Appendix J

FWS and NOAA Fisheries Biological Opinions

FWS Biological Opinion

United States Department of the Interior

FISH AND WILDLIFE SERVICE
New England Field Office
70 Commercial Street, Suite 300
Concord, New Hampshire 03301-5087
http://www.fws.gov/northeast/newenglandfieldoffice

Re: Final Biological Opinion, Cape Wind Associates, LLC, Wind Energy Project, Nantucket Sound, Massachusetts Formal Consultation # 08-F-0323

November 21, 2008

Mr. James Kendall
Chief, Environmental Division
Minerals Management Service
Washington, D.C. 20240

Dear Mr. Kendall:

This document transmits the Fish and Wildlife Service's (Service) biological opinion (BO) based on our review of the Minerals Management Service (MMS) proposed issuance of a lease or easement to Cape Wind Associates, LLC (CWA), to construct, operate and decommission a wind energy project on Horseshoe Shoal in the federal waters of Nantucket Sound, Massachusetts, and the effect on the threatened piping plover (*Charadrius melodus*) and endangered roseate tern (*Sterna dougalli dougalli*). This document was prepared in accordance with section 7 of the Endangered Species Act of 1973 (ESA), as amended (16 U.S.C. 1531 *et seq*.).

This BO is based on information provided in the MMS May 2008 biological assessment (BA), subsequently provided supplemental project information, and other sources of information cited herein. A complete administrative record of this consultation is on file at the Service's New England Field Office.

If you have any questions regarding this opinion, please contact Mr. Michael Amaral or Susi von Oettingen of my staff at (603)223-2541, or at the letterhead address.

Sincerely.

Thomas R. Chapman
Supervisor
New England Field Office

Attachment

Biological Opinion
for the
Cape Wind Energy Project
Nantucket Sound, Massachusetts

This document, dated November 21, 2008, is the U.S. Fish and Wildlife Service's (Service) biological opinion (BO) pursuant to our formal section 7 consultation under the Endangered Species Act of 1973 (ESA), as amended (16 U.S.C. 1531 *et seq*.). Minerals Management Service's (MMS) May 19, 2008 request to initiate formal consultation was received by the Service on May 20, 2008. MMS, the lead federal agency, is also consulting with the Service on behalf of the Army Corps of Engineers (ACOE) and the Environmental Protection Agency, the additional federal agencies with approval or permitting authorities for the Cape Wind Project. Cape Wind Associates, LLC (CWA) proposes to generate electricity from wind energy on the outer continental shelf; some components of the facility are located in waters within three miles of the coast and onshore. The proposal calls for construction of 130 wind turbine generators and associated infrastructure commencing in 2009 and operations beginning in 2010 (as described in the January 2008 Draft Environmental Impact Statement).

The ESA-listed species under the jurisdiction of the Service that are considered in this formal consultation are the threatened Atlantic Coast piping plover (*Charadrius melodus*) population and the endangered northeastern population of the roseate tern (*Sterna dougallii dougallii*). There is no critical habitat designated pursuant to section 4 of the ESA within the Horseshoe Shoal marine environment or elsewhere within the project area for either avian species. Similarly, there are no species currently proposed for ESA listing as threatened or endangered that may be present in the project area. We have also evaluated the potential effect of the project on the threatened northeastern beach tiger beetle (*Cicindela dorsalis dorsalis*), which occurs on the periphery of the project area, and concur with your evaluation, dated October 9, 2008, that the project is not likely to adversely affect this species. We based our concurrence on information provided in your October 9, 2008 letter and an analysis of the probability for an oil spill attributable to the Cape Wind Project (Etkin 2006) to reach a finding of not likely to adversely affect the northeastern beach tiger beetle or its habitat in Nantucket Sound.

The Service underscores that this BO only applies to the roseate tern and piping plover, as listed species under the ESA. Under numerous prior correspondence, the Service submitted comments regarding other avian trust resources, the potential for those resources to be impacted by the project, and MMS's treatment of those species in its draft environmental documentation. The contents of this BO should not be construed or extrapolated to apply to non-listed avian species, which are present in the project area in different numbers and frequency, and which exhibit different life histories, ecology and behaviors. For instance, the Service's conclusions about the level of anticipated effects, the vulnerability to collision, the utility of radar for research and monitoring apply only to the roseate tern and piping plover.

Information Standard
Section 7(a)(2) of the ESA requires that federal agencies undergoing consultation use the best scientific and commercial data available. The regulations implementing this section reiterate that both action agencies and the Service must employ this information standard in carrying out their consultation responsibilities [50 CFR §402.14(d) and (g)(8)].[1] The Service's Policy on Information Standards Under the Endangered Species Act [59 FR 34271 (July 1, 1994)] calls for the review of all scientific and other information to ensure that the information used by the Service to implement the ESA is reliable, credible, and represents the best scientific and commercial data available. The regulations [(50 CFR §402.12(d)(2)] also state that the Service may recommend discretionary studies or surveys that may provide a better information base for the preparation of a biological assessment. However, any recommendation for studies or surveys is not to be construed as the Service's opinion that the federal agency has failed to satisfy the information standard of section 7(a)(2) of the ESA. The Service's Consultation Handbook [section 1.2(D)] states that, where significant data gaps exist, the agencies can agree to extend the due date of the biological opinion until sufficient information is developed for a more complete analysis, or the Service can develop the biological opinion with the available information, giving the benefit of the doubt to the species. The Service's regulations again reiterate this point, noting "if no extension of formal consultation is agreed to, the Director will issue a biological opinion using the best scientific and commercial data available" [50 CFR §402.14(f)].

Uncertainty arising from a lack of information is inherent in biological evaluations and, when significant, can limit confidence in the conclusions drawn from the information or, in some cases, make drawing conclusions at all extremely difficult. Although Service policy regarding significant data gaps calls for giving the benefit of the doubt to the species, the Service cannot forgo addressing uncertainty within certain aspects of a project or its effects, to the best of its abilities. For example, the Service may find that surrogate species information can be used to address uncertainty arising from incomplete or contradictory information. When the scientific literature, surrogate species information, and expert opinion do not support consistent biological determinations, the Service must consider the weight of scientific authority in evaluating expert opinions and the basis for those opinions.

In the course of reviewing and commenting on the Cape Wind Project under all of its applicable authorities, the Service recommended several studies to more fully assess the project's impacts, particularly impacts on migratory birds. Certain information was collected, and some was not. While they would have generated information useful to assessment of migratory birds generally, the unimplemented studies would not necessarily yield information that would have significantly addressed the uncertainties in the analysis of impacts to the roseate tern and piping plover specifically. Other recommended studies (e.g., acoustic monitoring) involve techniques that are relatively untested in the offshore environment, and their potential to yield useful information about roseate terns and piping plovers is, as yet, unproven.

Another ubiquitous issue in ESA decisions is the robustness of available data. More samples over longer time periods increase confidence that natural variability inherent to natural systems

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1 Section 402.14(g)(8) also states that the Service "will give appropriate consideration to any beneficial actions taken by the Federal agency or applicant…"

has been captured. However, while cautious scientists always value additional data, benefit of doubt to the species can be conferred by other means. For example, cushions can be added to best existing estimates, or sensitivity tests can be performed to explore effects of higher or lower values. Information from one location can be compared with larger data sets collected elsewhere and potential reasons for any apparent differences can be evaluated.

This BO contains (among other things) a description of the project, species affected, and anticipated impacts. We based our findings on our independent review of the best scientific and commercial data available. In doing so, we reviewed field reports and investigations by Service staff and others, evaluated information in our files and the scientific literature, and conducted interviews with species and technical experts regarding species ecology, phenology, behavior, and the effects of wind turbines on birds. Service biologists also visited two wind power projects within the North Atlantic breeding range of the endangered roseate tern and threatened piping plover and independently reviewed post-construction avian mortality studies to assess their methodologies, limitations and utility.

Moreover, in preparing this BO, the Service reviewed the results of various models (e.g., collision, population viability, oil spill trajectory models) provided by CWA with respect to the effects of the proposed project on the local and rangewide populations of the roseate tern and piping plover. Models are used to synthesize complex data and assist in the formulation of predictions. Our confidence in a model's projections is determined by a number of factors, such as the amount and quality of the available data and understanding of relevant physical and biological processes; appropriate variance in input estimates and number of model iterations (for stochastic models); use of sensitivity tests to explore effects of changes in parameter estimates; consistency of model projections with those of other related models (or logical explanations for deviations); and comparisons between model projections and empirical evidence from past experience with similar or related questions. In assessing the results of the models used to explore the potential impacts of the Cape Wind Project, we carefully considered these factors and judiciously considered model predictions in the formulation of our findings.

We also considered and independently reviewed information from the following sources: MMS's May 2008 Biological Assessment (BA) for the Cape Wind Energy Project - Nantucket Sound; the January 2008 MMS Draft Environmental Impact Statement (DEIS); the February 15, 2007 final Environmental Impact Report (FEIR); the September 19, 2008 Framework for the Avian and Bat Monitoring Framework for the Cape Wind Proposed Offshore Wind Facility; numerous meetings, conference calls, workshops, and exchanges of information summarized in the Consultation History section (Appendix 1).

<u>Consultation History</u>

There is an extensive consultation history on the Cape Wind Project that spans the time frame from July 2001 to the present, and consists of several hundred articles of correspondence, including letters, telephone conversation records, electronic communications, and meeting summaries. A detailed chronological listing of the consultation history following the passage of the Energy Policy Act of 2005 in August 2005, when MMS assumed permitting and environmental review responsibility from the ACOE, to the present is provided in Appendix 1.

BIOLOGICAL OPINION

This BO addresses the effects of all activities associated with Cape Wind Associates' application to lease, construct, operate, maintain, and decommission a wind energy project on Horseshoe Shoal in Nantucket Sound, Massachusetts.

DESCRIPTION OF THE ACTION

As described in your letter of May 19, 2008 to the Service's Region 5 Regional Director Marvin Moriarty, the proposed project would consist of the construction of 130 wind turbine generators (WTGs) arranged in an array one-half mile to one-third mile apart across a 25-square-mile area on Horseshoe Shoal in Nantucket Sound. The project is approximately six miles offshore from Hyannis and 19 miles from South Beach, Chatham on the south shore of Cape Cod, 14 miles from Nantucket and nine miles from Edgartown, Massachusetts. The wind energy facility is designed for a maximum energy capacity of 454 megawatts (MW) with an average generation capacity of approximately 182.6 MW. A detailed description of the action and the project area is found in section 2.0 in the BA (pages 2-1 to 2-9), as well as in section 2.0 (pages 2-1 through 2-29) of the DEIS.

The BA concludes, and the Service concurs for the reasons explained in this BO, that the primary impact to ESA-listed birds is from collision injury and mortality with the WTGs, monopoles and electrical service platform (ESP), although there may be minor impacts from potential oil spills and associated construction activities. The following is a brief summary of the salient components of the proposed facility taken from pages 2-3 through 2-5 of the BA and the 2008 Framework for the Avian and Bat Monitoring Plan for the Cape Wind Proposed Offshore Wind Facility (Monitoring Framework):

- WTG Rotor Diameter: 364 ft (111 meters).
- WTG Monopole and Nacelle Hub Height: 275.5 ft (84 m) required to maintain a minimum of 75 ft (23 m) clearance between the rotor blade at the low point in its arc and the surface of the water.
- Overall WTG Height: 440 ft (134 m).
- Rotor-Swept Zone: 75 ft (23 m) to 440 ft (134 m), a circular area equal to 2.4 acres (1 ha) per WTG.
- Generating capacity: ±3.6 MW with a cut-in wind speed of approximately 8 mph (J. Lewandowski, MMS, electronic correspondence, October 1, 2008).
- WTGs have a stated life span of a minimum of twenty years.
- WTGs generate electricity independently of each other.
- Submarine inner array cables from each WTG interconnect within the grid and terminate on the ESP.
- The ESP is a fixed template platform of approximately 100 ft by 200 ft (30.5 m by 61 m), and approximately 39 ft (11.9 m) above mean lowest low water.
- The ESP is located within the approximate center of the WTG array and serves as the common connection point for all WTGs. Circuit breakers and transformers are interconnected with the cable systems in order to transmit power through shore-connected submarine cable systems.

- The shore-connected submarine cable travels northeast in Nantucket Sound into Lewis Bay, making landfall at New Hampshire Avenue in Yarmouth, Massachusetts.
- The proposed onshore transmission cable route to its intersection with the NSTAR Electric right-of-way (ROW) would be located entirely along existing paved underground utilities' ROWs. A portion of the onshore transmission cable route would be located underground within the existing maintained NSTAR Electric ROW.
- Construction equipment, wind turbine components and supplies will be staged at Quonset Point, Rhode Island. Maintenance vessels may be staged out of New Bedford and/or Falmouth, Massachusetts.
- CWA will prepare a Spill Prevention Control and Countermeasure Plan prior to operation of the facility in order to prevent contamination of wildlife and the environment, including roseate terns, piping plovers and their habitats (see 9.3.2 of the DEIS).

Installation of transmission cable

Approximately 12.5 circuit miles (20.1 km) of transmission cable will be installed in two circuits. The submarine transmission line will be installed via jet plow embedment over a period of two to four weeks. Installation in Lewis Bay will take one to two days, passing by Egg Island only in a few hours (this will occur twice to complete the circuit) (R. Pachter, electronic correspondence, 2008). Measures to minimize sedimentation resulting from the submarine cable embedment include 1) a turbidity curtain placed near the eelgrass bed near Egg Island, 2) monitoring of staging terns on Egg Island when the cable-laying ship passes by, and 3) to the maximum extent possible, CWA will avoid construction near Egg Island during low tide if construction occurs between mid-July and mid-September.

Framework for the Avian and Bat Monitoring Plan for the Cape Wind Proposed Offshore Wind Facility

The Framework for the Avian and Bat Monitoring Plan for the Cape Wind Proposed Offshore Wind Facility (Monitoring Framework) is an outline for developing pre-construction monitoring and post-constructing monitoring protocols for avian and bat presence in the project area and effects to these species from the wind project. The matrix provided in the Monitoring Framework summarizes pre- and post-construction survey methodology (with the exception of the anti-perching monitoring). The Service, MMS and CWA coordinated in the development of the Monitoring Framework with respect to pre- and post-construction monitoring for roseate terns and piping plovers. Monitoring protocols for other avian species and bats were not addressed in great detail during the coordination and review of draft versions of the Monitoring Framework.

This BO comments only on those provisions of the Monitoring Framework that relate specifically to roseate terns and piping plovers. For the roseate tern and piping plover, the Monitoring Framework proposes to:

1. Test the effectiveness of anti-perching devices (see Conservation Measures below) during pre-construction and post-construction. Adjustments to the anti-perching measures will be made in coordination with MMS and the Service based on the pre-construction monitoring results of perching deterrent devices on the met tower and ESP. Anti-perching device monitoring specifics are found on page 15 of the Monitoring Plan.

2. Monitor piping plover and roseate tern movement within the project area using radio telemetry. Pre-construction testing of the methodology would occur on surrogate species (common terns and semipalmated plovers). Post-construction radio tracking of piping plovers and roseate terns would be dependent on the outcome of the pre-construction radio telemetry monitoring effort.
3. Monitor avian use of the project area using acoustic technology. Acoustic monitoring is proposed for pre- and post-construction surveys to determine species' presence and possible passage rates.
4. Conduct visual aerial and boat surveys. For the purposes of endangered species monitoring, the visual surveys will be limited to supplement or ground-truth proposed radio telemetry monitoring (page 4 of the Monitoring Framework) and/or will be used to document and monitor post-construction changes in roseate tern use of the project area (page 16 of the Monitoring Plan).
5. Monitor post-construction collision mortality. A Thermal Animal Detecting System (TADS) consisting of a thermal video camera, data logger and software is tentatively proposed for monitoring the incidence of post-construction avian collisions at the WTGs and ESP. The methodology may be revised after coordination with MMS and the Service if newer technology is developed (e.g., blade collision sensors).
6. Provide monitoring reports at the end of the pre-construction surveys and annually by December 15 once post-construction surveys have been initiated.

Conservation measures implemented to reduce adverse effects
Conservation measures, as opposed to conservation recommendations discussed at the end of this BO, represent actions pledged in the project description that the action agency or CWA will implement to minimize adverse effects to roseate terns and piping plovers and to further the recovery of the species under review. As a result of extensive discussions with MMS and CWA during the informal consultation for this action, a number of conservation measures that make the Cape Wind Project less likely to adversely affect threatened and endangered species were incorporated into the project description. Some of these measures are described on pages 8-10 to 8-15 of the BA and in section 5.3.2.4.2 of the DEIS.

The following are conservation measures incorporated into the project proposal:

Perching Deterrents
Birds that perch on wind turbines are likely to be at increased risk of collision mortality, particularly if they use the structures for nesting or to initiate courtship displays. Roseate terns could be subjected to avian predation, if raptors opportunistically use the WTGs or ESP as hunting perches. CWA's proposed design, which utilizes monopole or tubular supporting towers for the WTGs, will minimize the likelihood that terns, plovers or other birds will find perching opportunities readily available on the structures. Additional perching deterrent devices (wires, paneling and fencing) will discourage birds from perching on railings and deck areas of the WTGs and the ESP. CWA has committed to field testing anti-perching wire at an alternate location prior to implementation of the project (Appendix B, section 2.2.3 of the BA; 2008 Monitoring Plan, pages 13-15). CWA proposes to monitor the effectiveness of the anti-perching devices for three months (May, June and July) during two years and evaluate the use of alternative or additional devices if warranted. If perching remains an issue based on these

monitoring results, CWA will propose additional anti-perching mechanisms for approval by MMS in coordination with the Service, and then consequently monitor the effectiveness of these new devices during the breeding season from mid-May to late July and the staging season from mid-August to late September.

Oil Spill Planning and Preparedness

MMS requires a draft Operation and Maintenance Plan that details standard operating and maintenance protocols to ensure proper operation of offshore facilities. The draft O&M Plan specifies operating guidelines, maintenance schedules, and materials approved for maintenance activities. Within the Operation and Maintenance Plan, CWA would be responsible for developing and implementing an Oil Spill Response Plan (OSRP) covering all phases of the proposed action. The OSRP would be prepared in accordance with the DOI's MMS regulations at 30 CFR §254, "Oil Spill Response Requirements for Facilities Located Seaward of the Coastline."

The Spill Prevention Control and Countermeasure Plan proposed by CWA states that it will meet the requirements of an MMS OSRP by addressing the vessels involved in construction and maintenance of the wind farm, and assessing the different types and amounts of oil products used in the WTGs and the ESP. MMS required that an oil spill risk assessment and fatality study be conducted; these are included in the DEIS (Etkin 2006; Knee *et al*. 2006) and meet the requirements of the OSRP according to MMS. In the event there is an oil spill related to the Cape Wind Project, response activities occurring within roseate tern and piping plover breeding, roosting and foraging habitat shall include measures to avoid adversely affecting these species and their habitats.

Lighting

The Cape Wind Project proposes to light 50 WTGs on the perimeter of the array and the eight WTGs adjacent to the ESP at night for aviation safety, each lit with a single flashing red light. The 72 interior WTGs would not be lit with aviation lighting at night. All aviation lights will be synchronized and flashing at 20 flashes per minute. WTG and ESP lighting meets FAA and U.S. Coast Guard requirements (DEIS) and uses a minimum number of lights of medium to low intensity and a minimum number of flashes per minute per the Service's interim guidelines on avoiding and minimizing wildlife impacts from wind turbines (USFWS 2003).

Bird Island Restoration Project

Bird Island is the second largest roseate tern nesting colony for the species in the western North Atlantic and is both the largest colony in Massachusetts and the largest colony in proximity to the Horseshoe Shoal project area. It is also consistently one of the most productive breeding sites for the species in the Northeast, with nesting roseate terns producing about 1.2 young per pair [Roseate Tern Recovery Team (RTRT) 2007].

Roseate tern nesting habitat on the 3-acre Bird Island is deteriorating due to erosion and salt water intrusion through the crumbling, 160-year-old revetment that surrounds and protects the island. Presently, only 1.5 acres is above the mean high water spring tide (ACOE 2005). Erosion of the island has lowered ground elevations (from 4-10 ft mean low water), changing portions of the island from sand and gravel to salt marsh and salt pannes (ACOE 2005). These lower areas are unsuitable as nesting habitat and have reduced the extent of the area available on the island for nesting by terns. As a consequence, the island can physically support fewer nesting pairs of terns, leading to crowding between the 1,800 pairs of common terns and ~1,000 pairs of roseate terns that nest there annually. Even without consideration that there may be an accelerated rate of erosion due to the deteriorating condition of the revetment and the potential for sea level rise, another 0.25 acre of tern nesting habitat is projected to be lost in the next 25 years if steps are not taken to "restore" the island (ACOE 2005).

The need to "physically maintain, enhance and expand nesting habitat with dredged material", task 1.3451 in the Roseate Tern Recovery Plan – Northeast Population, is identified as a priority 1 recovery task (USFWS 1998). The Service defines priority 1 recovery tasks as those that must be taken to prevent species extinction or to prevent the species from declining irreversibly in the foreseeable future.

The State of Massachusetts, the Town of Marion, the ACOE, the Service and other interested parties are actively studying alternative ways in which the revetment and tern nesting habitat on the island can be restored (ACOE 2005; DEIR 2002). The alternative recommended by the ACOE (2005) is to restore and repair the existing stone revetment in its current location on the island and to use clean dredged material to raise the elevation of 0.64 acre of habitat landward of the revetment. Re-vegetation of the filled area and the placement of artificial nest boxes will enhance the restored areas' suitability for tern nesting. This will result in about 2.2 acres of habitat suitable for tern nesting on the island. The ACOE (2005) considers the Bird Island restoration project (in planning) to have a 50-year life. On page 8-10 of the BA, MMS states that CWA will contribute $780,000 to the Bird Island restoration project (approximately 21% of the projected cost). Subsequently, the State of Massachusetts clarified that CWA will not directly contribute the funds, but that the state will dedicate $780,000 of Cape Wind-associated lease revenue to support the Bird Island restoration project.

Additional measures were identified as project components in the May 2008 BA. The State of Massachusetts included these measures as mitigation under the Massachusetts Environmental Protection Act (MEPA). However, MMS cannot specify how a state will spend future lease revenues, and subsequently withdrew these measures as part of the proposed action in an electronic transmission to the Service dated September 3, 2008. Despite the benefits that may occur in the long-term, these measures lack specifics and are unquantifiable. Consequently, although we recognize that future benefits may result from state-required measures identified in the BA, we have not incorporated them into our effects analysis and incidental take assessment.

STATUS OF THE SPECIES

Roseate Terns

The BA provides species description, life history, phenology, survivorship, population dynamics and distribution information on roseate terns on pages 3-35 – 3-40.

The following is a summary of general life history information and distribution information with emphasis on factors pertinent to the proposed Cape Wind Project. Information is excerpted primarily from the Roseate Tern Recovery Plan, Northeastern Population (USFWS 1998), Gochfeld *et al.* (1998), and additional publications as noted.

The roseate tern is a pale, medium-sized, black-capped sea tern [about 15 inches long (38 cm) including tail streamers up to eight inches (20.3 cm)] and weighs about four ounces (Gochfeld *et al.* 1998). Its plumage superficially resembles that of the common tern (*Sterna hirundo*), among which it invariably nests in the Northeast. On November 2, 1987, the Service determined the population that nests in the Northeast to be endangered, and the population that nests in the Caribbean to be threatened.

Historically, the breeding range of roseate terns in the northeastern population extended from Atlantic Canada south to Virginia and North Carolina. In recent decades, the breeding range has contracted and the population has become concentrated in Massachusetts and New York, with smaller colonies in Connecticut, New Hampshire and Maine. The current breeding distribution of roseate terns in the endangered northeastern population is as follows: birds breed from Long Island, New York, east and north to Nova Scotia and Quebec (Iles Madeleines). However, at present, less than 5% of the northeastern North American population nests in Canada (Environment Canada 2006). Approximately 87% of the endangered North Atlantic roseate population nests on just three colonies in Buzzards Bay, Massachusetts (Bird, Ram and Penikese Islands), and one colony off Long Island, New York (Great Gull Island) (RTRT 2007).

The basic breeding biology of the roseate tern is as follows: in spring, roseates make a long distance, northward migration traveling over open ocean. The terns arrive at Nantucket and Martha's Vineyard Islands "in large flocks", and then disperse to nesting colonies northward and westward (Gochfeld *et al.* 1998). Adult roseates arrive at nesting colony sites in late April to early May. Generally, courtship behavior is described as occurring at the breeding colonies and in the surrounding intertidal area (Nisbet 1981; Gochfeld *et al.* 1998). Roseate terns begin egg-laying in mid-to-late May. Typically, two eggs are laid and the incubation period lasts 23 days. Young tern chicks are fed small fish by both adults and grow rapidly. Re-nesting is common if the first clutch of eggs is lost.

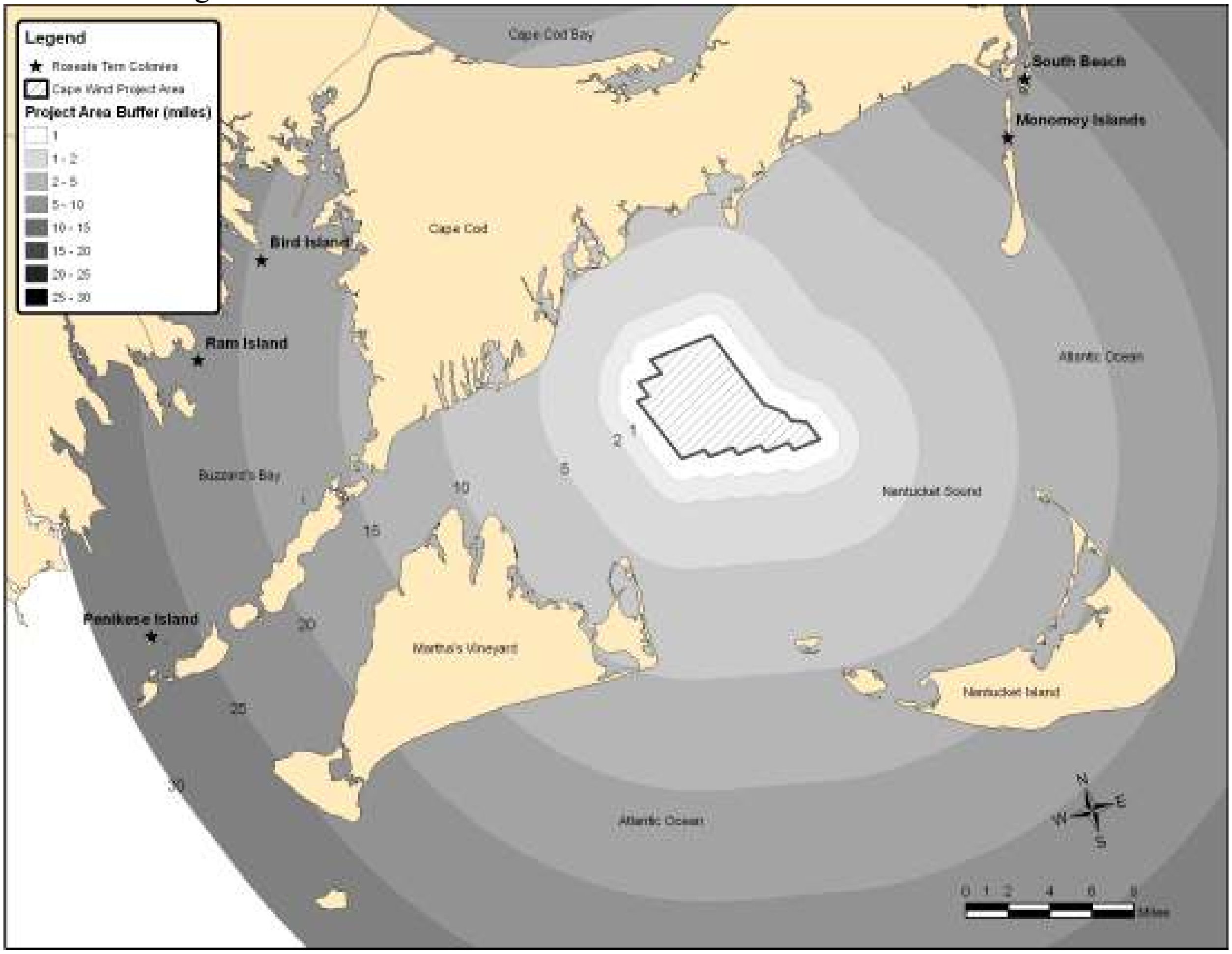
The roseate tern is a marine bird, usually breeding on small islands, but occasionally on sand spits and dunes at the ends of barrier beaches. All recorded nesting in the Northeast is within colonies of common terns. Within these mixed colonies, roseate terns usually select the more densely vegetated areas (Burger and Gochfeld 1988; Gochfeld *et al.* 1998) or other areas that provide dense cover. Unlike most other temperate zone terns, roseate terns usually nest under or adjacent to objects that provide cover or shelter (Nisbet 1981). These objects include clumps of vegetation, rocks, driftwood, or other man-made objects. Plants utilized for cover include beach grass (*Ammophila breviligulata*), seaside goldenrod (*Solidago sempervirens*), lambs quarter (*Chenopodium alba*), beach pea (*Lathyrus japonica*), and mustard (*Brassica* sp.). At some colony sites, vegetation grows to a height of 1-2 meters over the nesting sites during the breeding season, providing concealment for the eggs and chicks, but sometimes impeding access by the adults. At other colony sites, roseate terns nest under rocks, sometimes deep within crevices of rock riprap placed to protect island slopes from erosion. They readily adopt artificial sites such as wooden nest boxes or partially-buried automobile tires (Spendelow 1982, 1994). Nests typically are 24 to 71 inches (60 to 180 cm) apart, although density is sometimes as high as two or three nests per square meter within patches of suitable cover (Nisbet 1981; Burger and Gochfeld 1988).

Nisbet and Hatch (1999) examined the consequences of an imbalanced sex ratio among breeding-age roseate terns at Bird Island, Massachusetts. At Bird Island, they documented 127 females per 100 males, and found that supernormal clutches were often associated with female-female pairs. Female–female pairs exhibited lower fertility, hatching success and productivity than male-female pairs (Nisbet and Hatch 1999), but the cause of the imbalanced sex ratio remains unknown. Specifically, at Bird Island from 1970-1995, female terns within female-female pairs laid fewer eggs than females mated to males (1.20 versus 1.73), had lower fertility and hatching success (about 46% versus 98%), were less successful at raising young from eggs that did hatch (about 58% versus 73%), and their overall breeding success was 0.34 fledgling per female versus 1.35 (Nisbet and Hatch 1999). As recently as 2008, multi-female associations continue to be observed at many nesting colonies in Massachusetts (C. Mostello, Massachusetts Division of Fisheries and Wildlife, pers. comm. 2008).

Beginning in July and by mid-August, most terns have completed nesting and leave colony sites for pre-migratory staging areas. In August and September, staging birds are reported in large flocks with other species of terns at inlets and islands from Long Island, New York to Maine (Viet and Petersen 1993; Shealer and Kress 1994). From mid-August to mid-September, it is thought that most roseate terns have aggregated in coastal areas of Massachusetts, especially along outer Cape Cod. About 20 post-breeding staging areas have been identified around Cape Cod; South Beach and the Monomoy Islands (Figure 1) appear to be among the most important locations for roseates prior to fall migration (Gochfeld *et al.* 1998; Trull *et al*. 1999). Young-of-the-year roseate terns remain dependent on their parent(s) for at least six weeks after fledging and may remain dependent on parental feeding until after arrival in the winter quarters (Nisbet 1981).

After feeding for a matter of weeks, roseate terns migrate south through the West Indies to winter off the northern and eastern coasts of South America. The winter quarters are not fully known, but work by Hays *et al*. (1997 and 1999) documented concentrations of wintering birds along the Brazilian coast. A roseate tern recovered at Mangue Seco, Bahia, Brazil set a longevity record for the species at 25.6 years (Hays *et al*. 1999). Nearly all 1-year-old and most 2-year-old roseate terns are assumed to remain somewhere in the wintering area, based on banding studies (Nisbet 1984; Spendelow *et al*. 2002) and intensive observations of terns in the breeding grounds (J. Spendelow, U.S. Geological Survey, pers. comm. 2008).

Figure 1.Southern Cape Cod Roseate Tern staging areas and distances from Horseshoe Shoal to nearest breeding colonies.



Feeding Habits and Foraging Habitat

During the breeding season, roseate terns forage over shallow coastal waters, sometimes near the colony and at other times at distances of over 20 miles (32 km) (Heinemann 1992). Roseates tend to concentrate in places where prey fish are brought close to the surface by the vertical movement of water. Hence, they usually forage over shallow bays, tidal inlets and channels, tide-rips and sandbars over which tidal currents run rapidly (Nisbet 1981; Duffy 1986; Safina 1990; Heinemann 1992; Casey, Kilpatrick and Lima, unpubl. data, 1996 USFWS). Roseate studies strongly suggest that the species is a visual forager (Safina 1990; Heinemann 1992; Casey, Kilpatrick and Lima, unpubl. data, 1996 USFWS; Hatch and Brault 2007; Rock *et al*. 2007). Roseate terns forage mainly by plunge-diving and by contact-dipping or surface dipping over shallow sandbars, reefs or schools of predatory fish (Gochfeld *et al*. 1998). Gochfeld *et al*. (1998) also report that they tend to fly into the wind, hover and dive from a height of 3.3 – 20 ft (1-6 meters), but up to 40 ft (12 meters) at times.

In the only foraging study of roseate terns within the northeastern population that utilized telemetry, Rock *et al.* (2007) found that while roseates nesting at Country Island, Nova Scotia sometimes foraged as far as 7.2 miles (24 km) from the colony, on average they foraged much closer, 2.1 miles (7 km), and especially in locations within six miles (10 km) of the colony, at water depths less than 16.5 ft (5 m). The authors recommended that critical foraging habitat for the roseate terns at County Island, shallow areas (< 5 m depth) within 10 km of the colony, should be protected.

Anecdotal evidence suggests a reduction in provisioning of chicks during foggy conditions, therefore it appears that roseates continue to attempt to forage even when visibility is reduced, but do not venture far from the colony and are less successful in those conditions (S. Hall, National Audubon Soc., pers. comm. 2008; J. Spendelow, pers. comm. 2008). Roseate terns usually feed in clearer and deeper waters than those favored by common terns from the same colony sites (Gochfeld *et al.* 1998).

In recent years, the three most important roseate tern nesting sites in Massachusetts are Bird, Ram and Penikese Islands within Buzzards Bay (Figure 1). The closest edge of the proposed Cape Wind turbine array in Horseshoe Shoal is about 19 miles (31 km) from Bird Island, about 22 miles (35 km) from Ram Island, and 27 miles (43 km) from Penikese Island (BA). Heinemann (1992) studied the foraging behavior of roseate terns nesting at Bird Island during 1990 and 1991, and found that the shoal at Mashnee Flats between Bird Island and the entrance to the Cape Cod Canal was a favored foraging location. Other flats and shoals were also important, including Onset Bay, Buttermilk Bay, W. Falmouth Harbor, Quisset Harbor, Waquoit Bay, Woods Hole, and the extensive shallows between Ram Island and the mainland. Roseate terns from Bird Island were not found foraging any farther east (toward the Horseshoe Shoal project area) than Vineyard Sound (Lake Tashmoo to Vineyard Haven) and the westernmost portion of Nantucket Sound (Heinemann 1992). Presently, the Monomoy Islands (Minimoy and South Monomoy) are the closest locations to the project area (about 18 miles or 29 km) where roseate terns have nested in recent years. About 58 nesting pairs (total season) of roseates were recorded there in 2007. Horseshoe Shoal is close enough to the Monomoy Islands to be within the reported foraging distance for roseate terns (Heinemann 1992; Gochfeld *et al.* 1998).

Roseate terns feed almost exclusively on small, schooling marine fish. In the northeastern United States, they show a preference for sand eels (also called sand lance) (*Ammodytes* spp.). Also taken are various small fish, including bay anchovy (*Anchoa* spp.), juvenile herring (*Clupea* spp.), Atlantic menhaden (*Brevoortia tyannus*), Atlantic mackerel (*Scomber scombrus*), Atlantic silversides (*Menidia menidia*), juvenile bluefish (P*omatomus saltatrix*), and white hake (*Urophycis tenuis*) (Gochfeld *et al.* 1998). Roseates only rarely take insects, squid or small crustaceans.

Factors Affecting Roseate Terns

The numbers of roseate terns nesting in the Northeast were greatly reduced in the 19th century by commercial hunting for the millinery trade. With the cessation of market hunting, the population recovered, and by the 1930s, there were about 8,500 pairs. However, encroachment onto their nesting islands by increasing populations of gulls, and combined with habitat loss reduced numbers to a low of about 2,500 pairs in 1977.

The primary reasons for listing the northeast population of the roseate tern as endangered in 1987 were the concentration of the population into a small number of breeding sites, and to a lesser extent, a decline in total numbers (USFWS 1998). While roseates are now known to nest at about 20 different sites, they remain vulnerable because only small numbers of pairs occur at most colonies. In 2007, only six nesting colonies supported more than 100 pairs (four had >200 pairs), and more than 90% of the total population in the Northeast breeds on just five islands. Concentrated at so few nesting sites, the endangered northeast population of the roseate tern is susceptible to stochastic events, including erosion of nesting habitat, storms and over-washing of nests, prey food shortages, predation, oil spills and human disturbance. In addition, the roseate tern breeding population remains numerically and geographically reduced from historic levels.

Rangewide Status and Recovery Objectives

In the past 20 years (1988-2007), the total estimated breeding population has generally fluctuated in the range of about 3,000 pairs to 4,300 pairs, with a high of 4,926 pairs reached in 2000. During this period, the breeding population has exhibited an approximate 20% increase in the number of nesting pairs. Roseate terns have delayed maturity but are long-lived birds and appear capable of maintaining relatively stable populations from year to year (Spendelow *et al*. 2002; Spendelow *et al.* 2008). The greatest annual fluctuations in roseate breeding pair numbers recorded rangewide in the northeastern United States between 1988-2007 were declines of 17-20% from 1991-1992 and from 2000 to 2001 (Table 1). Preliminary data for the 2008 nesting season indicate another 16-20% decline (from 2007). If final colony census data confirm a 20% decline in 2008, the number of roseate tern pairs will have declined again to the approximate level recorded when the species was listed in 1988.

Table 1. Estimated "total season" nesting pairs of roseate tern in the northeastern United States.

Year	**1988**	**1989**	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**
No. pairs	3332	3164	3332	3718	3072	3400	3527	3633	3596	3980
Year	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**
No. pairs	4271	4284	4926	4012	3781	4129	3813	3435	3566	4066

During the past two decades, total numbers of roseate terns that nested in colonies from Long Island Sound to Buzzards Bay/Nantucket Sound increased about 6% per year (except for a 20% decline from 1991-1992), then decreased at about 4% per year between 2000-2006 (Spendelow *et al.* 2008). The increase noted from 2006 to 2007 may be short-term, as most preliminary colony census data for 2008 suggest a 16-20% decline in the number of breeding pairs. Spendelow *et al.* (2008) report that the annual survival rate (between 0.81-0.85) of the roseate tern population breeding in Massachusetts, Connecticut and New York has been fairly stable for the past 19 years. Fluctuations in the number of breeding pairs are thought to be mainly attributable to changes in the survival rate of juvenile terns and their rate of recruitment into the breeding population (J. Spendelow, pers. comm. 2008).

USFWS (1998) indicates that reclassification of the roseate tern from endangered to threatened should be evaluated when the northeastern nesting population achieves the following criteria:

- increase to 5,000 or more pairs, with high productivity (1.0 young per pair for five years);

- the pairs occur in six or more colonies of > 200 pairs; and
- the six colonies are distributed across the geographic range.

Delisting can be considered when, in addition to the above, the number of roseate tern nesting colonies has been expanded to 30 or more sites and the breeding range has been expanded to include historically-occupied areas south of the current range (USFWS 1998).

Only two prior biological opinions have been prepared involving roseate terns in the North Atlantic. In 1998, the Service submitted a non-jeopardy biological opinion to the ACOE for a shoreline protection project at Falkner Island, Connecticut. An unspecified level of incidental take was identified and phase 1 of the project was completed in 2000. Approximately nine roseate terns (one adult and eight chicks) are suspected of being "taken" (died as a result of entrapment in the revetment) (Spendelow and Kuter 2001). It is currently uncertain whether phase 2 of the project (completion of the revetment around the island periphery) will be built (R. Potvin USFWS, pers. comm. 2008). Completion of phase 2 could include conservation benefits to roseate terns if measures to correct the entrapment issue are included in the scope of work. For example, by filling the interstitial spaces within the revetment with crushed stone, the entrapment of chicks and adults can be greatly reduced. The second biological opinion developed for this species was submitted to the U.S. Coast Guard in October 2004, for the **response** to the 2003 Bouchard No. 120 (B-120) oil spill in Buzzards Bay, Massachusetts. This oil spill resulted in moderate oiling of Ram Island, one of the largest roseate nesting colonies, and slight oiling of Bird and Penikese Islands. Due to the necessity of hazing roseate terns from Ram Island to discourage them from settling into nesting habitat until it was cleaned of oil, many tern pairs moved to other islands, and/or delayed nesting. The Service determined that the take of roseate terns from oil spill response due to their delay in nesting and displacement from Ram Island was 350 chicks not produced (i.e., reduced productivity).

Studies to assess what effect the B-120 oil spill had on the northeastern roseate tern population are ongoing. In addition to the delay in nesting and the movement of pairs away from Ram Island caused by the oil spill response, some terns were oiled and may have been more directly injured. However, Spendelow *et al.* (2008) examined survival rates of roseate terns over a 19-year period and did not detect a lower survival of the birds nesting at the colonies near the spill compared to those nesting at other study sites in New York and Connecticut.

Piping plovers

Piping plovers are small, sand-colored shorebirds approximately seven inches long (18 cm) with a wing span of approximately 15 inches (38 cm). The piping plover was listed as threatened and endangered under provisions of the ESA on January 10, 1986. Three distinct populations were identified by the Service during the listing process: Atlantic Coast (threatened), Great Lakes (endangered), and Northern Great Plains (threatened). Protection of all three populations of this species under the ESA reflects its precarious status range-wide. The Atlantic Coast population, which is the focus of this BO, breeds on coastal beaches from Newfoundland to North Carolina (and occasionally in South Carolina), and winters along the Atlantic Coast from North Carolina southward, along the Gulf Coast, and in the Caribbean. In 1996, the Service approved a revised recovery plan for the Atlantic Coast piping plover population (USFWS 1996). No critical habitat, as defined by the ESA, has been designated for the breeding habitat of the Atlantic Coast population.

The following is a summary of general life history information with emphasis on factors pertinent to the proposed Cape Wind Project (e.g., movements during the breeding season, migration) and status. Information is drawn from the revised Atlantic Coast piping plover recovery plan (USFWS 1996), unless otherwise stated.

Breeding

Piping plovers begin returning to their Atlantic Coast nesting beaches in early March. By early April, males begin to establish and defend territories and court females. Piping plovers are generally monogamous within a given year, but usually shift mates between years, and only occasionally between nesting attempts in a given year. Plovers are known to breed at one year of age, but the rate at which this occurs is unknown.

Piping plovers nest above the high tide line on coastal beaches, sandflats at the ends of sandspits and barrier islands, gently sloping foredunes, blowout areas behind primary dunes, sparsely vegetated dunes, and washover areas cut into or between dunes. Clutch size is generally four eggs, and eggs are usually incubated for 27-30 days before hatching. Incubation is shared equally by both sexes. As a rule, piping plovers fledge only a single brood per season, but may re-nest several times if previous nests are lost.

Plover foods consist of invertebrates such as marine worms, fly larvae, beetles, crustaceans, and mollusks. Feeding areas include intertidal portions of ocean beaches, washover areas, mudflats, sandflats, wrack lines, and shorelines of coastal ponds, lagoons or salt marshes. Feeding activities of both adults and chicks occur during all hours of the day and night.

Upon hatching, flightless piping plover chicks may walk hundreds of yards from the nest site during their first week of life (Table 1 in USFWS 1996). Adults lead the chicks to and from feeding areas, shelter them from harsh weather and protect young from perceived predators. Jones (1997) studied home ranges of piping plovers at the Cape Cod National Seashore in Massachusetts and observed that most broods moved an average of 500 m (~1,600 ft) from their nests after hatching and before fledging. Two plover families with chicks within 16 to 21 days old were found to forage up to 1,000 m (~3,300 ft) from their nests. Plover broods have also been observed to move up to 1,600 m (~1 mile) from their nest and back in one day, and have moved maximum distances of more than 4,000 m (~2.5 miles) before fledging (Jones 1997).

Chicks remain together with one or both parents until they fledge at 25 to 35 days of age. Depending on the date of hatching, unfledged chicks may be present on beaches from late May through late August, although most have fledged by the end of July.

With the exception of aerial courtship displays performed over breeding territories that include circular or elliptical flights at observed heights up to approximately 35 m (114 ft), adult flights within breeding habitats or between nesting or brood-rearing areas and nearby intertidal sand or mudflats are low to the ground, less than 15 m (<50 ft). During three years of opportunistic observations of plovers crossing a road along Westhampton Island, Houghton (2005) recorded

550 and 913 observations of plovers flying, respectively, below and above truck height. At least two adult plovers were run over by vehicles on the road (Houghton 2005).
Feeding territories are generally contiguous to nesting territories, although adults may forage on the opposite side of small coastal inlets or on nearby bayside intertidal flats (e.g., Cohen 2005). MacIvor *et al.* (1985), however, observed a single plover from a pair breeding at one beach, feeding at another site 23 miles (37 kilometers) away.

Adult piping plovers generally demonstrate nest site fidelity, returning to the same breeding beach or a nearby beach in consecutive years, while fidelity of first-time Atlantic Coast breeders to natal sites is low. In New York, Wilcox (1959) recaptured 39% of the 744 adult plovers that he banded in prior years (many were recaptured during several successive seasons and all but three of them were re-trapped in the same nesting area), but recaptured only 4.7% of 979 plovers that he banded as chicks. He also observed that males exhibited greater fidelity to previous nest sites than females. Strauss (1990) observed individuals that returned to nest in his Massachusetts study area for up to six successive years, but mean distance between natal and breeding sites was 25.3 km (n=6, range 9 - 40 km). Also in Massachusetts, 13 of 16 birds banded on one site were resighted the following season, with 11 nesting on the same beach (MacIvor *et al.* 1987). Of 92 adults banded on Assateague Island, Maryland, and resighted the following year, 91 were seen on the same site, as were 8 of 12 first-year birds (Loegering 1992).

Piping plovers rarely move great distances from one nest site to another after a nest failure. In a four-year study on outer Cape Cod, MacIvor (1990) documented only three adult plovers among 101 uniquely identifiable color-banded breeding pairs that changed beaches between re-nest attempts in the same year. Distances from first nest site to re-nest site ranged from 8 to 23 miles (13 to 37 kilometers). Review of detailed breeding records for an estimated 501 pairs of piping plovers breeding in Massachusetts in 1999 by Melvin and Mostello (2000) included at least 14 pairs suspected of re-nesting at new territories on the same beach or at more than one site.

Migration
Both spring and fall migration routes of Atlantic Coast breeding piping plovers are thought to occur primarily within a narrow zone along the Atlantic Coast. Sightings away from the outer beaches, either inland or offshore, are rare (Bull 1964; Barbour *et al.* 1973; Imhoff 1975; Potter *et al.* 1980). Observations of color-marked birds from the Atlantic Coast suggest some crossover to Gulf Coast wintering areas (Haig and Plissner 1993). Occasional sightings of piping plovers at distant islands, such as Bermuda (American Birds 1987, 1990; D. Wingate, Bermuda Aquarium and Natural History Museum, *in litt.* 1988), demonstrate that long-distance migrations are possible.

Northward migration from wintering grounds to breeding grounds occurs during late February, March and early April (USFWS 1996). Reports of earliest arrivals in Massachusetts are concentrated in mid-March (e.g., MacIvor 1990; and Petersen 1993). Cairns (1982) states that most piping plovers arrive in Nova Scotia from mid-to-late April. A few failed breeders may return to their wintering grounds in early July, but the peak of southward migration begins as young plovers fledge in late July and extends through August, trailing off in early September. Transient plovers have been observed following early autumn hurricanes (USFWS 1996).

Relatively little is known about migration behavior, flight altitude or habitat use within the Atlantic Coast breeding range (USFWS 1996), but the pattern of both fall and spring counts at migration sites along the southeastern Atlantic Coast demonstrates that many piping plovers make intermediate stopovers lasting from a few days to one month during their migrations (NPS 2003; Noel *et al*. 2005; Stucker and Cuthbert 2006). Most reports indicate migrants congregating in small groups (e.g., Haig and Elliott-Smith 2004), but flocks of over 100 birds have been observed at Cape Lookout National Seashore in North Carolina during fall migration (Collazo *et al.* 1995). As many as 85 staging piping plovers have been tallied on South Beach in Chatham, Massachusetts (S. Perkins, Massachusetts Audubon Society, *in litt.* 2008), but this likely included adults that bred nearby and their fledged young of the year, as well as migrants from breeding sites farther north.

Factors affecting piping plovers
Loss and degradation of habitat due to development and shoreline stabilization have been major contributors to the species' decline. Beaches throughout the plover's range are affected by federal and non-federal actions, including inlet management, beach nourishment, dune construction, and dune stabilization. For example, throughout much of the New York-New Jersey Recovery Unit, periodic beach nourishment has interfered with natural coastal processes by precluding formation of newly-forming inlets, overwash zones, and accreting beach habitats that would create or revitalize piping plover nesting and foraging habitats (USFWS 2005).

Disturbance by humans and pets often reduces the functional suitability of habitat and causes direct and indirect mortality of eggs and chicks. Recreational use of piping plover beaches includes pedestrian and vehicular activities. Pedestrian and non-motorized recreational activities can be a source of both direct mortality and harassment of piping plovers. Pedestrians may disrupt plovers during territory establishment, courting, egg-laying and chick rearing. Intense pedestrian use of plover beaches may also prevent chicks from foraging, separate chicks from adults, and increase chicks' vulnerability to predation. Unmanaged off-road vehicle use will degrade plover nesting and foraging habitat or may crush chicks and occasionally adults. Intensive management, similar to that described in the Environmental Baseline section (below) to minimize the effects of recreational disturbance, is ongoing in most of the piping plover's Atlantic Coast breeding range.

Noncompliant pet owners who allow their dogs off leash have the potential to flush piping plovers, and these flushing events may be more prolonged than those associated with pedestrians or pedestrians with leashed dogs. Unleashed dogs may chase plovers, destroy nests, and kill chicks.

Predation has been identified as a major factor limiting piping plover reproductive success at many Atlantic Coast sites, and substantial evidence shows that human activities are affecting types, numbers, and activity patterns of predators, thereby exacerbating natural predation. Predators of piping plover eggs and chicks include foxes, skunks, raccoons, rats, opossums, crows, gulls, grackles, hawks and falcons, domestic dogs and cats, and ghost crabs (USFWS 1996). As with other limiting factors, the nature and severity of predation is highly site-specific.

Substantial evidence exists that human activities are affecting types, numbers, and activity patterns of predators, thereby exacerbating natural predation. Non-native species such as feral cats and rats are considered significant predators at some sites (Goldin *et al.* 1990; Post 1991). Humans have also indirectly influenced predator populations by abetting the expansions in the populations and/or range of other species such as gulls (Drury 1973). Strauss (1990) found that the density of fox tracks on a beach area was higher during periods of more intensive human use.

A variety of techniques that have been employed to reduce predation on plovers are discussed in the revised recovery plan (USFWS 1996). Most notably, the use of predator exclosures (fences around nests) has been used with demonstrated success to reduce predation on piping plover eggs (Melvin *et al.*, 1992; Rimmer and Deblinger 1990). Regional productivity increases (Melvin and Mostello 2003, 2007) resulting from higher nesting success are credited to the use of predator exclosures. However, these same devices have also been associated with serious problems, including entanglements of birds in the exclosure netting, and attraction of "smart" predators that have "learned" there is potential prey inside. The downside risks may include not only predation or nest abandonment, sometimes at rates exceeding those that might occur without exclosures, but also induced mortality of adult birds (Melvin and Mostello 2003, 2007). Exclosures provide no protection for mobile plover chicks, which generally leave the exclosure within one day of hatching and move extensively along the beach to feed. Selective predator removal at some sites is reducing losses of eggs and chicks and occasionally, adults (B. Clifford, New Hampshire Fish and Game Department, electronic transmission, 2008).

Since the 1986 listing, major oil spills affecting Atlantic Coast piping plovers have included the World Prodigy (RI - 1989), B.T. Nautilus (NY and NJ - 1990), North Cape (RI - 1996) and Anitra (NJ - 1996). Implementation of piping plover restoration plans using funds collected from the responsible party have been completed or are in progress for all of these spills.

In April 2003, the Bouchard No. 120 (B-120) fuel barge apparently struck bottom in Buzzards Bay, Massachusetts and released approximately 55,000 gallons of No. 6 fuel oil. Within 24 hours, an oil slick approximately 10 miles long and two miles wide was observed in the Bay. The spill continued to spread, affecting approximately 90 miles of shoreline in and beyond Buzzards Bay. Approximately 26 extant or historic piping plover beaches were located within the area affected by the B-120 oil spill. Of these 26 beaches, piping plovers were documented to have nested at 13 sites in 2003, of which 12 were oiled and subjected to clean-up activities. Over 60 oiled plovers were documented and up to 55 pairs of plovers could have been affected by the oil and response activities. A natural resources damage assessment is underway that will quantify the injury (oil spill-induced mortality and lost productivity) (S. von Oettingen, USFWS, pers. comm. 2008).

Rangewide Status and Recovery Objective

To facilitate an even distribution of the population, the Atlantic Coast piping plover recovery plan established four recovery units (Atlantic Canada, New England, New York-New Jersey, and Southern) and assigned a portion of the population target to each. These units are large enough that their overall carrying capacity should be buffered from changes due to natural habitat formation processes at individual nesting sites, while still assuring a geographically well-distributed population. Current information indicates that most Atlantic Coast piping plovers nest within their natal region and that intensive regional protection efforts contribute to increases in regional piping plover numbers (USFWS 1996).

Since listing under the ESA, the Atlantic Coast population estimate has increased 239%, from approximately 790 pairs to an estimated 1,890 pairs in 2007 (USFWS 2008), while the United States portion of the population has almost tripled, from approximately 550 pairs to an estimated 1,624 pairs. Even discounting apparent increases in New York, New Jersey, and North Carolina between 1986 and 1989, which likely were due in part to increased census effort, the population nearly doubled between 1989 and 2007. Population increases since 1989 have been highest in New England (242%), followed by New York-New Jersey (84%). Most growth in the Southern (DE-MD-VA-NC) recovery unit (67%) has occurred since 2003, while the Atlantic Canada population fluctuates from year to year with increases often quickly eroded in subsequent years (USFWS 2008).

While population growth is heartening, periodic rapid declines in populations at the level of the individual recovery unit raise concerns about the long-term risk of extirpation faced by the Atlantic Coast population. For example, the Atlantic Canada population declined by 21% in just three years (2002 - 2005), and the southern half of the Southern recovery unit population declined by 68% in seven years (1995 - 2001). Pressure on Atlantic Coast beach habitat from development, human disturbance, and predation is widespread and unrelenting. The recovery of the Atlantic Coast piping plover population is occurring in the context of extremely intensive annual management that is implemented on almost all plover beaches, in both the United States and Atlantic Canada (USFWS 1996; RENEW 2003, 2004).

The Revised Recovery Plan for the Atlantic Coast piping plover (USFWS 1996) identified a recovery objective for delisting the species, as well as five criteria for meeting the recovery objective. The overall objective is to ensure the long-term viability of the Atlantic Coast plover population in the wild. Delisting of the Atlantic Coast piping plover population may be considered when the following criteria have been met:

- increase and maintain for five years a total of 2,000 breeding pairs, distributed among four recovery units;
- verify the adequacy of a 2,000-pair population of piping plovers to maintain heterozygosity and allelic diversity over the long term;
- achieve a five-year average productivity of 1.5 fledged chicks per pair in each of the recovery units;
- institute long-term agreements to ensure protection and management are sufficient to maintain the population targets and average productivity in each recovery unit; and
- ensure long-term maintenance of wintering habitat, sufficient in quantity, quality, and distribution to maintain survival rates for a 2,000-pair population.

The New England recovery unit target is a minimum of 625 pairs. In 2007, there were approximately 754 nesting pairs of piping plovers in New England with an average productivity of 1.30 chicks per pair (USFWS 2008). Although the New England recovery unit population has exceeded (or been within two pairs of) the abundance goal since 1998, the average productivity is below the 1.5 chicks/pair threshold needed to maintain a secure population. Inclement weather and increased predation on both adults and young are the primary contributing factors that have been identified as limiting productivity.

The Atlantic Canada recovery unit has experienced the lowest population growth (net change between 1989 and 2007 is +14%), despite higher overall productivity than in the United States (1989-2006 average of 1.61 chicks/pair in Canada versus 1.31 chicks/pair in the United States). Based on estimates of survival derived from recent banding studies from 1998 through 2004, Calvert *et al.* (2006) estimated productivity of 1.63 chicks/pair required to maintain a stationary population at the sites surrounding the Gulf of St. Lawrence, compared with an estimate of 1.24 from banding data collected in Massachusetts in 1985-1988 (Melvin and Gibbs 1994).

The importance of productivity in driving Atlantic Coast piping plover population increases over the last 20 years notwithstanding, demographic models for piping plovers indicate that even small declines in adult and juvenile survival rates will cause substantial increases in extinction risk (Melvin and Gibbs 1994; Wemmer *et al.* 2001; Larson *et al.* 2002; Calvert *et al.* 2006). Elevated mortality of adults or post-fledglings has the potential to quickly undermine the progress toward recovery achieved at breeding sites. Calvert *et al.* (2006) found lower return rates of juvenile (first-year) birds to the breeding grounds than was documented for populations breeding in Massachusetts (Melvin and Gibbs 1994), Maryland (Loegering 1992), and Virginia (Cross 1996) in the late 1980s and early 1990s. This is consistent with low positive and negative growth in the Atlantic Canada population despite very high productivity (relative to other breeding populations) and extremely low rates of dispersal to the United States (Calvert *et al.* 2006). Thus, maximizing productivity does not ensure population increases; management must focus simultaneously on all sources of stress on the population.

Seven non-jeopardy formal consultations have been written for projects within the New England Recovery Unit since 1997. Most of the consultations were with the U.S. Coast Guard for marine event permits for fireworks events in coastal areas of Connecticut and Massachusetts (Table 2). These activities occur once a year and require follow-up reporting to assess take. Due to permit conditions incorporated in marine event permits issued by the U.S. Coast Guard, no plover egg or chick losses have been documented during the fireworks events. One consultation was written for the ACOE for maintenance dredging and disposal of dredged material on plover habitat and ultimately resulted in improved management and long-term benefits to the population utilizing the nourished beach. The B-120 oil spill consultation was based on spill response measures undertaken by the U.S. Coast Guard that resulted in the incidental take of eight eggs due to abandonment. The consultation identified measures to avoid adverse effects from oil spill response activities, thereby providing future protection to piping plovers under similar circumstances.

Table 2. Previous biological opinions completed for piping plovers in New England.

Year	Project	Incidental Take		Project Completed
		Amount/Extent of Take	Documented	
1997	Fireworks (Connecticut)	4 pairs of plovers and their broods/Harassment	No mortality or loss of productivity	Yes
1997	Fireworks (Massachusetts)	2 pairs of plovers/Harassment	No mortality or loss of productivity	Yes
1999	Beach nourishment/dredging (Maine)	2 pairs no productivity/harassment and mortality of young for the life of the project	1 pair 2002, no young, 1 pair 2003, 1 young	Yes, effects are ongoing
2000	Fireworks (Massachusetts)	1 egg /Mortality 4 broods/Harassment	No mortality or loss of productivity	Yes
2003	Fireworks (Connecticut)	2 pairs of plovers/Harassment	No plovers present during event	Yes
2004	B-120 Oil Spill Response – post spill consultation	8 eggs lost to abandonment	Additional unquantifiable take due to harassment	Yes
2005 - 2007	Fireworks (Massachusetts)	1 egg lost to temporary abandonment, harassment of chicks younger than 10 days, up to 2 broods	No loss of eggs or chicks documented	Yes

ENVIRONMENTAL BASELINE

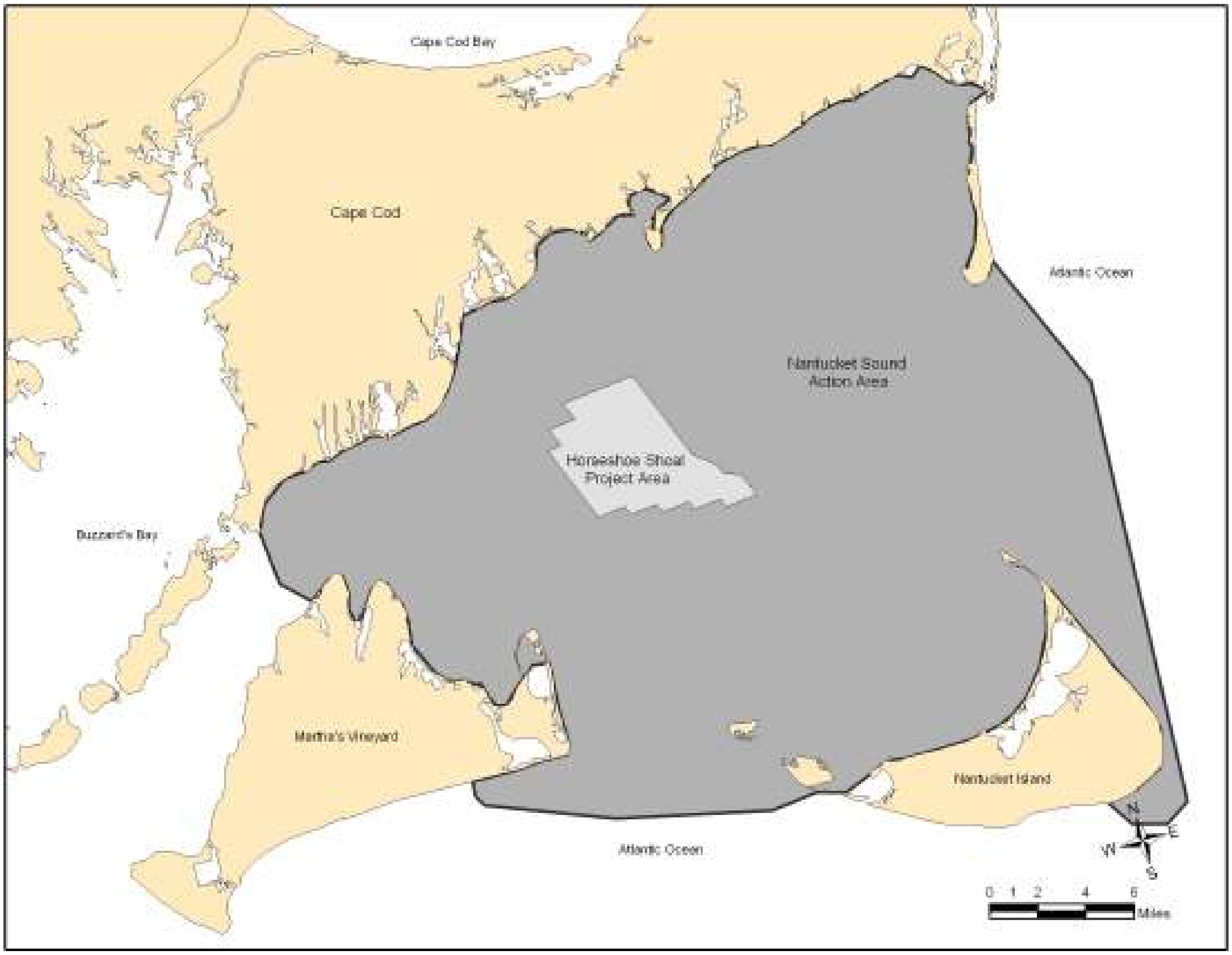
Status of the species within the action area

As defined in 50 CFR §402.02, "action" means all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by federal agencies in the United States or upon the high seas. The "action area" is defined as all areas to be affected directly or indirectly by the federal action, and not merely the immediate area involved in the action. The direct and indirect effects of the actions and activities resulting from the federal action must be considered in conjunction with the effects of other past and present federal, state, or private activities, as well as the cumulative effects of reasonably certain future state or private activities within the action area.

Although the primary potential effects are confined to the wind facility at Horseshoe Shoal, the Service considers the action area to encompass all of Nantucket Sound, the east coast of Nantucket and the northeast coast of Martha's Vineyard in order to include the physical facility (WTGs and ESP), maintenance vessel travel lanes, submarine cables and the potential oil spill impact area (Figure 2). The area depicted in Figure 2 was created by the Service as a schematic to highlight the potential area of impact due to an oil spill. The action area boundaries were not drawn specifically to include or exclude certain shore locations; rather, they are an attempt to depict the limits of a possible oil spill based on the oil spill risk and trajectory models provided by CWA (Etkin 2007; Knee *et al*. 2006).

The Bird Island restoration project (Buzzards Bay, MA) is also considered to lie within the roseate tern action area, since the project is considered part of the proposed action. The Service is not including the locations of other proposed beneficial actions (outlined in Section 8 of the BA) because insufficient information regarding the locations and duration of implementation precludes the assessment of their potential beneficial effects in this BO.

Figure 2. General schematic of the Nantucket Sound action area and Horseshoe Shoal project area.



Roseate Tern

Roseate terns are present in Massachusetts waters from late April to mid-September, when they depart for their wintering grounds in South America. The species has been studied by ornithologists in the Bay State for more than 70 years (see, for example, reports by C. Floyd from the 1920s cited in Nisbet 1981). Flocks of roseates arriving from the wintering grounds in the spring are reported "in large flocks" in Nantucket Sound before dispersing to breeding sites north, south and west (Gochfeld *et al.* 1998).

Most of the endangered northeastern population of the roseate tern (breeding range is the western North Atlantic) nests in Buzzards Bay, Massachusetts and in Long Island Sound, New York (about 90%), while about 10% migrate north to nest in the Gulf of Maine and Maritime Canada. Prior to the 1970s, many locations around Nantucket Sound, including Monomoy, Nantucket, Muskeget and Tuckernuck Islands, supported from the low hundreds to several thousands of nesting roseate terns (USFWS 1998). Presently, the Monomoy Islands (Minimoy and South Monomoy) are the closest locations to the project area (about 18 miles or 29 km) where roseate terns have nested in recent years. About 58 nesting pairs (total season) of roseates were recorded there in 2007. Despite a multi-year effort (1999 to at least 2007) to restore roseate terns to Muskeget Island (R. Veit, I. Nisbet, pers comm. 2007), roseates have not become re-established there or at the other large historic colonies listed above, except at Monomoy as noted (RTRT 2007; USFWS unpubl. data).

Although roseate terns do not nest at or immediately adjacent to Horseshoe Shoal, foraging activity during the pre-breeding, breeding and pre-migratory fall staging periods could result in roseate tern presence in the project area. Tern presence in the action area was investigated by CWA and by the Massachusetts Audubon Society (MAS) during observational surveys by boat and aircraft during 2002, 2003 and 2004, and by boat off South Beach, Chatham and Monomoy Island in 2006. The results of these studies and a summary of what is generally known about roseate tern presence in and around Nantucket Sound are discussed at length on pages 4-11 to 4-28 in the BA and on pages 2-7 to 2-13 of volume III of the DEIS. An abbreviated summary of the results from those studies is as follows:

During the month of May (pre-breeding and early breeding season) in 2002 and 2003, aerial surveys conducted by CWA estimated that a total of 784 terns (mixed species) were present within the Horseshoe Shoal project area (mean of 392 per survey). During June and July (breeding season months) of 2002-2004, only a combined, annual mean of 21 terns was observed [CWA and MAS surveys, see Hatch and Brault (2007) table 1]. During the post-breeding season (August and September, 2002-2004), a combined, annual mean of 86 terns was observed on Horseshoe Shoal. As a general index of relative occurrence, these data indicate more use of the project area after first arrival in spring, declining with the onset of the breeding season, and then increasing again during the post-breeding season.

From 2002-2006, ESS Group (on behalf of CWA) and MAS flew 125 systematic aerial surveys of Nantucket Sound to document avian species' presence and distribution. More than half of these surveys took place during winter (for waterfowl and sea ducks) when roseate terns and piping plovers are not present in Massachusetts. Fifteen of the surveys were conducted during the nesting season (mid-May to late July), and 37 of the surveys were conducted during fall staging of roseate terns in and around Nantucket Sound (mid-August to late September). ESS Group (2006) reports that surveys were flown at different times of the day, originated from different starting points and directions, at different tides, and in somewhat varying weather, but visibility was good or excellent during every survey. Surveys were not conducted in inclement weather or at night, and several other limitations of the survey methodologies are discussed on pages 2-12 and 2-13 of the DEIS.

The breeding season aerial surveys (mid-May to July) recorded significant common and roseate tern use of Nantucket Sound [e.g., 832 roseates, 4,779 commons and 12,646 mixed common-roseate terns (page 2-10 of the DEIS)]. However, of the total number of terns observed by MAS during the breeding season in 2003, only 1.5% were in the Horseshoe Shoal project area, and in 2004, 0% of the total terns observed by MAS were in Horseshoe Shoal. During aerial surveys conducted by CWA during the breeding season, about 10% of 2,888 total terns (includes both common and roseate terns) observed were in the Horseshoe Shoal project area. Of all terns observed at Horseshoe Shoal during the boat-based surveys by MAS, most (from 33-81%, median = 53%) were travelling across the Shoal (Hatch and Brault 2007), but some terns were actively foraging as well (BA).

The number of (post-breeding) terns staging on outer Cape Cod increases as summer progresses and peaks in mid-August to mid-September (Trull *et al.* 1999). However, staging by non-breeders and failed breeders may begin as early as June (J. Spendelow, pers. comm. 2008). The 37 fall staging period aerial surveys and 36 boat surveys conducted by CWA and MAS recorded relatively minimal tern activity in the Horseshoe Shoal proposed action area during 2002-2004 (DEIS), despite more than 16,500 tern sightings recorded by MAS throughout Nantucket Sound. Tern abundance was higher within a few miles of the south shore of Cape Cod. CWA's fall 2002 and 2003 surveys recorded nearly 3,000 terns (both species) using Nantucket Sound, but the vast majority were near Monomoy, Tuckernuck and Nantucket Islands, with fewer terns encountered over Horseshoe Shoal (BA). Most terns (>90%) that were recorded in Nantucket Sound were using areas other than the Horseshoe Shoal project area.

Combining all systematic survey data available, CWA noted a marked increase in the average densities[2] of terns in areas of Nantucket Sound during spring arrival and the post-breeding, pre-migratory period, when compared to surveys in the nesting season (BA). On Horseshoe Shoal, the average density of terns increased toward the southern boundary of the project area, with densities of 2 to 101-250 terns (mixed species) per square kilometer. These data are for all terns, and common terns are a more abundant species than roseate terns. For example, in nearby Buzzards Bay, common terns outnumber roseates by a factor of 3 to 1 (C. Mostello, pers. comm. 2007); the ratio of common to roseate terns in Nantucket Sound during the fall may be as high as 8:1 (Trull *et al.* 1999).

While we cannot completely discount concerns about difficulties in detecting roseate and common terns during boat and aerial surveys (i.e., they are relatively small light-colored birds that can be difficult to see against a light sky), the Service finds CWA results, based on more than 16,000 tern observations over multiple years, to be credible with respect to presence in the project area. Roseate terns occur with less frequency and regularity in the Horseshoe Shoal project area than in other coastal areas of Nantucket Sound.

For roseate terns across their breeding range in the Northeast, when young become capable of flight, family groups are no longer geographically restricted to the nesting colonies. During the early part of the post-nesting period, roseate terns will leave their nesting colonies (in mid-to-late summer) and may disperse widely to several coastal locations to feed, rest and roost (overnight)

[2] Average density was based on the count of individual terns observed along the survey transects within 1-mile-square (2.6 km square) grid cells and the number of survey dates within each grid cell.

prior to fall migration. One such fall staging location is reported by Shealer and Kress (1994) at Stratton Island, Maine, where banded birds from eight different colonies were observed. Others occur on Long Island, New York (Gochfeld *et al.* 1998). As the post-nesting period progresses, roseates increase in number in southeastern coastal Massachusetts. The best known and largest fall staging area used by roseate terns in the northeastern population occurs along outer Cape Cod and Nantucket Island (including Nantucket Sound). Here, roseates that have dispersed from their nesting colonies re-aggregate from late July to mid-September and prepare for their southward migration. A study by Trull *et al.* (1999) reported that numbers of roseate terns were staging and roosting at as many as 15 Cape Cod locations but that South Beach, Chatham supported the largest staging/roosting population of roseate terns in eastern North America. Peak numbers of terns were recorded at these outer Cape Cod locations between August 20 and September 10 (Trull 1998); whereas in Trull *et al*. (1999), a multi-year study, the largest roseate numbers were reported between August 26 and September 19 (the 20-25 days before departure). During this three-week period, there is the **potential** that every breeding adult roseate tern in the northeastern population (and their young of that year) will be in Nantucket Sound, within 20 miles of the Cape Wind Project area.

In 2007 and 2008, the post-breeding movement and aggregation of roseate terns in Massachusetts were followed by biologists of MAS and the U.S. Geological Survey (B. Harris, MAS; J. Spendelow, USGS, pers. comm. 2008). In 2007, terns were present in large numbers at Plymouth Beach/Duxbury (south of Boston in Cape Cod Bay), and Black Beach in Falmouth (Buzzards Bay). In 2008, these sites had relatively low use by post-breeding terns, and sites on the outer Cape, such as Provincetown (Cape Cod Bay/Atlantic Ocean) and Nauset/Coast Guard Beach (Atlantic Ocean) were heavily used. Terns that were roosting at night at Plymouth/Duxbury or at Provincetown and were foraging and resting during the day in these areas are less likely than terns staging at Monomoy/South Beach, Chatham to be making daily commuting flights across Nantucket Sound. Importantly, the Service interprets this information to mean that roseates during the post-breeding period do not always concentrate in Nantucket Sound, where they could have exposure to the Cape Wind Project. The observations by MAS and USGS also highlight that there are large within-year and year-to-year differences in tern distribution during this post-breeding period.

During the days and weeks that roseate terns are staging for fall migration on outer Cape Cod, they make daily commuting flights to forage. Some of these flights cross Nantucket Sound, where mixed species flocks of terns are encountered on Horseshoe Shoal, which they use for foraging, resting on the water and to commute back and forth across the Sound to more distant foraging areas (BA).

Hays *et al.* (1999) and Trull *et al.* (1999) are among the only published studies to report roseate terns in flight after dark. Hays *et al.* (1999) reported that at Mangue Seco, Brazil, wintering roseate and common terns came in after dark and left before first light. Trull *et al.* (1999) report that flocks of mixed species of terns arrived at overnight roosting locations from 8 pm to 10 pm and continued to arrive after dark when they could no longer be counted. Prior to the proposal to construct the Cape Wind Project, the dynamics of roseate tern presence (i.e., the location, number and characteristics of roseate tern commuting flights and foraging behavior) on and across Horseshoe Shoal was unstudied and little known. Similar data for nocturnal flights by roseate terns during the pre-breeding and breeding season are unavailable.

In recent years, the three most important roseate tern nesting sites in Massachusetts have been Bird, Ram and Penikese Islands within Buzzards Bay. The closest edge of the proposed WTG array in Horseshoe Shoal is about 19 miles (31 km) from Bird Island, about 22 miles (35 km) from Ram Island, and 27 miles (43 km) from Penikese Island (BA). Roseate terns actively breeding at these colonies are very unlikely to forage as far away as the Horseshoe Shoal project area.

In the five-year period from 2003 to 2007, from 1,480 to over 1,700 pairs, or 43-49% of all roseate terns recorded at breeding colonies in the northeastern United States population, nested at the three Buzzards Bay sites noted above. Accordingly, the Buzzards Bay roseate colonies play a vital role in both the survival and recovery of the species (Table 3). In comparison, the Monomoy Islands in Nantucket Sound have supported from 13-45 peak season pairs (and from 18-58 total season pairs) of nesting roseates during the same time frame.

Table 3. Peak season roseate tern nesting pairs and productivity (chicks fledged per pair) within Buzzards Bay and Monomoy Islands between 2003 and 2007. Peak season counts generally do not include late nesting pairs or re-nests by failed breeders.

Location	**Numbers of pairs/productivity**				
	2003	**2004**	**2005**	**2006**	**2007**
Bird Island, Marion, MA	904 (1.25)	554 (1.25)	680 (0.95)	1111 (1.29)	919 (1.26)
Ram Island, Mattapoisett, MA	557 (1.12)	936 (0.92)	724 (0.93)	463 (1.00)	661 (1.16)
Penikese Island, Gosnold, MA	251 (0.87)	9 (0.97)	76 (0.79)	48 (0.44)	102 (1.54)
Monomoy Islands	18 (1.70)	27 (1.13)	31 (0.73	29 (1.00)	58 (1.03)

Survival and productivity of roseate terns in Buzzards Bay are influenced by a number of factors, including but not limited to weather (particularly storms), predation, competition for nest sites, human disturbance and food availability. Bird and Ram Islands are low elevation islands that have lost upland habitat to erosion and over wash by storms and high tides (ACOE 2005; Ramsey and Osler 2008). Ram Island is estimated to have been reduced in size from 4.9 acres in 1935 to 2.0 acres (2 ha to 0.8 ha) in 2008, and if the current rate of loss is unmitigated, Ram Island will disappear in 40 years (Ramsey and Osler 2008). Bird Island, which has a partial revetment, has been reduced from approximately 3.0 acres (1.2 ha) in size in the 1850s to its present size of 1.5 acres (0.6 ha) (ACOE 2005). Both the ACOE (2005) study and the work of Applied Coastal Research and Engineering (Ramsey and Osler 2008) predict the rate of habitat loss for these islands will increase in coming decades due to erosional forces and sea level rise.

The Buzzards Bay tern colonies are monitored on a nearly daily basis by tern biologists that also serve as island wardens to minimize the loss of terns to predation and disturbance caused by recreational boaters coming ashore. Tern biologists also provide structures for roseate tern use for nesting. These structures sometimes alleviate competition for nest sites with common terns and minimize egg and chick loss due to predation.

Factors affecting the environment within the action area
Roseate tern use of the action area within Nantucket Sound is limited to the period of late April to mid-September. In the Monomoy Islands, nesting terns are relatively free from human disturbance, but adverse weather, predators of adults, eggs or chicks (e.g., coyotes, gulls and black-crowned night herons) and rank vegetation are factors influencing nesting success.

At Bird Island, crowding and competition for nesting space with the more abundant common tern is a factor influencing nest site selection and possibly, breeding success. The loss of upland nesting habitat due to erosion and deterioration of the revetment surrounding most of the island also contributes to crowding and competition for nesting sites. More importantly, it reduces the present carrying capacity of the island to below historic levels. Adverse weather and storms resulting in partial over wash of nest sites occasionally reduce hatching success. Sea level rise will exacerbate this factor if not mitigated by repairing the revetment. Predation periodically results in the loss of roseate adults, chicks and eggs. In the recent past, peregrine falcons, mink and gulls (C. Mostello pers. comm.) and Canada geese (J. Hatch pers. comm. 2008) have all caused injury or death of individual roseates at Bird Island.

During the fall staging period, roseate terns will use Nantucket Sound for feeding and travelling and will rest and roost overnight at several outer Cape and Island locations, such as South Beach, Chatham, Eel Point and Smith Point, Nantucket, Katama, Martha's Vineyard, and Tuckernuck Island. Human disturbance by pedestrians, dogs, and fishermen during this time can cause resting and roosting flocks to take flight, resulting in an energetic cost to the birds.

Piping plover

Approximately 50 extant piping plover breeding sites are located within the action area of the Cape Wind Energy Project (Table 4). Most of these beaches are intensively managed, although smaller, town-owned or privately-owned beaches may be minimally managed. The Massachusetts Division of Fisheries and Wildlife (MADFW) synthesizes annual estimates of abundance and productivity based on summaries of survey data from local monitors.

Table 4. Abundance of breeding piping plovers and fledged chicks within the action area by site, 2003, 2006, and 2007 (USFWS 2004; 2007; 2008).[3,4]

Plover Breeding Site, Town	Pairs (chicks)		
	2003	**2006**	**2007**
Arruda's Pt./The Jetties, Chappaquiddick	1 (0)	0 (0)	0
Bank St./Merkel Beach/Wychmere, Harwichport	4 (10)	4 (11)	4 (4)
Cape Pogue Elbow/The Narrows, Chappaquiddick	1 (1)	1 (1)	4 (2)
Coatue, Nantucket	1 (0)	1 (0)	5 (10)
Cockle Cove/Ridgevale Beach, Chatham	1 (0)	1 (0)	1 (3)
Coskata Beaches, Nantucket	1 (2)	1 (4)	5 (2)
Craigville Beach, Barnstable	ND	0	1 (0)
Crosby Landing Beach, Brewster	1 (1)	3 (9)	2 (2)
Dogfish Bar, Aquinnah	6 (0)	4 (3)	5 (7)

3 Bold text is used to indicate the sites on Martha's Vineyard and Nantucket.
4 Data for years 2004 and 2005 are still preliminary and incomplete (S. Melvin, pers. comm. 2008).

Plover Breeding Site, Town	Pairs (chicks)		
	2003	**2006**	**2007**
Dowses Beach, Barnstable	0	2 (0)	1 (4)
Eastville Point Beach, Oak Bluffs	1 (0)	1 (0)	0
Edgartown Great Pond/Job's Neck, Crackatuxet Pond, Edgartown	3 (0)	1 (4)	2 (0)
Eel Point, Nantucket	2 (0)	9 (12)	9 (9)
Eel Pond/Little Beach/Lighthouse Beach, Edgartown	1 (4)	0	0
Esther Is. / Smith Point, Nantucket	5 (2)	8 (15)	7 (11)
Forest Beach, Chatham	0	1 (3)	1 (1)
Gray's Beach, Yarmouth	1 (0)	0	0
Great Island, Yarmouth	3 (0)	3 (5)	2 (0)
Great Point, Nantucket	0	2 (0)	2 (0)
Harding Beach, Chatham	3 (6)	2 (4)	4 (5)
Harthaven, Oak Bluffs	2 (0)	0	0
Howes St./Corporation Beach, Dennis	1 (0)	0	0
Jetties Beach, Nantucket	1 (3)	3 (4)	3 (5)
Kalmus Park Beach, Barnstable	7 (0)	5 (1)	5 (0)
Leland/East Beaches, Chappaquiddick	1 (0)	4 (4)	4 (7)
Lobsterville Beach, Aquinnah	ND[5]	ND	1 (2)
Long Beach, Barnstable	5 (4)	5 (5)	4 (3)
Menauhant Yacht Club Beach, Falmouth	1 (1)	0	0
Minimoy Island, Chatham	1 (10	0	0
Miramar Beach (Swan River), Dennis	ND	6 (8)	1 (3)
Muskeget Island, Nantucket	5 (2)	4 (5)	5 (6)
New Seabury, Mashpee	2 (4)	2 (3)	2 (3)
North Monomoy Island, Chatham	2 (5)	1 (4)	1 (3)
Norton Point Beach, Edgartown	5 (0)	6 (9)	5 (1)
Pleasant St. Beach, Chatham	ND	ND	1 (2)
Popponesset Spit, Mashpee	3 (6)	3 (4)	6 (4)
Sampson's Is.-Dead Neck, Barnstable	16 (22)	26 (18)	17 (30)
Seagull Beach/Radio City, Yarmouth	3 (1)	3 (4)	4 (3)
Sippewisset, Falmouth	1 (0)	2 (6)	3 (7)
South Cape Beach, Mashpee	3 (2)	4 (0)	2 (2)
South Monomoy Island, Chatham	29 (44)	24 (21)	20 (14)
Squaw Island, Barnstable	2 (2)	3 (6)	5 (6)
Sylvia State Beach, Edgartown	5 (4)	3 (6)	5 (2)
Tashmoo, Tisbury	1 (0)	19 (2)	2 (5)
The Galls, Nantucket	2 (0)	1 (4)	2 (3)
Tuckernuck Island, Nantucket	2 (4)	7 (14)	7 (8)
Washburn Island, Falmouth	4 (1)	3 (5)	4 (3)
Wasque, Chappaquiddick	2 (3)	2 (2)	1 (0)
West Dennis Beach, Dennis	6 (14)	6 (8)	6 (7)
Wilfred's Pond and Mink Meadows Beach, Tisbury	3 (5)	1 (3)	2 (0)
Total pairs in Action Area	**150**	**168**	**173**

5 No Data.

Piping plovers migrating north in the spring and south in the fall may pass through or stage in the action area. Table 5 provides information on the number of breeding pairs that nest north of the action area and their productivity. Without allowance for post-breeding mortality of adults or post-fledging mortality of young of the year, the population of plovers migrating south averages 1.7 times that of the number of birds migrating north the previous spring.

Table 5. Abundance of breeding piping plovers and average productivity (chicks/pair) north of the action area, 2003, 2006, and 2007.

Plover Breeding Site, Town	Pairs[6] (productivity)			Average Pairs
	2003	2006	2007	
Massachusetts north of action area	300 (1.48)	283 (1.20)	349 (1.19)	311
New Hampshire	7 (1.00)	3 (0.67)	3 (0.33)	4
Maine	61 (1.28)	40 (1.35)	35 (1.06)	45
Atlantic Canada	256 (1.62)	256 (1.82)	266 (1.14)	259
Total (weighted average productivity)	**624 (1.51)**	**582 (1.48)**	**653 (1.16)**	**620 (1.38)**

Known migratory stopover areas within the action area include Chappaquiddick Beaches on Martha's Vineyard, South Beach Island and North and South Monomoy Islands, Chatham, and Great Point, The Galls, Smith Point/Esther's Island, Nantucket (USFWS 1996) (Figure 3). Plovers migrating to and from northern New England and Atlantic Canada may be using these beaches. Observations of banded Atlantic Coast piping plovers indicate that plovers are not non-stop migrants and may use multiple stopover areas along the coast. At least one banded Canadian Atlantic Coast plover was observed during fall migration at Esther's Island, a second banded Canadian migrant was observed at Parker River National Wildlife Refuge in Essex, Massachusetts, north of the project area (D. Amirault, Canadian Wildlife Service, pers. comm. 2008).

Avian surveys conducted by CWA and MAS (2002 – 2006, including 52 non-winter aerial surveys and 44 non-winter boat surveys) detected few shorebirds, and most of these were not identified to species. Shorebird observations included one American oystercatcher (Report No.4.2.4-8), one red knot in a mixed species flock with six unidentified sandpipers, and 20 dunlins observed on Muskeget Island (Report No. 4.2.4-9). Perkins *et al*. (2003, 2004) reported oystercatchers and unknown shorebirds were observed in Nantucket Sound. Most surveys were conducted during the day under good visibility conditions precluding documentation of shorebird or plover use of the project area at night (DEIS 5-85) and during inclement weather. No piping plovers were documented. Paucity of shorebird observations and absence of piping plovers may reflect limitations of survey methods, but it is also plausible that shorebirds (which feed and roost on land) make infrequent use of the surveyed areas or that they tend to transit the ocean at higher elevation during migration.

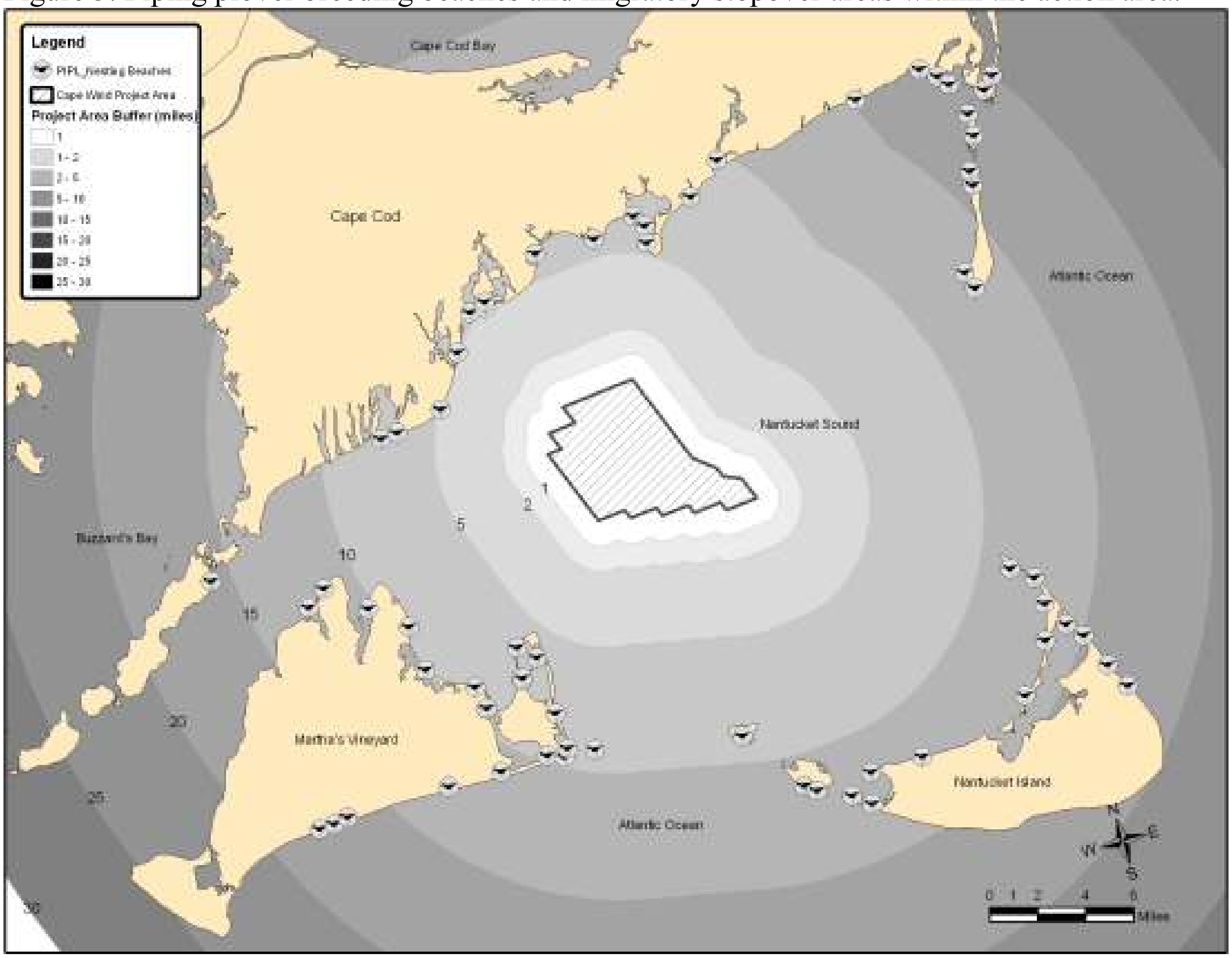
Factors affecting the environment within the action area

Male piping plovers first arrive in Massachusetts in mid-March to establish territories and begin nesting. Females generally arrive later, although plovers continue to arrive through April and

6 Total number of pairs observed.

into May. Egg-laying commences by mid-to-late April and chicks may begin to hatch shortly before Memorial Day weekend. Chicks fledge between late June and late August, with the peak in mid-to-late July. In Massachusetts, adult plovers may begin to stage and migrate shortly after their chicks have fledged. Plovers continue to stage and migrate throughout August and into early September.

Figure 3. Piping plover breeding beaches and migratory stopover areas within the action area.



Within the action area, piping plovers nest on private- and government (municipal, state and federal)-owned beaches in Nantucket and Vineyard Sounds. Most of these beaches are heavily used for recreation during the summer months when plovers are present and breeding.

Massachusetts state guidelines (MADFW 1993) for managing piping plovers have been in place since 1993, although intensive management of beaches was initiated prior to their publication. In 1994, the Service developed guidelines (USFWS 1994) for managing recreational activities on piping plover habitat and avoiding violations of the ESA.

Management at most sites in the state now conforms to both state and federal guidelines. All current nesting beaches and most historical or potential sites are censused each year, and more

than 80% of the major sites are monitored at least three times per week during periods of nesting and brood-rearing (Appendix C, USFWS 1996). Since 1995, estimates of productivity were obtained for more than 95% of all breeding pairs in the state.
On most Massachusetts beaches where nests are potentially threatened by pedestrian activities, nests are protected with buffers delineated by symbolic fencing and warning signs. Additionally, some nests are protected with wire predator exclosures. Within the action area, a number of nests are not exclosed at locations where predators have keyed into exclosures in the past and caused increased predation of eggs, chicks, and occasionally adults. Management of off-road vehicles at major beaches in Massachusetts conforms to most components of state and federal guidelines. Beginning in early April, and extending until the first egg hatches, off-road vehicles are restricted per guideline recommendations to discrete travel corridors along the outer edges of suitable plover nesting habitat. The guidelines call for sections of beach where unfledged plover chicks are present to be completely closed to recreational vehicles until chicks reach 35 days of age or are observed in flight. By requiring Orders of Conditions avoiding short- and long-term adverse effects on the habitat of listed species, the Massachusetts Wetlands Protection Act provides an effective regulatory tool to protect plover habitat from degradation caused by off-road vehicles and dune building activities.

Dog control is a continuing management problem on many Massachusetts plover beaches. Dogs disturb plovers and often prevent successful nesting by chasing adults and chicks and crushing eggs. Enforcement of town and state leash laws or dog prohibitions has been minimal at best. MAS manages plovers on more than 40 state and private beaches, and consistently reports dogs off-leash harassing plovers (E. Jedrey, Massachusetts Audubon Society, pers. comm. 2008). Some dogs are brought to beaches by owners; others swim a short distance from the mainland to barrier island plover nesting sites (e.g., Sampson's Island) and periodically disrupt breeding terns and plovers (B. Harris, Massachusetts Audubon Society, pers. comm. 2008).

Additional management challenges include increasing predation pressure, particularly from coyote, fox, cats, and avian predators, including crows and gulls. Predator control measures have not been implemented to a large extent due to restrictive state regulations, limited funding, or lack of support by the landowners. However, targeted predator management recently implemented at two beaches north of the action area (Plymouth Long Beach, Plymouth, Massachusetts and Crane Beach, Ipswich, Massachusetts) resulted in immediate increased productivity, although long-term benefits to productivity from targeted predator removal are unknown.

EFFECTS OF THE ACTION

The Service's evaluation of the effects of the federal action under consideration in this consultation is governed by its regulations and policies implementing the ESA. For instance, the Service is required to evaluate the direct, indirect and cumulative effects of the action on the species.[7] We conclude that construction, operation and decommissioning of offshore wind turbines may affect birds in several ways, including 1) risk of mortality from collision with the rotors and monopole or from turbulence behind the rotors (rotor wash), 2) short-term habitat loss or displacement during construction, 3) longer-term habitat loss or displacement due to the presence of the turbines and maintenance activities, 4) formation of barriers on migration routes (Exo *et al.* 2003), and 5) increased predation. Lighting of the facility could confuse or attract birds, potentially exacerbating the primary effects enumerated above, especially in poor weather or visibility conditions. Since the vessels involved in the construction, operation and decommissioning of the Cape Wind Project contain diesel and other oils, and the individual WTGs and the ESP contain cooling oil (some 40,000 gallons within the ESP), there is also the potential threat of an oil spill caused by a vessel accident, leakage or catastrophic failure of the ESP or one or more WTGs.

The ESA defines take to include the wounding and killing of listed species [16 USC 1532(19)]. Although both may occur with respect to this project, the Service has determined that it is highly unlikely that any plover or tern that collides with any part of the WTG or ESP would survive if injured. For example, injured birds would have a compromised ability to feed, defend against predation and complete a migration necessary for survival. As such, we refer to mortality as encompassing injury for the purpose of this BO.

The Service considered the effects to both species over 20 years, the anticipated project duration based upon CWA's estimate of a minimum 20-year life expectancy of the WTGs. Nevertheless, with respect to the beneficial effects that the Bird Island restoration project is anticipated to provide roseate terns, we examine it over 50 years, which is the designed life-expectancy of the restored revetment and re-nourishment operation (ACOE 2005).

As explained earlier, in applying the "best scientific and commercial data available" information standard in evaluating project impacts on listed species, the Service must carefully assess and address uncertainty. Uncertainty associated with evaluating the impacts of the Cape Wind Project on roseate terns and piping plovers arises largely from the fact that there are limited data for roseate tern presence within the project area in Horseshoe Shoal and no empirical data for

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[7] 50 CFR §402.02 provides that: Effects of the action refers to the direct and indirect effects of an action on the species or critical habitat, together with the effects of other activities that are interrelated or interdependent with that action, that will be added to the environmental baseline. The environmental baseline includes the past and present impacts of all federal, state, or private actions and other human activities in the action area, the anticipated impacts of all proposed federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process. Indirect effects are those that are caused by the proposed action and are later in time, but still are reasonably certain to occur. The Service concludes that "when effects on listed species are expected to be discountable, insignificant, or completely beneficial," it is appropriate to conclude that the action will not likely adversely affect the species. It defines those terms as follows: "Beneficial effects are contemporaneous positive effects without any adverse effects to the species. Insignificant effects relate to the size of the impact and should never reach the scale where take occurs. Discountable effects are those extremely unlikely to occur. Based on best judgment, a person would not: (1) be able to meaningfully measure, detect, or evaluate insignificant effects; or 2) expect discountable effects to occur."

piping plover presence within the action area other than at breeding sites on the periphery. Limited surveys for roseate tern presence at Horseshoe Shoal conducted by MAS and CWA documented use of the area by foraging, resting and travelling roseate terns. Avian surveys conducted sporadically over a five-year period were unable to conclusively document piping plover flights over Horseshoe Shoal, although breeding locations and productivity information are well documented for coastal beaches. The Service acknowledges these data gaps exist, but nevertheless is required to evaluate the project with the information that is available. We do so as transparently as possible, identifying the limitations of the methodologies used, the data assembled, and their resulting utility.

Given the limited data available concerning roseate tern and piping plover use of the project area, we depend to varying degrees on information collected from other wind projects where interaction with, or proximity to roseate terns, piping plovers, and similar species has been documented. In addition, where data is lacking, results are of questionable utility, or comparisons with other projects are not directly applicable, we account for this in the risk modeling upon which we have largely based our conclusions about the levels of anticipated take. For example, as is discussed below, the Service adjusted many of the factors in the model of collision risk (e.g., where information on the nighttime flights of terns is unavailable, we assumed some level of presence and incorporated a more conservative value).

Roseate Tern

Collision Risk

<u>*Other Wind Facilities*</u>
Some literature exists on offshore and onshore coastal wind projects of varying sizes and effects on bird species, including other tern and plover species that breed or roost in, forage within, or fly (during migration or commuting) through wind turbine fields. The Service evaluated information from what it considers to be the most relevant of these, given their proximity to known tern species and plover occurrences. Of these, the Service focused on one wind power facility in Belgium and three facilities within the western North Atlantic breeding range of the roseate tern and piping plover. These projects are not identical in design, size or location to Cape Wind. In fact, they differ in some notable respects, which we discuss in detail. Also, we recognize that the methodologies used to obtain data, particularly carcass searches used to obtain mortality data, in the studies of these other facilities have certain limitations. For these reasons, the data collected at these sites are not necessarily directly transferrable to our analyses. However, they do yield useful information about the species' behaviors and tendencies. We therefore thoroughly discuss the results, their utility, and how we apply the information to our modeling efforts.

<u>Zeebrugge</u>
The Zeebrugge commercial wind power facility in Belgium consists of 25 turbines located in a linear array on a breakwater (not in open water) between tern nesting habitat and foraging habitat. Fourteen of the turbines are directed at the sea and 11 are land-directed.

Everaert and Stienen (2006) recorded collision mortality at the Zeebrugge facility for three species of terns, common terns, sandwich terns (*S. sandvicensis*), and little terns (*S. albifrons*). They were not able to study roseate tern response to this facility because roseate terns do not nest or otherwise occur there. These studies showed that the Zeebrugge facility had an important negative effect on a tern colony due to collision mortality.

At Zeebrugge, over 6,000 pairs of terns nested between 100 ft (30 m) and 1,320 ft (400 meters) from the turbine array in 2004 and about 4,000 pairs nested between 165 ft (50 m) and 1,320 ft (400 m) from the turbines in 2005. Everaert and Stienen (2006) reported that an estimated 168 terns died from collisions with the turbines in 2004 (primarily the 14 turbines closest to the sea) and 161 were estimated to have died in 2005. This amounts to about 6.7 terns killed per turbine per year for the entire wind facility, and about 11 terns killed per turbine per year for the 14 turbines closest to the colony. Even within the 14 sea-directed turbine group, during 2004-2005, the four turbines closest to the tern colony caused the vast majority (90-92%) of the observed tern mortality. The authors attribute the recorded mortality of terns (and gulls) at Zeebrugge to the high number of daily crossings (>10,000 flights per day for common and sandwich terns) through the linear wind turbine array by the birds moving about the colony and travelling (commuting) from nesting habitat to their feeding grounds at sea.

The data from Everaert and Stienen imply little to no risk for adults and young terns at Zeebrugge during the post-fledging period and during migration, when passage rates through the turbine array are presumably much reduced. All tern mortality occurred between May and mid-August; no terns were found to have collided with the turbines the rest of the year (Everaert and Stienen 2006).

In a more recent finding from this study, Stienen *et al.* (2008) report strong evidence of a male bias among common tern collision fatalities at Zeebrugge during the egg-laying and incubation phases of the breeding season. The authors attribute the cause to behavioral differences between the sexes, and not to any morphological differences between the sexes, such as size, that could influence vulnerability to collision mortality. Specifically, female common terns spend more time in the colony defending their nests, incubating eggs and defending against kleptoparasitism, while males allocate more time to foraging and delivering food to their mates (Stienen *et al.* 2008). As a result, males were at greater risk of colliding with the turbines because as they foraged at sea and then returned to the colony, they made many more commuting flights past the turbine array than did females.

Among the limited studies that have been conducted to date, the tern mortality rate (6-11 terns per turbine per year) at Zeebrugge is the highest for those species reported in the scientific literature (e.g., Erickson *et al.* 2001; Perrow *et al.* 2006; Everaert and Kuijken 2007) and underscores the importance of site selection. Based on the Zeebrugge case study, the Service determines that placing wind turbines within or in very close proximity to tern nesting colonies is poor site selection that could result in high levels of mortality. It is similarly ill advised to encourage terns to nest in close proximity to turbines, as was the case at Zeebrugge.

However, the Service finds that there are substantial differences between the Zeebrugge facility and the Cape Wind Project. The Cape Wind Project is located > 19 miles (31 km) from the nearest major roseate tern colonies, while common, sandwich, and little terns nested between 100 ft (~30 m) and 1,320 ft (400 m) of wind turbines at the Zeebrugge facility (Everaert and Stienen 2006). Also, the small-to-medium-sized turbines at Zeebrugge are shorter with a smaller rotor-swept zone closer to the water surface, 52.5 to 164 ft (16 to 50 m) above water than those proposed for the Cape Wind Project, 75 to 440 ft (23 to 134 m) above water. In addition, the turbines at Zeebrugge are much more closely spaced, about 500 to 600 ft apart, than those proposed for the Cape Wind Project, which will be 0.33 to 0.5 mile apart.

Western North Atlantic

Presently, there are three small-to-mid-sized wind turbine projects within the breeding range of the roseate tern in the western North Atlantic. These are located at Sable Island and Pubnico Peninsula, Nova Scotia, and at the Massachusetts Maritime Academy (MMA) in Buzzards Bay, Massachusetts. The Sable Island wind generating station consists of five smaller, lattice tower mounted turbines 78 to 98 ft high (24 to 30 meters) with guy wires, the Pubnico Peninsula project consists of 17 large turbines (a 30 MW project), and the MMA single turbine is a Vestas V47-660kW model, 164-foot monopole with a blade tip height of 241 ft (50-meter monopole, blade tip height of 73.5 m). These projects differ in numerous and significant ways from the Cape Wind proposal. For example, they are much smaller in size and in the number of turbines, and they are land-based. However, because these wind facilities are all located closer to roseate tern nesting colonies than Horseshoe Shoal is to colonies in Nantucket Sound (the Monomoy Islands), they are pertinent to this analysis.

Sable Island: The Sable Island wind turbines were erected near the island's meteorological station but in a site used by nesting common and arctic terns (*S. paradisaea*) (Andrew Boyne, Canadian Wildlife Service, pers. comm. 2008). The few roseate tern pairs that are present on the island (four pairs were recorded in 2007) are apparently "on the other side of the island", but several hundred common and arctic terns nest within the scope of the supporting guy wires for the turbines (A. Boyne, pers. comm. 2008). Little information is available on this recently constructed project, however, approximately 10 terns may have been killed as a result of collisions and none were roseates (A. Boyne, pers. comm. 2008). A. Boyne (pers. comm. 2008). Photo-documentation found at www.greenhorsesociety.com/wind-energy/windfarm suggest that brightly colored flagging was added to the guy wires to reduce bird strike mortality at the structures. Indeed, A. Boyne (pers. comm. 2008) recently indicated that the turbine rotors are not spinning due to mechanical problems, thus tern mortality is almost certainly associated with strikes of the guy wires or possibly of other stationary parts of the latticed structures.

Pubnico Peninsula: The 17-turbine Pubnico Peninsula project was constructed in 2004 and 2005 and was visited by the Service and members of the United States and Canadian Roseate Tern Recovery Teams in October 2004. The project is located about 2.5 miles (4 km) from North and South Brothers Islands, the site of the largest roseate tern colony in Atlantic Canada. The turbines are land-based and are constructed on a peninsula adjacent to Pubnico Sound. About 50-75 pairs of roseate terns typically nest at the Brothers Islands (T. D'eon, c/o www.geocities.com/teddeon509/tern07), and, including common and arctic terns, the islands support from 450-750+ pairs of terns. Matkovich (2007) conducted two years of post-construction monitoring for avian and bat mortality from April 2005 to April 2007, and documented the mortality of 16 birds; all but one (a herring gull) were passerines. Mortality estimates adjusted for scavenger removal and searcher efficiency suggest 1-2.5 birds killed per turbine per year. No bats were found. All bird mortality was associated with periods of fog or overcast/windy weather, and most was associated with spring and fall passerine migration. Mr. Ted D'eon, Canadian Roseate Tern Recovery Team member and tern colony steward of the Brothers Islands, states, "I have no concerns with the wind farm disturbing the tern colony. I have seen no effect of the wind farm (negative or otherwise) on the tern colony" (T. D'eon, electronic correspondence 2008).

Massachusetts Maritime Academy: The single-wind turbine at the MMA in Buzzards Bay is a useful pilot project to examine the behavior of travelling and foraging common and roseate terns in proximity to a commercial scale WTG. Although the MMA turbine is land-based, its location on Taylors Point, a peninsula on the northern edge of Buzzards Bay, is useful because both common and roseate terns are present along the Cape Cod Canal on one side and along Buttermilk Bay on the other. Terns frequently cross the MMA campus in the area of the turbine to travel between these water bodies. Moreover, it is located much closer to the action area than any of the other projects we considered.

The Service recognizes the difficulty of extrapolating results from a one turbine project to a proposed 130-turbine facility. However, despite the small size of the MMA project, researchers studying tern behavior were able to obtain useful information about tern flight behavior, ecology and reactions to the turbine. Also, because there is only one turbine and it is land-based, monitoring (including post-construction collision mortality) could be more comprehensive and results therefore more accurate.

From April to November in both 2006 and 2007, Vlietstra (2008) recorded the abundance and flight altitude of common and roseate terns flying within 165 ft (50 m) of the MMA WTG. Although both tern species were present in the study area, common terns far outnumbered roseate terns. However, small numbers of roseate terns were present during mid-June to mid-to-late August in both years. Vlietstra observed 254 terns (both species) in 2006 and 294 terns in 2007 that were flying within 165 ft of the WTG.

An avian and bat mortality study was also conducted at the MMA during 2006 and 2007. No bat carcasses were found at the site but three birds (a laughing gull, an osprey and a great black-backed gull) were found and were likely casualties of a collision with the WTG. In July 2008, the Service (unpubl. data) observed a herring gull carcass that was also likely a collision fatality. Vlietstra (pers. comm. 2008) reported that European starlings were the only bird species observed to have briefly perched on the nacelle of the WTG.

The MMA WTG is lit by a flashing white aviation safety light. The light on the wind turbine is visible from the Bird Island tern colony, 7.5 miles (11 km) away. From the observations at the site, neither anecdotal nor empirical evidence suggests that the flashing light on the MMA WTG is an attractive nuisance to the common or roseate terns traveling through the area, although no specific study was undertaken to assess the effects of lighting on those birds. Similar to the Pubnico Peninsula wind power project, there is no evidence that the MMA wind turbine has had any adverse effect on the roseate tern (Matkovich 2007; T. D'eon, pers. comm. 2008; Vlietstra 2008).

To date, there is no evidence that any of the three wind power projects within the coastal breeding range of the roseate tern in the western North Atlantic have caused injury or mortality to roseate terns.

Collision Risk Exposure

Spring arrival and pre-migration fall staging

In the weeks when roseate terns first return to their northeast breeding range in late April and early May, they may be present in and around Nantucket Sound "in large flocks" (Gochfeld *et al.* 1998). During the post-breeding period, late July to September (to the date of departure when the North Atlantic population may congregate en masse and begin their migration to the wintering area), particularly a 21-25 day period in late August to mid-September, there is the potential that every breeding adult roseate tern in the North Atlantic population, and their young of that year, (e.g., about 8,000 adults and 4,000 young for the year 2007) will be in Nantucket Sound, within 20 miles of the Cape Wind Project area. During these times of year, roseate terns may enter the project area and be exposed to collision risk (See Collision Risk Assessment) from the Cape Wind Project as they forage or commute between foraging and staging habitats. Based on this information, the Service determines that the time of year when most roseate terns may be exposed to collision risk is during the few weeks in late April-mid-May and during the last three weeks of the post-breeding staging period (approximately August 20-September 15), when peak numbers of terns are reported along outer Cape Cod and in Nantucket Sound (Trull 1998; Trull *et al.* 1999).

Courtship

In courtship, roseate terns perform spectacular aerial displays at 9-91 ft (30-300 m). Courtship behavior is described as generally occurring at the breeding colonies and in the surrounding intertidal area (Nisbet 1981; Gochfeld *et al*. 1998). However, there is a remote possibility that some roseate terns might conduct courtship flights in the project area. Should this occur, the roseate terns could potentially fly within the rotor zones. The likelihood of courtship occurring in the project area is more remote if perching deterrents are installed, discouraging resting terns from gathering and possibly initiating courtship.

MMS and CWA have committed to field-test proposed perching deterrent measures during the pre-construction phase of the project (e.g., by using them on the meteorological tower already in place), to monitor the anti-perching measures post-construction with remote, motion-detecting cameras, and to alter methods if MMS and CWA, in coordination with the Service, find them unacceptable (BA). Based on roseate tern courtship primarily occurring at breeding colonies or adjacent intertidal areas and the CWA commitment to develop effective perching deterrent devices to further minimize the likelihood of courtship occurring in the project area, the Service finds that roseate terns are unlikely to collide with the WTGs or the ESP as a result of courtship flights.

Foraging
During the nesting season, roseate terns may be exposed to collision risk if they enter the project area to forage. The foraging range for nesting roseates is reported to be up to 30 km from the colony (Spendelow in Gochfeld. *et al.* 1998; Heineman 1992), although they often feed much closer [within 6.2 miles (10 km)] to the colony (Rock *et al.* 2007), depending on the availability of food (Nisbet 1981). Data collected in and around Cape Cod also suggest this to be the case. For instance, Nisbet (1981) and Heinemann (1992) reported on the feeding areas used by roseate terns nesting at Bird Island, the largest roseate colony in Massachusetts. Roseate terns nesting at Bird Island forage within Buzzards Bay and along the Elizabethan Island archipelago to Woods Hole and Vineyard Sound, but were not reported to venture farther east than the westernmost portion of Nantucket Sound. It is generally thought that what makes Bird and Ram Islands so attractive and productive for nesting terns is their proximity to nearby foraging locations at Mashnee Flats and the shallows surrounding Ram Island (Heinemann 1992; J. Spendelow, pers. comm. 2008). Heinemann (1992) states that the shoal at Mashnee Flats, and other feeding sites nearby, constitute the single-most important foraging area for the roseate terns nesting at Bird Island. The Service concludes that relatively few roseate terns will forage on Horseshoe Shoal during the breeding season due to its distance ($\geq$ 18 miles, 31 km) from the large nesting colonies in Buzzards Bay.

A male-biased mortality factor for roseate terns would be a very important concern, as the endangered northeast population already exhibits an imbalanced adult sex ratio skewed toward females (Nisbet and Hatch 1999; Arnold 2007). However, male-biased mortality of roseate terns from the Cape Wind Project is not anticipated, since unlike at Zeebrugge (for common terns), the Horseshoe Shoal project area is not within the foraging range of any major nesting colony for roseate terns, and likely only within the outer foraging area of terns nesting in the Monomoy Islands, 15+ miles (24 km) away.

Muskeget Island [8.6 miles (13.8 km) from Horseshoe Shoal] is historically the site of a large roseate tern colony. If Muskeget were to be re-occupied by nesting terns during the expected life of the project, there could be a future concern for overall and male-biased mortality of roseate terns. However, unlike at Zeebrugge, where terns nest immediately adjacent to the wind farm and pass through the turbine array as they commute to foraging areas, terns at Muskeget would be miles away from the nearest turbines and have alternative foraging locations along Nantucket Island and Tuckernuck Island that would not require crossing Horseshoe Shoal. The re-occupancy of Muskeget Island by large numbers of roseate terns is not reasonably certain to occur within the 20-year life of the Cape Wind Project.

Day versus night
In a comparison of collision risk by day or night, Everaert (2004) refers to a Dutch study in which it was reported that, in contrast to gulls, terns do not migrate much at night and therefore the chance for collision should be lower. Although roseate terns in Nantucket Sound are known to occasionally fly at night (Trull *et al.* 1999), what is known about roseate tern life history (e.g., Gochfeld *et al.* 1998) supports the conclusion that the species is diurnal and their use of the project area will primarily occur during daylight hours.

Juvenile roseate terns

For any particular year, there will be no exposure to collision mortality for about 18-20% of the individuals in the northeastern population of the roseate tern. Virtually all the one- and most of the two-year-old roseate terns do not migrate north to the breeding range, but rather are assumed to remain in the wintering grounds until their third year of age (Spendelow *et al.* 2002). Spendelow *et al.* (2002) report no cases where one- or two-year-old birds were detected breeding in their 10-year study of nesting roseates at Falkner Island. In examining the effect of Hurricane Bob, which struck the Cape Cod area on August 19, 1991, the authors found little impact of the storm on the survival of young from the two preceding years. They report that virtually no young from 1990 and relatively few from 1989 were expected to have been present around Cape Cod in August 1991 (Nisbet 1984), and these two cohorts showed no sign of less than typical survival (Spendelow *et al.* 2002). Spendelow (pers. comm. 2008) estimates that a small number of one-year-olds (~5%), and perhaps as many as 40% of the two-year-olds do migrate north in their first and second years, respectively. Spendelow *et al.* (2002) estimated that annualized survival probabilities for immature, pre-breeding roseate terns at Falkner Island were 0.53-0.57. Accordingly, in any given year, some 2,100 one-year-old roseate terns (4,000 X 0.55 survival rate X 95%) and another 726 two-year-old (4,000 X 0.55 X 0.55 X 60%), or ~2,800 pre-breeding roseate terns, will have no exposure to collision mortality from the proposed action, because they will not occur in the project area. Accordingly, subsequent discussions regarding the effects of the action on the roseate tern apply only to those birds that migrate north to the breeding range, and to their young of the year, that may stage in Nantucket Sound prior to their first migration south.

Collision Risk Factors

Flight height above water

Since the primary threat of injury and mortality to birds is the collision hazard posed by the spinning rotors and to a lesser extent to the vortex or turbulence effects behind the rotors (Desholm *et al.* 2004), the height of the rotors above the water and the flight altitude of roseate terns as they cross the project area is information essential to assess risk. For the Cape Wind Project, the rotor-swept zone is 75 ft (23 m) to 440 ft (134 m), an area of about 2.4 acres (1 ha) above the water surface. Birds flying below 75 ft at any given time are at no risk from the rotors but must avoid colliding with the monopole support towers. Birds flying above 440 ft should be at no risk.

In order to assess the presence of terns and their flight altitude above Horseshoe Shoal, CWA and MAS conducted aerial, boat and radar surveys. In 2003 and 2004, MAS observed more than 560 terns (both roseate and common) during boat surveys in Nantucket Sound. Of terns in flight (some were resting on the water), the average flight height was 29 ft (about 9 m), and 90% of the terns in flight were below 70 ft (21 m) [Perkins *et al.* 2004 and Sadoti *et al.* (2005a; 2005b) cited in BA]. During CWA's 2002 and 2003 aerial surveys, the flight heights of over 900 terns (both species) were categorized and 94% were at altitudes below the rotor-swept zone (RSZ); 6% were at heights within the RSZ. One roseate tern was among the terns identified to species that were flying at the height of the RSZ (page 4-22 of the BA). As summarized in the BA, CWA's breeding season surveys from 2002-2004 recorded 100 terns (both species) flying at altitudes within the RSZ (one was a roseate tern), but the majority of terns were flying below 39 ft (12 m).

During MAS boat surveys on Horseshoe Shoal in 2003 and 2004, 3% of about 450 terns (both species) flew within the rotor zone. Considering that common terns likely outnumber roseate terns in Nantucket Sound as they do in Buzzards Bay (perhaps by a ratio between 2:1 and 8:1) (Trull *et al.* 1999), very few of the terns observed in flight travelling through the Horseshoe Shoal project area within the rotor zone were likely to be roseate terns.

The difficulties of estimating flight height of small, fast moving birds during aerial and boat surveys over open water are discussed on pages 4-17 and 4-18 of the BA. In addition, the similarity of appearance between roseate and common terns, particularly when observed from an aircraft or boat at a distance, often prevented positive identification to species during CWA and MAS surveys. Other limitations of the CWA and MAS studies are discussed on pages 4-15 to 4-20 of the BA. We agree with MMS'ss assessment of the methodological limitations and resulting difficulties posed by studies that were of insufficient duration to capture seasonal and year-to-year variability. In summary, the most important limitations were that the aerial and boat surveys only obtained data during daylight hours, on days with light to moderate winds, and with good visibility. Accordingly, information about tern flight behavior over Horseshoe Shoal during night and other periods of low visibility (e.g., in fog or during inclement weather) when roseate terns and other birds may be at greatest risk to collision mortality remains unavailable. The Service further addresses these concerns when evaluating the assumptions and values employed in the collision risk model. However, based on the fact that the roseate tern is primarily a diurnal species and a visual forager, the Service concludes that during times when there is decreased visibility, roseate tern occurrence on Horseshoe Shoal will similarly decrease.

Additional questions remain concerning the proportion of terns in flight over the project area that will be at altitudes within the rotor zone; the Service nevertheless finds that the available data provide reasonable approximations if due allowances are made for inherent biases. Data obtained through radar were of little value to this evaluation because species identification for radar "targets" traversing the project area was not determined; no flight behavior of roseate terns or piping plovers was obtained from the radar data.

In the MMA post-construction monitoring study, Vlietstra (2008) reported that the flight altitude of about 550 common, roseate and unidentified terns that flew within 165 ft (50 m) of the MMA WTG averaged 63 ft (19.2 m), indicating that most terns flew below the rotor-swept zone, which occurs at this turbine between 86 ft (26.2 m) and 242 ft (73.8 m). Overall, the MMA study found that 85% of terns flew below the height of the rotor-swept zone, about 3% flew above the rotor zone and about 13% flew at the height of the rotor zone. When roseate terns were considered separately (n=8), all flew below the rotor zone. Roseates were found to pass through wind turbine airspace at a median height of 43 ft (13 m) (range 26-69 ft, 8-21 meters). Vlietstra noted that the likelihood of terns flying within 165 ft of the turbine was dependent on the operational status of the turbine rotors. During 2006, terns were 4-5 times less abundant when rotor velocity was > 1 rpm than when rotor velocity was < 1 rpm, leading her to conclude that when the rotor was operating, "terns seemed to avoid rotor-swept altitudes". Vlietstra (2008) concluded that when the rotor was operating, terns usually flew either below or above rotor-swept altitudes, whereas when the turbine was at rest, they flew at a variety of altitudes, including the height of the rotor.

At the MMA and nearby waters, Vlietstra (2008) found that in general, terns fly low to the water, about 3 to 39 ft (from 1-12 meters) when foraging, and it appears that they generally fly low when travelling short distances over land and water. When travelling, terns fly lower into the wind than when they fly with a tailwind (Hatch and Brault 2007; Nisbet 2008). This is important because when roseate terns on Horseshoe Shoal are foraging or flying into a head wind, they will likely be well under the RSZ and in no danger of collision mortality, but when they are travelling downwind, they are more likely to be flying higher and may be within the RSZ.

CWA and MAS also recorded flight heights of terns over Nantucket Sound during 37 aerial surveys and 36 boat surveys during the fall staging period (discussed on pages 4-22 to 4-27 of the BA). These studies suggest that from 0 – 31% (average about 6%) of travelling and foraging terns observed in the study area occurred at flight heights within the rotor zone (Hatch and Brault 2007). MAS boat surveys within Horseshoe Shoal found that 95% of terns flew below the rotor zone. In a recent study of visible bird migration during daytime in Helgoland, Germany, Exo *et al.* (2003) report less than 10% of divers (loons), grebes, ducks, mergansers, skuas, terns, gulls and auks were observed in flight at altitudes above 165 ft (50 m). Based on this information (from MMA, CWA and MAS), the Service finds that a large majority of terns of both species can be expected to fly below the rotor zone.

Visibility

Birds in flight, including roseate terns, are assumed to be at increased risk of colliding with WTGs when travelling during low visibility conditions. While terns could collide with the structures during different weather conditions, it is generally believed that birds are at greatest risk at night (Exo *et al.* 2003) and during other periods of limited visibility (Chamberlain *et al.* 2006). Roseates are generally considered a diurnal species, but they continue to forage during foggy weather (S. Hall, pers. comm. 2008; J. Spendelow, pers. comm. 2008), perhaps staying closer to the breeding colonies, and are known to fly at night during migration and during the post-breeding period (Hays *et al.* 1999, Trull *et al.* 1999) as they arrive and depart from overnight roosting locations.

The Service concurs with the discussion in the BA that terns are expected to regularly avoid collisions with the proposed WTGs, including the monopoles, on Horseshoe Shoal during favorable visibility and weather, because they are agile and maneuverable flyers (Gochfeld *et al.* 1998). At Zeebrugge, no dead terns were found beneath a disabled turbine which had no blades (Everaert pers. comm. cited in Hatch and Brault 2007). Unlike some birds, terns generally avoid colliding with man-made structures such as lighthouses (FWS unpublished data). Although researching the response of terns to a single onshore turbine, Vlietstra (2008) reports observing more than 550 tern flights past the MMA turbine, and none of the terns collided with either the rotors or the monopole.

The greatest potential for roseate terns to collide with the WTGs is during crossings of the Horseshoe Shoal project area during fog and rain, during nighttime and other low light conditions, or when the terns are flying in a downwind direction within the rotor-swept zone. As noted in the BA, there is a lack of observational data on tern occurrence in the project area during low visibility conditions. However, as previously noted, the Service believes that tern occurrence in the project area will decrease as visibility decreases.

The Service finds that the time of year when roseate terns are most exposed to collision risk is during the few weeks in late April-mid-May and during the last three weeks of the post-breeding staging period (approximately August 20-September 15) when peak numbers of terns are reported along outer Cape Cod and in Nantucket Sound (Trull 1998; Trull *et al.* 1999).
Weather data on visibility obtained from the Nantucket, Martha's Vineyard and Barnstable Airports for the years 2004 and 2005 were recently summarized by ESS Group on behalf of CWA (Table 6). At the Service's request, ESS analyzed what proportion of time (during daylight hours) for the month of May and for August/September visibility was recorded as being less than 0.25 mile. Combining two years (2004 and 2005) of data for these periods indicates that low visibility conditions for the two periods during the year when peak numbers of roseate terns were found in Nantucket Sound were very rare (only 3.67% of daylight hours during May and 1.63% of daylight hours during August/September). In view of this information, the Service concludes that when peak numbers of roseate terns are anticipated to be present in Nantucket Sound during daylight hours,[8] visibility will exceed 0.25 mile the vast majority of time, increasing the probability that terns and other birds will see and avoid the WTGs at Horseshoe Shoal.

Table 6. Visibility data from airports around Nantucket Sound.[9]

	Nantucket Airport				Martha's Vineyard Airport				Barnstable Airport			
Daylight hours[10]	**April/ May 2004**	**April/ May 2005**	**Aug/ Sept 2004**	**Aug/ Sept 2005**	**April/ May 2004**	**April/ May 2005**	**Aug/ Sept 2004**	**Aug/ Sept 2005**	**April/ May 2004**	**April/ May 2005**	**Aug/ Sept 2004**	**Aug/ Sept 2005**
No. hours visibility ≤0.25 miles	27	24	10	14	17	13	6	5	1	5	3	1
No. daylight hours in data set	383	383	430	430	382	381	431	431	383	383	431	431
% of daylight hours with visibility ≤0.25 miles	7.0%	6.3%	2.3%	3.3%	4.5%	3.4%	1.4%	1.2%	0.3%	1.3%	0.7%	0.2%

Review and Evaluation of Cape Wind Collision Risk Assessment

Models are frequently used as a tool in risk assessment when uncertainty prevents the prediction of an outcome (or outcomes) with accuracy. For the Cape Wind Project, the number of terns or plovers that will actually collide with the WTGs and ESP is not known, thus Hatch and Brault (2007) used a geometric model in which a series of factors (specific to roseate terns and piping plovers where available) are either measured, extrapolated from studies of closely related species or estimated from the scientific literature. Multiplying these factors together within the model provides a transparent way to develop an estimated probability of collision mortality to roseate terns and piping plovers from the project.

[8] Data from the 2006 ESS Group, Inc. study for CWA indicate that the majority of incoming and outgoing flights of the ±1,200 roseate terns observed over a four-day period occurred after sunrise and before sunset.

[9] Data Source: Nantucket, Martha's Vineyard and Barnstable Airport weather records, 2004-2005. Available at www.ncdc.noaa.gov/oa/ncdc.html.

[10] Daylight hours in April and May are from civil dawn to civil twilight; daylight hours in August and September are one hour before sunrise to one hour after sunset.

Hatch and Brault (2007), consultants for CWA, utilized tern passage rates, numbers flying at rotor height and other factors, and collision mortality data from sites such as Zeebrugge, Belgium to develop a collision risk assessment model for roseate terns occurring in or crossing the Cape Wind Project area. Collision probability at the Zeebrugge wind farm was believed to be influenced by a number of factors, including **flight behaviors**, e.g., irregular flight paths and circling around the colony, **flight height** as they commuted from nest sites to foraging locations and returned, **numbers of pairs nesting** at close range (most terns nested from 50-100 meters to 400 meters from the turbine array) and most importantly, **passage rates**, the number of terns flying past the turbines. An **avoidance rate** is the proportion of birds that could collide with a wind turbine generator but do not because they take effective avoiding action (Chamberlain *et al*. 2006). These and other factors for roseate terns at Horseshoe Shoal were similarly considered by Hatch and Brault to formulate an estimate of the number of terns that may collide with the WTGs of the Cape Wind Project.

It is important to note that tern mortality recorded at Zeebrugge included any and all collisions with the turbines (rotors and the monopoles) as well as any birds killed due to vortex (turbulence) effects of the rotors. Thus, estimated avoidance rates for common and sandwich terns at Zeebrugge take into consideration all potential causes of collision mortality. Accordingly, the collision risk assessment model developed by Hatch and Brault (2007) for this project, which utilized and adapted avoidance rates exhibited by common and sandwich terns at Zeebrugge, similarly accounts for any form of collision mortality with the wind turbines.

Hatch and Brault ran two versions of their geometric collision risk model for roseate terns in the Horseshoe Shoal project area (Hatch and Brault 2007; Appendix A of the BA). The result of the 2007 model estimated 0.3 to 2.3 (median of 0.8) roseate terns will be killed per project per year from collisions with the WTGs. Each of the parameters used in estimating the number of roseate tern collisions contains some uncertainty, due to variation in species presence/absence, numbers, or flight behavior and sampling variation. It is useful to know the range of this uncertainty because it can indicate what deviations from expected outcomes are possible and how likely they are. An uncertainty analysis was conducted by Hatch and Brault (2007) using a combination of Monte Carlo and data re-sampling methods to estimate the range of uncertainty surrounding the parameters of the model and to develop a probability distribution for the annual number of roseate tern collisions. Results of 5,000 simulations indicated a median mortality of 0.83 roseate tern per year with large uncertainty (5% to 95% probabilities: 0.01 to 8.2 roseate tern mortalities per year).

In the 2008 version of the model, the authors considered different parameter inputs, partially based on the comments of Nisbet (2008), and estimated that two roseate terns will be killed from collisions with the WTGs per year. Because inputs for many of the parameters used in both the 2007 and 2008 models, as well as those recommended by Nisbet (2008), were estimates or approximations, the authors acknowledge that the uncertainty surrounding the model inputs has a large impact on the variance of the mortality estimates. Using all the recommended revisions to the model suggested by Nisbet (2008) resulted in a mortality estimate of 17.5 roseate tern deaths per year. According to Nisbet (2008), 95% confidence limits for a “central” mortality estimate of 18 are 0.2 to 180 terns killed per year.

In review of the avian collision risk assessment for roseate terns presented in the DEIS and the BA, the Service is persuaded by some (but not all) of the revised collision risk model inputs recommended by Nisbet (2008). The Service independently reviewed all of the factors presented in Hatch and Brault's revised collision model and Nisbet (2008) as follows and estimates that, for a number of reasons described below, four to five roseate terns will be taken each year:

> Factor A. Numbers of terns (mixed species) present in the Horseshoe Shoal project area. The Service has reviewed the model inputs suggested by Hatch and Brault in the BA and those by Nisbet (2008) and finds that factor A, "numbers of terns", was underestimated in the BA but not likely by as much as 25% as suggested by Nisbet (2008). Nisbet's contention that the aerial surveys appear to have missed most of the high flying birds is based on questions about a large number of high flying targets detected in radar studies (and allegedly missed in aerial surveys) during 2002. The Service is persuaded by the discussion in Appendix A of the BA, that explains why more high flying targets were detected during the radar survey than during the aerial survey [different (larger) radar sampling area and longer duration]. However, the Service concludes that terns flying at dusk, night, dawn or otherwise in poor visibility would have been missed by CWA and MAS boat and aerial surveys. Therefore, although we disagree with Nisbet's proposed correction factor value, we find that a smaller correction factor of 10% is appropriate to account for undetected terns flying at night, dawn or in poor visibility, because the detectability issue he raised is addressed in part in Appendix A (BA). A=1.10
>
> Factor B. Proportion of total terns that were roseate terns. The Service agrees with Nisbet (2008) that the proportion of mixed species of terns recorded during surveys on Nantucket Sound that were ascribed by Hatch and Brault (2007) to be roseate terns, 3.2% and 10%, was too low. Trull *et al.* (1999) reports that common terns outnumbered roseate terns in ratios of 2:1 to 8:1 (33% to 11%) on outer Cape Cod during the post-breeding season. Trull (1998) reported that subsamples of roosting terns during this period showed an average of 35% roseates overall, but cautioned that because roseates call more frequently than do common terns at this time, observers can be misled if they rely on vocalizations alone to identify proportions between the species. CWA and MAS surveys during the pre-breeding and breeding seasons report the proportion of terns being roseates as from 2.4% to 20%. The Service finds that the roseate tern proportion estimated by Hatch and Brault (2007) and Appendix A (BA) (3.2% to 10%) does not sufficiently represent the range of variability for this factor, and finds that a correction factor of 15-20% is reasonable because it is intermediate between more historical estimates, for example Trull *et al*. (1999) and data from more contemporary CWA studies. B=1.15-1. 20
>
> Factor C is simply factor A multiplied by factor B. AXB=1.265–1.32
>
> Factor D is the fraction of roseate terns observed on Horseshoe Shoal that were travelling. As opposed to when terns rest on the water or forage from low flight heights, a proportion of terns using the Horseshoe Shoal project area will be travelling, which could, in certain circumstances, place them at the height of the rotors and in greater risk

of collision. Nisbet (2008) argues that Hatch and Brault (2007), as cited in the DEIS and the BA, underestimate not only the number of terns in Horseshoe Shoal but also the proportion of roseate terns in "travelling" flight that could be within the rotor-swept zone. Hatch and Brault (2007) analyzed tern flight height and wind direction data from the CWA and MAS surveys and found a clear effect of wind direction and speed on flight altitude. Notably, terns in flight with a tailwind will fly at higher altitudes. Based on that finding, Nisbet recommends that most or all terns travelling with a tailwind (within 45 degrees of the birds' heading) are likely to fly at rotor height. However, Hatch and Brault (2007) analyzed the height of terns in flight with a downwind and report that 40% were at rotor height, and not most or nearly all as Nisbet contends. Further, they dispute the detection questions raised by Nisbet, and contend that for the applicant's aerial surveys, all birds for the complete width of the transect to the height of the plane were recorded because "terns are relatively conspicuous and were detected over a wide area".

Nisbet (2008) accurately points out that no data were obtained on travelling terns or the height of their flight after dark. Nisbet (2008) argues that Hatch and Brault's estimate for this parameter, 5% travelling, was low and should be increased to 30%. However, this higher estimate is based on the assumption that the best available information underestimates that proportion by 600%. The Service is persuaded that a smaller correction factor is more reasonable because the detectability question has been partially addressed (see above).

Table 3 in Hatch and Brault (2007) indicates that from 0 to 31% of terns observed in flight from boat surveys were at heights in the rotor-swept zone, and an overall average is 6.2% of flying terns were at rotor zone height. However, there is additional uncertainty resulting from the lack of surveys conducted during low light and strong wind conditions. As discussed previously, although tern use of the project area is expected to decrease as visibility decreases, the Service nevertheless finds that some additional caution is warranted to account for uncertainty. Therefore, the Service determines that 6.2% is too low and that a correction factor of 10%-15%, should be applied to this factor. D=1.10-1.15

<u>Factor E</u> is the number of tern crossings of the project area. For this factor, Nisbet (2008) contends that Hatch and Brault (2007) underestimated the number of crossings by about one third (recommending a correction factor of 1.32). The Service does not concur with the correction factor recommended by Nisbet (2008) for the estimated number of roseate tern crossings of the project area per year. Nisbet reasons, in part, that the estimated number used by Hatch and Brault is too low and should be increased by about one third because terns flying at night were missed. Observations of terns departing South Beach, Chatham and Monomoy during four days in 2006 found that most tern flight activity occurred between 10 minutes of sunrise to 20 minutes after sunset (ESS 2006). Hatch and Brault (2007) report that these first and last birds were outliers, and that most tern flights occurred after sunrise and well before (~ 50 minutes) sunset. Although this study only occurred over the span of four days, it is supportive of Hatch and Brault's finding that day length (sunrise to sunset) provides a conservative measure of the period during which terns are active.

With due caution, the Service determines that reliance on limited data is preferable to findings of a speculative basis. The Service partially concurs with the discussion in Appendix A of the BA that Hatch and Brault's estimate is unlikely to underestimate the number of terns by 33% because although some terns do continue to arrive at night roosts after dark, this is a highly seasonal behavior restricted to several weeks in August through mid-September. Furthermore, roseate distribution on outer Cape Cod varies considerably from year to year. Recent roseate tern post-breeding distribution observations in 2007 and 2008 by MAS (B. Harris, pers. comm. 2008) and the U.S. Geological Survey (J. Spendelow, pers. comm. 2008) suggest that in some years, roseate terns were using Cape Cod Bay and the Atlantic Ocean more frequently than previously suspected. The Service interprets this to mean that in certain years, roseate terns may have less presence (i.e., fewer tern crossings) in Nantucket Sound than indicated by earlier fall staging period studies. A smaller correction factor, 10-15%, is therefore appropriate. E=1.10-1.15

<u>Factor F</u> is the fraction of travelling terns flying at rotor height. The uncertainty surrounding this factor is discussed above in factor A (i.e., detectability of high flying terns) and in particular, factor D above (% of terns flying downwind that were observed within the rotor zone). For reasons specified in those sections, the Service is not persuaded by Nisbet (2008) that a correction factor of 5 is appropriate. While Nisbet (2008) speculates that most or all of terns flying downwind fly at rotor height, Hatch and Brault (2007) rely on an empirical data set that includes 60% of downwind (day time) flyers observed not at rotor height. However, Hatch and Brault's (2007) estimate of 5% does not account for the range of uncertainty around this factor (e.g., terns that may have commuted back to an overnight roost at rotor zone height after dusk were not counted in the boat and aerial surveys because surveys only occurred during daylight hours). A higher proportion of the terns flying after dark may be at higher altitudes because unlike day time terns in flight, none of the birds after dark would be actively foraging (foraging terns generally fly low to the water). The Service concludes an appropriate correction factor for F is 3.0.

<u>Factor G</u> is number of rotors encountered. The figure used by Hatch and Brault (2007) is not in dispute, although the Service notes that some foraging terns may enter the turbine array, quickly catch a fish, and depart the wind project area. Terns behaving in this manner will not encounter as many turbines as estimated in the model. No correction factor is applied. G=1.0

<u>Factor H</u> is the species-specific probability of collision. Since no data for avoidance rates exist for roseate terns transiting WTGs, Hatch and Brault (2007) utilized common tern and sandwich tern data from the Zeebrugge, Belgium wind facility. Hatch and Brault then adjusted the common tern and sandwich tern avoidance rate based on the differences between Zeebrugge (a tern nesting colony) and Horseshoe Shoal (an area periodically used by foraging and travelling terns). Nisbet (2008) disputes the avoidance rate (0.953-0.983) for roseates used by Hatch and Brault (2007), and recommends a lower value based on the roseate tern being intermediate in size and flight characteristics between the two species.

> The Service is persuaded that the major differences between the Zeebrugge facility and the Cape Wind Project proposed for Horseshoe Shoal go beyond the size and flight speed of the birds, and support an avoidance rate in the range used by Hatch and Brault (2007; see also Appendix A of the BA). In particular, Everaert and Stienen (2006) report that while sandwich terns at Zeebrugge mainly flew a straight line (past the turbine array) to the feeding grounds and back, common terns flew more irregular flight paths and performed more circling movements around the colony (and presumably the turbine array). The flight patterns exhibited by common terns at Zeebrugge are associated with the breeding colony located nearby and resulted in increased collision mortality. Roseate terns flying across Nantucket Sound are not expected to fly in a similar manner, because Horseshoe Shoal is more than 15 miles from the nearest nesting colony. For the above reasons and other differences between the projects as previously discussed, the Service disagrees with the overly conservative avoidance rates recommended by Nisbet (2008) that are based on collision rates for common terns (in particular) and sandwich terns at the Zeebrugge facility in Belgium. Rather, the Service concludes that roseate terns will have a higher avoidance rate (and lower collision) rate through the Cape Wind Project area, compared to terns at Zeebrugge. No correction factor is applied. H=1.0

As described above, the Service has evaluated the input parameters for the collision model in light of rationale presented by Hatch and Brault (2007), Nisbet (2008), and Appendix A of the BA (2008), and exercised an abundance of caution by providing a reasonable but generous allowance for uncertainty around each parameter. Multiplication of our revised factors (A=1.1, B=1.15–1.2, D=1.1–1.15, E=1.1–1.15, F=3) yields a multiplier of 4.59 (using lower bounds for factors B, D, and E) and a multiplier of 5.24 (using the higher bounds). The multipliers applied to Hatch and Brault's (2007) estimate of 0.83 tern per year for the project yields an estimate of 3.8 – 4.3 roseate tern collisions per year for the proposed project. To further observe caution in our estimate, we rounded the estimate of 3.8 to 4.3 to four to five roseate tern collisions per year.

Indirect mortality to dependent juvenile roseate terns

Juvenile roseate terns remain dependent on their parent(s) for at least six weeks after fledging (Shealer and Kress 1994 in Gochfeld *et al.* 1998) and may remain dependent on parental feeding until after arrival in the winter quarters (Nisbet 1981). The loss of adults in late summer/fall may reduce the survival rate of their young. In this regard, the mortality of adults during the fall staging period may indirectly lead to the mortality of their dependent young, increasing the level of incidental take. If the take were to occur in May, prior to the breeding season, then no additional take (of dependent young) is anticipated. If the take were to consist of juveniles, non-breeders or failed breeders, then no additional indirect take of dependent young is anticipated. Accordingly, the Service acknowledges that a small number of dependent young, equal to but not exceeding the number of successful adult breeders taken during the post-breeding period, could also be taken as a result of the project.

Collision Risk Conclusion

After careful consideration of all available scientific literature, project-specific environmental documents, avian survey data, and known life history attributes of the roseate tern, the Service determines that the collision risk assessment [Hatch and Brault (2007), Appendix A in the BA, and Nisbet (2008)] should be adjusted to better represent the best available information. Accordingly, we adjusted the correction factors recommended by Nisbet (2008) and arrived at a collision mortality estimate of on average, four to five roseate terns per year for the Cape Wind Project. Some of these individuals may be adult breeding birds with dependent young, and their loss would reduce the survival rate of those young.

In light of concerns stemming from studies in Belgium, (Everaert and Kuijken 2007) concluding that wind farms can have a negative (population level) impact on birds, the Service reiterates that there are important differences between the proposed action and its likely effect on the roseate tern and effects on terns noted at the Zeebrugge, Belgium wind facility. The most important differences are:

1. At Zeebrugge, there are several thousands of pairs of terns, some nesting as close as ~100 ft (30 m) from the nearest turbine. Because thousands of terns nest nearby, tens of thousands of daily tern flights occur past the linear array of turbines during the breeding season. In addition, flight patterns near colonies can be more irregular and circling, placing more birds at altitudes within the rotor zone. At Horseshoe Shoal, the nearest roseate tern colony occurs at Minimoy Island (~18 miles or 29 km distant), and fewer than 60 pairs of roseates have nested there recently. Passage rates of roseate terns through the Horseshoe Shoal project area are much smaller than the passage rates for common and sandwich terns noted past the turbines at Zeebrugge.
2. The Zeebrugge WTGs are shorter and the rotor-swept zone is closer to the surface of the water than the Cape Wind Project WTGs. Zeebrugge WTGs are much more closely spaced, about 500 to 600 ft apart, vs. 0.33 mile to 0.5 mile apart for WTGs for the Cape Wind Project.

Even though the size of the rotor zone is much larger for the turbines proposed for the Cape Wind Project than the Zeebrugge turbines, more terns are likely to fly within the rotor zone at Zeebrugge because of low flight altitudes above water generally exhibited by terns in Nantucket Sound and the higher altitude of the rotor zone proposed by CWA. The wider spacing of turbines at Horseshoe Shoal will allow for more passive avoidance by terns transiting the project site. Based on the above, the Service concludes that the Cape Wind Project is unlike the Zeebrugge case study in many fundamental ways and therefore, is not likely to result in population level effects to the roseate tern.

Habitat Loss and Disturbance

Approximately 0.67 acre (0.003 square kilometer) of submerged land would be permanently occupied by the 130 WTGs and ESP, comprising less than 0.0042% of the project area and 0.0002% of Nantucket Sound (Table 5.3.2-2 in the DEIS). Rock armoring and scour mats would be placed underwater to protect the WTGs and ESP from scour, altering an additional 50.4 acres (20.4 ha) (MMS electronic transmission, accessed November 6, 2008). The total acreage affected by the WTGs, ESP, rock armoring and scour mats would be approximately 0.014% of Nantucket Sound. The Service finds that this amount of habitat loss, approximately 51.07 acres of 358,400 acres in Nantucket Sound, will have insignificant effects on the roseate tern due to the large amount of tern foraging habitat in the project area and throughout Nantucket Sound that will remain unaffected by the project. Moreover, studies conducted by CWA and MAS on tern travel corridors within Nantucket Sound do not suggest the existence of a foraging concentration site in the vicinity of Horseshoe Shoal that would be impacted by habitat disturbance or displacement. As such, aside from a coarse comparison regarding habitat availability, we find that these effects cannot be meaningfully measured, detected or evaluated.

The proposed landfall site for the submarine cable bringing power from the project to the onshore electrical distribution system is on the northeastern side of Lewis Bay in Yarmouth (BA). A cable laid along this route will pass near Egg Island, periodically used by roseate terns during the fall staging period in 2007 (B. Harris, *in litt*. 2008). The State of Massachusetts issued a 401 Water Quality Certificate (#W133633: finalized August 15, 2008) that established time-of-year restrictions relative to cable laying operations. In-water silt producing work is prohibited from January 15 to May 30. Jet plowing is only permitted from June 1 to January 14.

Short-term disturbance and reduced water clarity affecting roseate tern foraging activity or fall staging at Egg Island could result from this activity if the work occurred between July and mid-September. Disturbance during placement of the cable within the entire project area is likely to occur over a number of weeks, although water clarity in a particular area should quickly return to normal shortly after construction is concluded. Should roseate terns be foraging in the immediate vicinity of the jet plow, they may be briefly disturbed, although it is anticipated that the disturbance would not be any different than that of other construction vessels (barges or dredges) occurring in Nantucket Sound. Jet plow installation of the submarine cable near Egg Island is anticipated to be completed within several hours and suspended sediments should return to ambient conditions within two hours of disruption by the jet plow (MMS electronic transmission, accessed November 6, 2008). To further reduce adverse impacts, CWA proposes to avoid construction near Egg Island during low tide (when Egg Island is exposed) from mid-July to mid-September when roseate terns might be staging.

In summary, the Service concludes that foraging habitat loss and alteration from the Cape Wind Project in Horseshoe Shoal is not likely to adversely affect roseate terns due to the insignificant amount of foraging habitat that will be affected (<0.014%) within Nantucket Sound. Short-term disturbance and water quality effects from placement of the submarine cable to foraging or fall staging roseate terns are also not likely to adversely affect roseate terns due to the short-term duration of disturbance and water column sedimentation from submarine cable construction activities. The Service finds it unlikely that these activities will significantly alter tern breeding, foraging or others behaviors.

Piers as Fish Attractant Devices

During informal consultation on this project, the Service raised the concern that the WTG underwater monopole support structures will accumulate algae and be colonized by marine invertebrates and over time may serve to attract fish. If these prey species are present, the area around the WTGs may be utilized by marine birds such as terns, potentially putting them at greater risk of collision fatality. The potential for the monopole supports to become fish attractant devices is discussed in the DEIS on pages 5-16 and 5-17. The Service notes that the fish species likely attracted to these new underwater surfaces (for example, Atlantic cod, black sea bass, cunner, tautog and scup) are not those species commonly fed on (and fed to chicks) by roseate terns at breeding colonies (e.g., sand lance and silversides) (USFWS 1998).

The monopole foundations may, however, create tide rips as incoming and outgoing tidal currents surge past the structures. Roseate terns will feed in tide rips if the current brings schooling sand lance or silversides to the water's surface. Accordingly, the presence of the WTGs on Horseshoe Shoal may result in an increase in foraging opportunities and could increase their risk of collisions. However, because roseate terns feed predominantly by plunge-diving from 3.3-20 ft (1-6 m) and occasionally up to 40 ft (12 m) (Gochfeld *et al.* 1998), they are unlikely to be at risk of collision from the rotors when foraging. Furthermore, the presence of the monopoles may have the positive effect of increasing food availability in the project area.

In the event that the tidal flows around the turbine bases increase prey availability (a potential benefit) and attract roseate terns, the Service concludes that incremental collision risks are very low due to roseate tern foraging behavior occurring below the rotor-swept zone. Therefore, an increase in the prey availability within the project area is not likely to adversely affect roseate terns and may be considered a beneficial effect.[11]

Barrier or Displacement Effect

The Horseshoe Shoal project area is not a nesting location for roseate terns, and it is unlikely that during the nesting season, roseate terns from the large Buzzards Bay colonies would visit the project area on a daily basis when foraging for food for themselves or for their chicks. However, roseate terns that are travelling and feeding during early spring, small numbers of breeders (about 50 pairs) associated with the Monomoy Islands colonies, non-breeding roseate terns that may occur in Nantucket Sound and roseate terns staging in preparation for fall migration do use or commute through Horseshoe Shoal each year**.**

Some studies have shown that disturbance by operating wind turbines can displace birds from suitable breeding, roosting and feeding habitats (Exo *et al.* 2003). Exo *et al.* (2003) summarized studies undertaken at offshore and onshore wind facilities in Europe and identified a number of species that appeared to be especially sensitive to the presence of wind farms and that could be affected by habitat loss. These species included divers (loons), scoter, geese and waders. Exo *et al*. (2003) suggested that the degree of disturbance to these species could be determined by a number of factors, including availability of suitable foraging habitat, time of year and layout of the wind farm, among others.

The single turbine constructed at the MMA in Buzzards Bay, Massachusetts provided the first opportunity to examine this potential barrier effect for roseate terns. Vlietstra (2008) and the Service (unpubl. data, 2008) found no evidence that common and roseate terns crossing the MMA campus (>575 tern flights observed) were displaced from the area of the WTG (barrier effect). Common and roseate terns continued to forage along the shore of Buttermilk Bay, at its closest point about 330 ft (100 m) from the turbine. Terns using Buttermilk Bay exhibited no disturbance or displacement effect due to the presence of the turbine, its shadow, or noise it creates (Service, unpubl data, 2008). The Service acknowledges the disparity in the turbine size,

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[11] As noted in the preface to this BO, this conclusion only applies to the roseate tern; the Service previously expressed concern about potential risks that the monopole supports may pose to other avian species, e.g., waterfowl, that forage in the project area.

number of turbines, and areal extent of the wind turbine array proposed for the Cape Wind facility; nevertheless, we find these observations informative for our analysis.
Studies by Everaert (2004) on the Zeebrugge wind facility found that gulls and terns undertake thousands of local migration flights for feeding at sea and back. Gulls and terns at Zeebrugge crossed the dam (breakwater) where the turbines were located, adjusting their courses as needed to fly between the turbines. Everaert concluded that the turbines did not act as a barrier for these birds. Furthermore, the Service notes that roseates are known to nest on several islands with lighthouses, for example, Falkner Island, Bird Island, White Island, New Hampshire, and Petit Manan, Maine (USFWS unpublished data) and do not appear disturbed by the tall lighthouse structures, their flashing, rotating lights or their fog horns.

Langston and Pullan (2003) reviewed available literature and "on the basis of 10 years experience by the BirdLife partners…", developed a matrix of potential effects (disturbance/displacement, barrier to movement, collision, and direct habitat loss/damage), and evaluated sensitive bird groups. In this study, Langston and Pullan (2003) determined that among the four categories of potential effects, the *Sternidae* (terns) were sensitive to collision risk, and less so to the other potential effects of wind farms. Band *et al.* (2007) point out that post-construction disturbance (resulting in habitat avoidance or displacement) and collision risk are antagonistic processes that are spatially mutually exclusive. In other words, if birds stay away from a wind farm area, they are not at risk of colliding with the turbine rotors. They note that some bird species may avoid the area of a wind farm at first (no exposure to collision mortality), but then become habituated to its presence (introducing some exposure to collision mortality over time). It is important to note that short-term post-construction monitoring studies may miss detecting this dynamic.

In light of the available scientific literature discussed above, and observed roseate behavior around lighthouses and at the Zeebrugge facility, the Service concludes that the Cape Wind Project will not displace roseate terns from periodic and seasonal use of Horseshoe Shoal for foraging, resting on the water, or commuting from overnight roosting locations (i.e., South Beach) to feeding sites across Nantucket Sound. Therefore, the Service finds that the Cape Wind Project is unlikely to cause adverse effects by creating barriers for foraging or commuting roseate terns, or displacing roseate terns from foraging habitat. The Service will require that monitoring be designed to validate our reasoning during project construction and operation, so that we can revisit the attendant risks and develop minimization measures if necessary.

Increased Predation
The potential for the ESP and WTGs to act as opportunistic hunting perches for birds of prey is discussed on pages 5-60 to 5-61 in the BA. Some raptors such as the peregrine falcon (*Falco peregrinus*) are known to prey on terns, and their presence has been noted at roseate tern colonies in Connecticut, Massachusetts and Maine (J. Spendelow, pers. comm. 2008; C. Mostello pers. comm. 2008; L. Welch, USFWS unpubl. data).

If the anti-perching measures on the ESP and WTGs are ineffective, avian predators such as peregrine falcons could intercept roseate terns travelling across Horseshoe Shoal, resulting in occasional mortality of individuals. The presence of avian predators can also cause terns to take quick evasive flights, which could expose them to collision with the rotors. If anti-perching measures are effective, the effects from increased predation risk will be discountable. It remains unclear how the helipad on the ESP can be "bird proofed" while permitting unobstructed access to rotary aircraft. However, MMS and CWA have committed to field-test proposed perching deterrent measures during the pre-construction phase of the project (e.g., by using them on the meteorological tower already in place and an alternative location closer to tern activity), to monitor the measures with remote, motion-detecting cameras, and to alter methods if they are not acceptable as outlined in the 2008 Monitoring Framework.

The Service notes that predation is a much more important factor for communal nesting species such as roseate terns if it occurs at the breeding colonies, where predators can take multiple birds or eggs in a short time period and cause disruption or even colony abandonment. Predation in the project area (if it occurs at all) is likely to involve only the occasional, individual bird or birds. On this basis, the Service finds that the loss of roseate terns crossing Horseshoe Shoal to avian predators using the WTGs or the ESP as a hunting perch of opportunity will be a very rare event. The Service concludes that increased predation resulting from the Cape Wind Project is not likely to adversely affect roseate terns due to discountable effects.

Piping Plovers

Collision Risk

Of the limited studies conducted in the offshore or near-shore setting, review of available literature provides few documented observations of shorebird collisions with offshore wind generating facilities. Most studies investigating the effects of onshore and offshore wind generating facilities in Europe identify impacts to terns, seabirds, raptors, gulls and passerines (Chamberlain *et al*. 2006; Desholm *et al*. 2006; Hüppop *et al*. 2006; Petersen *et al*. 2006; Everaert and Kuijken 2007). Petersen *et al.* (2006) conducted extensive surveys and analyses of the impact of two wind generating facilities in Denmark and documented avoidance behavior and collisions for seabirds and passerines. On one occasion, four bar-tailed godwits were documented at a buoy-transect east of the Nysted offshore wind farm during fall migration (Petersen *et al*. 2006). At the onshore Belgian wind generating facilities of Zeebrugge and Brugge, the most common birds that collided with WTGs were gulls, ducks, pigeons and passerines (thrushes and pipits). Three shorebird species (common oystercatcher, black-tailed godwit and redshank) were documented collision fatalities, but numbers and season have not been reported (Everaert and Kuijken 2007). Everaert (Research Institute for Nature and Forest, Belguim, 21 and 22 October 2008 electronic correspondence to S. von Oettingen) reported finding one Kentish plover (*Charadrius alexandrinus*) in 2004 "that most likely collided with one of the turbines" at Zeebrugge, where 17 pairs of Kentish plovers and five pairs of ringed plovers (*Charadrius hiaticula*) bred nearby and foraged "very frequently near and even under the turbines." Everaert also noted that he and his colleagues have "only observed 1 ringed plover migrating at low height (below 30 m) during monitoring at sea."

The three onshore coastal North American wind generating facilities described under Roseate Tern Effects (Sable Island and Pubnico Peninsula facilities in Nova Scotia and the MMA facility) are located within the breeding range of piping plovers. Two piping plover nesting areas are within two miles of the MMA single wind generating turbine. Piping plovers have been observed on Sable Island, Nova Scotia during migration, although they are not known to nest on the island (D. Amirault-Langlais, Canadian Wildlife Service, electronic correspondence 2008). Post-construction surveys have not documented plover mortality or other shorebird fatalities attributed to wind generating turbines at any of these facilities (Matkovich 2007; Vlietstra 2008; Amirault-Langlais electronic correspondence 2008).

Collision Risk Exposure

Although piping plovers nest and forage on beaches encircling the project area, potential opportunities for plovers to cross the wind generating facility in Horseshoe Shoal include, to varying degrees discussed below: adults and young of the year during spring and fall migration, adults prospecting for breeding sites around Nantucket Sound in the early spring, breeding adults changing sites between nesting attempts, and adults foraging away from their nesting sites.

Non-migrating plovers

The Service finds that risks to plovers from crossings of Horseshoe Shoal not associated with migration are very low. Fidelity of adult plovers (especially males) to nest sites (MacIvor 1990; Strauss 1990; Loegering 1992) is consistent with low incidence of pre-breeding movements among potential nesting sites, although there may be more movements associated with first-year breeders prospecting for sites. Although a few plovers change sites between nest attempts within a season [three instances reported by MacIvor (1990), 14 suspected cases among 501 Massachusetts pairs in 1999], the vast majority of re-nests occur on the same site. Likewise, observations of adults foraging on beaches or intertidal flats away from their nest sites (MacIvor *et al*. 1985) within the range of possible movements between the mainland of Cape Cod and Nantucket and Martha's Vineyard are rare. Non-incubating adults are typically observed near the nest site, which facilitates their availability for nest-defense. Furthermore, foraging flights between the mainland and islands would be extremely inefficient compared with movements along the coastline over potential foraging habitats. While the Service has not completely discounted the potential for non-migrating piping plovers flying across Horseshoe Shoal, available data and basic piping plover breeding biology support an overall assessment of very low incidence and therefore, very low risk.

Migrating Plovers

There is no information regarding the altitude at which plovers migrate and whether they migrate over open water, close to the shore, over land or a combination of shore, water and land. Richardson (1979) conducted a radar study of shorebird migration over Nova Scotia and New Brunswick and documented shorebirds flying at a mean height of 1.2 mi (2 km). However, many of the shorebirds detected by Richardson are described as long-distance, non-stop migrants and their flight altitudes have little utility in understanding behaviors of piping plovers that appear to migrate shorter distances and make more frequent stops.

South Beach in Chatham, Massachusetts is located at the northeastern periphery of the action area (see Figure 1) and may be an important stop-over area for plovers in spring and fall migration. MAS recorded up to 85 piping plovers at one time on South Beach in August and up to 61 plovers in September (Perkins S., electronic correspondence 2008) between 1995 and 2005. Although there are no topographical features to funnel piping plovers through Horseshoe Shoal (BA), it is plausible that migrating piping plovers staging at South Beach, Monomoy Islands, or other migration stop-over sites within the action area (see Environmental Baseline section, Piping Plover) could cross the project area if they travel in a straight line to (or from) the south shore of Long Island. Absence of piping plover observations during aerial and boat surveys conducted by CWA and MAS may reflect limitations on the methodology to detect piping plovers (and other small shorebirds), infrequent flights of piping plovers through the area, or a combination of both. While it is possible that additional boat or aerial surveys might identify piping plovers transiting Nantucket Sound, the general paucity of small shorebird observations during past surveys suggests additional visual surveys are very unlikely to be informative.

Although reports of piping plovers in migration flights are non-existent, widespread observations of plovers roosting and foraging on beaches throughout their Atlantic Coast range during both the spring and fall support the idea that migration routes follow the coastline (USFWS 1996). Notwithstanding the lack of substantial evidence that piping plovers migrate over open water in Nantucket Sound, the Service contends that some migrating piping plovers may cross the Horseshoe Shoal project area. We base this conclusion on our independent review of the other information, including that referenced above, coupled with our understanding of the species ecology and its occurrence in the vicinity of the proposed project area.

Collision Risk Factors

Flight height is a significant factor in the assessment of collision risk; however, flight height has only been observed for breeding, courting or foraging plovers over and near their coastal shoreline habitat. Accordingly, there are no reported observations of piping plover flight height over offshore waters of Nantucket Sound, unlike the reports provided for roseate terns. If piping plovers migrate at high altitudes, then weather could be a factor in assessing the likelihood that a bird may fly at the height of the rotor-swept zone. The Service has considered information about general avian response to weather to assess the potential effects of weather on piping plovers.

Weather

Weather may be a factor in where and how plovers migrate, as with many other migrants (Chamberlain *et al*. 2006). Numerous studies indicate that the risk of bird collisions with turbines increases as weather conditions worsen and visibility decreases (Exo *et al*. 2003; Drewitt *et al*. 2006; Hüppop *et al*. 2006). The effect of weather on migrating birds' flight altitudes has been well documented through the use of radar and thermal imagery. During migration, birds may remain grounded during inclement weather; this is especially likely if, as commonly held, migrating piping plovers make relatively short flights along the coast. However, if birds are migrating at high altitudes and suddenly encounter fog, precipitation or strong head winds, they may be forced to fly at lower altitudes, exposing them to wind turbine collisions if they fly in the rotor-swept zone. (Drewitt *et al*. 2006). Hüppop *et al*. (2006) investigated year-round bird migration over the North Sea in Germany and the potential collision risk with offshore wind farms. The authors correlated weather factors with migrating bird flight altitudes or changes in flight altitude. For example, they found that tailwinds and light head winds were associated with higher flight altitudes, while a greater percentage of birds migrated below 200 m (600 ft) during nighttime rain events than on nights without rain. Based on the general assumption that piping plovers would respond to inclement weather and strong head winds similarly to other avian species, the Service finds that any high altitude migrating piping plovers encountering inclement weather or strong head winds while they are flying through the Horseshoe Shoal project area may be forced to fly below or within the rotor-swept zone.

Review and Evaluation of Cape Wind Collision Risk Assessment

The discussion in the BA relative to the collision probability assessment conducted by Hatch and Brault (2007) correctly states that "the numbers, height and course of [plover] flights are unknown." In the absence of empirical data, Hatch and Brault (Table 5) explore the ramifications of several hypothetical flight heights and avoidance rates for postulated 200 annual piping plover crossings of Horseshoe Shoal by analyzing a number of factors, including annual number of plovers migrating to breeding sites north of and in the action area, number of plovers crossing the Horseshoe Shoal project area, avoidance risk estimates and flight height. The Service independently reviewed all of the factors and found that some of the model inputs warranted revision. We therefore revised the collision assessment as discussed below, and estimate that, under a "worst plausible case scenario," collisions will not cause take of more than 10 piping plovers over the 20-year life of the project:

1. *Annual number of plovers migrating to and from breeding sites north of and in the action area***.** The Service concludes that Hatch and Brault's estimate that annually 2,458 piping plovers "cross the Massachusetts coastline" is too low for the purposes of this analysis. A post-breeding population of 260 pairs in Canada fledging 1.6 chicks/pair, for example, would yield an estimate of 1,456 combined northward and southward migrants (260 x 5.6, assuming no mortality between fledging and southward migration to Massachusetts), while approximately 525 pairs in Maine, New Hampshire, Massachusetts in and north of the project area (Tables 4 and 5) fledging 1.4 chicks/pair yield an estimate of 2,835 (525 x 5.4) northward and southward migration flights. Thus, the Service estimates approximately 4,300 northward and southward migration flights by piping plovers currently breeding or fledging in and north of the action area.

   Attainment of the 400-pair recovery objective in Atlantic Canada during the life of the proposed project would increase the post-fledging population to almost 1,450 piping plovers (400 X 3.6), assuming productivity of 1.6 chicks/pair to support a stationary population (Calvert *et al.* 2006). Thus, the Service finds that up to 5,000 migration flights per year (925 pairs x 5.5) to and from breeding sites north of, and in the action area could be attained during the life of the proposed project.

2. *Number of plovers annually crossing the Horseshoe Shoal project area*. Since the number of piping plover crossings through the Horseshoe Shoal project area is unknown, Hatch and Brault postulated 200 annual crossings discussed in a hypothetical example offered in a February 24, 2005 letter from the MADFW. The inference by Hatch and Brault that this is 10% of total migration flights is not consistent with our estimate in #1 (above). Furthermore, it seems plausible that greater than ten percent of plovers migrating to and from nesting sites north of Horseshoe Shoal might follow the relatively direct route across Nantucket Sound. Accordingly, the Service has evaluated ramifications of estimated crossings of up to 20% of 5,000 migration flights, or up to 1,000 annual crossings. This figure also provides an allowance for some crossings associated with non-migration activities such as within-season changes in breeding site,

first-year breeders looking for nesting sites, or birds traveling to post-breeding staging areas at South Beach or Monomoy from nesting sites in the southern or western parts of the action area. The Service also notes that current migration flights are estimated to be 4,300 per year, not 5,000, which provides additional leeway for within-season crossings.

3. *Avoidance rate estimates.*[12] In the absence of values specific to plovers, Hatch and Brault considered four avoidance rate values from Chamberlain *et al.* (2005)[13] for piping plovers (0.91, 0.95, 0.98 and 0.99). Notwithstanding the likelihood that low visibility (poor weather conditions, nighttime flights) decreases avoidance rates, the Service finds it highly probable that avoidance rates for piping plovers are toward the higher end of this range. We rely on the following factors in reaching this conclusion: evidence of good night vision inferred by nocturnal foraging behavior [albeit with lower peck rates than during the daytime (Staine and Burger 1994)], agility of adult plovers observed in distraction displays (including abrupt flights to escape the potential predator) on beaches[14], and the fact that any plovers in flight over Horseshoe Shoal will not be distracted by concurrent foraging or courting activities. Furthermore, several factors make it relatively unlikely that piping plover flights over Horseshoe Shoal will occur during periods of poor visibility in fog or storms often associated with low avoidance rates. Because plovers do not forage over water, the need to provision themselves or their young is not a factor compelling them to fly through the wind farm during periods of adverse weather. If, as is commonly held, migrating piping plovers make relatively short flights along the coast, they are likely to select good weather and land if they encounter poor conditions. A high altitude migration flight scenario, discussed under Collision Risk Factors (above), could entail occasional descents to rotor height during sudden periods of inclement weather. However, the frequency and duration of such poor weather (coinciding with one or more migrating plovers) is likely to affect only a very small fraction of the population and, therefore, exert only a small effect on the average avoidance rate.[15] Although avoidance rates as low as 0.91 cannot be completely discounted, the Service finds that 0.95 is a more likely lower bound for this parameter (even though the higher avoidance rates of 0.99 or 0.98 may be the most realistic for piping plovers over Horseshoe Shoal).

4. *Flight height estimates*. Hatch and Brault also evaluated three different height distributions: at or below the rotor-swept zone at approximately 0 to 100 ft [0 to 30 m] above sea level (asl), within the rotor-swept zone at 73 to 440 ft (23 to 134 m) asl and a

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[12] The Service is very mindful of the sensitivity of collision estimates to avoidance rates, as well as strong cautions pertinent to hazards associated with application of avoidance rates derived from one site or species to another (Chamberlain *et al*. 2005). We note, however, the impracticability of obtaining site-specific turbine avoidance rates <u>before</u> a project is constructed. Even the monitoring of a "test-turbine" will have difficulty capturing effects of long-term variability in bird behaviors under different conditions and suffer limitations due to small sample size.

[13] Chamberlain *et al*.'s Appraisal of Scottish Natural Heritage's Wind Farm Collision Risk Model and its Application (British Trust for Ornithology Research Report 401) provides a particularly thorough evaluation of avoidance rate estimates and their effect on collision risk.

[14] Although evidence of good nocturnal vision and agility are drawn from onshore behaviors, we know of no reason why fundamental physiological characteristics enabling these behaviors would change in flight or over water.

[15] We further note that a high altitude migration strategy would decrease the number of piping plovers likely to fly at rotor height (see *Flight height estimates* section, below).

range within and above the rotor-swept zone 100 to 2,000 ft (30 to 600 m) asl. Although Hatch and Brault's worst-case scenario (all birds flying at rotor height) cannot be completely discounted, the Service finds it is highly unlikely. Except for courtship displays, piping plover flights observed in and near beach habitats are low to the ground (well below rotor height). It also seems plausible that migrating plovers would seek high altitudes (above rotor height) for longer flights, as has been documented for other migrating shorebirds (Richardson 1979; Able 1999; Langston and Pullen 2003; Petersen *et al*. 2006). Under a "high altitude migration scenario," piping plovers could be forced into the rotor-swept zone during migration as a result of the sudden onset of unfavorable weather conditions such as strong head winds, precipitation, fog or low cloud ceiling (Able 1999; Exo *et al*. 2003; Drewitt and Langston 2006; Hüppop *et al*. 2006). Plovers taking off from or landing at staging and migratory stop-over beaches around Nantucket Sound may pass through altitudes that include the rotor height; however, this seems likely to occur closer to land than at Horseshoe Shoal. Furthermore, a high altitude migration strategy would also suggest relatively longer migration flight paths between piping plover wintering and breeding areas that would avoid the action area altogether (radically reducing the estimate of plover crossings of the wind facility). The Service finds that the more likely upper bound of collision risk within the project area is that associated with distribution of flight heights between 30 and 600 meters (the middle range evaluated by Hatch and Brault, Table 5).

We note that the risk assessment model does not account for the risk to piping plovers posed by the monopoles. The Service finds the risk of collision with these stationary structures is highly unlikely and therefore discountable. The stationary, 16.75- to 18-foot-wide monopoles are fundamentally different from the WTGs, that have a circular area equal to 2.4 acres (1 ha). We are not aware of instances of piping plovers colliding with the many human-made structures on and immediately adjacent to nesting beaches. The most ubiquitous structures are relatively short houses (many located in the flight paths of adults commuting between bayside foraging flats and oceanside nesting areas on barrier islands and spits, especially in New York and New Jersey). However, lighthouses are prominent features at a number of important piping plover breeding sites, many at the ends of sandspits and other promontories. Suggestions of collision hazards from lighthouses are notably absent from the piping plover literature.

Based on the above, we recalculated the risk and anticipated take to be caused by collisions. Using an avoidance rate of 0.91 and all piping plovers flying within the rotor-swept zone, Hatch and Brault projected 244 crossings per collision, which translates into 0.8 collision per year if there are 200 annual piping plover flights through the wind farm (Hatch and Brault incorrectly calculated 1.2 collisions per year for this scenario). Doubling the estimate of annual migration flights by piping plovers breeding or fledging in and north of the action area (the Service's revision explained under #1, above) during the life of the Cape Wind Project, and adjusting the estimate of annual crossings through the wind facility accordingly (to 400 per year), increases the collision estimate to 1.6 per year. As noted under discussion points #3 and #4, however, the Service finds that an avoidance rate of 0.95 and flight altitudes distributed between 30 and 600 meters provides a more reasonable upper bound estimate of one collision per 2,273 crossings. Therefore, the Service calculates that <u>four hundred estimated annual crossings of the wind farm</u>

result in 0.18 collisions per year. Increasing annual crossings to 1,000 (20% of 5,000 migration flights), with crossings distributed between 30 and 600 meters and an avoidance rate of 0.95, yields an estimate of 0.44 collision per year. Notwithstanding considerable uncertainty regarding all parameters, the Service finds that more than 0.5 piping plover collision per year (approximately 10 piping plovers averaged over the life of the project) is extremely unlikely. Indeed, we conclude that collisions may be much more infrequent.

Habitat Loss and Disturbance

Piping plovers nest on beaches and forage in the intertidal zone, on wrack at the high tide line, in ephemeral pools or sparsely vegetated dune grass on Cape Cod, Nantucket and Martha's Vineyard beaches. The Cape Wind Project and land-based components will not be situated in or near piping plover nesting, resting, or foraging habitat; therefore, the Service anticipates no adverse effects to plovers as a result of long- or short-term habitat loss.

The proposed submarine cable route through Lewis Bay is greater than 300 m (985 ft) from piping plover nesting habitat at Kalmus Beach, Hyannis to the west and Smiths Point, Yarmouth to the east. A cable vessel with jet plow equipment is expected to pass through the area in a matter of a few hours as it lays down cable. This equipment is similar to existing boat traffic (not noted in the piping plover literature as a source of direct disturbance) and will not adversely affect breeding piping plovers. The submarine cable landfall is not near breeding piping plovers.

Based on the sufficient distance of the cable laying activities from nesting piping plovers and disturbance effects no greater than existing marine vessel traffic, the Service anticipates no adverse effects as a result of disturbance from the installation of the submarine cable.

Barrier or Displacement Effects

There is no piping plover breeding or feeding habitat within Horseshoe Shoal, and within-season crossings are anticipated to be infrequent at best. The number of migrating piping plovers crossing Horseshoe Shoal in the vicinity of the proposed wind park is unknown. It is likely that most piping plovers migrate along the outer coastal beaches (USFWS 1996) since almost all observations are at stop-over locations along the coast. As discussed under Collision Risk (above), the most likely piping plover activity in the project area is by transiting northward or southward migrants.

Research indicates that some birds may fly around a wind farm instead of through, although avoidance could be dependent on the distance between turbines, the size of the wind farm and the extent of the displacement (Langston and Pullen 2003). Avoidance reactions to offshore and near-shore wind farms have been documented for some European birds, including divers, scoters, geese and waders (Drewitt and Langston 2006; Exo *et al.* 2003). As previously stated in the Roseate Tern section for Barrier and Displacement Effects, avoidance behavior, although it may increase the energetic requirements as a bird deviates from a routine flight path, will decrease the likelihood of collision with wind turbines. The Service anticipates that the greatest likelihood of plovers crossing Horseshoe Shoal will be during migration; should plovers perceive the wind farm as a barrier, we anticipate that the slight increase in the migratory distance would only marginally add to the overall energetic requirements of migration. Although longer migrations may contribute to patterns of lower survival rates of Atlantic Coast piping plovers

breeding at higher latitudes (Calvert *et al.* 2006), the contributing factors (e.g., foraging resources at stop-over locations, weather, predation by raptors) are unknown.
On the basis of the best available information, the Service finds that displacement effects of the proposed project on piping plovers are very likely to be inconsequential. Therefore, the Service concludes that the Cape Wind Project is not likely to adversely affect piping plovers through the creation of barriers for migration and commuting. There will be no adverse effects from displacement of plovers, since Horseshoe Shoal is not breeding or foraging habitat.

Roseate terns and piping plovers

Lighting
The effects of FAA lighting of the WTGs and the potential that lighting may become an attractive nuisance and cause disorientation of migrating plovers or terns at night or under poor visibility is discussed on pages 8-13 and 8-14 of the BA. The lighting proposed for 50 of the 130 WTGs and the ESP addresses Service interim guidelines to minimize and avoid impacts to migratory birds, by minimizing the use and intensity of lights and number of flashes per minute (USFWS 2003); the lighting scheme also complies with current FAA and U.S. Coast Guard requirements (see Project Description). The ESP and a portion of the WTGs will be lit with a single flashing red light. The 72 interior WTGs will not be lit with aviation lighting at night. All aviation lights will flash synchronously.

Studies cited in the BA, Gehring *et al.* (2007) and Shire *et al.* (2000) found that steady burning FAA obstruction lighting and some other types of lighting on mainly land-based tall structures [generally communication towers at heights of 1,000 ft (305 m)] can attract or disorient night migrating birds, resulting in collisions with those structures. In a Michigan study, Gehring *et al.* (2006, 2007) reported that where red, continuous lights were extinguished and replaced with flashing or strobe lights, there was a 71% reduction in avian collision mortality on communication towers. Jones and Francis (2003) reported a dramatic reduction in birds reported killed during a 41-year study of night-migrating bird mortality at a lighthouse in Canada, comparing number and species of birds killed before and after a change in the light signature. The reported bird mortality dropped significantly as a result of altering the intensity and nature of the light beam. Longcore *et al.* (2008) conducted a comprehensive review of research on the effects of lights from tall structures on night migrating birds and concluded that the use of synchronously flashing lights would reduce avian mortality at tall structures.

The Service concurs with the conclusion in the BA that the best information available does not support the hypothesis that terns are attracted to refracted light, and thus have an increased risk to collision mortality. Additionally, as previously noted, several contemporary roseate tern nesting colonies occur on (or near) islands with lighthouses. Examples include Falkner Island, Bird Island, White Island, and Petit Manan. Tern biologists have not reported that the lights at these locations are adversely affecting the terns that nest there (USFWS unpubl. data). In spite of the line-of-sight presence of the MMA turbine to the breeding colony at Bird Island and the Pubnico Peninsula wind turbines to the roseate tern colony at The Brothers Islands, no tern mortality has been detected at these facilities.

Despite the prevalence of artificial lighting from coastal developments, including beach homes and a lengthy history of lighthouses on offshore islands, promontories, and low-lying sandspits, attraction or disorientation of migrating piping plovers due to refracted light has never been reported as an actual or potential threat. During a 2005 study at Cape Lookout National Seashore, American oystercatchers (*Haematopus palliatus*) were observed running or flying directly into the headlights of vehicles transiting their beach foraging territories at night (Simons *et al.* 2005). However, neither USFWS (1994) nor MADFW (2003) guidelines for managing recreational vehicles in Atlantic Coast piping plover breeding habitats indicate that vehicle headlights are a hazard to adult or fledged juvenile piping plovers (even in close proximity to nesting and foraging habitats), nor has it been suggested that headlights or other artificial lights are a potential threat warranting further investigation.

Research demonstrates that some bird species are attracted to or confused by artificial lights. The Cape Wind proposed lighting incorporates flashing lights of low to medium intensity, which is consistent with recommended methods for avoiding or minimizing avian mortality from tall lighted structures (Jones and Francis 2003; Longcore *et al.* 2008). In view of the above, the Service finds that lighting of the WTGs and the ESP is not likely to adversely affect foraging, commuting or fall staging roseate terns or migrating or commuting piping plovers. We base our conclusion on the available literature, our current understanding of the species' reactions to artificial lights, and the implementation of synchronized, flashing lights per published recommendations and the interim Service guidelines.

Oil Spill Risk Assessment

The DEIS, BA and the supplementary report 5.2.1-1 (Etkin 2006) review the potential for an oil spill associated with about 500 marine vessel trips to Horseshoe Shoal during construction and maintenance of the proposed project. The possibility that other vessels (non-Cape Wind-related) may collide with the WTGs or the ESP and spill oil, or the catastrophic failure of the ESP (which will contain more than 40,000 gallons of mineral oil and 2,000 gallons of diesel), or one or more of the WTGs (each contain about 215 gallons of lubricating oil) is also considered in the oil spill probability analysis by Etkin (2006). There is no oil in the submarine interconnecting cables.

The earlier description of the baseline included discussions of numerous crude oil spills that impacted roseate terns and piping plovers. It is important to underscore that the type of oil maintained in the WTGs and ESP is different from crude oil in many important respects. For example, the majority of the oil that will be in the ESP is mineral oil. Mineral oil is light, floats on water and is generally non-persistent (rapidly breaks up into small droplets in the water column); about 12% of it is estimated to remain on the water surface after 36 hours post-spill (Etkin 2006).

The Service concurs with the assessment made in the BA that the potential impacts to plovers and terns from oil spills associated with the proposed action would depend on the season, the size and location of the spill, and the wind direction. It will also be influenced by oil spill planning and preparedness, the response action, and the type of oil that was spilled.

Roseate terns forage at the sea surface and frequently loaf and bathe in near-shore areas around their nesting colonies. As a result, they are vulnerable to oil spills in the marine environment. Piping plovers nest above the high tide line and forage along the wrack line and intertidal areas; an oil spill reaching piping plover foraging and breeding habitat would adversely affect them. Oil robs bird feathers of their insulating capacity and may cause oiled birds to die of exposure or of starvation if they are unable to fly. It may also cause hatching failure if the oil from an incubating bird spreads to its nest and eggs. Oil can be toxic to roseate terns and piping plovers if ingested when feeding or during preening.

Etkin (2006) analyzed the probability that an oil spill might occur at the wind energy complex. The analysis estimated the probability of a theoretical occurrence of an instantaneous release of 40,000 gallons (151,000 liters) of electric insulating oil and other oils from the ESP and the WTGs for a total worst case of 68,000 gallons (257,000 liters) of oil—an extremely unlikely scenario. Such a worst-case discharge event would only occur if something damaged the ESP *and* all 130 of the WTGs to the extent that the entire contents of all four electrical transformer insulating oil tanks, as well as the oil in each of the WTGs, would be released almost instantaneously. The analysis involved two major components: 1) determining the probability that any spill might occur from the ESP and WTGs: and 2) analyzing the range of spill sizes (and associated probabilities) that might be expected if a spill were to occur from the ESP and WTGs. The analysis involved a four-step process:

1. Evaluate and describe the events that might cause damage to the ESP and/or WTGs (e.g., extreme weather events, earthquakes, accidents, structural failures, oil transfers, etc.).
2. Estimate or qualitatively analyze the probability of each of these events occurring.
3. Estimate or qualitatively analyze the probability that for each of these events that damage occurs to the ESP and/or WTGs.
4. Estimate or qualitatively analyze the probability for each of these events to cause damage sufficient to cause an oil spill from the ESP and/or WTGs.

Etkin (2006) performed quantitative analyses for those events using previous spill/accident data records, and other events to the extent possible. Where quantitative analyses were not possible or practicable, Etkin performed qualitative evaluations. Once these probabilities were analyzed, the potential spill sizes that might occur (if a spill were to occur) were then analyzed using data from comprehensive oil spill databases. From this analysis, the probability that a worst-case discharge from the ESP and WTGs would occur was determined, as well as the probability of the smaller spill volumes.

Etkin's oil spill probability analysis concluded that the highest possibility of an oil spill occurring in the area in and around Nantucket Sound is related to vessels transiting the area, regardless of the presence of the Cape Wind facility and related work vessels, and that only 7% (two spills) of all spills expected in Nantucket Sound during a 30-year period (an estimated 29 spills) could be attributed to the addition of the Cape Wind facility. Of the two spills, there is a 90% chance that they would involve volumes of 50 gallons or less, and a 1% chance that they would involve volumes of 10,000 gallons or more. The probability of a spill in the same 30-year period involving the entire volume of 68,000 gallons of oil contained in the ESP and the 130 WTGs is less than one in a million.

In addition to estimating the probability of an oil spill associated with the proposed action, consultants for CWA also modeled the likely trajectories of oil released from the ESP and calculated probable estimates of its area coverage and travel time (Knee *et al.* 2006). The study used two models: HYDROMAP to calculate currents, and OILMAP to calculate oil spill trajectories and resulting oiled areas and travel times.

The OILMAP model was used to simulate spill trajectories and determine probabilities of areas being oiled and oil travel times for a instantaneous release of 40,000 gallons (151,000 liters) of electrical insulating oil at the ESP site in Nantucket Sound. This scenario (instantaneous release of entire tank contents) is highly unlikely and therefore conservative. The analysis estimates (not unexpectedly) that areas closest to the release site have the highest probability of surface water oiling and that lower probabilities generally spread radially outward from the site. While there is a >90% probability of oil impacting the shoreline somewhere in the action area within 12 hours, by the time the oil reaches the shoreline, 100 simulations of the model predicted that the probability of water surface oiling occurring within 10 days of a spill is generally reduced to 1-10%. Figure 4.6 in Knee *et al.* (2006, p. 21) shows that during the months of March-May, the model predicted a 1-10% probability of water surface oiling along the south shore of Cape Cod, eastern shore of Martha's Vineyard and western reaches of Nantucket; the Monomoy Islands are predicted to receive zero oiling. For the months of June-August, Figure 4.7 (p. 22) depicts both 1-10% and 10-20% oiling probability contours reaching the central Cape Cod (south coast) shoreline, with eastern Martha's Vineyard and western Nantucket being within the 1-10% contour. Nantucket Island is predicted to receive zero oiling. During September to November (Figure 4-8, p. 22), the model predicts 1-10% probability of water surface oiling along most of the Nantucket Sound shoreline, except for the Monomoy Islands and Nantucket, which have zero to very low probability of any surface water oiling.

The model projections of Knee *et al.* (2006) indicate the areas with highest probability of water surface oiling associated with an accidental release from the ESP in May through September occur in and near the project area, which coincides with roseate tern foraging habitat. As with collision risks, the peak potential exposure of the roseate tern population to any accidental oil spills occurs in May and then again in late August through mid-September. Piping plovers are also present in the action area from late March to September, but only a small subset of breeding sites are in areas with >10% probability of water surface oiling occurring within 10 days of a spill, and most sites known or suspected to receive concentrated plover use during migration (i.e., South Beach, Monomoy, Great Point, The Galls, Smiths Point and Esther's Island) have 1-10% or no probability of water surface oiling.

Although the oil spill trajectory modeling performed by Knee *et al.* (2006) implies some vulnerability to piping plovers along the Nantucket Sound shoreline and to the roseate terns using the waters of the Sound, consideration of these findings along with those of Etkin (2006) and the habitat use patterns and chronology of roseate terns and piping plovers support an overall assessment indicating very low risk to these bird species:

- 7% of all spills in Nantucket Sound in the next 30 years (two spills) will be attributable to Cape Wind;

- there is a 90% chance that these spills will be of 50 gallons or less;
- there is a 1% chance that these two spills will be >10,000 gallons;
- piping plovers are present in the action area less than half of the year and most of their habitat, including most known migratory stop-over concentration sites, is located in areas with <10% probability of contact with even the smallest amount of water surface oil dispersing from the ESP;
- roseate terns are present in the action area less than half of the year. Although they may forage close to sources of oil dispersing from a potential spill at the ESP, relatively small numbers of terns have been observed foraging on Horseshoe Shoal compared with other portions of Nantucket Sound.

In summary, there is a likelihood of at least two spills occurring during the life of the project as predicted by Etkin (2006) and the amount, time of occurrence and location are not predictable. The Service anticipates that should a spill occur as predicted by the model, there may be adverse effects to a few foraging roseate terns from oiling or displacement from their foraging habitat, but the likelihood of this occurrence is remote. Traveling roseate terns will not be adversely affected. The amount of oil from a spill most likely to occur (50 gallons or less) is unlikely to reach piping plover habitat. Accordingly, the Service finds that the effects to the roseate tern and piping plover from an oil spill associated with the proposed action are, for the purposes of this section 7 consultation, anticipated to be discountable.

In the unlikely event that an oil spill of a magnitude to affect piping plover beaches, or affect piping plover and roseate tern staging areas, and/or significantly affect roseate tern foraging habitat were to occur in Nantucket Sound, either project-related or independent of Cape Wind, the Service reserves the right under the Oil Pollution Act of 1990 (33 U.S.C. 2701 *et seq*.) to pursue damage claims for natural resources lost or injured.

We conclude our discussion by noting that MMS regulations at 30 CFR §254, "Oil Spill Response Requirements for Facilities Located Seaward of the Coastline", require owners/operators of oil handling, storage, or transportation facilities located seaward of the coastline to submit a spill response plan to MMS for approval prior to facility operation. In the event of a release of oil to the ocean, the applicant's employees, its contractors, and its responders would refer to the OSRP to ensure that the appropriate spill response actions are taken in a timely manner to minimize impacts to sensitive receptors and the environment. The OSRP (see Project Description - Conservation Measures) proposed in accordance with the Department of the Interior's regulations will minimize and avoid adverse impacts from possible oil spills to the maximum extent possible if appropriately implemented. Since the OSRP was unavailable to review prior to the completion of the BO, the Service will condition the BO on the development of measures that are protective of roseate terns and piping plovers.

The U.S. Coast Guard is the federal agency responsible for oil spill response in the coastal zone. The Massachusetts Department of Environmental Protection has developed a Geographic Response Plan (GRP) (http://grp.nukaresearch.com/CIgroup.htm), a component of which includes Nantucket Sound, the south shore of Cape Cod and the islands of Martha's Vineyard and Nantucket, to protect specific sensitive areas from impacts within 24 to 48 hours following a spill. However, adverse effects from the oil spill response on roseate terns and piping plovers must be balanced with the adverse effects from the oil spill. Therefore, there is a potential for unavoidable adverse effects, including harm and harassment, from oil spill response measures. These adverse effects will be addressed in a post-spill emergency consultation under section 7 of the ESA.

Short-term Effects from Pre- and Post-Construction, Routine Maintenance Activities and Decommissioning Activities

Roseate tern

The DEIS (Chapter 5) describes the various activities associated with the development, operation and decommissioning of the Cape Wind Project and the potential impacts on biological resources, including roseate terns and piping plovers (pages 5-172 through 5-175). These activities may occur during the construction, decommissioning and operation of the project and may incur short-term effects, primarily by disturbing foraging roseate terns or by the temporary displacement from preferred foraging habitat.

Activities that may temporarily affect roseate terns include increased vessel and helicopter traffic, and noise and vibrations from construction equipment. Roseate terns may be disturbed while foraging or temporarily displaced from foraging habitat as a result of increased boat traffic and sporadic helicopter flights during the construction, operation and decommissioning phases of the project. The DEIS also predicts that increased recreational fishing may occur in the Horseshoe Shoal project area if fish populations increase around the foundations of the wind turbines. Currently, there is considerable vessel traffic as well as helicopter and small plane traffic in Nantucket Sound. The additional increase in vessel and helicopter traffic from the Cape Wind Project is not anticipated to cause adverse effects that rise to the level of take. The disturbance and displacement of foraging roseate terns is anticipated to be negligible and of very short duration.

Since roseate terns are absent from Massachusetts waters each year from mid-September to mid-April, many of the activities described above will likely occur when roseate terns are not present in Nantucket Sound. During the breeding season, disturbance to foraging or traveling roseate terns from vibrations and noise resulting from construction of the WTGs, ESP and associated infrastructure is anticipated to be localized, of short duration, and result in insignificant and discountable effects. The area over which the noise and vibrations may occur is a fraction of the overall foraging habitat available to roseate terns in Nantucket Sound. Therefore, the Service has determined that these activities will not significantly alter essential roseate tern behaviors, and therefore will not cause harm or harassment, as those terms are defined by the ESA.

Piping plovers

The Service anticipates no adverse effects to breeding piping plovers from increased vessel or helicopter traffic or noise and vibration, since piping plovers are not found in the Horseshoe Shoal project area (with the possible exception of during migration or commuting), the center of most of this activity. Vessel and helicopter traffic will occur primarily in and over open water, well away from breeding and foraging piping plovers. Helicopters are expected to depart from local airports, none of which are located adjacent to breeding piping plovers, and fly directly to the Horseshoe Shoal project area.

Beneficial Effects to Roseate Terns from Bird Island Restoration

The preferred alternative recommended by the ACOE (2005) for the Bird Island restoration project is to restore and repair the existing stone revetment in its current location on the island and to use clean dredged material to raise the elevation of 0.64 acre of habitat landward of the revetment. The 0.64 acre of habitat to be filled is currently unsuitable for nesting by either common or roseate terns. Re-vegetation of the filled area and the placement of artificial nest boxes will further enhance the restored area's suitability for tern nesting. This will result in about 2.2 total acres of habitat suitable for tern nesting on the island for the projected 50-year life of the project.

Restoration of nesting habitat on Bird Island will benefit both common terns and roseate terns and is likely to result in measurable increases in the number of pairs of both species nesting there in the future and for the projected 50-year lifetime of the restored habitat (ACOE 2005). The following projection of the increase in the number of nesting pairs for both tern species that will be possible from the restoration work is based on the density of terns per unit of habitat present on the island now and the anticipated space available after an additional 0.64 acre is provided. Common tern pairs will have sufficient habitat to increase from about 1,800-1,900 pairs in 2007 to 2,890 pairs after the island is restored. Roseate tern pairs may increase from about 750 pairs in 2008 to over 1,150 pairs after the island is restored, an increase of 400 pairs. It is likely that contributions provided by Massachusetts from CWA's lease revenues will fill a critical gap (approx. 21%) in the funding needed for this project.

Since roseate terns nesting at Bird Island frequently exhibit higher productivity than roseate terns nesting at other Buzzards Bay colonies (an average of 1.17 chicks per pair at Bird Island, versus 1.03 at Ram and 0.92 at Penikese Islands during 2000-2007 (RTRT 2007), this aspect of the project is a substantial benefit to this endangered species. This benefit is realized even if roseate nesting pairs drawn to Bird Island after the habitat is restored are not "new" recruits to the regional population but rather are immigrants from Ram or Penikese Islands, because at Bird Island, they will likely breed more successfully and annually produce, on average, 12%-21% more young. Lastly, the Bird Island restoration project is critical in a more fundamental way. It is necessary to protect the habitat that exists there now. Without repairs to the revetment, the extent of suitable tern nesting habitat currently available on the island will continue to decline due to erosion and storm over wash, and the carrying capacity for both common and roseate terns will decline. For these reasons, should the restoration be undertaken, the Service concludes that all effects to roseate terns from the restoration of Bird Island will be beneficial. The design, scheduling and implementation of the Bird Island restoration plan is being closely coordinated among the Service, the ACOE and the State of Massachusetts.

The Bird Island restoration project is likely to have measurable beneficial effects for the roseate tern by preventing the further loss of existing essential nesting habitat, by creating additional suitable nesting habitat, and by increasing the carrying capacity of the island which is the most productive breeding site for the species in Buzzards Bay. The Service assumes that this project may not be completed and thus the benefits to terns may not accrue until some point after the

Cape Wind Project is constructed (and some take of terns may occur). However, the benefits will be long-term (50 years) and will persist even after the adverse effects of turbine mortality cease.

Population Level Implications of the Cape Wind Project on the Roseate Tern and Piping Plover

Effects of changes in vital rates (e.g., survival, fecundity) on the northeastern population of the roseate tern and the Atlantic Coast piping plover population vary with factors such as duration of the impact, age-class of the affected individuals, sex of the affected individuals (especially for roseates), or distribution of affected individuals in the breeding population (particularly for piping plovers). For the purposes of considering population level implications, the duration of the project and its impacts are presumed to continue for the indefinite future following construction. Although the projected life of the WTGs is 20 years, the Service recognizes that they may remain operational beyond that time span. Furthermore, while an ongoing activity with a short (e.g., five-year or even 10-year) life may have different population level implications than a 20-year project, our ability to reliably distinguish among population level effects due to varying duration diminishes markedly beyond the 20-year horizon. Therefore, it is appropriate (and errs on the side of the species) to assume that duration of the impact will continue for the indefinite future.

A Population Viability Analysis (PVA) is a method of estimating extinction probabilities using time series data of vital rates (e.g., survival, productivity, etc.) and/or population counts. Accordingly, a PVA may be useful in evaluating whether a range of potential additional mortalities (incidental take due to collisions with turbines) will have population level effects over time. Another value of PVAs is that sensitivity analyses can be evaluated to see what vital rate parameter is most important to changing the model's predicted outcome.

During this consultation, CWA's consultants made several attempts to model population level effects of hypothesized roseate tern mortality from the Cape Wind Project (Brault and Arnold 2004; Arnold 2007; PVA addendum of the BA), and did so once for the piping plover (Brault 2007). However, numerous reviews (e.g., Fieberg and Ellner 2000; Ralls *et al*. 2002; McCarthy *et al*. 2003) caution against over-reliance on the use of PVAs for assessing extinction risk, even when such models incorporate robust estimates of vital demographic rates, age structure, and habitat availability.

Roseate Tern

The roseate tern PVA model developed for CWA by Arnold (2007) is most sensitive to changes in adult survival rate followed by immature survival rate and population sex ratio. In it, the western North Atlantic roseate tern population exhibited a 95% probability of quasi-extinction (when the population of adult males drops below 500 individuals) within 50 years, even in the absence of any additional mortality due to the Cape Wind Project. Arnold (2007) estimated that the take of 1-5 males per year from collision mortality did not change the quasi-extinction probability, and the take of 10-15 males per year changed it 1%, from 95% to 96%. A take of 20 males per year increased the probability 2%, to 97%, and the take of 50 males per year brought it to 98% in 50 years. Essentially, the simulations run by Arnold (2007) suggest that the western

North Atlantic population of endangered roseate terns is likely to drop below a quasi-extinction threshold of 500 adult male breeders, with or without hypothesized mortality from Cape Wind.

The PVA addendum (BA) describes 12 new model runs and incorporates comments from Nisbet (2008), the Service and others, refined estimates of the number of young produced, adult survival and other vital rates, and current population estimates. It also incorporated "no take" scenarios, and different collision distribution estimates based on a 20- and 30-year project life. The PVA addendum also includes simulations run with and without the benefit of the Bird Island restoration project, which is estimated to provide additional nesting habitat for 300-400 nesting pairs of roseate terns. In summary, the 2008 PVA addendum model results deviate markedly from Arnold (2007). These alternative models find a quasi-extinction risk at 50 years of only 3.2% if there is no additional take from Cape Wind, and despite a range of collision probabilities, the quasi-extinction risk never exceeds 4%. More detail on the roseate tern PVA is provided on pages 5-56 to 5-60, and Appendix C of the BA.

Fieberg and Ellner (2000) found that even under optimistic assumptions (vital rate data are free from measurement error), the data requirements for estimates of extinction risk are overwhelming. Their analyses indicate further that estimates of extinction probabilities from time series data of vital rates or population counts will be unreliable measures of true extinction risk. They offer, however, that it may be possible to estimate short-term probability, but the predictive time period is only equal to 10-20% as long as the period over which the population has been monitored. Reliable population estimates for the endangered roseate tern in the western North Atlantic are only available for approximately 20 years, 1988-2007 (USFWS unpubl. data; RTRT 2007). Vital rates such as adult survival (Spendelow *et al.* 2008) are estimated from a similar time frame (19 years). Spendelow (pers. comm. 2008) notes that while some vital rates for the northeastern population of the roseate tern are reasonably well known, i.e., adult survival and annual productivity, other demographic parameters necessary for a PVA model, such as juvenile survival and recruitment of juvenile birds into the breeding population, are not well known. For roseate terns, juvenile survival and recruitment rates are not only less well known, they are also likely to be more variable than demographic parameters like adult survival and productivity (Monticelli *et al.* 2008). Because juvenile survival and recruitment rates can have a dramatic effect on population growth rate, predictions about long-term population trends when these parameters are not well studied must be made with extreme caution.

Given the above, as well as observed trends in abundance and productivity, the Service concludes that the above population viability analyses have limited value with regard to evaluating whether the Cape Wind Project is likely to reduce appreciably the likelihood of both the survival and recovery of the roseate tern in the western North Atlantic population by reducing its reproduction, numbers or distribution in the wild. Accordingly, the Service is not relying on the PVA to determine whether the proposed action will jeopardize the continued existence of the species. Instead, the Service is persuaded by the following summary which we find to be the best information available to assess population level effects of the project on the species:

First and foremost, the anticipated level of take, four to five roseate terns (and depending on the time of year and breeding status, possibly their dependent young), is very small in relation to the size of the northeast population. For example, the fall 2007 roseate population, including breeding age adults, their young of the year, and one- and two-year-old birds, is estimated at about 14,600 individuals (3,900 adults x 2, + approximately 4,000 young of the year, + 2,800 = 14,600). Even 10 birds taken per year is a very small fraction of a population that is estimated to number well over 10,000, even during periodic population lows (e.g., 2005, when only about 3,100 breeding pairs were recorded). In addition, for any given year, hatch-year roseate terns migrating to the wintering areas in Brazil will have virtually no exposure to collision mortality from the Cape Wind Project, as one-year-olds and most (~60%) of these birds will have no exposure as two-year-olds. Accordingly, in any given year, there will be two age classes of roseate terns that will have minimal exposure to collision mortality from the project or to other stochastic events that may occur during the fall staging period, such as Hurricane Bob in 1991.

In addition, the northeastern roseate tern population has exhibited resiliency during the past 20 years. Adult survival was lowered following a severe hurricane in August 1991, after which the breeding population at five key colonies south of Cape Cod (comprising more than 90% of the entire population) declined from 3,259 pairs in 1991 to 2,590 pairs in 1992 (Spendelow *et al.* 2008). The population then rebounded over the next several years to 3,623 pairs in 1998. Spendelow *et al.* (2008) report that for almost two decades, the annual survival rate of the roseate terns at these key colonies has been relatively stable in the range of 0.81-0.85. During this time, the population sustained the known loss of adult roseate terns to predators at several sites, including five at Bird Island in 1991 and five in 1993, 20 at Ram Island in 1997 and 20 in 2005, and 34 at Great Gull in 2004 and six in 2005 (Spendelow *et al.* 2008). Spendelow *et al.* (2008) concluded that despite the loss of these adult birds, the overall annual survival rate for the Long Island Sound and Massachusetts sub-population was little affected.

There are no recovery units identified for the endangered northeastern population, and terns from any of the colonies from Long Island, New York to Nova Scotia are equally at risk to collision mortality from the project. This is because the peak numbers of roseate terns were recorded in Nantucket Sound during the pre-breeding period of arrival in spring and the post-breeding period in late summer and fall, when terns from throughout the breeding range may have occurred in the Sound. As a result, the Service does not find that there will be any effects on the distribution of roseate terns as a result of the take of four to five individuals (and potentially their dependent young). Similarly, the loss of four to five terns will not appreciably reduce the likelihood of survival and recovery of the roseate tern because it is such a small fraction of the total northeastern population.

Piping Plover

The primary anticipated effect of the proposed project is mortality due to collisions during migration and, to a much lesser extent, collisions from non-migration flights by plovers breeding or fledging within the action area. Cape Wind Project-related mortality during migration could affect piping plovers breeding or fledging anywhere from Newfoundland south to the action area. However, it is reasonable to expect that piping plovers breeding on outer Cape Cod, Martha's Vineyard and Nantucket are the most likely to transit the project area while moving to or from their breeding grounds, while only a small portion of the population breeding at sites north of the project area will take migratory routes through Nantucket Sound. Furthermore, the potentially exposed population breeding in Canada is currently about one-third of the total number of pairs breeding in and north of the action area (even under a full-recovery scenario of 400 pairs, the Atlantic Canada population would be smaller than the New England population breeding in and north of the project). While we consider that quantitatively parsing the small number of projected collisions between breeding populations in New England and Atlantic Canada stretches the limits of available data, we estimate that markedly lower exposure will limit mortality of Canadian breeders to only one or two during the 20-year life of the project. Although it is plausible that young of the year on their southward migrations are likely the least skilled flyers with lowest avoidance rates, each adult has twice as many annual migration opportunities to transit the project area. The Service therefore projects that expected collision-induced mortality is about equally divided between adults and young of the year.

Unlike the situation for the roseate tern, previously published PVAs for the piping plover were available for use by consultants for CWA in examining the effects of potential mortality from the Cape Wind Project (Brault 2007). Brault's model assumptions and results provided to the Service appear appropriate, reasonable, and largely consistent with previous PVAs for piping plovers (Ryan *et al.* 1993; Melvin and Gibbs 1994; Plissner and Haig 2000; Wemmer *et al.* 2001; Larson *et al.* 2002; Calvert *et al*. 2006), as well as observed trends in abundance and productivity. The Service finds that Brault's PVA provides a useful framework for qualitative discussion of population level implications of the proposed project, but has not relied on its quantitative extinction risk estimates in formulating its conclusions in this BO.

The most consistent finding of all of the PVAs for piping plovers, including Brault (2007), is the sensitivity of extinction risk to even small declines in adult and/or juvenile survival rates. Changes in adult survival in Brault's model exert 2.25 times the effect on population growth than an equal change in productivity. In other words, a 1% reduction in annual adult survival would need to be offset by a 2.25% increase in fledglings produced, likely a formidable task given the baseline protection effort ongoing on the breeding grounds. Unsurprisingly, Brault's simulations indicated that the New England population is less sensitive to annual fatalities under intermediate growth (7%/year) than when the population is stationary. The smaller Atlantic Canada population is below-stationary growth, and Brault found that low numbers of fatalities (one to five birds/year) had large effects on extinction probabilities over 25-50 years.

Collision-induced mortality will exert the most significant population level effects if fatalities involve adults (birds >1 year old), especially from the Atlantic Canada breeding population (Brault 2007; Melvin and Gibbs 1994). This should not be construed as dismissing fatalities involving young of the year or plovers breeding in New England. Furthermore, relatively small numbers of sustained annual fatalities can exert noticeable effects on probabilities of population persistence, especially in a stationary or declining population. Relatively smaller demographic benefits from increasing piping plover productivity and the ongoing intensive recovery activities pose challenges to efforts to off-set mortalities that might be caused by the proposed project.

Notwithstanding potential demographic effects of small numbers of fatalities described above and the >20-year potential duration of exposure from the wind farm project, we return to the estimate of 0.44 collision-induced fatality per year, distributed across all age classes of piping plovers in the New England and Atlantic Canada recovery units, and emphasize the high likelihood that actual collision rates will be much lower. Demographic effect of any mortality is not trivial, but we find that mortality on the scale of one every two years will not rise to the level of an appreciable effect on probabilities of persistence of the Atlantic Coast piping plover population, which numbered 1,890 breeding pairs in 2007. The New England piping plover population has exceeded (or been within two pairs of) its 625 pair abundance goal for 10 years. While the Atlantic Canada breeding population remains more vulnerable, the Service finds that only a small proportion of the very limited mortality projected from the proposed project is likely to affect that portion of the population.

CUMULATIVE EFFECTS

Cumulative effects include the effects of future state, tribal, local or private actions that are reasonably certain to occur in the action area considered in this BO (50 CFR §402.02). Future federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

Human disturbance and domestic animals are generally not factors adversely affecting roseate tern breeding success, due to their nesting on offshore islands and the near daily presence of tern biologists that act as island stewards. However, what was once a summer tourist season is now extended into September and fall, and pedestrian and vehicular traffic on beaches, and unleashed dogs have been noted as having a disturbing effect on roseate terns (and piping plovers) present in the Plymouth area (S. Hecker pers. comm. 2008) and the Cape Cod area during the roseate pre-migratory, fall staging period (Trull *et al.* 1999). Indeed, Trull *et al*. (1999) report that staging flocks of terns were disturbed by human activities at 16 of the 20 sites they observed. Roseate terns that are repeatedly flushed from favored feeding, resting and overnight roosting areas may be unable to accumulate the necessary energy reserves needed for their long distance migration over water to their wintering grounds.

Commercial and recreational fishing activity may indirectly affect roseate terns if commercial herring harvest reduces the availability of juvenile herring used as food by terns. Commercial and recreational fishing may indirectly affect food availability for terns which often feed over schools of bluefish and striped bass, if harvest of those species diminishes their distribution or abundance in Nantucket Sound. There is a potential for an oil spill to occur unrelated to the Cape Wind Project that could adversely affect terns and plovers if it occurs at a time when the birds are present or at a location that impacts breeding, feeding, resting or roosting habitat.

Piping plover beaches in the action area are a mixture of publicly- and privately-owned land. On public beaches, recreational activity is expected to increase annually, as residential units are expanded and tourism of the area is promoted. Furthermore, ongoing disturbance and predation (resulting from human activities attracting predators to the area) are likely to continue throughout the action area. With the escalating numbers of beachgoers and their pets, disturbance to breeding piping plovers is expected to increase. Currently, most sites have effective management plans in place, although smaller, privately-owned or town-owned beaches lack specific management plans or agreements with qualified management entities such as MAS, and there is little-to-no enforcement of dog ordinances or leash laws. Therefore, it is expected that plover productivity may suffer some adverse effects due to increasing use of some beaches within the action area. Dredging and subsequent beach nourishment actions that may affect piping plovers will be addressed in future biological opinions.

Other wind energy projects (offshore, near shore and onshore) in the early planning stages in Massachusetts are not considered a cumulative effect because they are either outside of the action area or they will be the subject of future federal review under the ESA.

CONCLUSION

After reviewing the current status of the Atlantic Coast piping plover and the northeastern population of the roseate tern, the environmental baseline for the action area, and all effects of the proposed Cape Wind Project, it is the Service's biological opinion that the project is not likely to jeopardize the continued existence of these species. No critical habitat has been designated for the Atlantic Coast breeding ranges of these species; therefore, none will be affected.

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

In making this determination, the Service analyzed the potential impacts from collisions, habitat loss and disturbance, prey species attraction, barriers and displacement, increased predation, lighting, oil spills, pre- and post-construction activities, routine maintenance activities, and decommissioning activities. In analyzing these potential impacts, the Service determined that, in all cases except collisions, the effects were insignificant or discountable and would not result in take (mortality) of roseate terns and piping plovers. The Service estimates that the mortality of roseate terns due to collisions to be four to five roseate terns per year on average. Some of these individuals may be adult breeding birds with dependent young, and their loss would reduce the survival rate of those young. The Service also determines that mortality of piping plovers due to collisions is extremely unlikely to be more than 0.5 piping plover per year on average.

The Service assessed the population-level effects for roseate terns and piping plovers arising from the effects of the action, including the estimated collision mortality, and determined that these losses will not appreciably reduce the likelihood of survival and recovery of either species.

Roseate terns
No effect is anticipated on the distribution of roseate terns in the northeastern population, other than an increase in breeding pairs at Bird Island following the restoration of the habitat there (Beneficial Effects section). No effect on roseate tern breeding success is anticipated, because there is minimal use of the Horseshoe Shoal project area during the breeding season. There will be no measurable effects on roseate tern reproductive rates, although any collisions with the WTGs will foreclose future breeding of birds that are killed.

The population trend for roseate terns in the Northeast for the past two decades is characterized by fluctuations in the number of adult breeding pairs from about 3,000 – 4,300+ pairs. Short periods of population increase are followed by several years of decline. Although the loss of even a small number of birds per year due to the Cape Wind Project is a concern, particularly during periods of population decline, the take of even 10 individuals is a very small fraction of a population that is estimated to number well over 10,000, even during periodic population lows.

Piping Plovers

The Service's analysis finds that the greatest risk to piping plovers comes from potential collisions by migrating piping plovers that breed or fledge in or north of the action area (see Effects section). Although not entirely discountable, the best available information supports the Service's judgment that potential adverse effects of barriers, displacement, and collisions during flights not associated with migration, and oil spills are negligible, not rising to the level of "take". Thus, the primary potential effects of the proposed project will take the form of reductions in numbers of piping plovers incurred during northward and southward migration. The distribution of the potential mortality is among piping plovers that breed in the New England and Atlantic Canada recovery units. There will be no effects on reproductive rates, although it is axiomatic that any collisions will foreclose future breeding of the birds that are killed.

Although no piping plovers have been detected in the project area, this negative occurrence data is insufficient to discount potential risks. The Service has evaluated Hatch and Brault's deconstruction of collision risk to formulate its own reasoned upper bound estimate of <0.5 piping plover mortality per year. The Service recognizes more serious demographic implications if project-induced mortality involves adults and/or plovers breeding in Atlantic Canada. However, given that the total upper bound estimate is much less than one per year, the Service finds that it is not meaningful to attempt to parse quantitative mortality estimates by age class or breeding distribution. The Service cannot completely discount risks to the Atlantic Coast piping plover from the proposed project, but we emphasize that only 0.5 mortality per year is anticipated, even with conservative modeling. Notwithstanding the duration of potential exposure anticipated over 20 years or more, the Service determines that the proposed project will not appreciably reduce the likelihood of survival and recovery of the Atlantic Coast piping plover or populations breeding in the New England or Atlantic Canada recovery units.

INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and federal regulations pursuant to Section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without special exemption. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, collect, or attempt to engage in any such conduct. Harm is further defined to include significant habitat modification or degradation that results in death or injury to listed species by significantly impairing essential behavioral patterns including breeding, feeding, or sheltering. Harass is defined as intentional or negligent actions that create the likelihood of injury to listed species to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of Section 7(b)(4) and Section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the terms and conditions of an Incidental Take Statement.

Amount and Extent of Take

Roseate Tern

The Service estimates that on average, four to five roseate terns per year (80-100 terns over the 20-year life of the project) are likely to be taken (injured or killed) as a result of collisions with the WTGs on Horseshoe Shoal. If any of the four or five individuals are successful adult breeders with dependent young of the year, the survival rate of their young will be reduced, adding to the level of take. The Service bases this estimate on an independent review of the various collision modeling discussed previously and the modifications that reflect our full consideration of the best available scientific information and understanding of the species.

Piping Plover

The Service anticipates that a maximum of 10 piping plovers will be taken over the life of the Cape Wind Energy Project, based on our upper bound estimate of one piping plover collision every two years with the WTGs in the Horseshoe Shoal project area. As for roseate terns, the Service bases this estimate on an independent review of the various collision modeling discussed previously and includes our full consideration of the best available scientific information and understanding of the species. Because the formulation of mortality estimates is very complex, new empirical information demonstrating one or more of the following circumstances will constitute evidence that estimated take of piping plovers has been exceeded:

1. Annual flights across the project area exceed the total number of pairs breeding in and north of the action area. This is equivalent to approximately 18% of migration flights by adults and young of the year (pairs x 5.5).
2. More than 20% of flights occur at rotor height.
3. Avoidance rates <0.95.

Effect Of The Take

In this BO, the Service determined that the level of take is not likely to have jeopardized the continued existence of the piping plover and roseate tern. Furthermore, the Service estimates that implementation of the Bird Island restoration project will offset any potential roseate tern mortality that may occur from the Cape Wind Project.

REASONABLE AND PRUDENT MEASURES

Pursuant to Section 7(b)(4) of the Endangered Species Act, the Service finds the following reasonable and prudent measures are necessary and appropriate to minimize incidental take of roseate terns and piping plovers. In order to be exempt from prohibitions of Section 9 of the

ESA, MMS and CWA must comply with the following terms and conditions which implement the reasonable and prudent measures and outline reporting/monitoring requirements. These terms and conditions are non-discretionary. The term "reasonable and prudent measures" is defined by the Service's ESA implementing regulations (50 CFR §402.02) to mean "those actions that the [Service] believes are necessary to minimize the impacts of take, i.e., amount or extent, of incidental take". The Service's Section 7 Consultation Handbook (March 1998) further explains that measures are considered reasonable and prudent when they are consistent with the proposed action's basic design, location, scope, duration, or timing of the project" [Handbook at 4-50 (illustrations excluded)]. The Handbook also states that "the test of reasonableness is whether the proposed measure would cause more than a minor change to the project" and that RPMs can include only actions that occur within the action area.

1. Pre- and post-construction monitoring to assess the effects and incidental take of the Cape Wind Project
The MMS and CWA Monitoring Framework is a preliminary framework of methodologies for pre- and post-construction monitoring of the potential impacts of the Cape Wind Project on roseate terns and piping plovers. MMS, CWA and the Service will coordinate in the development of more detailed protocols to determine the extent of roseate tern and piping plover presence in the project area, the effects of the WTGs on roseate tern foraging and other use of Horseshoe Shoal and/or the level of incidental take as a result of the project.

2. Oil Spill Response Plan
Although MMS requires an oil spill response plan in the event of a spill related to the Cape Wind Project, specific response measures shall be identified for roseate tern and piping plover habitat in order to avoid or minimize take. Some adverse effects and possible take (primarily in the form of harm or harassment) may be unavoidable during an emergency response. These effects will be addressed in a post-spill emergency consultation as described in the BO.

3. Review of pre- and post-construction monitoring activities, perching deterrents and operational adjustments.
The Service, MMS and CWA will review the efficiency and efficacy of pre- and post-construction monitoring activities, and the implementation of perching deterrents to determine their effectiveness and/or make adjustments as needed, in order to continue or enhance avoidance and minimization of take.

4. Reporting requirements
Post-construction monitoring may not be able to sufficiently document take of roseate terns and piping plovers resulting from collisions with WTGs or the ESP. Nevertheless, MMA and CWA must report roseate tern and piping plover injury or mortality associated with the Cape Wind Project to the Service within 24 hours.

Operational adjustments
The Service also considered as a reasonable and prudent measure, an operational adjustment to the wind facility that would require the temporary and seasonal shut down of the WTGs through the feathering of the rotors. Feathering of the rotors causes them to face the wind and stop spinning, and would reduce the risk of collision by roseate terns and, to a limited extent,

migrating piping plovers transiting the Horseshoe Shoal project area. Although the Service considered that result in this "operational adjustment" would be based on weather and day light parameters that reduce visibility, and would be limited in time to seasons when plovers and peak numbers of roseate terns are expected to be present (a few weeks in early to mid-May and a few weeks in late August to mid-September), it was determined by MMS and CWA (J. Lewandowski, MMS electronic correspondence including Bennett *in litt*. as attachment, November 20, 2008) to **not** be reasonable and prudent based on the following:

The operational adjustment (shut down of turbine rotors to a neutral position) is not reasonable because it does not meet the RPM regulatory definition as a "reasonable measure" as it modifies the scope of the project in a manner that is adverse to the project's stated purpose and need, that is to make a substantial contribution to enhancing the region's electrical reliability and achieving the renewable energy requirements under the Massachusetts and regional renewable portfolio standards (DEIS 2008 at E-1). MMS considers that this may involve more than a "minor change" (50 C.F.R. § 402.14(i)(2).

MMS has also determined that the RPM is not reasonable because the uncertainty regarding the project's ability to generate electricity during the two time frames (late April to mid-May and late August to mid-September) reduces the project's predicted potential electrical output in a significant enough way to have a deleterious affect on anticipated revenues, financing and power purchasing agreements.

Furthermore, MMS indicates that the proposed timeframes for the operational adjustment, although limited by season, visibility and time of day, constitute peak period hours, when the energy supplied to the ISO New England (the regional transmission organization) has greater market value (see DEIS 2008 at 3-32). Therefore, the RPM may not be prudent because the economic cost makes this measure not feasible for project proponents to implement.

Terms and Conditions

In order to be exempt from the prohibitions of section 9 of the ESA, MMS and CWA must comply with the following terms and conditions to implement the reasonable and prudent measures described above. These terms and conditions are non-discretionary.

1. Monitoring:
    a. MMS, CWA and the Service will coordinate in the development of specific pre- and post-construction monitoring protocols discussed in the Framework for the Avian and Bat Monitoring Framework for the Cape Wind Proposed Offshore Wind Facility.

    b. Prior to implementation, monitoring protocols should be peer-reviewed, including at least one European scientist currently conducting similar monitoring efforts at off-shore wind projects. Peer review could allow data collection and analysis to be comparable with other ongoing off-shore monitoring efforts.

c. To the greatest extent practicable, all protocols shall incorporate methods to assess detectability and sufficiency of negative data. Examples might include double observer protocols for aerial and boat surveys, testing the range of radio transmitter reception and effects of flight altitude using transmitters affixed to small planes or boats, and use of recorded play-backs to evaluate acoustic monitoring in the project area.

d. Components of the Monitoring Framework, such as radio telemetry, that entail take (i.e., capture, some risk of injury) to roseate terns and/or piping plovers will be contingent on recovery permits under section 10(a)(1)(A) of the ESA.

e. The monitoring framework shall be adaptable and incorporate to the maximum extent practicable, new remote sensing technologies or other new technologies that will enhance data collection on avian use and collision risk in the project area.

2. Oil Spill Response Plan
   a. MMS and CWA will coordinate with the Service to develop an oil spill response plan (or section within CWA's proposed OSRP) that specifically addresses response activities that could occur in roseate tern and piping plover habitat (including breeding, foraging and resting habitat).

3. Review of pre- and post-construction monitoring activities, perching deterrents and operational adjustments.
   a. The Service, MMS and CWA will coordinate annually (or as needed) to review the results of pre- and post-construction monitoring efforts, and monitor the effectiveness of operational adjustments and perching deterrents.

   b. Based on the results of the reviews, adjustments to monitoring protocols and redesign of perching deterrents may be required. If operational adjustments are determined to be unnecessary, they may be discontinued.

4. Reporting requirements
   a. MMS and/or CWA shall report within 24 hours any roseate tern or piping plover mortality attributable to the Cape Wind Project to the Supervisor, New England Field Office, 70 Commercial St., Suite 300, Concord, NH 03301 or telephone 603-223-2541.

   b. MMS or CWA shall provide an annual summary report of pre- and post-constructing monitoring efforts to the Supervisor, New England Field Office, 70 Commercial St., Suite 300, Concord, NH 03301 or telephone 603-223-2541 no later than December 15 of each year.

CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the ESA directs federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. The following conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or proposed critical habitat, to help implement recovery plans, or to develop information.

1. MMS should develop and test technology to facilitate *pre-project* assessments of roseate tern and piping plover abundance in off-shore areas proposed for future WTG projects. Each additional project along these species' Atlantic Coast migration routes has potential to compound mortality risks from collisions, and extinction risks for both of these species are highly sensitive to increased mortality. Serious effects on piping plover extinction risk at relatively low sustained kill/year thresholds modeled by Brault (2007) underscore the importance of responding to this issue before off-shore wind projects proliferate, especially since piping plovers from the relatively precarious Atlantic Canada recovery unit are potentially exposed to every U.S. Atlantic off-shore wind generation proposal. The current dearth of information reflects the absence (until the advent of off-shore wind energy proposals) of perceived off-shore threats to these species.

   State-of-the-art radar technology is unable to identify birds to species. Off-shore visual surveys (boat and aerial) have only limited ability to identify flying terns to species and are virtually incapable of identifying small shorebirds such as piping plovers in flight. Acoustic monitoring and telemetry in the Cape Wind post-project Monitoring Framework have potential to refine assessments of risks to these species, but it is essential that planning and consultations for future projects have the benefit of pre-project assessments. The Service urges MMS to foster development of cost-effective means of stationing acoustic and telemetry receivers in off-shore areas where future wind energy projects may be proposed. Advances in other technology (e.g., harmonic radar) that might provide important pertinent information on the off-shore activities of these species (e.g., time of year, time of day, altitude, behaviors, effects of weather) are also needed. Every effort should be made to develop protocols for each technology that support assessments of detectability and sufficiency of negative data.

2. MMS should encourage CWA and its partners to fully implement all measures identified as state-required "compensatory mitigation" in section 8.2.1 of the BA. As a result of the Massachusetts Environmental Policy Act (MEPA) Certificate, CWA was required to establish a $10 million fund to compensate for unavoidable impacts to affected wildlife and habitat. The fund would come from anticipated lease revenues generated by Cape Wind. A portion of this fund is proposed for use in the following conservation measures:

   a. Predator Management: Predation on beach nesting piping plovers and terns has been a significant factor for reduced productivity at a number of Massachusetts beaches. Professional management of predators at carefully selected sites has yielded demonstrable benefits for piping plovers and terns in recent years. Successful implementation of predator management on select plover sites in Massachusetts should increase productivity and offset potential losses as a result of plover or tern collisions with WTGs (page 8-11 of the BA).

b. Population Monitoring, Site Protection and Management (Breeding Season): MADFW proposes to use mitigation funds to sustain and/or augment current statewide efforts to monitor and manage piping plovers and terns in Massachusetts. Priority locations where additional monitoring and protection is needed are identified on page 8-11 of the BA.

c. Identification and Protection of Piping Plover and Tern Post-Breeding Staging and Migration Areas: MADFW proposes to hire seasonal staff to identify post-breeding staging and migratory stopover areas for terns and piping plovers, and to develop and implement site management plans for high priority areas. Identification and evaluation of conservation needs at piping plover migration stop-over habitats is a heretofore rarely implemented and potentially beneficial action.

d. Coastal Waterbird Conservation Assistant: A permanent Coastal Waterbird Conservation Assistant may be hired by MADFW to oversee statewide conservation efforts for piping plovers and roseate terns. Augmentation of MADFW staff will enhance training of and coordination with local shorebird monitors and improve responsiveness to requests for assistance with the many unique and sometimes devastating problems that arise during a very short annual breeding window.

3. MMS should find mechanisms to implement the conservation measures for piping plovers discussed in section 8.2.1 of the BA in Atlantic Canada, Maine, and New Hampshire. It is the nature of off-shore wind generation facilities that they can affect birds breeding at locations far distant from the project area. Furthermore, any mortality of plovers that breed at higher latitudes will result in more serious demographic effects.

In order for the Service to be kept informed of actions minimizing or avoiding adverse effects or benefiting listed species or their habitats, the Service requests notification of the implementation of any conservation recommendations.

REINITIATION NOTICE

This concludes formal consultation regarding the MMS proposal to lease a portion of the outer continental shelf in Nantucket Sound, Massachusetts to Cape Wind Associates LLC, for development of a commercial wind energy facility. As provided in 50 CFR §402.16, reinitiation of formal consultation is required where discretionary federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded; (2) new information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not considered in this opinion; (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in this opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action. In instances where the amount or extent of incidental take is exceeded, any operations causing such take must cease pending reinitiation.

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Appendix 1. Consultation History

November 17, 2005 – Initial meeting between Minerals Management Service staff and U.S. Fish and Wildlife Service (FWS) Regional Office and New England Field Office staff to discuss environmental issues relative to the proposed Cape Wind project. The MMS briefed FWS on the upcoming NEPA review process and ESA consultation between the FWS and MMS.

December 27, 2005 – Electronic correspondence from MMS to the FWS requesting initiation of informal consultation on the Cape Wind project and FWS information needs required for consultation.

January 5, 2006. Conference call between MMS and FWS discussing the need to redo the Draft EIS and the desirability of conducting additional studies on terns and plovers.

February 22, 2006. Email from MMS to the FWS and others regarding establishing a date and time for a conference call to discuss avian studies in Nantucket Sound.

March 6, 2006. Notes from conference call between the MMS, FWS, U.S. Geological Survey USGS, Massachusetts Audubon Society (MAS), Cape Wind Associates (CWA) and consultants. Stated purpose of call was to close communication gaps, to discuss present body of avian studies conducted by Mass Audubon and the CWA, and work on-going on the population and viability analyses (PVA) for plovers and terns.

April 7, 2006. Protocol for Marine Radar Surveys of Birds. Geo-marine, Inc and ESS Group, Inc. for CWA.

April 12, 2006. Conference call between the MMS, FWS, USGS, CWA and consultants to discuss potential research for understanding plover and tern migratory routes within the project area.

April 27, 2006. Conference call between the MMS, FWS, Massachusetts Department of Fish and Wildlife (MADFW), CWA and consultants to discuss the roseate tern PVA.

April 27, 2006. Email from Jennifer Arnold USGS providing follow-up information and suggested changes for the roseate tern PVA.

June 26, 2006. Conference call between the MMS, FWS, MADFW, CWA and consultants regarding revision of the PVA for piping plovers.

July 10, 2006. Email from Anne Hecht FWS to MMS providing a research paper on movements, habitat use and survival rates of piping plovers and discussing need for research on piping plover migration.

December 6, 2006. Email correspondence from MMS to the FWS and others to coordinate a meeting to assimilate existing data on piping plover and roseate tern presence at two proposed off-shore wind energy projects and to discuss potential research avenues to assess plover and/or tern presence in the proposed sites.

January 5, 2007. Letter from FWS to ESS Group, Inc., providing an updated endangered species list for the HSS project area and transmission line corridor. Updated species distribution information was also provided.

January 8, 2007. Email correspondence from MMS to FWS and USGS regarding the purpose for a meeting on January 30, 2007.

January 30, 2007. Agenda, various notes, and copies of power point slides presented at the January 30th meeting at MMS offices in Herndon, VA., at which FWS, USGS, ESS Group, MADFW, Virginia Tech and several consultants were in attendance.

February 1, 2007. Email correspondence from USGS regarding the August 28-31 tern observations made by ESS Group near Monomoy Island.

February 15, 2007. Cape Wind Energy Project Final EIR/Development of Regional Impact.

February 16, 2007. Email correspondence from FWS to Woodlot Alternatives (consultant for MMS), MADFW and others, regarding four published PVAs for piping plovers for MMS and Woodlot review.

February 16, 2007. Woodlot Alternatives, summary of topics for Cape Wind biological assessment with notes from a conference call between MMS, FWS and Woodlot.

March 12, 2007. Email correspondence from MMS to FWS and Woodlot Alternatives, regarding a March 15, 2007 conference call to discuss issues relevant to the MMS'ss biological assessment.

March 15, 2007. Email correspondence from Woodlot Alternatives to MMS and FWS with several attachments, including the most recent versions of the PVAs and collision risk models and a list of the literature cited to date for the MMS draft biological assessment.

March 15, 2007. Notes from a meeting with Woodlot Alternatives, MMS and FWS staff regarding preparation of the biological assessment, particularly with respect to assumptions and inputs to roseate tern PVA.

March 16, 2007. Email correspondence from FWS to MMS reconfirming species list.

March 20, 2007. Email correspondence from Woodlot Alternatives to FWS and MMS containing a copy of the recommendations from MADFW regarding model assumptions for piping plovers, as requested by FWS on the March 15 call and again by email correspondence from FWS on March 19.

March 21, 2007. I. Nisbet comments on the Final EIR for the Cape Wind Project: Avian Impacts.

March 21, 2007. MADFW comments on tern sections of Cape Wind Final EIR.

March 22, 2007. Letter from MADFW to Secretary, Executive Office of Energy and Environmental Affairs' comments on the Cape Wind FEIR.

March 22, 2007. Multiple email correspondence from FWS to MMS staff and FWS staff about interagency communications regarding the Cape Wind project, the Long Island project and the programmatic EIS. Second email from FWS discussing designation of a non-federal representative, suggesting that the candidate species, red knot, be included in the species list for the Cape Wind project area and responding to questions from Woodlot Alternatives on the piping plover.

March 23, 2007. Email correspondence from FWS to Woodlot Alternatives in reply to a March 16 Woodlot Alternatives to FWS staff discussing information sources and assumptions for the PVAs and collision risk assessments.

March 24, 2007. Follow-up email correspondence from FWS to MMS with final feedback related to March 22, 2007 correspondences noted above.

March 29, 2007. The Commonwealth of Massachusetts, Certificate of the Secretary of Environmental Affairs on the Final EIR. Includes identification of ($780,000) from CWA lease revenues for the Bird Island restoration.

April 9, 2008. Email exchanges between MMS and FWS staff regarding interagency communications and ESA consultations for wind power.

April 10, 2007. Email correspondence from FWS to MMS providing follow-up answers to 15 questions from MMS regarding model inputs to the collision risk assessment to roseate terns. An April 10, 2007 email from USGS is appended.

April 2007. Roseate Tern Recovery Team c/o C. Mostello (MADFW), Final figures for roseate tern abundance, Northeast U.S. 1988-2006. Includes a table showing census estimates for all breeding colonies in the U.S.

April 27, 2007. Conference call between consultant to ESS Group, Inc., and roseate tern recovery team members, Ian Nisbet and Jeff Spendelow, regarding inputs and assumptions to the roseate tern PVA.

April 27, 2007. USGS electronic correspondence providing a summary of the April 27 roseate tern PVA conference call.

May 31, 2007. FWS electronic correspondence to MMS outlining preliminary questions regarding Appendices 3.6-I and 3.6-K in the Cape Wind FEIR as discussion points for the informal Section 7 consultation on piping plovers.

June 2007. Cape Wind Associates response to data requested by MMS regarding inputs to collision risk assessment models for plovers and terns.

June 20, 2007. Conference call between several MMS staff, FWS staff, CWA and consultants. Discussions focused on bird collision risk assessments derived from European projects and studies, and their application to the Cape Wind project and a bird monitoring plan.

August 7, 2007. Email from MMS to FWS staff coordinating a meeting date in Boston in September, 2007.

August 9, 2007. Telephone correspondence between FWS (red knot recovery team leader) and MMS regarding current status of red knot and potential for occurrence in the proposed action area.

September 13, 2007. Meeting in Boston between several MMS staff, several FWS staff, MADFW, consultants and CWA. Working MMS draft of existing recommendations for mitigation, monitoring and impact avoidance for birds and wind farms.

September 14, 2007. Email and fax correspondences from FWS to MMS, providing section 7 ESA guidance materials as follow-up to September 13 meeting.

October 9, 2007. Letter from MMS to FWS, requesting concurrence on the list of threatened and endangered species and critical habitat to be considered in the consultation for the Cape Wind project.

October 22, 2007. Email correspondence from MMS to FWS with update on progress of draft MMS biological assessment and request for one more piping plover call and a separate roseate tern call specifically addressing monitoring/mitigation issues.

November 12, 2007. Email from MMS to FWS staff, MADFW, USGS and Roseate tern recovery team member and others, regarding a conference call meeting to discuss the first draft of avian monitoring, mitigation and reporting requirements for the Cape Wind project.

November 12, 2007. Email from MMS to FWS staff, MADFW and piping plover recovery team member and others, regarding a November 14 conference call meeting to discuss the first draft of avian monitoring, mitigation and reporting requirements for the Cape Wind project.

November 12, 2007. Email from MMS to FWS staff, MADFW and piping plover recovery team member and others, regarding a November 20 conference call meeting to discuss the first draft of avian monitoring, mitigation and reporting requirements for the Cape Wind project.

November 16, 2007. Letter from FWS to MMS responding to concurrence request for the Cape Wind species list.

November 26, 2007. Outcome (meeting summary notes) from MMS Technical Meeting on ESA-Listed Bird Species and the Cape Wind Proposal held September 13, 2007 in Boston.

December 3, 2007. Email correspondence from FWS to MMS notifying of new article in Journal of Wildlife Management (Kuvlesky *et al.*, 2007) on Wind Energy Development and Wildlife Conservation.

December 4, 2007. Email correspondence from MMS to FWS, USGS and I. Nisbet with attached preliminary draft biological assessment seeking an initial review on completeness of information. Several additional emails between recipients of draft biological assessment during mid-December providing preliminary comments or questions.

January 2008. MMS provides latest version of draft MMS biological assessment for more extensive FWS review and comment.

February 5, 2008. FWS provided comments on the draft BA, comments of an anonymous reviewer and those of Dr. Ian Nisbet, roseate tern recovery team member.

April 21, 2008. Letter from FWS to MMS conveying FWS comments on the January 2008 draft EIS.

May 19, 2008. Biological Assessment and letter requesting initiation of formal section 7 consultation from MMS to FWS.

June 9-11, 2008. Several email correspondences between FWS and MMS on whether to post the biological assessment on line. Assessment was ultimately posted on the MMS website on June 16.

June 17, 2008. Letter from FWS to MMS acknowledging receipt of request to initiate formal section 7 consultation and anticipating completion date of October 2, 2008 for the consultation.

June 23, 2008. Notes from conference call between MMS staff, FWS staff and consultants. Discussion focused on on-going coordination MMS is having with multiple federal regulatory agencies [FAA, USCG, Army Corps of Engineers (ACOE), etc.] Discussion also addressed the FWS's comments on the DEIS and the research proposals for additional bird studies that MMS is reviewing.

July 17, 2008. Email from FWS to MMS and others, discussing FWS availability to assist MMS with bird study research proposals MMS has received.

July 21 and 23, 2008. Emails and notes to files to and from the FWS and MMS regarding FWS availability to participate in a panel to review and revise the avian monitoring plan.

August 15, 2008. Email correspondence from FWS to MMS requesting additional monitoring and mitigation information and confirmation of proposed action components for ongoing formal section 7 consultation.

August 18, 2008. Letter from FWS to MMS requesting extension of consultation period to October 16, 2008, with a complete biological opinion issued by December 1, 2008.

August 19 – 25, 2008. Conference call and email correspondence between MMS and FWS discussing development of draft avian monitoring plan.

August 20, 2008. Multiple email correspondence between FWS to ACOE to discuss Bird Island restoration project.

August 27, 2008. Conference call between MMS, FWS and consultants discussing post-construction monitoring options and methodologies.

September 3 4, 2008. Multiple email correspondence between MMS and FWS discussing submission of draft avian monitoring plan and timeline for completing ESA consultation and review of draft avian monitoring plan.

September 5, 2008. MMS electronically submitted draft avian monitoring plan to FWS for review and comment.

September 10, 2008. FWS requests missing draft avian monitoring plan references and methodology matrix from MMS.

September 11, 2008. Multiple email correspondence between MMS and FWS discussing missing references and timeline for ESA consultation.

September 12, 2008. FWS electronically submitted comments on the MMS draft avian monitoring plan.

September 15, 2008. Conference call between MMS, FWS and consultants to discuss draft avian monitoring plan.

September 19, 2008. MMS electronically submitted the final Framework for the Avian and Bat Monitoring Plan for the Cape Wind Proposed Offshore Wind Facility.

September 30, 2008. Letter from FWS (electronic) to MMS regarding the presence of the federally-threatened Northeastern beach tiger beetle within the periphery of the proposed Cape Wind project action area.

October 1, 2008. Multiple electronic correspondence between MMS and FWS discussing wind cut-in speed of the Cape Wind project wind generating turbines.

October 3 and 6, 2008. Multiple electronic correspondence between MMS and FWS discussing a potential Reasonable and Prudent Measure for the FWS biological opinion.

October 6, 2008. Letter from MMS to FWS responding to the Service's request for an extension on the formal consultation.

October 8, 2008. Telephone communication with MMS, FWS and representatives of CWA discussing draft RPMs.

October 9, 2008. Letter from MMS to FWS amending the Biological Assessment to include an effects analysis for the northeastern beach tiger beetle.

October 16, 2008. Additional telephone communication with MMS, FWS and representatives of CWA discussing draft RPMs.

October 21, 2008. Telephone conference with MMS, representatives of CWA and FWS to discuss availability of weather data for Nantucket Sound to clarify how often low visibility conditions (e.g., fog) occur on the sound during periods when tern numbers are highest.

October 22, 2008. Email transmittal from FWS to ESS Group, MMS and CWA requesting clarification of selected parameters of the roseate tern collision risk model.

October 23, 2008. Email correspondence from Jeremy Hatch, U. Mass Boston (ESS Group) to FWS, providing clarification on the Band *et al.* (2006) collision model adapted for use in the BA.

October 29, 2008. Telephone and email correspondence between FWS, and Jeff Spendelow, USGS, discussing the strengths and weaknesses of the roseate tern population viability analyses, and recent papers published in the scientific literature useful to the Service's on-going consultation with MMS.

October 31, 2008. Transmittal of FWS draft Biological Opinion, with appendices, to MMS.

November 6, 2008. Electronic submittal of MMS comments on the draft Biological Opinion to the FWS.

November 7, 2008. One additional comment on the draft Biological Opinion electronically submitted by MMS to FWS.

November 10, 2008. Conference call between MMS, FWS, CWA and consultants to discuss comments provided by MMS and CWA on the draft Biological Opinion.

November 12, 2008 Electronic submittal from MMS to FWS outlining MMS timeline and requesting FWS final Biological Opinion by November 21, 2008.

November 12, 2008. Electronic submittals relative to the Biological Opinion's Reasonable and Prudent Measures and consultation history.

November 13, 2008. Telephone call between MMS and FWS discussing additional information needs relative to the Reasonable and Prudent Measures.

November 18, 2008. Electronic transmittal from MMS to FWS relaying additional comments on the draft biological opinion regarding the collision model and oil spill risk analysis.

November 20, 2008. Electronic transmittal from MMS to FWS relaying MMS's response to a proposed Reasonable and Prudent Measure.

NOAA Fisheries Biological Opinion

NATIONAL MARINE FISHERIES SERVICE
ENDANGERED SPECIES ACT SECTION 7 CONSULTATION
BIOLOGICAL OPINION

Agency: Minerals Management Service, US Department of the Interior

Activity: Cape Wind Energy Project
F/NER/2008/03508

Conducted by: National Marine Fisheries Service
Northeast Regional Office

Date Issued: 11/13/08

Approved by: [signature] for Patricia Kurkul

This constitutes the biological opinion (Opinion) of NOAA's National Marine Fisheries Service (NMFS) on the effects of the Minerals Management Service's (MMS) proposed approval of an application by Cape Wind Associates, LLC for a lease, easement or right-of-way to construct, operate and decommission a wind energy project on Horseshoe Shoal in federal waters of Nantucket Sound, Massachusetts on threatened and endangered species in accordance with section 7 of the Endangered Species Act (ESA) of 1973, as amended (16 U.S.C. 1531 et seq.). The MMS's authority to approve, deny, or modify the proposed action derives from the Outer Continental Shelf Lands Act (43 U.S.C. § 1331 et seq.) as amended by the Energy Policy Act of 2005 (P.L. 109-58). This Opinion is based on information provided in MMS's Cape Wind Energy Project Nantucket Sound Biological Assessment (BA), the Draft Environmental Impact Statement for the project (DEIS), correspondence with the MMS, and other sources of information. A complete administrative record of this consultation will be kept on file at the NMFS Northeast Regional Office. Formal consultation was initiated on May 22, 2008.

CONSULTATION HISTORY

Cape Wind Associates (Cape Wind) began preliminary work on siting and designing a wind energy project in 2000. In November 2001, Cape Wind sought a permit from the US Army Corps of Engineers (ACOE) to construct and operate a wind-powered electrical generating facility on Horseshoe Shoal in Nantucket Sound, Massachusetts. Informal consultation between NMFS and the ACOE continued throughout 2001-2004. A DEIS was ultimately published by the ACOE in 2004. The DEIS included a draft BA dated May 2004. NMFS provided comments on the DEIS and indicated to the ACOE that consultation pursuant to Section 7 of the ESA would be necessary for the proposed project.

In August 2005, the Energy Policy Act of 2005 was passed which gave the Department of the Interior's (DOI) MMS authority for issuing leases, easements or rights-of-way for alternative energy projects on the Outer Continental Shelf (OCS). At this time, purview over the Cape Wind proposal was transferred from the ACOE to MMS. MMS then determined that a new DEIS was required given its different federal approval processes and requirements.

MMS and NMFS began discussing consultation requirements in January 2006. Throughout 2006 and 2007 NMFS provided technical assistance to MMS as they drafted a new DEIS and draft BA. The MMS published a DEIS on January 18, 2008. MMS provide NMFS with a final BA and request for formal consultation in a letter dated May 19, 2008. The date that the May 19 letter was received (May 22, 2008) serves as the date consultation was initiated.

DESCRIPTION OF THE PROPOSED ACTION

The proposed action entails the construction of a wind energy facility (wind park) consisting of 130 wind turbine generators (WTG) located on Horseshoe Shoal in Nantucket Sound, Massachusetts (see Figure 1 for map of project area). The northernmost WTGs would be located approximately 3.8 miles from the dry rock feature offshore near Bishop and Clerks and approximately 5.2 miles from Point Gammon on the mainland; the southernmost part of the Wind Park would be approximately 11 miles from Nantucket Island (Great Point) and the westernmost WTG would be approximately 5.5 miles from the island of Martha's Vineyard (Cape Poge). Installation of the WTGs will comprise of four activities: (1) installation of the foundation monopiles; (2) erection of the wind turbine generators; (3) installation of the submarine cables; and, (4) installation of the scour protection. The 130 WTGs and the ESP would occupy a total of 0.67 acres of submerged land. Scour protection for the WTGs would cover an additional 2.5 - 47.5acres, depending on whether scour mats or rock armoring was used. During installation of the WTGs, ESP, cable and scour protection, it is anticipated that approximately 867 acres would be temporarily disturbed.

Pre-Construction Geophysical and Geological Survey

Prior to construction, a supplemental geotechnical program may be conducted. Additionally, the applicant may conduct a high resolution marine shallow hazards survey. The geotechnical and geophysical (G&G) field investigations would be designed to collect sufficient information, coupled with previous site-specific field data, to further characterize the surface and subsurface geological conditions within the vertical and horizontal areas of potential physical effects (APPEs), in preparation for final design and construction. These areas include the offshore construction footprints and associated work areas for all facility components, including the WTGs, the ESP, the inner array cables and the 115kV transmission cables to shore. The supplemental geotechnical program would further analyze sediments and physical conditions within the proposed action APPEs, for use in final foundation design and to develop site-specific BMPs for constructability.

A one-time high resolution geophysical survey may be conducted to assess seafloor and/or shallow subsurface conditions. A survey vessel would be used to run rectilinear geophysical tracklines. If conducted, the survey is expected to take approximately 36 hours to complete. The applicant will use an airgun, boomer, sparker or chirper to obtain the necessary geophysical data. The survey equipment would be towed behind the survey vessel and the survey vessel will travel approximately 3-3.5 knots during the survey. As required by MMS, endangered species observers will be present during the survey and will maintain a 500 meter exclusion zone. Additional requirements for operation during the survey are outlined in Appendix A and include a ramp up procedure, continuous visual monitoring of the exclusion zone, and shut down requirements should a listed species enter the exclusion zone.

The supplemental geotechnical program involves the use of coring and boring equipment to collect sediment samples for laboratory analyses, which would disturb the seafloor in small discrete locations. Vibracores would be taken along the proposed submarine cable route with approximately 2 vibracores per mile and along the inner array cable route with 1 vibracore approximately every 3.5 miles. The vibracores would be advanced from a small (less than 45 feet) gasoline powered vessel. Approximately 50 vibracores are currently planned, with up to 6 collected during each field day. The diameter of the core barrel is approximately 4 inches and the cores are advanced up to a maximum of 15 feet. In addition to the vibracores, approximately 20 deep borings would be advanced at selected WTG sites. The borings would be advanced from a truck-mounted drill rig placed upon a jack-up barge that rests on spuds lowered to the seafloor. Each of the four spuds would be approximately 4 feet in diameter, with a pad approximately 10 feet on a side on the bottom of the spud. The barge would be towed from boring location to location by a tugboat. The drill rig would be powered using a gasoline or diesel powered electric generator. Crew would access the boring barge daily from port using a small boat. Borings generally can be advanced to the target depth (100 to 200 feet) within 1 to 3 days, subject to weather and substrate conditions. Drive and wash drilling techniques would be used; the casting would be approximately 6 inches in diameter. Cone Penetrometer Testing (CPT) or an alternative subsurface evaluation technique would be conducted prior to construction, to evaluate subsurface sediment conditions. A CPT rig would be mounted on a jack-up barge similar to that used for the borings. The top of a CPT drill probe is typically up to 3 inches in diameter, with connecting rods less than 6 inches in diameter

Construction of the Wind Energy Facility
Each WTG has an energy generating capacity of approximately 3.6 megawatts (MW) and the proposed action is designated for a maximum electrical energy capacity of 468 MW. In order to generate maximum wind energy production, the WTGs will be arranged in specific parallel rows in a grid pattern. For this area of Nantucket Sound, the wind power density analysis conducted by the applicant determined that operation of the array in a northwest to southeast alignment provides optimal wind energy potential for the WTGs. This alignment will position the WTGs perpendicular to prevailing winds, which are generally from the northwest in the winter and the southwest in the summer.

Each turbine is pitch-regulated with active yaw to allow it to turn into the wind, and has a three-blade rotor. The main components of the WTG are the rotor, transmission system, generator, yaw system, and the control and electrical systems, which are located within the nacelle. The nacelle is the portion of the WTG that encompasses the drive train and supporting electromotive generating systems that produce the wind-generated energy. The nacelle would be mounted on a manufactured tubular conical steel tower supported by a monopole foundation system. A prefabricated access platform and service vessel landing (approximately 32 feet from mean lower low water (MLLW) would be provided at the base of the tower. The rotor has three blades manufactured from fiberglass-reinforced epoxy, mounted on the hub. The monopiles would be of two different diameters, depending on the depth of the water. A 16.75 foot (5.1 meter) diameter monopile would be installed for WTGs in water depths of 0 to 40 feet and an 18 foot (5.5 meter) diameter monopile would be installed for WTGs in water depths of 40 to 50 feet.

A jack up barge with a crane would be used for the installation of the monopiles. The jack up barge would have four legs with pads a minimum of four meters on a side. The crane would lift the monopiles from the transport barge and place them into position. The monopiles would be installed into the seabed by means of a pile driving ram or vibratory hammer to an approximate depth of 85 feet. This would be repeated at all WTG locations. Only two pieces of pile driving equipment would be present within the action area at any one time and they will not operate simultaneously. Monopiles to be installed range in length from approximately 122 feet for those installed in the shallowest locations to approximately 172 feet for those to be installed at the deepest sites. The anticipated time to install all of the monopiles is expected to be approximately eight months.

The installation of the WTG itself would be from a specialized vessel configured specifically for this purposes. The vessel will be likely loaded at Quonset, Rhode Island with the necessary components to erect two to four WTGs. Approximately 43 trips will be needed to deliver the material to the work site. The vessel would transit from Quonset to the work site and set up adjacent to one of the previously installed monopiles. A jacking system would then stabilize the vessel in the correct location. A transition piece would then be grouted in place to the monopile. The crane would then place the lower half of the tower onto the deck of the transition piece. The upper tower section is then added and then the nacelle, hub and blades are raised to the top of the tower and secured. This process is anticipated to take approximately 30 to 40 hours for each WTG. This process is anticipated to take approximately 9 months to complete. The installation of the WTGs will overlap with the installation of the monopiles.

Each of the 130 WTGs will generate electricity independent of one another. Within the nacelle of each turbine, a wind-driven generator would produce low voltage electricity, which would be "stepped up" by an adjacent transformer to produce 33 kV electrical transmission capacity. Solid dielectric submarine cables from each WTG will interconnect within the grid and terminate at their spread junctions on the electrical service platform (ESP). The ESP will serve as the common interconnection point for all of the WTGs within the wind park. The proposed submarine cable system is approximately 12.5 miles in length from the ESP to the landfall location in Yarmouth, Massachusetts. The submarine cables would travel north to northeast in Nantucket Sound to Lewis Bay past the westerly side of Egg Island, and then make landfall at New Hampshire Avenue. The proposed onshore cable route to its intersection with the NSTAR Electric Right of Way (ROW) would be located entirely along existing paved ROWs where other underground utilities already exist.

As the monopiles and WTGs are completed, the submarine inner-array cables (see below) would be laid in order to connect each string of wind turbines, and then the scour control system would be installed on the seabed around each monopile. The scour control system would help to prevent underwater currents from eroding the substrate adjacent to the WTG foundation. The scour system would consist of either a set of six scour-control mats arranged to surround the monopile, or rock armoring. Each scour-control mat is 16.5 feet by 8.2 feet with eight anchors that securely tie to the seabed. It is anticipated that the process of completing one string of WTGs (10 WTGs with associated inner-array cable and scour mats) would take up to one month (approximately 13 months total). The scour mats are placed on the seabed by a crane or davit onboard the support vessel. Final positioning is performed with the assistance of divers. After the mat is placed on the bottom, divers use a hydraulic spigot gun fitted with an anchor drive spigot to drive the anchors into the seabed. In the event that scour mats are found to be less effective, rock armoring will be used. The rock armor scour control design requires the use of filter layer material and rock armor stones. The rock armor and filter material would be placed so that the final elevations approximate pre-installation bottom contours so that mounds of material would not be created. The rock armor stones would be placed on top of this filter material which is used to fill the majority of the scour hole that is predicted to develop after installation of each WTG and the ESP. The filter layer would also minimize the potential for the underlying natural sediment material to be removed by the wave action and would also minimize the potential for rock armor to settle into the underlying sediment material. The armor stones will be sized so that they are large enough not to be removed by the effects of the waves and current conditions, while being small enough to prevent the stone fill material placed underneath it from being removed. If it were used, the rock armor and filter layer (i.e., smaller stone fill) would be placed on the seabed using a clamshell bucket or chute.

An Electric Service Platform (ESP) will be installed and maintained within the approximate center of the WTG array. It would serve as the common interconnection point for all of the WTGs within the wind park. Each WTG would interconnect with the ESP via a 33 kV submarine cable system. These cable systems would interconnect with circuit breakers and transformers located on the ESP in order to transmit wind-generated power through the 115 kV shore-connected submarine cable systems. The inner-array cables would be arranged in strings, each of which would connect up to approximately 10 WTGs to a 33 kV circuit breaker on the ESP. The ESP would provide electrical protection and inner-array cable sectionalizing capability in the form of circuit breakers. It would also include voltage step-up transformers to step the 33 kV inner-array transmission voltage up to the 115 kV voltage level of the submarine cable connection to the land based system. The ESP would include a helipad to allow personnel access when conditions preclude vessel transport, and for emergency evacuation.

The ESP would be a fixed template type platform consisting of a jacket frame with six 42-inch driven piles to anchor the platform to the ocean floor. The six piles would be driven through pile sleeves to design tip elevation of approximately 150 feet below the surface of the sea bottom. The piles would be vibrated and hammered as required. The platform jacket and superstructure will be fully fabricated on shore and delivered to the work site by barges, where it will be installed. The platform would consist of a 100 foot by 200 foot steel superstructure. The installation of the ESP is anticipated to take approximately one month to complete. The platform would be placed approximately 39 feet above MLLW. Water depth at the site of the ESP installation is 28 feet. In addition to the electrical equipment, the ESP would include fire protection, battery backup units, and other ancillary systems. Maintenance and service access to the ESP would normally be by service boat. A boat landing dock consisting of a fender structure with ladder will be attached to the ESP to allow boat landing and transfer of personnel and equipment and temporary docking of the service craft. A crane will be mounted to the ESP to facilitate the transfer of equipment.

The submarine cable system interconnecting the WTGs with the ESP (the inner-array) would be of solid dielectric AC construction, using a three-conductor cable with all phases under a common jacket. The cables would be arranged in strings, each of which would connect up to approximately 10 WTGs to a 33 kV circuit breaker on the ESP. There would be a total of approximately 66.7 miles of inner-array cabling throughout the wind park. The proposed method of installation of the submarine cable is by the Hydroplow embedment process, commonly referred to as jet plowing. The cable laying barge would be loaded at the staging area (most likely in Quonset, RI) and then towed to the project site. This would be repeated as required to deliver and install all the required cable. This method involves the use of a positioned cable barge and a towed hydraulically-powered jet plow device that simultaneously lays and embeds the submarine cable in one continuous trench from WTG to WTG and then to the ESP. The barge would propel itself along the route with the forward winches, and the other moorings holding the alignment during the installation. The six point mooring system would allow a support tug to move anchors while the installation and burial proceeds uninterrupted on a 24-hour basis. The inner-array cables would be installed six feet below the seafloor. It is anticipated that three different cable sizes would be used with diameters ranging from 5.19 to 6.45 inches.

Jet plow equipment uses pressurized sea water from water pump systems on board the cable laying vessel to fluidize sediments. The jet plow device is typically fitted with hydraulic pressure nozzles that create a direct downward and backward "swept flow" force inside the trench. This provides a down and back flow of re-suspended sediments within the trench, thereby "fluidizing" the in situ sediment column as it progresses along the submarine cable route such that the submarine cable settles into the trench under its own weight to the planned depth of the burial. A skid/pontoon-mounted jet plow, towed by the cable-laying barge, is proposed for the submarine installation. This jet plow has no propulsion of its own. The cable system is deployed from the vessel to the funnel of the jet plow device. The jet plow blade is lowered onto the seabed, pump systems are initiated, and the jet plow progresses along the cable route, creating a fluidized sediment trench approximately 4 to 6 feet wide (top width) to a depth of 8 feet below the present bottom into which the cable system settles through its own weight. The jet plow does not create an open trench of these dimensions but rather fluidizes the sediment with enough injected water that the cable can settle into the "soupy" sediments to a minimum depth of 6 feet below the bottom. The installation of the submarine transmission cable is expected to take two to four weeks to complete.

The transition of the interconnecting 115 kV submarine transmission cables from water to land would be accomplished through the use of horizontal directional drilling (HDD) methodology. The HDD would be staged at the onshore landfall area and would involve the drilling of the boreholes from land toward the offshore exit point. Conduits would then be installed the length of the boreholes and the transmission cable would be pulled through the conduits from the seaward end toward the land. A transition manhole/transmission cable splicing vault would be installed using conventional excavation equipment at the onshore transition point where the submarine and land transmission cables would be connected.

Two 115 kV transmission circuits would interconnect the ESP with the existing NSTAR Electric transmission grid serving Cape Cod. Each of the two circuits consists of two three-conductor cables, resulting in a total of four cables. The four submarine transmission cables would be installed as two circuits by bundling two cables per circuit together during installation and installing the two circuits. The overall diameter of the cable is 7.75 inches. The submarine transmission cables would transition to the onshore transmission cable by using HDD methodologies to a transition vault positioned at the end of New Hampshire Avenue in Yarmouth, MA. Transmission cables would be installed six feet below the seafloor.

During construction, Quonset Point, Rhode Island will likely serve as the primary staging area. Vessels will transit between Quonset Point and the wind park to carry large equipment, components, personnel and supplies. During the operation phase, supplies, equipment and maintenance vessels are likely to be staged out of New Bedford and/or Falmouth. Approximately 43 trips are anticipated to move the monopiles to the work site.

It is anticipated that the main operation center for the wind park would be located in the Town of Yarmouth, MA. Cape Wind would operate a remote monitoring and command center where operational decisions could be made. Service and maintenance personnel would be stationed at one of two additional onshore locations: one for the parts storage and larger maintenance supply vessels and the second located closer to the site for crew transport. The maintenance operation would likely be based in New Bedford, Massachusetts and may also deploy several crew boats out of Falmouth, Massachusetts. The New Bedford facility would likely be located on Popes Island and would include dock space for two 65 foot maintenance vessels, as well as a warehouse for parts and tool storage, and crew parking. An off-site warehouse would also be utilized to increase parts storage. Maintenance vessels would be loaded with small containers at the Popes Island facility and transported to either the WTG or the ESP where the containers would be unloaded. Additional dock space would likely be rented in Falmouth Inner Harbor from which work crews would be deployed to either the WTG and/or the ESP in 35 and 45 foot long crew boats. In addition, a high speed emergency response boat (20 to 25 foot long) would be maintained in Falmouth Inner Harbor ready to respond whenever there is marine activity taking place.

Routine maintenance will occur on all WTGs once they become operational. Most planned preventative service and maintenance is expected to occur during the summer months when weather is most favorable. Routine service is usually a two day exercise and would include 3 to 4 crew members. Unplanned maintenance is carried out to any part of the WTG in response to a breakdown or failure. This could occur at any time of year but is unlikely to occur when wave heights exceed 5 feet.

The ESP could be serviced by vessel or by helicopter. This would allow for maintenance crews to be deployed to the ESP during periods when wind and wave conditions are unsuitable for boat transfers.

The anticipated schedule for the action, assuming all Federal and state permitting and approval processes are completed in the fourth quarter of 2008, is as follows: (1) during the winter of 2009-2010 the onshore ductbanks, landfall transition and the temporary cofferdam will be installed; (2) during the third and fourth quarter of 2009 and first quarter of 2010 the ESP, the submarine 115 kV cables, and the onshore 115 kV cables will be installed; and (3) beginning the first quarter of 2010, the WTGs the inner-array cables and the scour mats will be erected and installed.

Decommissioning

The WTGS have a stated design life span of twenty years. However, as this estimate is based on experience generated from land-based machines where winds are more turbulent, it is possible that the WTGs may be operational beyond the minimum design life of twenty years.

In the event that the proposed action ceases operations or at the end of its useful life, a decommissioning plan will be implemented to remove and, to the greatest degree possible, recycle equipment and associated materials, thereby returning the area essentially to pre-existing conditions, to the extent practicable. Any decision by the proposed action's owners to cease operation of individual WTGs or the entire proposed action and to decommission and remove the proposed action's components would require consultation with MMS. MMS would then consult with the FWS and NMFS to determine if reinitiation of section 7 consultation was required based on any decommissioning plans. If the entire proposed action ceases to operate for a period of time of 18 months or more, and during that time the proposed action's owners have made no good-faith effort to restart operation, upgrading or decommissioning, the proposed action may be determined to be inoperative and decommissioning instruments may be accessed by MMS to initiate decommissioning activities. Decommissioning of the proposed action is largely the reverse of the installation process.

It is anticipated that equipment and vessels similar to those used during installation would be used for decommissioning. For offshore work, this would include a jet plow, crane barges, jack up barges, tugs, crew boats and specialty vessels such as cable laying vessels. An onshore disposal and recycling facility would be used to handle the materials removed from the project site. A facility currently exists in Everett, Massachusetts that could be utilized for this aspect of the decommissioning.

The initial step in the decommissioning process would involve the disconnection of the inner-array cables from the WTGs. The cables would be removed from their embedded position in the seabed. Where necessary, the cable trench would be jet plowed to fluidize the sandy sediments covering the cables, and the cables would then be reeled up onto barges. The cable reels would then be transported to land based facilities for recycling. The WTGs would be prepared for dismantling by draining all fluids and then deconstructing the WTGs. Cranes and vessels would be used to remove the blades, hub, nacelle, and tower. Once the wind turbines and towers have been removed, the foundation components (transition piece, monopile, scour mats and rock armor) would be decommissioned. Sediments inside the monopile would be suctioned out to a depth of 15 feet below the existing sea bottom in order to allow for access for the cutting of the pile in preparation for removal. The sediments would be pumped from the monopile and stored on a barge. All scour mats would be recovered, brought to the surface by crane, placed on a barge and brought to shore. Any rock armoring would be excavated with a clamshell dredge, placed on a barge and disposed of at an upland location. The monopile would then be cut from the inside at approximately 15 feet below grade. The sediments removed from the inner space of the monopile would be returned to the depression left when the monopile is removed.

Action Area

The action area is defined in 50 CFR 402.02 as "all areas to be affected directly or indirectly by the Federal action and not merely the immediate area involved in the action." The action area includes the footprint of the energy project where the WTGs and ESP will be installed, the submarine cable route, the route between the staging and operations areas in Falmouth, MA, New Bedford, MA and Quonset Point, RI and the project site, as well as the underwater area where effects of the project (i.e., increases in suspended sediment and underwater noise) will be experienced. The action area is illustrated in Figure 2 and is largely consistent with the area identified as Nantucket Sound.

Water depths within Nantucket Sound range from 1 to 70 feet at mean lower low water (MLLW). Depths on Horseshoe Shoal where the WTGs will be installed range from 0.5 feet to 60 feet at MLLW. Along the cable interconnection corridor, between Horseshoe Shoal and the Cape Cod shoreline, water depths vary from 16 to 40 feet MLLW. Water depths within Lewis Bay and Hyannis Harbor range from 8 to 16 feet at MLLW in the center of the bay to less than 5 feet at MLLW along the perimeter and between Dunbar Point and Great Island.

STATUS OF AFFECTED SPECIES

Several species listed under NMFS' jurisdiction occur off of the Massachusetts coast and may occur seasonally within the action area. No critical habitat has been designated within the action area; as such, no critical habitat will be affected by this action.

In Massachusetts, the federally endangered shortnose sturgeon (*Acipenser brevirostrum*) is only known to occur in the Merrimack and Connecticut Rivers, neither of which are in the action area for this consultation (NMFS 1998b). As shortnose sturgeon do not occur in the action area, this species will not be considered further in this biological opinion.

The hawksbill turtle (*Eretmochelys imbricata*) is relatively uncommon in the waters of the continental US. Hawksbills prefer coral reefs, such as those found in the Caribbean and Central America; however, there are accounts of hawksbills in south Florida and Texas. Most of the

Texas records report small turtles, probably in the 1-2 year class range. Many captures or strandings are of individuals in an unhealthy or injured condition (Hildebrand 1982). The lack of sponge-covered reefs and the cold winters in the northern Gulf of Mexico probably prevent hawksbills from establishing a viable population in this area. No takes of hawksbill sea turtles have been recorded in northeast or mid-Atlantic fisheries covered by the NEFSC observer program. In the north Atlantic, small hawksbills have stranded as far north as Cape Cod, Massachusetts (STSSN database). Many of these strandings were observed after hurricanes or offshore storms. There have been no verified observations of hawksbills in the action area. Based on this information, NMFS has determined that hawksbill sea turtles are extremely unlikely to occur in the action area. As this species does not occur in the action area, this species will not be considered further in this consultation.

Sperm, blue and sei whales also occur in Northeast waters. However, all of these species occur in deep offshore waters. As none of these species occur in the action area, these species will not be considered further in this consultation.

NMFS has determined that the action being considered in this biological opinion may affect the following endangered or threatened species under NMFS' jurisdiction:

Cetaceans	
Right whale (*Eubalaena glacialis*)	Endangered
Humpback whale (*Megaptera novaeangliae*)	Endangered
Fin whale (*Balaenoptera physalus*)	Endangered
Sea Turtles	
Loggerhead sea turtle (*Caretta caretta*)	Threatened
Leatherback sea turtle (*Dermochelys coriacea*)	Endangered
Kemp's ridley sea turtle (*Lepidochelys kempii*)	Endangered
Green sea turtle (*Chelonia mydas*[1])	Endangered/Threatened

This section will focus on the status of the various species within the action area, summarizing information necessary to establish the environmental baseline and to assess the effects of the proposed action. Background information on the range-wide status of these species and a description of critical habitat can be found in a number of published documents including recent sea turtle status reviews and stock assessments(NMFS and USFWS 1995, USFWS 1997, TEWG 2000, NMFS SEFSC 2001), Recovery Plans for the humpback whale (NMFS 1991a), right whale (NMFS 2005), fin and sei whale (NMFS 1998a), loggerhead sea turtle (NMFS and USFWS 1991) and leatherback sea turtle (NMFS and USFWS 1992), and the 2007 marine mammal stock assessment reports (Waring et al. 2008).

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[1] Pursuant to NMFS regulations at 50 CFR 223.205, the prohibitions of Section 9 of the Endangered Species Act apply to all green turtles, whether endangered or threatened.

North Atlantic Right whales

Historically, right whales have occurred in all the world's oceans from temperate to subarctic latitudes (Perry *et al.* 1999). In both hemispheres, they are observed at low latitudes and in nearshore waters where calving takes place in the winter months, and in higher latitude foraging grounds in the summer (Clapham *et al.* 1999; Perry *et al.* 1999).

The North Atlantic right whale (*Eubalaena glacialis*) has been listed as endangered under the Endangered Species Act (ESA) since 1973. It was originally listed as the "northern right whale" as endangered under the Endangered Species Conservation Act, the precursor to the ESA in June 1970. The species is also designated as depleted under the Marine Mammal Protection Act (MMPA).

In December 2006, NMFS completed a comprehensive review of the status of right whales in the North Atlantic and North Pacific Oceans. Based on the findings from the status review, NMFS concluded that right whales in the northern hemisphere exist as two species: North Atlantic right whale (*Eubalaena glacialis*) and the North Pacific right whale (*Eubalaena japonica*). NMFS determined that each of the species is in danger of extinction throughout its range. In 2008, based on the status review, NMFS listed the endangered northern right whale (*Eubalaena spp.*) as two separate endangered species: the North Atlantic right whale (*E. glacialis*) and North Pacific right whale (*E. japonica*) (73 FR 12024).

The International Whaling Commission (IWC) recognizes two right whale populations in the North Atlantic: a western and eastern population (IWC, 1986). It is thought that the eastern population migrated along the coast from northern Europe to northwest Africa. The current distribution and migration patterns of the eastern North Atlantic right whale population, if extant, are unknown. Sighting surveys from the eastern Atlantic Ocean suggest that right whales present in this region are rare (Best *et al.*, 2001) and it is unclear whether a viable population in the eastern North Atlantic still exists (Brown 1986, NMFS 1991b). Photo-identification work has shown that some of the whales observed in the eastern Atlantic were previously identified as western Atlantic right whales (Kenney 2002). This Opinion will focus on the western North Atlantic subpopulation of right whales which occurs in the action area.

Habitat and Distribution

Western North Atlantic right whales generally occur from the southeast U.S. to Canada (*e.g.*, Bay of Fundy and Scotian Shelf) (Kenney 2002; Waring *et al.* 2007). Like other right whale species, they follow an annual pattern of migration between low latitude winter calving grounds and high latitude summer foraging grounds (Perry *et al.* 1999; Kenney 2002). Right whale movements and habitat have been described as follows:

> The distribution of right whales seems linked to the distribution of their principal zooplankton prey, calanoid copepods (Winn et al. 1986; NMFS 2005; Baumgartner and Mate 2005; Waring *et al.* 2007). Right whales are most abundant in Cape Cod Bay between February and April (Hamilton and Mayo 1990; Schevill *et al.* 1986; Watkins and Schevill 1982) and in the Great South Channel in May and June (Kenney *et al.* 1986; Payne *et al.*

1990; Kenney *et al.* 1995; Kenney 2001) where they have been observed feeding predominantly on copepods of the genera *Calanus* and *Pseudocalanus* (Baumgartner and Mate 2005; Waring *et al.* 2007). Right whales also frequent Stellwagen Bank and Jeffrey's Ledge, as well as Canadian waters including the Bay of Fundy and Browns and Baccaro Banks in the summer through fall (Mitchell *et al.* 1986; Winn *et al.* 1986; Stone *et al.* 1990). Calving occurs in the winter months in coastal waters off of Georgia and Florida (Kraus *et al.* 1988). In the North Atlantic it appears that not all reproductively active females return to the calving grounds each year (Kraus *et al.*, 1986; Payne, 1986). The location of the majority of the population during the winter months remains unknown (NMFS 2005).

While right whales are known to congregate in the aforementioned areas, much is still not understood and movements within and between these areas may be more extensive than thought (Waring *et al.* 2007). In the winter, only a portion of the known right whale population is seen on the calving grounds. The winter distribution of the remaining right whales remains uncertain (NMFS 2005, Waring *et al.* 2007). Results from winter surveys and passive acoustic studies suggest that animals may be dispersed in several areas including Cape Cod Bay (Brown *et al.* 2002) and offshore waters of the southeastern U.S. (Waring *et al.* 2007). Telemetry data have shown lengthy and somewhat distant excursions into deep water off of the continental shelf (Mate *et al.* 1997) as well as extensive movements over the continental shelf during the summer foraging period (Mate and Nieukirk 1992; Mate *et al.* 1997; Bowman 2003; Baumgartner and Mate 2005). Knowlton *et al.* (1992) reported several long-distance movements as far north as Newfoundland, the Labrador Basin, and southeast of Greenland; in addition, resightings of photographically identified individuals have been made off Iceland, arctic Norway, and in the old Cape Farewell whaling ground east of Greenland. The Norwegian sighting (September 1999) represents one of only two sightings this century of a right whale in Norwegian waters, and the first since 1926. Together, these long-range matches indicate an extended range for at least some individuals and perhaps the existence of important habitat areas not presently well described. Similarly, records from the Gulf of Mexico (Moore and Clark, 1963; Schmidly *et al.*, 1972) represent either geographic anomalies or a more extensive historic range beyond the sole known calving and wintering ground in the waters of the southeastern United States. The frequency with which right whales occur in offshore waters in the southeastern U.S. remains unclear (Waring *et al.*, 2007).

In 1994, critical habitat was designated for the North Atlantic right whales in accordance with the ESA. NMFS designated three critical habitat areas for right whales: (1) portions of Cape Cod Bay and Stellwagen Bank, (2) the Great South Channel, and (3) coastal waters off of Georgia and Florida's east coast (NMFS 1994). Right whale critical habitat in Northeast waters were designated for their importance as right whale foraging sites while the southeast critical habitat area was identified for its importance as a calving and nursery area (NMFS 1994). In 2002, NMFS received a petition to revise designated critical habitat for right whales by combining and expanding the existing Cape Cod Bay and Great South Channel critical habitats in the Northeast and by expanding the existing critical habitat in the Southeast (NMFS 2003). In response to the petition, NMFS (2003) recognized that there was new information on right whale distribution in areas outside of the designated critical habitat. However, the ESA requires that critical habitat be designated based on identification of specific habitat features essential to the conservation of the species rather than just known distribution (NMFS 2003). NMFS, therefore, concluded that more analyses of the sightings data and their environmental correlates are necessary to define and designate these areas as critical habitat (NMFS 2003).

Abundance estimates and trends

Although an estimate of the pre-exploitation population size for the North Atlantic right whale is not available, it is well known and documented that there are relatively few right whales remaining in the western North Atlantic. As is the case with most wild animals, an exact count cannot be obtained. However, abundance can be reasonably estimated as a result of the extensive study of this subpopulation. IWC participants from a 1999 workshop agreed to a minimum direct-count estimate of 263 right whales alive in 1996 and noted that the true population was unlikely to be greater than this estimate (Best *et al.* 2001). Based on a census of individual whales using photo-identification techniques and an assumption of mortality for those whales not seen in seven years, a total 299 right whales was estimated in 1998 (Kraus *et al.* 2001), and a review of the photo-ID recapture database on June 15, 2006, indicated that 313 individually recognized whales were known to be alive during 2001 (Waring *et al.* 2007). Because this 2006 review was a nearly complete census, it is assumed this estimate represents a minimum population size. Results from a review of the photo-ID recapture database on May 30, 2007, are still preliminary (matching photos from 2006 and 2007 is not complete), but indicate that 325 individual recognized whales in the catalog were known to be alive in 2003 (Waring *et al.* 2008).

A total of 156 right whale calves have been born from 2001-2007 (P. Hamilton, pers. comm.). The mean calf production for the 15-year period from 1993-2006 is estimated to be 11.2/year (Waring *et al.* 2007). Calving numbers have been sporadic, with large differences among years, including a record calving season in 2000/2001 with 31 right whale births (Waring *et al.* 2007). The three calving years (97/98; 98/99; 99/00) prior to this record year provided low recruitment levels with only 10 calves born. The last six calving seasons (2000-2006) have been remarkably better with 31, 21, 19, 16, 28 and 19 births, respectively (Waring *et al.* 2007). The calf count of 22 animals for the latest calving season (2006/2007) is still preliminary and additional calves may be observed (Waring *et al.* 2008). The subpopulation has also continued to experience losses of calves, juveniles and adults. As of August 1, 2008, there were 528 individually identified right whales in the photo-identification catalog of which 25 were known to be dead, 135 were presumed to be dead as they had not been sighted in the past six years and 368 were presumed to be alive (Hamilton *et al.* 2008). Although the population has seen some growth over the past 8 years, the level of growth is significantly lower than healthy populations of large whales (Pace *et al.* 2008).

As is the case with other mammalian species, there is an interest in monitoring the number of females in this right whale subpopulation since their numbers will affect the subpopulation trend (whether declining, increasing or stable). The sex ratio of the photo-identified and catalogued population of North Atlantic right whales appears to be slightly skewed toward males (196M:187F) (Waring *et al.* 2007). As of 2005, 92 reproductively-active females had been identified (Kraus *et al.* 2007). From 1983-2005, the number of new mothers recruited to the population (with an estimated age of 10 for the age of first calving), varied from 0-11 each year with no significant increase or decline over the period (Kraus *et al.* 2007). By 2005, 16 right whales had produced at least 6 calves each, and 4 cows had at least seven calves. Two of these cows were at an age which indicated a reproductive life span of at least 31 years (Kraus *et al.* 2007). As described above, the 2000/2001 - 2006/2007 calving seasons have had relatively high calf production (31, 21, 19, 16, 28, 19, and at least 22 respectively) and have included additional first time mothers (*e.g.*, eight new mothers in 2000/2001). These potential "gains" have been offset, however, by continued losses to the subpopulation including the death of mature females as a result of anthropogenic mortality (like that described in Glass *et al.* 2008, below). The period of November 2004 through May 2005 was particularly devastating with five right whale mortalities. One of the females, nicknamed "Stumpy" (so named because of an old wound on her left fluke), was killed in February 2004 of an apparent ship strike (NMFS 2006). One of the largest right whales on record, she was first sited in 1975 and known to be a prolific breeder, successfully rearing calves in 1980, 1987, 1990, 1993, and 1996 (Moore *et. al* 2007). At the time of her death, she was estimated to be 30 years of age and carrying her sixth calf; the near-term fetus also died (NMFS 2006).

Of the recent mortalities, including those in the first half of 2005, 6 were adult females, three of which were carrying near-term fetuses and 4 of which were just starting to bear calves. Since the average lifetime calf production is 5.25 calves (Fujiwara and Caswell 2001), the deaths of these females represent a loss of reproductive potential of as many as 21 animals (Waring *et al.* 2007). However, it is important to note though that not all right whale mothers are equal with regards to calf production (i.e. #1158 who had 1 calf over a 25-year period) (Kraus *et al.* 2007).

Abundance estimates are an important part of assessing the status of the species. However, for Section 7 purposes, the population trend (*i.e.,* whether increasing or declining) provides better information for assessing the effects of a proposed action on the species. As described in previous Opinions, data collected in the 1990s suggested that right whales were experiencing a slow but steady recovery (Knowlton *et al.* 1994). However, Caswell *et al.* (1999) used photo-identification data and modeling to estimate survival and concluded that right whale survival decreased from 1980 to 1994. Modified versions of the Caswell *et al.* (1999) model as well as several other models were reviewed at the 1999 IWC workshop (Best *et al.* 2001). Despite differences in approach, all of the models indicated a decline in right whale survival in the 1990s relative to the 1980s with female survival, in particular, apparently affected (Best *et al.* 2001, Waring *et al.* 2007). In 2002, NMFS' NEFSC hosted a workshop to review right whale population models to examine: (1) potential bias in the models and (2) changes in the subpopulation trend based on new information collected in the late 1990s (Clapham *et al.* 2002). Three different models were used to explore right whale survivability and to address potential sources of bias. Although biases were identified that could negatively affect the results, all three modeling techniques resulted in the same conclusion; survival has continued to decline and seems to be focused on females (Clapham *et al.* 2002). Mortalities, including those in the first half of 2005, suggest an increase in the annual mortality rate (Kraus *et. al* 2005). Calculations indicate that this increased mortality rate would reduce population growth by approximately 10%

per year (Kraus *et. al* 2005).

Reproductive Fitness

Healthy reproduction is critical for the recovery of the North Atlantic right whale (Kraus *et al.* 2007). While modeling work suggests a decline in right whale abundance as a result of reduced survival, particularly for females, some researchers have also suggested that the subpopulation is being affected by a decreased reproductive rate (Best *et al.* 2001; Kraus *et al.* 2001). Kraus *et al.* (2007) reviewed reproductive parameters for the period 1983-2005, and estimated calving intervals to have changed from 3.5 years in 1990 to over five years between 1998-2003, and then suddenly decreased to just over 3 years in 2004 and 2005.

Factors that have been suggested as affecting the right whale reproductive rate include reduced genetic diversity (and/or inbreeding), contaminants, biotoxins, disease, nutritional stress, and loss of critical habitat. Although it is believed that a combination of these factors is likely causing an effect on right whales (Kraus *et al.* 2007), there is currently no evidence available to determine their potential effect, if any. The dramatic reduction in the North Atlantic right whale population believed to have occurred due to commercial whaling may have resulted in a loss of genetic diversity which could affect the ability of the current population to successfully reproduce (*i.e.*, decreased conceptions, increased abortions, and increased neonate mortality). The current hypothesis is that the low level of genetic variability in this species produces a high rate of mate incompatibility and unsuccessful pregnancies (Frasier *et al.* 2007). Analyses are currently under way to assess this relationship further as well as the influence of genetic characteristics on the potential for species recovery (Frasier *et al.* 2007). Studies by Schaeff *et al.* (1997) and Malik *et al.* (2000) indicate that western North Atlantic right whales are less genetically diverse than southern right whales. However, several apparently healthy populations of cetaceans, such as sperm whales and pilot whales, have even lower genetic diversity than observed for western North Atlantic right whales (IWC 2001). Similarly, while contaminant studies have confirmed that right whales are exposed to and accumulate contaminants, researchers could not conclude that these contaminant loads were negatively affecting right whale reproductive success since concentrations were lower than those found in marine mammals proven to be affected by PCBs and DDT (Weisbrod *et al.* 2000). Another suite of contaminants (i.e. antifouling agents and flame retardants) that have been proven to disrupt reproductive patterns and have been found in other marine animals, have raised new concerns (Kraus *et al.* 2007). Recent data also support a hypothesis that chromium, an industrial pollutant, may be a concern for the health of the North Atlantic right whales and that inhalation may be an important exposure route (Wise *et al.* 2008). A number of diseases could be also affecting reproduction, however tools for assessing disease factors in free-swimming large whales currently do not exist (Kraus *et al.* 2007). Once developed, such methods may allow for the evaluation of disease effects on right whales. Impacts of biotoxins on marine mammals are also poorly understood, yet data is showing that marine algal toxins may play significant roles in mass mortalities of these animals (Rolland *et al.* 2007). Although there are no published data concerning the effects of biotoxins on right whales, researchers are now certain that right whales are being exposed to measurable quantities of paralytic shellfish poisoning (PSP) toxins and domoic acid via trophic transfer through the copepods upon which they feed (Durbin *et al.* 2002, Rolland *et al.* 2007).

Data indicating right whales are food-limited are difficult to evaluate (Kraus *et al.* 2007). Although North Atlantic right whales seem to have thinner blubber than right whales from the South Atlantic (Kenney 2000), there is no evidence at present to demonstrate that the decline in birth rate and increase in calving interval is related to a food shortage. Nevertheless, a connection among right whale reproduction and environmental factors may yet be found. Modeling work by Caswell *et al.* (1999) and Fujiwara and Caswell (2001) suggests that the North Atlantic Oscillation (NAO), a naturally occurring climatic event, does affect the survival of mothers and the reproductive rate of mature females, and it also seems to affect calf survival (Clapham *et al.* 2002). Greene *et al.* (2003) described the potential oceanographic processes linking climate variability to the reproduction of North Atlantic right whales. Climate-driven changes in ocean circulation have had a significant impact on the plankton ecology of the Gulf of Maine, including effects on *Calanus finmarchicus*, a primary prey resource for right whales. Researchers found that during the 1980's, when the NAO index was predominately positive, *C. finmarchicus* abundance was also high; when a record drop occurred in the NAO index in 1996, *C. finamarchicus* abundance levels also decreased significantly. Right whale calving rates since the early 1980's seem to follow a similar pattern, where stable calving rates were noted from 1982-1992, but then two major, multi-year declines occurred from 1993-2001, consistent with the drops in copepod abundance. It has been hypothesized that right whale calving rates are thus a function of food availability as well as the number of females available to reproduce (Greene et al 2003, Greene and Pershing 2004). Such findings suggest that future climate change may emerge as a significant factor influencing the recovery of right whales. Some believe the effects of increased climate variability on right whale calving rates should be incorporated into future modeling studies so that it may be possible to determine how sensitive right whale population numbers are to variable climate forcing (Greene and Pershing 2004).

Anthropogenic Mortality

There is general agreement that right whale recovery is negatively affected by anthropogenic mortality. From 2002-2006, right whales had the highest proportion of entanglement and ship strike events relative to the number of reports for a species (Glass *et al.* 2008). Given the small population size and low annual reproductive rate of right whales, human sources of mortality may have a greater effect to relative population growth rate than for other large whale species (Waring *et al.* 2007). For the period 2002-2006, the annual mortality and serious injury rate for the North Atlantic right whale averaged to be 3.8 per year (2.4 in U.S. waters; 1.4 in Canadian waters) (Glass *et al.* 2008, Waring *et. al.* 2008). Twenty-one confirmed right whale mortalities were reported along the U.S. east coast and adjacent Canadian Maritimes from 2002-2006 (Glass *et al.* 2008). These numbers represent the minimum values for human-caused mortality for this period. Given the range and distribution of right whales in the North Atlantic, and the fact that positively buoyant species like right whales may become negatively buoyant if injury prohibits effective feeding for prolonged periods, it is highly unlikely that all carcasses will be observed (Moore *et. al.* 2004, Glass *et al.* 2008)). Moreover, carcasses floating at sea often cannot be examined sufficiently and may generate false negatives if they are not towed to shore for further necropsy (Glass *et al.* 2008). Decomposed and/or unexamined animals represent lost data, some of which may relate to human impacts (Waring *et al.* 2007).

Considerable effort has been made to examine right whale carcasses for the cause of death (Moore *et al.* 2004). Because they live in an ocean environment, examining right whale carcasses is often very difficult. Some carcasses are discovered floating at sea and cannot be retrieved. Others are in such an advanced stage of decomposition when discovered that a complete examination is not possible. Wave action and post-mortem predation by sharks can also damage carcasses, and preclude a thorough examination of all body parts. It should also be noted that mortality and serious injury event judgments are based upon the best available data and additional information may result in revisions (Cole *et al.* 2005). Of the 21 total, confirmed right whale mortalities (2002-2006) described in Glass *et al.* (2008), 3 were confirmed to be entanglement mortalities (1 yearling female, 1 adult female, 1 calf [sex not listed]) and 10 were confirmed to be ship strike mortalities (1 yearling female, 1 yearling male, 6 adult females, 1 male calf, and 1 female of unknown age). Serious injury involving right whales was documented for 4 entanglement events (1 calf [sex not indicated], 2 adult females, and 1 individual whose sex and age were unknown) and 2 ship strike events (1 individual whose sex and age were unknown and 1 yearling male).

Although disentanglement is either unsuccessful or not possible for the majority of cases, during the period of 2001-2005, there were at least 5 documented cases of entanglements for which the intervention of disentanglement teams averted a likely serious injury determination (for 2002-2006, there were 4 documented cases of such disentanglement efforts) (Waring *et al.* 2007, Waring *et al.* 2008). Entanglement or vessel collisions may not cause direct mortalities, but may weaken or otherwise affect individuals so that further injury or death is likely (Waring *et. al* 2007). Some right whales that have been entangled were subsequently involved in ship strikes (Hamilton *et al.* 1998) suggesting that the animal may have become debilitated by the entanglement to such an extent that it was less able to avoid a ship. In the same, skeletal fractures and/or broken jaws sustained during a vessel collision may heal, but then compromise a whale's ability to efficiently filter feed (Moore *et al.* 2007). A necropsy of right whale #2143 ("Lucky) found dead in January 2005 suggested the animal (and her near-term fetus) died after healed propeller wounds from a previous ship strike re-opened and became infected as a result of pregnancy (Moore *et al.* 2007, Glass *et al.* 2008). Sometimes, even with a successful disentanglement, an animal may die of injuries sustained by fishing gear (e.g. RW #3107) (Waring *et al.* 2008).

Entanglement records from 1990-2006 maintained by NMFS include 45 confirmed right whale entanglement events (Waring *et al.* 2008). Because whales often free themselves of gear following an entanglement event, scarification analysis of living animals may provide better indications of fisheries interactions rather than entanglement records (Waring *et al.* 2008). Data presented in Knowlton *et al.* 2008 indicate the annual rate of entanglement interaction remains at high levels. Four hundred and ninety-three individual, catalogued right whales were reviewed and 625 separate entanglement interactions were documented between 1980 and 2004. Approximately 358 out of 493 animals (72.6% of the population) were entangled at least once; 185 animals bore scars from a single entanglement, however one animal showed scars from 6 different entanglement events. The number of male and female right whales bearing entanglement scars was nearly equivalent (142/202 females, 71.8%; 182/224 males, 81.3%), indicating that right whales of both sexes are equally vulnerable to entanglement. However, juveniles appear to become entangled at a higher rate than expected if all age groups were equally vulnerable. For all years but one (1998), the proportion of juvenile, entangled right whales exceeded their proportion within the population. Based on photographs of catalogued animals from 1935 through 1995, Hamilton *et al.* (1998) estimated that 6.4 percent of the North Atlantic right whale population exhibit signs of injury from vessel strikes. Preliminary data indicate 6 reported right whale ship strikes in 2006, 2 reported right whale strikes in 2007, and as of March 20, 2008, no reported right whale vessel strikes in 2008 (note: these numbers may include cases where whales were observed with indications of ship strike, however it could not be confirmed if the interaction was pre-mortem) (NMFS 2008 DRAFT).

Summary of Right Whale Status

In March 2008, NMFS listed the North Atlantic right whale as a separate, endangered species (*Eubalaena glacialis*) under the ESA. This decision was based on an analysis of the best scientific and commercial data available. The decision took into consideration current population trends and abundance, demographic risk factors affecting the continued survival of the species, and ongoing conservation efforts. NMFS determined that the North Atlantic right whale is in danger of extinction throughout its range because of: (1) overutilization for commercial, recreational, scientific or educational purposes; (2) the inadequacy of existing regulatory mechanisms; and (3) other natural and manmade factors affecting its continued existence.

In the Atlantic, there are an estimated 300 right whales (+/- 10%) (Best *et al.* 2001). The 2000/2001 - 2006/2007 calving seasons have had relatively high calf production (31, 21, 19, 16, 28, 19 and at least 22 calves, respectively) and have included additional first time mothers (*e.g.*, eight new mothers in 2000/2001) (Waring *et al.* 2008 DRAFT). These potential "gains" have been offset, however, by continued losses to the subpopulation including the death of mature females as a result of anthropogenic mortality (Glass *et al.* 2008). Of the recent mortalities, including those in the first half of 2005, 6 were adult females, three of which were carrying near-term fetuses and 4 of which were just starting to bear calves. There are some indications that climate-driven ocean changes impacting the plankton ecology of the Gulf of Maine, may, in some manner, be affecting right whale fitness and reproduction. However there is also general agreement that right whale recovery is negatively affected by anthropogenic mortality.

Over the five-year period (2002-2006), right whales had the highest proportion of entanglements and ship strikes relative to the number of reports for a species: of 54 reports involving right whales, 25 were confirmed entanglements and 16 were confirmed ship strikes. There were 21 verified right whale mortalities, three due to entanglements, and ten due to ship strikes (Glass *et al.* 2008). This represents an absolute minimum number of the right whale mortalities for this period. Given the range and distribution of right whales in the North Atlantic, it is highly unlikely that all carcasses will be observed. Scarification analysis indicates that some whales do survive encounters with ships and fishing gear. However, the long-term consequences of these interactions are unknown.

A number of different modeling exercises using the extensive data collected on this subpopulation have come to the same conclusion; right whale survival continues to decline (Clapham *et al.* 2002). Based on the information currently available, for the purposes of this Opinion, NMFS believes that the western North Atlantic right whale subpopulation numbers 300 (+/- 10%) and is declining.

Humpback Whales

Humpback whales inhabit all major ocean basins from the equator to subpolar latitudes. They generally follow a predictable migratory pattern in both hemispheres, feeding during the summer in the higher near-polar latitudes and migrating to lower latitudes in the winter where calving and breeding takes place (Perry *et al.* 1999). Humpbacks are listed under the ESA at the species level. Therefore, information is presented below regarding the status of humpback whales throughout their range.

North Pacific, Northern Indian Ocean and Southern Hemisphere

Humpback whales range widely across the North Pacific during the summer months; from Port Conception, CA, to the Bering Sea (Johnson and Wolman 1984, Perry *et al.* 1999). Although the IWC only considered one stock (Donovan 1991) there is evidence to indicate multiple populations migrating between their respective summer/fall feeding areas to winter/spring calving and mating areas within the North Pacific Basin (Anglis and Outlaw 2007, Carretta *et al.* 2007). NMFS recognizes three management units within the U.S. EEZ for the purposes of managing this species under the MMPA. These are: the eastern North Pacific stock, the central North Pacific stock and the western North Pacific stock (Anglis and Outlaw 2007, Carretta *et al.* 2007). Winter/spring populations of humpback whales also occur in Mexico's offshore islands, however the migratory destinations of these whales is currently not well known (Anglis and Outlaw 2007, Carretta *et al.* 2007). Recent research efforts via the Structure of Populations, Levels of Abundance, and Status of Humpback Whales (SPLASH) Project estimate the abundance of humpback whales to be just under 20,000 whales for the entire North Pacific, a number which doubles previous population predictions (Calambokidis *et al.* 2008). There are indications that the eastern North Pacific stock was growing in the 1980's and early 1990's with a best estimate of 6-8% growth per year (Carretta *et al.* 2007). The central North Pacific stock appears to also have increased in abundance between the 1980's -1990's (Anglis and Outlaw 2007). Although, there is no reliable population trend data for the western North Pacific stock, as surveys of the known feeding areas are incomplete and many feeding areas remain unknown, minimum population size is currently estimated at 367 whales (Anglis and Outlaw 2007).

Little or no research has been conducted on humpbacks in the Northern Indian Ocean so information on their current abundance does not exist (Perry *et al.* 1999). Since these humpback whales do not occur in U.S. waters, there is no recovery plan or stock assessment report for the northern Indian Ocean humpback whales. Likewise, there is no recovery plan or stock assessment report for southern hemisphere humpback whales, and there is also no current estimate of abundance for humpback whales in the southern hemisphere although there are estimates for some of the six southern hemisphere humpback whale stocks recognized by the

IWC (Perry *et al.* 1999). Like other whales, southern hemisphere humpback whales were heavily exploited for commercial whaling. Although they were given protection by the IWC in 1963, Soviet whaling data made available in the 1990's revealed that 48,477 southern hemisphere humpback whales were taken from 1947-1980, contrary to the original reports to the IWC which accounted for the take of only 2,710 humpbacks (Zemsky *et al.* 1995, IWC 1995, Perry *et al.* 1999).

Gulf of Maine (North Atlantic)
Humpback whales from most Atlantic feeding areas calve and mate in the West Indies and migrate to feeding areas in the northwestern Atlantic during the summer months. Most of the humpbacks that forage in the Gulf of Maine visit Stellwagen Bank and the waters of Massachusetts and Cape Cod Bays. Previously, the North Atlantic humpback whale population was treated as a single stock for management purposes, however due to the strong fidelity to the region displayed by many whales, the Gulf of Maine stock was reclassified as a separate feeding stock (Waring *et al.* 2007). Sightings are most frequent from mid-March through November between 41°N and 43°N, from the Great South Channel north along the outside of Cape Cod to Stellwagen Bank and Jeffrey's Ledge (CeTAP 1982) and peak in May and August. Small numbers of individuals may be present in this area year-round, including the waters of Stellwagen Bank. They feed on a number of species of small schooling fishes, particularly sand lance and Atlantic herring, targeting fish schools and filtering large amounts of water for their associated prey. It is hypothesized humpback whales may also feed on euphausiids (krill) as well as capelin (Waring et al. 2007, Stevick *et al.* 2006).

In winter, whales from waters off New England, Canada, Greenland, Iceland, and Norway, migrate to mate and calve primarily in the West Indies where spatial and genetic mixing among these groups does occur (Waring *et al.* 2007). Various papers (Clapham and Mayo 1990; Clapham 1992; Barlow and Clapham 1997; Clapham *et al.* 1999) summarize information gathered from a catalogue of photographs of 643 individuals from the western North Atlantic population of humpback whales. These photographs identified reproductively mature western North Atlantic humpbacks wintering in tropical breeding grounds in the Antilles, primarily on Silver and Navidad Banks, north of the Dominican Republic. The primary winter range also includes the Virgin Islands and Puerto Rico (NMFS 1991a).

Humpback whales use the Mid-Atlantic as a migratory pathway to and from the calving/mating grounds, but it may also be an important winter feeding area for juveniles. Since 1989, observations of juvenile humpbacks in the Mid-Atlantic have been increasing during the winter months, peaking January through March (Swingle *et al.* 1993). Biologists theorize that non-reproductive animals may be establishing a winter feeding range in the Mid-Atlantic since they are not participating in reproductive behavior in the Caribbean. Swingle *et al.* (1993) identified a shift in distribution of juvenile humpback whales in the nearshore waters of Virginia, primarily in winter months. Identified whales using the Mid-Atlantic area were found to be residents of the Gulf of Maine and Atlantic Canada (Gulf of St. Lawrence and Newfoundland) feeding groups, suggesting a mixing of different feeding populations in the Mid-Atlantic region. Strandings of humpback whales have increased between New Jersey and Florida since 1985 consistent with the increase in Mid-Atlantic whale sightings. Strandings were most frequent during September through April in North Carolina and Virginia waters, and were composed primarily of juvenile humpback whales of no more than 11 meters in length (Wiley *et al.* 1995).

Photographic mark-recapture analyses from the Years of the North Atlantic Humpback (YONAH) project gave an ocean-basin-wide estimate of 11,570 animals during 1992/1993 and an additional genotype-based analysis yielded a similar by less precise estimate of 10,400 whales (95% c.i. = 8,000 - 13,600) (Waring *et al.* 2007). For management purposes under the MMPA, the estimate of 11,500 individuals is regarded as the best available estimate for the North Atlantic population (Waring *et al.* 2007). Assessing abundance for the Gulf of Maine stock of humpback whales has proved problematic, however, the best, recent estimate for the Gulf of Maine stock is 847 whales, derived from the 2006 aerial survey (Waring *et al.* 2007).

As is the case with other large whales, the major known sources of anthropogenic mortality and injury of humpback whales occur from fishing gear entanglements and ship strikes.
For the period 2002 through 2006, the minimum annual rate of human-caused mortality and serious injury to the Gulf of Maine humpback whale stock averaged 4.4 animals per year (U.S. waters, 4.0; Canadian waters, 0.4) (Glass *et al.* 2008, Waring *et al.* 2008). Between 2002 and 2006 humpback whales were involved in 77 confirmed entanglement events and 9 confirmed ship strike events (Glass *et al.* 2008). Over the five-year period, humpback whales were the most commonly observed entangled whale species; entanglements accounted for 6 mortalities and nine serious injuries (Glass *et al.* 2008). Although ship strikes were relatively uncommon, 7 of the 9 confirmed events were fatal (Glass et al. 2008). It was assumed that all of these events involved members of the Gulf of Maine stock of humpback whales unless a whale was confirmed to be from another stock; in reports prior to 2007, only events involving whales confirmed to be members of the Gulf of Maine stock were included. As of February 2008, there was no available information to indicate that the events described here do *not* include a Gulf of Maine animal. There were also many carcasses that washed ashore or were spotted floating at sea for which the cause of death could not be determined. Given the number of decomposed and incompletely or unexamined animals in the records, there needs to be greater emphasis on the timely recovery of carcasses and complete necropsies; decomposed and/or unexamined animals (e.g., carcasses reported but not retrieved or no necropsy performed) represent 'lost data' some of which may relate to human impacts (Glass *et al.* 2008, Waring *et al.* 2008).

Based on photographs taken between 2000-2002 of the caudal peduncle and fluke of humpback whales, Robbins and Mattila (2004) estimated that at least half (48-57%) of the sample (187 individuals) was coded as having a high likelihood of prior entanglement. Evidence suggests that entanglements have occurred at minimum rate of 8-10% per year. Scars acquired by Gulf of Maine stock humpback whales between 2000 and 2002 suggest a minimum of 49 interactions with gear took place. Based on composite scar patterns, it was believed that male humpback whales were more vulnerable to entanglement than females. Males may be subject to other sources of injury that could affect scar pattern interpretation. Images were obtained from a humpback whale breeding ground; 24% exhibited raw injuries, presumable a result from agonistic interactions. However, current evidence suggests that breeding ground interactions alone cannot explain the higher frequency of healed scar patterns among Gulf of Maine stock male humpback whales (Robbins and Matilla 2004).

Humpback whales, like other baleen whales, may also be adversely affected by habitat degradation, habitat exclusion, acoustic trauma, harassment, or reduction in prey resources due to trophic effects resulting from a variety of activities including fisheries operations, vessel traffic, and coastal development. Currently, there is no evidence that these types of activities are affecting humback whales. In October 2006, NMFS declared an unusual mortality event (UME) for humpback whales in the Northeast United States. At least 17 dead humpback whales have been discovered since March 2006. There has also been a documented bloom of *Alexandrium* sp., a toxic dinoflagellate that causes red tide from Maine to Massachusetts. Prior to the most recent UME, there had been only three other known cases of a mass mortality involving large whale species along the east coast: 1987–1988, 2003, and 2005. Geraci *et al.* (1989) provide strong evidence that, in the former case, these deaths of humpback whales resulted from the consumption of mackerel whose livers contained high levels of saxitoxin, a naturally occurring red tide toxin; the origin of which remains unknown. It has been suggested that the occurrence of a red tide event is related to an increase in freshwater runoff from coastal development, leading some observers to suggest that such events may become more common among marine mammals as coastal development continues (Clapham *et al.* 1999).

Changes in humpback distribution in the Gulf of Maine have been found to be associated with changes in herring, mackerel, and sand lance abundance associated with local fishing pressures (Stevick *et al.* 2006, Waring *et al.* 2007). Shifts in relative finfish species abundance correspond to changes in observed humpback whale movements (Stevick *et al.* 2006). However, there is no evidence that humpback whales were adversely affected by these trophic changes.

Summary of Humpback Whales Status

The best available population estimate for humpback whales in the North Atlantic Ocean is estimated as 11,570 animals, and the best, recent estimate for the Gulf of Maine stock is 847 whales (Waring *et al.* 2007). Anthropogenic mortality associated with fishing gear entanglements and ship strikes remains significant. The winter, mating and calving occurs in areas located outside of the United States where the species is afforded less protection. Despite all of these factors, population modeling, using data obtained from photographic mark-recapture studies, estimates the growth rate of the Gulf of Maine stock to be at 6.5% (Barlow and Clapham 1997). Current productivity rates for the North Atlantic population overall are unknown, although Stevick *et al.* (2003) calculated an average population growth rate of 3.1% for the period 1979-1993 (Waring *et al.* 2007). With respect to the species overall, there are also indications of increasing abundance for the eastern and central North Pacific stocks. However, trend and abundance data is lacking for the western North Pacific stock, the Southern Hemisphere humpback whales, and the Southern Indian Ocean humpbacks. Given the best available information, changes in status of the North Atlantic humpback population are, therefore, likely to affect the overall survival and recovery of the species.

Fin Whale

Fin whales inhabit a wide range of latitudes between 20-75° N and 20-75° S (Perry *et al.* 1999). The fin whale is ubiquitous in the North Atlantic and occurs from the Gulf of Mexico and Mediterranean Sea northward to the edges of the arctic ice pack (NMFS 1998a). The overall pattern of fin whale movement is complex, consisting of a less obvious north-south pattern of migration than that of right and humpback whales. Based on acoustic recordings from hydrophone arrays Clark (1995) reported a general southward flow pattern of fin whales in the fall from the Labrador/Newfoundland region, south past Bermuda, and into the West Indes. The overall distribution may be based on prey availability as this species preys opportunistically on both invertebrates and fish (Watkins et al. 1984). Fin whales feed by filtering large volumes of water for the associated prey. Fin whales are larger and faster than humpback and right whales and are less concentrated in nearshore environments.

Within US waters of the Pacific, fin whales are found seasonally off of the coast of North America and Hawaii and in the Bering Sea during the summer (Angliss *et al.* 2001). NMFS recognizes three fin whale stocks in the Pacific for the purposes of managing this species under the MMPA. These are: Alaska (Northeast Pacific), California/Washington/Oregon, and Hawaii (Angliss *et al.* 2001). Reliable estimates of current abundance for the entire Northeast Pacific fin whale stock are not available (Angliss *et al.* 2001). Stock structure for fin whales in the southern hemisphere is unknown. Prior to commercial exploitation, the abundance of southern hemisphere fin whales is estimated to have been at 400,000 (IWC 1979, Perry *et al.* 1999). There are no current estimates of abundance for southern hemisphere fin whales. Since these fin whales do not occur in US waters, there is no recovery plan or stock assessment report for the southern hemisphere fin whales.

NMFS has designated one population of fin whale in US waters of the North Atlantic (Waring *et al.* 1998). This species is commonly found from Cape Hatteras northward. A number of researchers have suggested the existence of fin whale subpopulations in the North Atlantic based on local depletions resulting from commercial overharvesting (Mizroch and York 1984) or genetics data (Bérubé *et al.* 1998). Photoidentification studies in western North Atlantic feeding areas, particularly in Massachusetts Bay, have shown a high rate of annual return by fin whales, both within years and between years (Seipt *et al.* 1990) suggesting some level of site fidelity. In 1976, the IWC's Scientific Committee proposed seven stocks (or populations) for North Atlantic fin whales. These are: (1) North Norway, (2) West Norway-Faroe Islands, (3) British Isles-Spain and Portugal, (4) East Greenland-Iceland, (5) West Greenland, (6) Newfoundland-Labrador, and (7) Nova Scotia (Perry *et al.* 1999). However, it is uncertain whether these boundaries define biologically isolated units (Waring *et al.* 2005).

During 1978-1982 aerial surveys, fin whales accounted for 24% of all cetaceans and 46% of all large cetaceans sighted over the continental shelf between Cape Hatteras and Nova Scotia (Waring *et al.*1998). Underwater listening systems have also demonstrated that the fin whale is the most acoustically common whale species heard in the North Atlantic (Clark 1995). The single most important area for this species appeared to be from the Great South Channel, along the 50m isobath past Cape Cod, over Stellwagen Bank, and past Cape Ann to Jeffrey's Ledge (Hain *et al.*1992).

Like right and humpback whales, fin whales are believed to use North Atlantic waters primarily for feeding, and more southern waters for calving. However, evidence regarding where the majority of fin whales winter, calve, and mate is still scarce. Clark (1995) reported a general pattern of fin whale movements in the fall from the Labrador/Newfoundland region, south past Bermuda and into the West Indies, but neonate strandings along the US Mid-Atlantic coast from October through January suggest the possibility of an offshore calving area (Hain *et al.* 1992).

Fin whales achieve sexual maturity at 5-15 years of age (Perry *et al.* 1999), although physical maturity may not be reached until 20-30 years (Aguilar and Lockyer 1987). Conception is believed to occur during the winter with birth of a single calf after a 12 month gestation (Mizroch and York 1984). The calf is weaned 6-11 months after birth (Perry *et al.* 1999). The mean calving interval is 2.7 years (Agler *et al.* 1993).

The predominant prey of fin whales varies greatly in different geographical areas depending on what is locally available (IWC 1992). In the western North Atlantic, fin whales feed on a variety of small schooling fish (*i.e.*, herring, capelin, sand lance) as well as squid and planktonic crustaceans (Wynne and Schwartz 1999). Fin whales feed by filtering large volumes of water for their prey through their baleen plates.

Threats to fin whale recovery

The major known sources of anthropogenic mortality and injury of fin whales include entanglement in commercial fishing gear and ship strikes. Of 18 fin whale mortality records collected between 1991 and 1995, four were associated with vessel interactions, although the proximal cause of mortality was not known. From 1996-July 2001, there were nine observed fin whale entanglements and at least four ship strikes. From 2000-2004, the NEFSC has confirmed 9 entanglements (3 fatal; 1 serious injury) and 5 ship strikes (all fatal) (Cole et al. 2006). Since 2004, there have been an additional 2 new entanglements and 4 indications of ship strike reported (NMFS unpublished data), although these numbers are awaiting confirmation by the NEFSC. Fin whales are believed to be the most commonly struck cetacean by large vessels (Laist *et al.* 2001). In addition, hunting of fin whales continued well into the 20th century. Fin whales were given total protection in the North Atlantic in 1987 with the exception of a subsistence whaling hunt for Greenland (Gambell 1993, Caulfield 1993). However, Iceland reported a catch of 136 whales in the 1988/89 and 1989/90 seasons, and has since ceased reporting fin whale kills to the IWC (Perry *et al.* 1999). In total, there have been 239 reported kills of fin whales from the North Atlantic from 1988 to 1995. Fin whales may also be adversely affected by habitat degradation, habitat exclusion, acoustic trauma, harassment, or reduction in prey resources due to trophic effects resulting from a variety of activities.

Summary of Fin Whale Status

As noted above, the minimum population estimate for the western North Atlantic fin whale is 2,362 which is believed to be an underestimate. Fishing gear appears to pose less of a threat to fin whales in the North Atlantic Ocean than North Atlantic right or humpback whales. However, more fin whales are struck by large vessels than right or humpback whales (Laist *et al.* 2001). Some level of whaling for fin whales in the North Atlantic may still occur.

Information on the abundance and population structure of fin whales worldwide is limited. NMFS recognizes three fin whale stocks in the Pacific for the purposes of managing this species under the MMPA. Reliable estimates of current abundance for the entire Northeast Pacific fin whale stock are not available (Angliss *et al.* 2001). Stock structure for fin whales in the southern hemisphere is unknown and there are no current estimates of abundance for southern hemisphere fin whales.

Various estimates have been provided to describe the current status of fin whales in western North Atlantic waters. One method used the catch history and trends in Catch Per Unit Effort to obtain an estimate of 3,590 to 6,300 fin whales for the entire western North Atlantic (Perry *et al.* 1999). Hain *et al.* (1992) estimated that about 5,000 fin whales inhabit the Northeastern US continental shelf waters. The 2001 Stock Assessment Report (SAR) gives a best estimate of abundance for fin whales of 2,814 (CV = 0.21). The minimum population estimate for the western North Atlantic fin whale is 2,362 (Waring *et al.* 2001). However, this is considered an underestimate since the estimate was derived from surveys over a limited portion of the western North Atlantic. The 2005 SAR indicates that there are insufficient data at this time to determine population trends for the fin whale.

Status of Sea Turtles

Sea turtles continue to be affected by many factors occurring on the nesting beaches and in the water. Poaching, habitat loss, and nesting predation by introduced species affect hatchlings and nesting females while on land. Fishery interactions, vessel interactions, and (non-fishery) dredging operations, for example, affect sea turtles in the neritic zone (defined as the marine environment extending from mean low water down to 200m (660 foot) depths, generally corresponding to the continental shelf (Lalli and Parsons 1997; Encyclopedia Britannica 2008)). Fishery interactions also affect sea turtles when these species and the fisheries co-occur in the oceanic zone (defined as the open ocean environment where bottom depths are greater than 200m (Lalli and Parsons 1997))2. As a result, sea turtles still face many of the original threats that were the cause of their listing under the ESA.

Sea turtles are listed under the ESA at the species level rather than as subspecies or distinct population segments (DPS). Therefore, information on the range-wide status of each species is included to provide the reader with information on the status of each species, overall. Additional background information on the range-wide status of these species can be found in a number of published documents, including sea turtle status reviews and biological reports (NMFS and USFWS 1995; Hirth 1997; USFWS 1997; Marine Turtle Expert Working Group (TEWG) 1998; TEWG 2000; NMFS and USFWS 2007a; 2007b; 2007c; 2007d; Leatherback TEWG 2007), and

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2 As described in Bolten (2003), oceanographic terms have frequently been used incorrectly to describe sea turtle life stages. In turtle literature the terms benthic and pelagic were used incorrectly to refer to the neritic and oceanic zones, respectively. The term benthic refers to occurring on the bottom of a body of water, whereas the term pelagic refers to in the water column. Turtles can be "benthic" or pelagic" in either the neritic or oceanic zones.

recovery plans for the loggerhead sea turtle (NMFS and USFWS 1991a), leatherback sea turtle (NMFS and USFWS 1992; NMFS and USFWS 1998a;), Kemp's ridley sea turtle (USFWS and NMFS 1992), and green sea turtle (NMFS and USFWS 1991b; NMFS and USFWS 1998b).

Loggerhead sea turtle

Loggerhead sea turtles are a cosmopolitan species, found in temperate and subtropical waters. Loggerhead sea turtles are the most abundant species of sea turtle in U.S. waters.

Pacific Ocean. In the Pacific Ocean, major loggerhead nesting grounds are generally located in temperate and subtropical regions with scattered nesting in the tropics. The abundance of loggerhead turtles on nesting colonies throughout the Pacific basin have declined dramatically over the past 10-20 years. Loggerhead sea turtles in the Pacific are represented by a northwestern Pacific nesting group (located in Japan) and a smaller southwestern nesting group that occurs in Australia (Great Barrier Reef and Queensland), New Caledonia, New Zealand, Indonesia, and Papua New Guinea. Data from 1995 estimated the Japanese nesting group at 1,000 female loggerhead turtles (Bolten *et al.* 1996). More recent information suggests that nest numbers have increased somewhat over the period 1998-2004 (NMFS and USFWS 2007a). However, this time period is too short to make a determination of the overall trend in nesting (NMFS and USFWS 2007a). Genetic analyses of loggerhead females nesting in Japan indicates the presence of genetically distinct nesting colonies (Hatase *et al.* 2002).

In Australia, long-term census data has been collected at some rookeries since the late 1960's and early 1970's, and nearly all the data show marked declines in nesting since the mid-1980's (Limpus and Limpus 2003). The nesting group in Queensland, Australia, was as low as 300 females in 1997.

Pacific loggerhead turtles are captured, injured, or killed in numerous Pacific fisheries including gillnet, longline, and trawl fisheries in the western and/or eastern Pacific Ocean (NMFS and USFWS 2007a). In Australia, where turtles are taken in bottom trawl and longline fisheries, efforts have been made to reduce fishery bycatch (NMFS and USFWS 2007a).

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

In the southwestern Indian Ocean, loggerhead nesting has shown signs of recovery in South Africa where protection measures have been in place for decades. However, in other southwestern areas (*e.g.*, Madagascar and Mozambique) loggerhead nesting groups are still affected by subsistence hunting of adults and eggs (Baldwin *et al.* 2003). The largest known nesting group of loggerheads in the world occurs in Oman in the northern Indian Ocean. An estimated 20,000-40,000 females nest at Masirah, the largest nesting site within Oman, each year (Baldwin *et al.* 2003). In the eastern Indian ocean, all known nesting sites are found in Western

Australia (Dodd 1988). As has been found in other areas, nesting numbers are disproportionate within the area with the majority of nesting occurring at a single location. This may, however, be the result of fox predation on eggs at other Western Australia nesting sites (Baldwin *et al.* 2003).

Mediterranean Sea. Nesting in the Mediterranean is confined almost exclusively to the eastern basin (Margaritoulis *et al.* 2003). The greatest number of nests in the Mediterranean are found in Greece with an average of 3,050 nests per year (Margaritoulis *et al.* 2003; NMFS and USFWS 2007a). Turkey has the second largest number of nests with 2,000 nest per year(NMFS and USFWS 2007a). There is a long history of exploitation for loggerheads in the Mediterranean (Margaritoulis *et al.* 2003). Although much of this is now prohibited, some directed take still occurs (Margaritoulis *et al.* 2003). Loggerheads in the Mediterranean also face the threat of habitat degradation, incidental fishery interactions, vessel strikes, and marine pollution (Margaritoulis *et al.* 2003). Longline fisheries, in particular, are believed to catch thousands of juvenile loggerheads each year (NMFS and USFWS 2007a), although genetic analyses indicate that only a portion of the loggerheads captured originate from loggerhead nesting groups in the Mediterranean (Laurent *et al.* 1998).

Atlantic Ocean. Ehrhart *et al.* (2003) provided a summary of the literature identifying known nesting habitat of Atlantic loggerheads as well as known foraging areas within the Atlantic. Information is also provided in the 5-year status review (NMFS and USFWS 2007a). Briefly, nesting occurs on island and mainland beaches on both sides of the Atlantic and both north and south of the Equator (Ehrhart *et al.* 2003). By far, the majority of nesting occurs on beaches of the southeastern U.S. (NMFS and USFWS 2007a). Annual nest counts for loggerhead sea turtles on beaches from other countries are in the hundreds with the exception of Brazil where a total of 4,837 nests were reported for the 2003/2004 nesting season (Marcovaldi and Chaloupka 2007; NMFS and USFWS 2007a). In both the eastern and western Atlantic, waters as far north as 41°-42°N latitude are used for foraging by juveniles as well as adults (Shoop 1987; Shoop and Kenney 1992; Ehrhart *et al.* 2003; Mitchell *et al.* 2003). Of these, loggerheads that nest and/or forage in U.S. waters of the western Atlantic have been most extensively studied.

Loggerheads commonly occur throughout the inner continental shelf from Florida through Cape Cod, Massachusetts although their presence varies with the seasons due to changes in water temperature (Shoop and Kenney 1992; Epperly *et al.* 1995a; Epperly *et al.* 1995b; Braun and Epperly 1996; Epperly and Braun-McNeill 2002; Mitchell *et al.* 2003). Aerial surveys of continental shelf waters north of Cape Hatteras indicate that loggerhead sea turtles are most commonly sighted in waters with bottom depths ranging from 22 to 49 meters deep (Shoop and Kenney 1992). However, survey and satellite tracking data support that they occur in waters from the beach to beyond the continental shelf (Mitchell *et al.* 2003; Braun-McNeill and Epperly 2004; Blumenthal *et al.* 2006; Hawkes *et al.* 2006; McClellan and Read 2007). The presence of loggerhead turtles in an area is also influenced by water temperature. Loggerheads have been observed in waters with surface temperatures of 7-30°C but water temperatures of ≥11°C are favorable to sea turtles (Shoop and Kenney 1992; Epperly *et al.* 1995b).

In the western North Atlantic, loggerhead sea turtles occur year round in offshore waters off of North Carolina where water temperature is influenced by the Gulf Stream. As coastal water temperatures warm in the spring, loggerheads begin to migrate to North Carolina inshore waters (*e.g.*, Pamlico and Core Sounds) and also move up the coast (Epperly *et al.* 1995a; Epperly *et al.* 1995b; Epperly *et al.* 1995c; Braun-McNeill and Epperly 2004), occurring in Virginia foraging areas as early as April and on the most northern foraging grounds in the Gulf of Maine in June (Shoop and Kenney 1992). The trend is reversed in the fall as water temperatures cool. The large majority leave the Gulf of Maine by mid-September but some may remain in Mid-Atlantic and Northeast areas until late Fall. By December loggerheads have migrated from inshore North Carolina waters and more northern coastal waters to waters offshore of North Carolina, particularly off of Cape Hatteras, and waters further south where the influence of the Gulf Stream provides temperatures favorable to sea turtles (Shoop and Kenney 1992; Epperly *et al.* 1995b; Epperly and Braun-McNeill 2002).

Loggerheads mate in late March-early June, and eggs are laid throughout the summer, with a mean clutch size of 100-126 eggs in the southeastern United States. Individual females nest multiple times during a nesting season, with a mean of 4.1 nests/individual (Murphy and Hopkins 1984). Nesting migrations for an individual female loggerhead are usually on an interval of 2-3 years, but can vary from 1-7 years (Dodd 1988).

The scientific literature for loggerhead sea turtles recognizes five nesting groups in the western North Atlantic, divided geographically as follows: (1) a northern group of nesting females that nest from North Carolina to northeast Florida at about 29° N latitude; (2) a south Florida group of nesting females that nest from 29° N latitude on the east coast to Sarasota on the west coast; (3) a Florida Panhandle group of nesting females that nest around Eglin Air Force Base and the beaches near Panama City, Florida; (4) a Yucatán group of nesting females that nest on beaches of the eastern Yucatán Peninsula, Mexico (Márquez 1990; TEWG 2000); and (5) a Dry Tortugas group that nest on beaches of the islands of the Dry Tortugas, near Key West, Florida (NMFS SEFSC 2001). Genetic analyses of mitochondrial DNA, which a turtle inherits from its mother, indicate that there are genetic differences between turtles that nest at and originate from the beaches used by each of the five identified nesting groups of females (TEWG 2000). However, analyses of microsatellite loci from nuclear DNA, which represents the genetic contribution from both parents, indicates little to no genetic differences between turtles originating from nesting beaches of the five western North Atlantic loggerhead nesting groups (Pearce and Bowen 2001; Bowen 2003; Bowen *et al.* 2005; Shamblin 2007). These results suggest that female loggerheads have site fidelity to nesting beaches within a particular area, while males provide an avenue of gene flow between nesting groups by mating with females that originate from different nesting groups (Bowen 2003; Bowen *et al.* 2005). The extent of such gene flow, however, is unclear (Shamblin 2007).

In the western Atlantic, most loggerhead sea turtles nest from North Carolina to Florida and along the Gulf coast of Florida. In 1989, a statewide sea turtle Index Nesting Beach Survey (INBS) program was developed and implemented in Florida. There are currently 33 nesting beaches in the INBS program (letter to NMFS from the Director, Fish and Wildlife Research

Institute, Florida Fish and Wildlife Conservation Commission, October 25, 2006). As of 2006, 27 of the 33 beaches had reached the mandatory minimum of 10-years participation for their nest count data to be included in trend evaluations (letter to NMFS from the Director, Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission, October 25, 2006). Nesting recorded by the INBS program on the 27 beaches represented an average of 65% of all annual nesting by loggerheads in the state for the period 2001-2005 (letter to NMFS from the Director, Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission, October 25, 2006). Standardized nesting beach survey programs have been implemented in Georgia, South Carolina, and North Carolina as well (Dodd 2003; USFWS and NMFS 2003). A near complete census of the Dry Tortugas nesting beaches were conducted from 1995 – 2004 (excluding 2002). However, no trend in the number of nests laid was detected for the time period and no surveys have been conducted since 2004 (NMFS and USFWS 2007a). Survey effort to counts nests for loggerhead nesting beaches of the Yucatán, Mexico, was consistent from 1987-2001 for seven beaches in Quintana Roo, Mexico (NMFS and USFWS 2007a). However, nesting survey effort overall has been inconsistent among the Yucatán nesting beaches (Zurita *et al.* 2003).

Sea turtle nesting survey data is important in that it provides information on the relative abundance of nesting each year, and the contribution of each nesting group to total nesting of the species. Nest counts can also be used to estimate the number of reproductively mature females nesting annually. The 5-year review for loggerhead sea turtles (NMFS and USFWS 2007a) compiled the most recent information on mean number of loggerhead nests per year, and, where available, the approximated counts of nesting females for each of the five identified western north Atlantic loggerhead nesting groups. These are: (1) For the south Florida nesting group, a mean of 65,460 loggerhead nests per year with approximately 15,966 females nesting per year; (2) for the northern nesting group, a mean of 5,151 nests per year (no estimate of number of females nesting per year provided); (3) for the Florida panhandle nesting group, a mean of 910 nests per year with approximately 222 females nesting per year; (4) for the Dry Tortugas nesting group, a mean of 246 nests per year with approximately 60 females nesting per year; and (5) for the Yucatán nesting group, a range of 903-2,231 nests per year from 1987-2001 (no estimate of number of nesting females provided) (NMFS and USFWS 2007a). As is evident from this information, nests for the south Florida nesting group make up the majority of all loggerhead nests counted along the U.S. Atlantic and Gulf coasts and represents the largest known loggerhead nesting group (in terms of number of nesting females) in the Atlantic (USFWS and NMFS 2003; NMFS and USFWS 2007a). The northern nesting group is the second largest for loggerheads within the United States but smaller than the south Florida nesting group. The remaining three nesting groups (the Dry Tortugas, Florida Panhandle, and Yucatán) are, again, much smaller in terms of the number of nests laid and the estimated number of females laying nests.

In 2006, information was presented at an international sea turtle symposium (Meylan *et al.* 2006) and in a letter to NMFS (letter to NMFS from the Director, Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission, October 25, 2006) that a trend analysis of the nesting data collected for Florida's INBS program showed a decrease in nesting of 22.3% in the annual nest density of surveyed shoreline over the 17-year period and a 39.5% decline since 1998 (letter to NMFS from the Director, Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission, October 25, 2006). Data collected in Florida for the 2007 loggerhead nesting season reveals that the decline in nest numbers has continued, with even fewer nests counted in 2007 in comparison to any previous year of the period, 1989-2007 (Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission web posting November 2007). Declines in nesting have been noted for some of the other western Atlantic loggerhead nesting groups as well. Standardized ground surveys of 11 North Carolina, South Carolina, and Georgia nesting beaches showed a significant declining trend of 1.9% annually in loggerhead nesting from 1983-2005 (NMFS and USFWS 2007a). Aerial surveys conducted by the South Carolina Department of Natural Resources showed a 3.1% annual decline in nesting since 1980 (Dodd 2003; NMFS and USFWS 2007a). The South Carolina data represents approximately 59% of nesting by the northern nesting group (Dodd 2003). A significant declining trend (P=0.04) in loggerhead nesting of 6.8% annually from 1995-2005 has also been detected for the Florida Panhandle nesting group (NMFS and USFWS 2007a). Nesting for the Yucatán nesting group is characterized as having declined since 2001 (NMFS and USFWS 2007a) while no trend is detectable for the Dry Tortugas nesting group (NMFS and USFWS 2007a).

Unlike nesting beach data, in-water studies of sea turtles typically sample both sexes and multiple age classes. In-water studies have been conducted in some areas of the western Atlantic and provide data by which to assess the relative abundance of loggerhead sea turtles and changes in abundance over time (Maier *et al.* 2004; Morreale *et al.* 2004; Mansfield 2006; Ehrhart *et al.* 2007; Epperly *et al.* 2007). Maier *et al.* (2004) used fishery-independent trawl data to establish a regional index of loggerhead abundance for the southeast coast of the United States (Winyah Bay, South Carolina to St. Augustine, FL) during the period 2000 – 2003. A comparison of loggerhead catch data from this study with historical values suggested that in-water populations of loggerhead sea turtles along the southeastern United States appear to be larger, possibly an order of magnitude higher than they were 25 years ago (Maier *et al.* 2004). A comparison of catch rates for sea turtles in pound net gear fished in the Pamlico-Albemarle Estuarine Complex of North Carolina between the years 1995-1997 and 2001-2003 similarly found a significant increase in catch rates for loggerhead sea turtles for the latter period (Epperly *et al.* 2007). A long-term, on-going, study of loggerhead abundance in the Indian River Lagoon System of Florida found a significant increase in the relative abundance of loggerheads over the last 4 years of the study (Ehrhart *et al.* 2007). However, there was no discernible trend in loggerhead abundance during the 24-year time period of the study (1982-2006) (Ehrhart *et al.* 2007). In contrast to these studies, Morreale *et al.* (2004) observed a decline in the incidental catch of loggerhead sea turtles in pound net gear fished around Long Island, NY, during the period 2002-2004 in comparison to the period 1987-1992, with only two loggerhead sea turtles observed captured in pound net gear during the period 2002-2004. No additional loggerheads were reported captured in pound net gear through 2007, although 2 loggerhead sea turtles were found cold-stunned on Long Island bay beaches in the fall of 2007 (Memo to the File, L. Lankshear, December 2007). Using aerial surveys, Mansfield (2006) also found a decline in the densities of loggerhead sea turtles in Chesapeake Bay over the period 2001-2004 compared to aerial survey data collected in the 1980's. Significantly fewer turtles ($p<0.05$) were observed in both the spring (May-June) and the summer (July-August) of 2001-2004 compared to aerial surveys in the 1980's (Mansfield 2006). A comparison of median densities from the 1980's to the 2000's suggested that there had been a 63.2% reduction in densities during the spring residency period and a 74.9% reduction in densities during the summer residency period (Mansfield 2006).

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

The 5-year status review of loggerhead sea turtles recently completed by NMFS and the USFWS provides a summary of natural as well as anthropogenic threats to loggerhead sea turtles (NMFS and USFWS 2007a). Amongst those of natural origin, hurricanes are known to be destructive to sea turtle nests. Sand accretion and rainfall that result from these storms as well as wave action can appreciably reduce hatchling success. Other sources of natural mortality include cold stunning and biotoxin exposure.

Anthropogenic factors that impact hatchlings and adult female turtles on land, or the success of nesting and hatching include: beach erosion, beach armoring and nourishment; artificial lighting; beach cleaning; increased human presence; recreational beach equipment; beach driving; coastal construction and fishing piers; exotic dune and beach vegetation; and poaching. An increased human presence at some nesting beaches or close to nesting beaches has led to secondary threats such as the introduction of exotic fire ants, feral hogs, dogs and an increased presence of native species (*e.g.*, raccoons, armadillos, and opossums) which raid and feed on turtle eggs (NMFS and USFWS 2007a). Although sea turtle nesting beaches are protected along large expanses of the northwest Atlantic coast (in areas like Merritt Island, Archie Carr, and Hobe Sound National Wildlife Refuges), other areas along these coasts have limited or no protection. Sea turtle nesting and hatching success on unprotected high density east Florida nesting beaches from Indian River to Broward County are affected by all of the above threats.

Sea turtles, including loggerhead sea turtles, are affected by a completely different set of anthropogenic threats in the marine environment. These include oil and gas exploration, coastal development, and transportation; marine pollution; underwater explosions; hopper dredging; offshore artificial lighting; power plant entrainment and/or impingement; entanglement in debris; ingestion of marine debris; marina and dock construction and operation; boat collisions; poaching, and fishery interactions.

A 1990 National Research Council report concluded that for juvenile, subadults, and breeders in coastal waters, the most important source of human caused mortality in U.S. Atlantic waters was fishery interactions. Of these, the U.S. south Atlantic and Gulf of Mexico shrimp fisheries were considered to pose the greatest cause of mortality to neritic juvenile and adult age classes of loggerheads accounting for an estimated 5,000 – 50,000 loggerheads deaths each year (NRC 1990). Significant changes to the south Atlantic and Gulf of Mexico shrimp fisheries have occurred since 1990, and the effects of these shrimp fisheries on ESA-listed species, including loggerhead sea turtles, have been assessed several times through section 7 consultation. There is also a lengthy regulatory history with regard to the use of Turtle Excluder Devices (TEDs) in the U.S. south Atlantic and Gulf of Mexico shrimp fisheries (Epperly and Teas 2002; NMFS 2002; Lewison *et al.* 2003). Section 7 consultation was reinitiated in 2002 to, in part, consider the effect of a new rulemaking that would require increasing the size of TED escape openings to allow larger loggerheads (and green sea turtles) to escape from shrimp trawl gear. The resulting Opinion was completed in December 2002 and concluded that, as a result of the new rule, annual loggerhead mortality from capture in shrimp trawls would decline from an estimated 62,294 to 3,947 turtles assuming that all TEDs were installed properly and that compliance was 100% (Epperly *et al.* 2002; NMFS 2002). The total level of take (capture) for loggerhead sea turtles as a result of the U.S. south Atlantic and Gulf of Mexico shrimp fisheries was estimated to be 163,160 loggerheads per year (NMFS 2002) with up to 3,948 mortalities. On February 21, 2003, NMFS issued the final rule to require the use of the larger opening TED (68 FR 8456). The rule also provided the measures to disallow several previously approved TED designs that did not function properly under normal fishing conditions, and to require modifications to the trynet and bait shrimp exemptions to the TED requirements to decrease mortality of sea turtles.

The NRC report (1990) also stated that other U.S. Atlantic fisheries collectively accounted for 500-5,000 loggerhead deaths each year, but recognized that there was considerable uncertainty in the estimate. Subsequent studies suggest that these numbers were underestimated. For example, the first estimate of loggerhead sea turtle bycatch in U.S. Mid-Atlantic bottom otter trawl gear was completed in September 2006 (Murray 2006). Observers reported 66 loggerhead turtle interactions with bottom otter trawl gear during the period of which 38 were reported as alive and uninjured and 28 were reported as dead, injured, resuscitated, or of unknown condition (Murray 2006). Seventy-seven percent of observed turtle interactions occurred on vessels fishing for summer flounder (50%) and croaker (27%). The remaining 23% of observed takes occurred on vessels targeting weakfish (11%), long-finned squid (8%), groundfish (3%) and short-finned squid (1%) (Murray 2006). Based on observed interactions and fishing effort as reported on VTRs, the average annual loggerhead bycatch in these bottom-otter trawl fisheries combined was estimated to be 616 sea turtles for each year of the period 1996-2004 (Murray 2006).

The U.S. tuna and swordfish longline fisheries that are managed under the Highly Migratory Species Fishery Management Plan (HMS FMP), were estimated to capture 1,905 loggerheads (no more than 339 mortalities) for each 3-year period (NMFS 2004c). NMFS has mandated gear changes for the HMS fishery to reduce turtle bycatch and the likelihood of death from those takes that would still occur (Fairfield-Walsh and Garrison 2007). In 2006, there were 46 observed interactions between loggerhead sea turtles and longline gear used in the HMS fishery. Nearly all of the loggerheads (42 of 46) were released alive but with injuries (Fairfield-Walsh and Garrison 2007). The majority of the injured had been hooked internally (Fairfield-Walsh and Garrison 2007). Based on the observed take, an estimated 561 (range = 318-981) loggerhead sea turtles are estimated to have been taken (hooked and released live, with injuries) in the longline fisheries managed under the HMS FMP in 2006 (Fairfield-Walsh and Garrison 2007). This number is an increase from 2005 when 274 loggerheads were estimated to have been taken in the fisheries but is still lower than some previous years in the period of 1992-2006 (Fairfield-Walsh and Garrison 2007). This fishery represents just one of several longline fisheries operating in the Atlantic. Lewison *et al.* (2004) estimated that 150,000-200,000 loggerheads were taken (capture) in the Atlantic longline fisheries in 2000 (includes the U.S. Atlantic tuna and swordfish longline fisheries as well as others).

Summary of Status for Loggerhead Sea Turtles

Loggerheads are a long-lived species and reach sexual maturity relatively late; 20-38 years (NMFS SEFSC 2001). Loggerhead sea turtles are injured and killed by numerous human activities (NRC 1990; NMFS and USFWS 2007a). There are no population estimates for loggerhead sea turtles in any of the ocean basins in which they occur.

Genetic differences exist between turtles that nest and forage in the different ocean basins (Bowen 2003; Bowen and Karl 2007). Differences in the maternally inherited mitochondrial DNA also exist between loggerhead nesting groups that occur within the same ocean basin (TEWG 2000; Pearce 2001; Bowen 2003; Bowen *et al.* 2005; Shamblin 2007). Site fidelity of females to one or more nesting beaches in an area is believed to account for these genetic differences (TEWG 2000; Bowen 2003). Based on the most recent information, a decline in the annual nest counts has been measured or suggested for four of five western Atlantic loggerhead nesting groups. These include the south Florida nesting group which is the largest (in terms of number of nests laid) in the Atlantic.

NMFS has also convened a new loggerhead TEWG to review all available information on Atlantic loggerheads in order to determine what can be said about the status of this species in the Atlantic. A final report from the Loggerhead TEWG is not yet available. An interim update was provided by the Loggerhead TEWG to NMFS in December 2007. In summary, the memo stated that nest counts, fishery dependent data, and stranding data do not provide the necessary insight into loggerhead turtle population dynamics to properly assess species status (Loggerhead TEWG 2007). As has been stated in the literature (Meylan 1982; Ross 1996; Zurita *et al.* 2003; Hawkes *et al.* 2005), the TEWG remarked that nest counts alone provide no insight into the trend/abundance of sexually mature males or of other age classes of either sex (Loggerhead

TEWG 2007). In addition, the TEWG stated that interpreting the meaning of a decline in nest counts in terms of the status/trend of the number of nesting females in the population is difficult since converting nest counts to the number of nesting females is confounded by several issues such as variability in the number of nests per female per year; variability in remigration interval; and, as the ability to nest is resource dependent, the effect of habitat changes and the availability of food resources (Loggerhead TEWG 2007). The TEWG is continuing to explore several hypotheses for why nest counts have been declining. These hypotheses will be more fully discussed in the final report (Loggerhead TEWG 2007).

Leatherback sea turtle

Leatherback sea turtles are widely distributed throughout the oceans of the world, and are found in waters of the Atlantic and Pacific Oceans, the Caribbean Sea, and the Gulf of Mexico (Ernst and Barbour 1972). Leatherback sea turtles are the largest living turtles and range farther than any other sea turtles species; their large size and tolerance of relatively low temperatures allows them to occur in northern waters such as off Labrador and in the Barents Sea (NMFS and USFWS 1995).

In 1980, the leatherback population was estimated at approximately 115,000 adult females globally (Pritchard 1982). By 1995, this global population of adult females was estimated to have declined to 34,500 (Spotila *et al.* 1996). However, the most recent population size estimate for the North Atlantic alone is a range of 34,000-94,000 adult leatherbacks (Leatherback TEWG 2007). Thus, there is uncertainty with respect to global population estimates of leatherback sea turtles.

Pacific Ocean. Leatherback nesting has been declining at all major Pacific basin nesting beaches for the last two decades (Spotila *et al.* 1996; NMFS and USFWS 1998a; Sarti *et al.* 2000; Spotila *et al.* 2000). Leatherback turtles disappeared from India before 1930, have been virtually extinct in Sri Lanka since 1994, and appear to be approaching extinction in Malaysia (Spotila *et al.* 2000). For example, the nesting group on Terengganu (Malaysia) - which was one of the most significant nesting sites in the western Pacific Ocean - has declined severely from an estimated 3,103 females in 1968 to 2 nesting females in 1994 (Chan and Liew 1996). Nesting groups of leatherback turtles along the coasts of the Solomon Islands, which historically supported important nesting groups, are also reported to be declining (D. Broderick, pers. comm., *in* Dutton *et al.* 1999). In Fiji, Thailand, Australia, and Papua-New Guinea (East Papua), leatherback turtles have only been known to nest in low densities and scattered colonies.

Only an Indonesian nesting group has remained relatively abundant in the Pacific basin. The largest, extant leatherback nesting group in the Indo-Pacific lies on the north Vogelkop coast of Irian Jaya (West Papua), Indonesia, with over 1,000 nesting females during the 1996 season (Suarez *et al.* 2000). During the early-to-mid 1980s, the number of female leatherback turtles nesting on the two primary beaches of Irian Jaya appeared to be stable. However, in 1999, for example, local Indonesian villagers started reporting dramatic declines in sea turtles near their villages (Suarez 1999). Declines in nesting groups have been reported throughout the western

Pacific region where observers report that nesting groups are well below abundance levels that were observed several decades ago (*e.g.*, Suarez 1999).

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

In the eastern Pacific Ocean, leatherback nesting is declining along the Pacific coast of Mexico and Costa Rica. According to reports from the late 1970s and early 1980s, three beaches located on the Pacific coast of Mexico support as many as half of all leatherback turtle nests. Since the early 1980s, the eastern Pacific Mexican population of adult female leatherback turtles has declined to slightly more than 200 during 1998-99 and 1999-2000 (Sarti *et al.* 2000). Spotila *et al.* (2000) reported the decline of the leatherback nesting at Playa Grande, Costa Rica, which had been the fourth largest nesting group in the world. Between 1988 and 1999, the nesting group declined from 1,367 to 117 female leatherback turtles. Based on their models, Spotila *et al.* (2000) estimated that the group could fall to less than 50 females by 2003-2004. An analysis of the Costa Rican nesting beaches indicates a decline in nesting during the past 15 years of monitoring (1989-2004) with approximately 1,504 females nesting in 1988-89 to an average of 188 females nesting in 2000-2001 and 2003-2004 (NMFS and USFWS 2007b). A similar dramatic decline has been seen on nesting beaches in Pacific Mexico, where tens of thousands of leatherback nests were laid on the beaches in the 1980s but where a total of only 120 nests on the four primary index beaches (combined) were counted in the 2003-2004 season.

Commercial and artisanal swordfish fisheries off Chile, Columbia, Ecuador, and Peru, purse seine fisheries for tuna in the eastern tropical Pacific Ocean, and California/Oregon drift gillnet fisheries are known to capture, injure or kill leatherback turtles in the eastern Pacific Ocean. Given the declines in leatherback nesting in the Pacific, some researchers have concluded that the leatherback is on the verge of extinction in the Pacific Ocean (*e.g.*, Spotila *et al.* 1996; Spotila *et al.* 2000).

Indian Ocean. Leatherbacks nest in several areas around the Indian Ocean. These sites include Tongaland, South Africa (Pritchard 2002), and the Andaman and Nicobar Islands (Andrews *et al.* 2002). Intensive survey and tagging work in 2001 provided new information on the level of nesting in the Andaman and Nicobar Islands (Andrews *et al.* 2002). Based on the survey and tagging work, it was estimated that 400-500 female leatherbacks nest annually on Great Nicobar Island (Andrews *et al.* 2002). The number of nesting females using the Andaman and Nicobar Islands combined was estimated around 1000 (Andrews and Shanker 2002). Some nesting also occurs along the coast of Sri Lanka although in much smaller numbers than in the past (Pritchard 2002).

Atlantic Ocean. Evidence from tag returns and strandings in the western Atlantic suggests that adult leatherback sea turtles engage in routine migrations between boreal, temperate and tropical waters (NMFS and USFWS 1992). Leatherbacks are frequently thought of as a pelagic species that feed on jellyfish (*i.e.*, *Stomolophus, Chryaora*, and *Aurelia* (Rebel 1974)), and tunicates (salps, pyrosomas) in oceanic habitat. However, leatherbacks are also known to use coastal waters of the U.S. continental shelf (James *et al.* 2005b; Eckert *et al.* 2006; Murphy *et al.* 2006) as well as the European continental shelf on a seasonal basis (Witt *et al.* 2007).

A 1979 aerial survey of the outer Continental Shelf from Cape Hatteras, North Carolina to Cape Sable, Nova Scotia showed leatherbacks to be present throughout the area with the most numerous sightings made from the Gulf of Maine south to Long Island. Leatherbacks were sighted in water depths ranging from 1-4151m but 84.4% of sightings were in waters less than 180 m (Shoop and Kenney 1992). Leatherbacks were sighted in waters within a sea surface temperature range similar to that observed for loggerheads; from 7-27.2°C (Shoop and Kenney 1992). However, leatherbacks appear to have a greater tolerance for colder waters in comparison to loggerhead sea turtles since more leatherbacks were found at the lower temperatures as compared to loggerheads (Shoop and Kenney 1992). This aerial survey estimated the leatherback population for the northeastern U.S. at approximately 300-600 animals (from near Nova Scotia, Canada to Cape Hatteras, North Carolina). However, the estimate was based on turtles visible at the surface and does not include those that were below the surface out of view. Therefore, it likely underestimates the leatherback population for the northeastern U.S. Estimates of leatherback abundance of 1,052 turtles (C.V.= 0.38) and 1,174 turtles (C.V.= 0.52) were obtained from surveys conducted from Virginia to the Gulf of St. Lawrence in 1995 and 1998, respectively (Palka 2000). However, since these estimates were also based on sightings of leatherbacks at the surface, the author considered the estimates to be negatively biased and the true abundance of leatherbacks may be 4.27 times the estimates (Palka 2000). Studies of satellite tagged leatherbacks suggest that they spend a 10% - 41% of their time at the surface, depending on the phase of their migratory cycle (James *et al.* 2005a). The greatest amount of surface time (up to 41%) was recorded when leatherbacks occurred in continental shelf and slope waters north of 38° N (James *et al.* 2005a).

Leatherbacks are a long lived species (> 30 years). They mature at a younger age than loggerhead turtles, with an estimated age at sexual maturity of about 13-14 years for females with 9 years reported as a likely minimum (Zug and Parham 1996) and 19 years as a likely maximum (NMFS SEFSC 2001). In the U.S. and Caribbean, female leatherbacks nest from March through July. They nest frequently (up to 7 nests per year) during a nesting season and nest about every 2-3 years. During each nesting, they produce 100 eggs or more in each clutch and can produce 700 eggs or more per nesting season (Schultz 1975). However, a significant portion (up to approximately 30%) of the eggs can be infertile. Therefore, the actual proportion of eggs that can result in hatchlings is less than this seasonal estimate. As is the case with other sea turtle species, leatherback hatchlings enter the water soon after hatching. Based on a review of all sightings of leatherback sea turtles of <145 cm (56.55 in) curved carapace length (CCL), Eckert (1999) found that leatherback juveniles remain in waters warmer than 26° C until they exceed 100 cm (39 in) CCL.

As described in Section 3.1.1, sea turtle nesting survey data is important in that it provides information on the relative abundance of nesting, and the contribution of each population/subpopulation to total nesting of the species. Nest counts can also be used to estimate the number of reproductively mature females nesting annually, and as an indicator of the trend in the number of nesting females in the nesting group. The 5-year review for leatherback sea turtles (NMFS and USFWS 2007b) compiled the most recent information on mean number of leatherback nests per year for each of the seven leatherback populations or groups of populations that were identified by the Leatherback TEWG as occurring within the Atlantic. These are: Florida, North Caribbean, Western Caribbean, Southern Caribbean, West Africa, South Africa, and Brazil. In the U.S., the Florida Statewide Nesting Beach Survey program has documented an increase in leatherback nesting numbers from 98 nests in 1988 to between 800 and 900 nests in the early 2000s (NMFS and USFWS 2007b). An analysis of Florida's Index Nesting Beach Survey sites from 1989-2006 shows a substantial increase in leatherback nesting in Florida during this time, with an annual growth rate of approximately 1.17 (Leatherback TEWG 2007). The TEWG reports an increasing or stable trend for all of the seven populations or groups of populations with the exception of the Western Caribbean and West Africa. However, caution is also warranted even for those that were identified as stable or increasing. In St. Croix, for example, researchers have noted a declining presence of neophytes (first-time nesters) since 2002 (Garner *et al.* 2007). In addition, the leatherback rookery along the northern coast of South America in French Guiana and Suriname supports the majority of leatherback nesting in the western Atlantic (Leatherback TEWG 2007), and represents more than half of total nesting by leatherback sea turtles world-wide (Hilterman and Goverse 2004). Nest numbers in Suriname have shown an increase and the long-term trend for the Suriname and French Guiana nesting group seems to show an increase (Hilterman and Goverse 2004). In 2001, the number of nests for Suriname and French Guiana combined was 60,000, one of the highest numbers observed for this region in 35 years (Hilterman and Goverse 2004). The most recent Leatherback TEWG report (2007) indicates that using nest numbers from 1967-2005, a positive population growth rate was found over the 39-year period for French Guinea and Suriname, with a 95% probability that the population was growing. Nevertheless, given the magnitude of leatherback nesting in this area compared to other nest sites, impacts to this area that negatively impact leatherback sea turtles could have profound impacts on the species, overall.

Tag return data demonstrate that leatherbacks that nest in South America also use U.S. waters. A nesting female tagged May 29, 1990, in French Guiana was later recovered and released alive from the York River, VA. Another nester tagged in French Guiana was later found dead in Palm Beach, Florida (STSSN database). Many other examples also exist. For example, leatherbacks tagged at nesting beaches in Costa Rica have been found in Texas, Florida, South Carolina, Delaware, and New York (STSSN database). Leatherback turtles tagged in Puerto Rico, Trinidad, and the Virgin Islands have also been subsequently found on U.S. beaches of southern, Mid-Atlantic and northern states (STSSN database).

Of the Atlantic turtle species, leatherbacks seem to be the most vulnerable to entanglement in fishing gear. This susceptibility may be the result of their body type (large size, long pectoral flippers, and lack of a hard shell), and their attraction to gelatinous organisms and algae that collect on buoys and buoy lines at or near the surface, and perhaps to the lightsticks used to attract target species in longline fisheries. They are also susceptible to entanglement in gillnets (used in various fisheries) and capture in trawl gear (*e.g.*, shrimp trawls, bottom otter trawls). Sea turtles entangled in fishing gear generally have a reduced ability to feed, dive, surface to breathe or perform any other behavior essential to survival (Balazs 1985). In addition to drowning from forced submergence, they may be more susceptible to boat strikes if forced to remain at the surface, and entangling lines can constrict blood flow resulting in tissue necrosis.

Leatherbacks are exposed to pelagic longline fisheries in many areas of their range. According to observer records, an estimated 6,363 leatherback sea turtles were caught by the U.S. Atlantic tuna and swordfish longline fisheries between 1992-1999, of which 88 were released dead (NMFS SEFSC 2001). Since the U.S. fleet accounts for only 5-8% of the hooks fished in the Atlantic Ocean, adding up the under-represented observed takes of the other 23 countries actively fishing in the area would likely result in annual take estimates of thousands of leatherbacks over different life stages (NMFS SEFSC 2001).

Leatherbacks are susceptible to entanglement in the lines associated with trap/pot gear used in several fisheries. From 1990-2000, 92 entangled leatherbacks were reported from New York through Maine (Dwyer *et al.* 2002). Additional leatherbacks stranded wrapped in line of unknown origin or with evidence of a past entanglement (Dwyer *et al.* 2002). A review of leatherback mortality documented by the STSSN in Massachusetts suggests that vessel strikes and entanglement in fixed gear (primarily lobster pots and whelk pots) are the principal sources of this mortality (Dwyer *et al.* 2002). Fixed gear fisheries in the Mid-Atlantic have also contributed to leatherback entanglements. For example, in North Carolina, two leatherback sea turtles were reported entangled in a crab pot buoy inside Hatteras Inlet (NMFS SEFSC 2001). A third leatherback was reported entangled in a crab pot buoy in Pamlico Sound off of Ocracoke. This turtle was disentangled and released alive; however, lacerations on the front flippers from the lines were evident (NMFS SEFSC 2001). In the Southeast, leatherbacks are vulnerable to entanglement in Florida's lobster pot and stone crab fisheries as documented on stranding forms. In the U.S. Virgin Islands, where one of five leatherback strandings from 1982 to 1997 were due to entanglement (Boulon 2000), leatherbacks have been observed with their flippers wrapped in the line of West Indian fish traps (R. Boulon, pers. comm. to Joanne Braun-McNeill, NMFS SEFSC 2001).

Leatherback interactions with the U.S. south Atlantic and Gulf of Mexico shrimp fisheries, are also known to occur (NMFS 2002). Leatherbacks are likely to encounter shrimp trawls working in the coastal waters off the Atlantic coast (from Cape Canaveral, Florida through North Carolina) as they make their annual spring migration north. For many years, TEDs that were required for use in the U.S. south Atlantic and Gulf of Mexico shrimp fisheries were less effective for leatherbacks as compared to the smaller, hard-shelled turtle species, because the TED openings were too small to allow leatherbacks to escape. To address this problem, on February 21, 2003, NMFS issued a final rule to amend the TED regulations. Modifications to the design of TEDs are now required in order to exclude leatherbacks as well as large benthic immature and sexually mature loggerhead and green turtles.

Other trawl fisheries are also known to interact with leatherback sea turtles although on a much smaller scale. In October 2001, for example, a fisheries observer documented the take of a leatherback in a bottom otter trawl fishing for *Loligo* squid off of Delaware. TEDs are not required in this fishery. In November 2007, fisheries observers reported the capture of a leatherback sea turtle in bottom otter trawl gear fishing for summer flounder.

Gillnet fisheries operating in the nearshore waters of the Mid-Atlantic states are also known to capture, injure and/or kill leatherbacks when these fisheries and leatherbacks co-occur. Data collected by the NEFSC Fisheries Observer Program from 1994 through 1998 (excluding 1997) indicate that a total of 37 leatherbacks were incidentally captured (16 lethally) in drift gillnets set in offshore waters from Maine to Florida during this period. Observer coverage for this period ranged from 54% to 92%. In North Carolina, a leatherback was reported captured in a gillnet set in Pamlico Sound in the spring of 1990 (D. Fletcher, pers.comm. to Sheryan Epperly, NMFS SEFSC 2001). Five other leatherbacks were released alive from nets set in North Carolina during the spring months: one was from a net (unknown gear) set in the nearshore waters near the North Carolina/Virginia border (1985); two others had been caught in gillnets set off of Beaufort Inlet (1990); a fourth was caught in a gillnet set off of Hatteras Island (1993), and a fifth was caught in a sink net set in New River Inlet (1993). In addition to these, in September 1995, two dead leatherbacks were removed from a 11-inch (28.2 cm) monofilament shark gillnet set in the nearshore waters off of Cape Hatteras, North Carolina (STSSN unpublished data reported in NMFS SEFSC 2001).

Fishing gear interactions and poaching are problems for leatherbacks throughout their range. Entanglements are common in Canadian waters where Goff and Lien (1988) reported that 14 of 20 leatherbacks encountered off the coast of Newfoundland/Labrador were entangled in fishing gear including salmon net, herring net, gillnet, trawl line and crab pot line. Leatherbacks are known to drown in fish nets set in coastal waters of Sao Tome, West Africa (Castroviejo *et al.* 1994; Graff 1995). Gillnets are one of the suspected causes for the decline in the leatherback sea turtle population in French Guiana (Chevalier *et al.* 1999), and gillnets targeting green and hawksbill turtles in the waters of coastal Nicaragua also incidentally catch leatherback turtles (Lagueux *et al.* 1998). Observers on shrimp trawlers operating in the northeastern region of Venezuela documented the capture of six leatherbacks from 13,600 trawls (Marcano and Alio 2000). An estimated 1,000 mature female leatherback sea turtles are caught annually in fishing nets off of Trinidad and Tobago with mortality estimated to be between 50-95% (Eckert and Lien 1999). However, many of the turtles do not die as a result of drowning, but rather because the fishermen butcher them in order to get them out of their nets (NMFS SEFSC 2001).

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

Summary of Status for Leatherback Sea Turtles
In the Pacific Ocean, the abundance of leatherback turtles on nesting beaches has declined dramatically over the past 10 to 20 years: nesting groups throughout the eastern and western Pacific Ocean have been reduced to a fraction of their former abundance by the combined effects of human activities that have reduced the number of nesting females and reduced the reproductive success of females that manage to nest (for example, egg poaching) (NMFS and USFWS 2007b). No reliable long term trend data for the Indian Ocean populations are currently available. While leatherbacks are known to occur in the Mediterranean Sea, nesting in this region is not known to occur (NMFS and USFWS 2007b).

Nest counts in many areas of the Atlantic show increasing trends, including for beaches in Suriname and French Guiana which support the majority of leatherback nesting (NMFS and USFWS 2007b). The species as a whole continues to face numerous threats at nesting and marine habitats. The long term recovery potential of this species may be further threatened by observed low genetic diversity, even in the largest nesting groups like French Guiana and Suriname (NMFS and USFWS 2007b).

Based on its 5-year status review of the species, NMFS and the USFWS (2007b) determined that endangered leatherback sea turtles should not be delisted or reclassified as threatened. However, it was also determined that an analysis and review of the species should be conducted in the future to determine whether DPS's should be identified for the leatherback, and what the status of any DPSs should be (NMFS and USFWS 2007b).

Kemp's ridley sea turtle
The Kemp's ridley is one of the least abundant of the world's sea turtle species. In contrast to loggerhead, leatherback and green sea turtles which are found in multiple oceans of the world, Kemp's ridleys typically occur only in the Gulf of Mexico and the northern half of the Atlantic Ocean (USFWS and NMFS 1992).

The majority of Kemp's ridleys nest along a single stretch of beach near Rancho Nuevo, Tamaulipas, Mexico (Carr 1963; USFWS and NMFS 1992; NMFS and USFWS 2007c). There is a limited amount of scattered nesting to the north and south of the primary nesting beach (NMFS and USFWS 2007c). The number of nesting adult females reached an estimated low of 300 in 1985 (USFWS and NMFS 1992; TEWG 2000; NMFS and USFWS 2007c). Conservation efforts by Mexican and U.S. agencies have aided this species by eliminating egg harvest, protecting eggs and hatchlings, and reducing at-sea mortality through fishing regulations (TEWG 2000). From 1985 to 1999, the number of nests observed at Rancho Nuevo, and nearby beaches increased at a mean rate of 11.3% (95% C.I. slope = 0.096-0.130) per year (TEWG 2000). An estimated 5,500 females nested in Tamaulipas over a 3-day period in May 2007 (NMFS and USFWS 2007c).

Kemp's ridleys mature at 10-17 years (Caillouet *et al.* 1995; Schmid and Witzell 1997; Snover *et al.* 2007; NMFS and USFWS 2007c). Nesting occurs from April through July each year with hatchlings emerging after 45-58 days (USFWS and NMFS 1992). Once they leave the beach, neonates presumably enter the Gulf of Mexico where they feed on available sargassum and associated infauna or other epipelagic species (USFWS and NMFS 1992). The presence of juvenile turtles along both the Atlantic and Gulf of Mexico coasts of the U.S., where they are recruited to the coastal benthic environment, indicates that post-hatchlings are distributed in both the Gulf of Mexico and Atlantic Ocean (TEWG 2000).

The location and size classes of dead turtles recovered by the STSSN suggests that benthic immature developmental areas occur in many areas along the U.S. coast and that these areas may change given resource quality and quantity (TEWG 2000). Foraging areas documented along the Atlantic coast include Pamlico Sound (NC), Chesapeake Bay, Long Island Sound, Charleston Harbor (SC) and Delaware Bay. Developmental habitats are defined by several characteristics, including coastal areas sheltered from high winds and waves such as embayments and estuaries, and nearshore temperate waters shallower than 50m (NMFS and USFWS 2007c). The suitability of these habitats depends on resource availability, with optimal environments providing rich sources of crabs and other invertebrates. A wide variety of substrates have been documented to provide good foraging habitats, including seagrass beds, oyster reefs, sandy and mud bottoms and rock outcroppings (NMFS and USFWS 2007c). Adults are primarily found in near-shore waters of 37m or less that are rich in crabs and have a sandy or muddy bottom (NMFS and USFWS 2007c).

Next to loggerheads, Kemp's ridleys are the second most abundant sea turtle in Virginia and Maryland state waters, arriving in these areas during May and June (Keinath *et al.* 1987; Musick and Limpus 1997). In the Chesapeake Bay, where the seasonal juvenile population of Kemp's ridley sea turtles is estimated to be 211 to 1,083 turtles (Musick and Limpus 1997), ridleys frequently forage in submerged aquatic grass beds for crabs (Musick and Limpus 1997). Kemp's ridleys consume a variety of crab species, including *Callinectes* sp., *Ovalipes* sp., *Libinia* sp., and *Cancer* sp. Mollusks, shrimp, and fish are consumed less frequently (Bjorndal 1997). Upon leaving Chesapeake Bay in autumn, juvenile ridleys migrate down the coast, passing Cape Hatteras in December and January (Musick and Limpus 1997). These larger juveniles are joined there by juveniles of the same size from North Carolina sounds and smaller juveniles from New York and New England to form one of the densest concentrations of Kemp's ridleys outside of the Gulf of Mexico (Epperly *et al.* 1995a; Epperly *et al.* 1995b; Musick and Limpus 1997).

Kemp's ridleys face many of the same natural threats as loggerheads, including destruction of nesting habitat from storm events, natural predators at sea, and oceanic events such as cold-stunning. Although cold-stunning can occur throughout the range of the species, it may be a greater risk for sea turtles that utilize the more northern habitats of Cape Cod Bay and Long Island Sound. For example, as reported in the national STSSN database, in the winter of 1999/2000, there was a major cold-stunning event where 218 Kemp's ridleys, 54 loggerheads, and 5 green turtles were found on Cape Cod beaches. Annual cold stun events do not always occur at this magnitude; the extent of episodic major cold stun events may be associated with numbers of turtles utilizing Northeast waters in a given year, oceanographic conditions and the occurrence of storm events in the late fall. Although many cold-stun turtles can survive if found early enough, cold-stunning events can represent a significant cause of natural mortality.

Like other turtle species, the severe decline in the Kemp's ridley population appears to have been heavily influenced by a combination of exploitation of eggs and impacts from fishery interactions. From the 1940s through the early 1960s, nests from Ranch Nuevo were heavily exploited (USFWS and NMFS 1992), but beach protection in 1966 helped to curtail this activity (USFWS and NMFS 1992). Following World War II, there was a substantial increase in the number of trawl vessels, particularly shrimp trawlers, in the Gulf of Mexico where the adult Kemp's ridley turtles occur. Information from fishers helped to demonstrate the high number of turtles taken in these shrimp trawls (USFWS and NMFS 1992). Subsequently, NMFS has worked with the industry to reduce turtle takes in shrimp trawls and other trawl fisheries, including the development and use of TEDs. As described in Section 3.1.1 above, there is lengthy regulatory history with regard to the use of TEDs in the U.S. south Atlantic and Gulf of Mexico shrimp fisheries (Epperly and Teas 2002; NMFS 2002; Lewison *et al.* 2003). The Biological Opinion completed in 2002 concluded that 155,503 Kemp's ridley sea turtles would be taken annually in the fishery with 4,208 of the takes resulting in mortality (NMFS 2002).

Although changes in the use of shrimp trawls and other trawl gear has helped to reduce mortality of Kemp's ridleys, this species is also affected by other sources of anthropogenic impacts similar to those discussed above. For example, in the spring of 2000, a total of five Kemp's ridley carcasses were recovered from the same North Carolina beaches where 275 loggerhead carcasses were found. Cause of death for most of the turtles recovered was unknown, but the mass mortality event was suspected to have been from a large-mesh gillnet fishery operating offshore in the preceding weeks. The five ridley carcasses that were found are likely to have been only a minimum count of the number of Kemp's ridleys that were killed or seriously injured as a result of the fishery interaction since it is unlikely that all of the carcasses washed ashore.

Summary of Status for Kemp's ridley Sea Turtles

The majority of Kemp's ridleys nest along a single stretch of beach near Rancho Nuevo, Tamaulipas, Mexico (Carr 1963; USFWS and NMFS 1992; NMFS and USFWS 2007c). The number of nesting females in the Kemp's ridley population declined dramatically from the late 1940s through the mid 1980s, with an estimated 40,000 nesting females in a single arribada in 1947 and fewer than 250 nesting females in the entire 1985 nesting season (USFWS and NMFS 1992; TEWG 2000). However, the total annual number of nests at Rancho Nuevo gradually began to increase in the 1990's (NMFS and USFWS 2007c). Based on the number of nests laid in 2006 and the remigration interval for Kemp's ridley sea turtles, there were an estimated 7,000-8,000 adult female Kemps ridley sea turtles in 2006 (NMFS and USFWS 2007c). The number of adult males in the population is unknown but sex ratios of hatchlings and immature ridleys suggest that the population is female biased (NMFS and USFWS 2007c). Based on its 5-year status review of the species, NMFS and the USFWS (2007c) determined that Kemp's ridley sea turtles should not be reclassified as threatened under the ESA.

Green sea turtle
Green turtles are distributed circumglobally, and can be found in the Pacific, Indian and Atlantic Oceans as well as the Mediterranean Sea (NMFS and USFWS 1991b; Seminoff 2004; NMFS and USFWS 2007d). In 1978, the Atlantic population of the green sea turtle was listed as threatened under the ESA, except for the breeding populations in Florida and on the Pacific coast of Mexico, which were listed as endangered. As it is difficult to differentiate between breeding populations away from the nesting beaches, in water all green sea turtles are considered endangered.

Pacific Ocean. Green turtles occur in the eastern, central, and western Pacific. Foraging areas are also found throughout the Pacific and along the southwestern U.S. coast (NMFS and USFWS 1998b). Nesting is known to occur in the Hawaiian archipelago, American Samoa, Guam, and various other sites in the Pacific but none of these are considered large breeding sites (with 2,000 or more nesting females per year)(NMFS and USFWS 1998b). The main nesting sites for the green sea turtle in the eastern Pacific are located in Michoacan, Mexico, and in the Galapagos Islands, Ecuador (NMFS and USFWS 2007d). The number of nesting females per year exceed 1,000 females at each site (NMFS and USFWS 2007d). However, historically, greater than 20,000 females per year are believed to have nested in Michoacan, alone (Cliffton *et al.* 1982; NMFS and USFWS 2007d). Thus the current number of nesting females is still far below what has historically occurred.

Historically, green turtles were used in many areas of the Pacific for food. They were also commercially exploited and this, coupled with habitat degradation led to their decline in the Pacific (NMFS and USFWS 1998b). Green turtles in the Pacific continue to be affected by poaching, habitat loss or degradation, fishing gear interactions, and fibropappiloma (NMFS and USFWS 1998b; NMFS 2004d).

Indian Ocean. There are numerous nesting sites for green sea turtles in the Indian Ocean. One of the largest nesting sites for green sea turtles worldwide occurs on the beaches of Oman where an estimated 20,000 green sea turtles nest annually (Hirth 1997; Ferreira *et al.* 2003). Based on a review of the 32 Index Sites used to monitor green sea turtle nesting worldwide, Seminoff (2004) concluded that declines in green turtle nesting were evident for many of the Indian Ocean Index Sites. While several of these had not demonstrated further declines in the more recent past, only the Comoros Island Index Site in the Western Indian Ocean showed evidence of increased nesting (Seminoff 2004).

Atlantic Ocean. As has occurred in other oceans of its range, green turtles were once the target of directed fisheries in the United States and throughout the Caribbean. In 1890, over one million pounds of green turtles were taken in the Gulf of Mexico green sea turtle fishery (Doughty 1984). However, declines in the turtle fishery throughout the Gulf of Mexico were evident by 1902 (Doughty 1984).

In the western Atlantic, green sea turtles range from Massachusetts to Argentina, including the

Gulf of Mexico and Caribbean (Wynne and Schwartz 1999). Green turtles occur seasonally in Mid-Atlantic and Northeast waters such as Long Island Sound (Musick and Limpus 1997; Morreale and Standora 1998; Morreale *et al.* 2004), presumably for foraging.

Some of the principal feeding pastures in the western Atlantic Ocean include the upper west coast of Florida and the northwestern coast of the Yucatan Peninsula. Additional important foraging areas in the western Atlantic include the Mosquito and Indian River Lagoon systems and nearshore wormrock reefs between Sebastian and Ft. Pierce Inlets in Florida, Florida Bay, the Culebra archipelago and other Puerto Rico coastal waters, the south coast of Cuba, the Mosquito Coast of Nicaragua, the Caribbean Coast of Panama, and scattered areas along Colombia and Brazil (Hirth 1971).

Age at maturity for green sea turtles is estimated to be 20-50 years (Balazs 1982, Frazer and Ehrhart 1985; Seminoff 2004). As is the case with the other turtle species described above, adult females may nest multiple times in a season and typically do not nest in successive years (NMFS and USFWS 1991b; Hirth 1997).

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

By far, the most important nesting concentration for green turtles in the western Atlantic is in Tortuguero, Costa Rica (NMFS and USFWS 2007d). Nesting in the area has increased considerably since the 1970's and nest count data from 1999-2003 suggest nesting by 17,402-37,290 females per year (NMFS and USFWS 2007d). The number of females nesting per year on beaches in the Yucatán , at Aves Island, Galibi Reserve, and Isla Trindade number in the hundreds to low thousands, depending on the site (NMFS and USFWS 2007d). In the U.S., certain Florida nesting beaches have been designated index beaches. Index beaches were established to standardize data collection methods and effort on key nesting beaches. The pattern of green turtle nesting shows biennial peaks in abundance, with a generally positive trend during the ten years of regular monitoring since establishment of the index beaches in 1989, perhaps due to increased protective legislation throughout the Caribbean (Meylan *et al.* 1995).

An average of 5,039 green turtle nests were laid annually in Florida between 2001 and 2006 with a low of 581 in 2001 and a high of 9,644 in 2005 (NMFS and USFWS 2007d). Occasional nesting has been documented along the Gulf coast of Florida, at southwest Florida beaches, as well as the beaches on the Florida Panhandle (Meylan *et al.* 1995). More recently, green turtle nesting occurred on Bald Head Island, North Carolina just east of the mouth of the Cape Fear River, on Onslow Island, and on Cape Hatteras National Seashore. Increased nesting has also been observed along the Atlantic Coast of Florida, on beaches where only loggerhead nesting was observed in the past (Pritchard 1997).

Green turtles face many of the same natural threats as loggerhead and Kemp's ridley sea turtles. In addition, green turtles appear to be susceptible to fibropapillomatosis, an epizootic disease producing lobe-shaped tumors on the soft portion of a turtle's body. Juveniles are most commonly affected. The occurrence of fibropapilloma tumors may result in impaired foraging, breathing, or swimming ability, leading potentially to death.

As with the other sea turtle species, incidental fishery mortality accounts for a large proportion of annual human-caused mortality outside the nesting beaches, while other activities like dredging, pollution, and habitat destruction account for an unknown level of other mortality. Stranding reports indicate that between 200-400 green turtles strand annually along the Eastern U.S. coast from a variety of causes most of which are unknown (STSSN database). Sea sampling coverage in the pelagic driftnet, pelagic longline, southeast shrimp trawl, and summer flounder bottom trawl fisheries has recorded takes of green turtles.

Summary of Status of Green Sea Turtles

A review of 32 Index Sites3 distributed globally revealed a 48% to 67% decline in the number of mature females nesting annually over the last 3-generations4 (Seminoff 2004). An evaluation of green sea turtle nesting sites was also conducted as part of the 5-year status review of the species (NMFS and USFWS 2007d). Of the 23 nesting groups assessed in that report, 10 were considered to be increasing, 9 were considered stable, and 4 were considered to be decreasing (NMFS and USFWS 2007d). Nesting groups were considered to be doing relatively well (the number of sites with increasing nesting were greater than the number of sites with decreasing nesting) in the Pacific, western Atlantic, and central Atlantic (NMFS and USFWS 2007d). The report also estimates that 108,761 to 150,521 females nest each year among the 46 sites (NMFS

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3 The 32 Index Sites include all of the major known nesting areas as well as many of the lesser nesting areas for which quantitative data are available.

4 Generation times ranged from 35.5 years to 49.5 years for the assessment depending on the Index Beach site

and USFWS 2007d). However, given the late age to maturity for green sea turtles, caution is urged regarding the status for any of the nesting groups since no area has a dataset spanning a full green sea turtle generation (NMFS and USFWS 2007d).

There is cautious optimism that the green sea turtle abundance is increasing in the Atlantic. Seminoff (2004) and NMFS and USFWS (2007d) made comparable conclusions with regard to nesting for four nesting sites in the western Atlantic. Each also concluded that nesting at Tortuguero, Costa Rica represented the most important nesting area for green sea turtles in the western Atlantic and that nesting had increased markedly since the 1970's (Seminoff 2004; NMFS and USFWS 2007d). However, the 5-year review also noted that the Tortuguero nesting stock continued to be affected by ongoing directed take at their primary foraging area in Nicaragua (NMFS and USFWS 2007d). As with the other sea turtle species, fishery mortality accounts for a large proportion of annual human-caused mortality outside the nesting beaches, while other activities like dredging, pollution, and habitat destruction account for an unknown level of other mortality.

ENVIRONMENTAL BASELINE

Environmental baselines for biological opinions include the past and present impacts of all state, federal or private actions and other human activities in the action area, the anticipated impacts of all proposed federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions that are contemporaneous with the consultation in process (50 CFR 402.02). The environmental baseline for this biological opinion includes the effects of several activities that occur in the action area that may affect the survival and recovery of threatened and endangered species. The activities that shape the environmental baseline in the action area of this consultation include vessel operations, fisheries, and recovery activities associated with reducing those impacts.

The past impacts of each state, Federal, and private action or other human activity in the action area can not be particularized in their entirety. However, to the extent they have manifested themselves at the population level, such past impacts are subsumed in the information presented on the status and trends of the species in the Status of the Species sections, recognizing that the benefits to sea turtles as a result of recovery activities already implemented may not be evident in the status and trends of populations for years given the relatively late age to maturity for sea turtles, and depending on the age class(es) affected.

Federal Actions that have Undergone Formal or Early Section 7 Consultation

NMFS has undertaken several ESA section 7 consultations to address the effects of vessel operations and gear associated with federally-permitted fisheries on threatened and endangered species in the action area. Each of those consultations sought to develop ways of reducing the probability of adverse impacts of the action on listed species. Similarly, recovery actions NMFS has undertaken under both the Marine Mammal Protection Act (MMPA) and the ESA are addressing the problem of take of whales in the fishing and shipping industries.

Vessel Operations

Potential adverse effects from federal vessel operations in the action area of this consultation include operations of the US Navy (USN) and the US Coast Guard (USCG), which maintain the largest federal vessel fleets, the EPA, the National Oceanic and Atmospheric Administration (NOAA), and the ACOE. NMFS has conducted formal consultations with the USCG, the USN, EPA and NOAA on their vessel operations. In addition to operation of ACOE vessels, NMFS has consulted with the ACOE to provide recommended permit restrictions for operations of contract or private vessels around whales. Through the section 7 process, where applicable, NMFS has and will continue to establish conservation measures for all these agency vessel operations to avoid adverse effects to listed species. Refer to the biological opinions for the USCG (September 15, 1995; July 22, 1996; and June 8, 1998) and the USN (May 15, 1997) for detail on the scope of vessel operations for these agencies and conservation measures being implemented as standard operating procedures.

Federal Fishery Operations
Formal ESA section 7 consultation has been conducted on the fisheries authorized under the Atlantic mackerel, squid, and butterfish, monkfish, multispecies, skate, spiny dogfish, and summer flounder, scup, black sea bass FMP's as well as for the American lobster fishery. Given the size of the action area compared to the broad area of operation for these fisheries, only a small portion of the fishing effort for each of these is expected to occur within the action area of this consultation.

Gear used in the federal fisheries described below is expected to have an insignificant effect on sea turtle prey or the bottom habitat utilized by sea turtles. Sea turtle prey items such as horseshoe crabs, other crabs, and whelks are removed from the marine environment as fisheries bycatch in one or more of the aforementioned fisheries. While some of the bycatch is likely returned to the water dead or injured to the extent that the organisms will shortly die, they would still be available as prey for sea turtles which are known to eat a variety of live prey as well as scavenge dead organisms (Keinath *et al.* 1987; Lutcavage and Musick 1985; Dodd 1988; Burke *et al.* 1993; Morreale and Standora 2005). Several of the aforementioned fisheries (e.g., Atlantic mackerel, squid) use bottom-otter trawl gear. A panel of experts have previously concluded that the effects of even light weight otter trawl gear would include: (1) The scraping or plowing of the doors on the bottom, sometimes creating furrows along their path, (2) sediment suspension resulting from the turbulence caused by the doors and the ground gear on the bottom, (3) the removal or damage to benthic or demersal species, and (4) the removal or damage to structure forming biota. The panel also concluded that the greatest impacts from otter trawls occur in high and low energy gravel habitats and in hard clay outcroppings, and that sand habitats were the least likely to be impacted (NREFHSC 2002). The action area for this consultation does not include gravel habitats or hard clay outcroppings. Fixed gear such as pot/trap and sink gillnet gear is expected to have less of an effect on bottom habitat than mobile gear. Given this, the use of trawl gear and fixed gear in the aforementioned fisheries will have an insignificant effect on the bottom habitat utilized by sea turtles.

Other than entanglement in fishing gear, effects of *fishing vessels* on listed species may involve disturbance or injury/mortality due to collisions or entanglement in anchor lines. Sea turtles are known to be killed and injured as a result of being struck by vessels on the water. However, for the following reasons, the operation of fishing vessels used in the aforementioned fisheries will have discountable effects on loggerhead sea turtles. First, fishing vessels operate at relatively slow speeds, particularly when towing or hauling gear. Thus, sea turtles in the path of a fishing vessel would be more likely to have time to move away before being struck. Fishing effort for all of the federal fisheries within the action area is constrained in some way --- either through a limited access permit system or by fishing quotas, thus limiting the amount of time that vessels are on the water. Listed sea turtles occur seasonally in waters along the East Coast so that a portion of the fishing in these waters occurs at times when sea turtles are not likely to be present. Finally, sea turtles do not occur strictly at the water surface or strictly within close proximity of the water surface (Morreale 1999) meaning that sea turtles spend part of their time at depths out of range of a vessel collision with boats.

Listed species may also be affected by fuel oil spills resulting from fishing vessel accidents. Fuel oil spills could affect animals directly or indirectly through the food chain. Fuel spills involving fishing vessels are common events. However, these spills typically involve small amounts of material that are likely to have an insignificant effect on listed species. Larger spills may result from accidents, although these events would be rare and involve small areas. No direct effects on listed species resulting from fishing vessel fuel spills have been documented.

Formal ESA section 7 consultation has been conducted on the following fisheries which occur in the action area: Skate, Multispecies, Monkfish, Summer Flounder/Scup/Black Sea Bass, Mackerel/Squid/Butterfish, Lobster and Spiny Dogfish fisheries. These consultations are summarized below. These fisheries overlap with the action area to varying degrees.

Section 7 consultation on the Skate FMP was completed July 24, 2003, and concluded that authorization of the skate fishery may adversely affect ESA-listed sea turtles, including loggerheads, as a result of interactions with (capture in) gillnet and trawl gear. In August 2007, NMFS received an estimate of loggerhead sea turtle takes in bottom otter trawl gear used in the skate fishery (Memo from K. Murray, NEFSC to L. Lankshear, NERO, PRD). Using VTR data from 2000-2004 and the average annual bycatch of turtles as described in Murray 2006, the average annual bycatch of loggerhead sea turtles in bottom otter trawl gear used in the skate fishery was estimated to be 24 loggerhead sea turtles a year (Memo from K. Murray, NEFSC to L. Lankshear, NERO, PRD). This information represents new information on the capture of loggerhead sea turtles in skate fishery. NMFS has, therefore, reinitiated section 7 consultation on the continued authorization of the directed skate fishery under the Northeast skate FMP.

The *Northeast Multispecies fishery* operates throughout the year with peaks in spring, and from October through February. Multiple gear types are used in the fishery. However, the gear type of greatest concern is sink gillnet gear that can entangle whales and sea turtles (*i.e.*, in buoy lines and/or net panels). Data indicate that sink gillnet gear has seriously injured or killed North Atlantic right whales, humpback whales, fin whales, loggerhead and leatherback sea turtles. The most recent reinitiation of the Northeast Multispecies consultation was completed on June 14, 2001, and concluded that continued implementation of the Multispecies FMP may adversely affect loggerhead, Kemp's ridley and green sea turtles and is not likely to jeopardize the continued existence of the North Atlantic right whale. A new RPA was issued to avoid the likelihood that the operation of the gillnet sector of the multispecies fishery would result in jeopardy to right whales. The ITS exempted the lethal or non-lethal take (i.e., capture that may or may not result in mortality) of one loggerhead sea turtle, and one green, leatherback, or Kemp's ridley turtle annually. The northeast multispecies sink gillnet fishery has historically occurred from the periphery of the Gulf of Maine to Rhode Island in water to 60 fathoms. In recent years, more of the effort in the fishery has occurred in offshore waters and into the Mid-Atlantic. However, participation in this fishery has declined since extensive groundfish conservation measures have been implemented, particularly since implementation of Amendment 13 to the Multispecies FMP in April 2004. Additional management measures (*i.e.* Framework Adjustment 42) are expected to further reduce and control effort in the multispecies fishery. The exact relationship between multispecies fishing effort and the number of sea turtle interactions with gear used in the fishery is unknown. However, in general, less fishing effort results in less time that gear is in the water and therefore less opportunity for sea turtles to be captured or entangled in multispecies fishing gear. Section 7 consultation is on-going and will consider the information received from the NEFSC as well as information on changes to the fishery since 2004.

The federal *Monkfish fishery* occurs in all waters under federal jurisdiction from Maine to the North Carolina/South Carolina border. The monkfish fishery uses several gear types that may entangle protected species. In 1999, observers documented that turtles were taken in excess of the ITS as a result of entanglements in monkfish gillnet gear. NMFS reinitiated consultation on the Monkfish FMP on May 4, 2000, in part, to reevaluate the affect of the monkfish gillnet fishery on sea turtles. The Opinion also considered new information on the status of the North Atlantic right whale and new Atlantic Large Whale Take Reduction Plan (ALWTRP) measures, and the ability of the RPA to avoid the likelihood of jeopardy to right whales. The Opinion concluded that continued implementation of the Monkfish FMP was likely to jeopardize the existence of the North Atlantic right whale. A new RPA was provided that was expected to remove the threat of jeopardy to North Atlantic right whales. In addition, a new ITS was provided for the take of sea turtles in the fishery. However, consultation was once again reinitiated on the Monkfish FMP as of February 12, 2003, to consider the effects of Framework Adjustment 2 measures on ESA-listed species. This consultation was completed on April 14, 2003, and concluded that the proposed action is not likely to result in jeopardy to any ESA-listed species under NMFS jurisdiction. However, takes of sea turtles are still expected to occur, which was reflected in the ITS. In the Opinion, NMFS anticipated that over a five year period, the action would result in the capture of up to 25 sea turtles with no more than 15 of these being loggerheads captured in monkfish gillnet gear, no more than 5 of any combination of green, Kemp's ridley or leatherback sea turtles caught in monkfish gillnet gear, and no more than 5 being either loggerhead, green, Kemp's ridley or leatherback sea turtles captured in monkfish trawl gear. Of these, no more than 5 loggerheads are expected to die as a result of the capture in monkfish fishing gear. A maximum of 3 of any one of the other three species are expected to die as a result of capture in monkfish fishing gear.

The *Summer Flounder, Scup and Black Sea Bass fisheries* are known to interact with sea turtles. Significant measures have been developed to reduce the capture of sea turtles in summer flounder trawls and trawls that meet the definition of a summer flounder trawl by requiring the use of TEDs throughout the year for trawl nets fished from the North Carolina/South Carolina border to Oregon Inlet, NC, and seasonally (March 16-January 14) for trawl vessels fishing between Oregon Inlet, NC and Cape Charles, VA. Interactions with sea turtles may still occur with this gear type in other areas however. Based on the occurrence of gillnet entanglements in other fisheries, the gillnet portion of this fishery could entangle endangered whales. The pot gear and staked trap sectors could also entangle whales and sea turtles. The most recent (December 16, 2001) formal consultation on this fishery concluded that the operation of the fishery may adversely affect but is not likely to jeopardize the continued existence of listed species. The ITS anticipated that 19 loggerhead or Kemp's ridley takes (captures, with up to 5 lethal) and 2 green turtle takes (captures that may or may not result in mortality) may occur annually. However, as a result of new information not considered in previous consultations, NMFS has reinitiated section 7 consultation on this FMP to consider the effects of the fisheries on ESA-listed whales and sea turtles. Consultation is currently ongoing and to date, a revised Opinion has not yet been issued.

The primary gear types for the *Spiny dogfish fishery* are sink gillnets, otter trawls, bottom longline, and driftnet gear. Sea turtles can be incidentally captured in all gear sectors of this fishery. Turtle takes in 2000 included one dead and one live Kemp's ridley. Since the ITS issued with the August 13, 1999, Opinion anticipated the take (capture) of only one Kemp's ridley (that may or may not result in mortality), the incidental take level for the dogfish FMP was exceeded. In addition, a right whale mortality occurred in 1999 as a result of entanglement in gillnet gear that may (but was not determined to be) have originated from the spiny dogfish fishery. NMFS, therefore, reinitiated consultation on the Spiny Dogfish FMP on May 4, 2000, in order to reevaluate the ability of the RPA to avoid the likelihood of jeopardy to right whales, and the effect of the spiny dogfish gillnet fishery on sea turtles. The Opinion also considered new information on the status of the North Atlantic right whale and new ALWTRP measures. The Opinion, signed on June 14, 2001, concluded that continued implementation of the Spiny Dogfish FMP is likely to jeopardize the existence of the North Atlantic right whale. A new RPA was provided that was expected to remove the threat of jeopardy to North Atlantic right whales as a result of the gillnet sector of the spiny dogfish fishery. In addition, the ITS anticipated the annual take (capture) of 3 loggerheads (no more than 2 lethal), 1 green (lethal or non-lethal), 1 leatherback (lethal or non-lethal), and 1 Kemp's ridley (lethal or non-lethal).

The *American lobster trap fishery* has been identified as a source of gear causing serious injuries and mortality of endangered whales and leatherback sea turtles. Previous BOs for this fishery have concluded that operation of the lobster trap fishery is likely to jeopardize the continued existence of right whales and may adversely affect leatherback sea turtles. A Reasonable and Prudent Alternative (RPA) to avoid the likelihood that the lobster fishery would jeopardize the continued existence of right whales was implemented. However, these measures were not expected to reduce the number or severity of leatherback sea turtle interactions with the fishery. Subsequently, the death of a right whale was determined to be entanglement related and NMFS concluded that the death provided evidence that the RPA was not effective at removing the likelihood of jeopardy for right whales from the lobster trap fishery. Consultation was reinitiated and is in progress.

American lobster occur within U.S. waters from Maine to Virginia. They are most abundant from Maine to New Jersey with abundance declining from north to south (ASMFC 1997). An Interstate Fishery Management Plan (ISFMP) developed through the ASMFC provides management measures for the fishery that are implemented by the states. NMFS has issued regulations for the Federal waters portion of the fishery based on recommendations from the ASMFC. Of the seven lobster management areas (LMAs), only LMA 3 occurs entirely within Federal waters. The action area for this consultation overlaps with a portion of LMA 2. LMAs 1, 2, 4, 5, and the Outer Cape include both state and Federal waters (NMFS 1999; 2002b). Therefore, management of the Federal waters portion of LMAs 1, 2, 4, 5, and the Outer Cape must be consistent with management in the state waters portion of those areas to meet the objectives of the Lobster ISFMP. Management measures include a limited access permit system, gear restrictions, and other prohibitions on possession (*e.g.*, of berried or scrubbed lobsters), landing limits for lobsters caught by non-trap gear, a trap tag requirement, and trap limits. These measures include reduction of effort and capping of effort. The commercial lobster fishery is frequently described as an inshore fishery (typically defined as within state waters; 0-3 nautical miles from shore) and an offshore fishery (typically defined as nearshore Federal waters and the deepwater offshore fishery) (NMFS 1999).

Most lobster trap effort occurs in the Gulf of Maine. Maine and Massachusetts produced 93% of the 2004 total U.S. landings of American lobster, with Maine accounting for 78% of these landings (NMFS 2002b). Lobster landings in the other New England states as well as New York and New Jersey account for most of the remainder of U.S. American lobster landings. However, declines in lobster abundance and landings have occurred from Rhode Island through New Jersey in recent years. The Mid-Atlantic States from Delaware through North Carolina have been granted *de minimus* status under the Lobster ISFMP. Low landings of lobster in these *de minimus* states suggest that there is not a directed fishery for lobster in these territorial waters.

Non-Federally Regulated Actions

Private and Commercial Vessel Operations

Private and commercial vessels, including fishing vessels, operating in the action area of this consultation also have the potential to interact with listed species. Ship strikes have been identified as a significant source of mortality to the North Atlantic right whale population (Kraus 1990) and are also known to impact all other endangered whales. The Sea Turtle Stranding and Salvage Network (STSSN) also reports regular incidents of likely vessel interactions (e.g., propeller-type injuries) with sea turtles. Interactions with these types of vessels and sea turtles could occur in the action area, and it is possible that these collisions would result in mortality. The effects of fishing vessels, recreational vessels, or other types of commercial vessels on listed species may involve disturbance or injury/mortality due to collisions or entanglement in anchor lines. It is important to note that minor vessel collisions may not kill an animal directly, but may weaken or otherwise affect it so it is more likely to become vulnerable to effects such as entanglements. Listed species may also be affected by fuel oil spills resulting from vessel accidents. Fuel oil spills could affect animals directly or indirectly through the food chain. Fuel spills involving fishing vessels are common events. However, these spills typically involve small amounts of material that are unlikely to adversely affect listed species. Larger oil spills may result from accidents, although these events would be rare and involve small areas. No direct adverse effects on listed sea turtles resulting from fishing vessel fuel spills have been documented.

In addition to commercial traffic and recreational pursuits, private vessels also participate in high speed marine events. As these events require a Marine Event permit from the US Coast Guard, there is a federal action which may trigger section 7 consultation. While in some areas of the US these events may occur regularly, high speed marine events permitted by the USCG appear to be a relatively infrequent occurrence in the action area. NMFS is only aware of one such event that has occurred in the recent past in the action area (i.e., a high speed boat race sponsored by the Cape Cod Chamber of Commerce and held off Yarmouth, MA in September 2004). Endangered species observers were present on scene and no interactions with listed species were observed during this two day event.

Non-Federally Regulated Fishery Operations

Very little is known about the level of interactions with listed species in fisheries that operate strictly in state waters. However, depending on the fishery in question, many state permit holders also hold federal licenses; therefore, section 7 consultations on federal actions in those fisheries address some state-water activity. Nearshore entanglements of turtles have been documented; however, information is not currently available on whether the vessels involved were permitted by the state or by NMFS. Impacts of state fisheries on endangered whales are addressed as appropriate through the MMPA take reduction planning process. NMFS is actively participating in a cooperative effort with the Atlantic States Marine Fisheries Commission (ASMFC) and member states to standardize and/or implement programs to collect information on level of effort and bycatch of protected species in state fisheries. When this information becomes available, it can be used to refine take reduction plan measures in state waters.

With regard to whale entanglements, vessel identification is occasionally recovered from gear removed from entangled animals. With this information, it is possible to determine whether the gear was deployed by a federal or state permit holder and whether the vessel was fishing in federal or state waters. In 1998, 3 entanglements of humpback whales in state-water fisheries were documented. Nearshore entanglements of turtles have been documented; however, information is not available on whether the vessels involved were permitted by the state or by NMFS.

Other Potential Sources of Impacts in the Action Area

Sources of human-induced mortality, injury, and/or harassment of turtles in the action area that are reasonably certain to occur in the future include incidental takes in state-regulated fishing activities, vessel collisions, ingestion of plastic debris, and pollution. While the combination of these activities may affect populations of endangered and threatened sea turtles, preventing or slowing a species' recovery, the magnitude of these effects is currently unknown. A number of anthropogenic activities have likely directly or indirectly affect listed species in the action area of this consultation. These sources of potential impacts include previous dredging projects, pollution, water quality, and sonic activities. However, the impacts from these activities are difficult to measure. Where possible, conservation actions are being implemented to monitor or study impacts from these elusive sources.

Within the action area, sea turtles and optimal sea turtle habitat most likely have been impacted by pollution. Marine debris (e.g., discarded fishing line or lines from boats) can entangle turtles in the water and drown them. Turtles commonly ingest plastic or mistake debris for food, as observed with the leatherback sea turtle. The leatherback's preferred diet includes jellyfish, but similar looking plastic bags are often found in the turtle's stomach contents (Magnuson et al. 1990).

Sources of contamination in the action area include atmospheric loading of pollutants, stormwater runoff from coastal development, groundwater discharges, and industrial development. Chemical contaminants may also have an effect on sea turtle reproduction and survival. While the effects of contaminants on turtles is relatively unclear, pollution may be linked to the fibropapilloma virus that kills many turtles each year (NMFS 1997). If pollution is not the causal agent, it may make sea turtles more susceptible to disease by weakening their immune systems.

Pollution and Water Quality

Human activities in the action area causing pollution are reasonably certain to continue in the future, as are impacts from them on listed species. However, the level of impacts cannot be projected. Little data is available on water quality and pollutant levels in Nantucket Sound (Rivera 2007). Like other coastal waters, water quality in Nantucket Sound is influenced by pollution resulting from atmospheric loading of pollutants, storm water runoff from the coast, groundwater discharges and sewage treatment effluent. Concerns have been recently raised related to the effects of nutrient loading from land-based sources (Rivera 2007) which stimulate plankton blooms and result in eutrophication and lowered dissolved oxygen.

Marine debris (*e.g.*, discarded fishing line or lines from boats) can entangle turtles in the water and drown them. Turtles commonly ingest plastic or mistake debris for food. Chemical contaminants may also have an effect on sea turtle reproduction and survival. Excessive turbidity due to coastal development and/or construction sites could influence sea turtle foraging ability. As mentioned previously, turtles are not very easily affected by changes in water quality or increased suspended sediments, but if these alterations make habitat less suitable for turtles and hinder their capability to forage, eventually they would tend to leave or avoid these less desirable areas (Ruben and Morreale 1999). Noise pollution has been raised, primarily, as a concern for marine mammals but may be a concern for other marine organisms, including sea turtles. As described above, global warming is likely to negatively affect sea turtles – affecting when females lay their eggs, the survival of the eggs, sex ratios of offspring, and the stability of the Gulf Stream. To the extent that air pollution, for example from the combustion of fossil fuels by vessels, contributes to global warming, then it is also expected to negatively affect sea turtles.

NMFS and the US Navy have been working cooperatively to establish a policy for monitoring and managing acoustic impacts from anthropogenic sound sources in the marine environment. Acoustic impacts can include temporary or permanent injury, habitat exclusion, habituation, and disruption of other normal behavior patterns. It is expected that the policy on managing anthropogenic sound in the oceans will provide guidance for programs such as the use of acoustic deterrent devices in reducing marine mammal-fishery interactions and review of federal activities and permits for research involving acoustic activities.

Private and commercial vessels, including fishing vessels, operating in the action area of this consultation also have the potential to interact with sea turtles. The effects of fishing vessels, recreational vessels, or other types of commercial vessels on listed species may involve disturbance or injury/mortality due to collisions or entanglement in anchor lines. It is important to note that minor vessel collisions may not kill an animal directly, but may weaken or otherwise affect it so it is more likely to become vulnerable to effects such as entanglements. Listed species may also be affected by fuel oil spills resulting from vessel accidents. Fuel oil spills could affect animals directly or indirectly through the food chain. Fuel spills involving fishing vessels are common events. However, these spills typically involve small amounts of material that are unlikely to adversely affect listed species. Larger oil spills may result from accidents, although these events would be rare and involve small areas. No direct adverse effects on listed sea turtles resulting from fishing vessel fuel spills have been documented.

Global Climate Change

There is a large and growing body of literature on past, present, and future impacts of global climate change induced by human activities - frequently referred to in layman's terms as "global warming." Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. The Environmental Protection Agency's climate change webpage provides basic background information on these and other measured or anticipated effects (see www. epa.gov/climatechange/index.html). Activities in the action area that may have contributed to global warming include the combustion of fossil fuels by vessels.

The effects of global climate change on sea turtles is typically viewed as being detrimental to the species (NMFS and USFWS 2007a; 2007b; 2007c; 2007d). Changes in water temperature would be expected to affect prey distribution and/or abundance, salinity, and water circulation patterns perhaps even to the extent that the Gulf Stream is disrupted (Gagosian 2003; NMFS and USFWS 2007a; 2007b; 2007c; 2007d). The effects of these on sea turtles cannot, for the most part, be accurately predicted at this time. Several studies have, however, investigated the effects of changes in sea surface temperature and air temperatures on turtle reproductive behavior. For loggerhead sea turtles, warmer sea surface temperatures in the spring have been correlated to an earlier onset of nesting (Weishampel *et al.* 2004; Hawkes *et al.* 2007), shorter internesting intervals (Hays *et al.* 2002), and a decrease in the length of the nesting season (Pike *et al.* 2006). Green sea turtles also exhibited shorter internesting intervals in response to warming water temperatures (2002).

Air temperatures also play a role in sea turtle reproduction. In marine turtles, sex is determined by temperature in the middle third of incubation with female offspring produced at higher temperatures and males at lower temperatures within a thermal tolerance range of 25-35° C (Ackerman 1997). Based on modeling, a 2° C increase in air temperature is expected to result in a sex ratio of over 80% female offspring for loggerhead nesting beaches in the vicinity of Southport, NC. Farther to the south at Cape Canaveral, Florida, a 2°C increase in air temperature would likely result in production of 100% females while a 3°C increase in air temperature would likely exceed the thermal threshold of turtle clutches resulting in death (Hawkes *et al.* 2007). Thus changes in air temperature as a result of global climate change may alter sex ratios and may reduce hatchling production in the most southern nesting areas of the U.S. Given that the south Florida nesting group is the largest loggerhead nesting group in the Atlantic (in terms of nests laid), a decline in the success of nesting as a result of global climate change could have profound effects on the abundance and distribution of the loggerhead species in the Atlantic.

For green sea turtles, incubation temperatures also appeared to affect hatchling size with smaller turtles produced at higher incubation temperatures (Glen *et al.* 2003). It is unknown whether this effect is species specific and what impact it has on the survival of the offspring.

While the type and extent of effects to sea turtles as a result of global climate change are still speculative, a disruption of the Gulf Stream such as might occur as a result of global climate change (Gagosian 2003), would be expected to have profound effects on every aspect of sea turtle life history from hatching success, oceanic migrations at all life stages, foraging, and nesting.

The impact of climate change on cetaceans is likely to be related to changes in sea temperatures, potential freshening of sea water due to melting ice and increased rainfall, sea level rise, the loss of polar habitats and the potential decline of forage. Additionally, cetaceans may be affected by ocean acidification as more carbon dioxide is absorbed. These changes may effect the distribution of species and the fitness of individuals and populations due to the potential loss of foraging opportunities, displacement from ideal habitats and potential increase in susceptibility to disease (Elliot and Simmonds 2007). Similarly to sea turtles, a decline in the reproductive fitness as a result of global climate change could have profound effects on the abundance and distribution of large whales in the Atlantic.

Conservation and Recovery Actions Reducing Threats to Listed Species

A number of activities are in progress that may ameliorate some of the threat that activities summarized in the *Environmental Baseline* pose to threatened and endangered species in the action area of this consultation. These include education/outreach activities, specific measures to reduce the adverse effects of entanglement in fishing gear, including gear modifications, fishing gear time-area closures, and whale disentanglement, and measures to reduce ship and other vessel impacts to protected species. Many of these measures have been implemented to reduce risk to critically endangered right whales. Despite the focus on right whales, other cetaceans and some sea turtles will likely benefit from the measures as well.

Reducing threats of vessel collision on listed whales
In addition to the ESA measures for federal activities mentioned in the previous section, numerous recovery activities are being implemented to decrease the adverse effects of private and commercial vessel operations on the species in the action area and during the time period of this consultation. These include implementation of NOAA's Right Whale Ship Strike Reduction Strategy, extensive education and outreach activities, the Sighting Advisory System (SAS), other activities recommended by the Northeast Implementation Team for the recovery of the North Atlantic right whale (NEIT) and Southeast Implementation Team for the Right Whale Recovery Plan (SEIT), and NMFS regulations.

Northeast Implementation Team (NEIT)
The Northeast Large Whale Recovery Plan Implementation Team (NEIT) was founded in 1994 to help implement the right and humpback whale recovery plans developed under the ESA. The NEIT provided advice and expertise on the issues affecting right and humpback whale recovery, and was comprised of representatives from federal and state regulatory agencies and private organizations, and was advised by a panel of scientists with expertise in right and humpback whale biology. The Ship Strike Committee (SSC) was one of the most active committees of the NEIT, and NMFS came to recognize that vessel collisions with right whales was the recovery issue needing the most attention. As such, the NEIT was restructured in May 2004 to focus exclusively on right whale ship strike reduction research and issues and providing support to the NMFS Right Whale Ship Strike Working Group.

The Ship Strike Committee (SSC) of the former NEIT undertook multiple projects to reduce ship collisions with North Atlantic right whales. These included production of a video entitled: *Right Whales and the Prudent Mariner*, which provides information to mariners on the distribution and behavior of right whales in relation to vessel traffic. The video raises the awareness of mariners as to the plight of the right whale in the North Atlantic. NMFS and the NEIT also funded a project to develop recommended measures to reduce right whale ship strikes. The recommended measures project included looking at all possible options such as routing, seasonal and dynamic management areas, and vessel speed. It became evident in the process of meeting with the industry that a comprehensive strategy would have to be developed for the entire East coast. Development of NOAA's Ship Strike Reduction Strategy has been ongoing over the last number of years. The strategy is currently focused on protecting the North Atlantic right whale, but the operational measures are expected to reduce the incidence of ship strike on other large whales to some degree. The strategy consists of five basic elements and includes both regulatory and non-regulatory components: 1) operational measures for the shipping industry, including speed restrictions and routing measures, 2) section 7 consultations with Federal agencies that maintain vessel fleets, 3) education and outreach programs, 4) a bilateral conservation agreement with Canada, and 5) continuation of ongoing measures to reduce ship strikes of right whales (e.g., SAS, MSR, ongoing research into the factors that contribute to ship strikes, and research to identify new technologies that can help mariners and whales avoid each other). Progress made under these elements will be discussed further below.

Regulatory Actions to Reduce Vessel Strikes

In one recovery action aimed at reducing vessel-related impacts, including disturbance, NMFS published a proposed rule in August 1996 restricting vessel approach to right whales (61 FR 41116) to a distance of 500 yards. The Recovery Plan for the Northern Right Whale identified anthropogenic disturbance as one of many factors which had some potential to impede right whale recovery (NMFS 1991b). Following public comment, NMFS published an interim final rule in February 1997 codifying the regulations. With certain exceptions, the rule prohibits both boats and aircraft from approaching any right whale closer than 500 yds. Exceptions for closer approach are provided for the following situations, when: (a) compliance would create an imminent and serious threat to a person, vessel, or aircraft; (b) a vessel is restricted in its ability to maneuver around the 500-yard perimeter of a whale; (c) a vessel is investigating or involved in the rescue of an entangled or injured right whale; or (d) the vessel is participating in a permitted activity, such as a research project. If a vessel operator finds that he or she has unknowingly approached closer than 500 yds, the rule requires that a course be steered away from the whale at slow, safe speed. In addition, all aircraft, except those involved in whale watching activities, are excepted from these approach regulations. This rule is expected to reduce the potential for vessel collisions and other adverse vessel-related effects in the environmental baseline.

In April 1998, the USCG submitted, on behalf of the US, a proposal to the International Maritime Organization (IMO) requesting approval of a mandatory ship reporting system (MSR) in two areas off the east coast of the US, one which includes the right whale feeding grounds in the northeast, and one which includes the right whale calving grounds in the southeast. The USCG worked closely with NMFS and other agencies on technical aspects of the proposal. The package was submitted to the IMO's Subcommittee on Safety and Navigation for consideration and submission to the Marine Safety Committee at IMO and approved in December 1998. The USCG and NOAA play important roles in helping to operate the MSR system, which was implemented on July 1, 1999. Ships entering the northeast and southeast MSR boundaries are required to report the vessel identity, date, time, course, speed, destination, and other relevant information. In return, the vessel receives an automated reply with the most recent right whale sightings in the area and information on precautionary measures to take while in the vicinity of right whales.

A key component of NOAA's right whale ship strike reduction strategy is the proposed implementation of speed restrictions for vessels transiting the US Atlantic in areas and seasons where right whales predictably occur in high concentrations. The NEIT-funded "Recommended Measures to Reduce Ship Strikes of North Atlantic Right Whales" found that seasonal speed and routing measures could be an effective means of reducing the risk of ship strike along the US east coast. Based on these recommendations, NMFS published an Advance Notice of Proposed Rulemaking (ANPR) in June 2004 (69 FR 30857), and subsequently published a proposed rule on June 26, 2006 (71 FR 36299). NMFS published a final rule on October 6, 2008. The final rule implements a speed restriction of 10 knots in areas and at times when right whales are present.

Vessel Routing Measures to Reduce the Co-occurrence of Ships and Whales

Another critical, non-regulatory component of NOAA's right whale ship strike reduction strategy involves the development and implementation of routing measures that reduce the co-occurrence of vessels and right whales, thus reducing the risk of vessel collisions. Recommended routes were developed by overlaying right whale sightings data on existing vessel tracks, and plotting alternative routes where vessels could expect to encounter fewer right whales. Full implementation of these routes was completed at the end of November 2006. The routes are now charted on all NOAA electronic and printed charts, published in US Coast Pilots, and mariners have been notified through USCG Notices to Mariners.

Through a joint effort between NOAA and the USCG, the US also submitted a proposal to the IMO to shift the northern leg of the existing Boston Traffic Separation Scheme (TSS) 12 degrees to the north. Overlaying sightings of right whales and all baleen whales on the existing TSS revealed that the existing TSS directly overlaps with areas of high whale densities, while an area slightly to the north showed a considerable decrease in sightings. Separate analyses by the SBNMS and the NEFSC both indicated that the proposed TSS would overlap with 58% fewer right whale sightings and 81% fewer sightings of all large whales, thus considerably reducing the risk of collisions between ships and whales. The proposal was submitted to the IMO in April 2006, and was adopted by the Maritime Safety Committee in December 2006. The change was implemented domestically by the US Coast Guard on July 1, 2007.

Right Whale Sighting Advisory System

The right whale Sighting Advisory System (SAS) was initiated in early 1997 as a partnership among several federal and state agencies and other organizations to conduct aerial and ship board surveys to locate right whales and to alert mariners to right whale sighting locations in a near real time manner. The SAS surveys and opportunistic sightings reports document the presence of right whales and are provided to mariners via fax, email, NAVTEX, Broadcast Notice to Mariners, NOAA Weather Radio, several web sites, and the Traffic Controllers at the Cape Cod Canal. Fishermen and other vessel operators can obtain SAS sighting reports, and make necessary adjustments in operations to decrease the potential for interactions with right whales. The SAS has also served as the only form of active entanglement monitoring in Cape Cod Bay and the Great South Channel. Some of these sighting efforts have resulted in successful disentanglement of right whales. SAS flights have also contributed sightings of dead floating animals that can occasionally be retrieved to increase our knowledge of the biology of the species and effects of human impacts. The USCG has also played a vital role in this effort, providing air and sea support as well as a commitment of resources to NMFS operations. The Commonwealth of Massachusetts has been a key collaborator to the SAS effort and has continued the partnership. Other sources of opportunistic right whale sightings include whale watch vessels, commercial and recreational mariners, fishermen, the U.S. Navy, NMFS research vessels, and NEFSC cetacean abundance aerial survey data.

Education and Outreach Activities

NMFS, primarily through the NEIT and SEIT, is engaged in a number of education and outreach activities aimed specifically at increasing mariner awareness of the threat of ship strike to right whales. The NEIT and SEIT have developed a comprehensive matrix of mariner education and outreach tasks ranked by priority for all segments of the maritime industry, including both commercial and recreational vessels, and are in the process of implementing high priority tasks as funding allows. In anticipation of the 2006/2007 calving season, the SEIT is nearing completion of two new outreach tools—a multimedia CD to educate commercial mariners about right whale ship strike issues, and a public service announcement (PSA) targeted towards private recreational vessel operators to be distributed to media outlets in the southeast.

NMFS also distributes informational packets on right whale ship strike avoidance to vessels entering ports in the northeast. The informational packets contain various outreach materials developed by NMFS, including the video "Right Whales and the Prudent Mariner," a placard on the MSR system, extracts from the US Coast Pilots about whale avoidance measures and seasonal right whale distribution, and a placard on applicable right whale protective regulations and recommended vessel operating measures.

NMFS has also worked with the International Fund for Animal Welfare (IFAW) to develop educational placards for recreational vessels. These placards provide vessel operators with information on right whale identification, behavior, and distribution, as well as information about the threat of ship strike and ways to avoid collisions with whales.

The NEIT has contracted the development of a comprehensive merchant mariner education module for use and distribution to maritime academies along the east coast. The purpose of this program is to inform both new captains and those being re-certified about right whales and operational guidelines for minimizing the risk of collision. Development of the module is now complete and is in the process of being distributed and implemented in various maritime academies.

Reducing the Threat of Entanglement on Whales
Several efforts are ongoing to reduce the risk and impact of entanglement on listed whales, including both regulatory and non-regulatory measures. Most of these activities are captured under the Atlantic Large Whale Take Reduction Plan (ALWTRP). The ALWTRP is a multi-faceted plan that includes both regulatory and non-regulatory actions. Regulatory actions are directed at reducing serious entanglement injuries and mortality of right, humpback and fin whales from fixed gear fisheries (*i.e.*, trap and gillnet fisheries). The measures identified in the ALWTRP will also benefit minke whales (a non ESA-listed species). The non-regulatory component of the ALWTRP is composed of four principal parts: (1) gear research and development, (2) disentanglement, (3) the Sighting Advisory System (SAS), and (4) education/outreach. These components will be discussed in more detail below.

Regulatory Measures to Reduce the Threat of Entanglement on Whales
The regulatory component of the ALWTRP includes a combination of broad fishing gear modifications and time-area restrictions supplemented by progressive gear research to reduce the chance that entanglements will occur, or that whales will be seriously injured or die as a result of an entanglement. The long-term goal, established by the 1994 Amendments to the MMPA, was to reduce entanglement related serious injuries and mortality of right, humpback and fin whales

to insignificant levels approaching zero within five years of its implementation. The ALWTRP is a "work-in-progress", and revisions are made to the regulations as new information and technology becomes available. Because gear entanglements of right, humpback and fin whales have continued to occur, including serious injuries and mortality, new and revised regulatory measures are anticipated. These changes are made with the input of the Atlantic Large Whale Take Reduction Team (ALWTRT), which is comprised of representatives from federal and state government, the fishing industry, scientists and conservation organizations.

Lobster and gillnet gear are known to entangle endangered large whales. Regulations introduced in Massachusetts waters requiring modifications to lobster and gillnet fishing came into effect January 1, 2003. The purpose of the new requirements is to reduce the risk of right whale entanglements in an area that has a known congregation of right whales each year. From January 1 through April 30, single lobster pots are banned, and ground lines must be either sinking or neutrally buoyant. Buoy lines must also be mostly sinking line and must include a weak link. From May 1 through December 31, lobstermen must use at least two of the following gear configurations: buoy lines 7/16-inch diameter or less, a weak link at the buoy of 600 pounds breaking strength, sinking buoy lines, and sinking or neutrally buoyant ground lines.

Gear Modification and Research
Gear research and development is a critical component of the ALWTRP, with the aim of finding new ways of reducing the number and severity of protected species-gear interactions while still allowing for fishing activities. At the outset, the gear research and development program followed two approaches: (a) reducing the number of lines in the water without shutting down fishery operations, and (b) devising lines that are weak enough to allow whales to break free and at the same time strong enough to allow continued fishing. Development of gear modifications are ongoing and are primarily used to minimize risk of large whale entanglement. This regulatory development has now moved into the next phase and reducing the profile of groundlines in the water column is the focus and priority, while reducing risk associated with vertical lines is being discussed and assessed and ongoing research is continuing to develop and alleviate future risk. This aspect of the ALWTRP is important, in that it incorporates the knowledge and encourages the participation of industry in the development and testing of modified and experimental gear.

Large Whale Disentanglement Network
In recent years, NMFS has greatly increased funding for the Whale Disentanglement Network, purchasing equipment caches to be located at strategic spots along the Atlantic coastline, supporting training for fishers and biologists, purchasing telemetry equipment, etc. This has resulted in an expanded capacity for disentanglement along the Atlantic seaboard including offshore areas. The Center for Coastal Studies (CCS), under NMFS authorization, has responded to numerous calls since 1984 to disentangle whales entrapped in gear, and has developed considerable expertise in whale disentanglement. NMFS has supported this effort financially since 1995. Memorandum of Understandings developed with the USCG ensure their participation and assistance in the disentanglement effort. Hundreds of Coast Guard and Marine Patrol workers have received training to assist in disentanglements. As a result of the success of

the disentanglement network, NMFS believes that many whales that may otherwise have succumbed to complications from entangling gear have been freed and survived the ordeal. Humpback and right whales are two species that commonly become entangled due to fishing gear. Over the past five years the disentanglement network has been involved in many successes and has assisted many whales shed gear or freed them by disentangling gear from 35 humpback and 11 right whales (CCS web site).

Sighting Advisory System

Although the Sighting Advisory System (SAS) was developed primarily as a method of locating right whales and alerting mariners to right whale sighting locations in a real time manner, the SAS also addresses entanglement threats. Fishermen can obtain SAS sighting reports and make necessary adjustments in operations to decrease the potential for interactions with right whales. Some of these sighting efforts have resulted in successful disentanglement of right whales.

Education and Outreach

Education and outreach activities are considered one of the primary tools to reduce the threats to all protected species from human activities, including fishing activities. Outreach efforts for fishermen under the ALWTRP are fostering a more cooperative relationship between all parties interested in the conservation of threatened and endangered species. NMFS has also been active in public outreach to educate fishermen regarding sea turtle handling and resuscitation techniques. NMFS has conducted workshops with longline fishermen to discuss bycatch issues including protected species, and to educate them regarding handling and release guidelines. NMFS intends to continue these outreach efforts in an attempt to increase the survival of protected species through education on proper release techniques.

Reducing Threats to ESA-listed Sea Turtles

NMFS has implemented multiple measures to reduce the capture and mortality of sea turtles in fishing gear, and other measures to contribute to the recovery of these species. While some of these actions occur outside of the action area for this consultation, the measures affect sea turtles that do occur within the action area.

Sea Turtle Handling and Resuscitation Techniques

NMFS has developed and published as a final rule in the *Federal Register* (66 FR 67495, December 31, 2001) sea turtle handling and resuscitation techniques for sea turtles that are incidentally caught during scientific research or fishing activities. Persons participating in fishing activities or scientific research are required to handle and resuscitate (as necessary) sea turtles as prescribed in the final rule. These measures help to prevent mortality of hard-shelled turtles caught in fishing or scientific research gear.

Sea Turtle Entanglements and Rehabilitation

A final rule (70 FR 42508) published on July 25, 2005, allows any agent or employee of NMFS, the USFWS, the U.S. Coast Guard, or any other Federal land or water management agency, or any agent or employee of a state agency responsible for fish and wildlife, when acting in the course of his or her official duties, to take endangered sea turtles encountered in the marine environment if such taking is necessary to aid a sick, injured, or entangled endangered sea turtle, or dispose of a dead endangered sea turtle, or salvage a dead endangered sea turtle that may be useful for scientific or educational purposes. NMFS already affords the same protection to sea turtles listed as threatened under the ESA (50 CFR 223.206(b)).

Education and Outreach Activities
Education and outreach activities do not directly reduce the threats to ESA-listed sea turtles. However, education and outreach are a means of better informing the public of steps that can be taken to reduce impacts to sea turtles (*i.e.*, reducing light pollution in the vicinity of nesting beaches) and increasing communication between affected user groups (*e.g.*, the fishing community). For the HMS fishery, NMFS has been active in public outreach to educate fishermen regarding sea turtle handling and resuscitation techniques. For example, NMFS has conducted workshops with longline fishermen to discuss bycatch issues including protected species, and to educate them regarding handling and release guidelines. NMFS intends to continue these outreach efforts in an attempt to increase the survival of protected species through education on proper release techniques.

Sea Turtle Stranding and Salvage Network (STSSN)
As is the case with education and outreach, the STSSN does not directly reduce the threats to sea turtles. However, the extensive network of STSSN participants along the Atlantic and Gulf of Mexico coasts not only collects data on dead sea turtles, but also rescues and rehabilitates live stranded turtles. Data collected by the STSSN are used to monitor stranding levels and identify areas where unusual or elevated mortality is occurring. These data are also used to monitor incidence of disease, study toxicology and contaminants, and conduct genetic studies to determine population structure. All of the states that participate in the STSSN tag live turtles when encountered (either via the stranding network through incidental takes or in-water studies). Tagging studies help provide an understanding of sea turtle movements, longevity, and reproductive patterns, all of which contribute to our ability to reach recovery goals for the species.

Sea Turtle Disentanglement Network
NMFS Northeast Region established the Northeast Atlantic Coast Sea Turtle Disentanglement Network (STDN) in 2002. This program was established in response to the high number of leatherback sea turtles found entangled in pot gear along the U.S. Northeast Atlantic coast. The STDN is considered a component of the larger STSSN program. The NMFS Northeast Regional Office oversees the STDN program. In Massachusetts, NOAA Fisheries has partnered with the Provincetown Center for Coastal Studies (PCCS) for response to entangled sea turtles in MA. Since the programs inception in 2002, MA responders have received over 50 sea turtle entanglement reports, which resulted in over 20 live turtle disentanglements in MA waters.

Summary and synthesis of the Status of Species, Environmental Baseline, and Cumulative Effects sections

The Status of the Species, Environmental Baseline, and Cumulative Effects Sections, taken together, establish a "baseline" against which the effects of the proposed action are analyzed to determine whether the action—the proposed authorization of the Cape Wind project by MMS- is likely to jeopardize the continued existence of the species. To the extent available information allows, this "baseline" (which does not include the future effects of the proposed action) would be compared to the backdrop plus the effects of the proposed action. The difference in the two trajectories would be reviewed to determine whether the proposed action is likely to jeopardize the continued existence of the species. This section synthesizes the Status of the Species, the Environmental Baseline, and Cumulative Effects sections as best as possible given that some information on sea turtles is quantified, yet much remains qualitative or unknown.

Summary of status of species
Based on recent estimates, NMFS considers the best approximation for the number of *North Atlantic right whales* to be 300 +/- 10%. Losses of adult whales due to ship strikes and entanglements in fishing gear continue to depress the recovery of this species and the right whale population continues to be declining.

The best available population estimate for *humpback whales* in the North Atlantic Ocean is 10,600 animals. Anthropogenic mortality associated with ship strikes and fishing gear entanglements is significant. Modeling using data obtained from photographic mark-recapture studies estimates the growth rate of the Gulf of Maine feeding population at 6.5% (Barlow and Clapham 1997). With respect to the species as a whole, there are also indications of increasing abundance for the eastern and central North Pacific stocks. However, trend and abundance data is lacking for the western North Pacific stock, the Southern Hemisphere humpback whales, and the Southern Indian Ocean humpbacks.

The minimum population estimate for the western North Atlantic *fin whale* is 2,362 which is believed to be an underestimate. Information on the abundance and population structure of fin whales worldwide is limited. NMFS recognizes three fin whale stocks in the Pacific for the purposes of managing this species under the MMPA. Reliable estimates of current abundance for the entire Northeast Pacific fin whale stock are not available (Angliss *et al.* 2001). Stock structure for fin whales in the southern hemisphere is unknown and there are no current estimates of abundance for southern hemisphere fin whales. As this species continues to be subject to natural and anthropogenic mortality, this population is assumed to be at best stable and at worst declining.

Leatherback and *Kemp's ridley sea turtles* are endangered species, meaning that they are in danger of extinction throughout all or a significant portion of their ranges. The loggerhead sea turtle is a threatened species, meaning that it is likely to become an endangered species in the foreseeable future throughout all or a significant portion of its range. Green sea turtles in U.S. waters are listed as threatened except for the Florida breeding population which is listed as endangered. For purposes of this Opinion, NMFS considers the trend of the sea turtle species considered in this Opinion to be declining for loggerhead, leatherback, and green sea turtles, and stable for Kemp's ridley sea turtles. These trends are the result of past, present, and likely future human activities and natural events, some effects of which are positive, some negative, and some unknown, as discussed previously in the Status of the Species, Environmental Baseline, and

Cumulative Effects Sections taken together. Additional information is provided below.

Loggerhead Sea Turtles. Loggerhead sea turtles are listed as a single species classified as "threatened" under the ESA. Loggerhead nesting occurs on beaches of the Pacific, Indian, and Atlantic oceans, and Mediterranean Sea. Genetic analyses of maternally inherited mitochondrial DNA demonstrate the existence of separate, genetically distinct nesting groups between as well as within the ocean basins (TEWG 2000; Bowen and Karl 2007).

It takes decades for loggerhead sea turtles to reach maturity. Once they have reached maturity, females typically lay multiple clutches of eggs within a season, but do not typically lay eggs every season (NMFS and USFWS 1991a). There are many natural and anthropogenic factors affecting survival of turtles prior to their reaching maturity as well as for those adults who have reached maturity. As described in sections 3.1 and 4.0, negative impacts causing death of various age classes occur both on land and in the water. In addition, given the distances traveled by loggerheads in the course of their development, actions to address the negative impacts require the work of multiple countries at both the national and international level (NMFS and USFWS 2007a). Many actions have been taken to address known negative impacts to loggerhead sea turtles. However, many remain unaddressed, have not been sufficiently addressed, or have been addressed in some manner but whose success cannot be quantified. There are no population estimates for loggerhead sea turtles. Sea turtle nesting data, in terms of the number of nests laid each year, is collected for loggerhead sea turtles for at least some nesting beaches within each of the ocean basins and the Mediterranean Sea. From this, the number of reproductively mature females utilizing those nesting beaches can be estimated based on the presumed remigration interval and the average number of nests laid by a female loggerhead sea turtle per season. These estimates provide a minimum count of the number of loggerhead sea turtles in any particular nesting group. The estimates do not account for adult females who nest on beaches with no or little survey coverage, and do not account for adult males or juveniles of either sex. The proportion of adult males to females from each nesting group, and the age structure of each loggerhead nesting group is currently unknown. For these reasons, nest counts cannot be used to estimate the total population size of a nesting group and, similarly, trends in the number of nests laid cannot be used as an indicator of the population trend (whether decreasing, increasing or stable) (Meylan 1982; Ross 1996; Zurita *et al.* 2003; Hawkes *et al.* 2005; Loggerhead TEWG 2007).

Nevertheless, nest count data are a valuable source of information for each loggerhead nesting group and for loggerheads as a species since the number of nests laid reflect the reproductive output of the nesting group each year, and also provide insight on the contribution of each nesting group to the species. Based on a comparison of the available nesting data, the world's largest known loggerhead nesting group (in terms of estimated number of nesting females) occurs in Oman in the northern Indian Ocean where an estimated 20,000-40,000 females nest each year (Baldwin *et al.* 2003). The world's second largest known loggerhead nesting group occurs along the east coast of the United States where approximately 15,966 females nest per year on south Florida beaches (based on a mean of 65,460 nests laid per year from 1989-2006; NMFS and USFWS 2007a). The world's third largest loggerhead nesting group also occurs in the United States, from approximately northern Florida through North Carolina. However, the mean nest count for this nesting group, the third largest loggerhead nesting group in the world, is 5,151 nests laid per year (NMFS and USFWS 2007a) – less than 1/10th the mean number of nests laid by the south Florida nesting group. Thus, while loggerhead nesting occurs at multiple sites within multiple ocean basins and the Mediterranean Sea, the extent of nesting is disproportionate amongst the various sites and only two geographic areas, Oman and south Florida, U.S., account for the majority of nesting for the species, worldwide.

Declines in loggerhead nesting have been noted at nesting beaches throughout the range of the species. These include nesting for the south Florida nesting group – the second largest loggerhead nesting group in the world and the largest of all of the loggerhead nesting groups in the Atlantic (Dodd 2003; Meylan *et al.* 2006; Letter to NMFS from the Director, Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission, October 25, 2006; Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission web posting November 2007; NMFS and USFWS 2007a).

In light of the above, for purposes of this Opinion, NMFS considers the trend for loggerheads as a species to be declining. NMFS recognizes that the available nest count data only provides information on the number of females currently nesting, and is not necessarily a reflection of the number of mature females available to nest or the number of immature females that will reach maturity and nest in the future. Also, the trend in the number of nests laid is not a reflection of the overall trend in any nesting group given that the proportion of adult males to females, and the age structure of each loggerhead nesting group is currently unknown. This determination that the trend for loggerheads as a species is declining provides benefit of the doubt to the species given its threatened classification under the ESA, the many on-going negative impacts to the species across all areas of its range and to all age classes, and information to suggest that fewer nests are being laid (potentially reducing the number of offspring that will mature and contribute to the species' continued existence).

Leatherback turtles. Leatherback sea turtles are listed as a single species classified as "endangered" under the ESA. Leatherbacks are widely distributed throughout the oceans of the world, and are found in waters of the Atlantic, Pacific, and Indian Oceans, the Caribbean Sea, Mediterranean Sea, and the Gulf of Mexico (Ernst and Barbour 1972). Leatherback nesting occurs on beaches of the Atlantic, Pacific and Indian Oceans as well as in the Caribbean (NMFS and USFWS 2007b).

Like loggerheads, sexually mature female leatherbacks typically nest in non-successive years and lay multiple clutches in each of the years that nesting occurs. Leatherbacks face a multitude of threats that can cause death prior to and after reaching maturity. Some activities resulting in leatherback mortality have been addressed. However, many others remain to be addressed. Given their range and distribution, international efforts are needed to address all known threats to leatherback sea turtle survival (NMFS and USFWS 2007b).

There are some population estimates for leatherback sea turtles although there appears to be considerable uncertainty in the numbers. In 1980, the global population of adult leatherback females was estimated to be approximately 115,000 (Pritchard 1982). By 1995, this global population of adult females was estimated to be 34,500 (Spotila *et al.* 1996). However, the most recent population size estimate for the North Atlantic alone is 34,000-94,000 adult leatherbacks (Leatherback TEWG 2007; NMFS and USFWS 2007b).

Leatherback nesting in the eastern Atlantic (*i.e.*, off Africa) and in the Caribbean appears to be stable, but there is conflicting information for some sites and it is certain that some nesting groups (*e.g.*, St. John and St. Thomas, U.S. Virgin Islands) have been extirpated (NMFS and USFWS 1995). Data collected for some nesting beaches in the western Atlantic, including leatherback nesting beaches in the U.S. clearly indicate increasing numbers of nests (NMFS SEFSC 2001; NMFS and USFWS 2007b). However, declines in nesting have been noted for beaches in the western Caribbean (NMFS and USFWS 2007b). The largest leatherback rookery in the western Atlantic remains along the northern coast of South America in French Guiana and Suriname. More than half the present world leatherback population is estimated to be nesting on the beaches in and close to the Marowijne River Estuary in Suriname and French Guiana (Hilterman and Goverse 2004). The long-term trend for the Suriname and French Guiana nesting group seems to show an increase (Hilterman and Goverse 2004). In 2001, the number of nests for Suriname and French Guiana combined was 60,000, one of the highest numbers observed for this region in 35 years (Hilterman and Goverse 2004). Studies by Girondot *et al.* (2007) also suggest that the trend for the Suriname - French Guiana nesting population over the last 36 years is stable or slightly increasing.

Increased nesting by leatherbacks in the Atlantic is not expected to affect leatherback abundance in the Pacific where the abundance of leatherback turtles on nesting beaches has declined dramatically over the past 10 to 20 years (NMFS and USFWS 2007b). Although genetic analyses suggest little difference between Atlantic and Pacific leatherbacks (Bowen and Karl 2007), it is generally recognized that there is little to no genetic exchange between these turtles. In addition, Atlantic and Pacific leatherbacks are impacted by different activities (NMFS and USFWS 1992; 1998a). However, the ESA-listing of leatherbacks as a species means that the effects of a proposed action must, ultimately, be considered at the species level for section 7 consultations. In light of the above, for purposes of this Opinion, NMFS considers the trend for leatherbacks, as a species, to be declining. NMFS recognizes that the nest count data available for leatherbacks in the Atlantic clearly indicates increased nesting at many sites, and that the activities affecting declines in nesting by leatherbacks in the Pacific are not the same as those activities affecting leatherbacks in the Atlantic. However, NMFS also recognizes that the nest count data, including data for leatherbacks in the Atlantic, only provides information on the number of females currently nesting, and is not necessarily a reflection of the number of mature females in the Atlantic that are available to nest or the number of immature females that will reach maturity and nest in the future. Also, the trend in the number of nests laid is not a reflection of the overall trend in any leatherback population given that the proportion of adult males to females, and the age structure of the population(s) is unknown. This determination that the trend for leatherbacks as a species is declining provides benefit of the doubt to the species given its endangered classification under the ESA, the many on-going negative impacts to the species across all areas of its range and to all age classes, the uncertainty in the population estimates, the dramatic decline in leatherback nesting in the Pacific, and the disproportionate nesting of leatherbacks with more than half of the species nesting occurring in one area of the world (thus negative impacts to this area could have very large impacts on reproductive success of the species).

Kemp's Ridley Sea Turtles. Kemp's ridley sea turtles are listed as a single species classified as "endangered" under the ESA. Kemp's ridleys occur in the Atlantic Ocean and Gulf of Mexico. The only major nesting site for ridleys is a single stretch of beach near Rancho Nuevo, Tamaulipas, Mexico (Carr 1963; USFWS and NMFS 1992; NMFS and USFWS 2007c). Approximately 60% of its nesting occurs here with a limited amount of scattered nesting to the north and south of the primary nesting beach (NMFS and USFWS 2007c).

Age to maturity for Kemp's ridley sea turtles occurs earlier than for either loggerhead or leatherback sea turtles. However, maturation may still take 10-17 years (NMFS and USFWS 2007c). As is the case with the other turtle species, adult, female Kemp's ridleys typically lay multiple nests in a nesting season but do not typically nest every nesting season (TEWG 2000; NMFS and USFWS 2007c). Although actions have been taken to protect the nesting beach habitat, and to address activities known to be negatively impacting Kemp's ridley sea turtles, Kemp's ridleys continue to be impacted by anthropogenic activities.

Nest count data provides the best available information on the number of adult females nesting each year. As is the case with the other sea turtles species discussed above, nest count data must be interpreted with caution given that these estimates provide a minimum count of the number of nesting Kemp's ridley sea turtles. In addition, the estimates do not account for adult males or juveniles of either sex. Without information on the proportion of adult males to females, and the age structure of the Kemp's ridley population, nest counts cannot be used to estimate the total population size and, similarly, trends in the number of nests laid cannot be used as an indicator of the population trend (whether decreasing, increasing or stable) (Meylan 1982; Ross 1996; Zurita *et al.* 2003; Hawkes *et al.* 2005; Loggerhead TEWG 2007). Nevertheless, the nesting data does provide valuable information on the extent of Kemp's ridley nesting and the trend in the number of nests laid. Estimates of the adult female nesting population reached a low of approximately 250-300 in 1985 (USFWS and NMFS 1992; TEWG 2000). From 1985 to 1999, the number of nests observed at Rancho Nuevo, and nearby beaches increased at a mean rate of 11.3% per year (TEWG 2000). Current estimates suggest an adult female population of 7,000-8,000 Kemp's ridleys (NMFS and USFWS 2007c).

The most recent review of the Kemp's ridley as a species suggests that it is in the early stages of recovery (NMFS and USFWS 2007c). The nest count data indicates increased nesting and an increased number of nesting females in the population. In light of this information, for purposes of this Opinion, NMFS considers the trend for Kemp's ridley sea turtles to be stable. This determination that the trend for Kemp's ridleys as a species is stable provides benefit of the doubt to the species given the species classification of "endangered" under the ESA, the caveats associated with using nesting data as indicators of population size and population trends, that the estimated number of nesting females in the current population is still far below historical numbers (Stephens and Alvarado-Bremer 2003; NMFS and USFWS 2007c), the many on-going negative impacts to the species, and given that the majority of nesting for the species occurs in one area.

Green Sea Turtles. Green sea turtles are listed as both threatened and endangered under the ESA. Breeding colony populations in Florida and on the Pacific cost of Mexico are considered endangered while all others are considered threatened. Due to the inability to distinguish between these populations away from the nesting beach, for this Opinion, green turtles are considered endangered wherever they occur in U.S. waters. Green turtles are distributed circumglobally, and can be found in the Pacific, Indian and Atlantic Oceans as well as the Mediterranean Sea (NMFS and USFWS 1991; Seminoff 2004; NMFS and USFWS 2007d).

Green sea turtles appear to have the latest age to maturity of all of the sea turtles with age at maturity occurring after 2-5 decades (NMFS and USFWS 2007d). As is the case with all of the other turtle species mentioned here, mature green sea turtles typically nest more than once in a nesting season but do not nest every nesting season. As is also the case with the other turtle species, green sea turtles face numerous threats on land and in the water that affect the survival of all age classes.

A review of 32 Index Sites distributed globally revealed a 48% to 67% decline in the number of mature females nesting annually over the last 3-generations (Seminoff 2004). For example, in the eastern Pacific, the main nesting sites for the green sea turtle are located in Michoacan, Mexico, and in the Galapagos Islands, Ecuador where the number of nesting females exceed 1,000 females per year at each site (NMFS and USFWS 2007d). Historically, however, greater than 20,000 females per year are believed to have nested in Michoacan, alone (Cliffton *et al.* 1982; NMFS and USFWS 2007d). However, the decline is not consistent across all green sea turtle nesting areas. Increases in the number of nests counted and, presumably, the number of mature females laying nests, were recorded for several areas (Seminoff 2004; NMFS and USFWS 2007d). Of the 32 index sites reviewed by Seminoff (2004), the trend in nesting was described as: increasing for 10 sites, decreasing for 19 sites, and stable (no change) for 3 sites. Of the 46 green sea turtle nesting sites reviewed for the 5-year status review, the trend in nesting was described as increasing for 12 sites, decreasing for 4 sites, stable for 10 sites, and unknown for 20 sites (NMFS and USFWS 2007d). The greatest abundance of green sea turtle nesting in the western Atlantic occurs on beaches in Tortuguero, Costa Rica (NMFS and USFWS 2007d). Nesting in the area has increased considerably since the 1970's and nest count data from 1999-2003 suggest nesting by 17,402-37,290 females per year (NMFS and USFWS 2007d). One of the largest nesting sites for green sea turtles worldwide is still believed to be on the beaches of Oman in the Indian Ocean (Hirth 1997; Ferreira *et al.* 2003; NMFS and USFWS 2007d). However, nesting data for this area has not been published since the 1980's and updated nest numbers are needed (NMFS and USFWS 2007d).

The results of genetic analyses show that green sea turtles in the Atlantic do not contribute to green sea turtle nesting elsewhere in the species range (Bowen and Karl 2007). Therefore, increased nesting by green sea turtles in the Atlantic is not expected to affect green sea turtle abundance in other ocean basins in which the species occurs. However, the ESA-listing of green sea turtles as a species across ocean basins means that the effects of a proposed action must, ultimately, be considered at the species level for section 7 consultations. In light of the above, for purposes of this Opinion, NMFS considers the trend for green sea turtles, as a species, to be declining. NMFS recognizes that the nest count data available for green sea turtles in the Atlantic clearly indicates increased nesting at many sites. However, NMFS also recognizes that the nest count data, including data for green sea turtles in the Atlantic, only provides information on the number of females currently nesting, and is not necessarily a reflection of the number of mature females available to nest or the number of immature females that will reach maturity and nest in the future. Also, the trend in the number of green sea turtle nests laid is not an indication of the overall population trend given that the proportion of adult males to females, and the age structure of the population(s) is unknown. Finally, given the late age to maturity for green sea turtles (20 to 50 years; Balazs 1982, Frazer and Ehrhart 1985; Seminoff 2004), caution is urged regarding the trend for any of the nesting groups since no area has a dataset spanning a full green sea turtle generation (NMFS and USFWS 2007d). This determination that the trend for green sea turtles as a species is declining provides benefit of the doubt to the species given its endangered and threatened classification under the ESA, the many on-going negative impacts to the species across all areas of its range and to all age classes, the declining or uncertain trend in nesting for the majority of the world's nesting sites for green sea turtles, and the lack of up-to-date nesting information for the largest green sea turtle nesting site in the Indian Ocean and possibly the world.

EFFECTS OF THE ACTION

This section of an Opinion assesses the direct and indirect effects of the proposed action on threatened and endangered species or critical habitat, together with the effects of other activities that are interrelated or interdependent (50 CFR 402.02). Indirect effects are those that are caused later in time, but are still reasonably certain to occur. Interrelated actions are those that are part of a larger action and depend upon the larger action for their justification. Interdependent actions are those that have no independent utility apart from the action under consideration (50 CFR 402.02). Several listed species are likely to be present in the action area at various times of the year and may therefore be exposed to effects of the proposed action.

Summary of Information Related to Sea Turtle Presence in the Action Area

Leatherback sea turtles are the most common species of sea turtles in Massachusetts waters with frequent sightings in the summer and fall as this species pursues its preferred jellyfish prey. While in Massachusetts waters, loggerhead turtles feed on a variety of foods including hermit and spider crabs, whelks, blue mussels, and moon snails. During the summer months, Kemp's ridleys forage on mussels and crabs. The green sea turtle frequents Massachusetts waters with some degree of regularity but is not considered common as there are few records for it north of Cape Cod. The green turtles found in Massachusetts are three- to four -year-old subadults, 24-30 inches long, and weigh about 50lbs. Green turtles are the most herbivorous of all the sea turtles and feed mainly on submerged aquatic vegetation (SAV) including seagrasses and macroalgae.

One of the main factors influencing sea turtle presence in northern waters is seasonal temperature patterns (Ruben and Morreale 1999). Temperature is correlated with the time of year, with the warmer waters in the late spring, summer, and early fall being the most suitable for cold-blooded sea turtles. Nantucket Sound is not a concentration area for sea turtles but sea turtles are routinely documented in these waters. Observational data suggests that sea turtles are most common in eastern Nantucket Sound where waters are deepest and nearest to the coastal migratory path towards Cape Cod Bay. Sea turtles are most likely to occur in the action area between June and October, although individuals may be present in the early weeks of November as well.

To some extent, water depth also dictates the number of sea turtles occurring in a particular area. Waters in the action area range from approximately 0 to 70 feet deep. Satellite tracking studies of sea turtles in the Northeast found that foraging turtles mainly occurred in areas where the water depth was between approximately 16 and 49 ft (Ruben and Morreale 1999). This depth was interpreted not to be as much an upper physiological depth limit for turtles, as a natural limiting depth where light and food are most suitable for foraging turtles (Morreale and Standora 1990). Sea turtles are capable of dives to substantial depths (300-1000 m; Eckert et al. 1986 in Stabenau et al. 1991), and chelonid turtles have been found to make use of deeper, less productive channels as resting areas that afford protection from predators because of the low energy, deep water conditions. Leatherbacks have been shown to dive to great depths, often spending a considerable amount of time on the bottom (NMFS 1995).

The action area and the depths preferred by sea turtles do overlap and preferred sea turtle forage items occur in the action area (MMS 2008), suggesting that leatherbacks, loggerheads, Kemp's ridleys and green sea turtles are likely to be foraging while in the action area. Surveys reported in the BA indicate that there are several areas of SAV within the action area, including concentrations of macroalgae and some sea grass beds. Additionally, surveys indicate that there is a diverse and plentiful benthic community in Nantucket Sound. Sponges, bivalves, crabs, and other crustaceans all occur in the action area. Lazell (1980) confirms that arctic jellyfish, one of the preferred prey of leatherback sea turtles, also occur in Nantucket Sound in the summer months. In addition to foraging in the action area, migrating loggerhead, Kemp's ridley, green or leatherback sea turtles may be found swimming through the action area as they complete northward migrations in the spring and southward migrations in the fall. Sea turtles may also transit the action area while moving into or out of nearby foraging areas (i.e., Cape Cod Bay or Stellwagen Bank), or may be resting on or near the bottom.

While there have been no surveys of Nantucket Sound specifically designed to detect sea turtles, there is recent incidental observation data available for leatherback, loggerhead, Kemp's ridley and green sea turtles as well as historic records for each of these species. For example, several entangled leatherback sea turtles located in Nantucket Sound are reported to NMFS each year (NMFS unpublished data). A review of the OBIS SEAMAP database includes sightings data for all four sea turtle species in Nantucket Sound (OBIS SEAMAP online mapper, accessed on September 5, 2008). Satellite tracking data demonstrates the use of Nantucket Sound by Kemp's ridley, loggerhead and green sea turtles (seaturtle.org database, accessed on September 5, 2008). Lazell (1980) examined the data available on sea turtles in Massachusetts and in Nantucket Sound specifically. The paper includes information which confirms the use of Nantucket Sound by loggerheads, leatherbacks, Kemp's ridley and green sea turtles during the summer months.

More recently, Mass Audubon conducted surveys for terns over an approximately four week period in 2002, 2003 and 2004. Both shipboard and aerial surveys were conducted. In their reports, Mass Audubon includes information on sea turtle sightings. As this information was collected in the action area, it represents important information on the presence of sea turtles in this area. In each of the three study years, aerial surveys were conducted along sixteen fixed, parallel transects oriented north to south. The grid encompassed nearly all the waters south of Cape Cod between Martha's Vineyard and the Monomoy Island National Wildlife Refuge in Chatham (see Figure 3 for map of surveyed area and sea turtle sightings). The transects extended south to an east-west line roughly even with Great Point, Nantucket. Individual transects were positioned at 7,500 foot intervals, and the total combined linear length of all 16 transects was 247.4 miles (approximately 398 linear kilometers). The area surveyed was approximately 888 square kilometers. Flights were conducted at an average altitude of 500 feet on days with good atmospheric clarity (visibility >10 miles). Flights lasted approximately 2.5 hours each day. Several boat surveys also occurred but no sea turtle sightings were reported for these surveys.

In 2002, eleven aerial surveys were conducted between August 19 and September 19. Thirty-four sea turtles were observed (22 unidentified species, 1 Kemp's ridley, 6 leatherbacks and 5 loggerheads). In 2003, three aerial surveys occurred (June 3, July 14 and July 30). During these surveys, 28 sea turtles were observed (16 unidentified species, 8 leatherbacks, and 4 loggerheads). In 2004, eleven aerial surveys were conducted between August 7 and September 24. During these surveys, 53 sea turtles were observed (41 leatherbacks and 12 loggerheads). A total of 115 sea turtles were observed over the course of the three year study.

As sea turtles have been documented in the action area, the habitat is consistent with preferred foraging habitat of these species and forage is available, it is reasonable to expect that sea turtles will be present in the action area when project activities are occurring, most likely between June and October, and that sea turtles may be exposed to effects of the project during that time.

Summary of Information related to listed whales in the Action Area

Endangered whales migrate off the coast of Massachusetts area at various times of the year. North Atlantic right, humpback and fin whales have all been sighted in the near shore waters off Massachusetts with sightings most common in the waters of Stellwagen Bank, Cape Cod Bay and Great South Channel. In general, right whales can be anticipated to be in Massachusetts waters from December through July, humpback whales can be found in Massachusetts waters year-round, with peaks between May and August, and fin whales may be in Massachusetts waters year-round, with peaks during the summer months. A review of sightings data compiled by the Northeast Fisheries Science Center, CeTAP study data, the OBIS database, and status of the stock reports indicate that whales are rare visitors to Nantucket Sound.

In the Gulf of Maine and Cape Cod regions, humpback whales are found in three major concentration areas: Georges Bank, Stellwagen Bank, and in the northern Gulf of Maine (Waring et al. 2008). In the Gulf of Maine, humpback sightings are most frequent from mid-March through November in the Great South Channel north along the outside of Cape Cod to Stellwagen Bank and Jeffrey's Ledge. Sightings peak in May and August. NMFS Northeast Fisheries Science Center (NEFSC) has compiled humpback whale sightings data since 2002. In this time period, no humpback whales were observed in Nantucket Sound; the nearest observation to the action area was in the vicinity of Monomoy Shoals, near the northeastern tip of Nantucket Island (approximately 20km from the project footprint). Additionally, no humpback whales were sighted in the action area during NEFSC aerial and shipboard surveys (conducted in the summers of 1998, 1999, 2002, 2004 and 2006) (Waring et al 2008). While sightings data can not be used as absolute documentation of the occurrence of any particular species, it is helpful to determine patterns of occurrence and concentration areas. The best available information indicates that humpback whale occurrence in the action area is rare, with transient individuals likely to overlap only sporadically with the eastern extremes of the action area (i.e., near Monomoy). The shallow depths of Nantucket Sound and its location outside of the coastal migratory corridor likely minimizes the potential for humpback whales to occur in Nantucket Sound and, therefore, in the action area.

Similar to humpback whales, there are no documented occurrences of fin whales in Nantucket Sound (NEFSC unpublished data, and Waring et al. 2008). The nearest observations to the action area are one fin whale recorded near the Massachusetts coast near Martha's Vineyard and two fin whales observed near Monomoy Island (approximately 20km from the project footprint). The preferred feeding habitat for fin whales is over deeper waters of the continental shelf (300 to 600 feet). As depths in the action area are considerably shallower than the preferred foraging depths of this species the finding that fin whales are uncommon in the action area is consistent with what is known about their habitat preferences. The best available information indicates that fin whale occurrence in the action area is rare, with transient individuals likely to overlap only sporadically with the eastern extremes of the action area, most likely between April and October.

Sightings data of right whales in the Gulf of Maine and Cape Cod regions indicates that right whales congregate in three areas: Georges Bank, Stellwagen Bank and in the northern Gulf of Maine. Right whales are abundant in Cape Cod Bay between February and March and in the Great South Channel in May and June. They are also frequently sighted on Stellwagen Bank and Jeffrey's Ledge in the spring through fall. Right whale movements in the Gulf of Maine are understood in general; summer (June – October) foraging grounds are located in the Bay of Fundy, late spring (April – June) foraging grounds located in Great South Channel and winter foraging grounds are located in Cape Cod Bay (December – May).

Occasional right whales have been reported off Monomoy and off Great Point, Nantucket (northern tip of the island) but no right whales have been documented in Nantucket Sound (NEFSC unpublished data, Waring et al. 2008). Only one source included information on a whale in Nantucket Sound. Mate et al. (1997) reports data for several North Atlantic right whales outfitted with satellite tags. One right whale female, tagged in the Bay of Fundy on

August 24, 1990, transited Nantucket Sound in 1997 accompanied by her calf. However, this whale was only present in Nantucket Sound for a brief period of time (i.e., less than one day) and moved rapidly during that time (i.e., approximately 89.6km/day or 3.7km/hour). Right whales have been intensely studied in the Gulf of Maine and in Massachusetts waters. It is likely that if right whales were using Nantucket Sound on more than rare, unpredictable occasions, there would be documented sightings. The best available information indicates that like the other large whale species, right whale occurrence in the action area is extremely rare, with transient individuals likely to overlap only sporadically with the eastern extremes of the action area between December and June.

As explained above, only rare, transient whales occur in Nantucket Sound. As such, NMFS has determined that it is extremely unlikely that listed whales would occur within the project footprint. However, as occasional whales have been documented off of Monomoy and Great Point, these species may occasionally occur in the eastern extremes of the action area (i.e., near Monomoy Island and off Great Point). However, any occurrence of whales in the action area is expected to be sporadic and transient. The lack of whales in the action area is consistent with the finding that these habitats are shallower than the areas where these whales typically occur and are outside of their normal coastal migratory route.

Effects of the Project

As explained above, sea turtles may be distributed throughout the action area between June and October each year. Right, humpback and fin whales may occasionally occur near the eastern edge of the action area but, based on the best available information, are likely to be rare within the action area and extremely unlikely to occur in the project footprint (i.e., the WTG site or along the cable routes). The proposed action involves several stages of activity in various locations (i.e., submarine cable route and the WTG site on Horseshoe Shoal). The sections below will outline potential effects from the following sources: (1) construction of the facility include submarine cables and the WTGs themselves, (2) operation and maintenance of the facility, (3) pre-construction geotechnical and geophysical surveys, and, (4) decommissioning. In addition to these categories of effects, MMS provided information in the BA and DEIS on non-routine and accidental events. These events include oil spills, cable repair, and vessel collisions with a monopole. Effects of these non-routine and accidental events are also discussed below.

Construction and Operation of the Project

The major construction aspects of the project involve (1) the installation of the inner-array cables; (2) the installation of the submarine cables; and (3) the installation of monopiles associated with the WTGs and the ESP. Other construction activities include the assembly of the WTGs and ESP as well as the connection of the submarine cables to the land based cables at Lewis Bay. This section will also consider the effects of exposure to construction and operation related noise and construction and operation/maintenance vessel traffic.

Interactions with Cable Laying Equipment

Both the inner-array cables and the submarine cable will be installed with a jet plow and cable laying barge. Cables will be laid within the WTG array and from the ESP to Yarmouth, MA.

Due to the depths and location within Horseshoe Shoal and towards the south shore of Cape Cod and the lack of evidence of whales occurring in these areas, whales are expected to be extremely rare along the submarine cable route and along the inner-array cable route. As such, NMFS has determined that it is unreasonable to anticipate that a whale would occur along the cable route and subsequently it is unreasonable to expect that a whale would interact with cable laying operations. As such, NMFS has determined that the potential for the cable laying operations to affect whales is discountable.

The jet plow uses jets of water to liquefy the sediment, creating a trench in which the cable is laid. Sea turtles in the path of the cable could theoretically collide with the vessel towing the plow. Cable laying operations proceed at speeds of <1 knot. At these speeds, any sea turtle that is encountered on the bottom is expected to be able to avoid collision or interaction with the cable laying operations. Additionally, as the cable will be taut as it is unrolled and laid in the trench, there is no risk of entanglement. Although any sea turtles present in the vicinity of the cable laying may be displaced from the area, displacement would be temporary for the duration of the jet pass (i.e., several minutes). The cable trench will be no more than 6 feet wide. As such, any displacement would cause a turtle to make a temporary shift in swimming direction for up to several minutes. This is not likely to affect the ability of the individual to complete any essential function (i.e., foraging, resting, migrating) that may take place along the cable route. Based on this information, sea turtles colliding or directly interacting with cable laying and jetting equipment are extremely unlikely to occur and, therefore, discountable. The effects of suspended sediment and noise associated with the cable laying and impacts to benthic resources are discussed in detail below.

Light Pollution
Most construction activities (pile driving, WTG assembly) will be limited to daylight hours. However, cable laying operations would take place 24 hours per day, 7 days a week during installation. The submarine transmission cable will take approximately 2-4 weeks to complete and the inner array cable will be installed over several months. Construction and support vessels would be required to display lights when operating at night and deck lights would be required to illuminate work areas. However, lights would be downshielded to illuminate the deck, and would not intentionally illuminate surrounding waters. If sea turtles or their prey are attracted to the lights, it could increase the potential for interaction with equipment or associated turbidity. However, due to the nature of project activities, listed species and their prey are more likely to be displaced from the immediate area by seafloor disturbance, turbidity, and noise than attracted by lighting. As such, NMFS has determined that any effects of project lighting on sea turtles or whales will be insignificant.

In addition to vessel lighting, the WTGs will be lit for navigational and aeronautical safety. Sea turtle hatchlings are known to be attracted to lights and adversely affected by artificial beach lighting, which disrupts proper orientation towards the sea. However, nesting does not occur in Massachusetts, and hatchlings are not known to be present in Massachusetts waters. If this lighting resulted in the attraction of sea turtles or their prey, no effects to sea turtles would occur as they are not likely to collide with the stationary wind turbine monopile. As such, NMFS has determined that any adverse effects of project lighting on sea turtles or whales will be discountable.

Destruction of Prey Resources/Loss of Foraging Habitat

Activities that disturb the sea floor will also affect benthic communities, and can cause effects to sea turtles by reducing the numbers or altering the composition of the species upon which sea turtles prey. Activities that may affect the sea floor and result in the loss of foraging resources for listed species include:

- Cable installation;
- WTG and ESP installation;
- G and G surveys; and,
- Scour protection (scour mats and rock armoring).

Loss of Benthic Resources/Habitat

The proposed action will result in both the temporary disturbance and permanent loss of benthic habitat. Effects to benthic resources and habitat will be restricted to the area within the project footprint and along the cable route where sediment disturbing activities will occur. As no whales are expected to occur in the project footprint or along the cable route, whales will not be exposed to effects related to the loss of benthic resources or habitat. As such, the discussion below will focus on the effects on sea turtles. As noted above, surveys indicate that suitable depths and forage for leatherback, loggerhead, Kemp's ridley and green sea turtles exist in the action area and that individuals from any of these species are likely to be present in the action area between June and October.

The installation of the submarine transmission and inner-array cables will result in temporary impacts to approximately 866 acres (approximately 5% of the action area). This accounts for the 4-6 foot wide trench that will be jetted along the 12.5 mile submarine transmission cable and the 66.7 miles of inner-array cables. The jetting process will affect benthic resources and habitat in two ways: entrainment of microorganisms and displacement or burial of other benthic resources. This is likely to result in a temporary loss of forage items and a temporary reduction in the amount of benthic habitat available for foraging sea turtles. Impacts associated with cable installation, barge positioning, anchoring, anchor line sweep, and the pontoon on the jet plow device would be temporary and localized. Impacts from anchor line sweep would primarily affect the sediments to a depth of between 3 and 6 inches. Anchoring locations would have disturbances to the sediment to a depth of 4 to 6 feet at each anchor deployment, leaving a temporary irregularity to the seafloor with localized mortality of infauna. Jet plow embedment would directly disturb sediments to a depth of approximately 8 feet.

Modeling was presented by MMS in the DEIS which estimated seabed scar recovery from jet plow cable burial operations. Using the assumption that 3 percent of the sediments in the jetted cross section could be injected back into the water column and that the coarse sediment column is returned to the trench, it was estimated that the dimensions of the scar left along the cable routes would be 6 feet wide and from 0.75 to 1.7 feet deep. MMS also estimated approximate recovery times for the trench scars. Based on bedload transport rates for Horseshoe Shoal and throughout Nantucket Sound, recovery rates for jetting scars along the cable route are estimated to be between 0.2 and 38 days. Recovery of jetting scars on Horeshoe Shoal is anticipated to occur within a few days. It is likely that seabed scars from cable burial in Lewis Bay would last months or until a major storm occurs.

Egg and larval stages of demersal species would experience some mortality due to burial. The temporary displacement of benthic habitats is also likely to result in the mortality and/or dispersal of other benthic organism in the footprint of the construction activities. As the jetting and cable laying occurs very slowly, most mobile organisms (i.e., crabs, finfish) are likely to be able to avoid the area where the jet plow is operating. The cable route has been designed to avoid eel grass beds in Lewis Bay. There are very limited areas of SAV, mostly macroalgae as opposed to sea grass, that will be affected by construction on Horseshoe Shoal.

The alteration of benthic habitat and the loss of benthic resources during construction could impact sea turtles. However, most mobile organisms, including most sea turtle prey items, are likely to be able to avoid the jetting. While there is likely to be some loss of sea turtle forage items, the amount of habitat affected represents a very small percentage of the available foraging habitat in Nantucket Sound. Sea turtles may temporarily shift their foraging efforts to other areas within Nantucket Sound or in the most extreme instances leave Nantucket Sound for other undisturbed foraging areas. While this would effect the movements of individual sea turtles it is likely to be temporary and is not likely to affect the ability of the sea turtle to find adequate nourishment or result in any injury or mortality of sea turtles. Recolonization of temporarily disturbed areas is expected to be rapid, with colonization by mobile organisms beginning within days and complete recolonization occurring within 3-12 months. As cable laying will occur over several months and recovery of benthic communities will take another several months, foraging opportunities along the cable route may be reduced for one to two years. However, as only a small percentage of Nantucket Sound will be affected, any movements of sea turtles to other foraging sites are likely to be localized and the benthic disturbance is not likely to cause sea turtles to leave the action area. As whales are not expected to occur in the project footprint or along the cable route, no foraging whales will be affected by the proposed action. Additionally, the action will not result in the loss of potential forage for whales occurring outside of Nantucket Sound.

The installation of the WTG monopiles and the ESP will result in the permanent loss of 0.67 acres of benthic habitat (less than 0.0042% of the project area). Although these impacts would result in permanent loss of 0.67 acres of potential foraging habitat for sea turtles, loss of this habitat is not likely to have a measurable adverse impact on normal sea turtle foraging activity. The total impacted area represents only 0.0042% percent of the over 15,000 acres of similar bottom habitat surrounding the project area. Additionally, there is no evidence to suggest that the WTG or ESP sites offer more favorable foraging habitat for sea turtles than surrounding areas. Sea turtles are likely to find suitable foraging habitat in alternate areas nearby, and any effects from the permanent loss of habitat resulting from the proposed project will be insignificant.

Because the inner-array cables and the two submarine transmission cable circuits will be buried approximately 6 feet (1.8 m) below the seabed they will not pose a physical barrier to migratory animals, including sea turtles. 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.

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 to a structure-oriented system. This may create localized changes, 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 been reported to commonly occupy areas around oil platforms (NRC 1996) which also offer similar underwater vertical structure.

More specifically, loggerheads and Kemp's ridleys could be attracted to the monopiles to feed on attached organisms since they feed on mollusks and crustaceans. Loggerheads are frequently observed around wrecks, underwater structures and reefs where they forage on a variety of mollusks and crustaceans (USFWS 2005). Leatherback turtles and green turtles however should be less likely to be attracted to the monopiles for feeding since leatherbacks are strictly pelagic and feed from the water column primarily on jellyfish and green turtles are primarily herbivores feeding on seagrasses and algae. However, if either of these forage items occur in higher concentrations near the monopiles, these species of sea turtles could also be attracted to the monopiles.

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 (9.1m) below mean high water (MHW)) would be a minor addition to the hard substrate that is already present. Due to the small amount of additional surface area in relation to the total area of the proposed action 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 alter the species composition in the action area. While the increase in structure and localized alteration of species distribution in the action area around the WTG monopiles may affect the localized movements of sea turtles in the action area and provide additional sheltering and foraging opportunities in the action area for these species, any effects will be beneficial or insignificant.

Water Quality Degradation and Increased Marine Debris
Construction activities can impact water quality in various ways, including increased turbidity and resuspension of contaminated sediments due to seafloor disturbance.

Increased Turbidity and Exposure to Contaminated Sediments
Turbidity can interfere with the ability of sea turtles and whales to forage effectively by obscuring visual detection of or dispersing potential prey. Disturbance of the sea floor through jetting and other construction activities, including pile driving, can also release contaminated sediments back into the water column, thus exposing marine organisms to contaminants that were previously attached to sediment particles.

Increased turbidity and resuspension of sediments can be expected from the following activities:
- Cable installation;
- WTG and ESP pile installation; and,
- Vessel anchoring.

Of these activities, cable installation, including jetting and backfill, is expected to generate the most turbidity and disturbance of bottom sediments. Simulations of sediment transport and deposition from jet plow embedment of the submarine cable system and inner array cables were performed and reported in MMS's BA and DEIS. These simulations 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) to estimate the suspended sediment concentrations and deposition that could result from jet plow embedment of the cables. The model results demonstrate that concentrations of suspended sediment in the water column resulting from the jet plow embedment operations are largely below 50mg/L in Nantucket Sound. The modeling results indicate 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. For example, the duration of time when suspended sediment levels will be greater than 10mg/L above background levels is 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 100mg/L for 2-3 hours.

In Lewis Bay, suspended sediments are predicted to remain in suspension considerably longer than in Nantucket Sound due to weak tidal currents. Modeling demonstrates that the concentration of suspended sediment in the water column resulting from jet plow operations in Lewis Bay will be below 500mg/L. Suspended sediment concentrations in excess of 100mg/L are generally predicted to remain for less than 2 hours with the exception of some sections along the route where durations may be as long as 6 hours. Suspended sediment concentrations in excess of 10mg/L above background are generally predicted to remain for less than 24 hours after the jet plow has passed a given point, with the exception of the area near the Yarmouth landfall where concentrations in excess of 10mg/L are predicated to remain for up to 2 days after the jet plow passes as a result of very weak currents and fine bottom sediments.

Suspended sediment is most likely to affect sea turtles if a plume causes a barrier to normal behaviors or if sediment settles on the bottom affecting sea turtle prey. As sea turtles are highly mobile they are likely to be able to avoid any sediment plume and any effect on sea turtle movements is likely to be insignificant. Additionally, the TSS levels expected (less than 500mg/L) are below those shown to have an adverse effect on fish (580mg/L for the most sensitive species, with 1,000mg/L more typical; see summary of scientific literature in Burton 1993).

As whales are extremely rare in Nantucket Sound and are not expected to occur at all in Lewis Bay, no whales are expected to be exposed to increased levels of sediment associated with the cable laying operations. Any sea turtles in the area of the cable laying operations would be exposed to an increase in suspended sediment for a short duration (2-48 hours). However, as sea turtles are highly mobile and any suspended sediment plumes will be localized and temporary, it is not likely that sea turtles would be exposed to a high suspended sediment load for a significant amount of time. Sea turtles may temporarily avoid areas with high suspended sediment loads but as any effects will be temporary, there is not likely to be any long term effect or injury associated with these alterations of movement. Any alteration in movements is likely to be temporary and local.

As noted in MMS' BA, whales and sea turtles bioaccumulate contaminants from their environment, almost exclusively through their food sources. The potential mechanism by which sediments suspended during the proposed action's construction can harm whales is through bioaccumulation of sediment-associated chemicals through ingestion of contaminated prey.

MMS has reported that analysis of sediment core samples obtained from the area of the proposed action indicate that sediment contaminant levels were below established thresholds in reference Effect Range-Low and Effects-Range-Median marine sediment quality guidelines. Therefore, the temporary and localized disturbance of these sediments during the proposed action's construction activities are not anticipated to result in increased contaminants in lower trophic levels. Therefore, neither sea turtles nor whales are likely to experience increased bioaccumulation of chemical contaminants in their tissues from the consumption of prey items in the vicinity of the proposed action, and any effects to whales or sea turtles from the disturbance of these sediments will be discountable. Since other sources of turbidity and seafloor disturbance (i.e., pile installation and scour protection placement) will be minimal compared to that caused by cable installation, the overall effect of project construction on listed species due to turbidity and exposure to contaminants is insignificant or discountable.

Increased Marine Debris
Personnel will be present onboard the barges throughout construction activities, thus presenting some potential for accidental releases of debris overboard. As noted in the Environmental Baseline section, sea turtles may be adversely affected if they become entangled in or ingest marine debris, particularly plastics that are mistaken for prey items. The discharge and disposal of garbage and other solid debris from vessels by lessees is prohibited by the MMS (30 CFR 250.300) and the USCG (MARPOL Annex V, Public Law 100-220 [Statute 1458]). The

discharge of plastics is strictly prohibited. During construction, individual crew members will be responsible for ensuring that debris is not discharged into the marine environment. Additionally, training of construction crews will include a requirement explaining that the discharge of trash and debris overboard is harmful to the environment, and is illegal under the Act to Prevent Pollution from Ships and the Ocean Dumping Ban Act of 1988. Discharge of debris will be prohibited, and violations will be subject to enforcement actions. Therefore, construction activities are not likely to result in increased marine debris.

Exposure to Electro-magnetic field

The cable system (for both the inner-array cables and each of the submarine cable 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 submarine cable systems 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 submarine cable system and inner-array cables. As presented in the DEIS and accompanying Technical Report No. 5.3.2-3 the magnetic fields associated with the operation of the inner-array cables or the submarine cable system are not anticipated to result in any adverse impacts to marine life (ICNIRP 2000; Adai, 1994; Valberg *et al.* 1997 in MMS 2008).

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 produces 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 generated by the proposed action have not been reported to disrupt marine organism behavior, orientation, or migration. Based on the body of scientific literature presented by MMS in the DEIS and BA, there are no anticipated adverse impacts expected from the undersea power transmission cables or other components of the proposed action on the behavior, orientation, or navigation of marine organisms, including listed sea turtle species. Based on this analysis, potential direct impacts to listed sea turtles during the normal operation of the inner-array cables and the two submarine cable circuits will be discountable.

The burial depth of the cables (i.e., 6 feet below the seabed) also minimizes potential thermal impacts from operation of these cable systems. In addition, the inner-array and submarine cable systems 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 sea turtle species during the normal operation of the inner-array or submarine cable systems. There will also be no impacts to prey species of sea turtles during the normal operation of the inner-array or submarine cable systems.

Increased Risk of Vessel Strike

The construction and operation of the project will require the use of a variety of vessels. Vessels will be used to transport materials from the staging areas in Falmouth, MA and Quonset Point, RI to the project site and will also be used to deliver crew to the project site. Additionally, specialized vessels will be used during construction. These vessels will include barges and tugs used for cable installation and pile driving. An additional specialized vessel will be used to stage the assembly of the WTGs. Once construction is complete, maintenance vessels will visit the project site from New Bedford, MA. These vessels will represent an increase in vessel traffic in the action area.

During pile driving activities, it is estimated that 4 to 6 stationary or slow moving vessels would be present in the general vicinity of the pile installation (i.e., on Horseshoe Shoal). Vessels delivering construction materials or crews to the site will also be present in the area between the mainland and the proposed action site (a trip lasting approximately one hour). The barges, tugs and vessels delivering construction material generally will travel at speeds below 10 knots and may range in size from 90 to 400 feet, while the vessels carrying construction crews will be traveling at a maximum speed of 21 knots and will typically be 50 feet in length. While on site, vessels will be slow moving or stationary. Once construction is complete, maintenance vessels will continue to visit the site, with the highest number of maintenance vessels on site in the summer months when the weather is most favorable. As noted in the Description of the Action section above, Cape Wind will maintain two vessels for maintenance activities.

As discussed in the Environmental Baseline, collision with vessels remains a source of anthropogenic mortality for both sea turtles and whales. The proposed project will lead to increased vessel traffic during construction and long-term operation that would not exist but for the proposed action. This increase in vessel traffic will result in some increased risk of vessel strike of listed species. However, due to the limited information available regarding the incidence of ship strike and the factors contributing to ship strike events, it is difficult to determine how a particular number of vessel transits or a percentage increase in vessel traffic will translate into a number of likely ship strike events or percentage increase in collision risk. In spite of being one of the primary known sources of direct anthropogenic mortality to whales, and to a lesser degree, sea turtles, ship strikes remain relatively rare, stochastic events, and an increase in vessel traffic in the action area would not necessarily translate into an increase in ship strike events. No vessel strike events have been reported in the action area. Nonetheless, MMS and Cape Wind have proposed to implement the following mitigation measures to further reduce the likelihood of a project vessel interacting with a whale or sea turtle (see Appendix A for a complete listing of all mitigation measures):

- All vessels associated with the construction, operation/maintenance and/or decommissioning of the project will be required to abide by the (1) NMFS Northeast Regional Viewing Guidelines, as updated through the life of the project; and (2) MMS Gulf of Mexico Region's Notice to Lessee (NTL) No. 2007-G04.

- All vessel operators must undergo training to ensure they are familiar with the above requirements. These training requirements must be written into any contractor agreements.

Large whales, particularly right whales, are vulnerable to injury and mortality from ship strikes. Due to the overlap of heavy shipping traffic and high whale density, Massachusetts waters are a high risk area for ship strike events. All project vessels will be transiting between the project site and either Quonset, Rhode Island or Falmouth, Massachusetts. As explained throughout this document, whales are not expected to occur in the project footprint or along the cable route and only rarely would whales enter Nantucket Sound. As no whales are expected to occur along the routes where project vessels will transit or in the project footprint where construction and maintenance vessels will occur, the increase in vessel traffic attributable to the proposed project will not increase the likelihood of a whale being struck by a vessel. As no whales are likely to occur where project vessels will be operating, NMFS has determined that the likelihood of an interaction between a project vessel and a whale is discountable. However, as sea turtles are likely to occur in the area where increased vessel traffic will occur, this section will focus on the effects of an increase in vessel traffic on sea turtles.

Interactions between vessels and sea turtles occur and can take many forms, from the most severe (death or bisection of an animal or penetration to the viscera), to severed limbs or cracks to the carapace which can also lead to mortality directly or indirectly. Sea turtle stranding data for the U.S. Gulf of Mexico and Atlantic coasts, Puerto Rico, and the U.S. Virgin Islands show that between 1986 and 1993, about 9% of living and dead stranded sea turtles had propeller or other boat strike injuries (Lutcavage et al. 1997). According to 2001 STSSN stranding data, at least 33 sea turtles (loggerhead, green, Kemp's ridley and leatherbacks) that stranded on beaches within the northeast (Maine through North Carolina) were struck by a boat. From 2001-2006, an additional 14 sea turtles (12 leatherbacks, 1 Kemp's ridley, 1 loggerhead) have been documented with injuries consistent with propeller wounds (NMFS unpublished data) in the northeast. This number underestimates the actual number of boat strikes that occur since not every boat struck turtle will strand, every stranded turtle will not be found, and many stranded turtles are too decomposed to determine whether the turtle was struck by a boat. It should be noted, however, that it is not known whether all boat strikes were the cause of death or whether they occurred post-mortem (NMFS SEFSC 2001).

Information is lacking on the type or speed of vessels involved in sea turtle vessel strikes. However, there does appear to be a correlation between the number of vessel struck turtles and the level of recreational boat traffic (NRC 1990). Sea turtles have been reported with injuries consistent with propeller wounds, which are likely from interactions with small, fast moving vessels, such as recreational boats.

Although little is known about a sea turtle's reaction to vessel traffic, sea turtles are thought to be able to avoid injury from slower-moving vessels since the turtle has more time to maneuver and avoid the vessel. Vessels will only travel between 0-4 knots while actually engaged in construction activities, or 1-2 miles in a 24-hour period. At these speeds, vessel movements during construction are not likely to pose a vessel strike risk to sea turtles.

The risk of collision is greatest when vessels are moving at higher speeds when transiting between the staging areas and the project site. As such, the 10 knot speed of the construction vessels is likely to reduce the chance for collision. Crew support vessels may run at higher speeds, with a maximum speed of 21 knots. Lookouts will be posted on all vessel transits. All vessels would follow the vessel strike avoidance procedures discussed above. The presence of an experienced endangered species observer at the construction site who can advise the vessel operator to slow the vessel or maneuver safely when sea turtles are spotted will further reduce the potential for interaction with vessels.

Although the threat of vessel collision exists anywhere listed species and vessel activity overlap, ship strike is more likely to occur in areas where high vessel traffic coincides with high species density. In addition, ship strikes are more likely to occur and more likely to result in serious injury or mortality when vessels are traveling at speeds greater than ten knots. Although most construction vessel transits will occur at speeds of ten knots or less, some vessels may travel at speeds up to 21 knots. All vessel operators and lookouts will receive training on protected species identification and prudent vessel operating procedures in the presence of marine mammals and sea turtles. With these vessel strike avoidance measures in place, and due to the fact that the increase in vessel traffic will be insignificant compared to the number of vessels operating in the action area on a normal basis, NMFS has determined that the increased risk of vessel collision posed by project vessel operation in the action area is insignificant.

Acoustic Effects

When anthropogenic disturbances elicit responses from sea turtles and marine mammals, it is not always clear whether they are responding to visual stimuli, the physical presence of humans or manmade structures, or acoustic stimuli. However, because sound travels well underwater, it is reasonable to assume that, in many conditions, marine organisms would be able to detect sounds from anthropogenic activities before receiving visual stimuli. As such, exploring the acoustic effects of the proposed project provides a reasonable and conservative estimate of the magnitude of disturbance caused by the general presence of a manmade, industrial structure in the marine environment, as well as the specific effects of sound on marine mammal and sea turtle behavior.

Marine organisms rely on sound to communicate with conspecifics and derive information about their environment. There is growing concern about the effect of increasing ocean noise levels due to anthropogenic sources on marine organisms, particularly marine mammals. Effects of noise exposure on marine organisms can be characterized by the following range of physical and behavioral responses (Richardson et al. 1995):

1. Behavioral reactions – Range from brief startle responses, to changes or interruptions in feeding, diving, or respiratory patterns, to cessation of vocalizations, to temporary or permanent displacement from habitat.
2. Masking – Reduction in ability to detect communication or other relevant sound signals due to elevated levels of background noise.
3. Temporary threshold shift (TTS) – Temporary, fully recoverable reduction in hearing sensitivity caused by exposure to sound.

4. Permanent threshold shift (PTS) – Permanent, irreversible reduction in hearing sensitivity due to damage or injury to ear structures caused by prolonged exposure to sound or temporary exposure to very intense sound.
5. Non-auditory physiological effects – Effects of sound exposure on tissues in non-auditory systems either through direct exposure or as a consequence of changes in behavior, e.g., resonance of respiratory cavities or growth of gas bubbles in body fluids.

Several components of project construction and operation will produce sound that may affect listed sea turtles and whales. NMFS is in the process of developing a comprehensive acoustic policy that will provide guidance on managing sources of anthropogenic sound based on each species' sensitivity to different frequency ranges and intensities of sound. The available information on the hearing capabilities of cetaceans and the mechanisms they use for receiving and interpreting sounds remains limited due to the difficulties associated with conducting field studies on these animals. However, current thresholds for determining impacts to marine mammals typically center around root-mean-square (RMS) received levels of 180 dB re 1μPa for potential injury, 160 dB re 1μPa for behavioral disturbance/harassment from a non-continuous noise source, and 120 dB re 1μPa for behavioral disturbance/harassment from a continuous noise source. These thresholds are based on a limited number of experimental studies on captive odontocetes, a limited number of controlled field studies on wild marine mammals, observations of marine mammal behavior in the wild, and inferences from studies of hearing in terrestrial mammals. In addition, marine mammal responses to sound can be highly variable, depending on the individual hearing sensitivity of the animal, the behavioral or motivational state at the time of exposure, past exposure to the noise which may have caused habituation or sensitization, demographic factors, habitat characteristics, environmental factors that affect sound transmission, and non-acoustic characteristics of the sound source, such as whether it is stationary or moving (NRC 2003). Nonetheless, the threshold levels referred to above are considered conservative based on the best available scientific information at this time and will be used in the analysis of effects for this Opinion.

The acoustic effects analysis will:

- characterize the various sources of noise attributed to the proposed action
- determine which species are likely to be exposed to each type of noise
- characterize the range of expected or possible responses of sea turtles and marine mammals exposed to the noise; and,
- determine the significance of those effects to individuals and populations.

Characterization of Construction Noise Sources

Pile driving with an impact hammer produces impulsive sounds. All other noise sources associated with construction will be non-impulse sounds continuous for the duration of the activity. Sources of construction noise associated with the proposed project include the following:

- Cable laying and associated activities;
- Pile driving;
- Construction and maintenance vessel transits; and,

- Operation of the WTGs.

Right, Humpback, and Fin Whale Hearing

In order for right, humpback, and fin whales to be adversely affected by construction noise, they must be able to perceive the noises produced by the activities. If a species cannot hear a sound, or hears it poorly, then the sound is unlikely to have a significant effect (Ketten 1998). Baleen whale hearing has not been studied directly, and there are no specific data on sensitivity, frequency or intensity discrimination, or localization (Richardson et al. 1995) for these whales. Thus, predictions about probable impact on baleen whales are based on assumptions about their hearing rather than actual studies of their hearing (Richardson et al. 1995; Ketten 1998).

Ketten (1998) summarized that the vocalizations of most animals are tightly linked to their peak hearing sensitivity. Hence, it is generally assumed that baleen whales hear in the same range as their typical vocalizations, even though there are no direct data from hearing tests on any baleen whale. Most baleen whale sounds are concentrated at frequencies less than 1 kHz (Richardson et al. 1995), although humpback whales can produce songs up to 8 kHz (Payne and Payne 1985). Based on indirect evidence, at least some baleen whales are quite sensitive to frequencies below 1 kHz but can hear sounds up to a considerably higher but unknown frequency. Most of the manmade sounds that elicited reactions by baleen whales were at frequencies below 1 kHz (Richardson et al. 1995). Some or all baleen whales may hear infrasounds, sounds at frequencies well below those detectable by humans. Functional models indicate that the functional hearing of baleen whales extends to 20 Hz, with an upper range of 30 Hz. Even if the range of sensitive hearing does not extend below 20-50 Hz, whales may hear strong infrasounds at considerably lower frequencies. Based on work with other marine mammals, if hearing sensitivity is good at 50 Hz, strong infrasounds at 5 Hz might be detected (Richardson et al. 1995). Fin whales are predicted to hear at frequencies as low as 10-15 Hz. The right whale uses tonal signals in the frequency range from roughly 20 to 1000 Hz, with broadband source levels ranging from 137 to 162 dB (RMS) re 1 μPa at 1 m (Parks & Tyack 2005). One of the more common sounds made by right whales is the "up call," a frequency-modulated upsweep in the 50–200 Hz range (Mellinger 2004). The following table summarizes the range of sounds produced by right, humpback, and fin whales (from Au et al. 2000):

Table 1. Summary of known right, humpback, and fin whale vocalizations

Species	Signal type	Frequency Limits (Hz)	Dominant Frequencies (Hz)	Source Level (dB re 1μPa RMS)	References
North Atlantic Right	Moans	< 400	--	--	Watkins and Schevill (1972)
	Tonal	20-1000	100-2500	137-162	Parks and Tyack (2005)
	Gunshots		50-2000	174-192	Parks et al. (2005)
Humpback	Grunts	25-1900	25-1900	--	Thompson, Cummings, and Ha (1986)
	Pulses	25-89	25-80	176	Thompson, Cummings, and Ha (1986)
	Songs	30-8000	120-4000	144-174	Payne and Payne (1985)

Fin	FM moans Tonal Songs	14-118 34-150 17-25	20 34-150 17-25	160-186 186	Watkins (1981), Edds (1988), Cummings and Thompson (1994) Edds (1988) Watkins (1981)

Most species also have the ability to hear beyond their region of best sensitivity. This broader range of hearing probably is related to their need to detect other important environmental phenomena, such as the locations of predators or prey. Considerable variation exists among marine mammals in hearing sensitivity and absolute hearing range (Richardson et al. 1995; Ketten 1998); however, from what is known of right, humpback, and fin whale hearing and the source levels and dominant frequencies of the construction noise sources summarized in Table 3, it is evident that right, humpback, and fin whales are capable of perceiving construction noises, and have hearing ranges that are likely to have peak sensitivities in low frequency ranges that overlap the dominant frequencies of pile driving and vessel noise.

Sea Turtle Hearing
The hearing capabilities of sea turtles are poorly known. Few experimental data exist, and since sea turtles do not vocalize, inferences cannot be made from their vocalizations as is the case with baleen whales. Direct hearing measurements have been made in only a few species. An early experiment measured cochlear potential in three Pacific green turtles and suggested a best hearing sensitivity in air of 300–500 Hz and an effective hearing range of 60–1,000 Hz (Ridgway et al. 1969). Sea turtle underwater hearing is believed to be about 10 dB less sensitive than their in-air hearing (Lenhardt 1994). Lenhardt et al. (1996) used a behavioral "acoustic startle response" to measure the underwater hearing sensitivity of a juvenile Kemp's ridley and a juvenile loggerhead turtle to a 430-Hz tone. Their results suggest that those species have a hearing sensitivity at a frequency similar to those of the green turtles studied by Ridgway et al. (1969). Lenhardt (1994) was also able to induce startle responses in loggerhead turtles to low frequency (20–80 Hz) sounds projected into their tank. He suggested that sea turtles have a range of best hearing from 100–800 Hz, an upper limit of about 2,000 Hz, and serviceable hearing abilities below 80 Hz. More recently, the hearing abilities of loggerhead sea turtles were measured using auditory evoked potentials in 35 juvenile animals caught in tributaries of Chesapeake Bay (Bartol et al. 1999). Those experiments suggest that the effective hearing range of the loggerhead sea turtle is 250–750 Hz and that its most sensitive hearing is at 250 Hz. In general, however, these experiments indicate that sea turtles generally hear best at low frequencies and that the upper frequency limit of their hearing is likely about 1 kHz. As such, sea turtles are capable of hearing in low frequency ranges that overlap with the dominant frequencies of pile driving and vessel noise, and are therefore likely to be exposed to construction-related noise.

Effects of Exposure to Construction Noise – Pile Driving
Sound levels associated with the driving of monopiles have been modeled and results are presented in the BA. Modeling indicates that the source level of the noise (dB re 1uPa at 1 meter) will be 232 dB with a spectral energy of 1Hz to 20 kHz. Underwater noise from the installation of the monopiles has been modeled to be 178 dB re 1uPa at 500m, 172 dB re 1uPa at 1km and 166 dB re 1uPa at 2km. In order to minimize the effects of pile driving on listed

species, MMS will require and Cape Wind has agreed to implement several mitigation measures. These measures are detailed in Appendix A. The most significant of these measures requires that no pile driving occur if any whales or sea turtles are present within 750 meters of the pile to be driven. Outside the 750 m exclusion zone, noise levels are anticipated to be below 178dB re 1 uPa.

Exposure to Injurious Levels of Sound

As explained above, whales are not thought to normally occur in Nantucket Sound. However, right, humpback and fin whales have been documented off of the Northern tip of Nantucket Island and off of Monomoy (16-19 km from the project site). As no whales will occur within 500 meters of any pile driving, no whales will be exposed to sound levels greater than 178 dB and no whales will be exposed to sound levels at which injury could occur (i.e., 180dB re 1uPa).

As sea turtles could occur in the project area while pile driving is occurring, there is the potential for a sea turtle to be exposed to sound levels greater than 180 dB. Sound levels will have dissipated to below the 180 dB threshold within a distance of 500m. As no pile driving will occur if a sea turtle is within 750m of the pile, no sea turtles are likely to be exposed to potentially injurious levels of sound. Thus, sea turtles are not likely to be exposed to levels of construction-related noise that will result in injury.

Exposure to disturbing levels of sound

Although the potential for construction-related sounds to cause injury to whales and sea turtles is extremely low, there is greater potential for sea turtles to be exposed to disturbing levels of sound produced by these activities. For pile driving, potentially disturbing levels of sound (160-180dB) is expected to propagate over a distance of no more than 3.4km from the source.

Modeling presented by MMS in Appendix 5-11A (Noise Report) of the DEIS indicates that underwater noise levels may be greater than 160 dB re 1 uPa (i.e., NMFS threshold for behavioral disturbance/harassment from a non-continuous noise source) within approximately 3.4km of the pile being driven. As the nearest whale sighting was approximately 18km from the project site, it is extremely unlikely that any whales will be exposed to noise levels greater than 160 dB. Although construction noise may be audible for several kilometers from the source, right, humpback and fin whales are primarily found outside of Nantucket Sound, well beyond the distances over which the 160-180 dB contours are likely to extend. Based on the best available information and the analysis outlined herein, no right, humpback or fin whales will be exposed to noise levels greater than 160 dB. As such, no whales will be exposed to noise levels that could result in behavioral disturbance or harassment.

Since leatherback, green, Kemp's ridley and loggerhead sea turtles are known to occur in Nantucket Sound between June and October and construction will occur during this time period, these species are likely to be exposed to construction-related noise during the construction period.

There is very little information about sea turtle behavioral reactions to levels of sound below the thresholds suspected to cause injury or TTS. However, some studies have demonstrated that sea turtles have fairly limited capacity to detect sound, although all results are based on a limited number of individuals and must be interpreted cautiously. Ridgway et al. (1969) found that one green turtle with a region of best sensitivity around 400 Hz had a hearing threshold of about 126 dB in water. Streeter (in press) found similar results in a captive green sea turtle, which demonstrated a hearing threshold of approximately 125 dB at 400 Hz, but better sensitivity at 200 Hz (110-115 dB threshold). McCauley (2000) noted that dB levels of 166 dB re 1μPa were required before any behavioral reaction was observed.

As noted above, modeling results reported by MMS indicate that sound levels could be higher than 160 dB within 3.4 km of the pile being driven. As such, any sea turtles occurring within that area would be exposed to potentially disturbing sound levels. The available information on sea turtle behavioral responses to these sound levels indicates that individuals are likely to actively avoid areas with disturbing levels of sound. Avoidance behavior may shorten the exposure period; however, the avoidance behavior could potentially disrupt normal behaviors. Reactions of individual sea turtles to the pile driving is expected to be limited to an avoidance response. Only pile driving occurring during the June – November time frame has the potential to affect sea turtles, as sea turtles are not expected to occur in the action area outside of this time of year.

As explained throughout, there is limited information available specific to sea turtle presence in Nantucket Sound. There have been no systematic surveys to document the number of sea turtles in the action area or Nantucket Sound generally. Leatherback, loggerhead, green and Kemp's ridley sea turtles have all been documented in Nantucket Sound generally and/or the action area specifically (Lazell et al. 1980, Mass Audubon 2002, 2003 and 2004, as well as information at the OBIS and seaturtle.org databases). Sightings data indicate that leatherback sea turtles are the most common species of sea turtle in Massachusetts waters, including Nantucket Sound, followed by loggerheads, with fewer Kemp's ridley and green sea turtles. However, as all four sea turtle species have been documented to occur in Nantucket Sound and sea turtles are highly mobile, NMFS considers that any of these species could be present in the action area. NMFS considered several sources of information in order to estimate the number of sea turtles that could be exposed to sound levels between 160 and 180 dB. As noted above, the area where noise levels will be greater than 160dB extends approximately 3.4km from the pile being driven. This area includes the 750 meter exclusion zone. As no pile driving will take place when sea turtles are present within the exclusion zone, only sea turtles located in the area between 750 meters and 3.4 km from the pile being driven will be exposed to sound levels greater than 160dB. The size of this area is approximately of 160-180 dB will be experienced is limited to a roughly circular area extending from 750 m to 3.4km from the pile being driven. This results in an area of approximately 34.56km^2.

Few researchers have reported on the density of sea turtles in Northeastern waters. However, this information is available from one source (Shoop and Kenney 1992). Shoop and Kenney (1992) used information from the University of Rhode Island's Cetacean and Turtle Assessment

Program (CETAP[5]) as well as other available sightings information to estimate seasonal abundances of loggerhead and leatherback sea turtles in northeastern waters. The authors calculated overall ranges of abundance estimates for the summer of 7,000-10,000 loggerheads and 300-600 leatherbacks present in the study area from Nova Scotia to Cape Hatteras. Using the available sightings data (2841 loggerheads, 128 leatherbacks and 491 unidentified sea turtles), the authors calculated density estimates for loggerhead and leatherback sea turtles (reported as number of turtles per square kilometer). These calculations resulted in density estimates of 0.00164 – 0.510 loggerheads per square kilometer and 0.00209 – 0.0216 leatherbacks per square kilometer. It is important to note, however, that this estimate assumes that sea turtles are evenly distributed throughout the waters off the northeast, even though Shoop and Kenney report several concentration areas where loggerhead or leatherback abundance is much higher than in other areas. Further, the data do not include any sightings from Massachusetts generally, or Nantucket Sound specifically and only considered the presence of leatherback and loggerhead sea turtles. The Shoop and Kenney data, despite considering only the presence of loggerhead and leatherback sea turtles, likely overestimates the number of sea turtles present in the impact zone. This is due to the assumption that sea turtle abundance will be even throughout the Nova Scotia to Cape Hatteras study area, which is an invalid assumption. Sea turtles occur in high concentrations in several areas outside of the action area and the inclusion of these concentration areas in the density estimate skews the estimate for the action area.

As noted above (see pages 70-71), Mass Audubon conducted surveys for terns over an approximately four week period in 2002, 2003 and 2004. Both shipboard and aerial surveys were conducted. In their reports, Mass Audubon includes information on sea turtle sightings. As this information was collected in the action area, it represents important information on the presence of sea turtles in this area. There are limitations to the Mass Audubon data. As noted above, the aerial surveys were not designed to observe sea turtles. However, as the flights were flown at an elevation that is within the range known to be effective for observing sea turtles (i.e., 500 feet; Henwood and Epperly 1999) and flights were only taken on days when visibility was extremely good (i.e., greater than 10 miles), it is likely that the observations represent a reasonable estimate of the number of sea turtles at the surface during the survey. Further, when compared to a calculation made by Witzell and Azarovitz (1996) using aerial survey data where sea turtles were specifically targeted, the number of sea turtles observed per 100 km flown is nearly identical for the month of August (1.15 sea turtles observed per 100km flown in the Mass Audubon surveys and 1.1 sea turtles observed per 100km flown in the surveys reported in Witzell and Azarovitz).

It is likely that the Mass Audubon data underestimates the number of sea turtles present during the surveys. This is due to the fact that observations of sea turtles were incidental to the surveys for terns and other birds. Additionally, as sea turtles spend a considerable amount of time underwater, there were likely additional submerged sea turtles in the survey area that went uncounted. Sea turtles spend a significant amount of time underwater. Specifically, it has been

5 The CETAP survey consisted of three years of aerial and shipboard surveys conducted between 1978 and 1982 and provided the first comprehensive assessment of the sea turtle population between Nova Scotia, Canada and Cape Hatteras, North Carolina.

estimated that individual loggerhead sea turtles spend 80-94% of their time submerged, Kemp's ridleys spend approximately 96% of the time submerged and leatherbacks 74-91% of the time submerged (Lutcavage and Lutz 1997). One study of green sea turtles indicated that individual turtles spent between 81-98% of the time submerged, with an average of 91% (Renaud et al. 1995). It has been estimated that, on average, sea turtles spend only between 3-6% of the time at the surface, and cumulatively spend only approximately one hour a day at the surface (Spotila 2004; Lutcavage and Lutz 1997).

The 115 sea turtle observations occurred over 25 survey days where approximately 888 square kilometers were surveyed. It is important to note that these surveys coincided with the time of year when the highest numbers of sea turtles are expected to occur in the action area (i.e., July – September). Approximately 5 sea turtles were observed during each survey day. This translates into approximately 0.006 sea turtles observed per square kilometer surveyed. Based on the known amount of time that sea turtles spend submerged each day, it is likely that only 3-6% of the sea turtles present in the study area would have been at the surface at the time of the survey. In this case, the actual number of sea turtles present (i.e., submerged and at the surface) in the survey area during the aerial survey was more likely in the range of 83-166 sea turtles (i.e., 5 is 3% of 166 and 6% of 83). Using these estimates, the density of sea turtles per square kilometer can be calculated. The values calculated are 0.09 (which is equivalent to 83 sea turtles/888 square kilometers) and 0.19 (166 sea turtles/888 square kilometers).

Using these calculated densities, an estimate of the number of sea turtles likely to be exposed to noise levels between 160 and 180 dB can be calculated (i.e., number of sea turtles per square kilometer multiplied by 34.56 (the size of the area where noise levels will be between 160 and 180 dB)). This calculation results in an estimate of between 3 and 7 sea turtles likely to be present in any given 34.56 square kilometer area within Nantucket Sound.

Based on the available information it is likely that the number of sea turtles that would be exposed to noise levels between 160 and 180 dB ranges between 3 and 7. These numbers use the Mass Audubon data adjusted for the likely percentage of sea turtles that would have been submerged, and therefore not visible to observers, during the aerial surveys. The number of sea turtles exposed to these sound levels will be influenced by the depth of water at the particular site as well as the amount and type of forage present within the impact zone and the time of year when the pile driving is occurring (i.e., more sea turtles are likely to be present at sites with depths of 16-49 feet, with concentrations of preferred forage items, or during the months of August and September). As noted above, only pile driving occurring between June and November would result in the exposure of sea turtles to disturbing levels of noise.

Sea turtles behaviorally disrupted would be expected to resume their behavior after the pile driving has stopped. As pile driving will occur for approximately 4 hours a day, it is likely that sea turtles will be excluded from the area with disturbing levels of sound for at least this period each day. Available information indicates that sea turtle forage items are available throughout the action area; therefore, while sea turtles may move to other areas within the action area to forage during the times when pile driving is occurring, the ability of individual sea turtles to find suitable forage is not expected to be impacted. Likewise, if sea turtles were resting in a particular area they are expected to be able to find an alternate resting area within the action area. Additionally, if sea turtles are migrating through the action area, they may avoid the area with disturbing levels of sound and choose an alternate route through the action area. However, as at all times there will be areas of Nantucket Sound where noise levels are not at disturbing levels, the ability of sea turtles to migrate through the action area will not be affected. As such, while the movements of individual sea turtles while be affected by the sound associated with the pile driving, these effects will be temporary and localized and sea turtles are not expected to be excluded from Nantucket Sound and there will be only a minimal impact on foraging, migrating or resting sea turtles that will not result in injury or impairment in an individual's ability to complete essential behavioral functions. Major shifts in habitat use or distribution or foraging success are not expected. As changes to individuals movements are expected to be minor and short-term, and are therefore not likely to have population-level effects.

Effects of Noise Associated with Construction Vessel Traffic
Support and vessel transits will occur regularly throughout the construction period. These vessels will be shuttling personnel and supplies between Quonset, RI and Yarmouth, MA and the construction site, and will represent an additional transient source of noise along the transit path. During the construction period several vessels will transit to the work site each day, carrying supplies and equipment. Vessels transmit noise through water and cumulatively are a significant contributor to increases in ambient noise levels in many areas. The dominant source of vessel noise from the proposed action is propeller cavitation, although other ancillary noises may be produced. The intensity of noise from service vessels is roughly related to ship size and speed. Large ships tend to be noisier than small ones, and ships underway with a full load (or towing or pushing a load) produce more noise than unladen vessels. Vessel traffic associated with the proposed action would produce levels of noise of 150 to 170 dB re 1 μPa-m at frequencies below 1,000 Hz. A tug pulling a barge generates 164 dB re 1 μPa-m when empty and 170 dB re 1 μPa-m loaded. A tug and barge underway at 18 km/h can generate broadband source levels of 171 dB re 1 μPa-m. A small crew boat produces 156 dB re 1 μPa-m at 90 Hz.

As noted previously in relation to construction noise, sea turtles are thought to be far less sensitive to sound than marine mammals. Although vessel noises are within the limited range of frequencies they can detect, evidence suggests that sound levels of 110-126 dB re 1μPa are required before sea turtles can detect a sound (Ridgway 1969; Streeter, in press). McCauley (2000) noted that dB levels of 166 dB re 1μPa were required before any behavioral reaction was observed. As all operational noise sources are expected to diminish to below this threshold within very short distances, no sea turtles are expected to be exposed to injurious or harassing levels of sound. As no avoidance behaviors are anticipated, the distribution, abundance and behavior of sea turtles in the action area is not likely to be affected by noise associated with construction or maintenance vessels and any effects will be insignificant or discountable.

Effects of Exposure to Operational Noise Sources
In addition to construction-related noise, there is some noise associated with the long-term operation of the proposed WTG facility. Operational noise can be attributed to the following:

- Wind turbine operation
- maintenance and support vessel transits

Wind Turbine Operation
Once installed, the operation of the WTGs is not expected to generate substantial sound levels above baseline sound in the area. Preliminary results from noise studies conducted in the United Kingdom suggest that in general, the level of noise created during the operation of offshore windfarms is very low and does not cause avoidance of the area by marine species (Nedwell, unpub. data, reported in MMS 2008). Even in the area directly surrounding the wind turbines, noise was not generally found above the level of background noise, resulting in normal activity of marine animals (Nedwell, unpub. data, reported in MMS 2008).

Acoustic modeling of underwater operational sound at the proposed Cape Wind facility was performed for the design wind condition and reported in the BA and DEIS. Baseline underwater sound levels under the design wind condition are 107.2 dB. The predicted 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 sound level falls off to 107.5 dB at 164 ft (50 m) and declines to the baseline level by 361 ft (110 m)). 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.

As no whales are expected to occur within 360 feet of any of the WTGs, no whales will be exposed to operational noise associated with the project. As sea turtles are distributed throughout the project area, sea turtles are likely to be exposed to operational sound of the WTGs. However, as the sound (109.1 dB at 65.6 feet) will be less than 2dB above the baseline underwater noise levels (107.5 dB) and well below harassing noise levels (i.e., 120 dB re 1 uPa for a continuous noise source), the operational noise of the WTGs will not result in injury or disturbance of sea turtles. While sea turtles may be able to hear the noise associated with the operation of the WTGs the noise will not affect the distribution, abundance or behavior of sea turtles in the action area.

Geophysical and Geotechnical Surveys
The applicant may conduct a high resolution geophysical survey prior to construction. Only the project footprint on Horseshoe Shoal would be surveyed. The survey would investigate the shallow subsurface for geohazards and sediment conditions, as well as to identify potential benthic biological communities (or habitats) and archaeological resources. A typical high resolution seismic survey operation consists of a vessel towing an acoustic source (airgun, boomer, sparker, chirper) about 25 m behind the ship and a 600-m streamer cable with a tail buoy. In general, the ships travel at 3-3.5 knots (5.6-6.5 km/hour), and the source is activated every 7-8 seconds (or about every 12.5 m). All involved ships are designed to reduce self-noise, as the higher frequencies used in high-resolution work are easily masked by the vessel noise if special attention is not paid to keeping the ships quiet. If undertaken, this would involve one 36-hour sampling event. While the towed gear (i.e., the airgun, boomer, sparker or chirper) has the potential to result in interaction with sea turtles, the speed of towing (typically about 3 knots)

minimizes the potential for entanglement or vessel strikes during the survey as sea turtles would be able to avoid the slow moving gear and survey vessel.

The sound levels at the source (i.e., the survey vessel) will depend on the type of equipment used for the survey (i.e., airgun, boomer, sparker or chirper). If an airgun is used, noise levels at the source would range from 229-233 dB re 1uPa at 1 meter. A boomer has a sound pressure level of 205 re 1uPa at 1 meter, with an output-sound bandwith of 0.5-8 kHz, with the main peak at 4.5kHz. A sparker has an SPL of 209 dB re 1uPa at 1 meter at 150-1700 Hz, with a peak amplitude of 900 Hz. A chirper would have an output near 160 dB at the source. MMS has reported that if an airgun is used, at a distance of approximately 500 meters, the noise would be less than 180dB and at 1.5km, the noise would be less than 160 dB. For the other sources, the impact zones would be smaller. Given the likely maximum ranges of the 180 dB and 160 dB isopleths, it is highly unlikely that any whales would be exposed to injurious or disturbing sound levels associated with the survey. However, if the survey occurred between June and November, listed sea turtles could be exposed to effects of the survey. MMS is requiring that the applicant maintain a 500 meter exclusion zone during the survey. As such, no listed sea turtles will be exposed to noise levels greater than 180 dB and therefore, no sea turtles will be exposed to injurious levels of noise. However, sea turtles are likely to be exposed to disturbing levels of noise. Any sea turtles located within one km outside of the exclusion zone (i.e., from 0.5 – 1.5 km from the survey vessel) will be exposed to potentially disturbing levels of noise.

During the survey, an area of approximately 148 square kilometers will be surveyed. Based on the estimates of sea turtle density in the action area (see above), NMFS estimates that between 13 and 28 sea turtles would be exposed to disturbing levels of noise during the survey. At any given time during the survey, an approximately 3.14 square kilometer area will have noise levels between 160 and 180 dB.

Sea turtles whose behavior is disrupted would be expected to resume their behavior after the disturbance has stopped. While the total survey will take approximately 36 hours to complete, the time that any particular area will experience elevated sound levels will be significantly shorter. Available information indicates that sea turtle forage items are available throughout the action area; therefore, while sea turtles may move to other areas within the action area to forage during the times when the survey is occurring, the ability of individual sea turtles to find suitable forage is not expected to be impacted. Likewise, if sea turtles were resting in a particular area they are expected to be able to find an alternate resting area within the action area. Additionally, if sea turtles are migrating through the action area, they may avoid the area with disturbing levels of sound and choose an alternate route through the action area. However, as at all times there will be areas of Nantucket Sound where noise levels are not at disturbing levels, the ability of sea turtles to migrate through the action area will not be affected. As such, while the movements of individual sea turtles while be affected by the sound associated with the survey, these effects will be temporary and localized and sea turtles are not expected to be excluded from Nantucket Sound and there will be only a minimal impact on foraging, migrating or resting sea turtles that will not result in injury or impairment in an individual's ability to complete essential behavioral functions. Major shifts in habitat use or distribution or foraging success are not expected. As changes to individuals movements are expected to be minor and short-term, and are therefore not likely to have population-level effects.

The geotechnical surveys will result in small areas of the seafloor being disturbed, 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 geotechnical investigations would result in a negligible temporary loss of some benthic organisms (i.e., less than one foot diameter will be disturbed in the areas where cores are sampled), and a localized increase in disturbance due to vessel activity, including noise and anchor cable placement and retrieval. Effects of the disturbance of the seafloor and the effect on foraging sea turtles and whales are discussed in the "Destruction of Prey Resources/Loss of Foraging Habitat" section above. Additionally, the effect of the survey vessels on increasing the risk of vessel strikes is also discussed in the "Vessel Strike" section above. As noted in those sections, effects to listed species from these sources would be insignificant or discountable.

Decommissioning

At the conclusion of the life of the Cape Wind project, components would be retrieved and removed from the site. All components in the water column would be retrieved, including the ESP, WTGs, and submarine cables. At the end of the proposed action'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 to the proposed action. 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 WTG facility, 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.

These removal activities are expected to have impacts similar to those discussed above in relation to construction activities, including temporary seafloor disturbance, turbidity, and water withdrawal and discharge associated with flushing of the pipeline. However, all impacts would be of less magnitude than those resulting from construction activities. As such, effects of decommissioning activities will be insignificant or discountable.

Non-routine and Accidental Events

Cable Repair

Many of the types of disturbances that would occur during cable repair activities are smaller and of 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 cable 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. As explained in sections related to the effects of cable installation above, as no whales are expected to occur along the cable route, there would be no effects to whales from a cable repair. Depending on the time of year that the cable repair occurred, sea turtles may be present. However, as explained in the cable installation sections above, all effects of the cable laying process, and similarly, the cable repairing process, would be insignificant or discountable.

Vessel Collision with Monopile
The extent of potential impacts that could result from a vessel collision with a monopile largely depends on the extent of damage to the monopile or vessel. Some smaller vessels would merely strike a glancing blow and possibly 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 may cause a collapse of the monopile, also resulting in a small release of lubricating fluid. 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. The effects of a vessel collision on listed species are difficult to predict. However, as no whales are expected to occur in the action area, any effects of a vessel collision with a monopile with whales are discountable. Effects to sea turtles from a vessel collision with a monopile are more likely to be attributable to the debris that enters the water and effects of any repair activities. As any effects are likely to be on a small scale and temporary, any effects, if adverse, will be insignificant.

Oil Spill
Oil spills could occur either as a release from the ESP storage tank or from a vessel collision with a monopile. An oil spill would be an unintended, unpredictable event. Marine animals, including whales and sea turtles, are known to be negatively impacted by exposure to oil and other petroleum products. Without an estimate of the amount of oil released it is difficult to predict the likely effects on listed species. The applicant is required to develop an oil spill response plan which would ensure rapid response to any spill. As the effects of a spill are likely to be localized and temporary, sea turtles and whales are not likely to be exposed to oil and any effects would be discountable. Additionally, should a response be required by the US EPA or the USCG, there would be an opportunity for NMFS to conduct a consultation with the lead Federal agency on the oil spill response.

Electricity Production
The purpose of the Cape Wind project is to generate electricity. Electricity will travel from the WTGs to the ESP and then by submarine cable to on-land cables in Yarmouth, Massachusetts. From this point, electricity generated at the WTGs would be distributed to the New England Power Grid. Electricity will then be used to support existing uses. The total generating capacity in the New England power system in the year 2004 was 30,940 megawatts (MW; reported in MMS 2008). The maximum electric output of the Cape Wind project is predicted to be 468 MW, with an average output of 182.6 MW. Effects to listed species from the distribution and use of electricity generated by the Cape Wind project can not be predicted. However, as the electricity generated will support existing uses, any effects of these uses on sea turtles or whales

are expected to have been captured in the Status of the Species and Environmental Baseline sections above.

In the DEIS, MMS estimated that if the amount of energy produced by the proposed project was to be produced by fossil-fuel powered plants instead, it would result in about 0.88 million tons of carbon dioxide emitted per year. The projected increase in energy needs in New England between 2005 and 2014 would result in an increase of about 84 tons per year of carbon dioxide if the power were to be produced by fossil-fuel power plants. MMS estimated that the potential reduction in the growth of carbon dioxide emissions due to operation of the proposed project would be about 1 percent of the total projected increase. Whether there would be effects to listed species from a reduction in the growth of carbon dioxide emissions is unknown, as is what any such effect might be.

CUMULATIVE EFFECTS

Cumulative effects, as defined in the ESA, are those effects of future state or private activities, not involving federal activities that are reasonably certain to occur within the action area of the federal action subject to consultation. Future federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

Sources of human-induced mortality, injury, and/or harassment of listed sea turtles in the action area that are reasonably certain to occur in the future include state fisheries, vessel collisions and pollution. While the combination of these activities may affect loggerhead sea turtles, the magnitude of these effects is currently unknown.

Commercial Fishing - Future commercial fishing activities in state waters may take (capture, injure or kill) several protected species. However, it is not clear to what extent these future activities would affect listed species differently than the current state fishery activities described in the Environmental Baseline section. The Atlantic Coastal Cooperative Statistics Program (ACCSP) and the NMFS sea turtle/fishery strategy, when implemented, are expected to provide information on takes of protected species in state fisheries and systematically collected fishing effort data which will be useful in monitoring impacts of the fisheries. NMFS expects these state water fisheries to continue in the future, and as such, the potential for interactions with listed species will also continue.

Vessel Interactions – NMFS' STSSN data indicate that vessel interactions are responsible for a number of sea turtles strandings within the action area each year. Vessel use and collisions with sea turtles are reasonably certain to continue into the future. Collisions with boats can stun or easily kill sea turtles, and many stranded turtles have obvious propeller or collision marks (Dwyer *et al.* 2003). However, it is not always clear whether the collision occurred pre- or post-mortem. As a result an estimate of the number of sea turtles that will likely be killed by vessels is not possible.

Pollution and Contaminants - Human activities causing pollution are reasonably certain to continue in the future, as are impacts from them on loggerhead sea turtles in the action area. However, the level of impacts cannot be projected. Marine debris (*e.g.*, discarded fishing line or lines from boats) can entangle turtles in the water and drown them. Chemical contaminants may also have an effect on sea turtle reproduction and survival. Excessive turbidity due to coastal development and/or construction sites could influence sea turtle foraging ability. As mentioned previously, turtles are not very easily affected by changes in water quality or increased suspended sediments, but if these alterations make habitat less suitable for turtles and hinder their capability to forage, eventually they would tend to leave or avoid these less desirable areas (Ruben and Morreale 1999). Noise pollution has been raised, primarily, as a concern for marine mammals but may be a concern for other marine organisms, including sea turtles. The effects of increased noise levels can range from minor behavioral disturbance to injury and even death. Acoustic impacts can include auditory trauma, temporary or permanent loss of hearing sensitivity, habitat exclusion, habituation, and disruption of other normal behavior patterns such as feeding, migration, and communication. NMFS is working to develop policy guidelines for monitoring and managing acoustic impacts on marine mammals from anthropogenic sound sources in the marine environment. As described above, global warming is likely to negatively affect sea turtles – affecting when females lay their eggs, the survival of the eggs, sex ratios of offspring, and the stability of the Gulf Stream. To the extent that air pollution, for example from the combustion of fossil fuels by vessels, contributes to global warming, then it is also expected to negatively affect sea turtles in the action area.

INTEGRATION AND SYNTHESIS OF EFFECTS

In the effects analysis outlined above, NMFS considered potential effects from the following sources: (1) construction of the facility including the submarine cables and the WTGs, (2) operation and maintenance of the facility, (3) pre-construction geotechnical and geophysical surveys, and, (4) decommissioning. In addition to these categories of effects, NMFS considered the effects of non-routine and accidental events including oil spills, cable repair, and vessel collisions with a monopole.

Right, humpback, and fin whales

As noted in sections above, whales are extremely unlikely to occur in Nantucket Sound. The analysis contained above demonstrates that all effects of the project will be contained within Nantucket Sound. While noise associated with pile driving will extend several kilometers from the pile being driven, no whales are likely to be exposed to injurious or harassing sound levels. Additionally, as no whales will occur along the vessel transit routes or along the cable laying route, no interactions with project vessels or the cable laying are likely. As all effects to whales from the proposed project are likely to be insignificant or discountable, this action is not likely to adversely affect listed whales in the action area.

Kemp's ridley, loggerhead, green and leatherback sea turtles

As noted in sections above, the physical disturbance of sediments and associated benthic resources from various aspects of the project including cable laying and monopile installation, could reduce the availability of sea turtle prey in the affected areas, but these reductions will be localized and temporary, and foraging turtles are not likely to be limited by the reductions.

MMS will require several mitigation measures that will reduce the likelihood of interactions between sea turtles and project vessels, including the presence of observers. Based on the analysis presented above, the increase in risk of a vessel strike to a sea turtle in the action area is insignificant.

Marine animals are known to be injured and harassed by anthropogenic noise sources. In the Effects of the Action section above, NMFS has determined that any effects of exposure to construction and maintenance vessel noise, cable laying activities, and operation of the WTGs will be insignificant or discountable. However, sea turtles are likely to be exposed to disturbing levels of noise during pile driving and the high resolution shallow hazards survey.

Mitigation measures implemented during impact pile driving minimize the potential for acoustic-related injuries to sea turtles. Based on the analysis presented above, no sea turtles are likely to be exposed to potentially injurious levels of sound. However, sea turtles may be exposed to potentially disturbing levels of sound during pile driving activities. Any sea turtles located within 3.4km of a pile being driven are likely to be disturbed and exhibit avoidance behavior. As explained on page 90, NMFS has estimated that between 3 and 7 sea turtles are likely to be exposed to disturbing levels of noise during each 4 hour pile driving event that occurs between June and November.

Similarly to pile driving operations, mitigation measures implemented during the high resolution geophysical survey will minimize the potential for acoustic-related injuries to sea turtles. Based on the analysis presented above, no sea turtles are likely to be exposed to potentially injurious levels of sound resulting from the survey. However, sea turtles may be exposed to potentially disturbing levels of sound during the high resolution geophysical survey. Any sea turtles located within one km outside of the exclusion zone (i.e., from 0.5 – 1.5 km from the survey vessel) will be exposed to potentially disturbing levels of noise. At any given time during the survey, an approximately 3.14 square kilometer area will have noise levels between 160 and 180 dB. NMFS has estimated that, in total,between 13 and 28 sea turtles would be exposed to disturbing levels of noise during the survey.

Avoidance behavior may shorten the exposure period; however, the avoidance behavior could potentially disrupt normal behaviors. Sea turtles behaviorally disrupted would be expected to resume their behavior after the noise producing activity (i.e., pile driving or high resolution survey) has stopped. As pile driving will occur for approximately 4 hours a day, it is likely that sea turtles will be excluded from the area with disturbing levels of sound for at least this period each day. Likewise, during the time the high resolution geophysical survey is ongoing, sea turtles would be excluded from the area with disturbing levels of sound. While sea turtles may move to other areas within the action area to forage during the times when pile driving or the high resolution geophysical survey is occurring, the ability of individual sea turtles to find suitable forage is not expected to be impacted. Likewise, if sea turtles were resting in a particular area they are expected to be able to find an alternate resting area within the action area. Additionally, if sea turtles are migrating through the action area, they may avoid the area with disturbing levels of sound and choose an alternate route through the action area. However, as at all times there will be areas of Nantucket Sound where noise levels are not at disturbing levels, the ability of sea turtles to migrate through the action area will not be affected. As such, while the movements of individual sea turtles while be affected by the sound associated with the pile driving and the high resolution geophysical survey, these effects will be temporary and localized and sea turtles are not expected to be excluded from Nantucket Sound and there will be only a minimal impact on foraging, migrating or resting sea turtles that will not result in injury or impairment in individuals' ability to complete essential behavioral functions. Major shifts in habitat use or distribution or foraging success are not expected. Changes to individuals' movements are expected to be minor and short-term, and are, therefore, not likely to reduce numbers, reproduction or distribution. All other effects of the proposed project are expected to be insignificant or discountable and are not expected to reduce numbers, reproduction or distribution.

While the action may affect the distribution of sea turtles in the action area during the approximately four hours a day while pile driving is occurring (as sea turtles will avoid the 34.56 square kilometer impact zone), and during the 36-hour high resolution geophysical survey, the effect on distribution will be temporary and localized. As such, the action will not affect the overall long-term distribution of loggerhead, Kemp's ridley, green or leatherback sea turtles in the action area or throughout their range.

While the proposed action may temporarily affect the movement of individual sea turtles in the action area, NMFS has determined that this will not affect the overall distribution or abundance of sea turtles in the action area. Nor will it affect the ability of any individual sea turtles to complete any essential behavioral function such as foraging, resting or migrating. Therefore, the temporary disturbance caused by noise associated with pile driving will not negatively affect any sea turtles' chances of survival. Their ability to reproduce would be the same as for a sea turtle that had not been exposed to pile driving noise.

As no sea turtles will be injured or killed by the proposed action, either directly, through loss of prey and/or habitat, or other means, the action will not reduce the number of loggerhead, Kemp's ridley, green or leatherback sea turtles. Additionally, as the action will not affect the reproductive success of any individual turtle, it will not reduce the reproduction of loggerhead, Kemp's ridley, green or leatherback sea turtles. Therefore, the proposed action will not affect the numbers, reproduction or distribution of sea turtles in the western north Atlantic, and will not reduce their likelihood of survival. Since the proposed action has no direct or indirect effects on sea turtles that occur elsewhere in the Atlantic or outside of the Atlantic, the proposed action will not appreciably reduce the likelihood of survival of any species of sea turtle.

Section 4(a)(1) of the ESA requires listing of a species if it is in danger of extinction throughout all or a significant portion of its range (i.e., "endangered"), or likely to become in danger of extinction throughout all or a significant portion of its range in the foreseeable future (i.e., "threatened") because of any of the following five listing factors: (1) The present or threatened destruction, modification, or curtailment of its habitat or range, (2) overutilization for commercial, recreational, scientific, or educational purposes, (3) disease or predation, (4) the inadequacy of existing regulatory mechanisms, (5) other natural or manmade factors affecting its continued existence. Recovery of a species occurs when listing it as an endangered or threatened species is no longer warranted. The proposed action will not appreciably reduce the likelihood of recovery of any sea turtle species because it will not affect the numbers, reproduction or distribution of loggerhead, Kemp's ridley, green or leatherback sea turtles. Also, it is not expected to modify, curtail or destroy the range of the species since it does not reduce the number of loggerhead, Kemp's ridley, green or leatherback sea turtles in any geographic area or nesting group and since it will not affect the overall distribution of sea turtles other than to cause minor temporary adjustments in movements in the action area. The proposed action will not utilize loggerhead, Kemp's ridley, green or leatherback sea turtles for recreational, scientific or commercial purposes, affect the adequacy of existing regulatory mechanisms to protect any of these species of sea turtles, or affect their continued existence. The effects of the proposed action will not hasten the extinction timeline or otherwise increase the danger of extinction since the action will not result in mortality of loggerhead sea turtles or their ability to survive and reproduce. Therefore, the proposed action will have no effect on the ESA listing factors or the likelihood that loggerhead, Kemp's ridley, green or leatherback sea turtles can be brought to the point at which they are no longer listed as endangered or threatened. In light of the conclusions of the effect of the action relative to the ESA-listing factors, the proposed action will not appreciably reduce the likelihood of recovery for any of the sea turtle species.

CONCLUSION

After reviewing the best available information on the status of endangered and threatened species under NMFS jurisdiction, the environmental baseline for the action area, the effects of the action, and the cumulative effects, it is NMFS' biological opinion that the proposed action may adversely affect but is not likely to jeopardize the continued existence of the loggerhead, Kemp's ridley, leatherback or green sea turtles. Additionally, NMFS has concluded that the proposed action is not likely to adversely affect right, humpback or fin whales and, therefore, is not likely to jeopardize the continued existence of these whale species. NMFS has also concluded that the action will not affect hawksbill turtles, shortnose sturgeon, or sperm, blue or sei whales as these species do not occur in the action area. Because no critical habitat is designated in the action area, none will be affected by the proposed action.

INCIDENTAL TAKE STATEMENT

Section 9 of the ESA prohibits the take of endangered species. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. NMFS interprets the term "harm" as an act which actually kills or injures fish or wildlife. Such an act may include significant habitat modification or degradation where it actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns, including breeding, spawning, rearing, migrating, feeding or sheltering (50 CFR §222.102). Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. The term "harass" has not been defined by NMFS; however, it is commonly understood to mean to annoy or bother. In addition, legislative history helps elucidate Congress' intent: "[take] includes harassment, whether intentional or not. This would allow, for example, the Secretary to regulate or prohibit the activities of birdwatchers where the effect of those activities might disturb the birds and make it difficult for them to hatch or raise their young" (HR Rep. 93-412, 1973). Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited under the ESA provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement.

Amount or Extent of Incidental Take

The proposed action has the potential to directly affect loggerhead, Kemp's ridley, green and leatherback sea turtles by causing them to be exposed to potentially harassing levels of sound during pile driving and the high resolution geophysical survey. As explained in the "Effects of the Action" section of the accompanying Opinion, only sea turtles located within a 34.56 square km area surrounding the pile being driven will be exposed to noise levels between 160 and 180 dB. As explained on page 90 of the "Effects of the Action" section, NMFS has estimated that between 3 and 7 sea turtles are likely to be exposed to disturbing levels of noise during each 4 hour pile driving event. As pile driving will occur for approximately four hours per pile over a period of approximately eight months, the potential for exposure will be limited to that time period only. As explained in the "Effects of the Action" section, during the one time 36-hour high resolution geophysical survey, any sea turtles located within one km outside of the exclusion zone (i.e., from 0.5 – 1.5 km from the survey vessel) will be exposed to noise levels between 160 and 180 dB. During the survey, an area of approximately 148 square kilometers will be surveyed. Based on the estimates of sea turtle density in the action area (explained on page 90), NMFS estimates that between 13 and 28 sea turtles would be exposed to disturbing levels of noise during the survey. At any given time during the survey, an approximately 3.14 square kilometer area will have noise levels between 160 and 180 dB.

Exposure of sea turtles to sound levels greater than 160 dB will be considered harassment because that level of noise will disturb sea turtles and their normal behaviors (i.e., resting, foraging or migrating through the area) will be interrupted. Any sea turtles located within 3.4km of the pile being driven will be exposed to these disturbing noise levels and are likely to exhibit avoidance behavior which would cause the alteration of normal behaviors. As loggerhead, Kemp's ridley, green and leatherback sea turtles are likely to be present in the action area and exposed to potentially harassing sound levels, harassment of any of these species could occur and NMFS anticipates that the 3-7 sea turtles exposed to harassing noise levels during each pile driving event and the 13-28 sea turtles exposed to harassing levels of noise during the geophysical survey will be a combination of these species. As sea turtles are only likely to occur in the action area between June and November, only pile driving occurring during these months will result in the harassment of sea turtles. Similarly, effects to sea turtles from the high resolution geophysical survey would only occur if the survey took place between June and November. Incidental take via harassment will be limited to the spatial and temporal extent indicated above.

NMFS believes this level of incidental take is reasonable given the likely seasonal distribution and abundance of sea turtles in the action area and the modeling results provided by MMS in the

BA and DEIS. In the accompanying biological opinion, NMFS determined that this level of anticipated take is not likely to result in jeopardy to the species. As explained above, any incidental take will be limited to: the time period when pile driving is occurring and be limited to a 34.56 square kilometer area surrounding the pile being driven and the time period when the high resolution geophysical survey is occurring and be limited to a 3.14 square kilometer area at any given time during the survey.

Reasonable and prudent measures
Reasonable and prudent measures are those measures necessary and appropriate to minimize and monitor incidental take of a listed species. These reasonable and prudent measures are in addition to the mitigation measures proposed by MMS and agreed to by Cape Wind that will become a part of the proposed action (see Appendix A of the accompanying Biological Opinion). NMFS believes the following reasonable and prudent measures are necessary and appropriate to minimize and monitor impacts of incidental take of sea turtles:

1. MMS must ensure that any endangered species monitors contracted by Cape Wind are approved by NMFS.
2. During the conduct of pile driving activities related to turbine monopile and Electrical Service Platform (ESP) installation, the 750 meter exclusion zone must be monitored by a NMFS-approved endangered species monitor for at least 60 minutes prior to pile driving.
3. During the conduct of the high resolution geophysical survey, the 500 meter exclusion zone must be monitored by a NMFS-approved endangered species monitor for at least 60 minutes prior to the survey.
4. Acoustic measurement of the first pile being driven must be conducted to confirm the sound levels modeled by MMS and reported in the BA.
5. Prior to decommissioning, MMS must provide to NMFS a complete plan for decommissioning activities.

Terms and conditions
In order to be exempt from prohibitions of section 9 of the ESA, MMS must comply with the following terms and conditions, which implement the reasonable and prudent measures described above and which outline required minimization and monitoring requirements. These terms and conditions are non-discretionary.

1. To implement RPM #1, MMS shall provide NMFS with the names and resumes of all endangered species monitors to be employed at the project site at least 30 days prior to the start of construction. No observer shall work at the project site without written approval of NMFS. If during project construction or operations, additional endangered species monitors are necessary, MMS will provide those names and resumes to NMFS for approval at least 10 days prior to the date that they are expected to start work at the site.

2. To implement RPM #2, observers must begin monitoring at least 60 minutes prior to soft start of the pile driving. Pile driving must not begin until the zone is clear of all sea turtles for at least 60 minutes. Monitoring will continue through the pile driving period and end approximately 60 minutes after pile driving is completed.

3. To implement RPM #2 and #3, adequate lighting must be provided on all vessels used for endangered species observation to ensure that observers can monitor the exclusion zone for listed sea turtles. If sufficient lighting can not be provided, activities must be limited to daylight hours.

4. To implement RPM #3, observers must begin monitoring at least 60 minutes prior to the start of the high resolution geophysical survey. The survey must not begin until the zone is clear of all sea turtles for at least 60 minutes. Monitoring will continue through the survey period and end approximately 60 minutes after the survey is completed.

5. To implement RPM #4, acoustic monitoring must be conducted to verify that sound levels at 3.4km from the pile being driven is less than 160 dB. Results of this monitoring must be reported to NMFS prior to the driving of any subsequent piles.

6. To implement RPM #5, if the project is to be decommissioned, MMS must provide a complete decommissioning plan and analysis of effects on listed species to NMFS. NMFS would then review the plan to determine if reinitiation of this consultation is necessary.

The reasonable and prudent measures, with their implementing terms and conditions, are designed to minimize and monitor the impact of incidental take that might otherwise result from the proposed action. Specifically, these RPMs and Terms and Conditions will ensure that no listed species are exposed to injurious levels of sound and will verify the modeling results provided by MMS based on which NMFS has made conclusions regarding take. RPM and Term and Condition #1 is necessary and appropriate because it is specifically designed to ensure that all endangered species monitors employed by the applicant are qualified to conduct the necessary duties. Including this review of endangered species monitors by NMFS staff is only a minor change because it is not expected to result in any delay to the project and will merely enforce the qualifications of the endangered species monitors that are already required by MMS. RPM and Term and Condition #2 as well as RPM#3 and Term and Condition #4 are necessary and appropriate to provide adequate monitoring by extending the time that monitoring of the exclusion zone must occur from the 30 minutes required by MMS to 60 minutes. The normal duration of sea turtle dives ranges from 5-40 minutes depending on species, with a maximum duration of 45-66 minutes depending on species (Spotila 2004). As sea turtles can stay submerged for longer than 30 minutes, but typically surface at least every 60 minutes, it is reasonable to require that monitoring occur for at least 60 minutes to allow the endangered species monitor to detect any sea turtles that may be submerged in the exclusion zone. Increasing the time to 60 minutes is only a minor change because the observer will be on location already and an additional 30 minutes of observation is not expected to result in any effects to the project schedule. Term and Condition #3 is necessary and appropriate to provide adequate monitoring of the exclusion zone as if lighting is poor the endangered species monitors will not be able to effectively survey the exclusion zone. Requiring adequate lighting is only a minor change because the vessels will already have some lighting and the addition of extra lighting is not expected to be more than a minor cost and not cause any delay of the project. If sufficient lighting can not be provided and activities must be curtailed during the dark, the delay in project schedule will be only a few hours and this is not expected to result in more than a minor cost and minor effect on overall project schedule. RPM #4 and Term and Condition #5 are necessary and appropriate because they are designed to verify that the sound levels modeled by MMS are valid and that the 3.4km zone where sound levels are expected to be greater than 160dB is accurate. This RPM and Term and Condition does not cause more than minor changes because Cape Wind is already required by MMS to conduct monitoring of underwater sound levels associated with the driving of the first three piles. These measurements must be taken at 100m, 500m and 750m in two directions either west, east, south or north of the pile driving site. The addition of one additional monitoring site for one pile driving event will not cause delays to the project or add a significant cost. RPM #5 and Term and Condition #6 is necessary and appropriate as way to help monitor the proposed action and incidental take by ensuring that the effects of any decommissioning activities on listed species have been adequately analyzed. As it is impossible to predict the exact decommissioning scenario and the status of listed species at the time of decommissioning it is necessary to review the decommissioning plan when it is developed.

These RPMs and Terms and Conditions in conjunction with the mitigation measures proposed by MMS and agreed to by Cape Wind that will become a part of the proposed action will serve to minimize and monitor incidental take of listed species.

CONSERVATION RECOMMENDATIONS

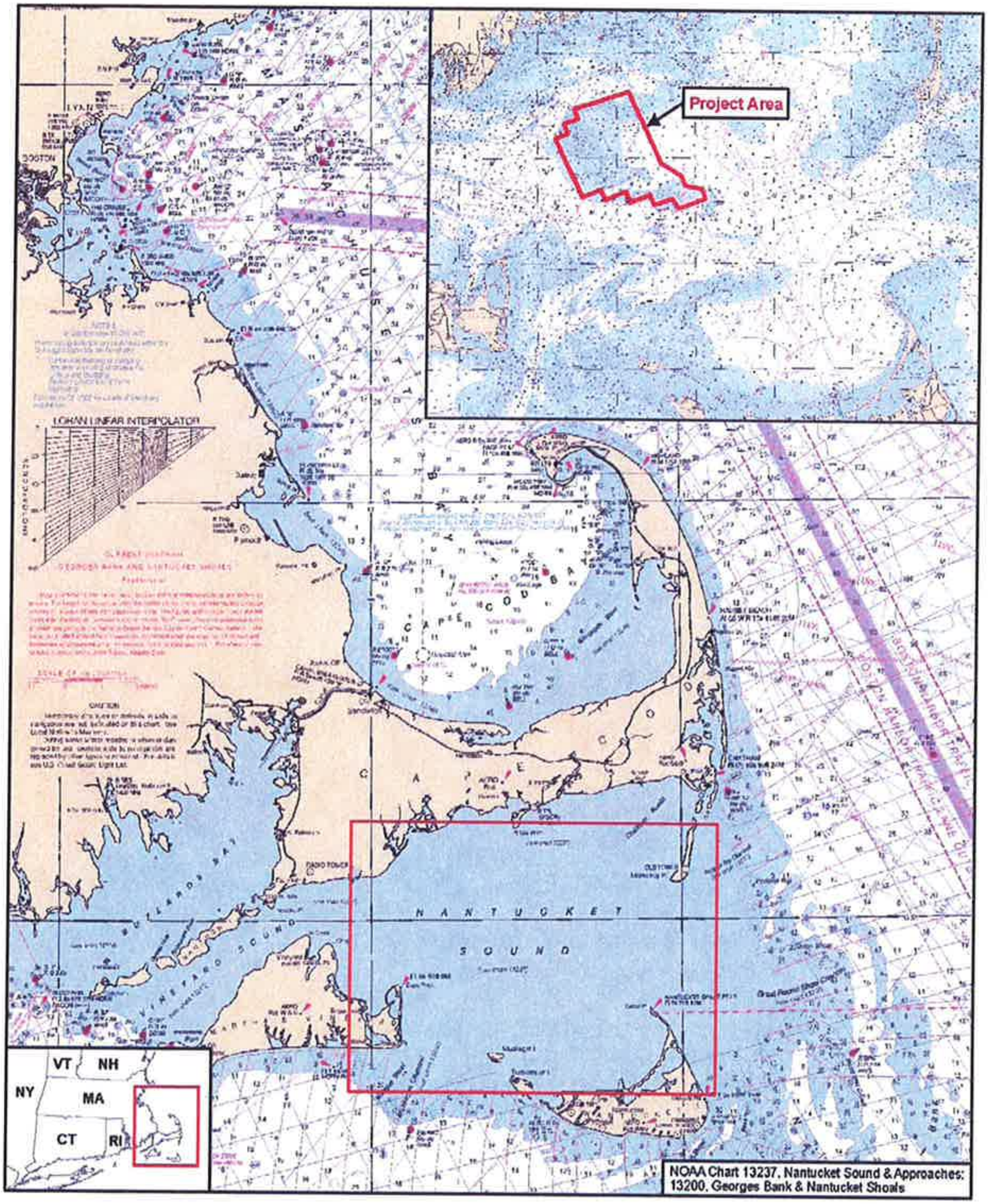
Section 7(a)(1) of the ESA directs Federal agencies to utilize their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information. NMFS has determined that the proposed action is not likely to jeopardize the continued existence of any listed species. To further reduce the adverse effects of the proposed actions, NMFS recommends that MMS work with the applicant, Cape Wind Associates, to implement the following conservation recommendations.

1. To the extent practicable, pile driving should be minimized during the June – October timeframe when sea turtles are expected to occur in the action area.
2. As there is limited data on use of Nantucket Sound by listed sea turtles, MMS and/or Cape Wind should support additional survey effort. This could include aerial surveys of the action area specifically targeting sea turtles.

REINITIATION OF CONSULTATION

This concludes formal consultation on MMS's proposed approval of an application by Cape Wind Associates, LLC for a lease, easement or right-of-way to construct, operate and decommission a wind energy project on Horseshoe Shoal. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded; (2) a new species is listed or critical habitat designated that may be affected by the action; (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in this opinion; or (4) new information reveals effects of the action that may affect listed species or critical habitat in a manner or to an extent not previously considered. If the amount or extent of incidental take is exceeded, the MMS must immediately request reinitiation of formal consultation.

Figure 1. Project Area



CAPE WIND ENERGY PROJECT

Figure BA-1
Project Locus Map

Figure 2. Action Area

= action area

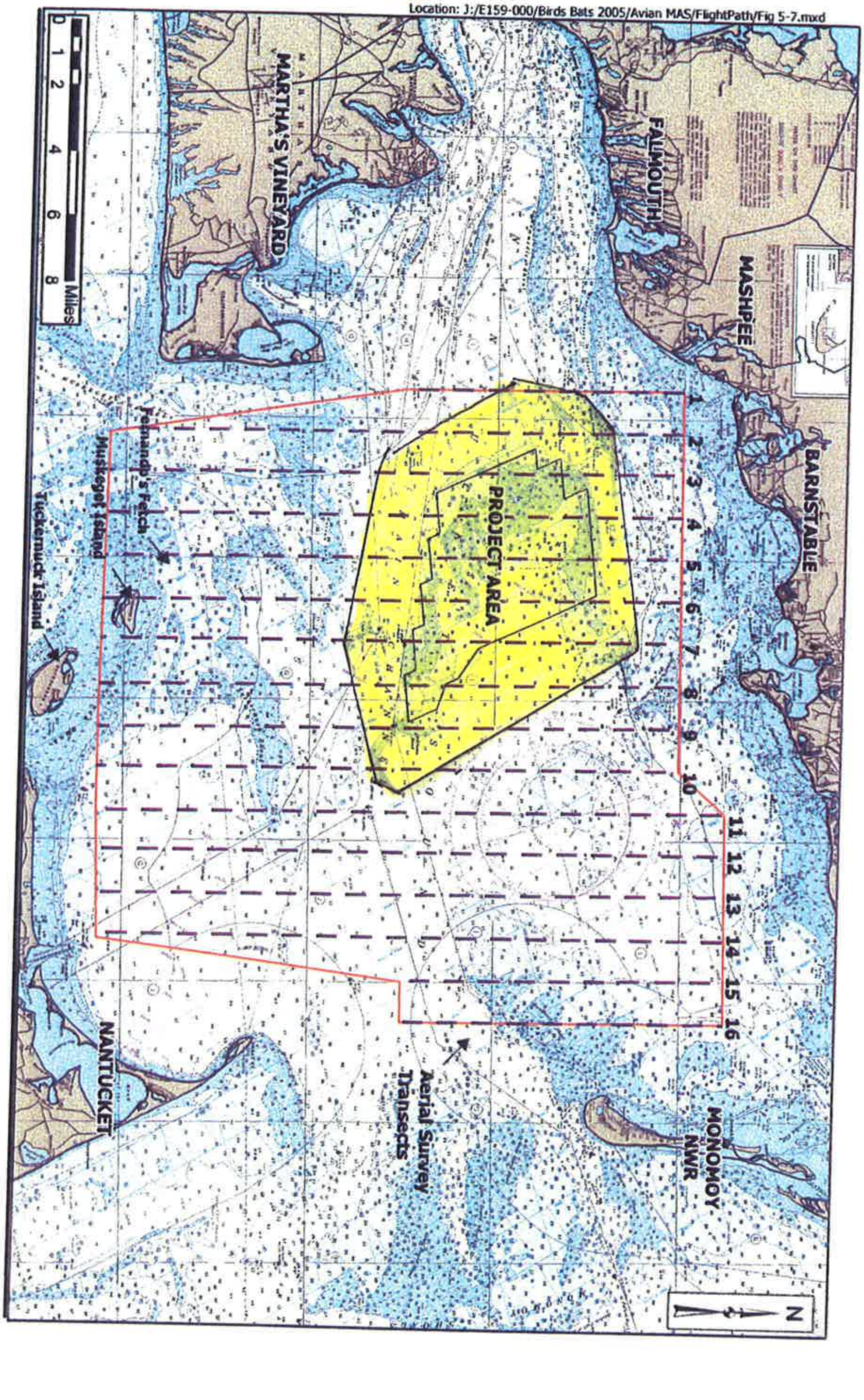


Figure 3
Mass Audubon
aerial survey area

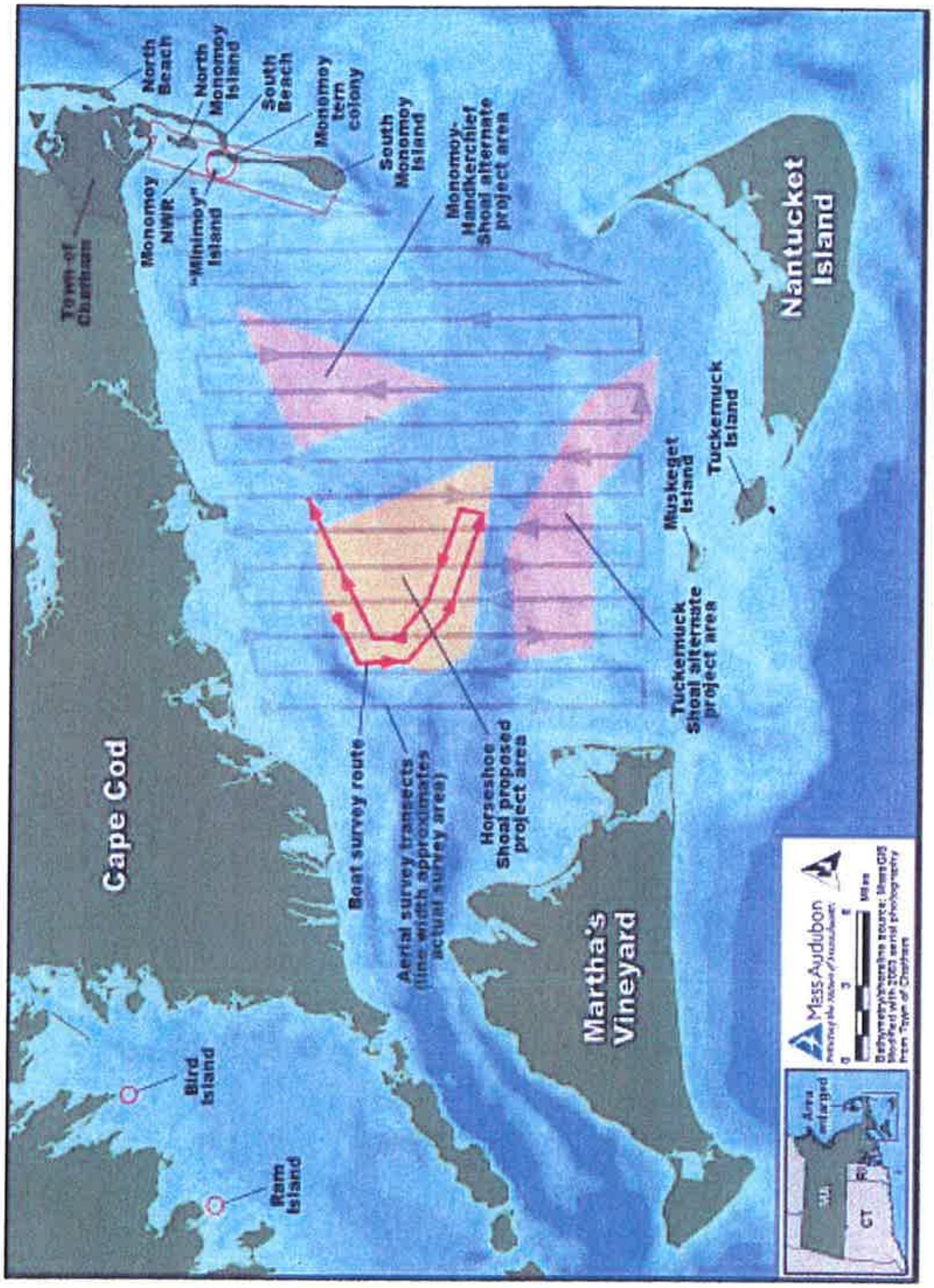
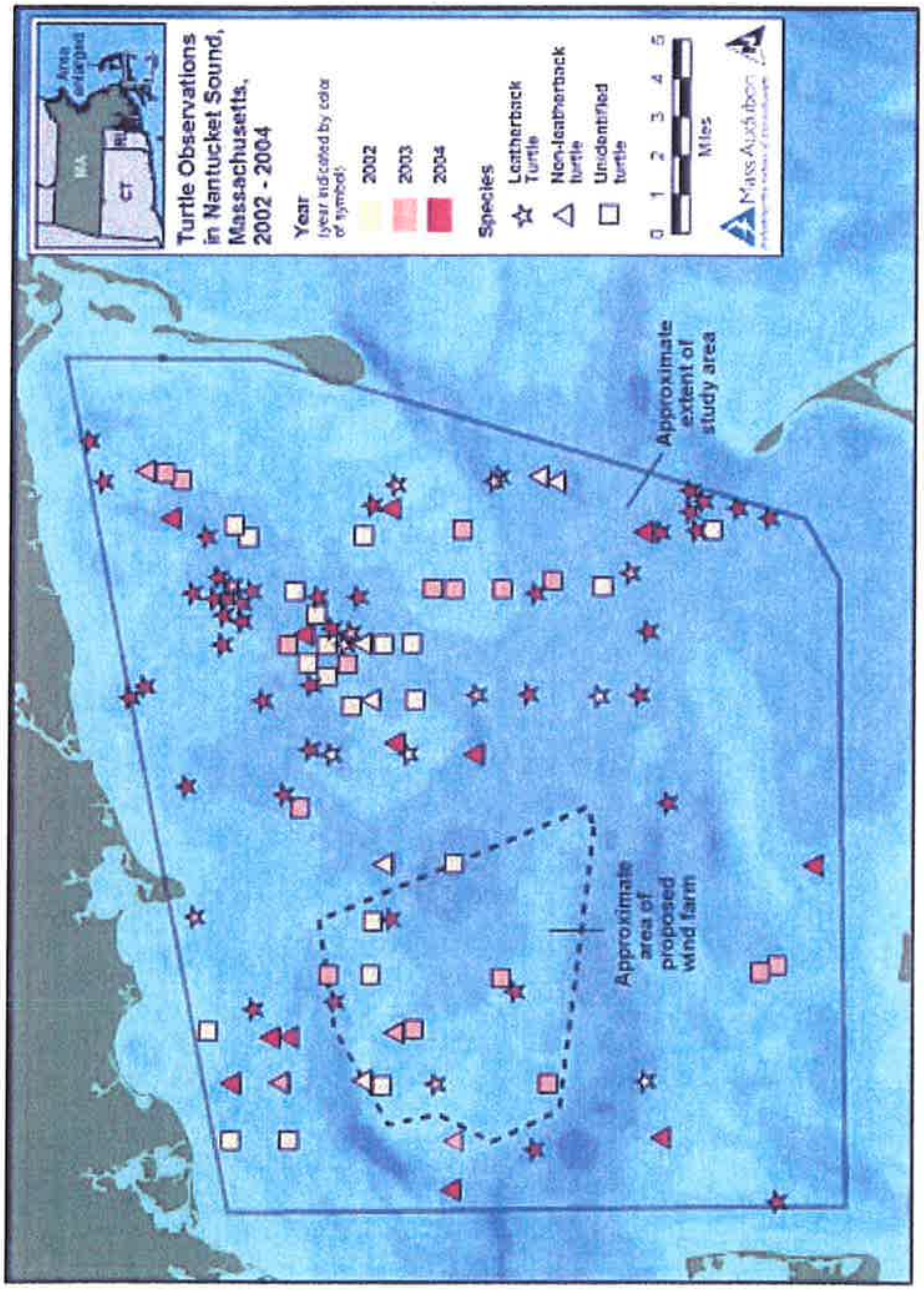


Figure 4
Mass Audubon
Sea turtle sightings



APPENDIX A
From MMS BA dated May 2008

8.0 MITIGATION, MONITORING AND REPORTING REQUIREMENTS FOR ESA LISTED SPECIES

This section outlines the specific mitigation, monitoring and reporting measures built into the proposed action to minimize or eliminate potential impacts to ESA-listed species of whales, sea turtles and birds. Any additional mitigation, monitoring or reporting measures may be added during the Federal ESA Section 7 process or through any issued MMS leases or other authorizations.

8.1 Measures for ESA-Listed Marine Mammals and Sea Turtles

The following measures are part of the proposed action and are meant to minimize or eliminate the potential for adverse impacts to ESA-listed whales and sea turtles. They are divided into the five sections: (1) those required during all phases of the project; (2) those required during pre-construction site assessment: (3) those required during construction; (4) those required during operation/maintenance; and (5) those required during decommissioning. These measures and those that may ultimately be required through the ESA consultation process will be included as requirements in any MMS lease or other authorization, if issued, for the proposed activity.

The applicant has informed MMS that it intends to seek authorization from NMFS under the MMPA. Therefore, MMS will require that the MMPA authorization be completed and a copy provided to MMS before activities are allowed to commence under any MMS issued lease or other authority that may result in the taking of marine mammals. This also includes any amended ESA incidental take statement, if issued, to include marine mammals. Any measures contained within any MMPA authorization, if issued, that are more conservative than those measures built into this proposed action will take precedence.

8.1.1 Requirements for All Phases of Project

As noted in Section 2.3 of the DEIS, the construction phase of the proposed action will 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 vessel traffic within the vicinity of the proposed action area. During construction activities, especially during pile driving activities, it is estimated that 4 to 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 will also be present in the area between the mainland and the proposed action site. The barges, tugs and vessels delivering construction materials generally will travel at 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 will typically be 50 ft (15 m) in length. The additional traffic from construction vessels may increase the chance of a strike or harassment of marine mammals or sea turtles.

Sections 2.3, 2.4 and 2.5 of the DEIS provides detail on the vessel and aircraft activity associated with the operations/maintenance and decommissioning phases of the project.

The following specific measures are meant to reduce the potential for vessel harassments or collisions with listed whales or sea turtles during all phases of the project.

- All vessels and aircraft associated with the construction, operation/maintenance and/or decommissioning of the project will be required to abide by the: (1) NOAA Fisheries Northeast Regional Viewing Guidelines, as updated through the life of the project (http://www.nmfs.noaa.gov/pr/pdfs/education/viewing_northeast.pdf); and (2) MMS Gulf of Mexico Region's Notice to Lessee (NTL) No. 2007-G04 (http://www.gomr.mms.gov/homepg/regulate/regs/ntls/2007NTLs/07-g04.pdf).

- All vessel and aircraft operators must undergo training to ensure they are familiar with the above requirements. These training requirements must be written into any contractor agreements.

- All vessel operators, employees and contractors actively engaged in offshore operations must be briefed on marine trash and debris awareness elimination as described in the MMS Gulf of Mexico Region's NTL No. 2007-G03 (http://www.gomr.mms.gov/homepg/regulate/regs/ntls/2007NTLs/07-g03.pdf). MMS will not require the applicant to undergo formal training or post placards, as described under this NTL. The applicant will be required to ensure that its employees and contractors are made aware of the environmental and socioeconomic impacts associated with marine trash and debris and their responsibilities for ensuring that trash and debris are not intentionally or accidentally discharged into the marine environment. The above referenced NTL provides information the applicant may use for this awareness training.

8.1.2 Requirements During Pre-Construction Site Assessment Geophysical Surveys

Section 2.7 of the DEIS describes the marine shallow hazards surveys and geotechnical program the applicant would undertake should MMS issue a lease for the proposal. These geophysical and geotechnical (G&G) field investigations would be conducted prior to construction.

The following mitigation, monitoring and reporting requirements will be implemented during the conduct of all high-resolution seismic surveying work proposed by the applicant. Additional detail on how these measures will be implemented is described in the MMS Gulf of Mexico (GOM) Notice to Lessee (NTL) No. 2007-G02 (see http://www.gomr.mms.gov/homepg/regulate/regs/ntls/2007NTLs/07-g02.pdf). Although this NTL focuses on seismic surveying with air guns in the GOM, the methodologies described in the NTL for exclusion zone monitoring, ramp up and shut down as the same as those that will be required under this proposed action.

- *Establishment of Exclusion Zone*: A 250 m (820.2 ft) radius exclusion zone for listed whales and sea turtles will be established around the seismic survey source vessel in order to reduce the potential for serious injury or mortality of these species.

- *Visual Monitoring of Exclusion Zone*: The exclusion zone around the seismic survey source vessel must be monitored for the presence of listed whales or sea turtles before, during and after any pile driving activity. The exclusion zone will be monitored for 30 minutes prior to the ramp up (if applicable) of the seismic survey sound source. If the exclusion zone is obscured by fog or poor lighting conditions, surveying will not be initiated until the entire exclusion zone is visible for the 30 minute period. If listed whales or sea turtles are observed within the zone during the 30 minute period and before the ramp up begins, surveying will be delayed until they move out of the area and until at least an additional 30 minutes have passed without a listed whale or sea turtle sighting. Monitoring of the zone will continue for 30 minutes following completion of the seismic surveying.

  Monitoring of the zones will be conducted by one qualified NMFS approved observer[3]. Visual observations will be made using binoculars or other suitable equipment during daylight hours. Data on all observations will be recorded based on standard marine mammal observer collection data. This will include: dates and locations of construction operations; time of observation, location and weather; details of marine mammal sightings (e.g., species, numbers, behavior); and details of any observed taking (behavioral disturbances or injury/mortality). Any significant observations concerning impacts on listed whales or sea turtles will be transmitted to NMFS and MMS within 48 hours. Any observed takes of listed whales or sea turtles resulting in injury or mortality will be immediately reported to NMFS and MMS.

- *Implementation of Ramp Up*: A "ramp up" (if allowable depending on specific sound source) will be required at the beginning of each seismic survey in order to by allowing them to vacate the area prior to the commencement of activities. Seismic surveys may not commence (i.e., ramp up) at night time or when the exclusion zone cannot be effectively monitored (i.e., reduced visibility).

- *Shut Down*: Continuous (day and night) seismic survey operations will be allowed. However, if a listed whale or sea turtle is spotted within or transiting towards the exclusion zone surrounding the sub-bottom profiler and the survey vessel, an immediate shutdown of the equipment will be required. Subsequent restart of the profiler will only be allowed following clearance of the exclusion zone and the implementation of ramp up procedures (if applicable).

- *Compliance with Equipment Noise Standards*: All seismic surveying equipment will comply as much as possible with applicable equipment noise standards of the U.S.

[3] Observer qualifications will include direct field experience on a marine mammal/sea turtle observation vessel and/or aerial surveys in the Atlantic Ocean/Gulf of Mexico. All observers will receive NMFS-approved marine mammal observer training and be approved in advance by NMFS after a review of their qualifications.

Environmental Protection Agency, and all equipment will have noise control devices no less effective than those provided on the original equipment.

- *Reporting for Seismic Surveys Activities*: The following reports must be submitted during the conduct of seismic surveys:

  - A report will be provided to MMS and NMFS within 90 days of the commencement of seismic survey activities that includes a summary of the seismic surveying and monitoring activities and an estimate of the number of listed whales and sea turtles that may have been taken as a result of seismic survey activities. The report will include information, such as: dates and locations of operations, details of listed whale or sea turtle sightings (dates, times, locations, activities, associated seismic activities), and estimates of the amount and nature of listed whale or sea turtle takings.

  - Any observed injury or mortality to a listed whale or sea turtle must be reported to NMFS and MMS within 24 hours of observation. Any significant observations concerning impacts on listed whales or sea turtles will be transmitted to NMFS and MMS within 48 hours.

8.1.3 Requirements During Construction

Acoustic harassment from construction activities hold the greatest potential for disturbance and impacts to listed whales and sea turtles due to the size and number of piles and the timeframe needed to complete the installation of all piles. Section 2.5.1 of the BA and Sections 2.3.2.2 of the DEIS describe the pile driving process in detail. Section 5.0 of the BA and Sections 5.3.2.9.1 of the DEIS outline the potential effects of pile driving activities on listed whales and sea turtles.

MMS has included the following specific measures as part of the proposed action and are meant to reduce or eliminate the potential for adverse impacts on listed whales or sea turtles during the construction phase of the project:

- *Pre-Construction Briefing*: Prior to the start of construction, a briefing will be held between the construction supervisors and crews, the marine mammal and sea turtle visual and acoustic observer(s) (see further below), and Cape Wind Associates. The purpose of the briefing will be to establish responsibilities of each party, define the chains of command, discuss communication procedures, provide an overview of monitoring purposes, and review operational procedures. The Resident Engineer will have the authority to stop or delay any construction activity, if deemed necessary. New personnel will be briefed as they join the work in progress.

- *Requirements for Pile Driving*: The following measures will be implemented during the conduct of pile driving activities related to turbine monopile and Electrical Service Platform (ESP) installation:

- Establishment of Exclusion Zone: A preliminary 750 m (2,461 ft)[4] radius exclusion zone for listed whales and sea turtles will be established around each pile driving site in order to reduce the potential for serious injury or mortality of these species. Once pile driving begins, the actual generated sound levels will be measured (see requirements below for *Field Verification of Zone*) and a new exclusion zone will be established based on the results of these field-verified measurements. This new exclusion zone will be based on the field inputs calculating the actual distance from the pile driving source where underwater sound levels are anticipated to equal or exceed 180 dB re 1 microPa rms (impulse). Based on the outcome of the field-verified sound levels and the calculated or measured distances as noted above, the applicant can either: (1) retain the 750 m zone or (2) establish a new zone based on field-verified measurements demonstrating the distance from the pile driving source where underwater SPLs are anticipated to equal or exceed the received the 180 dB re 1 microPa rms (impulse). Any new exclusion zone radius must be based on the most conservative measurement (i.e., the largest safety zone configuration), include an additional 'buffer' area extending out of the 180 dB zone and be approved by MMS and NMFS before implementing. Once approved, this zone will be used for all subsequent pile driving and will be periodically re-evaluated based on the regular sound monitoring described in the *Field Verification of Exclusion Zone* section described below.

- Field Verification of Exclusion Zone: Field verification of the exclusion zone will take during pile driving of the first three piles. The results of the measurements from the first three piles can then be used to establish a new exclusion zone which is greater than or less than the 750 m depending on the results of the field tests.

Acoustic measurements will take place during the driving of the last half (deepest pile segment) for any given open-water pile. One reference location will be established at a distance of 100 m (328 ft) from the pile driving. Sound measurements will be taken

4 Underwater sound pressure levels measured during impact pile driving to install the monopiles for the Utgrunden Wind Park in Sweden were used to derive the pile driving root mean square (RMS) sound level for the Cape Wind Project because the size of the monopiles and the installation techniques are similar. The RMS sound pressure level at 500 meters is 177.8 dB re 1 µPa for Utgrunden. The monopile diameters for the Cape Wind project, 5.1 to 5.5 meters, are slightly larger than monopiles for Utgrunden, and the cross-sectional area is 60 percent larger. Assuming pile driver blow energy (E) scales by the cross-sectional area and impulse noise is proportional to $10*\log(E_2/E_1)$ when blow energy increases from E_1 to E_2, the RMS sound pressure level for Cape Wind scales up to 179.8 dB re 1 µPa at 500 meters averaged over a 125-millisecond pulse duration. The SEL for Cape Wind also scales up in the same manner to 173 dB re 1 µPa at 500 meters. A recent COWRIE report suggests underwater SEL values of 171-173 dB re 1 µPa at 500 meters for piles with diameters equal to those proposed for Cape Wind (Nehls et al., 2007). Thus, the sound source data for Cape Wind are validated by recent COWRIE data at other wind farms. In order to apply an initial exclusion zone size that conservatively allows for an area that will avoid potential Level A harassment of marine mammals, MMS has established a preliminary 750-m zone. However, the applicant has the option to conduct field verification of this zone, as noted above, and change the size of the zone based on these measurements.

at the reference location at two depths (a depth near the mid-water column and a depth near the bottom of the water column but at least 1 m (3 ft) above the bottom) during the driving of the last half (deepest pile segment) for any given pile. Two additional in-water spot measurements will be conducted at appropriate depths (near mid water column), generally 500 m (1,640 ft) and 750 m (2,461 ft) in two directions either west, east, south or north of the pile driving site. These will be conducted at the same two depths as the reference location measurements. In cases where such measurements cannot be obtained due to obstruction by land mass, structures or navigational hazards, measurements will be conducted at alternate spot measurement locations. Measurements will be made at other locations either nearer or farther as necessary to establish the approximate distance for the zones. Each measuring system shall consist of a hydrophone with an appropriate signal conditioning connected to a sound level meter and an instrument grade digital audiotape recorder (DAT). Overall SPLs shall be measured and reported in the field in dB re 1 micro-Pa rms (impulse). An infrared range finder will be used to determine distance from the monitoring location to the pile. The recorded data will be analyzed to determine the amplitude, time history and frequency content of the impulse.

- Visual Monitoring of Exclusion Zone: Visual monitoring of the exclusion zone will be conducted during driving of all piles. Monitoring of the zones will be conducted by one qualified NMFS approved observer[5]. Multiple monitors will be required if pile driving is occurring at multiple locations at the same time.

  Observer(s) will begin monitoring at least 30 minutes prior to soft start of the pile driving. Pile driving will not begin until the zone is clear of all listed whales and sea turtles for at least 30 minutes. Monitoring will continue through the pile driving period and end approximately 30 minutes after pile driving is completed.

  Visual observations will be made using binoculars or other suitable equipment during daylight hours. Data on all observations will be recorded based on standard marine mammal observer collection data. This will include: dates and locations of construction operations; time of observation, location and weather; details of marine mammal sightings (e.g., species, numbers, behavior); and details of any observed taking (behavioral disturbances or injury/mortality). Any significant observations concerning impacts on listed whales or sea turtles will be transmitted to NMFS and MMS within 48 hours. Any observed takes of listed whales or sea turtles resulting in injury or mortality will be immediately reported to NMFS and MMS.

[5] Observer qualifications will include direct field experience on a marine mammal/sea turtle observation vessel and/or aerial surveys in the Atlantic Ocean/Gulf of Mexico. All observers will receive NMFS-approved marine mammal observer training and be approved in advance by NMFS after a review of their qualifications.

- Required Mitigation Should Listed Whales or Sea Turtles Enter the Exclusion Zone: The exclusion zone around the pile driving activity must be monitored for the presence of listed whales or sea turtles before, during and after any pile driving activity. The exclusion zone will be monitored for 30 minutes prior to the soft start of pile driving. If the safety radius is obscured by fog or poor lighting conditions, pile driving will not be initiated until the entire safety radius is visible for the 30 minute period. If listed whales or sea turtles are observed within the zone during the 30 minute period and before the soft start begins, pile driving of the segment will be delayed until they move out of the area and until at least an additional 30 minutes have passed without a listed whale or sea turtle sighting. Monitoring of the zone will continue for 30 minutes following completion of the pile driving activity.

  MMS recognizes that once the pile driving of a segment begins it cannot be stopped until that segment has reached its predetermined depth due to the nature of the sediments underlying the Sound. If pile driving stops and then resumes, it would potentially have to occur for a longer time and at increased energy levels. In sum, this would simply amplify impacts to listed whales and sea turtles, as they would endure potentially higher SPLs for longer periods of time. Pile segment lengths and wall thickness have been specially designed so that when work is stopped between segments (but not during a single segment), the pile tip is never resting in highly resistant sediment layers. Therefore, because of this operational situation, if listed whales or sea turtles enter the zone after pile driving of a segment has begun, pile driving will continue and observers will monitor and record listed whale and sea turtle numbers and behavior. However, if pile driving of a segment ceases for 30 minutes or more and a listed whale or sea turtle is sighted within the designated zone prior to commencement of pile driving, the observer(s) must notify the Resident Engineer (or other authorized individual) that an additional 30 minute visual and acoustic observation period will be completed, as described above, before restarting pile driving activities.

  In addition, pile driving may not be started during night hours or when the safety radius can not be adequately monitored (i.e., obscured by fog, inclement weather, poor lighting conditions) unless the applicant implements an alternative monitoring method that is agreed to by MMS and NMFS. However, if a soft start has been initiated before dark or the onset of inclement weather, the pile driving of that segment may continue through these periods. Once that pile has been driven, the pile driving of the next segment cannot begin until the exclusion zone can be visually or otherwise monitored.

- Implementation of Soft Start: A "soft start" will be required at the beginning of each pile installation in order to provide additional protection to listed whales and sea turtles near the project area by allowing them to vacate the area prior to the commencement of pile driving activities. The soft start requires an initial set of 3 strikes from the impact hammer at 40 percent energy with a one minute waiting period between subsequent 3-strike sets. If listed whales or sea turtles are sighted within the exclusion zone prior to pile-driving, or during the soft start, the Resident Engineer (or other authorized individual) will delay pile-driving until the animal has moved outside the exclusion zone.

- Compliance with Equipment Noise Standards: All construction equipment will comply as much as possible with applicable equipment noise standards of the U.S. Environmental Protection Agency, and all construction equipment will have noise control devices no less effective than those provided on the original equipment.

- *Reporting for Construction Activities*: The following reports must be submitted during construction:

  - Prior to any re-establishment of the exclusion zone, a report must be provided to MMS and NMFS detailing the field verification measurements and proposal for the new exclusion zone. This includes information, such as: a fuller account of the levels, durations, and spectral characteristics of the impact and vibratory pile driving sounds; and the peak, rms, and energy levels of the sound pulses and their durations as a function of distance, water depth, and tidal cycle. Any new zone may not be implemented until MMS and NMFS have reviewed and approved any changes.

  - Weekly status reports will be provided to MMS and NMFS that include a summary of the previous week's monitoring activities and an estimate of the number of listed whales and sea turtles that may have been taken as a result of pile driving activities. These reports will include information, such as: dates and locations of construction operations, details of listed whale or sea turtle sightings (dates, times, locations, activities, associated construction activities), and estimates of the amount and nature of listed whale or sea turtle takings. NMFS and MMS may reduce or increase the frequency of this reporting throughout the time period of pile driving activities dependent upon the outcome of these initial weekly reports.

  - Any observed injury or mortality to a listed whale or sea turtle must be reported to NMFS and MMS within 24 hours of observation. Any significant observations concerning impacts on listed whales or sea turtles will be transmitted to NMFS and MMS within 48 hours.

  - A final technical report within 120 days after completion of the pile driving and construction activities will be provided to MMS and NMFS that provides full documentation of methods and monitoring protocols, summarizes the data recorded during monitoring, estimates the number of listed whales and sea turtles that may have been taken during construction activities, and provides an interpretation of the results and effectiveness of all monitoring tasks.

- *Requirements for Cable Laying*: The following measures will be implemented during the conduct of cable laying activities:
    - The applicant must contact NMFS and MMS within 24-hours of the commencement of jet plowing activities and again within 24-hours of the completion of the activity.
    - All interactions with listed whales or sea turtles during cable laying activities must be reported to NMFS and MMS within 24 hours.
    - A final report must be submitted to NMFS and MMS within 60 days of completing cable laying activities which summarizes the results and any takes of listed species.

8.1.4 Requirements During Operation/Maintenance

Nedwell *et al.* (In press) measured and assessed the underwater noise and potential impacts to marine life during the construction and operations/maintenance phases of four offshore wind parks located in U.K. waters. For the operations/maintenance phase, they concluded that in general the level of underwater noise from the operation of a wind facility was very low and not above ambient levels even in close proximity to the turbines. Therefore, the underwater noise from the operation of offshore wind farms was unlikely to result in any behavioral response for the marine mammals and fish assessed in this study.

Given these results, the main mitigation required for the operations/maintenance phase of the proposed project, including standard and major repairs, inspections, etc. of the turbines, submarine cable and ESP, will include the vessel and aircraft measures outlined in section 8.1.1 of this BA. Section 2.4 of the DEIS outlines the anticipated vessel activity during the operations/maintenance phase of the proposal.

A yearly status report will also be provided to MMS that includes a summary of the year's operation and maintenance activities. In addition, any observed injury or mortality to a listed whale or sea turtle must be reported to NMFS and MMS within 24 hours of observation. Any significant observations concerning impacts on listed whales or sea turtles will be transmitted to NMFS and MMS within 48 hours.

8.1.5 Requirements During Decommissioning

Section 2.5.3 of the BA and Section 2.5.1 of the DEIS contain detail on the proposed methodology for decommissioning and removal of the wind turbines. Essentially, the decommissioning process is the reverse of the construction process (absent pile driving), and the impacts from decommissioning would likely mirror those of construction. In addition, vessel activity during decommissioning would be essentially the same as that required during construction. Therefore, the vessel and aircraft mitigation measures outlined in section 8.1.1 of this BA will be required.

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