Endangered Species Act - Section 7 Consultation
Biological Opinion

Action Agency: U.S. Army Corps of Engineers, Jacksonville District (USACE)

Activity: Permit for the construction of 1.1 miles of rubble-mounded emergent breakwaters off Singer Island, Palm Beach County, Florida. USACE Permit Number SAJ-2006-5344 (Consultation Number F/SER/2008/01896).

Consulting Agency: National Oceanic and Atmospheric Administration, National Marine Fisheries Service (NMFS), Southeast Regional Office, Protected Resources Division, St. Petersburg, Florida

Approved By:

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Background

Section 7(a)(2) of the Endangered Species Act (ESA) of 1973, as amended (16 U.S.C. § 1531 *et seq*.), requires that each federal agency shall ensure that any action authorized, funded, or carried out by such agency is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such species; section 7(a)(2) requires federal agencies to consult with the appropriate Secretary on any such action. The National Marine Fisheries Service (NMFS) and the U.S. Fish and Wildlife Service (USFWS) share responsibilities for administering the ESA.

Consultation is required when a federal action agency determines that a proposed action "may affect" listed species or designated critical habitat. Consultation is concluded after NMFS determines that the action is not likely to adversely affect listed species or critical habitat or issues a biological opinion (opinion) that identifies whether a proposed action is likely to jeopardize the continued existence of a listed species, or destroy or adversely modify critical habitat; if jeopardy or adverse modification are likely to result, reasonable and prudent alternatives (RPAs) that avoid these violations of section 7(a)(2) of the ESA must be implemented. Non-jeopardy opinions state the amount or extent of incidental take of the listed species that may occur, develop measures (i.e., reasonable and prudent measures - RPMs) to reduce the effect of take, and recommend conservation measures to further conserve the species.

This document represents NMFS' opinion based on our review of impacts associated with the proposed U.S. Army Corps of Engineers' (USACE) permitting of the construction of 1.1 miles of rubble-mounded emergent breakwaters off Singer Island, Palm Beach County, Florida, between DNR monuments R-61 and R-68. The purpose of the project is to attempt to stop erosion that is occurring along that stretch of beach, and to facilitate accretion of sand shoreward of the structures.

This opinion analyzes project effects on loggerhead, leatherback, hawksbill, and green sea turtles, in accordance with section 7 of the ESA, and is based on project information provided by the Florida Fish and Wildlife Conservation Commission (FWC), the Florida Department of Environmental Protection (FDEP), the Palm Beach County Department of Environmental Resource Management (DERM) and their consultants, the USACE, and other sources of information including the published literature cited herein.

BIOLOGICAL OPINION

1 CONSULTATION HISTORY

NMFS received a letter from the USACE dated September 28, 2007, regarding the Singer Island breakwater project. The letter requested ESA section 7 consultation on the project, and determined that the project may affect listed acroporid corals, and may affect but is not likely to adversely affect smalltooth sawfish and swimming sea turtles under NMFS' jurisdiction. NMFS requested additional information by letter dated October 4, 2007, regarding the need for a survey to determine if acroporid corals are present in the project area or downstream areas that may be impacted, as well as inquiring about other aspects of the project. The applicant's consultant, Humiston & Moore, responded directly to NMFS with a letter dated March 27, 2008, indicating that an acroporid survey following NMFS' prescribed protocol will be performed, providing additional design drawings, and answering questions regarding turbidity controls.

During the first half of 2008, a conference call with Palm Beach County, their consultants, NMFS, USFWS, and state agencies was held to discuss both the Singer Island project and a Juno Beach nourishment project. NMFS expressed concern regarding the applicant's and its consultant's supposition that the breakwaters would not pose a problem for hatchling sea turtles attempting to access open waters. On June 6, 2008, NMFS contacted Dr. Michael Salmon and Dr. Ken Lohman, researchers who have conducted numerous sea turtle hatchling orientation studies and are considered top experts on that subject. Both scientists replied that the breakwaters were in fact a likely impediment to hatchling sea turtles, and that it is inappropriate to assume that hatchlings will orient through gaps between breakwater segments, based on wave directions.

On August 29, 2008, the USACE forwarded reports from the applicant's consultant showing that a survey was performed according to the protocol for acroporid corals, and no *Acropora spp*. was found in the project area or nearby areas that may be impacted. Therefore, the formal consultation request for acroporid corals was terminated.

On February 24, 2009, NMFS and the USACE further discussed the project via telephone and NMFS recommended that the USACE change its effects determination for sea turtles to "likely to adversely affect" from "may affect, not likely to adversely affect," and enter into formal consultation. The fact that significant sea turtle impacts were expected was reiterated in a March 6, 2009, phone call. A meeting was then held in Palm Beach County on March 12, 2009, with NMFS, USFWS, Palm Beach County, Florida Wildlife Research Institute (FWRI), and FDEP to discuss the matter further. In a letter dated March 24, 2009, the USACE provided NMFS a biological assessment (BA) and analysis of lack of alternatives for the Singer Island breakwaters project.

Despite the information provided in the BA, the Services still maintained reservations about the lack of alternatives analysis provided by Palm Beach County, especially given the scope of the proposed project and the likely significant impacts that would result. At that point the USACE was still in the process of drafting an environmental assessment and also had concerns that additional alternatives needed to be better presented for analysis in order to make appropriate permitting decisions. After further internal NMFS discussions and analysis of the proposed project, a call was set up between the USACE, NMFS, and USFWS on January 20, 2010, to discuss the section 7 consultation and permitting. NMFS explained that the analysis done for the consultation up to that point indicated a very significant impact to sea turtles could be expected from this project. The USACE reiterated that the project was not simply awaiting ESA consultations for permit issuance, but that they also had substantial concerns with the project from both an ESA standpoint and design standpoints, and that they would push for further alternatives analyses. Additional meetings between NMFS and USFWS and FWRI staff in late January further solidified NMFS' analysis that the impacts of the project would be substantial.

On February 16, 2010, a meeting was held in Palm Beach County with representatives from Palm Beach County DERM, the Palm Beach County Commission, Humiston & Moore, the USACE, the DEP Bureau of Beaches, FWC, USFWS, and NMFS. The USACE discussed their concerns with the project and lack of alternatives, both from an ESA standpoint and engineering design. NMFS and USFWS explained the expected impacts to sea turtles and that although any extensive breakwater system will have potentially significant impacts, the fully emergent nature of the proposed design was the largest issue as it would compound other impacts on sea turtles. Palm Beach County concluded that there was no other feasible design, but that they would get engineering information from the USACE and determine if they would further explore alternatives. On February 23, 2010, there was a follow-up conference call between the USACE, Palm Beach County, USFWS, and NMFS during which the county announced it is still proceeding with the application process for the proposed design, but is also looking to develop possible alternatives. NMFS informed the participants that a draft of the biological opinion had been completed and our analysis determined that pending further review and editing, a jeopardy conclusion would likely result. Subsequently, NMFS had been asked to delay finalizing the opinion until March 31, 2010, so that the county can determine if they will pursue a different alternative. The USFWS is also conducting formal ESA section 7 consultation with the USACE regarding the project impacts to sea turtles under their purview, on the nesting beaches.

2 DESCRIPTION OF THE PROPOSED ACTION AND ACTION AREA

2.1 Proposed Action

The objective of the proposed action is to stop the erosion that has been occurring at the beach and increase the accretion of sand to rebuild the beach. Recent storms and regular wave action, coupled with the loss of dune systems due to placement of construction projects, have eroded the beach at approximately 15 feet per year since 2001, advancing to within a few feet of some condominium buildings. After conducting annual upland dune restoration projects since 2002, Palm Beach County believes that the best approach to restore the beach is to construct a series of breakwater structures to attempt to cause accretion of sand shoreward of the structures.

The applicant proposes to build a series of eleven emergent rubble-mound breakwaters stretching a length of approximately 1.1 miles north to south off Singer Island between DEP monuments R-60 and R-66. The breakwaters will be built from washed granite and limestone boulders. Each segment will be approximately 200 feet offshore at a depth of -10 to -16 feet NGVD. A typical segment will be 103 feet wide and 362 feet long at the base. The crest of each structure will be 240 feet in length and be continually emergent at +3 feet NGVD (mean high water is +1.98 feet NGVD). Each breakwater will be separated by approximately 300 feet crest to crest and 180 feet base to base.

Construction will be done using a long-armed excavator and rock grappler operated from a barge. Construction activities will be limited to daylight hours and no equipment will be left in the project area overnight. Construction will take place over the summer months when the waters are calmer, over a two year span. Some construction may take place over winter months weather permitting.

The breakwater structures will directly cover 0.09 acres of persistent hard bottom habitat, while an additional 1.4 acres of this low relief hard bottom habitat is expected to be permanently covered by sand as a result of the project. Periodic covering by sand as a result of the project will impact 1.3 acres, resulting in a total of 2.79 acres of low profile hard bottom habitat being lost through permanent and periodic covering.

Construction related to this project will occur south of the breakwaters. As part of the mitigation recommended by NMFS' Habitat Conservation Division and required by the state of Florida, the applicant will construct artificial reefs to help replace lost habitat value resulting from the project. This construction will occur on sand habitat. The artificial reefs are to be located in waters deeper than the natural hardbottom habitat and outside of the dynamic surf zone. Although some habitat value will be replaced, it will not fully replicate the lost habitat in functionality.

2.2 Action Area

50 CFR 404.02 defines action area as "all areas to be affected directly or indirectly by the federal action and not merely the immediate area involved in the action." The action area for this activity is Singer Island, Palm Beach County, Florida, between DNR monuments R-61 and R-68, from the dunes out past the breakwater segments (200 feet offshore), and adjacent areas which may be impacted by changes in sand transport. The submerged bottom is dynamic and includes a mixture of exposed nearshore hard bottom habitat, sediment over hard bottom, and sand. A small amount of the hard bottom in the area is persistently exposed, though of generally low profile. The remainder of the hard bottom is ephemeral, with periods of exposure and periods of burial dependent upon wave action and storms.

3 STATUS OF LISTED SPECIES AND CRITICAL HABITAT

The following endangered (E) and threatened (T) species under the jurisdiction of NMFS may occur in or near the action area:

Common Name	Scientific Name	Status
Sea Turtles		
Loggerhead sea turtle	*Caretta caretta*	T
Kemp's ridley sea turtle	*Lepidochelys kempii*	E
Green sea turtle	*Chelonia mydas*[1]	E/T
Leatherback sea turtle	*Dermochelys coriacea*	E
Hawksbill sea turtle	*Eretmochelys imbricata*	E
Coral		
Elkhorn coral	*Acropora palmata*	T
Staghorn coral	*Acropora cervicornis*	T
Fish		
Smalltooth sawfish	*Pristis pectinata*	T

There is no NMFS-designated critical habitat within the action area.

3.1 Species Not Likely to Be Adversely Affected

Sea Turtles

NMFS has analyzed the routes of potential effects of the proposed action and, based on our analysis, determined that potential effects are limited to the following: injury from interactions with construction equipment and breakwater boulder placement, temporary avoidance of the area during construction operations, loss of nearshore foraging and resting habitat, impedance of nesting females to reach the beach, and increased mortality of hatchling sea turtles in the water as a result of exhaustion attempting to get around the breakwaters and/or increased predation. Effects related to failure to nest (i.e., "false crawls") and/or loss of nests and nesting habitat are under USFWS' purview and will not be discussed in this biological opinion.

NMFS concludes that the project's construction effects on all sea turtle species are discountable because turtles are highly mobile and likely to avoid the area during construction activities. In addition, the applicant will be required to follow NMFS' March 23, 2006, Sea Turtle and Smalltooth Sawfish Construction Conditions, which will further reduce the potential for interactions with sea turtles from the proposed project.

NMFS believes the proposed project may affect, but is not likely to adversely affect, Kemp's ridley sea turtles. Kemp's ridley sea turtles rarely occur in the nearshore waters of Palm Beach County. Palm Beach County Reef Research Team divers observed 265 sea turtles of 4 species from January 2005 through November 2006 near natural and artificial reef habitats offshore

[1] Green turtles in U.S. waters are listed as threatened except for the Florida breeding population, which is listed as endangered.

Palm Beach County. During this period, divers observed only 1 Kemp's ridley sea turtle (compared to 130 hawksbill turtles, 111 loggerhead turtles, and 23 green turtles). NMFS analyzed FWC's stranding data in Palm Beach County for the years 1999-2004. During this period, only 5 Kemp's ridley turtles were reported stranded (compared to 221 loggerhead turtles, 218 green turtles, 29 hawksbill turtles, and 17 leatherback turtles). Adult Kemp's ridley sea turtles primarily occupy neritic habitats with muddy or sandy bottoms where prey can be found. Their diet consists mainly of swimming crabs, but may also include fish, jellyfish, and an array of mollusks (http://www.nmfs.noaa.gov/pr/species/turtles/kempsridley.htm). Because the action area does not contain preferred foraging habitat for Kemp's ridley sea turtles, NMFS believes any effects on this species of avoiding the area due to construction activities will be insignificant (i.e., would not result in take). Additionally, Kemp's ridley turtles are not known to nest on Palm Beach County beaches, with no recorded nests in the area. As a result, this species will not be discussed further in this opinion.

Smalltooth Sawfish

NMFS believes the proposed project may affect, but is not likely to adversely affect, smalltooth sawfish. No density or population estimates are available, but there have been a total of 14 sightings of smalltooth sawfish in waters off Palm Beach County between 2000 and 2006, with all of the offshore individuals being above 200 cm TL and most between 300-500+ cm TL (combined sightings data from Mote Marine Laboratory and FWCC). However, these sightings are typically offshore for larger individuals. Smaller individuals utilize shallow waters with mangrove or other such habitats. In the unlikely event that a sawfish were to utilize the action area, the construction activities are not expected to impact the individual as they are highly mobile and could move to a nearby area of the same habitat type. In addition, the applicant will be required to follow NMFS' March 23, 2006, Sea Turtle and Smalltooth Sawfish Construction Conditions, which will further reduce the potential for interactions with sawfish from the proposed project. The habitat impacts of the project are not expected to impact sawfish as the habitat types to be impacted are not typically utilized by this species. Therefore, this species will not be considered further in this biological opinion.

Acroporid Corals

NMFS believes the proposed project will not affect elkhorn and staghorn coral. On August 29, 2008, the USACE forwarded reports from the applicant's consultant showing that a survey was performed according to the protocol for acroporid corals, and no acropora was found in the project area or nearby areas that may be impacted. Designated critical habitat for acropora does not occur in the area of impact. Therefore, these species will not be considered further in this biological opinion.

3.2 Species Likely to Be Affected

NMFS believes the proposed action is likely to adversely affect green, hawksbill, leatherback, and loggerhead sea turtles. The species discussions in this section will focus primarily on the Atlantic Ocean populations of these species since these are the populations that may be affected by the proposed action. The following subsections are synopses of the best available information on the life history, distribution, population trends, and current status of the four species of sea turtles that are likely to be adversely affected by one or more components of the proposed action. Additional background information on the status of sea turtle species can be found in a number of published documents, including: recovery plans for the Atlantic green sea turtle (NMFS and

USFWS 1991), hawksbill sea turtle (NMFS and USFWS 1993), leatherback sea turtle (NMFS and USFWS 1992), and loggerhead sea turtle (NMFS and USFWS 2008); and sea turtle status reviews and biological reports (Conant et al. 2009, NMFS and USFWS 1995, Marine Turtle Expert Working Group [TEWG] 1998, 2000, 2007, and 2009, NMFS 2001a). However, because all four species are listed globally we also discuss their worldwide status.

3.2.1 Green Sea Turtle

Green turtles are distributed circumglobally and can be found in the Pacific, Indian, and Atlantic Oceans as well as the Mediterranean Sea (NMFS and USFWS 1991, Seminoff 2004, NMFS and USFWS 2007a). In 1978, the Atlantic population of the green sea turtle was listed as threatened under the ESA, except for the breeding populations in Florida and on the Pacific coast of Mexico, which were listed as endangered.

3.2.1.1 Pacific Ocean

Green turtles occur in the eastern, central, and western Pacific. Foraging areas are also found throughout the Pacific and along the southwestern U.S. coast (NMFS and USFWS 1998a). Nesting is known to occur in the Hawaiian archipelago, American Samoa, Guam, and various other sites in the Pacific. The only major population (>2,000 nesting females) of green turtles in the western Pacific occurs in Australia and Malaysia, with smaller colonies throughout the area. Green turtles have generally been thought to be declining throughout the Pacific Ocean, with the exception of Hawaii, from a combination of overexploitation and habitat loss (Seminoff 2002). Indonesia has a widespread distribution of green turtles, but has experienced large declines over the past 50 years. Historically, green turtles were used in many areas of the Pacific for food. They were also commercially exploited and this, coupled with habitat degradation, led to their decline in the Pacific (NMFS and USFWS 1998a). Green turtles in the Pacific continue to be affected by poaching, habitat loss or degradation, fishing gear interactions, and fibropapillomatosis (NMFS and USFWS 1998a, NMFS 2004).

Hawaiian green turtles are genetically distinct and geographically isolated, and the population appears to be increasing in size despite the prevalence of fibropapilloma and spirochidiasis (Aguirre et al. 1998 in Balazs and Chaloupka 2003). The East Island nesting beach in Hawaii is showing a 5.7 percent annual growth rate over 25 plus years (Chaloupka et al. 2007). In the Eastern Pacific, mitochondrial DNA analysis has indicated that there are three key nesting populations: Michoacán, Mexico; Galapagos Islands, Ecuador; and Islas Revillagigedos, Mexico (Dutton 2003). The number of nesting females per year exceeds 1,000 females at each site (NMFS and USFWS 2007a). However, historically, greater than 20,000 females per year are believed to have nested in Michoacán alone (Cliffton et al. 1982, NMFS and USFWS 2007a). Thus, the current number of nesting females is still far below what has historically occurred. There is also sporadic green turtle nesting along the Pacific coast of Costa Rica. At least a few of the non-Hawaiian nesting stocks in the Pacific have recently been found to be undergoing long-term increases. Datasets over 25 years in Chichi-jima, Japan; Heron Island, Australia; and Raine Island, Australia show increases (Chaloupka et al. 2007). These increases are thought to be the direct result of long-term conservation measures.

3.2.1.2 Indian Ocean

There are numerous nesting sites for green sea turtles in the Indian Ocean. One of the largest nesting sites for green sea turtles worldwide occurs on the beaches of Oman where an estimated 20,000 green sea turtles nest annually (Hirth 1997). Based on a review of the 32 index sites used to monitor green sea turtle nesting worldwide, Seminoff (2004) concluded that declines in green turtle nesting were evident for many of the Indian Ocean index sites. While several of these had not demonstrated further declines in the more recent past, only the Comoros Island index site in the western Indian Ocean showed evidence of increased nesting (Seminoff 2004).

3.2.1.3 Atlantic Ocean

Life History and Distribution

The estimated age at sexual maturity for green sea turtles is between 20-50 years (Balazs 1982, Frazer and Ehrhart 1985). Green sea turtle mating occurs in the waters off the nesting beaches. Each female deposits 1-7 clutches (usually 2-3) during the breeding season at 12-14 day intervals. Mean clutch size is highly variable among populations, but averages 110-115 eggs/nest. Females usually have 2-4 or more years between breeding seasons, whereas males may mate every year (Balazs 1983). After hatching, green sea turtles go through a post-hatchling pelagic stage during which they are associated with drift lines of algae and other debris. At approximately 20- to 25-cm carapace length, juveniles leave pelagic habitats and enter benthic foraging areas (Bjorndal 1997).

Green sea turtles are primarily herbivorous, feeding on algae and sea grasses, but also occasionally consume jellyfish and sponges. The post-hatchling, pelagic-stage individuals are assumed to be omnivorous, but little data are available.

Green sea turtle foraging areas in the southeastern United States include any coastal shallow waters having macroalgae or seagrasses. This includes areas near mainland coastlines, islands, reefs, or shelves, as well as open-ocean surface waters, especially where advection from wind and currents concentrates pelagic organisms (Hirth 1997, NMFS and USFWS 1991). Principal benthic foraging areas in the southeastern United States include Aransas Bay, Matagorda Bay, Laguna Madre, and the Gulf inlets of Texas (Doughty 1984, Hildebrand 1982, Shaver 1994), the Gulf of Mexico off Florida from Yankeetown to Tarpon Springs (Caldwell and Carr 1957, Carr 1984), Florida Bay and the Florida Keys (Schroeder and Foley 1995), the Indian River Lagoon system, Florida (Ehrhart 1983), and the Atlantic Ocean off Florida from Brevard through Broward Counties (Wershoven and Wershoven 1992, Guseman and Ehrhart 1992). Adults of both sexes are presumed to migrate between nesting and foraging habitats along corridors adjacent to coastlines and reefs.

Some of the principal feeding pastures in the western Atlantic Ocean include the upper west coast of Florida and the northwestern coast of the Yucatán Peninsula. Additional important foraging areas in the western Atlantic include the Mosquito Lagoon and Indian River Lagoon systems and nearshore wormrock reefs between Sebastian and Ft. Pierce Inlets in Florida, Florida Bay, the Culebra archipelago and other Puerto Rico coastal waters, the south coast of Cuba, the Caribbean coast of Panama, the Miskito Coast in Nicaragua, and scattered areas along Colombia and Brazil (Hirth 1997). The summer developmental habitat for green turtles also encompasses estuarine and coastal waters from North Carolina to as far north as Long Island Sound (Musick and Limpus 1997).

Population Dynamics and Status

Nest counts can also be used to estimate the number of reproductively mature females nesting annually. The 5-year status review for the species identified eight geographic areas considered to be primary sites for green sea turtle nesting in the Atlantic/Caribbean and reviewed the trend in nest count data for each (NMFS and USFWS 2007a). These sites include: (1) Yucatán Peninsula, Mexico; (2) Tortuguero, Costa Rica; (3) Aves Island, Venezuela; (4) Galibi Reserve, Suriname; (5) Isla Trindade, Brazil; (6) Ascension Island, United Kingdom; (7) Bioko Island, Equatorial Guinea; and (8) Bijagos Achipelago (Guinea-Bissau) (NMFS and USFWS 2007a). Nesting at all of these sites was considered to be stable or increasing with the exception of Bioko Island and the Bijagos Archipelago where the lack of sufficient data precluded a meaningful trend assessment for either site (NMFS and USFWS 2007a). Seminoff (2004) likewise reviewed green sea turtle nesting data for eight sites in the western, eastern, and central Atlantic, including all of the above with the exception that nesting in Florida was reviewed in place of Isla Trindade, Brazil. Seminoff (2004) concluded that all sites in the central and western Atlantic showed increased nesting with the exception of nesting at Aves Island, Venezuela, while both sites in the eastern Atlantic demonstrated decreased nesting. These sites are not inclusive of all green sea turtle nesting in the Atlantic. However, other sites are not believed to support nesting levels high enough that would change the overall status of the species in the Atlantic (NMFS and USFWS 2007a).

By far, the most important nesting concentration for green turtles in the western Atlantic is in Tortuguero, Costa Rica (NMFS and USFWS 2007a). Nesting in the area has increased considerably since the 1970s, and nest count data from 1999-2003 suggest nesting by 17,402-37,290 females per year (NMFS and USFWS 2007a). The number of females nesting per year on beaches in the Yucatán, Aves Island, Galibi Reserve, and Isla Trindade number in the hundreds to low thousands, depending on the site (NMFS and USFWS 2007a). The vast majority of green sea turtle nesting within the southeastern United States occurs in Florida (Meylan et al. 1995, Johnson and Ehrhart 1994). Green sea turtle nesting in Florida has been increasing since 1989 (Florida Fish and Wildlife Conservation Commission, Florida Marine Research Institute Index Nesting Beach Survey Database). Certain Florida nesting beaches have been designated index beaches. Index beaches were established to standardize data collection methods and effort on key nesting beaches. The pattern of green turtle nesting shows biennial peaks in abundance with a generally positive trend during the ten years of regular monitoring since establishment of the index beaches in 1989, perhaps due to increased protective legislation throughout the Caribbean (Meylan et al. 1995). An average of 5,039 green turtle nests were laid annually in Florida between 2001 and 2006, with a low of 581 in 2001 and a high of 9,644 in 2005 (NMFS and USFWS 2007a). Data from the index nesting beaches program in Florida substantiate the dramatic increase in nesting. In 2007, there were 9,455 green turtle nests found just on index nesting beaches, the highest since index beach monitoring began in 1989. The number fell back to 6,385 in 2008, but that is thought to be part of the normal biennial nesting cycle for green turtles (FWC Index Nesting Beach Survey Database). Occasional nesting has been documented along the Gulf coast of Florida, at southwest Florida beaches, as well as the beaches on the Florida Panhandle (Meylan et al. 1995). More recently, green turtle nesting occurred on Bald Head Island, North Carolina; just east of the mouth of the Cape Fear River; on

Onslow Island; and on Cape Hatteras National Seashore. Increased nesting has also been observed along the Atlantic coast of Florida, on beaches where only loggerhead nesting was observed in the past (Pritchard 1997). Recent modeling by Chaloupka et al. (2007) using data sets of 25 years or more has resulted in an estimate of the Florida nesting stock at the Archie Carr National Wildlife Refuge growing at an annual rate of 13.9 percent, and the Tortuguero, Costa Rica, population growing at 4.9 percent annually.

There are no reliable estimates of the number of immature green sea turtles that inhabit coastal areas of the southeastern United States, where they come to forage . However, information on incidental captures of immature green sea turtles at the St. Lucie Power Plant in St. Lucie County, Florida, show that the annual number of immature green sea turtles captured has increased significantly in the past 26 years, and has averaged 215 turtle captures since 1977 (FPL 2002). Ehrhart et al. (2007) has also documented a significant increase in in-water abundance of green turtles in the Indian River Lagoon area. It is likely that immature green sea turtles foraging in the southeastern United States come from multiple genetic stocks; therefore, the status of immature green sea turtles in the southeastern United States might also be assessed from trends at all of the main regional nesting beaches, principally Florida, Yucatán, and Tortuguero.

Threats

The principal cause of past declines and extirpations of green sea turtle assemblages has been the overexploitation of green sea turtles for food and other products. Although intentional take of green sea turtles and their eggs is not extensive within the southeastern United States, green sea turtles that nest and forage in the region may spend large portions of their life history outside the region and outside U.S. jurisdiction, where exploitation is still a threat. However, there are still significant and ongoing threats to green sea turtles from human-related causes in the United States. These threats include beach armoring, erosion control, artificial lighting, beach disturbance (e.g., driving on the beach), pollution, foraging habitat loss as a result of direct destruction by dredging, siltation, boat damage, other human activities, and interactions with fishing gear. Sea sampling coverage in the pelagic driftnet, pelagic longline, Southeast shrimp trawl, and summer flounder bottom trawl fisheries has recorded takes of green turtles. There is also the increasing threat from green sea turtle fibropapillomatosis disease. Presently, this disease is cosmopolitan and has been found to affect large numbers of animals in some areas, including Hawaii and Florida (Herbst 1994, Jacobson 1990, Jacobson et al. 1991).

There is a large and growing body of literature on past, present, and future impacts of global climate change exacerbated and accelerated by human activities. Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. NOAA's climate information portal provides basic background information on these and other measured or anticipated effects (see http://www.climate.gov).

Impacts on sea turtles currently cannot, for the most part, be predicted with any degree of certainty, however significant impacts to the hatchling sex ratios of green turtles may result (NMFS and USFWS 2007a). In marine turtles, sex is determined by temperature in the middle third of incubation, with female offspring produced at higher temperatures and males at lower temperatures within a thermal tolerance range of 25º-35°C (Ackerman 1997). Increases in global temperature could potentially skew future sex ratios toward higher numbers of females (NMFS and USFWS 2007a). Green sea turtle hatchling size also appears to be influenced by incubation temperatures, with smaller hatchlings produced at higher temperatures (Glen et al. 2003).

The effects from increased temperatures may be exacerbated on developed nesting beaches where shoreline armoring and construction has denuded vegetation. Sea level rise from global climate change is also a potential problem, for areas with low-lying beaches where sand depth is a limiting factor, as the sea may 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 increased 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).

Other changes in the marine ecosystem caused by global climate change (e.g., salinity, oceanic currents, dissolved oxygen levels, nutrient distribution, etc.) could influence the distribution and abundance of phytoplankton, zooplankton, submerged aquatic vegetation, forage fish, etc., which could ultimately affect the primary foraging areas of green sea turtles.

3.2.1.3 Summary of Status for Atlantic Green Sea Turtles

Green turtles range in the western Atlantic from Massachusetts to Argentina, including the Gulf of Mexico and the Caribbean Sea, but are considered rare in benthic areas north of Cape Hatteras (Wynne and Schwartz 1999). Green turtles face many of the anthropogenic threats for other sea turtles described herein. In addition, green turtles are also susceptible to fibropapillomatosis, which can result in death. In the continental United States, green turtle nesting occurs on the Atlantic coast of Florida (Ehrhart 1979). Recent population estimates for the western Atlantic area are not available. The pattern of green turtle nesting shows biennial peaks in abundance, with a generally positive trend during the almost 20 years of regular monitoring since establishment of index beaches in Florida in 1989.

3.2.2 Hawksbill Sea Turtle

The hawksbill turtle was listed as endangered under the precursor of the ESA on June 2, 1970, and is considered critically endangered by the International Union for the Conservation of Nature (IUCN). The hawksbill is a medium-sized sea turtle, with adults in the Caribbean ranging in size from approximately 62.5 to 94.0 cm straight carapace length. The species occurs in all ocean basins, although it is relatively rare in the Eastern Atlantic and Eastern Pacific, and absent from the Mediterranean Sea. Hawksbills are the most tropical sea turtle species, ranging from approximately 30°N latitude to 30°S latitude. They are closely associated with coral reefs and other hardbottom habitats, but they are also found in other habitats including inlets, bays, and coastal lagoons (NMFS and USFWS 1993). There are only five remaining regional nesting populations with more than 1,000 females nesting annually. These populations are in the Seychelles, Mexico, Indonesia, and two in Australia (Meylan and Donnelly 1999). There has been a global population decline of over 80 percent during the last three generations (105 years) (Meylan and Donnelly 1999).

3.2.2.1 Pacific Ocean

Anecdotal reports throughout the Pacific indicate the current Pacific hawksbill population is well below historical levels (NMFS 2004). It is believed that this species is rapidly approaching extinction in the Pacific because of harvesting for its meat, shell, and eggs as well as destruction of nesting habitat (NMFS 2004). Hawksbill sea turtles nest in the Hawaiian Islands as well as the islands and mainland of Southeast Asia, from China to Japan, and throughout the Philippines, Malaysia, Indonesia, Papua New Guinea, the Solomon Islands, and Australia (NMFS 2004). However, along the eastern Pacific Rim where nesting was common in the 1930s, hawksbills are now rare or absent (Cliffton et al. 1982, NMFS 2004).

3.2.2.2 Atlantic Ocean

In the western Atlantic, the largest hawksbill nesting population occurs on the Yucatán Peninsula of Mexico (Garduño-Andrade et al. 1999). With respect to the United States, nesting occurs in Puerto Rico, the U.S. Virgin Islands, and along the southeast coast of Florida. Nesting also occurs outside of the United States and its territories, in Antigua, Barbados, Costa Rica, Cuba, and Jamaica (Meylan 1999). Outside of the nesting areas, hawksbills have been seen off the U.S. Gulf of Mexico states and along the Eastern Seaboard as far north as Massachusetts, although sightings north of Florida are rare (NMFS and USFWS 1993).

Life History and Distribution

The best estimate of age at sexual maturity for hawksbill sea turtles is about 20-40 years (Chaloupka and Limpus 1997, Crouse 1999a). Reproductive females undertake periodic (usually non-annual) migrations to their natal beach to nest. Movements of reproductive males are less well known, but are presumed to involve migrations to their nesting beach or to courtship stations along the migratory corridor (Meylan 1999). Females nest an average of 3-5 times per season (Meylan and Donnelly 1999, Richardson et al. 1999). Clutch size is larger on average (up to 250 eggs) than that of other sea turtles (Hirth 1980). Reproductive females may exhibit a high degree of fidelity to their nest sites.

The life history of hawksbills consists of a pelagic stage that lasts from the time they leave the nesting beach as hatchlings until they are approximately 22-25 cm in straight carapace length (Meylan 1988, Meylan and Donnelly 1999), followed by residency in developmental habitats (foraging areas where juveniles reside and grow) in coastal waters. Adult foraging habitat, which may or may not overlap with developmental habitat, is typically coral reefs, although other hard-bottom communities and occasionally mangrove-fringed bays may be occupied. Hawksbills show fidelity to their foraging areas over several years (van Dam and Díez 1998).

The hawksbill's diet is highly specialized and consists primarily of sponges (Meylan 1988). Other food items, notably corallimorphs and zooanthids, have been documented to be important in some areas of the Caribbean (van Dam and Díez 1997, Mayor et al. 1998).

Population Dynamics and Status

Nesting within the southeastern United States and U.S. Caribbean is restricted to Puerto Rico (>650 nests/yr), the U.S. Virgin Islands (~400 nests/yr), and, rarely, Florida (0-4 nests/yr) (Eckert 1995, Meylan 1999, Florida Fish and Wildlife Conservation Commission, Florida Marine Research Institute's Statewide Nesting Beach Survey data 2002). At the two principal nesting beaches in the U.S. Caribbean where long-term monitoring has been carried out, populations appear to be increasing (Mona Island, Puerto Rico) or stable (Buck Island Reef National Monument, St. Croix, USVI) (Meylan 1999).

Threats
As with other sea turtle species, hawksbill sea turtles are affected by habitat loss, habitat degradation, marine pollution, marine debris, fishery interactions, and poaching in some parts of their range. There continues to be a black market for hawksbill shell products ("tortoiseshell"), which likely contributes to the harvest of this species.

There is a large and growing body of literature on past, present, and future impacts of global climate change exacerbated and accelerated by human activities. Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. NOAA's climate information portal provides basic background information on these and other measured or anticipated effects (see http://www.climate.gov).

Impacts on sea turtles currently cannot, for the most part, be predicted with any degree of certainty, however significant impacts to the hatchling sex ratios of hawksbill turtles may result (NMFS and USFWS 2007d). In marine turtles, sex is determined by temperature in the middle third of incubation with female offspring produced at higher temperatures and males at lower temperatures within a thermal tolerance range of 25º-35°C (Ackerman 1997). Increases in global temperature could potentially skew future sex ratios toward a higher numbers of females (NMFS and USFWS 2007d).

The effects from increased temperatures may be exacerbated on developed nesting beaches where shoreline armoring and construction has denuded vegetation. Sea level rise from global climate change is also a potential problem for areas with low-lying beaches where sand depth is a limiting factor, as the sea may 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 increased 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).

Other changes in the marine ecosystem caused by global climate change (e.g., salinity, oceanic currents, dissolved oxygen levels, nutrient distribution, etc.) could influence the distribution and abundance of phytoplankton, zooplankton, submerged aquatic vegetation, coral reefs, forage fish, etc. Since hawksbills are typically associated with coral reef ecosystems, increases in global temperatures leading to coral death (Sheppard 2006) could adversely affect the foraging habitats of this species.

3.2.2.3 Summary of Status for Hawksbill Sea Turtles

Worldwide, hawksbill sea turtle populations are declining. They face many of the same threats affecting other sea turtle species. In addition, there continues to be a commercial market for hawksbill shell products, despite protections afforded to the species under U.S. law and international conventions.

3.2.3 Leatherback Sea Turtle

The leatherback sea turtle was listed as endangered throughout its global range on June 2, 1970. Leatherbacks are widely distributed throughout the oceans of the world and are found in waters of the Atlantic, Pacific, and Indian Oceans (Ernst and Barbour 1972). Leatherback sea turtles are the largest living turtles and range farther than any other sea turtle species. The large size of adult leatherbacks and their tolerance to relatively low temperatures allows them to occur in northern waters such as off Labrador and in the Barents Sea (NMFS and USFWS 1995). Adult leatherbacks forage in temperate and subpolar regions from 71ºN to 47ºS latitude in all oceans and undergo extensive migrations to and from their tropical nesting beaches. In 1980, the leatherback population was estimated at approximately 115,000 adult females globally (Pritchard 1982). That number, however, is probably an overestimation as it was based on a particularly good nesting year in 1980 (Pritchard 1996). By 1995, the global population of adult females had declined to 34,500 (Spotila et al. 1996). Pritchard (1996) also called into question the population estimates from Spotila et al. (1996) and felt they may be somewhat low because it ended the modeling on data from a particularly bad nesting year (1994) while excluding nesting data from 1995, which was a good nesting year. The most recent population estimate for leatherback sea turtles from just the North Atlantic breeding groups is a range of 34,000-90,000 adult individuals (20,000-56,000 adult females) (TEWG 2007).

3.2.3.1 Pacific Ocean

Based on published estimates of nesting female abundance, leatherback populations have collapsed or have been declining at all major Pacific basin nesting beaches for the last two decades (Spotila et al. 1996, NMFS and USFWS 1998b, Sarti et al. 2000, Spotila et al. 2000). For example, the nesting assemblage on Terengganu, Malaysia–which was one of the most significant nesting sites in the western Pacific Ocean–has declined severely from an estimated 3,103 females in 1968 to 2 nesting females in 1994 (Chan and Liew 1996). Nesting assemblages of leatherback turtles are in decline along the coasts of the Solomon Islands, a historically important nesting area (D. Broderick, pers. comm., in Dutton et al. 1999). In Fiji, Thailand, Australia, and Papua New Guinea (East Papua), leatherback turtles have only been known to nest in low densities and scattered colonies.

Only an Indonesian nesting assemblage has remained relatively abundant in the Pacific basin. The largest extant leatherback nesting assemblage in the Indo-Pacific lies on the north Vogelkop coast of Irian Jaya (West Papua), Indonesia, with over 3,000 nests recorded annually (Putrawidjaja 2000, Suárez et al. 2000). During the early-to-mid 1980s, the number of female leatherback turtles nesting on the two primary beaches of Irian Jaya appeared to be stable. More recently, this population has come under increasing threats that could cause this population to experience a collapse that is similar to what occurred at Terengganu, Malaysia. In 1999, for example, local Indonesian villagers started reporting dramatic declines in sea turtle populations near their villages (Suárez 1999). Unless hatchling and adult turtles on nesting beaches receive more protection, this population will continue to decline. Declines in nesting assemblages of leatherback turtles have been reported throughout the western Pacific region, with nesting assemblages well below abundance levels observed several decades ago (e.g., Suárez 1999).

In the western Pacific Ocean and South China Seas, leatherback turtles are captured, injured, or killed in numerous fisheries, including Japanese longline fisheries. The poaching of eggs, killing of nesting females, human encroachment on nesting beaches, beach erosion, and egg predation by animals also threaten leatherback turtles in the western Pacific.

In the eastern Pacific Ocean, nesting populations of leatherback turtles are declining along the Pacific coast of Mexico and Costa Rica. According to reports from the late 1970s and early 1980s, three beaches on the Pacific coast of Mexico supported as many as half of all leatherback turtle nests for the eastern Pacific. Since the early 1980s, the eastern Pacific Mexican population of adult female leatherback turtles has declined to slightly more than 200 individuals during 1998-1999 and 1999-2000 (Sarti et al. 2000). Spotila et al. (2000) reported the decline of the leatherback turtle population at Playa Grande, Costa Rica, which had been the fourth largest nesting colony in the world. Between 1988 and 1999, the nesting colony declined from 1,367 to 117 female leatherback turtles. Based on their models, Spotila et al. (2000) estimated that the colony could fall to less than 50 females by 2003-2004. Leatherback turtles in the eastern Pacific Ocean are captured, injured, or killed in commercial and artisanal swordfish fisheries off Chile, Columbia, Ecuador, and Peru, and purse seine fisheries for tuna in the eastern tropical Pacific Ocean, and California/Oregon drift gillnet fisheries. Because of the limited data, we cannot provide high-certainty estimates of the number of leatherback turtles captured, injured, or killed through interactions with these fisheries. However, between 8-17 leatherback turtles were estimated to have died annually between 1990 and 2000 in interactions with the California/Oregon drift gillnet fishery; 500 leatherback turtles are estimated to die annually in Chilean and Peruvian fisheries; 200 leatherback turtles are estimated to die in direct harvests in Indonesia; and before 1992 the North Pacific driftnet fisheries for squid, tuna, and billfish captured an estimated 1,000 leatherback turtles each year, killing about 111 of them each year.

Although all causes of the declines in leatherback turtle colonies in the eastern Pacific have not been documented, Sarti et al. (1998) suggest that the declines result from egg poaching, adult and subadult mortalities incidental to high seas fisheries, and natural fluctuations due to changing environmental conditions. Some published reports support this suggestion. Sarti et al. (2000) reported that female leatherback turtles have been killed for meat on nesting beaches like Píedra de Tiacoyunque, Guerrero, Mexico. Eckert (1997) reported that swordfish gillnet fisheries in Peru and Chile contributed to the decline of leatherback turtles in the eastern Pacific. The decline in the nesting population at Mexiquillo, Mexico, occurred at the same time that effort doubled in the Chilean driftnet fishery. In response to these effects, the eastern Pacific population has continued to decline, leading some researchers to conclude that the leatherback is on the verge of extinction in the Pacific Ocean (e.g., Spotila et al. 1996, Spotila et al. 2000). The NMFS assessment of three nesting aggregations in its February 23, 2004, opinion supports this conclusion: If no action is taken to reverse their decline, leatherback sea turtles nesting in the Pacific Ocean either have high risks of extinction in a single human generation (for example, nesting aggregations at Terrenganu and Costa Rica) or they have a high risk of declining to levels where more precipitous declines become almost certain (e.g., Irian Jaya) (NMFS 2004).

3.2.3.2 Atlantic Ocean

In the Atlantic Ocean, leatherbacks have been recorded as far north as Newfoundland, Canada, and Norway, and as far south as Uruguay, Argentina, and South Africa (NMFS 2001a). Female leatherbacks nest from the southeastern United States to southern Brazil in the western Atlantic and from Mauritania to Angola in the eastern Atlantic. The most significant nesting beaches in the Atlantic, and perhaps in the world, are in French Guiana and Suriname (NMFS 2001).

Previous genetic analyses of leatherbacks using only mitochondrial DNA (mtDNA) resulted in an earlier determination that within the Atlantic basin there are at least three genetically different nesting populations: the St. Croix nesting population (U.S. Virgin Islands), the mainland nesting Caribbean population (Florida, Costa Rica, Suriname/French Guiana), and the Trinidad nesting population (Dutton et al. 1999). Further genetic analyses using microsatellite markers in nuclear DNA along with the mtDNA data and tagging data has resulted in Atlantic Ocean leatherbacks now being divided into seven groups or breeding populations: Florida, Northern Caribbean, Western Caribbean, Southern Caribbean/Guianas, West Africa, South Africa, and Brazil (TEWG 2007). When the hatchlings leave the nesting beaches, they move offshore but eventually utilize both coastal and pelagic waters. Very little is known about the pelagic habits of the hatchlings and juveniles, and they have not been documented to be associated with the *Sargassum* areas as are other species. Leatherbacks are deep divers, with recorded dives to depths in excess of 1,000 m (Eckert et al. 1989, Hayes et al. 2004).

Life History and Distribution

Leatherbacks are a long-lived species, living for well over 30 years. It has been thought that they reach sexual maturity somewhat faster than other sea turtles (except Kemp's ridley), with an estimated range from 3-6 years (Rhodin 1985) to 13-14 years (Zug and Parham 1996). However, some recent research using sophisticated methods of analyzing leatherback ossicles has cast doubt on the previously accepted age to maturity figures, with leatherbacks in the western North Atlantic possibly not reaching sexual maturity until as late as 29 years of age (Avens and Goshe 2007). Continued research in this area is vitally important to understanding the life history of leatherbacks and has important implications in management of the species.

Female leatherbacks nest frequently (up to 10 nests per year) during a nesting season and nest about every 2-3 years. During each nesting, they produce 100 eggs or more in each clutch and, thus, can produce 700 eggs or more per nesting season (Schultz 1975). However, a significant portion (up to approximately 30 percent) of the eggs can be infertile. Thus, the actual proportion of eggs that can result in hatchlings is less than this seasonal estimate. The eggs incubate for 55-75 days before hatching. Based on a review of all sightings of leatherback sea turtles of <145 cm curved carapace length (ccl), Eckert (1999) found that leatherback juveniles remain in waters warmer than 26ºC until they exceed 100 ccl.

Although leatherbacks are the most pelagic of the sea turtles, they enter coastal waters on an irregular basis to feed in areas where jellyfish are concentrated. Leatherback sea turtles feed primarily on cnidarians (medusae, siphonophores) and tunicates.

Evidence from tag returns and strandings in the western Atlantic suggests that adult leatherback sea turtles engage in routine migrations between boreal, temperate, and tropical waters (NMFS and USFWS 1992). A 1979 aerial survey of the outer continental shelf from Cape Hatteras, North Carolina, to Cape Sable, Nova Scotia, showed leatherbacks to be present throughout the area with the most numerous sightings made from the Gulf of Maine south to Long Island. Leatherbacks were sighted in waters where depths ranged from 1 to 4,151 m, but 84.4 percent of sightings were in areas where the water was less than 180 m deep (Shoop and Kenney 1992). Leatherbacks were sighted in waters of a similar sea surface temperature as loggerheads from 7ºC to 27.2ºC (Shoop and Kenney 1992). However, this species appears to have a greater tolerance for colder waters because more leatherbacks were found at the lower temperatures (Shoop and Kenney 1992). This aerial survey estimated the in-water leatherback population from near Nova Scotia, Canada, to Cape Hatteras, North Carolina, at approximately 300-600 animals.

General differences in migration patterns and foraging grounds may occur between the seven nesting assemblages identified by the TEWG in 2007, but data is limited:

> Marked or satellite tracked turtles from the Florida and North Caribbean assemblages have been re-sighted off North America, in the Gulf of Mexico and along the Atlantic coast and a few have moved to western Africa, north of the equator. In contrast, Western Caribbean and Southern Caribbean/Guianas animals have been found more commonly in the eastern Atlantic, off Europe and northern Africa, as well as along the North American coast. There are no reports of marked animals from the Western North Atlantic assemblages entering the Mediterranean Sea or the South Atlantic Ocean, though in the case of the Mediterranean this may be due more to a lack of data rather than failure of Western North Atlantic turtles moving into the Sea. The tagging data coupled with the satellite telemetry data indicate that animals from the western North Atlantic nesting subpopulations use virtually the entire North Atlantic Ocean. In the South Atlantic Ocean, tracking and tag return data follow three primary patterns. Although telemetry data from the West African nesting assemblage showed that all but one remained on the shallow continental shelf, there clearly is movement to foraging areas of the south coast of Brazil and Argentina. There is also a small nesting aggregation of leatherbacks in Brazil, and while data are limited to a few satellite tracks, these turtles seem to remain in the southwest Atlantic foraging along the continental shelf margin as far south as Argentina. South African nesting turtles apparently forage primarily south, around the tip of the continent.

Population Dynamics and Status

The status of the Atlantic leatherback population has been less clear than the Pacific population. This uncertainty has been a result of inconsistent beach and aerial surveys, cycles of erosion and reformation of nesting beaches in the Guianas (representing the largest nesting area), a lesser degree of nest-site fidelity than occurs with the hardshell sea turtle species, and inconsistencies in the availability and analyses of data. However, recent coordinated efforts at data collection and analyses by the Leatherback Turtle Expert Working Group have helped to clarify the understanding of the Atlantic population status (TEWG 2007).

The Southern Caribbean/Guianas stock is the largest known Atlantic leatherback nesting aggregation (TEWG 2007). This area includes the Guianas (Guyana, Suriname, and French Guiana), Trinidad, Dominica, and Venezuela, with the vast majority of the nesting occurring in the Guianas and Trinidad. Past analyses had shown that the nesting aggregation in French Guiana had been declining at about 15 percent per year since 1987 (NMFS 2001a). However, from 1979-1986, the number of nests was increasing at about 15 percent annually, which could mean that the current decline could be part of a nesting cycle that coincides with the erosion cycle of Guiana beaches described by Schultz (1975). It is thought that the cycle of erosion and reformation of beaches has resulted in shifting nesting beaches throughout this region. This was supported by the increased nesting seen in Suriname, where leatherback nest numbers have shown large recent increases concurrent with declines elsewhere (with more than 10,000 nests per year since 1999 and a peak of 30,000 nests in 2001), and the long-term trend for the overall Suriname and French Guiana population was thought to possibly show an increase (Girondot 2002 in Hilterman and Goverse 2003). In the past, many sea turtle scientists have agreed that the Guianas (and some would include Trinidad) should be viewed as one population and that a synoptic evaluation of nesting at all beaches in the region is necessary to develop a true picture of population status (Reichart et al. 2001). Genetics studies have added support to this notion and have resulted in the designation of the Southern Caribbean/Guianas stock. Using both Bayesian modeling and regression analyses, the TEWG (2007) determined that the Southern Caribbean/Guianas stock had demonstrated a long-term, positive population growth rate (using nesting females as a proxy for population). This positive growth was seen within major nesting areas for the stock, including Trinidad, Guyana, and the combined beaches of Suriname and French Guiana (TEWG 2007).

The Western Caribbean stock includes nesting beaches from Honduras to Colombia. The most intense nesting in that area occurs in Costa Rica, Panama, and the Gulf of Uraba in Colombia (Duque et al. 2000). The Caribbean coast of Costa Rica and extending through Chiriquí Beach, Panama, represents the fourth largest known leatherback rookery in the world (Troëng et al. 2004). Examination of data from three index nesting beaches in the region (Tortuguero, Gandoca, and Pacuare in Costa Rica) using various Bayesian and regression analyses indicated that the nesting population likely was not growing over the 1995-2005 time series of available data (TEWG 2007). Other modeling of the nesting data for Tortuguero indicates a possible 67.8 percent decline between 1995 and 2006 (Troëng et al. 2007).

Nesting data for the Northern Caribbean stock is available from Puerto Rico, the U.S. Virgin Islands (St. Croix), and the British Virgin Islands (Tortola). In Puerto Rico, the primary nesting beaches are at Fajardo and on the island of Culebra. Nesting between 1978 and 2005 has ranged between 469-882 nests, and the population has been growing since 1978, with an overall annual growth rate of 1.1 percent (TEWG 2007). At the primary nesting beach on St. Croix, the Sandy Point National Wildlife Refuge, nesting has fluctuated from a few hundred nests to a high of 1,008 in 2001, and the average annual growth rate has been approximately 1.1 percent from 1986-2004 (TEWG 2007). Nesting in Tortola is limited, but has been increasing from 0-6 nests per year in the late 1980s to 35-65 per year in the 2000s, with an annual growth rate of approximately 1.2 percent between 1994 and 2004 (TEWG 2007).

The Florida nesting stock nests primarily along the east coast of Florida. This stock is of growing importance, with total nests between 800-900 per year in the 2000s following nesting totals fewer than 100 nests per year in the 1980s (Florida Fish and Wildlife Conservation Commission, unpublished data). Using data from the index nesting beach surveys, the TEWG (2007) estimated a significant annual nesting growth rate of 1.17 percent between 1989 and 2005. In 2007, a record 517 leatherback nests were observed on the index beaches in Florida, with 265 in 2008 (FWC Index Nesting Beach database). The reduction in nesting from 2007 to 2008 is thought to be a result of the cyclical nature of leatherback nesting, similar to the biennial cycle of green turtle nesting.

The West African nesting stock of leatherbacks is a large, important, but mostly unstudied aggregation. Nesting occurs in various countries along Africa’s Atlantic coast, but much of the nesting is undocumented and the data are inconsistent. However, it is known that Gabon has a very large amount of leatherback nesting, with at least 30,000 nests laid along its coast in one season (Fretey et al. 2007). Fretey et al. (2007) also provide detailed information about other known nesting beaches and survey efforts along the Atlantic African coast. Because of the lack of consistent effort and minimal available data, trend analyses were not possible for this stock (TEWG 2007).

Two other small but growing nesting stocks utilize the beaches of Brazil and South Africa. For the Brazilian stock, the TEWG (2007) analyzed the available data and determined that between 1988 and 2003 there was a positive annual average growth rate of 1.07 percent using regression analyses and 1.08 percent using Bayesian modeling. The South African stock has an annual average growth rate of 1.06 based on regression modeling and 1.04 percent using the Bayesian approach (TEWG 2007).

Estimates of total population size for Atlantic leatherbacks are difficult to ascertain due to the inconsistent nature of the available nesting data. In 1996, the entire Western Atlantic population was characterized as stable at best (Spotila et al. 1996), with numbers of nesting females reported to be on the order of 18,800. A subsequent analysis by Spotila (pers. comm.) indicated that by 2000, the Western Atlantic nesting population had decreased to about 15,000 nesting females. Spotila et al. (1996) estimated that the leatherback population for the entire Atlantic basin, including all nesting beaches in the Americas, the Caribbean, and West Africa, totaled approximately 27,600 nesting females, with an estimated range of 20,082-35,133. This is consistent with the estimate of 34,000-95,000 total adults (20,000-56,000 adult females; 10,000-21,000 nesting females) determined by the TEWG (2007).

Threats
Zug and Parham (1996) pointed out that the main threat to leatherback populations in the Atlantic is the combination of fishery-related mortality (especially entanglement in gear and drowning in trawls) and the intense egg harvesting on the main nesting beaches. Other important ongoing threats to the population include pollution, loss of nesting habitat, and boat strikes.

Of sea turtle species, leatherbacks seem to be the most vulnerable to entanglement in fishing gear. This susceptibility may be the result of their body type (large size, long pectoral flippers, and lack of a hard shell), their attraction to gelatinous organisms and algae that collect on buoys and buoy lines at or near the surface, possibly their method of locomotion, and perhaps their attraction to the lightsticks used to attract target species in longline fisheries. They are also susceptible to entanglement in gillnets and pot/trap lines (used in various fisheries) and capture in trawl gear (e.g., shrimp trawls).

Leatherbacks are exposed to pelagic longline fisheries in many areas of their range. Unlike loggerhead turtle interactions with longline gear, leatherback turtles do not usually ingest longline bait. Instead, leatherbacks are typically foul-hooked by longline gear (e.g., on the flipper or shoulder area) rather than getting mouth-hooked or swallowing the hook (NMFS 2001a). A total of 24 nations, including the United States (accounting for 5-8 percent of the hooks fished), have fleets participating in pelagic longline fisheries in the area. Basin-wide, Lewison et al. (2004) estimated that 30,000-60,000 leatherback sea turtle captures occurred in Atlantic pelagic longline fisheries in the year 2000 alone (note that multiple captures of the same individual are known to occur, so the actual number of individuals captured may not be as high). Genetic studies performed within the Northeast Distant Fishery Experiment indicate that the leatherbacks captured in the Atlantic highly migratory species pelagic longline fishery were primarily from the French Guiana and Trinidad nesting stocks (over 95 percent); individuals from West African stocks were surprisingly absent (Roden et al. in press).

Leatherbacks are also susceptible to entanglement in the lines associated with trap/pot gear used in several fisheries. From 1990-2000, 92 entangled leatherbacks were reported from New York through Maine (Dwyer et al. 2002). Additional leatherbacks stranded wrapped in line of unknown origin or with evidence of a past entanglement (Dwyer et al. 2002). Fixed gear fisheries in the mid-Atlantic have also contributed to leatherback entanglements. In North Carolina, two leatherback sea turtles were reported entangled in a crab pot buoy inside Hatteras Inlet (D. Fletcher, pers. comm. to S. Epperly in NMFS 2001a). A third leatherback was reported entangled in a crab pot buoy in Pamlico Sound near Ocracoke. This turtle was disentangled and released alive; however, lacerations on the front flippers from the lines were evident (D. Fletcher, pers. comm. to S. Epperly in NMFS 2001a). In the Southeast, leatherbacks are vulnerable to entanglement in Florida's lobster pot and stone crab fisheries. In the U.S. Virgin Islands, where one of five leatherback strandings from 1982 to 1997 was due to entanglement (Boulon 2000), leatherbacks have been observed with their flippers wrapped in the line of West Indian fish traps (R. Boulon, pers. comm. to J. Braun-McNeill in NMFS 2001a). Because many entanglements of this typically pelagic species likely go unnoticed, entanglements in fishing gear may be much higher.

Leatherback interactions with the Southeast Atlantic shrimp fishery, which operates predominately from North Carolina through southeast Florida (NMFS 2002), have also been a common occurrence. Leatherbacks, which migrate north annually, are likely to encounter shrimp trawls working in the coastal waters off the Atlantic coast from Cape Canaveral, Florida, to the Virginia/North Carolina border. Leatherbacks also interact with the Gulf of Mexico shrimp fishery. For many years, TEDs required for use in these fisheries were less effective at excluding leatherbacks than the smaller, hard-shelled turtle species. To address this problem, on February 21, 2003, the NMFS issued a final rule to amend the TED regulations, which required modifications to the size and design of TEDs to exclude leatherbacks and large and sexually mature loggerhead and green turtles. Mortality of leatherbacks in the shrimp fishery is now estimated at 54 turtles per year.

Other trawl fisheries are also known to interact with leatherback sea turtles. In October 2001, a Northeast Fisheries Science Center (NEFSC) observer documented the take of a leatherback in a bottom otter trawl fishing for *Loligo* squid off Delaware; TEDs are not required in this fishery. The winter trawl flounder fishery, which did not come under the revised TED regulations, may also interact with leatherback sea turtles.

Gillnet fisheries operating in the nearshore waters of the mid-Atlantic states are also suspected of capturing, injuring, and/or killing leatherbacks when these fisheries and leatherbacks co-occur. Data collected by the NEFSC Fisheries Observer Program from 1994 through 1998 (excluding 1997) indicate that a total of 37 leatherbacks were incidentally captured (16 lethally) in drift gillnets set in offshore waters from Maine to Florida during this period. Observer coverage for this period ranged from 54 to 92 percent.

Poaching is not known to be a problem for nesting populations in the continental United States. However, in 2001 the NMFS Southeast Fisheries Science Center (SEFSC) noted that poaching of juveniles and adults was still occurring in the U.S. Virgin Islands and the Guianas. In all, four of the five strandings in St. Croix were the result of poaching (Boulon 2000). A few cases of fishermen poaching leatherbacks have been reported from Puerto Rico, but most of the poaching is on eggs.

Leatherback sea turtles may be more susceptible to marine debris ingestion than other species due to their pelagic existence and the tendency of floating debris to concentrate in convergence zones that adults and juveniles use for feeding areas and migratory routes (Lutcavage et al. 1997, Shoop and Kenney 1992). Investigations of the stomach contents of leatherback sea turtles revealed that a substantial percentage (44 percent of the 16 cases examined) contained plastic (Mrosovsky 1981). Along the coast of Peru, intestinal contents of 19 of 140 (13 percent) leatherback carcasses were found to contain plastic bags and film (Fritts 1982). The presence of plastic debris in the digestive tract suggests that leatherbacks might not be able to distinguish between prey items and plastic debris (Mrosovsky 1981). Balazs (1985) speculated that the object might resemble a food item by its shape, color, size, or even movement as it drifts about, and induce a feeding response in leatherbacks.

It is important to note that, like marine debris, fishing gear interactions and poaching are problems for leatherbacks throughout their range. Entanglements are common in Canadian waters where Goff and Lien (1988) reported that 14 of 20 leatherbacks encountered off the coast of Newfoundland/Labrador were entangled in fishing gear including salmon net, herring net, gillnet, trawl line and crab pot line. Leatherbacks are reported taken by many other nations that participate in Atlantic pelagic longline fisheries, including Taipei, Brazil, Trinidad, Morocco, Cyprus, Venezuela, Korea, Mexico, Cuba, U.K., Bermuda, People's Republic of China, Grenada, Canada, Belize, France, and Ireland (see NMFS 2001a for a description of take records). Leatherbacks are known to drown in fish nets set in coastal waters of Sao Tome, West Africa (Castroviejo et al. 1994, Graff 1995). Gillnets are one of the suspected causes of the decline in the leatherback sea turtle population in French Guiana (Chevalier et al. 1999), and gillnets targeting green and hawksbill turtles in the waters of coastal Nicaragua also incidentally catch leatherback turtles (Lageux et al. 1998). Observers on shrimp trawlers operating in the northeastern region of Venezuela documented the capture of six leatherbacks from 13,600 trawls (Marcano and Alio-M. 2000). A study by the Trinidad and Tobago's Institute for Marine Affairs (IMA) in 2002 confirmed that bycatch of leatherbacks is high in Trinidad. IMA estimated that more than 3,000 leatherbacks were captured incidental to gillnet fishing in the coastal waters of Trinidad in 2000. As much as one-half or more of the gravid turtles in Trinidad and Tobago waters may be killed (Lee Lum 2003), though many of the turtles do not die as a result of drowning, but rather because the fishermen butcher them in order to get them out of their nets (NMFS 2001a).

There is a large and growing body of literature on past, present, and future impacts of global climate change exacerbated and accelerated by human activities. Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. NOAA's climate information portal provides basic background information on these and other measured or anticipated effects (see http://www.climate.gov).

Impacts on sea turtles currently cannot, for the most part, be predicted with any degree of certainty, however significant impacts to the hatchling sex ratios of leatherback turtles may result (NMFS and USFWS 2007b). In marine turtles, sex is determined by temperature in the middle third of incubation with female offspring produced at higher temperatures and males at lower temperatures within a thermal tolerance range of 25º-35°C (Ackerman 1997). However, unlike other sea turtles species, leatherbacks tend to select nest locations in the cooler tidal zone of beaches (Kamel and Mrosovsky 2004). This preference may help mitigate the effects from increased beach temperature (Kamel and Mrosovsky 2004).

Sea level rise from global climate change is also a potential problem for areas with low-lying beaches where sand depth is a limiting factor, as the sea may 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 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).

Global climate change is likely to influence the distribution and abundance of jellyfish, the primary prey item of leatherbacks (NMFS and USFWS 2007b). Several studies have shown leatherback distribution is influenced by jellyfish abundance (e.g., Houghton et al. 2006, Witt et al. 2006, Witt et al. 2007). How these changes in jellyfish abundance and distribution will impact leatherback sea turtle foraging behavior and distribution is currently unclear (Witt et al. 2007).

3.2.3.3 Summary of Leatherback Status

In the Pacific Ocean, the abundance of leatherback turtle nesting individuals and colonies has declined dramatically over the past 10 to 20 years. Nesting colonies throughout the Eastern and Western Pacific Ocean have been reduced to a fraction of their former abundance by the combined effects of human activities that have reduced the number of nesting females. In addition, egg poaching has reduced the reproductive success of the remaining nesting females. At current rates of decline, leatherback turtles in the Pacific basin are a critically endangered species with a low probability of surviving and recovering in the wild.

In the Atlantic Ocean, our understanding of the status and trends of leatherback turtles is somewhat more confounded, although the overall trend appears to be stable to increasing. The data indicate increasing or stable nesting populations in all of the regions except West Africa (no long-term data are available) and the Western Caribbean (TEWG 2007). Some of the same factors that led to precipitous declines of leatherbacks in the Pacific also affect leatherbacks in the Atlantic (i.e., leatherbacks are captured and killed in many kinds of fishing gear and interact with fisheries in state, federal, and international waters). Poaching is also a problem that affects leatherbacks occurring in U.S. waters. Leatherbacks are also more susceptible to death or injury from ingesting marine debris than other turtle species.

3.2.4 Loggerhead Sea Turtle

The loggerhead sea turtle was listed as a threatened species throughout its global range on July 28, 1978. It was listed because of direct take, incidental capture in various fisheries, and the alteration and destruction of its habitat. Loggerhead sea turtles inhabit the continental shelves and estuarine environments along the margins of the Atlantic, Pacific, and Indian Oceans. The majority of loggerhead nesting occurs in the Western Atlantic Ocean (South Florida, United

States), and the western Indian Ocean (Masirah, Oman); in both locations nesting assemblages have more than 10,000 females nesting each year (NMFS and USFWS 2008). Loggerhead sea turtles are the most abundant species of sea turtle in U.S. waters.

3.2.4.1 Pacific Ocean

In the Pacific Ocean, major loggerhead nesting grounds are generally located in temperate and subtropical regions with scattered nesting in the tropics. Within the Pacific Ocean, loggerhead sea turtles are represented by a northwestern Pacific nesting aggregation (located in Japan) and a smaller southwestern nesting aggregation that occurs in Eastern Australia (Great Barrier Reef and Queensland) and New Caledonia (NMFS 2001a). There are no reported loggerhead nesting sites in the eastern or central Pacific Ocean basin. Data from 1995 estimated the Japanese nesting aggregation at 1,000 female loggerhead sea turtles (Bolten et al. 1996). More recent information suggests that nest numbers have increased somewhat over the period 1998-2004 (NMFS and USFWS 2007c). However, this time period is too short to make a determination of the overall trend in nesting (NMFS and USFWS 2007c). Recent genetic analyses on female loggerheads nesting in Japan suggest that this “subpopulation” is comprised of genetically distinct nesting colonies (Hatase et al. 2002) with precise natal homing of individual females. As a result, Hatase et al. (2002) indicate that loss of one of these colonies would decrease the genetic diversity of Japanese loggerheads; recolonization of the site would not be expected on an ecological time scale. In Australia, long-term census data have been collected at some rookeries since the late 1960s and early 1970s, and nearly all the data show marked declines in nesting populations since the mid-1980s; the nesting aggregation in Queensland, Australia, was as low as 300 females in 1997 (Limpus and Limpus 2003) .

Pacific loggerhead turtles are captured, injured, or killed in numerous Pacific fisheries including Japanese longline fisheries in the Western Pacific Ocean and South China Seas; direct harvest and commercial fisheries off Baja California, Mexico; commercial and artisanal swordfish fisheries off Chile, Columbia, Ecuador, and Peru; purse seine fisheries for tuna in the eastern tropical Pacific Ocean; and California/Oregon drift gillnet fisheries. In Australia, where turtles are taken in bottom trawl and longline fisheries, efforts have been made to reduce fishery bycatch (NMFS and USFWS 2007c). In addition, the abundance of loggerhead sea turtles in nesting colonies throughout the Pacific basin has declined dramatically over the past 10 to 20 years. Loggerhead turtle colonies in the Western Pacific Ocean have been reduced to a fraction of their former abundance by the combined effects of human activities that have reduced the number of nesting females and reduced the reproductive success of females that manage to nest (e.g., due to egg poaching).

In July 2007, NMFS received a petition requesting that loggerhead sea turtles in the North Pacific be classified as a distinct population segment (DPS) with endangered status and critical habitat designated. The petition also requested that if the North Pacific loggerhead is not determined to meet the DPS criteria that loggerheads throughout the Pacific Ocean be designated as a DPS and listed as endangered. NMFS’ 90-day finding for both petitions, published on November 16, 2007 (72 FR 64585 and 64587), was that the petition requests were “warranted” and that a full review would be conducted. A thorough review by the Loggerhead Turtle Biological Review Team determined that Pacific loggerheads can be divided into two DPSs, the North Pacific DPS and South Pacific DPS (Conant et al. 2009).

3.2.4.2 Indian Ocean

Loggerhead sea turtles are distributed throughout the Indian Ocean, along most mainland coasts and island groups (Baldwin et al. 2003). Throughout the Indian Ocean, loggerhead sea turtles face many of the same threats as in other parts of the world including loss of nesting beach habitat, fishery interactions, and turtle meat and/or egg harvesting.

In the southwestern Indian Ocean, loggerhead nesting has shown signs of recovery in South Africa where protection measures have been in place for decades. However, in other southwestern areas (e.g., Madagascar and Mozambique) loggerhead nesting groups are still affected by subsistence hunting of adults and eggs (Baldwin et al. 2003). The largest known nesting group of loggerheads in the world occurs in Oman in the Northern Indian Ocean. An estimated 20,000-40,000 females nest each year at Masirah, the largest nesting site within Oman (Baldwin et al. 2003). In the Eastern Indian Ocean, all known nesting sites are found in Western Australia (Dodd 1988). As has been found in other areas, nesting numbers are disproportionate within the area, with the majority of nesting occurring at a single location. This may, however, be the result of fox predation on eggs at other Western Australia nesting sites (Baldwin et al. 2003). A thorough review by the Loggerhead Turtle Biological Review Team determined that Indian Ocean loggerheads can be divided into three DPSs, the North Indian Ocean DPS, Southeast Indo-Pacific Ocean DPS, and Southwest Indian Ocean DPS (Conant et al. 2009).

3.2.4.3 Mediterranean Sea

Nesting in the Mediterranean is confined almost exclusively to the eastern basin. The highest level of nesting in the Mediterranean occurs in Greece, with an average of 3,050 nests per year. There is a long history of exploitation of loggerheads in the Mediterranean. Although much of this is now prohibited, some directed take still occurs. Loggerheads in the Mediterranean also face the threat of habitat degradation, incidental fishery interactions, vessel strikes, and marine pollution (Margaritoulis et al. 2003). Longline fisheries, in particular, are believed to catch thousands of juvenile loggerheads each year (NMFS and USFWS 2007c), although genetic analyses indicate that only a portion of the loggerheads captured originate from nesting groups in the Mediterranean (Laurent et al. 1998). A thorough review by the Loggerhead Turtle Biological Review Team determined that Mediterranean loggerheads could be designated as a separate DPS (Conant et al. 2009).

3.2.4.4 Atlantic Ocean

In the Western Atlantic, most loggerhead sea turtles nest from North Carolina to Florida and along the Gulf coast of Florida. Previous section 7 analyses have recognized at least five Western Atlantic subpopulations, divided geographically as follows: (1) a northern nesting subpopulation, occurring from North Carolina to Northeast Florida at about 29ºN; (2) a South Florida nesting subpopulation, occurring from 29ºN on the east coast to Sarasota on the west coast; (3) a Florida Panhandle nesting subpopulation, occurring at Eglin Air Force Base and the beaches near Panama City, Florida; (4) a Yucatán nesting subpopulation, occurring on the Eastern Yucatán Peninsula, Mexico (Márquez 1990 and TEWG 2000); and (5) a Dry Tortugas nesting subpopulation, occurring in the islands of the Dry Tortugas, near Key West, Florida (NMFS 2001b). The recently published recovery plan for the Northwest Atlantic population of loggerhead sea turtles concluded, based on recent advances in genetic analyses, that there is no genetic distinction between loggerheads nesting on adjacent beaches along the Florida Peninsula and that specific boundaries for subpopulations could not be designated based on genetic differences alone. Thus, the plan uses a combination of geographic distribution of nesting densities, geographic separation, and geopolitical boundaries, in addition to genetic differences, to identify recovery units. The recovery units are: (1) the Northern Recovery Unit (Florida/Georgia border north through southern Virginia); (2) the Peninsular Florida Recovery Unit (Florida/Georgia border through Pinellas County, Florida); (3) the Dry Tortugas Recovery Unit (islands located west of Key West, Florida); (4) the Northern Gulf of Mexico Recovery Unit (Franklin County, Florida, through Texas); and (5) the Greater Caribbean Recovery Unit (Mexico through French Guiana, the Bahamas, Lesser Antilles, and Greater Antilles) (NMFS and USFWS 2008). The recovery plan concluded that all recovery units are essential to the recovery of the species. A recent report by the Loggerhead Biological Review Team determined that loggerhead turtles in the Atlantic meet the required characteristics to be separated into three DPSs, the Northwest Atlantic DPS, Northeast Atlantic DPS, and South Atlantic DPS (Conant et al. 2009).

Life History and Distribution
Past literature gave an estimated age at maturity of 21-35 years (Frazer and Ehrhart 1985, Frazer et al. 1994) with the benthic immature stage lasting at least 10-25 years. However, based on new data from tag returns, strandings, and nesting surveys, NMFS SEFSC (2001) estimated ages of maturity ranging from 20-38 years and benthic immature stage lasting from 14-32 years.

Mating takes place in late March-early June, and eggs are laid throughout the summer, with a mean clutch size of 100-126 eggs in the southeastern United States. Individual females nest multiple times during a nesting season, with a mean of 4.1 nests per individual (Murphy and Hopkins 1984). Nesting migrations for an individual female loggerhead are usually on an interval of 2-3 years, but can vary from 1-7 years (Dodd 1988). Generally, loggerhead sea turtles originating from the Western Atlantic nesting aggregations are believed to lead a pelagic existence in the North Atlantic Gyre for as long as 7-12 years or more. Stranding records indicate that when pelagic immature loggerheads reach 40-60 cm straight-line carapace length, they begin to live in coastal inshore and nearshore waters of the continental shelf throughout the U.S. Atlantic and Gulf of Mexico, although some loggerheads may move back and forth between the pelagic and benthic environment (Witzell 2002). Benthic immature loggerheads (sea turtles that have come back to inshore and nearshore waters), the life stage following the pelagic immature stage, have been found from Cape Cod, Massachusetts, to southern Texas, and occasionally strand on beaches in northeastern Mexico.

Tagging studies have shown loggerheads that have entered the benthic environment undertake routine migrations along the coast that are limited by seasonal water temperatures. Loggerhead sea turtles occur year-round in offshore waters off North Carolina where water temperature is influenced by the Gulf Stream. As coastal water temperatures warm in the spring, loggerheads begin to immigrate to North Carolina inshore waters (e.g., Pamlico and Core Sounds) and also move up the coast (Epperly et al. 1995a-c), occurring in Virginia foraging areas as early as April and on the most northern foraging grounds in the Gulf of Maine in June. The trend is reversed in the fall as water temperatures cool. The large majority of loggerheads leave the Gulf of Maine by mid-September but some may remain in mid-Atlantic and Northeast areas until late fall. By December, loggerheads have emigrated from inshore North Carolina waters and coastal waters to the north to waters offshore of North Carolina, particularly off Cape Hatteras, and waters further south where the influence of the Gulf Stream provides temperatures favorable to sea turtles (≥ 11°C) (Epperly et al. 1995a-c). Loggerhead sea turtles are year-round residents of Central and South Florida.

Pelagic and benthic juveniles are omnivorous and forage on crabs, mollusks, jellyfish, and vegetation at or near the surface (Dodd 1988). Sub-adult and adult loggerheads are primarily coastal dwelling and typically prey on benthic invertebrates such as mollusks and decapod crustaceans in a variety of habitats.

More recent studies are revealing that the loggerhead's life history is more complex than previously believed. Rather than making discrete developmental shifts from oceanic to neritic environments, research is showing that both adults and (presumed) neritic stage juveniles continue to use the oceanic environment and will move back and forth between the two habitats (Witzell 2002, Blumenthal et al. 2006, Hawkes et al. 2006, McClellan and Read 2007). One of the studies tracked the movements of adult females post-nesting and found a difference in habitat use was related to body size, with larger turtles staying in coastal waters and smaller turtles traveling to oceanic waters (Hawkes et al. 2006). A tracking study of large juveniles found that the habitat preferences of this life stage were also diverse, with some remaining in neritic waters while others moved off into oceanic waters (McClellan and Read 2007). However, unlike the Hawkes et al. study (2006), there was no significant difference in the body size of turtles that remained in neritic waters versus oceanic waters (McClellan and Read 2007). In either case, the research not only supports the need to revise the life history model for loggerheads but also demonstrates that threats to loggerheads in both the neritic and oceanic environments are likely impacting multiple life stages of this species.

Population Dynamics and Status
A number of stock assessments and similar reviews (TEWG 1998, TEWG 2000, NMFS SEFSC 2001, Heppell et al. 2003, NMFS and USFWS 2008, Conant et al. 2009, TEWG 2009) have examined the stock status of loggerheads in the Atlantic Ocean, but none have been able to develop a reliable estimate of absolute population size.

Numbers of nests and nesting females can vary widely from year to year. However, nesting beach surveys can provide a reliable assessment of trends in the adult female population, due to the strong nest site fidelity of females turtles, as long as such studies are sufficiently long and effort and methods are standardized (see, e.g., NMFS and USFWS 2008, Meylan 1982). NMFS and USFWS (2008) concluded that the lack of change in two important demographic parameters of loggerheads, remigration interval and clutch frequency, indicate that time series on numbers of nests can provide reliable information on trends in the female population. Recent analysis of available data for the Peninsular Florida Recovery Unit has led to the conclusion that the observed decline in nesting for that unit over the last several years can best be explained by an actual decline in the number of adult female loggerheads in the population (Witherington et al. 2009).

Annual nest totals from beaches within what NMFS and USFWS have defined as the Northern Recovery Unit (NRU) averaged 5,215 nests from 1989-2008, a period of near-complete surveys of NRU nesting beaches (GDNR unpublished data, NCWRC unpublished data, SCDNR unpublished data), representing approximately 1,272 nesting females per year (4.1 nests per female, Murphy and Hopkins 1984). The loggerhead nesting trend from daily beach surveys showed a significant decline of 1.3 percent annually. Nest totals from aerial surveys conducted by SCDNR showed a 1.9 percent annual decline in nesting in South Carolina since 1980. Overall, there is strong statistical data to suggest the NRU has experienced a long-term decline. Data in 2008 has shown improved nesting numbers, but future nesting years will need to be analyzed to determine if a change in trend is occurring. In 2008, 841 loggerhead nests were observed compared to the 10-year average of 715 nests in North Carolina. In South Carolina, 2008 was the seventh highest nesting year on record since 1980, with 4,500 nests, but this did not change the long-term trend line indicating a decline on South Carolina beaches. Georgia beach surveys located a total of 1,648 nests in 2008. This number surpassed the previous statewide record of 1,504 nests in 2003. According to analyses by Georgia DNR, the 40-year time-series trend data show an overall decline in nesting, but the shorter comprehensive survey data (20 years) indicate a stable population (SCDNR 2008, GDNR unpublished data, NCWRC unpublished data, SCDNR unpublished data).

Another consideration that may add to the importance and vulnerability of the NRU is the sex ratio of this subpopulation. NMFS scientists have estimated that the Northern subpopulation produces 65 percent males (NMFS SEFSC 2001). However, research conducted over a limited time frame has found opposing sex ratios (Wyneken et al. 2004), so further information is needed to clarify the issue. Since nesting female loggerhead sea turtles exhibit nest fidelity, the continued existence of the Northern subpopulation is related to the number of female hatchlings that are produced. Producing fewer females will limit the number of subsequent offspring produced by the subpopulation.

The Peninsular Florida Recovery Unit (PFRU) is the largest loggerhead nesting assemblage in the Northwest Atlantic. A near-complete nest census undertaken from 1989 to 2007 showed a mean of 64,513 loggerhead nests per year, representing approximately 15,735 nesting females per year (from NMFS and USFWS 2008). An analysis of index nesting beach data shows a decline in nesting by the PFRU between 1989 and 2008 of 26 percent over the period, and a mean annual rate of decline of 1.6 percent despite a large increase in nesting for 2008 (Witherington et al. 2009, NMFS and USFWS 2008). In 2009, nesting levels dropped well below 2008 levels, to approximately 33,000 nests (FWRI website- Graph of Core Florida Index Nests for Loggerheads).

The remaining three recovery units—Dry Tortugas (DTRU), Northern Gulf of Mexico (NGMRU), and Greater Caribbean (GCRU)—are much smaller nesting assemblages but still considered essential to the continued existence of the species. Nesting surveys for the DTRU are conducted as part of Florida's statewide survey program. Survey effort has been relatively stable during the 9-year period from 1995-2004 (although the 2002 year was missed). Nest counts ranged from 168-270, with a mean of 246, but with no detectable trend during this period (Florida Fish and Wildlife Conservation Commission, Florida Marine Research Institute, Statewide Nesting Beach Survey Data, NMFS and USFWS 2008). Nest counts for the NGMRU are focused on index beaches rather than all beaches where nesting occurs. The 12-year dataset (1997-2008) of index nesting beaches in the area shows a significant declining trend of 4.7 percent annually (NMFS and USFWS 2008). Similarly, nesting survey effort has been inconsistent among the GCRU nesting beaches and no trend can be determined for this subpopulation. Zurita et al. (2003) found a statistically significant increase in the number of nests on seven of the beaches on Quintana Roo, Mexico, from 1987-2001, where survey effort was consistent during the period. However, nesting has declined since 2001, and the previously reported increasing trend appears to not have been sustained (NMFS and USFWS 2008).

Determining the meaning of the nesting decline data is confounded by various in-water research that suggests the abundance of neritic juvenile loggerheads is steady or increasing (Ehrhart et al. 2007, M. Bresette pers. comm. regarding captures at the St. Lucie Power Plant, SCDNR unpublished SEAMAP-SA data, Epperly et al. 2007). Ehrhart et al. (2007) found no significant regression-line trend in the long-term dataset. However, notable increases in recent years and a statistically significant increase in CPUE of 102.4 percent from the 4-year period of 1982-1985 to the 2002-2005 periods were found. Epperly et al. (2007) determined the trends of increasing loggerhead catch rates from all the aforementioned studies in combination provide evidence there has been an increase in neritic juvenile loggerhead abundance in the southeastern United States in the recent past. A study led by the South Carolina Department of Natural Resources found that standardized trawl survey CPUEs for loggerheads from South Carolina to North Florida was 1.5 times higher in summer 2008 than summer 2000. However, even though there were persistent inter-annual increases from 2000-2008, the difference was not statistically significant, likely due to the relatively short time series. Comparison to other datasets from the 1950s through 1990s showed much higher CPUEs in recent years regionally and in the South Atlantic Bight, leading SCDNR to conclude that it is highly improbable that CPUE increases of such magnitude could occur without a real and substantial increase in actual abundance (Arendt et al. 2009). Whether this increase in abundance represents a true population increase among juveniles or merely a shift in spatial occurrence is not clear. NMFS and USFWS (2008), citing Bjorndal et al. 2005, caution about extrapolating localized in-water trends to the broader population and relating localized trends in neritic sites to population trends at nesting beaches. The apparent overall increase in the abundance of neritic loggerheads in the southeastern U.S. may be due to increased abundance of the largest Stage III individuals (oceanic/neritic juveniles, historically referred to as small benthic juveniles), which could indicate a relatively large cohort that will recruit to maturity in the near future. However, such an increase in adults may be temporary, as in-water studies throughout the eastern U.S. also indicate a substantial decrease in the abundance of the smallest Stage III loggerheads, a pattern also corroborated by stranding data (TEWG 2009).

The NMFS SE Fishery Science Center has developed a preliminary stage/age demographic model to help determine the estimated impacts of mortality reductions on loggerhead sea turtle population dynamics (NMFS-SEFSC 2009). This model does not incorporate existing trends in the data (such as nesting trends) but instead relies on utilizing the available information on the relevant life-history parameters for sea turtles and then predicts future population trajectories based upon model runs using those parameters. Therefore, the model results do not build upon, but instead are complementary to, the trend data obtained through nest counts and other observations. The model uses the range of published information for the various parameters including mortality by stage, stage duration (years in a stage), and fecundity parameters such as eggs per nest, nests per nesting female, hatchling emergence success, sex ratio, and remigration interval. Model runs were done for each individual recovery unit as well as the western North Atlantic population as a whole, and the resulting trajectories were found to be very similar. One of the most robust results from the model was an estimate of the adult female population size for the western North Atlantic in the 2004-2008 time frame. The distribution resulting from the model runs suggest the adult female population size to be likely between approximately 20,000 to 40,000 individuals, with a low likelihood of being up to 70,000. A much less robust estimate for total benthic females in the western North Atlantic was also obtained, with a likely range of approximately 30,000-300,000 individuals, up to less than 1 million.

The results of one set of model runs suggest that the western North Atlantic population is most likely declining, but this result was very sensitive to the choice of the position of the parameters within their range and hypothesized distributions. This example was run to predict the distribution of projected population trajectories for benthic females using a range of starting population numbers from the 30,000 estimated minimum to the greater than the 300,000 likely upper end of the range and declining trajectories were estimated for all of the population estimates. After 10,000 simulation runs of the models using the parameter ranges, 14 percent of the runs resulted in growing populations, while 86 percent resulted in declining populations. While this does not translate to an equivalent statement that there is an 86 percent chance of a declining population, it does illustrate that given the life history parameter information currently thought to comprise the likely range of possibilities, it appears most likely that with no changes to those parameters the population is projected to decline. Additional model runs using the range of values for each life history parameter, the assumption of non-uniform distribution for those parameters, and a 5 percent natural (non-anthropogenic) mortality for the benthic stages resulted in a determination that a 60-70 percent reduction in anthropogenic mortality in the benthic stages would be needed to bring 50 percent of the model runs to a static (zero growth or decline) or increasing trajectory.

As a result of the large uncertainty in our knowledge of loggerhead life history, at this point predicting the future populations or population trajectories of loggerhead sea turtles with precision is very uncertain. The model results, however, are useful in guiding future research needs to better understand the life history parameters that have the most significant impact in the model. Additionally, the model results provide valuable insights into the likely overall declining status of the species and in the impacts of large-scale changes to various life history parameters (such as mortality rates for given stages) and how they may change the trajectories. The results of the model, in conjunction with analyses conducted on nest count trends (such as Witherington et al. 2009) which have suggested that the population decline is real, provides a strong basis for the conclusion that the western North Atlantic loggerhead population is in decline. NMFS also recently convened a new Turtle Expert Working Group (TEWG) for loggerhead sea turtles that gathered available data and examined the potential causes of the nesting decline and what the decline means in terms of population status. The TEWG ultimately could not determine whether or not decreasing annual numbers of nests among the Western North Atlantic loggerhead subpopulations were due to stochastic processes resulting in fewer nests, a decreasing average reproductive output of the adult females, decreasing numbers of adult females, or a combination of those factors. Past and present mortality factors that could impact current loggerhead nest numbers are many, and it is likely that several factors compound to create the current decline. Regardless of the source of the decline, it is clear that the reduced nesting will result in depressed recruitment to subsequent life stages over the coming decades (TEWG 2009).

Threats

The 5-year status review of loggerhead sea turtles recently completed by NMFS and the USFWS provides a summary of natural as well as anthropogenic threats to loggerhead sea turtles (NMFS and USFWS 2007c). The Loggerhead Recovery Team also undertook a comprehensive evaluation of threats to the species, and described them separately for the terrestrial, neritic, and oceanic zones (NMFS and USFWS 2008). The diversity of sea turtles' life history leaves them susceptible to many natural and human impacts, including impacts while they are on land, in the benthic environment, and in the pelagic environment. Hurricanes are particularly destructive to sea turtle nests. Sand accretion and rainfall that result from these storms, as well as wave action, can appreciably reduce hatchling success. For example in 1992 all of the eggs over a 90-mile length of coastal Florida were destroyed by storm surges on beaches that were closest to the eye of Hurricane Andrew (Milton et al. 1994). Also, many nests were destroyed during the 2004 and 2005 hurricane seasons. Other sources of natural mortality include cold-stunning and biotoxin exposure.

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

Loggerhead sea turtles are affected by a completely different set of anthropogenic threats in the marine environment. These threats include oil and gas exploration, coastal development, marine transportation, marine pollution (which may have a direct impact, or an indirect impact by causing harmful algal blooms), underwater explosions, hopper dredging, offshore artificial lighting, power plant entrainment and/or impingement, entanglement in debris, ingestion of marine debris, marina and dock construction and operation, boat collisions, poaching, and fishery interactions. Loggerheads in the pelagic environment are exposed to a series of longline fisheries, which include the highly migratory species' Atlantic pelagic longline fisheries, an Azorean longline fleet, a Spanish longline fleet, and various longline fleets in the Mediterranean Sea (Aguilar et al. 1995, Bolten et al. 1994). Loggerheads in the benthic environment in waters off the coastal United States are exposed to a suite of fisheries in federal and state waters including trawl, purse seine, hook-and-line, gillnet, pound net, longline, and trap fisheries. The sizes and reproductive values of sea turtles taken by fisheries vary significantly, depending on the location and season of the fishery, and size-selectivity resulting from gear characteristics. Therefore, it is possible for fisheries that interact with fewer, more reproductively valuable turtles to have a greater detrimental effect on the population than one that takes greater numbers of less reproductively valuable turtles if the fishery removes a higher overall reproductive value from the population (Wallace et al. 2008). The Loggerhead Biological Review Team determined that the greatest threats to the Northwest Atlantic DPS of loggerheads result from cumulative fishery bycatch in neritic and oceanic habitats (Conant, et al. 2009). Attaining a more thorough understanding of the characteristics, as well as the quantity, of sea turtle bycatch across all fisheries is of great importance.

There is a large and growing body of literature on past, present, and future impacts of global climate change exacerbated and accelerated by human activities. Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. NOAA's climate information portal provides basic background information on these and other measured or anticipated effects (see http://www.climate.gov).

Impacts on sea turtles currently cannot, for the most part, be predicted with any degree of certainty, however significant impacts to the hatchling sex ratios of loggerhead turtles may result (NMFS and USFWS 2007c). In marine turtles, sex is determined by temperature in the middle third of incubation with female offspring produced at higher temperatures and males at lower temperatures within a thermal tolerance range of 25º-35°C (Ackerman 1997). Increases in global temperature could potentially skew future sex ratios toward higher numbers of females (NMFS and USFWS 2007c). Modeling suggests an increase of 2°C in air temperature would result in a sex ratio of over 80 percent female offspring for loggerheads nesting near Southport, North Carolina. The same increase in air temperatures at nesting beaches in Cape Canaveral, Florida, would result in close to 100 percent female offspring. More ominously, an air temperature increase of 3°C is likely to exceed the thermal threshold of most clutches, leading to death (Hawkes et al. 2007).

Warmer sea surface temperatures have been correlated with an earlier onset of loggerhead nesting in the spring (Weishampel et al. 2004, Hawkes et al. 2007), as well as short inter-nesting intervals (Hays et al. 2002) and shorter nesting season (Pike et al. 2006).

The effects from increased temperatures may be exacerbated on developed nesting beaches where shoreline armoring and construction have denuded vegetation. Erosion control structures could potentially result in the permanent loss of nesting beach habitat or deter nesting females (NRC 1990). Alternatively, nesting females may nest on the seaward side of the erosion control structures, potentially exposing them to repeated tidal overwash (NMFS and USFWS 2007c). Sea level rise from global climate change is also a potential problem for areas with low-lying beaches where sand depth is a limiting factor, as the sea may 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).

Other changes in the marine ecosystem caused by global climate change (e.g., salinity, oceanic currents, dissolved oxygen levels, nutrient distribution, etc.) could influence the distribution and abundance of phytoplankton, zooplankton, submerged aquatic vegetation, crustaceans, mollusks, forage fish, etc., which could ultimately affect the primary foraging areas of loggerhead sea turtles.

Actions have been taken to reduce anthropogenic impacts to loggerhead sea turtles from various sources, particularly since the early 1990s. These include lighting ordinances, predation control, and nest relocations to help increase hatchling survival, as well as measures to reduce the mortality of pelagic immatures, benthic immatures, and sexually mature age classes in various fisheries and other marine activities. Recent actions have taken significant steps towards reducing the recurring sources of mortality of sea turtles in the environmental baseline and improving the status of all loggerhead subpopulations. For example, the TED regulation published on February 21, 2003 (68 FR 8456), represents a significant improvement in the baseline effects of trawl fisheries on loggerhead sea turtles, though shrimp trawling is still considered to be one of the largest source of anthropogenic mortality on loggerheads.

3.2.4.3 Summary of Status for Loggerhead Sea Turtles

In the Pacific Ocean, loggerhead sea turtles are represented by a northwestern Pacific nesting aggregation (located in Japan) and a smaller southwestern nesting aggregation that occurs in Australia (Great Barrier Reef and Queensland) and New Caledonia. The abundance of loggerhead sea turtles on nesting colonies throughout the Pacific basin has declined dramatically over the past 10 to 20 years. Data from 1995 estimated the Japanese nesting aggregation at 1,000 female loggerhead sea turtles (Bolten et al. 1996), but it has probably declined since 1995 and continues to decline (Tillman 2000). The nesting aggregation in Queensland, Australia, was as low as 300 females in 1997.

In the Atlantic Ocean, absolute population size is not known, but based on extrapolation of nesting information, loggerheads are likely much more numerous than in the Pacific Ocean. The NMFS recognizes five recovery units of loggerhead sea turtles in the western North Atlantic based on genetic studies and management regimes. Cohorts from all of these are known to occur within the action area of this consultation. There are long-term declining nesting trends for the two largest Western Atlantic recovery units: the PFRU and the NRU. Furthermore, no long-term data suggest any of the loggerhead subpopulations throughout the entire North Atlantic are increasing in annual numbers of nests (TEWG 2009). Additionally, using both computation of susceptibility to quasi-extinction and stage-based deterministic modeling to determine the effects of known threats to Northwest Atlantic loggerheads, the Loggerhead Biological Review Team determined that this population is likely to decline in the foreseeable future, driven primarily by the mortality of juvenile and adult loggerheads from fishery bycatch throughout the North Atlantic Ocean. These computations were done for each of the recovery units, and all of them resulted in an expected decline (Conant et al. 2009). Because of its size, the PFRU may be critical to the survival of the species in the Atlantic Ocean. In the past, this nesting aggregation was considered second in size only to the nesting aggregation on islands in the Arabian Sea off Oman (Ross 1979, Ehrhart 1989). However, the status of the Oman colony has not been evaluated recently; and it is located in an area of the world where it is highly vulnerable to disruptive events such as political upheavals, wars, catastrophic oil spills, and lack of strong protections for sea turtles (Meylan et al. 1995). Given the lack of updated information on this population, the status of loggerheads in the Indian Ocean basin overall is essentially unknown. On March 5, 2008, NMFS and USFWS published a 90-day finding that a petitioned request to reclassify loggerhead turtles in the Western North Atlantic Ocean as a distinct population segment may be warranted (73 FR 11849). NMFS and USFWS convened a Loggerhead Biological Review Team that determined that loggerhead turtles in the Atlantic meet the required characteristics to be separated into three DPSs, the Northwest Atlantic DPS, Northeast Atlantic DPS, and South Atlantic DPS (Conant et al. 2009). On March 10, 2010, NMFS and USFWS announced their determination that Loggerhead sea turtles should be listed as 9 separate DPSs, and that 7 of these, including Northwest Atlantic loggerheads, should be listed as endangered.

All loggerhead subpopulations are faced with a multitude of natural and anthropogenic effects that negatively influence the status of the species. Many anthropogenic effects occur as a result of activities outside of U.S. jurisdiction (i.e., fisheries in international waters).

4 ENVIRONMENTAL BASELINE

This section contains a description of the effects of past and ongoing human activities leading to the current status of the species, their habitat, and the ecosystem, within the action area. The environmental baseline is a snapshot of the factors affecting the species and includes federal, state, tribal, local, and private actions already affecting the species, or that will occur contemporaneously with the consultation in progress. Unrelated, future federal actions affecting the same species that have completed formal or informal consultation are also part of the environmental baseline, as are implemented and ongoing federal and other actions within the action area that may benefit listed species.

4.1 Status of Sea Turtles in the Action Area

Sea turtle species occurring in the Singer Island vicinity that may be adversely affected by the proposed action are green, hawksbill, leatherback, and loggerhead sea turtles. Sea turtles found in the immediate project area may travel widely throughout the Atlantic, Gulf of Mexico, and Caribbean Sea, and individuals found in the action area can potentially be affected by activities anywhere within this wide range. These impacts outside of the action area are discussed and incorporated as part of the overall status of the species as detailed in Section 3 above. The following environmental baseline includes past and ongoing human activities in the action area that relate to the status of the species.

Green turtles nesting in this area are listed as endangered (along with Mexican nesting populations). It is expected that a disproportionate number of green turtles in the aquatic environment of the action area will be from the endangered Florida nesting population. Within the action area there is limited information on the status of green turtles regarding estimated numbers or population trends in the aquatic environment. It is known that juvenile green sea turtles show a preferential use of nearshore, high-energy, hardbottom habitat in the area as both foraging grounds and refuge/rest areas. Even after scouring from storms eliminated the algal growth in an area for prolonged periods, juvenile green turtles continue to show a preferential use of that habitat. It is thought that foraging in such cases takes place in nearby deeper habitats but the turtles return to take refuge in the crevices of the nearshore rock (Dynamac Corporation 2005). There are no studies within the action area assessing the trend or density of in-water green turtle populations at this time, but nesting trends on Singer Island have mirrored the marked long-term increase occurring statewide over the past decade. As is typical with green turtles, nesting tends to be cyclical, with a high year followed by one or two lower years, and then another high year. A 5-year average of 247 green turtle nests are recorded annually on this beach, producing an estimated 16,638 hatchlings (FWRI analysis of nesting beach data 2004-2008).

Hawksbill sea turtles, listed as endangered, are not as prevalent in the action area as green turtles. Preferring coral and sponge reef habitats for foraging, the action area does not provide substantial foraging habitat for hawksbills. However, although no nests have been recorded directly in the project area, hawksbill nests have been recorded on several Palm Beach County beaches, including the beach immediately adjacent to the project area in Palm Beach Shores (FWC letter to FDEP, April 20, 2009; Palm Beach County DERM status report-www.pbcgov.com/erm/lakes/estuarine/lake-worth-lagoon/pdf/EER_Status_Reports/Nov_2008.pdf). It is reasonable to expect that hawksbills may occasionally nest on the project area beach.

Endangered leatherback sea turtles are nesting on Singer Island in small, but increasing numbers, mirroring the overall long-term increasing nesting trend that is being seen on Florida Atlantic coast beaches. According to FWRI nesting data, the recent average is now 16 nests per year, producing an estimated 500 hatchlings per year (FWRI analysis of nesting beach data 2004-2008). Leatherback use of the in-water environment in the project area is limited as they tend to be pelagic in their habits and they do not utilize nearshore areas as regular foraging habitat.

For loggerhead sea turtles, it is likely that since the nesting beach falls within the Peninsular Florida Recovery Unit a disproportionate number of individuals in the action area would be from that recovery unit, and thus the status of that unit is particularly relevant to this area. Loggerhead nesting in the action area is showing a long-term negative trend similar to that of the overall recovery unit. The 5-year average of loggerhead nesting in the project area is 937 nests, with an expected average emergence of 57,789 hatchlings (FWRI analysis of nesting beach data 2004-2008). Loggerhead sea turtles are year-round residents of Florida coastal waters, including those off the project area. There are no reliable studies documenting the abundance or trend of loggerhead sea turtles utilizing the waters off Singer Island. Loggerhead sea turtles are known to utilize hardbottom and reef structures further offshore and outside of the dynamic nearshore zone that is preferred by juvenile green turtles.

All of these species are highly migratory. The same individuals found in the action area may migrate into offshore waters, as well as other areas of the Gulf of Mexico, Caribbean Sea, and North Atlantic Ocean, and thus be impacted by activities occurring there; therefore, the species' statuses in the action area are considered to be the same as their range-wide statuses and supported by the species accounts in Section 3.0. Because these species travel widely throughout the Atlantic, Gulf of Mexico, and Caribbean Sea, individuals in the action area are impacted by activities that occur in other areas within their geographic range.

4.2 Factors Affecting the Species Environment within the Action Area

As stated in Section 2.2 ("Action Area"), the proposed project is located off Singer Island, Palm Beach County, Florida, between DNR monuments R-61 and R-68, from the dunes out past the breakwater segments (200 feet offshore), and adjacent areas which may be impacted by changes in sand transport. The following analysis examines actions that may affect these species' environment specifically within the defined action area.

4.2.1 Federal Actions

In recent years, NMFS has undertaken several ESA section 7 consultations to address the effects of federally-permitted fisheries and other federal actions on threatened and endangered sea turtle species, and when appropriate, has authorized the incidental taking of these species. Each of those consultations sought to develop ways of reducing the probability of adverse effects of the action on sea turtles. Similarly, NMFS has undertaken recovery actions under the ESA that are addressing the problem of take of sea turtles in the fishing and shipping industries and other activities such as Army Corps of Engineers (COE) dredging operations. The summary below of anticipated sources of incidental take of sea turtles includes only those federal actions in the action area which have already concluded or are currently undergoing formal section 7 consultation. Authorized take levels are presented in Appendix A.

Dredging

The construction and maintenance of federal navigation channels and sand mining ("borrow") areas has been identified as a source of sea turtle mortality. Hopper dredges move relatively rapidly (compared to sea turtle swimming speeds) and can entrain and kill sea turtles as the drag arm of the moving dredge overtakes the slower moving sea turtle. The COE has biological opinions from NMFS covering hopper dredging in the Atlantic and Gulf of Mexico. Along the Atlantic coast of the southeastern United States, NMFS estimates that annual observed injury or mortality of sea turtles from hopper dredging may total 35 loggerheads, 7 greens, 7 Kemp's ridleys, and 2 hawksbills (NMFS 1997).

Beach Nourishment

The activity of beach nourishment, especially when impacts include the loss of nearshore hardbottom habitat along the east coast of Florida, has been identified as a source of incidental take of juvenile green sea turtles. Juvenile green turtles are known to utilize these high-energy, dynamic habitats for foraging and as refugia, and show a preference for this habitat even when abundant deeper-water sites are available. The loss of such limited habitat, especially when considering the cumulative loss as a result of beach nourishment activities occurring along the entire range of the habitat and continually over time, is expected to result in loss of foraging opportunities and protective refuge, which constitutes take under the ESA. The stresses are also expected to contribute to additional mortality of individuals already in poor condition as a result of disease or other factors (NMFS 2009). Recent beach nourishment projects for which consultation has been completed and an incidental take statement issued for juvenile green turtles as a result of hardbottom habitat loss include the Reach 8 project in Palm Beach County (6.95 acres, 19 lethal takes), the Mid-Reach project in Brevard County (2.95 acres, 15 non-lethal takes), and the Juno Beach project in Palm Beach County (1.7 acres, 8 non-lethal and 1 lethal take). The Reach 8 state permit was later invalidated by an administrative court ruling, but Palm Beach County is examining other options for erosion response in that area. Continued loss of hardbottom habitat is expected to result from beach nourishment activities within the range of that habitat type.

ESA Permits

The ESA allows the issuance of permits to take ESA-listed species for the purposes of scientific research (section 10(a)(1)(a)). In addition, the ESA allows for NMFS and individual states to enter into cooperative agreements developed under section 6 of the ESA, to assist in recovery actions of listed species. Prior to issuance of these authorizations, the proposal must be reviewed for compliance with section 7 of the ESA.

Sea turtles are the focus of research activities authorized by a section 10 permit under the ESA. Authorized activities range from photographing, weighing, and tagging sea turtles incidentally taken in fisheries, blood sampling, tissue sampling (biopsy), and performing laparoscopy on intentionally captured turtles. The number of authorized takes varies widely depending on the research and species involved, but may involve the taking of hundreds of turtles annually. Most takes authorized under these permits are expected to be non-lethal. Before any research permit is issued, the proposal must be reviewed under the permit regulations (i.e., must show a benefit to the species). In addition, since issuance of the permit is a federal activity, issuance of the permit by NMFS must also be reviewed for compliance with section 7 of the ESA to ensure that issuance of the permit does not result in jeopardy to the species.

4.2.2 State or Private Actions

Vessel Traffic
Commercial traffic and recreational pursuits can have an adverse effect on sea turtles through propeller and boat strike damage. The extent of the impact on sea turtles in the action area is not known at this time.

State Fisheries
There are no commercial state fisheries located in the nearshore, hardbottom habitat areas that comprise the action area. However, recreational fishing from private vessels and from shore does occur in the area. Observations of state recreational fisheries have shown that loggerhead, leatherback, and green sea turtles are known to bite baited hooks, and loggerheads frequently ingest the hooks. Hooked turtles have been reported by the public fishing from boats, piers, and beach, banks, and jetties and from commercial fishermen fishing for reef fish and for sharks with both single rigs and bottom longlines (NMFS 2001b). Additionally, lost fishing gear such as line cut after snagging on rocks, or discarded hooks and line, can also pose an entanglement threat to sea turtles in the area. A detailed summary of the known impacts of hook-and-line incidental captures to loggerhead sea turtles can be found in the TEWG reports (1998; 2000). The incidental capture of sawfish by recreational fishermen is possible within the action area as well.

In-water Research Projects
In Florida, in-water sea turtle research has increased in recent years, but no coordinated trend monitoring program exists for in-water populations. The first step in developing such a program involves determining what research is actually taking place. In 2008, researchers in FWRI's marine turtle program inventoried all in-water marine turtle research that has been conducted in Florida. Through the use of interviews, questionnaires, and literature reviews, researchers compiled a comprehensive database containing detailed information on 36 research projects (21 active, 15 inactive) focusing on in-water aggregations of sea turtles. Geographic Information Systems (GIS) maps were also developed for each project that will serve as examples to in-water researchers of how GIS can be used to enhance their studies (FWRI online article 2008 - http://research.myfwc.com/features/view_article.asp?id=27486).

The vast majority of in-water projects (24) are, or were, located on the southeast coast of Florida. Based on the information compiled, candidate projects were identified for inclusion in a statewide in-water index monitoring program that would provide trend information on sea turtles in Florida's waters. Recommendations were presented on how to develop such a program, which would include the measurement of capture effort, promotion of cooperation among in-water research groups, and standardization of data collection methods resulting in a consistent set of measurements.

In addition to dedicated in-water studies, other projects and activities were identified that involve the collection of sea turtle data, often secondary to the primary purpose. These projects provide important data on general turtle distributions and can identify target areas for future in-depth studies. Many of these projects are conducted by other sections of FWRI, including capture efforts and aerial surveys for manatees or fish. Other data come from incidental capture in fisheries research projects, or by the fisheries themselves. Pre-dredge trawling, sea turtle aerial surveys, stranding networks, and satellite tracking of sea turtles also provide important distributional data. The end result of this project is a narrative document that will function as a guide to in-water research in Florida.

4.2.3 Other Potential Sources of Impacts in the Environmental Baseline

A number of activities that may indirectly affect sea turtles in the action area of this consultation include anthropogenic marine debris and acoustic impacts. The impacts from these activities are difficult to measure. Where possible, conservation actions are being implemented to monitor or study impacts from these sources.

Marine Pollution
Sources of pollutants along the Atlantic coastal regions include atmospheric loading of pollutants such as polychlorinated biphenyl compounds (PCBs), stormwater runoff from coastal towns and cities into rivers and canals emptying into bays and the ocean, and groundwater and other discharges. Nutrient loading from land-based sources such as coastal community discharges is known to stimulate plankton blooms in closed or semi-closed estuarine systems. The effects on larger embayments are unknown. Although pathological effects of oil spills have been documented in laboratory studies of marine mammals and sea turtles (Vargo et al. 1986), the impacts of many other anthropogenic toxins have not been investigated.

Acoustic Impacts
Acoustic impacts can include temporary or permanent injury, habitat exclusion, habituation, and disruption of other normal behavior patterns. NMFS and the U.S. Navy are working cooperatively to assess military acoustic impacts (e.g., mid-range sonar) along the east coast of the United States (i.e., primarily North Carolina through Florida). Although focused on marine mammals, sea turtles may benefit from increased research on acoustics and reduction in noise levels.

4.2.4 Conservation and Recovery Actions Shaping the Environmental Baseline

NMFS and cooperating states have established an extensive network of Sea Turtle Stranding and Salvage Network (STSSN) participants along the Atlantic and Gulf of Mexico coasts that not only collect data on dead sea turtles, but also rescue and rehabilitate any live stranded sea turtles.

In response to the growing awareness of recreational fishery impacts on sea turtles, the Marine Recreational Fishery Statistics Survey (MRFSS) added a survey question regarding sea turtle interactions within recreational fisheries in 2006. NMFS is exploring potential revisions to MRFSS to quantify recreational encounters with sea turtles on a permanent basis.

5 EFFECTS OF THE ACTION

In this section of the opinion, we assess the effects of the proposed action on green, hawksbill, leatherback, and loggerhead sea turtles. The analysis in this section forms the foundation for our jeopardy analysis in Section 7. A jeopardy determination is reached if we would reasonably expect a proposed action to cause reductions in numbers, reproduction, or distribution that would appreciably reduce a listed species' likelihood of surviving and recovering in the wild.

The quantitative and qualitative analyses in this section are based upon the best scientific and commercial data available on sea turtle biology and the effects of the proposed action. When analyzing the effects of any action, it is important to consider indirect effects as well as the direct effects. Indirect effects are caused by or result from the proposed action, are later in time, and are reasonably certain to occur. Indirect effects include aspects such as habitat loss and degradation, reduction of prey/foraging base, etc.

The proposed project has been determined to have three primary routes of effects: impacts to nesting females by construction activities and the breakwaters, habitat loss, and breakwater impacts to hatchlings attempting to reach the open ocean. Each of these routes of effects will be discussed for each species, a determination made whether an adverse impact is expected from that component of the project, and if an adverse impact is expected, an examination of that impact on the species in the action area will be presented. It is important to note that the breakwaters themselves will be permanent and thus any impacts to habitat or sea turtles will be in perpetuity. Additional significant project impacts may include nesting beach impacts and reduction in female nesting success. These nesting beach-related issues are under the purview of the USFWS and are being analyzed under a separate biological opinion to be provided by USFWS.

5.1 In-Water Adult Impacts

As female sea turtles approach the beach to nest, the proposed project could potentially impact them in the short term, from construction activities, and for the duration of the breakwaters' existence, from impeding their access to the beach. NMFS has examined these potential routes of effect and has determined that these aspects of the proposed project may affect, but are not likely to adversely affect, nesting females of any species. Construction activities will only take place during daylight hours, and no equipment will be left in the project area overnight. NMFS concludes that the risk of injury to nesting females from the project's construction is discountable because turtles are highly mobile, nest primarily at night when construction activities would not be occurring, and in the event that their presence coincides with construction activities, they would likely avoid the area undergoing construction at that time. Construction is proposed to occur in segments, and thus construction activities at any given time would be limited in area. In addition, the applicant will be required to follow NMFS' March 23, 2006, Sea Turtle and Smalltooth Sawfish Construction Conditions, which will further reduce the potential for interactions with sea turtles from the proposed project. The presence of the breakwaters, especially given their fully emergent design, could potentially interfere with nearshore sea turtle mating as well as nesting females attempting to access the nesting beach. The direct impact to the in-water females themselves, however, is not expected to rise to the level of take and thus will be insignificant, as they have the ability to go around the structures and injury is unlikely. The likelihood of the emergent breakwaters resulting in reduced nesting numbers and success as a result of the impact on nesting females falls under the jurisdiction of the USFWS, and will be analyzed under a separate biological opinion to be completed by the USFWS.

5.2 Habitat Loss Impacts

Habitat impacts are expected from this project via the loss of low relief hardbottom habitat. The breakwater structures will directly cover 0.09 acres of persistent (exposed for at least 3 consecutive years) hardbottom. An additional 1.4 acres of persistent low relief hardbottom is expected to be permanently covered by sand, with another 1.3 acres periodically covered by sand. Therefore, a total of 2.79 acres of hardbottom habitat will be lost permanently or periodically as a result of the project.

Leatherback sea turtles tend to be pelagic and are uncommon in shallow nearshore waters. Leatherback utilization of the project area is not expected to include use of the hardbottom habitat to be lost. Leatherback impacts from the project are expected to be limited to those related to nesting (under the jurisdiction of FWS) and hatchlings.

Loggerhead sea turtles are known to utilize reefs and hardbottom habitat along the east coast of Florida. However, this species tends to utilize somewhat deeper waters than those found in dynamic, high-energy, nearshore habitats to be impacted by the proposed project. In 2003, 2004, and 2005 surveys of nearshore hardbottom habitat in Brevard County by Dynamac Corporation, a total of 163 turtles were sighted during visual transect surveys, only one of which was a loggerhead, and the rest were green turtles. Net-capture studies over that same period yielded 29 captures, all of them green turtles (Dynamac Corporation 2005). It is likely that loggerhead usage of Palm Beach County nearshore habitat is similar.

Researchers have suggested that hawksbill sea turtles (as well as juvenile green sea turtles) utilize nearshore hardbottom habitat as nighttime resting areas. Wershoven and Wershoven (1988) captured 134 green turtles and 4 hawksbill turtles while diving on a 1.5-km stretch of reef in nearby Broward County (south of the project area). Lawrence Wood has surveyed reef habitats in Palm Beach County for the presence of hawksbill sea turtles (Wood 2005, 2006, 2007). Wood's study is the first effort to systematically survey Florida's reefs for hawksbill turtles. His study area consists of nearshore reef habitat that runs parallel to the Florida coast as close as a few meters from the beach to approximately 5 km offshore from the southern boundary of Palm Beach Island (located south of the action area) north to Juno Beach, Florida. Wood reports that habitats in which hawksbill turtles have been observed can be characterized as "steep ledges with undercuts that include artificial reef wrecks, thick octocoral/a.k.a. gorgonian pastures, and sparse sandy patch reefs." Based on Wood's observations, most of the hawksbill turtles he has observed have been seen foraging on reef habitats, located waterward of nearshore hardbottom habitat, where prey items, such as sponges, are more abundant. Although hawksbill turtles may use the nearshore hardbottom habitat in the action area for foraging and resting, NMFS believes that this species is more likely to utilize offshore reef habitats located outside of the action area, because their favored prey items (e.g., sponges) are more abundant on the offshore reefs.

Based on our review of the aforementioned in-water sea turtle studies in southeast Florida, NMFS believes that loggerhead and hawksbill sea turtles may utilize nearshore hardbottom habitat in the action area for foraging and resting. However, NMFS believes that these species are more likely to forage and rest on reef habitats located offshore from the action area because these areas support sponges and other benthic invertebrates (such as crustaceans and mollusks) that are common prey items for hawksbill sea turtles and loggerhead sea turtles, respectively.

We believe that their preferred foraging and resting habitat will not be affected by the proposed project. Therefore, NMFS believes the project’s hardbottom habitat loss effects on loggerhead and hawksbill sea turtles will be insignificant (i.e., will not result in take).

The use of nearshore hardbottom habitat by juvenile green turtles is well documented. Makowski et al. (2006) observed 141 juvenile green turtles in nearshore hardbottom and reef habitats from July 2001 to November 2003. Sonic telemetry data from six green turtles suggest that these turtles forage and rest within the nearshore hardbottom and reef habitats in Palm Beach County. Wershoven and Wershoven (1992) have observed green sea turtles during aerial surveys of nearshore areas in Palm Beach, Broward, and Martin counties.

Makowski et al. (2006) studied the habitat use and home range movements of juvenile green sea turtles along a shallow, worm-rock reef tract in Palm Beach County, Florida. The Makowski et al. (2006) study area is located in close proximity to the action area and contains similar nearshore habitats (i.e., nearshore hardbottom). Makowski et al. (2006) found that when sufficient resources are available, individual turtles may develop affinities to specific areas for activities, such as foraging and/or resting. In the case of green sea turtles, juveniles occupy shallow-water developmental habitats (e.g., nearshore hardbottom and reef habitats) where they occupy distinct home ranges as they feed and grow to maturity. Six turtles were tracked via ultrasonic telemetry from August to November 2003. Each turtle’s esophagus was flushed via lavage to determine recently ingested foods. All six turtles were found to have a mixed diet of similar macroalgae and sponge fragments. The red algae *Gracilaria mammillaris* was found in 100 percent of the lavage samples. The results of this study suggest that juvenile green turtles occupy stable home ranges along the nearshore, worm-rock reefs of southeast Florida, during summer and fall.

Makowski et al. (2006) found that the home ranges of all six turtles tracked in the study were distributed exclusively over the algae-rich nearshore reef habitat. The minimum convex polygon (MCP) and fixed kernel density estimator (FKD) methods were used to estimate home range size of each turtle. The MCP home ranges measured between 0.69 and 5.05 square kilometers and the FKD home ranges spanned from 0.73 to 4.89 square kilometers. Turtle core areas were determined to be associated with either foraging activity centers or nocturnal resting sites. Foraging activity centers (50 percent FKD) measured between 0.18 and 1.17 square kilometers. All core areas were confined to the narrow, nearshore reef track and showed considerable overlap with neighboring turtles; none were located over sand. At night, each turtle returned to its own exclusive resting site along the nearshore reef tract. Four of the six turtles returned to only one resting site (different for each turtle) and the two turtles with the largest home range means each used two resting sites on opposite ends of their respective home ranges. No two turtles were observed at any one resting site.

While overlaps in evening resting areas were not observed, the study did report that all six turtles exhibited an overlap of core areas with at least three other tracked green turtles. These home range overlaps suggest that areas within this developmental habitat offer sufficient resources to be commonly shared by neighboring turtles (Makowski et al. 2006). In addition to foraging efficiency, knowledge of their physical environment may provide turtles a greater familiarity with escape routes, hiding places, and shelter from environmental extremes (Bailey 1984; Alcock 2001).

With movements confined to specific home ranges over the nearshore reefs of Palm Beach County, Florida, it is apparent that these individuals were resident to the study area during the summer and fall (Makowski et al. 2006). Such habitat affinity among juvenile green turtles has been demonstrated at other foraging areas (e.g., Brill et al. 1995; Renaud et al. 1995; Seminoff et al. 2002). Renaud et al. (1995) recorded some of the smallest daily movements for juvenile green turtles at a jettied pass in Texas. Their study incorporated extremely narrow jetty channels, where marine algae were the most readily available food source. In contrast, Seminoff et al. (2002) reported home ranges of 12 green turtles in Bahia de los Angeles, Gulf of California, Mexico, from 5.84 to 39.08 square kilometers. As the largest green turtle home ranges reported to date, Seminoff et al. (2002) suggested these home ranges resulted from the substantial distance between macroalgae food sources and benthic shelter sites within the 60-square-kilometer Bahia de los Angeles. Based on these and other studies of green turtle home range areas, Makowski et al. (2006) suggest that differences among green turtle home range size in relation to the varying physical structure of each foraging site supply further evidence that both home range size and shape, and thus the behavior of the turtles, are determined by the distribution of resources in a specific area.

Because the Makowski et al. (2006) study area is located off Palm Beach County, as is Singer Island, and the nearshore habitat is similar (i.e., nearshore hardbottom), NMFS believes that the nearshore hardbottoms in the project area provide important foraging and resting habitat for juvenile green sea turtles. Due to the importance of these areas as foraging and resting habitat for juvenile green turtles, it is reasonable to believe that juvenile green turtles using the nearshore hardbottoms that will be lost due to the proposed action will be adversely affected. The loss of foraging and resting areas could lead to a decrease in health due to malnourishment, an increase in stress due to disorientation, and a greater exposure to predators. Causing a decrease in a turtle's health can lead to injury or death.

Now that we have established the probability of take of green sea turtles as a result of the proposed action's effects on their foraging and resting territories, we must determine the number of turtles that may be injured or killed. While the Makowski et al. (2006) study provided home range area estimates for the six turtles that were tracked in Palm Beach County, we were not able to use this study to estimate population density for two reasons: 1) the small sample size (n = 6) and 2) the study identifies considerable overlap in core areas with neighboring turtles, suggesting that the turtles in the study were using common foraging areas. The study found that all six turtles exhibited an overlap of core areas with *at least* three other tracked turtles. However, we were not able to determine approximately how many individuals were sharing a common foraging area.

The nearshore hardbottom and reef habitats in nearby Broward County (south of the project area) also serve as developmental foraging and resting habitat for juvenile green turtles. Wershoven and Wershoven (1992) completed a 5-year study (1986-1991) on juvenile green turtles in their nearshore habitat of Broward County. They conducted SCUBA dives on a year-round basis. In Broward County, the offshore area consists of three main reef tracts that run parallel to the shore (the inner, middle, and outer reef). The nearshore reef extends from 75 to 375 meters offshore, and is 4 to 7 meters in depth. Reefs are colonized by soft and hard corals, with algae proliferating on the limestone (Wershoven and Wershoven 1992). Wershoven and Wershoven conducted their study on the nearshore first reef tract, where juvenile green turtles are most frequently sighted. A 20- by 1600-meter (32,000 square meter) area was surveyed on the first reef tract using SCUBA. Dives were conducted during the day and night. Turtles in the study were captured and brought to the surface where they were examined, measured, and released. The turtles were also tagged prior to release. Recaptured turtles whose tags were over three years old were given additional tags. During the 5-year study period (1986-1991), 134 juvenile green turtles were captured; 37 of the juvenile green turtles were re-encountered; 97 of the juvenile green turtles were captured only once. Turtles were captured during all months, with April, May, June, and August being the most productive and September and December the least productive (Wershoven and Wershoven 1992).

The results of the Wershoven and Wershoven (1992) study indicate that nearshore reef habitats likely support some juvenile green sea turtles with strong site affinity or residency (the re-encounters), and other juvenile green sea turtles that use the habitat with less regularity or affinity. Because 37 turtles were re-encountered, it is reasonable to believe that these 37 individuals were residents of the area. The other 97 turtles captured were likely utilizing the area for foraging and resting during some part of the year, but it is not likely they had any strong affinity to the habitat since they were not recaptured.

Both types of juvenile green turtles (i.e., strong site affinity vs. less affinity) currently using the reef habitat that will be lost as a result of the proposed project will be affected by the loss of foraging and resting/refuge habitat. The loss of foraging and resting areas could lead to a decrease in health due to malnourishment and an increase in stress due to short-term disorientation and long-term displacement to other nearshore foraging and resting areas outside of the action area. Causing a decrease in a turtle's health can lead to injury or death. Stressed, disoriented, and displaced individuals may be more susceptible to predation (compared to juvenile green turtles that have adequate foraging and resting/refuge habitat) while they are searching for alternate foraging and resting/refuge habitat. We believe the effects of having to locate habitat on turtles that do not have a particular affinity to the study area will be less significant than the effects on turtles with home ranges or strong affinity to the project site, as those without the strong affinity are more likely transient or otherwise have not already established a dependence on the project site. This has the potential to become a greater issue if cumulative losses of hardbottom along the entire range of the habitat result in significant habitat decreases. In contrast, we believe that some or all juvenile green sea turtles with strong affinity or residency in a particular reef habitat will be adversely affected by loss of that habitat.

To determine the number of turtles that may be injured or killed by the proposed action, we will use the numbers from Wershoven and Wershoven (1992) as a proxy to determine the number of resident juvenile green turtles that will be affected. This study is the best scientific information available on juvenile green turtle densities on nearshore hardbottoms, and the study area in Wershoven and Wershoven (1992) is environmentally similar and geographically close to the action area. If we convert the size of their study area (32,000 square meters) to acres, then the study area encompassed 7.9 acres. If we assume that recapture indicates strong site affinity or residency, then 37 juvenile green turtles that were recaptured were resident in approximately 7.9 acres of nearshore habitat. If we divide 37 individuals by 7.9 acres, this equals 5 resident juvenile green sea turtles per acre of this habitat type (actual total is 4.68 turtles rounded up to 5 turtles).

The proposed action would impact a maximum of approximately 2.79 acres of nearshore hardbottom habitat via permanent and periodic covering by sand. If we assume the possible worst-case scenario of a permanent loss of 2.79 acres of nearshore hardbottom, then the proposed project would displace approximately 14 juvenile green turtles (i.e., 5 turtles per acre multiplied by 2.79 acres = 13.95 turtles). Given that studies of juvenile green turtles have shown that dynamic, nearshore, hardbottom habitat is a highly preferred habitat for foraging and resting/refuge, even when substantial areas of reef and other habitat are available in nearby waters seaward of the nearshore hardbottom, it is apparent that the habitat has a premium value to the juvenile greens as developmental habitat. Although there is the potential for the displaced resident juvenile sea turtles to move to nearby, non-impacted hardbottom, this is a limited habitat type for which we do not know the carrying capacity for green turtles. This habitat is being impacted by similar projects along its entire span from Brevard County to Broward County, which both reduces the total acreage available, as well as decreases the connectivity between areas of exposed hardbottom.

One confounding measure of decreased health of green sea turtles that could be exacerbated by displacement from residence habitat is the incidence of fibropapillomatosis, the most commonly observed disease in green sea turtles. The prevalence of fibropapillomatosis varies widely in Florida waters, even sometimes within nearby areas. In Ehrhart et al. (2007), 52.8 percent of the juvenile green turtles in their long-term study in the Indian River Lagoon system of Florida had symptoms of fibropapillomatosis (n = 2,214), but even within that system over time the percentage varied greatly (28-72 percent), and the disease was nonexistent in nearby Port Canaveral (Hirama and Ehrhart 2003). The overall occurrence of fibropapillomatosis (showing external tumors) in green turtles of Florida waters was 22 percent based on stranding records from 1980-2005 (FWC 2007). In nearshore reefs of central eastern Florida it has ranged from 8-21 percent (Hirama and Ehrhart 2003). This is likely an over-estimation of resident turtles infected with fibropapillomatosis, as turtles with this condition are more likely to strand compared to healthy turtles, and thus the percentage does not necessarily reflect the prevalence of the disease within the population. The extent of the fibropapillomatosis infection among turtles can also vary widely, with some having only minor tumor growth with minimal health consequences, and others experiencing severe effects from extensive tumor coverage. Likewise, tumor regression has been known to occur over time, with tumor number and size decreasing in individual turtles (Hirama and Ehrhart 2007); thus, the effects are not always lethal or detrimental over the long term. Nonetheless, this is the best available, most conservative information on fibropapillomatosis infection rates for central eastern Florida at this time.

To estimate the number of resident juvenile green turtles in the project area that may have fibropapillomatosis, we take the reasonable conservative approach and use the state average based on strandings (22 percent), which is higher than the top end of the range for central eastern Florida nearshore reefs (8-21 percent) but lower than the Indian River Lagoon system, which appears to be a unique situation. If we multiply 14 juvenile green turtles (see above) by 22 percent, then approximately 3 of the 14 turtles resident to the nearshore hardbottom expected to be displaced by the proposed action may be infected with fibropapillomatosis. Because of the high degree of variability in how the disease impacts individual turtles, the lack of data on how much additional physiological stress the loss of hardbottom habitat presents for an individual turtle, and the fact that tumor regression is not well understood, no data is available to quantify the effects of habitat loss on turtles with fibropapillomatosis. However, if we consider the synergistic effects of fibropapillomatosis combined with the possible additional decreased health related to the loss of important foraging and resting habitat from the proposed project, it is reasonable to believe that death could result in an extreme case. In order to take a conservative approach, we make the assumption that this extreme case would arise as a result of this project. Therefore, of the estimated 3 resident juvenile green turtles infected with fibropapillomatosis, one may have a case that is advanced enough that it may die as a result of this additional decrease in health. Thus, we believe the proposed action may result in the lethal take of up to 1 juvenile green turtle.

In summary, the proposed action would impact a maximum of approximately 2.79 acres of nearshore hardbottom habitat, displacing an estimated 14 juvenile green turtles with strong site affinity to the impacted area. This habitat has a premium value to the juvenile greens as developmental habitat, and its loss can lead to decreased health and fitness which can lead to injury, reduced growth (affecting future survival and fecundity), or even death if the turtles are already under other major physiological stresses. In the case of the 14 turtles expected to be displaced by this activity, a general non-lethal decrease in health and fitness (non-lethal take) can reasonably be expected for 13 of them, and lethal take for 1, as explained above.

5.3 Hatchling Impacts

As described in section 4.1 above, Singer Island beaches provide important nesting habitat for sea turtles. Although USFWS will be completing a biological opinion that will analyze the project impact on nesting females and nest numbers, NMFS is responsible for analyzing the impacts to hatchlings that emerge from nests once they reach the water. We expect that some nesting reduction will occur along the beach parallel to the breakwaters. Such impacts have been observed in other large-scale, emergent breakwater projects such as in Japan, with substantial nesting declines overall, and deflection of some nesting up drift of the structures (Imamura 2009). Although the deflection of nesting to unblocked areas near the breakwater system would allow the females to nest, it presents a problem of concentrating nesting in those areas. This can lead to density-dependent effects on hatching success and hatchling survivorship similar to the impacts of hatchlings emerging on Singer Island beaches discussed below. Beaches adjacent to the Singer Island project area, where any deflected nesting would be expected to occur, already have nest densities high enough to produce density-dependent mortality effects (FWC Statewide Nesting Beach Survey).

The potential impacts to hatchlings as a result of the proposed project are complex and extensive, including a number of routes of effects, some of which compound each other. This opinion examines: the impacts of increased predation as a result of predator concentration at the new, extensive high-relief structure; the barrier to egress out to the open sea which the emergent structures may present to the hatchlings; and the compounding impact of the barrier effect increasing the severity of the predation effect. As detailed in section 4.1 above, the 1.1-mile stretch of beach parallel to the proposed breakwater system has documented nesting by loggerhead, green, and leatherback sea turtles. Although hawksbill sea turtles have not been documented nesting along that stretch of beach, sporadic records of individual nests elsewhere along the Palm Beach County coast exist, including on beaches adjacent to the project area beach (August 17, 2009, FWC letter to FDEP). Thus it is reasonable to expect that an occasional hawksbill nesting event could occur along that stretch of beach at some point in future years. Since fewer than ten hawksbill nests have been found in all of Palm Beach County from the 1980's through 2008 (DERM Environmental Enhancement and Restoration Division, November 2008 Status Report), it is reasonable to expect that fewer than one hawksbill nest per year could be impacted by the project. To err on the side of the species, we round up and assume that one nest per year will be impacted.

Predation on hatchlings can be a significant source of mortality between the time of hatching and when the hatchlings reach open waters. On the beach hatchlings can be preyed upon by birds, raccoons, opossums, stray dogs, crabs, and other animals. Once they reach the water, predation by large fishes, and to a lesser extent, sea birds, begins. It has long been understood and is well documented that fish, including large predatory fish, are attracted to high-relief structure to a much greater extent than sand bottom or low-relief hardbottom such as naturally occurs at the project site. In fact such structure is often used as fish aggregating devices. Hatchling sea turtles are preyed upon by large predatory fishes such as jacks, tarpon, barracuda, grouper, etc. as they attempt to reach the open ocean (Stewart and Wyneken 2004, Whelan and Wyneken 2007). Studies such as Witherington and Salmon (1992) have shown that predation of hatchling sea turtles was substantially higher in the vicinity of reef structure, even patchy, low-relief reefs, than over open sand (9% vs. 0%), and other studies have also shown high fish predation near reef structures and/or shallow waters (Gyuris 1994, Wyneken and Salmon 1996, Stewart and Wyneken 2004). It is also expected that the concentration of predators will increase over time at the breakwaters until the ecological carrying capacity is reached, as predatory fish recruit to the structure over subsequent seasons. Studies have shown that predation rates on hatchling sea turtles are also much higher when hatchling are more concentrated in a particular area (Wyneken et al. 1998, Wyneken et al. 2000), such as would be expected to happen as hatchlings accumulate against the breakwaters and because the hatchlings that do manage to orient towards the breakwater openings will be more concentrated in those gaps. Marine predators are known to learn to wait at locations of increased concentration for the hatchlings (Wyneken et. al. 2000). These predation increases are not expected to be limited to fish predation. Increases in predation on hatchlings by avian predators are expected as in addition to concentrating the hatchlings and keeping them in the shallower waters for longer periods of time, the emergent breakwaters also provides substrate for seabirds, shorebirds, and wading birds to perch. This may facilitate or even enable in-water avian predation that would not typically occur (e.g. herons and other wading birds having access to hatchlings that approach the breakwaters).

The presence of the breakwaters will also impact sea turtle hatchlings through a combination of disorientation and acting as physical barriers. Hatchlings typically orient out to sea by swimming perpendicular to waves, but other cues may be used in the absence of wave activity. Per the information provided in the biological assessment for this project, the applicant's consultant had determined that disorientation would not occur, as the openings in the breakwater would diffract the waves into a near circular field, leading the hatchlings out between the gaps. Scientists who have done much of the research related to hatchling orientation, however, believe the possibility of disorientation as a result of the breakwaters is significant, and can result in delays in reaching the open waters (Salmon pers. comm. 2008, Lohman pers. comm. 2008, Wyneken pers. comm. 2009). Lateral drift potentially created by the nearshore structures may also impact the hatchlings as their swim speed can be low compared to the drift, thus decreasing the likelihood of all of the hatchlings being able to navigate out of the gaps (Witherington pers. comm. 2009). It is reasonable to expect significant impacts to hatchlings resulting from the breakwaters as disorientation or physical blockage delay the hatchlings' ability to reach the open waters. Migration to the open sea incorporates frenzied activity by the hatchlings, which is known to be energetically very demanding on the hatchlings (Clusella Trullas et al. 2006). Also, the swim frenzy is based upon an internal clock that determines when the hatchlings switch from frenzy to post-frenzy swimming (Wyneken and Salmon 1992, Wyneken 2000). Prolonging the time in which hatchlings must continually attempt to reach deeper, open waters can be expected to have a significant, though un-quantifiable, impact on the hatchlings, such as excess resource expenditures resulting in physiological effects reducing later fitness. Additionally, the physical barrier presented by the proposed always-emergent breakwater segments is expected to pose the threat of impingement or stranding of the hatchlings upon the breakwaters. Impingement on the breakwaters is likely to injure or kill hatchlings through traumatic injury, exposure, or predation.

The predation effect and the disorientation, barrier, and impingement or stranding effects detailed above are expected to compound the impact to hatchlings beyond the additive impact of each considered separately. As discussed previously, high-relief structure off sea turtle nesting beaches are known to result in increased predator concentrations, and highly increased predation of hatchlings. Predation in shallow waters is in part mitigated by the swim frenzy, which typically results in the hatchlings traversing the shallows and reaching deeper, open waters as quickly as possible. The majority of predation on hatchlings appears to occur during the first fifteen minutes in the water, as typically the hatchlings have passed the shallower, predator rich waters within that time frame (Stewart and Wyneken 2004). The presence of large, emergent barriers in the area can be expected to have a potentially significant impact on the time it takes for hatchlings to swim past the shallow area. The expected delays in reaching open waters as a result of the aforementioned disorientation and physical barrier effect will result in significant increases in the hatchlings' exposure time to the already increased predator population in the area. Delays that stretch into dawn/daylight will likely cause drastically increased predation rates from birds and potentially exposure to additional diurnally active fish predators. Therefore, we expect that predation rates would be substantially higher than that which currently occurs off Singer Island.

Evaluating the impact of expected hatchling mortality on sea turtle populations must take into account a number of factors. Hatchling mortality could be exacerbated by reductions in nesting that we expect will result from the breakwaters. Nesting for green and leatherback sea turtles tends to be cyclical, with up and down years, and nesting is variable for all species year to year. It is also known that very few hatchlings ultimately survive to reach sexual maturity even under natural conditions. Estimates of hatchling survival vary widely in part as a result of the inability to track the survival of individuals from a population over the long life span of these animals. Actual survival rates may vary widely from beach to beach or other geographical scales depending on many factors, but most values based on back-calculating from stage-specific estimated survivorship range from one in a few hundred to less than one in one thousand (Frazer 1986, 1987, NMFS and USFWS 2008). Given the potential dramatic increase in hatchling mortality that can be expected from this project compared to that prior to the project, and the reduced number of hatchlings expected as a result of nesting reductions, the likelihood of a hatchling from this beach reaching sexual maturity would be substantially lower than the very low odds already facing hatchlings.

NMFS is taking the conservative approach mandated by the ESA, and resolving uncertainty regarding level of effects by erring in favor of the species. We are assuming that between expected nesting and nest success reductions, breakwater blocking of hatchling egress, substantial predation increases as a result of numerous compounding factors greatly increasing hatchling mortality rates off the beach, and the already low likelihood of hatchlings to reach maturity, the nests parallel to the footprint of the breakwater system will result in almost no hatchlings from that stretch of beach reaching sexual maturity. It is likely that some very small proportion will reach maturity relative to the already small percentage that would have in the absence of the proposed project. However, this exact number is expected to be so low as to approach zero. Given the ESA-mandated requirement of making conservative assumptions that err on the side of the species, we are treating this de minimis value as zero. Thus, the biological significance of the 1.1-mile stretch of beach, as far as reproductive input into future adult green, hawksbill, leatherback, and loggerhead sea turtle populations, will be lost. The impacts are expected to be permanent as a result of the permanent nature of the breakwater system.

Assuming a loss of reproductive value for the stretch of beach parallel to the project area, and using nest numbers prior to any expected reduction in nesting, the following represents the maximum impact for each species to be used in the jeopardy analysis (nesting data from Palm Beach County DERM, attached to a February 12, 2010, e-mail; nesting data from FWC, in an August 17, 2009, letter to the FDEP):

- Data provided by DERM indicates that the 6-year (2003-2009) annual nesting average for green turtles on the stretch of beach parallel to the proposed breakwater system is 192 nests, with a range of 75 to 371. FWC estimated that green turtles average 115 eggs per nest in Florida (http://research.myfwc.com/features/view_article.asp?id=2496). Therefore, the 192 nest average for the area immediately parallel to the proposed breakwaters could be predicted to have over 22,000 eggs per year.

- Hawksbill nesting is rare and sporadic in the action area as it does not represent an important hawksbill nesting ground, and only an occasional nest over the course of multiple years is expected. Rounding up to one nest per year, and utilizing the average of 140 eggs per nest in Florida and the Caribbean (http://www.fws.gov/northflorida/SeaTurtles/Turtle%20Factsheets/hawksbill-sea-turtle.htm), the project area would be expected to have fewer than 140 hawksbill eggs per year.

- Data provided by DERM indicates that the 6-year (2003-2009) annual nesting average for leatherback turtles on the stretch of beach parallel to the proposed breakwater system is 7 nests, with a range of 2 to 17. FWC estimated that leatherback turtles average 80 fertilized eggs per nest in Florida (http://research.myfwc.com/features/view_article.asp?id=2479). Therefore, the 7 nest average for the area immediately parallel to the proposed breakwaters could be predicted to have over 560 fertilized eggs per year.

- Data provided by DERM indicates that the 6-year (2003-2009) annual nesting average for loggerhead turtles on the stretch of beach parallel to the proposed breakwater system is 774 nests, with a range of 685 to 965. FWC estimated that loggerhead turtles average 100-126 eggs per nest in Florida (http://research.myfwc.com/features/view_article.asp?id=2411). Therefore, using a mid-range value for eggs per nest, the 774 nest average for the area immediately parallel to the proposed breakwaters could be predicted to have over 87,000 eggs per year.

Different studies show variable hatching success rates for sea turtle eggs, including variation by species, time of year, and location, but rates are typically above 50 percent and often approaching 80+ percent in natural, in situ conditions (Fowler 1979, Mascarenhas 2004, Holloman and Godfrey 2005, Pike 2008). Therefore, even assuming a relatively low in situ hatchling rate of 50 percent, over 11,000 green, 280 leatherback, and 43,500 loggerhead hatchlings would be expected annually on average to enter the marine environment from the stretch of beach parallel to the proposed breakwater system.

5.4 Synthesis of Effects on Sea Turtles

As described in sections 5.1 to 5.3 above, there are a multitude of impacts expected to sea turtles as a result of the proposed project. Nesting beach impacts and in-water impacts that are expected to reduce nesting or nesting success are mentioned briefly, but fall under the purview of the USFWS, which is writing its own biological opinion. Habitat loss impacts, primarily to green turtles, are expected to occur and result in take of green turtles, both non-lethal and lethal. The most significant impact of the project will occur to hatchling sea turtles that emerge from the beaches parallel to the project area. The beach along this area represents a very important, high density nesting beach for three species of turtles, and is especially important to the Peninsular Florida Recovery Unit of loggerhead turtles. This recovery unit of loggerhead turtles is one of the largest and most important in the world, and is currently experiencing a decade-long, significant declining trend, as are many other loggerhead nesting populations in the northwest Atlantic Ocean. The project is expected to create a gauntlet of compounding effects impeding the hatchlings' access to open waters. The construction of extensive high-relief structure in the water is expected to result in the concentration of both piscine and avian predators of hatchlings. This effect will be further compounded by the nesting locations, and thus hatchling emergence, becoming more concentrated as the females are funneled through the breakwater openings to the beaches. Likewise, the fully emergent design of the breakwaters will result in the hatchlings being more concentrated within the gaps as they attempt to bypass the breakwater system, thus providing predatory species with locations in which to focus. The predation effect will be further compounded by the barrier effect of the emergent breakwaters, as hatchlings will experience increased times in the nearshore waters as they attempt to navigate their way around the structures, substantially increasing their exposure time to predators. The barrier can also result in impingement or stranding of hatchlings. The multiple, compounding impacts of the breakwater system are expected to be profound and all the more significant given that the impacts will be permanent as a result of the presence of the structures.

6 CUMULATIVE EFFECTS

Cumulative effects are the effects of future state, local, or private activities that are reasonably certain to occur within the action area considered in this biological opinion. Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA. Within the action area, major future changes are not anticipated in ongoing human activities described in the environmental baseline. The present, major human uses of the action area such as commercial fishing, recreational boating and fishing, and the transport of petroleum and other chemical products through the sound, are expected to continue at the present levels of intensity in the near future as are their associated risks of injury or mortality to sea turtles posed by incidental capture by fishermen, accidental oil spills, vessel collisions, marine debris, chemical discharges, and man-made noises.

As discussed earlier, however, listed species of turtles migrate throughout the Gulf of Mexico and Atlantic and may be affected during their life cycles by non-Federal activities outside the action area.

Beachfront development, lighting, and beach erosion control are all ongoing activities along the southeastern coast of the United States. These activities potentially reduce or degrade sea turtle nesting habitats or interfere with hatchling movement to sea. Nocturnal human activities along nesting beaches may also discourage sea turtles from nesting sites. The extent to which these activities reduce sea turtle nesting and hatchling production is unknown. However, more and more coastal counties have or are adopting more stringent protective measures to protect hatchling sea turtles from the disorienting effects of beach lighting. Some of these measures were drafted in response to law suits brought against the counties by concerned citizens who charged the counties with failing to uphold the ESA by allowing unregulated beach lighting which results in takes of hatchlings. The cumulative impacts of large-scale breakwater projects are also important to consider. As stand-alone projects on a limited stretch of beach, an emergent breakwater might not jeopardize any listed sea turtle species. However, the cumulative impact of multiple such projects along significant stretches of nesting beach on Florida's east coast, such as the proposed Jupiter Beach breakwater, in conjunction with impacts from other beach protection and nourishment projects, may cumulatively present significant adverse impacts to sea turtle species.

7 JEOPARDY ANALYSIS

This section considers the likelihood that the proposed action will jeopardize the continued existence of green, hawksbill, leatherback, and loggerhead sea turtles in the wild. To *jeopardize the continued existence of* is defined as "to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 CFR 402.02). Sub-sections 3.2.1.3 ("Summary of Green Sea Turtle Status"), 3.2.2.3 ("Summary of Hawksbill Sea Turtle Status"), 3.2.3.3 ("Summary of Leatherback Sea Turtle Status"), and 3.2.4.3 ("Summary of Loggerhead Sea Turtle Status"), describe the status of the species affected by the proposed action. Section 5 ("Effects of the Action") describes the effects of the proposed action on green, hawksbill, leatherback, and loggerhead sea turtles, and the extent of those effects in terms of an estimate of the number of sea turtles that would be killed or otherwise taken. The following jeopardy analysis first considers the effects of the action to determine if we would reasonably expect the action to result in reductions in reproduction, numbers, or distribution of these sea turtle species. The analysis next considers whether any such reduction would in turn result in an appreciable reduction in the likelihood of survival of these species in the wild, and the likelihood of recovery of these species in the wild. In the following analysis, we demonstrate that the anticipated take of 14 juvenile green turtles (1 lethal) via loss of 2.79 acres of hardbottom habitat, and the loss of reproductive contribution to the populations of green, hawksbill, and leatherback turtles from the 1.1-mile stretch of beach parallel to the proposed breakwater system is not expected to appreciably reduce any of the three species' likelihood of survival or recovery in the wild. However, our analysis concludes that the loss of reproductive contribution from the project area to the Peninsular Florida Recovery Unit of loggerhead sea turtles is expected to appreciably reduce the likelihood of survival and recovery of loggerhead sea turtles in the wild.

7.1 Effects of the Action on the Likelihood of Survival and Recovery in the Wild

This section analyzes the effects of the action on the likelihood of survival of each species in the wild. In this context, the survival of the species refers to the continued existence of the species in the wild, and whether or not any anticipated take of that species will result in any reduction in reproduction, numbers, or distribution of that species that may appreciably increase a species' risk of extinction in the wild.

In the following analysis, we find that although some reduction in numbers and reproduction is expected, the anticipated take of green, hawksbill, and leatherback sea turtles will not appreciably increase the risk of extinction of these species in the wild. We also find that the reduction in numbers and reproduction of loggerhead sea turtles is expected to appreciably increase the risk of extinction in the wild for that species.

All life stages are important to the survival and recovery of the species; however, it is important to note that individuals of one life stage are not equivalent to those of other life stages. For example, the take of male juveniles may affect survivorship and recruitment rates into the reproductive population in any given year, and yet not significantly reduce the reproductive potential of the population. A very low percent of hatchlings is typically expected to survive to reproductive age. The death of mature, breeding females can have an immediate effect on the reproductive rate of the species. Sub-lethal effects on adult females may also reduce reproduction by hindering foraging success, as sufficient energy reserves are probably necessary for producing multiple clutches of eggs in a breeding year. Different age classes may experience varying rates of mortality and resilience.

Green Sea Turtles
The anticipated annual lethal take of up to 1 juvenile green turtle and the loss of all hatchling contribution to the breeding population from the 1.1-mile stretch of beach parallel to the breakwater system is a reduction in numbers. This lethal take, as well as the non-lethal take of an additional 13 juveniles due to hardbottom habitat loss, is expected to result in a reduction in reproduction as well, as a result of reductions in fitness and growth prior to maturity of the juveniles that survive to reproductive age.

Data provided by DERM indicates that the 6-year (2003-2009) annual nesting average for green turtles on the stretch of beach parallel to the proposed breakwater system is 192 nests, with a range of 75 to 371. FWC estimated that green turtles average 115 eggs per nest in Florida (http://research.myfwc.com/features/view_article.asp?id=2496). Therefore, the 192 nest average for the area immediately parallel to the proposed breakwaters could be predicted to have over 22,000 eggs per year. This does not account for the fact that green turtle nesting has been steadily increasing in Florida in recent years. Based upon statewide nesting data from FWC the 5-year average for green turtle nesting in Florida is 8,034 nests per year, and thus nesting on the stretch of beach parallel to the proposed breakwater system represents about 2.4 percent of the average annual nesting in Florida. Thus, a loss of this productivity would represent a considerable loss of current green turtle nesting in Florida.

However, as reported in the August 2007 ESA 5-year review of the green sea turtle (NMFS and USFWS 2007a), nesting populations are stable or increasing in all rookery areas in the Western Atlantic Ocean, including rookeries in Costa Rica, Florida, Mexico, Venezuela, and Suriname.

Further, based on the results from the first 24 years of an ongoing study of the composition, population structures, and population trends of green sea turtles in the central region of the Indian River Lagoon in Florida, Ehrhart et al. (2007) reported a 661 percent increase in juvenile green turtle capture rates at their study area. This increase in capture rates is similar to those recorded at the St. Lucie Power Plant over a similar period (Wilcox et al. 1998). During the 24-year period studied by Ehrhart et al. (2007), green turtle nest deposition in Florida has increased exponentially. Since 1982, Ehrhart et al. (2007) have surveyed marine turtle nesting on a 21-km stretch of beach in southern Brevard County, Florida, now part of the Archie Carr National Wildlife Refuge. From 1990-91 to 2004-05, green turtle nest deposition increased 358 percent in southeast Florida (Ehrhart et al. 2007). Since 1989, the Florida Fish and Wildlife Research Institute's results of monitoring from index nesting beaches shows that 90 percent of Florida green turtle nest deposition occurs in southeast Florida (Brevard through Miami-Dade counties). The pattern of green turtle nesting shows biennial peaks in abundance, with a generally positive trend during the twenty years of regular monitoring since establishment of index beaches in Florida in 1989.

We believe that the expected impact of one juvenile green turtle mortality, the injury through displacement of up to 13 resident juveniles, and the loss of the reproductive value of an area representing about 2.4 percent of the average annual nesting in Florida represents a substantial adverse impact to the species. However, this species is currently showing a very large increasing nesting trend in Florida, a population that is listed separately as endangered, with nesting numbers already approaching or exceeding those required by the recovery plan for the species. Therefore, we believe that the reduction in reproduction as a result of the anticipated takes detailed above is not expected to appreciably reduce the likelihood of survival of the species, and the reduction in species numbers is not expected to appreciably reduce the likelihood of survival of green sea turtles in the wild.

Green sea turtles are highly migratory, and individuals from all Atlantic nesting populations may range throughout the Gulf of Mexico, Atlantic Ocean, and Caribbean Sea. While the potential take would result in a displacement of individuals from important developmental habitat as well as the loss of reproductive value for the 1.1-mile stretch of beach, the loss is not significant in terms of local, regional, or global distribution as a whole. The Florida population distribution would be expected to remain the same, but with a loss of productivity in the 1.1-mile stretch. Therefore, we believe the anticipated impacts will not affect the species' distribution, and the reductions in numbers and reproduction are not expected to appreciably reduce the species' likelihood of survival in the wild.

We also consider the recovery objectives in the recovery plan prepared for the U.S. populations of green sea turtles that may be affected by the predicted reduction in numbers and reproduction. The recovery plan for green sea turtles (NMFS and USFWS 1991) lists the following relevant recovery objectives:

(1) The level of nesting in Florida has increased to an average of 5,000 nests per year for at least 6 years. Nesting data must be based on standardized surveys.

    Status: An average of 5,039 green turtle nests were laid annually in Florida between 2001 and 2006, with a low of 581 in 2001 and a high of 9,644 in 2005 (NMFS and USFWS 2007a). That average increased to 8,034 nests per year for the 5 year period of 2004-2008. Data from the index nesting beach program in Florida support the dramatic increase in nesting. In 2007, there were 9,455 green turtle nests found just on index nesting beaches, the highest since index beach monitoring began in 1989. The number fell back to 6,385 in 2008, but that is thought to be part of the normal biennial nesting cycle for green turtles (FWC Index Nesting Beach Survey Database).

(2) A reduction in stage class mortality is reflected in higher counts of individuals on foraging grounds.

Status: There are no reliable estimates of the number of immature green sea turtles that inhabit coastal areas (where they come to forage) of the southeastern United States. However, information on incidental captures of immature green sea turtles at the St. Lucie Power Plant (they have averaged 215 green sea turtle captures per year since 1977) in St. Lucie County, Florida, show that the annual number of immature green sea turtles captured has increased significantly in the past 26 years (FPL 2002). Ehrhart et al. (2007) has also documented a significant increase in in-water abundance of green turtles in the Indian River Lagoon area.

The expected take described above will result in a reduction in numbers and reproduction, but will not have any detectable influence on the population and nesting trends noted above. The average loss per year will not have an appreciable impact on total recruitment of new sea turtles to the population given the extent of the impact versus the very rapid population increases occurring over the past decade. The estimated non-lethal take described above would not affect these trends either as they are not expected to impact the survival, distribution, or fecundity of individuals taken in an appreciable manner relative to the population size. Thus, the proposed action will not interfere with achieving the recovery objectives above and will not result in an appreciable reduction in the likelihood of green sea turtles' recovery in the wild.

Hawksbill Sea Turtles

The anticipated loss of reproductive value for hawksbill sea turtles as a result of the project is expected to be minimal. As detailed previously, hawksbill nesting is rare and sporadic in the action area as it does not represent an important hawksbill nesting ground, and only an occasional nest over the course of multiple years is expected. Rounding up to the very conservative estimate of up to one nest per year, and utilizing the average of 140 eggs per nest in Florida and the Caribbean (http://www.fws.gov/northflorida/SeaTurtles/Turtle%20Factsheets/hawksbill-sea-turtle.htm), the project area would be expected to have fewer than 140 hawksbill eggs per year.

Therefore, we believe there will be no appreciable reduction in reproduction as a result of the anticipated takes detailed above, and the loss of reproductive value via decreased nesting and hatchling mortality is not expected to appreciably reduce the likelihood of survival of hawksbill sea turtles in the wild.

Hawksbill sea turtles are highly migratory, and individuals may range throughout the Gulf of Mexico, Atlantic Ocean, and Caribbean Sea. While the potential take would result in a loss of reproductive value for the action area, the loss is not significant in terms of local, regional, or global distribution as a whole, especially given the very minimal nesting by hawksbills in the action area and surrounding beaches. Therefore, we believe the anticipated impacts will not affect the species' distribution, and the reductions in numbers and reproduction are not expected to appreciably reduce the species' likelihood of survival in the wild.

We also consider the recovery objectives in the recovery plan prepared for the U.S. populations of hawksbill sea turtles that may be affected by the predicted reduction in numbers. The recovery plan for hawksbill sea turtles (NMFS and USFWS 1993) concludes that the U.S. populations of hawksbill turtles can be considered for delisting if, over a period of 25 years, the following recovery criteria are met:

(1) The adult female population is increasing, as evidenced by a statistically significant trend in the annual number of nests on at least five index beaches, including Mona Island and Buck Island Reef National Monument (BIRNM).

Status: To date hawksbill nesting on U.S. beaches does not show a clear trend. Hawksbill nesting is solitary, and often occurs at remote beaches. Nesting is also very limited, with Mona Island as the predominant site with only 500-1000 (Diez and van Dam 2006) nests per year, and thus determining a trend is difficult. The two largest nesting populations, Mona Island and BIRNM do appear to have been experiencing an increase over the last few decades (Meylan 1999) but the overall U.S. nesting shows no clear signs of recovery after the severe population reductions that occurred in the 20th century (NMFS and USFWS 1993, Meylan and Donnelly 1999). U.S. Caribbean nesting is not expected to be impacted by the loss of rare, sporadic nesting on the east coast of Florida

(2) Numbers of adults, subadults, and juveniles are increasing, as evidenced by a statistically significant trend on at least five key foraging areas within Puerto Rico, USVI, and Florida.

Status: There are no reliable data to determine the trend of hawksbill turtle abundance in the key foraging areas within U.S. waters.

The potential take described above will result in a reduction in numbers and reproduction, but will not have any detectable influence on the population and nesting trends noted above because the average loss per year will not have a measurable, discernible, or appreciable impact on total recruitment of new sea turtles to the population. Thus, the proposed action will not interfere with achieving the recovery objectives above and will not result in an appreciable reduction in the likelihood of hawksbill sea turtles' recovery in the wild.

Leatherback Sea Turtles

Data provided by DERM indicates that the 6-year (2003-2009) annual nesting average for leatherback turtles on the stretch of beach parallel to the proposed breakwater system is 7 nests, with a range of 2 to 17. FWC estimated that leatherback turtles average 80 fertilized eggs per nest in Florida (http://research.myfwc.com/features/view_article.asp?id=2479). Therefore, the 7 nest average for the area immediately parallel to the proposed breakwaters could be predicted to have over 560 eggs per year. This does not account for the fact that leatherback turtle nesting has been steadily increasing in Florida in recent years. Based upon statewide nesting data from FWC the 5-year average (2004-2008) for leatherback turtle nesting in Florida is 793 nests per year, and thus nesting on the stretch of beach parallel to the proposed breakwater system represents just under one percent of the average annual nesting in Florida. As reported in the leatherback sea turtle population assessment (TEWG 2007), the Florida nesting stock has been increasing dramatically over the past two decades, and nesting stocks throughout much of the Atlantic are stable or increasing. Additionally, Florida represents the northern limits of leatherback nesting in the Atlantic and is one of the smallest nesting grounds for this species in the western hemisphere.

Leatherback sea turtles are highly migratory, and individuals may range throughout the Gulf of Mexico, Atlantic Ocean, and Caribbean Sea. While the potential take would result in a loss of reproductive value for the action area, given the limited leatherback nesting that occurs in the action area and Florida in general compared to other nesting grounds for the species, and the increasing nesting trend in Florida as well as at the larger nesting beaches, the loss is not significant in terms of local, regional, or global distribution as a whole. Therefore, we believe the anticipated impacts will not affect the species' distribution, and the reductions in numbers and reproduction are not expected to appreciably reduce the species' likelihood of survival in the wild.

We also consider the recovery objectives in the recovery plan prepared for the U.S. populations of leatherback sea turtles that may be affected by the predicted reduction in numbers and reproduction. The recovery plan for leatherback sea turtles (NMFS and USFWS 1992) lists the following relevant recovery objectives:

(1) The adult female population increases over the next 25 years, as evidenced by a statistically significant trend in the number of nests at Culebra, Puerto Rico, St. Croix, USVI, and along the east coast of Florida.

> Status: Leatherback nesting has been increasing at these beaches. The estimated annual growth rates of the nesting female population using those beaches was 1.1 for Puerto Rico (1984-2005) and the USVI (1986-2004), and 1.17 for the east coast of Florida (1989-2005) (TEWG 2007).

The potential take described above will result in a reduction in numbers and reproduction, but will not have any detectable influence on the population and nesting trends noted above because the average loss per year will not have a measurable, discernible, or appreciable impact on total recruitment of new sea turtles to the population. Thus, the proposed action will not interfere with achieving the recovery objectives above and will not result in an appreciable reduction in the likelihood of leatherback sea turtles' recovery in the wild.

Loggerhead Sea Turtles

Data provided by DERM indicates that the 6-year (2003-2009) annual nesting average for loggerhead turtles on the stretch of beach parallel to the proposed breakwater system is 774 nests, with a range of 685 to 965. FWC estimated that loggerhead turtles average 100-126 eggs per nest in Florida (http://research.myfwc.com/features/view_article.asp?id=2411). Therefore, using a mid-range value for eggs per nest, the 774 nest average for the area immediately parallel to the proposed breakwaters could be predicted to have over 87,000 eggs per year. Based upon statewide nesting data from FWC the 5-year average (2004-2008) for loggerhead turtle nesting in Florida is 51,194 nests per year, and thus nesting on the stretch of beach parallel to the proposed breakwater system represents one and a half percent of the average annual nesting in Florida.

The Florida statewide nesting number is roughly equal to that of the Peninsular Florida Recovery Unit designated in the latest recovery plan (NMFS and USFWS 2008), minus the much smaller amount of nesting in the panhandle and the islands west of Key West, Florida.

The Peninsular Florida Recovery Unit, as explained previously, is by far the largest recovery unit among Northwest Atlantic loggerhead populations. It also represents one of the two largest and thus most important loggerhead nesting populations in the world. Conservation of this recovery unit is therefore critical to the conservation of loggerhead turtles in the Atlantic and worldwide. Although the Peninsular Florida Recovery Unit and the Northwest Atlantic populations are presently large in terms of absolute numbers of individuals, nesting and presumably population trends for the recovery unit have experienced significant declines over the past decade as described in the Status of the Species section above, and they continue to decline. This decline has occurred despite protections afforded to loggerheads since their listing in 1978, including strong management of nesting beaches. Leatherback and green turtle nesting in Florida have been increasing dramatically under the same management regime; therefore it appears that the existing protections of loggerhead reproductive output have not been sufficient to offset at-sea mortality of loggerheads in this recovery unit.

The significance of the expected impacts from the proposed action results from a combination of factors: the loss would represent at least one and a half percent of the annual nesting production for the largest recovery unit in the western hemisphere; the loss would occur annually; the loss would be permanent; and the loss would be a wholly new impact to be added to the baseline of impacts that is already resulting in significant population decline. Stabilization of declining loggerhead nesting will require reducing at-sea mortality and maintaining or improving beach protection measures (NMFS and USFWS 2008). Reducing the long-term effective reproductive output of Florida loggerheads by an additional one and a half percent would further reduce the population's resilience to mortality that is already exceeding the population's capacity to replace losses. These additional impacts will further, perhaps significantly, extend the time required to stabilize nesting and population trends. Even if measures to increase survival rates of juvenile and adult loggerheads at sea were to improve in the near future, which is not at all certain, the impacts of the proposed action would accrue and accumulate annually until the lost hatchling production becomes apparent approximately 30 years in the future when those hatchlings would have recruited into the adult population. We believe that the impacts of the proposed action on the numbers and reproduction of the Peninsular Florida Recovery Unit of loggerheads – a considerable decrease in the reproductive capacity of a population that is already declining – represents a marked increase in the extinction risk for that population. Given the importance of the Peninsular Florida Recovery Unit to the Northwest Atlantic populations of loggerheads, which is the one of the two largest populations globally, we believe that the reduction in reproduction and numbers as a result of the anticipated takes detailed above would appreciably reduce the likelihood of survival of loggerhead sea turtles in the Atlantic Ocean.

Loggerhead sea turtles are highly migratory, and individuals may range throughout the Gulf of Mexico, Atlantic Ocean, and Caribbean Sea. While the potential take would result in the loss of reproductive value for the 1.1-mile stretch of beach, a significant loss in terms of reproduction and numbers, the loss is in terms of local, regional, or global in-water distribution is not expected to be significant. Although the population of the recovery unit and the NW Atlantic population would be reduced, the range of nesting and in-water habitat use would not. Therefore, we believe the anticipated impacts will not affect the species' distribution.

We also consider the recovery objectives in the recovery plan prepared for the U.S. populations of loggerhead sea turtles that may be affected by the predicted reduction in numbers and reproduction. The recovery plan for loggerhead sea turtles (NMFS and USFWS 2008) lists the following relevant recovery criteria for the PFRU. However, loggerhead aggregations on foraging grounds tend to be from mixed recovery units and thus recovery goals for other recovery units would also be pertinent, and their status is also listed below.

(1) There is statistical confidence (95%) that the annual rate of increase over a generation time of 50 years is statistically detectable (1%) resulting in a total annual number of nests of 106,100 or greater for this recovery unit.

(2) This increase in number of nests must be a result of corresponding increases in number of nesting females (estimated from nests, clutch frequency, and remigration interval).

Status: The South Florida nesting population showed a decrease in nests of 26 percent over the 20-year period from 1989-2008. Analysis of nesting data from 1989-2005 by the Florida Wildlife Research Institute indicates there is a significant declining trend in nesting at beaches utilized by the South Florida nesting subpopulation (McRae letter to NMFS, October 25, 2006). Data obtained for the 2006 and 2007 nesting seasons are also consistent with the decline in loggerhead nests. In 2008, overall nesting trend data still indicate a significant declining trend (FWRI Index Nesting Beach web site: http://research.myfwc.com/features/view_article.asp?id=10690). It has been unclear whether the nesting decline reflects a decline in population, or is indicative of a failure to nest by the reproductively mature females as a result of other factors (resource depletion, nesting beach problems, oceanographic conditions, etc.). However, recent analysis of the data has led to the conclusion that the nesting decline can best be explained by an actual decline in the number of adult female loggerheads in the population (Witherington et al. 2009).

In North Carolina, South Carolina, and Georgia, an average of 5,215 nests were documented from 1989-2008. Standardized ground surveys of 11 North Carolina, South Carolina, and Georgia nesting beaches showed a significant decline of 1.3 percent annually in loggerhead nesting from 1989-2008. In addition, standardized aerial nesting surveys in South Carolina have shown a 1.9 percent annual decline in nesting in South Carolina since 1980. Overall, there is strong statistical data to suggest a long-term decline.

The Florida Panhandle nesting subpopulation showed a significant declining trend of 4.7 percent annually. No trend in the annual number of nests was detected in the Dry Tortugas nesting subpopulation from 1995-2004; because of the annual variability in nest totals, a longer time series is needed to detect a trend.

The projected take described above will result in a reduction in numbers and reproduction, and is expected to have a considerable adverse impact on the population and nesting trends noted above because the average loss per year (approximately one and a half percent of the PFRU nesting) will have a measurable, discernible, and appreciable impact on total recruitment of new sea turtles to what is an already declining population. Thus, the proposed action will interfere with achieving the recovery objectives above and is expected to result in an appreciable reduction in the likelihood of loggerhead sea turtles' recovery in the Atlantic Ocean.

8 CONCLUSION

We have analyzed the best available data, the current status of the species, environmental baseline, effects of the proposed action, and cumulative effects and determined that the proposed action is not likely to jeopardize the continued existence of green, hawksbill, and leatherback sea turtles. However, we have determined that the proposed action, given the current status of the species, environmental baseline, and cumulative effects, is likely to jeopardize the continued existence of loggerhead sea turtles in the Atlantic Ocean. Given that these populations of loggerhead sea turtles are proposed to be listed as an endangered Distinct Population Segment, and that this proposed action is likely to jeopardize this DPS' continued existence, ESA section 7(a)(4) requires a conference consultation regarding these impacts.

It is also our biological opinion that the action as proposed will jeopardize the continued existence of loggerhead sea turtles in their current listing as a single species distributed globally. Given the depleted and precarious status of the other loggerhead populations summarized in section 3 above, jeopardizing the continued existence of Northwest Atlantic loggerheads would appreciably increase the extinction risk of the entire species. As discussed above, the abundance of loggerhead sea turtles in nesting colonies throughout the Pacific basin has declined dramatically over the past 10 to 20 years. Loggerhead turtle colonies in the Western Pacific Ocean have been reduced to a fraction of their former abundance by the combined effects of human activities that have reduced the number of nesting females and reduced the reproductive success of females that manage to nest. Though southwestern Indian Ocean loggerhead nesting has shown signs of recovery in South Africa where protection measures have been in place for decades, in other southwestern areas (e.g., Madagascar and Mozambique) loggerhead nesting groups are still affected by subsistence hunting of adults and eggs. The largest known nesting group of loggerheads in the world occurs in Oman in the Northern Indian Ocean, with an estimated 20,000-40,000 females nesting each year at the largest nesting site. In the Eastern Indian Ocean, all known nesting sites are found in Western Australia where nesting occurs disproportionately at a single location. Nesting in the Mediterranean is confined almost exclusively to the eastern basin, and the highest level of nesting occurs in Greece, with an average of 3,050 nests per year. As explained in section 3 above, the Northwest Atlantic populations of loggerhead sea turtles, and the Florida Peninsular Recovery Unit in particular, rivals the Oman population as being the largest population worldwide. These two populations are far larger than any other worldwide, and are of utmost importance to the survival of the species. The stability and level of protection for the Oman population is unclear and may be tenuous, and thus the importance of the Northwest Atlantic population in terms of global survival of the species, as currently listed, is of paramount importance. Therefore, it is our biological opinion that jeopardizing the continued existence of Northwest Atlantic populations of loggerhead sea turtles will appreciably reduce the likelihood of survival and recovery of the species worldwide.

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

No RPA is being provided for this biological opinion. Given the nature of the intended purpose of the action, the lack of alternatives proposed by the applicant, and NMFS' lack of coastal engineering expertise to propose appropriate alternatives, we cannot include an RPA. If the applicant and the USACE wish to explore other methods for meeting the applicant's stated needs and goals NMFS will work with them to help ensure that a solution is derived that will not result in jeopardy to any listed species.

Because this opinion concludes the proposed action is likely to jeopardize the continued existence of the threatened loggerhead sea turtle and no RPA has been identified, no incidental take statement is provided (ESA § 7(b)(4)(A). The USACE is required to notify NMFS of its final decision on this action [50 CFR 402.15(b)]. Application for an exemption from the ESA 7(a)(2) requirement may be sought, pursuant to ESA § 7(g), following the procedures in 50 CFR 451.

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APPENDIX A

Summary of annual incidental take levels anticipated under the incidental take statements associated with NMFS' existing biological opinions in the action area. Note that while these activities overlap the action area, the takes include the entire range of the activity which often far exceeds the geographical scope of the action area.

Federal Action	**Sea Turtle Species** (numbers represents lethal takes unless otherwise noted)				
	Loggerhead	**Leatherback**	**Green**	**Kemp's Ridley**	**Hawksbill**
Coast Guard Vessel Operation	1 (combined)				
Navy – SE Ops Area[1]	91	17	16	16	4
COE Dredging – S. Atlantic	35	0	7	7	2
Dolphin/Wahoo Fishery	16 (No more than 2 lethal)	16 (No more than 1 lethal)	2 (No more than 1 lethal)	2 (No more than 1 lethal)	2 (No more than 1 lethal)
Monkfish Fishery	6 (No more than 3 lethal)	1	1	1	0
Summer Flounder, Scup, and Black Sea Bass Fishery	15 (No more than 5 lethal)	3	3	3	3
Shrimp Fishery[2]	163,160 (No more than 3,948 lethal)	3,090 (No more than 80 lethal)	155,503 (No more than 4,208 lethal)	18,757 (No more than 514 Lethal)	640[3] (All lethal)
Weakfish Fishery	20	0	0	2	0
Atlantic HMS-Shark Fisheries (**Note: this is 3-year take, not annual**)	679 (No more than 346 lethal)	74 (No more than 47 lethal)	2 (No more than 1 lethal)	2 (No more than 1 lethal)	2 (No more than 1 lethal)
Coastal Migratory Pelagic	33 (No more than 33 lethal)	2 (No more than 2 lethal)	4 (No more than 4 lethal)	14 (No more than 14 lethal)	2 (No more than 2 lethal)
Pamlico Sound Section 10 Permit	38 live, 3 lethal	2 (live or lethal, observed not extrapolated)	120 live, 48 lethal	27 live, 14 lethal	2 (live or lethal, observed not extrapolated)

[1]Total estimated take includes acoustic harassment
[2]Represents estimated take (interactions between sea turtles and trawls). Lethal take in parentheses.
[3]Actual mortalities of hawksbills, as a result of sea turtle/trawl interactions, is expected to be much lower than this number. This number represents the estimated total number of mortalities of hawksbill sea turtles from all sources in areas where shrimp fishing takes place.

APPENDIX B

SEA TURTLE HANDLING AND RESUSCITATION GUIDELINES

Any sea turtles taken incidentally during the course of fishing or scientific research activities must be handled with due care to prevent injury to live specimens, observed for activity, and returned to the water according to the following procedures:

A) Sea turtles that are actively moving or determined to be dead (as described in paragraph (B)(4) below) must be released over the stern of the boat. In addition, they must be released only when fishing or scientific collection gear is not in use, when the engine gears are in neutral position, and in areas where they are unlikely to be recaptured or injured by vessels.

B) Resuscitation must be attempted on sea turtles that are comatose or inactive by:

1) Placing the turtle on its bottom shell (plastron) so that the turtle is right side up and elevating its hindquarters at least 6 inches (15.2 cm) for a period of 4 to 24 hours. The amount of elevation depends on the size of the turtle; greater elevations are needed for larger turtles. Periodically, rock the turtle gently left to right and right to left by holding the outer edge of the shell (carapace) and lifting one side about 3 inches (7.6 cm) then alternate to the other side. Gently touch the eye and pinch the tail (reflex test) periodically to see if there is a response.

2) Sea turtles being resuscitated must be shaded and kept damp or moist but under no circumstance be placed into a container holding water. A water-soaked towel placed over the head, carapace, and flippers is the most effective method in keeping a turtle moist.

3) Sea turtles that revive and become active must be released over the stern of the boat only when fishing or scientific collection gear is not in use, when the engine gears are in neutral position, and in areas where they are unlikely to be recaptured or injured by vessels. Sea turtles that fail to respond to the reflex test or fail to move within 4 hours (up to 24, if possible) must be returned to the water in the same manner as that for actively moving turtles.

4) A turtle is determined to be dead if the muscles are stiff (rigor mortis) and/or the flesh has begun to rot; otherwise, the turtle is determined to be comatose or inactive and resuscitation attempts are necessary.

Any sea turtle so taken must not be consumed, sold, landed, offloaded, transshipped, or kept below deck.

These requirements are excerpted from 50 CFR 223.206(d)(1). Failure to follow these procedures is therefore a punishable offense under the Endangered Species Act.