 1995). In Greece, total catch (presented as 95% confidence intervals) in 2000 was estimated at 0 to 418 turtles in the northern Aegean Sea and at 0 to 448 turtles in the Ionian Sea (Laurent *et al*. 2001).

Although juveniles are incidentally captured in trawl gear in many areas of the Mediterranean (Casale *et al*. 2004, 2007a; Jribi *et al*. 2007), adult turtles are also found. In Egypt, 25% of loggerheads captured in bottom trawl gear (n=16) were ≥ 70 cm CCL, and in Tunisia, 26.2% (n=62) were of this larger size class (Laurent *et al*. 1996). Off Lampedusa Island, Italy, the average size of turtles caught by bottom trawlers was 51.8 cm CCL (range 22-87 cm CCL; n=368), and approximately 10% of the animals measured greater than 75 cm CCL (Casale *et al*. 2007a). For all areas of the Mediterranean, Casale (2008) reported that medium to large turtles are generally caught by bottom trawl gear (mean 53.9 cm CCL; range 22-87 cm CCL; n=648).

While there is a notable interaction rate in the Mediterranean, it appears that the mortality associated with trawling is relatively low. Incidents of mortality have ranged from 3.3% (n=60) in Tunisia (Jribi *et al*. 2007) and 3.3% (n=92) in France (Laurent 1991) to 9.4% (n=32) in Italy (Casale *et al*. 2004). Casale *et al*. (2004) found that mortality would be higher if all comatose turtles were assumed to die. It also should be noted that the mortality rate in trawls depends on the duration of the haul, with longer haul durations resulting in higher mortality rates (Henwood and Stuntz 1987, Sasso and Epperly 2006). Jribi *et al*. (2007) stated that the low recorded mortality in the Gulf of Gabès is likely due to the short haul durations in this area. Based on available information from multiple areas of the Mediterranean, and assuming that comatose animals die if released in that condition, the overall average mortality rate for bottom trawlers was estimated to be 20% (Casale 2008). This results in at least 7,400 turtles killed per year by bottom trawlers in all of the Mediterranean, but the number is likely more than 10,000 (Casale 2008).

Bottom trawl fisheries are likely impacting the Mediterranean Sea DPS. Although in a previous assessment of bottom trawl fisheries sampled from Tunisia, Egypt, and Turkey, all loggerheads captured were from the Mediterranean nesting population (as opposed to Atlantic nesting animals; Laurent *et al*. 1998), a recent genetic study with a larger sample size found a minor but significant contribution from the Atlantic populations in the turtles foraging in the neritic habitats off Tunisia (Casale *et al*. 2008b).

Mid-water trawling may have less total impact on sea turtles found in the Mediterranean than some other gear types, but interactions still occur. Casale *et al*. (2004) found that while no turtles were caught on observed mid-water trawl trips in the North Adriatic Sea, vessel captains reported 13 sea turtles captured from April to September. Considering total fishing effort, these reports resulted in a minimum total catch estimate of 161 turtles/year in the Italian mid-water trawl fishery. Off Turkey, 71 loggerheads were captured in mid-water trawls from 1995-1996, while 43 loggerheads were incidentally taken in bottom trawls (Oruç 2001). In this same study, of a total 320 turtles captured in mid-water trawls (loggerheads and greens combined), 95% were captured alive and apparently healthy. While the total catch numbers throughout the Mediterranean have not been estimated, mid-water trawl fisheries do present a threat to loggerhead sea turtles.

Other gear types

Seine, dredge, trap/pot, and hook and line fisheries operate in Mediterranean waters and may affect loggerhead turtles, although incidental captures in these gear types are largely unknown (Camiñas 2004). Artisanal fisheries using a variety of gear types also have the potential for sea turtle takes, but the effects of most artisanal gear types on sea turtles have not been estimated.

Other Manmade and Natural Impacts

Other anthropogenic threats, such as interactions with recreational and commercial vessels, marine pollution, and intentional killing, also impact loggerheads found in the Mediterranean. Propeller and collision injuries from boats and ships are becoming more common in sea turtles, although it is unclear as to whether the events are increasing or just the reporting of the injuries. Speedboat impacts are of particular concern in areas of intense tourist activity, such as Greece and Turkey. Losses of nesting females from vessel collisions have been documented in Zakynthos and Crete in Greece (Camiñas 2004). In the Gulf of Naples, 28.1% of loggerheads recovered from 1993-1996 had injuries attributed to boat strikes (Bentivegna and Paglialonga 1998). Along the Greece coastline from 1997-1999, boat strikes were reported as a seasonal phenomenon in stranded turtles (Kopsida *et al*. 2002), but numbers were not presented.

Direct or indirect disposal of anthropogenic debris introduces potentially lethal materials into loggerhead foraging habitats. Unattended or discarded nets, floating plastics and bags, and tar balls are of particular concern (Camiñas 2004, Margaritoulis 2007). Monofilament netting appears to be the most dangerous waste produced by the fishing industry (Camiñas 2004). In the Mediterranean, 20 out of 99 loggerhead turtles examined from Maltese fisheries were found contaminated with plastic or metal litter and hydrocarbons, with crude oil being the most common pollutant (Gramentz 1988). Of 54 juvenile loggerhead turtles incidentally caught by fisheries in Spanish Mediterranean waters, 79.6% had debris in their digestive tracts (Tomás *et al*. 2002). In this study, plastics were the most frequent type of marine debris observed (75.9%), followed by tar (25.9%). However, an examination of stranded sea turtles in Northern Cyprus and Turkey found that only 3 of 98 animals were affected by marine debris (Godley *et al*. 1998b).

Pollutant waste in the marine environment may impact loggerheads, likely more than other sea turtle species. Omnivorous loggerheads stranded in Cyprus, Greece, and Scotland had the highest organochlorine contaminant concentrations, as compared to green and leatherback turtles (Mckenzie *et al*. 1999). In northern Cyprus, Godley *et al*. (1999) found heavy metal concentrations (mercury, cadmium, and lead) to be higher in loggerheads than green turtles. Even so, concentrations of contaminants from sea turtles in Mediterranean waters were found to be comparable to other areas, generally with levels lower than concentrations shown to cause deleterious effects in other species (Godley *et al*. 1999, Mckenzie *et al*. 1999). However, lead concentrations in some Mediterranean loggerhead hatchlings were at levels known to cause toxic effects in other vertebrate groups (Godley *et al*. 1999).

As in other areas of the world, intentional killing or injuring of sea turtles has been reported to occur in the Mediterranean. Of 524 strandings in Greece, it appeared that 23% had been intentionally killed or injured (Kopsida *et al*. 2002). While some turtles incidentally captured are used for consumption (as noted in Section 5.2.8.2 above), it has been reported that some fishermen kill the sea turtles they catch for a variety of other reasons, including non-commercial use, hostility, prejudice, recovery of hooks, and ignorance (Laurent *et al*. 1996, Godley *et al*. 1998a, Gerosa and Casale 1999, Casale 2008).

Natural environmental events also may affect loggerheads in the Mediterranean. Cyclonic storms that closely resemble tropical cyclones in satellite images occasionally form over the Mediterranean Sea (Emanuel 2005). While hurricanes typically do not occur in the Mediterranean, researchers have suggested that climate change could trigger hurricane development in this area in the future (Gaertner *et al*. 2007). Any significant storm event that may develop could disrupt loggerhead nesting activity and hatchling production, but the results are generally localized and rarely result in whole-scale losses over multiple nesting seasons.

Similar to other areas of the world (see Section 5.1.5.), climate change and sea level rise have the potential to impact loggerheads in the Mediterranean. Over the long term, Mediterranean turtle populations could be threatened by the alteration of thermal sand characteristics (from global warming), resulting in the reduction or cessation of female hatchling production (Camiñas 2004). Further, a significant rise in sea level would restrict loggerhead nesting habitat in the eastern Mediterranean.

5.2.9. South Atlantic Ocean DPS

5.2.9.1. Present or Threatened Destruction, Modification, or Curtailment of Its Habitat or Range

Terrestrial Zone
As described in Section 5.1.1., destruction and modification of loggerhead nesting habitat in the South Atlantic result from coastal development and construction, placement of erosion control structures and other barriers to nesting, beachfront lighting, vehicular and pedestrian traffic, sand extraction, beach erosion, beach sand placement, beach pollution, removal of native vegetation, and planting of non-native vegetation (D'Amato and Marczwski 1993; Marcovaldi and Marcovaldi 1999; Naro-Maciel *et al*. 1999; Marcovaldi *et al*. 2002, 2005; Marcovaldi 2007).

The primary nesting areas for loggerheads in the South Atlantic are in the states of Sergipe, Bahia, Espírito Santo, and Rio de Janeiro in Brazil (Marcovaldi and Marcovaldi 1999). These primary nesting areas are monitored by Projeto TAMAR, the national sea turtle conservation program in Brazil. Since 1980, Projeto TAMAR has worked to establish legal protection for nesting beaches (Marcovaldi and Marcovaldi 1999). As such, human activities, including sand extraction, beach nourishment, seawall construction, beach driving, and artificial lighting, that can negatively impact sea turtle nesting habitat, as well as directly impact nesting turtles and their eggs and hatchlings during the reproductive season, are restricted by various state and federal laws (Marcovaldi and Marcovaldi 1999; Marcovaldi *et al*. 2002, 2005). Nevertheless, tourism development in coastal areas in Brazil is high, and Projeto TAMAR works toward raising awareness of turtles and their conservation needs through educational and informational activities at their Visitor Centers that are dispersed throughout the nesting areas (Marcovaldi *et al*. 2005).

In terms of non-native vegetation, the majority of nesting beaches in northern Bahia, where loggerhead nesting density is highest in Brazil (Marcovaldi and Chaloupka 2007), have coconut plantations dating back to the 17th century backing them (Naro-Maciel *et al*. 1999). It is impossible to assess whether this structured habitat has resulted in long-term changes to the loggerhead nesting rookery in northern Bahia.

Neritic/Oceanic Zone
As described in Section 5.1.1., human activities that impact bottom habitat in the loggerhead neritic and oceanic zones include fishing practices, channel dredging, sand extraction, marine pollution, and climate change. These threats also occur in the South Atlantic Ocean (e.g., Ibe 1996, Silva *et al*. 1997). General human activities have altered ocean ecosystems, as identified by ecosystem models (http://www.lme.noaa.gov). On the western side of the South Atlantic, the Brazil Current Large Marine Ecosystem (LME) region is characterized by the Global International Waters Assessment (GIWA) as suffering severe impacts in the areas of pollution,

coastal habitat modification, and overexploitation of fish stocks (Marques *et al*. 2004). The Patagonian Shelf LME is moderately affected by pollution, habitat modification, and overfishing (Mugetti *et al*. 2004). On the eastern side of the South Atlantic, the Benguela Current LME has been characterized as moderately impacted in the area of overfishing, with future conditions expected to worsen by the GIWA (Prochazka *et al*. 2005). Climate change also may result in future trophic changes, thus impacting loggerhead prey abundance and/or distribution.

5.2.9.2. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes

Deliberate hunting of loggerheads for their meat, shells, and eggs is reduced from previous exploitation levels, but still exists. Limited numbers of eggs are taken for human consumption in Brazil, but the relative amount is considered minor when compared to historical rates of egg collection (Marcovaldi and Marcovaldi 1999, Marcovaldi *et al*. 2005, Almeida and Mendes 2007). Use of sea turtles including loggerheads for medicinal purposes occasionally occurs in northeastern Brazil (Alves and Rosa 2006). Use of bycaught loggerheads for subsistence and medicinal purposes is likely to occur in southern Atlantic Africa, based on information from central West Africa (Fretey 2001, Fretey *et al*. 2007).

5.2.9.3. Disease or Predation

The potential effects of diseases and endoparasites, as described in Section 5.1.3., also exist for loggerheads found in the South Atlantic Ocean. There have been five confirmed cases of fibropapillomatosis in loggerheads in Brazil (Baptistotte 2007). There is no indication that this disease poses a major threat for this species in the eastern South Atlantic (Formia *et al*. 2007).

Eggs and nests in Brazil experience depredation, primarily by foxes (Marcovaldi and Laurent 1996). Nests laid by loggerheads in the southern Atlantic African coastline, if any, likely experience similar predation pressures to those on nests of other species laid in the same area (e.g., jackals depredate green turtle nests in Angola; Weir *et al*. 2007).

Loggerheads in the South Atlantic also may be impacted by harmful algal blooms (Gilbert *et al*. 2005).

5.2.9.4. Inadequacy of Existing Regulatory Mechanisms

Unless identified to a specific geographic area, the regulatory mechanisms described in Section 5.1.4. also apply to sea turtles found in the South Atlantic Ocean. The following additional instruments also apply to loggerheads found in this area.

Convention on the Conservation of Migratory Species of Wild Animals, Memorandum of Understanding Concerning Conservation Measures for Marine Turtles of the Atlantic Coast of Africa (Abidjan Memorandum)
For information about this instrument, see section 5.2.7.4. for the Northeast Atlantic Ocean DPS.

The Convention for the Co-operation in the Protection and Development of the Marine and Coastal Environment of the West and Central African Region (Abidjan Convention)
For information about this instrument, see section 5.2.7.4. for the Northeast Atlantic Ocean DPS.

South-East Atlantic Fisheries Organization (SEAFO)
SEAFO manages fisheries activities in the Southeast Atlantic high seas area, excluding tunas and billfish. SEAFO adopted Resolution 01/06, "to Reduce Sea Turtle Mortality in Fishing Operations," in 2006. The Resolution requires Members to: (1) implement the FAO Guidelines; and (2) establish on-board observer programs to collect information on sea turtle interactions in SEAFO-managed fisheries. This Resolution is not legally binding. Additional information is available at http://www.seafo.org.

Inter-American Convention for the Protection and Conservation of Sea Turtles (IAC)
For information about this instrument, see section 5.2.1.4. for the North Pacific Ocean DPS.

Nairobi Convention for the Protection, Management and Development of the Marine and Coastal Environment of the Eastern African Region
For information about this instrument, see section 5.2.3.4. for the North Indian Ocean DPS.

5.2.9.5. Other Natural or Manmade Factors Affecting Its Continued Existence

Incidental Bycatch in Fishing Gear
Incidental capture of sea turtles in artisanal and commercial fisheries is a significant threat to the survivability of loggerheads in the South Atlantic. Sea turtles may be caught in pelagic and demersal longlines, drift and set gillnets, bottom and mid-water trawling, fishing dredges, pound nets and weirs, haul and purse seines, pots and traps, and hook and line gear. In the western South Atlantic, there are various efforts aimed at mitigating bycatch of sea turtles in various fisheries. In Brazil, there is the *National Action Plan to Reduce Incidental Capture of Sea Turtles in Fisheries*, coordinated by Projeto TAMAR (Marcovaldi *et al*. 2006). This action plan focuses on both artisanal and commercial fisheries, and collects data directly from fishers as well as on-board observers. Although loggerheads have been observed as bycatch in all fishing gear and methods identified above, Marcovaldi *et al*. (2006) have identified longlining as the major source of incidental capture of loggerhead turtles. Reports of loggerhead bycatch by pelagic longlines come mostly from the southern portion of the Brazilian Exclusive Economic Zone, between 20°S and 35°S. Bugoni *et al*. (2008) reported a loggerhead bycatch rate of 0.52 juvenile turtles/1000 hooks by surface longlines targeting dolphinfish. Pinedo *et al*. (2004) reported seasonal variation in bycatch of juvenile loggerheads (and other sea turtle species) by pelagic longlines in the same region of Brazil, with the highest rates (1.85 turtles/1000 hooks) in the austral spring. Kotas *et al*. (2004) reported the highest rates of loggerhead bycatch (>10 turtles/1000 hooks) by pelagic longlines in the austral summer/fall months. A study based on several years found that the highest rate of loggerhead bycatch in pelagic longlines off of Uruguay and Brazil was in the late austral summer month of February: 2.72 turtles/1000 hooks (López-Medilaharsu *et al*. 2007). Sales *et al*. (2004) reported a loggerhead bycatch rate of 0.87/1000 hooks near the Rio Grande Elevação do Rio Grande, about 600 nm off the coast of southern Brazil. In Uruguayan waters, the primary fisheries with loggerhead bycatch are bottom trawlers and longlines (Domingo *et al*. 2006). Domingo *et al*. (2008) reported bycatch rates of

loggerheads of 0.9-1.3/1000 hooks by longline deployed south of 30°S. In waters off Argentina, bottom trawlers also catch some loggerheads (Domingo *et al*. 2006).

In the eastern South Atlantic, sea turtle bycatch in fisheries has been documented from Gabon to South Africa (Fretey 2001). Limited data are available on bycatch of loggerheads in coastal fisheries, although loggerheads are known (or strongly suspected) to occur in coastal waters from Gabon to South Africa (Fretey 2001, Bal *et al*. 2007, Weir *et al*. 2007). Coastal fisheries implicated in bycatch of loggerheads and other turtles include gillnets, beach seines, and trawlers (Bal *et al*. 2007).

In the high seas, longlines are used by fishing boats targeting tuna and swordfish in the eastern South Atlantic. A recent study by Honig *et al*. (2008) estimates 7,600-120,000 sea turtles are incidentally captured by commercial longlines fishing in the Benguela Current LME; 60% of these are loggerheads. Petersen *et al*. 2007, 2009) report that the rate of loggerhead bycatch in South African longliners was around 0.02 turtles/1000 hooks, largely in the Benguela Current LME. In the middle of the South Atlantic, loggerhead bycatch by longlines was reported to be low, relative to other regions in the Atlantic (Mejuto *et al* 2008).

Other Manmade and Natural Impacts

As described in Section 5.1.5., other anthropogenic impacts, such as boat strikes and ingestion or entanglement in marine debris, also apply to loggerheads in the South Atlantic. Bugoni *et al*. (2001) have suggested the ingestion of plastic and oil may contribute to loggerhead mortality on the southern coast of Brazil. Plastic marine debris in the eastern South Atlantic also may pose a problem for loggerheads and other sea turtles (Ryan 1996). Similar to other areas of the world (see Section 5.1.5.), climate change and sea level rise have the potential to impact loggerheads in the South Atlantic.

Oil reserve exploration and extraction activities also may pose a threat for sea turtles in the South Atlantic. Seismic surveys in Brazil and Angola have recorded sea turtle occurrences near the seismic work (Gurjão *et al*. 2005, Weir *et al*. 2007). While no sea turtle takes were directly observed on these surveys, increased equipment and presence in the water that is associated with these activities also increases the likelihood of sea turtle interactions (Weir *et al*. 2007).

Natural environmental events may affect loggerheads in the South Atlantic. However, while a rare hurricane hit Brazil in March 2004, typically hurricanes do not occur in the South Atlantic (McTaggart-Cowan *et al*. 2006). This is generally due to higher windspeeds aloft, preventing the storms from gaining height and therefore strength.

SECTION 6—CONCLUSIONS—SYNTHESIS & INTEGRATION

6.1. North Pacific Ocean DPS

We used two approaches to compute extinction risk for the North Pacific Ocean DPS: (1) computation of susceptibility to quasi-extinction (SQE), and (2) a stage-based deterministic model to determine negative effects of known threats to the DPS. Based on nest count data for nearly the past two decades, the North Pacific population of loggerheads is small. The SQE approach suggested that the North Pacific Ocean DPS appears to be declining, is at risk, and is thus likely to decline in the foreseeable future. These results are based on nesting beach census data from 33 nesting beaches in Japan over a 17-year period (in Snover 2008). We note that the SQE approach we used is based on past performance of the DPS (nesting data) and does not fully reflect ongoing and future threats to all life stages within the DPS. The stage-based deterministic modeling approach used a hypothetical population that was at its maximum population growth rate and computed the population growth potential under known or suspected threats to different life stages of the North Pacific Ocean DPS. This approach provided a range of results: in the case of the lowest anthropogenic mortality rates (or the best case scenario), the deterministic model suggests that the North Pacific Ocean DPS would grow slightly, but in the worst-case scenario, the model indicates that the population would be likely to substantially decline in the foreseeable future. These results are largely driven by the mortality of juvenile and adult loggerheads from fishery bycatch that occurs throughout the North Pacific Ocean, including the coastal pound net fisheries off Japan, coastal fisheries impacting juvenile foraging populations off Baja California, Mexico, and undescribed fisheries likely affecting loggerheads in the South China Sea and the North Pacific Ocean. Although national and international governmental and non-governmental entities on both sides of the North Pacific are currently working toward reducing loggerhead bycatch, and some positive actions have been implemented, it is unlikely that this source of mortality can be sufficiently reduced in the near future due to the challenges of mitigating illegal, unregulated, and unreported fisheries, the lack of comprehensive information on fishing distribution and effort, limitations on implementing demonstrated effective conservation measures, geopolitical complexities, limitations on enforcement capacity, and availability of comprehensive bycatch reduction technologies. In addition to fishery bycatch, coastal development and coastal armoring on nesting beaches in Japan continues as a substantial threat. Coastal armoring, if left unaddressed, will become an even more substantial threat as sea level rises. Therefore, the BRT concluded that the North Pacific Ocean DPS is currently at risk of extinction.

6.2. South Pacific Ocean DPS

We used two approaches to compute extinction risk for the South Pacific Ocean DPS: (1) computation of susceptibility to quasi-extinction (SQE), and (2) a stage-based deterministic model to determine negative effects of known threats to the DPS. The SQE approach suggested that, based on nest count data for the past three decades, the population is at risk and thus likely to decline in the foreseeable future. These results are based on recently published nesting census data for loggerheads at index beaches in eastern Australia (Limpus 2009). The stage-based deterministic modeling approach used a hypothetical population that was at its maximum population growth rate and computed the population growth potential under known or suspected threats to different life stages of the South Pacific Ocean DPS. This approach provided a wide range of results: in the case of the lowest anthropogenic mortality rates (or the best case scenario), the deterministic model suggests that the South Pacific Ocean DPS will grow slightly, but in the worst-case scenario, the model indicates that the population is likely to substantially decline in the foreseeable future. These results are largely driven by mortality of juvenile and adult loggerheads from fishery bycatch that occurs throughout the South Pacific Ocean. Although national and international governmental and non-governmental entities on both sides of the South Pacific are currently working toward reducing loggerhead bycatch, and some positive actions have been implemented, it is unlikely that this source of mortality can be sufficiently reduced in the near future due to the challenges of mitigating illegal, unregulated, and unreported fisheries, the continued expansion of artisanal fleets in the southeastern Pacific, the lack of comprehensive information on fishing distribution and effort, limitations on implementing demonstrated effective conservation measures, geopolitical complexities, limitations on enforcement capacity, and availability of comprehensive bycatch reduction technologies. Therefore, the BRT concluded that the South Pacific Ocean DPS is currently at risk of extinction.

6.3. North Indian Ocean DPS

Two approaches to computing extinction risk for the North Indian Ocean DPS were considered: (1) computation of susceptibility to quasi-extinction (SQE), and (2) a stage-based deterministic model to determine negative effects of known threats to the DPS. The SQE approach is based on nesting data; however, an adequate time series of nesting data for this DPS was not available to the BRT. Therefore, we could not use this approach to evaluate extinction risk. The stage-based deterministic modeling approach used a hypothetical population that was at its maximum population growth rate and computed the population growth potential under known or suspected threats to different life stages of the North Indian Ocean DPS. The model indicated the North Indian Ocean DPS is likely to decline in the foreseeable future. These results are driven by cumulative mortality from a variety of sources across all life stages. Threats to nesting beaches are likely to increase, which would require additional and widespread nesting beach protection efforts. Little is currently being done to monitor and reduce mortality from neritic and oceanic fisheries in the range of the North Indian Ocean DPS; this mortality is likely to continue and increase with expected additional fishing effort from commercial and artisanal fisheries. Reduction of mortality would be difficult due to a lack of comprehensive information on fishing distribution and effort, limitations on implementing demonstrated effective conservation measures, geopolitical complexities, limitations on enforcement capacity, and availability of comprehensive bycatch reduction technologies. Therefore, the BRT concluded that the North Indian Ocean DPS is currently at risk of extinction.

6.4. Southeast Indo-Pacific Ocean DPS

Two approaches to computing extinction risk for the Southeast Indo-Pacific Ocean DPS were considered: (1) computation of susceptibility to quasi-extinction (SQE), and (2) a stage-based deterministic model to determine negative effects of known threats to the DPS. The SQE approach is based on nesting data; however, an adequate time series of nesting data for this DPS was not available to the BRT; therefore, we could not use this approach to evaluate extinction risk. The stage-based deterministic modeling approach used a hypothetical population that was at its maximum population growth rate and computed the population growth potential under known or suspected threats to different life stages of the Southeast Indo-Pacific Ocean DPS. This approach provided a wide range of results: in the case of the lowest anthropogenic mortality rates, the deterministic model suggests that the South Pacific Ocean DPS will grow slightly, but in the worst-case scenario, the model indicates that the population is likely to substantially decline in the foreseeable future. These results are largely driven by mortality of juvenile and adult loggerheads from fishery bycatch that occurs throughout the region, as can be inferred from data from Australia's Pacific waters. Although national and international governmental and non-governmental entities are currently working toward reducing loggerhead bycatch, and some positive actions have been implemented, it is unlikely that this source of mortality can be sufficiently reduced in the near future due to the challenges of mitigating illegal, unregulated, and unreported fisheries, the continued expansion of artisanal fleets, the lack of comprehensive information on fishing distribution and effort, limitations on implementing demonstrated effective conservation measures, geopolitical complexities, limitations on enforcement capacity, and availability of comprehensive bycatch reduction technologies. Therefore, the BRT concluded that the Southeast Indo-Pacific Ocean DPS is currently at risk of extinction.

6.5. Southwest Indian Ocean DPS

Two approaches to computing extinction risk for the Southwest Indian Ocean DPS were considered: (1) computation of susceptibility to quasi-extinction (SQE), and (2) a stage-based deterministic model to determine negative effects of known threats to the DPS. These two approaches provided different, although not incompatible, results for the Southwest Indian Ocean DPS. The SQE approach, based on a 37-year time series of nesting female counts at Tongaland, South Africa (1963-1999), indicated this segment of the population, while small, has increased, and the likelihood of quasi-extinction is negligible. We note that the SQE approach we used is based on past performance of the DPS (nesting data from 1963-1999) and does not fully reflect ongoing and future threats to all life stages within the DPS. The stage-based deterministic modeling approach used a hypothetical population that was at its maximum population growth rate and computed the population growth potential under known or suspected threats to different life stages of the Southwest Indian Ocean DPS. This approach provided a wide range of results: in the case of the lowest anthropogenic mortality rates, the deterministic model suggests that the South Pacific Ocean DPS will grow slightly, but in the worst-case scenario, the model indicates that the population is likely to substantially decline in the foreseeable future. These results are largely driven by mortality of juvenile loggerheads from fishery bycatch that occurs throughout the Southwest Indian Ocean. This mortality is likely to continue and may increase with expected additional fishing effort from commercial and artisanal fisheries. Reduction of mortality would be difficult due to a lack of comprehensive information on fishing distribution and effort, limitations on implementing demonstrated effective conservation measures, geopolitical complexities, limitations on enforcement capacity, and availability of comprehensive bycatch reduction technologies. Therefore, the BRT concluded that although the Southwest Indian Ocean DPS is likely not currently at immediate risk of extinction, the extinction risk is likely to increase in the foreseeable future.

6.6. Northwest Atlantic Ocean DPS

We used two approaches to compute extinction risk for the Northwest Atlantic Ocean DPS: (1) computation of susceptibility to quasi-extinction (SQE), and (2) a stage-based deterministic model to determine negative effects of known threats to the DPS. The SQE approach suggested that, based on nest count data for the past two decades, the population is at risk and thus likely to decline in the foreseeable future. These results are based on nesting data for loggerheads at index/standardized nesting survey beaches in the USA and the Yucatan Peninsula, Mexico (NMFS and FWS 2008). The stage-based deterministic modeling approach used a hypothetical population that was at its maximum population growth rate and computed the population growth potential under known or suspected threats to different life stages of the Northwest Atlantic Ocean DPS. This approach indicated the Northwest Atlantic Ocean DPS is likely to decline in the foreseeable future, even under the scenario of the lowest anthropogenic mortality rates. These results are largely driven by mortality of juvenile and adult loggerheads from fishery bycatch that occurs throughout the North Atlantic Ocean. Although national and international governmental and non-governmental entities on both sides of the North Atlantic are currently working toward reducing loggerhead bycatch, and some positive actions have been implemented, it is unlikely that this source of mortality can be sufficiently reduced across the range of the DPS in the near future because of the diversity and magnitude of the fisheries operating in the North Atlantic, the lack of comprehensive information on fishing distribution and effort, limitations on implementing demonstrated effective conservation measures, geopolitical complexities, limitations on enforcement capacity, and availability of comprehensive bycatch reduction technologies. Therefore, the BRT concluded that the Northwest Atlantic Ocean DPS is currently at risk of extinction.

6.7. Northeast Atlantic Ocean DPS

Two approaches to computing extinction risk for the Northeast Atlantic Ocean DPS were considered: (1) computation of susceptibility to quasi-extinction (SQE), and (2) a stage-based deterministic model to determine negative effects of known threats to the DPS. The SQE approach is based on nesting data. However, we had insufficient nest count data over an appropriate time series for this DPS and could not use this approach to evaluate extinction risk. The stage-based deterministic modeling approach used a hypothetical population that was at its maximum population growth rate and computed the population growth potential under known or suspected threats to different life stages of the Northeast Atlantic Ocean DPS. The model indicated the Northeast Atlantic Ocean DPS is likely to decline in the foreseeable future, even under the scenario of the lowest anthropogenic mortality rates. These results are largely driven by the ongoing directed lethal take of nesting females, low hatching and emergence success, and mortality of juveniles and adults from fishery bycatch that occurs throughout the Northeast Atlantic Ocean. Currently, conservation efforts to protect nesting females are growing, and a reduction in this source of mortality is likely to continue in the near future. Although national and international governmental and non-governmental entities in the Northeast Atlantic are currently working toward reducing loggerhead bycatch, and some positive actions have been implemented, it is unlikely that this source of mortality can be sufficiently reduced across the range of the DPS in the near future because of the lack of bycatch reduction in high seas fisheries operating within the range of this DPS, lack of bycatch reduction in coastal fisheries in Africa, the lack of comprehensive information on fishing distribution and effort, limitations on implementing demonstrated effective conservation measures, geopolitical complexities, limitations on enforcement capacity, and availability of comprehensive bycatch reduction technologies. Therefore, the BRT concluded that the Northeast Atlantic Ocean DPS is at immediate risk of extinction.

6.8. Mediterranean Sea DPS

Two approaches to computing extinction risk for the Mediterranean Sea DPS were considered: (1) computation of susceptibility to quasi-extinction (SQE), and (2) a stage-based deterministic model to determine negative effects of known threats to the DPS. The SQE approach is based on nesting data; however, region-wide nesting data for this DPS were not available to the BRT. Therefore, we could not use this approach to evaluate extinction risk. The stage-based deterministic modeling approach used a hypothetical population that was at its maximum population growth rate and computed the population growth potential under known or suspected threats to different life stages of the Mediterranean Sea DPS. The model indicated the Mediterranean Sea DPS is likely to decline in the foreseeable future, even under the scenario of the lowest anthropogenic mortality rates. These results are largely driven by mortality of juvenile and adult loggerheads from fishery bycatch that occurs throughout the Mediterranean Sea, as well as anthropogenic threats to nesting beaches and eggs/hatchlings. Although conservation efforts to protect some nesting beaches are underway, more widespread and consistent protection is needed. Although national and international governmental and non-governmental entities in the Mediterranean Sea are currently working toward reducing loggerhead bycatch, it is unlikely that this source of mortality can be sufficiently reduced across the range of the DPS in the near future because of the lack of bycatch reduction in commercial and artisanal fisheries operating within the range of this DPS, the lack of comprehensive information on fishing distribution and effort, limitations on implementing demonstrated effective conservation measures, geopolitical complexities, limitations on enforcement capacity, and availability of comprehensive bycatch reduction technologies. Therefore, the BRT concluded that the Mediterranean Sea DPS is at immediate risk of extinction.

6.9. South Atlantic Ocean DPS

We used two approaches to compute extinction risk for the South Atlantic Ocean DPS: (1) computation of susceptibility to quasi-extinction (SQE), and (2) a stage-based deterministic model to determine negative effects of known threats to the DPS. These two approaches provided different, although not incompatible, results for the South Atlantic Ocean DPS. The SQE approach suggested that, based on nest count data for the past two decades, the population is unlikely to decline in the foreseeable future. These results are consistent with Marcovaldi and Chaloupka's (2007) nesting beach trend analyses. We note that the SQE approach is based on past performance of the DPS (nesting data) and does not fully reflect ongoing and future threats to all life stages within the DPS. The stage-based deterministic modeling approach used a hypothetical population that was at its maximum population growth rate and computed the population growth potential under known or suspected threats to different life stages of the South Atlantic Ocean DPS. This approach indicated the South Atlantic Ocean DPS is likely to decline in the foreseeable future, even under the scenario of the lowest anthropogenic mortality rates.

This result is largely driven by mortality of juvenile loggerheads from fishery bycatch that occurs throughout the South Atlantic Ocean. Although national and international governmental and non-governmental entities on both sides of the North Atlantic are currently working toward reducing loggerhead bycatch in the South Atlantic, it is unlikely that this source of mortality can be sufficiently reduced across the range of the DPS in the near future because of the diversity and magnitude of the commercial and artisanal fisheries operating in the South Atlantic, the lack of comprehensive information on fishing distribution and effort, limitations on implementing demonstrated effective conservation measures, geopolitical complexities, limitations on enforcement capacity, and availability of comprehensive bycatch reduction technologies. Therefore, the BRT concluded that although the South Atlantic Ocean DPS is not currently at immediate risk of extinction, the extinction risk is likely to increase substantially in the foreseeable future.

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