Appendix C

Information on T&E Species and Potential Effects to T&E Species

Cape Wind Energy Project
Nantucket Sound

Information on the Presence and Potential Effects to Endangered and Threatened Species in the Proposed Action Area

Minerals Management Service

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1.0 ENDANGERED AND THREATENED SPECIES AND CRITICAL HABITAT DESCRIPTION

The information to follow provides detail on the endangered and threatened species potentially found in the proposed action area. This includes three species of whales, four species of sea turtles, and two species of birds. There is also one candidate species, the red knot, being considered for listing by the FWS. This species is discussed within Sections 4.2.9 and 5.3.2.9 of the DEIS. Species descriptions are summarized below and Table C-1 lists the endangered and threatened species that may occur in the proposed action area.

Table C-1.
Listed Species

Group	Common Name	Scientific Name	Status	Critical Habitat
Whales	Humpback Whale	*Megaptera novaeangliae*	E	No
	North Atlantic Right Whale	*Eubalaena glacialis*	E	Yes
	Fin Whale	*Balaenoptera physalus*	E	No
Sea Turtle	Kemp's ridley	*Lepidochelys kempii*	E	No
	Leatherback	*Dermochelys coriacea*	E	No
	Loggerhead	*Caretta caretta*	T	No
	Green	*Chelonia mydas*	T	No
Avian	Roseate Tern	*Sterna dougallii*	E	No
	Piping Plover	*Charadrius melodus*	T	No

Critical habitat has been designated for the North Atlantic Right whale in southeastern Massachusetts, but outside of Nantucket Sound. The designated critical habitats are located within Cape Cod Bay and along the Great South Channel. These areas do not overlap with the proposed action area.

1.1 Whales

1.1.1 Humpback Whale

1.1.1.1 Species Description

Humpback whales (*Megaptera novaeangliae*) occur in all the oceans of the world, often in areas of upwelling, along the edges of banks, and over rapidly changing bathymetry along the continental shelf, and along frontal zones between well-mixed and stratified water masses (Waring et al., 2006). The fifth largest of the baleen whales, humpback whales are approximately 13 ft (4 m) long at birth and reach a maximum size of 59 ft (18 m) and a weight of 48 metric tons (106,000 lbs) (Winn and Reichley 1985). Females are slightly larger than males. Distinguishable features of humpback whales include extremely long flippers that may reach 16.4 ft (5 m) in length, well-defined ventral grooves and fleshy protuberances (tubercles) that cover the whale's rostrum and a small, variable shaped dorsal fin. Color patterns are used to

identify individuals, specifically by the black and white patterns on the underside of the fluke and pectoral fins (Katona et al., 1980; Katona and Whitehead 1981). Calves also appear to inherit the fluke pigmentation patterns of their mothers (Rosenbaum and Clapham 1993).

1.1.1.2 Life History

Female humpback whales reach sexual maturity after four to six years and thereafter give birth approximately every two or three years (Waring et al., 2006; Clapham 1992). The gestation period is ten to twelve months and most births take place in the winter in the West Indies (Waring et al., 2006). Mothers usually nurse their calves for a year or less, as weaning may begin when the calves are five to six months old and still in the northern feeding grounds (Clapham 1992; Baraff and Weinrich 1993; Waring et al., 2006). After weaning their calves, the adult females are ready to mate again. The life span of humpback whales is approximately 30 years (Chittleborough 1959).

Humpback whales are migratory species that spends winter breeding in the Lesser and Greater Antilles Islands of the eastern Caribbean Sea (Waring et al., 2006). Summers are spent in northern latitude feeding grounds (40° to 75° N latitude) in areas of high productivity off the coasts of Iceland, southwestern Greenland, Newfoundland and Labrador, the Gulf of St. Lawrence, and the Gulf of Maine (Waring et al., 2006). Movements of humpbacks within the northern latitude feeding grounds are controlled by prey densities favoring areas of upwelling on the shelf supporting dense aggregations of near-surface zooplankton and shoaling, plankton-feeding fish upon which the whales feed (Brodie et al., 1978; Gaskin 1982; Kenney and Winn 1986; Dolphin 1987a, b; Mayo et al., 1988; Waring et al., 2006).

Humpback whales feed opportunistically on a wide variety of species of pelagic crustaceans and small fish including sand lance (*Ammodytes americanus* and *A. dubius)*, caplin (*Mallotus villosus*), and euphausiids (*Meganyctophanes norvegica*) (Nemoto 1970; Hain et al., 1982; Kreiger and Wing 1984; Whitehead and Glass 1985; Waring et al., 2006). During their seasonal northern residency in the area, humpbacks may also feed on several commercially important fish and invertebrates, such as herring (*Clupea harengus*), mackerel (*Scomber scombrus*), menhaden (*Brevoortia tyrannus*), pollock (*Pollachius virens*), small haddock (*Melanogrammus aeglfinus*), and squid (*Illex illecebrosus*) (Overholtz and Nicolas 1979; Meyer et al., 1979; Whitehead and Glass 1985; Whitehead 1987; Piatt et al., 1989; Waring et al., 2006). They may feed singly or in closely coordinated groups. Groups of up to 22 individuals may lunge in unison at surface schools of fish (Hain et al., 1982; Würsig 1990).

1.1.1.3 Population Dynamics

As reported in the 2005 Marine Mammal Stock Assessment Report (Waring et al., 2006), 902 whales is considered the best abundance estimate for humpbacks in the Gulf of Maine, with a minimum population estimate of 647 individuals. Currently the best available estimate of the North Atlantic population is 11,570 (coefficient of variation (CV) = 0.069) individuals based on mark and recapture studies in 1992 and 1993 (Waring et al., 2006).

Western North Atlantic humpbacks belong to four primary feeding aggregations: United States east coast (including the Gulf of Maine), the Gulf of St. Lawrence, Newfoundland/Labrador, and western Greenland (Waring et al., 2006). Two other North Atlantic feeding grounds have been identified off Iceland and northern Norway (Waring et al., 2006). Composition of these feeding aggregations is determined by matrilineal fidelity. Based on genetic evidence supporting the distinction of the western North Atlantic feeding, the Gulf of Maine is now viewed as a distinct feeding stock for management purposes (Waring et al., 2006). Humpbacks from these three feeding aggregations seem to congregate preferentially in winter breeding and calving areas off the Dominican Republic and eastern Puerto Rico (Waring et al., 2006) and may follow similar southward migration routes from summer feeding areas to winter breeding areas.

Little is known about natural mortality in humpback whales. Parasites, toxics, ice entrapment, predation by killer whales, and fluctuating prey populations due to events such as El Niño may contribute to natural humpback mortality rates (Waring et al., 2006). Young or sick humpbacks seem to be particularly vulnerable to attacks by killer whales (*Orcinus orca*) and occasionally by larger predatory sharks (Waring et al., 2006). In the western North Atlantic, 14 percent (n=3365) of the appropriately photographed humpback whales bear scars, primarily on their flukes, from killer whale attacks (Katona et al., 1988; Waring et al., 2006). Although humpback whales and killer whales have been observed feeding near one another without aggressive interactions (Dolphin 1987c), killer whales have been observed attacking and killing other species of baleen whales (Hancock 1965; Baldridge 1972; Silber et al., 1990).

Humpback whales are the top carnivores in a relatively simple food chain consisting of phytoplankton, zooplankton, small forage fish and crustaceans. Although the food chain is short, it does afford a mechanism for accumulation of natural and anthropogenic toxins from prey species to whale tissues through trophic transfer and biomagnification.

1.1.1.4 Status and Distribution

Humpback whales were an important commercial species throughout most of their range, including New England waters, until early in the twentieth century (Allen 1916). Some taking of humpback whales occurred in northwest Atlantic waters until the mid-1950s. The International Convention for the Regulation of Whaling, Washington 1946, afforded the North Atlantic population of humpback whales full protection in 1955 (Best 1993). Humpback whales were afforded endangered species status in the United States in 1970 (USFWS 1986), and retain that status today. Although severely depleted by whaling, the species has shown good recovery over most of its range.

The most common anthropogenic source of mortality for humpback whales in the western North Atlantic is entanglement in commercial fishing gear (O'Hara et al., 1986; Lien et al., 1989a, b; Hofman 1990; Volgenau and Kraus 1990; Waring et al., 2006). Humpback whales in the Gulf of Maine stock become entangled most frequently in gill nets and trap/pot gear (such as lobster or crab), followed by weirs and seines (Waring et al., 2004). In inshore waters of Newfoundland, entanglement occurs most frequently in cod traps, followed by groundfish gill nets and salmon gill nets (Hofman 1990, Waring et al., 2006). A total of 18 humpback whales

were reported entangled in lobster gear in coastal waters between New Jersey and New England between 1976 and 1993 (NMFS 1994). Two humpback whales were entangled in New Jersey, one in New York, and eleven in coastal waters of Massachusetts. Only one of the whales died as a probable result of the entanglement (NMFS 1994). A review of mortalities and serious injuries for the years 1999 to 2003 reveal that five mortalities and nine serious injuries in the Gulf of Maine stock were attributable to fishery interactions (Waring et al., 2006). For the period of 1999 to 2003, three additional fishery-related mortalities and three serious injuries are on record for southeastern and mid-Atlantic waters; it is uncertain if any of these whales are from the Gulf of Maine stock (Waring et al., 2006).

Humpback whales are relatively tolerant of boats (Pett and McKay 1990) and are seen frequently in the Great South Channel and Stellwagen Bank in the vicinity of commercial and recreational fishing vessels and whale watch boats. During the early 1970s, before whale watching became popular in Massachusetts Bay, humpback whales were difficult to approach in a small boat (Watkins 1986). The whales usually diminished surface activities and moved away, emitting agonistic trumpeting sounds when approached too closely. However, during recent years humpback whales in nearshore waters often readily accept the presence of vessels, and some even "perform" various surface behaviors when approached by a whale watch vessel. Humpbacks in the western North Atlantic are more habituated to vessel approach than any other whale in the area (Watkins 1986). As whales become more habituated to whale-watch and other vessel traffic, the chance of collision increases (Beach and Weinrich 1989). There is some evidence of increased incidents of ship collisions in the Gulf of Maine (Waring et al., 2006). In a recent study of stranded humpback whales along the Middle-Atlantic and southeast United States, 30 percent (n=20) had injuries potentially associated with a ship strike (Waring et al., 2006).

Figure C-1 provides data on the sightings of humpback whales in the Gulf of Maine and Cape Cod regions. The sighting data indicates three areas of congregation, Georges Bank, Stellwagen Bank, and in the northern Gulf of Maine. One humpback whale has been reported in the vicinity of Monomoy Shoals, but none have been observed within Nantucket Sound. The sightings data should not be considered as an absolute documentation of the occurrence of any particular species, because the results are dependent in part on the level of effort that is expended in looking for whales. However, as a general indicator of where humpback whales are likely versus unlikely to occur, the data suggest that few or no humpback whales are likely to occur in or near the site of the proposed action during construction, operation or decommissioning.

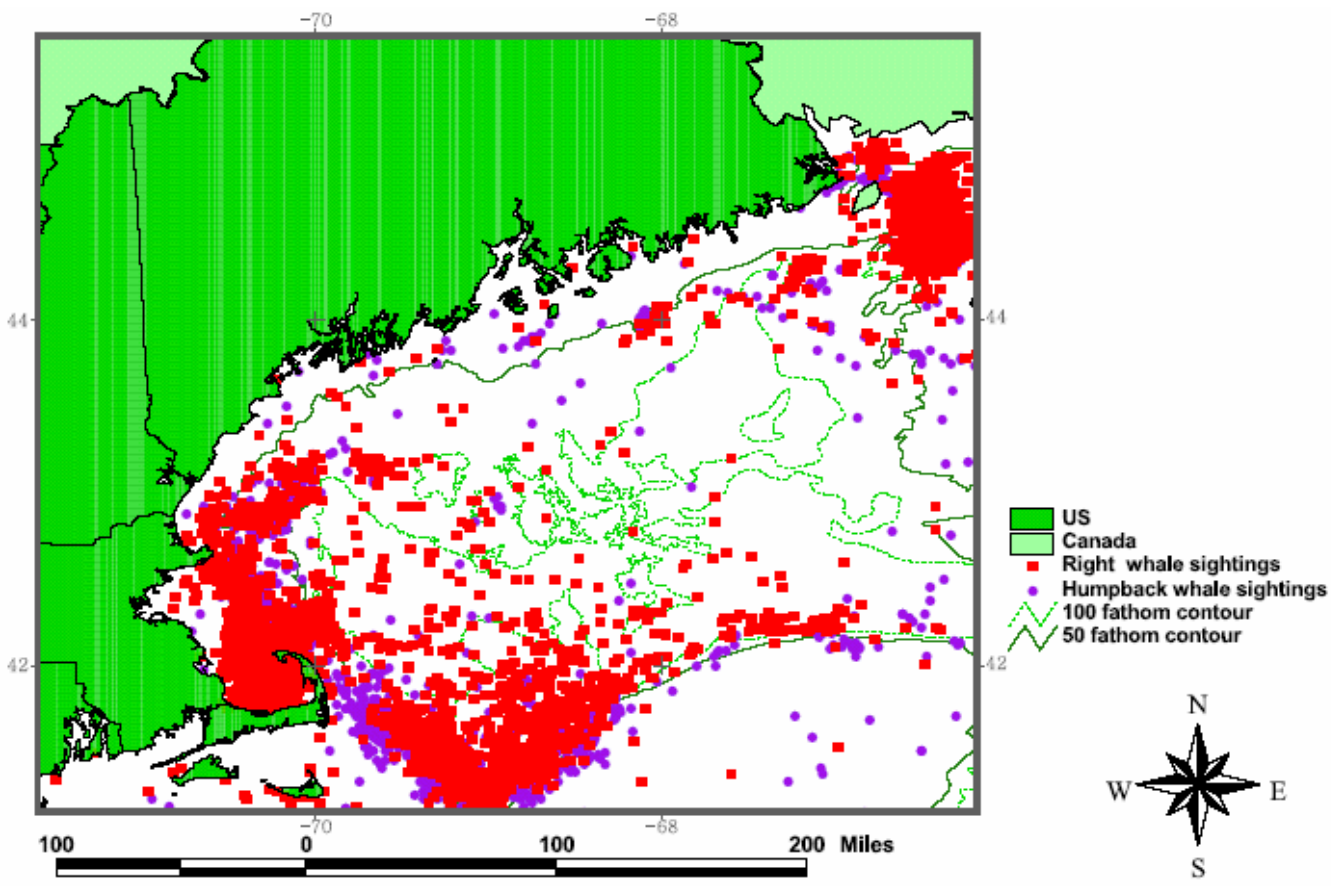

Figure C-1.
North Atlantic Right and Humpback whale sightings through 2002 – Cape Cod North
(NMFS Northeast Fisheries Science Center, unpublished data)

1.1.2 Fin Whale

1.1.2.1 Species Description

Fin whales (*Balaenoptera physalus*) are present in all the major oceans of the world from the Arctic to the tropics, with greatest numbers in temperate and boreal latitudes (Evans 1987). They are the most common of the large whales in the temperate waters of the western North Atlantic, and are found along the continental shelf between Cape Hatteras and southeastern Canada in all seasons at depths of 69 to 328 ft (21 to 100 m) (Waring et al., 2006). Because the fin whale is the most numerous of the large whales with the largest food requirements, it has the largest impact on the continental shelf ecosystem of any whale species, and may be a valuable indicator of the health of this area (Hain et al., 1982).

Fin whales are long and slender, growing to a maximum size of 88 ft (27 m) and 73,000 kg (161,000 lbs), and individuals from the Southern Hemisphere tend to grow to a larger size than those in the Northern Hemisphere with females are generally larger than males (Slijper 1978;

Minasian et al., 1984). Color variations in fin whales range from dark gray to brown, asymmetrical coloring on their head, and white on the ventral sides of their belly, flukes and flippers. Distinctive features include the tall, falcate dorsal fin, the light pigmentation (or blaze) on the right side of the head, and the V-shaped gray-white "chevron" on the back and sides (Agler et al., 1990).

1.1.2.2 Life History

Relatively little is known about reproduction in North Atlantic fin whales. Presumably, reproduction takes place during their winter sojourn off the mid- and south-Atlantic states. Some evidence suggests that calving occurs in coastal or offshore waters south of New Jersey between October and January, and wintering grounds are south of Cape Hatteras (Waring et al., 2006). No mating or breeding is known to occur in the Gulf of Maine and Canadian waters (Waring et al., 2006).

Females reach sexual maturity after four to seven years, apparently depending on availability of food (Ohsumi 1986). The size at sexual maturity 49.2 to 52.5 ft (15 to 16 m). Newborn fin whales are approximately 26.2 ft (8 m) long and are weaned at age seven months to one year (Waring et al., 2006). Fin whales may have a calf every two years (Agler et al., 1993). The average life span for fin whales ranges from 85 to 90 years (Evans 1987).

Fin whales are euryphagous (*broad diet*), and feed on a wide variety of small schooling fish and crustaceans such as sand lance, capelin, euphausiids, copepods, squid (*Loligo* spp. and *Ilex* spp.) and myctophid fish (Katona et al., 1977; Overholtz and Nicolas 1979; McKenzie and Nicolas 1988; Piatt et al., 1989; Waring et al., 2006). Fin whales eat many of the same foods as humpback whales and the two species frequently are seen feeding together in spring and summer feeding areas (CeTAP 1982). Fin and humpback whales likely compete directly with cod, haddock, other piscivorous ground fish, and humans for food (Overholtz and Nicolas 1979; Hain et al., 1985).

Feeding behaviors of fin whales are less well known than those of right and humpback whales. They appear to feed individually or in groups of two to 50 animals (CeTAP 1982). Fin whales are streamlined, fast swimmers and typically cruise at speeds of five to ten km/hr (Hain 1991b). They apparently use this speed to feed on less dense, more widely separated patches of prey species (Whitehead and Carlson 1988). However, Brodie et al., (1978) observed high densities of euphausiids in fin whale stomach contents, suggesting that fin whales focus their feeding efforts on dense aggregations of prey when available.

1.1.2.3 Population Dynamics

The estimated modern worldwide population of fin whales is 105,000 to 125,000 individuals (Würsig 1990). Fin whales are the most abundant and frequently sighted of the endangered great whales visiting the coastal waters of the northeastern United States. The 2001 NMFS Marine Mammal Stock Assessment Report estimates the western North Atlantic stock abundance at 2,814 (CV=0.21), with a minimum size estimate of 2,362, based on a 1999 line-transect sighting survey conducted by a ship and airplane covering waters from Georges Bank to the mouth of the

Gulf of St. Lawrence (Waring et al., 2006). Because of the fin whale's extended distribution and poorly understood population structure, this is considered to be an extremely conservative estimate.

Stocks of fin whales in the United States, Nova Scotia, and Labrador are believed by some to be from one or a few closely related populations, whereas the Icelandic population is distinct (Waring et al., 2006). However, the population structure is not well understood. Waring et al., (2006) reported that a recent genetic study published by Berube et al., in 1998 supports the presence of several subpopulations first suggested by Kellogg in 1929. Because little is known about the winter breeding and calving areas of fin whales, it is uncertain whether fin whales segregate into a few separate breeding populations that form several distinct summer feeding aggregations occupying different feeding grounds. It has been suggested different subpopulations use the same feeding grounds (Waring et al., 2006). It is unclear whether fin whales in the North Atlantic split into separate feeding stocks. Mitchell (1974) suggested that fin whales seen off the United States, Nova Scotia, and Labrador coasts were from one or a few closely related populations. Fin whales often travel alone, but an average group size ranges from two to three individuals. Groups can get as large as 65 individuals, though groups of more than ten are unusual.

Because fin whales are the fastest swimmers of the baleen whales, it is unlikely that predation by killer whales and large sharks is an important cause of natural injury and death, except possibly among the very young or sick. Nevertheless, the literature contains some records of attacks by killer whales on fin whales (Tomlin 1957). Other natural mortality factors may include accumulation of biological toxins from prey species to whale tissues through trophic transfer and biomagnification. At the time in 1987 when fourteen humpback whales died from consumption of mackerel contaminated with phytoplankton toxin (see Section 4.1.4, above), two partly decomposed fin whales washed up on the western shore of Cape Cod Bay at Marshfield and Manomet, MA (Waring et al., 2006). The cause of death was not determined, but may have been consumption of the contaminated fish, as fin and humpback whales eat similar foods.

1.1.2.4 Status and Distribution

Fin whales were listed as endangered throughout their range in 1970. Because of their high cruising speed, fin whales were not harvested commercially in large numbers until other species, such as slow-moving right whales, were depleted and whalers developed high-speed boats (Leatherwood et al., 1976). Nonetheless, more than 700,000 fin whales were harvested worldwide in the twentieth century (NMFS 1994). A fishery for this species existed in Nova Scotia from 1964 to 1972 (Waring et al., 2006). During this period, 3,528 individuals were harvested. Commercial harvesting of fin whales elsewhere in the world continued at least into the early 1990s. However, stocks of fin whales have not been as severely depleted by commercial whaling as other stocks of large whales.

It is probable that the hazards that affect humpback whales also affect fin whales. Fin whales often are caught in fish traps deployed in offshore Canadian waters. Between 1969 and 1986, twelve fin whales were entangled in fishing gear, usually groundfish gill nets, in inshore waters of Newfoundland (Hofman 1990). Five of these whales (42 percent) died. Between 1975 and

1990, three fin whales were observed entangled in fishing gear in the Gulf of Maine (Volgenau and Kraus 1990). All entanglements were in lobster gear. The commercial lobster industry reported six instances of fin whale entanglements in lobster gear between November 1975 and January 1991 (NMFS 1994). All but one of the whales was alive when sighted. Three of the entangled whales were sighted in Massachusetts, two in New York, and one in Maine. Such entanglements may indicate that fin whales sometimes feed near or at the bottom. NMFS fisheries observers reported no fisheries-related fin whale mortalities for the period 1999-2003 (Waring et al., 2006). Anecdotal records from NMFS for the same period found two records of fishery related mortality or serious injury to fin whales, yielding a minimum annual rate of mortality and serious injury from fishery interaction of 0.4 individuals for United States and Bermuda's waters (Waring et al., 2006). An additional five records for the same period did not contain enough information to determine if the entanglement was severe enough to cause serious injury or if the entanglement contributed to mortality. In 2004, the CCS documented three entangled fin whales from Stellwagen Bank, North Carolina, and the Bay of Fundy (CCS 2004).

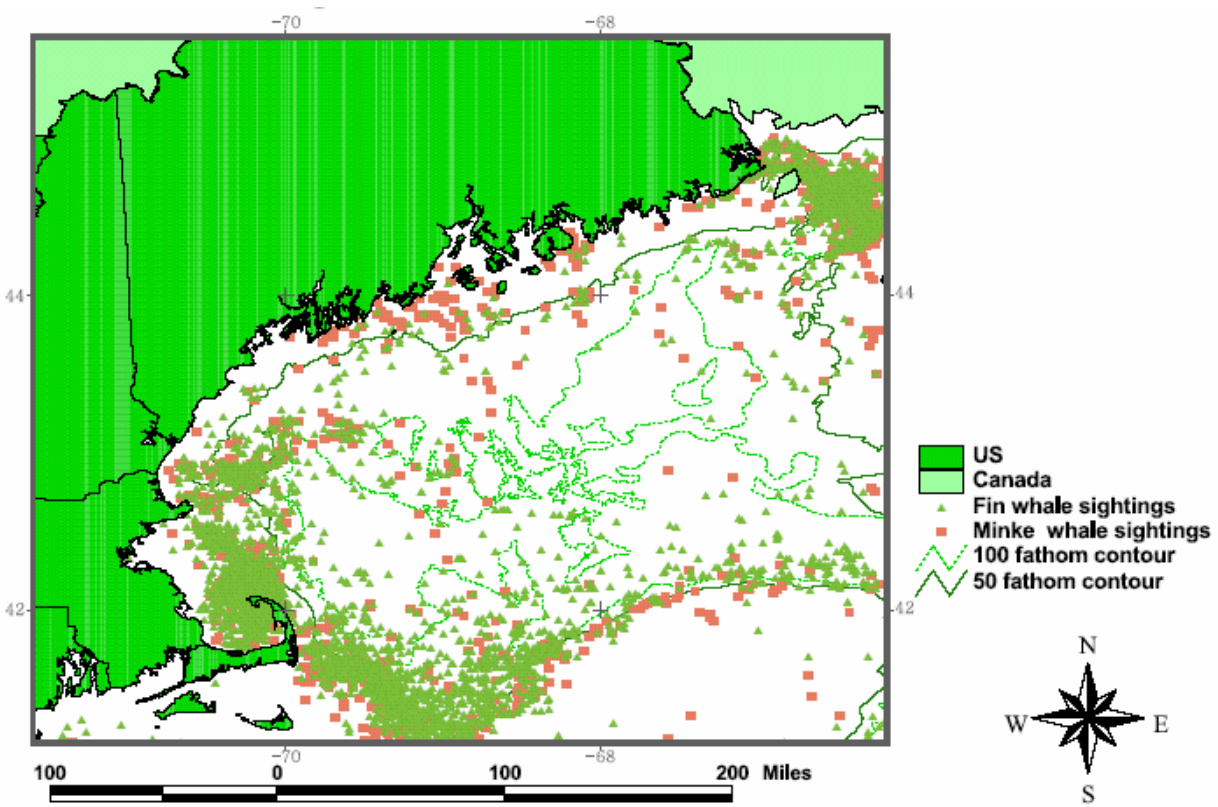

Figure C-2.
North Atlantic Fin and Minke Whale sitings through 2002 – Cape Cod North
(NMFS Northeast Fisheries Science Center, unpublished data)

Figure C-2 provides data on the sightings of fin whales in the Gulf of Maine and Cape Cod regions. The sighting data indicates three areas of congregation, Georges Bank, Stellwagen Bank, and in the northern Gulf of Maine. One fin whale has been reported in the area off the Falmouth coast, but none have been observed within the rest of Nantucket Sound or Horseshoe

Shoal. The sightings data should not be considered as an absolute documentation of the occurrence of any particular species, because the results are dependent in part on the level of effort that is expended in looking for whales. However, as a general indicator of where fin whales are likely versus unlikely to occur, the data suggest that very few fin whales are likely to occur in or near the site of the proposed action during construction, operation or decommissioning.

1.1.3 North Atlantic Right Whale

1.1.3.1 Species Description

The North Atlantic right whale (*Eubalaena glacialis*) was a prime target of early whale fisheries from the 1100s through the early 1900s, due to its coastal distribution, slow swimming speed, high oil yield, and characteristic of floating when dead (Brown 1986; Aguilar 1986). Due to intense exploitation, it is one of the rarest of the large whales and is in danger of extinction. Historically, there was an eastern and western stock of right whales in the North Atlantic, but current evidence suggests that the eastern stock may be extinct or on the verge of extinction (Brown 1986; Best 1993).

The majority of right whales sighted in the North Atlantic Ocean are approximately 36.1 to 49.2 ft (11 to 15 m) in length and weigh up to 140,000 lbs (63,503 kilograms) (Kraus et al., 1988). Females are larger than males. Right whales can be distinguished from other baleen whales by their black color, stocky body, the absence of a dorsal fin, short, paddle-shaped flippers, large head (more than 25 percent of the total body length), and a strongly bowed lower jaw. The distinct "V-shaped" blow provides a means of identification from a distance. Individuals are identified through the distribution and size of thickened, patches of epidermis called callosities on the rostrum, chin, and lower lips varies among right whales and can be used in conjunction with other unique features, such as scars and pigmentation patterns (Payne et al., 1984; Kraus et al., 1986).

1.1.3.2 Life History

In both the northern and southern hemisphere, females give birth to their first calf at an average age of nine years (NMFS 2005). Calves are 18.0 to 19.7 ft (5.5 to 6 m) in length at birth (Best 1994). Gestation lasts about a year, and the mean calving interval for female right whales is less than four years (NMFS 2005; Waring et al., 2006). The coastal waters of Georgia and northeastern Florida are the only known calving grounds of the North Atlantic right whale, but limited surveys recently conducted along the Mid Atlantic suggest some mother-calf pairs use the area from Cape Fear North Carolina to South Carolina as a wintering/calving area as well (NMFS 2005). The calving season extends from late November to early March, and appears to peak in January.

Generally, right whales are found along the east coast of North America (CeTAP 1982) but, in the last century, have been seen as far north as Greenland, Iceland, and arctic Norway, as far east as Bermuda, and as far south as the Gulf of Mexico (Waring et al., 2006). Right whales, like other large whales, are migratory animals (Gaskin 1982). Some female right whales have been

observed to migrate more than 1800 mi (2900 km) from their northern feeding grounds to the southern calving/wintering grounds (Waring et al., 2006). Right whale seasonal movements occur among the following five areas of "high use" 1) Coastal Florida and Georgia (Sebastian Inlet, Florida to the Altamaha River, Georgia); 2) The Great South Channel (east of Cape Cod); 3) Massachusetts Bay and Cape Cod Bay; 4) The Bay of Fundy; and 5) The Scotian Shelf, including Browns and Baccaro Banks, Roseway Basin and areas to the east (NMFS, 2005).

The primary prey of right whales in the western North Atlantic is calanoid copepods, *Calanus finmarchicus*, and juvenile euphausiids (Nemoto 1970; Watkins and Schevill 1979; Kraus and Prescott 1982; Murison and Gaskin 1989), and secondarily the copepods *Pseudocalanus minutus* and *Centropages* sp. (Marx and Mayo 1992). Both the density of plankton patches and the proportion of caloric-rich adult (Stage V) copepods appear to be factors influencing the foraging threshold of right whales (Murison and Gaskin 1989; Marx and Mayo 1992; Waring et al., 2006).

1.1.3.3 Population Dynamics

A census in 1992, based on photo-identification techniques, estimates the western North Atlantic population at 295 individuals (NMFS 2005). An updated analysis using the same method gave an estimate of 299 animals in 1998 (Waring et al., 2006). Because this was nearly a complete census, it is assumed that this represents a minimum population size estimate (Waring et al., 2006). The 1998 IWC right whale workshop accepted an estimate of about 300 individuals for the western North Atlantic population based on this information (NMFS 2005). The 2005 Marine Mammal Stock Assessment Report (Waring et al., 2006) has not updated these population estimates.

The use of a given nursery by females is culturally transmitted (Schaeff et al., 1992). Not all mother-calf pairs that are seen in the southeastern United States region wintering grounds are observed the following summer in the Bay of Fundy (Waring et al., 2006). In addition, based on mitochondrial DNA (mtDNA) data, one of the three known matrilines does not appear to bring its calves to the Bay of Fundy summer nursery area (Schaeff et al., 1992). Therefore, it is likely that at least one other nursery area exists. Further work has identified two additional matrilines (NMFS 2005).

Sei whales (*Balaenoptera borealis*) (Mitchell 1975; Mitchell et al., 1986), sand lance (*Ammodytes spp*.) (Waring et al., 2006), and planktivorous species could represent a source of competition for the right whale's preferred prey *(Calanus finmarchicus)*. In 1986, when *C. finmarchicus* levels were high in the Gulf of Maine, right whales, fin whales, and sei whales were the dominant whales in the area. Although Waring et al., (2006) reported an increase in sei whales in the Great South Channel and Nova Scotian Shelf, there is little quantitative evidence of direct competition between right whales and these other species. In addition, *C. finmarchicus* populations are highly variable, and little of this variation is due to predation pressure (McLaren et al., 1989; Tande and Slagstad 1992).

1.1.3.4 Status and Distribution

The pre-exploitation western North Atlantic population is estimated to have numbered 10,000 animals (Waring et al., 2006). Commercial harvest of the species over the centuries resulted in the decimation of the population to possibly less than 50 animals at the turn of the century (Waring et al., 2006). Although protected by international law since 1935, currently the population is believed to contain only about 300 individuals and it's unclear as to whether its abundance is remaining static, undergoing modest growth or, as recent modeling exercises suggest, in decline (NMFS 2005). The North Atlantic right whale has been listed as endangered under the Endangered Species Act (ESA) since its passage in 1973 (35 FR8495, June 2, 1970). The NMFS approved a recovery plan in December 1991, under Section 4(f) of the Endangered Species Act (NMFS 1994), and a revised plan incorporating revisions from July 2001 and August 2004, was approved on May 26, 2005.

The greatest known current cause of right whale mortality in the western North Atlantic is collision with ships (NMFS 2005). Of 45 confirmed deaths of western North Atlantic right whales between 1970 and 1999, 16 (35.6 percent) are known to have been caused by ship strikes and two additional collisions were judged as possibly fatal (Waring et al., 2006). Entanglement in fishing gear is the second largest human-related threat to right whales. In a recent analysis of the scarification of right whales, more than half (61.6 percent) of the appropriately photographed population of right whales have scars indicative of entanglement in commercial fishing gear (Waring et al., 2006). Habitat change and degradation is also a key environmental factor affecting the rate of recovery of the right whale (NMFS 2005; Gaskin 1982). Neonatal and juvenile right whales appear to be the most vulnerable and impacted part of the population. Analyses of sighting data between the northern feeding areas and the southern calving areas indicate that about 17 percent of calves die within their first year of life. After the first year, mortality rates drop to an average of 3 percent for the next three years, or a total of 27 percent mortality for the first four years of life (Kraus 1990). Of the known 45 right whale mortalities that occurred from 1970 to 1999, 13 (28.9 percent) deaths were neonates believed to have died from perinatal or other natural causes (Waring et al., 2006). Thus, even a few incidental deaths may greatly affect the rate of recovery in a drastically reduced population with such a long reproductive cycle (Best 1988). Thirteen documented right whale mortalities and serious injuries were attributed to human causes for the period 1999 to 2003 (Waring et al., 2006).

In May 2000, NMFS requested consultation on Fisheries Management Plans for multiple fisheries because of several right whale entanglements in 1999 and new information provided by the International Whaling Commission indicating that the North Atlantic right whale population may be declining. As described in NMFS (2005), the Biological Opinions identified reasonable and prudent alternatives to be implemented by NMFS which included: developing and implementing annual restrictions to fishing operations aimed at minimizing interactions between fisheries and right whales; implementing a "dynamic area management" program to supplement annual restrictions with temporary closures at times and in places where right whales aggregate; and expanding gillnet gear modification research program and extension of gear modification requirements to include waters off mid-Atlantic and southeast states. NMFS implemented these fishery restrictions through three rules that (1) make further modifications to commercial fishing gear (67 FR 1300, January 10, 2002; see also 67 FR 15493, April 2, 2002; 67 FR 59471,

September 23, 2002), (2) establish restricted areas based on annual, predictable aggregations of right whales (67 FR 1142, January 9, 2002), and (3) establish a system for restricting fishing in areas where unexpected aggregations of right whales are observed (67 FR 1133, January 9, 2002; see also 68 FR 51195, August 26, 2003).

The revised recovery plan for the North Atlantic right whale (NMFS 2005) identified that the most immediate need for the North Atlantic Right Whale is to reduce or eliminate human-related deaths and injuries. At present, these result primarily from ship collisions and fishing gear entanglement (NMFS 2005). The new recovery plan also identified that the secondary, but still high priority, needs to involve other actions of importance to the species' management, including characterization and monitoring of important habitat, and protection of this habitat; and identification and monitoring of the status, trends, distribution and health of the species (NMFS 2005).

Figure C-1 provides data on the sightings of right whales in the Gulf of Maine and Cape Cod regions. The sighting data indicates three areas of congregation, Georges Bank, Stellwagen Bank, and in the northern Gulf of Maine. Right whales have been reported off Monomoy and the northern tip of Nantucket Island, but none have been observed within the rest of Nantucket Sound nor Horseshoe Shoal. The sightings data should not be considered as an absolute documentation of the occurrence of any particular species, because the results are dependent in part on the level of effort that is expended in looking for whales. However, as a general indicator of where right whales are likely versus unlikely to occur, the data suggest that very few right whales are likely to occur in or near the site of the proposed action during construction, operation, or decommissioning.

1.2 Sea Turtles

1.2.1 Kemp's Ridley Sea Turtle

1.2.1.1 Species Description

The Kemp's ridley (*Lepidochelys kempi*) sea turtle is distributed throughout coastal areas of the Gulf of Mexico and the northwestern Atlantic Ocean, and is assumed to constitute a single stock (TEWG 1998). A small sea turtle, adult Kemp's ridley females have carapace lengths of 2 to 2.25 ft (0.6 to 0.7 m) and weigh 77 to 100 lbs (35 to 45 kg) (NRC 1990). Most ridleys that visit the east coast of the United States are juveniles, averaging 0.8 to 1 ft (0.25 to 0.3 m) long and weighing about 6.6 lbs (3 kg) (NMFS 1988; NOAA 1991).

1.2.1.2 Life History

Female ridleys reach sexual maturity when they reach a carapace length of about 22 to 23 in (0.58 to 0.60 m) and are six to nine years old (Márquez 1994). The mature females nest annually and produce one to three (average about 1.7) clutches per season containing a total of about 120 to 190 eggs. Little is known about the sex ratio of ridley turtles or about the life history of the males. Nearly all reproduction takes place along a single 9-mi (15-km) stretch of beach near Rancho Nuevo, Mexico, about 200 mi (322 km) south of Brownsville, Texas. Nesting occurs in

a highly synchronized manner with large numbers of females coming ashore within a period of a few daylight hours (NRC 1990). Longevity is greater than 20 years.

Following a pelagic feeding stage shortly after hatching and lasting for several months (Carr 1986a), juvenile ridleys move into shallow coastal waters to feed and grow. The young juveniles often forage in water less than one meter deep, but they tend to move into deeper water as they grow (Ogren 1989). In New England waters, nearly all feeding takes place on or near the bottom in shallow water (Morreale and Standora 1989, 1992). The deepest recorded dive of a juvenile ridley was to 69 ft (21 m); dives usually level off at about 50 ft (15 m) if the bottom isn't reached (Morreale and Standora 1989).

All the ridley turtles in New England waters are two- to five-year old juveniles with carapace lengths of 8.6 to 15 in (22 to 38 cm) (Burke et al., 1989; Morreale and Standora 1989). They begin arriving in northern waters in late May and June each year and remain in shallow nearshore waters, particularly in the bays on eastern Long Island, during the summer (Keinath et al., 1987; Musick and Limpus 1997). They begin leaving the area in mid-September and most have left for warmer southern waters by the beginning of November. Some ridleys may hibernate over the winter in nearshore sediments (Carminati et al., 1994). Most of the ridleys observed after the beginning of November are cold-stunned. Ridleys become sluggish and have labored breathing when the temperature falls below 55.4 ^{o}F (13 ºC); feeding ceases below 50 ^{o}F (10 ºC), and they die when water temperatures reach between 43.7 and 41 ^{o}F (6.5 and 5.0 ºC) (Schwartz 1978).

In coastal waters of New York and New England, young ridleys consume several species of crabs, including in order of decreasing preference, spider crabs (*Libinia emarginata*), lady crabs (*Ovalipes ocellatus*), and rock crabs (*Cancer irroratus*) (Morreale and Standora 1992, 1989). Crustaceans represent more than 80 percent of the diet of juvenile ridleys in the New York Bight (Burke et al., 1994). Other food items found in ridley stomachs include mollusks and algae. The preference for spider crabs over lady crabs, despite the fact that the latter is more abundant in ridley foraging habitat, is probably due to the greater ease of capture of the slower moving spider crabs by the small turtles (Morreale and Standora 1989). Ridley turtles make long dives to the bottom and may feed on the bottom for an hour or more at a time; one turtle was observed burrowing in the bottom of Long Island Sound (NMFS 1988). During daylight hours, ridleys spend most of their time under water. In a typical dive the turtle spends about 56 percent of its time in the upper third of the water column, 12 percent in the middle, and 32 percent of its time on the bottom (Morreale and Standora 1989). In water deeper than about 50 ft (15 m), the turtles usually dive to a depth of 19.7 to 32.8 ft (6 to 10 m) where they appear to be swimming in a directed manner.

1.2.1.3 Population Dynamics

Estimates of adult abundance indicated 9,600 individuals in 1966, 1,050 in 1985, and 3,000 in 1995 (TEWG 2000). The total population, adults and juveniles, may number 22,000 to 110,000 individuals. The total nesting population of females during the mid- to late-1980s has been estimated to number about 600 individuals, with each female laying about two clutches of eggs per year (Pritchard 1990). When compared to the estimated number of nests in 1947

(92,000), this is the most severe population decline documented for any species of sea turtles (NRC 1990).

Several stages in the life cycle of Kemp's ridley turtles are sensitive to natural and anthropogenic disturbance. Each year between November and January when ocean water temperatures are falling, small numbers of ridley turtles become stranded and die on beaches of the north and east shores of Long Island and Cape Cod Bay (see Figure C-3), due to cold stunning (NOAA 1991; Morreale and Standora 1992). When the water temperature drops below about 53.6 ^{o}F (12ºC), the metabolic rate of these cold-blooded reptiles decreases to the point where they are unable to swim and digest food; they become comatose and may die if not warmed quickly. A total of 115 ridley turtles were stranded on Cape Cod beaches between 1977 and 1987 (Danton and Prescott 1988). For the period of 1990 to 2001, between nine and 216 ridleys strandings were reported in Massachusetts waters, and one ridley stranding was reported in Rhode Island waters (Sea Turtle Stranding and Salvage Network, unpublished data). Sea turtle stranding data collected by the Massachusetts Audubon Wellfleet Bay Wildlife Sanctuary (MA Audubon 2005b) indicate that 79 and 32 kemp's ridleys were reported stranded on Cape Cod beaches in 2003 and 2004, respectively. Cold stunned ridleys have stranded as far south as the Indian River Lagoon, FL (Wilcox 1986). However, as shown in Figure C-3, ridley sea turtles are much more likely to become stranded on the north shore of Cape Cod, and in the1980-1997 timeframe, only one was found in the vicinity of Waquoit Bay. The strandings data are not the equivalent of sightings data, and may or may not represent density or abundance of occurrence, but may rather reflect a behavioral component of their seasonal movement patterns.

1.2.1.4 Status and Distribution

The Kemp's ridley sea turtle was listed under the Endangered Species Act as endangered on December 2, 1970, and status has remained unchanged as it is the most endangered sea turtle in the world.

A major cause of sea turtle mortality attributable to humans is entanglement in fishing gear, particularly shrimp nets before the implementation of turtle excluder devices (NRC 1990). Most of the mortalities attributable to entanglement in shrimp nets are in the Gulf of Mexico. Other fishing-related deaths, caused by entanglement in lobster gear and pound nets, may result in an additional 50 to 500 deaths of Kemp's ridley turtles each year (O'Hara et al., 1986; Morreale and Standora 1989). Ridley turtles, being benthic feeders, tend to become entangled in debris, including abandoned fish and crab traps, on the bottom. This incidental catch could represent as much as 7.5 percent of the hatchling ridleys produced each year, assuming that the 800 nests produced a total of 80,000 hatchling ridley turtles each year. This additional mortality undoubtedly is contributing to the rapid decline in the population of Kemp's ridley turtles.

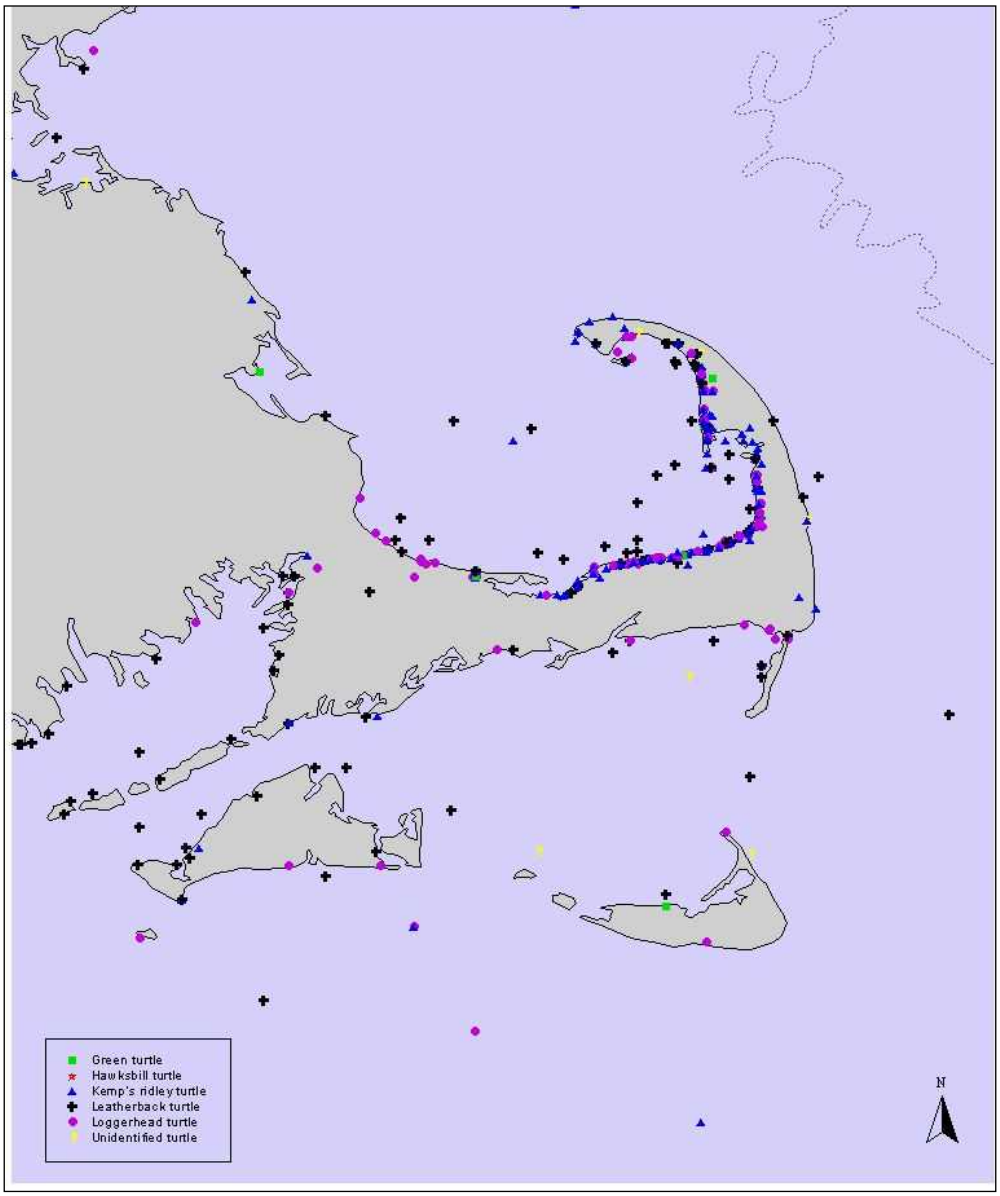

Figure C-3.
Sea turtle strandings on Cap Cod, Nantucket, and Martha's Vineyard Areas for the period 1980 to 1997 (NMFS unpublished data)

Under some circumstances, chemical pollution may be a threat to ridley turtles. As part of the Sea Turtle Head Start Program, 12,422 one-year-old ridley turtles were tagged and released between 1979 and 1987 (Manzella et al., 1988). In 1982, 1,325 ridleys were released 3.7 to 6.2 mi (6 to 10 km) off the Texas coast in floating patches of *Sargassum* weed. More than 28 percent of the turtles washed ashore within 14 days of release, and most were coated with oil or had ingested tar balls, probably associated with the *Sargassum*. Because early pelagic stage ridleys are thought to congregate and feed in rafts of *Sargassum*, they may be vulnerable, as juvenile loggerhead turtles are (Carr 1987), to floating oil and nondegradable debris that tends to

collect in drift lines of *Sargassum*. Ridleys feeding in *Sargassum* rafts or on benthic prey may accumulate metal and organic contaminants from their prey.

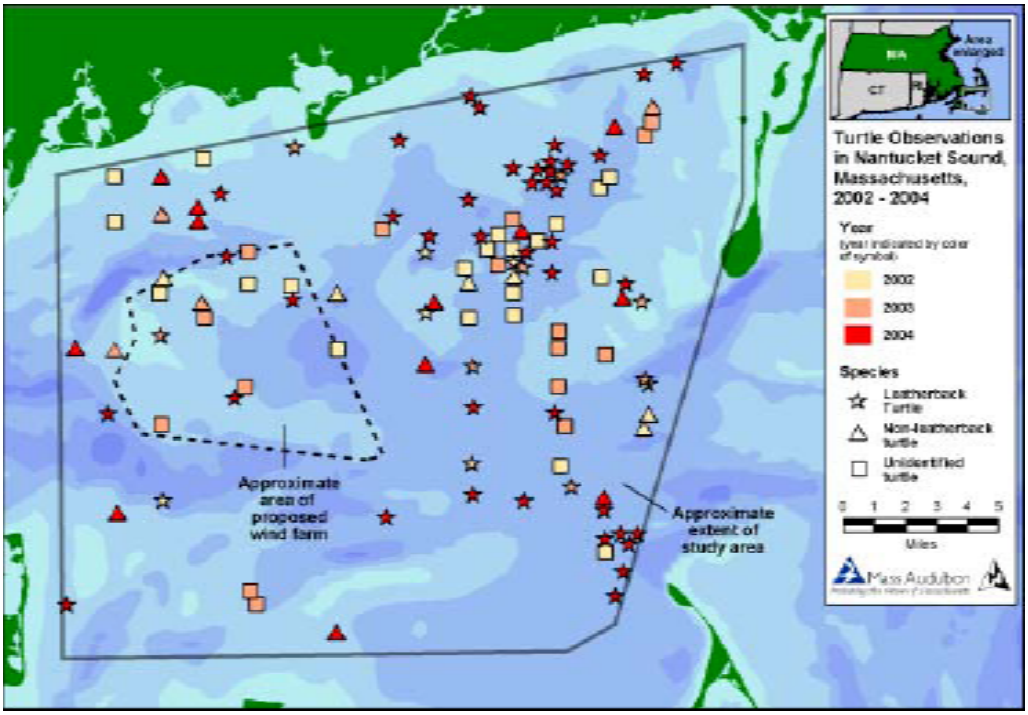

Figure C-4.
Incidental observations of 115 sea turtles made during 35 pre-migratory staging period aerial surveys of Nantucket Sound, Massachusetts, 2002-2004 (MA Audubon 2005a)

In comparison to Figure C-3, Figure C-4 presents incidental observations of sea turtles made during late summer while performing surveys of birds from an airplane. Given the challenges of identifying sea turtle species from a moving airplane, the majority of the sightings are not species specific. During the 2002-2004 study period, sea turtles where observed within the proposed action area of Nantucket Sound. If Ridley sea turtles were among those observed, and they were feeding, they would most likely be found in the shallower portions of Horseshoe Shoal at the western end where some scattered sea grass and macroalgae were observed by the Applicant. However, the greatest number of observations are located to the east of the proposed action area in deeper water.

1.2.2 Leatherback Sea Turtle

1.2.2.1 Species Description

Leatherback turtles (*Dermochelys coriacea*) forage in temperate and subpolar waters and nest on tropical beaches. The leatherback is the second most common turtle along the eastern seaboard of the United States, and the most common north of the 42°N latitude. Leatherback turtles are the largest and most distinctive of the living sea turtles. Because of their unique

anatomy and physiology, they are classified in a separate family, the Dermochelyidae, containing a single species (NMFS & USFWS 1992). Leatherbacks reach a length of 5 to 5.5 ft (1.5 to 1.7 m) SLCL and a weight of 110 lbs (500 kg) (exceptionally 1985 lbs [900 kg]). Large outstretched front flippers may span 8.9 ft (2.7 m) in an adult. Lacking a keratinized shell, they are covered instead with a tough hide. Leatherbacks have a layer of subcutaneous fat that is 2.3 to 2.7 in (6 to 7 cm) thick and circulatory adaptations to reduce the rate of heat loss through the flippers (Greer et al., 1973). They respond to drops in ambient temperature by increasing metabolic heat production and so can maintain an internal body temperature well above ambient (Standora et al., 1984; Paladino et al., 1990). A leatherback in 45.5 ^{o}F (7.5 °C) seawater was able to maintain its core body temperature at 77.9 ^{o}F (25.5 °C) (Friar et al., 1972). This endothermy allows leatherbacks to survive and feed in colder temperate waters than other sea turtles can tolerate. Therefore, adult leatherbacks are more widely distributed than other sea turtles in temperate and boreal waters throughout the world. However, all leatherbacks return to subtropical and tropical shores to nest.

1.2.2.2 Life History

Each female may nest up to ten times (mean frequency five to seven times, depending on year) in a single season at intervals of about ten days (Tucker 1989). Females usually nest only every other year (NRC 1990; Boulon et al., 1994). Most nesting takes place during March and April (NOAA 1991). A typical nest on a Culebra beach contains about 30-115 eggs (mean 70), each about 2.1 in (5.4 cm) in diameter (Hall 1990). Some of the eggs do not have a yolk and are infertile. The eggs hatch after about 65 days.

The seasonal distribution of leatherback sea turtles in the North Atlantic waters range from Cape Sable, Nova Scotia south to Puerto Rico and the US Virgin Islands. Leatherback turtles are common during the summer in North Atlantic waters from Florida to the Gulf of Maine, the Canadian Maritime Provinces, and occasionally as far north as Baffin Island, Canada (Goff and Lien 1988). New England and Long Island Sound waters support the largest populations on the Atlantic coast during the summer and early fall (Lazell 1980; Prescott 1988; Shoop and Kenney 1992). Leatherbacks are observed frequently in lower Chesapeake Bay and off the mouth of the Bay during the summer, where they probably are feeding on locally abundant jellyfish (Barnard et al., 1989).

Leatherback turtles are pelagic feeders, though they can dive to considerable depths. They feed throughout the water column to depths of at least 3280 ft (1000 m) (Eckert et al., 1989) on jellyfish and other gelatinous zooplankton, such as salps, ctenophores, and siphonophores (Limpus 1984). Most feeding dives average about 197 ft (60 m), but frequently extend to 985 to 1312 ft (300 to 400 m) (Eckert et al., 1986, 1989) where they feed on deep-water gelatinous zooplankton, such as siphonophores and salps. Their seasonal inshore movements in New England waters have been linked to inshore movements of their preferred prey, the jellyfish (*Cyanea capillata)* (Lazell 1980; Payne and Selzer 1986). Leatherbacks have a notched upper jaw, an adaptation for grasping soft prey (Pritchard 1971). They also possess a long digestive tract, about nine times longer than the length of the carapace, and a large caecum for holding the quantities of watery, gelatinous prey they need to consume to fulfill their caloric needs (Bjorndal 1985).

1.2.2.3 Population Dynamics

Because they are a largely oceanic, pelagic species, estimates of their population status and trends have been difficult to obtain. In addition, only a small fraction of the North Atlantic population nests on beaches of the continental United States, mostly in Florida (NRC 1990; Meylan et al., 1994) and the U.S. Virgin Islands (Boulon et al., 1994). Leatherbacks that visit U.S. Atlantic waters nest primarily along the coasts of Surinam and French Guiana, and to a lesser extent on the island of St. Croix and at Culebra, Puerto Rico (NRC 1990; NMFS & USFWS 1992; Boulon et al., 1994). Nesting is scattered along isolated beaches throughout the Caribbean. Nesting females do not have the nest-site fidelity exhibited by Kemp's ridley turtles and tend to move to different beaches in different years (Tucker 1990). Therefore, it has been difficult to estimate temporal trends in population size.

Between 100 and 900 leatherbacks visit coastal and continental shelf waters of the western North Atlantic ocean between Canada and North Carolina each year, with peak abundance in summer (Shoop and Kenney 1992). As many as 115,000 adult female leatherbacks remain worldwide (Pritchard 1982), though a more recent estimate places the adult female population at 34,500 (Spotila et al., 1996). Spotila et al., (1996) estimate that between 7,813 and 13,833 female leatherbacks visit the largest Atlantic nesting colonies in French Guiana, Suriname, other locations in the Caribbean, and Gabon, West Africa each year.

1.2.2.4 Status and Distribution

The leatherback sea turtle was listed as endangered throughout its range on June 2, 1970, and status has remained unchanged (USFWS 1986). Current estimates are that 20-30,000 females exist worldwide. Of all Atlantic sea turtles, leatherbacks are the most vulnerable to entanglement in fishing gear. Between 1992 and 1999, 6,363 leatherback turtles were caught by US Atlantic tuna and swordfish longlines; 88 of those turtles died (NOAA Fisheries Service, SEFSC 2001). Many of the same natural and anthropogenic factors that affect survival of loggerhead and Kemps ridley turtles also affect leatherbacks. Being a temperate water species, leatherbacks do not seem to be sensitive to cold temperatures, and strandings cannot be attributed to cold stunning.

Between 1986 and 1999, 42 to 170 leatherback turtles were reported stranded on the U.S. Atlantic coast each year. Most strandings were in Florida and New York. Between four and 39 leatherbacks stranded each year in Massachusetts and Rhode Island waters during the period of 1990 to 2001 (Sea Turtle Stranding and Salvage Network, unpublished data). For the period of 1980 to 1997, twelve leatherback strandings were recorded in Nantucket Sound (NMFS unpublished data) (see Figure C-3), primarily scattered along the southern shore of Cape Cod and along the northern shore of Martha's Vineyard. The causes of these strandings are not known, but entanglement in fishing gear may be a major factor. Leatherbacks are also very susceptible to entanglement in other fishing gear and in plastic debris (Mager 1985; Witzell and Teas 1994). Because of their preferred diet of gelatinous zooplankton, particularly jellyfish, leatherback turtles often ingest floating plastic debris, mistaking it for food (Wallace 1985; O'Hara 1989). Plastic bags blocked the stomach openings of 11 of 15 leatherbacks that washed

ashore on Long Island during a two-week period (Balazs 1985). Leatherbacks have also been entangled in lobster gear (O'Hara et al., 1986; Sadove and Morreale 1990) and long-lines (Balazs 1985) in New York Bight and New England waters. Subsistence harvesting also places pressure leatherback populations. Although leatherbacks are not harvested commercially for meat or other products, extensive subsistence harvesting of the females that come ashore to nest occurs throughout much of the tropical nesting range, including Guyana, Trinidad, and Columbia (NRC 1990). Egg collecting is also intense in some areas.

Figure C-4 presents incidental observations of sea turtles made during late summer while performing surveys of birds from an airplane. Given the challenges of identifying sea turtle species from a moving airplane, the majority of the sightings are not species specific. However, the leatherback turtle is the one species that occurs in the proposed action area that was able to be identified to species, for at least some of the observations. During the 2002 to 2004 study period, leatherbacks where observed within the proposed action area of Nantucket Sound, although they were most frequently observed east of Horseshoe Shoal where water depths tend to be 40 to 50 feet.

1.2.3 Loggerhead Sea Turtle

1.2.3.1 Species Description

The loggerhead sea turtle (*Caretta caretta*) is the most common and seasonally abundant turtle in inshore coastal waters of the western North Atlantic. Most sightings of loggerheads are in shallow coastal bays and estuaries where the turtles feed on benthic invertebrates, particularly crabs (Morreale and Standora 1989). They rarely are observed in water depths of 65 ft (20 m) or more, and rarely feed at depths greater than about 50 ft (15 m).

Juvenile loggerhead turtles grow rapidly during their summer visits to northern coastal waters (Morreale and Standora 1989). The increase in SLCL of juvenile turtles in New York ranges from 4.1 in/year (10.6 cm/year) for 15.7 to 19.7 inch (40 to 50 cm) individuals to 1.2 in/year (3.0 cm/year) for 19.7 to 23.6 inch (50 to 60 cm) individuals. Growth rate slows as the turtles approach sexual maturity, which may occur after 12 to 45 years in the wild (Zug et al., 1983; Frazer and Ehrhart 1985; Foster 1994) when the turtles are about 29 to 35.5 inches (74 to 90 cm) SLCL (Dodd 1988; Foster 1994). Adult loggerheads from the Florida population may grow to more than 47 inches (120 cm) SLCL and weight more than 400 lbs (180 kg) (Ehrhart and Yoder 1978).

1.2.3.2 Life History

Hatchling loggerheads crawl from their nests to the sea and then float at the surface entrained in surface currents that may transport them far out to sea and across ocean basins. They may lead a pelagic life for up to 6.5 to 12 years, with an average of eight years, drifting and feeding in the *Sargassum* community (Carr 1986a, b, 1987; Bjorndal et al., 1994; Bolten et al., 1994; Bjorndal et al., 2000). During this long pelagic period, the young turtles, termed "pelagic immatures" may make several transits of the North Atlantic Ocean in the Great Gyre of the Gulf Stream and grow from a newly-hatched size of 1.8 inch (4.5 cm) to about 15.7 to 23.6 inch (40-

60 cm) straight carapace length (SCL) before moving inshore to coastal waters (Carr 1987; Dodd 1988) to benthic habitats. Loggerheads settle at an average size of 19.3 inch (49 cm) SCL, and are then known as "small benthic immatures" (Bjorndal et al., 2000) and will occur from Cape Cod through southern Texas. "Large benthic immatures" sized at 27.6 to 35.5 inch (70 to 90 cm) SCL will continue to mature for up to another ten years before reaching reproductive maturity (Carr 1987). Thirty five and one half inch (90 cm) SCL is considered the average size-to-maturity (NMFS-SEFSC 2001).

Migratory behavior seems to be cued to sea surface temperatures, with preferred water temperatures off Cape Hatteras falling in the range of 57.2 to 82.4 ^{o}F (14 ºC to 28 ºC) (Shoop and Kenney 1992; Coles et al., 1994). The Atlantic range of loggerhead turtles extends from Newfoundland to Argentina. Loggerheads occur in the Northeast from May 1 through November 15 (NOAA 2005a). Typically loggerhead turtles are abundant during spring and summer months in coastal waters off New York and the Mid-Atlantic States, and a small number of individuals may reach as far north as New England. In Northeast waters, these individuals consist mainly of juveniles (NOAA 2005a). In the fall, loggerheads migrate southward to coastal waters off the south Atlantic states, particularly Florida, and the Gulf of Mexico. During the winter, the turtles tend to aggregate in warmer waters along the western boundary of the Gulf Stream off Florida (Thompson 1988). In the spring, they congregate off southern Florida before migrating north to their summer feeding ranges (CeTAP 1982). Loggerheads are commonly found in waters of 72.2 to 160.8 ft (22 to 49 m); they range from the beach to the continental shelf (Shoop and Kenney).

Both benthic immature and adult loggerheads may travel great distances to foraging grounds (TEWG 1998). Adult loggerheads are primarily bottom feeders, foraging in coastal waters for benthic mollusks and crustaceans (Bjorndal 1985). During feeding, they spend more than 57 minutes of each hour submerged (Thompson 1988). Stomach contents from sub-adult loggerheads collected in Chesapeake Bay contained horseshoe crabs, cancer crabs, and blue crabs, with traces of *Sargassum* weed (Lutcavage 1981). In New England coastal waters, they feed primarily on small benthic crabs, such as spider crabs, rock crabs, and green crabs (Burke et al., 1989; Morreale and Standora 1989; Morreale and Standora 1992). Loggerhead turtles stranded on Cumberland Island, Georgia, had been feeding on a variety of crabs, whelks, and mantis shrimp (Ruckdeschel and Shoop 1988). Some turtles had large numbers of barnacles in their stomachs. Although loggerheads appear to feed primarily on the bottom on benthic invertebrates, they also take food from the water column or the water surface. Turtles frequently contain large amounts of sediment in their guts, probably ingested during feeding on benthic prey (Ruckdeschel and Shoop 1988).

1.2.3.3 Population Dynamics

Loggerhead turtle population estimates are best obtained from nesting data. One modeling method incorporated nesting and stranding data to estimate the loggerhead population size for the period of 1989 to 1995; a second method incorporated aerial survey data in addition to nesting and stranding data, for the same period of time. The two mean post-pelagic (i.e., benthic immature and adult) loggerhead population estimates were 224,321 and 234,355, respectively

(TEWG 1998). Due to sampling biases, these numbers are believed to be underestimates (TEWG 1998).

The TEWG (2000) reports that the South Florida subpopulation appears to be increasing and that no trends are apparent in the northern subpopulation. Confounding data for loggerheads in the Panhandle area, and sparse and/or incomplete data for the Yucatan and Dry Tortugas subpopulations cannot support trends analyses at this time (TEWG 2000; NMFS-SEFSC 2001). Fishery-independent trawl survey data from the Southeast Area Monitoring and Assessment Program (SEAMAP) covering the nearshore area from Cape Canaveral, FL to Cape Hatteras, NC for the period 1990 to 2000 have recently been analyzed (SCMRD 2000). While catches have increased over time, the error around each year's point estimate is large and no statistically significant trend of increase in the in-water loggerhead population in the Western North Atlantic is apparent (SCMRD 2000; NMFS-SEFSC 2001).

At least five genetically distinct nesting assemblages exist in the western North Atlantic: the Florida Panhandle subpopulation, the South Florida subpopulation, the northern subpopulation (Amelia Island, Volusia County, FL and northward), the Yucatan Peninsula subpopulation, and the Dry Tortugas subpopulation (TEWG 1998, 2000; NMFS-SEFSC 2001). Most recent evidence suggests that the number of nesting females in South Carolina and Georgia may be declining, while the number of nesting females in Florida appears to be stable (The Oceanic Resource Foundation 2005). As distinct reproductive populations, these nesting assemblages will not be replenished by regional dispersal if depleted.

The major sources of mortality of sea turtles, including loggerheads, caused by human activities include incidental take in bottom trawls, particularly shrimp trawls (Henwood and Stuntz 1987; Thompson 1988; NRC 1990; Anonymous 1992), coastal gill net fisheries, marine debris, and channel dredging (Thompson 1988; NMFS 1992). Loss of nesting habitat along the south Atlantic coast caused by coastal development has also likely slowed recruitment of sea turtles.

1.2.3.4 Status and Distribution

The loggerhead sea turtle (*Caretta caretta*) was listed as threatened throughout its range on July 27, 1978, under the Endangered Species Act and its status has not changed.

Strandings have caused a high number of mortalities for loggerhead sea turtles; however, natural causes of these strandings are not well understood. Between four and seventeen loggerheads stranded each year in Massachusetts and Rhode Island waters during the period from 1990 to 2001, though atypically high numbers of 72 and 56 loggerhead strandings were reported in Massachusetts in 1995 and 1999, respectively (Sea Turtle Stranding and Salvage Network, unpublished data). For the period of 1980 to 1997, seven loggerheads strandings were recorded on the shorelines in Nantucket Sound, primarily along the south coast of Cape Cod to the east of the proposed action area, and four strandings were reported on the southern shorelines of Martha's Vineyard and Nantucket (NMFS unpublished data) (see Figure C-3). Strandings occur most frequently in the fall and winter; these strandings may be caused by cold stunning (Morreale et al., 1992; Matassa et al., 1994). As with most marine turtles, prolonged exposure of

loggerheads to low water temperatures, below about 46.4 °F (8 ºC), may result in dormancy, shock, and death. During the winters of 1985, 1986, and 1987, 28 loggerhead turtles became cold-stunned and washed ashore in the Bay system of eastern Long Island and along the north shore of the island (Morreale et al., 1992). The turtles became cold-stunned between early November and late January each year. However, cold stunning is not restricted to northern waters, as demonstrated by several documented cold stunning incidents involving loggerheads in the northern part of the Indian River Lagoon system in east central Florida (Witherington and Ehrhart 1989; Schroeder et al., 1990).

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Figure C-4 presents incidental observations of sea turtles made during late summer while performing surveys of birds from an airplane. Given the challenges of identifying sea turtle species from a moving airplane, the majority of the sightings are not species specific. During the 2002-2004 study period, sea turtles where observed within the proposed action area of Nantucket Sound, some of which could have been loggerhead turtles. However, within the bird aerial survey study area, the greatest number of observations occurred in the deeper water areas east of the proposed action area.

1.2.4 Green Sea Turtle

1.2.4.1 Species Description

The green sea turtle (*Chelonia mydas*) is largely distributed in tropical and subtropical waters worldwide. Major green turtle nesting colonies occur in the central Atlantic on the beaches of Ascension Island (British Territory, South Atlantic Ocean), Aves Island (Venezuela), Costa Rica, and Surinam. The species has been observed along the United States' Pacific coastline from southern Alaska to Baja California, in U.S. Hawaiian waters, in the waters of most tropical islands in the central Pacific Ocean, and circumglobal in tropical and subtropical waters (NOAA 2002; USFWS 2002). In the Atlantic Ocean the green turtle occurs in Puerto Rico and the U.S. Virgin Islands and along the coast from Texas to as far north as Massachusetts waters with some degree of regularity; however, is considered an 'oceanic straggler in southern New England' by the USFWS (USFWS 2006). In comparison to other sea turtle species, there have been minimal recordings of the green turtle as far north as Cape Cod. The green turtles found near Cape Cod are three to four year old subadults, which are approximately 24 to 30 inches (61.0 to 76.2 cm) long and weigh about 50 lbs (22.7 kilograms) (Prescott 2000). Green turtle population trends are particularly difficult to assess because of annual fluctuations in numbers of nesting females, difficulties of conducting research on early life stages, and long generation time (NOAA 2002).

Newly-hatched green turtles are approximately 0.2 inches (5 mm) straight carapace length (SCL) and weigh approximately 0.9 ounces (25 grams). The adult stage of the species are the

largest of the marine turtles, during which the green turtle ranges in size from approximately 2.3 feet (0.71 m) to 5 ft (1.5 m) SCL, and can weigh up to 440 lbs (200 kg) (Crite 2000).

1.2.4.2 Life History

Green turtle hatchlings make their way from the nest to the sea, where they migrate along the coast from the rookeries to feeding grounds. They float at the surface and are carried by tropical and subtropical currents that may transport them far out to sea and across ocean basins, up to several thousand kilometers away (EuroTurtle 2005). Little is known regarding the duration of the juveniles' pelagic life until they reach a late-juvenile/subadult phase. It is believed that the green turtle may spend up to ten years in the juvenile, pelagic life stage before shifting to a benthic feeding stage (Crite 2000; Luschi et al., 2003). Numerous sources document that green turtles reach sexual maturity at various ages between 10 to 50 years of age (Crite 2000; NOAA 2002; EuroTurtle 2005).

While nesting on the beach and beginning their oceanic/pelagic phase hatchlings forage on available planktonic organisms (Crite 2000). As they increase in size, juvenile pelagic green turtles tend to feed on marine plants and organisms such as jellyfish, crabs, sponges, snails, and worms (Crite 2000). After a number of years in the oceanic zone, these late juvenile/subadult turtles recruit to neritic developmental areas that are rich in sea grass and marine algae where they forage and grow until maturity. Upon attaining sexual maturity, once every few years, green turtles commence breeding migrations between foraging grounds and nesting areas (Seminoff 2004). In years when they mate, green turtles migrate from several hundred to over a thousand miles across the ocean to the beaches where they hatched (Crite 2000). Adult green turtles are primarily herbivorous, feeding on shallow-growing algae and sea grasses in the protected waters of reefs, bays, inlets, lagoons, and shoal areas (USFWS 2002; Crite 2000). During non-breeding periods adults reside at coastal neritic feeding areas that sometimes coincide with juvenile developmental habitats (Seminoff 2004).

1.2.4.3 Population Dynamics

While there are no known records of documented green turtle feeding or nesting grounds along the beaches of New England, or more specifically, Nantucket Sound, documented accounts of green turtles in New England are most commonly instances of reported strandings.

In 2004, the World Conservation Union's (IUCN) Marine Turtle Specialist Group (MTSG) conducted a global assessment of the green turtle's worldwide population status. The MTSG's assessment focused on the number of mature individuals. Therefore, to accurately assess the population this assessment measured changes in the annual number of nesting females from 32 representative sites around the world. The index sites included all of the known major nesting areas and many of the lesser nesting areas for which quantitative data are available (IUCN 2004). Turtle population estimates are best obtained from nesting data. Scientists can use annual estimates of the number of nests laid each year to determine indirectly the number of adult females nesting in a given year and the number of adult females in the population (TEWG 1998). The main limitation to using nesting data to estimate population size is that the most mature life history stage of one gender is used to make estimates about the entire population. However,

nesting data are becoming available over large geographic areas and longer time periods, lending to their utility in population modeling.

Eighteen, or 56 percent, of the 32 sites the MTSG assessed displayed overall declining trends in green turtle nesting activity. Subpopulation declines of over 50 percent have been identified in the eastern and western Atlantic Ocean; and declines of greater than 80 percent have been shown in the eastern, southern, and western Pacific Ocean, Southeast Asia, Indian Ocean, and Mediterranean Sea. The most common intentional impact that has been documented among declining subpopulations is egg harvest, which occurs at beaches along the Pacific Coast of Mexico and along islands of Indonesia, the Philippines, and Malaysia (IUCN 2004).

The remaining 14, or 44 percent, of the 32 sites the MTSG assessed displayed stable or overall increasing trends in green turtle nesting activity. Stable populations were reported at the Galapagos Islands (Ecuador), Karan Island (Saudi Arabia), and Ras al Hadd (Oman). Increasing populations were reported at the following 11 sites: Ascension Island, Bijagos Islands (Guinea-Bissau, Africa), Comoros Islands (Indian Ocean), Florida, Hawaii, Heron Island (Australia), Raine Island, Sabah Turtle Islands (Malaysia), Surinam, Tortuguero (Costa Rica), and Yucatan (Mexico) (IUCN 2004).

Specifically in the Atlantic waters of the United States, green turtles nest in small numbers in the U.S. Virgin Islands and Puerto Rico, and they nest in larger numbers along the east coast of Florida. Although the Florida nesting subpopulation appears to be increasing, annual nesting effort fluctuates by a factor of ten. In more abundant years, annual nest counts have reached the low thousands (SCDNR Undated).

1.2.4.4 Status and Distribution

The green sea turtle was protected by the Endangered Species Act on July 27, 1978. The species is currently listed as threatened, except for populations of breeding colonies in the waters of Florida and along the Pacific coast of Mexico, where it is endangered. Threatened populations exist in all other areas where the species is known to occur.

The Sea Turtle Stranding and Salvage Network (STSSN) was formally established in 1980 to collect information regarding, and document strandings of marine turtles along the United States' Gulf of Mexico and Atlantic coastlines. The STSSN encompasses the coastal areas of an eighteen-state region from Maine through Texas, including portions of the U.S. Caribbean (NOAA 2005a). The STSSN reported that a total of nine green turtles were stranded in Massachusetts and Rhode Island waters between 1998 and 2001 (STSSN 2005) provides regional data, which is not available on a state-specific basis, and reports a total of 14 green turtles were stranded in New York, Massachusetts, Connecticut, and Rhode Island waters between 2002 and 2005 (STSSN 2005).

The Wellfleet Bay Wildlife Sanctuary, which is part of the Massachusetts Audubon Society (Mass Audubon), coordinates the Cape Cod Sea Turtle Stranding Network (CCSTSN). For the period of 2003 to 2004, CCSTSN reported five green turtle strandings on Cape Cod beaches, and one green turtle stranding on the shoreline of Chappaquiddick, Martha's Vineyard (MA

Audubon 2005b). Additionally, for the period of 1980 to 1997, the National Marine Fisheries Service (NMFS) has record of four green turtle strandings on the shorelines of Cape Cod Bay, and one stranding was reported in Nantucket Sound along the northern shoreline of Nantucket (NMFS unpublished data) (see Figure C-3). Of all the sea turtle strandings reported on Figure C-3, the green sea turtle occurred the least, supporting other evidence that it is the least likely to occur in the area, given its stronger preference for more tropical areas.

Sea turtle strandings in New England waters are most likely caused by cold stunning (Morreale et al., 1992; Matassa et al., 1994). As with most marine turtles, prolonged exposure of green turtles to water temperatures below 50 ^{o}F (10 ^{o}C) (Shaver 1990), may result in dormancy, shock, and death. In the cold water, the juveniles have a harder time maintaining an elevated body temperature and getting out of the bays and into warmer water than larger, adult turtles. In addition to cold stunning, the major sources of mortality of green turtles in United States waters are caused by anthropogenic activities which include the following: the incidental bycatch of turtles by various fisheries methods including shrimp trawls, gillnet fisheries and hook and line fishing; entanglement in ghost fishing gear and marine debris; entrapment in power plants' saltwater intake infrastructure; oil spills and the explosive removal of offshore oil platforms; and ingestion of marine pollutants including debris, pesticides, metals, and polychlorinated biphenyls (PCBs) (Thompson 1988; NOAA 2002; Seminoff 2004; NMFS & USFWS 1991).

Degradation of both nesting beach habitat and marine habitats also play a role in the decline of the green sea turtle. Nesting habitat degradation results from activities such as beach erosion, beachfront development; beach armoring and re-nourishment, beach raking, cleaning, and maintenance; the operation of recreational equipment and vehicles on the beach; sand extraction; the presence of exotic and invasive beach vegetation; and predation by raccoons, foxes, coyotes, ants, and ghost crabs. Habitat degradation in the marine environment results from dredging, increased contamination from coastal development, marina construction, and increased levels of recreational and commercial boat traffic. The presence of lights on or adjacent to nesting beaches can also alter the behavior of nesting adults and is potentially fatal to emerging hatchlings as they are attracted to light sources and drawn away from the water (USFWS 2002; NOAA 2002; Seminoff 2004; NMFS & USFWS 1991).

Green turtles mortality is also caused by the disease fibropapillomatosis, or the development of tumors on the turtles' skin, scales, scutes, eyes, oral cavities, and internal organs. The tumors interfere with the turtles' ability to swim, eat, breathe, see, and reproduce; and lead to the eventual death of the infected reptile (USFWS 2002; SCDNR undated; Seminoff 2004).

Figure C-4 presents incidental observations of sea turtles made during late summer while performing surveys of birds from an airplane. Given the challenges of identifying sea turtle species from a moving airplane, the majority of the sightings are not species specific. During the 2002-2004 study period, sea turtles where observed within the proposed action area of Nantucket Sound, some of which could have been green sea turtles. However, within the bird aerial survey study area, the greatest number of observations occurred in the deeper water areas east of the proposed action area.

1.3 Avian Species

1.3.1 Piping Plover

1.3.1.1 Species Description

The piping plover is a small, light-colored shorebird with distinctive orange legs, a black neck band, and black brow during the breeding season. It forages in the wrack line, at the water's edge along coastal and island beaches, in mudflats, and sandflats. Piping plovers' primary foods are invertebrates including worms, tiny crustaceans and mollusks, insects, other invertebrates, and their eggs (Haig and Elliot-Smith 2004).

The United States Fish and Wildlife Service (USFWS) recognizes three distinct breeding populations; the Great Lakes population is designated as endangered, and the Northern Plains and Atlantic coast populations are designated as threatened (USFWS 1996). The Great Lakes population breeds on the shorelines of Lake Superior, Lake Michigan, and Lake Huron (Haig and Elliot-Smith 2004). The Northern Plains population breeds along prairie rivers and reservoirs from southeastern Alberta, to Manitoba, southwestern Ontario south to Colorado, Kansas, Montana, North and South Dakota, Nebraska, and Iowa (Haig and Elliot-Smith 2004). Along the Atlantic Coast, piping plover nest on sandy beaches in New Brunswick, Prince Edward Island, Nova Scotia, and from southern Maine south to North Carolina, as well as the French Islands of Saint Pierre and Miquelon (Haig and Elliot-Smith 2004). The piping plover is divided into two subspecies based on geographic distribution, neck-band extent and brightness, and mitochondrial DNA (Haig and Elliot-Smith 2004). Ornithologists continue to debate the validity of subspecies separation; however, the American Ornithologists' Union (AOU) maintains recognition of the two subspecies. Nonetheless, existing evidence indicates that the Atlantic coast plovers form a distinct breeding population.

Piping plover winter on beaches and mudflats from North Carolina to the Yucatan, with large concentrations in Texas (Haig and Elliot-Smith 2004). Islands in the Caribbean also serve as important wintering habitat (USGS 2007). Band recovery results from birds banded during the breeding season indicate that most Atlantic Coast breeders winter along the southern Atlantic Coast, although some Massachusetts birds have been reported to winter in Texas (Haig and Elliot-Smith 2004). Evidence suggests that most of the Northern Plains population winters on the Gulf Coast, and most Great Lakes breeders winter south along the Atlantic (Haig and Elliot-Smith 2004). Winter studies of marked birds indicate that they generally return to the same wintering ground each year (Nichols and Baldasarre 1990, Haig and Elliot-Smith 2004, USFWS 1996).

Over the past fifty years, the historical breeding ranges of the piping plover populations have decreased, particularly in the Great Lakes range. Plovers no longer breed in Illinois, Ohio, Indiana, Pennsylvania, on the Gaspe Peninsula, or on the North Shore of Quebec (Haig and Elliot-Smith 2004). Threats to the Great Plains population include drought and unsuitable water management practices (Haig and Elliot-Smith 2004). Population decline in the interior of Canada and the U.S., as well as along the Atlantic, is partially attributed to such factors as: egg and chick mortality; high tides and frequent storms which result in the destruction of nests and

beach erosion; and loss of habitat due to human recreational activities and development (Haig and Elliot-Smith 2004).

Some information about piping plovers provided in this biological assessment may include that derived from observations of the Great Lakes and Northern Plains populations. Discussion, however, will focus primarily on the Atlantic Coast population.

1.3.1.2 Life History

1.3.1.2.1 Breeding Season

Arrival and Courtship

Plovers arrive at breeding sites in early spring; sightings suggest some individuals arrive in Massachusetts as early as March 15 (Haig and Elliot-Smith 2004). Spring arrival peaks in late April or early May (Haig and Elliot-Smith 2004, O'Brien et al., 2006). Piping plovers exhibit breeding site fidelity associated with areas of reproductive success, and will also use other locations in the vicinity if more available (Haig and Elliot-Smith 2004). Breeding site fidelity is lower in areas where habitat is unreliably suitable from year to year (Haig and Elliot-Smith 2004).

Pairs bond typically within a day or two of the birds' arrival at breeding locations. Piping plovers are primarily monogamous. Most birds retain the same mate throughout the breeding season but often change mates between years (Haig and Elliot-Smith 2004). At the beginning of courtship, males undertake flights and vocalizations over breeding territories to advertise their availability to females. These displays could occur at elevations as high as 33 ft (10 m), but are restricted to nesting beaches (Report No. 4.2.9-1). Frequency and duration of the display decreases once a female begins to lay eggs, although males occasionally perform displays during the rest of the breeding cycle (Cairns 1982). Unpaired males may continue to perform aerial displays until mid-summer, or until they disperse south. Nest-scraping rituals occur during courtship and just prior to copulation (Haig and Elliot-Smith 2004). A courting male will call and walk about his territory conducting a series of movements that demonstrate nest-scraping. Following nest-scraping, a male may stand in a scrape with a female nearby and perform several head-tilt displays accompanied by stomping (Cairns 1982). Females also dig scrapes, but less conspicuously than males. The pair may dig several scrapes during courtship but eventually one scrape will be chosen as the nest site. It typically takes five to ten days for a pair to settle down to one scrape, during which time a nest lining is added as part of courtship (Haig and Elliot-Smith 2004).

Nesting and Brood-rearing

Piping plover mainly create nests on open sand or gravel beaches that are sparsely vegetated (Haig and Elliot-Smith 2004). Nest sites are characterized by open sand, gravel, or shell-covered substrates, often close to small clumps of evenly distributed vegetation or near large objects (e.g., stones, logs), above the high water mark (Cairns 1982, Haig and Elliot-Smith 2004). Sites close to ephemeral pools, salt-ponds, or bays may be associated with increased fledging success (Goldin and Regosin 1997, USFWS 1996). The typical clutch size is four eggs, however late initiated nests may only have three eggs. Most first clutches in breeding locations in Canada (for

both Northern Plains and Atlantic Coast populations) are initiated during the first two weeks of May (Haig and Elliot-Smith 2004). Copulation generally continues until the last egg is laid. Males are very attentive and guard their mates prior to clutch completion (Haig and Elliot-Smith 2004). The average egg laying time for first clutches is six days (Wilcox 1959, Haig and Elliot Smith 2004).

Females may lay several clutches if nests are predated, inundated by the tide, or abandoned due to a disturbance. However, piping plovers will usually raise and fledge one brood per year; very rare cases have been reported where two clutches were brooded (Haig and Elliot-Smith 2004). Both sexes incubate the clutch. Eggs, therefore, are usually covered, unless a potential predator approaches, and the attending adult leaves the nest to lure the predator away by feigning an injury (Cairns 1982). It is also possible that, in warm weather, eggs may be left exposed for several minutes while the adult forages close by (Haig and Elliot-Smith 2004). Males exhibit intense territory and mate defense during incubation, especially if an unpaired male approaches.

A clutch is incubated, on average, for a period of 28 days. Peak hatching for most North American breeders occurs in the first two weeks of June (Haig and Elliot-Smith 2004). The downy chicks are precocial and depart the nest to forage within hours of hatching. Chicks remain in the vicinity of their parents while foraging and are periodically brooded by either adult. Chicks younger than 20 days spend the majority of their time feeding (Haig and Elliot-Smith 2004) and typically triple their weight within two weeks of hatching. If chicks do not achieve 60 percent of their tripled birth weight within 12 days of hatching, they are unlikely to survive (Cairns 1977 *as cited by* USFWS 1996). On ocean beaches, young chicks forage in wrack lines. Adults will vocalize the presence of a potential predator, and chicks will either freeze-up or take cover in vegetation. Chicks will begin to visit the inter-tidal zone to forage as they approach fledgling status. Foraging occurs throughout the day, and at night. Brooding adults will alternate guarding while the chicks forage. When both parents are tending a brood, one may lead the chicks away from a potential predator while the other adult feigns an injury (Haig and Elliot-Smith 2004).

Both parents brood and tend to chicks after hatching; however, females may desert broods after 5 to 17 days (Haig and Elliot-Smith 2004). Generally, males stay with the young until they can fly. The chicks reach fledgling status after 30-35 days; yet, some birds are known to fledge as early as 27 days after hatching (Haig and Elliot-Smith 2004). Telemetry data gathered in North Dakota suggests that juveniles may travel more than 32 mi (50 km) within a few days of being able to fly (Haig and Elliot-Smith 2004).

Productivity data (the mean number of chicks fledged per breeding pair per year) is variable among breeding populations and locations. The destruction of nests by storms and high tides, as well as predation of eggs and chicks, has limited plover productivity along the Atlantic Coast. Table C-2 shows the preliminary productivity estimates for Atlantic U.S. and Canada breeding pairs from 2003 to 2006.

Table C-2. Preliminary Productivity Estimates (# of chicks/pair) for the Atlantic Coast 2003-2006 (USFWS 2006)				
	2003	**2004**	**2005**	**2006**
U.S.	1.24 (1,287)	1.40 (1,379)	1.20 (1,383)	1.26 (1,368)
Canada	1.62 (219)	1.93 (223)	1.82 (192)	1.80 (222)
Parentheses indicate the # of breeding pairs on which productivity estimates are based.				

1.3.1.2.2 Migration

Information regarding Atlantic Coast piping plover migration routes and stop-over sites are based on available observations at breeding and stopover locations. Although detailed information about the exact migration routes for the population is incomplete (USFWS 1996), general migration patterns are available. Northern interior breeding birds leave breeding locations as early as late June or early July, occasionally earlier if adverse weather destroys initial nests (Haig and Elliot-Smith 2004). Plovers may depart for fall migration as late as September if they have late-hatching nests (Haig and Elliot-Smith 2004). In Massachusetts, birds depart breeding sites by late-August (Haig and Elliot-Smith, O'Brien et al., 2006). Peak numbers of adults pass through the mid-Atlantic and Gulf coasts from mid-July to mid-August; peak numbers of juveniles pass through New Jersey in August, through Virginia in September and October; and through Texas in October and early November; some late moving juveniles will remain along the mid-Atlantic into November or December (O'Brien et al., 2006). Generally, females will leave first, followed by unpaired males, then males with fledglings, then unaccompanied juveniles (Haig and Elliot-Smith 2004, USFWS 1996).

Migration stop-overs occur largely at beaches that are predominantly mudflat. When moving south in the fall, small groups of Atlantic birds are believed to follow the coastline and may stop at several places before reaching their wintering destination (Haig and Elliot-Smith 2004, USFWS 1996). Migrants along the Atlantic Coast are believed to follow a 'narrow' path along the coast; observations of plovers away from coastal beaches (inland or offshore) during migration are rare (USFWS 1996).

There have been observations of concentrations of what were assumed to be pre-migratory staging plovers at breeding locations along the Atlantic Coast (USFWS 1996). Some mid-coast sites experience larger numbers of plovers during fall migration than during the breeding season or the winter, suggesting that at least some Atlantic Coast birds use stop-over locations (Haig and Elliot Smith 2004). Flocks as large as 100 birds have been observed at Cape Lookout, North Carolina (Haig and Elliot-Smith 2004). Other locations in North Carolina that have experienced large flocks of migrants include Oregon Inlet, Ocracoke Inlet/Portsmouth Flats, and New Drum Inlet, within the Cape Hatteras National Seashore (USFWS 1996). In Massachusetts, South Beach, Chatham, Monomoy Island, and other beaches have been identified as fall stop-over areas that experience numbers of migrating plovers (USFWS 1996). Eel Point on Nantucket has been identified as a staging area for pre-breeding piping plover (USFWS 1996). The birds congregate

at stop-over sites in large groups of up to 100 birds; but are believed to travel in small flocks typically consisting of three to six birds, but sometimes up to as many as 15 birds (Haig and Elliot-Smith 2004). Stop-over observations are not as well documented in the spring as in the fall but northward movements are believed to be similar to fall movements (Haig and Elliot-Smith 2004). Arrival to New England and the Maritime Provinces occurs between mid-April and mid-May (Haig and Elliot-Smith 2004).

Most interior Canada and U.S. breeding birds are believed to make non-stop migrations overland to wintering grounds on the Gulf of Mexico or the Atlantic Coast (Haig and Elliot-Smith 2004, Haig and Plissner 1993). Most interior birds are anticipated to fly south-southeast in the fall; however, very rare observations suggest that some inland birds may fly east to reach the Atlantic Coast, then fly south to the wintering grounds (Haig and Elliot-Smith 2004). Inland birds' spring movements are also expected to be similar to fall movements (Haig and Elliot-Smith 2004).

Sightings away from inland or offshore outer beaches are rare (USFWS 1996). Observations of color-marked birds indicate that some Atlantic Coast birds cross over to Gulf Coast wintering grounds; however, the actual routes are unknown (O'Brien et al., 2006, USFWS 1996). Sightings of plovers at islands in the Caribbean as far as Bermuda and the West Indies indicate that piping plover do undertake long-distance offshore movements (USFWS 1996). To develop a more complete understanding of piping plover migration, additional information would be needed about piping plover migration patterns in a variety of weather conditions during both day and night, including: flight heights; flight patterns (e.g., coastal, off-shore, well off-shore); flight directions; and additional important stop-over habitats (USFWS 1996).

1.3.1.2.3 Survivorship

Plovers may breed during their first year and all birds are thought to have bred by their fourth year (Calvert et al., 2006). Evidence suggests that many first year birds return to the vicinity of their natal sites for the breeding season (Haig and Elliot-Smith 2004).

Plovers have been known to live as long as 5 to 11 years of age. Sources of mortality for eggs, chicks, and adults include predation by raccoon (*Procyon lotor*), striped skunk (*Mephitis mephitis*), Virginia opossum (*Didelphis virginiana*), red fox (*Vulpes vulpes*), mink (*Mustela vison*), ermine (*M. ermina*), coyote (*Canis latrans*), domestic dog (*C. familiaris*), domestic cat (*Felis domesticus*), old-world rat (*Rattus* sp.), American crow (*Corvus brachyrhynchos*), Northern raven (*Corvus corax*), and several species of gull (*Larus* spp.) (Haig and Elliott-Smith 2004). Migrating peregrine falcon (*Falco peregrinus*) are occasional predators of piping plover (USFWS 1996). On the wintering grounds, hurricanes, extreme cold weather, and oil spills are potential sources of mortality (USFWS 1996). Estimated survival rates for adults range from 67 to 83 percent, and 41 to 48 percent for chicks (USFWS 1996).

1.3.1.3 Population Dynamics

There are no comprehensive estimates of historical population numbers prior to 1980 (USFWS 1996). Over-hunting around the turn of the century was a major factor contributing to

the decline of the historical piping plover population (USFWS 1996). Since 1918, the take of piping plovers and other migratory birds has been prohibited by the Migratory Bird Treaty Act. The more recent decline of the Atlantic Coast piping plover population is attributed to habitat loss and degradation, disturbances caused by humans and pets, and increased predator populations in coastal environments (USFWS 1996). Excessive disturbance may cause the parents to flee the nest, exposing eggs or chicks to the hot sun or predators. Vehicle traffic can destroy unprotected nests and can crush chicks (USFWS 1996). High disturbance levels around nest sites can also result in the abandonment of nests, and ultimately, decreased breeding success. Causing parents or juveniles to flush while foraging may stress juveniles enough to negatively influence critical growth and development (see Effects of the Action, Disturbance, for discussion of disturbance distances related to habitat loss).

Along the Atlantic Coast, commercial, residential, and recreational development have decreased the amount of coastal habitat available for piping plovers. Development such as seawalls and jetties, and the manipulation of beach dunes, has resulted in beach erosion and the degradation of nesting habitat (USFWS 1996). Developments near beaches provide suitable habitat for predators such as raccoons, skunks, and foxes. Domestic and feral cats, often associated with developed beaches, have been identified as a significant source of mortality for piping plover eggs and chicks (Winter and Wallace 2006, Melvin et al., 1991 *as cited by* Haig and Elliot-Smith 2004). Extreme tides and storm tides can inundate nests; however, plovers will re-nest after destruction of nests early in the breeding season. Unusual stochastic events, such as hurricanes, can impact hundreds of individuals. Storms can also, over a period of time, positively affect local piping plover populations by leveling dunes and creating suitable nesting habitat (Wilcox 1959, USFWS 1996). Beach development and stabilization activities, dredging, recreational activities, and pollution are factors that are impacting the plover population on wintering grounds (USFWS 1996). There are additional unknown sources of mortality experienced during migration or on the wintering grounds (Root et al., 1992, Calvert et al., 2006).

During the past 50 years, breeding piping plovers have disappeared from several places in the Midwest and a few places on the Atlantic Coast, specifically the Gaspé Peninsula and the North Shore of Quebec (Haig and Elliot-Smith 2004). The Atlantic Coast population has gradually increased, particularly in the central part of the range (New England and New York), mainly due to management practices. Conversely, numbers have declined in eastern Canada and along the southern edge of the range in North Carolina. Numbers of piping plovers continue to increase in the Great Lakes, but this population segment represents less than two percent of the species and most historic sites have not been re-colonized, except those in Wisconsin (Haig and Elliott-Smith 2004).

1.3.1.4 Status and Distribution

On January 10, 1986, the piping plover was listed as endangered and threatened under provisions of the Endangered Species Act (50 FR 50726 50734). USFWS designated the Great Lakes population as endangered and the Atlantic Coast and Northern Great Plains populations as threatened. The 1991 international census documented 5,482 total piping plover (Plissner 1993). The 2001 total population estimate was 5,945 total birds (Haig and Elliot-Smith 2004).

Preliminary results from the 2006 international census conducted on the breeding grounds resulted in approximately 3,800 birds counted in the U.S., Mexico, and the Caribbean on the wintering grounds (USGS 2007).

The Atlantic Coast population nests along beaches in New Brunswick, Prince Edward Island, Nova Scotia, Quebec, Newfoundland, Saint Pierre and Miquelon (France), southern Maine, New Hampshire, Rhode Island, Massachusetts, Connecticut, New York, New Jersey, Delaware, Maryland, Virginia, and North Carolina (Haig and Elliott-Smith 2004).

Since being listed as federally threatened in 1986, the Atlantic Coast population has increased from 800 pairs to approximately 1,350 pairs in 1995 (however, increases from 1986 to 1989 were attributed to increases in survey effort) (USFWS 1996). The current estimated number of Atlantic Coast breeding pairs for 2006 is 1,743 (Table C-3).

Table C-3.
Estimates of the number of Atlantic Coast Breeding Pairs 1996-2006 (USFWS 2006)

	1996	1997	1998	1999	2000	2001	2002	2003	2004*	2005*	2006*
U.S.	1,162	1,187	1,168	1,156	1,207	1,280	1,415	1,420	1,423	1,415	1,491
Canada	186	197[a]	212	240	231	245	275	256	237[b]	217	252
Atlantic Coast	1,384	1,384	1,380	1,396	1,438	1,525	1,690	1,676	1,660	1,632	1,743

*Numbers represent preliminary estimates.
[a] Used 1996 Newfoundland estimate of 11 pairs because 1997 estimate unavailable.
[b] Two sites totaling 10 pairs in 2003 not included in estimate.

The Atlantic Canada population has generally declined in recent years despite relatively high productivity (Table C-2, Table C-3, Figure C-5). The Canadian Wildlife Service reports that banding studies are showing relatively low rates of returns of second-year birds, even following years of high productivity, suggesting mortality outside of the breeding season (USFWS 1996). The Atlantic U.S. population has generally been increasing over the past 10 years, likely due to intensive management practices (Figure C-5).

The Atlantic Coast recovery goal is 2,000 pairs maintained over a period of at least 5 years, including 1,600 breeding pairs for the U.S. (New England 625 breeding pairs; New York and New Jersey 575 breeding pairs; and Delaware, Maryland, Virginia, and North Carolina 400 breeding pairs), and 400 breeding pairs for Atlantic Canada (USFWS 1996). Another recovery objective is a five-year average productivity of 1.5 fledged chicks per pair in each of the four recovery units (USFWS 1996). Atlantic Canada has exceeded the average productivity goal; however, the number of breeding pairs does not reflect an increase. Overall, the Atlantic Coast population is approaching its recovery goal mainly due to intense annual management activities (USFWS 1996).

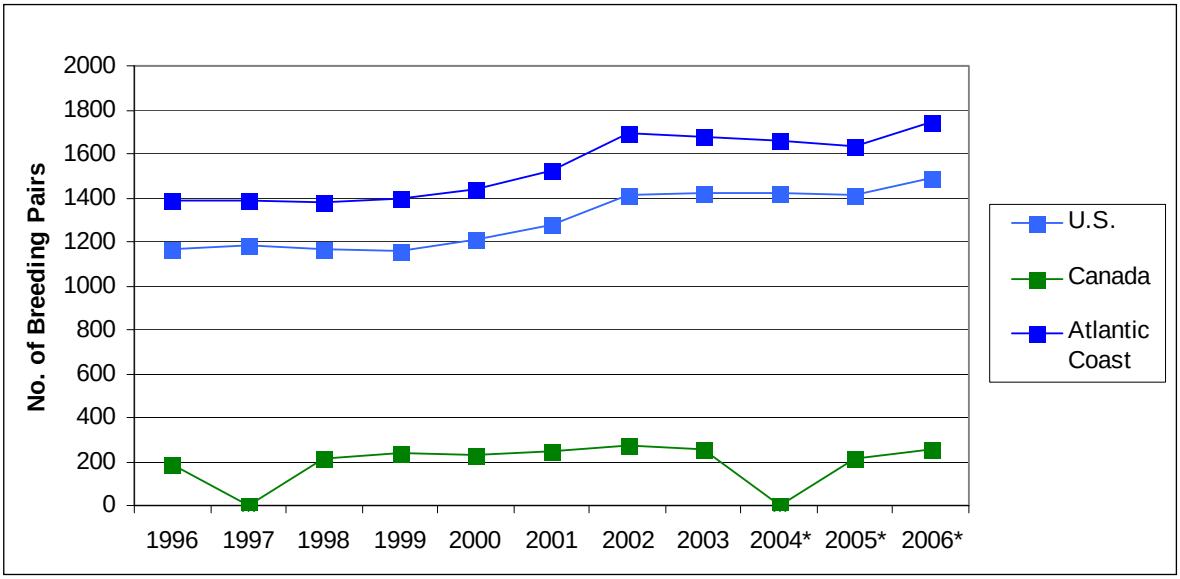

Figure C-5.
Estimates of the number of Atlantic Coast breeding pairs 1996-2006 (USFWS2006)
***preliminary estimates**

As part of the management practices being implemented on behalf of the plover, on June 12, 2001 a proposed rule for the designation of critical habitat was published for the Northern Great Plains breeding population. On May 3, 2001, USFWS designated critical habitat for endangered Great Lakes birds. The final rule designating critical habitat for the wintering population of the piping plover was published in the Federal Register on July 10, 2001. There have been no critical habitat designations established for Atlantic Coast breeding areas; however, protection of wintering grounds provide benefits to the Atlantic Coast breeding population. The USFWS designated 137 areas along the coasts of North Carolina, South Carolina, Georgia, Florida, Alabama, Mississippi, Louisiana, and Texas as critical habitat for wintering populations. This includes approximately 1,798.3 mi (2,891.7 km) of mapped shoreline and approximately 165,211 acres (ac) (66,881 hectares (ha)) of mapped area along the Gulf and Atlantic coasts and along interior bays, inlets, and lagoons (USFWS 2001).

1.3.2 Roseate Tern

1.3.2.1 Species Description

Roseate terns (*Sterna dougallii*) are distinguished from similar tern species by their overall pale color and, during the breeding season, their long-forked tail 5.5 to 9 inches (14 to 23 cm) and the rosy tinge to their belly. Roseate terns are medium sized terns13 to 16 inches (33 to 41 cm) in length) with a swift wing-beat during flight. During the breeding season, the base of the otherwise black bill and the legs are orange. The black cap is replaced by a black face mask and white forehead, the rosy tinge fades, and the tail is shorter during adult non-breeding stages. Roseate terns breed in colonies along marine coasts and migrate long distances during the spring and fall (Gochfeld et al., 1998). As of 1987 when the species became listed as Endangered in the U.S., the number of roseate tern colony sites had decreased from 30 historical nesting locations

to 4 major colony sites (USFWS 1987). The narrowing of their breeding range is believed to be influenced by an increase of gull predation, loss of habitat, and the species' restrictive habitat requirements. Currently, there are seven major breeding colonies in the Northeast and Atlantic Canada including The Brothers, Nova Scotia; Eastern Egg Rock and Stratton Island, Maine; Bird and Ram Island, Massachusetts; Falkner Island, Connecticut; and Great Gull Island, New York.

1.3.2.2 Life History

1.3.2.2.1 Foraging

Roseate terns forage by plunge diving at heights ranging from 3.3 to 39.4 ft (1 to 12 m) for small fish, primarily sand eels (*Ammodytes americanus*), and sometimes take insects, small crustaceans, and squid (Gochfeld et al., 1998). Roseate terns forage in mixed flocks with other terns and gulls over shoals, shallow water, and schools of predatory fish where bait fish are pushed to the surface (Heinemann 1992). They seldom forage over diving cormorants that chase bait fish to the surface and have been known to kleptoparasitize fish from other terns at breeding colonies (Heinemann 1992). During the breeding season, roseate terns will travel as far as 16 to 19 miles (25 to 30 km) from the colony to access foraging habitat and are believed to regularly return to productive foraging locations (Heinemann 1992).

1.3.2.2.2 Pre-breeding Period

The northeastern population of roseate terns travels long-distances in the spring and fall, mainly over the ocean (Gochfeld et al., 1998). During spring migration, roseate terns travel in large flocks, often mixed with common terns (*S. hirundo*). Northeast birds arrive and aggregate in Nantucket Sound before dispersing to breeding locations in the Sound, Buzzards Bay, Maine, and Canada. An estimated 10-12 percent of the roseate population travels north of Nantucket Sound, while 88-90 percent move south and west to locations in Buzzards Bay, Long Island Sound, and to Falkner Island in Connecticut (Spendelow et al., unpublished). Pair bonds are formed just before arrival at breeding colonies or during territory establishment (Gochfeld et al., 1998). Bonds continue through the breeding season and can end after a single season or last for multiple seasons (Gochfeld et al., 1998). Roseate terns return to breeding grounds in the Northeast and Atlantic Canada from late-April to mid-May and can occupy the colony site for up to 3 weeks before egg laying (Gochfeld et al., 1998).

1.3.2.2.3 Nesting Period

Roseate terns prefer nest sites on rocky island beaches, barrier beaches, or salt marsh islands that provide some vegetative or debris cover. They require locations that are protected from human disturbance and predation from gulls or mammals. Roseate terns share nesting colonies with common terns, benefiting from the protection of their more defensive behavior (Gochfeld et al., 1998). Roseate terns exhibit strong colony site fidelity though there is some individual movement among colony sites of the northeastern meta-population (the Northeast and Atlantic Canada) (Lebreton et al., 2003). Data suggest that 5 percent of breeding individuals move between northeast colonies among years (Nisbet et al., 1999).

Observations made in the vicinity of Bird Island indicate that, during pre-incubation, roseate terns spend 5 to 8 percent of daylight time in territorial defense, 20 to 30 percent of the time in courtship behavior at the territory, 20 percent of the time resting on the territory, and for the remainder of daytime, males forage to courtship-feed females (Gochfeld et al., 1998). Females usually remain at the breeding territory but may join the male at feeding areas where courtship feeding can also occur (Gochfeld et al., 1998). Roseate terns undertake elaborate courtship flights in the vicinity of breeding grounds that involve multiple individuals and range from heights of 98 to 980 ft (30 to 300 m) (Gochfeld et al., 1998). Most courtship display behavior and copulation takes place on the edge of the colony and rarely at nest sites, however, it also occurs at resting locations far from the colony (Gochfeld et al., 1998).

The timing of initiation of egg-laying has been found to be largely dependent on the availability of prey (Heinemann 1992). The availability of prey influences the ability of females to build up their nutritional reserves prior to egg-laying (Safina and Gochfeld 1988). Studies suggest that declines in food availability are associated with the presence of schools of predatory bluefish (*Pomatomus saltatrix*) (Safina et al., 1988).

Eggs are laid in nest 'scrapes' in substrates of sand, shell, or rock. Spacing of nests among breeding pairs at colony sites is dependent on the vegetative and debris cover available and nests have been reported as close as 11.8 to 19.7 inches (30 to 50 cm) (Gochfeld et al., 1998). Clutch sizes range from one to four eggs (usually 1 or 2 eggs) and are laid over a period of two to four days.

Both male and females share incubating and foraging responsibilities and rear a single brood per season. Due to the species' unbalanced sex ratio, which is 45 percent male (Report No. 5.3.2-5), female-female pair bonds are estimated to account for about 7.8 to 12 percent of nests at Bird and Falkner Islands (Nisbet et al., 1999, Szczys et al., 2005). Females that share incubation responsibility with other females mate with already paired males and usually produce larger clutches (3 to 4 eggs) (Gochfeld et al., 1998). However, infertility and incubation problems resulting from the large clutches of female-female pairs attribute to the low hatching success of these pairs (40 to 50 percent hatching success). There is a 98 percent hatching success among male-female pairs (Nisbet et al., 1999).

A second clutch, usually of only one egg, is often initiated by breeding pairs if eggs or chicks are lost. After an incubation period of about 23 days, the semi-precocial chicks hatch and require 22 to 30 days before fledgling. Nesting success is seasonally variable and is generally lower for later clutches (nests initiated in late June and July). Birds that nest later typically produce fewer fledglings. Additionally, the first chicks to hatch from a clutch have better survival rates (97 percent) than the second chicks to hatch (30 to 60 percent) (Nisbet et al., 1999). The mean annual productivity (fledglings per breeding pair) for the Northeast population (productivity data is unavailable for breeding sites in Canada) from 1998 to 2005 is 1.056 (Report No. 5.3.2-5). Figure C-6 shows the variable productivity trends of the major North Atlantic colonies from 1999 to 2006. Productivity is limited by chick mortality, which is usually caused by starvation or predation. Chicks are most vulnerable to mortality for five days after hatching. Productivity may also be limited by the populations' unbalanced sex ratio.

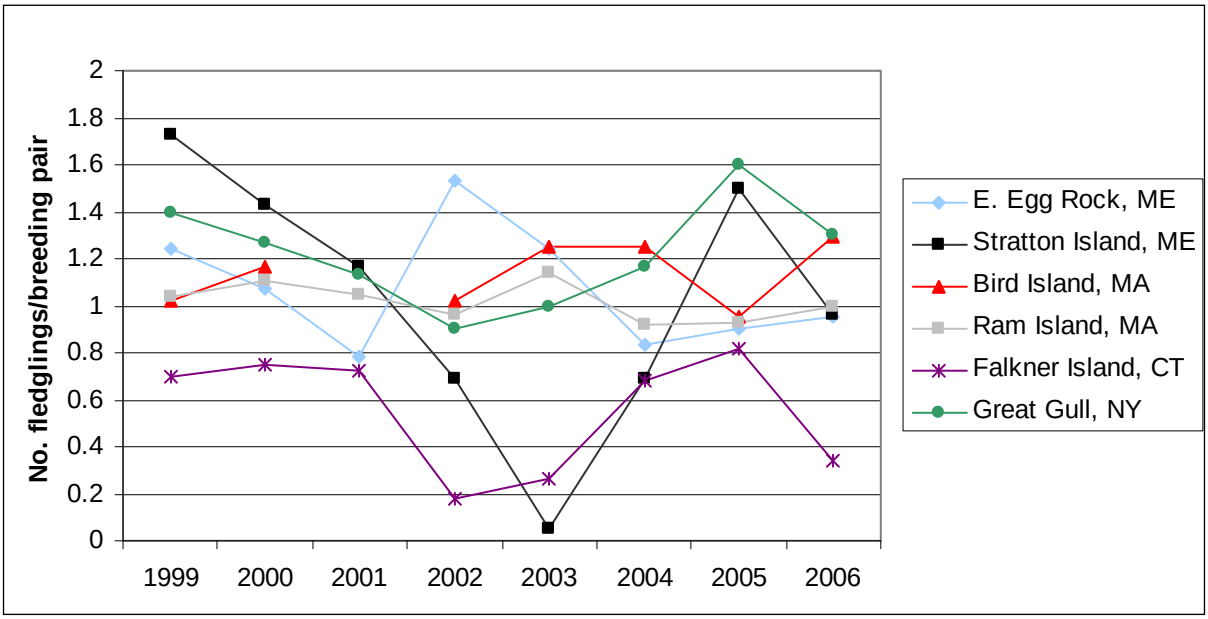

Figure C-6.
Roseate tern productivity at the major Northeast colonies 1999-2006 (Spendelow et al., **unpub.)**

After the breeding season, the fledglings leave the colony with their parents and gather with other roseate terns at pre-migratory staging areas where they are taught to forage and are fed and attended by their parents at least until migration (Gochfeld et al., 1998; Shealer and Kress 1994). A study investigating the efficiency of roseate tern fledgling foraging success determined that by the end of the staging period, roseate fledglings are not likely to be skilled enough as foragers to survive migration without parental care (Watson and Hatch 1999). Therefore, parental care may extend through migration so that juveniles can learn advanced foraging techniques (Watson and Hatch 1999). During fall migration, the terns travel in groups from post-breeding staging areas to the West Indies and South America (Gochfeld et al., 1998).

1.3.2.2.4 Survivorship

Most roseate terns remain at the wintering grounds during their first summer. Some migrate north at the age of two and occur as non-breeders at breeding colonies (Nisbet et al., 1999). Most birds begin to breed at an age of three years, though few begin as early as two years and some wait until after four years of age. Breeding usually occurs every year after initiation, however, it is believed that some pairs do not breed during seasons with low food availability (Gochfeld et al., 1998). Additionally, some individuals (mostly females), that were known to nest in previous years, were observed at nesting areas during seasons when they did not breed (Nisbet et al., 1999).

Studies conducted at four colonies in Connecticut and New York suggest that annual adult survival rates range from 74 to 84 percent. The survival rate from fledgling to first breeding is estimated at 20 percent (Gochfeld et al., 1998). Adult mortality is uncommon during the breeding season; however great horned owl (*Bubo virginianus*), red fox (*Vulpes vulpes*), and

Norway rat (*Rattus norvegicus*) have been known to take adults (Gochfeld et al., 1998). Human predation on wintering grounds is believed to be a significant source of adult mortality (Gochfeld et al., 1998). Males are believed to have a lower survival rate than females (Nichols et al., 2004 *as cited by* Spendelow et al., unpublished). Roseate terns often live over 20 years (Gochfeld et al., 1998). The oldest banded bird on record, banded as a chick in Massachusetts and recaptured in Brazil, was 25.6 years old (Gochfeld et al., 1998).

1.3.2.3 Population Dynamics

In the late 1890s, due to the millinery trade, the northeastern population (Northeast and Atlantic Canada) began to decline from an estimated 8,500 breeding pairs to 2,000 breeding pairs. After the passing of the Migratory Bird Treaty Act in 1918, the population quickly recovered to roughly 8,500 breeding pairs in 1930, remaining stable until the 1950s. After 1978, 90 percent of the Northeast population was limited to four breeding colonies: Bird Island and Monomoy Island in Massachusetts; Great Gull Island in New York; and Falkner Island in Connecticut. The northeastern population of roseate terns formerly bred from Sable Island, Nova Scotia to Virginia, however, the birds no longer breed south of Long Island, New York, at former sites in Virginia, Maryland, or New Jersey. By 1979, the population had dropped to 2,500 breeding pairs.

Due to management activities after listing of the species in 1987, the population gradually increased until about 2000, despite a dramatic decline between 1991 and 1992, the year of Hurricane Bob (Spendelow et al., unpublished). Since 2000 the breeding population has generally been declining (Figure C-8). The northeastern and Atlantic Canada population is currently at 3,500-3,600 breeding pairs (Spendelow et al., unpublished) among 7 major colonies and a few small colonies.

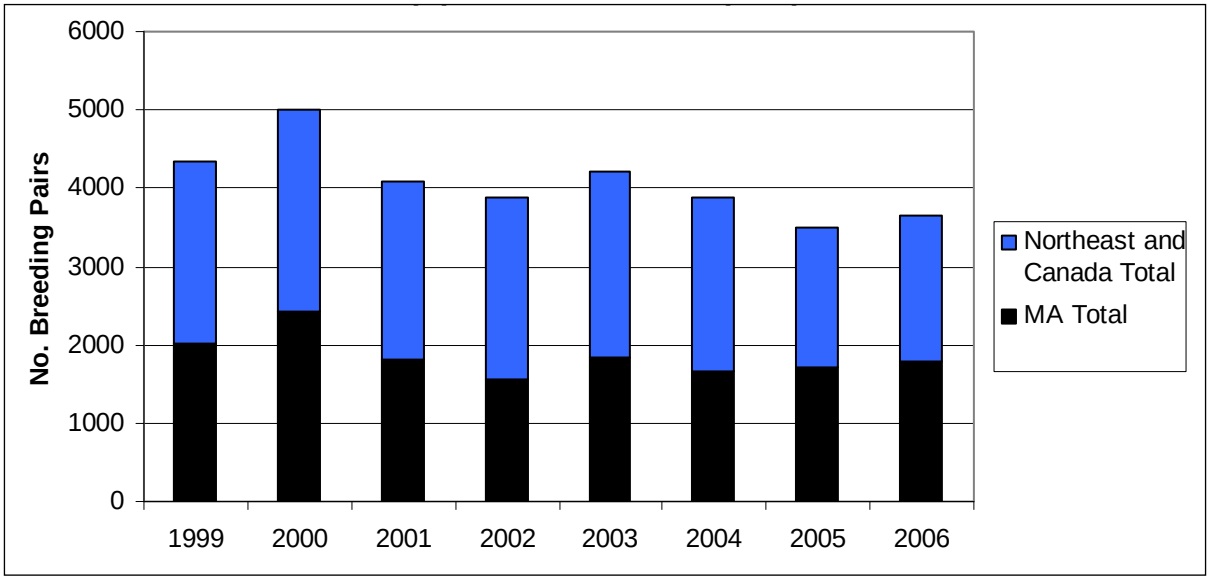

Figure C-7.
Reported totals of roseate tern breeding pairs 1999-2006 – MA and Northeast and Canada (Spendelow et al., unpub.)

1.3.2.4 Status and Distribution

Roseate terns were listed as Endangered by the U.S. Fish and Wildlife Service (USFWS) in the U.S. (Atlantic Coast south to Virginia) (USFWS 1987). Roseate terns that occur in Canada and Bermuda are part of this population and their status is also monitored by the USFWS. The Canadian population (Newfoundland, Nova Scotia, Quebec) of roseate terns was designated as Threatened by the Committee on the Status of Endangered Wildlife in Canada in 1985. The Western Hemisphere population, including birds that occur in Bermuda, the southern U.S. Atlantic Coast, Puerto Rico, and the Virgin Islands, are listed as Threatened.

Factors contributing to the generally declining population of roseate terns include loss of suitable nesting habitat caused by human coastal development and beach erosion, gull predation and competition at colony sites, low productivity due to decreases in food availability and possibly the female biased sex ratio, as well as adult mortality at wintering grounds (Gochfeld et al., 1998). The roseate tern's specific breeding habitat requirements, which include vegetative or debris cover and a lack of competitors or predators, make it difficult for the birds to establish colonies at other locations.

The northeastern roseate tern population (numbers have been derived from the past 8 years of data as reported by Spendelow et al., unpublished) currently consists of approximately 70 to 130 pairs that nest on islands off of Nova Scotia, particularly The Brothers Island, as well as Pointe de l'Est, Quebec; approximately 250 to 280 pairs that breed in Maine on ten different islands; roughly 25-70 pairs that breed on White and Seavey Islands in New Hampshire; approximately 1,500 to 1,800 breeding pairs in Massachusetts (99 percent of which breed in Buzzards Bay primarily on Bird and Ram Islands and occasionally on Penikese and Nashewena Islands, representing 45 to 50 percent of the North American population) and other small colonies located on Minimoy and South Monomoy Islands in Nantucket Sound; approximately 40 to 110 pairs that breed on Falkner Island in Connecticut; and 1,430-1,850 pairs that nest on Great Gull and Cartwright Islands in New York.

2.0 SPECIES PRESENCE WITHIN PROPOSED ACTION AREA

2.1 Status of the Species within the Action Area

2.1.1 Whales

The three species of whales that have the potential to occur in the area of proposed activities are highly migratory, with seasonal distribution in New England waters and a limited potential to occur year-round. Further, the potential for these whales to occur in Nantucket Sound is lessened by the limited presence of adequate food sources. Most whales are found in areas where their primary food source can be easily located. The bathymetric and oceanographic features identified above that favor dense aggregations of whale prey species are not developed in Nantucket Sound to the extent that they are farther north around Stellwagen Bank, Jeffreys Ledge, Browns and Bacaro Banks, and in the Great South Channel (Kenney and Winn 1986). Therefore, the preferred foods of whales and the whales themselves occur in Nantucket Sound with far less abundance and frequency than in high-use areas farther north.

2.1.1.1 Humpback Whale

Humpback whales are present in New England waters in greatest abundances between June and September, with the potential to occur in Massachusetts waters from mid-March to the end of November (Payne and Heinemann 1990; Sadove and Cardinale 1993; NOAA Fisheries 2005a). All age classes, including mother/calf pairs, are present during the summer. Smaller numbers, nearly exclusively solitary juveniles, frequently are observed in December and January.

Although the primary feeding grounds for humpback whales are located further offshore from Nantucket Sound, very few whales are sighted within the Sound itself. According to the Center for Coastal Studies (CCS) (CCS, 2006 personal communication), humpback whales may pass through Nantucket Sound occasionally, but they are "simply not forming aggregations there."

2.1.1.2 Fin Whale

Fin whales are the most common of the large whales in the temperate waters of the western North Atlantic, and may be present in Massachusetts waters mid- March to the end of November (Waring et al., 2006; NOAA Fisheries 2005a). New England waters provide important feeding grounds for fin whales, particularly in Jeffreys Ledge, Stellwagen Bank, and Cape Cod Bay (CeTAP 1982). Very few whales are found in Nantucket Sound, even though the Stellwagen Bank feeding grounds are in adjacent offshore waters. Similar to humpback whales, Nantucket Sound does not support dense aggregations of fin whale prey species as other areas. Therefore, the preferred foods of fin whales and the whales themselves occur in Nantucket Sound with far less abundance and frequency than in high-use areas farther north.

2.1.1.3 North Atlantic Right Whale

Right whales are a migratory species that may be found in New England waters between February and May, with peak abundance in late March (NMFS 2005). The potential for the presence of right whales in Massachusetts waters exists from early December to the end of June (NOAA 2005a). Feeding, nursing and mating behavior has been observed in Cape Cod Bay, and the Great South Channel provides important feeding and nursery grounds with peak abundances in spring (Schevill et al., 1986; Hamilton and Mayo 1990; Kraus and Kenney 1991; Marx and Mayo 1992). As such, Cape Cod Bay and the Great South Channel have been designated Critical Habitat for the recovery of the North Atlantic right whale.

Although important seasonal feeding and nursery grounds for right whales are located further offshore from Nantucket Sound in the Great South Channel and Stellwagen Bank, very few whales have been sighted in Nantucket Sound. Nantucket Sound is too shallow and not productive enough in terms of copepods to support right whales (Kraus 2006). No right whale surveys have targeted Nantucket Sound, though many ship-based and aerial surveys for great whales (including right whales) in North Atlantic waters pass over or through Nantucket Sound, including the CeTAP and POP surveys. Neither the North Atlantic Right Whale Consortium database nor NMFS data contain any records of right whale sightings in Nantucket Sound (Kenney 2002, personal communication; NMFS-NEFSC 2002, unpublished data).

2.1.2 Sea Turtles

The four species of sea turtles that occur in the action area are highly migratory, and no individual members of any of the species are likely to be year-round residents of the action area. Incidental observations made during 2002 to 2004 pre-migratory staging season survey of Nantucket Sound showed 115 sea turtles, of which only 14 were found within the proposed action area (MA Audobon 2005a).

2.1.2.1 Kemp's Ridley Sea Turtle

The Kemp's ridley sea turtle is found mainly in the Gulf of Mexico, but juveniles may occur during the summer along the Atlantic seaboard from Florida to Long Island Sound, with some individuals occasionally visiting Cape Cod Bay, Massachusetts Bay and the Gulf of Maine (Lasell 1980; Hildebrand 1982). Juveniles may be found in northern waters from late May to mid-September primarily in shallow nearshore waters (Keinath et al., 1987; Musick and Limpus 1997). Most ridleys that remain in northern waters in the beginning of November are cold-stunned; however some may hibernate over winter in the nearshore sediments (Carminati et al., 1994).

2.1.2.2 Leatherback Sea Turtle

The seasonal distribution of leatherback sea turtles in the North Atlantic waters range from Cape Sable, Nova Scotia to Puerto Rico and the U.S. Virgin Islands. The waters of New England and Long Island Sound support the largest populations on the Atlantic coast during the summer and early fall (Lazell 1980; Prescott 1988; Shoop and Kenney 1992). Leatherbacks

have the potential to occur in Massachusetts waters from June through November in coastal and outer continental shelf waters (NOAA Fisheries 2005a). During summer months, leatherbacks move into fairly shallow coastal waters following prey, while during the fall they move offshore and begin their migration south (Payne et al., 1984). Incidental sea turtle observation data obtained by MA Audubon during 2002 to 2004 avian aerial and boat surveys identified 55 leatherbacks out of 115 total sea turtles observed, of which only 4 of the 55 turtles identified as leatherbacks were observed within the proposed action area.

2.1.2.3 Loggerhead Sea Turtle

Loggerhead sea turtles occur in the Northeast waters from May 1 through November 15, with higher abundances in the waters of f New York and the Mid-Atlantic States in the spring and summer (NOAA 2005a). A small number of individuals that may reach the waters of New England consist mainly of juveniles (NOAA 2005a). The presence of loggerhead sea turtles in New England waters has only been documented through incidental observations, strandings, entanglements and mariner reports, and the actual occurrence of loggerhead sea turtles in Nantucket Sound is expected to be rare (Cheryl Ryder 2002, personal communication).

2.1.2.4 Green Sea Turtle

The range of the green turtle in the continental United States extends from Massachusetts to Texas. However, as the green turtle is typically a tropical and subtropical species, the occurrence of this species north of Virginia during any month of the year is considered unusual (NOAA 2002; Thompson 1988). Green turtles are typically considered stragglers when found in New England waters (USFWS 2006). Therefore, in comparison to other species that may be seasonally observed in Nantucket Sound (i.e., Kemp's ridley, leatherback and loggerhead turtles), the green turtle is the least likely to be observed in Nantucket Sound.

2.1.3 Avian Species

2.1.3.1 Piping Plover

2.1.3.1.1 New England and Massachusetts

In Massachusetts, piping plover breed on beaches along the North Shore, South Shore, Upper Cape, Lower Cape, Bristol County, Elizabeth Islands, Martha's Vineyard, and Nantucket (Melvin and Mostello 2007). Figure C-8 (Plover) shows the breeding locations in the vicinity of Nantucket Sound. During the 2002 season, observers reported 106 active nest sites in Massachusetts (Melvin and Mostello 2003). The Lower and Upper Cape Cod regions represented nearly 60 percent of Massachusetts' breeding pairs in 2002. These regions continue to support most of the Massachusetts population. Roughly 120 breeding pairs of piping plover occur along the coastal areas of Nantucket Sound. South Beach, Chatham, and the Monomoy Islands support over 60 percent of these breeding pairs (Melvin and Mostello 2007). Within Nantucket Sound and Vineyard Sound, piping plover nest on island beaches. The 2002 Massachusetts piping plover census reported 84 breeding pair of piping plover dispersed on the islands of Nantucket and Martha's Vineyard (Figure C-8).

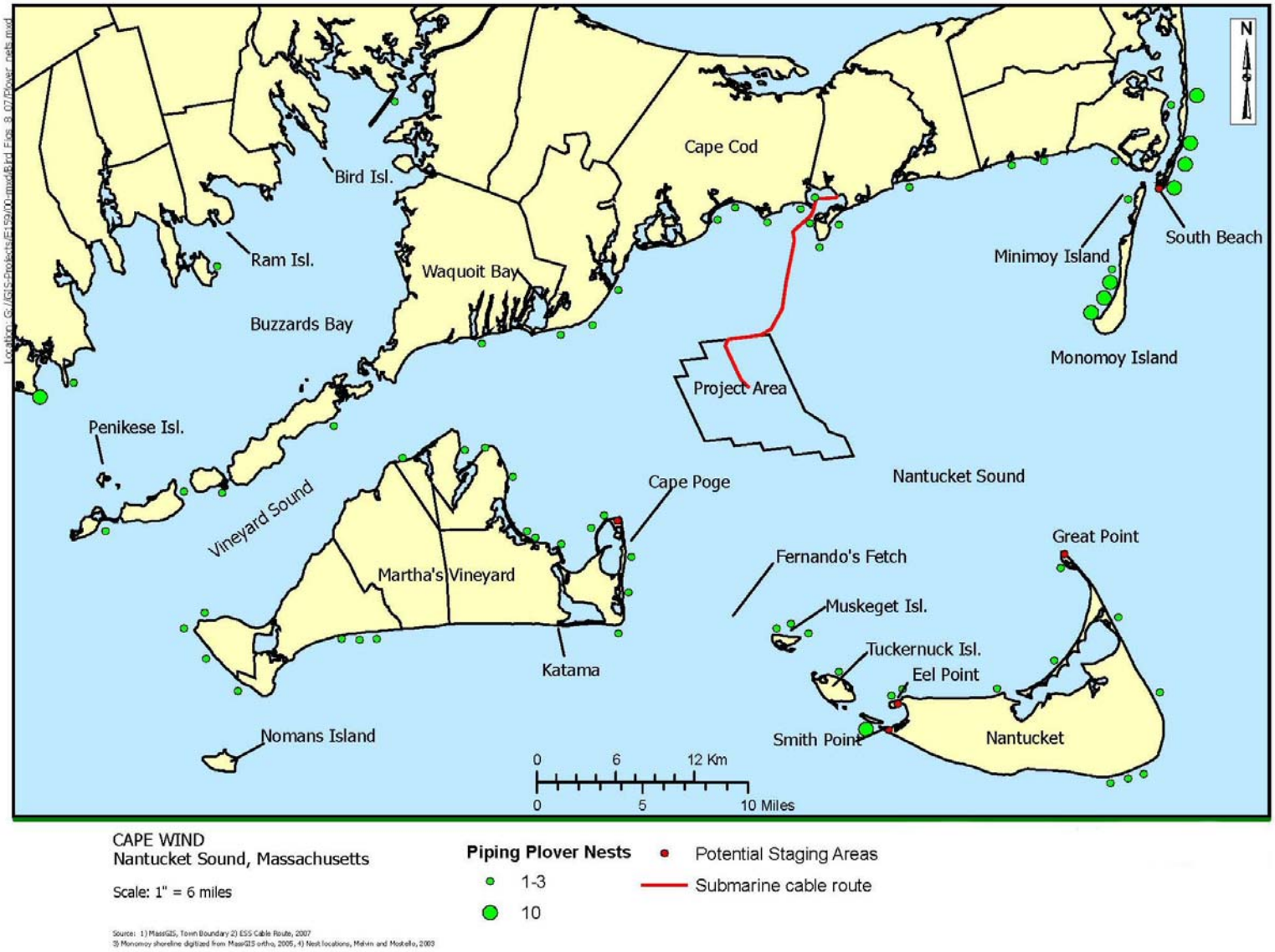

Figure C-8.
Piping plover nests in Nantucket Sound and Buzzards Bay

Preliminary estimates of the number of breeding pairs in Massachusetts are 475 and 480 for 2005 and 2006, respectively. Massachusetts breeding pairs represent over 75 percent of the New England population (Figure C-9).

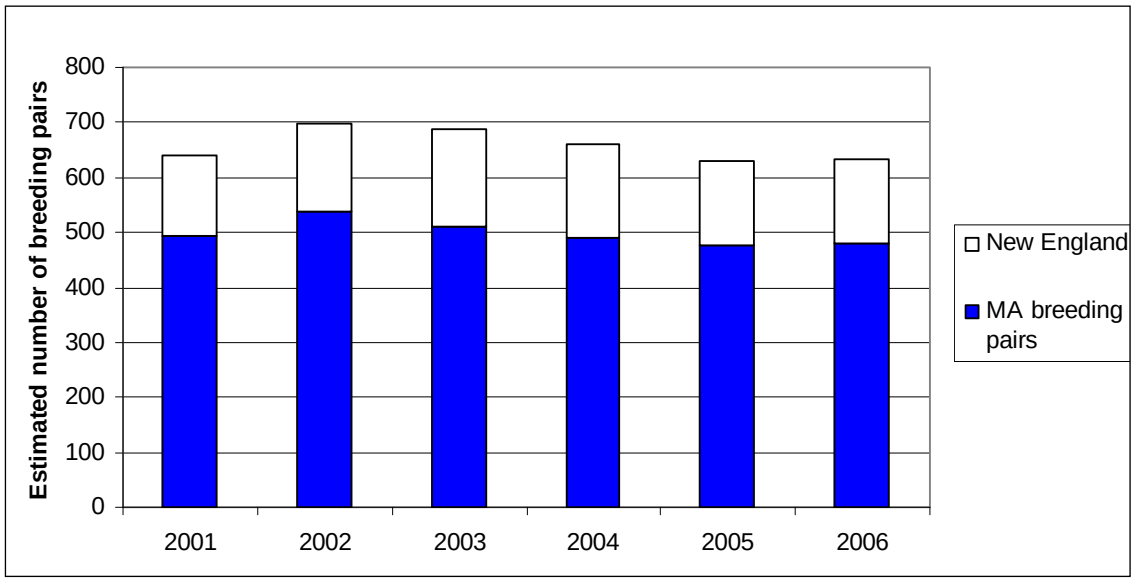

Figure C-9.
Estimated number of piping plover breeding pairs 2001-2006 – MA and New England (USFWS 2006)

In Massachusetts, the number of chicks fledged per nest in 2005 and 2006 was estimated to be 1.0 and 1.23 for 470 pairs, respectively (Figure C-10). Figure 6 shows that, at least for the past six years, declines or increases in productivity are not correlated to the number of breeding pairs the following year.

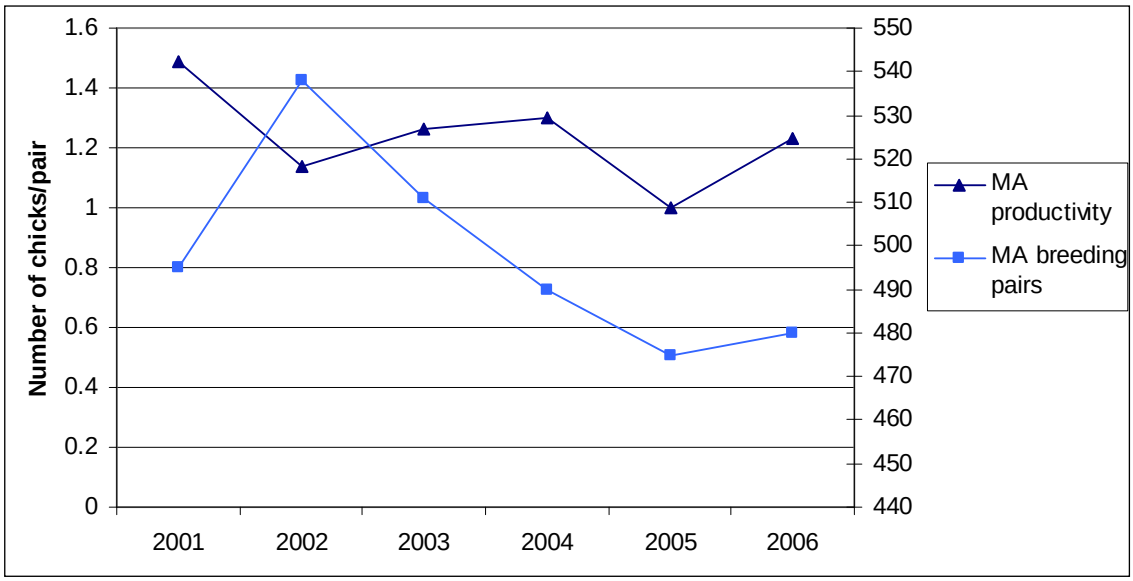

Figure C-10.
Estimated productivity and number of piping plover breeding pairs 2001-2006 in MA (USFWS 2006)

2.1.3.1.2 Piping Plover Use of the Proposed Action Area

Beginning in 2002, the Applicant and the Massachusetts Audubon Society (MAS) conducted studies to characterize bird use of Nantucket Sound. Between 2002 and 2004, 66 aerial surveys and 65 boat surveys were conducted in study areas within Nantucket Sound between March and September, the timeframe when piping plover could be present in the area.

No piping plovers were observed in the proposed action area, Horseshoe Shoals (HSS), or in other areas of Nantucket Sound included in the studies. The results of the surveys suggest that crossings do not occur in HSS during the day.

However, there are limitations to the data available, particularly related to the potential occurrences in the proposed action area at night, or during periods of inclement weather, which . results in some level of uncertainty about their use of the proposed action area. There are also difficulties in detecting the presence of small, light-colored birds from 250 ft (75 m) or greater above the water's surface during aerial surveys. Also, during boat surveys conducted at the water's surface, it may be difficult to detect small, light-colored high-flying birds. Few observations of shorebirds took place during the surveys. Twenty dunlin (*Calidris alpina*) were observed in the study area during a fall aerial survey in 2002 (Report No. 4.2.4-9). Dunlin is a notably darker, larger shorebird species than piping plover and would therefore be easier to detect. Few observations during surveys could indicate minimal use of the Sound by shorebird species, or could reflect the difficulties in adequately surveying for this group of birds during aerial or boat surveys. Aerial and boat surveys were conducted during the day, during periods of fair weather and good visibility. Radar surveys, in the absence of infrared visual confirmation, can not describe shorebird movements through the project area at night; radar surveys conducted during the day with targets analyzed according to size can not adequately describe individual shorebird species' movement patterns through the proposed action area. Therefore, there is no information on the potential of piping plover occurrences in the proposed action area at night, or during periods of inclement weather.

Breeding Period

Once nesting locations are established, piping plovers are relatively sedentary. Most of their movements involve walking or running as opposed to flying because of their cryptic coloration on the ground (Haig and Elliot-Smith 2004). Because nesting and feeding habitats are proximal, movements between the locations are relatively short. Once the chicks have hatched, the area between nesting locations and foraging areas increases (USFWS 1996). Once the chicks have reached fledgling status, family groups become more mobile. Telemetry data gathered in North Dakota suggests that juveniles may travel more than 32 miles (50 km) within a few days of being able to fly (Haig and Elliot-Smith 2004).

It is important to note that failed breeders and unpaired birds could be more mobile during the breeding season. Crossings of the project area are a potential during the breeding season, but no data has been recorded. Therefore, potential flight paths are not known. There is the potential that individuals could travel between the mainland to Nantucket or Martha's Vineyard in search of a mate or habitat during the breeding season. Aerial and boat surveys conducted in HSS indicate that these movements do not occur in the proposed action area during the day;

however, more data collected in a variety of weather conditions and at night is required to more fully assess these potential movements.

Migration

In addition to the breeding season, birds could appear in the proposed action area during migration. Migrant birds from interior U.S. and Canada, as well as Atlantic Canada and New England, have been documented along the Atlantic Coast during migration (Haig and Elliot-Smith 2004). Little is known of the actual migration routes of Atlantic Coast piping plover during migration. However, observations of staging birds along the Atlantic Coast indicate that piping plover make short-distance movements to stop-over sites while following a narrow corridor along the coastline (Haig and Elliot-Smith 2004, USFWS 1996). There are very few sightings of Atlantic Coast plovers offshore or inland during migration (USFWS 1996). Plovers have been observed at wintering grounds on islands in the Caribbean (USGS 2007, Haig and Elliot-Smith 2004); therefore, they will undertake long-distance ocean crossings. There are documented reports of pre-migratory staging plovers on the Cape Poge Elbow, the eastern point of Martha's Vineyard (USFWS 1996). There is potential migrant use of Smith Point and Great Point, Nantucket (USFWS 1996). South Beach, Chatham is considered a stop-over site for migratory piping plover, and if they were to depart the island for a destination in New York, they could cross areas of Nantucket Sound. Pre-breeding plovers have been observed staging on Eel Point, Nantucket Sound (USFWS 1996).

Piping plover could potentially cross areas of the Vineyard Sound or Nantucket Sound in order to access or depart breeding or staging locations on Martha's Vineyard and Nantucket. Additionally, weather events could potentially push migrants following the coastline either inland or offshore. It is unknown, however, if these movements would result in crossings of HSS. Aerial and boat surveys conducted during the early-spring and fall dispersal periods in 2002, 2003, and 2004 suggest that these crossings do not occur in HSS during the day. However, further study is required to assess these movements in a variety of weather conditions and at night. Due to the restriction of aerial and boat surveys to periods of fair weather and good visibility, as well as the limitations of visual data collected from a boat or a plane, telemetry surveys would be beneficial in determining piping plover movement patterns in Nantucket Sound during a variety of conditions.

2.1.3.2 Roseate Tern

2.1.3.2.1 Breeding Season

The majority of roseate terns in North America breed between Long Island and Cape Cod, with the largest colonies located at Great Gull Island, New York, and Bird Island in Buzzards Bay. In Massachusetts, roseate terns are present from late April until mid-September, during the breeding and pre-migratory staging periods. Northeast birds arrive from late-April to mid-May and aggregate in Nantucket Sound before dispersing to breeding locations in the Sound, Buzzards Bay, Maine, and Canada. An estimated 10 to 12 percent of the roseate population travels north of Nantucket Sound, while 88-90 percent move south and west in Buzzards Bay, Long Island Sound, and to Falkner Island in Connecticut (Spendelow et al., unpublished).

Figure C-11 shows the roseate tern active breeding and potential breeding locations in Massachusetts. In 2005, there were 27 breeding pairs of roseate terns on South Monomoy Island (1 pair) and Minimoy Island (located off of the Monomoy Refuge) (26 pairs) (USFWS 2005), located approximately 12 mi (19.3 km) from the proposed action area in Horseshoe Shoals (HSS). There were 27 pairs on Minimoy and 2 pairs on South Monomoy in 2006. Since 1999, there have been small numbers of pairs believed to be nesting on Smith Point, Nantucket (16.8 mi [27 km] from the center of the proposed action area in HSS) and Muskeget Island (11.5 mi [18.5 km] from HSS) in Nantucket Sound. However, in recent years, neither nests nor chicks have been confirmed (Spendelow et al., unpublished). In Buzzards Bay, Bird and Ram Island are designated Audubon Important Bird Areas because of the islands' significant tern breeding habitat. Bird Island once supported the largest population of breeding pairs in North America, however, due to erosion on the island, nearby Ram Island has recently supported an increase in pairs. Between the two islands, in the past 8 years there have been 1,500 to 1,800 breeding pairs of roseate terns (Spendelow et al., unpublished), as well as hundreds of breeding common terns. The center of the project area in HSS is located approximately 23 mi (37 km) from Bird Island and 25.8 mi (41.5 km) from Ram Island. In 2003, after the Bouchard oil spill, roseate and common terns were discouraged from nesting on Ram Island until the oil was removed. As a result, 250 pairs of roseate terns nested on Penikese Island in Buzzards Bay, located 30.8 mi (49.6 km) from the center of the proposed action area in HSS. In 2004, 9 pairs of roseate terns nested on Penikese, 84 pairs in 2005, and there were 50 nesting pairs in 2006 (Spendelow et al., unpublished). There are additional locations in Massachusetts where a few pairs of roseate terns have nested in past years, including Gray's Beach, Yarmouth, and Nauset Inlet, Eastham, however, there have been no breeding pairs at these locations reported in recent years (Spendelow et al., unpublished).

Important roseate tern foraging locations vary seasonally and are dependent on prey location and availability (Heinemann 1992). Roseate terns are known to forage as far as 16 to 19 miles (25 to 30 km) from colony sites (Heinemann 1992). Bird surveys conducted in the early 1990s indicate that popular foraging areas vary during the day and throughout the season, dependent on the tide and prey movements. Locations that have been found to be popular roseate tern foraging sites among different years include Woods Hole, Falmouth (10 mi [16 km] from Bird Island), near the entrance to the western end of the Cape Cod Canal (3 to 7 mi [5 to 11 km] from Bird Island); and the Mashnee Flats Shoal near the Cape Cod Canal (5.3 mi [9 kilometers (km)] from Bird Island) (Heinemann 1992). Roseate terns nesting at Bird Island were regularly observed traveling (12 to 19 mi [20 to 30 km] from Bird Island) to forage in Vineyard Sound. Also, flocks of terns were regularly observed foraging along the shore of Cape Cod from Woods Hole to Poponesset Bay (Heinemann 1992).

Although the proposed action area in Horseshoe Shoals (HSS) does not occur within nesting or popular foraging habitat, daily foraging activity during the breeding and pre-migratory staging periods could result in roseate tern crossings of the area. In order to determine tern use of the proposed action area, boat and aerial visual surveys were conducted. From 2002 to 2004, Massachusetts Audubon Society (MAS) and the Applicant conducted aerial surveys (May through mid-September) over the proposed action area in HSS and surrounding areas in Nantucket Sound.

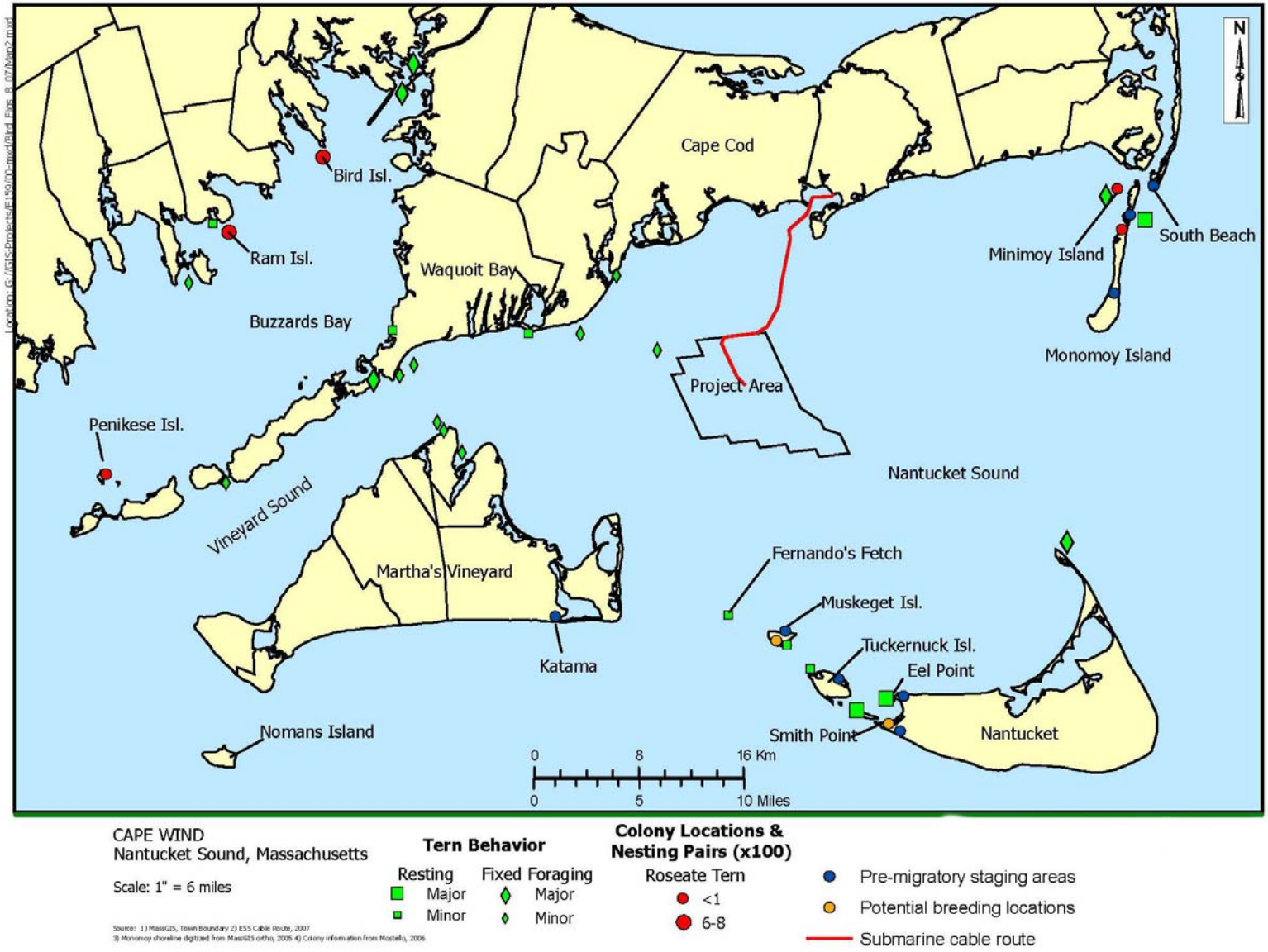

Figure C-11.
Roseate terns in Nantucket Sound and Buzzards Bay

Breeding Season Aerial Surveys

During the breeding seasons (mid-May to late July), MAS and the Applicant conducted a total of 15 aerial surveys and documented relatively minimal tern activity in the HSS proposed action area. Due to the inability to consistently identify terns to the species level during surveys, the majority of terns were counted as common-roseate-type terns during both MAS and the Applicant's surveys. MAS breeding season surveys documented a total of 832 roseate terns, 4,779 common terns, and 12,646 common-roseate-type terns throughout Nantucket Sound (Perkins et al., 2004a, Sadoti et al., 2005a). Of the total number of terns observed in 2003 during the MAS surveys, 1.5 percent occurred in HSS, 5.3 percent in Monomoy-Hankerchief Shoals (MHS), 2.5 percent in Tuckernuck Shoals (TS), and 90.7 percent were outside of shoal study areas (Perkins et al., 2004a, Sadoti et al., 2005a). In 2004, of all the terns observed, 0 percent occurred in HSS, 0.7 percent in MHS, 0 percent in TS, 99.3 percent were outside of the shoal study areas. During surveys, the majority of terns were observed over shallow waters close to the shorelines of Cape Cod, Martha's Vineyard, and Nantucket (Figure C-11). Of the 2,888 total terns observed during the Applicant's breeding season surveys, 9.6 percent occurred in HSS, 2.6 percent in MHS, 5.7 percent in TS, and 82.1 percent were outside of the shoal study areas (Report No. 4.2.4-3, 4.2.4-10, and 4.2.4-8). The Applicant surveys documented large aggregations of resting terns at Fernando's Fetch, a sandbar northwest of Muskeget Island (Figure C-11). Of all terns observed in HSS during the Applicant's aerial surveys, the majority were traveling, though terns were also observed actively foraging (Report No. 4.2.4-3, 4.2.4-10, and 4.2.4-8).

During the Applicant's aerial surveys, the altitude of flying birds was estimated in relation to the surface of the water and the altitude of the plane (250 ft [75 m]). During boat surveys, when birds were in the vicinity of the vessels, their flight height was estimated. During the breeding season surveys, the majority of terns observed by the Applicant were flying well below the proposed rotor zone (75.5 to 440 ft [23 to 134 m]), mainly below 39.4 ft (12 m) (Report No. 4.2.4-3, 4.2.4-10, and 4.2.4-8). During a four day tern survey in 2006, a total of 966 commuting terns were observed and 89 percent of their documented flight altitudes were categorized as below the rotor zone (Report No. C-1). During the Applicant's breeding season surveys from 2002-2004, 32 terns flew within the rotor zone, including one roseate tern at 75.5 ft (23 m) (Report No. 4.2.4-3, 4.2.4-10, and 4.2.4-8). MAS generally did not estimate flight heights below 300 ft (91 m) during aerial surveys conducted at an average height of 500 ft (150 m), however, flight height observations were made during boat surveys. During MAS boat surveys, of those terns observed flying in HSS, 3.2 percent of 130 traveling terns flew within the rotor zone in 2003 and 2.9 percent of 317 traveling terns flew within the rotor zone in 2004 (Perkins et al., 2004a, Sadoti et al., 2005a). Other terns were reported flying well above the proposed rotor zone.

2.1.3.2.2 Staging and Migration Season

After breeding, roseate terns move to popular staging areas in the vicinity of favorable feeding sites (Trull 1998). Staging areas are where mixed flocks gather to rest between foraging activities during the day (Trull et al., 1999). In the Northeast, the number of birds at staging grounds peaks during mid-August, though staging can begin as early as mid-June for some non-

breeders or failed breeders (Spendelow et al., unpublished). Banded birds have been observed at staging grounds from 1 to 26 days (Shearler and Kress 1994). The birds then disperse to their wintering grounds during August and September. Roseate terns are known to travel far from their colony sites to popular staging areas, and sometimes travel north to staging areas before traveling south to wintering grounds. For example, Shealer and Kress (1994) reported staging and roosting roseate terns from 8 different colonies in the Northeast in August at Stratton Island, Maine. Terns that were known to breed in Maine and New York were observed at the South Beach staging location off of Chatham, Massachusetts (Trull et al., 1999). Cape Cod, Massachusetts, Sable Island off of Nova Scotia, and Long Island, New York, also provide important staging habitat for the northeastern population. Because of the species' wide range of dispersal during the migration and staging periods, there is a potential that any individual from the northeastern population could occur in the vicinity of the proposed action over the course of a year.

Cape Cod, Massachusetts supports the largest pre-migratory staging habitat for roseate terns in North America (Trull, 1998). For example, during a peak survey conducted on September 1, 1998, 4,500 roseate terns were counted off of Cape Cod, with 3,850 counted on South Beach in Chatham, Massachusetts. Other known staging areas include sand flats or beaches located on South and North Monomoy Islands, Sandy Neck and Jeremy's Point in Cape Cod Bay, Katama on Martha's Vineyard, Smith and Eel Points on Nantucket Island, and Tuckernuck and Muskeget Islands in Nantucket Sound (Trull et al., 1999). Terns have occasionally been observed staging at Nauset Inlet on the east coast of Cape Cod and Hatch's Harbor at the northern tip of Cape Cod (Trull et al., 1999). Figure C-11 shows the tern pre-migratory staging locations that are located in Nantucket Sound.

Not all staging locations are used as nighttime roosting areas. Trull et al., (1999) observed that roseate terns only roosted at one of the staging areas in Nantucket Sound, South Beach (Figure C-11). Sandy Neck in Cape Cod Bay was the other staging area observed to be used as a roosting site. During the staging period, terns arrive at roosting sites around sunset and continue to arrive after dark (Trull et al., 1999). Terns are assumed to depart staging grounds before dawn and at sunrise to travel to foraging locations. While making these movements, roseate terns are known to travel overland across Cape Cod, and are also anticipated to pass over waters in Nantucket Sound (Spendelow et al., unpublished).

Staging and Migration Season Aerial Surveys

Surveys were recently conducted to document tern use of the proposed action area during the fall staging and migration periods. In total, 37 aerial surveys were conducted by MAS and the Applicant during the fall pre-migratory staging periods from 2002-2004. Relatively minimal tern activity was detected in the HSS proposed action area during these surveys. MAS fall surveys documented a total of 16,550 terns throughout the Sound. In 2002, 94 percent of terns observed were outside of shoal study areas; in 2003, 92 percent were observed outside of the shoals; and in 2004, 93 percent were observed outside of the shoals (Perkins et al., 2003, Perkins et al., 2004b, Sadoti et al., 2005b). Of all terns observed in HSS during aerial surveys, the majority were traveling, though some were observed actively foraging. Surveys in 2002-2004 documented thousands of terns staging at South Beach in Chatham, Massachusetts (approximately 23.2 mi [37.3 km] from the center of the project area in HSS). An additional

boat survey was conducted from August 28 to August 31, 2006, to observe the movement of terns near the popular staging area on South Beach and Monomoy Island. The survey documented 932 common terns and 63 roseate terns commuting, foraging, or kettling (Report No. C-1). It was determined that terns staging on Monomoy likely pass through HSS during the day while traveling to foraging locations and when returning to Monomoy to roost for the night.

Similar to breeding season surveys, the majority of flight heights observed in the proposed action area during the fall occurred well below the rotor zone. During the 2002-2004 fall boat and aerial surveys, MAS observed only one tern flying within the rotor zone of the proposed turbines within HSS (Perkins et al., 2003, Perkins et al., 2004b, Sadoti et al., 2005b). Of the 5,112 terns observed during the Applicant's fall aerial surveys, 5,012 flew below 75.5 ft (23 m) (Report No. 4.2.4-9).

In summary, surveys conducted by the Applicant suggest that only 5 percent of traveling and foraging terns observed in the study area occurred at flight heights within the rotor zone (Report No. C-1). The results of MAS boat surveys indicate that the majority (95+ percent) of terns observed within HSS occurred at heights below the rotor zone (MAS 2006). The majority of tern observations in Nantucket Sound occurred outside of the Shoal study areas. Terns were generally concentrated around the mainland and island coasts of the Sound, particularly Monomoy Island during the late-August and early-September staging period (Figure C-11). During this period HSS likely had the lowest level of activity out of any similar habitat surveyed in the Sound (MAS 2006). Tern nesting and foraging locations are dynamic. If the locations of colony sites or popular foraging areas change, the occurrence and density of roseate terns in HSS could change. However, multiple years of data collected in HSS suggest minimal tern activity occurs in HSS relative to other locations in the Sound. Additionally, the major historical locations of breeding terns and their preferred foraging locations are well studied and there is no data that suggest that HSS would become a high tern use area.

It should be noted that there are limitations to the available data regarding roseate tern occurrence and behavior in HSS, especially during crepuscular periods and inclement weather. Aerial and boat surveys can only be conducted during periods of good visibility and decent weather. Nighttime radar data in the absence of infrared visual confirmation cannot provide species specific information. Similarly, radar data obtained during the day with targets classified into size-categories cannot be used to describe tern movement patterns or flight behaviors through HSS. Therefore, data is lacking for tern activity in HSS at night, during crepuscular periods, and during periods of inclement weather. The available survey data provides presence and density information. However, it does not describe movement patterns through the proposed action area (Nisbet 2005). Additionally, the majority of observations of terns were grouped as common-roseate-type terns and therefore, the data results in generalizations about tern behavior when there are known differences between common and roseate tern flight behaviors, as well differences in the species' densities within the proposed action area.

Flight height behavior observed during aerial surveys conducted at 250 ft (75 m) or greater should be interpreted with caution due to the difficulties detecting small, light-colored birds from the water's surface to the height of the aircraft, and because birds greater than the height of the aircraft may not have been detected (Nisbet 2005). Some high flying terns may have been

undetected due to the 'wedge-shaped' area of visibility from an aircraft (Nisbet 2007). Nisbet (2007) suggests that the volume of space visible from an aircraft is widest at the surface, and 'tapers to a line along the path of the aircraft'. Therefore the overall percent of terns flying within the rotor zone may have been underestimated (Nisbet 2007). There is a possibility that small numbers of high flying terns were undetected, however, it is unlikely that large groups of terns would have been undetected. MAS generally did not estimate flight heights below 300 ft (91 m) during aerial surveys conducted at an average height of 500 ft (150 m). However, the results of boat surveys indicate that that few terns flew within the rotor zone within HSS. Estimates of flight heights from boat surveys provide more accurate information for low flying birds than do aerial surveys.

Given that some high flying terns may not have been detected, aerial surveys may have also underestimated the density of terns observed in HSS. However, aerial survey sample biases would have been ubiquitous throughout the Sound survey areas so that the proportion of terns in HSS in relation to other areas is accurate.

Particularly in the breeding season, roseate terns are under intense energy demands as they access foraging locations as far as 16 to 19 miles (25 to 30 km) from colony sites. Therefore, if a tern were undertaking such a movement across HSS it can be assumed that it would travel in the most energy efficient manner. Therefore, a tern's height of flight would depend largely on wind speed and direction. Nisbet (2007) suggests that it would be safe to assume, without more data, that most terns or all terns flying with a following wind (± 45° of the birds' heading) are likely to fly at rotor height through HSS. Although there are limitations to the data available, surveys conducted in a variety of wind conditions by MAS and the Applicant indicate that this is not accurate. Overall, a small percent of terns were observed within the rotor zone. It can be assumed, however, that the majority of roseate terns would fly below the rotor zone when flying into a headwind. When traveling into a strong headwind, other species of tern have been observed to fly closer to the surface of the waves (Alerstam 1985). As the predominant wind direction is from the southwest during the summer and the average annual wind speed is 19.75 mph (8.8 m/s) in Nantucket Sound, it can be assumed that when roseate terns are flying in a south, southwest, or westerly direction across HSS, they would regularly occur at heights below the proposed rotor zone. Additionally, if making shorter, more localized flights in an effort to forage in HSS, roseate terns would be expected to occur at flight heights at or below their maximum foraging height of 39.4 ft (12 m). These assumptions are speculative and demonstrate that additional data is needed to assess roseate tern flight behavior in HSS. However, there is credibility in the data that is available despite the limitations of survey methods, because results are consistent between both the MAS and the Applicant's surveys. Both parties produced comparable results suggesting that the majority of terns observed in HSS were below the rotor zone and that relatively low numbers of terns were observed in HSS overall.

3.0 EFFECTS OF THE ACTION

Potential effects on listed species or critical habitat may occur either from routine activities or from accidental events and may be direct or indirect. Discussed below are general sources of potential impacts to endangered and threatened species and/or designated critical habitat from the proposed action. Following that, more specific impact factors related to individual species are included. An impact determination is then made for each listed species and designated critical habitat.

3.1 Whales

Potential impacts to listed whale species can occur from planned, routine, or anticipated impacts during the construction, operation, and decommissioning phases. In addition, there are several potential impacts that could occur as a result of accidental or unplanned events.

3.1.1 Whales-Routine Activities

The major impact-producing factors affecting whales as a result of the proposed action include noise associated with construction and operation activities, vessel traffic, changes in water quality, and habitat.

3.1.1.1 Construction and Operational Noise

The maximum submarine sound generated during construction will occur during installation of the monopile foundations. Sound levels measured during impact pile driving operations at the Utgrunden Wind Park in Sweden were used to model underwater sound expected from installation of the monopiles since the size of the monopiles and the installation techniques proposed for the proposed action are the same as for the Utgrunden Wind Park (Report No. 4.1.2-1). The Utgrunden data show a maximum (L_{max}) sound level of 178 dBL at 1,640 ft (500 m). Frequency plots from the Utgrunden data show the peak energy from pile driving occurred between 200 and 1,000 Hz, with underwater sound levels falling below background levels (inaudible) for frequencies below 5 Hz.

The jet plow embedment process for laying the offshore transmission cable system circuits and inner-array cables produces no sound beyond that produced by typical vessel traffic and the cable installation barge will produce sound typical of vessel traffic already occurring in Nantucket Sound. No substantial underwater sound will be generated during horizontal directional drilling (HDD) operations used to transition the offshore transmission cable system to the upland cable system in Lewis Bay. Due to the sound-insulating qualities of earthen materials (the sediment), and the fact that the drilling would take place through unconsolidated material, the HDD transition is not anticipated to transmit vibration from the sediment to the water (i.e., it would not add appreciable sound into the water column).

The sound source level for a tug and barge traveling at low speed, typical construction vessels, is 162 dB re 1 μPa @ 1 m (Malme et al,. 1989).

Pile Driving

In general, toothed whales have a hearing bandwidth of 100 Hz to over 100 kHz, with the most sensitive hearing in the high frequency range of 10 kHz to 65 kHz where their hearing threshold is 40 to 60 dB (Richardson et al., 1995). Baleen whales react primarily to sounds at low frequencies below 1 kHz, which is consistent with the fact these whales usually communicate at frequencies in the 20 Hz to 500 Hz range (Richardson et al., 1995). The hearing threshold for baleen whales ranges from 82 dB at 500 Hz to 88 dB at 20 Hz (Nedwell et al., 2004).

A dB_{ht} was calculated for several species of whales to determine the actual underwater sound level that is heard by whales from monopile installation at different distances from construction activities. The results of the dB_{ht} analysis, provided in Table 6 in Report 5.3.2-1 of the DEIS, show that no injury to whales are predicted if an individual were to approach as close as 30 m to the pile driving because all dB_{ht} values at this minimum distance are well below 130 dB.

Vessels

Maximum whale hearing thresholds for vessels were calculated for a distance of 100 ft (30.5 m). Levels of 42 dB_{ht} and 45 dB_{ht} were calculated for whales and for toothed whales, respectively. These levels are well below the injury threshold of 130 dB_{ht} and the harassment threshold of 90 dB_{ht}. The animal would be able to hear the vessel, but no physical harm or behavioral effects would occur.

The jet plow embedment process for laying the offshore transmission cable system circuits and inner-array cables produces no sound beyond that produced by typical vessel traffic and the cable installation barge will produce sound typical of vessel traffic already occurring in Nantucket Sound. Furthermore, no substantial underwater sound will be generated during horizontal directional drilling.

Any whales are likely to temporarily avoid a given area around the construction, and only minor impacts would be anticipated due to construction generated noises. Any noise should not affect the migration, nursing/breeding, feeding/sheltering or communication of whales.

Wind Turbine Operational Noise

Once installed, the operation of the WTGs is not expected to generate substantial sound levels above baseline sound in the area. Acoustic modeling of underwater operational sound at the Wind Park was performed for the design wind condition (see Section 3.13 of ESS, 2007). Baseline underwater sound levels under the design wind condition are 107.2 dB re 1 μPa. The predicted sound level from operation of a WTG is 109.1 dB re 1 μPa at 65.6 ft (20 m) from the monopile (i.e., only 1.9 dB re 1 μPa above the baseline sound level), and this total sound level falls off to 107.5 dB re 1 μPa at 164 ft (50 m) and declines to the baseline level at a relatively short distance of 110 m (361 feet). Since the WTGs will be spaced farther apart than 360 ft (110 m) (approximately 629 to 1,000 m or 0.34 to 0.54 nautical miles apart), no cumulative impacts from the operation of the 130 WTGs in the Wind Park are anticipated.

An analysis of predicted underwater sound levels perceived by whales during operations show that no injury or harassment to whales are predicted even if an individual were to approach as close as 65.6 ft (20 m) to a monopile when the turbines are operating at the design wind speed as all dB_{ht} values at this minimum distance are well below 90 dB. In fact, operation sounds will be inaudible for toothed whales, and only slightly audible to baleen whales at the extremely close distance of 65.6 ft (20 m). Therefore, no behavioral effects to whales are anticipated even if an individual were to approach within 65.6 ft (20 m) of the structures.

Decommissioning Noise

Noise produced by the decommissioning is expected to be similar to those produced during construction. However, decommissioning will not require pile driving activities, which cause the highest sound levels of any activities associated with construction. Decommissioning will involve the use of similar vessels, cranes, jet plow, cutting and welding equipment and other tools that were involved in construction, but would not include any pile driving, blasting or activities which approach the noise level of pile driving. During decommissioning, the monopiles and transition pieces would be cut off at the mudline. As such, the noise impacts from decommissioning activities would appear to be less than the worst case impacts already presented for construction and will be minor. However, consultations with other Federal agencies will be conducted in order to confirm the potential for impact.

3.1.1.2 Vessel Traffic

Humpback whales are relatively tolerant of boats, but, due to this habituation, may be more susceptible to ship collisions. Right whales continue to die from vessel collisions, even though they can theoretically hear approaching ships (Richardson *et al* 1995; Nowacek et al., 2004). A study by Nowacek et al., (2004), reported that right whales did not respond to the sounds of approaching vessels or the actual vessels. Some anecdotal observations suggest that right whales only respond when vessels approach to within a very close range. Right whales off the eastern coast of North America are frequently exposed to vessels, and they may have habituated to the sounds of approaching vessels at greater distances (Richardson et al., 1995; Terhune & Verboom 1999; Laist et al., 2001). The greatest known current cause of right whale mortality in the western North Atlantic is collision with large ships such as container ships, military vessels and tankers. There were 27 documented deaths from 1970 through 1991 (NMFS 2005). From 1991 through the beginning of 1993, an additional 3 deaths were reported as a result of collisions with vessels (NMFS 2005). According to a recently published large whale ship strike database based on public information collected by NOAA Fisheries from 1975 to 2002 (Jensen and Silber 2003), finback whales are the most often reported species hit by ships (75 records of strike) followed by humpback (44 records), North Atlantic right (38 records), gray (24 records), minke (19 records), southern right (15 records), and sperm whales (17 records).

As discussed above in the species descriptions for humpback, fin, and right whales, respectively, each of these species are known to seasonally migrate between their fall/winter mating, birthing, and nursing grounds in the southern waters of the West Indies and the mid- and south-Atlantic states (including the Carolinas, Georgia, and Florida), and their spring/summer feeding grounds in the western North Atlantic (Clapham 1992; Baraff and Weinrich 1993; Waring et al., 2006; NMFS 2005; CeTAP 1982; USEPA Region 1 1988). Therefore, whales are

only expected to be within the vicinity of New England waters during the spring and summer feeding seasons. While the endpoints of the whales' migration are well established (Martin et al., 1984; Mattila et al., 1989; Waring et al., 2006), the exact route between the summer and wintering grounds is unknown, although it is likely to be well offshore (Clapham and Mattila 1990).

Once the north-bound migrating whales (cow-calf pairs included) reach their feeding grounds in New England waters, as discussed above, their fine-scale movements have generally been observed to follow dense aggregations of their preferred prey species, which are not developed in Nantucket Sound, and tend to occur in greater abundance in waters further offshore, around Stellwagen Bank, Jeffreys Ledge, Browns and Bacaro Banks, and in the Great South Channel (Kenney and Winn 1986). Additionally, both feeding and nursing behaviors have been observed in Cape Cod Bay and the lower Bay of Fundy (Schevill et al., 1986; Hamilton and Mayo 1990; Marx and Mayo 1992; Kraus and Kenney 1991; NMFS 1994; NMFS 2005; Waring et al., 2006).

It has been reported that vessel traffic also may physically displace some whale species from feeding areas. There is evidence that some whales may have been displaced from traditional feeding and wintering areas due to increased vessel traffic in Pacific waters (Baker et al., 1982; Forestell 1986). Hawaiian research of Pacific humpback populations have observed cow-calf pairs to move away from areas presumed to be favored habitat where human activities were also common (Lien 2005). Canadian research regarding humpbacks' response to whale watching activities also observed cow-calf pairs to be especially sensitive to human presence (Lien 2005). However, evidence from whale watching and fishing activities in Massachusetts waters indicates that humpback and fin whales readily habituate to the presence of large and small motor vessels (Watkins 1986).

3.1.1.3 Water Quality

Increased TSS

The primary water quality concern to the listed species is elevated concentrations of Total Suspended Solids (TSS) associated with construction and decommissioning activities. Sustained elevated concentrations of TSS may generate an avoidance response from the protected species (direct impact) and may potentially affect prey species (indirect impact) of the whales and sea turtles (i.e., zooplankton and fish). However, as indicated below, construction and decommissioning activities are expected to result in only temporary and localized increases in TSS and therefore will have minimal impacts to the listed species.

Construction activities associated with installing the monopile foundations, scour control, and offshore cables will result in a temporary and localized increase in suspended sediment concentrations. Decommissioning-related impacts will be short-term and localized and are expected to be similar to impacts during construction. The pile driving hammer and jet plow technology that will be used to install the monopile foundations and the offshore cables, respectively, were selected specifically for their ability to keep sediment disturbance to a minimum. Due to the predominant presence of fine to coarse-grained sands in Nantucket Sound, localized turbidity associated with construction or decommissioning is anticipated to be minimal and confined to the area immediately surrounding the monopiles and the offshore cable route.

Sediments disturbed by construction or decommissioning activities are expected to settle back to the sea floor within a short period of time (one to two tidal cycles). In addition, the proposed action area is situated in a dynamic environment that is subject to naturally high suspended sediment concentrations in near-bottom waters as a result of relatively strong tidal currents and wind and storm generated waves, particularly in shoals areas. Therefore, marine organisms in this area are accustomed to substantial amounts of suspended sediment on an irregular basis and should not be substantially impacted by a temporary increase in turbidity.

Simulations of sediment transport and deposition from jet plow embedment of the offshore transmission cable system and the inner-array cables were performed. These simulations, which used two models (HYDROMAP to calculate currents and SSFATE to calculate suspended sediments in the water column and bottom deposition from the jet plow operations), estimated the suspended sediment concentrations and deposition that could result from jet plow embedment of the cables. The full analysis is included in Report No. 4.1.1-2.

The model results demonstrate that concentrations of suspended sediment in the water column resulting from jet plow embedment operations (i.e., concentrations above natural background conditions) are largely below 50 mg/L. The effect of grain size distribution is evident since the finer sediments present in portions of the Lewis Bay area, the area at the southern half of the north-south portion of the route, and the area just northwest of the ESP remain in suspension longer due to higher silt and clay fraction. This results in larger predicted plume extents.

It is important to note that the suspended sediment concentration levels are short lived due to the tides flushing the plume away from the jetting equipment and the sediments rapidly settling out of the water column. To put the water column concentrations in perspective, Figure 4.1.1-2 of the DEIS shows the duration that a 10 mg/L excess (above background) suspended sediment concentration is seen. Most of the area shows a duration of less than 3 hours after the jet plow has passed a given point along the route. In places along and immediately adjacent to the cable route, suspended sediment concentrations are predicted to remain at 100 mg/L for approximately 2 to 3 hours.

In Lewis Bay, suspended sediments are predicted to remain in suspension considerably longer than in Nantucket Sound due to weak tidal currents. As a result, water column concentrations are predicted to build-up rather than quickly disperse. The model results demonstrate that concentrations of suspended sediment in the water column resulting from jet plow embedment operations (i.e., concentrations above natural background conditions) in Lewis Bay are largely below 500 mg/L. Suspended sediment concentrations in excess of 100 mg/L are generally predicted to remain for less than 2 hours with the exception of some sections along the cable route showing durations at 6 hours. Suspended sediment concentrations in excess of 10 mg/L are generally predicted to remain for less than 24 hours after the jet plow has passed a given point along the route, except near the Yarmouth landfall where concentrations in excess of 10 mg/L are predicted to remain for up to 2 days after the jet plow passes as a result of very weak currents and fine bottom sediments.

These TSS concentrations are still minimal when compared to the active bed load sediment transport known to exist in Nantucket Sound (between 45 and 71 mg/L under natural tidal conditions and up to 1,500 mg/L as a result of trawling operations (see Section 3.16.2.2 of ESS 2007). Sediment suspension during construction and decommissioning activities will not result in long-term or environmentally significant elevations in water column TSS. Zooplankton or fish species may be temporarily affected or displaced in the immediate vicinity of the area of the activity; however, they are likely to rapidly return to these areas once construction in the specific area is ceased or completed. In addition, since the proposed action area is situated in a dynamic environment that is subject to naturally high suspended sediment concentrations in near-bottom waters, these organisms would be accustomed to substantial amounts of suspended sediment on an irregular basis and should not be substantially impacted by a temporary increase in turbidity. Whales and sea turtles that may be present in the vicinity of the proposed action area during construction are not expected to be adversely affected by temporary increases in TSS and since they are mobile, are capable of avoiding or moving away from the disturbances associated with construction.

Sediment suspension during excavation of the HDD borehole ends in Lewis Bay will be minimal since these activities will be contained within the cofferdam and the top of the sheet piles for the cofferdam will contain turbidity associated with dredging for the HDD borehole end transition. Furthermore, it is unlikely that the protected whale or sea turtle species would be present this close to shore in Lewis Bay. Therefore, no impacts to these protected marine species will occur from the limited, contained sediment suspension during excavation of the HDD borehole ends in Lewis Bay. These activities will not be required during decommissioning.

Protected whales within the area Nantucket Sound are accustomed to substantial amounts of suspended sediment on an irregular basis, from storms and strong tidal currents, and a temporary increase in turbidity from construction and decommissioning activities would have minor impacts.

Contaminated Sediments

Whales bioaccumulate contaminants from their ocean environment, almost exclusively through their food sources. The potential mechanism by which sediments suspended during construction can harm whales is through bioaccumulation of sediment-associated chemicals through ingestion of contaminated prey (indirectly).

Analysis of sediment core samples obtained from the proposed action area indicate that sediment contaminant levels were below established thresholds in reference Effect Range-Low (ER-L) and Effects-Range-Median (ER-M) marine sediment quality guidelines (Long et al., 1995). Therefore the temporary and localized disturbance and suspension of these sediments during project construction activities are not anticipated to result in increased contaminants in lower trophic levels. Therefore, whales are unlikely to experience increased bioaccumulation of chemical contaminants in their tissues from the consumption of prey items in the proposed action area, and any impacts are expected to be negligible.

During the nearshore installation, the release of material from the Horizontal Direction Drilling (HDD) operation within Lewis Bay will be minimized through a drilling fluid fracture or

overburden breakout monitoring program, minimizing the potential of drilling fluid breakout into the water.

Decommissioning-related impacts will be short-term and localized and are expected to be similar to or less than impacts during construction. The suspension of solids are expected to be temporary and localized, as the removal technology that will be used to install the monopile foundations and the offshore cables, respectively, were selected specifically for their ability to keep sediment disturbance to a minimum. Further, the physical composition of the sands and the physical characteristics of the sound environment provide reason to believe that any localized turbidity will settle back to the sea floor within a short period of time (one to two tidal cycles).

3.1.1.4 Temporarily Reduced Habitat

Activities related to construction may cause a temporary reduced availability of habitat for whales in the vicinity of the proposed action. The main anticipated impact would be avoidance of areas of high traffic. However, as under normal conditions the whales are exposed to high volumes of vessel traffic due to commercial and recreational ships within Nantucket Sound, the increase in traffic is not anticipated to displace whales for long periods of time. Some avoidance may also occur during construction activities due to acoustical harassment, as mentioned previously, however this disturbance will be temporary and will not result in any major effects on the listed whales. Studies at off-shore Danish Wind Farms showed that harbor porpoises temporarily avoided the area in the vicinity of the turbines only during construction, and mainly during pile driving activities (Danish Offshore Wind – Key Environmental Impacts, 2006). Abundances for harbor porpoises slowly returned to close to pre-construction values for most of the area, with only a limited area with strong negative impacts that continued through the operation of the Wind Farm. Therefore any impacts are expected to be minor for the whales in Nantucket Sound, and any changes would be temporary.

Construction and decommissioning are not anticipated to result in changes in whale prey abundance or distribution. Some temporary displacement may occur during periods of noise or high suspended sediments, but this will be limited to areas directly surrounding the given activities, causing both prey species and whales moving to an undisturbed area. Pelagic prey tends to be highly variable and animals foraging on these sources move with the food source, as seen with many whales and their prey species. Any temporary disturbance to pelagic prey is likely to mimic typical temporal and spatial variability, and is likely available in other areas of Nantucket Sound and surrounding waters for foraging by whales.

3.1.1.5 Habitat Shift

The presence of 130 monopile foundations, 6 ESP piles and their associated scour control mats in Nantucket Sound has the potential to shift the area immediately surrounding each monopile from soft sediment, open water habitat system to a structure-oriented system, with minor effects to whales.

At the end of the project lifespan, removal of the WTG monopile foundations and ESP piles at the time of decommissioning would result in a localized shift from a structure-oriented habitat

near the WTGs and ESP to the original shoal-oriented habitat present prior to construction. However, as the addition of the monopiles would be a minor addition to the hard substrate that was present prior to the construction of the Wind Park, the removal of the WTGs and ESPs will only have a minor effect on the overall habitat structure. Therefore, any whale that may have been attracted to the WTGs for feeding would have to forage in other locations within Nantucket Sound and surrounding waters, which has ample supplies of prey species.

3.1.1.6 EMF

Potential direct or indirect impacts to listed whales or sea turtles during the normal operation of the inner-array cables and the offshore transmission cable system circuits are expected to be negligible. The cable system (for both the inner-array cables and the offshore transmission cable system circuits) is a three-core solid dielectric AC cable design, which was specifically chosen for its minimization of environmental impacts and its reduction of any electromagnetic field. The proposed inner-array and the offshore transmission cable system will contain grounded metallic shielding that effectively blocks any electric field generated by the operating cabling system. Since the electric field will be completely contained within those shields, impacts are limited to those related to the magnetic field emitted from the offshore transmission cable system and inner-array cables. As presented in Report No. 5.3.2-3 of the DEIS, the magnetic fields associated with the operation of the inner-array cables or the submarine cable system are not anticipated to result in an adverse impact to marine mammals, sea turtles, or their prey (ICNIRP 2000; Adair, 1994; Valberg et al., 1997).

The research presented in the technical report on EMF indicates that although high sensitivity has been demonstrated by certain species (especially sharks) for weak electric fields, this sensitivity is limited to steady (DC) and slowly-varying (near-DC) fields. The proposed action would produce 60-Hz time-varying fields and no steady or slowly-varying fields. Likewise, evidence exists for marine organisms utilizing the geomagnetic field for orientation, but again, these responses are limited to steady (DC) and slowly-varying (near-DC) fields. 60-Hz alternating power-line EMF fields such as those that would be generated by the Project have not been reported to disrupt marine organism behavior, orientation, or migration. Based on the body of scientific literature examined, there are no anticipated adverse impacts expected from the undersea cables on the behavior, orientation, or navigation of marine organisms, including listed whale and sea turtle species (Report No. 5.3.2-3).

3.1.1.7 Proposed Action Impact Analysis- Routine Conditions

The major impact-producing factors affecting whales as a result of project activities include noise generated by construction and operational activities; vessel traffic; temporary reduced habitat; and degradation of water quality.

The main underwater acoustical impacts during construction activities will be limited to that generated by installation of the monopile foundations and vessel traffic.

Although vessel collisions are a primary cause of large whale mortality in the western North Atlantic, the proposed action is not expected to put whales at increased risk for vessel collisions.

Vessels moving at slower speeds (less than 14 knots [26 km/h]), such as the construction vessels to be used, are less likely to cause collisions (Laist et al., 2001). In addition, the vessel routes proposed to be used do not occur in areas where there have been high concentrations of whale sightings. Based upon the underdevelopment of whale prey species in Nantucket Sound, it is highly unlikely that whales would be migrating through, nursing, or feeding in Nantucket Sound. Therefore, the physical presence of vessels associated with the construction, operation, and decommissioning of the proposed action will not contribute to the harassment of migrating, nursing, or feeding humpback, fin, or right whales. As previously discussed, whales do not appear to be exceedingly disturbed by the physical presence and sound produced by vessels, and the vessel traffic itself (NMFS 2001; NMFS 2002). Whales should be able to detect and move away from vessels by diving into deeper waters. Therefore, the impacts of increased vessel traffic should have minor impacts on listed whales.

It is possible, yet difficult to predict, whether there will be increased fishing activity during operation of the proposed action. Such fishing efforts will mainly be by private and recreational charter boats using hook and line fishing gear, which should not adversely impact any whale or dolphin species.

Construction and decommissioning are not anticipated to result in changes in whale prey abundance or distribution. Some temporary displacement may occur during periods of noise or high suspended sediments, but this will be limited to areas directly surrounding the given activities, causing both prey species and whales moving to an undisturbed area. Pelagic prey tends to be highly variable and animals foraging on these sources move with the food source, as seen with many whales and their prey species. Any temporary disturbance to pelagic prey is likely to mimic typical temporal and spatial variability, and is likely available in other areas of Nantucket Sound and surrounding waters for foraging by whales. Construction is therefore anticipated to have minor impacts on whales in regards to reduced habitat and prey availability.

As discussed previously in construction impacts, there is little potential for whales to bioaccumulate chemical contaminants in their tissue from consuming prey within the proposed action area. The suspension of the sediments due to decommissioning activities is not anticipated to increase the amount of contaminants found within lower trophic levels.

3.1.1.8 Summary and Conclusion – Routine Activities

Routine activities associated with construction, operation and maintenance, and decommissioning will have minor impacts on the whale species that may be found in the proposed action area. Some temporary avoidance of the area may occur due to elevated acoustic, vessel harassment, and degraded water quality however this should be short-lived and normal conditions are expected to resume once construction and decommissioning activities have ceased. During operations, there will little disturbance of the habitat, and maintenance vessels will generally travel at less than 14 knots, thereby reducing the potential for vessel strikes.

3.1.2 Whales-Non-routine or Accidental Events

Accidental and unexpected events associated with the proposed action could impact whales. Such impacts would primarily be the result of oil spills, but may also relate to cable repair, collapse of a monopile, vessel collision, and geotechnical and geophysical investigations.

3.1.2.1 Oil Spills

Oil spills could occur either as a release from the ESP storage tank or from a vessel collision with a monopile. Little species-specific information is available regarding the effects of oil spills on whales. Past studies suggest that large whale species do not seem to be particularly sensitive to oil spills. As discussed in Section 4.2.4 of this appendix, a study of fin whales found the whales swimming in an oil slick on Nantucket Shoals following the spill of almost eight million gallons of fuel oil. None of the whales observed showed any obvious signs of distress in the short term (Grose and Mattson 1977). Another study reported fin whales surfacing in heavy slicks following a spill of fuel oil southeast of Cape Cod with no apparent adverse impacts (Goodale et al., 1981).

Because they rely on blubber for insulation, whales are less vulnerable to oil spills than fur-coated marine mammals which can die from hypothermia when coated in oil. In addition, humpback whales, fin whales and right whales are all migratory which may limit their exposure to a persistent oil slick in a small geographic area. Of the three listed whale species, the right whale population should be considered at greatest risk to being negatively impacted by an oil spill because of the small population size and slow recovery of their numbers from earlier depletion from whaling. Pollution from various anthropogenic sources has been suggested as one possible cause of the slow right whale recovery. The right whale population may be more vulnerable to long-term oil spill effects than humpback and fin whales whose larger numbers provide greater resiliency to their populations.

Although most research suggests that whales do not appear to be especially sensitive to spills, other studies have shown that there are negative long-term effects to whales from exposure to oil. Direct mortality as a result of contact with oil and development of brain lesions were reported after the Exxon Valdez Spill in Alaska. When surfacing, oil may irritate whale's eyes and skin and they may breathe in harmful fumes. Other symptoms of acute exposure to oil include lethargy, poor coordination and difficulty breathing which can lead to drowning (Hammond et al., 2001). However, the case of the Exxon Valdez should be considered an extreme example which represented a spill much larger than any worst-case scenario from the proposed action. Given the probable infrequency of listed whales occurring in the vicinity of Horseshoe Shoalls, and the low probability of a large oil spill, the potential adverse affects from an oil spill event due to a vessel colliding with a project component is considered negligible.

3.1.2.2 Cable Repair

Many of the types of disturbances that would occur during cable repair activities are smaller and shorter duration, but of similar type, to those that would occur during cable installation. A relatively short distance along the sea floor would be disturbed by the jetting process used to

uncover the cable and allow it to be cut so that the ends could be retrieved to the surface. In addition to the temporary loss of some benthic organisms, there would be increased turbidity for a short period, and a localized increase in disturbance due to vessel activity, including noise and anchor cable placement and retrieval. Given the small area, short duration, and infrequency of occurrence of listed whales in the project area, potential adverse impacts from cable repair activities on the listed whales would be negligible.

3.1.2.3 Vessel Collision with Monopile

The extent of potential impacts that could result from a vessel collision with a monopile largely depend on the extent of damage to the monopile or vessel, as well as the nature of the vessel. Some smaller vessels would merely strike a glancing blow and suffer some hull damage but not sink. Other vessels may suffer enough damage to sink, causing a small release of fuel and debris. A larger vessel, such as an oil tanker, would most likely cause a collapse of the monopile, also resulting in a small release of lubricating fluid. If oil being transported were to be released, then depending upon the quantity released, an oil spill that escapes Nantucket Sound could directly affect listed whales (see section 5.1.2.1). Repair of a damaged or collapsed monopile would create short term and localized disturbances to the benthos, water column, and pelagic organisms similar to the construction and decommissioning of a single monopile, albeit in reverse order and combined in a single event. Since these disturbances are localized to the monopile they are unlikely to adversely affect listed whale species, and therefore potential adverse impacts resulting from a vessel collision with a monopile and the associated repair activities on the listed whales would be negligible.

3.1.2.4 Geotechnical and Geophysical Investigations

Many of the types of disturbances that would occur during the geotechnical and geophysical investigations are short term and very localized. A very small area of the sea floor would be disturbed by coring activities, either at the core hole or associated with the coring vessel anchor placements. It is likely that the duration of activity at any one coring location would be no more than a few days. The geophysical survey work uses mobile gear towed behind a vessel, and would not result in bottom disturbance, nor does it result in activity at a fixed location. The geotechnical investigations would result in a negligible temporary loss of some benthic organisms, and a localized increase in disturbance due to vessel activity, including noise and anchor cable placement and retrieval. Given the small area of disturbance, short duration of activities, and infrequency of occurrence of listed whales in the project area, potential adverse impacts from geotechnical and geophysical investigations to the listed whales would be negligible.

3.1.2.5 Proposed Action Analysis-Non-Routine or Accidental Scenarios

While improbable, an oil spill would have moderate to major impacts on whales within Nantucket Sound. However, if such an event occurs the whales within the vicinity of the spill will be impacted.

As they rely on blubber for insulation, whales are less vulnerable to oil spills than fur-coated marine mammals, such as seals. Whales are most vulnerable to oil spills when they are surfacing for air when skin and eyes can be irritated. Direct exposure to oil spills can result in the inhalation of harmful fumes, lethargy, poor coordination and difficulty breathing which can lead to drowning (Hammond et al., 2001). Migratory whales may limit their exposure to a persistent oil slick in a small geographic area by avoiding that area. Of the three listed large whale species, the right whale population should be considered at greatest risk to being negatively impacted by an oil spill because of the small population size and slow recovery of their numbers from earlier depletion events.

The other potential non-routine or accidental events that have been evaluated all have very localized and short term affects on the habitat, and when combined with the low probability of a listed whale species occurring in the project area when one of these activities is occurring, the potential impacts to listed whales is negligible.

3.1.2.6 Summary and Conclusion – Non-routine Conditions

Non-routine or accidental activities associated with the construction, operation and maintenance, and decommissioning will have negligible impacts on the whale species that may be found in the project area. Either the activity is so short term or localized, or is unlikely to occur, that alteration of habitat or direct affects on listed whale species are unlikely to occur, or if they do occur, would have essentially no adverse affects.

3.2 Sea Turtles

3.2.1 Routine Activities

The major impact-producing factors affecting sea turtles as a result of proposed action include noise associated with construction and operation activities, vessel traffic, changes in water quality and the establishment of "fouling" communities on monopiles.

3.2.1.1 Construction and Operational Noise

The Applicant conducted short-term noise level measurements of underwater noise at Buoy G5 in the North Shipping Channel and at Buoy R20 at the edge of the Main Channel. Measured L_{eq} underwater sound levels were found to be 90 dB and 93 dB at Buoys G5 and R20, respectively. The sound level at Buoy R20 is slightly higher due to the shallower water and greater current. The depth at this location is also more representative of the water depth on Horseshoe Shoal, and accordingly, the Buoy R20 data were used as a baseline for the proposed action.

Underwater sound levels with higher wind speeds (as would occur when the turbines are operating) would be higher. Studies conducted in other coastal water areas indicate that the sound level increases 7.2 dB per doubling of wind speed. Accordingly, the estimated underwater L_{eq} sound level for the design wind speed would be 107.2 dB.

Underwater sound level measurements conducted on Horseshoe Shoals at the site of the Scientific Measurement Devices Station (SMDS) at the time that three support piles were driven into the seabed were also utilized for this study. The weather conditions at the time of the measurements included moderate winds of less than 12 mph (5.4 m/s) and moderate seas of less than 5 ft (1.5 m). The measured existing underwater L_{max} level (without pile driving) was 123 dB.

Once installed, the operation of the WTGs is not expected to generate substantial sound levels above baseline sound in the area. Existing underwater sound levels for the design condition are 107.2 dB. The calculated sound level from operation of a WTG is 109.1 dB at 65.6 ft (20 m) from the monopile (i.e., only 1.9 dB above the baseline sound level), and this total falls off to 107.5 dB at 164 ft (50 m) and declines to the baseline level at a relatively short distance of 360 ft (110 m).

An analysis of predicted underwater sound levels perceived by sea turtles from operation show that no injury or harassment to sea turtles are predicted even if an individual were to approach as close as 65.6 ft (20 m) to a monopile when the turbines are operating at the design wind speed as all dB_{ht} values at this minimum distance are well below 90 dB. In fact, operational noise will be inaudible for sea turtles. Therefore, no behavioral effects to sea turtles are anticipated even if an individual were to approach within 65.6 ft (20 m) of the structures. Construction and operational noise levels will result in negligible impacts on sea turtles in Nantucket Sound.

3.2.1.2 Vessel Traffic

Although sea turtles are likely to dive at the approach of a vessel, they are still at risk of boat-related injuries. Between 1987 and 1993, up to 17 percent of all stranded sea turtles on the U.S. Atlantic coast had boat-related injuries (Teas 1994a, b). Ship strikes appear to be a significant source of mortality for sea turtles, and vessel-related injuries have increased in recent years (Teas 1994a, b). However, vessels moving at slower speeds, such as those associated with construction, are less likely to cause collisions (NMFS 2001; NMFS 2002). In addition, sea turtles present in Nantucket Sound are likely to be foraging and their feeding behaviors may also reduce the risk of collisions. While feeding, these turtles spend most of their time submerged. Ridleys and loggerheads can spend more than 57 minutes of each hour submerged (Thompson 1988) and between 25 and 58 percent of their time is directly on the bottom (Standora et al., 1994). Feeding dives last from about four minutes to as long as two hours (Renaud and Carpenter 1994). During these long periods of submergence, loggerhead and ridley turtles are not particularly vulnerable to collisions with barges.

It is possible that some increased fishing effort could occur after the Wind Park is operational, but that is difficult to predict. It is not likely that increased trawling activity would occur after construction of the monopile structures since the fish attracted to these structures would tend to remain fairly close to each monopile. For safety reasons and to protect their gear, trawlers would not want to deploy their gear immediately next to a monopile. Trawlers would, however be able to continue trawling in the general vicinity and between the monopiles leaving enough room to safely navigate their vessel and gear.

If there is increased fishing effort, it is more likely to consist of private and charter recreational boats. It is true that this could result in increased fishing effort and boat traffic which may increase the risk of boat collisions and/or impacts from fishing gear to sea turtles. However, recreational fishing gear is likely to consist primarily of hook and line which should not adversely impact any sea turtles that have the potential to occur in the proposed action area.

Loggerhead and Kemp's ridley sea turtles could be attracted to the monopile foundations for food and shelter. Any sea turtles that may be attracted to the Wind Park area are likely to remain near each monopile except for the times transiting the Wind Park. While close to the monopile, they are less likely to be subject to vessel interaction since prudent vessel captains would reduce speeds when approaching a monopile. It is possible that sea turtles could be at risk of interaction with vessels while transiting from one place to the next within the Wind Park; however, this risk should be similar to risks that turtles face throughout Nantucket Sound.

3.2.1.3 Effects on Sea Turtle Behaviors

While project vessel activity has the potential to cause alteration of sea turtle behaviors, the likely incidence of the co-occurrence of vessels and sea turtles is likely to be rare. Since the Nantucket Sound region is near the northern limit of the ranges for the sea turtles (except leatherback), nesting does not occur, and the sea turtles are not present during winter months. Elevated turbidity levels may cause an avoidance response around cable jetting activities, should a turtle be in the vicinity. Otherwise, proposed action activities are likely to only have minor affects on sea turtle behaviors.

3.2.1.4 Habitat Shift

Activities related to construction may cause a temporary reduced availability of habitat for sea turtles in the vicinity of the construction equipment. The main anticipated impact would be avoidance of areas of high traffic or the immediate vicinity of active construction. Construction is not anticipated to result in permanent changes in sea turtle prey abundance or distribution. Some temporary displacement may occur during periods of noise or high suspended sediments, but this will be limited to areas directly surrounding the given activities, causing both prey species and sea turtles moving to an undisturbed area. Benthic habitat loss due to construction activities may cause mortality to benthic organisms in the area, but similar benthic communities are found throughout Nantucket Sound, enabling sea turtles to find suitable prey in other areas.

The presence of 130 monopile foundations, 6 ESP piles and their associated scour control mats or rock armor in Nantucket Sound has the potential to shift the area immediately surrounding each monopile from a soft sediment, open water habitat system to a structure-oriented system, with potential localized changes to benthic feeding sea turtle prey, namely the establishment of "fouling communities" within the Wind Park and an increased availability of shelter among the monopiles.

The WTG monopile foundations will represent a source of new substrate with vertical orientation in an area that has a limited amount of such habitat, and as such may attract finfish

and benthic organisms, potentially affecting sea turtles by causing changes to prey distribution and/or abundance. While the aggregation of finfish around the monopiles will not attract sea turtles, some sea turtle species may be attracted to the WTGs for the fouling community and epifauna that may colonize the monopiles as an additional food source for certain sea turtle species, especially loggerhead and Kemp's ridley turtles. All four species may be attracted to the monopiles for shelter, especially loggerheads that have reported to commonly occupy areas around oil platforms (NRC 1996).

Some of the sea turtle species however could be attracted to the WTGs for food or shelter. Potential finfish aggregations at the monopiles are not likely to affect foraging sea turtles; however, the benthic invertebrates that may attach to the monopiles could provide an additional food source for certain sea turtle species. Loggerheads and Kemp's ridleys could be attracted to the monopiles to feed on attached organisms since they feed on mollusks and crustaceans. According to USFWS (2005), loggerheads are frequently observed around wrecks, underwater structures and reefs where they forage on a variety of mollusks and crustaceans. Leatherback turtles and green turtles however should not be attracted to the monopiles for feeding since leatherbacks are strictly pelagic and feed from the water column primarily on jellyfish (OBIS-SEAMAP 2002) and green turtles are primarily herbivores feeding on seagrasses and algae. In addition, green turtles are much more likely to be found in shallow warmer waters and are not expected to frequent the Nantucket Sound area with any regularity. All four species of sea turtles have been observed around oil platforms, especially loggerheads which are reportedly the most common species sighted around oil platforms and have been observed sleeping under platforms or next to support structures (NRC 1996). Kemp's ridley turtles, however appear to prefer more sheltered areas along the coast or in estuaries, bays and lagoons (FWIE 1996). Therefore, although it is possible that any of the four sea turtle species could be attracted to the monopiles for shelter, the loggerhead is the most likely species to be attracted to the structures for both food and shelter.

Although the monopile foundations would create additional attachment sites for benthic organisms that require fixed (non-sand) substrates and additional structure that may attract certain finfish species, the additional amount of surface area being introduced (approximately 1,200 square feet (111 square meters) per tower, assuming an average water depth of 30 feet (9.1 m) below mean high water (MHW)) would be a minor addition to the hard substrate that is already present (see Section 3.9 of ESS, 2007). Due to the small amount of additional surface area in relation to the total proposed action area and Nantucket Sound and the spacing between WTGs (0.34 to 0.54 nautical miles (0.63 to 1.0 km) apart), the new additional structure is not expected to affect the overall environment, benthic community composition, finfish species composition, or populations of foraging sea turtles in the area.

At the end of the project's lifespan, removal of the WTG monopile foundations and ESP piles at the time of decommissioning would result in a localized shift from a structure-oriented habitat near the WTGs and ESP to the original shoal-oriented habitat present prior to construction. However, as the addition of the monopiles would be a minor addition to the hard substrate that was present prior to the construction of the Wind Park, the removal of the WTGs and ESPs will not cause a great impact in the overall habitat structure. Therefore, sea turtle

populations that consume colonizing benthic invertebrate prey are not likely to increase due solely to the presence of the monopiles and hence would not be greatly affected by their removal.

3.2.1.5 Water Quality

An increase in the total suspended solids (TSS) within the water can impact the foraging abilities of the sea turtles, decreasing the visibility of prey species. The suspension of solids produced by construction are expected to be temporary and localized, as the pile driving hammer and jet plow technology that will be used to install the monopile foundations and the submarine cables, respectively, were selected specifically for their ability to keep sediment disturbance to a minimum. Further, the physical composition of the sands and the physical characteristics of the sound environment provide reason to believe that any localized turbidity will settle back to the sea floor within a short period of time (one to two tidal cycles). Simulations of sediment transport and deposition demonstrate that jet plow embedment operations will result in a sediment plume below 50 mg/L, and would settle in less than 2 to 3 hours. Within Lewis Bay, suspended sediments are expected to remain in suspension for longer periods due to the weak tidal currents, with a plume in excess of 100 mg/L remaining for 2 to 6 hours depending on location and period of cycle. Decommissioning-related impacts will be short-term and localized and are expected to be similar to or less than impacts during construction.

3.2.1.6 EMF

Potential direct impacts to listed sea turtles during the normal operation of the inner-array cables and the offshore transmission cable system circuits are expected to be negligible. The cable system (for both the inner-array cables and the offshore transmission cable system circuits) is a three-core solid dielectric AC cable design, which was specifically chosen for its minimization of environmental impacts and its reduction of any electromagnetic field. The proposed inner-array and offshore transmission cable system will contain grounded metallic shielding that effectively blocks any electric field generated by the operating cabling system. Since the electric field will be completely contained within those shields, impacts are limited to those related to the magnetic field emitted from the offshore transmission cable system and inner-array cables. As presented in Section 5.3.2.9.2 in ACOE, 2004 and Report No. 5.3.2-3 the magnetic fields associated with the operation of the inner-array cables or the offshore transmission cable system are not anticipated to result in an adverse impact to sea turtles or their prey (ICNIRP 2000; Adair, 1994; Valberg et al., 1997).

The research presented in the technical report on EMF indicates that although high sensitivity has been demonstrated by certain species (especially sharks) for weak electric fields, this sensitivity is limited to steady (DC) and slowly-varying (near-DC) fields. The proposed action would produce 60-Hz time-varying fields and no steady or slowly-varying fields. Likewise, evidence exists for marine organisms utilizing the geomagnetic field for orientation, but again, these responses are limited to steady (DC) and slowly-varying (near-DC) fields. 60-Hz alternating power-line EMF fields such as those that would be generated by the proposed action have not been reported to disrupt marine organism behavior, orientation, or migration. Based on the body of scientific literature examined, there are no anticipated adverse impacts expected from the undersea cables on the behavior, orientation, or navigation of sea turtles (Report No. 5.3.2-3).

Because the inner-array cables and the offshore transmission cable system circuits connecting the Wind Park to the landfall will be buried approximately 6 feet (1.8 m) below the seabed, they will not pose a physical barrier to fish passage. The considerable depth to which the cables will be buried will allow benthic organisms to colonize and demersal fish species to utilize surface sediments without being affected by the cable operation. The burial depth of the cables also minimizes potential thermal impacts from operation of these cable systems. In addition, the inner-array and offshore transmission cable system utilize solid dielectric AC cable designed for use in the marine environment that does not require pressurized dielectric fluid circulation for insulating or cooling purposes. There will be no direct impacts to the listed sea turtle species during the normal operation of the inner-array or offshore transmission cable systems. . There will also be no impacts to finfish, invertebrate, or plankton prey species of sea turtles (indirect impact) during normal operations.

3.2.1.7 Proposed Action Analysis – Routine Activities

An analysis of predicted underwater sound levels perceived by sea turtles from operation show that no injury or harassment to sea turtles are predicted even if an individual were to approach as close as 65.6 ft (20 m) to a monopile when the turbines are operating at the design wind speed as all dB_{ht} values at this minimum distance are well below 90 dB. In fact, operational sounds will be inaudible for sea turtles. Therefore, no behavioral effects to sea turtles are anticipated even if an individual were to approach within 65.6 ft (20 m) of the structures. Proposed action operations will result in negligible impacts on sea turtles in Nantucket Sound.

As previously discussed, sea turtles do not appear to be exceedingly disturbed by the physical presence and sound produced by vessels, and the vessel traffic itself (NMFS 2001; NMFS 2002). Sea turtles should be able to detect and move away from vessels by diving into deeper waters. Any impact will be limited to temporary avoidance of an area; however this is unexpected due to the high volumes of vessel traffic that normally travel the waters of Nantucket Sound. Therefore, the impacts of increased vessel traffic should have minor impacts on listed sea turtles.

Activities related to construction may cause a temporary reduced availability of habitat for sea turtles in the vicinity of the proposed action. Construction is not anticipated to result in permanent changes in sea turtle prey abundance or distribution. Some temporary displacement may occur during periods of noise or high suspended sediments, but this will be limited to areas directly surrounding the given activities, causing both prey species and sea turtles moving to an undisturbed area. Benthic habitat loss due to construction activities may cause mortality to benthic organisms in the area, but similar benthic communities are found throughout Nantucket Sound, enabling sea turtles to find suitable prey in other areas.

Sea turtles that forage within the area of Nantucket Sound are naturally accustomed to substantial amounts of suspended sediment on a regular basis, from storms and strong tidal currents, and should be minimally impacted by a temporary increase in turbidity associated with construction and decommissioning activities, including the sea turtles that may inhabit or forage within Lewis Bay. Further, sea turtles are mobile and can move away from any disturbance,

including any increases in suspended sediments. The impacts of increased turbidity on the foraging abilities of sea turtles are expected to be minor.

3.2.1.8 Summary and Conclusion – Routine Activities

Routine activities associated with construction, operation and maintenance, and decommissioning will have minor impacts on the sea turtles species that may be found in that area. Some temporary avoidance of the proposed action area may occur due to elevated acoustic, vessel harassment and decreased water quality, however this should be short-lived and normal conditions are expected to resume once construction and decommissioning activities have ceased. During operations, very few activities would occur that could have long term or extensive effects on the sea turtles using the area. The one affect that may result in attraction of benthic feeding sea turtles to the area is the creation of hard substrate fouling and macroninvertebrate community on the rock armor and monopiles. This could result in increased feeding opportunities for the loggerhead and Kemp's ridley sea turtle species.

3.2.2 Sea Turtles-Non-routine and Accidental Events

Accidental, unexpected industrial events associated with the proposed action could impact sea turtles. Such impacts would primarily be the result of oil spills, cable repair, vessel collisions, and geotechnical and geophysical investigations.

3.2.2.1 Oil Spills

Compared with whales, there is greater concern of the impact of oil/chemical spills on sea turtles. Sea turtle populations are in decline world-wide and studies suggest that they are more sensitive to oil spills than whales. Turtles can be harmed if they surface in an oil slick to breathe. Oil can affect their eyes and damage airways or lungs. Sea turtles can also be affected by oil through contamination of the food supply or by absorption through the skin. In addition, sea turtles are vulnerable through all life stages. Other aspects of sea turtle biology including their lack of avoidance behavior in regard to oil slicks, indiscriminate feeding in convergence zones and large predive air inhalations are all reasons for their extra sensitivity to oil/chemical spills (Shigenaka et al., 2003).

Existing research suggests that sea turtle eggs, embryos and hatchlings are more vulnerable to oil than adults. However, none of these life stages occur in Nantucket Sound. Though less vulnerable than earlier life stages, research suggests that adult sea turtles are still quite sensitive to oil/chemical spills. A study of loggerhead turtles found that they showed no avoidance behavior when an encountering oil slicks. They also seemed unable to distinguish between food and tar balls. Because they inhale large volumes of air before diving and resurface regularly, the turtles are exposed to oil vapors for longer periods of time. Research on the loggerheads also found that any oil ingested was retained for several days, leading to greater risk to internal organs.

There are very few laboratory studies that investigate the impact of oil on adult sea turtles. One such study conducted by the Minerals Management Service in 1986 on loggerhead turtles

found the turtles' health were adversely affected by both chronic and acute exposures to crude oil (Lutcavage et al., 1995). Results showed that the turtles' skin sloughed off, with inflamed, abnormal and dead cells. It took several weeks for the turtles to recover which led to an increased risk of infection. Exposure to oil also led to changes in blood chemistry with a decrease in the volume of red blood cells and an increase in white blood cell production. In addition, the turtles in the study did not display any avoidance behavior towards oil. Another study found similar results with cell abnormalities in the skin, alteration of respiratory patterns and blood cell dysfunctions in turtles exposed to crude oil (Lutz and Lutcavage 1989). In addition, the research found that the turtles had ingested oil and it appeared in their feces. Ingestion could have physiological effects that could be fatal.

Overall, there is very little species-specific information on the impact of oil to sea turtles. However, because of its small population size and limited nesting distribution, Kemp's Ridley sea turtle is considered to be especially vulnerable to oil/chemical spills (Lutz and Lutcavage 1989). In summary, existing literature suggests that sea turtles are especially sensitive to oil/chemical spills. However, the type of oil, length of exposure, condition of the oil in terms of weathering, and life stage at which sea turtles are exposed all play a role in the impact to the species. While the probability of occurrence of a large oil spill is very small, should one occur during the season that sea turtles are present in Nantucket Sound, the potential impacts would be moderate, requiring implementation of special procedures to reduce potential harm to sea turtles.

3.2.2.2 Cable Repair

Many of the types of disturbances that would occur during cable repair activities are smaller and shorter duration, but of similar type, to those that would occur during cable installation. A relatively short distance along the sea floor would be disturbed by the jetting process used to uncover the cable and allow it to be cut so that the ends could be retrieved to the surface. In addition to the temporary loss of some benthic organisms, there would be increased turbidity for a short period, and a localized increase in disturbance due to vessel activity, including noise and anchor cable placement and retrieval. Given the small area, short duration, and low probability of a cable repair occurrence, potential adverse impacts from cable repair activities on the listed sea turtles would be negligible.

3.2.2.3 Vessel Collision with Monopile

The extent of potential impacts that could result from a vessel collision with a monopile largely depend on the extent of damage to the monopile or vessel, as well as the nature of the vessel. Some smaller vessels would merely strike a glancing blow and suffer some hull damage but not sink. Other vessels may suffer enough damage to sink, causing a small release of fuel and debris. A larger vessel, such as an oil tanker, would most likely cause a collapse of the monopile, also resulting in a small release of lubricating fluid. If oil being transported were to be released, then depending upon the quantity released, an oil spill could directly affect listed whales (see section 5.2.2.1). Repair of a damaged or collapsed monopile would create short term and localized disturbances to the benthos, water column, and pelagic organisms similar to the construction and decommissioning of a single monopile, albeit in reverse order and combined in a single event. Other than the oil spill scenario, since these disturbances are localized to the

monopile they are unlikely to adversely affect listed sea turtle species, and therefore potential adverse impacts resulting from a vessel collision with a monopile and the associated repair activities on the listed sea turtles would be negligible.

3.2.2.4 Geotechnical and Geophysical Investigations

Many of the types of disturbances that would occur during the geotechnical and geophysical investigations are short term and very localized. A very small area of the sea floor would be disturbed by coring activities, either at the core hole or associated with the coring vessel anchor placements. It is likely that the duration of activity at any one coring location would be no more than a few days. The geophysical survey work uses mobile gear towed behind a vessel, and would not result in bottom disturbance, nor does it result in activity at a fixed location. The geotechnical investigations would result in a negligible temporary loss of some benthic organisms, and a localized increase in disturbance due to vessel activity, including noise and anchor cable placement and retrieval. While the towed gear has the potential to result in interaction with sea turtle, the speed of towing, typically about 1 knot, minimizes the potential for entanglement or vessel strikes. Given the small area of disturbance, short duration of activities, and slow speed of mobile surveys, potential adverse impacts from geotechnical and geophysical investigations to the listed sea turtles would be negligible.

3.2.2.5 Proposed Action Analysis – Non-routine or Accidental Condition

While improbable, an oil spill would have moderate impacts on sea turtles within Nantucket Sound. The type of oil, length of exposure, condition of the oil in terms of weathering and life stage at which the sea turtle is exposed to the spill will all play a role in the impact on the animal. While some oil products will be present within the WTG structures, the amount of oil being used will lead to less severe impacts in the event of a spill. In general, researchers concluded that as oil weathers in the marine environment over time, its toxic effects on sea turtles decreases. The negative effects on sea turtles discussed above likely represent a worst-case scenario based on the impact from a large, fresh oil spill which reaches turtle breeding areas on shore. In the case of the proposed action, the amount of oil being used and distance to shore would most likely lead to less severe impacts than described above in the event of a spill. In addition, turtle breeding areas are located well south of Nantucket Sound; therefore, no breeding areas or early life stages would be affected by a potential spill.

3.2.2.6 Summary and Conclusion – Non-routine or Accidental Condition

Activities associated with construction, operation and maintenance, and decommissioning that are non-routine or accidental will have minor impacts overall on the sea turtle species that may be found in the area. The unplanned event that may have the greatest adverse effect on sea turtle species includes the unlikely chance of an oil spill, in which case impacts could be moderate.

3.3 Birds

Piping Plover

Potential piping plover impacts associated with construction, operation and maintenance, and decommissioning could include loss of habitat; disturbances associated with the presence or activity of construction equipment; disturbances such as barriers to flight paths due to the presence of the turbines; and the risk of collision with WTG structures. Impacts associated with decommissioning activities are expected to be similar to or less than construction activities. Individuals that could potentially be impacted include those from the New England and Atlantic Canada subpopulations, as well as a few interior breeding birds that migrate east before migrating south to locations along the Atlantic Coast.

There are potential impacts that could occur as a result of non-routine, accidental, or unplanned events including oil spills or cable repair activities.

Roseate Tern

Potential impacts to roseate terns could occur from routine activities during construction, operation and maintenance, and decommissioning including short or long-term habitat loss or modification; disturbances due to the presence and maintenance of the WTGs; risk of collision; increased predation; and increased human activity such as increased vessel traffic. In addition, there are potential impacts that could occur as a result of non-routine, accidental, or unplanned events associated with oil spills, monopile collapse, cable repair, and geotechnical and geophysical investigations.

3.3.1 Routine Activities

3.3.1.1 Piping Plover

3.3.1.1.1 Habitat Loss

Habitat loss or alteration associated with construction/decommissioning or operation is not anticipated. The proposed wind turbine generators (WTGs) would be located offshore, at least 5 mi (8 km) from the nearest nesting or staging habitat (Figure C-8). The proposed landfall of the offshore transmission cable system would not occur within breeding habitat. The proposed action would not impact critical habitat as there are no designations in Massachusetts.

The proposed location of the landfall of the offshore transmission cable system is on the northeastern side of Lewis Bay at the end of New Hampshire Avenue in Yarmouth. Neither the proposed cable or landfall would cross piping plover breeding habitat. The closest nesting location to the proposed landfall is approximately 1.5 mi (2.4 km) at Kalmar Beach/Dunbar Point in Hyannis. The closest distance of the offshore transmission cable system to the nearest piping plover nest site on the seaward side of Great Island is 0.8 mi (1.3 km) (Figure C-8). The buried cables at their closest point would occur approximately 820 ft (250 m) from Kalmar Point/Dunbar Beach and approximately 1,210 ft (369 m) from Great Island. In addition, since the shoreline would be drilled under for cable placement, there would be no disturbance of beach areas.

Loss of foraging habitat is not anticipated as the majority of foraging during the breeding season takes place in proximity to nesting locations. Loss of pre-migratory staging or foraging habitat is not anticipated as the landfall would not occur in proximity to reported staging areas (USFWS 1996).

3.3.1.1.2 Disturbance

High disturbance levels around nest sites can result in the abandonment of nests, and ultimately, decreased breeding success. Causing parents or juveniles to flush while foraging may stress juveniles enough to negatively influence critical growth and development. Potential disturbances during construction and decommissioning associated with increased human activity, the presence and operation of large equipment, and increased boat traffic offshore of nesting sites located closest to the proposed landfall, would be temporary and are not anticipated to impact breeding piping plover. It is possible that a tracking system consisting of a wire, for the operation of the drill head may be placed across the beach. This would be a minor, temporary activity that would not disturb the area more than a person walking on the beach.

Available data on disturbance distances suggest that flushing distances of incubating birds vary among sites and individuals. Disturbances resulting in flushing occurred as far away from nests as 689 ft (210 m), 984 ft (300 m), and 571 ft (174 m) at Nova Scotia, Virginia, and Maryland beaches, respectively (USFWS 1996). The recommended disturbance buffer around nest sites is typically a 164 ft (50 m) buffer; however, at Maryland sites it is 738 ft (225 m) (USFWS 1996). The mean flushing distance at Massachusetts nest sites is 24 m (USFWS 1996). For non-incubating birds, the maximum disturbance distances reported for pedestrian, vehicles, pets, and kites are 197 ft (60 m), 230 ft (70 m), 328 ft (100 m), and 394 ft (120 m), respectively (USFWS 1996).

Due to the 820 ft (250 m) (or greater) separation of the offshore transmission cable system from the nearest nesting beaches, disturbances associated with offshore construction or operation activities are not anticipated for nesting piping plover. In addition, since the shoreline would be drilled under for cable placement, there would be no disturbance of beach areas. The placement of a wire on the beach (and seafloor) to help guide and track the drill head would result in disturbance essentially equal to a person walking on the beach. The proposed landfall site is 1.5 mi (2.4 km) from the nearest nesting beach and, therefore, onshore construction or decommissioning activities are not anticipated to impact nesting piping plover.

3.3.1.1.3 WTG Presence and Rotor Movement

Breeding Season

During the breeding season, piping plover remain in close proximity to nests as they forage on invertebrates in the inter-tidal zone near nest sites. During this period, plovers mainly travel by walking or running between foraging and nearby breeding sites. Their regular daily movements would not result in crossings of the proposed action area. However, there have been some observations of plovers during the breeding season departing land and heading for the horizon. There was a potential that these observed departures resulted in crossings of Nantucket

Sound (Report No. 4.2.9-1). There are no known flight corridors for plovers over the Sound during the breeding season. There are no topographical features such as shortest crossings that would direct occasional flights over the Sound into HSS. Therefore, the presence and operation of turbines is not anticipated to present a major barrier to the flight paths of piping plover.

Other unusual crossings of Nantucket Sound during the breeding season could include the crossings of failed breeders or unpaired birds seeking alternate habitat or a mate. However, aerial and boat surveys conducted in 2002, 2003 and 2004 in Nantucket Sound did not detect such movements in any of the study areas. There is no data available that suggest piping plover would cross HSS during such movements; therefore, the WTGs are not anticipated to create a major barrier to the flight paths of piping plover.

Migration and Dispersal

The majority of Atlantic Coast piping plover migratory movements is believed to take place along the outer beaches of the coastline (USFWS 1996). Most movements are believed to occur along a narrow flight corridor, and offshore and inland observations are rare (USFWS 1996). Some birds may occur inland or offshore if blown off course by weather events. The birds that breed or stop-over on islands in Nantucket Sound and Vineyard Sound would make over-water crossings while accessing these locations. Therefore, there is a potential that piping plover could occur in the proposed action area during migratory or post-breeding dispersal movements. However, there are no topographical features that would funnel piping plover through HSS. Therefore, the presence of the WTGs is not expected to present a major barrier to the flight path of migrating piping plover.

3.3.1.1.4 Risk of Collision

The potential exists for piping plover to collide with WTG structures, including the blades and tubular towers during the breeding and migration periods. The results of available mortality studies indicate that the majority of collisions with man-made structures take place at night during periods of inclement weather (Kerlinger 2000). Birds that fly within the rotor zone of the proposed turbines (75.5 - 440 ft [23 - 134 m]) during periods of low visibility would be at greatest risk of collision.

Breeding Season

The risk of collision of piping plover with the proposed WTG structures during the breeding season is low. Daytime aerial and boat surveys conducted in 2002, 2003 and 2004, during the timeframe when plover are breeding in the area, reported no observations of piping plover in any of the Nantucket Sound study areas. Few crossings of Nantucket Sound are expected during the breeding season as plovers are mainly sedentary and make small scale movements between nesting and foraging locations along the beach. Regular daily movements would not result in crossings of Nantucket Sound. The exception would be occasional crossings of Nantucket Sound as individuals access alternate nesting or foraging areas. Other unusual crossings could be conducted by failed nesters or unpaired individuals traveling between the mainland and Nantucket or Martha's Vineyard in search of habitat or a mate. More information would be useful in determining with a high level of certainty the likelihood of piping plover over-water movements during the breeding season.

Potential crossings of Nantucket Sound during the breeding season would be sporadic. There are no topographical features that would funnel piping plover through HSS if they were to cross Nantucket Sound. Therefore, the potential of piping plover encountering a WTG is low. The risk of collision during a crossing of the project area depends on the height of flight. During the breeding season, piping plover have been documented flying low over the water (or adjacent land) while accessing foraging locations, typically less than 33 ft (10 m), but sometimes higher (Report No. 4.2.9-1). These observations included departures toward land, beyond the horizon, that may have resulted in crossings of Nantucket Sound (Report No. 4.2.9-1).

Hatch and Brault (2007) (Report No. 5.3.2-1) estimated the number of piping plover turbine encounters per crossing of the wind turbine array, assuming all turbines were aligned perpendicular to each bird's path, based on three different flight height scenarios: If all individuals fly below 30 m, the expected encounters per crossing would be 0.07; if all birds fly in the rotor swept zone (75.5 - 440 ft [23-134 m]), there would be 0.67 encounters; and if flights are evenly distributed from 30 to 600 m then there would be 0.13 encounters. The authors suggest that, based on high avoidance rates estimated for other species, the likelihood of collision resulting from encounters is low (see Collision Probability Modeling for description of avoidance rates). The authors assume that all encounters with stationary monopiles would be avoided.

Migration

Migratory movements and post-breeding dispersal could result in piping plover crossings of Nantucket Sound, and potentially HSS. Miscellaneous reports indicate peaks of 30 to 100 staging birds observed in the Nauset Beach to South Beach area of Chatham during the late summer staging period (Report No. 4.2.9-1). There is known staging habitat on Cape Poge, Martha's Vineyard and possible staging habitat on Nantucket (USFWS 1996). Individuals that stage in areas of Nantucket Sound could include birds from Atlantic Canada, northern New England, and potentially a few interior breeding birds that migrated east before heading south to Atlantic Coast wintering grounds (Haig and Elliot-Smith 2004). There is the potential that spring and fall migratory movements among stop-over sites on the Atlantic Coast could result in piping plover crossings of HSS. Hatch and Brault (2007) (Report No. 5.3.2-1) suggest that if piping plover were to depart South Beach for a stop-over destination on Long Island, NY, and if the birds were to fly at rotor height, they would be at risk of collision with the proposed WTGs.

The potential for piping plover crossings of HSS and the potential for collision are based on speculation. The risk of collision of piping plover would be based on the flight frequency through the project area, the height of flight, and turbine avoidance behavior. However, it is not known how many birds cross Nantucket Sound during migration, what flight paths they take, or the average flight height. Piping plover migrate both day and night, and an unknown percentage of the Atlantic population could move through Nantucket Sound during migration.

There are some data available on piping plover flight altitudes over water, but it is not extensive. Observations suggest piping plover fly over water at relatively low heights, generally less than 33 ft (10 m), but sometimes higher (Report No. 4.2.9-1). Other species with long-distance migrations, like golden plover, have been documented as migrating at altitudes of

15,000 to 20,000 feet (4,500-6,000 m) (Sibley 2001). These birds are known to cross large expanses of land and water and make stop-overs at staging areas along the way. However, they make much larger scale movements than piping plover. Other species of long-distance migrating plover appear to target departure during optimal flight conditions with following winds, and appear to choose flight trajectories that would result in the least flight resistance (Sibley 2001).

The periods of greatest risk of collision are during periods of low visibility, at night, during crepuscular periods, and during periods of inclement weather (Kerlinger 2000). If piping plover were to occur in HSS at night, they would be expected to visually detect turbines due to the lighting mounted on turbine nacelles as well as natural sources of light and therefore avoid collisions. At the Nysted and Horns Rev wind farm in Denmark, all the wind turbines are equipped with yellow navigational lighting. In addition, all wind turbines positioned at the outer edge of the wind farm are equipped with two medium intensity flashing red lights on the top of the nacelles. The lights operate at a frequency of 20 to 60 fpm (Petersen et al., 2006). Radar observations suggest that birds approached the turbines at closer distances at night than during the day, and that more birds entered the wind farm at night than during the day, however, observations indicated avoidance behavior of the turbines by nighttime migrants. The typical distance at which an avoidance reaction occurred was 1,640 ft (500 m) from turbines at night and 1.9 mi (3 km) during the day (Petersen et al., 2006). It may be that that migrating birds react later to the turbines at night due to decreased visibility, but are eventually able to detect the turbines due to lighting mounted on the nacelles or natural sources of night lighting. Another study conducted with a vertically oriented radar suggests that migrating birds may also react to turbines by 'vertical deflection' at night instead of the linear avoidance primarily observed during the day (Blew et al., 2006 as cited by Petersen et al., 2006).

There is data that suggest refraction caused by lighting mounted on tall structures during periods of fog and rain can disorient traveling birds (Huppop et al., 2006), however, Petersen et al., (2006) observed a substantial decrease in the volume of migrating birds during weather periods of elevated collision risk. Fewer waterbirds migrated during periods of low visibility and strong headwinds (Petersen et al., 2006). It is likely that plover wait out inclement weather conditions prior to departing staging locations. However, there is the potential that plovers could depart in fair weather conditions but could encounter severe weather en route. At staging locations, migrants have been observed following early fall hurricanes (USFWS 1996). Weather events could potentially push migrants inland or offshore. More data, collected in a variety of weather conditions during the day and at night, is needed to determine if piping plover occur in HSS during migration, particularly during periods of elevated risk of collision. Radar data collected at night in the absence of infrared visual confirmation surveys can not be used to describe shorebird species movement patterns through the project area. Due to the restriction of aerial and boat surveys to periods of fair weather, as well as the significant limitations of obtaining visual data, telemetry surveys would be beneficial in determining piping plover movement patterns in Nantucket Sound during a variety of conditions.

Collision Probability Modeling

Hatch and Brault (2007) (Report No. 5.3.2-1) used the Band Collision Risk Model to estimate a 91 to 99 percent plover turbine avoidance rate based a range of known avoidance rates calculated for other species. These avoidance rates are consistent with rates calculated at a few

existing wind farms in the U.S. where mainly geese and raptor species were estimated to have avoidance rates greater than 95 percent. Fernley et al., (2006) calculated the avoidance rates of geese at four operating land-based wind farms in the U.S. using the Band Collision Risk Model. The avoidance rates calculated at the four facilities ranged from 99.82 percent to 100 percent despite high usage by geese at these wind farm sites. Whitfield and Madders (2006) used the Band Collision Risk Model to estimate the avoidance rate of hen harriers (*Circus cyaneus*) at eight wind farms in the U.S. Estimates were: 100 percent at 6 sites, 99.8 percent at 1 site, and 93.2 percent at 1 site. Other avoidance rates reported include: 99.62 percent mainly for gull species at Blyth Harbor in Northeast England, 99.5 percent for golden eagle (*Aquila chrysaetos*) at a U.S. facility, and 99.98 percent for passerines at the Oosterbierum wind farm in the Netherlands (Chamberlain et al., 2006). There are, however, limitations to the Band Collision Risk Model, as it does not account for differences among bird activities and behaviors under a range of conditions, and because avoidance rates exhibited by a range of species are understudied (Chamberlain et al., 2006).

Chamberlain et al., (2006) warned against the inaccuracies that can result in collision models that are based on the avoidance rates calculated for other species. Hatch and Brault (2007) (Report No. 5.3.2-1) provided an estimate of the number of plover crossings of the project area per year. This estimate was based on the number of breeding plovers from Massachusetts northwards, including the Atlantic Canada population. It was estimated that 2,458 plovers cross the Massachusetts coastline over the course of a year (based on adults in spring and fall, and fledglings). MassWildlife suggested that less than 200 piping plover would cross HSS in a year (Report No. 5.3.2-1). This figure was applied to the model with varying scenarios of flight height and collision probability. Based on an avoidance rate of 98 percent, if all flights occurred in the rotor zone, one piping plover collision would occur in 5.5 years; if all flew below 98 ft (30 m), there would be one collision in 50 years; if flight heights were distributed between 98 to 1,968 ft (30 to 600 m), there would be one collision in 28 years. Using the avoidance rate of 91 percent, there would be 1.2 collisions per year if birds flew exclusively in the rotor zone, 1 collision in 12 years if all birds flew below 98 ft (30 m), and 1 collision in 6 years if flight heights were distributed between 98 and 1,968 (30 and 600 m). The authors emphasize the uncertainties surrounding the model including the lack of information regarding piping plover occurrence and flight behavior in HSS, as well as the lack of a species-specific avoidance rate.

A population viability analysis (PVA) was developed by Brault (2007) (Report No. 5.3.2-4) using the most recent breeding population trends of both the Atlantic Canada and New England population. The model estimated a range of mortality associated with the project that could be tolerated by the population without increased risk of extinction or decreased probability of recovery goals (the author used 600 breeding pairs for New England, although the current recovery goal is 625 pairs; the correct recovery goal of 400 pairs was used as the Atlantic Canada threshold). The author modeled varying kill rates with no growth and intermediate growth scenarios. It was estimated that a take of up to 5 piping plover per year would not influence the likelihood of achieving Atlantic Coast recovery goals, or influence the probability of extinction. It was estimated that the increase in the risk of extinction was low over a period of 50 years with wind farm fatalities up to 20 birds per year, given that there are no changes in available breeding and wintering habitat. It was determined that changes in the annual survival rate had 2.25 times the effect on population dynamics than did changes in productivity. The

author emphasized that the potential impacts associated with the proposed action are greatly dependent on the level of management efforts. The PVA used a New England recovery goal of 600 breeding pairs instead of the actual 625 breeding pairs. This discrepancy in 25 birds is likely an insignificant factor to the wide range of parameters factored into the model; however, it represents a flaw in the model.

Assuming the estimated worst case scenario of 1.2 wind farm-related piping plover fatalities per year with the low turbine avoidance rate of 91 percent, calculated by Hatch and Brault (2007) (Report No. 5.3.2-1), the recent PVA model suggests that the proposed action would not significantly impact the probability of achieving recovery goals or the influence the probability of extinction. However, there is a large range of uncertainty surrounding the collision mortality estimate. The actual number of crossings of the proposed action area per year, the average height of flight during crossings, and the turbine avoidance rates specific to piping plover are not known. The estimate of 1.2 wind farm-related fatalities is conservative because it assumed that piping plover exhibit a low turbine avoidance rate and that all birds fly through HSS at rotor height. The assumption that all piping plover would cross the project area at rotor height is likely inaccurate; however, it is appropriate to be conservative until more data is available.

Aerial and boat surveys conducted in Nantucket Sound documented no piping plover crossings of the site of the proposed action. However, these surveys were restricted to periods during the day and during fair weather. Additionally, there are significant limitations to the detection of small light-colored shorebirds during visual surveys conducted from a boat or a plane. There are no topographical features that would funnel piping plover through the HSS, therefore crossings of the proposed action area are expected to be small in relation to the number of birds that could potentially cross Nantucket Sound over the course of a year.

To gain greater certainty in the collision probability model, Hatch and Brault (2007) documented that more information would be required to assess piping plover use of the proposed action area during migration and the breeding season, during the day and at night, and during a range of weather conditions. As in any modeling effort, more information is always better than less in terms of reducing uncertainty in extrapolating the model to real-life conditions. In this case, some information is available to assess piping plover use of the proposed action area during the day for migration and the breeding season; however the quantity is limited, especially during the night and during inclement weather, and leads to some amount of uncertainty. However, conservative collision rate estimates and the results of the recent PVA, based on available data, suggest that potential collisions with the proposed turbines would result in minor adverse impacts to piping plover.

3.3.1.2 Roseate Tern

3.3.1.2.1 Habitat loss or modification

Terns traveling or foraging in the proposed action area could potentially be impacted by habitat loss or modification during construction, operation, and decommissioning activities. Some species of birds are more sensitive to disturbances than others and can be displaced up to hundreds of meters from the source of the activity (Gill 2005). Breeding terns would be most

sensitive to construction and operation disturbances during the breeding season when they have increased energy demands.

There is no available breeding habitat within or in close proximity to the proposed action area boundary, the offshore transmission cable system and proposed landfall would not cross breeding locations. All points along the transmission cable would be greater than 15 mi (24 km) to the nearest breeding location in Nantucket Sound on Monomoy Island. The center of the turbine array in HSS would be greater than 19.8 mi (31.8 km) to Monomoy Island and 11.5 mi (18.5 km) from the closest potential breeding habitat on Muskeget Island. Therefore, construction and decommissioning activities would not result in the loss of breeding habitat. However, terns travel substantial distances (16 to 19 mi [25.8 to 30.6 km]) from their breeding locations to access foraging habitat and terns may be affected as they travel or forage in the vicinity of the proposed action area.

Construction, operation, and decommissioning activities could directly deter roseate terns or their prey from the proposed action area resulting in the temporary or permanent loss of habitat. Baseline surveys conducted in Nantucket Sound, documented minimal tern use of the HSS area in relation to other locations in the Sound. Most terns were observed traveling, fewer were seen actively foraging. Terns are known to regularly forage near recreational fishing boats, ships, and other man-made structures. Terns and gulls are among species of birds that have been observed in the vicinity of operating turbines at European offshore facilities (Everaert and Stienen 2006, Petersen et al., 2006, Pettersson 2005). Roseate terns would likely continue to forage and travel in the vicinity of construction activities and operating WTGs, assuming that their food sources are not displaced.

Roseate terns have been observed to have decreased breeding success during periods of low food availability (Safina et al., 1988). The effects of habitat loss due to development are dependent on the amount of habitat lost and the food resources available at alternative sites (Maclean 2006). The roseate tern's primary food source is sand eel (*Ammodytes americanus*) and its locations are variable. Popular foraging habitat varies seasonally and on a daily basis with the tide cycle.

Vibrations from pile-driving could startle and temporarily displace prey fish from the proposed action area. Increases in turbidity from cable trenching could temporarily impede fish foraging and navigation in disturbed areas (Jarvis 2005). Construction activities could affect fish and benthic communities up to 328 ft (100 m) from the activity (Nedwell et al., 2004 *as cited by* Gill 2005). However, impacts to foraging habitat are anticipated to be minimal as construction activities would be temporary and localized. A jack-up barge with a crane would be used to install the monopiles. There would be a total of two pile driving rams used to fix the 130 monopile structures into the seabed and it is unlikely that both rams would be used simultaneously. The hollow monopiles are expected to trap the majority of sediment displaced during pile driving.

Sediment suspended by trenching during cable installation is expected to be localized (20 milligrams/liter within 1,500 feet [457 m] from the trench) and is expected to quickly resettle (within minutes or up to a few hours) (Report No. 4.1.1-2). Jet plow embedment would allow

for simultaneous plowing and cable-laying to minimize impacts. As a result of disturbances to sediment during trenching and pile driving, small benthic organisms would be stirred up and prey fish may be attracted to the area to forage. This in turn could attract roseate terns to forage.

Rock may be placed at the base of each monopile to prevent scour. The rock and the monopiles would increase the available surface area and provide substrate for the colonization of benthic invertebrates and habitat for prey fish. Fish may concentrate around turbine foundations similar to how invertebrates cluster around oil platforms (Vella 2002 *as cited by* Jarvis 2005). Habitat with more 'physical heterogeneity' can result in greater fish abundance (Jenkins et al., 1997 and Charbonnel et al., 2002 *as cited by* Gill 2005). The underwater structures could create a localized 'artificial reef effect', providing foraging habitat for terns. Wide spacing of turbines (0.34 to 0.54 nautical miles [0.63 to 1.0 km] apart) would allow for tern foraging between turbines (see section Risk of Collision below).

The boundary of the proposed action area would include approximately 25 square miles (6474 hectares) of WTGs and ESP (electrical service platform) foundations, and 5.89 acres (2.4 hectares) of transmission cable. The total area represents 11 percent of Nantucket Sound (Jarvis 2005). However, the total area of seabed that would permanently be disturbed would be less than 1 percent of the total wind farm area: including approximately 1 acre (0.4 hectares) for the 130 turbines, 100 by 200 ft (30.5 to 61 m) for the ESP platform, and over 45 acres (18 hectare) for rock scour protection (Jarvis 2005). The additional amount of surface area (approximately 1,200 square feet [111 square meters] per tower would result in a minor addition to the substrate that is currently available. Due to the small amount of additional surface area in relation to the total proposed action area in Nantucket Sound, and the spacing between WTGs, the proposed structures are not expected to have a significant affect on the benthic community, the presence of prey fish, or foraging terns. However, the additional substrate would be oriented vertically in the water column, and could result in a localized and minor increase in certain fish prey species.

As HSS is not a significant foraging location or traveling corridor, and because of the small footprint of the actual development area, minimal habitat loss is anticipated during construction and operation activities. Impacts associated with displacement of prey fish during construction are anticipated to be minimal and temporary. The natural benthic substrate and prey fish communities would be essentially maintained after a short recovery period, therefore, adverse impacts associated with loss of habitat or modification are not anticipated. The impacts associated with decommissioning are anticipated to be similar to or less than construction activities because pile driving would not be required (Jarvis 2005).

3.3.1.2.2 WTG presence and rotor movement

The presence of wind turbines and the spinning of the blades could present barriers to the flight paths of birds and could potentially affect or restrict access to breeding, staging, or foraging habitat. A wind farm could potentially lead to significant impacts if it were to occur in an area of high use by birds (Drewitt and Langston 2006). Barriers can result in increases in energy expenditure if birds are forced to travel greater distances while accessing foraging habitats or while undertaking migration movements. However, there are no known situations

where a wind farm has created a 'barrier effect' resulting in an avian population level impact (Drewitt and Langston 2006).

Terns have been observed to continue to use WTG areas at existing offshore facilities during both migration and breeding periods. Post-construction radar studies during migration at the Nysted and Horns Rev wind farms in Denmark indicate that, although the greatest levels of movement occurred outside of the wind farms, terns continued to migrate through the wind farm areas (Petersen et al., 2006). The facility is located 8.7 mi (14 km) offshore and is comprised of 80 turbines with a rotor zone of 98 to 360 ft (30 to 110 m). The turbines are spaced 1,640 ft (500 m) apart, half the distance of the proposed Project turbines. Visual data collected at the Nysted and Horns Rev facility indicate that the majority of terns generally avoided the direct wind farm area but increased their use of the 1.2 mi (2 km) zone surrounding the facility (Petersen et al., 2006). Terns were observed foraging at the outer edges of the facility around turbine structures. Small flocks flew into the farm, but then exited the area after passing through the second row of turbines (Petersen et al., 2006). Sandwich terns (*S. sandvicensis*) entered the wind farm between two turbines more frequently when one or both of the turbines were not active (Petersen et al., 2006). Common and artic terns (*S. paradisaea*), observed flying in the vicinity of turbines at a facility in Kalmar Sound, Sweden, flew between turbines or right next to the turbines instead of veering off in wide curves as waterfowl species were observed to do (Pettersson 2005). The Kalmar facility is located 1.9 to 7.8 mi (3 to 12.5 km) from the shore with 12 turbines spread out over two locations positioned 20 to 30 km apart. The rotor zone is 115 to 328 ft (35 to 100m) above the water surface. The facility is located along a major migration corridor for water birds. Most birds were observed making slight alterations to their flight paths while traveling past turbines to avoid approaching individual turbines. It was estimated that the presence of the turbines resulted in a minor increase (0.2 to 0.5 percent) to the overall distance traveled by most birds during migration (Pettersson 2005).

A post-construction study at the Zeebrugge wind farm in Belgium investigated the level of project disturbance on nesting terns. An artificial peninsula, created to provide nesting habitat for common, sandwich, and little (*S. albifrons*) terns, was built adjacent to 25 small to medium-sized turbines on a jetty. In 2004, terns nested as close as 98 ft (30 m) from the turbines, while the majority of nests were situated 328 ft (100 m) or further from the turbines (Everaert and Stienen 2006). In 2005, terns nested as close as 164 ft (50 m) from the turbines. The greater distance between nests and turbines in 2005 was believed to be a result of the distribution of vegetative growth on the peninsula and not due to the operation of the turbines themselves (Everaert and Stienen 2006). While terns traveled to and from the colony past the turbines, many made no apparent changes in their flight paths. The terns that exhibited a reaction to the turbines made slight changes in their flight paths to fly between turbines (Everaert 2004). The turbines did not present barriers to the flight paths of terns and observations suggest the presence of turbines resulted in minimal increases in energy expenditure for the terns. It was concluded that the presence of the turbines represented little disturbance to the activity of breeding terns (however, the project has resulted in high numbers of collisions due to the facility's location in close proximity to the colony, discussed in the following section, Risk of Collision).

A more local tern-turbine interaction study was conducted at the Massachusetts Maritime Academy (MMA) campus turbine. The MMA turbine has a maximum height of 74 m (243 ft)

(85 to 243 ft [26 to 74 m] rotor zone) and is located at the western entrance of the Cape Cod Canal. The turbine is situated 328 ft (100 m) from the water's edge on a landmass adjacent to a popular common and roseate tern foraging location, the Mashnee Flats Shoal located 5.3 mi (9 km) from one of the largest roseate tern breeding colonies, Bird Island. Visual surveys and mortality searches were conducted from April 24 to November 30, 2006, during the breeding, staging, and fall migration periods (see Section 5.3.2 Risk of Collision for information regarding mortality searches). Terns were most abundant in the area during the post-breeding period when they were foraging in large, mixed-species flocks. Terns were most abundant in the turbine airspace (within 164 ft [50 m] of turbine tower, rotor, and blades) during the chick-rearing period and least abundant during the nesting period. The average flight height of terns in the turbine airspace was 83 ft (25.3 m), the mean flight height was 50 ft (15 m). The one positively identified roseate tern observed in the turbine airspace flew at 26 ft (8 m). In summary, of the terns observed in the 164 ft (50 m) airspace surrounding the turbine: 17 percent flew within, 74 percent flew below, and 9 percent flew above the rotor zone. The study demonstrated that terns continued to use the 50 m [164 ft] airspace around the turbine while traveling between foraging locations (Vlietstra 2007). However, the operating rotors and spinning blades were observed to deter terns from flying directly within the rotor zone of the turbine when the rotor velocity was greater than 1 rotation per minute (rpm). Under these conditions, terns were found to be 4 to 5 times less abundant in the turbine airspace. Therefore, it was assumed that the terns visually and acoustically detected the spinning blades when the rotor was operating (Vlietstra 2007). Despite the turbine's location in between foraging locations, terns continued to use the area and their access to habitat was not evidently restricted.

As terns are known to travel and forage around other man-made structures, including lighthouses, bridges, and wind turbines, it is likely that roseate terns would continue to use the proposed action area after construction. Although the majority of terns are expected to avoid the direct WTG rotor swept area (refer to the following section, Risk of Collision, for detailed information of avoidance behavior), it is anticipated that terns would continue to travel and forage in the vicinity of the WTG array. Also, because the turbines are widely spaced (0.34 to 0.54 NM [0.63 to 1.0 km] apart), it is anticipated that most terns would occur between turbines while traveling at heights within the rotor swept zone.

3.3.1.2.3 Risk of Collision

The potential exists for roseate terns to collide with WTG structures, including the blades and tubular towers during the breeding, staging, and migration periods when any individual from the northeastern population could occur in the vicinity of the proposed action area. The results of available mortality studies indicate that the majority of collisions with man-made structures take place at night during periods of inclement weather (Kerlinger 2000). Birds that fly within the rotor zone of the proposed turbines (75.5 to 440 ft [23 to 134 m]) during periods of low visibility would be at greatest risk of collision.

Collision Risk during Periods of Good Visibility

Outside of migration, terns are mainly active during the day except at dusk and dawn when they have been observed to depart or arrive at roosting locations (Trull et al., 1999, Hays et al., 1999 as cited by Nisbet 2007). During daytime periods of good visibility, there is a low risk of

roseate tern collision with the proposed WTG structures. Results of post-construction studies at existing European facilities suggest offshore wind farms, when properly sited, do not impose adverse impacts to local tern populations. Studies conducted at the wind farms in Sweden and Denmark showed continued tern use of turbine areas after development, as well as collision avoidance behaviors when terns approached individual turbines (Peterson et al., 2006, Pettersson 2005). The study conducted at the MMA turbine, near the Cape Cod Canal in Buzzards Bay, indicated continued use of the area by roseate terns and avoidance of the rotor zone when the turbine was operating at greater than 1 rpm (Vlietstra 2007). When the rpm was greater than 1, terns were 4 to 5 times less abundant in the 50 m (164 ft) airspace surrounding the turbine blades. Mortality surveys at the MMA turbine indicated no tern fatalities. When searcher efficiency and scavenger data were factored into the number of bird fatalities found (3 total birds were found but only one gull was suspected to have collided with the turbine), it was estimated that the MMA turbine would result in 2.15 bird fatalities per year. This rate is within the range of avian fatalities, 0.01 and 7.5 per turbine per year, reported at other existing onshore facilities (Erickson 2001, Erickson 2004 as cited by Vliestra 2007). There is a new wind farm located in Pubnico Novia Scotia (17 turbines) that is located 2.4 miles (4 km) from the largest population of roseate terns in Canada. Mortality surveys are currently being conducted, however, the findings are not yet available.

Poorly sited facilities can result in high collision rates. A mortality study conducted at the Zeebrugge, Belgium facility reported notably high tern collision mortality. At this facility, terns have nested on a peninsula as close as 98 ft (30 m) from a string of 25 small to medium sized turbines located on an adjacent breakwater. The mean number of terns killed for all turbines was 6.7 terns per turbine per year, and the mean number of terns killed at the 14 turbines closest to the colony was 11.2 terns per turbine per year for 2004 and 2005 (Everaert and Stienen 2006). The rotor zone (52.5 to 164 ft [16 to 50 m]) of the 14 turbines that are responsible for the high number of tern fatalities is lower than that of the proposed turbines, and the turbines are spaced closer together (394 ft [120 m] apart) (Report No. 5.3.2-1). The collision mortality observed at the Belgium facility was determined to have an adverse impact on the breeding terns. However, the majority of these collisions occurred at the 14 turbines located closest to the tern colony and most collisions may have been associated with the circular, erratic flight behaviors that terns exhibited near the colony (Everaert and Stienen 2006). Ten tern collisions were documented when observers were onsite and it is likely that the observers themselves caused a disturbance to the colony, resulting in the observed collisions (Everaert and Stienen 2006). If the peninsula colonized by the terns had not been created adjacent to the string of turbines on the breakwater, it is very likely that the observed collision mortality would have been substantially lower.

The particular flight behaviors of roseate terns that could put them at a greater risk of collision with man-made structures include aerial displays (involving dives from heights of 98 to 980 ft [30 to 300 m]) (Gochfeld et al., 1998) and defensive flights (involving aerial circling and dives) around the colony. Also, the approach of vessels or helicopters could cause terns to quickly depart the proposed action area. Similar flight behaviors put the tern species at the Belgium facility at a greater risk of collision. Sandwich terns exhibited a lower collision probability than other species of tern and this was believed to be due to the fact that sandwich terns generally flew in a straight line toward feeding locations and back, while common terns typically exhibited more erratic, circular movements around the colony (Everaert and Stienen

2006). Roseate terns would not exhibit high risk flight behaviors in the vicinity of HSS as these activities are conducted at the edge of colonies or at resting areas (Gochfeld et al., 1998) and these habitats do not occur within HSS. Terns departing the area due to approaching vessels or helicopters would be expected to make direct flights out of the area because they would not exhibit the defensive behaviors they direct toward intruders at the colonies. The flight behavior expected of roseate terns in the proposed action area, based on bird surveys conducted in HSS and roseate tern biology, would predominantly be direct paths to and from breeding, staging, or foraging locations at heights below the proposed rotor zone. A minimal amount of foraging is expected in HSS and roseate terns typically forage at heights (3.3 to 39.4 ft [1 to 12 m]) below the proposed rotor zone (Gochfeld et al., 1998).

Above water foundations are not anticipated to create perching habitat or result in increased risk of collision for roseate terns. The above water foundations, WTGs, and the ESP would be equipped with stainless wire and vision restriction perch deterrent devices. Each turbine foundation would have a deck which would be covered by aluminum chain link fencing to discourage access on the sides (and the deck overhangs the access ladder). There would be a taught 3 mm stainless steel wire on top of the railing, and a 2.1 ft (0.65 m) solid panel to restrict the view of birds from the deck (some species prefer perches with views). The spacing between the wire and the rail would be 1.2 in (3 cm). The ESP would have a perimeter railing and the ladders and railing would be equipped with stainless steel wire, chain-link fence, and panels similar to the WTG foundations. The use of perch deterrent devices has discouraged terns perching on the fence and deck of the platforms supporting the Cape Wind Scientific Measurement Devices (SMDS). The final design of perch deterrents would be based on recommendations from USFSW. The use of tubular towers instead of lattice towers also discourages perching under the rotors. Vibrations and low level noise created by operating WTGs may also deter terns from perching.

The potential exists for roseate terns to occur at heights within the rotor zone while traveling. There is the potential that additional mortality or injury could result from birds not actually colliding with the turbines, but getting caught in the turbulence behind rotors (Winkelman 1994). Winkelman's 1992 study suggests that approximately 20 percent of avian mortality found at the shore-based Oosterbierum wind farm in the Netherlands was caused by such turbulence 'strikes', however, there are no other studies that have reported observations of this phenomenon for birds (Desholm 2006). Turbulence effects may also increase avian avoidance of turbines. Daytime visual surveys, radar, and nocturnal surveys with a thermal image intensifier suggested that birds flying into a headwind were more likely to react to turbines perhaps because they approached the rotor wake before reaching the rotor (Winkelman 1994). The MMA turbine study suggests that birds can, in addition to visually detecting and avoiding turbines, detect the 'whooshing' sound that rotors create (Vliestra 2007). Potential turbulence effects to birds would depend on the wind speed and direction, and the direction from which a bird approaches a turbine. However, further studies are required to determine the level of impact rotor turbulence poses to birds.

Turbine Avoidance Behavior

Terns, during periods of good visibility, are expected to avoid collisions with the proposed turbines based on observations of tern behavior at existing facilities. Roseate terns are agile fliers and they would be expected to visually detect and avoid the moving blades and stationary

towers of the proposed turbines during periods of good visibility. Additionally, modern turbines are more avoidable as they have only three, relatively slow rotating blades. Roseate terns regularly avoid collisions with other man-made structures including moving vessels, piers, bridges, and lighthouses, as well as wind turbines.

Hatch and Brault (2007) (Report No. 5.3.2-1) used the Bolker model to estimate the number of turbines encountered during a single crossing of the WTG array area to assess the potential for collisions. The proposed turbines would be spaced 0.34 to 0.56 NM (0.63 to 1.0 km) apart and would be oriented in a single direction during a single tern crossing. Therefore, assuming terns travel in straight paths from all angles through the WTG array area at flight heights distributed evenly through the rotor zone, the estimated number of turbine encounters during a single crossing at rotor height is 0.43. Under the same assumptions, the estimated number of monopiles encountered during a single crossing is 0.04. These numbers are likely accurate estimates of the number of potential roseate tern encounters with turbine structures because the majority of terns in HSS would be expected to move through the WTG array area along straight flight paths. The potential for collisions with WTG structures is influenced by wind direction and its influences on tern flight behavior. The authors suggest that birds flying downwind in the rotor zone are at a decreased risk of collision because of the rapid groundspeed at which they would pass through the blade area. Alternatively, the risk of collision is greater when birds pass turbines while traveling into a strong headwind because of the slow groundspeed at which they travel through the blades. However, terns are less likely to be flying within the rotor zone while traveling into a strong headwind as they tend to fly closer to the water surface under high winds (see following section, Collision Risk Modeling). Because monopiles are stationary and do not have moving parts, terns are expected to avoid collisions with these structures except during the most severe weather conditions, when the activity of terns in HSS is expected to be very low. Additionally, terns have been observed to avoid approaching individual turbines at other facilities, by making slight alterations to their flight paths when passing turbines (Pettersson 2005, Petersen et al., 2006, Everaert and Stienen 2006).

Hatch and Brault (2007)(Report No. 5.3.2-1) estimated a turbine avoidance rate for roseate terns traveling through the HSS WTG array area, based on a rate calculated for common, sandwich, and little terns at the Zeebrugge, Belgium facility (91 percent turbine avoidance). The avoidance rate observed at the Belgium facility was based on common tern observations in the vicinity of a nesting colony where their flight behavior was circular and erratic. Since roseate terns are smaller and faster flying than common terns and because the proposed rotor zone is above the majority of flight heights observed in HSS, Hatch and Brault used an estimated range of avoidance rates for roseate terns in HSS, 95.3 percent and 98.3 percent. These avoidance rates are consistent with rates calculated at a few existing wind farms in the U.S. where mainly geese and raptor species were estimated to have avoidance rates greater than 95 percent. Fernley et al., (2006) calculated the avoidance rates for geese at four operating land-based wind farms in the U.S using the Band Collision Risk Model. The avoidance rates calculated at the four facilities ranged from 99.82 percent to 100 percent despite high usage by geese at these wind farm sites. Whitfield and Madders (2006) used the Band Collision Risk Model to estimate the avoidance rate of hen harriers (*Circus cyaneus*) at eight wind farms in the U.S.: estimates were 100 percent at 6 sites, 99.8 percent at 1 site, and 93.2 percent at 1 site. Other avoidance rates reported include: 99.62 percent mainly for gull species at Blyth Harbor in Northeast England, 99.5

percent for golden eagle (*Aquila chrysaetos*) at a U.S. facility, and 99.98 percent for passerines at the Oosterbierum wind farm in the Netherlands (Chamberlain et al., 2006). There are, however, limitations to the Band Collision Risk Model as it does not account for differences among bird activities and behaviors under a range of conditions, and because avoidance rates exhibited by a range of species are understudied (Chamberlain et al., 2006). Chamberlain et al., (2006) had concerns with applying an avoidance rate that was calculated for a different species because inaccuracies in avoidance rate calculations can have magnified effects on estimated mortality rates.

Collision Risk during Periods of Low Visibility

There is the potential for roseate tern crossings of HSS during crepuscular periods, periods of fog or rain, and during nighttime migration movements when the risk of collision is elevated. Due to the difficulties of surveying during these conditions, there is no information about roseate tern occurrence and behavior in HSS during these periods.

During the breeding and staging periods, terns arrive at roosting sites around sunset and continue to arrive after dark (Trull et al., 1999). Terns are assumed to depart staging grounds before dawn and at sunrise to travel to foraging locations. While making daily movements to and from staging habitat, roseate terns are known to travel overland across Cape Cod and are also anticipated to pass over waters in Nantucket Sound (Spendelow et al., unpublished).

Roseate terns disperse to their wintering grounds during August and September and return to breeding locations from late-April to mid-May. Some terns depart for wintering grounds during the day but it is likely that terns also depart in the evening. There is a scarcity of data available specifically on roseate tern nighttime migration that results in a level of uncertainty and the inability to forecast future risk in an absolute manner; however, other species of tern are known to complete the majority of their movements at night (Alerstam 1985). Additionally, observations have been made of mixed flocks of terns departing staging grounds in Nantucket Sound around sunset (Report No. C-1). On August 29, 2006, around sunset, approximately one mile northwest of South Monomoy Island, a total of 75 terns were observed soaring to heights of 427 ft (130 m), the highest birds disappeared into the clouds (Report No. C-1). It was believed that the terns were departing South Beach and preparing to migrate. However, their flight direction indicated that they would not cross the WTG array area. Although, in varying wind conditions, the potential exists that roseate terns could cross HSS while departing staging grounds or when arriving at breeding areas in the spring. If migrating terns were to occur over HSS, the risk of collision would be dependent upon the flight height of the migrants. There are limitations to available data, and assessing whether terns do or do not cross HSS during crepuscular or nighttime migration movements would be based on speculation.

Based on the available data, terns have been observed at heights well above the rotor zone when making migratory movements. There have been observations of what were assumed to be both roseate and common terns departing South Beach in the fall around sunset, apparently heading toward their wintering grounds, and quickly gaining altitudes of hundreds of meters (Veit and Petersen 1993). Other species of terns have been observed migrating at heights above 9,842 ft (3,000 m) when migrating over land (Alerstam 1985). It is likely that nighttime migration movements, if they were to cross HSS, would occur well above the rotor zone. The

flight height, however, would be dependent on weather conditions. If terns were to depart in unfavorable conditions such as strong headwinds, their flight heights would likely be lower as other tern species have been observed flying close to the water's surface during strong headwinds (Alerstam 1985). However, it is unlikely that flocks of staging terns would depart for migration in unfavorable weather conditions during the day, and less likely at night. More data is required to assess roseate tern migratory behavior in HSS during a variety of weather conditions.

If roseate terns were to cross the WTG array area during crepuscular periods or at night, aspects of the facility design are expected to minimize the risk of collision. Each perimeter WTG nacelle would be lit at night with one red flashing FAA light. Corner WTGs would be equipped with medium intensity FAA L-864 lighting. The other perimeter WTGs would be equipped with low intensity lights visible up to 1 NM (1.9 km). The eight turbines adjacent to the ESP would have one L-810 flashing red light. FAA lighting would be synchronized to flash in unison at 20 flashes per minute (fpm). Construction structures and equipment would be lit at night and construction activities would be limited during inclement conditions, particularly at night.

Although passerine species are known to be attracted to the refracted lighting of tall structures during periods of fog or rain (Huppop et al., 2006), there is no data available that suggests terns are attracted to refracted lighting during these conditions. The lighting mounted on nacelles as well as natural sources of nighttime lighting are expected to decrease the risk of tern collisions if their migratory movements result in nighttime crossings of the WTG array area. At the Nysted and Horns Rev wind farm in Denmark, all the wind turbines are equipped with yellow navigational lighting. In addition, all wind turbines positioned at the outer edge of the wind farm are equipped with two medium intensity flashing red lights on the top of the nacelles. The lights operate at a frequency of 20 to 60 fpm (Peterson et al., 2006). Radar observations suggest that birds approached the turbines at closer distances at night than during the day, and that more birds entered the wind farm at night than during the day, however, observations indicated avoidance behavior of the turbines by nighttime migrants. The typical distance at which an avoidance reaction occurred was 1,640 ft (500 m) from turbines at night and 1.9 mi (3 km) during the day (Peterson et al., 2006). It may be that that migrating birds react later to the turbines at night due to decreased visibility, but are eventually able to detect the turbines due to lighting mounted on the nacelles or natural sources of night lighting. Another study conducted with a vertically oriented radar suggests that migrating birds may also react to turbines by 'vertical deflection' at night instead of the linear avoidance primarily observed during the day (Blew et al., 2006 as cited by Peterson et al., 2006). Peterson et al., (2006) observed a substantial decrease in the volume of migrating waterbirds during weather periods of elevated collision risk. Fewer waterbirds migrated during periods of low visibility and strong headwinds (Peterson et al., 2006).

During the breeding season, terns would likely continue to forage during the day in most areas of the Sound during light rain and periods of moderately high surf. However, during extreme high surf and wind conditions, it is likely that they would forage most efficiently in sheltered bays or salt marshes than in more exposed areas like HSS. Therefore they would not be expected in HSS during periods of extreme weather when the risk of collision is elevated.

However, surveys targeting tern behavior in the proposed action area during these conditions would be beneficial to the assessment of collision risk. If terns were flying into a strong headwind, they would likely fly just above the surface of the waves (Alerstam 1985). Terns flying above the surface of the water would be at a decreased risk of collision with turbine blades. Additionally, turbine rotation would shut down when wind speed exceeds 55 miles/hour (24.5 m/s), decreasing the risk of collision with turbine blades.

Collision Risk Modeling

Hatch and Brault (2007) (Report No. 5.3.2-1) used a geometric model to determine the number of potential roseate tern collisions with the proposed turbines based on estimated annual averages of the number of roseate tern crossings of the proposed action area. The authors calculated the estimated average number of terns based on the mean of the average survey totals and the mean flight altitudes observed during MAS and the Applicant aerial and boat surveys conducted in May, June, July, August, and September of 2002-2004. The model suggests that the median mortality rate resulting from roseate tern collisions with the proposed turbines would be 0.8 individuals per year. The model was based on an estimated number of tern encounters with turbine structures (as discussed previously in *Turbine Avoidance Behavior*), and on an estimated number of roseate tern crossings of the proposed action area per year. There is uncertainty surrounding the number of crossings as it is based on estimated proportions of 3.2 percent and 10 percent roseate terns out of mixed-flocks observed in HSS. Some studies suggest that the proportion of roseate terns within mixed flocks in areas of Cape Cod range from 11 to 33 percent (Trull et al., 1999 as cited by Nisbet 2007). The number of crossings and potential encounters with turbines were inaccurately estimated because the model assumes that traveling terns continue to cross the wind farm, and are replaced by others as they exit the proposed action area during all daylight hours, during the time that they are present in Nantucket Sound. The model also does not account for the periods 1 hour before and 1 hour after sunset when terns are known to depart for and return from foraging habitat (Nisbet 2007). Additionally, the turbine avoidance rate factored into the model was based on an avoidance rate that assumed collisions are less likely to occur at the proposed facility because it would have a higher rotor zone (75 to 440 ft [23 to 134 m]) than the Belgium facility (52.5 to 164 ft [16 to 50 m]). Roseate terns are also smaller and are known to fly faster than common terns (Report No. 5.3.2-1, Gochfeld et al., 1998). The authors used a combination of Monte Carlo and data resampling methods to estimate the range of uncertainty surrounding the parameters of the model. The range of uncertainty surrounding the median value is 5 percent and 95 percent, resulting in a range of 0.01 to 8.2 roseate tern mortalities per year. These extreme values are influenced by the lack of data regarding roseate tern occurrence in HSS and species specific avoidance behaviors. However, the median values of the sensitivity analysis demonstrate a narrow range of annual kill rates, suggesting the robustness of the median value of 0.8 kills per year.

Surveys conducted in the proposed action area suggest that there is minimal tern use of HSS as traveling and foraging habitat. The majority of roseate terns that do occur in HSS would be expected at heights well below the rotor zone. Surveys in the study area indicated that the majority of flight heights (95 percent) did not occur within the rotor zone. Aerial survey data should be interpreted with caution due to the inaccuracies of determining heights of small, light-colored birds from 246 ft (75 m). However, available data describing low-flying terns from boat

surveys conducted at the surface of the water is more reliable. Boat surveys reported the majority of heights below the rotor zone.

Data on tern occurrence, flight behavior, and movement patterns in HSS during periods of low visibility and at night during migration is limited, which causes there to be a level of uncertainty about the true potential for collision risk. The potential remains for roseate tern crossings of the WTG array area during these periods of increased risk of collision. However, roseate terns would be expected to visually detect and avoid structures and equipment during periods of good visibility, and are expected to avoid turbines due to FAA lighting and natural light sources if they were to occur in the area at night. The potential of roseate terns occurring in the proposed action area during extreme weather conditions is low. The risk of roseate tern collisions is low based on the site's distance from breeding colonies, the minimal use of HSS demonstrated by roseate terns, and the overall low flight heights observed in the proposed action area. However, any level of collision mortality would be detrimental to the Atlantic coast population; therefore, impacts associated with risk of collision are anticipated to be moderate.

3.3.1.2.4 Increased Predation

There is a potential that WTG and ESP foundation structures may provide perching habitat for predatory peregrine falcons (*Falco peregrinus*), which could result in the mortality of roseate terns. Peregrine falcons aerial hunt or hunt from perches while they take avian prey. They are known to rarely take tern species (Wheeler 2003). Banding and telemetry data indicate that peregrine "migration routes are distinctly centered along the Atlantic Coast" during fall migration (Wheeler 2003). Peregrine falcons will also make major over-water crossings from Baffin Island or Labrador to the mid-Atlantic Coast (Wheeler 2003).

There is little seasonal overlap between roseate terns and migrating peregrine falcons within the Atlantic Coast region. Peregrine falcon fall migration peaks in late-September to early-October (Wheeler 2003) and roseate terns migrate south by mid-September. Limited information is available regarding peregrine falcon spring migration. However, telemetry data indicates that peregrine falcons reach breeding grounds by May (Wheeler 2003), at which time roseate terns return to Nantucket Sound. Mainly the winter range of the peregrine falcons overlaps with the proposed action area (Wheeler 2003) when roseate terns are not present. Additionally, above water foundations would have perch deterrent devices. The use of tubular towers instead of lattice towers would also discourage perching.

It is anticipated that development of the proposed action would not result in the addition of hunting opportunities for predatory species or increased predation of roseate terns.

3.3.1.2.5 Vessel traffic

Increases in vessel traffic could result in impacts to roseate terns during the construction, operation, and decommissioning phases. A large vessel(s) would be used to transport and install the monopiles, towers, nacelles, hubs, and blades during construction and decommissioning. The vessel would be loaded in Quonset, Rhode Island, and would be anchored near the monopiles that are undergoing construction. During installation and decommissioning of the

WTGs, the large vessel would make several trips from Quonset to the proposed action area. Additionally, small vessels from Falmouth, Massachusetts, and a maintenance support vessel from New Bedford would make regular trips to HSS during the construction period. While the proposed turbines are in operation, there would be regular vessel trips made from Falmouth and New Bedford harbors to the proposed action area. The expected maintenance schedule would be approximately 2 vessel trips per day for 252 days per year (5 maintenance days per turbine per year) (see Section 2 of the DEIS for a description of maintenance activities).

During high surf conditions, workers may be transported by helicopter to the platform on the ESP. There may also be occasional helicopter landings at the ESP in association with some regular maintenance activities. An increase in recreational fishing may occur around the WTGs if fish populations aggregate around foundations. The arrival of vessels and helicopters could temporarily displace terns from localized areas within the larger proposed action area. This type of disturbance already occurs to some extent within and adjacent to the proposed action area due to existing levels of vessel activity.

Terns appear to be less sensitive to human disturbances than other species of birds, and are also thought to be attracted to some areas of human activity (Borberg et al., 2005, Drewitt and Langston 2006, Sadoti et al., 2005a). Terns are known to habituate to some levels of human presence and disturbance. Terns are regularly observed traveling and foraging in the vicinity of vessels and other man-made structures. The major northeast roseate tern breeding colonies on Ram and Bird Islands in Buzzards Bay are located near the entrance of the Cape Cod Canal which receives frequent recreational boating and commercial shipping activity, yet terns continue to colonize these islands. Biologists frequently visit the large roseate tern colonies on the Atlantic Coast and consequently, roseate terns have become habituated to their presence and their handling of eggs, chicks, and adults (Nisbet et al., 1999). An increase in the presence of terns and gulls observed in areas around the Horns Rev offshore facility in Denmark was believed to be associated with increased boat activity for maintenance activities (Petersen et al., 2006). Therefore, roseate terns are expected to continue their traveling and foraging activities despite the presence of increased boat traffic and the few anticipated helicopter landings in HSS. Terns would be expected to return to the area after the departure of the vessels.

Roseate terns are expected to be among those species of bird that would habituate to the presence of increased boat traffic associated with maintenance activities. Therefore disturbances associated with the operation of the facility are anticipated to have minimal effects on roseate terns.

3.3.1.3 Proposed Action Analysis-Routine Conditions

Piping Plover

Potential avian impacts associated with construction and operation could include loss of habitat, disturbances associated with the presence or activity of construction equipment or maintenance vessels, disturbances such as barriers to flight paths due to the presence of the turbines, and risk of collision.

Minor to moderate adverse impacts to piping plover are anticipated during the routine activities associated with construction, operation, and decommissioning. The proposed action area does not occur within breeding or staging habitat, or within a known migration or movement corridor: therefore, the effects of loss of habitat or habitat modification would be negligible. Since the shoreline would be drilled under for cable placement, there would be no disturbance of beach areas. There would be an 820 ft (250 m) (or greater buffer) between the closest breeding beach and the proposed offshore transmission cable system. Therefore, disturbances associated with offshore construction activities including increased vessel traffic and the presence and operation of construction equipment are not anticipated. A wire may placed across the beach located closest to the offshore transmission cable system in order to power the drill head, however, disturbance to nesting piping plover would be equal to a person walking on the beach. The proposed landfall is greater than 1.5 miles (2.4 km) to the nearest breeding locations; therefore, disturbances associated with construction and operation of the facility are not anticipated.

There are no features that would funnel piping plover across the proposed action area if their movements were to result in crossings of Nantucket Sound during the breeding season or migration season. Therefore, the presence and operation of the WTGs is not expected to present a major barrier to the flight paths of transient plovers. Piping plover that encounter turbines during crossings of the Sound are generally expected to avoid collisions with WTG structures depending on visibility. These avoidance behaviors are expected to result in minor changes to piping plover flight behavior and minimal increases in energy expenditure. Therefore, the presence of WTGs in HSS may affect piping plover, but minor adverse impacts are anticipated.

Piping plover cross areas of Nantucket Sound to access breeding locations during migration or dispersal, and may sporadically cross the Sound during the breeding period. However, the flight paths of piping plover through the Sound are not known. The migration flight paths of piping plover along the Atlantic Coast are expected to occur within a narrow corridor along the coast but some birds may occur offshore or inland. Piping plover migrate both day and night and could travel during periods of inclement weather when visibility is reduced. However, studies suggest that migration of birds is reduced during periods of inclement weather (Petersen et al., 2006). Birds have demonstrated turbine avoidance behaviors both during the day and at night. If piping plover were to occur within the proposed action area, they would generally be expected to visually detect and avoid collisions due to FAA lighting on the nacelles as well as sources of natural lighting. More information is required to assess the effects of refracted light during periods of rain or fog to traveling piping plover. There are no topographical features that would funnel piping plover through HSS, therefore, crossings of the WTG array area are expected to be few in relation to the number of birds that could potentially cross Nantucket Sound over the course of a year.

Aerial and boat surveys conducted in Nantucket Sound documented no piping plover crossings of the WTG array area. However, these surveys were restricted to periods during the day and during fair weather. Additionally, there are significant limitations to the detection of small light-colored shorebirds during visual surveys conducted from a boat or a plane. More information is required to assess piping plover use of the proposed action area during migration and the breeding season, during the day and at night, and during a range of weather conditions.

Assuming the estimated worst case scenario of 1.2 wind farm-related piping plover fatalities per year with the low turbine avoidance rate of 91 percent, calculated by Hatch and Brault (2007) (Report No. 5.3.2-1), the recent PVA model suggests that the proposed action would not significantly impact the probability of achieving recovery goals or the influence the probability of extinction. The results suggest that collision mortality would result in moderate adverse impacts to piping plover but would not jeopardize the Atlantic Coast population. These results should be interpreted with caution as there is a large range of uncertainty surrounding the estimate of collision mortality. The actual number of crossings of the proposed action area per year, the average height of flight during crossings, and the turbine avoidance rates specific to piping plover are not known.

Roseate Tern

The routine activities associated with construction, operation, and decommissioning that could potentially result in impacts to roseate terns include loss of habitat or prey displacement during construction, barriers to flight paths due to the presence of WTGs, collisions with structures, increased predation, and/or disturbances associated with increased vessel traffic.

Bird surveys in Nantucket Sound indicate that the WG array area is not located within significant roseate tern foraging habitat, and it is not a major traveling corridor. Because of the small footprint of the actual development area, minimal habitat loss is anticipated during construction, operation, and decommissioning activities. Impacts associated with displacement of prey fish during construction are anticipated to be minimal and temporary. The natural benthic substrate and prey fish communities would be maintained to the greatest extent possible. Terns have demonstrated minimal use of the WTG array area in relation to other more significant tern habitat locations in the area, therefore, adverse effects associated with loss of habitat or modification are not anticipated.

As terns are known to travel and forage in the vicinity of other man-made structures, including wind turbines, it is likely that roseate terns would continue to use the WTG array area after construction. Although the majority of terns are expected to avoid the direct WTG rotor swept area, it is anticipated that terns would continue to travel and forage in the vicinity of the turbines. Tern surveys in HSS documented minimal use of the WTG array area, therefore, the array is not anticipated to present a major barrier to the flight paths of terns. The proposed action is not expected to substantially increase energy expenditure as terns travel around the direct area of WTGs. Also, because turbines are widely spaced (0.34 to 0.56 NM [0.63 to 1.0 km] apart), it is anticipated that some terns would occur between turbines while traveling or foraging as they have been observed to do at existing offshore facilities with smaller spacing between turbines. Therefore, the presence of the turbines may affect roseate tern behavior to some extent, but is not anticipated to adversely affect the population of roseate terns.

Roseate terns are anticipated to continue minimal foraging and traveling activities in the vicinity of the WTGs with a low risk of collision given the turbine-avoidance behavior exhibited by terns at the majority of existing offshore and near-shore facilities. The exception is the high collision mortality observed at the Belgium facility located adjacent to a tern colony where terns exhibited high risk flight behaviors and frequent flights through the turbines which put them at a greater risk of collision. Because no colony is located adjacent to HSS, this data is not

particularly relevant to the proposed action's potential impacts. Roseate terns would be expected to make direct flights while traveling through HSS to access foraging or breeding locations. The majority of flight heights observed in the proposed action area occurred below the rotor zone. During conditions of good visibility, roseate terns would be expected to visually detect and react to turbine structures. Roseate terns are not expected to frequent the WTG array area during those periods of inclement weather or at night, however, surveys have not been conducted under these conditions and therefore the potential for collision under these conditions can not be ruled out. However, if flying into strong headwinds, terns would be expected to fly closer to the water's surface. If flying at night, they would be expected to avoid encountering the proposed turbines based on the observed turbine avoidance behavior observed by other waterbirds at night. These factors decrease the risk of roseate tern collisions with structures.

Arnold (2007) (Report No. 5.3.2-5) developed a population viability analysis (PVA) to demonstrate the range of mortality that the Northeast population of breeding roseate terns (excluding the Canada population) could tolerate without an increased risk of extinction. The probability analysis assumed that all individuals from the Northeast population have an equal opportunity for encountering the proposed turbines. At the time of modeling, the most updated life history data was used to model three adult survival rate scenarios: 'best-case' (mean survival rates from 1988 to 1998, 'worst-case' (the lowest survival rates from 1988 to 1998), and 'recent trend' (mean survival rates from 1999 to 2005). The model considered annual variation in survival rates depending on environmental stochasticity as well as uncertainty in the life history data. Under the most updated survival and productivity parameters at the time of modeling, it was determined that there is a 95 percent chance that the population would fall below the threshold of 500 males after 50 years in the absence of additional mortality resulting from the Project (the risk of extinction at 15 and 25 years without additional Project associated mortality is 9 percent and 42 percent, respectively). Hatch and Brault (2007) (Report No. 5.3.2-1) determined that 0.8 roseate terns per year could die as a result of collision with the proposed turbines. The results of the population viability analysis indicate that a take of 0.8 individuals per year would lead to a minimal increase in the risk of population extinction. For a 20 year life of the proposed action, the take of the maximum uncertainty parameter of 8.2 individuals per year after 15, 25, and 50 years would result in the risk of population extinction of 11 to 12 percent, 46 to 50 percent, and 95 to 96 percent, respectively. These extinction probabilities are only slightly higher than the probability of extinction in the absence of the wind farm. The PVA suggests that the proposed action would result in a minimal increase in the risk of population extinction. Therefore, the potential for collision may result in moderate adverse impacts to roseate terns, but is not anticipated to jeopardize the population.

The collision probability model and the PVA suggest that the development of the proposed action would result in moderate adverse impacts to the population of roseate terns. However, there is uncertainty surrounding the collision risk probability model because the actual number of roseate terns that occur in the proposed action area each year and the turbine-avoidance rates are estimates only. There is uncertainty surrounding the population viability analysis because of the unpredictability of stochastic events. However, these models are based on the most current life history data and were developed in consultation with roseate tern experts. The models are therefore acceptable projections of potential impacts.

It is unlikely that development of the proposed action would result in additional hunting habitat for predatory species. Peregrine falcons aerial and perch hunt, and are known to take species of tern. However, their seasonal occurrence in the proposed action area does not overlap with roseate terns. Additionally, proposed action structures would be equipped with perch deterrents and modern tubular towers do not provide perch habitat as did the older, outdated lattice towers. Therefore, increased predation may affect, but is not likely to adversely affect roseate terns.

Roseate terns are expected to be among those species of bird that would habituate to the presence of increased boat traffic and other disturbances associated with construction and maintenance activities. Roseate terns tolerate a range of human disturbances at breeding locations where they would be considered most vulnerable to impacts. Terns do not breed in the vicinity of the proposed action and have demonstrated minimal use of the area for traveling, and less use for foraging. Therefore, disturbances resulting from the routine activities associated with construction, operation and maintenance, and decommissioning of the proposed action may affect roseate terns, however, these disturbances are not likely to adversely affect the population.

3.3.1.4 Summary and Conclusion-Routine Conditions

Piping Plover

The preliminary assessment determines that routine activities associated with project construction, operation, and decommissioning would result in minor to moderate impact to piping plover. There would be no loss of critical habitat and no loss of nesting or staging habitat. Due to the 250 m (or greater) buffer of the offshore transmission cable system from the nearest nesting beaches, disturbances associated with offshore construction or operation activities are not anticipated for nesting piping plover. In addition, since the shoreline would be drilled under for cable placement, there would be no disturbance of beach areas. The placement of a wire to help assess the position of the drill head on the beach would result in disturbance roughly equal to a person walking on the beach. Construction activities may result in minimal disturbances to piping plover, however, these impacts are anticipated to be temporary and minor. The proposed landfall site is 1.5 miles (2.4 km) from the nearest nesting beach and, therefore, onshore construction activities are not expected to impact nesting piping plover.

Relatively few piping plover crossings of Nantucket Sound are expected and there are no topographical features that would funnel transients through HSS. Therefore, the presence of the WTGs may affect the flight behaviors of piping plover, however, minor adverse impacts are anticipated.

There is a risk of collision of piping plover with WTG structures during nighttime migration movements or crossings of HSS during periods of reduced visibility. However, crossings of HSS are expected to be few in relation to the potential number of piping plover crossings of Nantucket Sound that may occur over the course of a year. Piping plover are expected to visually detect and avoid turbine structures based on FAA lighting and natural sources of lighting. Piping plover are not anticipated to regularly occur within the project area during periods of inclement weather, however, surveys have not been conducted during these conditions so piping plover occurrence in HSS during these periods can not be ruled out. The conservative results of the

2007 collision risk model suggest there could be as many as 1.2 wind-farm related fatalities per year. The recent PVA, however, suggests that the Atlantic Coast population could withstand this impact without increased risk of extinction or decreased risk of achieving recovery goals. Therefore, the potential for collision may result in moderate adverse impacts to piping plover, however, these impacts are not anticipated to jeopardize the Atlantic Coast population.

Roseate Tern

Roseate terns are sensitive to additional sources of mortality due to their generally declining population and certain characteristics of their reproductive biology. The preliminary assessment determines that routine activities associated with construction, operation and maintenance, and decommissioning would result in minor to moderate adverse impacts and would therefore not jeopardize the population of roseate terns.

Displacement of prey fish may occur during construction activities. Temporary but local displacement of roseate terns resulting from an increase in vessel traffic may occur during construction and operation. Roseate terns are known to habituate to the presence of man-made structures and a range of human disturbances. Disturbances associated with construction and operation are expected to be minor and temporary.

The presence of WTGs and the spinning blades may cause terns to avoid the rotor zone of the WTGs, depending on lighting and weather conditions, although proposed turbine spacing would allow for continued use of the area outside of the proposed action boundary and in between turbines.

Roseate terns have demonstrated minimal use of the WTG array area and the majority of flight heights observed in the WTG array area were below the proposed rotor zone. Roseate terns are not expected to regularly occur within the rotor zone of HSS during periods of greatest risk of collision: crepuscular periods, periods of inclement weather, or at night. Roseate terns would be expected to visually detect and avoid turbine structures during periods of good visibility, and would be expected to avoid turbines if they were to occur in the proposed action area at night, depending on visibility. Therefore, the risk of collision of roseate terns with turbine structures is considered low. The recent PVA determines that the predicted collision mortality resulting from the proposed action, 0.8 roseate tern kills per year, would not jeopardize the roseate tern population. Therefore, the potential for collision is expected to have minor adverse impacts to the roseate tern population. The risk of collision for roseate terns should remain low in HSS given that major breeding colonies are not established closer than approximately 16 to 19 mi (25 to 30 km) (distance roseate terns will travel to forage from breeding colonies) from HSS and that dynamic foraging locations do not result in the greater abundance of terns in HSS. Tern nesting and foraging locations are dynamic. If the locations of colony sites or popular foraging areas change, the occurrence and density of roseate terns in HSS could be subject to change. However, multiple years of data collected in HSS have suggested minimal tern activity in HSS relative to other locations in the Sound. Additionally, the major historical locations of breeding terns and their preferred foraging locations are well studied and there is no data that suggest that HSS would become a high tern use area.

It is anticipated that development of the proposed action would not result in an increase of hunting habitat for predatory species. Additionally, there is little overlap of roseate terns and peregrine falcons in the proposed action area. Therefore, increased predation may affect but is not likely to adversely affect roseate terns.

Adverse impacts associated with routine activities of the proposed action are anticipated to have minor to moderate adverse impacts to the roseate tern population. However minor, the project may effect roseate terns, even if only modifying flight and foraging behavior within the Project area, and therefore, mitigation measures have been proposed and further consultation with the USFWS service is anticipated.

3.3.2 Non-routine, Accidental or Unplanned Events

Potential sources of adverse impacts to piping plover during non-routine, accidental, or unplanned events associated with construction, operation and maintenance, and decommissioning include oil spills, cable repair, monopile collapse, geotechnical and geophysical investigations.

Potential sources of adverse impacts to roseate terns during non-routine, accidental, or unplanned events associated with Project construction, operation, and decommissioning include oil spills, cable repair, monopile collapse, and geotechnical and geophysical investigations.

3.3.2.1 Oil Spills

Piping Plover

The presence of WTG and ESP foundations in the vicinity of oil tanker shipping lanes increases the risk of ship collisions, and possibly oil spills. Depending on the size and location of an area impacted by an oil spill, spills could result in the direct mortality or decreased breeding success of piping plovers. If the feathers become coated with oil, birds loose their ability to repel water and to insulate (Jarvis 2005). Potential impacts include mortality from heat loss, starvation, or drowning. Some birds may loose their ability to fly. Mortality can result if toxins are ingested through water or during preening. Also, nesting birds can transfer oil to their eggs resulting in decreases in hatching success, developmental problems, or the mortality of embryos (Jarvis 2005).

Oil spills can impact large areas if the spills are not immediately contained. The coastline of Buzzards Bay was impacted when the *Bouchard No. 120* collided with rocks off the coast of Westport Massachusetts in 2003. Oil was reported as far as Block Island and Middleton, Rhode Island (BBNEP 2003). Piping plover were impacted by the oil spill, particularly at Barney's Joy, Dartmouth Massachusetts. Two piping plover were reported dead as a result of oil slicking. However, overall nesting success that year was not believed to be adversely impacted (BBNEP 2003).

The potential impacts of oil spills associated with the proposed action would be situational depending on the location and size of the area affected by a spill. Large spills or spills that are not quickly contained could result in the loss of piping plover adults or could lead to decreased

nesting success. Oil spills that occur outside of the breeding or dispersal periods could result in no impact to piping plover. Due to the distance between the WTG area and the closest piping plover nesting location (approximately 5 mi [8 km]), the potential for impacts are reduced.

Roseate Tern

The presence of WTG and ESP foundations in the vicinity of oil tanker shipping lanes increases the risk of ship collisions and possibly oil spills. Because terns forage at the water's surface, they are among those species of birds that are particularly vulnerable to oil spills (Jarvis 2005). If the feathers become coated with oil, birds loose their ability to repel water and to insulate, and in some instances, loose the ability to fly. Potential impacts include mortality from heat loss, starvation, or drowning. Mortality can result if toxins are ingested through water or during preening. Also, nesting birds can transfer oil to their eggs resulting in decreases in hatching success, developmental problems, or the mortality of embryos (Jarvis 2005).

Oil spills can impact large areas if the spills are not immediately contained. The coastline of Buzzards Bay was impacted when the *Bouchard No. 120* collided with rocks off the coast of Westport Massachusetts in 2003. Oil was reported as far as Block Island and Middleton, Rhode Island (BBNEP 2003). At least three adult roseate terns were found dead with traces of oil. Roseate terns were discouraged from nesting on Ram Island in 2003 because it was soiled from the oil spill. Consequently, 250 pairs nested on Penikese Island that year and productivity suffered due to the late initiation of egg-laying (BBNEP 2005).

The potential impacts of oil spills associated with the proposed action would be situational depending on the location and size of the area affected by a spill. Large spills or spills that are not quickly contained could result in the loss of roseate tern adults or could lead to decreased nesting success. Oil spills could directly impact roseate tern colonies, as the Ram Island colony was impacted in 2003. However, due to the distance of the proposed action from nesting colonies, oil spills associated with the proposed action are unlikely to impact nesting colonies. Therefore, potential oil spills are anticipated to result in minor adverse impacts to roseate terns.

3.3.2.2 Cable Repair

Piping Plover

The disturbances associated with cable repair activities are expected to be similar to or less than construction activities. Maintenance activities would be restricted to a small area and would be temporary. Due to the 820 ft (250 m) (or greater) buffer of the offshore transmission cable system from the nearest nesting beaches, disturbances including increased human presence and vessel traffic associated with offshore maintenance activities are not anticipated for nesting piping plover. These activities would not impact beach areas as the cables would be under the shoreline.

Roseate Tern

Cable repair activities would be similar to cable installation activities, but would occur for a short period in a small discrete location. Cable jetting, splicing, and re-jetting would result in minor and temporary increases in suspended sediments and would temporarily disturb benthos.

Tern foraging in areas of elevated suspended sediments would be reduced. In both instances the habitat and species would recover and no impacts to roseate terns are anticipated from cable repair activities.

3.3.2.3 Monopile Collapse

Piping Plover

Piping plover are mostly present along the shoreline and coastal beaches, particularly during the breeding season, and monopile collapse would have a negligible affect on this species as this would occur miles offshore.

Roseate Tern

In the event of a monopile collapse, recovery and replacement activities would be similar to decommissioning and construction of a single WTG. A very minor amount of benthic habitat would be disturbed with a short term and localized increase in suspended sediments. Foraging opportunities for terns would be reduced in areas of elevated suspended sediments. Some lubricating fluid would likely leak from the submerged nacelle, but would rapidly disperse given the small quantity involved. However, should a tern dive for fish within this small plume, it could be harmed. There is a low likelihood of this occurrence and low probability of it occurring coincidentally with tern use of the immediate area. Potential impacts to roseate tern in the event of a monopile collapse would therefore be negligible.

3.3.2.4 Geotechnical and Geophysical Investigations

Piping Plover

Prior to receiving authorization to construct the proposed action, the Applicant would be performing additional geotechnical and geophysical investigations to assist in final analysis of certain design elements of the project, such as depth of monopile foundations. Geotechnical investigation methods such as borings would result in negligible effects on benthos and water column characteristics, and these activities would be localized and short term, such that no affects on piping plover habitat or use of the proposed action area are anticipated. In addition, these activities will be focused on the Horseshoe Shoal portion of the proposed action area, and this is distant from most of the frequently used habitats in Nantucket Sound. Geophysical investigation methods, such as sidescan sonar, are even less intrusive and have less habitat altering capabilities, and would, therefore, also have no adverse effects on piping plover.

Roseate Tern

The geotechnical investigation methods such as borings would result in negligible effects on benthos and water column characteristics, and these activities would be localized and short term, such that no affects on roseate tern habitat or use of the proposed action area are anticipated, even though much of this activity will be focused on the Horseshoe Shoal area. Geophysical investigation methods, such as sidescan sonar, are even less intrusive and have less habitat altering capabilities, and would, therefore, also have no adverse effects on roseate terns.

3.3.2.5 Proposed Action Analysis-Non-routine, Accidental, or Unplanned Events

Piping Plover

The impacts associated with accidental events such as oil spills would be situational depending on the location and size of the spill, how quickly the spill is contained, and the season during which a spill were to take place. However, given the distance of the proposed action area from piping plover nest locations, potential impacts are reduced. A spill during the breeding season could result in decreases in piping plover reproductive success or increased mortality of adults. If a spill were to occur when piping plover are not present within the region, no impacts could result. Additionally, there is low probability of the event of an oil spill.

Maintenance activities associated with cable repair would result in minor and temporary disturbances to nesting piping plover due to the 820 ft (250 m) or greater buffer between the offshore transmission cable system and nesting locations. Therefore, disturbances associated with cable repair including increased human presence and vessel traffic are expected to have negligible to minor impacts to nesting piping plover.

Roseate Tern

Roseate terns could be affected by non-routine, accidental, or unplanned events associated with the construction, operation, and decommissioning of the project. These events include oil spills, monopile collapse, cable repair, and geotechnical and geophysical investigations. Depending upon the size and season of an oil spill it could result in a minor impact to the population of roseate terns. Depending on the season, and the size and location of the area affected by an oil spill, a spill in Nantucket Sound could result in the decreased breeding success or mortality of roseate terns. However, an oil spill is an unlikely event. Furthermore, if a spill were quickly detected and contained, negative impacts could be minimized or avoided.

Other activities such as cable repair or monopile collapse are expected to have negligible to minor impacts to the population of the roseate terns because these impacts would be localized and temporary.

The unlikeliness of most of the described non-routine, accidental, or unplanned events, associated with the three phases of the proposed action, combine with the roseate tern habitat use to result in negligible to minor adverse impacts to roseate terns. Therefore, it is anticipated that such unplanned events would not result in ecologically significant or adverse impacts to the roseate tern population.

3.3.2.6 Summary and Conclusion- Non-routine, Accidental, or Unplanned Events

Piping Plover

Potential impacts associated with non-routine, accidental, and unplanned events during construction, operation, and decommissioning could result in negligible to minor impacts to piping plover.

The chance of an event of an oil spill is unlikely, however, potential impacts associated with a spill would depend on the size and location of the spill, the season in which it were to occur, and how quickly it would be contained. The range of impacts associated with an oil spill include no impacts, to decreased breeding success, or the mortality of piping plover. Due to the distance of the proposed action area from nesting locations, potential impacts are reduced. Therefore, oil spills may result in minor adverse impacts to piping plover. Activities associated with cable repair are expected to be temporary and minor. Therefore, these impacts are not predicted to jeopardize the Atlantic Coast population.

Roseate Tern

Potentially minor impacts to roseate terns could result from non-routine, accidental, or unplanned events associated with development of the proposed action. Sources of potential impacts include oil spills, monopile collapse, cable repair, and geotechnical and geophysical investigations.

The impacts associated with accidental events such as oil spills would be situational depending on the location and size of the spill, how quickly the spill is contained, and the season during which a spill were to take place. However, given the distance of the proposed action area from roseate tern colonies, potential impacts are low. A spill during the breeding season could result in decreases in roseate tern reproductive success or increased mortality of adults. If a spill were to occur when roseate terns are not present within the region, no impacts could result. Additionally, there is low probability of an oil spill occurrence.

Cable repair, monopile collapse and geotechnical and geophysical investigations would result in minor and temporary disturbances to roseate tern behavior and use of the proposed action area, therefore, these activities are expected to have negligible to minor impacts to roseate terns.

4.0 CUMULATIVE EFFECTS

Other marine-based activities in the past, present or future that may contribute to cumulative impacts in Nantucket Sound would include activities such as submarine electric cable or gas pipeline installations, harbor and channel dredging and disposal activities, commercial fishing (bottom dragging), commercial shipping, sand and gravel mining, installation of pile-supported or solid fill marine structures, residential and commercial coastal development, coastal recreational activities, beach management activities including beach nourishment, communication tower and navigational structure installations, and other offshore wind power or other sources of alternative energy installations. Of these activities, those that have the potential to occur within the location and/or timeframe of the proposed action and that may contribute to overall impacts to protected whale, sea turtle, and avian species include: offshore sand and gravel mining, offshore wind energy projects and other sources of alternative energy installations, maintenance dredging/beach nourishment, coastal development, coastal recreational activities, and the installation of communication towers and navigational structures. Although other activities listed above could occur in the general proposed action area, a detailed cumulative impact analysis determined that these activities would have little potential for any cumulative impacts to environmental resources, including listed whale, sea turtle, and avian species.

An assessment of the possible cumulative effect of impacts caused by offshore sand and gravel mining, offshore wind energy projects, maintenance dredging/beach nourishment, coastal development, coastal recreational activities, and the installation of communication towers and navigational structures activities when combined with proposed action impacts is provided below.

4.1 Offshore Sand and Gravel Mining

The mining of sand resources for beach nourishment may lead to impacts or increase stress on commercial and noncommercial living resources that utilize the subject extraction sites. The demand for sand to nourish eroding beaches has risen in recent time and will be expected to increase given the rising sea levels and eroding shorelines. Although the direct and indirect impacts are not completely understood, it is expected that benthic communities will be directly disturbed because of the extractive nature of the activity and because the dynamics of the water movement may be influenced which could have the potential to alter sediment dynamics. As for cumulative impacts, it is important to note that if the proposed action is permitted and constructed, sand and gravel extraction within the designated MMS lease area would be precluded.

Presently, there is one proposal for an offshore sand mining project in the vicinity of Nantucket Sound. The Sconset Beach Nourishment Project is proposing a 345 acre (140 hectare) borrow site approximately 3 miles east of Nantucket Island outside of Nantucket Sound and just outside the Cape & Islands Ocean Sanctuary. This project is currently under MEPA review and is contingent upon approval and licensing from several other state and federal agencies including the MMS and the U.S. Army Corps of Engineers.

There is also an expressed interest by the Town of Barnstable to conduct sand mining projects within Nantucket Sound in the vicinity of the proposed action area for future beach nourishment. Although there are presently no approvals for sand mining projects by MMS, the potential for future sand mining activities and associated construction do exist. In the event that the proposed action and an offshore sand mining project occur concurrently in close proximity, there is a potential risk for cumulative impacts to protected marine species associated with the combined activities. However, as noted previously, if the proposed action is permitted, sand mining would be precluded within the designated MMS lease area.

Any cumulative effects associated with the proposed action are expected to be localized and temporary. In addition, as discussed earlier, Nantucket Sound is not considered a high use area for whales and sea turtles have only been occasionally observed in the Sound as transient species. If any listed whales or sea turtle species were to be present in the proposed action area, they would likely temporarily avoid the area during construction activities. If a sand mining project were to occur concurrently with proposed action construction, the sand mining project would have to occur outside of the MMS lease area (the 25 mi^2 [6475 hectare] area). Due to the geographical separation of the proposed action and any future sand mining project, cumulative impacts would be negligible.

Because the proposed action has been sited and designed to avoid, minimize or mitigate potential impacts to protected marine species, any cumulative impacts from concurrent construction of the proposed action and a sand mining project are likely to be limited to those associated with the sand mining activities, with no significant adverse impacts taking place as a result of the proposed action.

4.2 Offshore Wind Projects

Aside from the proposed action, the only other offshore wind power installation that may have the potential to contribute to cumulative impacts to protected marine species is the Patriot Renewables, LLC proposed wind farm in the Cape and Islands Ocean Sanctuary of Buzzards Bay. No other offshore wind installations of any significant size have been proposed off the New England coast. Based on European experiences, each wind farm is expected to have relatively similar impacts on environmental resources although some discrepancies will arise given the differences in scale and location for each commercially-sized project. The potential cumulative impacts associated with the proposed action and the Patriot Renewables, LLC proposed project, if they were to be constructed and/or operated concurrently, are described below.

Impacts associated with the proposed action are expected to be localized and temporary and since the location and schedule for the Patriot Renewables, LLC project are different, the potential for cumulative impacts to listed whale or sea turtle species would be negligible.

Potential cumulative effects may accrue for the Federally Endangered roseate tern or Federally Threatened piping plover as a result of the development of the Patriot Renewables, LLC wind farm. The proposed location of this facility in Buzzards Bay would be in proximity of the major Massachusetts roseate tern nesting colonies, including Bird and Ram Island, and

Penikese Island. Buzzards Bay provides nesting habitat for 99 percent of Massachusetts breeding pairs and for 45-50 percent of North America's breeding pairs. Piping plover also nest along the mainland and island shorelines of Buzzards Bay. The cumulative effects for roseate tern and piping plover may include habitat modification, influences to flight behaviors, or increased collision mortality. Because of these potential impacts, the two projects combined do have the potential for at least moderate levels of cumulative impacts.

The MMA turbine has a maximum height of 74 m (243 ft) and is located at the western entrance of the Cape Cod Canal. The turbine is situated 100 m (328 ft) from the water's edge on a landmass adjacent to a popular common and roseate tern foraging location, the Mashnee Flats Shoal, located 5.3 miles (9 km) from Bird Island. Post-construction surveys indicate that the turbine has not presented a barrier to traveling terns as they have been observed to occur within the 50 m airspace of the turbine, with no apparent hindrances to their movements as they travel between foraging locations. No tern or shorebird mortalities were reported from the 2006 survey period. The turbine has resulted in no measurable impacts to avian species and would not result in cumulative impacts with the proposed action.

4.3 Other Offshore Power Projects

While it is possible that other energy project developers could pursue a generation site within or nearby Nantucket Sound and the proposed action area, only one tidal power project is currently known.

An application has been filed with FERC for a potential tidal power site in the area between Martha's Vineyard and Falmouth/Woods Hole. Construction and operation of a tidal power facility at this location would have cumulative effects on some of the resources also present at the site of the proposed action. For instance, construction could result in noise, elevated turbidity, increased vessel activity, all of which would be temporary, but may result in avoidance behavior of the construction area by terns as well as sea turtles and whales. Given that there is likely to be a schedule difference of at least a year, if not more, this cumulative effect would not adversely affect the protected species.

Operation of the known tidal power project could result in some cumulative impacts over time, since some of the listed species would occur in both project areas, however, there is not enough information known about the tidal power project to predict or discuss these cumulative impacts for all species. However, the potential development of tidal power in the region may result in cumulative effects to avian species in the form of habitat modification or decreased breeding success. Disruptions to the dynamic sedimentation process of inter-tidal areas would result in changes to sediment and turbidity within potential foraging habitat. Avian species may suffer from indirect impacts due to changes of prey base in impacted areas. Impediments to foraging can result in increases in energy expenditure while bird species access alternative resources, as well as decreased breeding success.

4.4 Maintenance Dredging/Beach Nourishment

The offshore transmission cable system for the proposed action would be placed adjacent to the eastern edge of the Federal Navigation Project in Hyannis Harbor. Maintenance dredging of the channel, if initiated at the same time as the jet plow installation of the cable system, could result in additional concurrent, cumulative sediment suspension and deposition and some mortality to benthic resources. Hyannis Harbor was dredged in 1985, 1991, 1998, and 1999. No future dredging activities are currently scheduled. Nonetheless, the potential for cumulative impacts to protected marine species resulting from Hyannis Harbor dredging activities possibly occurring concurrently with the jet-plow installation of the offshore transmission cable system into Lewis Bay is discussed below.

As discussed in Report No. 4.1.1-2, sediment deposition resulting from cable installation would be minimal and localized and would not substantially contribute to any cumulative impact. It is expected that the permits authorizing dredging in Hyannis Harbor would stipulate that proper measures be taken to avoid/minimize impacts during the dredging event. Although interactions between sea turtles and dredging activities have been documented, potential dredging of Hyannis Harbor will not likely occur simultaneously to the proposed action's offshore transmission cable system cable installation and Hyannis Harbor is not an area that has high reported observations of sea turtles. Therefore, the potential for cumulative impacts to occur to listed whale and sea turtle species from concurrent dredging and proposed action cable installation is minimal.

Effects of dredging and beach nourishment activities on shorebirds and terns can be both positive and negative, and the effects appear to be situational. Dredging may change foraging habitat by re-releasing settled toxicants into the water and by altering predator dynamics in the region (Burger 1995). Increased human presence during dredging can affect nesting and foraging tendencies of both terns and shorebirds (Burger 1994). Dredging can also positively affect shorebirds by providing material for creation of habitat when deposed on land. However, in some cases, artificial and stabilized dunes and vegetation may impair avian nesting success by disrupting the natural processes of dune vegetative growth and sand accretion (USFWS 1996). The erection of snow fencing to stabilize dunes may degrade nesting habitat (USFWS 1996). In some cases, beach stabilization has led to the decrease of natural tidal inundations that create favored habitat for nesting piping plover (USFWS 1996).

At sites throughout New England, New York, and New Jersey, beach nourishment has created habitat for piping plover (Burger 1995, USFWS 1996). Additionally, beach nourishment may cause the depletion of bird nesting habitat in nearby areas if the relocated sand becomes ultimately unavailable to the natural sand deposition processes that occur at other habitats in the area. The proper regulation of dredging and beach nourishment activities to benefit avian species of conservation concern are anticipated to negate any potential negative cumulative effects of such activities on piping plover.

4.5 Commercial and Recreational Fishing

Commercial fishing and recreational fishing may result in effects to foraging terns. The roseate tern relies on American sandlance (*Ammodytes americanus*) for approximately 70 percent of its diet (Rock et al., 2007). Studies suggest that declines in food availability are associated with the presence of schools of predatory bluefish (*Pomatomus saltatrix*) (Safina et al., 1988). The catch of large, predatory fish by the fishing industry may positively affect roseate tern by reducing competition for bait fish. Alternately, terns benefit from the presence of predatory fish that push bait fish to the surface where they become more available to foraging terns. Therefore, overfishing may result in negative effects to terns as well. However, terns also catch prey over the shallow water of shoals. Terns are among species of birds that are believed to be attracted to fishing vessels, therefore, the presence of commercial and recreational fishing boats may attract terns to an area to forage (Borberg et al., 2005, Drewitt and Langston 2006, Sadoti et al., 2005a). Commercial and recreational fishing do not have any effect on piping plover, and contrary effects on roseate terns; therefore cumulative effects are expected to be negligible between fishing and the proposed action.

4.6 Commercial Shipping/Vessel Collisions

The presence of WTG and ESP foundations in the vicinity of oil tanker shipping lanes increases the risk of ship collisions and possibly oil spills. The potential cumulative effects of oil spills associated with the proposed action would be situational depending on the location and size of the area affected by a potential spill. Large spills or spills that are not quickly contained could result in the mortality of terns or shorebirds, or could lead to decreased nesting success. Oil spills could directly impact roseate tern colonies or piping plover nesting habitat as occurred in 2003. However, due to the distance of the proposed action from nesting locations, and the fact that oil spills are unlikely events, oil spills associated with the project are expected to result in negligible cumulative impacts.

4.7 Military Operations

Military operations in the project area are minimal, and none occur that are likely to have adverse affects on the listed avian species, and would therefore not result in cumulative effects.

4.8 Bird Collisions with Human Structures

There is the potential for avian cumulative effects in the form of collision mortality associated with other tall, man-made structures. The results of mortality surveys indicate that birds killed by communication towers are mainly neotropical migratory songbirds. The majority of collisions of migratory songbirds appear to occur at night when they may become disoriented by required navigational lights on towers, particularly during inclement weather (Shire et al., 2000). The US Fish and Wildlife Service estimates that 4 to 5 million birds are killed annually at communication towers (Shire et al., 2000). Over 200 species of birds have been documented to die as a result of collision with communication structures. A summary of 47 mortality studies conducted in the U.S. indicates that tern, gull, and petrel species represent 2 percent of tower collision fatalities; and shorebird species represent 5 percent of documented mortality. Roseate

terns may be susceptible to collisions with land-based man-made structures including communication towers and lighthouses as they will travel as far as 16 to 19 miles (25 to 30 km) from the breeding colonies to access foraging habitat (Heinemann 1992), and some of their movements result in crossings of sections of land. However, there is a lighthouse on Bird Island, one of the most important colony sites in the region, and the lighthouse has resulted in no apparent detrimental impacts to the nesting terns. Outside of migration, piping plover are less susceptible to collision with land-based structures as the majority of their movements are small-scale between nesting areas and inter-tidal zones.

Lighthouses are among the first man-made structures known to result in avian collision mortality (Jones and Francis 2003). Photopollution associated with lighthouse beams can have detrimental effects on the navigation systems of insects, sea turtles, and birds (Jones and Francis 2003). Collisions with lighthouse structures may occur, particularly during cloudy or foggy conditions; disoriented migrants may also die as a result of exhaustion after circling light sources; and migrants may also become more susceptible to predation after expending energy to circle around a light source (Jones and Francis 2003). A study on the impacts to migrants posed by a lighthouse on Lake Erie in Ontario documented up to 2000 bird kills in a single night. Neotropical migratory songbirds consisted of the majority of collisions. However, after alterations to the light beam (decreasing the intensity and narrowing the beam) the kill rate decreased.

Terns and shorebirds are among species of birds that have experienced relatively low collision mortality with other man-made structures. Therefore, the cumulative impacts associated with collision mortality due to other man-made structures and those associated with the proposed action are anticipated to be negligible.

4.9 Coastal Development

Coastal development has contributed to the population declines of both roseate tern and piping plover. Residential and commercial development decreases suitable habitat for these species while increasing disturbances associated with human presence in the area. Of particular concern is the construction of hardened structures, including jetties, piers and groins, designed to impede natural variability in inter-tidal zones. Studies have found that shorebirds, including the piping plover, rely on many of the characteristics of a volatile coastal environment including over-wash fans, sand pits, open vegetation, and ephemeral pools for nesting and foraging habitat (USFWS 1997, Houghton 2005). Loss of sand accretion during natural offshore drift processes due to artificial beach barriers may result in the erosion of habitat, therefore, such coastal development is directly related to habitat modification or loss. Alternately, artificial beach barriers such as jetties may result in the creation of habitat on the up-drift side of an artificial structure (USFWS 1996). However, excessive sand accretion may result in vegetative growth that may eventually make habitat less suitable to piping plover. Continued management efforts to maintain or restore habitat, the creation of additional nesting habitat, and the regulation of additional development activities are anticipated to mitigate the cumulative effects of coastal development. Since the proposed action avoids shoreline disturbance through the use of HDD to installed the cables, no cumulative effects with other shoreline disturbing activities are anticipated.

4.10 Coastal Recreational Activities

Human recreation in coastal areas results in disturbances to breeding or staging shorebirds and terns. Human disturbances may result in increased energy expenditure or decreased nesting success for avian species. Piping plovers, for example, have been shown to choose nesting and foraging habitat of sub-optimal quality to avoid nesting in areas with a relatively high human presence. Piping plovers will permanently abandon nest due to significant levels of disturbance. Terns may abandon their nests in the presence of humans, leaving eggs or chicks vulnerable to chilling or predation (USFWS 2001).

Where permitted, the use of motor vehicles on beaches has negative impacts on foraging and nesting shorebird and tern species. Motorized vehicles can crush eggs and chicks, cause noise pollution, damage breeding or foraging habitat, create ruts that disrupt foraging patterns and trap chicks, and can scare brooding adults away from their nests leaving eggs vulnerable to cooling as well as predation (USFWS 1996). However, management practices are geared to minimizing the impacts of human activities at important nesting areas. Efforts include fencing and signs and the restriction of access of pedestrians and vehicles from breeding areas between the period of hatching to fledging. The Applicant will perform work on the shoreline crossing during non-breeding seasons, and would therefore not generate additional effects related to human activities on the beach.

5.0 EFFECTS DETERMINATION FOR LISTED SPECIES AND DESIGNATED CRITICAL HABITATS

Though there is potential for adverse impacts on the listed species from the proposed action, these impacts will be temporary and localized to the vicinity of the proposed action area. No actions are expected to be lethal or long-term for either the group of whales or sea turtles described in this report. The extent of any impacts of proposed actions is further reduced in that those whales and sea turtles described are rarely observed in the vicinity of the proposed action area and any occurrence is presumed to be transient and temporary.

The greatest risks to sea turtles and whales are posed by acoustical harassment and water quality degradation. Increased noise levels due to pile driving and jet plowing are expected to result in only minor impacts to both whales and sea turtles, primarily resulting in the temporary avoidance of the area where construction is occurring. Once construction ceases, it is expected that the whales and sea turtles will resume their normal activities in the area. Table C-4 provides a summary of the determination of effect for each of the listed species covered in this evaluation. Only four of the nine species covered were determined to be affected in some manner. However, only the roseate tern and piping plover are likely to be adversely affected from construction, operation and maintenance, or decommissioning activities.

Table C-4. Effects Determination Summary

Group	Common Name	Effect Determination	Basis
Whale	Fin	No Effect	Frequency of occurrence in the proposed action area is extremely low. Vessels would be relatively slow moving. Affects of disturbance within HSS unlikely to influence higher use areas outside of Nantucket Sound
	Northern Right	No Effect	Frequency of occurrence in the proposed action area is extremely low. Vessels would be relatively slow moving. Affects of disturbance within HSS unlikely to influence higher use areas outside of Nantucket Sound
	Humpback	No Effect	Frequency of occurrence in the proposed action area is extremely low. Vessels would be relatively slow moving. Affects of disturbance within HSS unlikely to influence higher use areas outside of Nantucket Sound

Table C-4. Effects Determination Summary			
Group	**Common Name**	**Effect Determination**	**Basis**
Sea Turtle	Leatherback	No Effect	Frequency of occurrence in the proposed action area is extremely low. Vessels would be relatively slow moving. Affects of disturbance within HSS unlikely to influence higher use areas outside of Nantucket Sound
	Green	No Effect	Frequency of occurrence in the proposed action area is extremely low, and they tend to be in shallow waters with seagrasses. Affects of disturbance within HSS unlikely to influence higher use areas outside of Nantucket Sound.
	Loggerhead	May affect, but not likely to adversely affect	Feeds on benthic fouling organisms likely to develop on monopiles and rock scour armor so could be attracted to WTGs where they could experience greater interaction with maintenance vessels and recreational fishermen. One of the more common sea turtles in the area so more potential for interaction with construction and operation vessels.
	Kemp's ridley	May affect, but not likely to adversely affect	Feeds on benthic organisms likely to develop on monopiles and rock scour armor. One of the more common sea turtles in the area so more potential for interaction with construction and operation vessels.
Bird	Piping Plover	Is likely to adversely affect, not likely to jeopardize	It has been predicted that there would be some level of collision mortality associated with the Project. The conservative/high results of the collision probability analyses would not substantially increase the risk of extinction for the species. There is uncertainty in collected data, and future predictions.
	Roseate Tern	Is likely to adversely affect, not likely to jeopardize	It has been predicted that there would be some level of collision mortality associated with the Project. The conservative/high results of the collision probability analyses would not substantially increase the risk of extinction for the species. There is uncertainty in collected data, and future predictions.

6.0 IMPACT AVOIDANCE, MINIMIZATION, AND MITIGATION MEASURES

6.1 Whales and Sea Turtles

Consultations with other Federal agencies are being conducted in order to confirm the potential for impact, and to confirm the appropriate mitigation measures to minimize potential impacts. While limited localized impacts are anticipated during construction and operations, measures would be implemented to prevent and minimize these impacts. These measures include posting NMFS-approved observer(s) on-site during pile driving activities; using state-of-the-art hydraulic jet plow technology for cable installation; and implementing post-construction monitoring to document habitat disturbance and recovery. Specific mitigation measures that would be employed to minimize the potential for vessel collisions, vessel harassment and acoustic disturbance to ESA protected marine species are described below.

6.1.1 Measures to Reduce Likelihood of Vessel Collisions and Harassment

As previously mentioned, the proposed action would temporarily increase the number of vessels within the vicinity of the construction area, especially in the route between Quonset, Rhode Island and the proposed action area. Several shipping lanes and two navigational channels exist within the vicinity of the proposed action area, normally producing high volumes of vessel traffic within the vicinity of the proposed action area. During construction activities, especially during pile driving activities, it is estimated that 4-6 stationary or slow moving vessels would be present in the general vicinity of the pile installation. Vessels delivering construction materials or crews to the site would also be present in the area between the mainland and the WTG array site. The barges, tugs and vessels delivering construction materials would be limited to speeds below 10 knots (18.5 km/h) and may range in size from 90 to 400 ft (27.4 to 122 m), while the vessels carrying construction crews will be traveling at a maximum speed of 21 knots (39 km/h) and would typically be 50 ft (15 m) in length. The additional traffic from construction vessels may increase the chance of a strike or harassment of ESA protected marine species.

Potential vessel impacts (collisions and harassment) to marine mammals and sea turtles would be minimized by requiring the vessels to follow *NOAA Fisheries Regional Viewing Guidelines-Northeast Region* (NMFS and NOS, 2006) while in transit to and from the proposed action area so as to minimize disturbance of any individuals that may be in the area. In addition, vessel operators would follow guidelines implemented for the Gulf of Mexico oil and gas lease industry for vessel strike avoidance (MMS 2003). These guidelines are summarized below:

- Vessel operators and crews should maintain a vigilant watch for marine mammals and sea turtles and slow down or stop their vessel to avoid striking protected species.
- When whales are sighted, maintain a distance of 295 ft (90 m) or greater from the whale.
- When sea turtles or small whales are sighted, attempt to maintain a distance of 148 ft (45 m) or greater whenever possible.

- When whales are sighted while a vessel is underway, attempt to remain parallel to the animal's course. Avoid excessive speed or abrupt changes in direction until the whale has left the area.
- Reduce vessel speed to 10 knots (18 km/h) or less when pods or large assemblages or whales are observed near an underway vessel. Whales at the surface may indicate the presence of submerged animals near the vessel.
- Whales may surface in unpredictable locations or approach slowly moving vessels. When you sight animals in the vessel's path or in close proximity to a moving vessel, reduce speed and shift the engine to neutral. Do not engage the engines until the animals are clear of the area.

Project vessel operators and crew would be required to undergo training to ensure they are familiar with NOAA regional viewing guidelines and MMS guidelines for vessel strike avoidance to minimize encounters and interactions with marine mammals and sea turtles. These training requirements would be written into contractor agreements/contracts.

6.1.2 Measures to Reduce Likelihood of Acoustic Harassment

The Applicant has proposed to mitigate any minor noise impacts that may occur through utilization of the following measures:

Underwater Sound Monitoring

Underwater sound monitoring is proposed during the installation of the first three monopiles. The monitoring would be conducted in order to confirm that actual pile driving noise levels do not exceed the levels utilized for conducting the analysis presented. MMS recommends that underwater sound monitoring be conducted during installation of all monopiles because differences in subsurface materials from site to site could affect actual underwater noise levels during pile driving.

NMFS Approved Observer

The Applicant has proposed to utilize a marine mammal observer to insure that no marine mammals or sea turtles are within the 1640 ft (500 m) safety radius during pile driving. MMS recommends that the observer be utilized during all pile driving activities at all monopile locations.

Soft Start Pile Driving

The Applicant has proposed to utilize a "soft-start" of pile driving equipment in order to allow any marine life not visible from the surface to move away from the monopiles. MMS recommends the use of soft-start pile driving at all monopile locations.

Underwater Electromagnetic Field (EMF)

The proposed inner-array and offshore transmission cable systems would contain grounded metallic shielding that effectively blocks any electric field generated by voltages on the conductors within the cable systems. Since the electric field would be completely contained within those shields, impacts are limited to those related to the magnetic field emitted from the offshore transmission cable system and inner-array cables. As described in Report No. 5.3.2-3, the magnetic fields associated with the operation of the inner-array cables or the offshore transmission cable system are not anticipated to result in an adverse impact to marine organisms, including T&E species.

6.2 Avian Species

6.2.1 Measures to Minimize Bird Strikes

Roseate Terns

The above water foundations, WTGs, and the ESP would be equipped with stainless steel wire and vision restriction perch deterrent devices. Each turbine foundation would have a deck which would be covered by aluminum chain link fencing to discourage access on the sides (the deck overhangs the access ladder). There would be a taught 0.1 in (3 mm) stainless steel wire on top of the railing, and a 2.1 ft (0.65 m) solid panel to restrict the view of birds from the deck (some species prefer perches with views). The spacing between the wire and the rail would be 1.2 in (3 cm). The ESP would have a perimeter railing and the ladders and railing of the ESP would be equipped with stainless steel wire, chain-link fence, and panels similar to the WTG foundations. The use of perch deterrent devices has discouraged terns perching on the fence and deck of the platforms supporting the SMDS. The final design of perch deterrents would be based on recommendations from USFSW. The use of tubular towers instead of lattice towers also discourages perching under the rotors. Vibrations and low level noise created by operating WTGs may also deter terns from perching.

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Report reference format, i.e. Report No C-1, refers to a report referenced within this appendix only.

Numbered reports reference format, i.e. Report No. 4.1.1-2, refers to a report originally referenced in Section 4.1.1 of the DEIS. All numbered reports occur in order of first appearance within the DEIS.

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