

Appendix 5.5-A

Marine Biological Assessment
for the Cape Wind Project

**MARINE BIOLOGICAL ASSESSMENT
FOR THE CAPE WIND PROJECT**

NANTUCKET SOUND

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Prepared for the U.S. Army Corps of Engineers

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1 INTRODUCTION

1.1 Endangered Species Act Regulations

This Biological Assessment was prepared in accordance with Section 7 of the Endangered Species Act (ESA) of 1973 (ESA, P.L. 93-205) which requires that all Federal agencies ensure that any action they authorize, fund, or execute will not jeopardize the continued existence of any endangered or threatened species (*i.e.*, listed species) or result in the destruction or adverse modification of any critical habitat of such species. The “action” under consideration is the construction, operation, and maintenance of an offshore wind energy project, proposed by Cape Wind Associates, LLC (the “Applicant” or “Cape Wind”), and consists of the installation and operation of 130 Wind Turbine Generators (WTGs) and associated equipment on Horseshoe Shoal in Nantucket Sound along with a submarine electric transmission cable system that connects the power to the mainland electric grid.

Because the action will occur in federal marine waters, the ESA mandates that the federal agency authorizing, funding, or carrying out the action (*i.e.*, U.S. Army Corps of Engineers (USACE)) consult with the Department of Commerce, National Marine Fisheries Service (NMFS). Consultation with the Secretary of Interior, U.S. Fish and Wildlife Service (USFWS) is required if any birds or other non-marine endangered or threatened species may be affected by the action. This consultation includes preparation of a Biological Assessment (BA) to determine if the proposed action is likely to result in adverse effects to threatened or endangered marine species. The BA for marine species is presented herein and the BA for avian species is presented in Appendix 5.7-H. Accordingly, USACE is consulting with NMFS and USFWS to ensure that the proposed action does not “...jeopardize the continued existence of endangered or threatened species or result in the destruction or adverse modification of the critical habitat of such species” (50 CFR Part 402). Letters documenting the consultation process are included as Attachment A to this Appendix.

1.2 Species Considered

This BA was developed to determine if the proposed Project is likely to have adverse effects on federally-listed threatened and endangered species of marine animals that may reside in or visit the Project Area and the barge routes used in project construction, maintenance, and decommissioning. In accordance with NMFS recommendations and the USACE scope, three species of endangered great whales (humpback, fin, and right whales) and three species of threatened or endangered sea turtles (loggerhead, Kemp’s Ridley, and leatherback turtles) are discussed in this BA. These species are permanent or seasonal residents of coastal and ocean waters of the western North Atlantic, including Nantucket Sound, and may visit Nantucket Sound seasonally to feed. It should be noted that the BA for the two listed avian species in the Project Area, the piping plover and roseate tern, is presented in Section 5.7 and Appendix 5.7-H of the DEIS.

1.3 Format of the Biological Assessment

A summary of the proposed action and site alternatives is included in Section 2 of this BA. Section 3 describes the affected physical and biological environments. Section 4 presents life history information for each species considered, including population status and trends, seasonal distribution in North Atlantic waters, preferred food and feeding behaviors, and known disturbances and mortality factors. Potential impacts of the proposed action on the listed species, and the management practices to minimize those impacts are discussed in Section 5. Literature cited in this document and other relevant references are presented in Section 6.

2 PROPOSED ACTION AND ALTERNATIVES

2.1 Description of Proposed Action

The offshore wind energy project proposed by Cape Wind consists of the installation and operation of 130 Wind Turbine Generators (WTGs) on Horseshoe Shoal (Alternative #1) in Nantucket Sound. Two additional sites within Nantucket Sound have been evaluated as alternatives for construction of the WTG array. All three sites, depicted in Figure 1, are located outside of the Massachusetts’ three-mile state jurisdictional limit, exclusively within federal waters of Nantucket Sound.

The WTGs will produce an average of 170 megawatts (MW) (up to a maximum output of 454 MW) of clean renewable energy using the natural wind resources off the coast of Massachusetts. Wind-generated energy

produced by the WTGs will be transmitted via a 33 kV submarine transmission cable system (inner-array cables) to the Electric Service Platform (ESP) centrally located within the WTG array. The ESP will then take the wind-generated energy from each of the WTGs and transform and transmit this electric power to the mainland electric transmission system via two 115 kV alternating current (AC) submarine cable circuits (submarine cable system) to the selected landfall site at New Hampshire Avenue in Yarmouth, Massachusetts. The submarine cable system will then interconnect via horizontal directional drilling (HDD) with the upland cable system. The upland cable system will be installed underground within existing rights of way (ROWs) and roadways in the Towns of Yarmouth and Barnstable, where it will interconnect with the existing NSTAR Electric Barnstable Switching Station. The energy produced by the Wind Park will be transmitted by this cable system to the electric transmission system serving Cape Cod, the Islands of Nantucket and Martha's Vineyard ("the Islands"), and the New England region.

Construction and operation of the Project will not preclude or prohibit traditional uses of the water-sheet area within or around the turbine array. Use of the water sheet area within the turbine array that could still continue during operation include general commercial and recreational navigation, commercial and recreational aviation, commercial and recreational fishing, and other traditional water-based activities that promote the use and enjoyment of this area of Nantucket Sound.

2.2 Alternatives to the Proposed Action

The Applicant has conducted a thorough analysis of alternative technologies and site locations for the Project (see Section 3.0 of the DEIS), considering both terrestrial and offshore locations throughout New England. The alternative analysis determined that Nantucket Sound is the only technically and economically feasible environment for installation of an offshore wind park based on the application of preliminary screening criteria. Additional siting analysis was then conducted to evaluate specific locations within the Sound. Nantucket Sound alternative WTG array sites (Figure 1) include Horseshoe Shoal (Site 1 – the Proposed Alternative Site), eastern Nantucket Sound near the Monomoy Island area (Site 2 – Monomoy-Handkerchief Shoal), and southern Nantucket Sound near the Hawes and Tuckernuck Shoals area (Site 3 – Tuckernuck Shoal)

3 AFFECTED ENVIRONMENT

Environmental variables such as benthic topography (Evans 1975, Hui 1979), fronts and mixing regimes (Volkov and Moroz 1977), sea surface temperature (Au and Perryman 1985), and sea salinity (Thomson *et al.* 1986) may be related to the distribution of cetaceans and sea turtles. This section describes the physical and biological characteristics in Nantucket Sound in general and in the proposed Alternative Site and other alternative sites in Nantucket Sound when possible. Much of the information presented in this section is found in the DEIS/DEIR/DRI prepared by the USACE pursuant to the National Environmental Policy Act (NEPA). Readers are encouraged to review referenced literature (Section 6 of this BA) and the DEIS/DEIR/DRI for more detailed information on the affected environment of Nantucket Sound and potential environmental consequences of the proposed action.

3.1 Physical Environment

This section describes the physical environment of Nantucket Sound, and includes subsections on hydrography, currents, salinity, temperature, sediment distribution, sediment quality, and sediment transport. Information is drawn from published literature and from studies conducted by the Applicant. The following description of the physical environment of Nantucket Sound provides a basis for understanding the oceanographic processes that affect the habitat potentially used by endangered and threatened species.

3.1.1 Hydrography

In general, the bathymetry in Nantucket Sound is irregular, with a large number of shoals present in various locations throughout the glacially formed basin. Charted water depths in the Sound range between one and 70 feet at mean lower low water (MLLW). Each of the alternatives sites is located in a shoal area within Nantucket Sound. The shoals have complex shapes, as shown in Figure 1. Water depths on each shoal range from less than 10 feet deep to more than 50 feet deep at low tide. The Proposed Alternative Site (Site 1) is located on Horseshoe Shoal, a prominent geological feature in the center of the Sound. Depths on Horseshoe Shoal are as shallow as 0.5 feet at MLLW. Site 2 is located on Monomoy-Handkerchief Shoal, in the eastern part of Nantucket

Sound west of Monomoy Island. Monomoy-Handkerchief Shoal has an extensive area of shallows averaging 6 to 8 feet deep. Site 3 is located on Tuckernuck Shoal, in the southern portion of Nantucket Sound, northwest of Nantucket and Muskeget Islands and east of the opening between Nantucket and Martha's Vineyard. Localized areas on the crest of Tuckernuck Shoal are as shallow as 2 feet.

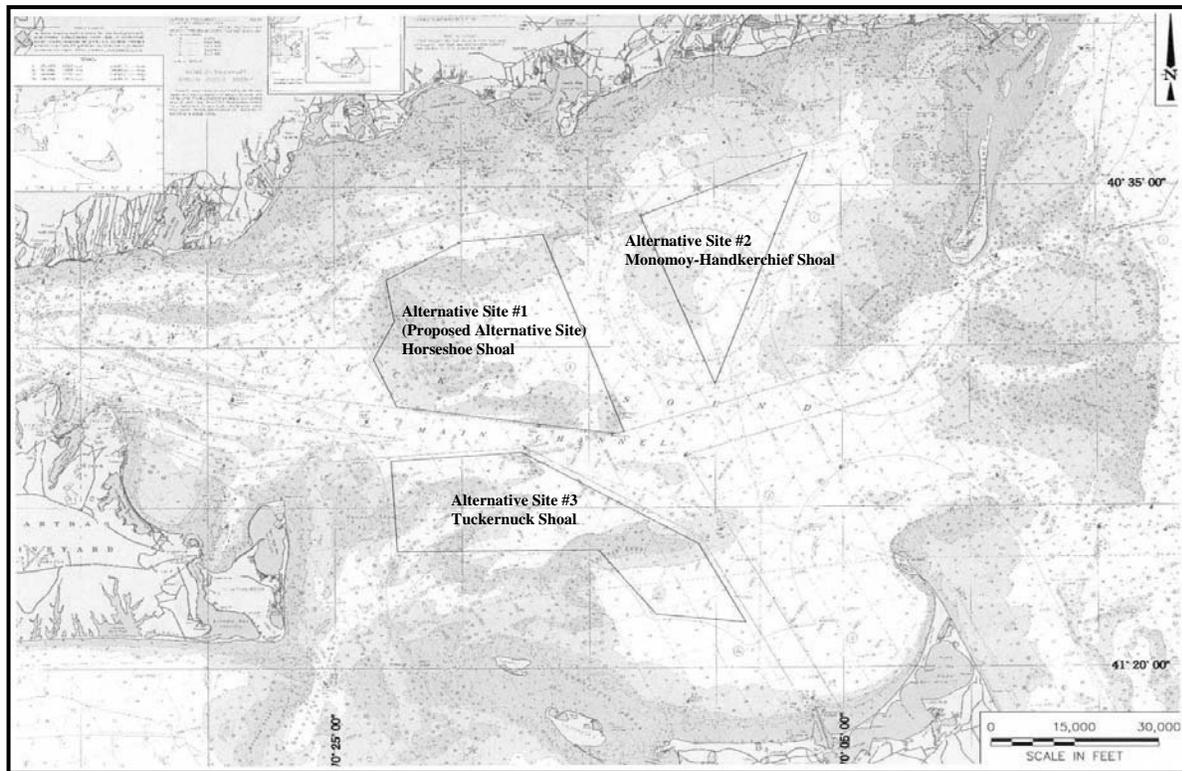


Figure 1. Location of the Proposed and alternative wind park sites in Nantucket Sound

Water depths between Horseshoe Shoal and the Cape Cod shoreline are variable, with an average depth of approximately 15 to 20 feet at MLLW. Along the submarine cable system route, depths vary from about 16 to 40 feet at MLLW, with an average depth of approximately 30 feet at MLLW. Water depths in Lewis Bay and Hyannis Harbor are variable ranging from eight to 14 feet at MLLW in the center of the Bay to less than five feet at MLLW along the perimeter and between Dunbar Point and Great Island. There are three navigation channels in Lewis Bay: the Federal Navigation Channel providing access to Hyannis Inner Harbor (authorized depth –13 feet MLLW); and two privately maintained channels, one into Mill Creek (reported depth of –two feet MLLW in 1983) and the other northeast of Great and Pine Islands (approximately seven feet deep at MLLW).

The submarine cable system route will extend outside the eastern edge of the federal channel into Lewis Bay and will then turn east, north of Egg Island, to make landfall between Mill Creek and the privately maintained channel northeast of Great and Pine Islands. Water depths along this route in Lewis Bay range from five to 15 feet, with an average of ten feet. The shallowest portions of Lewis Bay/Hyannis Harbor along this route exist between Great Island and Dunbar Point, with depths of one to four feet at MLLW.

3.1.2 Oceanographic Conditions

3.1.2.1 Currents

The water currents in Nantucket Sound are driven by strong, reversing, semidiurnal tidal flows. Wind-driven currents are only moderate because of the sheltering effect of Nantucket and Martha's Vineyard. The tidal range and diurnal timing are variable because of the semi-enclosed nature of the Sound and the regional variations in bathymetry. Typical tidal heights are in the range of one to four feet with tidal surges of up to approximately ten feet having been recorded during hurricanes (Bumpus *et al.* 1973; Gordon and Spaulding 1979). Times of high and low tides vary in different parts of the Sound by up to two hours.

Tidal flow and circulation within the Sound generate complex currents, the direction of which form an ellipse during the two tidal cycles each day. The complex bathymetry of Nantucket Sound forces the tidal ellipses to take different shapes in different regions of the Sound. Just off the coast of the south shore of Cape Cod, there is a strong rectilinear, semi-diurnal tidal flow approximately parallel to the coast (Goud and Aubrey 1985). The tidal current flows to the east during the flood tide (incoming) and to the west during the ebb tide (outgoing). Peak tidal currents often exceed two knots (Bumpus *et al.* 1973). The intensity of tidal flow, in general, decreases from west to east. There is a slow net drift of the water mass toward the east in the Sound. The net drift is about 200m² per tidal cycle, roughly 5% of the total easterly and westerly tidal flows (Bumpus *et al.* 1971).

To characterize site-specific tidal and wind-driven currents at the Proposed and alternative sites in Nantucket Sound, analytical models were applied (Appendix 5.2-A), with the results as follows. Flood currents on the shoals are generally directed easterly and ebb currents are generally directed westerly. Local changes in tidal current direction occur on the shoals due to the nearby shoreline shape and bathymetric features. For example, the direction of tidal currents at Handkerchief Shoal is directed around Monomoy Island and has more of a southeast (flood)/northwest (ebb) tendency. Currents at Horseshoe Shoal are diverted slightly around the shallowest portion of the shoal. Flood currents also are generally stronger than ebb currents and spring tidal currents are approximately 15-20 percent stronger than mean tidal currents. Mean tidal current velocities were calculated to be approximately 2 feet/second at Horseshoe Shoal; less than 2 feet/second at Tuckernuck Shoal, and more than 2.5 feet/second at Monomoy-Handkerchief Shoal. Wind-driven current velocities modeled at Horseshoe Shoal were found to be much lower than tidal velocities and concentrated over the crest of the shoal (Appendix 5.2-A).

3.1.2.2 Salinity

Salinities in Nantucket Sound are near oceanic and salinity gradients are small due to strong lateral and vertical mixing. River runoff into Nantucket Sound is low, so there is little dilution of ocean waters with fresh water. Surface and bottom water salinities vary seasonally and spatially from about 30 to 32.5 ppt (Bumpus *et al.* 1973). Surface water salinities throughout the Sound are just over 31 ppt during the summer, and are uniformly about 32 ppt in the winter (Limeburger *et al.* 1980).

3.1.2.3 Temperature

The annual cycle of surface and bottom water temperatures in Nantucket Sound encompasses a range of about 45°F, from nearly 30°F (-1° C) in the winter to as high as 75°F (24° C) in the late summer (Bumpus *et al.* 1973). Temperature extremes are greatest in coastal ponds and estuaries and the seasonal temperature cycle is smallest in the deeper parts of the Sound. However, because the Sound is shallow and well mixed, there is little lateral temperature variation and vertical temperature stratification. There is a tendency in the summer for surface water temperature to increase from east to west in Nantucket Sound. In the winter, the gradient is in the opposite direction (Limeburger *et al.* 1980). This change is caused by the intrusion of warmer continental shelf water into the Sound from the east during the summer months.

Bottom water temperature varies less and changes more slowly on a seasonal basis than surface water temperature. The highest bottom water temperature in Nantucket Sound during summer is in the range of 61 to 66°F (16 to 19° C) (Theroux and Wigley 1998). Warmest bottom water temperatures are near the coast of the south shore of Cape Cod, and temperature decreases with distance offshore. Coolest bottom water temperatures in Nantucket Sound are in the range of 32 to 35.6°F (0 to 2° C), and become warmer with distance from the Cape Cod and Nantucket shorelines.

3.1.3 Sediment Distribution, Quality, and Transport

3.1.3.1 Sediment Distribution

Nantucket Sound generally contains sand- and silt-sized surficial marine sediments, with localized patches of clay, gravel and/or cobbles. The sediments were derived from material originally transported from upland areas during glacial and post-glacial processes, and are now continually sorted and reworked by tidal, current, wave and storm actions. Shallow marine sediments were collected in vibracores and benthic grabs during 2001 and 2002 across the Proposed and alternative sites. Visual analysis of sediments within the 0- to 2-foot depth range beneath the seabed indicates the presence of fine- to coarse-grained sands in areas of relatively shallow bathymetry, with fine to silty sands and silts predominating in deeper surrounding waters across the three sites. This distribution is

consistent with the higher-energy marine environments typically found in shallower waters, where finer sediments are winnowed away by current and wave action. The fines then settle out and deposit in the surrounding lower-energy deeper water areas.

Medium-grained sands predominate atop the U-shaped Horseshoe Shoal, with fine-grained sands found in the east-opening embayment. Localized fractions of silt, gravel and/or cobbles, consistent with glacial drift may also be present in the area. Fine to silty sands were encountered in the deeper water portions surrounding the shoal area. Fine sands predominate in the western and central portions of Monomoy-Handkerchief Shoal, with silty sands to the east in deeper waters. Across Tuckernuck Shoal, fine sands predominate, with an area of medium to coarse sands traversing the center of the shoal and oriented parallel to the tidal currents sweeping between Martha's Vineyard and Nantucket. Silty sands were encountered to the east of Tuckernuck Shoal, again in the deeper water areas surrounding the shoal.

A geophysical survey across Horseshoe Shoal conducted in 2001 identified areas of sand waves, especially in the south central portion of the shoal. The sand wave crests were oriented generally in a north-south direction, with long period wavelengths ranging between 100 to 600 feet. Short period sand waves are located between the larger crests. The average sand wave height was 4 to 5 feet, but waves as high as 15 feet were found. The size of the sand waves attest to the dynamic shallow water environment on Horseshoe Shoal. The symmetry of the sand waves indicates migration to the east or west, depending on where they formed on the shoal. In other areas of the shoal, the majority of the seafloor contained few significant features and smooth sandy bottoms (Ocean Surveys, Inc., July 2002).

Sand waves were also identified within the Tuckernuck Shoal area (as well as across Horseshoe Shoal) during a geophysical survey conducted by USGS in 1976 and 1977. Sand waves were not identified by USGS at that time across what is now the Monomoy-Handkerchief Shoal alternative site (O'Hara and Oldale, 1987). The geophysical survey conducted in 2001 did not include those two alternative site areas.

Along the submarine cable system route, seabed sediments contain fine to coarse size sands, with patches of clay, silt, gravel and/or cobbles. Intermittent glacially transported boulders may also be present along the route.

3.1.3.2 Sediment Quality

Bulk chemical analyses were performed on selected core samples obtained from the WTG array area and along the proposed submarine cable route into Lewis Bay to determine whether the sediments could pose an environmental concern. To assess the relative environmental quality of these sediments, the analytical laboratory results for the targeted chemical constituents were compared to sediment guidelines typically used by agencies to evaluate risk from contaminants in marine and estuarine sediments (Effects Range-Low (ER-L) and Effects Range-Median (ER-M) guidelines). None of the targeted chemical constituents were detected in the samples above ER-L or ER-M guidelines (Long et al., 1995) for marine sediments. The ER-L and ER-M guidelines use numerous modeling, laboratory, and field studies to establish values for evaluating marine and estuarine sediments. Concentrations below the ER-L represent a concentration range in which adverse effects are rarely observed. Section 5.1 of the DEIS has more detailed information on sediment quality in the Project Area.

3.1.3.3 Sediment Transport

Analytical sediment transport modeling was performed to determine the extent to which existing wave and current conditions are likely to lift and move sand at the Proposed and alternative project sites (see Appendix 5.2-A of the DEIS).

Generally the analysis found that active sediment transport occurs at all of the shoals, even under typical wave and tidal current conditions. The highest sediment transport rates are focused locally on the shallowest portions of the shoals, and there is relatively little sediment transport in the deeper regions for typical conditions. The most dynamic transport conditions are shown to be on Monomoy-Handkerchief Shoal. This is expected due to the extensive shallow flats in this area, relatively swift tidal currents that funnel at this location between the Sound and Ocean, open western exposure to waves generated within the Sound, and relatively fine sediment grain size at this location. Although Tuckernuck Shoal experiences the lowest tidal currents, the potential sediment transport rate for typical conditions is on the order of Horseshoe Shoal due to the fine grain size of sediments at Tuckernuck Shoal.

Bed load transport on Horseshoe Shoal is typically an order of magnitude greater than suspended load transport. This is expected at the Horseshoe Shoal site, where sediments are relatively coarse. It is also expected since the level of wave and current energy under typical conditions is not sufficient to lift and suspend large volumes of sediment within the water column.

At all sites, spring tidal currents initiate approximately 20 percent more transport than mean tidal currents, and wind-driven currents from a sustained 15 knot westerly wind have a similar effect by comparison. The greatest impact on sediment transport initiation is due to waves. Larger locally generated waves within Nantucket Sound can cause a significant increase in sediment transport. If swell waves from the ocean impact the Proposed or alternative project sites, sediment transport rates can increase as much as one hundred fold, even for typical swells propagating from the Atlantic Ocean (e.g., four to five foot height with an eight second period). Since flood currents are stronger than ebb currents, there is a long-term forcing mechanism to cause the net transport of sediment to the east, particularly at Horseshoe Shoal.

3.2 Biological Environment

This section describes the biological environment of Nantucket Sound, and includes subsections on submerged aquatic vegetation, the plankton community, benthic communities, and fish. Information was drawn from published literature and from studies conducted by the Applicant. The following description of the biological environment of Nantucket Sound provides a basis for understanding the biological and ecological conditions of this area potentially used by endangered and threatened species.

3.2.1 Submerged Aquatic Vegetation

Seagrass beds and other submerged aquatic vegetation (SAV) provide habitat for many species of benthic invertebrates and fish. The MADEP Wetlands Conservancy Program has mapped SAV beds one quarter acre or larger in size along the coast using aerial photography, GPS, and a digital base map. Mapping was completed in 1995 and 2000; the 1995 data is available from MassGIS. One SAV bed has been mapped within Lewis Bay, located to the west of Egg Island in the Town of Barnstable. A December 2002 telephone conversation with Mr. Charles Costello of the MADEP Wetlands Conservancy Program indicates that the mapped SAV bed has not changed much in size between 1995 and 2000. In addition to the mapped SAV in Lewis Bay, MADEP has mapped areas of SAV in Hyannis Harbor in the Town of Barnstable and to the west of Great Island in the Town of Yarmouth. Field investigations have been conducted to determine the extent of mapped SAV beds in the vicinity of the proposed Project. The submarine cable system will be no closer than 70 feet from the edge of the eelgrass bed located near Egg Island.

3.2.2 Plankton Communities

Plankton refers to those plants (phytoplankton) and animals (zooplankton) that cannot maintain their distribution against the movement of water masses. Individual plankters are generally very small or microscopic; however, organisms such as jellyfish are often considered with the plankton community. Review of the scientific literature suggests that little information exists describing the plankton communities of Nantucket Sound. However, because all coastal and offshore waters contain both phytoplankton and zooplankton communities, it is expected that the waters within Nantucket Sound also support a diverse and abundant plankton community. These planktonic communities are generally variable in time and space resulting in relative patchy distributions.

Various zooplankton species serve as prey for higher trophic level organisms such as fish and whales. Many species of whales found in the northeast Atlantic feed on zooplankton such as copepods and euphausiids during their migrations up and down the coastline. The zooplankton communities within Nantucket Sound are likely to contain these organisms as well as a variety of other planktonic crustaceans such as amphipods and isopods. Fish eggs and larvae resulting from the spawning of local fish populations would also be found within the plankton community of Nantucket Sound. Various jellyfish species are also likely inhabitants of the planktonic community in Nantucket Sound. The leatherback turtle is known to feed on jellyfish and other gelatinous organisms floating in the plankton.

3.2.3 Benthic Communities

Based on a literature reviewed, the most abundant benthic fauna taxa in Nantucket Sound are crustaceans and mollusks, followed by polychaete worms (annelids) (Sanders, 1956; Wigley, 1968; Pratt, 1973; Theroux and Wigley, 1998). Among the crustaceans, amphipods are reported to be by far the most abundant. Bivalves are reported to be the most abundant and diverse of the mollusks in Nantucket Sound (Pratt, 1973). MDMF (2001) reports that a heavily populated area of northern quahog (*Mercenaria mercenaria*) exists in the shoals east of Horseshoe Shoal. The annelid fauna is also reported to be diverse (Theroux and Wigley, 1998). Maurer and Leathem (1981) identified 333 species of polychaete worms in sandy sediments from Georges Bank and Nantucket Shoals. Many of these species occur in the deeper waters of Nantucket Sound. Biomass is reported to be lower in shallow areas of Nantucket Sound, including the Proposed Alternative Site (Theroux and Wigley, 1998). This is most likely due to the unstable sandy sediments in these shallow waters. These polychaetes are a favorite prey of several species of demersal fish, particularly winter flounder (*Pseudopleuronectes americanus*) (Buckley, 1989).

Based on the benthic survey conducted in the late summer of 2001, ninety-five taxa were identified in the benthic grab samples collected for the Project from the Horseshoe Shoal area. Consistent with previous research, the most diverse and abundant taxonomic class found was *Amphipoda* (amphipods, or scuds). Benthic surveys conducted during late spring of 2002 also found that amphipods were a dominant group, however, abundances observed during late spring 2002 were significantly lower ($p < 0.10$) than those observed during late summer of 2001. The late spring 2002 survey was conducted to assess Horseshoe Shoal as well as two alternative areas, Tuckernuck Shoal and Monomoy Shoal. The most dominant taxon found during 2002 was Nematoda, followed by Ampeliscidae (four-eyed amphipod). With regard to differences among the three alternative sites, benthic diversity was found to be significantly higher ($p < 0.10$) on Monomoy Shoal than on Tuckernuck Shoal while no significant difference ($p > 0.10$) was found between the benthic diversity of Horseshoe Shoal and either of the two other alternative areas assessed. Benthic organism abundance did not differ significantly ($p > 0.10$) among the three alternative sites.

Differences in benthic organism abundance and community composition were expected to be related to differences in bottom substrate type, water depth or possibly due to the presence or absence of sand wave formations (unstable, shifting sediment). These physical habitat features were assessed during the 2002 study. In general, results indicated that benthic diversity was significantly higher ($p < 0.10$) in shallow waters characterized by fine-grained sediments and absent of sand waves. Organism abundance was found to be much less dependent on depth and sediment type. Abundances were generally found to be significantly lower ($p < 0.10$) in areas with sand waves. Overall, the benthic community composition and relative abundance documented as part of the 2001 and 2002 studies (Appendices 5.3A and 5.3B) was consistent with data reported in earlier studies on Nantucket Sound, Georges Bank, and the Southern New England Shelf (Sanders, 1956; Wigley, 1968; Pratt, 1973; Theroux and Wigley, 1998). The sandy substrate of Nantucket Sound is dynamic and mobile, as is indicated by ripple marks and sand waves. The magnitude and frequency of sand movements has a marked influence on the composition and abundance of the benthic communities. Organisms living on or in these sandy sediments are expected to be adapted for movement or settlement in sand and recovery from burial. Section 5.3 of the DEIS has more detailed information on benthic communities in Nantucket Sound.

3.2.4 Finfish

The waters of Nantucket Sound support a diverse fish community. Many of the fish found within the region are local inhabitants that remain year-round, while other species are migratory and move into and out of the Sound in response to temperature changes. The migration patterns of the fish that are prey of several whale species influence the distribution of these whales. Herring, mackerel, capelin and sand lance are small schooling fish that are the primary prey of humpback whales and fin whales. Although not particularly common, all of these species except for capelin, which is a boreal arctic species, have been observed in Nantucket Sound during some time of the year.

Herring (*Clupea harengus*) form large schools in coastal waters throughout the Gulf of Maine and off southern New England (Reid *et al.* 1999). In the summer and fall, juveniles move from nearshore waters to overwinter in deep bays or near bottom in offshore areas (Reid *et al.* 1999). Some juveniles spend at least the spring and early summer off southern New England, especially off southern Massachusetts (through at least mid-June) before moving into the Gulf of Maine or offshore, presumably east of Cape Cod (Reid *et al.* 1999). According to

the Massachusetts inshore trawl surveys (1978-1996), as reported by Reid *et al.* (1999), juveniles in spring were most abundant northwest of Cape Ann, throughout Cape Cod Bay, along the northern shore of Nantucket Island and southern shore of Martha's Vineyard, and Buzzard's Bay. Juveniles were also found to a lesser degree in the northeast corner of Nantucket Sound near Monomoy Island and off the south shore of Dennis, MA. In the fall, the largest catches of juveniles occurred around Cape Ann, in central and western Cape Cod Bay, off Buzzard's Bay, and off the southern shore of Martha's Vineyard. Therefore, herring are found in Nantucket Sound, but data indicate that they are likely more prevalent outside of Nantucket Sound or on the outer reaches of the Sound, outside of the Project Area.

Mackerel (*Scomber scombrus*) have the same geographic distribution as herring, but the migration patterns differ. Juveniles are common in Nantucket Sound from August to November and adults in March, April, and October to December as indicated by the Estuarine Living Marine Resources (ELMR) Program database provided by NOAA. Occurrences of juvenile Atlantic mackerel were highest in the fall and occurrence of adults were highest in the spring (Studholme *et al.* 1999). Yet, based on a Massachusetts coastal zone survey in Studholme *et al.* (1999), juvenile and adult mackerel in Nantucket Sound occur only randomly.

Sand lance or sand eels (*Ammodytes*) occur in estuarine, open coastal, and offshore habitats, generally over sandy substrates. They are habitat dependant and are typically found in areas with high bottom current velocities over sandy substrates. Juveniles and adults are generally found in schools during the day with larger schools found in deep waters, and smaller schools found over shoal habitat (Auster and Stewart, 1986). No specific data on the abundance and distribution of sand lance in Nantucket Sound was found, but the habitat characteristics of Nantucket Sound indicate that sand lance are likely present in the Project Area. Section 5.4 of the DEIS has more detailed information on finfish in Nantucket Sound.

4 PROTECTED SPECIES

This section summarizes the natural history of the three species of endangered whales and three species of threatened or endangered sea turtles considered in this BA.

4.1 The Humpback Whale (*Megaptera novaeangliae*)

Humpback whales (*Megaptera novaeangliae*) occur in all the oceans of the world, except possibly the Arctic (NMFS 1991a). The fifth largest of the baleen whales, humpback whales are approximately four meters long at birth and reach a maximum size of 18 meters and a weight of 48 metric tons (Winn and Reichley 1985). Females are slightly larger than males. The unique feature of humpback whales that distinguishes them from all other baleen whales is their extremely long flippers that may reach five meters in length, or half of the entire body length. Other distinguishing features include well-defined ventral grooves and fleshy protuberances (tubercles) that cover the whale's rostrum. Also, the small, variable shaped dorsal fin acts as an identifying feature. The body of the humpback is generally black in color, but individually distinctive black and white pigment and scar patterns occur on the underside of the broad tail (flukes), the belly, and the pectoral fins. These patterns, along with dorsal fin shape and scarring, are used to identify whales (Katona *et al.* 1980; Katona and Whitehead 1981). Calves also appear to inherit the fluke pigmentation patterns of their mothers (Rosenbaum and Clapham 1993).

Female humpback whales reach sexual maturity after four to six years and thereafter give birth approximately every two or three years, although intervals of one to five years between successive calvings have been observed (Clapham and Mayo 1987; NMFS 1991a; Clapham 1992). Most births take place in the winter in the West Indies. Mothers usually nurse their calves for a year or less (Clapham 1992; Baraff and Weinrich 1993). Weaning may begin when the calves are five to six months old and still in the northern feeding grounds (Clapham and Mayo 1987; Baraff and Weinrich 1993). After weaning their calves, the adult females are ready to mate again. The gestation period is ten to twelve months (NMFS 1991a) and some females have been sighted with newborn calves two or three years in a row (Weinrich *et al.* 1993), suggesting that mating may occur while the female is still nursing. The life span of humpback whales is at least 30 years (Chittleborough 1959).

4.1.1 Population Status and Trends

Humpback whales were an important commercial species throughout most of their range, including New England waters, until early in the twentieth century (Allen 1916). Some taking of humpback whales occurred in northwest Atlantic waters until the mid-1950s. The International Convention for the Regulation of Whaling, Washington

1946, afforded the North Atlantic population of humpback whales full protection in 1955 (Best 1993). Humpback whales were afforded endangered species status in the United States in 1970 (USFWS 1986), and retain that status today. Although severely depleted by whaling, the species has shown good recovery over most of its range.

Western North Atlantic humpbacks belong to four primary feeding aggregations: United States east coast (including the Gulf of Maine), the Gulf of St. Lawrence, Newfoundland/Labrador, and western Greenland (Katona and Beard 1990). Two other North Atlantic feeding grounds have been identified off Iceland and northern Norway (Christensen *et al.* 1992; Palsboll *et al.* 1997). Composition of these feeding aggregations is determined by matrilineal fidelity (Clapham and Mayo 1987). Based on genetic evidence supporting the distinction of the western North Atlantic feeding aggregations (Palsboll *et al.* 2001), the Gulf of Maine is now viewed as a distinct feeding stock for management purposes (Waring *et al.* 2002; IWC 2002).

Humpback whales that are observed with increasing frequency in New England waters each winter may be from the Gulf of Maine, Newfoundland, and St. Lawrence River estuary feeding aggregations. Individuals identified in one feeding aggregation occasionally are observed in another aggregation (Katona and Beard 1990). Whales from the Gulf of Maine aggregation have been observed in the Gulf of St. Lawrence and Newfoundland aggregations. Humpbacks from these three feeding aggregations seem to congregate preferentially in winter breeding and calving areas off the Dominican Republic and eastern Puerto Rico (Katona and Beard 1990) and may follow similar southward migration routes from summer feeding areas to winter breeding areas.

Prior to exploitation, the worldwide population of humpback whales was thought to number more than 125,000 individuals (Braham 1984; NMFS 1991a). Best (1993) reviewed recent sightings data for ten of the eleven putative stocks of humpback whales in the world's oceans and concluded that the oceans of the northern hemisphere and Australia support more than 17,500 humpback whales; data for Southern Ocean waters south of 30° S latitude are less certain. The three Antarctic humpback stocks may contain as many as 20,000 individuals, bringing the current world total to more than 37,000 individuals, representing approximately 30 percent of the pre-exploitation population size.

In 1932, the western North Atlantic population was estimated to contain as few as 700 animals (Breiwick *et al.* 1983), though this may have been an underestimate (Reeves and Mitchell 1986). Three population estimates are available for the Gulf of Maine stock. Mark and recapture methods from 1992 and 1993 yield an estimate of 652 (CV = 0.29) whales (Clapham *et al.* 2001). Photo-identification data from 1997 provide a minimum population estimate of 497 individuals (Waring *et al.* 2001). An estimate of 816 whales is derived from line transect survey data from 1999 (Palka 2000; Clapham *et al.* 2001b); this number increases to 902 whales when exchange between Scotian Shelf and Gulf of Maine populations is considered. Some whales from the St. Lawrence River estuary and Canadian Maritimes (Bay of Fundy and Scotian Shelf) feeding aggregations move through New England waters during their biannual migrations between summer and winter habitats. As reported in the 2002 Marine Mammal Stock Assessment Report (Waring *et al.* 2002), this combined number of 902 whales is considered the best abundance estimate for humpbacks in the Gulf of Maine, with a minimum population estimate of 647 individuals. Currently the best available estimate of the North Atlantic population is 11,570 (coefficient of variation (CV) = 0.069) individuals based on mark and recapture studies in 1992 and 1993 (Stevick *et al.* 2001).

4.1.2 Seasonal Distribution in North Atlantic Waters

The humpback whale is a migratory species, and spends the summer in northern latitude feeding grounds (40° to 75° N latitude) in areas of high productivity (NMFS 1991a). Because of the patchy distribution of their prey, humpback whales must target places where prey abundance is high. Humpbacks are found most often in areas of upwelling, along the edges of banks, and over rapidly changing bathymetry along the continental shelf, and along frontal zones between well-mixed and stratified water masses (Figure 2). An abrupt change in water depth on the shelf may cause upwelling or other oceanographic conditions that favor dense aggregations of near-surface zooplankton and shoaling, plankton-feeding fish upon which the whales feed. Movements of humpback whales along these features probably are controlled by the distribution of their prey (Brodie *et al.* 1978; Gaskin 1982; Kenney and Winn 1986; Dolphin 1987a,b; Mayo *et al.* 1988; Payne *et al.* 1990). Although there appears to be some broad-scale matrilineal feeding site fidelity (Clapham and Mayo 1987; 1990), shifts in summer

distributions of humpbacks along the Newfoundland coast (Whitehead and Carscadden 1985), and in the Gulf of Maine (Payne *et al.* 1986) have occurred in apparent responses to changes in prey abundance.

The seasonal distribution of humpback whales in New England waters suggests that most of the humpbacks sighted are part of the Gulf of Maine feeding aggregation (Wiley *et al.* 1995). Humpback whales regularly visit the southern New England area, where they are present in greatest abundance between June and September (Payne and Heinemann 1990; Sadove and Cardinale 1993). All age classes, including mother/calf pairs, are present during the summer. Smaller numbers, nearly exclusively solitary juveniles, frequently are observed in December and January. One of the primary feeding grounds is Stellwagen Bank. On November 4, 1992, this area was designated as a National Marine Sanctuary under Title III of the Marine Protection, Research, and Sanctuaries Act. Since 1988, a dramatic decline in the use of Stellwagen Bank by adult humpback whales has occurred, apparently due to the decline in sand lance populations in the area, a primary food source (Weinrich *et al.* 1993).

Humpbacks remain in the Gulf of Maine for nearly eight months each year, feeding on the abundant populations of schooling fish and crustaceans (Kenney *et al.* 1981; Kenney and Winn 1986). Individuals or small groups of whales from this feeding aggregation may periodically move southward into southern New England and coastal waters of the middle Atlantic during the feeding season in search of food (Lee and Socci 1989; Wiley *et al.* 1995). Some juvenile humpbacks do not make the migration to the tropical breeding grounds, but instead remain in the north to feed over the winter and congregate along the middle Atlantic coast (Swingle *et al.* 1993). Recent studies report that the composition of mid-Atlantic humpbacks is dominated by whales from the Gulf of Maine (Barco *et al.* 2001), but note that the Canadian whale populations are not thoroughly and recently catalogued and so their identifications among mid-Atlantic sightings may be underrepresented (Waring *et al.* 2002). The mid-Atlantic region may be a supplemental feeding ground with multiple other uses (Barco *et al.* 2001).

The humpback whale population can be divided into eleven to thirteen breeding stocks, each of which winters and reproduces in a different clearly-defined tropical and sub-tropical area worldwide (NMFS 1991a; Best 1993). The western North Atlantic breeding stock winters in the Lesser and Greater Antilles Islands of the eastern Caribbean Sea. During the spring and summer, whales from this stock split into five feeding aggregations that migrate to and feed along the coasts of Iceland, southwestern Greenland, Newfoundland and Labrador, the Gulf of St. Lawrence, and the Gulf of Maine (Payne *et al.* 1986; Katona and Beard 1990; NMFS 1991a). While the endpoints of this migration are well established (Martin *et al.* 1984; Mattila *et al.* 1989; Katona and Beard 1990), the exact route between the summer and wintering grounds is unknown, although it is likely to be well offshore (Clapham and Mattila 1990).

Although the primary feeding grounds for humpback whales are located further offshore from Nantucket Sound, very few whales are sighted within the Sound itself (Figure 2). Most whales are found in areas where their primary food source can be easily located. The bathymetric and oceanographic features identified above that favor dense aggregations of humpback whale prey species are not developed in Nantucket Sound to the extent that they are farther north around Stellwagen Bank, Jeffreys Ledge, Browns and Bacaro Banks, and in the Great South Channel (Kenney and Winn 1986). Therefore, the preferred foods of humpback whales and the whales themselves occur in Nantucket Sound with far less abundance and frequency than in high-use areas farther north.

Allen (1974) documented only two instances of humpback whale sightings within Nantucket Sound between 1757 and 1913. The first was struck by a boat near Yarmouth, MA in 1757 and the other was located in Nantucket Harbor in 1825. Cumulative sightings from 1969 to 1992 did not record any humpback whales within Nantucket Sound. Though no humpback whale surveys have targeted Nantucket Sound specifically, many ship-based and aerial surveys for great whales (including right whales) in North Atlantic waters pass over or through Nantucket Sound, including the CeTAP and POP surveys (Robert Kenney 2002, personal communication). The Center for Coastal Studies Humpback Whale Studies Program is currently reviewing opportunistic humpback sightings data (Jooke Robbins 2002, personal communication). These data will be reviewed for Nantucket Sound sightings when available and will be included in the Final EIR/EIS/DRI. Most references show humpback whales moving from the eastern shores of Long Island Sound, out around the southern shores of Martha's Vineyard and Nantucket towards Nantucket Shoals and the Great South Channel. Therefore, historically and at present, Nantucket Sound does not appear to be an important area for humpback whales.

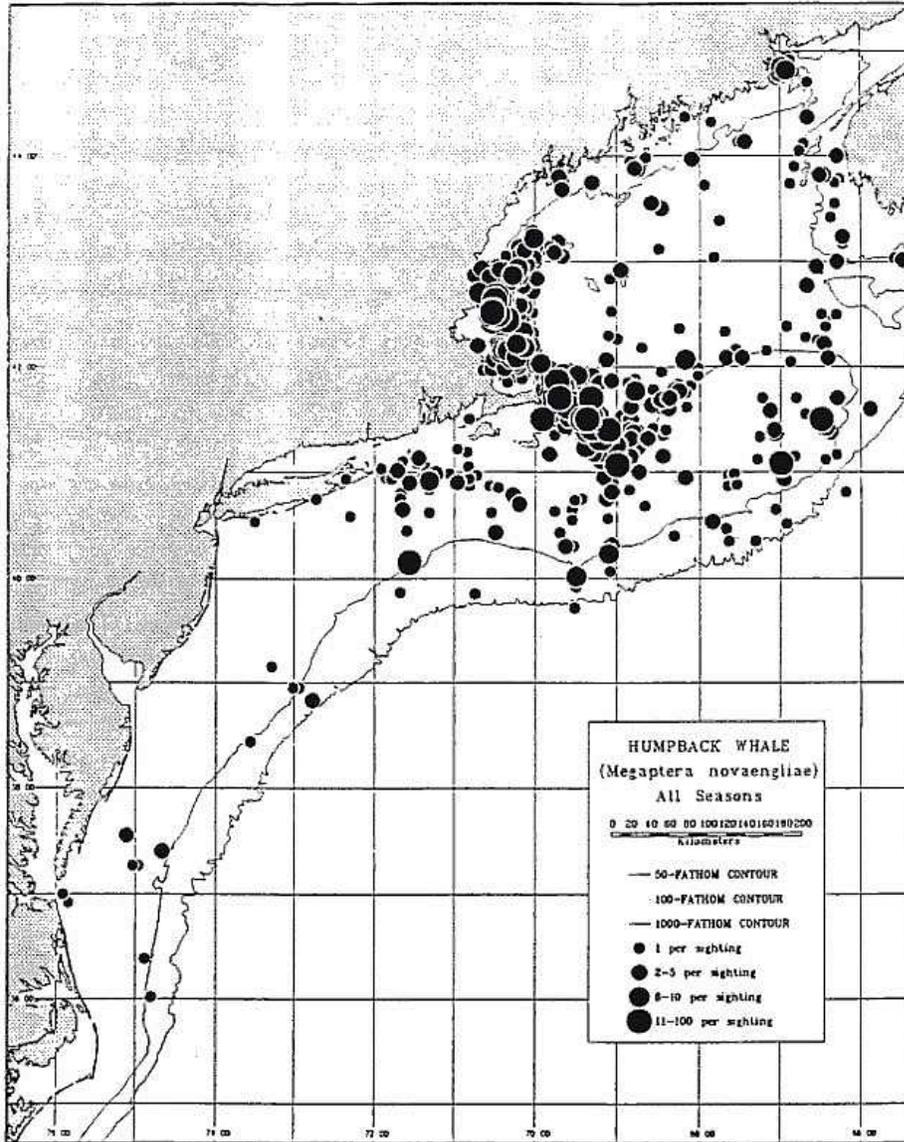


Figure 2. Humpback whale sightings in the Western North Atlantic (NMFS Northeast Fisheries Science Center, unpublished data).

4.1.3 Food and Feeding Behaviors

Humpback whales feed opportunistically on a wide variety of species of pelagic crustaceans and small fish (Nemoto 1971; Kreiger and Wing 1984). Sand lance (*Ammodytes americanus* and *A. dubius*) currently is an important food of humpback whales in the western Gulf of Maine (Hain *et al.* 1982; Payne *et al.* 1986, 1990). Capelin (*Mallotus villosus*) and euphausiids (*Meganyctophanes norvegica*) are preferred foods of humpback whales in more northern U.S. and Canadian waters (Whitehead and Glass 1985). Humpbacks observed south of Cape Cod and off Rhode Island in the spring were feeding on dense patches of euphausiids (Kenney and Winn 1986). During their seasonal northern residency in the area, humpbacks may also feed on several commercially important fish and invertebrates, such as herring (*Clupea harengus*), mackerel (*Scomber scombrus*), menhaden (*Brevoortia tyrannus*), pollock (*Pollachius virens*), small haddock (*Melanogrammus aeglefinus*), and squid (*Illex illecebrosus*) (Overholtz and Nicolas 1979; Meyer *et al.* 1979; Whitehead and Glass 1985; Whitehead 1987; Piatt *et al.* 1989; NMFS 1991a).

Fine-scale movements of humpback whales are probably controlled by the distribution of their prey (Brodie *et al.* 1978; Gaskin 1982; Kenney and Winn 1986; Dolphin 1987a,b; Mayo *et al.* 1988; Payne *et al.* 1990). Apparent

declines in the abundance of humpback whales feeding on Georges Bank may be related in part to declines in the local abundance of some commercial fisheries species due to overfishing. There were dramatic increases in the abundance of sand lance in both the western Gulf of Maine (especially on Stellwagen Bank) and on Georges Bank in the late 1970s and early 1980s (Meyer *et al.* 1979; Payne *et al.* 1986, 1990). The increase in the abundance of sand lance coincided with a large decline in the abundance of several commercial fish species, particularly herring and mackerel (Clark and Brown 1977; Anthony and Waring 1980; Grosslein *et al.* 1980), leading Sherman *et al.* (1981) to speculate that sand lance had replaced these species in the zooplanktivore niche throughout the middle Atlantic, Gulf of Maine, and Georges Bank. The large increase since the mid 1970s in the abundance of humpback whales feeding in the western Gulf of Maine, particularly on Stellwagen Bank, is directly related to the increase in abundance of sand lance populations there (Payne *et al.* 1986, 1990). However, humpbacks on Georges Bank apparently do not feed preferentially on sand lance and their reduced abundance there may be due to declines in the abundance of some other forage species (Payne *et al.* 1986) or to climatic changes that may also have affected the abundance of zooplankton on the bank (Kane 1993; Kann and Wishner 1995). In the middle to late 1980s, a gradual decline began in sand lance abundance, and was accompanied by an increase in herring abundance in the Gulf of Maine (Fogarty *et al.* 1991). This change in fish populations has been accompanied by a decrease in the numbers of humpback whales in the vicinity of Stellwagen Bank and an increase in their numbers farther north in waters of Cultivator Shoals, the northwest peak of Georges Bank, and Jeffreys Ledge, where herring are more abundant (Blaylock *et al.* 1995). More recently, sand lance reappeared on Stellwagen Bank, and herring remained abundant; following this trend, humpbacks also reappeared on Stellwagen Bank (Center for Coastal Studies and College of the Atlantic, unpublished data, in Waring *et al.* 2001).

Humpback whales have the most diverse repertory of feeding behaviors among the great whales. They may feed singly or in closely coordinated groups. Groups of up to 22 individuals may lunge in unison at surface schools of fish (Hain *et al.* 1982; Würsig 1990). During lunge-feeding, a whale rushes at a school of fish or euphausiids near the water surface at an angle of 20° to 40° (occasionally up to 90°) opens its mouth just before reaching the school, engulfing the school, and breaking the water surface with mouth agape (Watkins and Schevill 1979). As much as one third of the body may clear the water surface during a lunge. The whale then closes its mouth and forces the water out through the baleen plates by contracting the mouth cavity, trapping the prey inside. Between lunges, the whale often rests for several seconds to a few minutes, often at a depth of four to six meters, presumably swallowing its prey, before lunging again. Humpbacks also lunge-feed on schools of fish and crustaceans located at mid depths or near the bottom. When feeding at depth, the whale rarely breaks the water surface at the end of the lunge.

Humpbacks may use a variety of herding strategies to force potential prey into a tight aggregation easy to consume in a single gulp. They may use their flukes and fins to slap the water, possibly confusing or stunning prey. Humpback whales, singly or in groups, also produce bubble clouds (single, four- to seven-meter diameter dome-shaped clouds composed of small uniformly sized bubbles) and bubble columns (smaller, 1-1.5-m structures composed of randomly sized bubbles) that encircle or confuse prey long enough for the whales to consume them (Hain *et al.* 1982). Several neighboring whales may bubble net at the same time, increasing the effectiveness of the feeding stratagem.

Approximately 65 percent of the humpback whales feeding in Massachusetts Bay, particularly on Stellwagen Bank, have scuff marks on their lower jaws, suggesting that they have been feeding on or in the bottom (Hain 1991a). This may be a specialized behavior of humpbacks feeding on sand lance (when not schooling, sand lance spend much time buried in sandy sediments (Meyer *et al.* 1979)). Stellwagen Bank is the only location where humpbacks have been observed feeding on the bottom.

4.1.4 Known Disturbance and Mortality Factors

The most common anthropogenic source of mortality for humpback whales in the western North Atlantic is entanglement in commercial fishing gear, particularly off Newfoundland (O'Hara *et al.* 1986; Lien *et al.* 1989a,b; Hofman 1990; Volgenau and Kraus 1990; NMFS 1991a). Nearly 600 humpback whale entanglements, leading to 93 verified deaths of whales (15%), were recorded in Newfoundland waters between 1979 and 1989 (Lien *et al.* 1989b). Between 1975 and 1990, 47 humpback whales were reported entangled in various types of fishing gear in United States waters. Five of these entanglements were fatal (10.6%). The NMFS (1991a) reported 18 entanglements on the outer continental shelf of the northeastern United States. Of these entanglements, nine of the whales were freed by volunteers, six whales died, and the fate of the other three is unknown.

Overall, 12.4% of the photographed flukes and 6.3% of the tail stocks of the western North Atlantic population are scarred due to encounters with fishing gear (Hofman 1990; Volgenau and Kraus 1990). Entanglement mortalities tended to increase as the size of the whale decreased. Twenty-five percent (n=20) of juvenile humpback whales stranded along the central and southeast Atlantic coastlines had injuries indicative of entanglement in fishing gear (Wiley *et al.* 1995). Of 134 whales studied in the Gulf of Maine, entanglement-related scarring on the caudal peduncle was evident in 48% to 65% of the whales (Robbins and Matile 2001). This study suggests that males are more likely to be entangled than females, that yearlings are entangled more than other age classes, and that females that have been entangled may experience reduced reproductive success. Volgenau and Kraus (1990) estimated that the annual loss of humpback whales to entanglement mortality is about 0.3% for the Newfoundland population and 0.2% for the Gulf of Maine population.

Barco *et al.* reviewed length data of 48 of the 52 known mortalities on record for mid-Atlantic waters for the period of 1990 to 2000. 81.2 % (n=39) were first year whales, 14.6% (n=7) were subadults, and 4.2% (n=2) were adults (Barco *et al.* 2001). It should be noted that more mature whales are present in the populations than these stranding numbers indicate (Waring *et al.* 2002).

For the Gulf of Maine populations, the total annual mortality and serious injury rate attributable to human impacts for the period of 1996 to 2000 is estimated at 3.0 (Waring *et al.* 2002). Of this rate 2.8 are linked to incidental fishery interaction records, and 0.4 to vessel collision records. An additional 1.6 mortalities and serious injuries (1.0 from fishery interactions and 0.6 from vessel collisions) are known for this period though it cannot be determined if the whales involved are from the Gulf of Maine population (Waring *et al.* 2002).

Commercial fisheries may compete directly with whales for a particular species of fish, as evidenced by the capelin fishery off Newfoundland (Lien *et al.* 1988), or the whales may become entangled in fishing gear, as happens frequently in both Newfoundland and the Gulf of Maine (Hofman 1990; Volgenau and Kraus 1990). Humpback whales in the Gulf of Maine become entangled most frequently in gill nets, followed by weirs and seines (Volgenau and Kraus 1990; Volgenau *et al.* 1995). In inshore waters of Newfoundland, entanglement occurs most frequently in cod traps, followed by groundfish gill nets and salmon gill nets (Hofman 1990, Volgenau *et al.* 1995). A total of 18 humpback whales were reported entangled in lobster gear in coastal waters between New Jersey and New England between 1976 and 1993 (NMFS 1994). Two humpback whales were entangled in New Jersey, one in New York, and eleven in coastal waters of Massachusetts. Only one of the whales died as a probable result of the entanglement (NMFS 1994). In Canadian waters, a yearly average of 365 (range 174-813) humpback collisions with fixed fishing gear were reported from 1979 to 1987. For the period of 1979 to 1988, an average of 50 entanglements were reported annually in Canadian waters (Lien *et al.* 1988). A review of mortalities and serious injuries for the years 1996 to 2000 reveal that two mortalities and twelve serious injuries in the Gulf of Maine stock were attributable to fishery interactions (Waring *et al.* 2002). Five additional fishery-related mortalities and serious injuries are on record for southeastern and mid-Atlantic waters; it is uncertain if any of these whales are from the Gulf of Maine stock.

Humpback whales are relatively tolerant of boats (Pett and McKay 1990) and are seen frequently in the Great South Channel and Stellwagen Bank in the vicinity of commercial and recreational fishing vessels and whale watch boats. During the early 1970s, before whale watching became popular in Massachusetts Bay, humpback whales were difficult to approach in a small boat (Watkins 1986). The whales usually diminished surface activities and moved away, emitting agonistic trumpeting sounds when approached too closely. However, during recent years humpback whales in nearshore waters often readily accept the presence of vessels, and some even “perform” various surface behaviors when approached by a whale watch vessel. Humpbacks in the western North Atlantic are more habituated to vessel approach than any other cetacean in the area (Watkins 1986). As whales become more habituated to whale-watch and other vessel traffic, the chance of collision increases (Beach and Weinrich 1989). There is some evidence of increased incidents of ship collisions in the Gulf of Maine (NMFS 1991a). In a recent study of stranded humpback whales along the Middle-Atlantic and southeast United States, 30% (n=20) had injuries potentially associated with a ship strike (Wiley *et al.* 1995).

Little is known about natural mortality in humpback whales. Parasites, ice entrapment, predation by killer whales, and fluctuating prey populations due to events such as El Niño may contribute to natural humpback mortality rates (NMFS 1991a). Young or sick humpbacks seem to be particularly vulnerable to attacks by killer whales

(*Orcinus orca*) and occasionally by larger predatory sharks (NMFS 1991a). In the western North Atlantic, 14% (n=3365) of the appropriately photographed humpback whales bear scars, primarily on their flukes, from killer whale attacks (Katona *et al.* 1988; NMFS 1991a). Although humpback whales and killer whales have been observed feeding near one another without aggressive interactions (Dolphin 1987c), killer whales have been observed attacking and killing other species of baleen whales (Hancock 1965; Baldrige 1972; Silber *et al.* 1990).

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

An example of this phenomenon occurred in late November, 1987, when 14 humpback whales died in Cape Cod Bay and Nantucket Sound after eating Atlantic mackerel containing a dinoflagellate toxin, saxitoxin (Geraci *et al.* 1989). A few species of dinoflagellates (phytoplankton) produce powerful neurotoxins. These phytoplankton may periodically experience an explosive increase in population size (a "bloom") in restricted locations along the coast. Primary consumers feed on the toxic blooms and may accumulate high concentrations of the toxins, leading to outbreaks of paralytic shellfish poisoning (when commercially exploited bivalves consume the phytoplankton). The primary consumers may be resistant to the toxins or may not accumulate doses that are toxic. However, secondary or higher level consumers that eat the contaminated primary consumers may accumulate a toxic or lethal dose. This is apparently what happened to the humpback whales. The evidence accumulated by Geraci *et al.* (1989) suggests that a northern stock of mackerel, that had accumulated saxitoxin while feeding on their preferred food of zooplankton and small fish in the Gulf of St. Lawrence, migrated into Massachusetts and Cape Cod Bays, where they were consumed by humpback whales. Some of the whales received a lethal dose of neurotoxin and died almost instantly while feeding. Although this is the first documented case in the world of a large kill of humpbacks or other whales attributable to ingestion of foods contaminated with phytoplankton toxins, it is possible that periodic mortalities were caused by phytotoxins, but went unrecorded. During the first six months of 1990, seven juvenile humpback whales stranded on the shore between North Carolina and New Jersey with no apparent cause of death (NMFS 1991a). The young whales may have been killed by consuming saxitoxin-contaminated food.

Some nonpolar organic contaminants, such as chlorinated pesticides and polychlorinated biphenyls (PCBs), may biomagnify through the marine food web and bioaccumulate to potentially toxic concentrations in the tissues of humpback whales and other piscivorous cetaceans (Reijnders 1986; Aguilar 1987). There are only limited published data on contaminant residues in tissues of northwest Atlantic humpback whales (Taruski *et al.* 1975; Geraci 1989). Concentrations of synthetic organochlorines in blubber of four humpback whales from the western North Atlantic stock (Geraci 1989) were higher than concentrations of organochlorines in blubber of several other Atlantic baleen whales, including right whales (Woodley *et al.* 1991) and fin whales (Aguilar and Borrell 1991; 1994). The lower organochlorine residues in right whales probably are caused by the lower trophic status of these zooplankton-feeding species. The lower residues in fin whales may be due to geographic differences; the fin whales were sampled at a whaling station in northwestern Spain and the humpback whales were collected in United States waters of the northwestern Atlantic.

Accumulated organochlorines may lead to a variety of pathological conditions in cetaceans, in particular reproductive impairment (Addison 1989). Nonpolar organic contaminants tend to accumulate in lipids, such as blubber, while the whale is feeding, and may be mobilized during fasting (female humpback whales are not known to feed when they are in the winter calving grounds). These mobilized contaminants may accumulate in the lipid rich milk when the female whale is lactating and nursing its newborn calf (Aguilar 1987; Aguilar and Borrell 1994).

4.2 The Fin Whale (*Balaenoptera physalus*)

Fin whales (*Balaenoptera physalus*) are present in all the major oceans of the world from the Arctic to the tropics, with greatest numbers in temperate and boreal latitudes (Evans 1987). Fin whales are long and slender, growing to a maximum size of 27 meters and 73,000 kg (Minasian *et al.* 1984). As with most cosmopolitan whales, animals from the Southern Hemisphere tend to grow to a larger size than those in the Northern Hemisphere (Slijper 1978), and females are generally larger than males. The average adult size calculated for fin whales in the western North Atlantic is 16.1 meters, though the largest fin whale sighted in the northwest Atlantic during

the CeTAP program was 21.6 meters in length. North Atlantic fin whales appear to be smaller than adults captured in Iceland (18.3 m), Canada (16.9 to 18.4 m), and Norway (17.6 to 18.9 m) (Hain *et al.* 1992). This may be due to sexual dimorphism, seasonal or environmental factors, latitudinal differences, or a sampling bias. It may also be due to population segregation (Seargent 1977). It is unclear whether fin whales in the North Atlantic split into separate feeding stocks. Mitchell (1974) suggested that fin whales seen off the United States, Nova Scotia, and Labrador coasts were from one or a few closely related populations. Fin whales often travel alone, but an average group size ranges from two to three individuals. Groups can get as large as 65 individuals, though groups of more than ten animals are uncommon (CeTAP 1982).

Fin whales are primarily dark gray or brown in color, and the ventral sides of the belly, flukes, and flippers are white. Like humpback whales, fin whales can be individually identified from their natural marks and scars. Distinctive features include the tall, falcate dorsal fin, the light pigmentation (or blaze) on the right side of the head, and the V-shaped gray-white "chevron" on the back and sides (Agler *et al.* 1990). One of the most unusual features of the fin whale is its asymmetrical coloration. The right side of the head, lower lip, upper lip, and a portion of the baleen is white, while the entire left side of the head is dark in color. It has been hypothesized that this coloration is a feeding related adaptation (Katona *et al.* 1993) though there is no evidence to date to support this (Tershey and Wiley 1992).

Newborn fin whales are just under eight meters long (Hain *et al.* 1992). Typically, the rapidly growing calves are weaned at age seven months to one year. Like most baleen whales, fin whales may have a calf every two years. Female fin whales that summer in the Gulf of Maine average one birth every 2.71 years (Agler *et al.* 1993). The average rate of increase in the size of the northwestern Atlantic fin whale population is approximately 8% per year (Agler *et al.* 1993). Females reach sexual maturity after four to seven years, apparently depending on availability of food (Ohsumi 1986). The size at sexual maturity is fifteen to sixteen meters. Fin whales may live for 85 to 90 years (Evans 1987).

Relatively little is known about reproduction in North Atlantic fin whales. Presumably, reproduction takes place during their winter sojourn off the mid- and south-Atlantic states. Based on the distribution of neonate strandings, some of which were premature, calving appears to occur in coastal or offshore waters south of New Jersey between October and January (Hain *et al.* 1992). Hain *et al.* (1992) hypothesized that the Charleston Bight south of Cape Hatteras is the wintering ground for some of the fin whale population that occupies New England waters during the summer. No mating or breeding is known to occur in the Gulf of Maine and Canadian waters.

4.2.1 Population Status and Trends

Stocks of fin whales in the United States, Nova Scotia, and Labrador are believed by some to be from one or a few closely related populations, whereas the Icelandic population is distinct (Mitchell 1974; Donovan 1991). However, the population structure is not well understood. A recent genetic study (Berube *et al.* 1998 in Waring *et al.* 2001) supports the presence of several subpopulations first suggested by Kellogg (1929). Observations of fin whales from the United States, Canada, and Iceland indicate that the average size of fin whales from more northern Atlantic waters off Canada and Iceland is larger than that of fin whales sighted off the U.S. North Atlantic coast. Sergeant (1977) suggested that stocks of fin whales along the North American coast may be segregated latitudinally by length during part of the year. Because little is known about the winter breeding and calving areas of fin whales, it is uncertain whether fin whales segregate into a few separate breeding populations that form several distinct summer feeding aggregations occupying different feeding grounds. Berube *et al.* (1998) suggested different subpopulations use the same feeding grounds.

Fin whales were listed as endangered throughout their range in 1970. Because of their high cruising speed, fin whales were not harvested commercially in large numbers until other species, such as slow-moving right whales, were depleted and whalers developed high-speed boats (Leatherwood *et al.* 1976). Nonetheless, more than 700,000 fin whales were harvested worldwide in the twentieth century (NMFS 1994). A fishery for this species existed in Nova Scotia from 1964 to 1972 (Mitchell 1974). During this period, 3,528 individuals were harvested. Commercial harvesting of fin whales elsewhere in the world continued at least into the early 1990s. However, stocks of fin whales have not been as severely depleted by commercial whaling as other stocks of large whales.

The estimated modern worldwide population of fin whales is 105,000 to 125,000 individuals (Würsig 1990). Fin whales are the most abundant and frequently sighted of the endangered great whales visiting the coastal waters of the northeastern United States. There are several estimates of the Western North Atlantic population size. Mitchell (1974) estimated that approximately 7,200 fin whales occupy the outer continental shelf between Cape Cod and Labrador on a seasonal basis. Allen (1974) estimated that 2,000 fin whales visited Newfoundland waters each year in the early 1970s, but the number appears to have dropped during the 1980s (Lynch and Whitehead 1984). Fin whales in outer continental shelf waters off the eastern United States from Cape Hatteras to the Canadian border are estimated to range from 5,000 individuals in the spring and summer to approximately 1,500 individuals in the fall and winter (Hain *et al.* 1992). The portion of the northwest Atlantic population that visited the Gulf of Maine in the late 1970s and early 1980s range from about 3,000 individuals in the spring and summer to 200 individuals in the fall and winter (CeTAP 1982). The 2001 NMFS Marine Mammal Stock Assessment Report estimates the western North Atlantic stock abundance at 2,814 (CV=0.21), with a minimum size estimate of 2,362, based on a 1999 line-transect sighting survey conducted by a ship and airplane covering waters from Georges Bank to the mouth of the Gulf of St. Lawrence (Waring *et al.* 2001). Because of the fin whale's extended distribution and poorly understood population structure, this is considered to be an extremely conservative estimate.

Prior to exploitation, the size of the fin whale population in the western North Atlantic Ocean was likely between 30,000 and 50,000 individuals (CeTAP 1982). Hain *et al.* (1992) suggested that the 1992 population of fin whales off the northeast coast of the United States might be in the range of 9,000 to 10,000 animals. The fin whale population may be increasing, though insufficient data are available to determine population trends.

4.2.2 Seasonal Distribution in North Atlantic Waters

Fin whales are the most common of the large whales in the temperate waters of the western North Atlantic, and are found all along the continental shelf between Cape Hatteras and southeastern Canada in all seasons (Hain *et al.* 1992) (Figure 3). Their distribution is cosmopolitan, with a less distinct, seasonal latitudinal migration than other baleen whales (Evans 1987). The distribution, abundance, and general ecology of the species is poorly understood, primarily because fin whales were not heavily exploited by commercial whalers in United States waters to the degree that they were in other areas. However, studies have recently been organized to fill these gaps in our understanding of fin whale ecology. They are commonly seen on the continental shelf in waters less than 100m deep, and rarely on the continental slope or beyond.

In spring and summer, approximately 5,000 fin whales occupy the continental shelf between Cape Hatteras and the Canadian border; numbers decrease to about 1,500 during the fall and winter each year (Hain *et al.* 1992). Thus, fin whales are by far the most abundant baleen whales in coastal waters of the middle Atlantic. They are common in waters out to the shelf edge at 200 m, but rarely are sighted in waters deeper than 2,000 m. Sixty-five percent of sightings are in water depths of 21 to 100 m. Because the fin whale is the most numerous of the large cetaceans with the largest food requirements, it has the largest impact on the continental shelf ecosystem of any cetacean species, and may be a valuable indicator of the health of this area (Hain *et al.* 1982). There is some evidence of maternally-directed feeding site fidelity in females (Clapham and Seipt 1991), although this varies among individuals (Seipt *et al.* 1990).

New England waters are important feeding grounds for fin whales. Jeffreys Ledge, Stellwagen Bank, and Cape Cod Bay experience a spring influx of fin whales, reaching maximum numbers during the summer (CeTAP 1982). They are most abundant along the 40 to 50-m depth contour, particularly in the Great South Channel, across Stellwagen Bank and northeastward to Jeffreys Ledge (Hain *et al.* 1992). The summer distribution of fin whales is similar to that of humpback whales, and the two species are considered sympatric throughout much of their range in United States waters of the Atlantic during the summer feeding season.

During the fall and winter, three quarters of these whales leave the area, and the distribution of the remaining whales contracts to the mid-shelf east of New Jersey and to Stellwagen and Georges Banks. The majority moves south and offshore, starting in October, to wintering grounds off the Delmarva Peninsula and the Outer Banks of North Carolina (CeTAP 1982; EPA Region 1 1988), and perhaps further south. Hain *et al.* (1992) speculate that the large numbers of fin whales sometimes sighted in waters off Cape Hatteras in spring and fall are moving between northern summer feeding grounds and southern over-wintering grounds in the Charleston Bight off South Carolina. However, very few surveys have been performed in this area, so it is uncertain if fin whales

actually occur there in large numbers during the winter. Acoustic data indicate that fin whales are present far offshore during the winter months (Clark *et al.* 1993).

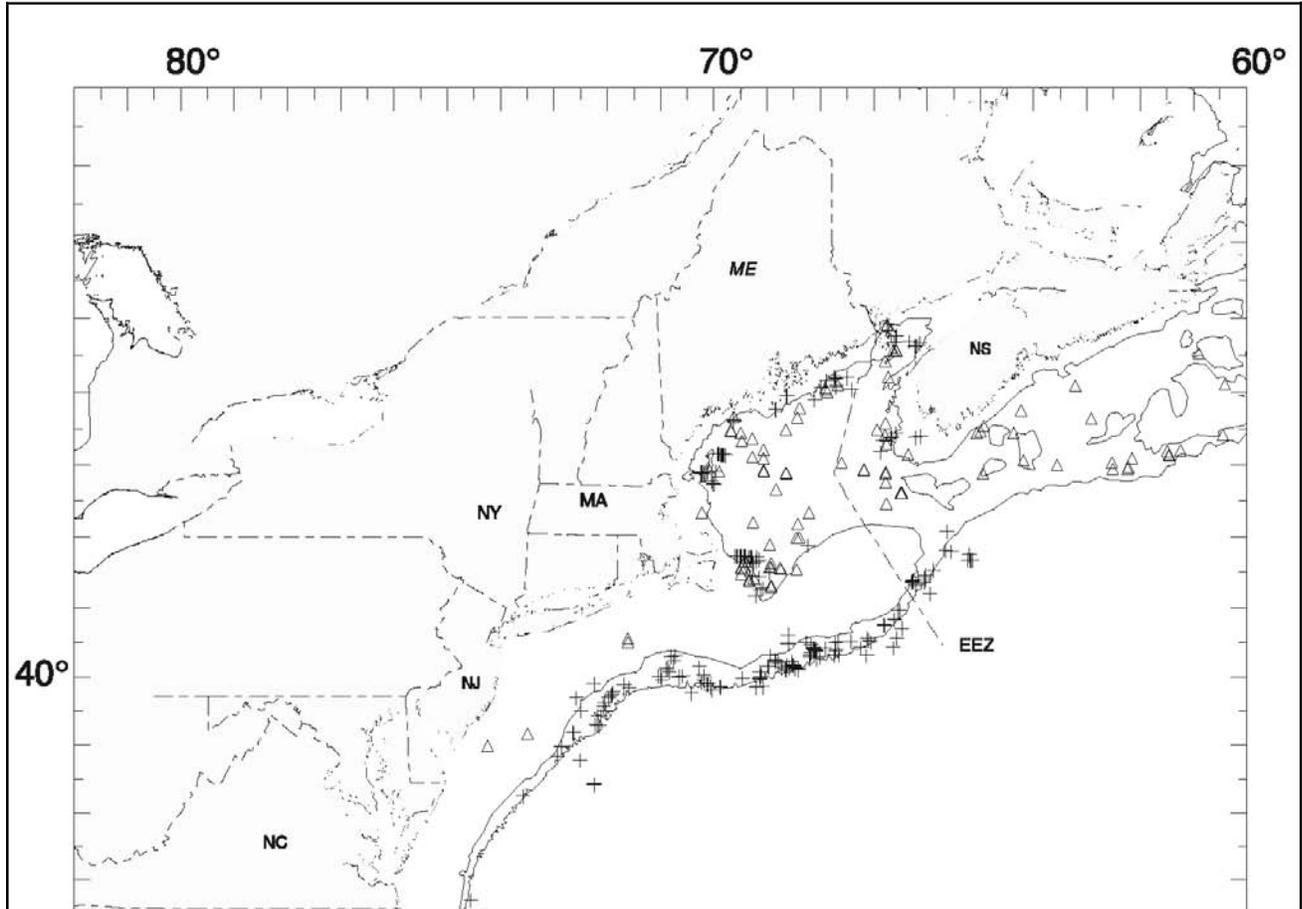


Figure 3. Distribution of fin whale sightings from summer NEFSC and SEFSC shipboard and aerial surveys for the period 1990 to 1998.

Fin whale calves arrive in the summer feeding areas in Rhode Island Sound, off New England, and in eastern Canada with their mothers. Even after separation from their mothers, which usually takes place after about a year, most juveniles return to the same feeding areas they first visited with their mothers, suggesting that migratory behavior and preferred feeding locations are maternally derived (Seipt *et al.* 1990; Clapham and Seipt 1991).

Fin whale distributions have changed somewhat in recent years, similar to the changes observed for humpback whales. Both species feed on similar species and changes in seasonal distributions reflect changes in the relative abundance and distribution of certain forage species, such as sand lance and herring (Payne *et al.* 1986; Fogarty *et al.* 1991).

Very few whales are found in Nantucket Sound, even though the Stellwagen Bank feeding grounds are in adjacent offshore waters. Most whales are found in areas where their primary food source can be easily located. The bathymetric and oceanographic features that favor dense aggregations of fin whale prey species are not developed in Nantucket Sound to the extent that they are farther north around Stellwagen Bank, Jeffreys Ledge, Browns and Bacaro Banks, and in the Great South Channel (Kenney and Winn 1986). Therefore, the preferred foods of fin whales and the whales themselves occur in Nantucket Sound with far less abundance and frequency than in high-use areas farther north.

Allen (1974) documented four or five instances of fin whale sightings within Nantucket Sound between 1854 and 1913. The first was captured outside Nantucket Harbor in 1854 and one other was found floating dead in

Nantucket Harbor in 1881. NMFS cumulative sightings data listed one sighting of a fin whale between Falmouth and Martha's Vineyard. The North Atlantic Right Whale Consortium database contains a record of three fin whales in Nantucket Sound in 1977 (Robert Kenney 2002, personal communication). Most references show fin whales moving from the eastern shores of Long Island Sound, out around the southern shores of Nantucket and Martha's Vineyard towards Nantucket Shoals and the Great South Channel. Therefore, historically and at present, Nantucket Sound does not appear to be an important area for fin whales.

4.2.3 Food and Feeding Behaviors

The distribution of fin whales is likely a function of the distribution of their food (Katona and Whitehead 1988). Capelin abundance alone accounted for 63% of the seasonal variation in baleen whale abundance in Newfoundland waters (Piatt *et al.* 1989). Because of their large size, fin whales may depend on higher density prey patches than other smaller baleen whales. However, the foraging thresholds of baleen whales may vary in relation to the overall abundance of their prey (Piatt and Methven 1992). Fin whales are euryphagous (*broad diet*), and therefore in years when their preferred prey is scarce (*i.e.* 1986), distribution within the Gulf of Maine varied to a lesser degree than that of stenophagous (*selective diet*) species (Payne *et al.* 1990).

Fin whales feed on a wide variety of small schooling fish and crustaceans. Since the mid 1970s, the preferred prey of fin whales on Stellwagen Bank in Massachusetts Bay is sand lance (Overholtz and Nicolas 1979; McKenzie and Nicolas 1988). In coastal waters off Newfoundland, they feed primarily on capelin (Piatt *et al.* 1989). North Atlantic populations also have been reported to feed on euphausiids, copepods, squid (*Loligo* spp. and *Illex* spp.), and myctophid fish when locally abundant (Mitchell 1974; Katona *et al.* 1977).

Brodie *et al.* (1978) reported that a 17.8 meter fin whale caught off Nova Scotia contained approximately 560 kg of euphausiids in its fore-stomach, probably having eaten more than that amount in less than eight hours. Hain *et al.* (1992) estimated that a "typical" 25.7-ton fin whale eats about 533 kg of prey daily during the summer feeding period. He estimated that the entire fin whale population of the northeast coast of the United States consumes about 150,000 metric tons of prey during the fall and winter and 494,000 metric tons of food during the more active spring and summer feeding periods. Baleen whale populations, including fin whales, off Newfoundland may consume about 400,000 metric tons of capelin each year (Winters 1975). A similar amount of food may be consumed by baleen whales on Georges Bank (Hain *et al.* 1985). Thus, fin whales have an important effect on the pelagic marine ecosystem of the western North Atlantic.

Fin whales eat many of the same foods as humpback whales and the two species frequently are seen feeding together in spring and summer feeding areas (CeTAP 1982). Fin and humpback whales likely compete directly with cod, haddock, other piscivorous ground fish, and humans for food (Overholtz and Nicolas 1979; Hain *et al.* 1985). Hain *et al.* (1992) estimated that the amount of food eaten by baleen whales each year was equivalent to the amount of the prey species harvested by humans. However, Piatt *et al.* (1989) estimated that fin, humpback, and minke whales consumed less than 2% of the available capelin during heavy feeding in Witless Bay, Newfoundland. The large breeding population of seabirds in the area probably consumed more capelin than the whales. The three species of whales consumed 60 to 100 metric tons of capelin each year from the bay, with the less abundant fin whales taking about 6.6 to 11 tons, and the more abundant humpback whales taking 47 to 80 tons per year. These observations suggest that, although fin and humpback whales do compete for some of the same foods with some commercially valuable groundfish, the available food resource is not limiting to either group of consumers.

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

Lunge-feeding in fin whales involves moving rapidly parallel to the water surface, as opposed to swimming at an acute angle to the sea surface (Watkins and Schevill 1979). Typically, a fin whale approaches a school of fish or euphausiids at a speed of five to eleven km/h (Orton and Brodie 1987). It waits to open its mouth until close to or in the school (Watkins and Schevill 1979). It opens its mouth to an angle of 10° to 30°, depending on the size

of the school, and closes it slowly over a period of one to three seconds, engulfing a massive volume of water as it continues to swim forward. The grooved pouch forming the floor of the mouth of fin whales is extremely elastic and can extend tremendously during feeding, allowing the mouth to accommodate a large amount of water, equivalent to as much as 50% of the volume of the whale (Pivorunas 1979; Orton and Brodie 1987). The force of water rushing into the open mouth causes the elastic pouch to expand, often doubling the diameter of the whale. The rapid expansion of the mouth cavity allows water to enter with minimal disturbance, avoiding a bow wave that might allow prey to escape. After the whale closes its mouth, a complex network of muscles lining the grooved pouch contracts slowly, compressing the water and forcing it out through the baleen plates on either side of the jaws. The food retained by the baleen is swallowed slowly through the small throat.

4.2.4 Known Disturbance and Mortality Factors

There is little published information about natural and anthropogenic causes of death and disease in fin whales. It is probable that the hazards that affect humpback whales also affect fin whales. Fin whales often are caught in fish traps deployed in offshore Canadian waters. Between 1969 and 1986, twelve fin whales were entangled in fishing gear, usually groundfish gill nets, in inshore waters of Newfoundland (Hofman 1990). Five of these whales (42%) died. Between 1975 and 1990, three fin whales were observed entangled in fishing gear in the Gulf of Maine (Volgenau and Kraus 1990). All entanglements were in lobster gear. The commercial lobster industry reported six instances of fin whale entanglements in lobster gear between November 1975 and January 1991 (NMFS 1994). All but one of the whales was alive when sighted. Three of the entangled whales were sighted in Massachusetts, two in New York, and one in Maine. Such entanglements may indicate that fin whales sometimes feed near or at the bottom. NMFS fisheries observers reported no fisheries-related fin whale mortalities for the period 1995-1999 (Waring et al. 2001). Anecdotal records from NMFS for the same period found three records of fishery related mortality or serious injury to fin whales, yielding a minimum annual rate of mortality and serious injury from fishery interaction of 0.6 individuals for United States and Canadian waters (Waring et al. 2001). An additional eight records for the same period did not contain enough information to determine if the entanglement was severe enough to cause serious injury or if the entanglement contributed to mortality. The 2002 Marine Mammal Stock Assessment Report (Waring et al. 2002) reports two mortalities or serious injuries from vessel collisions (0.4 minimum annual mortality or serious injury rate) and four additional inconclusive entanglement reports for the period of 1996 to 2000.

Fin whales seem to be the most wary of the great whales when approached by whale watch boats and other vessels in Massachusetts Bay (Watkins 1986). Fin whales react strongly to low-frequency ship sounds which are near the frequency of their own vocalizations (14 to 750 Hz) (Cummings *et al.* 1986; Watkins *et al.* 1987). In the early 1970s, they actively avoided approaching vessels and would often dive if approached. In recent years, however, they have either ignored small vessels or actually approached to investigate them (Watkins, 1986). Although they have become accustomed to small vessel activity in recent years, they apparently are not often harmed by it.

High swimming speeds may protect fin whales from collisions with motor vessels. The Smithsonian Institution Marine Mammal Database contains nine records of collisions or propeller scaring of fin whales between 1980 and 1994 (NOAA 1995). However, several of the documented and photographed fin whales in the western North Atlantic population have prominent scars indicative of boat collisions (Agler *et al.* 1990; Seipt *et al.* 1990). Anecdotal records from NMFS contain six documented cases of mortality or serious injury from vessel collisions, yielding a minimum annual mortality or serious injury rate of 1.2 individuals attributable to vessel collisions. An additional seven records make reference to fin whale collisions with vessels, but were inconclusive as to resulting serious injury or mortality (Waring et al. 2001). The 2002 Marine Mammal Stock Assessment Report (Waring et al. 2002) reports six mortalities or serious injuries from vessel collisions (1.2 minimum annual mortality or serious injury rate) and five additional inconclusive collision reports for the period of 1996 to 2000.

There have been 72 verified strandings and nine “floaters” of fin whales along the U.S. Atlantic coast during this century (Hain *et al.* 1992). The years with the highest number of strandings were 1983, with twelve strandings, and 1986, with six strandings. Strandings have occurred most frequently on Cape Cod, Cape Hatteras, and Long Island. All strandings of neonates (less than eight meters long) occurred south of New Jersey. The cause of death of most of these whales is unknown. However, a yearling female fin whale stranded in New England in 1977 apparently died of a massive infection of giant nematode parasites (*Crassicauda boopis*) in the kidneys (Lambertsen 1986). This parasitic disease has a prevalence of nearly 95% in the Icelandic population of fin

whales and appears to be very common in other fin whale stocks as well (Lambertsen 1986). The parasite may cause renal failure and mild anemia in severely infected whales (crassicaudiosis). It may be passed from mothers to their suckling calves in the urine (the urethral opening and the mammary grooves are close together in most whales). Lambertsen (1986) suggested that crassicaudiosis is a major natural cause of mortality in fin whales.

Because fin whales are the fastest swimmers of the baleen whales, it is unlikely that predation by killer whales and large sharks is an important cause of natural injury and death, except possibly among the very young or sick. Nevertheless, the literature contains some records of attacks by killer whales on fin whales (Tomlin 1957).

Other natural mortality factors may include accumulation of biological toxins from prey species to whale tissues through trophic transfer and biomagnification. At the time in 1987 when fourteen humpback whales died from consumption of mackerel contaminated with phytoplankton toxin (see Section 4.1.4, above), two partly decomposed fin whales washed up on the western shore of Cape Cod Bay at Marshfield and Manomet, MA (Geraci *et al.* 1989). The cause of death was not determined, but may have been consumption of the contaminated fish, as fin and humpback whales eat similar foods.

Limited information exists on concentrations of anthropogenic chemical contaminants in the tissues of fin whales (Taruski *et al.* 1975; Wagemann and Muir 1984; Aguilar and Borrell 1991). Blubber of fin whales harvested off Spain contained 100 to 1,000 µg/kg lipid weight (parts contaminant per billion parts lipid) total DDT and up to about 1,800 µg/kg lipid total PCBs (Aguilar and Borrell 1991). Females contained lower concentrations of these organochlorines than males, probably due to transfer of these lipophilic compounds to embryos during gestation and to calves in the lipid-rich milk (Aguilar 1987). These concentrations are higher than those reported by Woodley *et al.* (1991) for right whales from the western North Atlantic, but are comparable to concentrations in other fish-eating baleen whales (Taruski *et al.* 1975; Wagemann and Muir 1984). Blubber from Icelandic sei whales (planktivorous) analyzed by Aguilar and Borrell (1991) contained about half the concentrations of total DDT and PCB of blubber from fin whales. The differences probably related to differences in the feeding habits of the two cetaceans.

Like most large whales, fin whales do not seem to be particularly sensitive to spilled oil. Following a spill of nearly eight million gallons of heavy bunker C fuel oil from the tanker *Argo Merchant* on Nantucket Shoals in 1976, large numbers of whales, including 21 fin whales, were observed in the area of the oil slick (Grose and Mattson 1977). Two fin whales were seen in a heavily oiled area and apparently were not bothered by the oil. None of the whales showed obvious distress from coming in direct contact with the oil. Following another spill of bunker C and No. 2 fuel oils from the *Regal Sword* southeast of Cape Cod, fin whales were observed surfacing in heavy slicks with no apparent adverse effects (Goodale *et al.* 1981).

4.3 The North Atlantic Right Whale (*Eubalaena glacialis*)

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

The majority of right whales sighted in the North Atlantic Ocean are approximately eleven to fifteen meters in length and weigh up to 70 tons (Kraus *et al.* 1988). Females are larger than males. Right whales can be distinguished from other baleen whales by their black color, the absence of a dorsal fin, short, paddle-shaped flippers, large head (more than 25% of the total body length), and a strongly bowed lower jaw. The distinct "V-shaped" blow provides a means of identification from a distance. The distribution and size of thickened, cornified patches of epidermis called callosities on the rostrum, chin, and lower lips varies among right whales and can be used in conjunction with other unique features, such as scars and pigmentation patterns, to identify individuals (Kraus *et al.* 1986; Payne *et al.* 1984).

The coastal waters of Georgia and northeastern Florida are the only known calving grounds of the North Atlantic Right Whale. The calving season extends from late November to early March, and appears to peak in January. Females give birth to a single 4.0 to 5.5 meter calf after a gestation period of at least twelve months (Klumov

1962; IWC 1986). The estimated age of first birth is 7.57 years, which is lower than that estimated for the Argentine population and may be artificially low due to a lack of data. The mean calving interval for female right whales is 3.67 years and appears to be increasing (Knowlton *et al.* 1994).

Sixty-six calves and 87 photo-identified non-calves, or 48% (153/319) of all cataloged right whales have been observed in the southeastern United States region. Cows with newborn calves appear to stay in the southeast region longer than other classes of right whales. This, combined with the tendency of cow-calf pairs to stay significantly closer to shore than other right whales (Kraus *et al.* 1993), may increase their risk of human interactions.

The use of a given nursery by females is culturally transmitted (Schaeff *et al.* 1992). Not all mother-calf pairs that are seen in the southeastern United States region wintering grounds are observed the following summer in the Bay of Fundy (Knowlton *et al.* 1994). In addition, based on mitochondrial DNA (mtDNA) data, one of the three known matriline does not appear to bring its calves to the Bay of Fundy summer nursery area (Schaeff *et al.* 1993). Therefore, it is likely that at least one other nursery area exists.

4.3.1 Population Status and Trends

The pre-exploitation western North Atlantic population is estimated to have numbered 10,000 animals (NMFS 1991b). Commercial harvest of the species over the centuries resulted in the decimation of the population to possibly less than 50 animals at the turn of the century (Reeves *et al.* 1992; Kenney *et al.* 1995). Although protected by international law since 1935, current studies indicate that there are fewer than 350 right whales in the western North Atlantic (Knowlton *et al.* 1994). Based on three years of aerial survey data, CeTAP researchers (1982) estimated the population size at approximately 380 whales. A 1996 population study reported a minimum direct-count estimate of 263 animals (Best *et al.* in Waring *et al.* 2001). A recent census, based on photo-identification techniques, estimates the population at 291 individuals. This is assumed to be the best available population estimate because it is believed to be a nearly complete census (Kraus *et al.* 2001 in Waring *et al.* 2001). However, some of these animals have not been seen in several years and could be dead; to account for this potential bias, the 1998 population estimate does not include catalogued animals that had not been seen for five or more years and were presumed dead for the purpose of this census (Knowlton *et al.*, 1992 in Waring *et al.* 2001). The 2002 Marine Mammal Stock Assessment Report (Waring *et al.* 2002) does not update the 1998 population estimate.

It appears that animals in the western North Atlantic are from a single stock (Knowlton *et al.* 1992). Although reduced to very low numbers, this is the largest remaining population of North Atlantic Right Whales, and it stands to benefit most from recovery actions (NMFS 1991b, 1994; Kenney *et al.* 1995). The western North Atlantic population will be considered “recovered” when it reaches 60-80% of its pre-exploitation number (NMFS 1991b), or about 7000 animals.

From 1980 to 1992, 65 photo-identified cows gave birth to 145 calves, for a mean calving interval of 3.67 years (Knowlton *et al.* 1994). Incorporating birthing data from 1992 through 1999, the mean calving interval has increased significantly from 3.67 to over 5 years, suggesting reproductive failure (Kraus *et al.* 2001 in Waring *et al.* 2001). Possible causes of reproductive failure may include contaminants, biotoxins, nutrition/food limitation, disease, and inbreeding problems (Reeves *et al.* 2001 in Waring *et al.* 2001). Of note, the 2002 Marine Mammal Stock Assessment Report (Waring *et al.* 2001) indicates that a record number of calves were born in 2001.

Despite the cessation of whaling, and the implementation of the Marine Mammal Protection Act (1972) and the Endangered Species Act (1973), the population of North Atlantic Right Whales appears to be growing at a very slow rate. In contrast to the closely related southern right whale (*Eublaena australis*) which is exhibiting signs of recovery in the eastern and western South Atlantic populations and in the Australian population, the situation for North Atlantic Right Whales is less encouraging. The North Atlantic population was estimated to be increasing at rates of 2.5% for the period 1986-1992 based on photoidentification techniques (Knowlton *et al.* 1994) and 3.8% for the period 1979-1989 based on aerial sighting rates in the Great South Channel (Kenney *et al.* 1995). However, updated reproduction and population data and status and trends modeling indicated that the population was in decline in the 1990s (Caswell 1999, Best *et al.* 2001 in Waring *et al.* 2001).

Numerous causes of this negative rate of recovery have been proposed. Because female right whales were preferentially targeted by whalers, it is possible that there is a shortage of females in the population, though recent mitochondrial DNA (mtDNA) evidence indicates that the ratio of males to females is not significantly different than unity (Brown *et al.* 1994). However, there are proportionally fewer parous females of reproductive age in the North Atlantic population (58/152 or 38%) than there are in the South Atlantic population (320/595 or 54%) (Brown *et al.* 1994). Overall, the northern population is increasing at a lower rate than expected. The pool of reproductively active females is not increasing, and calving intervals are longer than expected. This may be evidence of poor reproductive health in this population (Knowlton *et al.* 1994). This slow recovery could also be caused by inherently low reproductive rates (Reeves *et al.* 1978; Brown *et al.* 1994), inbreeding (Kraus *et al.* 1988; Schaeff *et al.* 1992), or reduction of the population below some “critical population size” (Allen 1974). Juveniles represent a smaller proportion of the population than expected (Hamilton *et al.* 1998, Best *et al.* 2001 in Waring *et al.* 2001), possibly due to high juvenile mortality and/or lower recruitment (Waring *et al.* 2001). Though undocumented, an unstable age structure and reproductive senescence in females have also been suggested as potential contributions to the low reproductive rate (Waring *et al.* 2001).

4.3.2 Seasonal Distribution in North Atlantic Waters

Generally, right whales are found along the east coast of North America (CeTAP 1982) but, in the last century, have been seen as far north as Greenland, Iceland, and arctic Norway, as far east as Bermuda, and as far south as the Gulf of Mexico (Waring *et al.* 2001). Right whales, like other large whales, are migratory animals (Gaskin 1982). Some female right whales have been observed to migrate more than 2900 km from their northern feeding grounds to the southern calving/wintering grounds (Knowlton *et al.* 1992). Right whale seasonal movements occur among the following six “high use” areas in the North Atlantic: (1) Cape Cod and Massachusetts Bays, (2) the Great South Channel, (3) the Bay of Fundy, (4) the Scotian Shelf, (5) the coastal waters of the southeastern United States, and (6) Georges Bank/Gulf of Maine.

Originally, it was assumed that right whales remained in these discrete high-use areas for well-defined periods of time (NMFS 1991b). However, satellite-telemetry data have shown that some individuals regularly move among these areas within seasons (Mate *et al.* 1992). In addition, right whale use of preferred habitats may vary with prey availability. During 1986, major shifts in the distribution of many cetaceans occurred apparently in response to changes in prey abundance. Right whales remained in Cape Cod Bay and were also regularly seen on Stellwagen Bank and Jeffrey's Ledge throughout the summer (Payne *et al.* 1990).

The North Atlantic Right Whale was listed as endangered on June 2, 1970 (35 FR 8495). The NMFS approved a recovery plan in December 1991, under Section 4(f) of the Endangered Species Act (NMFS 1994), and a revised plan was under review in 2001 (Waring *et al.* 2001). One of the recommendations of the plan was that designation of critical habitat was essential to the recovery of the North Atlantic Right Whale. On June 3, 1994, NMFS published the “Final Rule Designating Critical Habitat for the North Atlantic Right Whale” (50 CFR Part 226). Based on the best available scientific information and after considering public comment, three areas were designated as critical habitat for the North Atlantic Right Whale: the Great South Channel, Cape Cod Bay, and Southeastern U.S. waters fifteen miles offshore from the Alameda River in Georgia to Sebastian Inlet in Florida. These regions are considered to be essential for the reproduction, rest and refuge, health, continued survival, conservation, and recovery of the North Atlantic Right Whale population. This designation does not restrict human activities within the critical habitat, but instead serves as a means of alerting interested parties, including Federal agencies, to the importance of the area, and helps to focus conservation efforts.

New England waters are important feeding and nursery grounds for right whales (Figure 4). In February through April, an average of 40 animals arrive and feed in Cape Cod Bay (Marx and Mayo 1992). Between 1978 and 1987, more than one half of all photographically identified animals were seen in this area. Peak abundance, including cow-calf pairs, is in April (Hamilton and Mayo 1990). Feeding, nursing, and mating behavior have all been observed in Cape Cod Bay (Schevill *et al.* 1986; Hamilton and Mayo 1990; Marx and Mayo 1992). In the spring, many animals (6 to 22% of the population, and 0 to 57% of all calves), also use the Great South Channel as feeding and nursery grounds (Kraus and Kenney 1991). Utilization peaks in May, when up to 179 animals have been observed in the area. Individuals are usually in temperature-stratified waters north of a persistent thermal front and in water deeper than 100 m.

The movement of whales into the Great South Channel is apparently in response to extremely dense aggregations of zooplankton. It is likely that this is the primary feeding ground for the North Atlantic Right Whale (Kenney *et al.* 1995). In the summer and fall, the lower Bay of Fundy is used as a feeding and nursery area for some animals, including nearly all mother/calf pairs. An additional summer/fall feeding ground, on the southern Nova Scotian shelf, is used almost exclusively by mature right whales (NMFS 1994).

The coastal waters of Georgia and Florida are the only known calving ground and winter nursery area for the North Atlantic Right Whale. Typically, the majority of animals seen in this area are females about to give birth, females with their newborn calves, and some juveniles. In the winter of 1993-1994, there were 54 sightings of right whales in this region. Of these, 30 sightings were of mother/calf pairs, and 11 were of juveniles in surface-active groups (Slay *et al.* 1994). The winter distribution of the remaining population, including all adult males and most of the juveniles, is unknown.

Although important seasonal feeding and nursery grounds for right whales are located further offshore from Nantucket Sound in the Great South Channel and Stellwagen Bank, very few whales have been sighted in Nantucket Sound (Figure 4). Most whales are found in areas where their primary food sources can be easily located. The bathymetric and oceanographic features that favor dense aggregations of prey species are not developed in Nantucket Sound to the extent that they are farther north around Stellwagen Bank, Jeffreys Ledge, Browns and Bacaro Banks, and in the Great South Channel (Kenney and Winn 1986). Therefore, the preferred foods of right whales and the whales themselves occur in Nantucket Sound with far less abundance and frequency than in high-use areas farther north.

Only seven instances of right whales within Nantucket Sound have been documented. Allen (1974) documented four to five instances of a right whale within Nantucket Sound between 1697 and 1913. In 1697, a cow and calf pair was captured near Yarmouth MA and a right whale was spotted in waters around Martha's Vineyard in 1703. In 1800 and 1886, small schools of right whales were documented on the north side of Nantucket. Schevill *et al.* (1981) documented a sighting of two right whales in April 1975 southeast of the western end of Nantucket Island, and an adult female and her calf were monitored via satellite telemetry as having been present in Nantucket Sound just north of the western portion of Nantucket Island and east of Martha's Vineyard. Although more right whales were sighted in Nantucket Sound than fin and humpback whales, Allen (1974) noted that whales were more common on the seaward side of Nantucket Island. Most references show right whales moving from the eastern shores of Long Island Sound, out around the southern shores of Martha's Vineyard and Nantucket towards Nantucket Shoals and the Great South Channel. No right whale surveys have targeted Nantucket Sound, though many ship-based and aerial surveys for great whales (including right whales) in North Atlantic waters pass over or through Nantucket Sound, including the CeTAP and POP surveys. Neither the North Atlantic Right Whale Consortium database nor NMFS data (Figure 4) contain any records of right whale sightings in Nantucket Sound (Robert Kenney 2002, personal communication; NMFS-NEFSC 2002, unpublished data).

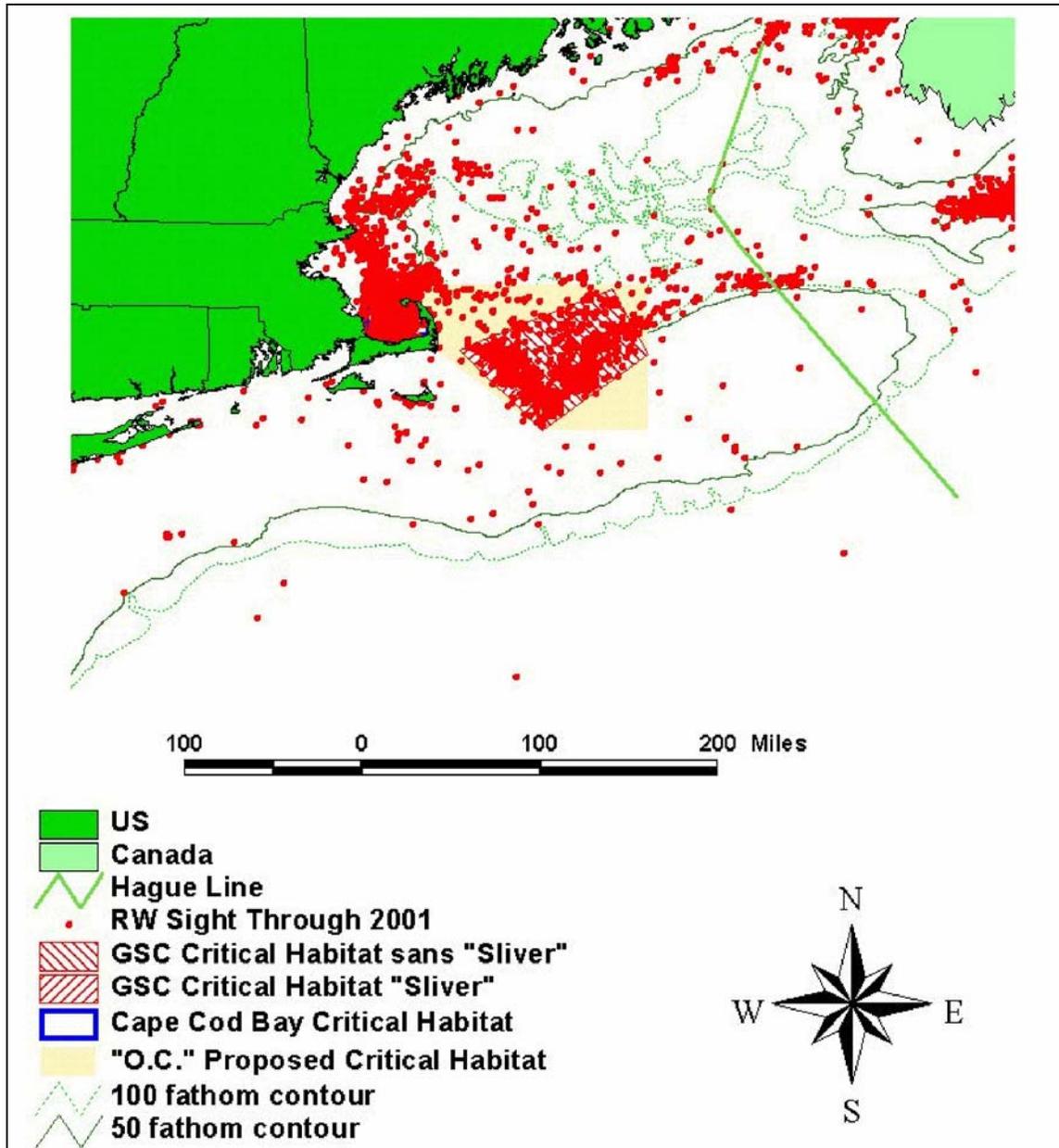


Figure 4. North Atlantic Right Whale Sightings through 2001 (NMFS Northeast Fisheries Science Center, unpublished data).

4.3.3 Food and Feeding Behaviors

The primary prey of right whales in the western North Atlantic are carangid copepods, *Calanus finmarchicus*, and juvenile euphausiids (Nemoto 1970; Watkins and Schevill 1979; Kraus and Prescott 1982; Murison and Gaskin 1989), and secondarily the copepods *Pseudocalanus minutus* and *Centropages sp.* (Marx and Mayo 1992). Both the density of plankton patches and the proportion of caloric-rich adult (Stage V) copepods appear to be factors influencing the foraging threshold of right whales (Kenney *et al.* 1986; Murison and Gaskin 1989; Marx and Mayo 1992; Payne *et al.* 1990). Kenney *et al.* (1986) estimated that the "average" 40,000-kg right whale would need up to 2.4×10^3 kcal m^{-3} and must target extremely dense patches of zooplankton. A group of right whales was associated with such a patch (4.16×10^4 copepods m^{-3} or a median of 2.8×10^3 kcal m^{-3}) for four days while in the Great South Channel. It is not known how right whales locate these dense patches of food.

Sei whales (*Balaenoptera borealis*) (Mitchell 1975; Mitchell *et al.* 1986), sand lance (*Ammodytes spp.*) (Payne *et al.* 1990; Kenney *et al.* 1986), and planktivorous species could represent a source of competition for the right whales's preferred prey (*Calanus finmarchicus*). In 1986, when *C. finmarchicus* levels were high in the Gulf of Maine, right whales, fin whales, and sei whales were the dominant cetaceans in the area. Although Kenney *et al.* (1995) and Knowlton *et al.* (1994) reported an increase in sei whales in the Great South Channel and Nova Scotian Shelf, there is little quantitative evidence of direct competition between right whales and these other species. In addition, *C. finmarchicus* populations are highly variable, and little of this variation is due to predation pressure (McLaren *et al.* 1989; Tande and Slagstad 1992).

The majority of right whale feeding occurs at depth, but occasionally skim feeding occurs at the surface (Nemoto 1970). As right whales swim through the water with their mouth agape, large volumes of seawater are filtered through a triangular opening in the baleen at the front of the mouth. As water flows through the mouth, zooplankton are trapped on the fine fringe of the inner surface of their baleen plates (Watkins and Schevill 1979; Kraus *et al.* 1982; Marx and Mayo 1992). The whale then closes its mouth periodically to swallow its prey. When skim feeding, individuals change swimming direction more often than when traveling (Marx and Mayo 1992).

Feeding behavior has been observed in Cape Cod Bay, Stellwagen Bank (Watkins and Schevill 1979, 1979; Payne *et al.* 1990), the Great South Channel (CeTAP 1982; Winn *et al.* 1995), Jeffreys Ledge, the lower Bay of Fundy (Kraus *et al.* 1982; Gaskin 1982) and the Scotian Shelf (Brownell *et al.* 1986; NMFS 1991b), and is likely to occur in other areas as well when planktonic conditions are suitable. The broad-scale migratory movements of right whales appear to be correlated with zooplankton "blooms" in areas such as Cape Cod Bay (Marx and Mayo 1992) and the Great South Channel (Kenney *et al.* 1995). The majority of feeding in these areas occurs underwater. Feeding has not been observed in the southern wintering grounds off Georgia and Florida, and it is possible that right whales fast while in that area (Kraus and Kenney 1991).

The vertical distribution of right whales is also influenced by the distribution of their prey. Recent evidence indicates that a foraging right whale modifies its dive patterns to follow the vertical movements of *Calanus finmarchicus*. In the Great South Channel, during years when zooplankton exhibited diel vertical migration patterns, diel differences in right whale diving behavior were observed. However, in other years, vertical plankton distribution was more stable throughout the day, and there were no diel differences in right whale diving patterns (Winn *et al.* 1995). Individuals studied by satellite-monitored radio tags exhibited tremendous variation in their dive patterns (Mate *et al.* 1992).

4.3.4 Known Disturbance and Mortality Factors

Many investigators consider habitat change to be the key environmental factor affecting the rate of recovery of the right whale (NMFS 1991b; Gaskin 1982). Of primary concern are the anthropogenic sources of change such as pollution, oil and gas exploration, sea-bed mining, and a general increase in coastal activities due to an increase in human population along the east coast (NMFS 1994; EPA 1993). Numerous wastewater discharges and dredge disposal sites are located in Cape Cod Bay, near Stellwagen Bank (NMFS 1991b) and along the East coast of the United States. These discharges, as well as dredging and dredge material disposal activities, may alter the physical and chemical properties of nearshore waters and sediments, making them unsuitable for right whale feeding and reproduction (EPA 1993). In September 2000, a new offshore sewage outfall serving the Boston, Massachusetts area came online; habitat degradation in Massachusetts and Cape Cod Bays resulting from this outfall is a concern, though impacts have not yet been defined (Waring *et al.* 2001). Intensive human use of areas such as Delaware Bay, the New York Bight, and Long Island Sound may have resulted in the exclusion of right whales from areas they once frequented (Reeves *et al.* 1978). Pollution resulting from intentional or accidental releases of chemicals to coastal waters has also been suggested as an important factor in the apparent poor recovery of North Atlantic right whale populations (Gaskin 1982). Although trace concentrations of several chemicals have been found in tissue samples from right whales (Woodley *et al.* 1991), there is no direct evidence to date that right whales have been adversely affected by pollutants, either through a pollution-induced increase in mortality rates or decrease in reproductive rate or success (EPA 1993).

The most significant impacts to right whales not attributable to habitat change are ship strikes and entanglement in fishing gear. A total of 45 right whale mortalities are on record for the period of 1970 to 1999 (IWC 1999 and Knowlton and Kraus 2001 in Waring *et al.* 2001). Of these deaths, 41.3% were human-induced, from ship strikes (35.6%) and fishing gear entanglement (6.7%). A revised estimate of human-induced mortality and serious

injury rates for the period of 1995 to 1999 is 2.2 individuals per year (Waring et al. 2001). This rate is derived from non-observed fishery entanglement records and ship strike records for United States and Canadian waters. The 2002 Marine Mammal Stock Assessment Report (Waring et al. 2002) adjusts this rate to 1.8 for the years 1996 to 2000. The overall mortality rate for adults has been estimated at 1% (Kraus 1990) to 4% (Gaskin 1982).

Neonatal and juvenile right whales appear to be the most vulnerable and impacted part of the population. Analyses of sighting data between the northern feeding areas and the southern calving areas indicate that about 17% of calves die within their first year of life. After the first year, mortality rates drop to an average of 3% for the next three years, or a total of 27% mortality for the first four years of life (Kraus 1990). Of the known 45 right whale mortalities that occurred from 1970 to 1999, 13 (28.9%) deaths were neonates believed to have died from perinatal or other natural causes (Waring et al. 2001). Thus, even a few incidental deaths may greatly affect the rate of recovery in a drastically reduced population with such a long reproductive cycle (Best 1988). The eleven documented right whale mortalities and serious injuries attributed to human causes for the period 1995 to 1999 include one juvenile mortality and one juvenile serious injury (Waring et al. 2001). Both of these animals were estimated to be two years of age and both were entangled with fishing gear or other rope.

Although right whales spend a great deal of time underwater (Mate *et al.* 1992) they also spend prolonged periods at the surface while surface skim-feeding, resting, and in surface courtship groups (NMFS 1991b). This, and the fact that many of the high-use areas for right whales include major shipping lanes or high-traffic areas along the east coast, makes them susceptible to interactions with ships. Vessel activities can change whale behavior, disrupt feeding practices, disturb courtship rituals, disperse food sources, and injure or kill whales through collisions (NMFS 1994). Twelve percent of all photo-cataloged individuals have scars from ship propellers, and 35.6% (n=45) of right whale mortalities documented between 1970 and 1999 were due to collisions with ships (Knowlton and Kraus 2001 in Waring et al. 2001). Lately, research has pointed to ship-whale interactions as a possible barrier to the recovery of the species (Reeves *et al.* 1978; Kraus *et al.* 1988; Kraus 1990; IWC 1999 in Waring et al. 2001; Knowlton and Kraus 2001 in Waring et al. 2001). Right whales monitored by satellite telemetry frequently swam through or near the shipping lanes off Boston, Portland, Maine, and New York (Mate *et al.* 1992). The size and extent of scarring among right whales indicates that collisions are primarily with large vessels such as container ships and tankers. These collisions are fatal to right whales approximately 19% of the time (Kraus 1990). Adjusting shipping lanes to reduce ship/whale collisions may be only partly effective because right whales appear to use much of the North Atlantic coastline (Mate *et al.* 1997).

Entanglement in fishing gear is the second largest human-related threat to right whales. More than half (61.6%) of the appropriately photographed population of right whales have scars indicative of entanglement in commercial fishing gear (Hamilton *et al.* 1998 in Waring et al. 2001). Between 1970 and 1999, at least 62 entanglements or possible entanglements are on record (NMFS unpublished data in Waring et al. 2001). While entanglement records may not contain enough information to attribute the entanglement to a particular fishery, individuals appear to swim through all types of gear including gill nets, weirs, trailing lines, buoys, seines, and cod traps. Gear and lines become wrapped around the peduncle, the pectoral fins, or are caught in the gape of the mouth and become wrapped around the head (Kraus, 1990; NMFS 1994). If animals are unable to surface to breathe, they will drown. Nets and lines may stay attached for long periods of time due to the use of synthetic, rot-resistant materials by the fishing industry. This may be especially dangerous for juveniles that become entangled while still actively growing.

Of the 45 known mortalities since 1970, three (6.7%) have been attributed to entanglement in fishing gear (Knowlton and Kraus, 2001). In 1994, three whales were reported entangled in gear in the Gulf of Maine and the Bay of Fundy, and two or three additional animals were reported to be injured by gill nets in the southeastern United States (NMFS 1995). At least two individuals ("Stars" and "Necklace") were entangled for more than four years and have been recently photographed without the gear (NMFS 1991b). Although entanglement is less likely to result in a direct mortality (2.9% of gear entanglements are fatal, based on revised Kenney and Kraus 1993 data), it may weaken an animal, making it more susceptible to disease, killer whale attacks, or collisions with ships (Kenney and Kraus 1993). Seasonal and regional restrictions on fishing areas have been proposed as a means of minimizing interactions between the fishing industry and right whales. However, recent studies indicate that individual right whales do not remain in discrete areas for well-defined periods or seasons. Regional closures may therefore be ineffective, and alternatives related to gear modifications or fishing methods may be necessary (Mate *et al.* 1997).

Drilling for oil and gas may introduce adverse effects to right whales, including acoustic disturbance from seismic vessels and drilling rigs, and pollution resulting from accidental releases during performance of these activities. At present, there is no active drilling for oil and gas along the North Atlantic coastline. Exploration for the development of oil and gas resources is currently occurring along the Scotian shelf and exploratory drilling may begin in the next few years in the Canadian section of George's Bank. Previous studies of oil exploration activities conducted off the east coast in the 1980s concluded that cetacean distributions around oil rigs were no different than distributions in undisturbed areas (Sorenson *et al.* 1984). Studies off the California and Alaska coastlines have shown that most species of cetaceans adjust to the presence of drilling equipment (Geraci and St. Aubin 1987). Oil and gas exploration inevitably leads to increased ship traffic in the area, which can be problematic for right whales.

Predation by killer whales may be a source of mortality for right whales. At least 3% (NMFS 1991b) to 9% (Kraus 1990; Kenney and Kraus 1993) of cataloged right whales bear scars, primarily on the flukes, from killer whale attacks (Kraus *et al.* 1986; Kraus 1990). Killer whales are relatively uncommon in the North Atlantic, but have been observed in the coastal waters of Georgia and Florida (Layne 1965), and in the Gulf of Maine (Katona *et al.* 1988). Deaths due to killer whale attacks have been documented for other species of baleen whales (Hancock 1965; Baldrige 1972; Silber *et al.* 1990).

4.4 The Loggerhead Turtle (*Caretta caretta*)

The loggerhead sea turtle (*Caretta caretta*) is listed as threatened under the Endangered Species Act. It is the most common and seasonally abundant turtle in inshore coastal waters of the western North Atlantic. At least five genetically distinct nesting assemblages exist in the western North Atlantic: the Florida Panhandle subpopulation, the South Florida subpopulation, the northern subpopulation (Amelia Island, Volusia County, FL and northward), the Yucatan Peninsula subpopulation, and the Dry Tortugas subpopulation (TEWG 1998, 2000; NMFS-SEFSC 2001). As distinct reproductive populations, these nesting assemblages will not be replenished by regional dispersal if depleted.

Hatchling loggerheads crawl from their nests to the sea and then float at the surface entrained in surface currents that may transport them far out to sea and across ocean basins. They may lead a pelagic life for up to 6.5 to 12 years, with an average of eight years, drifting and feeding in the *Sargassum* community (Carr 1986a, 1986b, 1987; Bjorndal *et al.* 1994; Bolten *et al.* 1994; Bjorndal *et al.* 2000). During this long pelagic period, the young turtles, termed "pelagic immatures" may make several transits of the North Atlantic Ocean in the Great Gyre of the Gulf Stream and grow from a newly-hatched size of 4.5 cm to about 40-60 cm straight carapace length (SCL) before moving inshore to coastal waters (Carr 1987; Dodd 1988) to benthic habitats. Loggerheads settle at an average size of 49 cm SCL, and are then known as "small benthic immatures" (Bjorndal *et al.* 2000) and will occur from Cape Cod through southern Texas. "Large benthic immatures" sized at 70-90 cm SCL will continue to mature for up to another ten years before reaching reproductive maturity (Carr 1987). 90 cm SCL is considered the average size-to-maturity (NMFS-SEFSC 2001). Both benthic immature and adult loggerheads may travel great distances to foraging grounds, and genetic evidence shows that foraging groups are comprised of individuals from multiple nesting assemblages (TEWG 1998).

4.4.1 Population Status and Trends

Multiple in-water surveys of loggerheads and other sea turtles have been conducted at several sites in the western North Atlantic and the Gulf of Mexico. However, due to environmental and biological conditions, changes in sampling effort, variable temporal sampling, and/or different sampling methodologies, the NMFS has been unable to compare results of these studies to determine the status and trends of in-water loggerhead populations (TEWG 2000; NMFS-SEFSC 2001). Loggerhead turtle population estimates are best obtained from nesting data. Scientists can use annual estimates of the number of nests laid each year to determine indirectly the number of adult females nesting in a given year and the number of adult females in the population (TEWG 1998). Limitations to using nesting data to estimate population size are that the most mature life history stage of one gender are used to make estimates about the entire population. However, nesting data are becoming available over large geographic areas and longer time periods, lending to their utility in population modeling.

One modeling method incorporated nesting and stranding data to estimate the loggerhead population size for the period of 1989 to 1995; a second method incorporated aerial survey data in addition to nesting and stranding data, for the same period of time. The two mean post-pelagic (*i.e.* benthic immature and adult) loggerhead population estimates were 224,321 and 234,355, respectively (TEWG 1998). Due to sampling biases, these numbers are believed to be underestimates (TEWG 1998). It should be noted that the TEWG (2000) decided not to update these figures with recent data using the same models because of potential size biases in stranding data and nesting female recapture data.

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

4.4.2 Seasonal Distribution in North Atlantic Waters

Loggerhead turtles are abundant during spring and summer months in coastal waters off New York and the mid-Atlantic states, and a small number of individuals may reach as far north as New England. In the fall, loggerheads migrate southward to coastal waters off the south Atlantic states, particularly Florida, and the Gulf of Mexico. During the winter, the turtles tend to aggregate in warmer waters along the western boundary of the Gulf Stream off Florida (Thompson 1988). In the spring, they congregate off southern Florida before migrating north to their summer feeding ranges (CeTAP 1982).

Following their juvenile pelagic phase, benthic immature loggerheads begin appearing in coastal waters along the United States Atlantic coast. Surface density estimates for summer months obtained from aerial surveys (SETS, CeTAP, GoMEX) indicate the following approximate regional distribution of post-pelagic loggerheads: southeastern U.S. Atlantic, 54%; northeastern U.S. Atlantic, 29%; eastern Gulf of Mexico, 12%; and western Gulf of Mexico, 5% (TEWG, 1998). The abundance of loggerhead turtles is much lower north of Cape Hatteras and nearly all the turtles that visit northern waters during the summer are juveniles with carapace lengths less than about 60 cm (Morreale and Standora 1989).

The center of distribution of juvenile loggerheads along the United States Atlantic coast is in central Florida off Cape Canaveral (Schmid 1995). Many of the young turtles migrate north during the spring and early summer to nearshore feeding areas, such as Core Sound, North Carolina (Epperly *et al.* 1995a,b), southern Chesapeake Bay (Keinath *et al.* 1987; Schmid 1995), and Gardners Bay and Long Island Sound, New York (Morreale *et al.* 1989; Shoop and Kenney 1992; Morreale and Standora 1989). Between 2,000 and 10,000 sub-adult loggerhead turtles use Chesapeake Bay south of the Potomac River for feeding during the summer (Keinath *et al.* 1987). Smaller numbers are encountered, particularly in July, in Delaware Bay (Eggers 1989). Loggerheads also are encountered frequently in Long Island Sound, New York Harbor-Raritan Bay, and along the south coast of Long Island during the summer (Morreale *et al.* 1989). Loggerheads frequently strand due to cold stunning between November and January each year along the north shore of Long Island Sound and in the Bays of eastern Long Island (Morreale *et al.* 1992). When the water temperature drops below about 12°C, the metabolic rate of these cold-blooded reptiles decreases to the point where they are unable to swim and digest food; they become comatose and may die if not warmed quickly. Loggerheads rarely occur north of Long Island around Cape Cod and in the Gulf of Maine (Shoop and Kenney 1992). Several sub-adult loggerheads strand along the south shore of Cape Cod Bay each winter (Matassa *et al.* 1994). The stranded turtles measure 27 to 47 cm SCL, indicating that they are late juvenile pelagic loggerheads. Information from strandings, entanglements, mariner reports, and the U.S. Coast Guard suggest that loggerheads can be expected to occur rarely in Nantucket Sound as well, in the summer and fall months, though no systematic surveys have been conducted in the Nantucket Sound area (Cheryl Ryder 2002, personal communication).

Migratory behavior seems to be cued to sea surface temperatures, with preferred water temperatures off Cape Hatteras falling in the range of 14°C to 28°C (Shoop and Kenney 1992; Coles *et al.* 1994). Loggerhead turtles

first appear in waters around New York and New England in early June, where they remain in New York waters, mostly in coastal for the entire summer (Morreale and Standora 1989). Loggerhead turtles begin to leave northern waters, swimming first eastward and offshore and then southward, in late September through mid-October each year. Nearly all loggerheads remaining in northern waters after the beginning of November are cold-stunned and were likely caught by rapidly declining water temperatures during their southward migration (Morreale and Standora 1989). Loggerheads migrate southward to coastal waters off the south Atlantic states, particularly from Cape Hatteras, North Carolina, to Florida, with peak numbers passing Cape Hatteras in November (Morreale and Standora 1989; Musick *et al.* 1994). Some juvenile loggerheads remain through the winter in nearshore waters of North Carolina south of Cape Hatteras where water temperatures remain at or above 11°C (Epperly *et al.* 1995b). During the winter, the turtles tend to aggregate in warmer waters along the western boundary of the Gulf Stream off Florida (Thompson 1988). They also may hibernate in bottom waters and soft sediments of channels and inlets along the Florida coast (Ogren and McVea 1981; Butler *et al.* 1987). In the winter and spring, they congregate off southern Florida before migrating northward to their summer feeding ranges (CeTAP 1982). Peak numbers of northward-migrating sub-adult loggerheads occur off Cape Hatteras in April and May each year (Musick *et al.* 1994).

4.4.3 Food and Feeding Behaviors

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

Loggerhead turtles in Nantucket Sound are expected to be found in similar shallow coastal areas as those in the New York Bight area. In the New York Bight area, nearly all sightings of juvenile loggerheads (the only life stage present) are in shallow coastal bays and estuaries where the turtles feed on benthic invertebrates, particularly crabs (Morreale and Standora 1989). They rarely are observed in water depths of 20 m or more, and rarely feed at depths greater than about 15 m.

Juvenile loggerhead turtles grow rapidly during their summer visits to northern coastal waters (Morreale and Standora 1989). The increase in SLCL of juvenile turtles in New York ranges from 10.6 cm/year for 40- to 50-cm individuals to 3.0 cm/year for 50- to 60-cm individuals. These growth rates are slightly lower than those of loggerheads in Florida and the Bahamas (Mendonça 1981; Bjorndal and Bolten 1988). Schmid (1995) estimated, based on tag-recapture studies, that loggerheads along the east coast of central Florida grow at a rate of 5.56 cm/year. Growth rate slows as the turtles approach sexual maturity, which may occur after 12 to 45 years in the wild (Zug *et al.* 1983; Frazer and Ehrhart 1985; Foster 1994) when the turtles are about 74 to 90 cm SLCL (Dodd 1988; Foster 1994). Adult loggerheads from the Florida population may grow to more than 120 cm SLCL and weight more than 180 kg (Ehrhart and Yoder 1978). Data on the growth of juvenile loggerhead turtles in New York waters suggests that the turtles are not casual visitors to these northern habitats, but visit coastal bays and nearshore waters of the region intentionally each summer to feed (Morreale and Standora 1989).

4.4.4 Known Disturbance and Mortality Factors

Strandings are a major source of mortality for loggerhead sea turtles; however, natural causes of these strandings are not well understood. One of the largest reported strandings occurred between 1980 and 1983 when there were 6,691 reported strandings of loggerhead turtles along the U.S. Atlantic and Gulf of Mexico coasts (Thompson 1988). Most strandings (77%) were along the southeast coast from North Carolina to Florida; about 11% of strandings occurred north of Cape Hatteras. Most strandings occurred during the spring and

summer; 79% of the strandings north of North Carolina were between April and July. The cause of these strandings has not been determined.

Between four and seventeen loggerheads stranded each year in Massachusetts and Rhode Island waters during the period from 1990 to 2000, though atypically high numbers of 72 and 56 loggerhead strandings were reported in Massachusetts in 1995 and 1999, respectively (Table 1) (Sea Turtle Stranding and Salvage Network, unpublished data). For the period of 1980 to 1997, seven loggerheads strandings were recorded on the shorelines in Nantucket Sound, and four strandings were reported on the southern shorelines of Martha's Vineyard and Nantucket (Figure 5) (NMFS, unpublished data). Strandings occur most frequently in the fall and winter; these strandings may be caused by cold stunning (Morreale *et al.* 1992; Matassa *et al.* 1994). As with most marine turtles, prolonged exposure of loggerheads to low water temperatures, below about 8°C, may result in dormancy, shock, and death. During the winters of 1985, 1986, and 1987, 28 loggerhead turtles became cold-stunned and washed ashore in the Bay system of eastern Long Island and along the north shore of the island (Morreale *et al.* 1992). The turtles became cold-stunned between early November and late January each year. However, cold stunning is not restricted to northern waters, as demonstrated by several documented cold stunning incidents involving loggerheads in the northern part of the Indian River Lagoon system in east central Florida (Witherington and Ehrhart 1989; Schroeder *et al.* 1990).

Table 1. Strandings of loggerhead turtles (*Caretta caretta*) in Massachusetts and Rhode Island from 1990 to 2000, all months combined each year (Sea Turtle Stranding and Salvage Network, unpublished data).

State	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000
MA	4	6	17	9	12	72	4	10	6	56	6
RI	-	1	-	-	1	1	-	3	1	1	4

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

Shrimp fishing is the best quantified and probably the dominant source of anthropogenic mortality among North Atlantic loggerhead turtles (Thompson 1988; NRC 1990). An estimated 7,913 to 18,148 loggerheads are killed each year in shrimp nets along the southeast coast of the United States. An additional 3,555 to 4,716 loggerhead turtles are killed in shrimp nets each year in the Gulf of Mexico, bringing the total killed in the shrimp industry to approximately 10,000 to 23,000 individuals per year (Henwood and Stuntz 1987). The National Research Council (1990) estimated an annual mortality of loggerheads due to the commercial shrimping industry of 5,000 to 50,000 individuals in United States waters. These mortality estimates were obtained prior to the introduction of turtle excluder devices (TEDs). Crowder *et al.* (1994b) estimated that implementation of TED regulations has resulted in a decline of about 5% to 6% per year in loggerhead strandings. Recent models indicate that it may take up to 70 years or more of deployment of TEDs on shrimp trawls for an increase in the numbers of nesting females to become evident (Crowder *et al.* 1994).

Other fisheries account for 500 to 5,000 mortalities per year (NRC 1990). Three loggerhead turtles (two in New Jersey and one in New York) were reported entangled in lobster gear between 1983 and 1991 by the Sea Turtle Stranding and Salvage Network (NMFS 1994). Nationally, dredging operations and collisions with boats may take an additional 50 to 500 loggerheads per year each (NRC 1990). Dickerson *et al.* (1992) reported that winter dredging of the Cape Canaveral Channel, FL and King's Bay, GA resulted in the deaths of hibernating sea turtles. Entrainment in electric power plant cooling water intakes accounts for fewer than 50 loggerhead deaths per year (NRC 1990).

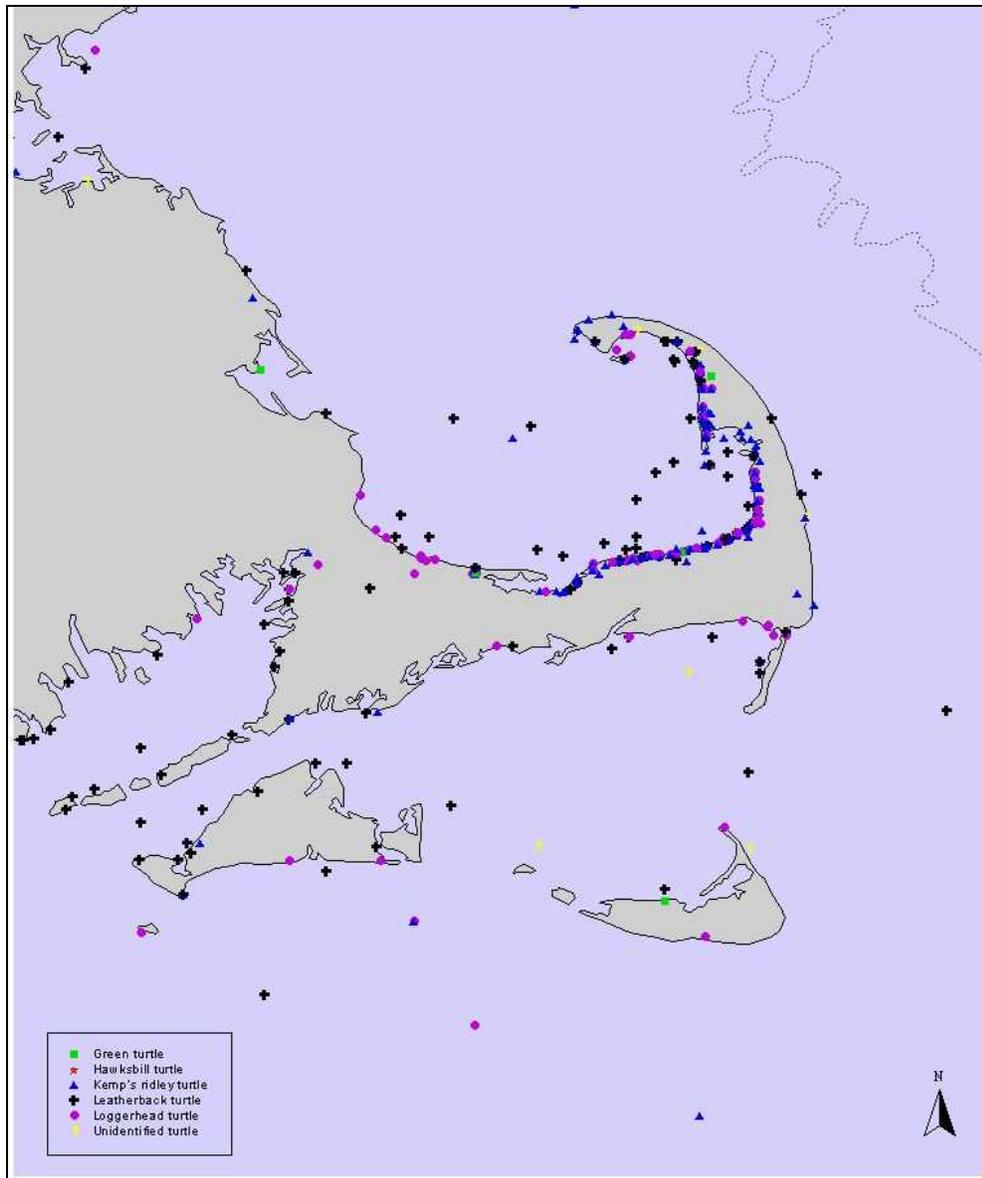


Figure 5. Sea turtle strandings on Cape Cod, Nantucket, and Martha's Vineyard Areas for the period 1980 to 1997 (NMFS unpublished data).

Ingestion of or entanglement in plastic debris contributes to the death of many loggerhead turtles each year, though the magnitude of this mortality is difficult to estimate (NRC 1990). One study found that ten percent of 33 necropsied loggerheads that stranded in the New York Bight contained ingested synthetic materials, mostly plastics (Sadove and Morreale 1990). Loggerheads in the New York Bight become entangled most frequently in pound nets and lobster pot lines. Of 22,547 sea turtles (72.4% of them loggerheads) stranded on shores of the Atlantic and Gulf of Mexico coasts of the United States between 1980 and 1992, 676 (3%) were affected in some way by debris (Witzell and Teas 1994). Of the different species of sea turtles, loggerheads were least affected by entanglement; when entanglement occurred, it most frequently involved monofilament lines with fish hooks, fishing nets, and rope. More than 40 loggerheads stranded along the south Atlantic coast of the United States had ingested monofilament lines or hooks; a few had ingested plastic or balloons. Fourteen loggerheads stranded on the south Atlantic coast had ingested or become contaminated with oil or tar balls.

There is little published information on the role or importance of chemical pollution in the mortality of sea turtles, including loggerhead turtles. Stoneburner *et al.* (1980) reported that loggerhead eggs collected from the shores

of Florida, Georgia, and North Carolina contained 0.41 to 1.39 mg/kg dry wt. total mercury. Several loggerheads collected south of Malta in the Mediterranean Sea were contaminated with tarry residues of petroleum, probably derived from encounters with floating tar balls, which the turtles often mistake for food (Gramentz 1988). Juvenile loggerhead and Kemp's ridley turtles that stranded in Virginia and North Carolina contained 55 to 1,730 µg/kg wet wt. total organochlorines in subcutaneous fat, and 7.5 to 607 µg/kg wet wt. total organochlorines in liver tissue (Rybitski *et al.* 1995). The most abundant organochlorines were polychlorinated biphenyls (PCBs) and DDT and its degradation products.

4.5 Kemp's Ridley Turtle (*Lepidochelys kempii*)

The Kemp's ridley (*Lepidochelys kempii*) sea turtle is distributed throughout coastal areas of the Gulf of Mexico and the northwestern Atlantic Ocean, and is assumed to constitute a single stock (TEWG 1998). A small sea turtle, adult Kemp's ridley females have carapace lengths of 60 to 70 cm and weigh 35 to 45 kg (NRC 1990). Most ridleys that visit the east coast of the United States are juveniles, averaging 25 to 30 cm long and weighing about 3 kg (NMFS 1988; NOAA 1991). Female ridleys reach sexual maturity when they reach a carapace length of about 58 to 60 cm and are six to nine years old (Márquez 1994). The mature females nest annually and produce one to three (average about 1.7) clutches per season containing a total of about 120 to 190 eggs. Longevity probably is greater than 20 years. Little is known about the sex ratio of ridley turtles or about the life history of the males. Nearly all reproduction takes place along a single 15-km stretch of beach near Rancho Nuevo, Mexico, about 322 km south of Brownsville, Texas. Nesting occurs in a highly synchronized manner with large numbers of females coming ashore within a period of a few daylight hours (NRC 1990).

4.5.1 Population Status and Trends

The Kemp's ridley turtle is the most endangered sea turtle in the world. The entire Atlantic population, consisting almost exclusively of juveniles, probably does not exceed 500 individuals (Carr and Mortimer 1980). The total world population of adult ridleys, mostly in the Gulf of Mexico, is approximately 2,200 individuals, down from an estimated 162,400 adult individuals in 1947 (Márquez 1989). The total population, adults and juveniles, may number 22,000 to 110,000 individuals. The total nesting population of females during the mid- to late-1980s has been estimated to number about 600 individuals, with each female laying about two clutches of eggs per year (Pritchard 1990). When compared to the estimated number of nests in 1947 (92,000), this is the most severe population decline documented for any species of sea turtles (NRC 1990).

In 1947, an estimated 40,000 females nested in one day on the Rancho Nuevo beach. Only 842 nests were found in 1988 (Ross *et al.* 1989). The number of nesting females declined at a rate of 3% per year from 1978 to the mid-1980's (Thompson 1988). However, the number of nests observed at Rancho Nuevo has increased at a mean rate of 11.3% per year from 1985 to 1999, and currently exceed 3000 nests per year (TEWG 2000). In-water surveys also indicate increasing numbers of ridleys (TEWG 2000). A multi-year time lag exists between the release of new hatchlings and the appearance of neophyte nesters; thus the annual number of nesting females or annual number of nests will not reflect this apparent population increase for several years (TEWG 1998, 2000).

4.5.2 Seasonal Distribution in North Atlantic Waters

The Kemp's ridley sea turtle is found mainly in the Gulf of Mexico (Hildebrand 1982), but juveniles also occur during the summer along the Atlantic seaboard from Florida to Long Island Sound, Martha's Vineyard, and occasionally north of Cape Cod, in Cape Cod Bay, Massachusetts Bay, the Gulf of Maine, and as far north as the Canadian Maritime Provinces (Lazell 1980). Groups of dozens of young ridleys are observed frequently during the summer feeding in shallow coastal waters of Vineyard Sound, Buzzards Bay, MA, and in the eastern Bays of Long Island, NY (Carr 1967; Lazell 1980; Morreale and Standora 1989).

Hatchlings and young juveniles from the western Gulf of Mexico are thought to drift east in the Gulf gyres and become caught in the eastern Gulf Loop Current. They are then carried by the Florida Current through the Straits of Florida into the Gulf Stream, which transports them up the eastern seaboard of the United States (Collard 1987; Márquez 1994). They may be carried around the entire North Atlantic in the circular gyre of the Gulf Stream before swimming into shallow coastal waters along the U.S Atlantic coast. When they move into coastal waters of New England, the juvenile ridleys are 24 to 30 cm long. They forage in shallow coastal waters of New

England, New York, and New Jersey during the spring and summer and then migrate to southern waters in the fall.

Turtles that were tagged off Cape Canaveral, FL, migrated north in the spring as water temperatures increased and south in the fall as water temperatures dropped (Henwood and Ogren 1987). The longest recorded northward migration was about 880 km. Three juvenile ridleys that were tagged and released in the fall at Virginia Beach, VA migrated south in nearshore waters (Keinath *et al.* 1992). One turtle got as far south as Cape Canaveral, FL, before the transmitter stopped. Two ridleys that were tagged in eastern Long Island were subsequently recaptured in coastal waters of North Carolina (Morreale and Standora 1989). A juvenile ridley turtle from eastern Long Island was tracked by satellite telemetry as it traveled south in the fall of 1991 (Morreale and Standora 1989). It traveled westward along the south shore of Long Island and then continued southwestward toward the New Jersey coast. It then swam directly south until it reached the coast of North Carolina. The entire trip of 709 km was performed within 60 km of shore in water depths less than about 40 m. The average swimming speed was 22 km/day. These studies show that ridley turtles that visit New England waters during the summer to feed are a part of the larger population centered in the Gulf of Mexico. These animals, despite their small size can migrate back to southern waters and are not lost to the breeding population, as had been previously suggested (Carr 1980).

All the ridley turtles in New England waters are two- to five-year old juveniles with carapace lengths of 22 to 38 cm (Burke *et al.* 1989; Morreale and Standora 1989). They begin arriving in northern waters in July or August each year and remain in shallow nearshore waters, particularly in the bays on eastern Long Island, during the summer (Burke *et al.* 1989; Morreale and Standora 1989). They begin leaving the area in mid-September and most have left for warmer southern waters by the beginning of November. Some ridleys may hibernate over the winter in nearshore sediments (Carminati *et al.* 1994). Most of the ridleys observed after the beginning of November are cold-stunned. Ridleys become sluggish and have labored breathing when the temperature falls below 13°C; feeding ceases below 10°C, and they die when water temperatures reach between 6.5 to 5.0°C (Schwartz 1978).

4.5.3 Food and Feeding Behaviors

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

In coastal waters of New York and New England, young ridleys consume several species of crabs, including in order of decreasing preference, spider crabs (*Libinia emarginata*), lady crabs (*Ovalipes ocellatus*), and rock crabs (*Cancer irroratus*) (Morreale and Standora 1992, 1989). Crustaceans represent more than 80% of the diet of juvenile ridleys in the New York Bight (Burke *et al.* 1994). Other food items found in ridley stomachs include mollusks and algae. The preference for spider crabs over lady crabs, despite the fact that the latter is more abundant in ridley foraging habitat, is probably due to the greater ease of capture of the slower moving spider crabs by the small turtles (Morreale and Standora 1989).

In Chesapeake Bay, sub-adult ridleys concentrate in seagrass (*Zostera* and *Ruppia*) beds and feed primarily on blue crabs (*Callinectes sapidus*) and cancer crabs (*Cancer* spp.) (Lutcavage 1981; Byles 1989). Juvenile to adult ridleys stranded on Texas beaches contained a wide variety of foods in their digestive tracts; crabs were most abundant, followed by mollusks and small fish (Shaver 1991). More than 60% of the turtles contained some plant materials in their stomachs, but it represented less than one percent of the total gut contents.

Juvenile and sub-adult ridleys in Florida and Georgia were observed to feed on the crabs *Ovalipes ocellatus* and *Heppatus ephilliticus* (De Sola and Abrams 1933; Carr 1952). Blue crabs (*Callinectes sapidus*) are the favorite food of sub-adult ridleys in Virginia (Hardy 1962; Musick 1979). Because of their preference for crabs and other primarily shallow-water demersal prey, juvenile and adult ridley turtles concentrate in coastal waters less than 100 m deep throughout their range (Thompson 1988).

Ridley turtles make long dives to the bottom and may feed on the bottom for an hour or more at a time; one turtle was observed burrowing in the bottom of Long Island Sound (NMFS 1988). During daylight hours, ridleys spend most of their time under water. In a typical dive the turtle spends about 56% of its time in the upper third of the water column, 12% in the middle, and 32% of its time on the bottom (Morreale and Standora 1989). In water deeper than about 15 m, the turtles usually dive to a depth of six to 10 m where they appear to be swimming in a directed manner.

In New York and Southern New England waters, ridley turtles spend most of their time feeding in shallow water and growing rapidly. It is assumed that ridley turtles visiting Southern New England waters are part of the group that migrates into New York waters in search of food. Individuals 20 to 30 cm long grow at an average rate of 2.2 cm/year; larger 30 to 40 cm individuals grow at a rate of 4.5 cm/year. These rates are slower than those for ridleys in Texas (McVey and Wibbles 1984), possibly because New York ridleys are able to occupy optimal foraging areas for only a few months each year. A growth model proposed by Márquez (1972) indicates that ridleys may reach a length of 40 cm after about four years and reach sexual maturity at a carapace length of about 60 cm after about six or seven years.

4.5.4 Known Disturbance and Mortality Factors

Several stages in the life cycle of Kemp's ridley turtles are sensitive to natural and anthropogenic disturbance. Each year between November and January when ocean water temperatures are falling, small numbers of ridley turtles become stranded and die on beaches of the north and east shores of Long Island and Cape Cod Bay, due to cold stunning (NOAA 1991; Morreale and Standora 1992). When the water temperature drops below about 12°C, the metabolic rate of these cold-blooded reptiles decreases to the point where they are unable to swim and digest food; they become comatose and may die if not warmed quickly. A total of 115 ridley turtles stranded on Cape Cod beaches between 1977 and 1987 (Danton and Prescott 1988). While ridley strandings are common on Cape Cod beaches, no ridleys strandings were reported in Nantucket Sound for the period 1980 to 1997 (see Figure 5) (NMFS, unpublished data).

For the period of 1990 to 2000, between nine and 216 ridleys strandings were reported in Massachusetts waters, and one ridley stranding was reported in Rhode Island waters (Table 2) (Sea Turtle Stranding and Salvage Network, unpublished data). Cold stunned ridleys have stranded as far south as the Indian River Lagoon, FL (Wilcox 1986).

Table 2. Strandings of Kemp's ridley turtles (*Lepidochelys kempii*) in Massachusetts and Rhode Island from 1990 to 2000, all months combined each year (Sea Turtle Stranding and Salvage Network, unpublished data).

State	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000
MA	43	13	9	36	23	101	9	29	36	216	34
RI	-	-	-	-	-	-	-	-	1	-	-

A major cause of sea turtle mortality attributable to humans is entanglement in fishing gear, particularly shrimp nets (NRC 1990). Henwood and Stuntz (1987) estimated an annual incidental capture of approximately 47,000 sea turtles of all species, with an estimated mortality of about 11,000 individuals in the shrimp fisheries of the Gulf of Mexico and Atlantic coastal waters from Florida to North Carolina. These estimates are thought to be conservative (NRC 1990). Of all the turtles killed each year by U.S. commercial shrimping, 500 to 5,000 are juvenile and adult Kemp's ridley turtles. Most of the mortalities attributable to entanglement in shrimp nets are in the Gulf of Mexico.

Other fishing-related deaths, caused by entanglement in lobster gear (O'Hara *et al.* 1986) and pound nets (Morreale and Standora 1989), may result in an additional 50 to 500 deaths of Kemp's ridley turtles each year. Ridley turtles, being benthic feeders, tend to become entangled in debris, including abandoned fish and crab traps, on the bottom. This incidental catch could represent as much as 7.5% of the hatchling ridleys produced

each year, assuming that the 800 nests produced a total of 80,000 hatchling ridley turtles each year. This additional mortality undoubtedly is contributing to the rapid decline in the population of Kemp's ridley turtles.

Large numbers of sea turtles, including some Kemp's ridley turtles, die from eating or becoming entangled in plastic debris (O'Hara 1989; NRC 1990). Plastic bags and plastic particles are the most common forms ingested; they probably are mistaken for food. Sea turtles are also particularly prone to becoming entangled in monofilament fishing line and phantom fishing nets (Balazs 1985).

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

Ridley turtles that were cold-stunned on eastern Long Island between 1980 and 1989 contained mean concentrations of total PCBs ranging from 218 to 738 µg/kg, mean concentrations of DDT ranging from 156 to 300 µg/kg, and mean concentrations of trans-nonachlor ranging from 27.5 to 86 µg/kg in their livers (Lake *et al.* 1994). These concentrations were much higher in body fat. There was a general trend for mean concentrations of these organochlorines in the turtle tissues to decrease between 1980 and 1989. The concentrations of these compounds in ridleys from Long Island were higher than those found in ridley and loggerhead turtles from southern Chesapeake Bay (Rybitski *et al.* 1995) and were higher than concentrations known to cause reproductive effects in snapping turtles.

4.6 The Leatherback Turtle (*Dermochelys coriacea*)

Leatherback turtles (*Dermochelys coriacea*) forage in temperate and subpolar waters and nest on tropical beaches. The leatherback is the second most common turtle along the eastern seaboard of the United States, and the most common north of the 42°N latitude. Leatherback turtles are the largest and most distinctive of the living sea turtles. Because of their unique anatomy and physiology, they are classified in a separate family, the Dermochelyidae, containing a single species (NMFS & USFWS 1992). Leatherbacks reach a length of 150-170 cm SLCL and a weight of 500 and exceptionally 900 kg. Large outstretched front flippers may span 270 cm in an adult. Lacking a keratinized shell, they are covered instead with a tough hide. Leatherbacks have a layer of subcutaneous fat that is 6-7 cm thick and circulatory adaptations to reduce the rate of heat loss through the flippers (Greer *et al.* 1973). They respond to drops in ambient temperature by increasing metabolic heat production and so can maintain an internal body temperature well above ambient (Standora *et al.* 1984; Paladino *et al.* 1990). A leatherback in 7.5° C seawater was able to maintain its core body temperature at 25.5° C (Friar *et al.* 1972). This endothermy allows leatherbacks to survive and feed in colder temperate waters than other sea turtles can tolerate. Therefore, adult leatherbacks are more widely distributed than other sea turtles in temperate and boreal waters throughout the world. However, all leatherbacks return to subtropical and tropical shores to nest.

4.6.1 Population Status and Trends

The leatherback sea turtle is listed as endangered throughout its range (USFWS 1986). Between 100 and 900 leatherbacks visit coastal and continental shelf waters of the western North Atlantic ocean between Canada and North Carolina each year, with peak abundance in summer (Shoop and Kenney 1992). As many as 115,000 adult female leatherbacks remain worldwide (Pritchard 1982), though a more recent estimate places the adult female population at 34,500 (Spotila *et al.* 1996). Spotila *et al.* (1996) estimate that between 7,813 and 13,833 female leatherbacks visit the largest Atlantic nesting colonies in French Guiana, Suriname, other locations in the Caribbean, and Gabon, West Africa each year.

Because they are a largely oceanic, pelagic species, estimates of their population status and trends have been difficult to obtain. In addition, only a small fraction of the North Atlantic population nests on beaches of the continental United States, mostly in Florida (National Research Council 1990; Meylan *et al.* 1994) and the U.S. Virgin Islands (Boulon *et al.* 1994). Leatherbacks that visit U.S. Atlantic waters

nest primarily along the coasts of Surinam and French Guiana, and to a lesser extent on the island of St. Croix and at Culebra, Puerto Rico (National Research Council 1990; NMFS & USFWS 1992; Boulon *et al.* 1994). Nesting is scattered along isolated beaches throughout the Caribbean. Nesting females do not have the nest-site fidelity exhibited by Kemp's ridley turtles and tend to move to different beaches in different years (Tucker 1990). Therefore, it has been difficult to estimate temporal trends in population size.

Between ten and 188 leatherback nests are reported each year along the Atlantic coast of Florida (NMFS & USFWS 1992; Meylan *et al.* 1994). Between ten and 25 female leatherbacks account for all the nests deposited each year along the Atlantic coast of Florida (NMFS & USFWS 1992). Nesting in Florida is widespread but erratic amongst year and location. Most of the remaining leatherback nesting on United States shores occurs in the U.S. Virgin Islands and in Puerto Rico, including the small islands of Culebra, Vieques, and Mona (NMFS & USFWS 1992). Between 50% and 70% of the total nesting on St. Croix occurs at Sandy Point (NMFS & USFWS 1992). Between 18 and 55 leatherback turtles nest each year at Sandy Point, a 3-km beach on St. Croix (Boulon 1992; Boulon *et al.* 1994). Because of the importance of Sandy Point, St. Croix for leatherback nesting, it has been designated as critical habitat for leatherback turtles (NMFS 1994). There is one record of a leatherback turtle nesting on a North Carolina beach (Pritchard 1989). Nesting data from St. Croix and Florida indicate increasing numbers of nests over the past twenty years (approximately 10.3% and 7.5%, respectively), but nesting in Suriname and French Guiana appears to be on the decline (NMFS-SEFSC, 2001). The nesting aggregation in French Guiana is potentially the largest in the world, and the number of nests has declined approximately 15% per year since 1987 (NMFS-SEFSC, 2001).

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

4.6.2 Seasonal Distribution in North Atlantic Waters

Leatherback turtles are common during the summer in North Atlantic waters from Florida to Massachusetts, the Canadian Maritime Provinces, and occasionally as far north as Baffin Island, Canada (Goff and Lien 1988). New England and Long Island Sound waters support the largest populations on the Atlantic coast during the summer and early fall (Lazell 1980; Prescott 1988; Shoop and Kenney 1992). Leatherbacks are observed frequently in lower Chesapeake Bay and off the mouth of the Bay during the summer, where they probably are feeding on locally abundant jellyfish (Barnard *et al.* 1989).

Leatherbacks are rarely sighted north of Cape Hatteras during the winter. Three leatherbacks were sighted in Core Sound, just south of Cape Hatteras, in December 1989 (Epperly *et al.* 1992). In some years, they are abundant in nearshore waters off the East Coast of Florida. Knowlton and Weigle (1989) reported sighting 168 leatherbacks in coastal waters between Sebastian Inlet and St. Augustine, Florida in February 1988. During most of the year, they are pelagic and remain far offshore in oceanic waters. However, periodically, especially during the summer, they may come relatively close to shore pursuing their jellyfish prey (Lee and Palmer 1981).

Leatherback turtles nest on tropical beaches, after which the adults move into temperate waters to feed. Most leatherbacks that visit New England waters are adult males, usually >150 cm and weighing >450 kg (NOAA 1991). Adults migrate extensively throughout the Atlantic basin in search of food. There are numerous records of leatherback turtles in New England, and as far north as Nova Scotia and Newfoundland (Goff and Lien 1988). Sightings off Massachusetts are most frequent in the late summer months (Shoop *et al.* 1981; CeTAP 1982; Shoop and Kenney 1992).

In the spring, following breeding and nesting in the tropical Caribbean and Florida, and aided by the northward flow of the Gulf Stream, leatherback turtles move northward beyond the shelf break. For this reason, there are few sightings of leatherbacks in coastal and outer continental shelf waters in the spring months (CeTAP 1982). They appear in offshore waters of the mid-Atlantic states and in the Gulf of Maine in late May and June, and in shelf waters from June through October (Shoop *et al.* 1981; Shoop and Kenney 1992). In New England waters, they are seen most frequently in the southern Gulf of Maine, including Cape Cod and Massachusetts Bays.

Leatherbacks occur most frequently in coastal waters of Newfoundland in August and September when water temperatures are at their highest (Goff and Lien 1988).

During summer months, leatherbacks move into fairly shallow coastal waters, apparently following their preferred jellyfish prey. In the fall, they move offshore and begin their migration south to the winter breeding grounds in the Caribbean (Payne *et al.* 1984). Leatherbacks may travel great distances between nesting and feeding areas. Tagging studies have shown that some of the leatherbacks that visit New England waters nested in the U.S. Virgin Islands and along the southern coast of the Caribbean or in the Guianas (Boulon 1989; National Research Council 1990). A 157-cm leatherback found entangled in fishing nets near Fox Harbor, Newfoundland, on 17 September 1987, bore a tag indicating that it had migrated 5000 km from French Guiana, South America, in 128 days at an estimated speed of at least 39 km/day (Goff *et al.* 1994).

4.6.3 Food and Feeding Behaviors

Leatherback turtles are pelagic feeders, though they can dive to considerable depths. They feed throughout the water column to depths of at least 1000 m (Eckert *et al.* 1989) on jellyfish and other gelatinous zooplankton, such as salps, ctenophores, and siphonophores (Limpus 1984). Most feeding dives average about 60 m, but frequently extend to 300-400 m (Eckert *et al.* 1986, 1989) where they feed on deep-water gelatinous zooplankton, such as siphonophores and salps. Their seasonal inshore movements in New England waters have been linked to inshore movements of their preferred prey, the jellyfish *Cyanea capillata* (Lazell 1980; Payne and Selzer 1986).

Leatherbacks have a notched upper jaw, an adaptation for grasping soft prey (Pritchard 1971). They also possess a long digestive tract, about nine times longer than the length of the carapace, and a large caecum for holding the quantities of watery, gelatinous prey they need to consume to fulfill their caloric needs (Bjorndal 1985).

4.6.4 Known Disturbance and Mortality Factors

Many of the same natural and anthropogenic factors that affect survival of loggerhead and Kemps ridley turtles also affect leatherbacks. In 1987 and 1988, 119 and 63 leatherbacks, respectively, stranded along the United States coast (National Research Council 1990). Most of the strandings occurred along the coasts of Delaware, New Jersey, and New York. There was only one stranding in New England. The cause of death of most of these turtles was not known. Being a temperate water species, leatherbacks do not seem to be sensitive to cold temperatures, and strandings can not be attributed to cold stunning.

Between 1986 and 1999, 42 to 170 leatherback turtles were reported stranded on the U.S. Atlantic coast each year. Most strandings were in Florida and New York. Between four and 39 leatherbacks stranded each year in Massachusetts and Rhode Island waters during the period of 1990 to 2000 (Table 3) (Sea Turtle Stranding and Salvage Network, unpublished data). For the period of 1980 to 1997, twelve leatherback strandings were recorded in Nantucket Sound (Figure 5) (NMFS, unpublished data). The causes of these strandings are not known, but entanglement in fishing gear may be a major factor.

Table 3. Strandings of leatherback turtles (*Dermochelys coriacea*) in Massachusetts and Rhode Island from 1990 to 2000, all months combined each year (Sea Turtle Stranding and Salvage Network, unpublished data).

State	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000
MA	5	5	8	5	1	7	5	11	7	30	1
RI	2	11	9	13	12	25	8	5	3	9	3

Approximately 650 leatherbacks are entangled in commercial shrimp nets each year (NMFS-SEFSC, 2001). Leatherbacks are also very susceptible to entanglement in other fishing gear and in plastic debris (Mager 1985; Witzell and Teas 1994). Because they are adapted to a pelagic existence, leatherbacks have trouble swimming backwards, maneuvering in tight places and, avoiding obstructions in shallow waters (Payne and Selzer 1986; NOAA 1991). In January and February 1992, a leatherback turtle became entangled and died in a summer flounder trawl south of Cape Hatteras (Epperly *et al.* 1995b). Leatherbacks have also been entangled in lobster

gear (O'Hara *et al.* 1986; Sadove and Morreale 1990) and long-lines (Balazs 1985) in New York Bight and New England waters. In 1992, 50 leatherbacks were taken in the long-line fishery between Cape Hatteras and the Grand Banks (Brady and Boremen 1994). An estimated 356 leatherbacks were captured in 1992 and 242 were captured in 1993 in the entire long-line fisheries for tuna and swordfish in the western North Atlantic Ocean (Witzell and Cramer 1995). Each year, during the period of 1992 to 1999, approximately 308 to 1054 leatherbacks become entangled in fishing gear in the U.S. longline fishery, though not all of these entanglements may be fatal (NMFS-SEFSC, 2001). Records from the Sea Turtle Stranding and Salvage Network show that 45 leatherback turtles became entangled in lobster gear between 1983 and 1993 in coastal waters of New Jersey, New York, and southern New England (NMFS 1994). Eleven of the entangled turtles died. The leatherbacks large front flippers (often 1 m long) often bear cuts, chafing marks, or are severed altogether, possibly due to entanglement (Fretey 1982).

Because of their preferred diet of gelatinous zooplankton, particularly jellyfish, leatherback turtles often ingest floating plastic debris, mistaking it for food (Wallace 1985; O'Hara 1989). Plastic bags blocked the stomach openings of 11 of 15 leatherbacks that washed ashore on Long Island during a two-week period (Balazs 1985). The largest leatherback ever recorded washed ashore on the coast of Wales, dead in tangled fishing gear and with a large piece of plastic blocking the entrance to its small intestine (Eckert and Eckert 1988). Subsistence harvesting also places pressure leatherback populations. Although leatherbacks are not harvested commercially for meat or other products, extensive subsistence harvesting of the females that come ashore to nest occurs throughout much of the tropical nesting range, including Guyana, Trinidad, and Columbia (National Research Council 1990). Egg collecting is also intense in some areas.

5 POTENTIAL IMPACTS TO PROTECTED SPECIES

5.1 Summary of Construction, Operation/Maintenance and Decommissioning Methodology

5.1.1 Project Construction

The construction of the Project will involve the installation of 130 wind turbine generators (WTGs) in Nantucket Sound, an Electrical Service Platform (ESP) within the WTG array, inner-array cables to connect each WTG to the ESP, and two submarine cable circuits to connect the ESP to the landfall area in Yarmouth, Massachusetts. One monopile foundation will be constructed to support each of the 130 WTGs and six smaller monopile foundations will support the ESP. The monopiles will be installed using pile driving hammer technology and will be driven approximately 85 feet into the seabed. To prevent scour around the monopiles, seabed scour control systems will be installed. These systems consist of mats of seagrass-like polypropylene "fronds" that serve to reduce the velocity of water circulation around the foundations, thereby preventing scour at the base of the monopiles (please refer to Section 4.0 of the DEIS-DEIR and Appendix 4.0-A for more a detailed discussion on scour control mats). Anchors and jack-up barges will be used to facilitate the installation of the monopiles.

The two submarine cable circuits connecting the Wind Park to the landfall location and the inner-array cables connecting each WTG to the ESP will be installed in the seabed using hydraulic jet-plow embedment technology. This method utilizes pressurized water jets to create a localized path along the seafloor into which the cable system is immediately positioned. The sediment displaced by the jet-plow then begins to settle over the created path, thereby burying and protecting the cable. The localized pathway disturbed to install each circuit will be approximately four to six feet wide and eight feet deep to reach an approximate 6 foot burial depth. Additionally, anchoring will be required for cable installation barge positioning.

The transition of the interconnecting submarine cable system from water to land will be accomplished through the use of Horizontal Directional Drill (HDD) methodology in order to minimize disturbance within the intertidal zone and near shore area. HDD would be staged at the upland landfall area and 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 line would be pulled through the conduits from the seaward end toward the land.

The offshore end of the conduits will terminate in a pre-excavated pit where the jet plow cable burial machine will start. To further facilitate the HDD operation, a temporary cofferdam will be constructed using steel sheet piles

at the end of the boreholes. Approximately 840 cubic yards of sediment will be excavated from the area inside the cofferdam to expose the seaward end of the borehole. The top of the sheet piles will be cut-off approximately 2 feet above mean high water to contain any turbidity associated with the dredging. The excavated material will be disposed of at an approved upland disposal location. The area enclosed by the cofferdam will be approximately 2,925 square feet, a minimal area compared to surrounding habitat in Lewis Bay. See Section 4.0 of the DEIS for more detailed information on construction and installation methodologies.

5.1.2 Project Operation/Maintenance

As previously discussed, the Project will consist of 130 WTGs on Horseshoe Shoal in Nantucket Sound. Wind-generated energy produced by the WTGs will be transmitted via a 33 kV submarine transmission cable system (inner-array cables) to the Electric Service Platform (ESP) centrally located within the WTG array. The ESP will then take the wind-generated energy from each of the WTGs and transform and transmit this electric power to the mainland electric transmission system via two 115 kV alternating current (AC) cable circuits.

Maintenance required for the 130 WTGs would be distributed among two to three crews, thus likely resulting in daily trips to the Wind Park estimated to be at least 250 days per year. In the event that a WTG or a section of the inner-array or submarine cable systems require repair during operation, methodologies for conducting this repair are expected to be similar to those used during construction; however, impacts would be limited to the immediate vicinity of the WTG or portion of the cable system requiring repair. The maintenance program will include preventive and emergency maintenance functions including shore based predictive maintenance analysis of the WTG and ESP. These visits cover two days of planned or preventative maintenance, and three days of unplanned or forced outage emergency maintenance. See Section 4.0 of the DEIS for a more detailed description of operation and maintenance procedures.

5.1.3 Project Decommissioning

The approximate design life of the Project is 20 years, after which the decommissioning of the Project will occur. Decommissioning the Project involves dismantling the WTGs and ESP, removing scour control mats, removing the inner-array cables and submarine cable system, and transporting all parts to shore for recycling. In deconstructing the WTGs down to the transition piece, the blades, hub, nacelle and tower would come apart in the same manner that they were put together utilizing similar equipment. The parts would be brought to shore for reuse or recycling. The monopile, with the transition piece, would be cut off at the mud line followed by the removal of the sediment within it to a suitable depth (approximately 6.5 feet (2 meters) below the level of the seabed). Once the sediments have been removed, the remaining monopile would be cut off at a depth of approximately 6.5 feet (2.0 meters) below the surface. The monopile would be placed on a barge and brought to shore for recycling.

Decommissioning for the ESP will be a reverse process of the construction activities and will commence when the 33 kV and 115 kV cables have been disconnected and removed from the ESP. The heliport, ladders and boat platform will be removed by cutting and placed on a barge. The superstructure will then be lifted onto a vessel and moved to port. The balance of the jacket structure will be cut from the piles and lifted out of the water for placement on barges. The piles will be cut below the sand line and removed.

During decommissioning, the submarine cables will be disconnected and pulled out of the J-tubes on both the WTG and the ESP, and the cables will be cut below the seafloor. The cables will then be reeled in after being water jetted free of the bottom sand. The reels will be transported to the staging area for further handling. It is expected that all metal from the cable will be reused via recycling. The equipment used to remove the submarine cables will be similar to that used for installation (barge, attendant tugs and jet plow equipment). The objective of the decommissioning process will be to return the Project Area to its pre-Project state (see Section 4.0 of the DEIS for a complete discussion of the decommissioning process). Following decommissioning, there should be no interferences with normal uses of the region nor should there be any adverse environmental impacts.

5.2 Analysis of Impacts

This section discusses potential water quality, biological, and physical impacts to the protected whale and sea turtle species that could occur during construction, operation/maintenance, and decommissioning of the Project. The level of effect - direct, indirect, or cumulative - is indicated for each impact.

5.2.1 Total Suspended Solids

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

Construction activities associated with installing the monopile foundations, scour control mats, and submarine cables will result in a temporary and localized increase in suspended sediment concentrations. Decommissioning-related impacts will be short-term and localized and are expected to be similar to or less than impacts during construction. The pile driving hammer and jet plow technology that will be used to install the monopile foundations and the submarine cables, respectively, were selected specifically for their ability to keep sediment disturbance to a minimum. Due to the predominant presence of fine to coarse-grained sands in Nantucket Sound, localized turbidity associated with Project construction or decommissioning is anticipated to be minimal and confined to the area immediately surrounding the monopiles and the submarine cable route. Sediments disturbed by construction or decommissioning activities are expected to settle back to the sea floor within a short period of time (one to two tidal cycles). In addition, the Project Area is situated in a dynamic environment that is subject to naturally high suspended sediment concentrations in near-bottom waters as a result of relatively strong tidal currents and wind and storm generated waves, particularly in shoals areas. Therefore, marine organisms in this area are accustomed to substantial amounts of suspended sediment on an irregular basis and should not be substantially impacted by a temporary increase in turbidity from Project activities.

Sediment suspension during construction and decommissioning activities will not result in long-term or environmentally significant elevations in water column TSS. Zooplankton or fish species may be temporarily affected or displaced in the immediate vicinity of the area of the activity; however, they are likely to rapidly return to these areas once construction in the specific area is ceased or completed. Whales and sea turtles that may be present in the vicinity of the Project Area during construction are not expected to be adversely affected by temporary increases in TSS and since they are mobile, are capable of avoiding or moving away from the disturbances associated with construction.

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

5.2.2 Sediment Contaminants

Sea turtles and whales bioaccumulate contaminants from their ocean environment almost exclusively through their food sources. As air-breathing animals, sea turtles and whales have an outer integument that is much less permeable than the gills of fish or the body wall of most marine invertebrates. Therefore, passive or active uptake of chemical contaminants directly from solution across permeable body surfaces is much less important than bioaccumulation of chemical contaminants from food. The potential mechanism by which sediments suspended during project construction and decommissioning can harm sea turtles and whales is through bioaccumulation of sediment-associated chemicals through ingestion of contaminated prey (indirect impact).

There is little potential for whales and sea turtles to bioaccumulate chemical contaminants in their tissues from consuming prey in the Project Area because analysis of sediment core samples obtained from the Project Area indicated that sediment contaminant levels were below established thresholds in reference sediment guidelines.

Specifically, all of the chemical constituents detected in the sediment core samples obtained from the WTG site and along the submarine cable route had concentrations below Effects Range-Low (ER-L) and Effects Range-Median (ER-M) marine sediment quality guidelines (Long et al., 1995) (see Section 5.1 of the DEIS). Therefore, the temporary and localized disturbance and suspension of these sediments during project construction and decommissioning activities is not likely to result in increased incorporation of contaminants at low trophic levels. Sea turtles or whales are unlikely to experience increased bioaccumulation of chemical contaminants in their tissues from the consumption of prey items in the Project vicinity.

During the nearshore installation, the release of contaminants from the HDD operation within Lewis Bay will be minimized through a drilling fluid fracture or overburden breakout monitoring program. The drilling fluid will consist of water (approximately 95%) and an inorganic, bentonite clay (approximately 5%). The bentonite clay is a naturally occurring hydrated aluminosilicate composed of sodium, calcium, magnesium, and iron. This program will minimize the potential of drilling fluid breakout into waters of Lewis Bay. Although it is anticipated that drilling depths in the overburden will be sufficiently deep to avoid pressure-induced breakout of drilling fluids through the seafloor bottom, a bentonite monitoring program will be implemented for the detection of possible fluid loss (see Section 4.3.5 of the DEIS). In the unlikely event of drilling fluid release, the bentonite fluid density and composition will cause it to remain as a cohesive mass on the seafloor in a localized slurry pile similar to the consistency of gelatin. This cohesive mass can be quickly cleaned up and removed by divers and appropriate diver-operated vacuum equipment; thereby minimizing any long-term impacts to protected marine species. Short-term impacts would consist of the covering of benthic organisms in the immediate area of release.

Operations and maintenance of the WTG will involve the utilization of lubricating oil, cooling liquids, and grease, all of which will be located in the nacelle or hub. The WTG has been carefully configured to contain any fluid leakage and prevent overboard discharges. The possibility of leaks may occur in two different situations: (1) during service and maintenance and (2) during operation. During the servicing and maintenance of a WTG, a spill could happen during oil changes of hydraulic pump units or the gearbox oil conditioning system. During WTG operation leakage may occur as the result of broken gear oil hoses/pipes, and / or broken coolant hoses/pipes. Gear oil leaks will be contained within the hub and main bed frame and/or tower as described above. Coolant leaks can occur on a number of locations within the nacelle and will be contained inside the nacelle fiberglass cover.

In order to minimize and mitigate any minor spill incidents, all service vessels will be equipped with oil spill handling equipment. In addition, waste collection systems will be installed on board each WTG. The waste collection system is based on a container system for easy and safe handling during transfer from/to turbine-service vessel-dock. The waste will be separated (i.e., used oil, coolant liquids, filters, paper/rags, etc.) for correct disposal once the containers are off-loaded at the dock.

The ESP will have small amounts of lubricating oil, greases and coolants in pumps, fans, air compressors, emergency generators and miscellaneous equipment plus diesel fuel. The ESP will also have four oil-cooled step up transformers. The ESP will have sealed, leak-proof decks, which will act as fluid containment. In addition, spill containment kits will be available near all equipment. The details of spill containment equipment and related spill control measures will be provided in Spill Prevention Control and Countermeasure (SPCC) Plan prior to operation of the facility.

Due to the amount of oil, required for the WTGs and the ESP, the Project requires development of an SPCC Plan according to 40 CFR 112. The SPCC Plan will address all applicable components of the Project including the WTGs and the ESP.

For more details on WTG and ESP fluid containment and the SPCC plan, please refer to Sections 4.6 and 4.7 of the DEIS-DEIR.

In open water, marine organisms such as fish and whales have the ability to swim away from a spill by going deeper in the water or further out to sea, reducing the likelihood that they will be harmed by even a major spill. Marine animals that generally live closer to shore, such as turtles, seals, and dolphins, risk contamination by oil that washes onto beaches or by consuming oil-contaminated prey (USEPA, 2004)

5.2.3 Vessel harassment

Several shipping lanes transect Nantucket Sound, which like the rest of the Atlantic seaboard, has witnessed an increase in vessel traffic independent of the larger vessels (*e.g.*, fishing boats, recreational boats from nearby areas). This Project will introduce additional vessel traffic steaming from Quonset, RI to the Project Area during construction and decommissioning and from New Bedford, MA to the Project Area during operation/maintenance. There have been many studies of the effects of vessels on cetaceans, particularly the underwater noises they make (Richardson *et al.* 1985; 1991). It is likely that whales react primarily to the sound generated by vessels, and not to their physical presence. Sea turtles do not appear to be overly disturbed by the physical presence of and sounds produced by vessels and vessel traffic; they may simply dive when approached by a vessel and avoid areas of intensive human activity (NMFS, 2001; NMFS, 2002).

The most serious risks of vessels to any of the listed species is from a collision and the physical harassment resulting from increased interaction between humans and these species. It has been reported that vessel traffic also may physically displace some whale species from feeding areas and may interrupt courtship, breeding, and other social activities if the vessel makes repeated approaches or if vessel traffic is heavy. There is evidence that some cetaceans may have been displaced from traditional feeding and wintering areas due to increased vessel traffic in Pacific waters (Baker *et al.* 1982; Forestell 1986). However, evidence from whale watching and fishing activities in Massachusetts waters indicates that humpback and fin whales readily habituate to the presence of large and small motor vessels (Watkins 1986). There is no evidence of long-term effects due to physical disturbance from motor vessels. Moreover, the central portion of Nantucket Sound and the vessel routes proposed to be used by Project vessels do not occur in areas where there have been high concentrations of whale or sea turtle sightings. If listed whales or sea turtles are present in the Project Area, temporary avoidance behavior is expected, though behavior changes will be short-term and will likely be similar to the avoidance behaviors observed during pleasure boat activity, ferry traffic, or fishing activity in the area. Close encounters between Project vessels and protected species is likely to be rare and result in minimal physical disturbance to the animals.

5.2.4 Vessel strikes

As previously mentioned, this Project will introduce additional vessel traffic steaming from Quonset, RI (construction and decommissioning) and New Bedford, MA (operation/maintenance) to the Project Area in Nantucket Sound, as barges and other vessels construct, maintain, and decommission the Project. Listed whales and sea turtles may be at risk of collisions with Project-related vessels; however, as stated above, the vessel routes proposed to be used by Project vessels do not occur in areas where there have been high concentrations of whale or sea turtle sightings.

Vessel strikes are a significant source of mortality for inshore species of baleen whales (Kenney and Kraus 1993; Wiley *et al.* 1995; NMFS 1991c). Some evidence exists of increased incidents of vessel collisions with large whales in northeastern U.S. waters in recent years (NMFS 1991c; Wiley *et al.* 1995). As has been documented for bowhead whales (George *et al.* 1994), the size and extent of scarring on whales indicate that collisions are primarily with large, faster moving vessels such as container ships, tankers, or military vessels. Collisions with vessels moving at slower speeds (less than 14 knots), such as the construction vessels to be used for the Project, are less likely to cause collisions (Laist *et al.*, 2001) and can be avoided by most marine mammals. Humpback, right, and fin whales should be able to detect and avoid tugboats, barges, and other slow-moving Project vessels. A tugboat cruising at two to six knots with a barge in tow generates underwater sounds with peak intensities in the optimum range of hearing in whales (Miles *et al.* 1987; McCauley 1994). Baleen whales can easily detect and respond to sounds of the frequency range and intensity of those produced by tugboats and barges (Richardson *et al.* 1991; Advanced Research Projects Agency 1995). Thus, right, humpback, and fin whales are likely to detect and respond to the sounds of an approaching tug and barge. Whale responses, however, are unpredictable and may depend on the activity of the whale at the time, or its previous experience with other motor vessels. Humpback whales are relatively tolerant of boats, but, due to this habituation, may be more susceptible to ship collisions. Fin and right whales appear to be more wary of approaching boats, and are likely to move away from vessels.

Although sea turtles are likely to dive at the approach of a vessel, they are still at risk of boat-related injuries. Between 1987 and 1993, up to 17% of all stranded sea turtles on the U.S. Atlantic coast had boat-related injuries (Teas 1994a,b). Ship strikes appear to be a significant source of mortality for sea turtles, and vessel-related

injuries have increased in recent years (Teas 1994a,b). However, as discussed above, sea turtles should be able to easily avoid vessels moving at slower speeds, such as those associated with Project construction (NMFS, 2001; NMFS, 2002). In addition, sea turtles present in Nantucket Sound are likely to be foraging and their feeding behaviors may also reduce the risk of collisions. While feeding, these turtles spend most of their time submerged. Ridleys and loggerheads can spend more than 57 minutes of each hour submerged (Thompson 1988) and between 25 and 58% of their time is directly on the bottom (Standora *et al.* 1994). Feeding dives last from about four minutes to as long as two hours (Renaud and Carpenter 1994). During these long periods of submergence, loggerhead and ridley turtles are not particularly vulnerable to collisions with barges.

The listed whales and sea turtles considered in this BA will likely be able to avoid Project-related vessel traffic easily. In addition, because vessel traffic associated with the Project will not occur in areas where there have been high concentrations of marine mammal and sea turtle sightings, the probability of Project-related vessels colliding with marine mammals and sea turtles is unlikely.

5.2.5 Acoustic harassment

Section 5.11 of the DEIS discusses the anticipated acoustic effects and potential impacts of the Project. Based on modeling and results from other wind projects, it is concluded that the Project will have no adverse impacts to wildlife. A small amount of localized and temporary sound will be generated in the marine environment from construction of the Project. The operation and maintenance phases will have very low-level acoustic effects, and underwater sound will not be measurable above background levels beyond a short distance from each monopile.

Sound can be measured in many terms, including frequency and sound pressure. Frequency is the rate of the sound wave vibration and is measured in cycles per second or hertz (Hz) (Richardson *et al.*, 1995). For airborne and underwater sound pressure, the standard unit of measurement is the decibel (dB), a logarithmic scale formed by taking 20 times the \log_{10} of a ratio of two pressures: the measured sound pressure divided by a reference sound pressure. Above air sound is referenced to $20 \mu\text{Pa}^1$, while underwater sound is referenced to $1 \mu\text{Pa}$. As a result, an identical sound pressure wave in air and underwater is recorded differently in the two fluids. For example, a sound pressure of 80 dB in air is equivalent to 106 dB underwater, i.e., the underwater scale is shifted 26 dB higher than the air scale. There are also substantial differences in ambient (background) sound levels in air and in the ocean, and in the frequency weighting that is used in the two media. Thus, the reader should not try to equate dB levels reported for water with those in air, or vice-versa.

A sound can also be transient or continuous. A transient sound (i.e., an explosion) has an obvious starting and stopping point while a continuous sound (e.g., offshore oil drill) is more or less persistent. The monopiles for the Project will be installed using pile driver technology and a pile driver is categorized as a repeating transient sound.

The total background ambient sound in the open ocean is about 74 to 100 dB re $1 \mu\text{Pa}$. However, several natural sound sources, such as earthquakes, lightning strikes, and some biological sounds, such as vocalizations of baleen whales and some swimbladder sounds of fish, may temporarily increase natural ambient sound above these levels. Sound pressure source levels for several different types of natural ambient sound in the marine environment are presented in Table 4. In comparison, vessel sounds generally fall in the range of 150 to 200 dB re $1 \mu\text{Pa}$ at a distance of 1 meter (3.3 feet) with peak frequencies in the 5 to 2000 Hz range and highest intensities below 100 Hz (Scrimger and Hietmeyer 1991). Table 5 lists estimated sound pressure source levels for vessels of different sizes and at different speeds. Sound intensity, particularly at higher frequencies, tends to increase with the size of the vessel. Supertankers and large container ships may have a maximum broad-band sound source level of 190 to 200 dB re $1 \mu\text{Pa}$ at 1m from the source (Table 5). Small outboard motor vessels produce broad-band sounds of about 150 dB re $1 \mu\text{Pa}$ at 1 meter (3.3 feet); these sounds are attenuated to the range of 85 to 140 dB re $1 \mu\text{Pa}$ at a distance of 50 meters (164 feet) from the source (Richardson *et al.* 1991).

¹ MicroPascals = 10^{-6} Newton/m².

Table 4. Maximum Broad-Band (20-1000 Hz) Sound Pressure Source Levels for Different Types of Natural Ambient Noise in the Marine Environment.

Noise Source	Maximum Source Level (dB re 1 μ Pa @ 1 m)	Remarks
Undersea Earthquake	272	Magnitude 4.0 on Richter scale (energy integrated over 50-Hz band width)
Seafloor Volcanic Eruption	255+	Massive steam explosions
Lightning Strike on Water Surface	250	Random events during storm at sea
Baleen Whales	to 188	<2000 Hz simple and complex calls, clicks, pulses, knocks, grunts, moans
Swimbladder Sounds of Fish	140	Marked spectral peaks in 50-3000 Hz range
Dugong	<90	2000-5000 Hz simple chirps and squeaks
Total Open-Ocean Ambient Noise	74-100	Estimate for offshore central California, sea state 3-5; expected to be higher (120 dB) when vessels are present
Rain Storm	80	Heavy rain shower, flat frequency spectrum
Wind	66	Force 3 wind over water

Sources: Richardson *et al.* (1991), McCauley (1994), and Advanced Research Projects Agency (1995).

Table 5. Estimated Peak 1/3-Octave Sound Pressure Source Levels for Vessels of Different Sizes and Speeds.

Vessel	Speed (knots)	Sound Pressure Level (dB re 1 μ Pa @ 1 m)	Reference
>250-m Large Oil Tanker	16	203	Cybulski 1977
274-m Container Ship (23 Hz)	--	198	Richardson <i>et al.</i> 1991
340-m Supertanker	20	190	Buck and Chalfant 1972
WWII Battleship	20	183	Urick 1984
337-m Tanker (20 Hz)	16	177	Cybulski 1977
Icebreaker	10	174	Malme <i>et al.</i> 1989
135-m Freighter	--	172	Richardson <i>et al.</i> 1991
Large Ferry	16	171	Malme <i>et al.</i> 1989
Tug and Loaded Barge	--	170	Miles <i>et al.</i> 1987
210-m Container Ship	19	169	Jennette <i>et al.</i> 1987
Cruise Ship	19	168	Malme <i>et al.</i> 1989
20-m Tug and Empty Barge	--	166	Buck and Chalfant 1972
200-m Roll On/Off	15	165	Jennette <i>et al.</i> 1987
190-m Car Carrier	16	162	Jennette <i>et al.</i> 1987
Tug and Barge	10	162	Malme <i>et al.</i> 1989
34-m Twin-Diesel Tour Boat	10	159	Malme <i>et al.</i> 1989
Fishing Trawler (transit)	10	158	Malme <i>et al.</i> 1989
Fishing Trawler (trawling)	5	147	Malme <i>et al.</i> 1989
16-m Crew Boat	--	156	Greene 1985
7-m Boat with 2 x 80-hp outdrive	20	156	Malme <i>et al.</i> 1982
8-m Boat with 260-hp outdrive	10	156	Malme <i>et al.</i> 1982
4-m Boston Whaler/20-hp outboard	20	153	Malme <i>et al.</i> 1982
5-m Zodiac with 20-hp outboard	20	152	Malme <i>et al.</i> 1982
4-m Boat with 25-hp outboard	20	152	Malme <i>et al.</i> 1982
20-m Tour Boat	10	150	Malme <i>et al.</i> 1989
Small Boat with 18-hp outboard	5	150	Evans 1982

The range of human hearing extends from about 20 Hz to 20,000 Hz (lowest to highest tones), though sensitivity falls off sharply at high and low frequencies. Most marine animals can perceive underwater sounds over a broad range of frequencies from 10 Hz to more than 10,000 Hz. Peak acoustic sensitivity of sea turtles and baleen whales generally is below 1,000 Hz, usually in the range of 20 to 200 Hz (McCauley 1994). Mysticete (baleen) whale sensitivity may be centered in the range of 100 to 200 Hz (Potter and Delroy 1998). Mysticete whales produce a wide variety of sounds, some of them of considerable intensity. They are thought to use sound for intraspecific communication and possibly also for echolocation and to aid in prey capture. Sounds often are frequency-modulated and range from short chirps and clicks to complex songs. Most baleen whale sounds fall in the frequency range of 12 to about 1,000 Hz, with some sounds as high as 8,000 Hz (Richardson *et al.* 1991; McCauley 1994). The high-frequency clicks (up to 31 kHz) sometimes recorded in the vicinity of fin, blue, minke, and gray whales have not been attributed with certainty to the whales themselves. If produced by the whales, they could be used for echolocation, as high frequency sound is by toothed cetaceans. The low-frequency moans, grunts, rumbles, and pulsive sounds produced by many baleen whales are in the frequency range of highest intensity sounds produced by motor vessels, particularly large vessels, such as tankers and container ships (Table 5). Low-frequency vocalizations in the 10 to 40 Hz range may be used by baleen whales for long-range communication (Payne and Webb 1971). Humpback whales (Thompson *et al.* 1986) and fin whales (Edds 1988) produce low frequency moans, tonal, pulsive and grunting sounds in the 10 to 40 Hz range. These low frequency sounds travel great distances with little attenuation in ocean waters, making them ideal for long-distance communication.

Underwater sounds, if they are intense enough, may cause injury to the ears of whales and sea turtles, or even death from concussion. Lower intensity sounds in the frequency ranges heard and used for communication by these animals may interfere with normal intraspecies communication, detection (by echo-location) of prey, and orientation (Richardson *et al.* 1991). Most baleen whales respond to constant, low-frequency sounds with broadband intensities of more than 120 dB re 1 μ Pa (Advanced Research Projects Agency 1995). However, actual thresholds for behavioral responses to sounds in the natural environment depend on the level of natural ambient noise. Whales apparently are able to distinguish sounds in their optimum frequency range that are 10 to 20 dB re 1 μ Pa above ambient noise at the same frequency (Richardson *et al.* 1991). The threshold intensity of constant or impulsive sounds for injury to the hearing apparatus of marine mammals is about 200 to 220 dB re 1 μ Pa (Greenlaw 1987; McCauley 1994). The sensitivity of sea turtles to acoustic disturbance has not been well studied. Sea turtles may use acoustic signals within their environment for orientation to natal beaches (Lenhardt *et al.* 1983). In addition, loggerhead turtles swam towards the surface when exposed to low frequency, high-intensity sounds (20-80 Hz, 175-180 dB) while underwater (Lenhardt 1994). However, typical vessel sounds, which can sometimes be in this range, do not seem to disturb sea turtles.

The maximum submarine sound generated during construction of the Wind Park will occur during installation of the monopile foundations, and as discussed below that maximum sound level is 178 dBL at the 500-meter Initial Safety Radius (ISR). Since foundation work will not proceed when listed species are within the ISR, construction of the Project is not anticipated to cause physical harm to marine mammals or sea turtles. The jet plow embedment process for laying the two submarine cable circuits and inner-array cables produces no sound beyond that produced by typical vessel traffic and the cable installation barge will produce sound typical of vessel traffic already occurring in Nantucket Sound. No substantial underwater sound will be generated during horizontal directional drilling (HDD) operations used to transition the submarine cable to the upland cable system in Lewis Bay. Due to the sound-insulating qualities of earthen materials (the sediment), and the fact that the drilling would take place through unconsolidated material, the HDD transition is not anticipated to transmit vibration from the sediment to the water (i.e. it would not add appreciable sound into the water column). As a result, the main underwater acoustical impacts during construction activities will be limited to that generated by installation of the monopile foundations and vessel traffic.

Sound levels measured during impact pile driving operations at the Utgrunden Wind Park in Sweden were used to model underwater sound impacts for this Project because the monopiles are similar in size and the same pile driving installation technique is proposed by the Applicant (see Section 2.3 of Appendix 5.11-A of the DEIS). The Utgrunden data show a maximum (L_{max}) sound level of 178 dBL at 500 m (1,640 feet). Frequency plots from the Utgrunden data show the peak energy from pile driving occurred between 200 and 1,000 Hz, with underwater sound levels falling below background levels (inaudible) for frequencies below 5 Hz. NMFS suggests that 180 dBL

represents the threshold level for preventing injury or harassment to marine mammals and sea turtles². The sound levels anticipated to occur during Project construction at and beyond the Initial 500 m Safety Radius³ are below this level (see Section 5.11.2.6 of the DEIS). Therefore, construction of the Project is not anticipated to cause physical harm to marine mammals or sea turtles. Underwater sound monitoring will be performed during initial monopile construction (as was done to ensure protection of marine mammals during the installation of the SMDS foundation piles). This will include posting a NMFS-approved observer on-site during initial pile driving activities to monitor the area during construction. If listed species are observed by the NMFS-approved observer within the 500 m Safety Zone, the observer will ensure that work will cease until the animal is clear of the work area and safety zone (see Section 5.11.2.7 of the DEIS).

If marine mammals or sea turtles are present in the Project area, they are likely to temporarily avoid the area during construction activities. Studies off the California and Alaska coastlines have shown that most species of cetaceans adjust to the presence of offshore drilling equipment (Geraci and St. Aubin 1987). However, studies of bowhead whales in the Arctic indicate that individuals will often change course and behavior when exposed to the intense noise generated by active rigs and seismic vessels (Ljungblad *et al.* 1988; Richardson *et al.* 1985; 1991). Bowhead whales in the Beaufort Sea react, at least briefly, to aircraft, ships, seismic exploration, marine construction, and offshore drill sites (Richardson and Malme 1993). To date, there is no conclusive evidence that this short-term disturbance leads to long-term effects on individuals or populations (Richardson *et al.* 1991).

Increased project-related vessel traffic will occur during all phases: construction, operation and maintenance, and decommissioning. Sounds produced by tugboats towing barges probably produce underwater sounds with peak intensities in the frequency range of whale and sea turtle auditory capability of about 165 dB re 1 μ Pa at 1 meter (3.3 feet) (Buck and Chalfant 1972; Miles *et al.* 1987; Malme *et al.* 1989). These sounds attenuate naturally in the water to about 120 dB re 1 μ Pa at about 2 nautical miles from the source. There are conflicting reports of the short-term effects of vessel engine sound on marine mammals (*i.e.*, some species of whales react to noise at great distances (as far as 4 km), some do not) (Ljungblad *et al.* 1988; MMS 1992). There is limited evidence that abrupt changes in vessel RPMs may disturb whales (Watkins 1986); however, it appears that they readily acclimate to the noise in their environment. Overall, reactions to human-generated noise vary not only between species, but also within species (Richardson *et al.* 1991). Some studies suggest that whales may react to short-term acoustic disturbances by moving away from the sound source, changing breathing and diving patterns, or through possible distressed behavioral displays (NMFS 1991c).

The sounds of tugs and barges will be clearly audible to whales and sea turtles in the vicinity of the Project site and transit routes. However, these sounds would be too weak to cause outright harm to the whales or sea turtles. Nantucket Sound frequently experiences heavy vessel traffic from a variety of vessels; therefore, the slight increase in vessel traffic caused by the Project should not substantially alter the acoustic environment in this area above the normal baseline sound.

Once installed, the operation of the WTGs is not expected to generate substantial sound levels above baseline sound in the area. Acoustic modeling of underwater operational sound at the Wind Park was performed for the design wind condition (see Section 5.11.2.4 of the DEIS). Baseline underwater sound levels under the design wind condition are 107.2 dBL (see Section 5.11.2.3.1 of the DEIS). The predicted sound level from operation of a WTG is 109.1 dBL at 20 m (65.6 feet) from the monopile (*i.e.*, only 1.9 dBL above the baseline sound level), and this total sound level falls off to 107.5 dBL at 50 m (164 feet) and declines to the baseline level at a relatively short distance of 110 m (361 feet). Since the WTGs will be spaced farther apart than 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. The operational effects of the Project are anticipated to be minimal, with no adverse effects on marine mammals or sea turtles. Submarine sound levels generated during Wind Park operation are unlikely to adversely affect the listed whale or sea turtle species or their prey.

² National Marine Fisheries Service, letter from Ms. Patricia Kurkul, Regional Administrator to Ms. Christine Godfrey, U.S. Army Corps of Engineers, June 27, 2002.

³ The 500 meter safety radius is based on a condition in the USACE Permit granted to Cape Wind for construction and operation of the Scientific Measurement Devices Station (SMDS) [Permit No. 199902477]. The condition requires that sound level monitoring during pile driving procedures be conducted at an initial safety zone radius of 500 meters to determine compliance with the 180 dBL NMFS threshold. A similar safety radius was established by NMFS for pile installation at the San Francisco-Oakland Bay Bridge [SRS Technologies. 2004. San Francisco-Oakland Bay Bridge East Span Seismic Safety Project. Revised Marine Mammal Monitoring Plan.] [Illingworth & Rodkin, Inc. 2001. Pile Installation Demonstration Project Construction Report. *In*: San Francisco-Oakland Bay Bridge East Span Seismic Safety Project.]

In summary, sound levels produced during construction, operation, and decommissioning activities and by Project-related vessels are not expected to be of sufficient intensity to cause physical damage to listed whales or sea turtles that may be in the Project Area. Temporary avoidance behavior to Project related noise and to heavy vessel traffic is likely to occur. Sound levels expected from Project related activities are not expected to cause lasting behavioral alterations to the protected species (Richardson *et al.* 1991).

5.2.6 Electromagnetic/Thermal Emissions

Potential direct impacts to listed whales or sea turtles during the normal operation of the inner-array cables and the two submarine cable circuits are expected to be negligible. 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 for the Project 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 described in Section 5.13 of the DEIS, the magnetic fields associated with the operation of the inner-array cables or the submarine cable system are not anticipated to result in an adverse impact to marine mammals, sea turtles, or their prey (ICNIRP 2000; Adair, 1994; Valberg *et al.* 1997).

Because the inner-array cables and the two submarine cable circuits connecting the Wind Park to the landfall will be buried approximately 6 feet below the seabed, they will not pose a physical barrier to fish passage. The considerable depth to which the cables will be buried will allow benthic organisms to colonize and demersal fish species to utilize surface sediments without being affected by the cable operation. The burial depth of the cables also minimizes potential thermal impacts from operation of these cable systems. In addition, the inner-array and 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 the listed whale or sea turtle species during the normal operation of the inner-array or submarine cable systems. There will also be no impacts to finfish, invertebrate, or plankton prey species of marine mammals or sea turtles (indirect impact) during the normal operation of the inner-array or submarine cable systems.

5.2.7 WTGs as Fish Aggregating Devices

The WTG monopile foundations would represent a source of new hard substrate with a vertical orientation in an area that has a limited amount of such habitat. Therefore, the WTG monopile foundations and ESP piles may attract finfish and benthic organisms, potentially indirectly affecting listed whale and sea turtle species by causing changes to prey distribution and/or abundance. Bohnsack (1989) found that fish species most likely to benefit from artificial structures, such as the monopiles, are those with demersal, philopatric (*i.e.*, non-dispersing), territorial, and reef-obligate life histories. Several finfish species within the Proposed Alternative Site and other alternative sites in Nantucket Sound display these characteristics in some or all of their life history stages, and thus may benefit from the presence of the WTG monopiles. These species include Atlantic cod, black sea bass, cunner, tautog, and scup. Benthic species that may initially be supported by such structures are likely to include algae, barnacles, hydroids, sponges, tunicates, bryozoans, anemones, and possibly mussels, all of which occur in other areas of hard substrate within Nantucket Sound. Additional organisms, such as crabs, gastropods, nudibranchs, polychaetes, oligochaetes, and nematodes may also utilize the structures once algal growth becomes established.

The listed whale species do not rely on these finfish or benthic organisms as prey, occur only rarely in Nantucket Sound, and therefore are not expected to be influenced by potential finfish or benthic organism aggregations at the individual WTG monopiles. Potential finfish aggregations at the monopiles are not likely to affect foraging sea turtles; however, the benthic invertebrates that may attach to the monopiles could provide an additional food source for sea turtles. 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 would be a relatively inconsequential addition to the hard substrate that is already present (see Section 5.3 of the DEIS). Due to small amount of additional surface area in relation to the total Project Area and Nantucket Sound and the spacing between WTGs (0.34 to

0.54 nautical miles apart), the new additional structure is not expected to affect the overall environment, benthic community composition, finfish species composition, or populations of foraging marine mammals or sea turtles in the area.

5.2.8 Habitat Shift from Open Shoals to Structure-Oriented

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 a non-structured system to a structure-oriented system, with potential localized changes to marine mammal and sea turtle movement in the Project Area within the Sound, as well as to the benthic and finfish community assemblages which serve as prey for these species. However, because the WTGs and the scour control mats within the array will be spaced approximately 0.34 by 0.54 nautical miles apart, movement and populations of marine mammals and sea turtles that may occur in the vicinity of the WTG array are not predicted to substantially change from pre-Project conditions. The additional amount of surface area being introduced is relatively inconsequential (approximately 1,200 square feet per tower assuming an average water depth of 30 feet below mean high water (MHW)) and the wide spacing of these monopiles is also not expected to greatly increase the production of finfish and benthic invertebrates in the Project Area (see Sections 5.3 and 5.4 of the DEIS). Therefore, the overall environment, benthic community composition, and finfish species composition in the Project Area and Nantucket Sound is also not predicted to substantially change from pre-Project conditions. Additionally, installation of scour control mats around the monopile foundations will help ensure that scour around the WTGs and changes to the overall sediment transport within the array will be limited, further reducing potential impacts to listed whale and sea turtle species that may visit the Project Area. For more information on WTG and scour control mat installation and impacts, please refer to Section 4.0 of the DEIS, Appendix 4.0-A and Section 5.4 of the DEIS.

5.2.9 Cumulative Impacts

No significant cumulative impacts to the listed whale or sea turtle species are expected from construction of the WTGs, the inner-array cables, or the two submarine cable circuits. Any impacts from construction activities are expected to be localized and temporary. In addition, the rarity with which the protected whale species and sea turtles occur within Nantucket Sound further reduces any of these temporary or localized impacts. Other projects that are proposed to occur in the vicinity of the Project could contribute additional impacts to protected marine species.

In addition to the proposed Project, other activities which may contribute to cumulative impacts to protected marine species would include other submarine cable or pipeline installations, dredging activities, trawling, pile supported marine structures and other offshore wind installations (which at this time are limited to a small scale project proposed off the coast of Hull Massachusetts, and a large installation proposed by Long Island Power Authority (LIPA) off the southern coast of Long Island). The cumulative impacts from various potential activities that may occur within the location and timeframe of the proposed Project are discussed below.

The submarine cable system would be placed adjacent to the eastern edge of the Federal Navigation Project in Hyannis Harbor. Maintenance dredging of the channel, if initiated at the same time as the jet plow installation of the cable system, could result in additional concurrent, cumulative sediment suspension and deposition. Hyannis Harbor was dredged in 1985, 1991, and 1998. No dredging is currently scheduled, but based on recent experience it could be needed in the next 3-4 years. However, as discussed above, protected species, seal haul-out and breeding sites are not common in the vicinity of Hyannis Harbor and, individually, these projects are not likely result in impacts to protected marine species. If the cable installation is completed in 2006 as expected, these activities will not be concurrent. In any event, as discussed in Appendix 5.2-C, sediment deposition resulting from the cable installation would be minimal and localized, and would not substantially contribute to any cumulative impact. Since potential dredging will not likely occur simultaneously to the submarine cable installation, no other significant cumulative impacts to protected marine species (i.e. noise, habitat disturbance) are expected.

A new submarine transmission cable has been proposed by National Grid between Cape Cod and Nantucket. Its proposed route would only cross the Project's submarine cable route in the vicinity of Hyannis Harbor. Outside of Massachusetts waters, at its closest point the proposed route of the Nantucket Cable would be approximately 2 miles (3.2 km) from the Wind Park and its inner array cables in Nantucket Sound. Prior to final design and construction, the Applicants for both projects would need to coordinate plans, design, and schedule for

installation of the cables at this crossing point. At this crossing, and in its near vicinity, the impacts of each project would be coincident in nature. However, as discussed above, protected species are not common in Nantucket Sound and, individually, these projects are not likely result in impacts to protected marine species. Thus, the area would not likely have increased water column sediment loadings from the first project installation at the time the second project is constructed and no significant cumulative impacts to protected marine species are anticipated.

The submarine cable installation for the Cape Wind Project would cross Nantucket Sound's North Channel. North Channel is a naturally occurring and maintained passageway marked by USCG aids-to-navigation and is not designated as a Corps of Engineers Federal Navigation Project, and therefore is not subjected to maintenance dredging. Therefore, no significant cumulative effects to protected marine species are expected in the area of the North Channel crossing.

There are existing submarine cables that cross from Falmouth to Martha's Vineyard and from Harwich to Nantucket. These submarine cables require routine maintenance. However, there are no significant cumulative impacts expected to finfish since the existing cables are approximately 13 miles (21 km) and 8 miles (13 km) away from the Project area, respectively.

It is possible that additional dredging may occur at shore-based marinas supporting boating activities throughout the Project area. However, these marina dredging projects, if they were to occur, are very localized and not likely to result in sediment suspension and deposition that would be coincident with the Project's cable installation (the closest point of which would be a minimum of .5 miles (805 meters) from the closest marina), nor would the impacts to finfish from these activities be substantial. Thus no significant cumulative impacts are anticipated from such activities

As discussed above, based upon the lack of any other active USACE Section 10 Applications proposing similar large-scale offshore wind power generation projects or other offshore projects in Federal waters off the New England coast, it is anticipated that the cumulative impacts from this project and other potential offshore facilities will be negligible. It is anticipated that smaller projects ranging from single turbines to less than ten turbines will make up the bulk of the offshore wind generation in the foreseeable near term. These are likely to be municipally sponsored, nearshore projects, and not developed in sufficient numbers to create any significant cumulative impacts.

Furthermore, the increased traffic from operation and maintenance activities (estimated to be 2 to 3 vessels per day) and any potential increase in recreational vessel activity in the project vicinity is not expected to significantly alter the behavior of protected marine species since they are not typically found in the Project area.

5.3 Summary/Conclusion of Impacts

Based on the analysis of potential impacts discussed above, it is unlikely that substantial adverse effects to the listed whales and sea turtles will result from the construction, operation, maintenance, and decommissioning of this Project. Water quality is not expected to be impaired by the Project, and bioaccumulation from consumption of contaminated food sources is not expected to occur. Although vessel traffic will increase during the period of the Project, collisions between Project tugboats and barges and whales and sea turtles are unlikely. These species can detect the presence of barges and tugs, and because of the slow speed at which barges and tugs operate, whales and sea turtles would likely avoid them. Acoustic disturbance from the increased vessel traffic and Project construction is likely to cause only minor and temporary disturbance to the listed whales and sea turtles. Design and placement of the inner-array and submarine cable systems will ensure that no impacts to listed species or their prey occur during normal Wind Park operation. The presence of the WTGs may result in localized changes to marine mammal and sea turtle movement in the Project Area within the Sound, as well as to the benthic and finfish community assemblages which serve as prey for these species. However, as noted above, because the WTGs within the array will be spaced 0.34 by 0.54 nautical miles apart, the overall environment and species composition in the Project Area and Nantucket Sound is not predicted to substantially change from pre-Project conditions. The management actions discussed below in Section 5.4 will minimize potential vessel collisions and potential acoustic disturbance during the Project.

5.4 Management Alternatives for the Potential Impacts of the Proposed Action

1. Possible event: Collision with Project-related vessels

Possible result: Injury or death of sea turtle or whale

Management Action or Practice: Vessels transporting construction materials to the Project site in Nantucket Sound will travel at slow speeds, usually well below 14 knots. Nevertheless, potential vessel impacts to whales and sea turtles will be minimized by requiring that Project vessels follow *NOAA Fisheries Regional Viewing Regulations and Guidelines for Marine Mammals and Sea Turtles* (NOAA Fisheries, 2003) while in transit to and from the WTG site so as not to disturb any individuals that may be in the area.

2. Possible event: Physical harassment from Project-related vessels

Possible result: Alter “normal” behavior, stop feeding, abandon feeding area, decrease maternal care.

Management Action or Practice: Evidence from whale watching activities in Massachusetts Bay indicates that humpback and fin whale species readily acclimate to the presence of large and small motor vessels. There is no evidence of long-term effects due to physical disturbance from motor vessels. To prevent physical harassment and minimize impacts to listed species during the Project, Project vessels will be required to follow *NOAA Fisheries Regional Viewing Regulations and Guidelines for Marine Mammals and Sea Turtles* (NOAA Fisheries, 2003) while in transit to and from the WTG site to minimize or eliminate the potential for harassment.

3. Possible event: Acoustic harassment

Possible result: Short-term: change in swimming direction and/or breathing patterns; Long-term: unknown

Management Action or Practice: Little is known of the long-term effects of acoustic harassment. Humpback and fin whales seem to acclimate readily to underwater sounds produced by motor vessels. However, it is unclear if chronic elevated noise levels lead to behavioral modification in the whales. Barge traffic will not contribute significantly to the total underwater background sound in the area and is not likely to be disturbing to whales and sea turtles. Construction and operation sounds are not expected to reach levels known to be harmful to whales. The sensitivity of sea turtles to acoustic disturbances has not been well studied. Available evidence indicates that loggerhead and ridley turtles are only disturbed by intense underwater sounds, well above those produced by tugs towing loaded barges. Acoustic harassment during monopile construction may be minimized by conducting a “soft start” to each pile event. This will allow listed species in the immediate vicinity of Project activities, the opportunity to leave the area. In addition, underwater sound monitoring will be performed during initial monopile construction identical to that done to protect marine mammals during the installation of the SMDS foundation piles. This will include posting a NMFS-certified observer on-site during initial pile driving activities to monitor the area during construction. If listed species are observed within the Safety Zone by the NMFS-approved observer, the observer will ensure that work will cease until the animal is clear of the work area and safety zone (see Section 5.11.2.7 of the DEIS). These measures should provide adequate protection to avoid and minimize acoustic impacts to marine mammals and sea turtles.

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