Biological Opinion

Folly Beach Renourishment and Groin Rehabilitation Project

FWS Log #: 04ES1000-2017-F-0746-R001/04ES1000-2018-F-0273

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January 19, 2018

Date

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CONSULTATION HISTORY

This section lists key events and correspondence during the course of this consultation. A complete administrative record of this consultation is on file in the U.S. Fish and Wildlife Service's (Service) South Carolina Field Office (SCFO).

2017-06-13 – The Service received the U.S. Army Corps of Engineers' (Corps) and South Carolina Department of Health and Environmental Control – Office of Ocean and Coastal Resource Management's joint public notice for the City of Folly Beach's (City) beach nourishment and groin rehabilitation project on Folly Beach from 8th Street to 14th Street East.

2017-06-28 – The Service provided comments to the Corps regarding the proposed project. The Service stated that formal consultation needed to be initiated and a Biological Assessment (BA) needed to be provided.

2017-07-03 – The Service received a letter from the Corps requesting to initiate formal consultation and the BA for the proposed project.

2017-07-20 – The Service sent a letter to the Corps acknowledging receipt of all information necessary to initiate the consultation.

2017-11-01 – The Service issued its biological opinion to the Corps.

2017-11-02 – Ms. Melissa Chaplin of the Service received an email from Ms. Bethney Ward of the Corps regarding the Folly Beach Shore Protection Project. Funding became available for the Corps' Planning Division to do the beach renourishment portion of the proposed project.

2017-12-01 – The Service received a letter from the Corps' Planning Division requesting to initiate formal consultation and the BA for the proposed project.

2017-12-21 – The Service sent a letter to the Corps acknowledging receipt of all information necessary to initiate the consultation.

2018-01-11 – Ms. Melissa Chaplin of the Service attended a site visit on Bird Key Stono with Ms. Bethney Ward and Mr. Alan Shirey of the Corps and Ms. Felicia Sanders of the South Carolina Department of Natural Resources (SCDNR).

BIOLOGICAL OPINION

1. INTRODUCTION

A biological opinion (BO) is the document that states the opinion of the Service under the Endangered Species Act of 1973, as amended (ESA), as to whether a Federal action is likely to:

- Jeopardize the continued existence of species listed as endangered or threatened; or
- Result in the destruction or adverse modification of designated critical habitat.

The Federal action addressed in this BO is a Corps permit for the City of Folly Beach's proposed Folly Beach Renourishment and Groin Rehabilitation Project (the Action). This BO considers the effects of the Action on the loggerhead sea turtle (*Caretta caretta*) and its designated critical habitat, piping plover (*Charadrius melodus*) and its critical habitat, and red knot (*Calidris canutus rufa*).

The Corps determined that the Action is not likely to adversely affect the green sea turtle (*Chelonia mydas*), the leatherback sea turtle (*Dermochelys coriacea*), the West Indian manatee (*Trichechus manatus*), and the wood stork (*Mycteria americana*). The Service concurs with these determinations, for reasons we explain in section 2 of the BO.

A BO evaluates the effects of a Federal action along with those resulting from interrelated and interdependent actions, and from non-Federal actions unrelated to the proposed Action (cumulative effects), relative to the status of listed species and the status of designated critical habitat. A Service opinion that concludes a proposed Federal action is *not* likely to jeopardize species and is *not* likely to destroy or adversely modify critical habitat fulfills the Federal agency's responsibilities under §7(a)(2) of the ESA.

"*Jeopardize the continued existence*" means to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species (50 CFR §402.02). "*Destruction or adverse modification*" means a direct or indirect alteration that appreciably diminishes the value of designated critical habitat for the conservation of a listed species. Such alterations may include, but are not limited to, those that alter the physical or biological features essential to the conservation of a species or that preclude or significantly delay development of such features (50 CFR §402.02).

This BO uses hierarchical numeric section headings. Primary (level-1) sections are labeled sequentially with a single digit (e.g., 2. PROPOSED ACTION). Secondary (level-2) sections within each primary section are labeled with two digits (e.g., 2.1. Action Area), and so on for level-3 sections. The basis of our opinion for each listed species and each designated critical habitat identified in the first paragraph of this introduction is wholly contained in a separate level-1 section that addresses its status, environmental baseline, effects of the Action, cumulative effects, and conclusion.

2. PROPOSED ACTION

Folly Beach

The proposed project includes: (1) The Corps placing 755,000 cubic yards (cy) of beach quality sand along 13,000 linear feet of shoreline from 8^{th} Street East to the last groin past the last structure on the east end of the island; (2) The City placing 3,470 cy of rock, 1,160 cy of concrete, and 778 cy of marine mattresses to repair nine existing groins between 8th Street and 14th Street East; and (3) The Corps placing an additional 200,000 cy to create a wider protective berm if funding is available (**Figures 1 and 2**). The proposed project will take place in conjunction with the upcoming Corps' Folly River navigation channel dredging project, which involves dredging the channel and using the beach quality sand from the project for beach renourishment via a hydraulic dredge. Construction is expected to take nine months to complete. The construction window is anticipated to extend from March through September.

Bird Key Stono

The proposed project includes the Corps placing up to 40,000 cy of beach quality sand from the borrow area on Bird Key Stono in a six acre area on the northeast edge of the island (**Figure 3**). The sand will be placed above the high tide line to minimize impacts to intertidal benthic invertebrates. Construction is expected to take up to one week to complete. The timing of the sand placement will be coordinated with the Service and the South Carolina Department of Natural Resources (SCDNR) and will occur prior to mid-March to minimize impacts to nesting shorebirds and seabirds and migratory nonbreeding piping plovers and red knots.

2.1. Action Area

For purposes of consultation under ESA §7, the action area is defined as "all areas to be affected directly or indirectly by the Federal action and not merely the immediate area involved in the action" (50 CFR §402.02). The "Action Area" for this consultation includes the entire shoreline of Folly Beach and Bird Key Stono (**Figures 1, 2, and 3**).

Figure 1. Location of the beach renourishment (Corps 2017).

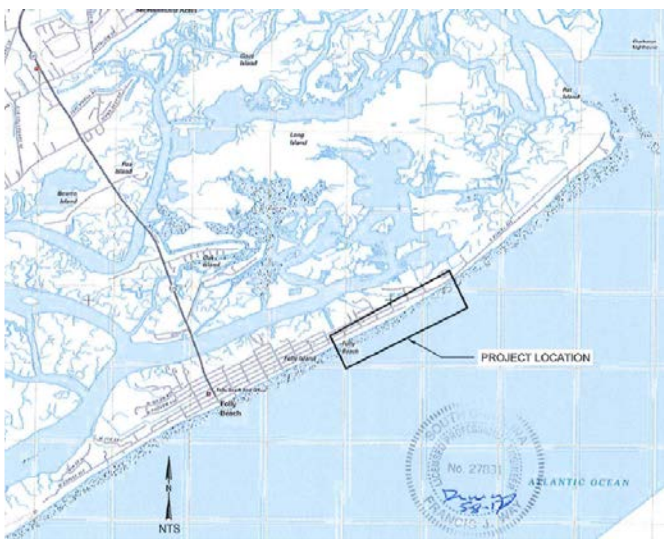

Figure 2. Location of the groin construction (ATM 2017).

Figure 3. Location of sand placement on Bird Key Stono (Corps 2017).

2.2. Renourishment

The renourishment portion of the proposed Action consists of the beneficial use of dredged material from the Corps' navigation maintenance dredging of the Folly River Federal Navigation Channel (**Figure 4**). The Corps plans to pump up to 955,000 cy of the dredged material via hydraulic dredge between 8th Street and 14th Street East to the last groin past the last structure on the east end of the island and up to 40,000 cy on the northeast edge of Bird Key Stono above the high tide line. The renourishment portion of the Action is anticipated to take up to nine months to complete operating 24 hours a day. Heavy equipment and pipeline will be stored on the beach and used to complete the Action.

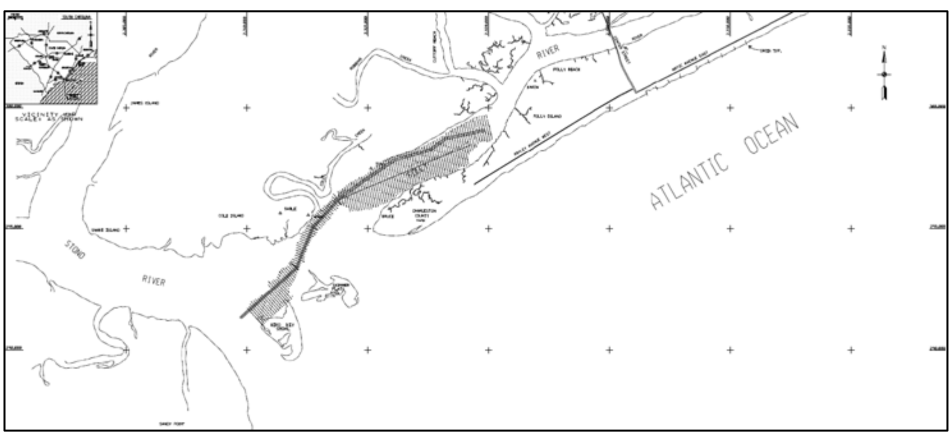

Figure 4. Location of borrow area for the proposed action (ATM 2017).

2.3. Groin Rehabilitation

The groin rehabilitation portion of the proposed Action consists of restoring the original functionality of nine groins. The existing groins will not be extended or expanded. Additional armor stone will be added to the existing rock that can be salvaged from the groins and re-stacked into an engineered section along the entire length of the structure footprint. In cases where all the rock structure is gone, marine mattresses will be added to form a base for the new armor stone and to minimize scour and settling. The rehabilitation materials will consist of rock, marine mattresses, geogrid composite, small stone, poured concrete, and grout. Construction materials for the groins will be delivered to the site via barge or trucks. Heavy equipment will be used to rehabilitate the groins and each of the nine groins is anticipated to take up to one month to complete. Since the rehabilitation of all nine groins will take up to nine months, the work will likely coincide with the loggerhead sea turtle nesting season.

2.4. Interrelated and Interdependent Actions

A BO evaluates the effects of a proposed Federal action. For purposes of consultation under ESA §7, the effects of a Federal action on listed species or critical habitat include the direct and indirect effects of the action, and the effects of interrelated or interdependent actions. "Indirect effects are those that are caused by the proposed action and are later in time, but still are reasonably certain to occur. Interrelated actions are those that are part of a larger action and depend on the larger action for their justification. Interdependent actions are those that have no independent utility apart from the action under consideration" (50 CFR §402.02).

In its request for consultation, the Corps did not describe, and the Service is not aware of, any interrelated or interdependent actions to the Action. Therefore, this BO does not further address the topic of interrelated or interdependent actions.

3. CONCURRENCE

The Corps determined that the Action is not likely to adversely affect the green sea turtle (*Chelonia mydas*), the leatherback sea turtle (*Dermochelys coriacea*), the West Indian manatee (*Trichechus manatus*), and the wood stork (*Mycteria americana*). The Service concurs with these determinations, for reasons we explain in this section (**Table 1**).

Table 1. Species Evaluated for Effects from the Action.

SPECIES OR CRITICAL HABITAT	PRESENT IN ACTION AREA	PRESENT IN ACTION AREA BUT "NOT LIKELY TO BE ADVERSELY AFFECTED" BASED ON
Green sea turtle	Possible, but rare. No green sea turtle nests have been documented on Folly Beach to date.	Protection measures in place for the loggerhead sea turtle
Leatherback sea turtle	Yes, but rare. Leatherback sea turtle nests were documented on Folly Beach in 2003, 2010, and 2012.	Protection measures in place for the loggerhead sea turtle
West Indian manatee	Possible if water temperatures are >68°F.	Implementation of Standard Manatee Construction Conditions (**Appendix A**)
Wood stork	Possible in shallow water along the edges of the Folly River.	Limited foraging habitat adjacent to dredge operations.

This concurrence concludes consultation for the listed species named in this section, and these are not further addressed in this BO. The circumstances described in the Reinitiation Notice of this BO that require reinitiating consultation for the Action, except for exceeding the amount or extent of incidental take, also apply to these species and critical habitats.

4. LOGGERHEAD SEA TURTLE

The Service and National Marine Fisheries Service (NMFS) share Federal jurisdiction for sea turtles under the ESA. The Service has responsibility for sea turtles on the nesting beach. The NMFS has jurisdiction for sea turtles in the marine environment. In accordance with the ESA, the Service completes consultations with all Federal agencies for actions that may adversely affect sea turtles on the nesting beach. The Service's analysis only addresses activities that may impact nesting sea turtles, their nests and eggs, and hatchlings as they emerge from the nest and crawl to the sea. NMFS assesses and consults with Federal agencies concerning potential impacts to sea turtles in the marine environment, including updrift and downdrift nearshore areas affected by sand placement projects on the beach.

4.1. Status of Species

This section summarizes best available data about the biology and current condition of the loggerhead sea turtle throughout its range that are relevant to formulating an opinion about the Action. The Service published its decision to list the loggerhead sea turtle as threatened on July 28, 1978 (43 FR 32800). On September 22, 2011, the loggerhead sea turtle's listing under the

ESA was revised from a single threatened species to nine distinct population segments (DPS) listed as either threatened or endangered. The nine DPSs and their statuses are:

Northwest Atlantic Ocean DPS – threatened
Northeast Atlantic Ocean – endangered
Mediterranean Sea DPS – endangered
South Atlantic Ocean DPS – threatened
North Pacific Ocean DPS – endangered
South Pacific Ocean DPS – endangered
North Indian Ocean DPS – endangered
Southwest Indian Ocean – threatened
Southeast Indo-Pacific Ocean DPS – threatened

4.1.1. Species Description

The loggerhead sea turtle grows to an average weight of about 200 pounds and is characterized by a large head with blunt jaws. Adults and subadults have a reddish-brown carapace. Scales on the top of the head and top of the flippers are also reddish-brown with yellow on the borders. Hatchlings are a dull brown color (NMFS 2009). The loggerhead feeds on mollusks, crustaceans, fish, and other marine animals. The loggerhead may be found hundreds of miles out to sea, as well as in inshore areas such as bays, lagoons, salt marshes, creeks, ship channels, and the mouths of large rivers. Coral reefs, rocky places, and shipwrecks are often used as feeding areas. Within the Northwest Atlantic, the majority of nesting activity occurs from April through September, with a peak in June and July (Williams-Walls *et al.* 1983, Dodd 1988, Weishampel *et al.* 2006). Nesting occurs within the Northwest Atlantic along the coasts of North America, Central America, northern South America, the Antilles, Bahamas, and Bermuda, but is concentrated in the southeastern U.S. and on the Yucatán Peninsula in Mexico on open beaches or along narrow bays having suitable sand (Sternberg 1981, Ehrhart 1989, Ehrhart *et al.* 2003, NMFS and USFWS 2008).

4.1.2. Life History

Loggerheads are long-lived, slow-growing animals that use multiple habitats across entire ocean basins throughout their life history. This complex life history encompasses terrestrial, nearshore, and open ocean habitats. The three basic ecosystems in which loggerheads live are the:

1. Terrestrial zone (supralittoral) - the nesting beach where oviposition (egg laying) and embryonic development and hatching occur.

2. Neritic zone - the inshore marine environment (from the surface to the sea floor) where water depths do not exceed 656 feet. The neritic zone generally includes the continental shelf, but in areas where the continental shelf is very narrow or nonexistent, the neritic zone conventionally extends to areas where water depths are less than 656 feet.

3. Oceanic zone - the vast open ocean environment (from the surface to the sea floor) where

water depths are greater than 656 feet.

Maximum intrinsic growth rates of sea turtles are limited by the extremely long duration of the juvenile stage and fecundity. Loggerheads require high survival rates in the juvenile and adult stages, which are common constraints critical to maintaining long-lived, slow-growing species, to achieve positive or stable long-term population growth (Congdon *et al.* 1993, Heppell 1998, Crouse 1999, Heppell *et al.* 1999, 2003, Musick 1999).

The generalized life history of Atlantic loggerheads is shown in **Figure 5** (from Bolten 2003).

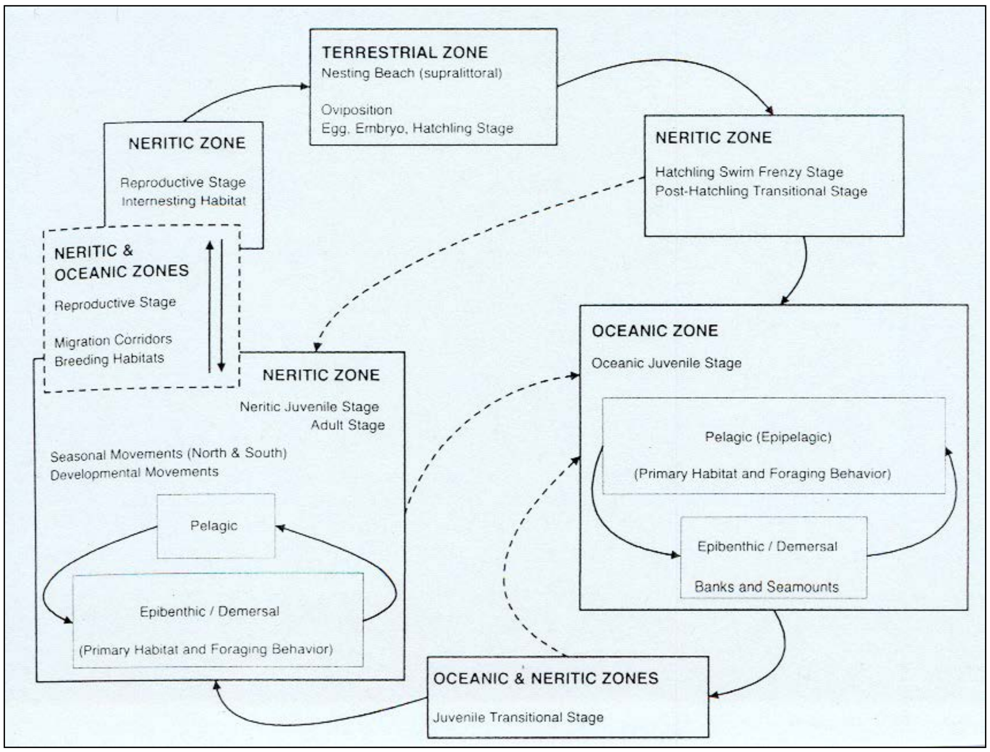

Figure 5. Life history stages of a loggerhead turtle. The boxes represent life stages and the corresponding ecosystems, solid lines represent movements between life stages and ecosystems, and dotted lines are speculative (Bolten 2003).

Numbers of nests and nesting females are often highly variable from year to year due to a number of factors including environmental stochasticity, periodicity in ocean conditions, anthropogenic effects, and density-dependent and density-independent factors affecting survival, somatic growth, and reproduction (Meylan 1982, Hays 2000, Chaloupka 2001, Solow *et al.* 2002). Despite these sources of variation, and because female turtles exhibit strong nest site fidelity, a nesting beach survey can provide a valuable assessment of changes in the adult female population, provided that the study is sufficiently long and effort and methods are standardized (Meylan 1982, Gerrodette and Brandon 2000, Reina *et al.* 2002). **Table 2** summarizes key life history characteristics for loggerheads nesting in the U.S.

Table 2. Typical values of life history parameters for loggerheads nesting in the U.S. (NMFS and Service 2008).

Life History Trait	**Data**
Clutch size (mean)	100-126 eggs[1]
Incubation duration (varies depending on time of year and latitude)	Range = 42-75 days[2,3]
Pivotal temperature (incubation temperature that produces an equal number of males and females)	84˚F[5]
Nest productivity (emerged hatchlings/total eggs) x 100 (varies depending on site specific factors)	45-70 percent[2,6]
Clutch frequency (number of nests/female/season)	3-4 nests[7]
Internesting interval (number of days between successive nests within a season)	12-15 days[8]
Juvenile (<34 inches Curved Carapace Length) sex ratio	65-70 percent female[4]
Remigration interval (number of years between successive nesting migrations)	2.5-3.7 years[9]
Nesting season	late April-early September
Hatching season	late June-early November
Age at sexual maturity	32-35 years[10]
Life span	>57 years[11]

1 Dodd (1988).
2 Dodd and Mackinnon (1999, 2000, 2001, 2002, 2003, 2004).
3 Witherington (2006) (information based on nests monitored throughout Florida beaches in 2005, n = 865).
4 NMFS (2001); Foley (2005).
5 Mrosovsky (1988).
6 Witherington (2006) (information based on nests monitored throughout Florida beaches in 2005, n = 1,680).
7 Murphy and Hopkins (1984); Frazer and Richardson (1985); Hawkes *et al.* 2005; Scott 2006.
8 Dodd (1988).
9 Richardson *et al.* (1978); Bjorndal *et al.* (1983).
10 Snover (2005).
11 Dahlen *et al.* (2000).

Loggerheads nest on ocean beaches and occasionally on estuarine shorelines with suitable sand. Nests are typically laid between the high tide line and the dune front (Routa 1968, Witherington 1986, Hailman and Elowson 1992). Wood and Bjorndal (2000) evaluated four environmental factors (slope, temperature, moisture, and salinity) and found that slope had the greatest influence on loggerhead nest-site selection on a beach in Florida. Loggerheads appear to prefer relatively narrow, steeply sloped, coarse-grained beaches, although nearshore contours may also play a role in nesting beach site selection (Provancha and Ehrhart 1987).

The warmer the sand surrounding the egg chamber, the faster the embryos develop (Mrosovsky and Yntema 1980). Sand temperatures prevailing during the middle third of the incubation period also determine the sex of hatchling sea turtles (Mrosovsky and Yntema 1980). Incubation temperatures near the upper end of the tolerable range produce only female hatchlings while incubation temperatures near the lower end of the tolerable range produce only male hatchlings.

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

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

Loggerheads in the Northwest Atlantic display complex population structure based on life history stages. Based on mitochondrial deoxyribonucleic acid (mtDNA), oceanic juveniles show no structure, neritic juveniles show moderate structure, and nesting colonies show strong structure (Bowen *et al.* 2005). In contrast, a survey using microsatellite (nuclear) markers showed no significant population structure among nesting populations (Bowen *et al.* 2005), indicating that while females exhibit strong philopatry, males may provide an avenue of gene flow between nesting colonies in this region.

4.1.3. Numbers, Reproduction, and Distribution

The loggerhead occurs throughout the temperate and tropical regions of the Atlantic, Pacific, and Indian Oceans (Dodd 1988). However, the majority of loggerhead nesting is at the western rims of the Atlantic and Indian Oceans. The most recent reviews show that only two loggerhead nesting beaches have greater than 10,000 females nesting per year (Baldwin *et al.* 2003, Ehrhart *et al.* 2003, Kamezaki *et al.* 2003, Limpus and Limpus 2003, Margaritoulis *et al.* 2003): Peninsular Florida (U.S.) and Masirah (Oman). Those beaches with 1,000 to 9,999 females nesting each year are Georgia through North Carolina (U.S.), Quintana Roo and Yucatán (Mexico), Cape Verde Islands (Cape Verde, eastern Atlantic off Africa), and Western Australia (Australia). Smaller nesting aggregations with 100 to 999 nesting females annually occur in the Northern Gulf of Mexico (U.S.), Dry Tortugas (U.S.), Cay Sal Bank (Bahamas), Sergipe and Northern Bahia (Brazil), Southern Bahia to Rio de Janerio (Brazil), Tongaland (South Africa), Mozambique, Arabian Sea Coast (Oman), Halaniyat Islands (Oman), Cyprus, Peloponnesus (Greece), Island of Zakynthos (Greece), Turkey, Queensland (Australia), and Japan.

The loggerhead is commonly found throughout the North Atlantic including the Gulf of Mexico, the northern Caribbean, the Bahamas archipelago, and eastward to West Africa, the western Mediterranean, and the west coast of Europe.

The major nesting concentrations in the U.S. are found in South Florida. However, loggerheads nest from Texas to Virginia. Total estimated nesting in the U.S. has fluctuated between 49,000 and 90,000 nests per year from 1999-2010 (NMFS and Service 2008, FWC/FWRI 2010). About 80 percent of loggerhead nesting in the southeast U.S. occurs in six Florida counties (Brevard, Indian River, St. Lucie, Martin, Palm Beach, and Broward Counties). Adult loggerheads are known to make considerable migrations between foraging areas and nesting beaches (Schroeder *et al.* 2003, Foley *et al.* 2008). During non-nesting years, adult females from U.S. beaches are distributed in waters off the eastern U.S. and throughout the Gulf of Mexico, Bahamas, Greater Antilles, and Yucatán.

From a global perspective, the U.S. nesting aggregation is of paramount importance to the survival of the species as is the population that nests on islands in the Arabian Sea off Oman (Ross 1982, Ehrhart 1989, Baldwin *et al.* 2003). Based on standardized daily surveys of the highest nesting beaches and weekly surveys on all remaining island nesting beaches, approximately 50,000, 67,600, and 62,400 nests, were estimated in 2008, 2009, and 2010, respectively (Conant *et al.* 2009). The status of the Oman loggerhead nesting population, reported to be the largest in the world (Ross 1979), is uncertain because of the lack of long-term standardized nesting or foraging ground surveys and its vulnerability to increasing development pressures near major nesting beaches and threats from fisheries interaction on foraging grounds and migration routes (Possardt 2005). The loggerhead nesting aggregations in Oman and the U.S. account for the majority of nesting worldwide.

Distribution

Five recovery units have been identified in the Northwest Atlantic based on genetic differences and a combination of geographic distribution of nesting densities, geographic separation, and geopolitical boundaries (NMFS and USFWS 2008). Recovery units are subunits of a listed species that are geographically or otherwise identifiable and essential to the recovery of the species. Recovery units are individually necessary to conserve genetic robustness, demographic robustness, important life history stages, or some other feature necessary for long-term sustainability of the species. The five recovery units identified in the Northwest Atlantic are:

1. Northern Recovery Unit (NRU) - defined as loggerheads originating from nesting beaches from the Florida-Georgia border through southern Virginia (the northern extent of the nesting range);

2. Peninsula Florida Recovery Unit (PFRU) - defined as loggerheads originating from nesting beaches from the Florida-Georgia border through Pinellas County on the west coast of Florida, excluding the islands west of Key West, Florida;

3. Dry Tortugas Recovery Unit (DTRU) - defined as loggerheads originating from nesting beaches throughout the islands located west of Key West, Florida;

4. Northern Gulf of Mexico Recovery Unit (NGMRU) - defined as loggerheads originating from nesting beaches from Franklin County on the northwest Gulf coast of Florida through Texas; and

5. Greater Caribbean Recovery Unit (GCRU) - composed of loggerheads originating from all other nesting assemblages within the Greater Caribbean (Mexico through French Guiana, The Bahamas, Lesser Antilles, and Greater Antilles).

The mtDNA analyses show that there is limited exchange of females among these recovery units (Ehrhart 1989, Foote *et al.* 2000, NMFS 2001, Hawkes *et al.* 2005). Based on the number of haplotypes, the highest level of loggerhead mtDNA genetic diversity in the Northwest Atlantic has been observed in females of the GCRU that nest at Quintana Roo, Mexico (Encalada *et al.* 1999, Nielsen 2010).

Nuclear DNA analyses show that there are no substantial subdivisions across the loggerhead nesting colonies in the southeastern U.S. Male-mediated gene flow appears to be keeping the subpopulations genetically similar on a nuclear DNA level (Francisco-Pearce 2001).

Historically, the literature has suggested that the northern U.S. nesting beaches (NRU and NGMRU) produce a relatively high percentage of males and the more southern nesting beaches (PFRU, DTRU, and GCRU) a relatively high percentage of females (e.g., Hanson *et al.* 1998, NMFS 2001, Mrosovsky and Provancha 1989). The NRU and NGMRU were believed to play an important role in providing males to mate with females from the more female-dominated subpopulations to the south. However, in 2002 and 2003, researchers studied loggerhead sex ratios for two of the U.S. nesting subpopulations, the northern and southern subpopulations (NGU and PFRU, respectively) (Blair 2005, Wyneken *et al.* 2005). The study produced interesting results. In 2002, the northern beaches produced more females and the southern beaches produced more males than previously believed. However, the opposite was true in 2003 with the northern beaches producing more males and the southern beaches producing more females in keeping with prior literature. Wyneken *et al.* (2005) speculated that the 2002 result may have been anomalous; however, the study did point out the potential for males to be produced on the southern beaches. Although this study revealed that more males may be produced on southern recovery unit beaches than previously believed, the Service maintains that the NRU and NGMRU play an important role in the production of males to mate with females from the more southern recovery units.

The NRU is the second largest loggerhead recovery unit within the Northwest Atlantic Ocean DPS. Annual nest totals from northern beaches averaged 5,215 nests from 1989-2008, a period of near-complete surveys of NRU nesting beaches, representing approximately 1,272 nesting females per year (4.1 nests per female, Murphy and Hopkins 1984) (NMFS and Service 2008). Since 2008, annual nests totals from NRU beaches have steadily increased with a record high of

11,275 nests in 2016 (www.seaturtle.org). Nesting in Georgia reached a new record in 2016 (3289) but only reached 2144 nests in 2017. South Carolina had the two highest years of nesting on record in 2016 (6,446 nests) and 2017 (5,226 nests). North Carolina had 1622 nests in 2016 and 1193 nests in 2017, which is well above the average of 715. The Georgia, South Carolina, and North Carolina nesting data come from the seaturtle.org Sea Turtle Nest Monitoring System, which is populated with data input by the State agencies. The loggerhead nesting trend from daily beach surveys was declining significantly at 1.3 percent annually from 1983 to 2007 (NMFS and Service 2008). Nest totals from aerial surveys conducted by the SCDNR showed a 1.9 percent annual decline in nesting in South Carolina from 1980-2007. Overall, there is strong statistical data to suggest the NRU experienced a long-term decline in the past (NMFS and USFWS 2008). Currently, however, nesting for the NRU is showing possible signs of stabilizing (76 FR 58868).

The PFRU is the largest loggerhead recovery unit within the Northwest Atlantic Ocean DPS and represents approximately 87 percent of all nesting effort in the DPS (Ehrhart *et al.* 2003). A near-complete nest census of the PFRU undertaken from 1989 to 2007 revealed a mean of 64,513 loggerhead nests per year representing approximately 15,735 females nesting per year (4.1 nests per female, Murphy and Hopkins 1984) (FWC 2008, NMFS and USFWS 2008). This near-complete census provides the best statewide estimate of total abundance, but because of variable survey effort, these numbers cannot be used to assess trends. Loggerhead nesting trends are best assessed using standardized nest counts made at Index Nesting Beach Survey (INBS) sites surveyed with constant effort over time. In 1979, the Statewide Nesting Beach Survey (SNBS) program was initiated to document the total distribution, seasonality, and abundance of sea turtle nesting in Florida. In 1989, the INBS program was initiated in Florida to measure seasonal productivity, allowing comparisons between beaches and between years (FWC 2009). Of the 190 SNBS surveyed areas, 33 participate in the INBS program (representing 30 percent of the SNBS beach length). Using INBS nest counts, a significant declining trend was documented for the Peninsular Florida Recovery Unit, where nesting declined 26 percent over the 20-year period from 1989–2008, and declined 41 percent over the period 1998-2008 (NMFS and USFWS 2008, Witherington *et al.* 2009). However, with the addition of nesting data through 2016, the nesting trend for the PFRU did not show a nesting decline statistically different from zero (76 FR 58868, Brost 2017).

The NGMRU is the third largest nesting assemblage among the four U.S. recovery units. Nesting surveys conducted on approximately 186 miles of beach within the NGMRU (Alabama and Florida only) were undertaken between 1995 and 2007 (statewide surveys in Alabama began in 2002). The mean nest count during this 13-year period was 906 nests per year, which equates to about 221 females nesting per year (4.1 nests per female, Murphy and Hopkins 1984, FWC 2008, NMFS and USFWS 2008). Evaluation of long-term nesting trends for the NGMRU is difficult because of changed and expanded beach coverage. Loggerhead nesting trends are best assessed using standardized nest counts made at INBS sites surveyed with constant effort over time. Using Florida INBS data for the NGMRU (FWC 2008), a log-linear regression showed a significant declining trend of 4.7 percent annually from 1997-2008 (NMFS and USFWS 2008).

The DTRU, located west of the Florida Keys, is the smallest of the identified recovery units. A near-complete nest census of the DTRU was undertaken from 1995 to 2004, excluding 2002, (9 years surveyed) revealed a mean of 246 nests per year, which equates to about 60 females nesting per year (4.1 nests per female, Murphy and Hopkins 1984, FWC 2008, NMFS and USFWS 2008). The nesting trend data for the DTRU are from beaches that are not part of the INBS program, but are part of the SNBS program. A simple linear regression of 1995-2004 nesting data, accounting for temporal autocorrelation, revealed no trend in nesting numbers. Because of the annual variability in nest totals, it was determined that a longer time series is needed to detect a trend (NMFS and USFWS 2008).

The GCRU is composed of all other nesting assemblages of loggerheads within the Greater Caribbean and is the third largest recovery unit within the Northwest Atlantic Ocean DPS, with the majority of nesting at Quintana Roo, Mexico. Statistically valid analyses of long-term nesting trends for the entire GCRU are not available because there are few long-term standardized nesting surveys representative of the region. Additionally, changing survey effort at monitored beaches and scattered and low-level nesting by loggerheads at many locations currently precludes comprehensive analyses. The most complete data are from Quintana Roo and Yucatán, Mexico, where an increasing trend was reported over a 15-year period from 1987-2001 (Zurita *et al.* 2003). However, TEWG (2009) reported a greater than 5 percent annual decline in loggerhead nesting from 1995-2006 at Quintana Roo.

4.1.4. Conservation Needs and Threats

Conservation needs: Recovery Criteria (only the Demographic Recovery Criteria are presented below; for the Listing Factor Recovery Criteria, see NMFS and USFWS 2008)

1. Number of Nests and Number of Nesting Females
   a. NRU
      i. There is statistical confidence (95 percent) that the annual rate of increase over a generation time of 50 years is 2 percent or greater resulting in a total annual number of nests of 14,000 or greater for this recovery unit (approximate distribution of nests is North Carolina =14 percent [2,000 nests], South Carolina =66 percent [9,200 nests], and Georgia =20 percent [2,800 nests]); and
      ii. 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).

   b. PFRU
      i. There is statistical confidence (95 percent) that the annual rate of increase over a generation time of 50 years is statistically detectable (one percent) resulting in a total annual number of nests of 106,100 or greater for this recovery unit; and
      ii. 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).

c. DTRU
   i. There is statistical confidence (95 percent) that the annual rate of increase over a generation time of 50 years is three percent or greater resulting in a total annual number of nests of 1,100 or greater for this recovery unit; and
   ii. 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).

d. NGMRU
   i. There is statistical confidence (95 percent) that the annual rate of increase over a generation time of 50 years is three percent or greater resulting in a total annual number of nests of 4,000 or greater for this recovery unit (approximate distribution of nests (2002-2007) is Florida= 92 percent [3,700 nests] and Alabama =8 percent [300 nests]); and
   ii. 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)

e. GCRU
   i. The total annual number of nests at a minimum of three nesting assemblages, averaging greater than 100 nests annually (e.g., Yucatán, Mexico; Cay Sal Bank, Bahamas) has increased over a generation time of 50 years; and
   ii. 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).

2. Trends in Abundance on Foraging Grounds
   A network of in-water sites, both oceanic and neritic across the foraging range is established and monitoring is implemented to measure abundance. There is statistical confidence (95 percent) that a composite estimate of relative abundance from these sites is increasing for at least one generation.

3. Trends in Neritic Strandings Relative to In-water Abundance
   1. Stranding trends are not increasing at a rate greater than the trends in in-water relative abundance for similar age classes for at least one generation.

Threats

Anthropogenic (human) factors that impact hatchlings and adult female turtles on land, or the success of nesting and hatching include: beach erosion, 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 increased human presence at some nesting beaches or close to nesting beaches has led to secondary threats such as the introduction of exotic fire ants, feral hogs, dogs, and an increased presence of native species (*e.g.*, raccoons, armadillos, and opossums), which raid and feed on turtle eggs. Although sea turtle nesting beaches are protected along large expanses of the western North Atlantic coast, other areas along these coasts have limited or no protection.

Loggerhead turtles are affected by a completely different set of anthropogenic threats in the marine environment. These include oil and gas exploration and transportation; marine pollution; underwater explosions; hopper dredging, offshore artificial lighting; power plant entrainment and/or impingement; entanglement in debris; ingestion of marine debris; marina and dock construction and operation; boat collisions; poaching, and fishery interactions. In the oceanic environment, loggerheads are exposed to a series of longline fisheries that include the U.S. Atlantic tuna and swordfish longline fisheries, an Azorean longline fleet, a Spanish longline fleet, and various fleets in the Mediterranean Sea (Aguilar *et al.* 1995; Bolten *et al.* 1994; Crouse 1999). There is particular concern about the extensive incidental take of juvenile loggerheads in the eastern Atlantic by longline fishing vessels. In the neritic environment in waters off the coastal U.S., loggerheads are exposed to a suite of fisheries in Federal and State waters including trawl, purse seine, hook and line, gillnet, pound net, longline, dredge, and trap fisheries (NMFS and USFWS 2007).

Coastal Development

Loss of nesting habitat related to coastal development has had the greatest impact on nesting sea turtles. Beachfront development not only causes the loss of suitable nesting habitat, but can result in the disruption of powerful coastal processes accelerating erosion and interrupting the natural shoreline migration (National Research Council 1990b). This may in turn cause the need to protect upland structures and infrastructure by armoring, groin placement, beach emergency berm construction and repair, and beach nourishment, all of which cause changes in, additional loss of, or impact to the remaining sea turtle habitat. Rice (2012a) identified that approximately 856 miles (40%) of sandy beaches from North Carolina to Texas have been developed (**Table 3**).

Table 3. The lengths and percentages of sandy oceanfront beach in each state in the Southeastern U.S. that are developed, undeveloped, and preserved (Rice 2012b).

State	Approximate Shoreline Beach Length (miles)	Approximate Miles of Beach Developed (percent of total shoreline length)	Approximate Miles of Beach Undeveloped (percent of total shoreline length)[a]	Approximate Miles of Beach Preserved (percent of total shoreline length)[b]
NC	326	159 (49%)	167 (51%)	178.7 (55%)

SC	182	93 (51%)	89 (49%)	84 (46%)
GA	90	15 (17%)	75 (83%)	68.6 (76%)
FL	809	459 (57%)	351 (43%)	297.5 (37%)
-Atlantic	*372*	*236 (63%)*	*136 (37%)*	*132.4 (36%)*
-Gulf	*437*	*223 (51%)*	*215 (49%)*	*168.0. (38%)*
AL	46	25 (55%)	21 (45%)	11.2 (24%)
MS - *barrier island coast*	27	0 (0%)	27 (100%)	27 (100%)
MS - *mainland coast*	51[c]	41 (80%)	10 (20%)	12.6 (25%)
LA	218	13 (6%)	205 (94%)	66.3 (30%)
TX	370	51 (14%)	319 (86%)	152.7 (41%)
TOTAL	2,119	856 (40%)	1,264 (60%)	901.5 (43%)

[a] Beaches classified as "undeveloped" occasionally include a few scattered structures.
[b] Preserved beaches include public ownership, ownership by non-governmental conservation organizations, and conservation easements. The miles of shoreline that have been preserved generally overlap with the miles of undeveloped beach but may also include some areas (e.g., in North Carolina) that have been developed with recreational facilities or by private inholdings.
[c] The mainland Mississippi coast along Mississippi Sound includes 51.3 miles of sandy beach as of 2010-2011, out of approximately 80.7 total shoreline miles (the remaining portion is non-sandy, either marsh or armored coastline with no sand). See Rice 2012b for details.

Hurricanes

Hurricanes were probably responsible for maintaining coastal beach habitat upon which sea turtles depend through repeated cycles of destruction, alteration, and recovery of beach and dune habitat. Hurricanes generally produce damaging winds, storm tides and surges, and rain, which can result in severe erosion of the beach and dune systems. Overwash and blowouts are common on barrier islands. Hurricanes and other storms can result in the direct loss of sea turtle nests, either by erosion or washing away of the nests by wave action and inundation or "drowning" of the eggs or pre-emergent hatchlings within the nest, or indirectly by causing the loss of nesting habitat. Depending on their frequency, storms can affect sea turtles on either a short-term basis (nests lost for one season and/or temporary loss of nesting habitat) or long-term, if frequent (habitat unable to recover). The manner in which hurricanes affect sea turtle nesting also depends on their characteristics (winds, storm surge, rainfall), the time of year (within or outside of the nesting season), and where the northeast edge of the hurricane crosses land.

Because of the limited remaining nesting habitat in a natural state with no immediate development landward of the sandy beach, frequent or successive severe weather events could threaten the ability of certain sea turtle populations to survive and recover. Sea turtles evolved under natural coastal environmental events such as hurricanes. The extensive amount of predevelopment coastal beach and dune habitat allowed sea turtles to survive even the most severe hurricane events. It is only within the last 20 to 30 years that the combination of habitat loss to beachfront development and destruction of remaining habitat by hurricanes has increased the threat to sea turtle survival and recovery. On developed beaches, typically little space remains for sandy beaches to become reestablished after periodic storms. While the beach itself moves landward during such storms, reconstruction or persistence of structures at their pre-storm locations can result in a loss of nesting habitat.

Erosion

A critically eroded area is a segment of shoreline where natural processes or human activity have caused or contributed to erosion and recession of the beach or dune system to such a degree that upland development, recreational interests, wildlife habitat, or important cultural resources are threatened or lost. Critically eroded areas may also include peripheral segments or gaps between identified critically eroded areas because, although they may be stable or slightly erosional now, their inclusion is necessary for continuity of management of the coastal system or for the design integrity of adjacent beach management projects (FDEP 2009). It is important to note that for an erosion problem area to be critical there must be an existing threat to or loss of one of four specific interests – upland development, recreation, wildlife habitat, or important cultural resources.

Beachfront Lighting

Artificial lights along a beach can deter females from coming ashore to nest or misdirect females trying to return to the surf after a nesting event. A significant reduction in sea turtle nesting activity has been documented on beaches illuminated with artificial lights (Witherington 1992). Artificial beachfront lighting may also cause disorientation (loss of bearings) and misorientation (incorrect orientation) of sea turtle hatchlings. Visual signs are the primary sea-finding mechanism for hatchlings (Mrosovsky and Carr 1967, Mrosovsky and Shettleworth 1968, Dickerson and Nelson 1989, Witherington and Bjorndal 1991). Artificial beachfront lighting is a documented cause of hatchling disorientation and misorientation on nesting beaches (Philibosian 1976, Mann 1977, Witherington and Martin 1996). The emergence from the nest and crawl to the sea is one of the most critical periods of a sea turtle's life. Hatchlings that do not make it to the sea quickly become food for ghost crabs, birds, and other predators, or become dehydrated and may never reach the sea. In addition, research has documented significant reduction in sea turtle nesting activity on beaches illuminated with artificial lights (Witherington 1992). During the 2010 sea turtle nesting season in Florida, over 47,000 turtle hatchlings were documented as being disoriented (FWC/FWRI 2011).

Predation

Predation of sea turtle eggs and hatchlings by native and introduced species occurs on almost all nesting beaches. Predation by a variety of predators can considerably decrease sea turtle nest hatching success. The most common predators in the southeastern U.S. are ghost crabs (*Ocypode quadrata*), raccoons (*Procyon lotor*), feral hogs (*Sus scrofa*), foxes (*Urocyon cinereoargenteus* and *Vulpes vulpes*), coyotes (*Canis latrans*), armadillos (*Dasypus novemcinctus*), and fire ants (*Solenopsis invicta*) (Dodd 1988, Stancyk 1995). In the absence of nest protection programs in a number of locations throughout the southeast U.S., raccoons may depredate up to 96 percent of all nests deposited on a beach (Davis and Whiting 1977, Hopkins and Murphy 1980, Stancyk *et al.* 1980, Talbert *et al.* 1980, Schroeder 1981, Labisky *et al.* 1986).

Beach Driving

The operation of motor vehicles on the beach affects sea turtle nesting by interrupting or striking a female turtle on the beach, headlights disorienting or misorienting emergent hatchlings, vehicles running over hatchlings attempting to reach the ocean, and vehicle tracks traversing the beach that interfere with hatchlings crawling to the ocean. Hatchlings appear to become diverted not because they cannot physically climb out of the rut (Hughes and Caine 1994), but because the sides of the track cast a shadow and the hatchlings lose their line of sight to the ocean horizon (Mann 1977). The extended period of travel required to negotiate tire tracks and ruts may increase the susceptibility of hatchlings to dehydration and depredation during migration to the ocean (Hosier *et al.* 1981).

Driving on the beach can cause sand compaction, which may result in adverse impacts on nest site selection, digging behavior, clutch viability, and emergence by hatchlings, decreasing nest success and directly killing pre-emergent hatchlings (Mann 1977, Nelson and Dickerson 1987, Nelson 1988).

Additionally, the physical changes and loss of plant cover caused by vehicles on dunes can lead to various degrees of instability, and therefore encourage dune migration. As vehicles move either up or down a slope, sand is displaced downward, lowering the trail. Since the vehicles also inhibit plant growth, and open the area to wind erosion, dunes may become unstable, and begin to migrate. Unvegetated sand dunes may continue to migrate across stable areas as long as vehicle traffic continues. Vehicular traffic through dune breaches or low dunes on an eroding beach may cause an accelerated rate of overwash and beach erosion (Godfrey *et al.* 1978). If driving is required, the area where the least amount of impact occurs is the beach between the low and high tide water lines. Vegetation on the dunes can quickly reestablish provided the mechanical impact is removed.

Climate Change

The varying and dynamic elements of climate science are inherently long term, complex, and interrelated. Regardless of the underlying causes of climate change, glacial melting and expansion of warming oceans are causing sea level rise, although its extent or rate cannot as yet be predicted with certainty. At present, the science is not exact enough to precisely predict when and where climate impacts will occur. Although we may know the direction of change, it may not be possible to predict its precise timing or magnitude. These impacts may take place gradually or episodically in major leaps.

Climate change has the potential to impact loggerhead sea turtles. Global sea level during the 20th century rose at an estimated rate of about 1.7 millimeters (mm) (0.7 in) per year or an estimated 17 centimeters (cm) (6.7 in) over the entire 100-year period, a rate that is an order of magnitude greater than that seen during the several millennia that followed the end of the last ice age (Bindoff et al. 2007). Global sea level is projected to rise in the 21st century at an even greater rate. Potential impacts to nesting loggerheads include beach erosion from rising sea levels, repeated inundation of nests, skewed hatchling sex ratios from rising incubation temperatures, and abrupt disruption of ocean currents used for natural dispersal during the complex life cycle (Fish et al. 2005, Fish et al. 2008, Hawkes et al. 2010, Poloczanska et al. 2009). Along developed coastlines, and especially in areas where shoreline protection structures have been constructed to limit shoreline movement, rising sea levels will cause severe effects on loggerhead nesting habitat and nesting females and their eggs. 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 intensity of storms and/or changes in prevailing currents, both of which could lead to increased beach loss due to erosion (Meehl and Teng 2007).

Recreational Beach Use

There is increasing popularity in the southeastern U.S., especially in Florida, for beach communities to carry out beach cleaning operations to improve the appearance of beaches for visitors and residents. Beach cleaning occurs on private beaches and on some municipal or county beaches that are used for nesting by loggerhead sea turtles. Beach cleaning activities effectively remove “seaweed, fish, glass, syringes, plastic, cans, cigarettes, shells, stone, wood, and virtually any unwanted debris” (Barber and Sons 2012). Removal of wrack material (organic material that is washed up onto the beach by surf, tides, and wind) reduces the natural sand-trapping abilities of beaches and contributes to their destabilization. As beach cleaning vehicles and equipment move over the sand, sand is displaced downward, lowering the substrate. Although the amount of sand lost due to single sweeping actions may be small, it adds up considerably over a period of years (Neal *et al.* 2007). In addition, since the beach cleaning vehicles and equipment also inhibit plant growth and open the area to wind erosion, the beach and dunes may become unstable. According to Defeo *et al.* 2009, beach cleaning “can result in abnormally broad unvegetated zones that are inhospitable to dune formation or plant colonization, thereby enhancing the likelihood of erosion”. This is also a concern because dunes and vegetation play an important role in minimizing the impacts of artificial beachfront lighting, which causes disorientation of sea turtle hatchlings and nesting turtles, by creating a barrier that prevents residential and commercial business lighting from being visible on the beach.
Human presence on the beach at night during the nesting season can reduce the quality of nesting habitat by deterring or disturbing and causing nesting turtles to avoid otherwise suitable habitat. In addition, human foot traffic can make a beach less suitable for nesting and hatchling emergence by increasing sand compaction and creating obstacles to hatchlings attempting to reach the ocean (Hosier *et al.* 1981).

The use and storage of lounge chairs, cabanas, umbrellas, catamarans, and other types of recreational equipment on the beach at night can also make otherwise suitable nesting habitat unsuitable by hampering or deterring nesting by adult females and trapping or impeding hatchlings during their nest to sea migration. The documentation of non-nesting emergences (also referred to as false crawls) at these obstacles is becoming increasingly common as more recreational beach equipment is left on the beach at night. Sobel (2002) describes nesting turtles being deterred by wooden lounge chairs that prevented access to the upper beach.

Sand Placement

Sand placement projects may result in changes in sand density (compaction), beach shear resistance (hardness), beach moisture content, beach slope, sand color, sand grain size, sand grain shape, and sand grain mineral content if the placed sand is dissimilar from the original beach sand (Nelson and Dickerson 1988a). These changes could result in adverse impacts on nest site selection, digging behavior, clutch viability, and hatchling emergence (Nelson and Dickerson 1987, Nelson 1988).

Beach nourishment projects create an elevated, wider, and unnatural flat slope berm. Sea turtles nest closer to the water the first few years after nourishment because of the altered profile (and perhaps unnatural sediment grain size distribution) (Ernest and Martin 1999, Trindell 2005) Rice (2012a) identified that approximately 32% of sandy shorelines from North Carolina to Texas have been modified by sand placement projects (**Table 4**).

Table 4. Approximate shoreline miles of sandy beach that have been modified by sand placement activities for each state in the Southeastern U.S. These totals are minimum numbers, given missing data for some areas (Rice 2012b).

State	Known Approximate Miles of Beach Receiving Sand	Proportion of Modified Sandy Beach Shoreline
NC	91.3	28%
SC	67.6	37%
GA	5.5	6%
FL - *Atlantic coast*	189.7	51%
FL - *Gulf coast*	189.9	43%
AL	7.5	16%
MS - *barrier island coast*	1.1	4%
MS - *mainland coast*	43.5	85%
LA	60.4	28%
TX	28.3	8%
TOTAL	684.8+	32%

Beach compaction and unnatural beach profiles resulting from beach nourishment activities could negatively impact sea turtles regardless of the timing of projects. Very fine sand or the use of heavy machinery can cause sand compaction on nourished beaches (Nelson *et al.* 1987, Nelson and Dickerson 1988a). Significant reductions in nesting success (*i.e.*, false crawls occurred more frequently) have been documented on severely compacted nourished beaches (Fletemeyer 1980, Raymond 1984, Nelson and Dickerson 1987, Nelson *et al.* 1987), and increased false crawls may result in increased physiological stress to nesting females. Sand compaction may increase the length of time required for female sea turtles to excavate nests and cause increased physiological stress to the animals (Nelson and Dickerson 1988b). Nelson and Dickerson (1988c) concluded that, in general, beaches nourished from offshore borrow sites are harder than natural beaches, and while some may soften over time through erosion and accretion of sand, others may remain hard for 10 years or more.

These impacts can be minimized by using suitable sand and by tilling (minimum depth of 36 inches) compacted sand after project completion. The level of compaction of a beach can be assessed by measuring sand compaction using a cone penetrometer (Nelson 1987). Tilling of a nourished beach with a root rake may reduce the sand compaction to levels comparable to unnourished beaches. However, a pilot study by Nelson and Dickerson (1988c) showed that a tilled nourished beach will remain uncompacted for only up to 1 year. Thus, multi-year beach compaction monitoring and, if necessary, tilling would help to ensure that project impacts on sea turtles are minimized.

A change in sediment color on a beach could change the natural incubation temperatures of nests in an area, which, in turn, could alter natural sex ratios. To provide the most suitable sediment for nesting sea turtles, the color of the nourished sediments should resemble the natural beach sand in the area. Natural reworking of sediments and bleaching from exposure to the sun would help to lighten dark nourishment sediments; however, the timeframe for sediment mixing and bleaching to occur could be critical to a successful sea turtle nesting season.

In-water and Shoreline Alterations

Many navigable mainland or barrier island tidal inlets or beaches along the Atlantic and Gulf of Mexico coasts are stabilized with jetties or groins. Jetties are built perpendicular to the shoreline and extend through the entire nearshore zone and past the breaker zone to prevent or decrease sand deposition in the channel (Kaufman and Pilkey 1979). Groins are also shore-perpendicular structures that are designed to trap sand that would otherwise be transported by longshore currents and can cause downdrift erosion (Kaufman and Pilkey 1979). Rice (2012b) identified over half of inlets from North Carolina to Texas have been modified by some type of structure (**Table 5**).

Table 5. The number of open tidal inlets, inlet modifications, and artificially closed inlets in each state in the Southeastern U.S. (Rice 2012a).

State	Existing Inlets								
	Number of Inlets	Total Number of Modified Inlets	Habitat Modification Type						
			Structures[a]	Dredged	Relocated	Mined	Artificially opened	Artificially closed
NC	20	17 (85%)	7	16	3	4	2	11
SC	47	21 (45%)	17	11	2	3	0	1
GA	23	6 (26%)	5	3	0	1	0	0
FL - *Atlantic*	21	19 (90%)	19	16	0	3	10	0
FL - *Gulf*	48	24 (50%)	20	22	0	6	7	1
AL	4	4 (100%)	4	3	0	0	0	2
MS	6	4 (67%)	0	4	0	0	0	0
LA	34	10 (29%)	7	9	1	2	0	46
TX	18	14 (78%)	10	13	2	1	11	3
TOTAL	221	119 (54%)	89 (40%)	97 (44%)	8 (4%)	20 (9%)	30 (14%)	64 (N/A)

[a] Structures include jetties, terminal groins, groin fields, rock or sandbag revetments, seawalls, and offshore breakwaters.

These in-water structures have profound effects on adjacent beaches (Kaufman and Pilkey 1979). Jetties and groins placed to stabilize a beach or inlet prevent normal sand transport, resulting in accretion of sand on updrift beaches and acceleration of beach erosion downdrift of the structures (Komar 1983, Pilkey *et al.* 1984). Witherington *et al.* (2005) found a significant negative relationship between loggerhead nesting density and distance from the nearest of 17 ocean inlets on the Atlantic coast of Florida. The effect of inlets in lowering nesting density was observed both updrift and downdrift of the inlets, leading researchers to propose that beach instability from both erosion and accretion may discourage loggerhead nesting.

Following construction, the presence of groins and jetties may interfere with nesting turtle access to the beach, result in a change in beach profile and width (downdrift erosion, loss of sandy berms, and escarpment formation), trap hatchlings, and concentrate predatory fishes, resulting in higher probabilities of hatchling predation. In addition to decreasing nesting habitat suitability, construction or repair of groins and jetties during the nesting season may result in the destruction of nests, disturbance of females attempting to nest, and disorientation of emerging hatchlings from project lighting.

4.2. Environmental Baseline

This section is an analysis of the effects of past and ongoing human and natural factors leading to the current status of the loggerhead sea turtle, its habitat, and ecosystem within the Action Area. The environmental baseline is a “snapshot” of the species’ health in the Action Area at the time of the consultation, and does not include the effects of the Action under review.

South Carolina barrier beaches are part of a complex and dynamic coastal system that continually respond to inlets, tides, waves, erosion and deposition, longshore sediment transport, and depletion, fluctuations in sea level, and weather events. The location and shape of the coastline perpetually adjusts to these physical forces. Winds move sediment across the dry beach forming dunes and the island interior landscape. The natural communities contain plants and animals that are subject to shoreline erosion and deposition, salt spray, wind, drought conditions, and sandy soils. Vegetative communities include foredunes, primary, and secondary dunes, interdunal swales, sand pine scrub, and maritime forests. However, the protection or persistence of these important natural land forms, processes, and wildlife resources is often in conflict with long-term beach stabilization projects and their indirect effects, i.e., increases in residential development, infrastructure, and public recreational uses.

South Carolina has approximately 182 miles of coastline and approximately 51% (93/182 miles) of the coastline is developed (SCDHEC 2010). Approximately 37% (67.6/182 miles) of the state's coastline has received sand placement via beach nourishment or dredge disposal placement (Rice 2012a). South Carolina currently has 47 tidal inlets open and 36% (17/47 inlets) have been stabilized with some type of hard structure(s) along at least one shoreline (Rice 2012b).

Folly Beach is a six mile long barrier island located four miles south of the entrance to Charleston Harbor and south of the Charleston Harbor jetties, which have interrupted the natural sediment transport by eliminating sediment bypassing around the harbor's ebb tidal delta causing continuous erosion (City of Folly Beach 2015). Before the jetties were built in the late 1800s, sand that bypassed the Charleston Harbor was transported to Folly Beach. Now the sand either accumulates on the north side of the jetties or is dredged from the Charleston Harbor entrance channel and disposed of offshore. In the 1940s and 1950s, the accelerated erosion resulted in the loss of some beachfront homes and roads (Levine et al. 2009). The South Carolina State Highway Department installed 48 timber and rock groins along the beachfront in the mid-1900s as an erosion control measure (City of Folly Beach 2015). These groins are now dilapidated and are no longer effective for trapping sand. In 1993, the Corps completed the first beach renourishment under the previously authorized Folly Beach Shore Protection Project. The Corps completed three additional projects in 2005, 2007, and 2014. In addition to the ongoing Federal project, the Charleston County Parks and Recreation Commission completed a renourishment of the Folly Beach County Park, which is located on the west end of the island, and constructed a terminal groin in 2013 (CSE 2013).

Bird Key Stono

This undeveloped island is a SCDNR Heritage Preserve located in Stono Inlet between Kiawah Island and Folly Beach. It is managed by SCDNR for seabird and shorebird nesting.

4.2.1. Action Area Numbers, Reproduction, and Distribution

Folly Beach has a nest protection project permitted through SCDNR to conduct daily nesting surveys, nest relocations, predator control measures, and nest inventories. Folly Beach averages 63.9 nests per year based on a 17-year average (**Figure 6**). Loggerhead sea turtle nesting occurs along the entire shoreline. Sea turtles do nest on the Bird Key Stono, but not in large numbers. Any nests on the island are left in situ regardless of nest location because this island does not have a turtle project and it is not monitored daily. Nesting on the island is typically documented by aerial survey or at the time of a bird survey on the island. Beginning in May nesting females come ashore to lay their eggs and will nest multiple times on the same beach or on adjacent beaches through mid-August. Beginning in mid-July, nests will hatch and hatchlings will emerge at night to head to the water through the end of October.

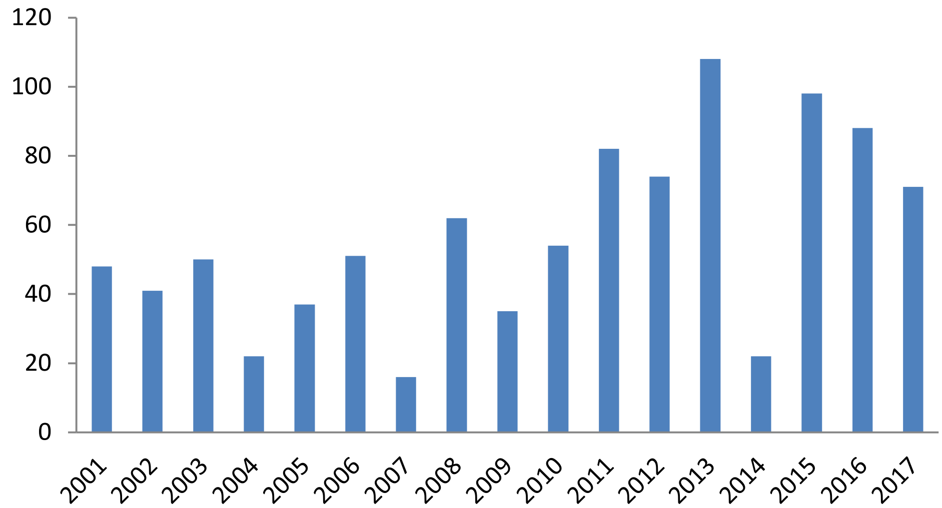

Figure 6. Annual number of loggerhead sea turtle nests on Folly Beach 2001-2017.

4.2.2. Action Area Conservation Needs and Threats

The Action Area is within the NRU. The Action Area is also designated critical habitat (**see Section 5**). Within the Action Area, sea turtle nests are subject to severe erosion, tidal inundation, storms, and predation. Sea turtle hatchlings are subject to disorientations caused by artificial lighting, predation, and entrapment. Nesting sea turtles are subject to disorientations due to artificial lighting and entrapment. The Folly Beach nest protection project volunteers address some of these threats by relocating nests and calling in lighting ordinance violations near nests that are about to hatch. Despite these threats, nesting on Folly Beach follows the increasing trend of nesting within the NRU.

4.3. Effects of the Action

This section analyzes the direct and indirect effects of the Action on the loggerhead sea turtle, which includes the direct and indirect effects of interrelated and interdependent actions. Direct effects are caused by the Action and occur at the same time and place. Indirect effects are caused by the Action, but are later in time and reasonably certain to occur. Our analyses are organized according to the description of the Action in section 2 of this BO.

4.3.1. Effects of Beach Renourishment

Beneficial Effects

The placement of sand on a beach with reduced dry foredune habitat may increase sea turtle nesting habitat if the placed sand is highly compatible (*i.e.*, grain size, shape, color, etc.) with naturally occurring beach sediments in the area, and compaction and escarpment remediation measures are incorporated into the project. In addition, a nourished beach that is designed and constructed to mimic a natural beach system may benefit sea turtles more than an eroding beach it replaces.

Direct Effects

Potential adverse effects during the project construction phase include disturbance of existing nests, which may have been missed by surveyors and thus not marked for avoidance, disturbance of females attempting to nest, and disorientation of emerging hatchlings. In addition, heavy equipment will be required to construct the beach profile. This equipment will have to traverse the beach portion of the Action Area, which could result in harm to nesting sea turtles, their nests, and emerging hatchlings.

1. *Equipment during construction*

The use of heavy machinery on beaches during a construction project may also have adverse effects on sea turtles. Equipment left on the nesting beach overnight can create barriers to nesting females emerging from the surf and crawling up the beach, causing a higher incidence of false crawls and unnecessary energy expenditure.

The operation of motor vehicles or equipment on the beach to complete the project work at night affects sea turtle nesting by: interrupting or colliding with a nesting turtle on the beach, headlights disorienting or misorienting emergent hatchlings, vehicles running over hatchlings attempting to reach the ocean, and vehicle ruts on the beach interfering with hatchlings crawling to the ocean. Apparently, hatchlings become diverted not because they cannot physically climb out of a rut (Hughes and Caine 1994), but because the sides of the track cast a shadow and the hatchlings lose their line of sight to the ocean horizon (Mann 1977). The extended period of travel required to negotiate tire ruts may increase the susceptibility of hatchlings to dehydration and depredation during migration to the ocean (Hosier et al. 1981). Driving directly above or over incubating egg clutches or on the beach can cause sand compaction, which may result in adverse impacts on nest site selection, digging behavior, clutch viability, and emergence by hatchlings, as well as directly kill pre-emergent hatchlings (Mann 1977, Nelson and Dickerson 1987, Nelson 1988).

The physical changes and loss of plant cover caused by vehicles on vegetated areas or dunes can lead to various degrees of instability and cause dune migration. As vehicles move over the sand, sand is displaced downward, lowering the substrate. Since the vehicles also inhibit plant growth, and open the area to wind erosion, the beach and dunes may become unstable. Vehicular traffic on the beach or through dune breaches or low dunes may cause acceleration of overwash and erosion (Godfrey et al. 1978). Driving along the beachfront should be between the low and high tide water lines. To minimize the impacts to the beach, dunes, and dune vegetation, transport and access to the construction sites should be from the road to the maximum extent possible. However, if vehicular access to the beach is necessary, the areas for vehicle and equipment usage should be designated and marked.

2. *Artificial lighting as a result of an unnatural beach slope on the adjacent beach*

Visual cues are the primary sea-finding mechanism for hatchling sea turtles (Mrosovsky and Carr 1967, Mrosovsky and Shettleworth 1968, Dickerson and Nelson 1989, Witherington and Bjorndal 1991). When artificial lighting is present on or near the beach, it can misdirect hatchlings once they emerge from their nests and prevent them from reaching the ocean (Philibosian 1976, Mann 1977, FWC 2007). In addition, a significant reduction in sea turtle nesting activity has been documented on beaches illuminated with artificial lights (Witherington 1992). Therefore, construction lights along a project beach may deter females from coming ashore to nest, misdirect females trying to return to the surf after a nesting event, and misdirect emergent hatchlings from adjacent non-project beaches.

The unnatural sloped beach adjacent to the structure exposes sea turtles and their nests to lights that were less visible, or not visible, from nesting areas before the sand placement activity, leading to a higher mortality of hatchlings. Review of over 10 years of empirical information from beach nourishment projects indicates that the number of sea turtles impacted by lights increases on the post-construction berm. A review of selected nourished beaches in Florida (South Brevard, North Brevard, Captiva Island, Ocean Ridge, Boca Raton, Town of Palm Beach, Longboat Key, and Bonita Beach) indicated disorientation reporting increased by approximately 300 percent the first nesting season after project construction and up to 542 percent the second year compared to pre-nourishment reports (Trindell et al. 2005).

Specific examples of increased lighting disorientations after a sand placement project include Brevard and Palm Beach Counties, Florida. A sand placement project in Brevard County, completed in 2002, showed an increase of 130 percent in disorientations in the nourished area. Disorientations on beaches in the County that were not nourished remained constant (Trindell 2007). This same result was also documented in 2003 when another beach in Brevard County was nourished and the disorientations increased by 480 percent (Trindell 2007). Installing appropriate beachfront lighting is the most effective method to decrease the number of disorientations on any developed beach including nourished beaches. A shoreline protection project was constructed at Ocean Ridge in Palm Beach County, Florida, between August 1997 and April 1998. Lighting disorientation events increased after nourishment. In spite of continued aggressive efforts to identify and correct lighting violations in 1998 and 1999, 86 percent of the disorientation reports were in the nourished area in 1998 and 66 percent of the reports were in the nourished area in 1999 (Howard and Davis 1999).

3. *Missed nests*

Although a nesting survey and nest marking program would reduce the potential for nests to be impacted by construction activities, nests may be inadvertently missed (when crawls are obscured by rainfall, wind, and/or tides) or misidentified as false crawls during daily patrols. Even under the best of conditions, about seven percent of the nests can be misidentified as false crawls by experienced sea turtle nest surveyors (Schroeder 1994).

4. *Nest relocation*

Besides the potential for missing nests during surveys, there is a potential for eggs to be damaged by nest relocation, particularly if eggs are not relocated within 12 hours of deposition (Limpus *et al.* 1979). Relocated nests can incubate at different temperatures than nests left to incubate in place (*in situ*) (Mrosovosky and Yntema 1980, Hoekert *et al.* 1998, Başkale and Kaska 2005, Tuttle 2007, Bimbi 2009, Tuttle and Rostal 2010, Pintus *et al.* 2009) and cause skewed sex ratios (Morreale *et al.* 1982, Godfrey *et al.* 1997). Relocated nests can also have higher or lower hatch success and hatchling emergence than *in situ* nests (Wyneken *et al.* 1988, Hoekert *et al.* 1998, García *et al.* 2003, Moody 2000, Kornaraki *et al.* 2006, Tuttle 2007, McElroy 2009, Pintus *et al.* 2009) depending on relocation technique and environmental conditions.

Nest relocation can have adverse impacts on gas exchange parameters and the hydric environment of nests (Limpus *et al.* 1979, Ackerman 1980, Parmenter 1980, Spotila *et al.* 1983, McGehee 1990). Nests relocated into sands deficient in oxygen or moisture can result in mortality, morbidity, and reduced behavioral competence of hatchlings. Water availability is known to influence the incubation environment of the embryos and hatchlings of turtles with flexible-shelled eggs, which has been shown to affect nitrogen excretion (Packard *et al.* 1984), mobilization of calcium (Packard and Packard 1986), mobilization of yolk nutrients (Packard *et al.* 1985), hatchling size (Packard *et al.* 1981, McGehee 1990), energy reserves in the yolk at hatching (Packard *et al.* 1988), and locomotory ability of hatchlings (Miller *et al.* 1987).

<u>*Indirect Effects*</u>

Many of the direct effects of shoreline stabilization projects may persist over time and become indirect impacts. These indirect effects include increased susceptibility of relocated nests to catastrophic events, the consequences of potential increased beachfront development, changes in the physical characteristics of the beach, and the formation of escarpments.

1. *Increased susceptibility to catastrophic events*

Nest relocation within a nesting season may concentrate eggs in an area making them more susceptible to catastrophic events. Hatchlings released from concentrated areas also may be subject to greater predation rates from both land and marine predators, because the predators learn where to concentrate their efforts (Glenn 1998, Wyneken *et al.* 1998).

2. *Changes in the physical environment*

The use of heavy machinery can cause sand compaction (Nelson et al. 1987, Nelson and Dickerson 1988a). Significant reductions in nesting success (i.e., false crawls occurred more frequently) have been documented on severely compacted beaches (Fletemeyer 1980, Raymond

1984, Nelson and Dickerson 1987, Nelson et al. 1987), and increased false crawls may result in increased physiological stress to nesting females.

3. *Escarpment formation*

Escarpments may develop on beaches between groins as the beaches equilibrate to their final profiles. Escarpments can hamper or prevent access to nesting sites (Nelson and Blihovde 1998) and can cause adult females to choose unsuitable nesting areas, such as seaward of an escarpment. These nest sites commonly receive prolonged tidal inundation and erosion, which results in nest failure.

4.3.2. Effects of Groin Rehabilitation

<u>*Beneficial Effects*</u>

Groins constructed in appropriate high erosion areas, or to offset the effects of shoreline armoring, may reestablish a beach where none currently exists, stabilize the beach in rapidly eroding areas and reduce the potential for escarpment formation, reduce destruction of nests from erosion, and reduce the need for future sand placement events by extending the interval between sand placement events. However, caution should be exercised to avoid automatically assuming the reestablishment of a beach will wholly benefit sea turtle populations without determining the extent of the groin effect on nesting and hatchling sea turtle behavior.

<u>*Direct Effects*</u>

The presence of the groins has the potential to adversely affect sea turtles. For instance, they may interfere with the egress and ingress of adult females at nesting sites; alter downdrift beach profiles through erosion, escarpment formation, and loss of berms; trap or obstruct hatchlings during a critical life-history stage; increase hatchling and adult female energy expenditure in attempts to overcome the structures; and attract additional predatory fish or concentrate existing predatory fish, thereby increasing the potential of hatchling predation.

1. *Equipment during construction* (see **Section 4.3.1**)

2. *Entrapment/physical obstruction*

Groins have the potential to interfere with the egress or ingress of adult females at nesting sites where they may proceed around them successfully, abort nesting for that night, or move to another section of beach to nest. This may cause an increase in energy expenditure, and, if the bodies of the groins are exposed, may act as a barrier between beach segments and also prevent nesting on the adjacent beach. In general, the groins are exposed to dissipate wave energy and facilitate sand bypass, functioning in many cases to stabilize the beach and adjacent areas.

Typically, hatchlings emerge from the nest at night when lower sand temperatures elicit an increase in hatchling activity (Witherington et al. 1990). After emergence, approximately 20 to 120 hatchlings crawl en *masse* immediately to the surf using predominately visual cues to orient them (Witherington and Salmon 1992, Lohmann et al. 1997). Upon reaching the water, sea turtle hatchlings orient themselves into the waves and begin a period of hyperactive swimming activity, or swim frenzy, which lasts for approximately 24 hours (Salmon and Wyneken 1987, Wyneken et al. 1990, Witherington 1991). The swim frenzy effectively moves the hatchling quickly away from shallow, predator rich, nearshore waters to the relative safety of deeper water (Gyuris 1994, Wyneken et al. 2000). The first hour of a hatchling's life is precarious and predation is high, but threats decrease as hatchlings distance themselves from their natal beaches (Stancyk 1995, Pilcher et al. 2000). Delays in hatchling migration (both on the beach and in the water) can cause added expenditures of energy and an increase of time spent in predator rich nearshore waters. On rare occasions hatchlings will encounter natural nearshore features that are similar to the emergent structures proposed for this project. However, observations of hatchling behavior during an encounter with a sand bar at low tide, a natural shore-parallel barrier, showed the hatchlings maintained their shore-perpendicular path seaward, by crawling over the sand bar versus deviating from this path to swim around the sand bar through the trough, an easier alternative. In spite of the design features, the groins may adversely affect sea turtle hatchlings by serving as a barrier or obstruction to sea turtle hatchlings and delaying offshore migration; depleting or increasing expenditure of the "swim frenzy" energy critical for allowing hatchlings to reach the relative safety of offshore development areas; and possibly entrapping hatchlings within the groins or within eddies or other associated currents.

Indirect Effects

Some of the direct effects of groins may persist over time and become indirect impacts. These indirect effects include changes in future sand migration and breakdown of erosion control structures.

1. *Future sand migration and erosion*

Groins and jetties are shore-perpendicular structures that are designed to trap sand that would otherwise be transported by longshore currents. Jetties are defined as structures placed to keep sand from flowing into channels (Kaufman and Pilkey 1979, Komar 1983). In preventing normal sand transport, these structures accrete updrift beaches while causing accelerated beach erosion downdrift of the structures (Komar 1983, Pilkey et al. 1984, National Research Council 1987), a process that results in degradation of sea turtle nesting habitat. As sand fills the area updrift from the groin or jetty, some littoral drift and sand deposition on adjacent downdrift beaches may occur due to spillover. However, these groins and jetties often force the stream of sand into deeper offshore water where it is lost from the system (Kaufman and Pilkey 1979). The greatest changes in beach profile near groins and jetties are observed close to the structures, but effects eventually may extend many miles along the coast (Komar 1983).

Erosion control structures (e.g., terminal groins, T-groins, and breakwaters), in conjunction with beach nourishment, can help stabilize U.S. Gulf and Atlantic coast barrier island beaches (Leonard et al. 1990). However, groins often result in accelerated beach erosion downdrift of the structures (Komar 1983, National Research Council 1987) and corresponding degradation of suitable sea turtle nesting habitat (NMFS and Service 1991, 1992). Initially, the greatest changes are observed close to the structures, but effects may eventually extend significant distances along the coast (Komar 1983).

Groins operate by blocking the natural longshore transport of littoral drift (Kaufman and Pilkey 1979, Komar 1983). Once sand fills the updrift groin area, some littoral drift deposition on adjacent downdrift beaches occurs due to spillover. However, groins often force the river of sand into deeper offshore water, and sand that previously would have been deposited on downdrift beaches is lost from the system (Kaufman and Pilkey 1979). Conventional rubble mound groins control erosion by trapping sand and dissipating some wave energy. In general, except for terminal groins at the downdrift limit of a littoral cell, groins are not considered favorable erosion control alternatives because they usually impart stability to the updrift beach and transfer erosion to the downdrift side of the structure. In addition, groins deflect longshore currents offshore, and excess sand builds up on the updrift side of the structure which may be carried offshore by those currents. This aggravates downdrift erosion and erosion escarpments are common on the downdrift side of groins (Humiston and Moore 2001).

2. *Changes in the physical environment*

The presence of the groins may alter the natural coastal processes and result in an unnatural beach profiles resulting from the presence of groins, which could negatively impact sea turtles regardless of the timing of projects.

3. *Erosion control structure breakdown*

If erosion control structures fail and break apart, they spread debris on the beach, which may further impede nesting females from accessing suitable nesting sites (resulting in a higher incidence of false crawls) and trap hatchlings and nesting turtles (NMFS and Service 1991, 1992, 1993).

4.3.3. Summary of the Effects of the Action

Beach renourishment and groin rehabilitation will occur within loggerhead sea turtle nesting habitat and construction activities will overlap with the nesting season. Potential effects include destruction of nests deposited within the boundaries of the Action, harassment in the form of disturbing or interfering with female turtles attempting to nest within the construction area or on adjacent beaches as a result of construction activities, disorientation of hatchling turtles on beaches adjacent to the construction area as they emerge from the nest and crawl to the water as a result of project lighting or presence of the groins, and behavior modification of nesting females during the nesting season resulting in false crawls or situations where they choose marginal or unsuitable nesting areas to deposit eggs due to escarpment formation or presence of the groins within the Action Area. The rehabilitation of the groins could affect the movement of sand by altering the natural coastal processes and could affect the ability of female turtles to nest, the suitability of the nest incubation environment, and the ability of hatchlings to emerge from the nest and crawl to the ocean.

Some individuals in a population are more "valuable" than others in terms of the number of offspring they are expected to produce. An individual's potential for contributing offspring to future generations is its reproductive value. Because of delayed sexual maturity, reproductive longevity, and low survivorship in early life stages, nesting females are of high value to a population. The loss of a nesting female in a small recovery unit would represent a significant loss to the recovery unit. The reproductive value for a nesting female has been estimated to be approximately 253 times greater than an egg or a hatchling (NMFS and Service 2008).

With regard to indirect loss of eggs and hatchlings, on most beaches, nesting success typically declines for the first year or two following sand placement, even though more nesting habitat is available for turtles (Trindell et al. 1998, Ernest and Martin 1999, Herren 1999). Reduced nesting success on constructed beaches has been attributed to increased sand compaction, escarpment formation, and changes in beach profile (Nelson et al. 1987, Crain et al. 1995, Lutcavage et al. 1997, Steinitz et al. 1998, Ernest and Martin 1999, Rumbold et al. 2001). In addition, even though constructed beaches are wider, nests deposited there may experience higher rates of wash out than those on relatively narrow, steeply sloped beaches (Ernest and Martin 1999). This occurs because nests on constructed beaches are more broadly distributed than those on natural beaches, where they tend to be clustered near the base of the dune. Nests laid closest to the waterline on constructed beaches may be lost during the first year or two following construction as the beach undergoes an equilibration process during which seaward portions of the beach are lost to erosion. As a result, the sand project is anticipated to result in decreased nesting and loss of nests that do get laid within the project area for two subsequent nesting seasons following the completion of the proposed sand placement. However, it is important to note that it is unknown whether nests that would have been laid in the Action Area during the two subsequent nesting seasons had the project not occurred are actually lost from the population or if nesting is simply displaced to adjacent beaches.

During construction, direct mortality of the developing embryos in nests within the project area may occur for nests that are missed and not relocated. The exact number of these missed nests is not known. However, in two separate monitoring programs on the east coast of Florida where hand digging was performed to confirm the presence of nests and thus reduce the chance of missing nests through misinterpretation, trained observers still missed about 6 to 8 percent of the nests because of natural elements (Martin 1992, Ernest and Martin 1993). This must be considered a conservative number, because missed nests are not always accounted for. In another study, Schroeder (1994) found that even under the best of conditions, about 7 percent of nests can be misidentified as false crawls by highly experienced sea turtle nest surveyors. Missed nests are usually identified by signs of hatchling emergences in areas where no nest was previously documented. Signs of hatchling emergence are very easily obliterated by the same elements that interfere with detection of nests.

The presence of the rehabilitated groin may create a physical obstacle to nesting sea turtles. The interaction between the groin and the hydrodynamics of tide and current often results in the alteration of the beach profile seaward and in the immediate vicinity of the structure (Pilkey and Wright 1988, Terchunian 1988, Tait and Griggs 1990, Plant and Griggs 1992), including increased erosion seaward of structures, increased longshore currents that move sand away from the area, loss of interaction between the dune and ocean, and concentration of wave energy at the ends of an armoring structure (Schroeder and Mosier 1996). These changes or combination of changes can have various detrimental effects on sea turtles and their nesting habitat.

Increased erosion downdrift of the rehabilitated groins may take a few years to develop post construction depending on environmental conditions since the compartments within the groin rehabilitation area will be filled over capacity and the area immediately downdrift of the groin will be filled to mitigate downdrift impacts. The downdrift impacts will likely be exacerbated once the rehabilitated groins' trapping capacity is maximized but local conditions no longer allow longshore transport between compartments downdrift. At that time, the impacts can only be mitigated by the placement of more sand, which will largely depend on available funding.

Under these conditions, the rehabilitated groins are anticipated to result in increased false crawls and the relocation of nests laid downdrift of the construction area. However, it is important to note that it is unknown whether nests that would have been laid in the area had the Action not occurred are actually lost from the population or if nesting is simply displaced to other areas of the island or to adjacent beaches. Regardless, eggs and hatchlings have a low reproductive value; each egg or hatchling has been estimated to have only 0.004 percent of the value of a nesting female (NMFS and Service 2008). The Service would not expect this loss to have a significant effect on the recovery and survival of the species, for the following reasons: 1) some nesting is likely just displaced to adjacent non-project beaches, 2) not all eggs will produce hatchlings, and 3) destruction and/or failure of nests will not always result from construction activities. A variety of natural and unknown factors negatively affect incubating egg clutches, including tidal inundation, storm events, and predation.

4.4. Cumulative Effects

For purposes of consultation under ESA §7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under §7 of the ESA. The Service is not aware of any cumulative effects in the Action Area at this time; therefore, cumulative effects are not relevant to formulating our opinion for the Action.

4.5. Conclusion

In this section, the Service summarizes and interprets the findings of the previous sections for the loggerhead sea turtle (status, baseline, effects, and cumulative effects) relative to the purpose of a BO under §7(a)(2) of the ESA, which is to determine whether a Federal action is likely to:

a) Jeopardize the continued existence of species listed as endangered or threatened; or
b) Result in the destruction or adverse modification of designated critical habitat.

"Jeopardize the continued existence" means to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species (50 CFR §402.02).

After reviewing the current status of the species, the environmental baseline for the Action Area, the effects of the Action and the cumulative effects, it is the Service's biological opinion that the Action is not likely to jeopardize the continued existence of the loggerhead sea turtle due to: (1) Nesting within the NRU appears to be increasing despite current threats; (2) nesting within the Action Area is following the same trend as the NRU despite current threats and environmental conditions; (3) effects due to construction activities are expected to be short term and become beneficial once construction is completed.

"Take" of sea turtles will be minimized by implementation of the Reasonable and Prudent Measures, and Terms and Conditions outlined in **Section 9**. These measures have been shown to help minimize adverse impacts to sea turtles.

5. CRITICAL HABITAT FOR THE LOGGERHEAD SEA TURTLE

5.1. Status of Critical Habitat

This section summarizes best available data about the current condition of all designated units of critical habitat for the Northwest Atlantic Ocean DPS loggerhead sea turtle (*Caretta caretta*) that are relevant to formulating an opinion about the Action. The Service published its decision to designate critical habitat for loggerhead sea turtle on July 10, 2014, (79 FR 39755).

5.1.1. Description of Critical Habitat

Critical habitat is comprised of 1,102.1 kilometers (km) (684.8 miles) in 88 separate units located in the terrestrial environment in the States of North Carolina, South Carolina, Georgia, Florida, Alabama, and Mississippi. The designated units include habitats that support loggerhead sea turtle oviposition (egg laying), embryonic development, and hatching.

Critical habitat designation for the loggerhead sea turtle used the term "primary constituent elements" (PCEs) to identify the key components of critical habitat that are essential to its conservation and may require special management considerations or protection. Revisions to the critical habitat regulations in 2016 (81 FR 7214, 50 CFR §4.24) discontinue use of the term PCEs, and rely exclusively the term "physical and biological features" (PBFs) to refer to these key components, because the latter term is the one used in the statute. This shift in terminology does not change how the Service conducts a "destruction or adverse modification" analysis. In this BO, we use the term PBFs to label the key components of critical habitat that provide for the conservation of the loggerhead sea turtle that were identified in its critical habitat designation rule as PCEs.

The PBFs of loggerhead sea turtle critical habitat are (79 FR 39755):

PBF 1 – Suitable nesting beach habitat that has (a) relatively unimpeded nearshore access from the ocean to the beach for nesting females and from the beach to the ocean for both post-nesting females and hatchlings, and (b) is located above mean high water to avoid being inundated frequently by high tides.

PBF 2 – Sand that (a) allows for suitable nest construction, (b) is suitable for facilitating gas diffusion conducive to embryo development, and (c) is able to develop and maintain temperatures and a moisture content conducive to embryo development.

PBF 3 – Suitable nesting beach habitat with sufficient darkness to ensure nesting turtles are not deterred from emerging onto the beach and hatchlings and post nesting females orient to the sea.

PBF 4 – Natural coastal processes or artificially created or maintained habitat mimicking natural conditions.

5.1.2. Conservation Value of Critical Habitat

Since loggerhead sea turtles nest on dynamic ocean beaches that may be significantly degraded or lost through natural processes (e.g., erosion) or coastal development (e.g., armoring, artificial lighting), it is important to conserve: (1) Beaches that have the highest nesting densities (by State or region within a State); (2) Beaches that have a good geographic spatial distribution to ensure protection of genetic diversity; (3) Beaches that collectively provide a good representation of total nesting; and (4) Beaches adjacent to the high density nesting beaches that can serve as expansion areas (79 FR 39755).

5.1.3. Conservation Needs for Critical Habitat

Special management considerations and/or protection are needed within critical habitat areas to address threats to the essential features of loggerhead sea turtle terrestrial habitat. The Service identified 12 categories that necessitate special management or protection including: (1) Recreational beach use; (2) Beach driving; (3) Predation; (4) Beach sand placement activities; (5) In-water and shoreline alterations; (6) Coastal development; (7) Artificial lighting; (8) Beach erosion; (9) Climate Change; (10) Habitat obstructions; (11) Human-caused disasters and response to natural and human-caused disasters; (12) Military testing and training activities (79 FR 39755). Many of these threats currently exist outside of and within critical habitat areas.

5.2. Environmental Baseline for Critical Habitat

This section is an analysis of the effects of past and ongoing human and natural factors leading to the current status of designated critical habitat for the loggerhead sea turtle within the Action Area. The environmental baseline is a “snapshot” of the condition of the physical and biological features (PBFs) that are essential to the conservation of the species within designated critical of the Action Area at the time of the consultation, and does not include the effects of the Action under review.

5.2.1. Action Area Conservation Value of Critical Habitat

The Action Area is within designated critical habitat unit LOGG-T-SC-09 (**Figure 7**) and currently contains all PBFs. Each unit within the NRU is essential to the recovery of the species. The text description of the unit is as follows:

LOGG-T-SC-09 – Folly Island, Charleston County: This unit consists of 11.2 km (7.0 miles) of island shoreline along the Atlantic Ocean. The island is separated from the mainland by the Atlantic Intracoastal Waterway, Folly River, a network of coastal islands, and salt marsh. The unit extends from Lighthouse Inlet to Folly River Inlet. The unit includes lands from the MHW line to the toe of the secondary dune or developed structures. Land in this unit is in State, and private and other ownership. The Lighthouse Inlet Heritage Preserve, is owned by the County, with a 10 percent undivided interest from the South Carolina Department of Natural Resources (SCDNR). The Folly Beach County Park is owned by the County. Both are managed by the Charleston County Park and Recreation Commission. This unit was occupied at the time of listing and is currently occupied. This unit supports expansion of nesting from an adjacent unit (LOGG-T-SC-10) that has high-density nesting by loggerhead sea turtles in South Carolina. This unit contains all of the PBFs. The PBFs in this unit may require special management considerations or protections to ameliorate the threats of recreational use, beach sand placement activities, in-water and shoreline alterations, coastal development, beach erosion, climate change, artificial lighting, human-caused disasters, and response to disasters. The City of Folly Beach has a beach management plan that includes measures to protect nesting and hatchling loggerhead sea turtles from anthropogenic disturbances (City of Folly Beach 1991). These measures apply to both the private and other lands within this critical habitat unit.

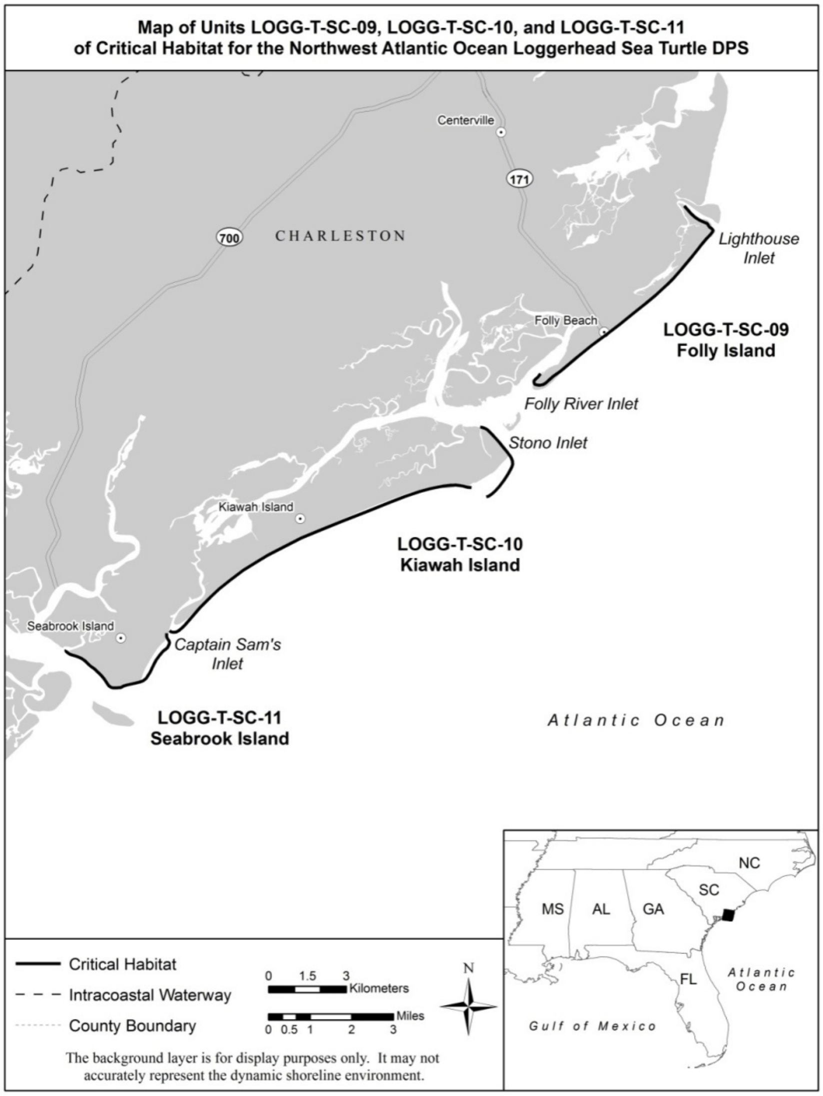

Figure 7. Map of Units LOGG-T-SC-09, LOGG-T-SC-10, and LOGG-T-SC-11

5.2.2. Action Area Conservation Needs for Critical Habitat

The Action Area on Folly Beach (LOGG-T-SC-09) has ten of the 12 categories necessitating special management or protection. Current threats within the Action Area are being managed by SCDNR, the Folly Beach nest protection project volunteers, the City of Folly Beach, and the Service.

5.3. Effects of the Action on Critical Habitat

This section analyzes the direct and indirect effects of the Action on critical habitat for the loggerhead sea turtle, which includes the direct and indirect effects of interrelated and interdependent actions. Direct effects are caused by the Action and occur at the same time and place. Indirect effects are caused by the Action, but are later in time and reasonably certain to occur. Our analyses are organized according to the description of the Action in section 2 of this BO.

5.3.1. Effects of Renourishment on Critical Habitat

Beach renourishment may alter the PBFs (see **Section 5.1.1**) that currently exist within the Action Area. Regarding PBF 1, construction on the beach during sea turtle nesting and hatchling season can obstruct nesting females from accessing the beach and hatchlings from entering the water unimpeded. More nests are washed out on the wide, flat beaches of the nourished areas than on the narrower steeply sloped natural beaches. This phenomenon may persist through the second post construction year and result from the placement of nests near the seaward edge of the beach berm where dramatic profile changes, caused by erosion and scarping, occur as the beach equilibrates to a more natural contour. Regarding PBF 2, a significantly larger proportion of turtles emerging on engineered beaches abandon their nesting attempts than turtles emerging on natural or prenourished beaches, even though more nesting habitat is available (Trindell *et al.* 1998; Ernest and Martin 1999; Herren 1999), with nesting success approximately 10 to 34 percent lower on nourished beaches than on control beaches during the first year post-nourishment. This reduction in nesting success is most pronounced during the first year following project construction and is most likely the result of changes in physical beach characteristics (beach profile, sediment grain size, beach compaction, frequency and extent of escarpments) associated with the nourishment project (Ernest and Martin 1999). This impact directly impacts PBF 2, however, on severely eroded sections of beach like the one where the Action is proposed where little or no suitable nesting habitat exists, sand placement can result in increased nesting (Ernest and Martin 1999). The placement of sand on a beach with reduced dry foredune habitat may increase sea turtle nesting habitat if the placed sand is highly compatible (i.e., grain size, shape, color, etc.) with naturally occurring beach sediments in the area. In addition, a nourished beach that is designed and constructed to mimic a natural beach system may benefit sea turtles more than an eroding beach it replaces. Regarding PBF 3, during nighttime construction lights directly visible on the beach during nesting and hatching season will illuminate suitable nesting habitat and may deter nesting females from nesting within the area or disorient nesting females or hatchlings on their way to the ocean. The newly created wider and flatter beach berm exposes sea turtles and their nests to lights that were less visible, or not visible, from nesting areas before the sand placement activity leading to a higher mortality of hatchlings. Regarding PBF 4, on severely eroded sections of beach like the one where the Action is proposed where little or no suitable nesting habitat exists, sand placement can result in increased nesting (Ernest and Martin 1999).

5.3.2. Effects of Groin Rehabilitation on Critical Habitat

Groin rehabilitation may directly and indirectly alter the PBFs (see **Section 5.1.1**) that currently exist within the Action Area. Regarding PBF 1, construction activities for groin rehabilitation on the beach during sea turtle nesting and hatchling season can obstruct nesting females from accessing the beach and hatchlings from entering the water unimpeded. Regarding PBF 2, construction activities for groin rehabilitation are not anticipated to adversely affect sand suitable for nest construction and embryonic development. Regarding PBF 3, lighting associated with nighttime construction will illuminate suitable nesting habitat and may deter nesting females from nesting within the area or disorient nesting females or hatchlings on their way to the ocean. Regarding PBF 4, groin rehabilitation may help trap dry sand and improve habitat suitability within this area on the island. However, it may cause downdrift impacts that degrade habitat conditions and exacerbate existing erosion west of the area by trapping sand.

5.4. Cumulative Effects on Critical Habitat

For purposes of consultation under ESA §7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under §7 of the ESA. The Service is not aware of any cumulative effects in the Action Area at this time; therefore, cumulative effects are not relevant to formulating our opinion for the Action.

5.5. Conclusion for Critical Habitat

In this section, we summarize and interpret the findings of the previous sections for the Northwest Atlantic Ocean DPS loggerhead sea turtle critical habitat (status, baseline, effects, and cumulative effects) relative to the purpose of a BO under §7(a)(2) of the ESA, which is to determine whether a Federal action is likely to:

a) jeopardize the continued existence of species listed as endangered or threatened; or
b) result in the destruction or adverse modification of designated critical habitat.

"*Destruction or adverse modification*" means a direct or indirect alteration that appreciably diminishes the value of designated critical habitat for the conservation of a listed species. Such alterations may include, but are not limited to, those that alter the physical or biological features essential to the conservation of a species or that preclude or significantly delay development of such features (50 CFR §402.02).

After reviewing the current status of the critical habitat, the environmental baseline for the Action Area, the effects of the Action, and the cumulative effects, it is the Service's biological opinion that the Action is not likely to destroy or adversely modify designated critical habitat for the loggerhead sea turtle due to: (1) Nesting within the NRU critical habitat units appears to be increasing despite current threats; (2) nesting within LOGG-T-SC-09 is following the same trend as the NRU despite current threats and environmental conditions; (3) effects due to construction activities are expected to be short term and become beneficial once construction is completed.

6. PIPING PLOVER

6.1. Status of the species

On January 10, 1986, the piping plover (*Charadrius melodus*) was listed under the Endangered Species Act (ESA) as endangered in the Great Lakes watershed and threatened elsewhere within its range, including migratory routes outside of the Great Lakes watershed and wintering grounds (USFWS 1985). However, the final listing rule did not utilize subspecies. The preamble of this rule acknowledged the continuing recognition of two subspecies, *Charadrius melodus melodus*

(Atlantic Coast of North America) and *Charadrius melodus circumcinctus* (Northern Great Plains of North America) in the American Ornithologist Union's most recent treatment of subspecies (AOU 1957). However, it also noted that allozyme studies with implications for the validity of the subspecies were in progress. The final rule determined the species as endangered in the Great Lakes watershed of both the United States (U.S.) and Canada and as threatened in the remainder of its range in the U.S. (Northern Great Plains, Atlantic and Gulf Coasts, Puerto Rico, and Virgin Islands), Canada, Mexico, Bahamas, and the West Indies (USFWS 1985).

Subsequent ESA actions have consistently recognized three separate breeding populations of piping plovers on the Atlantic Coast (threatened), Great Lakes (endangered) and Northern Great Plains (NGP) (threatened). Piping plovers that breed on the Atlantic Coast of the U.S. and Canada belong to the subspecies *C. m. melodus*. The second subspecies, *C. m. circumcinctus,* is comprised of two Distinct Population Segments (DPS). One DPS breeds on the Northern Great Plains of the U.S. and Canada, while the other breeds on the Great Lakes. Each of these three entities is demographically independent. The piping plover winters in coastal areas of the U.S. from North Carolina to Texas, and along the coast of eastern Mexico and on Caribbean islands from Barbados to Cuba and the Bahamas (Elliott-Smith and Haig 2004) (**Figure 8**).

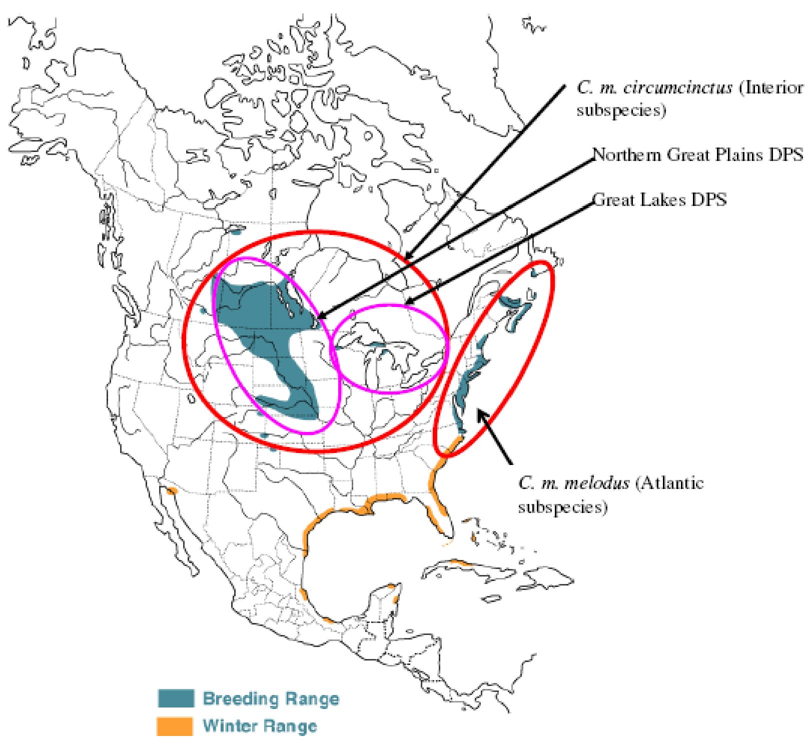

Figure 8. **Distribution and range of piping plovers (base map from Elliott-Smith and Haig 2004). Conceptual presentation of subspecies and DPS ranges are not intended to convey precise boundaries.**

Two successive recovery plans established delisting criteria for the threatened Atlantic Coast breeding population (USFWS 1988a, 1996). A joint recovery plan specified separate criteria for the endangered Great Lakes and threatened Northern Great Plains populations (USFWS 1988b), and the Service later approved a recovery plan exclusive to the Great Lakes population (USFWS 2003).

6.1.1. Species Description

The piping plover, named for its melodic call, is a small North American shorebird approximately 17 centimeters (7 inches) long with a wingspan of about 38 cm (15 in) and weighing 40-65 grams (1.4-2.3 oz.) (Palmer 1967, Elliot-Smith and Haig 2004). Adult piping plovers exhibit breeding and nonbreeding plumage. Plovers can arrive on wintering grounds with partial breeding plumage remaining (a single black breastband, which is often incomplete, and a black bar across the forehead). During the late summer or early autumn, the birds lose the black bands, the legs fade from orange to pale yellow, and the bill turns from orange and black to mostly black (**Figure 9**). Most adults begin their molt into breeding plumage before northward migration and complete the molt before arrival on their breeding sites. Piping plover subspecies are considered phenotypically indistinguishable, although slight clinal breeding plumage variations between populations have been noted (Elliot-Smith and Haig 2004).

Figure 9. Adult breeding plumage (left) and nonbreeding plumage (right).

6.1.2. Life History

Piping plovers live an average of five years, although studies have documented birds as old as 11 (Wilcox 1959) and 15 years. Breeding activity begins in mid-March when birds begin returning to their nesting areas (Coutu *et al*. 1990; Cross 1990; Goldin *et al*. 1990; MacIvor 1990; Hake 1993). Plovers are known to begin breeding as early as one year of age (MacIvor 1990; Haig 1992); however, the percentage of birds that breed in their first adult year is unknown. Piping plovers generally fledge only a single brood per season, but may re-nest several times if previous nests are lost.

Plovers depart their breeding grounds for their wintering grounds from July through late August, but southward migration extends through November. Piping plovers spend up to 10 months of their life cycle on their migration and winter grounds, generally July 15 through as late as May 15. Piping plovers migrate through and winter in coastal areas of the U.S. from North Carolina to Texas and in portions of Mexico and the Caribbean. Migration routes and habitats overlap breeding and wintering habitats, and, unless banded, migrants passing through a site usually are indistinguishable from other breeding or wintering piping plovers.

Habitat Use

Wintering piping plovers utilize a mosaic of habitat patches and move among these patches in response to local weather and tidal conditions (Nicholls and Baldassarre 1990a, Nicholls and Baldassarre 1990b, Drake *et al*. 2001, Cohen *et al*. 2008). Preferred coastal habitats include sand spits, small islands, tidal flats, shoals (usually flood tidal deltas), and sandbars that are often associated with inlets (Nicholls and Baldassarre 1990b, Harrington 2008, Addison 2012). Sandy mud flats, ephemeral pools, seasonally emergent seagrass beds, mud/sand flats with scattered oysters, and overwash fans are considered primary foraging habitats (Nicholls and Baldassarre 1990b, Cohen *et al*. 2008). A South Carolina study strongly links plover habitat use to the abundance of key invertebrate taxa (SCDNR 2011). Plovers vary their use of ocean beaches and bay shorelines and flats in Texas depending on season and in response to weather conditions (Zdravkovic and Durkin 2011, Zonick 2000).

Studies in North Carolina, South Carolina, Texas, and Florida complement earlier investigations of the habitat use patterns (Zivojnovich and Baldassarre 1987, Johnson and Baldassarre 1988, Nicholls and Baldassarre 1990a and 1990b, Fussell 1990, Drake *et al*. 2001). Nonbreeding piping plovers in North Carolina primarily used sound (bay or bayshore) beaches and sound islands for foraging. On ocean beaches they exhibited roosting, preening, and alert behaviors (Cohen *et al*. 2008). The probability of piping plovers being present on the sound islands increased as exposure of the intertidal areas increased (Cohen *et al*. 2008). Maddock *et al*. (2009) also observed shifts in roosting habitats and behaviors during high-tide periods in South Carolina. Similar patterns in Gulf Coast studies confirm high plover numbers on Gulf beaches during migration (July-October) and when wind conditions inundate bayside flats (Zdravkovic and Durkin 2011, Pinkston 2004, Zonick 2000).

Several studies identified wrack (organic material including seaweed, seashells, driftwood, and other materials deposited on beaches by tidal action) as an important component of roosting habitat for nonbreeding piping plovers[1]. Lott *et al*. (2009b) found that more than 90% of roosting piping plovers in southwest Florida were roosting in old wrack. In South Carolina, 45% of roosting piping plovers were in old wrack, and 18% were in fresh wrack (Maddock *et al*. 2009). Thirty percent of roosting piping plovers in northwest Florida were observed in wrack substrates (Smith 2007). In Texas, seagrass debris (bayshore wrack) was found to be an important feature of piping plover roost sites (Drake 1999).

Intertidal areas provide key foraging habitats. Exposed intertidal areas were the dominant foraging substrate, both in South Carolina (accounting for 94% of observed foraging piping plovers; Maddock *et al*. 2009) and in northwest Florida (96% of foraging observations; Smith 2007). In southwest Florida, Lott *et al*. (2009b) found approximately 75% of foraging piping plovers on intertidal substrates with bay beaches (bay shorelines as opposed to ocean-facing beaches) as the most common landform used by foraging piping plovers. In northwest Florida, however, Smith (2007) reported that landform use by foraging piping plovers was almost equally divided between Gulf (ocean-facing) and bay beaches. Zonick (2000) found dietary differences across the range of piping plovers in Texas, with plovers along the northern Texas coast feeding

[1] Wrack also contains invertebrate organisms consumed by piping plovers and other shorebirds.

predominantly on polychaetes while those observed further south largely fed on insects and other arthropods.

Atlantic and Gulf Coast studies highlighted the importance of inlets for nonbreeding piping plovers. Almost 90% of observations of roosting piping plovers at ten coastal sites in southwest Florida were on inlet shorelines (Lott *et al*. 2009b). In an evaluation of 361 International Shorebird Survey sites from North Carolina to Florida (Harrington 2008), piping plovers were among seven shorebird species found more often than expected ($p = 0.0004$; Wilcoxon Scores test) at inlet versus non-inlet locations. Wintering plovers on the Atlantic Coast prefer wide beaches near inlets (Nicholls and Baldassarre 1990b, Wilkinson and Spinks 1994). At inlets, foraging plovers are associated with moist substrate features such as intertidal flats, algal flats, and ephemeral pools (Nicholls and Baldassarre 1990b, Wilkinson and Spinks 1994, Dinsmore *et al*. 1998, Addison 2012).

In South Carolina, multivariate analyses showed that many of the taxa responsible for the temporal changes in composition of the invertebrate community at occupied foraging sites were also responsible for the changes associated with site abandonment by piping plovers (SCDNR 2011). This suggests that taxa changes in the diets of migratory and overwintering piping plovers were occurring both within individual foraging sites (leading to subsequent site-abandonment) and within the larger Kiawah Island/Bird Key system, potentially contributing to declines in the overwintering population. The study further suggests that larger, errant polychaetes such as the families Nereididae, Glyceridae, and Oenonidae may be particularly important to piping plover overwintering in this region. Consequently, habitat changes, whether natural or anthropogenic in origin, that affect polychaete densities may also affect overwintering populations of the piping plover (SCDNR 2011).

Geographic analysis of piping plover distribution on the upper Texas coast noted major concentration areas in washover passes (low, sparsely vegetated barrier island habitats created and maintained by temporary, storm-driven water channels) and at the mouths of rivers feeding into major bay systems (Arvin 2008). Earlier studies in Texas indicated the importance of washover passes or fans, which were commonly used by piping plovers during periods of high bayshore tides and during the spring migration period (Zonick 1997, Zonick 2000). Surveys of the Lower Laguna Madre in Texas found piping plovers using both Gulf beach and bayside areas during the fall 2009 migratory period. These include Gulf beaches, inlet shorelines, bay shorelines of barrier islands, shorelines of islands in the bay (natural and dredged-material), mainland bay shorelines, tidal flats and other habitats such as isolated "pools" of evaporating water associated with bay habitats. A clear shift from Gulf beaches to bay habitats occurred during the wintering period, as well as during certain wind and weather conditions (Zdravkovic and Durkin 2011). Piping plovers have also been observed in high numbers on seasonally emergent seagrass beds and oyster-studded mud flats in several central Texas coastal bays (Cobb *in* Elliott-Smith *et al*. 2009).

Winter Site Fidelity

Piping plovers exhibit a high degree of intra- and inter-annual fidelity to wintering areas, which often encompass several relatively nearby sites (Drake *et al*. 2001, Noel and Chandler 2008, Stucker *et al*. 2010). Gratto-Trevor *et al*. (2012) found little movement between or among regions (**Figure 10**), and reported that 97% of the birds they surveyed remained in the same region, often at the same beach. Between August of 2010 and December of 2014, 44 piping plovers wintering in the Bahamas were seen either on the beach where they were banded or within six km of that beach (Gratto-Trevor *et al*. 2016). Only six of 259 banded piping plovers were observed more than once per winter moving across boundaries of seven U.S. regions. Of 216 birds observed in multiple years, only eight changed regions between years, and several of these shifts were associated with late summer or early spring migration periods (Gratto-Trevor *et al*. 2012). Although many sites on the northern Gulf Coast of Texas and in Louisiana were affected by hurricanes after the 2008 fall migration, none of the 17 birds known to have wintered in these areas before the hurricane and resighted afterward moved from their original areas (Gratto-Trevor *et al*. 2012).

The areas used by wintering piping plovers often comprise habitats on both sides of an inlet, nearby sandbars or shoals, and ocean and bayside shorelines. In South Carolina, Maddock *et al*. (2009) documented many movements back and forth across inlets by color-banded piping plovers, as well as occasional movements of up to 18 km by approximately 10% of the banded population. Similarly, eight banded piping plovers that were observed in two locations during the 2006-2007 surveys in Louisiana and Texas were all in close proximity to their original location, such as on the bay and ocean side of the same island or on adjoining islands (Maddock 2008).

The mean-average home-range size for 49 radio-marked piping plovers in southern Texas in 1997-1998 was 12.6 km^2; the mean core area was 2.9 km^2; and the mean linear distance moved between successive locations, averaged across seasons, was 3.3 km (Drake *et al*. 2001). Seven radio-tagged piping plovers used a 20.1 km^2 area at Oregon Inlet, North Carolina, in 2005-2006, and piping plover activity was found to be concentrated in 12 areas totaling 2.2 km^2 that were located on both sides of the inlet (Cohen *et al*. 2008). Noel and Chandler (2008) also observed high site fidelity of banded piping plovers to 1-4.5 km sections of beach on Little St. Simons Island, Georgia.

Intra- and Inter-specific Interactions

Piping plovers are often found in association with other shorebird species during the nonbreeding season, as many shorebird species utilize the southern Atlantic and Gulf Coasts for migration and wintering (Nicholls and Baldassarre 1990b, Eubanks 1992, Helmers 1992). Migrating and wintering piping plovers often roost close to conspecifics, as well as in multi-species flocks (Nicholls and Baldassarre 1990b, Zonick and Ryan 1993, Elliott and Teas 1996, Drake 1999). During foraging, however, territorial and agonistic interactions with other piping plovers and with similar-sized plover species, including semipalmated and snowy plovers, are relatively common (Johnson and Baldassarre 1988, Zonick and Ryan 1993, Elliott and Teas 1996, Drake 1999). Burger *et al*. (2007) observed competition for foraging space among shorebird species foraging in Delaware Bay, especially between shorebirds and larger gulls. Intra- and inter-specific competition for foraging habitat may be increased by continuing habitat loss and degradation, as well as by disturbance due to human recreation, forcing some piping plovers to forage or roost in suboptimal habitats and thereby affecting their energetic budgets. Shorebirds require extensive fat reserves to complete migrations. Birds with less than maximum fat reserves are expected to show reduced survival rates (Brown *et al*. 2001).

6.1.3. Numbers, Reproduction, and Distribution

The data from the International Piping Plover Breeding Censuses represent a minimum estimate of all three breeding populations (**Table 6**). Although the effort is as comprehensive as possible, some populations and some areas are able to be more intensively monitored than others outside of Census years. However, some portions of populations are only monitored during Census years Northern Great Plains (NGP) Canada) so this data is currently the best way to get a rough estimate of the status of all three breeding populations. The data from the most recent (2016) Census is still being compiled so the final results are not available at this time. However, the 2011 Piping Plover Breeding Census documented 2,391 breeding pairs with a total of 5,723 birds throughout Canada and U.S (Elliott-Smith *et al*. 2015) (**Table 6**).

Table 6. Number of Adults Documented During the 1991, 1996, 2001, 2006, and 2011 International Piping Plover Breeding Census (Haig *et al*. 2005, Elliott-Smith *et al*. 2009, Elliott-Smith *et al*. 2015).

Population	**Number of piping plovers**				
	1991	**1996**	**2001**	**2006**	**2011**
NGP	3469	3286	2953	4662	3486
Canada	*1437*	*1687*	*972*	*1703*	*2249*
U.S.	*2032*	*1599*	*1981*	*2959*	*1237*
Great Lakes	40	48	72	110	112
Canada	*0*	*1*	*1*	*1*	*14*
U.S.	*40*	*47*	*71*	*109*	*98*
Atlantic Coast	1641	2591	2911	3312	3362
Canada	*509*	*422*	*481*	*457*	406
U.S.	*1462*	*2169*	*2430*	*2855*	2952
Total	**5480**	**5925**	**5936**	**8084**	**5723**

Northern Great Plains Population

The NGP plover breeds from Alberta to Manitoba, Canada and south to Nebraska; although some nesting has recently occurred in Oklahoma. Currently, the most westerly breeding piping plovers in the U.S. occur in Montana and Colorado.

The decline of piping plovers on rivers in the Northern Great Plains has been largely attributed to the loss of sandbar island habitat and forage base due to dam construction and operation. Nesting occurs on sand flats or bare shorelines of rivers and lakes, including sandbar islands in the upper Missouri River system, and patches of sand, gravel, or pebbly-mud on the alkali lakes of the northern Great Plains. Plovers do nest on shorelines of reservoirs created by the dams, but reproductive success is often low and reservoir habitat is not available in many years due to high water levels or vegetation. Dams operated with steady constant flows allow vegetation to grow on potential nesting islands, making these sites unsuitable for nesting. Population declines in alkali wetlands are attributed to wetland drainage, contaminants, and predation.

Since the NGP population is geographically widespread, with many birds in very remote places, especially in the U.S. and Canadian alkali lakes. Thus, determining the number of birds or even identifying a clear trend in the population is a difficult task. The International Piping Plover Census (IPPC) was designed, in part, to help deal with this problem by instigating a large effort every five years in which an attempt is made to survey every area with known or potential piping plover breeding habitat during a two-week window (i.e., the first two weeks of June). The relatively short window is designed to minimize double counting if birds move from one area to another. The 1988 recovery plan, which is currently being revised, uses the numbers from the IPPC as a major criterion for delisting, as does the 2006 Canadian Recovery Plan (Environment Canada 2006).

Great Lakes Population

The Great Lakes plovers once nested on Great Lakes beaches in Illinois, Indiana, Michigan, Minnesota, New York, Ohio, Pennsylvania, Wisconsin, and Ontario. Great Lakes piping plovers nest on wide, flat, open, sandy or cobble shoreline with very little grass or other vegetation. Reproduction is adversely affected by human disturbance of nesting areas and predation by foxes, gulls, crows and other avian species. Shoreline development, such as the construction of marinas, breakwaters, and other navigation structures, has adversely affected nesting and brood rearing.

Atlantic Coast Population

The Atlantic Coast piping plover breeds on coastal beaches from Newfoundland and southeastern Quebec to North Carolina. Historical population trends for the Atlantic Coast piping plover have been reconstructed from scattered, largely qualitative records. Nineteenth-century naturalists, such as Audubon and Wilson, described the piping plover as a common summer resident on Atlantic Coast beaches (Haig and Oring 1987). However, by the beginning of the 20th Century, egg collecting and uncontrolled hunting, primarily for the millinery trade, had greatly reduced the population, and in some areas along the Atlantic Coast, the piping plover was close to extirpation. Following passage of the Migratory Bird Treaty Act (40 Stat. 775; 16 U.S.C. 703-712) in 1918, and changes in the fashion industry that no longer exploited wild birds for feathers, piping plover numbers recovered to some extent (Haig and Oring 1985).

Available data suggest that the most recent population decline began in the late 1940s or early 1950s (Haig and Oring 1985). Reports of local or statewide declines between 1950 and 1985 are numerous, and many are summarized by Cairns and McLaren (1980) and Haig and Oring (1985). While Wilcox (1939) estimated more than 500 pairs of piping plovers on Long Island, New York, the 1989 population estimate was 191 pairs (see Table 4, USFWS 1996). There was little focus on gathering quantitative data on piping plovers in Massachusetts through the late 1960s because the species was commonly observed and presumed to be secure. However, numbers of piping plover breeding pairs declined 50 to 100 percent at seven Massachusetts sites between the early 1970s and 1984 (Griffin and Melvin 1984). Piping plover surveys in the early years of the recovery effort found that counts of these cryptically colored birds sometimes went up with increased census effort, suggesting that some historic counts of piping plovers by one or a few observers may have underestimated the piping plover population. Thus, the magnitude of the species decline may have been more severe than available numbers imply.

Survival

Population viability analyses (PVAs) conducted for piping plovers (Ryan *et al.* 1993, Melvin and Gibbs 1996, Plissner and Haig 2000, Wemmer *et al*. 2001, Larson *et al*. 2002, Calvert *et al*. 2006, Brault 2007, McGowan and Ryan 2009) all demonstrate the sensitivity of extinction risk in response to small declines in adult and/or juvenile survival rates. These results further emphasize the importance of nonbreeding habitat to species recovery (Roche *et al*. 2010). Poor overwintering and stopover habitat has been shown to have a negative effect on survival of other shorebird species, which contributed to breeding population declines (Gill *et al*. 2001, Baker *et al*. 2004, Morrison and Hobson 2004).

There is limited information specific to survival rates during the nonbreeding portion of the annual cycle. Catlin *et al.* (2015) summarized survival estimates for piping plovers from 1959-2014 and found average true survival of after hatch year birds ranged from 0.70 to 0.80 in four studies. Drake *et al*. (2001) observed no mortality among 49 radio-marked piping plovers (total of 2,704 transmitter-days) in Texas in the 1990s. Cohen *et al*. (2008) also reported no mortality among a small sample (n=7) of radio-marked piping plovers at Oregon Inlet, North Carolina in 2005-2006. Analysis of resighting data for 87 banded piping plovers observed in South Carolina during 2006-2007 and 2007-2008 found 100% survival from December to April[2] (J. Cohen, Virginia Polytechnic Institute and State University, pers. comm. 2009). At Little St. Simons Island, Georgia, Noel *et al*. (2007) inferred two winter mortalities among 21 banded (but not radio-tagged) overwintering piping plovers in 2003-2004, and nine mortalities among 19 overwintering birds during the winter of 2004-2005. In a study of 150 after-hatch-year Great Lakes piping plovers, LeDee (2008) found higher apparent survival[3] rates during breeding and southward migration than during winter and northward migration.

Analysis of piping plover mark-recapture data by Roche *et al*. (2010) found that after-hatch-year apparent survival declined in four of their seven study populations. They found evidence of correlated year-to-year fluctuations in annual survival among populations wintering primarily along the southeastern U.S. Atlantic Coast, as well as indications that shared overwintering or stopover sites may influence annual variation in survival among geographically disparate breeding populations. Additional mark-resighting analysis of color-banded individuals across piping plover breeding populations has the potential to shed light on threats that may affect

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[2] However, two of those birds were seen in the first winter and resighted in the second fall, but were not seen during the second winter (Maddock et al. 2009).

[3] "Apparent survival" does not account for permanent emigration. If marked individuals leave a survey site, apparent survival rates will be lower than true survival. If a survey area is sufficiently large, such that emigration out of the site is unlikely, apparent survival will approach true survival.

survival in the migration and wintering range, and to further elucidate survival within the annual cycle (Cohen 2009, Roche *et al*. 2010).

Status and distribution

<u>Breeding Range</u>

Northern Great Plains Population

The Northern Great Plains population is geographically widespread, with many birds in unpopulated areas, especially in the U.S. and Canadian alkaline lakes region. Determining the number of birds or even identifying a clear trend in the population is challenging. The International Piping Plover Census was designed, in part, to address this problem by implementing a range-wide survey every five years, starting in 1991. During a two-week window, monitors attempt to survey every area with known or potential piping plover breeding habitat. The relatively short window is designed to minimize double counting if birds move from one area to another.

Participation in the International Piping Plover Census has been excellent in the Northern Great Plains (Elliot-Smith *et al.* 2009). The large area to be surveyed and sparse human population in the Northern Great Plains make annual surveys of the entire area impractical. Many areas are only surveyed during the Census years.

The wide swings in bird numbers appear closely tied to the amount of habitat available for nesting (**Table 6**). The amount of available habitat, in turn, is largely caused by multi-year wet and dry cycles in the Northern Great Plains. The International Census may not be sufficiently robust in statistical design to inform our understanding of the population's dynamics. In the 2009 status review, the Service concluded that the Northern Great Plains piping plover population remains vulnerable, especially due to management of river systems throughout the breeding range (USFWS 2009b). Many of the threats identified in the 1988 recovery plan, including those affecting Northern Great Plains piping plover population during the two-thirds of its annual cycle spent in the wintering range, remain today or have intensified.

Great Lakes Population

The population has shown significant growth, from approximately 17 pairs at the time of listing in 1986, to 75 pairs in 2017. The 75 breeding pairs represent approximately 50% of the current recovery goal of 150 breeding pairs for the Great Lakes population. Although initial information considered at the time of the 2003 recovery plan suggested the population may be at risk from a lack of genetic diversity, currently available information suggests that genetic diversity may not pose a high risk to the Great Lakes population. Additional genetic information is needed to assess genetic structure of the population and verify the adequacy of a 150 pair population to maintain long-term heterozygosity and allelic diversity.

Population growth is evidence of the effectiveness of the ongoing Great Lakes piping plover recovery program. Most major threats, however, including habitat degradation, predation, and human disturbance remain persistent and pervasive. Severe threats from human disturbance and predation remain ubiquitous within the Great Lakes. Expensive labor-intensive management to minimize the effects of these continuing threats, as specified in recovery plan tasks, are implemented every year by a network of dedicated governmental and private partners. Because threats to Great Lakes piping plovers persist, reversal of gains in abundance and productivity are expected to quickly follow if current protection efforts are reduced.

In the 2009 status review, the Service concluded that the Great Lakes population remains at considerable risk of extinction due to its small size, limited distribution and vulnerability to stochastic events, such as disease outbreak (USFWS 2009b). In addition, the factors that led to the piping plover's 1986 listing remain present.

Atlantic Coast Population

Substantial population growth, from approximately 790 pairs in 1986 to1,941 pairs in 2016 (2017 preliminary estimate not available), has decreased the Atlantic Coast piping plover's vulnerability to extinction since ESA listing (USFWS unpublished data). Annual estimates of breeding pairs of Atlantic Coast piping plovers are based on multiple surveys at most occupied sites. Sites that cannot be monitored repeatedly in May and June (primarily sites with few pairs or inconsistent occupancy) are surveyed at least once during a standard nine-day count period (Hecht and Melvin 2009).

Considerable progress has been made towards the overall goal of 2,000 breeding pairs articulated in recovery criterion 1. As discussed in the 1996 revised recovery plan, however, the overall security of the Atlantic Coast piping plover is fundamentally dependent on even distribution of population growth, as specified in subpopulation targets, to protect a sparsely-distributed species with strict biological requirements from environmental variation (including catastrophes) and increase the likelihood of interchange among subpopulations.

In the 2009 status review, the Service concluded that the Atlantic Coast piping plover remains vulnerable to low numbers in the Southern and Eastern Canada (and, to a lesser extent, the New York-New Jersey) Recovery Units (USFWS 2009b). Furthermore, the factors that led to the piping plover's 1986 listing remain operative rangewide (including in New England), and many of these threats have increased. Interruption of costly, labor-intensive efforts to manage these threats would quickly lead to steep population declines.

<u>Nonbreeding Range</u>

Piping plovers spend up to 10 months of their annual cycle on their migration and winter grounds, typically from 15 July through 15 May (Elliott-Smith and Haig 2004, Noel *et al*. 2007, Stucker *et al*. 2010). Southward migration from the breeding grounds primarily occurs from July to September, with the majority of birds initiating migration by the end of August (USFWS 1996, USFWS 2003). However, the New Jersey Division of Fish and Wildlife documented sustained presence of low numbers of piping plovers at several sites through October 2011 (C. Davis, New Jersey Division of Fish and Wildlife, pers. comm. 2012). Piping plovers depart the wintering grounds as early as mid-February and as late as mid-May, with peak migration in

March (Haig 1992). In their analysis of 10 years of band sightings, Stucker *et al*. (2010) found that wintering adult males and females from the Great Lakes population exhibit latitudinal segregation. Female plovers arrived on the winter grounds before males and returned later to breeding sites. Second year birds arrived latest on the breeding grounds, rarely appearing on the breeding grounds before the third week of May (Stucker *et al*. 2010).

Routes of migration and habitat use overlap breeding and wintering habitats and, unless the birds are banded, migrants passing through a site are indistinguishable from breeding or wintering piping plovers. Coastal migration stopovers of plovers banded in the Great Lakes region have been documented in New Jersey, Maryland, Virginia, North Carolina, South Carolina and Georgia (Stucker *et al*. 2010). Migrating birds from eastern Canada have been observed in Massachusetts, New Jersey, New York, and North Carolina (Amirault *et al*. 2005). Piping plovers banded in the Bahamas have been sighted during migration in nine Atlantic Coast states and provinces between Florida and Nova Scotia (C. Gratto-Trevor, Environment Canada, pers. comm. 2012a). In general, the distance between stopover locations and the duration of stopovers throughout the coastal migration range remain poorly understood.

International Piping Plover Winter Censuses, which began in 1991, have been conducted during mid-winter at five-year intervals across the species' range (**Table 7**). Results of the 2015 Census are not available at this time. Total numbers have fluctuated over time, with some areas increasing while other areas showed declines. Regional and local fluctuations may reflect changes in the quantity and quality of suitable foraging and roosting habitat, which vary in response to natural coastal formation processes as well as anthropogenic habitat changes (e.g., inlet relocation, dredging of shoals and spits). See, for example, discussions of survey number changes in Mississippi, Louisiana, and Texas in Elliott-Smith *et al*. (2009). Fluctuations may also reflect localized weather conditions during surveys or different survey coverage; for example, changes in wind-driven tides can cause large rapid shifts in the distribution of piping plovers on the Texas Laguna Madre (Zonick 2000). In another example, Cobb (*in* Elliott-Smith *et al*. 2009) notes that use of airboats during the 1991 and 2006 censuses facilitated greater coverage in central Texas than in 1996 and 2001, when airboats were not used and counts were lower. Changes in wintering numbers within a given area may also be influenced by growth or decline in particular breeding populations.

Increased survey effort in the Bahamas since approximately 2006 resulted in dramatic increases in wintering population estimates. Although the 2016 International Piping Plover Winter Census are not yet available, over 1,000 birds were counted in the Bahamas during 2011 (Elliott-Smith *et al.* 2015), compared to 417 birds in 2006 (Elliott-Smith *et al*. 2009) and 35 birds in 2001(Haig *et al*. 2005). Additional habitat in the Bahamas remains to be surveyed, as do many other sites in the Caribbean. Piping Plovers have been reported from Nicaragua, St. Vincent and the Grenadines, Turks and Caicos Islands, and St. Croix (L. Schibley, Manomet Center for Conservation Science, pers. comm. 2011, and C. Lombard, USFWS, pers. comm. 2010), but follow-up is needed to determine where and in what numbers piping plovers were seen and if the sites are used regularly.

Table 7. Results of the 1991, 1996, 2001, 2006, and 2011 international piping plover winter censuses (Haig *et al*. 2005, Elliott-Smith *et al*. 2009, Elliott-Smith *et al*. 2015).

Location	Number of piping plovers				
	1991	1996	2001	2006	2011
Virginia	ns[a]	Ns	ns	1	1
North Carolina	20	50	87	84	43
South Carolina	51	78	78	100	86
Georgia	37	124	111	212	63
Florida	551	375	416	454	306
-Atlantic	*70*	*31*	*111*	*133*	*83*
-Gulf	*481*	*344*	*305*	*321*	*223*
Alabama	12	31	30	29	38
Mississippi	59	27	18	78	88
Louisiana	750	398	511	226	86
Texas	1,904	1,333	1,042	2,090	2,145
Puerto Rico	0	0	6	ns	2
U.S. Total	**3,384**	**2,416**	**2,299**	**3,355**	**2,858**
Mexico	27	16	ns	76	30
Bahamas	29	17	35	417	1066
Cuba	11	66	55	89	19
Other Caribbean Islands	0	0	0	28	0
GRAND TOTAL	**3,451**	**2,515**	**2,389**	**3,884**	**3,973**

[a] ns = not surveyed

Survey timing and intensity affect abundance estimates and the ability to detect local movements of nonbreeding piping plovers. Mid-winter surveys (such as the International Census) may substantially underestimate the number of nonbreeding piping plovers using a site or region during other months. Along the central Texas Gulf Coast, Pinkston (2004) observed much heavier use of ocean-facing beaches between early September and mid-October (approximately 16 birds per mile) than during the period from December to March (approximately two birds per mile). Zdravkovic and Durkin (2011) reported a similar pattern in southern Texas. In late September, 2007, 104 piping plovers were counted at the south end of Ocracoke Island, North Carolina (NPS 2007), where none were seen during the 2006 International Piping Plover Winter Census (Elliott-Smith *et al*. 2009). Differences among fall, winter, and spring counts in South Carolina were less pronounced, but large inter-year fluctuations (e.g., 108 piping plovers in spring 2007 versus 174 piping plovers in spring 2008) were observed (Maddock *et al*. 2009). Noel *et al*. (2007) observed up to 100 piping plovers during peak migration and only about 40 overwintering at Little St. Simons Island, Georgia in 2003-2005. Monthly counts at Phipps Preserve in Franklin County, Florida ranged from a mid-winter low of four piping plovers in December 2006 to peak counts of 47 in October 2006 and March 2007 (Smith 2007).

Zdravkovic and Durkin (2011) attributed substantially higher counts during surveys in the Lower Laguna Madre, Texas in 2010 compared with the 2006 International Census (881 plovers versus 459 plovers) to more complete survey coverage.

The number of surveyor visits to the site may also affect abundance estimates for nonbreeding piping plovers. A preliminary analysis found 87% detection during the mid-winter period at South Carolina sites surveyed three times a month during fall and spring and one time per month during winter, compared with 42% detection at sites surveyed only three times per year (J. Cohen, pers. comm. 2009, review of data by Maddock *et al*. 2009).

Gratto-Trevor *et al*. (2012) found distinct patterns (but no exclusive partitioning) in winter distribution of banded piping plovers from four breeding areas (**Figure 10**). Resightings of more than 700 uniquely marked birds from 2001 to 2008 were used to analyze winter distributions along the Atlantic and Gulf Coasts. Plovers from eastern Canada and most Great Lakes birds wintered from North Carolina to Southwest Florida. However, eastern Canada birds were more heavily concentrated in North Carolina, while a larger proportion of Great Lakes piping plovers were found in South Carolina, Georgia, and Florida. This pattern is consistent with analysis of band sightings of Great Lakes plovers from 1995-2005 by Stucker *et al*. (2010). Gratto-Trevor *et al*. (2012) also found that Northern Great Plains populations were primarily seen farther west and south, especially on the Texas Gulf Coast. The majority of birds from the Canadian Prairie were observed in Texas (particularly southern Texas), while individuals from the U.S. Great Plains were more widely distributed on the Gulf Coast from Texas to Florida. Seventy-nine percent of 57 piping plovers banded in the Bahamas in 2010 have been reported breeding on the Atlantic Coast, and none have been resighted at interior locations (preliminary results, Gratto-Trevor pers. comm. 2012a). However, consistent with patterns observed in other parts of the wintering range, a few banded individuals from the Great Lakes and Northern Great Plains populations have been observed in the Bahamas (Gratto-Trevor pers. comm. 2012b, D. Catlin, Virginia Polytechnic Institute, pers. comm. 2012a). Collectively, these studies demonstrate an intermediate level of connectivity between breeding and wintering areas. Specific breeding populations will be disproportionately affected by habitat and threats occurring where they are most concentrated in the winter.

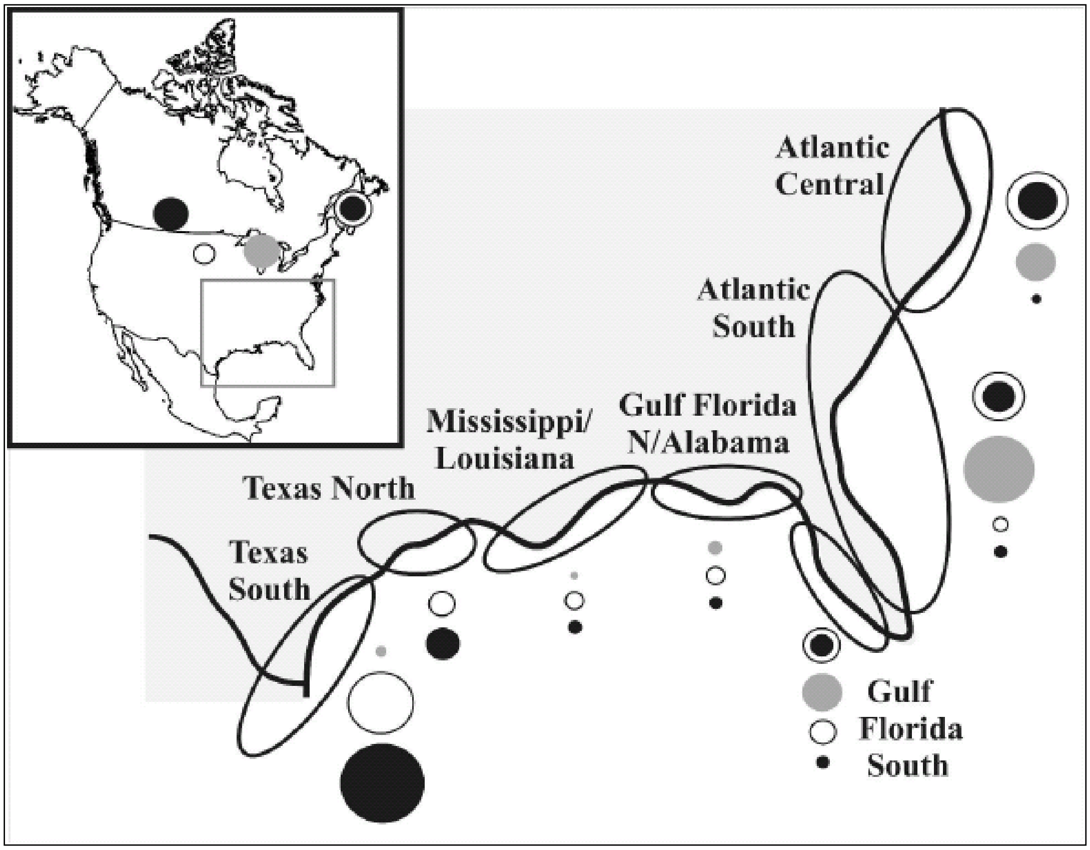

Figure 10. The winter distribution in the continental U.S. of piping plovers from four breeding locations (inset), including eastern Canada (white circle with central black dot), Great Lakes (gray circle), U.S. Northern Great Plains (white circle), and Prairie Canada (black circle). The wintering range is expanded to the right, divided into different wintering regions. The size of the adjacent circles relative to the others represents the percentage of individuals from a specific breeding area reported in that wintering region (from Gratto-Trevor *et al*. 2012; reproduced by permission).

6.1.4. Conservation Needs and Threats

Threats to Piping Plovers

The three recovery plans stated that shoreline development throughout the wintering range poses a threat to all populations of piping plovers. The plans further stated that beach maintenance and nourishment, inlet dredging, and artificial structures, such as jetties and groins, could eliminate wintering areas and alter sedimentation patterns leading to the loss of nearby habitat.

Loss, Modification, and Degradation of Habitat

The wide, flat, sparsely vegetated barrier beaches, spits, sandbars, and bayside flats preferred by piping plovers in the U.S. are formed and maintained by natural forces and are thus susceptible to degradation caused by development and shoreline stabilization efforts. As described below, barrier island and beachfront development, inlet and shoreline stabilization, inlet dredging, beach maintenance and nourishment activities, seawall installations, and mechanical beach grooming continue to alter natural coastal processes throughout the range of migrating and wintering piping plovers. Dredging of inlets can affect spit formation adjacent to inlets, as well as ebb and flood tidal shoal formation. Jetties stabilize inlets and cause island widening and subsequent vegetation growth on the updrift inlet shores; they also cause island narrowing and/or erosion on the downdrift inlet shores. Seawalls and revetments restrict natural island movement and exacerbate erosion. Although dredge and fill projects that place sand on beaches and dunes may restore lost or degraded habitat in some areas, in other areas these projects may degrade habitat quality by altering the natural sediment composition, depressing the invertebrate prey base, hindering habitat migration with sea level rise, and replacing the natural habitats of the dune-beach-nearshore system with artificial geomorphology. Construction of any of these projects during months when piping plovers are present also causes disturbance that disrupts the birds' foraging and roosting behaviors. These threats are exacerbated by accelerating sea level rise, which increases erosion and habitat loss where existing development and hardened stabilization structures prevent the natural migration of the beach and/or barrier island. Although threats from sea level rise are discussed on page 41, its specific synergistic effects on threats from coastal development and artificial coastal stabilization are also described in the pertinent subsections, below.

Development and Construction

Development and associated construction threaten the piping plover in its migration and wintering range by degrading, fragmenting, and eliminating habitat. Constructing buildings and infrastructure adjacent to the beach can eliminate roosting and loafing habitat within the development's footprint and degrade adjacent habitat by replacing sparsely vegetated dunes or back-barrier beach areas with landscaping, pools, fences, etc. In addition, bayside development can replace foraging habitat with finger canals, bulkheads, docks and lawns. High-value plover habitat becomes fragmented as lots are developed or coastal roads are built between oceanside and bayside habitats. Development activities can include lowering or removing natural dunes to improve views or grade building lots, planting vegetation to stabilize dunes, and erecting sand fencing to establish or stabilize continuous dunes in developed areas; these activities can further degrade, fragment, and eliminate sparsely vegetated and unvegetated habitats used by the piping plover and other wildlife. Development and construction of other infrastructure in close proximity to barrier beaches often creates economic and social incentives for subsequent shoreline stabilization projects, such as shoreline hardening and beach nourishment.

At present, there are approximately 2,119 miles of sandy beaches within the U.S. continental wintering range of the piping plover (**Table 3**). Approximately 40% (856 miles) of these sandy beaches are developed, with mainland Mississippi (80%), Florida (57%), Alabama (55%), South Carolina (51%), and North Carolina (49%) comprising the most developed coasts, and Mississippi barrier islands (0%), Louisiana (6%), Texas (14%) and Georgia (17%) the least developed (Rice 2012b). As discussed further below, developed beaches are highly vulnerable to further habitat loss because they cannot migrate in response to sea level rise.

Several studies highlight concerns about adverse effects of development and coastline stabilization on the quantity and quality of habitat for migrating and wintering piping plovers and other shorebirds. For example, Zdravkovic and Durkin (2011) observed fewer plovers on the developed portions of the Laguna and Gulf beach sides of South Padre Island than on undeveloped portions during both migratory and wintering surveys. Drake *et al*. (2001) observed that radio-tagged piping plovers overwintering along the southern Laguna Madre of Texas seldom used tidal flats adjacent to developed areas (five of 1,371 relocations of radio-marked individuals), suggesting that development and associated anthropogenic disturbances influence piping plover habitat use. Detections of piping plovers during repeated surveys of the upper Texas coast in 2008 were low in areas with significant beach development (Arvin 2008).

The development of bayside or estuarine shorelines with finger canals and their associated bulkheads, docks, buildings, and landscaping have led to direct loss and degradation of plover habitat. Finger canals are channels cut into a barrier island or peninsula from the soundside to increase the number of waterfront residential lots. Finger canals can lead to water pollution, fish kills, loss of aquatic nurseries, saltwater intrusion of groundwater, disruption of surface flows, island breaching due to the funneling of storm surge, and a perpetual need for dredging and disposal of dredged material in order to keep the canals navigable for property owners (Morris *et al*. 1978, Bush *et al*. 1996).

Rice (2012b) has identified over 900 miles (43%) of sandy beaches in the wintering range that are currently “preserved” through public ownership, ownership by non-governmental conservation organizations, or conservation easements (**Table 3**). These beaches may be subject to some erosion as they migrate in response to sea level rise or if sediment is removed from the coastal system, and they are vulnerable to recreational disturbance. However, these are the areas most likely to maintain the geomorphic characteristics of suitable piping plover habitat.

In summary, approximately 40% of the sandy beach shoreline in the migration and wintering range is already developed, while 43% are largely preserved. This means, however, that the remaining 17% of shoreline habitat (that which is currently undeveloped but not preserved) is susceptible to future loss to development and the attendant threats from shoreline stabilization activities and sea level rise.

Dredging and Sand Mining

The dredging and mining of sediment from inlet complexes threatens the piping plover on its wintering grounds through habitat loss and degradation. The maintenance of navigation channels by dredging, especially deep shipping channels such as those in Alabama and Mississippi, can significantly alter the natural coastal processes on inlet shorelines of nearby barrier islands, as described by Otvos (2006), Morton (2008), Otvos and Carter (2008), Beck and Wang (2009), and Stockdon *et al*. (2010). Cialone and Stauble (1998) describe the impacts of mining ebb shoals within inlets as a source of beach fill material at eight locations and provide a recommended monitoring protocol for future mining events; Dabees and Kraus (2008) also describe the impacts of ebb shoal mining in southwest Florida.

Forty-four percent of the tidal inlets within the U.S. wintering range of the piping plover have been or continue to be dredged, primarily for navigational purposes (**Table 5**). States where more than two-thirds of inlets have been dredged include Alabama (three of four), Mississippi (four of six), North Carolina (16 of 20), and Texas (13 of 18), and 16 of 21 along the Florida Atlantic coast. The dredging of navigation channels or relocation of inlet channels for erosion-control purposes contributes to the cumulative effects of inlet habitat modification by removing or redistributing the local and regional sediment supply; the maintenance dredging of deep shipping channels can convert a natural inlet that normally bypasses sediment from one shoreline to the other into a sediment sink, where sediment no longer bypasses the inlet.

Among the dredged inlets identified in Rice (2012a), dredging efforts began as early as the 1800s and continue to the present, generating long-term and even permanent effects on inlet habitat; at least 11 inlets were first dredged in the 19th century, with the Cape Fear River (North Carolina) being dredged as early as 1826 and Mobile Pass (Alabama) in 1857. Dredging can occur on an annual basis or every two to three years, resulting in continual perturbations and modifications to inlet and adjacent shoreline habitat. The volumes of sediment removed can be major, with 2.2 million cubic yards (mcy) of sediment removed on average every 1.9 years from the Galveston Bay Entrance (Texas) and 3.6 mcy of sediment removed from Sabine Pass (Texas) on average every 1.4 years (USACE 1992).

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As sand sources for beach nourishment projects have become more limited, the mining of ebb tidal shoals for sediment has increased (Cialone and Stauble 1998). This is a problem because exposed ebb and flood tidal shoals and sandbars are prime roosting and foraging habitats for piping plovers. In general, such areas are only accessible by boat; and as a result, they tend to receive less human recreational use than nearby mainland beaches. Rice (2012a) found that the ebb shoal complexes of at least 20 inlets within the wintering range of the piping plover have been mined for beach fill. Ebb shoals are especially important because they act as "sand bridges" that connect beaches and islands by transporting sediment via longshore transport from one side (updrift) to the other (downdrift) side of an inlet. The mining of sediment from these shoals upsets the inlet system equilibrium and can lead to increased erosion of the adjacent inlet shorelines (Cialone and Stauble 1998). Rice (2012a) noted that this mining of material from inlet shoals for use as beach fill is not equivalent to the natural sediment bypassing that occurs at unmodified inlets for several reasons, most notably for the massive volumes involved that are "transported" virtually instantaneously instead of gradually and continuously and for the placement of the material outside of the immediate inlet vicinity, where it would naturally bypass. The mining of inlet shoals can remove massive amounts of sediment, with 1.98 mcy mined for beach fill from Longboat Pass (Florida) in 1998, 1.7 mcy from Shallotte Inlet (North Carolina) in 2001 and 1.6 mcy from Redfish Pass (Florida) in 1988 (Cialone and Stauble 1998, USACE 2004). Cialone and Stauble (1998) found that monitoring of the impacts of ebb shoal mining has been insufficient, and in one case the mining pit was only 66% recovered after five years; they conclude that the larger the volume of sediment mined from the shoals, the larger the perturbation to the system and the longer the recovery period.

Information is limited on the effects to piping plover habitat of the deposition of dredged material, and the available information is inconsistent. Drake *et al*. (2001) concluded that the conversion of bayshore tidal flats of southern Texas mainland to dredged material impoundments results in a net loss of habitat for wintering piping plovers because such impoundments eventually convert to upland habitat. Zonick *et al*. (1998) reported that dredged material placement areas along the Gulf Intracoastal Waterway in Texas were rarely used by piping plovers, and noted concern that dredge islands block the wind-driven water flows that are critical to maintaining important shorebird habitats. Although Zdravkovic and Durkin (2011) found 200 piping plovers on the Mansfield Channel dredge material islands during a survey in late 2009, none were counted there in early 2011. By contrast, most of the sound islands where Cohen *et al*. (2008) found foraging piping plovers at Oregon Inlet, North Carolina were created by the Corps from dredged material. Another example is Pelican Island, in Corpus Christi Bay, Texas, where dredged material is consistently used by piping plovers (R. Cobb, USFWS, pers. comm. 2012a). Research is needed to understand why piping plovers use some dredge material islands, but are not regularly found using many others.

In summary, the removal of sediment from inlet complexes via dredging and sand mining for beach fill has modified nearly half of the tidal inlets within the continental wintering range of the piping plover, leading to habitat loss and degradation. Many of these inlet habitat modifications have become permanent, existing for over 100 years. The expansion of several harbors and ports to accommodate deeper draft ships poses an increasing threat as more sediment is removed from the inlet system, causing larger perturbations and longer recovery times; maintenance dredging conducted annually or every few years may prevent full recovery of the inlet system. Sand removal or sediment starvation of shoals, sandbars and adjacent shoreline habitat has resulted in habitat loss and degradation, which may reduce the system's ability to maintain a full suite of inlet habitats as sea level continues to rise at an accelerating rate. Rice (2012a) noted that the adverse impacts of this threat to piping plovers may be mitigated; however, by eliminating dredging and mining activities in inlet complexes with high habitat value, extending the interval between dredging cycles, discharging dredged material in nearshore downdrift waters so that it can accrete more naturally than when placed on the subaerial beach, and designing dredged material islands to mimic natural shoals and flats.

Inlet Stabilization and Relocation

Many navigable tidal inlets along the Atlantic and Gulf coasts are stabilized with hard structures. A description of the different types of stabilization structures typically constructed at or adjacent to inlets – jetties, terminal groins, groins, seawalls, breakwaters and revetments – can be found in Rice (2009) as well in the *Manual for Coastal Hazard Mitigation* (Herrington 2003, available online) and in *Living by the Rules of the Sea* (Bush *et al*. 1996).

The adverse direct and indirect impacts of hard stabilization structures at inlets and inlet relocations can be significant. The impacts of jetties on inlet and adjacent shoreline habitat have been described by Cleary and Marden (1999), Bush *et al*. (1996, 2001, 2004), Wamsley and Kraus (2005), USFWS (2009a), Thomas *et al*. (2011), and many others. The relocation of inlets or the creation of new inlets often leads to immediate widening of the new inlet and loss of adjacent habitat, among other impacts, as described by Mason and Sorenson (1971), Masterson *et al*. (1973), USACE (1992), Cleary and Marden (1999), Cleary and Fitzgerald (2003), Erickson *et al*. (2003), Kraus *et al*. (2003), Wamsley and Kraus (2005) and Kraus (2007).

Rice (2012a) found that, as of 2011, an estimated 54% of 221 mainland or barrier island tidal inlets in the U.S continental wintering range of the piping plover had been modified by some form of hardened structure, dredging, relocation, mining, or artificial opening or closure (**Table 5**). On the Atlantic Coast, 43% of the inlets have been stabilized with hard structures, whereas 37% were stabilized on the Gulf Coast. The Atlantic coast of Florida has 17 stabilized inlets adjacent to each other, extending between the St. John's River in Duval County and Norris Cut in Miami-Dade County, a distance of 341 miles. A shorebird would have to fly nearly 344 miles between unstabilized inlets along this stretch of coast.

The state with the highest proportion of natural, unmodified inlets is Georgia (74%). The highest number of adjacent unmodified, natural inlets is 15, which is the number of inlets found in Georgia between Little Tybee Slough at Little Tybee Island Nature Preserve and the entrance to Altamaha Sound at the south end of Wolf Island National Wildlife Refuge, a distance of approximately 54 miles. Another relatively long stretch of adjacent unstabilized inlets is in Louisiana, where 17 inlets between a complex of breaches on the West Belle Pass barrier headland (in Lafourche Parish) and Beach Prong (near the western boundary of the state

Rockefeller Wildlife Refuge) have no stabilization structures; one of these inlets (the Freshwater Bayou Canal), however, is dredged (Rice 2012a).

Unstabilized inlets naturally migrate, reforming important habitat components over time, particularly during a period of rising sea level. Inlet stabilization with rock jetties and revetments alters the dynamics of longshore sediment transport and the natural movement and formation of inlet habitats such as shoals, unvegetated spits and flats. Once a barrier island becomes "stabilized" with hard structures at inlets, natural overwash and beach dynamics are restricted, allowing encroachment of new vegetation on the bayside that replaces the unvegetated (open) foraging and roosting habitats that plovers prefer. Rice (2012a) found that 40% (89 out of 221) of the inlets open in 2011 have been stabilized in some way, contributing to habitat loss and degradation throughout the wintering range.
Accelerated erosion may compound future habitat loss, depending on the degree of sea level rise (Titus *et al*. 2009). Due to the complexity of impacts associated with projects such as jetties and groins, Harrington (2008) noted the need for a better understanding of potential effects of inlet-related projects, such as jetties, on bird habitats.

Relocation of tidal inlets also can cause loss and/or degradation of piping plover habitat. Although less permanent than construction of hard structures, the effects of inlet relocation can persist for years. For example, December-January surveys documented a continuing decline in wintering plover numbers from 20 birds pre-project (2005-2006) to three birds during the 2009-2011 seasons (SCDNR 2011). Subsequent decline in the wintering population on Kiawah is strongly correlated with the decline in polychaete worm densities, suggesting that plovers emigrated to other sites as foraging opportunities in these habitats became less profitable (SCDNR 2011). At least eight inlets in the migration and wintering range have been relocated; a new inlet was cut and the old inlet was closed with fill. In other cases, inlets have been relocated without the old channels being artificially filled (**Table 5** and Rice 2012a).

The artificial opening and closing of inlets typically creates very different habitats from those found at inlets that open or close naturally (Rice 2012a). Rice (2012a) found that 30 inlets have been artificially created within the migration and wintering range of the piping plover, including 10 of the 21 inlets along the eastern Florida coast (**Table 5**). These artificially created inlets tend to need hard structures to remain open or stable, with 20 of the 30 (67%) of them having hard structures at present. An even higher number of inlets (64) have been artificially closed, the majority in Louisiana (**Table 5**). One inlet in Texas was closed as part of the Ixtoc oil spill response efforts in 1979 and 32 were closed as part of Deepwater Horizon oil spill response efforts in 2010-2011. Of the latter, 29 were in Louisiana, two in Alabama and one in Florida. To date only one of these inlets, West (Little Lagoon) Pass in Gulf Shores, Alabama, has been reopened, and the rest remain closed with no plans to reopen any of those identified by Rice (2012a). Most other artificial inlet closures in Louisiana are part of barrier island restoration projects, because much of that state's barrier islands are disintegrating (Otvos 2006, Morton 2008, Otvos and Carter 2008). Inlets closed during coastal restoration projects in Louisiana are purposefully designed to approximate low, wide naturally closed inlets and to allow overwash in the future. By contrast, most artificially closed inlets have higher elevations and tend to have a constructed berm and dune system. Overwash may occur periodically at a naturally closed inlet but is prevented at an artificially closed inlet by the constructed dune ridge, hard structures, or sandbags (Rice 2012a).

The construction of jetties, groins, seawalls and revetments at inlets leads to habitat loss and both direct and indirect impacts to adjacent shorelines. Rice (2012a) found that these structures result in long-term effects, with at least 13 inlets across six of the eight states having hard structures initially constructed in the 19th century. The cumulative effects are ongoing and increasing in intensity, with hard structures built as recently as 2011 and others proposed for 2012.

With sea level rising and global climate change altering storm dynamics, pressure to modify the remaining half of sandy tidal inlets in the range is likely to increase, notwithstanding that this would be counterproductive to the climate change adaptation strategies recommended by the USFWS (2010d), CCSP (2009), Williams and Gutierrez (2009), Pilkey and Young (2009), and many others.

Groins

Groins pose an ongoing threat to piping plover beach habitat within the continental wintering range. Groins are hard structures built perpendicular to the shoreline (sometimes in a T-shape), designed to trap sediment traveling in the littoral drift and to slow erosion on a particular stretch of beach or near an inlet. "Leaky" groins, also known as permeable or porous groins, are low-crested structures built like typical groins but which allow some fraction of the littoral drift or longshore sediment transport to pass through the groin. They have been used as terminal groins near inlets or to hold beach fill in place for longer durations. Although groins can be individual structures, they are often clustered along the shoreline in "groin fields." Because they intentionally act as barriers to longshore sand transport, groins cause downdrift erosion, which degrades and fragments sandy beach habitat for the piping plover and other wildlife. The resulting beach typically becomes scalloped in shape, thereby fragmenting plover habitat over time.

Groins and groin fields are found throughout the southeastern Atlantic and Gulf Coasts and are present at 28 of 221 sandy tidal inlets (Rice 2012a). Leaky terminal groins have been installed at the south end of Amelia Island, Florida, the west end of Tybee Island, Georgia, and the north end of Hilton Head Island, South Carolina. Permeable or leaky groins have also been constructed on the beaches of Longboat Key and Naples, Florida, and terminal groins were approved in 2011 for use in up to four inlet locations in North Carolina (reversing a nearly 30-year prohibition on hard stabilization structures in that state).

Although most groins were in place before the piping plover's 1986 ESA listing, new groins continue to be installed, perpetuating the threat to migrating and wintering piping plovers. Two groins were built in South Carolina between 2006 and 2010, bringing the statewide total to 165 oceanfront groins (SC DHEC 2010). Eleven new groins were built in Florida between 2000 and 2009. The East Pass Navigation Project in Okaloosa County, Florida (USFWS 2009a) illustrates the negative impacts to plover habitat that can be associated with groins, which are often built as one component of a much larger shoreline or inlet stabilization project. The East Pass Navigation Project includes two converging jetties, one with a groin at the end, with dredged material placed on either side to stabilize the jetties; minimal piping plover foraging habitat remains due to changed inlet morphology. As sea level rises at an accelerating rate, the threat of habitat loss, fragmentation and degradation from groins and groin fields may increase as communities and beachfront property owners seek additional ways to protect infrastructure and property.

Seawalls and Revetments

Seawalls and revetments are hard vertical structures built parallel to the beach in front of buildings, roads, and other facilities. Although they are intended to protect human infrastructure from erosion, these armoring structures often accelerate erosion by causing scouring both in front of and downdrift from the structure, which can eliminate intertidal plover foraging and adjacent roosting habitat. Physical characteristics that determine microhabitats and biological communities can be altered after installation of a seawall or revetment, thereby depleting or changing composition of benthic communities that serve as the prey base for piping plovers (see *Loss of Macroinvertebrate Prey Base due to Shoreline Stabilization*). Dugan and Hubbard (2006) found in a California study that intertidal zones were narrower and fewer in the presence of armoring, armored beaches had significantly less macrophyte wrack, and shorebirds responded with significantly lower abundance (more than three times lower) and species richness (2.3 times lower) than on adjacent unarmored beaches. As sea level rises, seawalls will prevent the coastline from moving inland, causing loss of intertidal foraging habitat (Galbraith *et al*. 2002, Defeo *et al*. 2009). Geotubes (long cylindrical bags made of high-strength permeable fabric and filled with sand) are less permanent alternatives, but they prevent overwash and thus the natural production of sparsely vegetated habitat.

Rice (2012b) found that at least 230 miles of beach habitat has been armored with hard erosion-control structures. Data were not available for all areas, so this number is a minimum estimate of the length of habitat that has been directly modified by armoring. Out of 221 inlets surveyed, 89 were stabilized with some form of hard structure, of which 24 had revetments or seawalls along their shorelines (Rice 2012b). The Texas coast is armored with nearly 37 miles of seawalls, bulkheads and revetments, the mainland Mississippi coast has over 45 miles of armoring, the Florida Atlantic coast has at least 58 miles, and the Florida Gulf coast over 59 miles (Rice 2012b). Shoreline armoring has modified plover beachfront habitat in all states, but Alabama (4.7 miles), Georgia (10.5 miles) and Louisiana (15.9 miles) have the fewest miles of armored beaches.

Although North Carolina has prohibited the use of hard erosion-control structures or armoring since 1985 the "temporary" installation of sandbag revetments is allowed. As a result the precise length of armored sandy beaches in North Carolina is unknown, but at least 350 sandbag revetments have been constructed (Rice 2012b). South Carolina also limits the installation of some types of new armoring but already has 24 miles (27% of the developed shoreline or 13% of the entire shoreline) armored with some form of shore-parallel erosion-control structure (SC DHEC 2010).

The repair of existing armoring structures and installation of new structures continues to degrade, destroy, and fragment beachfront plover habitat throughout its continental wintering range. As sea level rises at an accelerating rate, the threat of habitat loss, fragmentation and degradation from hard erosion-control structures is likely to increase as communities and property owners seek to protect their beachfront development. As coastal roads become threatened by rising sea level and increasing storm damage, additional lengths of beachfront habitat may be modified by riprap, revetments, and seawalls.

Sand Placement Projects

Sand placement projects threaten the piping plover and its habitat by altering the natural, dynamic coastal processes that create and maintain beach strand and bayside habitats, including the habitat components that piping plovers rely upon. Although specific impacts vary depending on a range of factors, so-called "soft stabilization" projects may directly degrade or destroy roosting and foraging habitat in several ways. Beach habitat may be converted to an artificial berm that is densely planted in grass, which can in turn reduce the availability of roosting habitat. Over time, if the beach narrows due to erosion, additional roosting habitat between the berm and the water can be lost. Berms can also prevent or reduce the natural overwash that creates and maintains sparsely vegetated roosting habitats. The growth of vegetation resulting from impeding the natural overwash can also reduce the availability of bayside intertidal feeding habitats.

Overwash is an essential process, necessary to maintain the integrity of many barrier islands and to create new habitat (Donnelly *et al*. 2006). In a study on the Outer Banks of North Carolina, Smith *et al*. (2008) found that human "modifications to the barrier island, such as construction of barrier dune ridges, planting of stabilizing vegetation, and urban development, can curtail or even eliminate the natural, self-sustaining processes of overwash and inlet dynamics." They also found that such modifications led to island narrowing from both oceanside and bayside erosion. Lott (2009) found a strong negative correlation between ocean shoreline sand placement projects and the presence of piping and snowy plovers in the Panhandle and southwest Gulf Coast regions of Florida.

Sand placement projects threaten migration and wintering habitat of the piping plover in every state throughout the range (**Table 4**). At least 684.8 miles (32%) of sandy beach habitat in the continental wintering range of the piping plover have received artificial sand placement via dredge disposal activities, beach nourishment or restoration, dune restoration, emergency berms, inlet bypassing, inlet closure and relocation, and road reconstruction projects. In most areas, sand placement projects are in developed areas or adjacent to shoreline or inlet hard stabilization structures in order to address erosion, reduce storm damages, or ameliorate sediment deficits caused by inlet dredging and stabilization activities.

The beaches along the mainland coast of Mississippi are the most modified by sand placement activities with at least 85% affected (**Table 4**). Of the oceanfront beaches, the Atlantic coast of Florida has had the highest proportion (at least 51%) of beaches modified by sand placement activities. Approximately 47% of Florida's sandy beach coastline has received sand placement of some type, with many areas receiving fill multiple times from dredge disposal, emergency berms, beach nourishment, dune restoration and other modifications (Rice 2012b).

In Louisiana, the sustainability of the coastal ecosystem is threatened by the inability of the barrier islands to maintain geomorphologic functionality. The state's coastal systems are starved for sediment sources (USACE 2010). Consequently, most of the planned sediment placement projects in Louisiana are conducted as environmental restoration projects by various Federal and State agencies because without the sediment many areas would erode below sea level. Several Louisiana Coastal Wetland Planning, Protection, and Restoration Act projects have been constructed on portions of undeveloped islands within the Terrebonne Basin to restore and maintain the diverse functions of those barrier island habitats (USFWS 2010). Altogether over 60 miles of sandy beaches have been modified with sand placement projects in Louisiana, both through restoration projects and in response to the Deepwater Horizon oil spill (Rice 2012b).

Both the number and the size of sand projects along the Atlantic and Gulf coasts are increasing (Trembanis *et al*. 1998), and these projects are increasingly being chosen as a means to combat sea level rise and related beach erosion problems (Klein *et al*. 2001).

Loss of Macroinvertebrate Prey Base due to Shoreline Stabilization

Wintering and migrating piping plovers depend on the availability and abundance of macroinvertebrates as an important food item. Studies of invertebrate communities have found that communities are richer (greater total abundance and biomass) on protected (bay or lagoon) intertidal shorelines than on exposed ocean beach shorelines (McLachlan 1990, Cohen *et al*. 2006, Defeo and McLachlan 2011). Polychaete worms tend to have a more diverse community and be more abundant in more protected shoreline environments, and mollusks and crustaceans such as amphipods thrive in more exposed shoreline environments (McLachlan and Brown 2006). Polychaete worms comprise the majority of the shorebird diet (Kalejta 1992, Mercier and McNeil 1994, Tsipoura and Burger 1999, Verkuil *et al*. 2006); and of the piping plover diet in particular (Hoopes 1993, Nicholls 1989, Zonick and Ryan 1996).

The quality and quantity of the macroinvertebrate prey base is threatened by shoreline stabilization activities, including the approximately 685 miles of beaches that have received sand placement of various types. The addition of dredged sediment can temporarily affect the benthic fauna of intertidal systems. Invertebrates may be crushed or buried during project construction. Although some benthic species can burrow through a thin layer of additional sediment (38-89 cm for different species), thicker layers (i.e., >1 meter) are likely to smother these sensitive benthic organisms (Greene 2002). Numerous studies of such effects indicate that the recovery of benthic fauna after beach nourishment or sediment placement projects can take anywhere from six months to two years, and possibly longer in extreme cases (Thrush *et al*. 1996, Peterson *et al*. 2000, Zajac and Whitlatch 2003, Bishop *et al*. 2006, Peterson *et al*. 2006).

Invertebrate communities may also be affected by changes in the physical environment resulting from shoreline stabilization activities that alter the sediment composition or degree of exposure. For example, SCDNR (2011) found the decline in piping plovers to be strongly correlated with a decline in polychaete densities on the east end of Kiawah Island, South Carolina, following an inlet relocation project in 2006. Similar results were documented on Bird Key, South Carolina, in 2006 when rapid habitat changes occurred within the sheltered lagoon habitat following dredge disposal activities, and piping plovers shifted to more exposed areas. Their diet also appeared to have shifted to haustoriid amphipods, based on analysis of fecal samples containing pieces of *Neohaustorius schmitzi*, *Lepidactylus dytiscus*, and *Acanthohaustorius* sp., which were also found during the invertebrate sampling in both locations (SCDNR 2011).

Shoreline armoring with hard stabilization structures such as seawalls and revetments can also alter the degree of exposure of the macroinvertebrate prey base by modifying the beach and intertidal geomorphology, or topography. Seawalls typically result in the narrowing and steepening of the beach and intertidal slope in front of the structure, eventually leading to complete loss of the dry and intertidal beach as sea level continues to rise (Pilkey and Wright 1988, Hall and Pilkey 1991, Dugan and Hubbard 2006, Defeo *et al*. 2009, Kim *et al*. 2011).

Sand placement projects bury the natural beach with up to millions of cubic yards of new sediment, and grade the new beach and intertidal zone with heavy equipment to conform to a predetermined topographic profile. This can lead to compaction of the sediment (Nelson *et al*. 1987, USACE 2008, Defeo *et al*. 2009). If the material used in a sand placement project does not closely match the native material on the beach, the sediment incompatibility may result in modifications to the macroinvertebrate community structure, because several species are sensitive to grain size and composition (Rakocinski *et al*. 1996; Peterson *et al*. 2000, 2006; Peterson and Bishop 2005; Colosio *et al*. 2007; Defeo *et al*. 2009).

Delayed recovery of the benthic prey base or changes in their communities due to physical habitat changes may affect the quality of piping plover foraging habitat. The duration of the impact can adversely affect piping plovers because of their high site fidelity. Although recovery of invertebrate communities has been documented in many studies, sampling designs have typically been inadequate and have only been able to detect large-magnitude changes (Schoeman *et al*. 2000, Peterson and Bishop 2005). Therefore, uncertainty persists about the impacts of various projects to invertebrate communities and how these impacts affect shorebirds, particularly the piping plover. Rice (2009) has identified several conservation measures that can avoid and minimize some of the known impacts.

Invasive Vegetation

The spread of invasive plants into suitable wintering piping plover habitat is a relatively recently identified threat (USFWS 2012). Such plants tend to reproduce and spread quickly and to exhibit dense growth habits, often outcompeting native plants. Uncontrolled invasive plants can shift habitat from open or sparsely vegetated sand to dense vegetation, resulting in the loss or degradation of piping plover roosting habitat, which is especially important during high tides and migration periods. The propensity of invasive species to spread, and their tenacity once established, make them a persistent threat that is only partially countered by increasing landowner awareness and willingness to undertake eradication activities.

Many invasive species are either currently affecting or have the potential to affect coastal beaches and thus plover habitat. Beach vitex (*Vitex rotundifolia*) is a woody vine introduced into the southeastern U.S. as a dune stabilization and ornamental plant which has spread to coastal communities throughout the southeastern U.S. from Virginia to Florida, and west to Texas (Westbrooks and Madsen 2006). Hundreds of beach vitex occurrences and targeted eradication efforts in North and South Carolina and a small number of known locations in Georgia and Florida are discussed in the 5-Year Review (USFWS 2009b). Crowfootgrass (*Dactyloctenium aegyptium*), which grows invasively along portions of the Florida coastline, forms thick bunches or mats that can change the vegetative structure of coastal plant communities and thus alter shorebird habitat (USFWS 2009b, Florida Exotic Pest Plant Council 2009). Australian pine (*Casuarina equisetifolia*) affects piping plovers and other shorebirds by encroaching on foraging and roosting habitat (Stibolt 2011); it may also provide perches for avian predators. Japanese sedge (*Carex kobomugi*), which aggressively encroaches into sand beach habitats (USDA plant profile website), was documented in Currituck County, North Carolina, in the mid-1970s and as recently as 2003 on Currituck National Wildlife Refuge (J. Gramling, Department of Biology, The Citadel, pers. comm. 2011), at two sites where migrating piping plovers have also been documented. Early detection and rapid response are the keys to controlling this and other invasive plants (R. Westbrooks, U.S. Geological Survey, pers. comm. 2011).

Defeo *et al*. (2009) cite biological invasions of both plants and animals as global threats to sandy beaches, with the potential to alter the food web, nutrient cycling and invertebrate assemblages. Although the extent of the threat is uncertain, this may be due to poor survey coverage more than an absence of invasions.

Wrack Removal and Beach Cleaning

Wrack on beaches and baysides provides important foraging and roosting habitat for piping plovers (Drake 1999, Smith 2007, Maddock *et al*. 2009, Lott *et al*. 2009b; see also discussion of piping plover use of wrack substrates in *Habitat Use*) and for many other shorebirds. Because shorebird numbers are positively correlated both with wrack cover and the biomass of their invertebrate prey that feed on wrack (Tarr and Tarr 1987, Hubbard and Dugan 2003, Dugan *et al*. 2003), beach grooming has been shown to decrease bird numbers (Defeo *et al*. 2009).
It is increasingly common for beach-front communities to carry out “beach cleaning” and “beach raking” activities. Beach cleaning is conducted on private beaches, where piping plover use is not well documented, and on some municipal or county beaches used by piping plovers. Most wrack removal on state and Federal lands is limited to post-storm cleanup and does not occur regularly. Wrack removal and beach raking both occur on the Gulf beach side of the developed portion of South Padre Island in the Lower Laguna Madre in Texas, where plovers have been documented during both the migratory and wintering periods (Zdravkovic and Durkin 2011). Wrack removal and other forms of beach cleaning have been the subject of formal consultations between the U.S. Army Corps of Engineers, municipalities, and Service in Neuces County, Texas (USFWS 2008b, 2009c).

Although beach cleaning and raking machines effectively remove human-made debris, these efforts also remove accumulated wrack, topographic depressions, emergent foredunes and hummocks, and sparse vegetation nodes used by roosting and foraging piping plovers (Nordstrom 2000, Dugan and Hubbard 2010). Removal of wrack also reduces or eliminates natural sand-trapping, further destabilizing the beach. Cathcart and Melby (2009) found that beach grooming and raking beaches “fluffs the sand” whereas heavy equipment compacts the sand below the top layer; the fluffed sand is then more vulnerable to erosion by storm water runoff and wind. These authors found that beach raking and grooming practices on mainland

Mississippi beaches "exacerbate the erosion process and shorten the time interval between renourishment projects" (Cathcart and Melby 2009). Furthermore, the sand adhering to seaweed and trapped in the cracks and crevices of wrack also is lost to the beach when the wrack is removed. Although the amount of sand lost during a single sweeping activity may be small, over a period of years this loss could be significant (Neal *et al*. 2007).

Tilling beaches to reduce soil compaction, which is sometimes required by the Service for sea turtle protection after beach nourishment activities, has similar impacts to those described above. In northwest Florida, tilling on public lands is currently conducted only if the land manager determines that it is necessary. Where tilling is needed, adverse effects are reduced by Florida USFWS sea turtle protection provisions that require tilling to be above the primary wrack line, rather than within it.

As of 2009, the Florida Department of Environmental Protection's Beaches and Coastal Management Systems section had issued 117 permits allowing multiple entities to conduct beach raking or cleaning operations. The Florida Department of Environmental Protection estimated that 240 of 825 miles (29%) of sandy beach shoreline in Florida are cleaned or raked on varied schedules, i.e., daily, weekly, monthly (L. Teich, Florida DEP, pers. comm. 2009). Beach cleaning along 45 miles of coastline in Nueces, Kleberg, and Cameron Counties in Texas was addressed in five USFWS biological opinions completed between 2008 and 2012 (Cobb pers. comm. 2012c).

Dugan and Hubbard (2010), studying beach grooming activities on the beaches and dunes of southern California, concluded that "beach grooming has contributed to widespread conversion of coastal strand ecosystems to unvegetated sand" by removing wrack cover, increasing the transport of windblown sediment, lowering the seed bank and the survival and reproduction of native plants, and decreasing native plant abundance and richness. They argue that conserving beach ecosystems by reducing beach grooming and raking activities "could help retain sediment, promote the formation of dunes, and maintain biodiversity, wildlife, and human use in the face of rising sea level (Dugan and Hubbard 2010)."

Accelerating Sea Level Rise and other Climate Change Impacts

Accelerating sea level rise poses a threat to piping plovers during the migration and wintering portions of their life cycle. As noted in the previous section, threats from sea level rise are tightly intertwined with artificial coastal stabilization activities that modify and degrade habitat. Potential effects of storms, which could increase in frequency or intensity due to climate change, are discussed in the Storm Events section. If climate change increases the frequency or magnitude of extreme temperatures (see discussion in Severe Cold Weather), piping plover survival rates may be affected. Other potential adverse and beneficial climate change-related effects (e.g., changes in the composition or availability of prey, emergence of new diseases, fewer periods of severe cold weather) are poorly understood, but cannot be discounted.

Numerous studies have documented accelerating rise in sea levels worldwide (Rahmstorf *et al*. 2007, Douglas *et al*. 2001 as cited in Hopkinson *et al*. 2008, CCSP 2009, Pilkey and Young 2009, Vermeer and Rahmstorf 2009, Pilkey and Pilkey 2011). Predictions include a sea level rise of between 50 and 200 cm above 1990 levels by the year 2100 (Rahmstorf 2007, Pfeffer *et al*. 2008, Vermeer and Rahmstorf 2009, Grinsted *et al*. 2010, Jevrejeva *et al*. 2010) and potential conversion of as much as 33% of the world's coastal wetlands to open water by 2080 (IPCC 2007a, CCSP 2008). Potential effects of sea level rise on piping plover roosting and foraging habitats may vary regionally due to subsidence or uplift, the geological character of the coast and nearshore, and the influence of management measures such as beach nourishment, jetties, groins, and seawalls (CCSP 2009, Galbraith *et al*. 2002, Gutierrez *et al*. 2011). Sea level rise along the U.S. Gulf Coast exceeded the global average by 13-15 cm because coastal lands there are subsiding (EPA 2009). The rate of sea level rise in Louisiana is particularly high (Louisiana Coastal Wetlands Conservation and Restoration Task Force and the Wetlands Conservation and Restoration Authority 1998). Sediment compaction and oil and gas extraction compound tectonic subsidence along the Gulf of Mexico coastline (Penland and Ramsey 1990, Morton *et al*. 2003, Hopkinson *et al*. 2008).

Low elevations and proximity to the coast make all nonbreeding piping plover foraging and roosting habitats vulnerable to the effects of rising sea level. Areas with small tidal ranges are the most vulnerable to loss of intertidal wetlands and flats (EPA 2009). Sea level rise was cited as a contributing factor in the 68% decline in tidal flats and algal mats in the Corpus Christi, Texas region (i.e., Lamar Peninsula to Encinal Peninsula) between the 1950s and 2004 (Tremblay *et al*. 2008). Mapping by Titus and Richman (2001) showed that more than 80% of the lowest land along the Atlantic and Gulf coasts was in Louisiana, Florida, Texas, and North Carolina. Gutierrez *et al*. (2011) found that along the Atlantic coast, the central and southern Florida coast is the most likely Atlantic portion of the wintering and migration range to experience moderate to severe erosion with sea level rise.

Inundation of piping plover habitat by rising seas could lead to permanent loss of habitat, especially if those shorelines are armored with hardened structures (Brown and McLachlan 2002, Dugan and Hubbard 2006, Fish *et al*. 2008, Defeo *et al*. 2009). Overwash and sand migration are impeded on the developed portions of sandy ocean beaches (Smith *et al*. 2008) that comprise 40% of the U.S. nonbreeding range (Rice 2012b). As the sea level rises, the ocean-facing beaches erode and attempt to migrate inland. Buildings and artificial sand dunes then prevent sand from washing back toward the lagoons (i.e., bayside), and the lagoon side becomes increasingly submerged during extreme high tides (Scavia *et al*. 2002). Barrier beach shorebird habitat and natural features that protect mainland developments are both diminished as a result.

Modeling by Galbraith *et al*. (2002) for three sea level rise scenarios at five important U.S. shorebird staging and wintering sites predicted aggregate loss of 20-70% of current intertidal foraging habitat. The most severe losses were projected at sites where the coastline is unable to move inland due to steep topography or seawalls. Of five study sites, the model predicted the lowest loss of intertidal shorebird foraging habitat at Bolivar Flats, Texas (a designated piping plover critical habitat unit) by 2050 because the habitat at that site will be able to migrate inland in response to rising sea level. The potential for such barrier island migration with rising sea level is most likely in the 42% of plover's U.S. nonbreeding range that is currently preserved from development (Rice 2012b). Although habitat losses in some areas are likely to be offset by gains in other locations, Galbraith *et al*. (2002) noted that time lags between these losses and the creation of replacement habitat elsewhere may have serious adverse effects on shorebird populations. Furthermore, even if piping plovers are able to move their wintering locations in response to accelerated habitat changes, there could be adverse effects on the birds' survival rates or subsequent productivity.

In summary, the magnitude of threats from sea level rise is closely linked to threats from shoreline development and artificial stabilization. These threats will be perpetuated in places where damaged structures are repaired or replaced, exacerbated where the height and strength of structures are increased, and increased at locations where development and coastal stabilization is expanded. Sites that are able to adapt to sea level rise are likely to become more important to piping plovers as habitat at developed or stabilized sites degrades.

Weather events

Storm Events

Storms are an integral part of the natural processes that form coastal habitats used by migrating and wintering piping plovers, and positive effects of storm-induced overwash and vegetation removal have been noted in portions of the wintering range. For example, biologists reported piping plover use of newly created habitats at Gulf Islands National Seashore in Florida within six months of overwash events that occurred during the 2004 and 2005 hurricane seasons (M. Nicholas, Gulf Islands National Seashore, pers. comm. 2005). Hurricane Katrina created a new inlet and improved habitat conditions on some areas of Dauphin Island, Alabama, but subsequent localized storms contributed to habitat loss there (D. LeBlanc, USFWS, pers. comm. 2009) and the inlet was subsequently closed with a rock dike as part of Deepwater Horizon oil spill response efforts (Rice 2012a). Following Hurricane Ike in 2008, Arvin (2009) reported decreased numbers of piping plovers at some heavily eroded Texas beaches in the center of the storm impact area and increases in plover numbers at sites about 100 miles to the southwest. Piping plovers were observed later in the season using tidal lagoons and pools that Hurricane Ike created behind the eroded beaches (Arvin 2009).

Adverse effects attributed to storms alone are sometimes actually due to a combination of storms and other environmental changes or human use patterns. For example, four hurricanes between 2002 and 2005 are often cited in reference to rapid erosion of the Chandeleur Islands, a chain of low-lying islands in Louisiana where the 1991 International Piping Plover Winter Census (Haig and Plissner 1992) tallied more than 350 birds. Comparison of imagery taken three years before and again several days after Hurricane Katrina found that the Chandeleur Islands had lost 82% of their combined surface area (Sallenger 2010). A review of aerial photographs taken before the 2006 Census suggested that little piping plover habitat remained (Elliott-Smith *et al*. 2009). However, Sallenger *et al*. (2009) noted that habitat changes in the Chandeleur Islands stem not only from the effects of these storms, but rather from the combined effects of the storms, and more than a thousand years of diminishing sand supply and sea level rise. Although the Chandeleur Islands marsh platform continued to erode for 22 months post-Katrina, some sand was released from the marsh sediments which in turn created beaches, spits, and welded swash bars that advanced the shoreline seaward. Despite the effects of intense erosion, the Chandeleur Islands are still providing high quality shorebird habitat in the form of sand flats, spits, and beaches used by substantial numbers of piping plovers (Catlin *et al*. 2011), a scenario that could continue if restoration efforts are sustainable and successful from a shorebird perspective (USACE 2010).

Storm-induced adverse effects include post-storm acceleration of human activities such as beach nourishment, sand scraping, closure of new inlets, and berm and seawall construction. As discussed previously, such stabilization activities can result in the loss and degradation of feeding and resting habitats. Land managers sometimes face public pressure after big storm events to plant vegetation, install sandfences, and bulldoze artificial “dunes.” For example, national wildlife refuge managers sometimes receive pressure from local communities to “restore” the beach and dunes following blowouts from storm surges that create the overwash foraging habitat preferred by plovers (C. Hunter, USFWS, pers. comm. 2011). At least 64 inlets have been artificially closed, the vast majority of them shortly after opening in storm events (**Table 5**). Storms also can cause widespread deposition of debris along beaches. Subsequent removal of this debris often requires large machinery that in turn can cause extensive disturbance and adversely affect habitat elements such as wrack. Challenges associated with management of public use can grow when storms increase access (e.g., merger of Pelican Island with Dauphin Island in Alabama following a 2007 storm (Gibson et. al. 2009, D. LeBlanc pers. comm. 2009)).

Some available information indicates that birds may be resilient, even during major storms, and move to unaffected areas without harm. Other reports suggest that birds may perish in or following storm events. Noel and Chandler (2005) suspected that changes in habitat caused by multiple hurricanes along the Georgia coastline altered the spatial distribution of piping plovers and may have contributed to the winter mortality of three individuals. Wilkinson and Spinks (1994) suggested that low plover numbers in South Carolina in January 1990 could have been partially influenced by effects on habitat from Hurricane Hugo the previous fall, while Johnson and Baldassarre (1988) found a redistribution of piping plovers in Alabama following Hurricane Elena in 1985.

Climate change studies indicate a trend toward increasing numbers and intensity of hurricane events (Emanuel 2005, Webster *et al*. 2005). Combined with the predicted effects of sea level rise, this trend indicates potential for increased cumulative impact of future storms on habitat. Major storms can create or enhance piping plover habitat while causing localized losses elsewhere in the wintering and migration range.

Severe Cold Weather

Several sources suggest the potential for adverse effects of severe winter cold on survival of piping plovers. The Atlantic Coast piping plover recovery plan mentioned high mortality of coastal birds and a drop from approximately 30-40 to 15 piping plovers following an intense 1989 snowstorm along the North Carolina coast (Fussell 1990). A preliminary analysis of survival rates for Great Lakes piping plovers found that the highest variability in survival occurred in spring and correlated positively with minimum daily temperature (weighted mean based on proportion of the population wintering near five weather stations) during the preceding winter (E. Roche, Univ. of Tulsa, pers. comm. 2010 and 2012). Catlin (pers. comm. 2012b) reported that the average mass of ten piping plovers captured in Georgia during unusually cold weather in December 2010 was 5.7 grams (g) less than the average for nine birds captured in October of the same year (46.6 g and 52.4 g, respectively; $p = 0.003$).

Disturbance from Recreation Activities

Increasing human disturbance is a major threat to piping plovers in their coastal migration and wintering range (USFWS 2012). Intense human disturbance in shorebird winter habitat can be functionally equivalent to habitat loss if the disturbance prevents birds from using an area (Goss-Custard *et al*. 1996). Nicholls and Baldassarre (1990a) found less people and off-road vehicles at sites where nonbreeding piping plovers were present than at sites without piping plovers. Pfister *et al*. (1992) implicate anthropogenic disturbance as a factor in the long-term decline of migrating shorebirds at staging areas. Disturbance can cause shorebirds to spend less time roosting or foraging and more time in alert postures or fleeing from the disturbances (Burger 1991, 1994; Elliott and Teas 1996; Lafferty 2001a, 2001b; Thomas *et al*. 2003). Shorebirds that are repeatedly flushed in response to disturbance expend energy on costly short flights (Nudds and Bryant 2000).

Shorebirds are more likely to flush from the presence of dogs than people, and breeding and nonbreeding shorebirds react to dogs from farther distances than people (Lafferty 2001a, 2001b; Lord *et al*. 2001, Thomas *et al*. 2003). Hoopes (1993) found that dogs flush breeding piping plovers from further distances than people and that both the distance the plovers move and the duration of their response is greater. Foraging shorebirds at a migratory stopover on Delaware Bay, New Jersey responded most strongly to dogs compared with other disturbances; shorebirds often failed to return within ten minutes after the dog left the beach (Burger *et al*. 2007). Dogs off-leash were disproportionate sources of disturbance in several studies (Thomas *et al*. 2003, Lafferty 2001b), but leashed dogs also disturbed shorebirds. Pedestrians walking with dogs often go through flocks of foraging and roosting shorebirds; some even encourage their dogs to chase birds.

Off-road vehicles can disrupt piping plover's normal behavior patterns. The density of off-road vehicles negatively correlated with abundance of piping plovers on the ocean beach in Texas (Zonick 2000). Cohen *et al*. (2008) found that radio-tagged wintering piping plovers using ocean beach habitat at Oregon Inlet in North Carolina were far less likely to use the north side of the inlet where off-road vehicle use was allowed. Ninety-six percent of piping plover detections occurred on the south side of the inlet even though it was more than four times farther away from foraging sites, prompting a recommendation that controlled management experiments be conducted to determine if recreational disturbance drives roost site selection (Cohen *et al*. 2008). Zdravkovic and Durkin (2011) stated that Laguna Madre Gulf beaches are considered part of the Texas state highway system and are severely impacted by unrestricted public recreational off-road vehicle use.

In a study of migrating shorebirds in Maryland, Forgues (2010) found that shorebird abundance declined with increased off-road vehicle frequency, as did the number and size of roosts. Migrants spent less time foraging in the presence of vehicles. In a before-after control-impact experiment, densities of three focal species were significantly reduced after a vehicle closure was lifted, while densities outside the closure zone exhibited little change; densities of two other species also decreased more in the area where the closure was removed, but the difference was not significant (Forgues 2010). In North Carolina, a before-after control-impact experiment using the undisturbed plots as the controls found that vehicle disturbance decreased abundance of shorebirds and altered their habitat use during fall migration (Tarr 2008).

Recreational activities, especially off-road vehicles, may degrade piping plover habitat. Tires that crush wrack into the sand render it unavailable as a roosting habitat or foraging substrate (Goldin 1993, Hoopes 1993). At four study beaches in New York and Massachusetts, Kluft and Ginsberg (2009) found that abundance of invertebrates in pitfall trap samples and abundance of wrack was higher on vehicle-free beaches, although invertebrate abundance in wrack clumps and cores taken below them did not show consistent differences between areas open and closed to vehicles. Off-road vehicles significantly lessened densities of invertebrates on intertidal flats on the Cape Cod National Seashore in Massachusetts (Wheeler 1979). In eastern Australia, off-road vehicles use has been documented as a significant cause of invertebrate mortality on beaches (Schlacher *et al*. 2008a, 2008b). Results of Schlacher and Thompson (2012) in eastern Australia also suggest that channeling major pedestrian access points away from key shorebird habitat may enhance protection of their prey base.

Various local and regional examples also illustrate threats from recreation. On a 12-kilometer stretch of Mustang Island in Texas, Foster *et al*. (2009) observed a 25% decline in piping plover abundance and a simultaneous five-fold increase in human use over a 29-year study period, 1979 – 2007. This trend was marginally significant, but declines in two other plover species were significant; declining shorebird abundance was attributed to a combination of human disturbance and overall declines in shorebird populations (Foster *et al*. 2009). In South Carolina, almost half of sites with five or more piping plovers had ten or more people present during surveys conducted in 2007-2008 and more than 60% allow dogs (Maddock and Bimbi unpubl. data). Zdravkovic and Durkin (2011) noted disturbance to piping plovers in Texas from kite boarding, windsurfing, and horseback riding.

LeDee *et al*. (2010) surveyed land managers of designated critical habitat sites across seven southern states and documented the extent of beach access and recreation. All but four of the 43 reporting sites owned or managed by Federal, state, and local governmental agencies or by non-governmental organizations allowed public beach access year-round (88% of the sites). At the sites allowing public access, 62% of site managers reported more than 10,000 visitors during September-March, and 31% reported more than 100,000 visitors in this period. However, more than 80% of the sites allowing public access did not allow vehicles on the beach and half did not allow dogs during the winter season.

Oil Spills and Other Contaminants

Piping plovers may accumulate contaminants from point and non-point sources at migratory and wintering sites. Depending on the type and degree of contact, contaminants can have lethal and sub-lethal effects on birds, including behavioral impairment, deformities, and impaired reproduction (Rand and Petrocelli 1985, Gilbertson *et al*. 1991, Hoffman *et al*. 1996). Notwithstanding documented cases of lightly oiled piping plovers that have survived and successfully reproduced (Amirault-Langlais *et al*. 2007, A. Amos, University of Texas Marine

Science Institute, pers. comm. 2009, 2012), contaminants have both the potential to cause direct toxicity to individual birds and to negatively impact their invertebrate prey base (Chapman 1984, Rattner and Ackerson 2008). Piping plovers' extensive use of the intertidal zone puts them in constant contact with coastal habitats likely to be contaminated by water-borne spills. Negative impacts can also occur during rehabilitation of oiled birds. Frink *et al*. (1996) describe how standard treatment protocols were modified to reflect the extreme susceptibility of piping plovers to handling and other stressors.

<u>Oil Spills</u>

Following the Ixtoc spill, which began on June 3, 1979, off the coast of Mexico, approximately 350 metric tons of oil accumulated on South Texas barrier beaches, resulting in a 79% decrease in the total number of infaunal organisms on contaminated portions of the beach (Kindinger 1981, Tunnell *et al*. 1982)? Chapman (1984) collected pre- and post-spill data on the abundance, distribution, and habitat use of shorebirds on the beaches in the affected area and saw declines in the numbers of birds as well as shifts in the habitats used. Shorebirds avoided the intertidal area of the beach, occupying the backshore or moving to estuarine habitats when most of the beach was coated. Chapman surmised that the decline in infauna probably contributed to the observed shifts in habitats used. His observations indicated that all the shorebirds, including piping plovers, avoided the contaminated sediments and concentrated in oil-free areas. Amos, however, reported that piping plovers ranked second to sanderlings in the numbers of oiled birds he observed on the beach, although there was no recorded mortality of plovers due to oil (Amos pers. comm. 2009, 2012). Oiled birds were seen for a year or more following the initial spill, likely due to continued washing in of sunken tar; but there were only occasional subsequent observations of oiled or tarred plovers (Amos pers. comm. 2009).

According to government estimates, the 2010 Deepwater Horizon Mississippi Canyon Well #252 oil spill discharged more than 200 million gallons of oil into the Gulf of Mexico (U.S. Government 2010). Containment activities, recovery of oil-water mix, and controlled burning removed some oil, but additional impacts to natural resources may stem from the 1.84 million gallons of dispersant that were applied to the spill (U.S. Government 2010). At the end of July 2010, approximately 625 miles of Gulf of Mexico shoreline was oiled. This included approximately 360 miles in Louisiana, 105 miles in Mississippi, 66 miles in Alabama, and 94 miles in Florida (U.S. Government 2010). These numbers do not address cumulative impacts or include shoreline that was cleaned earlier. The U.S. Coast Guard, the states, and responsible parties that form the Unified Command (with advice from federal and state natural resource agencies) initiated protective measures and clean-up efforts as provided in contingency plans for each state's coastline. The contingency plans identified sensitive habitats, including all ESA-listed species' habitats, which received a higher priority for response actions.

Efforts to prevent shoreline oiling and cleanup response activities can disturb piping plovers and their habitat. Although most piping plovers were on their breeding grounds in May, June, and early July when the Deepwater well was discharging oil, oil was still washing onto Gulf beaches when the plovers began arriving back on the Gulf in mid-July. Ninety percent of piping plovers detected during the prior four years of surveys in Louisiana were in the Deepwater Horizon oil spill impact zone, and Louisiana's Department of Wildlife and Fisheries reported significant disturbance to birds and their habitat from response activities. Wrack lines were removed, and sand washing equipment "cleansed" beaches (M. Seymour, Louisiana Natural Heritage Program, pers. comm. 2011). Potential long-term adverse effects stem from the construction of sand berms and closing of at least 32 inlets (Rice 2012a). Implementation of prescribed best management practices reduced, but did not negate, disturbance to plovers (and to other beach-dependent wildlife) from cleanup personnel, all-terrain vehicles, helicopters, and other equipment. USFWS and state biologists present during cleanup operations provided information about breeding, migrating, and wintering birds and their habitat protection needs. However, high staff turnover during the extended spill response period necessitated continuous education and training of clean up personnel (M. Bimbi, USFWS, pers. comm. 2011). Limited clean-up operations were still ongoing throughout the spill area in November 2012 (H. Herod, USFWS, pers. comm. 2012). Results of a natural resources damage assessment study to assess injury to piping plovers (Fraser *et al*. 2010) are not yet available.

More subtle but cumulatively damaging sources of oil and other contaminants are leaking vessels located offshore or within the bays on the Atlantic and Gulf coasts, offshore oil rigs and undersea pipelines in the Gulf of Mexico, pipelines buried under the bay bottoms, and onshore facilities such as petroleum refineries and petrochemical plants. In Louisiana, about 2,500-3,000 oil spills are reported in the Gulf region each year, ranging in size from very small to thousands of barrels (L. Carver, Louisiana Department of Wildlife and Fisheries, pers. comm. 2011). Chronic spills of oil from rigs and pipelines and natural seeps in the Gulf of Mexico generally involve small quantities of oil. The oil from these smaller leaks and seeps, if they occur far enough from land, will tend to wash ashore as tar balls. In cases such as this, the impact is limited to discrete areas of the beach, whereas oil slicks from larger spills coat longer stretches of the shoreline (K. Rice, USFWS, pers. comm. 2009). In late July and early August 2009, for example, oil suspected to have originated from an offshore oil rig in Mexican waters was observed on plumage or legs of 14 piping plovers in south Texas (Cobb pers. comm. 2012b).

<u>Pesticides and Other Contaminants</u>

A piping plover was found among dead shorebirds discovered on a sandbar near Marco Island, Florida following the county's aerial application of the organophosphate pesticide Fenthion for mosquito control in 1997 (Pittman 2001, Williams 2001). Subsequent to further investigations of bird mortalities associated with pesticide applications and to a lawsuit being filed against the Environmental Protection Agency in 2002, the manufacturer withdrew Fenthion from the market, and Environmental Protection Agency banned all use after November 30, 2004 (American Bird Conservancy 2007).

Absent identification of contaminated substrates or observation of direct mortality of shorebirds on a site used by migrating and wintering piping plovers, detection of contaminants threats is most likely to occur through analysis of unhatched eggs. Contaminants in eggs can originate from any point in the bird's annual cycle, and considerable effort may be required to ascertain where in the annual cycle exposure occurred (see, for example, Dickerson *et al*. 2011 characterizing contaminant exposure of mountain plovers).

There has been limited opportunistic testing of piping plover eggs. Polychlorinated biphenol (PCB) concentrations in several composites of Great Lakes piping plover eggs tested in the 1990s had potential to cause reproductive harm. Analysis of prey available to piping plovers at representative Michigan breeding sites indicated that breeding areas along the upper Great Lakes region were not likely the major source of contaminants to this population (D. Best, USFWS, pers. comm. 1999 in USFWS 2003). Relatively high levels of PCB, dichloro diphenyl dichloroethylene (DDE), and polybrominated diphenyl ether (PBDE) were detected in one of two clutches of Ontario piping plover eggs analyzed in 2009 (V. Cavalieri, USFWS, pers. comm. 2011). Results of opportunistic egg analyses to date from Atlantic Coast piping plovers did not warrant follow-up investigation (Mierzykowski 2009, 2010, 2012; S. Mierzykowski, USFWS pers. comm. 2012). No recent testing has been conducted for contaminants in the Northern Great Plains piping plover population.

Energy Development

Land-based Oil and Gas Exploration and Development

Various oil and gas exploration and development activities occur along the Gulf Coast. Examples of conservation measures prescribed to avoid adverse effects on piping plovers and their habitats include conditions on driving on beaches and tidal flats, restrictions on discharging fresh water across unvegetated tidal flats, timing exploration activities during times when the plovers are not present, and use of directional drilling from adjacent upland areas (USFWS 2008c; B. Firmin, USFWS, pers. comm. 2012). With the implementation of appropriate conditions, threats to nonbreeding piping plovers from land-based oil and gas extraction are currently very low.

Wind Turbines

Wind turbines are a potential future threat to piping plovers in their coastal migration and wintering range. Relatively small single turbines have been constructed along the beachfront in at least a few locations (e.g., South Carolina; M. Caldwell, USFWS, pers. comm. 2012). Current risk to piping plovers from several wind farms located on the mainland north and west of several bays in southern Texas is deemed low during months of winter residency because the birds are not believed to traverse these areas in their daily movements (D. Newstead, Coastal Bend Bays and Estuaries Program, pers. comm. 2012a). To date, no piping plovers have been reported from post-construction carcass detection surveys at these sites (P. Clements, USFWS, pers. comm. 2012). However, Newstead (pers. comm. 2012a) has raised questions about collision risk during migration departure, as large numbers of piping plovers have been observed in areas of the Laguna Madre east of the wind farms during the late winter. Furthermore, there is concern that, as sea level rises, the intertidal zone (and potential piping plover activity) may move closer to these sites. Several off-shore wind farm proposals in South Carolina are in various stages of early scoping (Caldwell pers. comm. 2012). A permit application was filed in 2011 for 500 turbines in three areas off the coast of south Texas (USACE 2011), but it is unknown whether piping plovers transit these areas.

In addition to uncertainty regarding the location and design (e.g., number and height of turbines) of future wind turbines, the magnitude of potential threats is difficult to assess without better information about piping plover movements and behaviors. For wind projects situated on barrier beaches, bay shorelines, or within bays, relevant information includes the flight routes of piping plovers moving among foraging and roosting sites, flight altitude, and avoidance rates under varying weather and light conditions. For offshore wind projects, piping plover migration routes and altitude, as well as avoidance rates will be key determinants of threats.

Predation

The extent of predation on migrating or wintering piping plovers remains largely unknown and is difficult to document. Avian and mammalian predators are common throughout the species' wintering range. Human activities affect the types, numbers, and activity patterns of some predators, thereby exacerbating natural predation on breeding piping plovers (USFWS 1996). One incident involving a cat observed stalking piping plovers was reported in Texas (NY Times 2007). It has been estimated that free-roaming cats kill over one billion birds every year in the U.S., representing one of the largest single sources of human-influenced mortality for small native wildlife (Gill 1995, Sax and Gaines 2008).

Predatory birds, including peregrine falcons, merlin, and harriers, are present in the nonbreeding range. Newstead (pers. comm. 2012b) reported two cases of suspected avian depredation of piping plovers in a Texas telemetry study, but he also noted that red tide may have compromised the health of these plovers. It has been noted, however, that the behavioral response of crouching when in the presence of avian predators may minimize avian predation on piping plovers (Morrier and McNeil 1991, Drake 1999, Drake *et al*. 2001). Drake (1999) theorized that this piping plover behavior enhances concealment associated with roosting in depressions and debris in Texas.

Nonbreeding piping plovers may reap some collateral benefits from predator management conducted for the primary benefit of other species. Florida Keys Refuges National Wildlife Refuge (USFWS 2011a), for example, released a draft integrated predator management plan that targets predators, including cats, for the benefit of native fauna and flora. Other predator control programs are ongoing in North Carolina, South Carolina, Florida, and Texas beach ecosystems (USFWS 2009b).

Although the extent of predation to nonbreeding piping plovers is unknown, it remains a potential threat. At this time, however, the USFWS considers predator control and related research on wintering and migration grounds to be a low priority.

Military Operations

Five of the eleven coastal military bases located in the U.S. continental range of nonbreeding piping plovers have consulted with the USFWS about potential effects of military activities on plovers and their habitat (USFWS 2009b, USFWS 2010). Formal consultation under section 7 of the ESA with Camp Lejeune, North Carolina in 2002, provided for year-round piping plover surveys, but restrictions on activities on Onslow Beach only pertain to the plover breeding season (J. Hammond, USFWS, pers. comm. 2012). Informal consultations with three Florida bases (Naval Station Mayport, Eglin Air Force Base, and Tyndall Air Force Base) addressed training activities that included beach exercises and occasional use of motorized equipment on beaches and bayside habitats. Eglin Air Force Base conducts twice-monthly surveys for piping plovers, and habitats consistently used by piping plovers are posted with avoidance requirements to minimize direct disturbance from troop activities. Operations at Tyndall Air Force Base and Naval Station Mayport were determined to occur outside optimal piping plover habitats. A 2001 consultation with the Navy for one-time training operations on Peveto Beach in Louisiana concluded informally (USFWS 2010). Current threats to wintering and migrating piping plovers posed by military activities appear minimal.

Disease

No instances of disease have been documented in piping plovers outside the breeding range. In the southeastern U.S., the cause of death of one piping plover received from Texas was emaciation (C. Acker, U.S. Geological Survey, pers. comm. 2009). Newstead (pers. comm. 2012b) reported circumstantial evidence that red tide weakened piping plovers in the vicinity of the Laguna Madre and Padre Island, Texas during the fall of 2011. Samples collected in Florida from two live piping plovers in 2006 both tested negative for avian influenza (M. Hines, U.S. Geological Survey, pers. comm. 2009). The 2009 5-Year Review concluded that West Nile virus and avian influenza remain minor threats to piping plovers on their wintering and migration grounds.

Summary and Synthesis of Threats

A review of threats to piping plovers and their habitat in their migration and wintering range shows a continuing loss and degradation of habitat due to sand placement projects, inlet stabilization, sand mining, groins, seawalls and revetments, dredging of canal subdivisions, invasive vegetation, and wrack removal. This cumulative habitat loss is, by itself, of major threat to piping plovers, as well as the many other shorebird species competing with them for foraging resources and roosting habitats in their nonbreeding range. However, artificial shoreline stabilization also impedes the processes by which coastal habitats adapt to storms and accelerating sea level rise, thus setting the stage for compounding future losses. Furthermore, inadequate management of increasing numbers of beach recreationists reduces the functional suitability of coastal migration and wintering habitat and increases pressure on piping plovers and other shorebirds depending upon a shrinking habitat base. Experience during the Deepwater Horizon oil spill illustrates how, in addition to the direct threat of contamination, spill response activities can result in short- and long-term effects on habitat and disturb piping plovers and other shorebirds. If climate change increases the frequency and magnitude of severe weather events, this may pose an additional threat. The best available information indicates that other threats are currently low, but vigilance is warranted, especially in light of the potential to exacerbate or compound effects of very significant threats from habitat loss and degradation and from increasing human disturbance.

Recovery criteria

Northern Great Plains Population (USFWS 1988b, 1994)

1. Increase the number of birds in the U.S. Northern Great Plains states to 2,300 pairs (USFWS 1994).
2. Increase the number of birds in the prairie region of Canada to 2,500 adult piping plovers (USFWS 1988).
3. Secure long-term protection of essential breeding and wintering habitat (USFWS 1994).

Great Lakes Population (USFWS 2003)

1. At least 150 pairs (300 individuals), for at least 5 consecutive years, with at least 100 breeding pairs (200 individuals) in Michigan and 50 breeding pairs (100 individuals) distributed among sites in other Great Lakes states.
2. Five-year average fecundity within the range of 1.5-2.0 fledglings per pair, per year, across the breeding distribution, and ten-year population projections indicate the population is stable or continuing to grow above the recovery goal.
3. Protection and long-term maintenance of essential breeding and wintering habitat is ensured, sufficient in quantity, quality, and distribution to support the recovery goal of 150 pairs (300 individuals).
4. Genetic diversity within the population is deemed adequate for population persistence and can be maintained over the long-term.
5. Agreements and funding mechanisms are in place for long-term protection and management activities in essential breeding and wintering habitat.

Atlantic Coast Population (USFWS 1996)

1. Increase and maintain for 5 years a total of 2,000 breeding pairs, distributed among 4 recovery units.

Recovery Unit	*Minimum Subpopulation*
Atlantic (eastern) Canada	*400 pairs*
New England	*625 pairs*
New York-New Jersey	*575 pairs*
Southern (DE-MD-VA-NC)	*400 pairs*

2. Verify the adequacy of a 2,000 pair population of piping plovers to maintain heterozygosity and allelic diversity over the long term.
3. Achieve a 5-year average productivity of 1.5 fledged chicks per pair in each of the 4 recovery units described in criterion 1, based on data from sites that collectively support at least 90% of the recover unit’s population.
4. Institute long-term agreements to assure protection and management sufficient to maintain the population targets and average productivity in each recovery unit.

5. Ensure long-term maintenance of wintering habitat, sufficient in quantity, quality, and distribution to maintain survival rates for a 2,000-pair population.

Conservation Recommendations

<u>Nonbreeding plovers from all three breeding populations (USFWS 2012)</u>

1. Maintain natural coastal processes that perpetuate wintering and coastal migration habitat.
2. Protect wintering and migrating piping plovers and their habitat from human disturbance.
3. Monitor nonbreeding plovers and their habitat.
4. Protect nonbreeding plovers and their habitats from contamination and degradation from oil or other chemical contaminants.
5. Assess predation as a potential limiting factor for piping plovers on wintering and migration sites.
6. Improve application or regulatory tools.
7. Develop mechanisms to provide long-term protection of nonbreeding plovers and their habitat.
8. Conduct scientific investigations to refine knowledge and inform conservation of migrating and wintering piping plovers.

6.2. Environmental Baseline

This section is an analysis of the effects of past and ongoing human and natural factors leading to the current status of the piping plover, its habitat, and ecosystem within the Action Area. The environmental baseline is a “snapshot” of the species’ health in the Action Area at the time of the consultation, and does not include the effects of the Action under review (see **Section 4**).

6.2.1. Action Area Numbers, Reproduction, and Distribution

Piping plovers have been documented during migration and/or winter on the ends of Folly Beach and Bird Key Stono within the Action Area (Maddock *et al.* 2013, USFWS unpublished data). The migrant population is typically larger than the winter population. Although piping plovers that winter at sites, meaning they spend the majority of their nonbreeding season at one location, can arrive at their winter site as early as August and depart as late as April (Maddock *et al.* 2009), the best winter population estimate cannot be determined until December and/or January. Results of a band re-sighting analysis for birds documented at sites in South Carolina showed zero immigration or emigration during the months of December and January (J. Cohen, pers. comm. 2009). Therefore, the Service determines the local winter population by using the single highest count of birds during surveys conducted between December 1 and January 31. Since the majority of piping plovers are unbanded, the number of migrants as well as the passage population (the total number of birds that use a site during the entire nonbreeding season) for the sites within the action area are currently unknown.

Uniquely banded piping plovers have been seen on both islands and are known to cross the Stono Inlet based on band re-sights (**Table 8**). Therefore, both sites are considered to be one site, which makes determining the winter population number difficult. Numbers at each site cannot simply be added together unless band resightings are taken into consideration. **Table 9** represents the maximum number of birds counted at each site during a winter survey. Piping plovers from all three breeding populations, the Northern Great Plains, Great Lakes, and Atlantic Coast, have been documented using the west end of Folly Beach and Bird Key Stono.

Table 8. Confirmed band re-sightings of piping plovers on the west end of Folly Beach and Bird Key Stono from 2006 to 2017.

Band Combinations[1]	Winter or Migrant Bird[2]	Breeding Population[3]
--,WA:-Gf,GL	W	NGP U.S.
-Gf(YX6),--:-O,--	M	AC U.S.
-Gf(2MU),--:-B,--	M	AC U.S.
-O,--:-X,-R	W	GL U.S.
-Of,Lb:-X,-O	W	GL U.S.
-X,-B:-Of,OB	W	GL U.S.
-X,-b:-Of,OG	W	GL U.S.

[1]All band combinations have been confirmed by the banders. Re-sight data is a compilation of the SC Shorebird Project (2006-2008), required monitoring set forth by two Service BOs (2006-2014), and Service and SCDNR re-sight surveys (2010-2017). Band combinations were recorded in the following order: upper left, lower left: upper right, and lower right using the following abbreviations:
X: metal, b: light blue, f: flag, G: dark green, R: red, g: light green, /: split color band (2 colors on the same band), Y: yellow, L: black, //: triple split color band (1 color separated by another color on the same band), O: orange, W: white, B: dark blue, A: gray, –: no band.
[2]The local winter population is determined by the highest count of birds during surveys conducted between December 1 and January 31. This is consistent with the results of a band re-sighting analysis for birds seen in South Carolina that showed zero immigration or emigration during the months of December and January (J. Cohen, pers. comm. 2009). W=A winter bird is a bird that has been documented at a site between December 1 and January 31. M=A migrant bird is a bird that has not been documented at a site between December 1 and January 31.
[3]Breeding populations: NGP=Northern Great Plains, AC=Atlantic Coast, GL=Great Lakes.

Table 9. Piping plover winter population numbers in the action area from 2005-2018.

Season	West end of Folly[2]	Bird Key Stono[2]
2005-2006	1	3
2006-2007	ND[3]	8
2007-2008	ND[3]	7
2008-2009	ND[3]	ND[2]
2009-2010	ND[3]	ND[2]
2010-2011	0	4
2011-2012	ND[3]	3
2012-2013	ND[3]	5
2013-2014	1	3
2014-2015	1	5
2015-2016	1	3
2016-2017	1	5
2017-2018	0	6

[1]The local winter population is determined by the highest count of birds during surveys conducted between December 1 and January 31. This is consistent with the results of a band re-sighting analysis for birds seen in South Carolina that showed zero immigration or emigration during the months of December and January (J. Cohen, pers. comm. 2009). The same uniquely banded birds have been documented at these sites between December 1 and January 31.

[2]The 2005-2006 and 2010-2011 season data for the west end of Folly Beach, Bird Key, and Deveaux Bank and the 2013-2014 season data for the west end of Folly Beach only represent 1 survey during the winter and may not accurately represent the true winter population.

[3]ND=no data

If project construction occurs during the piping plover nonbreeding season, plovers may avoid the construction area on Bird Key Stono and use other areas on Bird Key Stono and/or the west end of Folly Beach. Therefore, Folly Beach and Bird Key Stono are considered to be part of the action area for piping plovers.

6.2.2. Action Area Conservation Needs and Threats

Recreational Disturbance

Intense human disturbance in winter habitat can be functionally equivalent to habitat loss. If the disturbance prevents birds from using an area (Goss-Custard *et al*. 1996), this can lead to roost abandonment and population declines (Burton *et al*. 1996). Disturbance from human and pet presence alters plover behavior and often negatively influences distribution.

West End of Folly Beach

The west end of Folly Beach provides public access and a portion of the beach is within the Folly Beach County Park. Dogs are allowed on the beach, but must be on leash at all times unless the dog owner is a member of the Folly Island Dog Owner (FIDO) club, which allows special privileges for "responsible dog owners and their dogs."

Bird Key

This undeveloped island is a SCDNR Heritage Preserve. It is managed for seabird and shorebird nesting and is within designated critical habitat for wintering piping plovers (see **Section 7**). No dogs are allowed on the island and pedestrian access is seasonally restricted.

6.3. Effects of the Action

This section analyzes the direct and indirect effects of the Action on the piping plover, which includes the direct and indirect effects of interrelated and interdependent actions. Direct effects are caused by the Action and occur at the same time and place. Indirect effects are caused by the Action, but are later in time and reasonably certain to occur. Our analyses are organized according to the description of the Action in section 2 of this BO.

6.3.1. Effects of beach renourishment on piping plovers

Beneficial Effects

The renourishment project may temporarily increase roosting habitat on the northeast end of Bird Key Stono due to the sand placement covering existing dune vegetation.

Adverse Effects

Shoreline stabilization projects have been documented to have adverse effects on nonbreeding piping plover habitat and piping plover abundance and distribution. Results of monitoring piping plovers and their habitat provide additional information on how piping plovers respond to these projects, minimization measures, and other factors that influence piping plover abundance, distribution, and site selection.

Direct effects: Direct effects are those direct or immediate effects of a project on the species or its habitat. The construction window overlaps with one nonbreeding season for piping plovers. Heavy machinery and equipment (e.g., trucks and bulldozers operating on project area beaches, sand excavation, and berm construction) may adversely affect migrating and wintering piping plovers in the project area by disturbance and disruption of normal activities such as roosting and foraging, and possibly forcing birds to expend valuable energy reserves to seek available habitat elsewhere.

Burial and suffocation of invertebrate species will occur during each nourishment and renourishment cycle. Impacts will affect the 40,000 feet of shoreline. Timeframes projected for benthic recruitment and re-establishment following beach nourishment are between 6 months to 2 years (Thrush *et al*. 1996, Peterson *et al*. 2000, Zajac and Whitlatch 2003, Bishop *et al*. 2006, Peterson *et al*. 2006). Depending on actual recovery rates, impacts may occur even if nourishment activities occur outside the migration and wintering seasons. The sand is being placed above the high tide line on a small section of the island, which will limit impacts to intertidal foraging habitat.

Indirect effects: Indirect effects are effects caused by or result from the proposed action, are later in time, and are reasonably certain to occur. The proposed project may facilitate increased access to currently occupied roosting and foraging habitat. Recreational activities that potentially adversely affect plovers include disturbance by unleashed pets and increased pedestrian use.

6.3.2. Effects of Groin Rehabilitation

No additional effects due to groin rehabilitation are anticipated based on the location of the groins. Piping plovers tend to use the ends of the islands, which is outside of the area of groin rehabilitation. Folly Beach will continue to be renourished on a regular interval, which will allow sediment transport to continue to the west end of the island.

6.4. Cumulative Effects

For purposes of consultation under ESA §7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under §7 of the ESA. The Service is not aware of any cumulative effects in the Action Area at this time; therefore, cumulative effects are not relevant to formulating our opinion for the Action.

6.5. Conclusion

In this section, the Service summarizes and interprets the findings of the previous sections for the piping plover (status, baseline, effects, and cumulative effects) relative to the purpose of a BO under §7(a)(2) of the ESA, which is to determine whether a Federal action is likely to:

a) jeopardize the continued existence of species listed as endangered or threatened; or
b) result in the destruction or adverse modification of designated critical habitat.

"Jeopardize the continued existence" means to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species (50 CFR §402.02).

After reviewing the current status of the Northern Great Plains, Great Lakes, and Atlantic Coast piping plover populations, the environmental baseline for the proposed project, associated construction activities, and the cumulative effects, it is the Service's Biological Opinion that implementation of the project, as proposed, is not likely to jeopardize the continued existence of the piping plover because effects due to construction activities are expected to be short term and become beneficial once construction is completed.

"Take" of piping plovers will be minimized by the implementation of the Reasonable and Prudent Measures, and Terms and Conditions outlined in **Section 9**.

7. CRITICAL HABITAT FOR THE PIPING PLOVER

7.1. Status of Critical Habitat

The Service has designated critical habitat for the piping plover on three occasions. Two of these designations protected different breeding populations. Critical habitat for the Great Lakes breeding population was designated May 7, 2001, (66 [FR] (Federal Register) 22938, USFWS 2001a), and critical habitat for the northern Great Plains breeding population was designated September 11, 2002, (67 FR 57637, USFWS 2002). No critical habitat has been proposed or designated for the Atlantic Coast breeding population, but the needs of all three breeding populations were considered in the 2001 critical habitat designation for wintering piping plovers (66 FR 36038, USFWS 2001b) and subsequent redesignations (USFWS 2008d, 2009d). Wintering piping plovers may include individuals from the Great Lakes and northern Great Plains breeding populations as well as birds that nest along the Atlantic coast.

7.1.1. Description of Critical Habitat

Critical habitat for wintering piping plovers currently comprises 141 units totaling 256,513 acres along the coasts of North Carolina, South Carolina, Georgia, Florida, Alabama, Mississippi, Louisiana, and Texas. The original designation included 142 areas (the rule erroneously states 137 units) encompassing approximately 1,798 miles of mapped shoreline and 165,211 acres of mapped areas (USFWS 2001b). A revised designation for four North Carolina units was published in 2008 (USFWS 2008d). Eighteen revised Texas critical habitat units were designated in 2009, replacing 19 units that were vacated and remanded by a 2006 court order

(USFWS 2009c). Designated areas include habitats that support roosting, foraging, and sheltering activities of piping plovers.

Critical habitat designation for nonbreeding piping plovers used the term "primary constituent elements" (PCEs) to identify the key components of critical habitat that are essential to its conservation and may require special management considerations or protection. Revisions to the critical habitat regulations in 2016 (81 FR 7214, 50 CFR §4.24) discontinue use of the term PCEs, and rely exclusively the term "physical and biological features" (PBFs) to refer to these key components, because the latter term is the one used in the statute. This shift in terminology does not change how the Service conducts a "destruction or adverse modification" analysis. In this BO, we use the term PBFs to label the key components of critical habitat that provide for the conservation of the nonbreeding piping plover that were identified in its critical habitat designation rule as PCEs.

The PBFs of nonbreeding piping plover critical habitat are sand or mud flats or both with no or sparse emergent vegetation for foraging piping plovers and adjacent unvegetated or sparsely vegetated sand, mud, or algal flats above high tide for roosting piping plovers (66 FR 36038). Important components of the beach/dune ecosystem include surf-cast algae, sparsely vegetated back beach and salterns, spits, and washover areas. Washover areas are broad, unvegetated zones, with little or no topographic relief, that are formed and maintained by the action of hurricanes, storm surge, or other extreme wave action.

7.1.2. Conservation Value of Critical Habitat

Designation of critical habitat can help focus conservation activities for a listed species by identifying areas that contain PBFs that are essential for the conservation of that species. Recovery of piping plovers is dependent upon available habitat throughout the range of the species.

7.1.3. Conservation Needs for Critical Habitat

All critical habitat units were occupied at the time of designation. Due to the dynamic nature of these ephemeral habitats, all units are needed for the recovery of the species. Natural coastal processes are also necessary to ensure the existence and functionality of these units in the future. When these processes are limited or altered habitat quality diminishes.

7.2. Environmental Baseline for Critical Habitat

This section is an analysis of the effects of past and ongoing human and natural factors leading to the current status of designated critical habitat for the piping plover within the Action Area. The environmental baseline is a "snapshot" of the condition of the PBFs that are essential to the conservation of the species within designated critical of the Action Area at the time of the consultation, and does not include the effects of the Action under review.

7.2.1. Action Area Conservation Value of Piping Plover Critical Habitat

The Action Area is within designated critical habitat unit SC-9 (**Figure 11**) and currently contains all PBFs. Each unit within the nonbreeding piping plover designation is essential to the recovery of the species. The text description of the unit is as follows:

Unit SC-9: Stono Inlet: 495 ha (1223 ac) in Charleston County.

Most of this unit is privately owned. It includes the eastern end of Kiawah Island (approximately 4.0 km (2.5 mi)) from mean low low water (MLLW) on the Atlantic Ocean running north to MLLW on the first large tributary connecting east of Bass Creek running northeast into the Stono River. It includes MLLW up to where densely vegetated habitat, not used by the piping plover, begins and where the constituent elements no longer occur along the Stono Inlet and River. All of Bird Key Stono Heritage Preserve and all of Skimmer Flats to MLLW are included. The golf course and densely vegetated areas are not included.

Regarding the PBFs for this project, the placement of sand and resulting burial of the prey base is anticipated to temporarily degrade foraging habitat, but increase roosting habitat. This BO includes required terms and conditions that minimize the incidental take of piping plovers. The PBFs are expected to recover and roosting habitat may increase immediately after project construction.

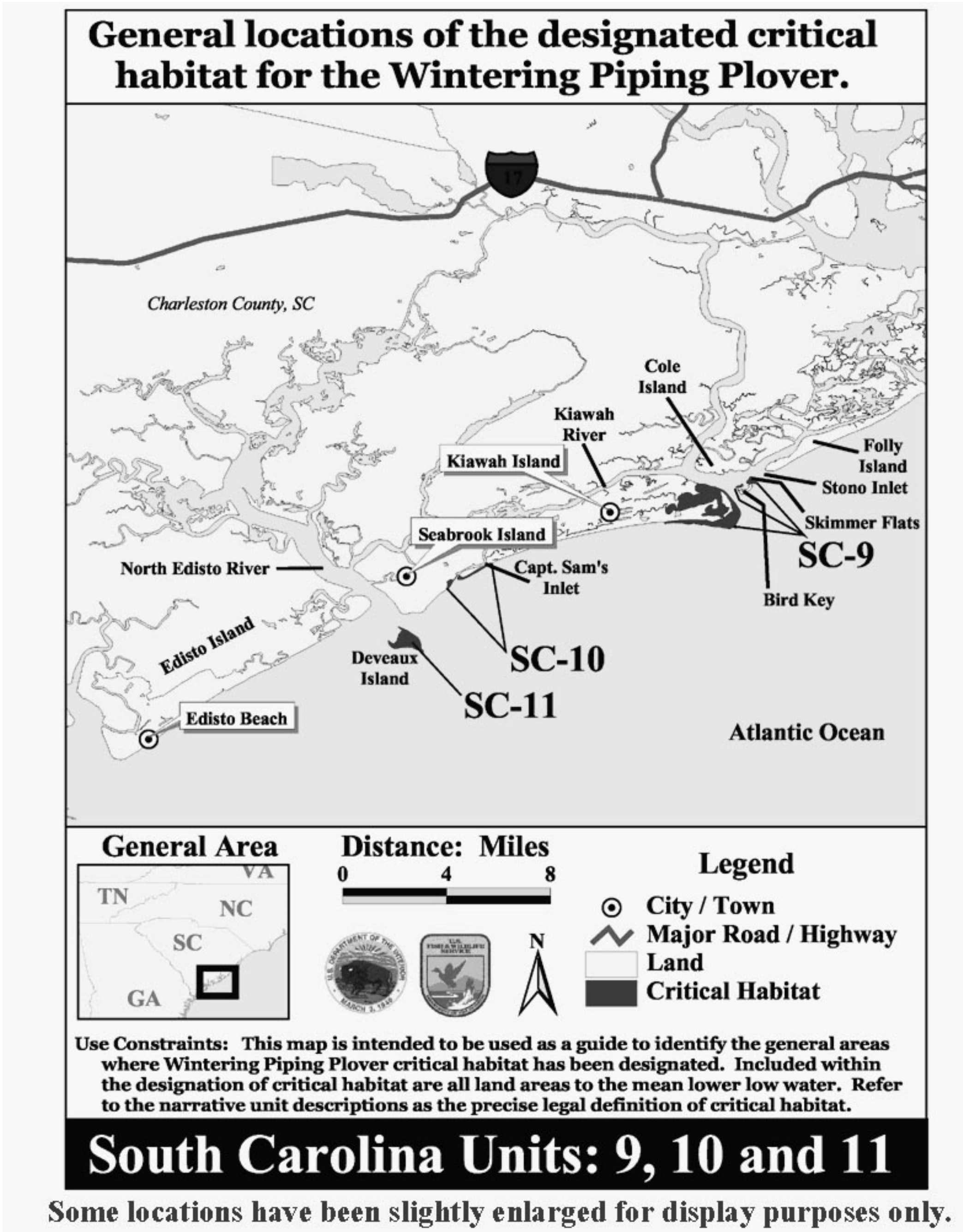

Figure 11. Map of piping plover designated critical habitat units SC-9, SC-10, and SC-11.

7.2.2. Action Area Conservation Needs for Critical Habitat

Recreational Disturbance

Intense human disturbance in winter habitat can be functionally equivalent to habitat loss. If the disturbance prevents birds from using an area (Goss-Custard *et al*. 1996), this can lead to roost abandonment and population declines (Burton *et al*. 1996). Disturbance from human and pet presence alters plover behavior and often negatively influences distribution.

7.3. Effects of the Action on Critical Habitat

This section analyzes the direct and indirect effects of the Action on critical habitat for the piping plover, which includes the direct and indirect effects of interrelated and interdependent actions. Direct effects are caused by the Action and occur at the same time and place. Indirect effects are caused by the Action, but are later in time and reasonably certain to occur. Our analyses are organized according to the description of the Action in section 2 of this BO.

7.3.1. Effects of Renourishment on Critical Habitat

The renourishment on Bird Key Stono will occur on a six acre area of the island and will be limited to an area above the high tide line. Construction will take up to one week to complete. PBFs that support roosting are anticipated to increase in size and quality. PBFs that support foraging may temporarily be impacted adjacent to the construction area. However, these impacts will be minimized due to the location of the sand placement and the short construction window, which will allow for faster benthic invertebrate recruitment.

7.3.2. Effects of Groin Rehabilitation on Critical Habitat

Groin rehabilitation will occur outside of critical habitat. Therefore, no adverse effects are anticipated.

7.4. Cumulative Effects on Piping Plover Critical Habitat

For purposes of consultation under ESA §7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under §7 of the ESA. The Service is not aware of any cumulative effects in the Action Area at this time; therefore, cumulative effects are not relevant to formulating our opinion for the Action.

7.5. Conclusion for Piping Plover Critical Habitat

In this section, we summarize and interpret the findings of the previous sections for nonbreeding piping plover critical habitat (status, baseline, effects, and cumulative effects) relative to the purpose of a BO under §7(a)(2) of the ESA, which is to determine whether a Federal action is likely to:

a) jeopardize the continued existence of species listed as endangered or threatened; or
b) result in the destruction or adverse modification of designated critical habitat.

"*Destruction or adverse modification*" means a direct or indirect alteration that appreciably diminishes the value of designated critical habitat for the conservation of a listed species. Such alterations may include, but are not limited to, those that alter the physical or biological features essential to the conservation of a species or that preclude or significantly delay development of such features (50 CFR §402.02).

After reviewing the current status of the critical habitat, the environmental baseline for the Action Area, the effects of the Action, and the cumulative effects, it is the Service's biological opinion that the Action is not likely to destroy or adversely modify designated critical habitat for the piping plover because effects due to construction activities are expected to be short term and become beneficial once construction is completed.

8. RED KNOT

8.1. Status of the species

On December 11, 2014, the Service published the final rule to list the rufa red knot (*Calidris canutus rufa*) as a threatened subspecies under the ESA (79 FR 73706).

8.1.1. Description of the species

The rufa red knot is a medium-sized migratory shorebird that breeds in the Canadian Arctic, winters in parts of the Southeastern U.S., the Caribbean, and South America, and primarily uses well-known spring and fall stopover areas on the Atlantic coast of the U.S., although some follow a midcontinental migratory route.

8.1.2. Life History

The rufa red knot migrates annually between its breeding grounds in the central Canadian Arctic and several wintering regions, including the Southeast United States (Southeast), the Northeast Gulf of Mexico, northern Brazil, and Tierra del Fuego at the southern tip of South America. The rufa red knot's typical life span is at least 7 years (J. Parvin pers. comm. March 14, 2014, Niles *et al.* 2008), with the oldest known wild bird at least 21 years old as of 2014 (Bauers 2014, Jordan 2014). Age of first breeding is at least 2 years (S. Koch, L. Niles, and R. Porter pers. comm. August 12, 2014, Harrington 2001).

On the breeding grounds, pair bonds form soon after the birds arrival, in late May or early June, and remain intact until shortly after the eggs hatch (Niles *et al.* 2008, Harrington 2001). Female rufa red knots lay only one clutch (group of eggs) per season, and, as far as is known, do not lay a replacement clutch if the first is lost. The usual clutch size is four eggs, though three-egg clutches have been recorded. The incubation period lasts approximately 22 days from the last egg laid to the last egg hatched, and both sexes participate equally in egg incubation. Young are precocial, leaving the nest within 24 hours of hatching and foraging for themselves (Niles *et al.* 2008). Females are thought to leave the breeding grounds and start moving south soon after the chicks hatch in mid-July. Thereafter, parental care is provided solely by the males, but about 25 days later (around August 10) males also abandon the newly fledged juveniles and move south. Not long after, they are followed by the juveniles (Niles *et al.* 2008). Breeding success of High Arctic shorebirds such as *Calidris canutus* varies dramatically among years in a somewhat cyclical manner. Two main factors seem to be responsible for this annual variation: abundance of arctic lemmings (*Dicrostonyx torquatus* and *Lemmus sibericus*) (by indirectly affecting predation pressure on shorebirds) and weather (Piersma and Lindström 2004, Blomqvist *et al*. 2002, Summers and Underhill 1987). Growth rate of *C. canutus* chicks is very high compared to similarly sized shorebirds nesting in more temperate climates and is strongly correlated with weather-induced and seasonal variation in availability of invertebrate prey (Schekkerman *et al*. 2003).

During both the northbound (spring) and southbound (fall) migrations, red knots use key staging and stopover areas to rest and feed (**Figure 12**). Each year some red knots make one of the longest distance migrations known in the animal kingdom, traveling up to 19,000 mi (30,000 km) annually. Red knots undertake long flights that may span thousands of miles without stopping. As *Calidris canutus* prepare to depart on long migratory flights, they undergo several physiological changes. Before takeoff, the birds accumulate and store large amounts of fat to fuel migration and undergo substantial changes in metabolic rates. In addition, the leg muscles, gizzard (a muscular organ used for grinding food), stomach, intestines, and liver all decrease in size, while the pectoral (chest) muscles and heart increase in size. Due to these physiological changes, *C. canutus* arriving from lengthy migrations are not able to feed maximally until their digestive systems regenerate, a process that may take several days. Because stopovers are time-constrained, *C. canutus* requires stopovers rich in easily digested food to achieve adequate weight gain (Niles *et al.* 2008, van Gils *et al.* 2005a, van Gils *et al.* 2005b, Piersma *et al*. 1999) that fuels the next migratory flight and, upon arrival in the Arctic, also fuels a body transformation to breeding condition (Morrison 2006). At some stages of migration, very high proportions of entire shorebird populations may use a single migration staging site to prepare for long flights. High fractions of the red knot's rangewide population can occur together at a small number of nonbreeding locations, leaving populations vulnerable to loss of key resources (Harrington 2001). For example, Delaware Bay provides the final Atlantic coast stopover for a significant majority (50 to 80 percent) of the red knot population making its way to the arctic breeding grounds each spring (Clark *et al*. 2009, Brown *et al.* 2001). Individual red knots show moderate fidelity to particular migration staging areas between years (French Guiana Regional Scientific Council for Natural Heritage (CSRPN) 2013, Duerr *et al.* 2011, Watts 2009a, Harrington 2001).

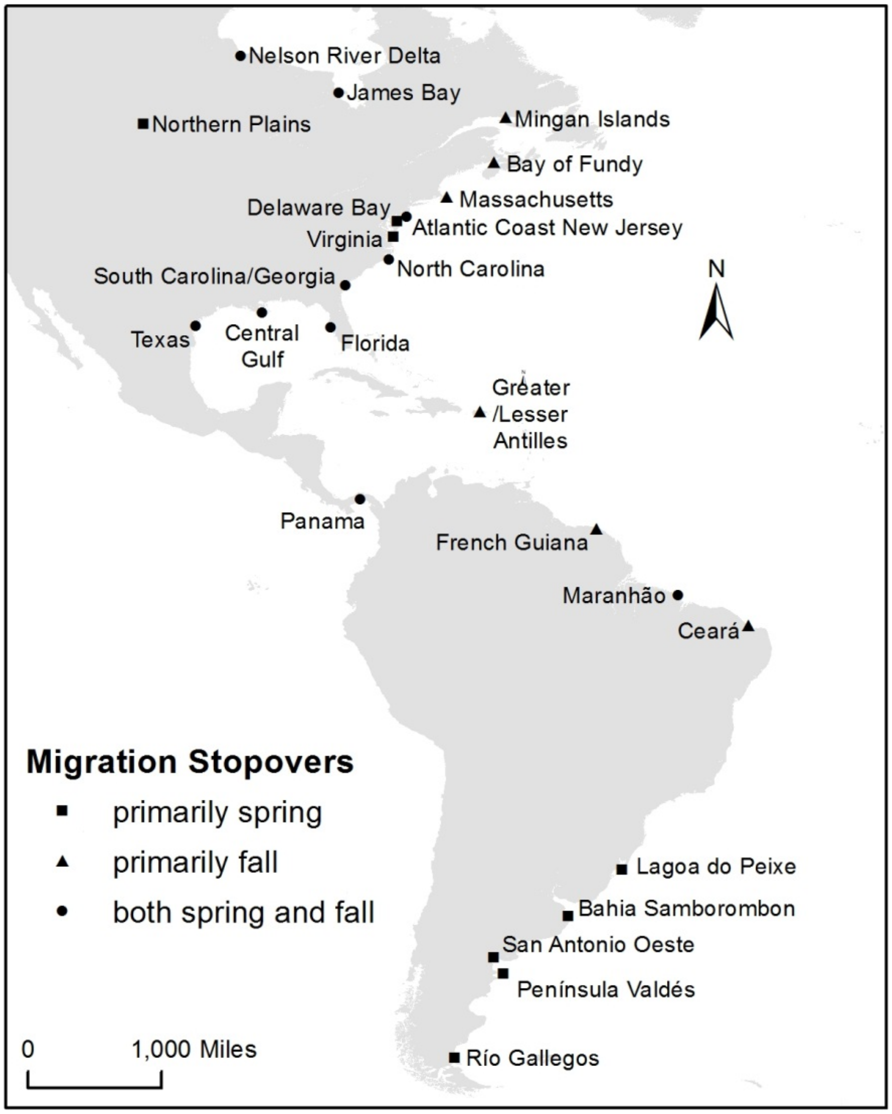

Figure 12. Well-known rufa red knot migration stopover areas.

On the wintering grounds, fidelity appears to be high, with minimal movement of birds among wintering regions (Georgia Department of Natural Resources (GDNR) 2013, BandedBirds.org 2012, Schwarzer *et al.* 2012, Niles *et al.* 2008, Harrington *et al.* 1988). Researchers often distinguish between those rufa red knots that winter the farthest south (in Argentina and Chile) and therefore undertake the longest-distance migrations ("southern-wintering"), from those that winter farther north in northern Brazil and the Southeast ("northern-wintering"), with some notable physiological and ecological differences between the two groups (B. Harrington pers. comm. November 14, 2013).

Nonbreeding (Migration and Winter) Habitat Use

Coastal habitats used by red knots in migration and wintering areas are similar in character (Harrington 2001), generally coastal marine and estuarine (partially enclosed tidal area where fresh and salt water mixes) habitats with large areas of exposed intertidal sediments. Migration and wintering habitats include both high-energy ocean- or bay-front areas, as well as tidal flats in more sheltered bays and lagoons (Harrington 2001). Preferred wintering and migration microhabitats are muddy or sandy coastal areas, specifically, the mouths of bays and estuaries, tidal flats, and unimproved tidal inlets (North Carolina Wildlife Resources Commission (NCWRC) 2013, Lott *et al.* 2009, Niles *et al.* 2008, Harrington 2001). Along the U.S. Atlantic coast, dynamic and ephemeral (lasting only briefly) features are important red knot habitats, including sand spits, islets, shoals, and sandbars, features often associated with inlets (Harrington 2008, Harrington in Guilfoyle *et al.* 2007, Winn and Harrington in Guilfoyle *et al.* 2006). In many wintering and stopover areas, quality high-tide roosting habitat (i.e., close to feeding areas, protected from predators, with sufficient space during the highest tides, free from excessive human disturbance) is limited (CSRPN 2013, K. Kalasz pers. comm. November 26, 2012, L. Niles pers. comm. November 19 and 20, 2012, Kalasz 2008). In nonbreeding habitats, *Calidris canutus* require sparse vegetation to avoid predation (Niles *et al.* 2008, Piersma *et al.* 1993).

Available information suggests that red knots use inland saline lakes as stopover habitat in the Northern Great Plains (Newstead *et al.* 2013, North Dakota Game and Fish Department (NDGFD) 2013, Western Hemisphere Shorebird Reserve Network (WHSRN) 2012, Skagen *et al.* 1999). There is little information indicating whether or not red knots may also utilize inland freshwater habitats during migration, but data suggest that certain freshwater areas may warrant further study as potential stopover habitats (C. Dovichin pers. comm. May 6, 2014, eBird.org 2014, Russell 2014). Best available data indicate that small numbers of red knots sometimes use manmade freshwater habitats (e.g., impoundments) along inland migration routes (eBird.org 2014, Russell 2014, Central Flyway Council 2013, NDGFD 2013, Oklahoma Department of Wildlife Conservation (ODWC) 2013, A. Simnor pers. comm. October 15, 2012).

Foraging Habits

Across all (six) subspecies, *Calidris canutus* is a specialized molluscivore, eating hard-shelled mollusks, sometimes supplemented with easily accessed softer invertebrate prey, such as shrimp- and crab-like organisms, marine worms, and horseshoe crab eggs (Piersma and van Gils 2011, Harrington 2001). The mollusk prey is swallowed whole and crushed in the gizzard, which in *C. canutus* is the largest (relative to body size) among any shorebird species evaluated (Piersma and van Gils 2011). Large gizzards are among this species' adaptations to a mollusk diet, allowing *C. canutus* to grind the hard shells of its prey. *Calidris canutus* prefer thin-shelled to thick-shelled prey species because they are easier to digest and provide a more favorable meat to mass ratio (higher prey quality) (van Gils *et al.* 2005a, Harrington 2001, Zwarts and Blomert 1992). From studies of other subspecies, Zwarts and Blomert (1992) concluded that *C. canutus* cannot ingest prey with a circumference greater than 1.2 in (30 millimeters (mm)). For rufa red knots, prey lengths of 0.16 to 0.79 in (4 to 20 mm) have been observed (Cohen *et al.* 2010b, González *et al.* 1996). Foraging activity is largely dictated by tidal conditions, as *C. canutus* rarely wade in water more than 0.8 to 1.2 in (2 to 3 cm) deep (Harrington 2001). Due to bill morphology, *C. canutus* is limited to foraging on only shallow-buried prey, within the top 0.8 to 1.2 in (2 to 3 cm) of sediment (Gerasimov 2009, Zwarts and Blomert 1992). Along the U.S. coast, *Donax* and *Mulinia* clams and blue mussel (*Mytilus edulis*) spat are key prey items. A prominent departure from typical prey items occurs each spring when red knots feed on the eggs of horseshoe crabs

(*Limulus polyphemus*), particularly during the key migration stopover within the Delaware Bay. Delaware Bay serves as the principal spring migration staging area for the red knot because of the abundance and availability of horseshoe crab eggs (Clark *et al.* 2009, Harrington 2001, Harrington 1996, Morrison and Harrington 1992). In Delaware Bay, horseshoe crab eggs are a superabundant source of easily digestible food.

8.1.3. Numbers, Reproduction, and Distribution

After a thorough review of the best available population data, a precise rangewide population estimate for the rufa red knot cannot be derived at this time. For example, there are no rangewide population estimates for fall migration or breeding areas because birds are too dispersed. However, population trend information from some areas can be reliably inferred. A high confidence data set of long-term surveys from two key red knot areas, Tierra del Fuego (wintering) and Delaware Bay (spring), show declines of 70 to 75 percent over roughly the same period, since about 2000 (Dey *et al.* 2014, Dey *et al.* 2011a, Clark *et al.* 2009, Morrison *et al*. 2004, Morrison and Ross 1989, Kochenberger 1983, Dunne *et al.* 1982, Wander and Dunne 1982). Data sets associated with lower confidence, from the Brazil wintering region and three South American spring stopovers, also suggest declines roughly over this same timeframe (Niles *et al.* 2008, Baker *et al.* 2005, González 2005, Morrison and Ross 1989, Harrington *et al.* 1986), however, more recently a substantial increase was documented in Brazil (Dey *et al.* 2014). Emerging information from Virginia also suggests a decline relative to the 1990s (B. Watts pers. comm. August 22, 2014). The Southeast wintering region has not declined over this period despite some years of lower counts in Florida, due to the likelihood that the birds' usage shifts geographically within this region from year to year (Harrington 2005a). In summary, the best available data indicate a sustained decline occurred in the 2000s, and may have stabilized at a relatively low level in the last few years. Attempts to evaluate long-term population trends using national or regional data from volunteer shorebird surveys and other sources have also generally concluded that red knot numbers have declined, probably sharply (National Park Service (NPS) 2013, Andres 2009, Morrison *et al.* 2006).

Breeding Range

The red knot breeds in the central Canadian Arctic, from the islands of northern Hudson Bay to the Foxe Basin shoreline of Baffin Island, and west to Victoria Island (Niles *et al.* 2008, Morrison and Harrington 1992). Potential breeding habitat extends farther north the southern Queen Elizabeth Islands (Niles *et al.* 2008) (**Figure 13**). The extent to which rufa red knots from different wintering areas mix on the breeding grounds, and therefore potentially interbreed, is poorly known (Harrington *et al.* 1988). Red knots generally nest in dry, slightly elevated tundra locations, often on windswept slopes with little vegetation. Breeding areas are located inland, but near arctic coasts. Nests may be scraped into patches of mountain avens (*Dryas octopetala*) plants, or in low spreading vegetation on hummocky (characterized by knolls or mounds) ground containing lichens, leaves, and moss. After the eggs hatch, red knot chicks and adults quickly move away from high nesting terrain to lower, freshwater wetland habitats. On the breeding grounds, the red knot's diet consists mostly of terrestrial invertebrates such as insects and other arthropods (Niles *et al.* 2008, Harrington 2001).

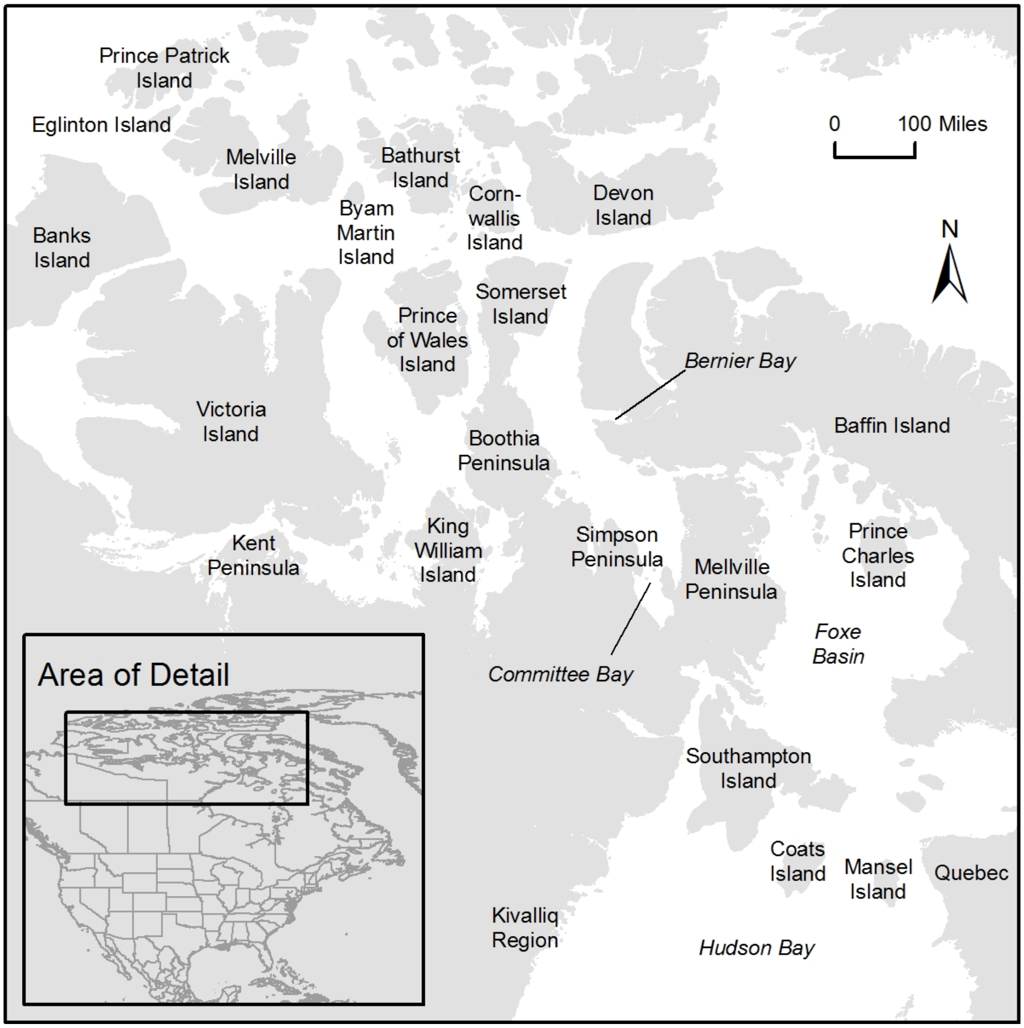

Figure 13. Known and potential breeding range of the rufa red knot

Nonbreeding Range

Geolocator and resightings data show definitively that the *rufa* nonbreeding range includes the entire Atlantic and Caribbean coasts of South America and the Caribbean islands; Chiloé Island on the central Pacific coast of Chile; the Pacific coast of Panama; the North American Gulf and Atlantic coasts from Tamaulipas, Mexico through Quebec, Canada; the interior of South America; and the interior of the United States and Canada west at least as far as the Great Plains (Bimbi *et al.* 2014, S. Koch, L. Niles, R. Porter, and F. Sanders pers. comm. August 8 and 12, 2014; Newstead 2014a, D. Newstead pers. comm. May 8, 2014, Niles 2014, J. Parvin pers. comm. March 13, 2014, Newstead *et al.* 2013, Burger *et al.* 2012b, Niles 2012a, Niles *et al.* 2012a, Niles 2011a, Niles 2011b, Niles *et al.* 2010a, Niles *et al.* 2008, B. Paxton pers. comm. November 9, 2008, Buehler 2002, Morrison and Harrington 1992).

Wintering

Wintering areas for the rufa red knot include the Atlantic coasts of Argentina and Chile (particularly the island of Tierra del Fuego that spans both countries), the north coast of Brazil (particularly in the State of Maranhão), the Northwest Gulf of Mexico from the Mexican State of Tamaulipas through Texas (particularly at Laguna Madre) to Louisiana, and the Southeast United States from Florida (particularly the central Gulf coast) to North Carolina (Newstead 2014a, Newstead *et al.* 2013, L. Patrick pers. comm. August 31, 2012, Niles *et al.* 2008) (**Figure 14**). Smaller numbers of knots winter in the Caribbean, and along the central Gulf coast (Alabama, Mississippi), the mid-Atlantic, and the Northeast United States (eBird.org 2014; Russell 2014, Burger *et al.* 2012b, A. Dey pers. comm. November 19, 2012, H. Hanlon pers. comm. November 22, 2012, Niles *et al.* 2012a, L. Patrick pers. comm. August 31, 2012, Morrison and Harrington 1992). *Calidris canutus* is also known to winter in Central America, northwest South America, and along the Pacific coast of South America, but it is not yet clear if all these birds are the *rufa* subspecies (Carmona *et al.* 2013).

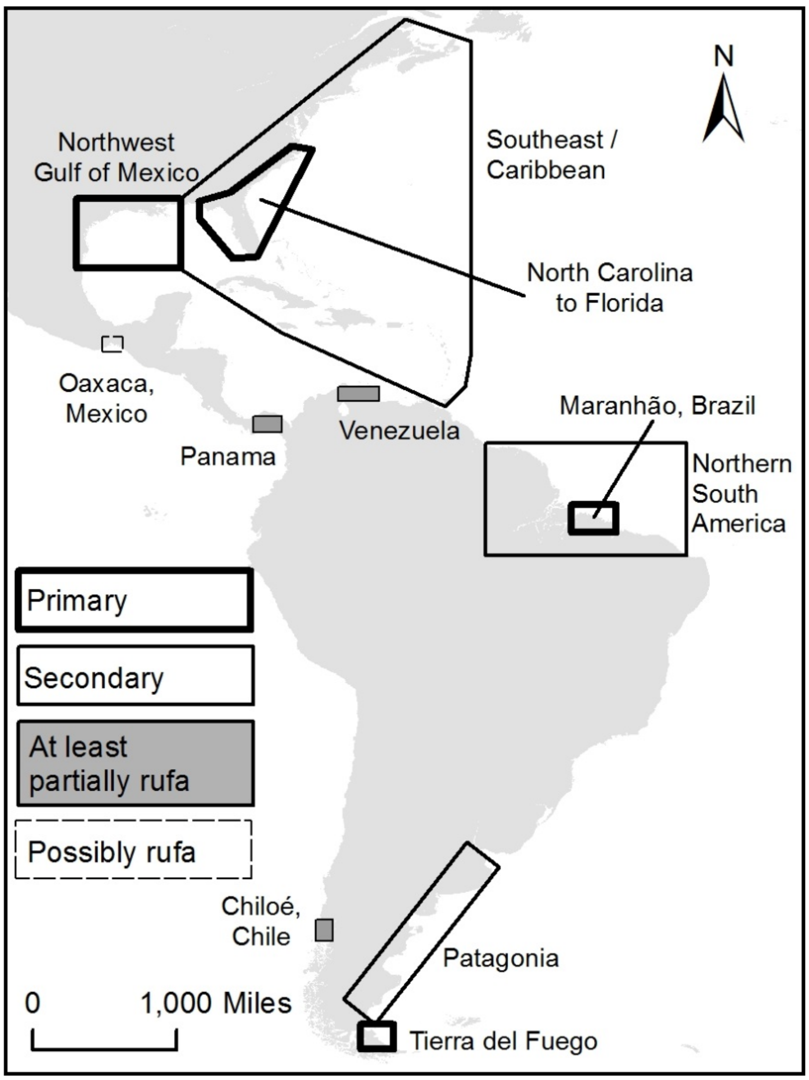

Figure 14. Known rufa red knot wintering areas.

Spring Migration

Well-known spring stopover areas along the Atlantic coast include Río Gallegos, Península Valdés, and San Antonio Oeste (Patagonia, Argentina); Lagoa do Peixe (eastern Brazil, State of Rio Grande do Sul); Maranhão (northern Brazil); the Southeast United States (e.g., the Carolinas to Florida); the Virginia barrier islands (United States); and Delaware Bay (Delaware and New Jersey, United States) (A. Dey pers. comm. April 21, 2014, Wallover *et al.* 2014, GDNR 2013, South Carolina Department of Natural Resources (SCDNR) 2013, Cohen *et al.* 2009, Niles *et al.* 2008, González 2005). However, large and small groups of red knots, sometimes numbering in the thousands, may occur in suitable habitats all along the Atlantic and Gulf coasts from Argentina to Massachusetts (Niles *et al.* 2008).

Although a few birds may depart before the end of January, the main red knot movement north from Tierra del Fuego occurs in February. The northward migration through South America is typically rapid, with only brief stopovers (Niles *et al*. 2008), although longer stops in Argentina (17 to 22 days) have been reported (Musmeci *et al.* 2012). Birds moving north from Argentina typically arrive in Brazil in April (Scherer and Petry 2012, Niles *et al.* 2008). Departure from Brazil tends to occur in the first half of May (Niles *et al.* 2010a, Niles *et al*. 2008). Many knots marked in Argentina and Chile are seen on the Atlantic coasts of Florida, Georgia, South Carolina, and North Carolina during, but not before, May (B. Harrington pers. comm. November 14, 2013, GDNR 2013, SCDNR 2013). Available data indicate that red knots wintering in the Southeast use at least two different spring migration routes—coastal (moving north along the coast to the mid-Atlantic before departing for the Arctic) and inland (departing overland for the Arctic directly from the Southeast coast) (Bimbi *et al.* 2014, SCDNR 2013, Niles *et al.* 2012a, Harrington 2005a, Morrison and Harrington 1992).

Fall Migration

Departure from the breeding grounds begins in mid-July and continues through August. Females are thought to leave first, followed by males and then juveniles (Niles *et al.* 2008, Harrington 2001). Adult *Calidris canutus* pass through stopover sites along the migratory route earlier in years with low reproductive success than in years with high reproductive success (Blomqvist *et al*. 2002). Along the U.S. Atlantic coast, southbound red knots start arriving in July. Numbers of adults peak in mid-August and most depart by late September, although geolocators and resightings have shown some birds (especially northern-wintering knots) stay through November (Wallover *et al.* 2014, Niles *et al.* 2012a, Harrington *et al.* 2010b, Harrington 2001). Well-known fall stopover sites include southwest Hudson Bay (including the Nelson River delta), James Bay, the north shore of the St. Lawrence River, the Mingan Archipelago, and the Bay of Fundy in Canada; the coasts of Massachusetts and New Jersey and the mouth of the Altamaha River in Georgia in the U.S.; the Caribbean (especially Puerto Rico and the Lesser Antilles); and the northern coast of South America from Brazil to Guyana (eBird.org 2014, Autoridad de Energía Eléctrica (Electric Energy Authority, or (AEE) 2013, Newstead *et al.* 2013, Niles 2012a, D. Mizrahi pers. comm. October 16, 2011, Niles *et al.* 2010a, Schneider and Winn 2010, Niles *et al.* 2008, B. Harrington pers. comm. March 31, 2006, Antas and Nascimento 1996, Morrison and Harrington 1992, Spaans 1978). However, birds can occur all along the coasts in suitable habitats. In one study of northern-wintering red knots, the total time spent along the U.S. Atlantic coast (including spring, fall, and for some birds winter) averaged 218 days (range 121 to 269 days) (Burger *et al.* 2012b), or about 60 percent of the calendar year.

Midcontinental Migration

Geolocator results from red knots wintering in Texas have shown that these birds typically use a central, overland flyway across the midcontinental United States, with birds departing Texas between May 16 and May 21 and using stopover areas in the Northern Great Plains and along southern Hudson Bay (Newstead *et al.* 2013). Texas-wintering birds typically use a similar and direct interior flyway across the midcontinental United States during the southbound migration, using a southbound stopover site on the south shore of Hudson Bay (Nelson River Delta to James Bay). Geolocator results (Bimbi *et al.* 2014, Niles 2014, Newstead *et al.* 2013, Niles *et al.* 2012a, Niles 2011a, Niles 2011b, Niles *et al.* 2010a) have suggested that rufa red knots exhibit strong flyway fidelity (Newstead *et al.* 2013) (i.e., not switching between Atlantic coast and midcontinental routes). However, newer geolocator data, as yet unpublished, do show some switching between these two flyways. Several Texas-wintering birds have been shown to use the "typical" midcontinental flyway in spring, but then follow a fall migration route along the U.S. Atlantic coast before returning Texas via the Gulf coast. To date, no known geolocator tracks from Texas birds have shown use of the Atlantic coast during spring migration, but some resighting data suggest that this may also occur (D. Newstead pers. comm. May 8, 2014). Even for the same individual bird, the actual routes and number of stopovers can vary considerably from year to year (D. Newstead pers. comm. May 8, 2014). In one study, red knots wintering in the Northwest Gulf of Mexico spent nearly the entire nonbreeding phase of their annual cycle (286 days, or 78.4 percent of the calendar year) on the Texas coast (Newstead *et al.* 2013).

8.1.4. Conservation Needs and Threats

Red knots face a wide range of threats across their range on multiple geographic and temporal scales. A number of threats are likely contributing to habitat loss, anthropogenic mortality, or both, and thus contribute to the red knot's threatened status, particularly considering the cumulative and synergistic effects of these threats, and that several key populations of this species have already undergone considerable declines. The effects of some smaller threats may act in an additive fashion to ultimately impact populations or the subspecies as a whole (cumulative effects). Other threats may interact synergistically to increase or decrease the effects of each threat relative to the effects of each threat considered independently (synergistic effects). For example, reduced food availability has been shown to interact synergistically with asynchronies and several other threats, such as asynchronies, disturbance, predation pressure, and competition with gulls (Escudero *et al.* 2012, Dey *et al.* 2011a, Breese 2010, Niles *et al*. 2008, Atkinson *et al.* 2007, Niles *et al.* 2005, Baker *et al.* 2004).

Threats to Red Knots

Threats to the red knot from habitat destruction and modification are occurring throughout the entire range of the subspecies. These threats include climate change, shoreline stabilization, and coastal development, exacerbated regionally or locally by lesser habitat-related threats such as beach cleaning, invasive vegetation, agriculture, and aquaculture. The subspecies-level impacts from these activities are expected to continue into the future.

Accelerating Sea Level Rise

Due to background rates of sea level rise and the naturally dynamic nature of coastal habitats, it is likely that red knots are adapted to moderate (although sometimes abrupt) rates of habitat change in their wintering and migration areas. However, rates of sea level rise have accelerated beyond those that have occurred over recent millennia and continue to increase (IPCC 2013a). In most of the red knot's nonbreeding range, shorelines are expected to undergo dramatic reconfigurations over the next century as a result of accelerating sea level rise (CCSP 2009b). Extensive areas of marsh are likely to become inundated, which may reduce foraging and roosting habitats. Marshes may be able to establish farther inland, but the rate of new marsh formation (e.g., intertidal sediment accumulation, development of hydric soils, colonization of marsh vegetation) may be slower than the rate of deterioration of existing marsh, particularly under the higher sea level rise scenarios (Nikitina *et al.* 2013, Glick *et al.* 2008). The primary red knot foraging habitats, intertidal flats and sandy beaches, will likely be locally or regionally inundated or eroded, but replacement habitats are likely to reform along the shoreline in its new position (CCSP 2009b, Scavia *et al*. 2002). However, if shorelines experience a decades-long period of high instability and landward migration (e.g., under higher rates of sea level rise), the formation rate of new beach habitats may be slower than the rate of loss of existing habitats (Iwamura *et al.* 2013). In addition, low-lying and narrow islands (e.g., in the Caribbean and along the Gulf and Atlantic coasts) may disintegrate rather than migrate (Chapter 5 in IPCC 2014, Titus 1990), representing a net loss of red knot habitat.

Shoreline Stabilization and Coastal Development

Nonbreeding Range

Within the nonbreeding portion of the range, red knot habitat is primarily threatened by the highly interrelated effects of sea level rise, shoreline stabilization, and coastal development. Superimposed on changes from sea level rise are widespread human efforts to stabilize the shoreline, which are known to exacerbate losses of intertidal habitats by blocking their landward migration. About 40 percent of the U.S. coastline within the range of the red knot is already developed, and much of this developed area is stabilized by a combination of existing hard structures and ongoing beach nourishment programs (Rice 2012a, Titus *et al.* 2009). Hard stabilization structures and dredging degrade and often eliminate existing intertidal habitats, and in many cases prevent the formation of new shorebird habitats (CCSP 2009b, Nordstrom 2000). Beach nourishment may temporarily maintain suboptimal shorebird habitats where they would otherwise be lost as a result of hard structures or sea level rise (Nordstrom and Mauriello 2001), but beach nourishment can also have adverse effects to red knots and their habitats (Defeo *et al.* 2009, Rice 2009, Peterson *et al.* 2006, Peterson and Bishop 2005, Greene 2002). Demographic and economic pressures remain strong to continue existing programs of shoreline stabilization, and to develop additional areas (Melillo *et al.* 2014, Nordstrom 2000), with an estimated 20 to 33 percent of the coast still available for development (Rice 2012a, Titus *et al.* 2009). However, we expect existing beach nourishment programs will likely face eventual constraints of budget and sediment availability as sea level continues to rise (BOEM 2014b, NJDEP 2010, Titus *et al.* 1991, Weggel 1986). In those times and places where artificial beach maintenance is abandoned (e.g., due to constraints on funding or sediment availability), the remaining alternatives available to coastal communities would likely be limited to either a retreat from the coast or increased use of hard structures to protect development (CCSP 2009b, Defeo *et al*. 2009). The quantity of red knot habitat would be markedly decreased by a proliferation of hard structures. Red knot habitat would be significantly increased by retreat, but only where hard stabilization structures do not exist or where they get dismantled. We have little information about coastal development in the red knot's non-U.S. migration and wintering areas, compared to U.S. nonbreeding areas. However, escalating pressures caused by the combined effects of population growth, demographic shifts, economic development, and global climate change pose unprecedented threats to sandy beach ecosystems worldwide (Defeo *et al.* 2009, Schlacher *et al.* 2008a). However, in some key international wintering and stopover sites, development pressures are likely to exacerbate habitat impacts caused by sea level rise (CSRPN 2013, WHSRN 2012, Niles *et al.* 2008, Ferrari *et al.* 2002). The cumulative loss of habitat across the nonbreeding range could affect the ability of red knots to complete their annual cycles, possibly affecting fitness and survival, and is thereby likely to negatively influence the long-term survival of the rufa red knot.

Beach Cleaning

On beaches that are heavily used for tourism, mechanical beach cleaning (also called beach grooming or raking) is a common practice to remove wrack (seaweed and other organic debris are deposited by the tides), trash, and other natural or manmade debris by raking or sieving the sand, often with heavy equipment (Defeo *et al.* 2009). Beach raking became common practice in New Jersey in the late 1980s (Nordstrom and Mauriello 2001) and is increasingly common in the Southeast, especially in Florida (M. Bimbi pers. comm. November 1, 2012). In Texas, wrack removal and beach raking occur on the Gulf beach side of the developed portion of South Padre Island in the Lower Laguna Madre (USFWS 2012a), as well as on North Padre Island, Mustang Island, and Galveston Island (D. Newstead pers. comm. May 8, 2014), all known red knot areas. Along with beach nourishment, intensive beach grooming has probably reduced the capacity the southern edge of South Padre Island to support red knots (Newstead 2014a). On the Southeast Atlantic and Gulf coasts, beach cleaning occurs on private beaches and on some municipal or county beaches that are used by red knots (M. Bimbi pers. comm. November 1, 2012). Most wrack removal on State and Federal lands is limited to post-storm cleanup and does not occur regularly (USFWS 2012a).

Practiced routinely, beach cleaning can cause considerable physical changes to the beach ecosystem. In addition to removing humanmade debris, beach cleaning and raking machines remove accumulated wrack, topographic depressions, emergent foredunes and hummocks, and sparse vegetation (USFWS 2012a, Defeo *et al.* 2009, Nordstrom and Mauriello 2001, Nordstrom 2000), all of which can be important microhabitats for shorebirds and their prey. Many of these changes promote erosion. Grooming loosens the beach surface by breaking up surface crusts (salt and algae) and lag elements (shells or gravel), and roughens or "fluffs" the sand, all of which increase the erosive effects of wind (Cathcart and Melby 2009, Defeo *et al.* 2009, Nordstrom 2000). Grooming can also result in abnormally broad unvegetated zones that are inhospitable to dune formation or plant colonization, thereby enhancing the likelihood of erosion

(Defeo *et al.* 2009). By removing vegetation and wrack, cleaning machines also reduce or eliminate natural sand-trapping features, further destabilizing the beach (USFWS 2012a, Nordstrom *et al*. 2006b, Nordstrom 2000). Further, the sand adhering to seaweed and trapped in the cracks and crevices of wrack is lost to the beach when the wrack is removed; although the amount of sand lost during a single sweeping activity is small, over a period of years this loss could be significant (USFWS 2012a). Cathcart and Melby (2009) found that beach raking and grooming practices on mainland Mississippi beaches exacerbate the erosion process and shorten the time interval between beach nourishment projects (see discussion of shoreline stabilization, above). In addition to promoting erosion, raking also interferes with the natural cycles of dune growth and destruction on the beach (Nordstrom and Mauriello 2001).

Wrack removal also has significant ecological consequences, especially in regions with high levels of marine macrophyte (e.g., seaweed) production. The community structure of sandy beach macroinvertebrates can be closely linked to wrack deposits, which provide both a food source and a microhabitat refuge against desiccation (drying out). Wrack-associated animals, such as amphipods, isopods, and insects, are significantly reduced in species richness, abundance, and biomass by beach grooming (Defeo *et al.* 2009). Invertebrates in the wrack are a primary prey base for some shorebirds such as piping plovers (USFWS 2012a), but generally make up only a secondary part of the red knot diet. Overall shorebird numbers are positively correlated with wrack cover and the biomass of their invertebrate prey that feed on wrack; therefore, grooming can lower bird numbers (USFWS 2012a, Defeo *et al.* 2009). Due to their specialization on benthic, intertidal mollusks, red knots may be less impacted by these effects than some other shorebird species. However, removal of wrack may cause more significant localized effects to red knots at those times and places where abundant mussel spat are attached to deposits of tide-cast material, or where red knots become more reliant on wrack-associated prey species such as amphipods, insects, and marine worms. In Delaware Bay, red knots preferentially feed in the wrack line because horseshoe crab eggs become concentrated there (Nordstrom *et al.* 2006a, Karpanty *et al.* 2011). However, removal of wrack material is not practiced along Delaware Bay beaches (K. Clark pers. comm. February 11, 2013, A. Dey and K. Kalasz pers. comm. February 8, 2013).

The heavy equipment used in beach grooming can cause disturbance to roosting and foraging red knots. Because beach cleaning generally occurs on beaches intensively used for human recreation, disturbance to red knots from these recreational activities may, on many beaches, be greater than the disturbance from the beach cleaning machines. However, beach cleaning may occur at times of day (e.g., early morning, evening) when few recreational activities are taking place, thus increasing total daily duration that knots are disturbed by human activities. Conversely, many raked beaches may have such high levels of human recreational use that red knots are precluded from using them entirely; in such cases there would be no incremental additional disturbance from the raking activities. Where it occurs, disturbance from beach grooming may be more problematic for roosting than foraging birds because roosting red knots are particularly vulnerable to disturbance, and because beach grooming is typically focused along or landward of the high tide line where red knots may roost but are unlikely to forage. On mid-Atlantic and northern Atlantic beaches, raking is most prevalent from Memorial Day to Labor Day. In the latter part of this period (late July and August), hundreds to thousands of red knots may occur at stopover habitats in this region (B. Harrington pers. comm. November 14, 2013, eBird.org 2014). There is no information regarding the extent to which raking is practiced in fall stopover areas when red knots are present. Further south, raking may occur year-round.

In summary, the practice of intensive beach raking may cause physical changes to beaches that degrade their suitability as red knot habitat. Removal of wrack may also have an effect on the availability of red knot food resources, particularly in those times and places that birds are more reliant on wrack-associated prey items. Beach cleaning machines are likely to cause disturbance to nonbreeding red knots, particularly roosting birds. Mechanized beach cleaning is widespread within the red knot's U.S. range, particularly in developed areas. Beach grooming may expand in some areas that become more developed but may decrease in other areas due to increasing environmental regulations, such as restrictions on beach raking in piping plover nesting areas (e.g., Nordstrom and Mauriello 2001).

Invasive Vegetation

Defeo *et al.* (2009) cited biological invasions of both plants and animals as global threats to sandy beaches, with the potential to alter food webs, nutrient cycling, and invertebrate assemblages. Although the extent of the threat is uncertain, this may be due to poor survey coverage more than an absence of invasions (USFWS 2012a). The propensity of invasive species to spread, and their tenacity once established, make them a persistent problem that is only partially countered by increasing awareness and willingness of beach managers to undertake control efforts (USFWS 2012a). Like most invasive species, exotic coastal plants tend to reproduce and spread quickly and exhibit dense growth habits, often outcompeting native plants (USFWS 2012a, Bahamas National Trust 2010, True 2009, Invasive Plant Atlas of New England undated). If left uncontrolled, invasive plants can cause a habitat shift from open or sparsely vegetated sand to dense vegetation (USFWS 2012a, True 2009, City of Sanibel undated, Invasive Plant Atlas of New England undated). Many invasive species are either affecting or have the potential to affect coastal beaches (USFWS 2012a), and thus red knot habitat. In nonbreeding habitats, *Calidris canutus* require sparse vegetation to avoid predation (Niles *et al.* 2008, Piersma *et al.* 1993).

Beach vitex (*Vitex rotundifolia*) is a woody vine introduced into the Southeast as a dune stabilization and ornamental plant that has spread from Virginia to Florida and west to Texas (Westbrooks and Madsen 2006). There are hundreds of beach vitex occurrences in North and South Carolina, and a small number of known locations in Georgia and Florida. Targeted beach vitex eradication efforts have been undertaken in the Carolinas (USFWS 2012a). Crowfootgrass (*Dactyloctenium aegyptium*), which grows invasively along portions of the Florida coastline, forms thick bunches or mats that can change the vegetative structure of coastal plant communities and thus alter shorebird habitat (USFWS 2009).

Japanese (or Asiatic) sand sedge (*Carex kobomugi*) is a 4- to 12-in (10- to 30-cm) tall perennial sedge adapted to coastal beaches and dunes (Plant Conservation Alliance 2005, Invasive Plant Atlas of New England undated). The species occurs from Massachusetts to North Carolina (U.S. Department of Agriculture (USDA) 2013) and spreads primarily by vegetative means through production of underground rhizomes (horizontal stems) (Plant Conservation Alliance 2005). Japanese sand sedge forms dense stands on coastal dunes, outcompeting native vegetation and increasing vulnerability to erosion (Plant Conservation Alliance 2005, Invasive Plant Atlas of

New England undated). In the 2000s, Wootton (2009) documented rapid (exponential) growth in the spread of Japanese sand sedge at two New Jersey sites that are known to support shorebirds.

Australian pine (*Casuarina equisetifolia*) is not a true pine, but is actually a flowering plant. Australian pine affects shorebirds by encroaching on foraging and roosting habitat and may also provide perches for avian predators (USFWS 2012a, Bahamas National Trust 2010). Native to Australia and southern Asia, Australian pine is now found in all tropical and many subtropical areas of the world. This species occurs on nearly all islands of the Bahamas (Bahamas National Trust 2010), and is among the three worst invasive exotic trees damaging wildlife habitat throughout South Florida (City of Sanibel undated). Growing well in sandy soils and salt tolerant, Australian pine is most common along shorelines (Bahamas National Trust 2010), where it grows in dense monocultures with thick mats of acidic needles (City of Sanibel undated). In the Bahamas, Australian pine often spreads to the edge of the intertidal zone, effectively usurping all shorebird roosting habitat (A. Hecht pers. comm. December 6, 2012). In addition to directly encroaching into shorebird habitats, Australian pine contributes to beach loss through physical alteration of the dune system (Stibolt 2011, Bahamas National Trust 2010, City of Sanibel undated). The State of Florida prohibits the sale, transport, and planting of Australian pine (Stibolt 2011, City of Sanibel undated).

In summary, red knots require open habitats that allow them to see potential predators and that are away from tall perches used by avian predators. Invasive species, particularly woody species, degrade or eliminate the suitability of red knot roosting and foraging habitats by forming dense stands of vegetation. Although not a primary cause of habitat loss, invasive species can be a regionally important contributor to the overall loss and degradation of the red knot's nonbreeding habitat.

Agriculture and Aquaculture

In some localized areas within the red knot's range, agricultural activities or aquaculture are impacting habitat quantity and quality. For example, on the Magdalen Islands, Canada (Province of Quebec), clam farming is a growing local business. The clam farming location overlaps with the feeding grounds of transient red knots, and foraging habitats are being affected. Clam farming involves extracting all the juvenile clams from an area and relocating them in a "nursery area" nearby. The top sand layer (upper 3.9 in (10 cm) of sand) is removed and filtered. Only the clams are kept, and the remaining fauna are rejected on the site. This disturbance of benthic fauna could affect foraging rates and weight gain in red knots by removing prey, disturbing birds, and altering habitat. This pilot clam farming project could expand into more demand for clam farming in other red knot feeding areas in Canada (USFWS 2011b).

Luckenbach (2007) found that aquaculture of clams (*Mercenaria mercenaria*) in the lower Chesapeake Bay occurs in close proximity to shorebird foraging areas. The current distribution of clam aquaculture in the very low intertidal zone minimizes the amount of direct overlap with shorebird foraging habitats, but if clam aquaculture expands farther into the intertidal zone, more shorebird impacts (e.g., habitat alteration) may occur. However, these Chesapeake Bay intertidal zones are not considered the primary habitat for red knots (Cohen *et al.* 2009), and red knots were not among the shorebirds observed in this study (Luckenbach 2007).

Oyster aquaculture is practiced in Delaware Bay (NJDEP 2011), and this practice, to date, has had minimial documented effects to red knots. However, as of fall 2014, the Service is aware of two proposed nearshore (intertidal) aquaculture projects on New Jersey's side of Delaware Bay, and three existing operations. Some of the existing operations may wish to expand. Nearshore aquaculture could result in more substantial knot effects than offshore (subtidal) operations. For example, if aquaculture structures or activities are permitted in intertidal habitats during the spring stopover period, they would likely disturb red knots and could create a barrier to horseshoe crab movement. Federal and State agencies are working to minimize adverse effects to red knots from Delaware Bay aquaculture activities.

Shrimp (Family Penaeidae, mainly *Litopenaeus vannamei*) farming has expanded rapidly in Brazil in recent decades. Particularly since 1998, extensive areas of mangroves and salt flats, important shorebird habitats, have been converted to shrimp ponds (Carlos *et al.* 2010). In addition to causing habitat conversion, shrimp farm development has caused deforestation of river margins (e.g., for pumping stations), pollution of coastal waters, and changes in estuarine and tidal flat water dynamics (Campos 2007, Zitello 2007). Ninety-seven percent of Brazil's shrimp production is in the Northeast region of the country (Zitello 2007). Carlos *et al.* (2010) evaluated aerial imagery from 1988 to 2008 along 435 mi (700 km) of Brazil's northeast coastline in the States of Piauí, Ceará, and Rio Grande do Norte, covering 20 estuaries. Over this 20-year period, shrimp farms increased by 36,644 acres (ac) (14,829 hectares (ha)), while salt flats decreased by 34,842 ac. (14,100 ha.) and mangroves decreased by 2,876 ac. (1,164 ha.) (Carlos *et al.* 2010).

In the region of Brazil with the most intensive shrimp farming (the Northeast), newer surveys from the 2000s have documented more red knots than were previously known to use this area from earlier surveys in the 1980s. However, considering the extensive loss of shorebird habitat over this period, the difference between these two surveys does not likely represent a population increase, but rather likely reflects differences in survey methodology, intensity, and coverage. In winter aerial surveys of Northeast Brazil in 1983, Morrison and Ross (1989) documented only 15 red knots in the States of Ceará, Piauí, and eastern Maranhão. However, ground surveys in the State of Ceará in December 2007, documented an average peak count of 481 ± 31 wintering red knots at just one site, Cajuais Bank (Carlos *et al.* 2010), which is located immediately adjacent to the 1983 survey area. Cajuais Bank also supports considerable numbers of red knots during migration, with an average peak count of 434 ± 95 in September 2007, (Carlos *et al.* 2010). Over this 1-year study, red knots were the most numerous shorebird at Cajuais Bank, accounting for nearly 25 percent of observations (Carlos *et al.* 2010). Red knots that utilize Northeast Brazil were likely affected by recent habitat losses and degradation from the expansion of shrimp farming.

Farther west along the North-Central coast of Brazil, the western part of Maranhão and extending into the State of Pará is considered an important red knot concentration area during both winter and migration (D. Mizrahi pers. comm. November 17, 2012, Niles *et al.* 2008, Baker *et al.* 2005, Morrison and Ross 1989). Shrimp farm development has been far less extensive in Maranhão and Pará than in Brazil's Northeast region (Campos 2007). However, rapid or unregulated expansion of shrimp farming in Maranhão and Pará could pose an important threat to this key red knot wintering and stopover area (WHSRN 2012). In addition to aquaculture, some fishing is practiced in Maranhão, but the area is fairly protected from conversion to land-based agriculture by its high salinity and inaccessibility (WHSRN 2012). Fishing activities could potentially cause disturbance or alter habitat conditions.

On the east coast of Brazil, Lagoa do Peixe serves as an important migration stopover for red knots. The abundance and availability of the red knot's food supply (snails and other invertebrates) are dependent on the lagoon's water levels. The lagoon's natural fluctuations, and the coastal processes that allow for an annual connection of the lagoon with the sea, are altered by farmers draining water from farm fields into the lagoon. The hydrology of the lagoon is also affected by upland pine (*Pinus* spp.) plantations that cause siltation and lower the water table (Niles *et al*. 2008). These coastal habitats are also degraded by extensive upland cattle grazing, farming of food crops, and commercial shrimp farming. Fishermen also harvest from the lagoon and the sea, with trawlers setting nets along the coast (WHSRN 2012). Fishing activities could potentially cause disturbance or alter habitat conditions.

The red knot wintering and stopover area of Río Gallegos is located on the south coast of Argentina, just north of Tierra del Fuego. The lands surrounding the estuary have historically been used for raising cattle. During the past few years significant areas of brush land (that had served as a buffer) next to the shorebird reserve have been cleared and designated for agricultural use and the establishment of small farms. This loss of buffer areas may cause an increase in disturbance of the shorebirds (WHSRN 2012) because agricultural activities within visual distance of roosting or foraging shorebirds, including red knots, may cause the birds to flush. Regarding aquaculture, Goldfeder and Blanco in Boere *et al.* (2006) cited sea farming projects as a potential threat to the red knot in Argentina. Likewise, aquaculture and seaweed farming could alter prey composition for *Calidris canutus* on Chiloè Island on the Pacific coast of Chile (B. Andres pers. comm. July 21, 2014).

Grazing of the upland buffer is also a problem at Bahía Lomas in Chilean Tierra del Fuego. The government owns all intertidal land and an upland buffer extending 262 ft (80 m) above the highest high tide, but ranchers graze sheep into the intertidal vegetation. Landowners have indicated willingness to relocate fencing to exclude sheep from the intertidal area and the upland buffer, but as of 2011, funding was needed to implement this work (L. Niles pers. comm. March 2, 2011). Grazing in the intertidal zone could potentially displace roosting and foraging red knots, as well as degrade the quality of habitat through trampling, grazing, and feces.

In summary, moderate numbers of red knots that winter or stopover in Northeast Brazil are likely impacted by past and ongoing habitat loss and degradation due to the rapid expansion of shrimp farming. Expansion of shrimp farming in North-Central Brazil, if it occurs, would affect far more red knots. Localized clam farming in Canada could degrade habitat quality and prey availability for transient red knots, and aquaculture may be impacting red knot habitats in Argentina and on Chiloé Island, Chile. Farming practices around Lagoa do Peixe are degrading habitats at this red knot stopover site. Agriculture is contributing to shorebird habitat loss and degradation at Río Gallegos in Argentina, and probably at other localized areas within the range of the red knot. However, clam farming in the Chesapeake Bay and Delaware Bay aquaculture do not appear to be impacting red knots at this time. Agriculture and aquaculture activities are a minor but locally important contributor to overall loss and degradation of the red knot's nonbreeding habitat.

Breeding Range

Within the breeding portion of the range, the primary threat to red knot habitat is from climate change. With arctic warming, vegetation conditions on the breeding grounds are changing, which is expected to eventually cause the zone of nesting habitat to shift north and contract (Feng *et al.* 2012, Meltofte *et al.* 2007, Arctic Climate Impact Assessment (ACIA) 2005).

Habitat Loss and Ecosystem Change from Arctic Warming

Arctic regions are warming about twice as fast as the global average rate (IPCC 2013a), and the Canadian Archipelago is predicted to experience some of the fastest warming in the Arctic (ACIA 2005). Red knots currently breed in a region of sparse, low tundra vegetation within the southern part of the High Arctic and the northern limits of the Low Arctic (CAFF 2010, Niles *et al.* 2008, Niles *et al.* in Baker 2001). Forests are expected to colonize the southern part of the red knot's current breeding range by 2100 (ACIA 2005), and vegetation throughout the entire breeding range is likely to become taller and denser and with less bare ground, potentially making it unsuitable for red knot nesting, possibly as soon as mid-century (Galbraith *et al.* 2014, Pearson *et al.* 2013, COSEWIC 2007, Zöckler and Lysenko 2000). Studies have already documented changes in arctic vegetation, including increases in peak "greenness" of North American tundra vegetation since 1982; increases in plant biomass linked to warming arctic temperatures; advancing of the arctic tree line; increased shrub abundance, biomass, and cover; increased plant canopy heights; and decreased prevalence of bare ground (Summary for Policymakers in IPCC 2014, Chapter 28 in IPCC 2014). Vegetation changes are likely accelerated near coastlines, where red knots breed, due to the loss of sea ice that currently cools the adjacent land (Bhatt *et al*. 2010). Loss of sea ice may also make the central Canadian island habitats more maritime-dominated and, therefore, less suitable for breeding shorebirds (Meltofte *et al.* 2007). The red knot's breeding range is also experiencing changes in freshwater wetland foraging habitats (Meltofte *et al.* 2007, ACIA 2005), as well as unpredictable but profound ecosystem changes (e.g., changing interactions among predators, prey, and competitors) (Meltofte *et al.* 2007). The IPCC notes early warning signs that arctic ecosystems are already experiencing irreversible regime shifts (Summary for Policymakers in IPCC 2014). Ecosystem changes in the Arctic are already underway and likely to continue, and arctic ecosystems likely face much greater future change that may be abrupt, irreversible, or both. The red knot's adaptive capacity to withstand these changes in place, or to shift its breeding range northward, is unknown.

Hunting

Legal and illegal sport and market hunting in the mid-Atlantic and Northeast United States substantially reduced red knot populations in the 1800s, and we do not know if the subspecies ever fully recovered its former abundance or distribution (Karpanty *et al.* 2014, Cohen *et al.* 2008, Harrington 2001). Neither legal nor illegal hunting are currently a threat to red knots in the U.S., but both occur in the Caribbean and parts of South America (Harrington 2001).

Hunting pressure on shorebirds in the Lesser Antilles (e.g., Barbados, Guadeloupe) is very high (USFWS 2011c), but only small numbers of red knots have been documented on these islands, so past mortality may not have exceeded tens of birds per year (G. Humbert pers. comm. November 29, 2013). Red knots are no longer being targeted in Barbados or Guadeloupe, and other measures to regulate shorebird hunting on these islands are being negotiated (G. Humbert pers. comm. November 29, 2013, McClain 2013, USFWS 2011c). Much larger numbers (thousands) of red knots occur in the Guianas, where legal and illegal subsistence shorebird hunting is common (CSRPN 2013, Niles 2012b, Ottema and Spaans 2008). About 20 red knot mortalities have been documented in the Guianas (D. Mizrahi pers. comm. October 16, 2011, Harrington 2001), but total red knot hunting mortality in this region cannot be surmised. As of 2013, shorebird hunting was unregulated in French Guiana (A. Levesque pers. comm. January 8, 2013, D. Mizrahi pers. comm. October 16, 2011). However, a ban on hunting all shorebird species has been proposed in French Guiana (CSRPN 2013), and the red knot was designated a protected species in October 2014 (C. Carichiopulo and N. de Pracontal pers. comm. October 10, 2014). Subsistence shorebird hunting was also common in northern Brazil, but has decreased in recent decades (Niles *et al.* 2008).

There is no evidence that hunting was a driving factor in red knot population declines in the 2000s, or that hunting pressure is increasing. While only low to moderate red knot mortality is documented, additional undocumented mortality is likely. The findings of Watts (2010) suggest that even moderate (hundreds of birds) direct human-caused mortality may begin to have population-level effects on the red knot. There is no reliable information to reasonably know if hunting mortality is or was previously at this level in the Guianas, though it was likely much lower (tens of birds) in the Caribbean islands.

Disease

Red knots may be adapted to parasite-poor habitats and may, therefore, be susceptible to parasites when migrating or wintering in high-parasite regions (Piersma 1997). However, there is no evidence that parasites have affected red knot populations beyond causing normal, background levels of mortality (D'Amico *et al*. 2008, Harrington 2001), and there are no indications that parasite infection rates or red knot fitness impacts are likely to increase. For the most prevalent viruses found in shorebirds within the red knot's geographic range (e.g., avian influenza, avian paramyxovirus), infection rates in red knots are low, and health effects are minimal or have not been documented (D. Stallknecht pers. comm. January 25, 2013, Maxted *et al.* 2012, Coffee *et al.* 2010, Escudero *et al.* 2008, Niles *et al*. 2008, D'Amico *et al.* 2007). However, an unlikely but potentially high-impact, synergistic effect among avian influenza, environmental contaminants, and climate change could produce a population-level impact in Delaware Bay.

Predation

Outside of the breeding grounds, predation is not directly effecting red knot populations despite some mortality (Niles *et al.* 2008). At key stopover sites, however, localized predation pressures exacerbate other threats to red knot populations by pushing red knots out of otherwise suitable foraging and roosting habitats, causing disturbance, and possibly causing changes to stopover duration or other aspects of the migration strategy (Niles 2010a, Watts 2009b, Niles *et al.* 2008, Lank *et al.* 2003). In addition, predation pressure may induce sublethal physiological stress that can impact shorebird fitness (Clark and Clark 2002). The direct and indirect effects of predators are likely to continue at the same level or decrease slightly over the next few decades.

Within the breeding range, normal 3- to 4-year cycles of high predation, mediated by rodent (e.g., lemming) cycles, result in years with extremely low reproductive output but do not threaten the survival of the red knot at the subspecies level (Niles *et al.* 2008, Meltofte *et al.* 2007). It is believed shorebirds, such as red knots, have adapted to these cycles, therefore these natural cycles are not considered a threat to the red knot. What is a threat, however, is that these natural rodent/predator cycles are being disrupted by climate change, which may increase predation rates on shorebirds over the long term and have subspecies-level effects (IPCC 2014, Fraser *et al.* 2013, Brommer *et al*. 2010, Ims *et al.* 2008, Kausrud *et al.* 2008). Disruptions in the rodent-predator cycle pose a substantial threat to the red knot, as they may result in prolonged periods of very low reproductive output (Meltofte *et al.* 2007). Such disruptions have already occurred and may increase due to climate change (IPCC 2014, Fraser *et al.* 2013, Brommer *et al*. 2010, Ims *et al.* 2008, Kausrud *et al.* 2008). The substantial impacts of elevated egg and chick predation on shorebird reproduction are well known (Smith and Wilson 2010, Meltofte *et al.* 2007), although the red knot's capacity to adapt to long-term changes in predation pressure is unknown (Meltofte *et al.* 2007). The threat of persistent increases in predation in the Arctic may already be having subspecies-level effects (Fraser *et al.* 2013) and is anticipated to increase into the future. Further, warming temperatures and changing vegetative conditions in the Arctic are likely to bring additional changes in the predation pressures faced by red knots, such as colonization by new predators from the south, though we cannot forecast how such ecosystem changes are likely to unfold.

Reduced Prey Availablity

Reduced food availability at the Delaware Bay stopover site due to commercial harvest of the horseshoe crab is considered a primary causal factor in the decline of rufa red knot populations in the 2000s (Escudero *et al.* 2012, McGowan *et al.* 2011a, Niles *et al.* 2008, Baker *et al.* 2004). Under the current Adaptive Resource Management (ARM), framework the present horseshoe crab harvest is not considered a threat to the red knot. However, continued implementation of the ARM is imperiled by lack of funding to support the requisite monitoring programs. With or without the ARM, it is not yet known if the horseshoe crab egg resource will continue to adequately support red knot population growth over the next decade. Notwithstanding the importance of the horseshoe crab and Delaware Bay, the red knot faces a range of ongoing and emerging threats to its food resources throughout its range, including small prey sizes from unknown causes (Escudero *et al.* 2012, Espoz *et al.* 2008), warming water and air temperatures (Jones *et al.* 2010), ocean acidification (International Geosphere-Biosphere Programme (IGBP) *et al.* 2013, NRC 2010b), physical habitat changes (IPCC 2014, Rehfisch and Crick 2003, Najjar *et al.* 2000), possibly increased prevalence of disease and parasites (Ward and Lafferty 2004), marine invasive species (Seebens *et al.* 2013, Ruesink *et al.* 2005, Grosholz 2002), and burial and crushing of invertebrate prey from sand placement and recreational activities (Sheppard *et al.* 2009, Schlacher *et al.* 2008b, Schlacher *et al.* 2008c, Greene 2002).

The quantity and quality of red knot prey may be affected by the placement of sediment for beach nourishment or disposal of dredged material. Invertebrates may be crushed or buried during project construction. Although some benthic species can burrow through a thin layer of additional sediment, thicker layers (over 35 in. (90 cm)) smother the benthic fauna (Greene 2002). By means of this vertical burrowing, recolonization from adjacent areas, or both, the benthic faunal communities typically recover. Recovery can take as little as 2 weeks or as long as 2 years, but usually averages 2 to 7 months (Greene 2002, Peterson and Manning 2001). Although many studies have concluded that invertebrate communities recovered following sand placement, study methods have often been insufficient to detect even large changes (e.g., in abundance or species composition), due to high natural variability and small sample sizes (Peterson and Bishop 2005). Therefore, uncertainty remains about the effects of sand placement on invertebrate communities, and how these impacts may affect red knots.

The invertebrate community structure and size class distribution following sediment placement may differ considerably from the original community (Zajac and Whitlatch 2003, Peterson and Manning 2001, Hurme and Pullen 1988). Recovery may be slow or incomplete if placed sediments are a poor grain size match to the native beach substrate (Bricker 2012, Peterson *et al.* 2006, Greene 2002, Peterson *et al.* 2000, Hurme and Pullen 1988), or if placement occurs during a seasonal low point in invertebrate abundance (Burlas 2001). Recovery is also affected by the beach position and thickness of the deposited material (Schlacher *et al.* 2012). If the profile of the nourished beach and the imported sediments do not match the original conditions, recovery of the benthos is unlikely (Defeo *et al.* 2009). Reduced prey quantity and accessibility caused by a poor sediment size match have been shown to affect shorebirds, causing temporary but large (70 to 90 percent) declines in local shorebird abundance (Peterson *et al.* 2006).

Beach nourishment is a regular practice in Delaware Bay and can affect spawning habitat for horseshoe crabs. Although beach nourishment generally preserves horseshoe habitat better than hard stabilization structures, nourishment can enhance, maintain, or decrease habitat value depending on beach geometry and sediment matrix (Smith *et al.* 2002a). In a field study in 2001 and 2002, Smith *et al.* (2002a) found a stable or increasing amount of spawning activity at beaches that were recently nourished while spawning activity at control beaches declined. These authors also found that beach characteristics affect horseshoe crab egg development and viability. Beach nourishment can alter both the beach foreshore (sediment size distribution, slope, and width) and low tide terrace (sediment size distribution, elevation, and width) (Smith *et al.* 2002b). Avissar (2006) modeled nourished versus control beaches and found that nourishment may compromise egg development and viability. Although nourishment is generally considered to be environmentally compatible, the effect of nourishment on horseshoe crab spawning, egg development, and survival of juveniles is understudied (Smith *et al.* 2002b). Evaluating the impacts of beach nourishment projects on horseshoe crab populations has been identified as a high research priority by ASMFC (2013a). Despite possible drawbacks, beach nourishment is often successfully used to restore and maintain horseshoe crab spawning habitat on both sides of Delaware Bay.

Although threats to food quality and quantity are widespread, red knots in localized areas have shown some adaptive capacity to switch prey when the preferred prey species became reduced (Escudero *et al.* 2012, Musmeci *et al.* 2011), suggesting some adaptive capacity to cope with this threat. Nonetheless, based on the combination of documented past impacts and a spectrum of ongoing and emerging threats, reduced quality and quantity of food supplies is a threat to the rufa red knot at the subspecies level, and the threat is likely to continue into the future.

Asynchronies

The red knot's life-history strategy makes this species inherently vulnerable to mismatches in timing between its annual cycle and those periods of optimal food and weather conditions upon which it depends (Galbraith *et al.* 2014, Liebezeit *et al.* 2014, Conklin *et al.* 2010, Gill *et al.* 2013, Hurlbert and Liang 2012, McGowan *et al*. 2011a, Smith *et al.* 2011a, Meltofte *et al.* 2007). The red knot's sensitivity to timing asynchronies has been demonstrated through a population-level response, as the late arrivals of birds in Delaware Bay is generally accepted as a key causative factor (along with reduced supplies of horseshoe crab eggs) behind population declines in the 2000s (Baker *et al*. 2004). The factors that caused delays in the spring migrations of red knots from Argentina and Chile are still unknown (Niles *et al*. 2008), and there is no information to indicate if this delay will reverse, persist, or intensify in the future. Superimposed on the existing threat of late arrivals in Delaware Bay are new threats emerging due to climate change (IPCC 2014, Root *et al.* 2013, Hurlbert and Liang 2012), such as changes in the timing of reproduction for both horseshoe crabs and mollusks (Burrows *et al.* 2011, Poloczanska *et al.* 2013, Smith *et al.* 2010b, van Gils *et al.* 2005a, van Gils *et al.* 2005b, Philippart *et al.* 2003). Climate change may also cause shifts in the period of optimal arctic insect and snow conditions relative to the time period when red knots currently breed (Grabowski *et al.* 2013, McGowan *et al*. 2011a, Smith *et al*. 2010a, Tulp and Schekkerman 2008, Meltofte *et al.* 2007, Piersma *et al.* 2005, Schekkerman *et al.* 2003). The red knot's adaptive capacity to deal with numerous changes in the timing of resource availability across its geographic range is largely unknown (Liebezeit *et al.* 2014, Grabowski *et al.* 2013, Meltofte *et al.* 2007). A few examples suggest some flexibility in red knot migration strategies (D. Newstead pers. comm. May 8, 2014, Grabowski *et al.* 2013, Smith *et al*. 2010a, González *et al.* 2006, González *et al.* <u>in</u> International Wader Study Group (IWSG) 2003), but differences between the annual timing cues of red knots (at least partly celestial and endogenous) (Liebezeit *et al.* 2014, Conklin *et al.* 2010, Gill *et al.* 2013, McGowan *et al.* 2011a, Cadée *et al.* 1996) and their prey (primarily environmental) (Smith *et al.* 2010b, Philippart *et al.* 2003) suggest there are limitations on the adaptive capacity of red knots to cope with increasing frequency or severity of asynchronies.

Disturbance from Recreational Activities

Red knots are exposed to disturbance from recreational and other human activities throughout their nonbreeding range (B. Andres pers. comm. July 21, 2014, B. Harrington pers. comm. November 14, 2013, CSRPN 2013; Escudero *et al.* 2012, WHSRN 2012, USFWS 2011b Niles *et al.* 2008). Excessive disturbance has been shown to preclude red knot use of otherwise preferred habitats (Burger and Niles 2013a, Burger and Niles 2013b, Escudero *et al.* 2012, Foster *et al.* 2009, Karpanty *et al.* 2006, Harrington 2005b) and can impact shorebird energy budgets (Burger and Niles 2013a, Escudero *et al.* 2012, Harrington 2005b, Burger 1986). Both of these effects are likely to exacerbate other threats to the red knot, such as habitat loss, reduced food availability, asynchronies in the annual cycle, and competition with gulls.

Competition with Gulls

Competition with gulls can exacerbate food shortages in Delaware Bay (Dey *et al.* 2011b, Kalasz *et al*. 2010, Niles *et al.* 2008, Burger *et al.* 2007, Hernandez 2005). Despite the growth of gull populations in southern New Jersey, numbers of gulls using Delaware Bay in spring decreased considerably from the early 1990s to the early 2000s (Dey *et al.* 2011b, Sutton and Dowdell 2002). Gull competition was likely not a driving cause of red knot population declines in the 2000s, but was likely one of several factors (along with predation, storms, late arrivals of migrants, and human disturbance) that likely exacerbated the effects of reduced horseshoe crab egg availability.

Gull competition has not been reported as a threat to red knots outside of Delaware Bay (e.g., S. Koch pers. comm. March 5, 2013, K. Iaquinto pers. comm. February 22, 2013), but is likely to exacerbate other threats throughout the knot's range due to gulls' larger body sizes, high aggression (Burger undated, Niles *et al.* 2008, Burger *et al.* 1979), tolerance of human disturbance (Burger *et al.* 2007), and generally stable or increasing populations. However, outside of Delaware Bay, there is typically less overlap between the diets of red knots and most gulls species, which are generalist feeders. The effects of gulls are likely to be most pronounced where red knots become restricted to reduced areas of foraging habitat, which can occur as a result of reduced food resources, human disturbance or predation that excludes knots from quality habitats, or outright habitat loss.

Harmful Algal Blooms (HABs)

Direct mortality of red knots from HABs have been documented only in Texas (Newstead 2014a), although there are anecdotal reports that red tide has also caused red knot sickness and mortality on Florida's west coast (B. Harrington pers. comm. November 14, 2013). A large die-off in Uruguay may have also been linked to an HAB, but this link was not substantiated (J. Aldabe pers. comm. February 4, 2013). Some level of undocumented red knot mortality from HABs likely occurs most years, based on probable underreporting of shorebird mortalities from HABs and the direct exposure of red knots to algal toxins (particularly via contaminated prey) throughout the knot's nonbreeding range. There is no documented evidence that HABs were a driving factor in red knot population declines in the 2000s. However, HAB frequency and duration have increased and do not show signs of abating over the next few decades (Melillo *et al.* 2014, Anderson 2007, FAO 2004). Combined with other threats, ongoing and possibly increasing effects from HABs may be a regionally important contributor to red knot mortality.

Oil Spills and Leaks

Red knots are exposed to large-scale petroleum extraction and transportation operations in many key wintering and stopover habitats including Tierra del Fuego, Patagonia, the Gulf of Mexico, Delaware Bay, and the Gulf of St. Lawrence (NOAA 2013d, Anderson *et al.* 2012, BSEE 2012, WHSRN 2012, USFWS 2011b, Niles *et al.* 2008, Ottema and Spaans 2008, COSEWIC 2007, Gappa and Sueiro 2007, Ferrari *et al.* 2002, Philadelphia Area Committee 1998, Harrington and Morrison 1980). The documented effects to red knots from oil spills and leaks have been minimal; however, information regarding any oiling of red knots during the Deepwater Horizon spill has not yet been released (Natural Resource Trustees 2012). High potential exists for small or medium spills to impact moderate numbers of red knots or their habitats, such that one or more such events is likely over the next few decades, based on the proximity of key red knot habitats to high-volume oil operations. Risk of a spill may decrease with improved spill contingency planning, infrastructure safety upgrades, and improved spill response and recovery methods. However, these decreases in risk (e.g., per barrel extracted or transported) could be offset if the total volume of petroleum extraction and transport continues to grow. A major spill affecting habitats in a key red knot concentration area (e.g., Tierra del Fuego, Gulf coasts of Florida or Texas, Delaware Bay, Mingan Archipelago) while knots are present is less likely but would be expected to cause population-level impacts. Oil spills are not a current threat to the red knot on its arctic breeding grounds. A substantial increase in commercial vessel traffic through the red knot's breeding grounds is likely over coming decades (NRC 2013, Smith and Stephenson 2013), but there is no data to evaluate the risks of this potential future threat.

Environmental Contaminants

Although red knots are exposed to a variety of contaminants across their nonbreeding range, there is no evidence that such exposure is impacting health, survival, or reproduction at the subspecies level. Exposure risks exist in localized red knot habitats in Canada, but best available data suggest shorebirds in Canada are not impacted by background levels of contamination (WHSRN 2012, Braune and Noble 2009, COSEWIC 2007). Levels of most metals in red knot feathers from the Delaware Bay have been somewhat high but generally similar to levels reported from other studies of shorebirds (Burger *et al.* 1993). One preliminary study suggests organochlorines and trace metals are not elevated in Delaware Bay shorebirds, although this finding cannot be confirmed without updated testing (USFWS 1996). Levels of metals in horseshoe crabs are generally low in the Delaware Bay region and not likely impacting red knots or recovery of the crab population (Burger *et al.* 2003, Burger *et al.* 2002, Burger 1997b).

Horseshoe crab reproduction does not appear impacted by the mosquito control chemical methoprene (at least through the first juvenile molt) or by ambient water quality in mid-Atlantic estuaries (USFWS 2007). Exposure of shorebirds to agricultural pollutants in rice fields may occur regionally in parts of South America, but red knot usage of rice field habitats was low in the several countries surveyed (Blanco *et al.* 2006). Finally, localized urban pollution has been shown to impact South American red knot habitats (WHSRN 2012, Niles *et al.* 2008, Ottema and Spaans 2008, COSEWIC 2007, Atkinson *et al.* 2005, Ferrari *et al.* 2002), but there are no available documented health effects or population-level impacts.

Wind Energy and Development

The Service analyzed shorebird mortality at land-based wind turbines in the United States (Akios 2011, Erickson *et al.* 2001), and we considered the red knot's vulnerability factors for collisions with offshore wind turbines that we expect will be built in the next few decades (Burger *et al.* 2011). We have minimal information regarding wind energy development in other countries. Based on our analysis of wind energy development in the U.S., we expect ongoing improvements in turbine siting, design, and operation will help minimize bird collision hazards (USFWS 2012d). However, we also expect cumulative avian collision mortality to increase through 2030 as the number of turbines continues to grow, and as wind energy development expands into coastal and offshore environments (DOE 2008). Shorebirds as a group have constituted only a small percentage of collisions with U.S. turbines in studies conducted to date (Akios 2011, NJAS 2009, NJAS 2008a, NJAS 2008b, Erickson *et al.* 2001), but wind development along the coasts (where shorebirds might be at greater risk) did not begin until 2005 (New Jersey Clean Energy Program undated). Based on the higher frequency and lower altitudes of red knot flights along the coasts, as well as the coastal location of most well-known U.S. nonbreeding red knot roosting and foraging areas, we conclude that collision and displacement risks per turbine (notwithstanding differences in specific factors such as turbine size, design, operation, siting) are likely higher along the coasts (both on land and nearshore) than in areas either far offshore or far inland (D. Newstead pers. comm. March 5, 2103; Burger *et al.* 2012c, Burger *et al.* 2011, Stewart *et al.* 2007, Alerstam *et al.* 1990). Likewise, hazards to red knots from offshore wind energy development likely increase for facilities situated closer to shore, particularly near bays and estuaries that serve as major stopover or wintering areas (Burger *et al.* 2012c, Burger *et al.* 2011).

The Service is not aware of any documented red knot mortalities at any wind turbines to date, but low levels of red knot mortality from turbine collisions may be occurring now based on the number of turbines along the red knot's migratory routes and the frequency with which red knots traverse these corridors. Based on the current number and geographic distribution of turbines, if any such mortality is occurring, it is likely not causing subspecies-level effects. However, our primary concern is that, as buildout of wind energy infrastructure progresses, especially near the coast, increasing mortality from turbine collisions may contribute to a subspecies-level effect due to the red knot's modeled vulnerability to direct human-caused mortality (Watts 2010). We anticipate that the threat to red knots from wind turbines will be primarily related to collision or behavioral changes during migratory or daily flights.

8.1.5. Summary of Status

Summary of Threats

Threats to the red knot from habitat destruction and modification are occurring throughout the entire range of the subspecies. These threats include climate change, shoreline stabilization, and coastal development, exacerbated regionally or locally by lesser habitat-related threats such as beach cleaning, invasive vegetation, agriculture, and aquaculture. The subspecies-level impacts from these activities are expected to continue into the future.

Other factors are likely to exacerbate the effects of reduced prey availability and asynchronies, including human disturbance (Burger and Niles 2013a, Burger and Niles 2013b, Escudero *et al.* 2012), competition with gulls (Niles *et al.* 2008, Burger *et al.* 2007), and behavioral changes from wind energy development (Kuvlesky *et al.* 2007). Additional factors are likely to increase the levels of direct red knot mortality, such as harmful algal blooms (HABs) (Newstead 2014a, Anderson 2007), oil spills (Anderson *et al*. 2012, WHSRN 2012, Kalasz 2008, Niles *et al.* 2008), and collisions with wind turbines (D. Newstead pers. comm. March 5, 2013; Burger *et al*. 2012c, Burger *et al.* 2011, Watts 2010, Kuvlesky *et al.* 2007). In addition to elevating background mortality rates, these three factors pose the potential for a low-probability but high-impact event if a severe HAB or major oil spill occurs when and where large numbers of red knots are present, or if a mass-collision event occurs at wind turbines during migration.

Red knots face a wide range of threats across their range on multiple geographic and temporal scales. The effects of some smaller threats may act in an additive fashion to ultimately impact populations or the subspecies as a whole (cumulative effects). Other threats may interact synergistically to increase or decrease the effects of each threat relative to the effects of each threat considered independently (synergistic effects). A number of threats are likely contributing to habitat loss, anthropogenic mortality, or both, and thus contribute to the red knot's threatened status, particularly considering the cumulative and synergistic effects of these threats, and that several key populations of this species have already undergone considerable declines.

Recovery Criteria

Recovery planning and development of recovery criteria are still underway since the final rule to list this subspecies was published on December 11, 2014.

8.2. Environmental Baseline

This section is an analysis of the effects of past and ongoing human and natural factors leading to the current status of the red knot, its habitat, and ecosystem within the Action Area. The environmental baseline is a "snapshot" of the species' health in the Action Area at the time of the consultation, and does not include the effects of the Action under review (see **Section 4**).

8.2.1. Action Area Numbers, Reproduction, and Distribution

Red knots have been documented on Bird Key Stono and Folly Beach during the fall, winter, and/or spring from 2006 to 2012 (Maddock *et al.* 2013, Thibault 2013). Flocks tend to be largest in the spring and smallest in the winter (Maddock *et al.* 2013, Thibault 2013). Red knots have been documented in larger numbers on Bird Key Stono than Folly Beach and their consistency of use and the duration of their stay varies by year and season.

8.2.2. Action Area Conservation Needs and Threats

Recreational Disturbance

Intense human disturbance in winter habitat can be functionally equivalent to habitat loss. If the disturbance prevents birds from using an area (Goss-Custard *et al*. 1996), this can lead to roost abandonment and population declines (Burton *et al*. 1996). Disturbance from human and pet presence alters plover behavior and often negatively influences distribution.

8.3. Effects of the Action

This section analyzes the direct and indirect effects of the Action on the red knot, which includes the direct and indirect effects of interrelated and interdependent actions. Direct effects are caused by the Action and occur at the same time and place. Indirect effects are caused by the Action, but are later in time and reasonably certain to occur. Our analyses are organized according to the description of the Action in section 2 of this BO.

8.3.1. Effects of Beach Renourishment

See **Section 6.3.1**. The Service anticipates the same effects to red knots as described for piping plovers.

8.3.2. Effects of Groin Rehabilitation

See **Section 6.3.2**. The Service anticipates the same effects to red knots as described for piping plovers.

8.4. Cumulative Effects

For purposes of consultation under ESA §7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under §7 of the ESA.

8.5. Conclusion

In this section, the Service summarizes and interprets the findings of the previous sections for the red knot (status, baseline, effects, and cumulative effects) relative to the purpose of a BO under §7(a)(2) of the ESA, which is to determine whether a Federal action is likely to:

a) jeopardize the continued existence of species listed as endangered or threatened; or
b) result in the destruction or adverse modification of designated critical habitat.

"Jeopardize the continued existence" means to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species (50 CFR §402.02).

After reviewing the current status of the migrating and wintering subpopulation of the rufa red knot, the environmental baseline for the proposed beach renourishment, associated construction activities, and the cumulative effects, it is the Service's biological opinion that implementation of the project, as proposed, is not likely to jeopardize the continued existence of the rufa red knot because effects due to construction activities are expected to be short term and become beneficial once construction is completed. Critical habitat for this species has not been designated at this time.

Red knots have been documented on Bird Key Stono and Folly Beach, but not in consistent numbers within and between migration and winter seasons. "Take" of red knots will be minimized by implementation of the Reasonable and Prudent Measures, and Terms and Conditions outlined in **Section 9**.

9. INCIDENTAL TAKE STATEMENT

ESA §9(a)(1) and regulations issued under §4(d) prohibit the "take" of endangered and threatened fish and wildlife species without special exemption. The term "take" in the ESA means "to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct" (ESA §3). In regulations at 50 CFR §17.3, the Service further defines:

- "Harass" as "an intentional or negligent act or omission which creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering;"
- "Harm" as "an act which actually kills or injures wildlife. Such act may include significant habitat modification or degradation where it actually kills or injures wildlife by significantly impairing essential behavioral patterns, including breeding, feeding or sheltering;" and
- "Incidental take" as "any taking otherwise prohibited, if such taking is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity."

Under the terms of ESA §7(b)(4) and §7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered prohibited, provided that such taking is in compliance with the terms and conditions of an incidental take statement (ITS).

For the exemption in ESA §7(o)(2) to apply to the Action considered in this BO, the Corps must undertake the non-discretionary measures described in this ITS, and these measures must become binding conditions of any permit, contract, or grant issued for implementing the Action. The Corps has a continuing duty to regulate the activity covered by this ITS. The protective coverage of §7(o)(2) may lapse if the Corps fails to:

- Assume and implement the terms and conditions; or
- Require a permittee, contractor, or grantee to adhere to the terms and conditions of the ITS through enforceable terms that are added to the permit, contract, or grant document.

In order to monitor the impact of incidental take, the Corps must report the progress of the Action and its impact on the species to the Service as specified in this ITS.

9.1. Amount or Extent of Take

The Service anticipates that the Action is reasonably certain to cause incidental take of individual loggerhead sea turtles consistent with the definition of harass resulting from construction activities during the nesting season. The Service anticipates that the Action is reasonably certain to cause incidental take of individual loggerhead sea turtles consistent with the definition of harm resulting from construction activities during the nesting season.

The Action considered in this BO includes a conservation measure to relocate any nests within the construction area if beach renourishment overlaps with the nesting season. Through this statement, the Service authorizes this conservation measure as an exception to the prohibitions against trapping, capturing, or collecting listed species. This conservation measure is identified as a Reasonable and Prudent Measure below, and we provide Terms and Conditions for its implementation.

Anticipated Take of Loggerhead Sea Turtles

The Service anticipates 13,000 lf of nesting beach habitat could be taken as a result of the Action. The take is expected to be in the form of: (1) Harm in the form of destruction of all nests that may be constructed and eggs that may be deposited and missed by a nest survey and nest relocation program (May 1 – October 31) within the boundaries of the proposed project; (2) harm in the form of destruction of all nests deposited during the period when a nest survey and nest relocation program is not required to be in place (November 1 – April 30) within the boundaries of the proposed project; (3) harassment in the form of disturbing or interfering with female turtles attempting to nest within the construction area or on adjacent sections of beach as a result of construction activities; (4) harassment in the form of misdirection of nesting sea turtles or hatchling turtles on beaches within the boundaries of the construction area or sections of beach adjacent to the construction area as nesting females emerge from the water or hatchlings emerge from the nest and crawl to the water as a result of increased sand accretion due to the presence of the groins; (5) harassment in the form of behavior modification of nesting females due to escarpment formation, resulting in false crawls or situations where they choose marginal or unsuitable nesting areas to deposit eggs; (6) harm in the form of destruction of nests from escarpment leveling within a nesting season when such leveling has been approved by the Service; (7) harassment in the form of behavior modification of nesting females or hatchlings due to the presence of groin, which may act as barriers to movement or cause disorientation of turtles while on the nesting beach; (8) harassment in the form of physical entrapment of hatchling sea turtles on the nesting beach due to the presence of the groin; behavior modification of nesting females if they dig above a buried portion of the structure, resulting in false crawls or situations where they choose marginal or unsuitable nesting areas; and (9) harassment of an unknown number of adult and hatchling sea turtles in the form of obstruction or entrapment during ingress or egress at nesting sites.

The Service anticipates incidental take of sea turtles will be difficult to detect for the following reasons: (1) the turtles nest primarily at night and all nests are not found because [a] natural factors, such as rainfall, wind, and tides may obscure crawls and [b] human-caused factors, such as pedestrian and vehicular traffic, may obscure crawls, and result in nests being destroyed because they were missed during a nesting survey and nest mark and avoidance program (2) the total number of hatchlings per undiscovered nest is unknown; (3) an unknown number of females may avoid the project beach and be forced to nest in a less than optimal area; (5) lights may misdirect an unknown number of hatchlings and cause death; (6) an unknown number of adult and hatchling sea turtles may be obstructed or entrapped during ingress or egress at nesting sites; and (7) escarpments may form and prevent an unknown number of females from accessing a suitable nesting site. However, the take of this species can be anticipated for the Action because: (1) turtles nest within the Action Area; (2) construction will likely occur during a portion of the nesting season; (3) the construction activities will modify the beach profile and width and increase the presence of escarpments; and (4) artificial lighting will deter and/or misdirect nesting hatchling turtles.

Anticipated Take of Piping Plovers

The Service anticipates that directly and indirectly an unspecified amount of piping plovers along six acres of shoreline on Bird Key Stono, all at some point, potentially usable by piping plovers, could be taken in the form of harm and harassment as a result of this proposed action; however, incidental take of piping plovers will be difficult to detect for the following reasons:

(1) Harassment to the level of harm may only be apparent on the breeding grounds the following year; and
(2) Dead plovers may be carried away by waves or predators.

The Service has reviewed the biological information and other information relevant to this action. The take is expected in the form of harm and harassment because of: (1) decreased fitness and survivorship of wintering plovers due to a temporary loss and degradation of a section of foraging habitat; and (2) decreased fitness and survivorship of plovers attempting to migrate to breeding grounds due to a temporary loss and degradation of a section of foraging habitat.

Anticipated Take of Red Knots

The Service anticipates that directly and indirectly an unspecified number of red knots along six acres of shoreline on Bird Key Stono, all at some point, potentially usable by red knots, could be taken in the form of harm and harassment as a result of this proposed action; however, incidental take of red knots will be difficult to detect for the following reasons:

(1) Harassment to the level of harm may only be apparent on the breeding grounds the following year; and
(2) Dead knots may be carried away by waves or predators.

The Service has reviewed the biological information and other information relevant to this action. The take is expected in the form of harm and harassment because of: (1) decreased fitness and survivorship of migrating or wintering knots during the nonbreeding season due to a temporary loss and degradation of a section of foraging habitat; and (2) decreased fitness and survivorship of knots attempting to migrate to breeding grounds due to a temporary loss and degradation of a section of foraging habitat.

9.2. Reasonable and Prudent Measures

The Service believes the following reasonable and prudent measures (RPMs) are necessary or appropriate to minimize the impact of incidental take caused by the Action on listed wildlife species. RPMs are described for the species below.

RPM#1. Conservation Measures included in the permit application/project plans must be implemented (unless revised below in the Terms and Conditions) in the proposed project.

RPM#2. A meeting/conference call between representatives of the Corps, City, contractor, SCFO, SCDNR, and the permitted sea turtle surveyor(s) must be held prior to the commencement of work on this Action.

RPM#3. The Corps will use beach quality sand for sand placement.

RPM#4. The Corps will remove all derelict concrete, metal, coastal armoring material or other debris from the beach prior to any material placement.

RPM#5. The Corps must be responsible for reviewing the project design to ensure the predicted project performance and minimization of downdrift impacts is probable.

RPM#6. The Corps and/or City will hire sea turtle monitors to survey the project area during the sea turtle nesting season (May 1 – October 31). Surveys for nesting sea turtles must be conducted within the project area if work will occur during a portion of the nesting season. If nests are constructed in the project footprint, the eggs must be relocated to minimize sea turtle nest burial, crushing of eggs, or nest excavation. Nest relocation will be on a selected area of beach that is not expected to experience daily inundation by high tides or known to routinely experience severe erosion and egg loss, predation, or subject to artificial lighting.

RPM#7. The Corps' and City's contractor(s) will store construction equipment and materials for project construction in a manner that will minimize impacts to sea turtles to the maximum extent practicable.

RPM#8. The Corps' contractor will install and maintain predator-proof trash receptacles during project construction at all beach access points used for project construction to minimize the potential for attracting predators of sea turtles.

RPM#9. The City must complete post construction surveys of all artificial lighting visible from the project beach.

RPM#10. Prior to the beginning of the project, the Corps must submit a lighting plan for the dredge that will be used in this project.

RPM#11. The Corps must hire nighttime monitors with sea turtle experience to patrol the beach at night in the project area if nighttime construction activities and equipment occur during the nesting season.

RPM#12. The Corps must take actions to minimize sea turtle misorientation/disorientations on the beach caused by the projects' construction-related lighting and artificial lighting associated with oceanfront development adjacent to the project area and within the project limits during the nesting season from May 1 through October 31.

RPM#13. The Corps must monitor compliance with construction related lighting during the sea turtle nesting season (May 1 – October 31).

RPM#14. During the sea turtle nesting season, the Corps' contractor must not extend the beach fill more than 500 feet along the shoreline and must confine work activities within this area between dusk and the following day's nesting survey unless nighttime monitors patrol the beach to reduce the impacts to emerging sea turtles and burial of new nests.

RPM#15. The Corps will monitor sand compaction and conduct tilling (non-vegetated areas) if needed immediately after completion of the sand placement work and prior to the next three nesting seasons to reduce the likelihood of impacting sea turtle nesting and hatching activities.

RPM#16. The Corps will monitor escarpment formation and conduct leveling if needed immediately after completion of the sand placement project and prior to the next three nesting seasons to reduce the likelihood of impacting nesting and hatchling sea turtles.

RPM#17. During the portion of the nesting season that overlaps with the construction window, the Corps' contractor will restrict on-beach access to the construction site to the wet sand below mean high water (MHW).

RPM#18. The Corps will minimize impacts to piping plovers, red knots, and their habitat by completing the work on Bird Key Stono prior to March 15 unless an extension is agreed to by the SCDNR and the Service.

9.3. Terms and Conditions

In order for the exemption from the take prohibitions of §9(a)(1) and of regulations issued under §4(d) of the ESA to apply to the Action, the Corps must comply with the terms and conditions (T&Cs) of this statement, provided below, which carry out the RPMs described in the previous section. These T&Cs are mandatory. As necessary and appropriate to fulfill this responsibility, the Corps must require any permittee, contractor, or grantee to implement these T&Cs through enforceable terms that are added to the permit, contract, or grant document.

T&C#1. Conservation Measures included in the permit application/project plans must be implemented in the proposed project. Project construction via hydraulic dredge will be limited to November 1 through July 15. Project construction via hopper dredge will be limited to November 1 through March 31.

T&C#2. A meeting or conference call between representatives of the Corps, City, contractor, SCFO, SCDNR, and the permitted sea turtle surveyors must be held prior to the commencement of work on this project. At least ten business days advance notice will be provided prior to conducting this meeting. The meeting/conference call will provide an opportunity for explanation and/or clarification of the protection measures.

T&C#3. Beach compatible fill must be placed on the beach or in any associated dune system. Beach compatible fill is material that maintains the general character and functionality of the material occurring on the beach and in the adjacent dune and coastal system. Such material must be predominately of carbonate, quartz or similar material with a particle size distribution ranging between 0.062mm and 4.76mm (classified as sand by either the Unified Soils or the Wentworth classification), must be similar in color and grain size distribution (sand grain frequency, mean and median grain size and sorting coefficient) to the material in the historic beach sediment at the disposal site, and must not contain:

a. Greater than five percent, by weight, silt, clay or colloids passing the #230 sieve;

b. Greater than five percent, by weight, fine gravel retained on the #4 sieve (-2.25φ);

c. Coarse gravel, cobbles or material retained on the 3/4 inch sieve in a percentage or size greater than found on the native beach;

d. Construction debris, toxic material or other foreign matter; and

e. Material that will result in cementation of the beach.

If rocks or other non-specified materials appear on the surface of the filled beach in excess of 50% of background in any 10,000 square foot area, then surface rock should be removed from those areas. These areas must also be tested for subsurface rock percentage and remediated as required. If the natural beach exceeds any of the limiting parameters listed above, then the fill material must not exceed the naturally occurring level for that parameter on nearby native beaches.

These standards shall not be exceeded in any 10,000 square foot section extending through the depth of the nourished beach. If the native beach exceeds any of the limiting parameters listed above, then the fill material shall not exceed the naturally occurring level for that parameter on nearby native beaches.

T&C#4. All derelict concrete, metal, and coastal armoring geotextile material and other debris must be removed from the beach prior to any sand placement to the maximum extent possible. If debris removal activities will take place from May 1 through October 31, the work must be conducted during daylight hours only and must not commence until completion of the sea turtle survey each day.

T&C#5. The groins must be removed if they are determined to not be effective as determined by the City's monitoring and mitigation plan following groin repair. In the event the structure begins to disintegrate, all debris and structural material must be removed from the nesting beach area and deposited off site immediately. If removal of the structure is required during the period from May 1 to October 31, no work will be initiated without prior coordination with the SCDNR and the SCFO.

T&C#6. Daily early morning surveys for sea turtle nests will be required if construction coincides with the sea turtle nesting season. Nesting surveys must be conducted May 1–October 31 in the project area if work will begin before October 31. If nests are constructed in areas where they may be affected by construction activities, the nests must be relocated per the following requirements.

a. Nesting surveys and nest relocation will only be conducted by personnel with prior experience and training in nesting survey and nest marking procedures. Surveyors must have a valid SCDNR permit. Nesting surveys must be conducted daily between sunrise and 9:00 AM.

b. Only those nests that may be affected by sand placement activities will be relocated. Nests requiring relocation will be moved no later than 9:00 AM the morning following deposition to a nearby self-release beach site in a secure setting where artificial lighting will not interfere with hatchling orientation. Relocated nests will not be placed in organized groupings. Relocated nests will be randomly staggered along the length and width of the beach in settings that are not expected to experience daily inundation by high tides or known to routinely experience severe erosion and egg loss, or subject to artificial lighting. Nest relocations in association with construction activities must cease when construction activities no longer threaten nests.

c. Nests deposited within areas where construction activities have ceased or will not occur for 75 days or nests laid in the nourished berm prior to tilling must be marked and left in situ unless other factors threaten the success of the nest. The turtle permit holder will install an on-beach marker at the nest site. No activity will occur within this area nor will any activities occur which could result in impacts to the nest. Nest sites will be inspected daily to assure nest markers remain in place and the nest has not been disturbed by the project activity.

T&C#7. During the sea turtle nesting season, nighttime storage of construction equipment not in use must be off the beach to minimize disturbance to sea turtles. Staging areas for construction equipment must be located off the beach. Nighttime storage of construction equipment not in use must be off the beach to minimize disturbance to sea turtle nesting and hatching activities. In addition, all construction pipes placed on the beach must be located as far landward as possible without compromising the integrity of the dune system. Pipes placed parallel to the dune must be 5 to 10 feet away from the toe of the dune if the width of the beach allows. Temporary storage of pipes must be off the beach to the maximum extent possible. If the pipes are stored on the beach, they must be placed in a manner that will minimize the impact to nesting habitat and must not compromise the integrity of the dune systems.

T&C#8. Predator-proof trash receptacles must be installed and maintained during construction at all beach access points used for the project construction to minimize the potential for attracting predators of sea turtle (**Appendix B**). The contractors conducting the work must provide predator-proof trash receptacles for the construction workers. All contractors and their employees must be briefed on the importance of not littering and keeping the project area trash and debris free.

T&C#9. Two post-construction surveys must be conducted of all lighting visible from the beach placement area using standard techniques for such a survey (**Appendix C**). The timing of these surveys will be coordinated with the SCFO prior to commencement of the work. Summary reports of both surveys will be provided to the SCFO. The summary report from the post-construction surveys (including the following information: methodology of the survey, a map showing the position of the lights visible from the beach, a description of each light source visible from the beach, recommendations for remediation, and any actions taken) will be provided to the SCFO within three months after the survey is conducted. After the report is completed, a meeting or conference call must be set up with the Corps, the project sponsors, SCDNR, and the SCFO to discuss the survey report, as well as any documented sea turtle disorientations in or adjacent to the project area. Any action related to artificial beachfront lighting will be addressed by the appropriate project sponsor. If the project is completed during the nesting season and prior to May 1, the lighting surveys may be conducted during the year of construction.

T&C#10. Prior to the beginning of the project, the Corps must submit a lighting plan for the dredge that will be used in the project if it will overlap with the nesting season. The plan must include a description of each light source that will be visible from the beach and the measures implemented to minimize this lighting. This plan must be reviewed and approved by the SCFO.

T&C#11. The Corps must hire nighttime monitors with sea turtle experience and a valid SCDNR permit to patrol the beach at night in the project area if nighttime construction activities and equipment occur during the nesting season. Monitors must patrol the length of the pipeline within the active nighttime construction area for nesting females May 1 – August 15. From July 1 - October 15, sea turtle monitors must check all nests on a nightly basis after 10 pm within 1,000 feet of the active nighttime project area that have been incubating for 45 days until three nights after the first sign of emergence or the inventory of the nest contents.

T&C#12. Light visible from the beach will be documented and the source identified. Lighting will be classified into five categories: 1) sky glow (ambient light from coastal development), 2) construction related (light coming from the active nighttime project area), 3) residential or municipal (light coming from a house, condo, pier, or street light), 4) personal use (light from a flashlight), or 5) monitoring related (light from headlights of vehicles used to conduct night monitoring). The applicant or its representative will take corrective measures to address construction and monitoring related lighting visible from nests due to hatch. Sea turtle monitors will contact the appropriate code enforcement officials for residential or municipal lighting visible from the beach.

T&C#13. Direct lighting of the beach and nearshore waters must be limited to the immediate construction area during nesting season and must comply with safety requirements. Lighting on all equipment must be minimized through reduction, shielding, lowering, and appropriate placement to avoid excessive illumination of the water's surface and nesting beach while meeting all Coast Guard, Corps EM 385-1-1, and OSHA requirements. Light intensity of lighting equipment must be reduced to the minimum standard required by OSHA for General Construction areas, in order not to misdirect sea turtles. Shields must be affixed to the light housing and be large enough to block light from all on-beach lamps from being transmitted outside the construction area or to the adjacent sea turtle nesting beach (**Figure 8**).

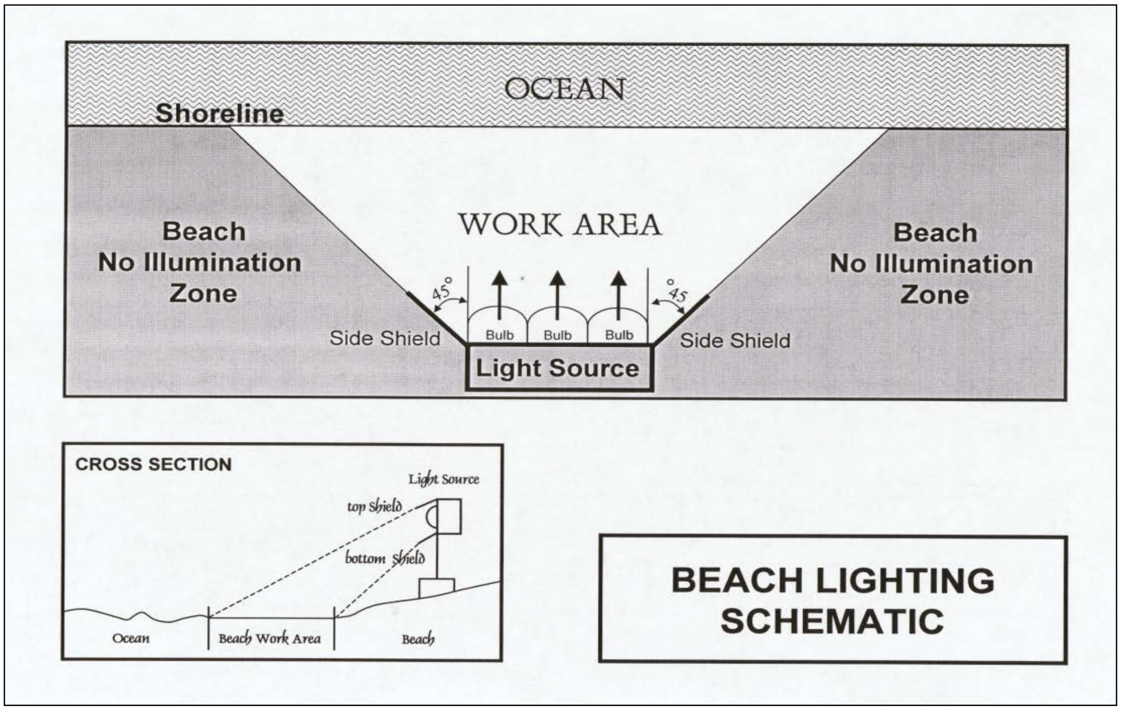

Figure 8. Beach lighting schematic.

T&C#14. During the sea turtle nesting season, the contractor must not extend the beach fill more than 500 feet (or other agreed upon length) along the shoreline between dusk and dawn and the following day until the daily nesting survey has been completed and the beach cleared for fill advancement. An exception to this may occur if there is permitted sea turtle surveyor present on-site to ensure no nesting and hatching sea turtles are present within the extended work area. If the 500 feet is not feasible for the project, an agreed upon distance will be decided on during the preconstruction meeting. Once the beach has been cleared and the necessary nest relocations have been completed, the contractor will be allowed to proceed with the placement of fill and work activities during daylight hours until dusk at which time the 500-foot length (or other agreed upon length) limitation must apply. If any nesting turtles are sighted on the beach within the immediate construction area, activities must cease immediately until the turtle has returned to the water and the sea turtle permit holder responsible for nest monitoring has relocated the nest.

T&C#15. Sand compaction must be monitored in the area of sand placement immediately after completion of the project and prior to May 1 for three subsequent years. If tilling is needed, the area must be tilled to a depth of 24 inches. Each pass of the tilling equipment must be overlapped to allow more thorough and even tilling. All tilling activity must be completed at least once prior to nesting season. An electronic copy of the results of the compaction monitoring must be submitted to the SCFO prior to any tilling actions being taken or if a request not to till is made based on compaction results. The requirement for compaction monitoring can be eliminated if the decision is made to till regardless of post construction compaction levels. Additionally, out-year compaction monitoring and remediation are not required if placed material no longer remains on the dry beach.

a. Compaction sampling stations must be located at 500-foot intervals along the sand placement template. One station must be at the seaward edge of the dune/bulkhead line (when material is placed in this area), and one station must be midway between the dune line and the high water line (normal wrack line).

b. At each station, the cone penetrometer must be pushed to a depth of 6, 12, and 18 inches three times (three replicates). Material may be removed from the hole if necessary to ensure accurate readings of successive levels of sediment. The penetrometer may need to be reset between pushes, especially if sediment layering exists. Layers of highly compact material may lie over less compact layers. Replicates must be located as close to each other as possible, without interacting with the previous hole or disturbed sediments. The three replicate compaction values for each depth must be averaged to produce final values for each depth at each station. Reports will include all 18 values for each transect line, and the final six averaged compaction values.

c. If the average value for any depth exceeds 500 pounds per square inch (psi) for any two or more adjacent stations, then that area must be tilled immediately prior to May 1.

d. If values exceeding 500 psi are distributed throughout the project area but in no case do those values exist at two adjacent stations at the same depth, then consultation with the SCFO will be required to determine if tilling is required. If a few values exceeding 500 psi are present randomly within the project area, tilling will not be required.

e. Tilling must occur landward of the wrack line and avoid all vegetated areas three square feet or greater with a three square foot buffer around the vegetated areas.

T&C#16. Visual surveys for escarpments along the project area must be made immediately after completion of the sand placement and within 30 days prior to May 1 for three subsequent years if sand in the project area still remains on the dry beach.

Escarpments that interfere with sea turtle nesting or that exceed 18 inches in height for a distance of 100 feet must be leveled and the beach profile must be reconfigured to minimize scarp formation by the dates listed above. Any escarpment removal must be reported by location. If the project is completed during the early part of the sea turtle nesting and hatching season, escarpments may be required to be leveled immediately, while protecting nests that have been relocated or left in place. The SCFO must be contacted immediately if subsequent reformation of escarpments that interfere with sea turtle nesting or that exceed 18 inches in height for a distance of 100 feet occurs during the nesting and hatching season to determine the appropriate action to be taken. If it is determined that escarpment leveling is required during the nesting or hatching season, the SCFO will provide a brief written authorization within 30 days that describes methods to be used to reduce the likelihood of impacting existing nests. An annual summary of escarpment surveys and actions taken must be submitted to the SCFO.

T&C#17. During the sea turtle nesting season, on-beach access to the construction site will be restricted to the wet sand below MHW.

T&C#18. The Corps will coordinate with SCDNR and the Service before sand is placed on Bird Key Stono. All material will be placed above the high tide line in the area agreed upon during the January 11, 2018, site visit.

9.4. Monitoring and Reporting Requirements

In order to monitor the impacts of incidental take, the Corps must report the progress of the Action and its impact on the species to the Service as specified in the incidental take statement (50 CFR §402.14(i)(3)). This section provides the specific instructions for such monitoring and reporting (M&R). As necessary and appropriate to fulfill this responsibility, the Corps must require any permittee, contractor, or grantee to accomplish the monitoring and reporting through enforceable terms that are added to the permit, contract, or grant document. Such enforceable terms must include a requirement to immediately notify the Corps and the Service if the amount or extent of incidental take specified in this ITS is exceeded during Action implementation.

M&R#1. A report describing the work conducted and actions taken to implement the reasonable and prudent measures and terms and conditions of this incidental take statement including the information listed below must be submitted to the SCFO within three months of either the completion of construction or the end of the nesting season depending on construction timing.

- Project location (latitude and longitude coordinates)
- Project description (include linear feet and acres of beach, access points used for construction equipment)
- Dates of actual construction activities
- Names and qualifications of personnel involved in sea turtle nesting surveys and nest relocation
- Number of nests laid within the construction area, number of nests

relocated due to construction activities, nest hatch and emergence success
- Number of false crawls within the construction area
- Number of encounters with nesting females within the construction area
- Number of misorientations/disorientations of nesting sea turtles or hatchlings from lighting of construction area
- Escarpment formation and remedial action
- Compaction testing results and remedial action

M&R#2. Upon locating a dead or injured sea turtle adult, hatchling, or egg that may have been harmed or destroyed as a direct or indirect result of the project, the Corps, permittee, and/or local sponsor will be responsible for notifying the SCDNR Hotline (1-800-922-5431) and the SCFO (843-727-4707). Care must be taken in handling injured sea turtles or eggs to ensure effective treatment or disposition, and in handling dead specimens to preserve biological materials in the best possible state for later analysis.

10.CONSERVATION RECOMMENDATIONS

§7(a)(1) of the ESA directs Federal agencies to use their authorities to further the purposes of the ESA by conducting conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary activities that an action agency may undertake to avoid or minimize the adverse effects of a proposed action, implement recovery plans, or develop information that is useful for the conservation of listed species. The Service offers the following recommendations that are relevant to the listed species addressed in this BO and that we believe are consistent with the authorities of the Corps.

CR#1. Educational signs should be placed where appropriate at beach access points explaining the life history of beach dependent species.

CR#2. Piping plover and red knot surveys should be continued to assess the status of the local migratory and winter populations.

11.REINITIATION NOTICE

Formal consultation for the Action considered in this BO is concluded. Reinitiating consultation is required if the Corps retains discretionary involvement or control over the Action (or is authorized by law) when:

a. The amount or extent of incidental take is exceeded;
b. New information reveals that the Action may affect listed species or designated critical habitat in a manner or to an extent not considered in this BO;
c. The Action is modified in a manner that causes effects to listed species or designated critical habitat not considered in this BO; or
d. A new species is listed or critical habitat designated that the Action may affect.

In instances where the amount or extent of incidental take is exceeded, the Corps is required to immediately request a reinitiation of formal consultation.

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