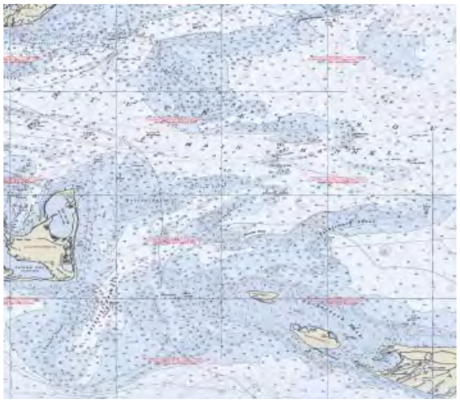
Marine Megavertebrates and Fishery Resources in the Nantucket Sound – Muskeget Channel Area

Ecology and Effects of Marine Renewable Energy Installations



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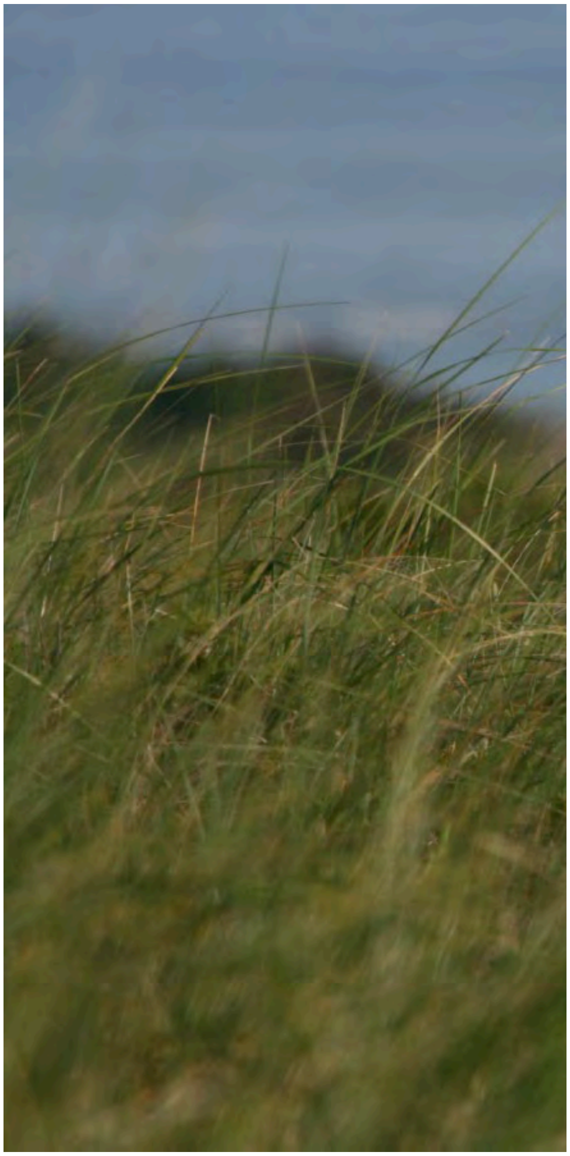

A Report to

Harris Miller Miller & Hanson Inc

for HMMH Project # 303910

Environmental Impacts of Sediment Transport Alteration and Impacts on Protected Species: Edgartown Tidal Energy Project

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Executive Summary

Marine Megavertebrates and Fishery Resources

The authors found that there has been little or no directed research on marine megavertebrates in the Nantucket Sound – Muskeget Channel area. While there has been directed research on some species in the Gulf of Maine, survey effort has been very low in the waters immediately south of Cape Cod, including the Muskeget Channel area.

Surveys have been done to estimate population size of harbor and gray seals in this area; however these are now out of date. Most of the data on cetaceans and sea turtles discussed in this report are from opportunistic sightings, strandings and entanglements. With the exception of a tagging program on leatherbacks, there is no systematic survey effort on sea turtles in this area. The lack of systematic survey efforts in the study area precludes an accurate assessment of the abundance and distribution of cetaceans and sea turtles in the Nantucket Sound – Muskeget Channel area. This is also true for basking sharks and ocean sunfish.

There is little readily available data with which to evaluate the specific importance of the Muskeget Channel study area to commercial and recreational fisheries. The Massachusetts Division of Marine Fisheries conducts fall and spring trawl surveys that measure relative abundance of species throughout state waters; however, these surveys are not designed to measure fine-scale distribution patterns.

Section IV of this report discusses these data gaps.

Effects of Marine Renewable Energy Installations on Marine Megavertebrates

This section provides a summary of the existing literature and knowledge regarding the effects of marine renewable energy installations (MREIs) on marine megavertebrates. The review notes that MREIs have the potential to be both detrimental and beneficial to the environment, and the effects will likely be site-specific. Most studies to date have investigated the effects of offshore wind turbines on marine fauna; data is lacking on other technologies. There is therefore a great need for focused research to address the potential effects of tidal devices on marine ecosystems.

Continuous assessment over longer time periods, in different locations and with appropriate control sites for comparison will be necessary, as marine organisms may respond or adapt differently in different habitats. The many possible impacts of MREIs on marine ecosystems, both positive and negative, will have complex interactions which are difficult to predict. Such interactions will likely be cumulative both temporally and as the number of MREIs grows. In carrying out an environmental impact assessment at any given MREI site, it will be crucial to incorporate solid study design into any monitoring program to allow for reliable detection of effects. It will also be important to assess whether the effects on individual megavertebrates at specific sites will translate into population-level effects.

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Background

Photo: Harbor seals, Provincetown Center for Coastal Studies image taken under NOAA permit 775-1875

Muskeget Channel is located between the islands of Martha's Vineyard and Nantucket. Water depths in the channel range between 40 and 160 feet, with Wasque Shoals to the west and Mutton Shoal to the east. Muskeget Channel allows for the exchange of water between Nantucket Sound to the north and the Atlantic Ocean and continental shelf to the south.

The Town of Edgartown is proposing to develop an initial 5MW tidal energy pilot project in Muskeget Channel. Edgartown holds a Preliminary Permit from the Federal Energy Regulatory Commission (FERC), giving it the exclusive right to explore the development of the resource for energy. Edgartown is required to submit a Draft Pilot License Application that will allow the town to deploy, operate and monitor this pilot-scale turbine installation. This application must include information on initial consultation with cooperating federal resource agencies; draft study plans, including one on protected species, and an outline of work that will be completed during deployment of the pilot project.

The Town of Edgartown engaged Harris Miller Miller & Hanson (HMMH) as its Principal Investigator (PI) and program manager. HMMH was successful in obtaining U.S. Department of Energy funding for the study: *Environmental Effects of Sediment Transport Alteration and Impacts on Protected Species: Edgartown Tidal Energy Project.*

The Provincetown Center for Coastal Studies (PCCS) is one of four organizations working on this study under the direction of HMMH. The PCCS tasks were to:

1. Conduct a literature review of
   - current information on the documented occurrence and habitats of marine megavertebrates – cetaceans, pinnipeds, turtles, basking sharks and sunfish – in the Muskeget Channel region;
   - documented distribution of fishery resources and habitats and commercial and recreational fishing activity;
   - studies and assessments on the environmental impacts of marine energy conversion projects on marine megavertebrates.
2. Prepare protocols for environmental studies and monitoring of marine megavertebrates specific to Muskeget Channel sufficient to collect data needed to define baseline conditions and evaluate impacts from the operation and maintenance of the tidal energy project.
3. Prepare a synthesis report on the permitting and planning framework for marine energy conversion projects, focusing on the Muskeget Channel region.

This report includes work PCCS completed under Task 1. Work completed under Tasks 2 and 3 is presented in separate reports.

2 Review of the Distribution, Abundance and Habitats of Marine Megavertebrates (Cetaceans, Pinnipeds, Sea Turtles, Basking Sharks and Ocean Sunfish) in the Nantucket Sound – Muskeget Channel Area

Photo: North Atlantic right whales, Provincetown Center for Coastal Studies image taken under NOAA permit 633-1763, with authority of the U.S. Endangered Species Act

2.1 Cetaceans

2.1.1 Introduction

Broad-scale seasonal distribution patterns of most cetacean species in the waters of the Northeastern United States are relatively well understood (Kenney and Winn, 1986; Kenney *et al.,* 1996; Pittman *et al.,* 2006). However, systematic survey effort has been very low in the waters immediately south of Cape Cod, including the Muskeget Channel area (Pittman *et al.,* 2006; see also Data Summary). The intent of this section is to summarize available information on the occurrence of cetaceans in the vicinity of the Muskeget Channel region. A separate section is devoted to historical and present-day occurrence of North Atlantic right whales due to the species' critical status (Kraus *et al.,* 2005). The remainder of this section reviews the occurrence of other baleen whales, including endangered fin, sei and humpback whales, and a summary of the occurrence of the endangered sperm whale and several other species of toothed whales.

2.1.2 Species Descriptions

2.1.2.1 Mysticetes (Baleen Whales)

North Atlantic right whale (*Eubalaena glacialis*):

North Atlantic right whales are listed as endangered under the U.S. Endangered Species Act. A minimum of 415 individuals were thought to be alive in 2007 (Pettis, 2009). Right whales are distributed from winter calving grounds in the waters of the Southeastern United States north to summer feeding grounds in the Bay of Fundy and on the Scotian Shelf, with rare sightings in the Gulf of Mexico and off Greenland and Norway (Winn *et al.,* 1986; Waring *et al.,* 2009). Right whales are present in Cape Cod Bay in winter and spring (Hamilton and Mayo, 1990; Nichols *et al.,* 2008) and the Great South Channel in late spring (Kenney *et al.,* 1995), where they feed on dense concentrations of zooplankton, particularly calanoid copepods (Mayo and Marx, 1990; Beardsley *et al.,* 1996). Pittman *et al*. (2006) analyzed the limited systematic survey effort available in the area south of Cape Cod, including Muskeget Channel, and noted that right whale sightings-per-unit effort (SPUE) was very low, with 0.1-8.2 whales sighted per 1,000 km of survey effort in most of the area. Given the low survey effort in the area, opportunistic sightings warrant further discussion, as do patterns of historical occurrence.

The nearshore waters off Nantucket Island were productive hunting grounds for shore-based whalers as early as the mid-1600s. Reeves *et al.* (1999) reviewed catch histories of whalers targeting right whales in Nantucket waters, primarily citing the monographs by Allen (1916) and Little (1981, 1988). The above authors gathered information from a variety of sources, including whaling logbooks and Nantucket newspapers, and primarily made reference to the abundance and/or seasonality of right whale presence. In the instances where catch locations were reported relative to the location of Nantucket Island, most were distributed to the south or east, and a few were documented in Nantucket Sound.

Schevill *et al*. (1981) recorded a number of winter and spring right whale sightings in Nantucket waters from 1956-1980, and Mate *et al*. (1997) tracked a female accompanied by her calf in Nantucket Sound using satellite telemetry in the summer of 1990. The carcass of an entangled one-year-old female drifted ashore on Nantucket in October 2002 (Moore *et al.,* 2004). A number of right whale sightings recorded in the area from assorted platforms (n = 111) are archived in the North Atlantic Right Whale Consortium sightings database (Right Whale Consortium, 2010; Figure 2.1). Of particular interest is the occurrence of a relatively large number of right whales in winter/spring 2010 in Nantucket, Vineyard and Rhode Island Sounds and the waters immediately south of Nantucket (Kenney, 2010). Sightings in the Sounds reported to the North Atlantic Right Whale Sighting Advisory System during winter and spring 2010 (n = 105; most of which occurred in Rhode Island Sound) presented by Kenney (2010) are included in Figure 2.1. These sightings were not yet entered into the Right Whale Consortium sightings database at the time of this writing. The combination of opportunistic sightings in the past few decades and present-day reports suggests that in some years, particularly during the winter and early spring, right whales may still be found in the near-shore waters of Nantucket Island and Martha's Vineyard as well as Nantucket Sound.

Four other species of baleen whales occur frequently in Northeastern U.S. waters: fin (*Balaenoptera physalus*), sei (*Balaenoptera borealis*), humpback (*Megaptera novaeangliae*) and minke whales (*Balaenoptera acutorostrata*). Sightings of these species are most common in spring, summer and fall (Kenney and Winn, 1986; Kenney *et al.,* 1996; Pittman *et al.,* 2006).

Fin whales are listed as endangered under the U.S. Endangered Species Act. The most recent abundance estimate available for the western North Atlantic fin whale stock is 2,269 (CV = 0.37; Waring *et al.,* 2009). Occurrence in northeast U.S. waters from spring through the fall is associated with distribution of prey, in particular small schooling fish (Kenney *et al.,* 1996). The location of calving, mating and wintering is unknown.

Sei whales are listed as endangered under the Endangered Species Act; no population estimate is available for the Nova Scotia stock, which includes the waters of the Northeastern U.S. in its range (Waring *et al.,* 2009). The species is generally distributed offshore towards the outer continental shelf, although episodic incursions into near-shore waters are associated with reduced competition for their zooplankton prey (Kenney *et al.,* 1996).

Humpback whales are listed as endangered under the Endangered Species Act. Humpback whales off the Northeastern U.S. are considered to be part of the Gulf of Maine stock, which is defined by high individual fidelity to the region during seasonal migrations away from calving and mating grounds in the West Indies and surrounding low-latitude waters (Robbins, 2007; Waring *et al.,* 2009). Several abundance estimates have been generated for this stock, all averaging approximately 500 animals (Waring *et al.,* 2009).

Minke whales are not listed under the Endangered Species Act. Minke whales that occur off the eastern coast of the United States are considered to be part of the Canadian East Coast stock, which is found from the Davis Strait south to the Gulf of Mexico. No population estimate exists for this stock. Peak abundance in continental shelf waters off New England occurs during spring and summer, while during winter the species appears to be largely absent (Waring *et al.,* 2009). Like humpback and fin whales, minke whales in the region are largely piscivorous, and their distribution is affected by that of their prey (Kenney *et al.,* 1996).

2.1.2.2 Odontocetes (Toothed Whales, Dolphins & Porpoises)

Numerous species of toothed whales occur off the northeastern U.S., including Atlantic white-sided dolphin (*Lagenorhynchus acutus*), bottlenose dolphin (*Tursiops truncatus*), common dolphin (*Delphinus delphis*), harbor porpoise (*Phocoena phocoena*), pilot whales (*Globicephala* spp.), pygmy sperm whale (*Kogia breviceps*), Risso's dolphin (*Grampus griseus*), sperm whale (*Physeter macrocephalus*), striped dolphin (*Stenella coeruleoalba*) and white-beaked dolphin (*Lagenorhynchus albirostris*) (Katona *et al.,* 1993). Of these species, only the sperm whale is listed as endangered under the Endangered Species Act.

Atlantic white-sided dolphins occur in shelf waters from Greenland south to the Carolinas and are most common off the Northeastern U.S. in spring and summer (Pittman *et al.,* 2006; Waring *et al.,* 2009). Like many of the region's toothed whales, white-sided dolphins feed on fish and squid, and their distribution often reflects that of their prey (Kenney *et al.,* 1996). They are mostly found in deeper offshore waters, but can be seen quite close to shore around the Cape Cod region. The best estimate of abundance of the Western North Atlantic stock is 63,368 (CV = 0.27; Waring *et al.,* 2009).

Bottlenose dolphins are rare in Northeastern U.S. waters and are more commonly noted in coastal waters in the Southeastern U.S. (Katona *et al.,* 1993).

Common dolphins occur most often in fall in offshore waters of the Northeastern U.S., although they are known to mass strand on Cape Cod along with white-sided dolphins (Bogomolni *et al.,* 2010).

Harbor porpoises move through the Northeastern U.S. shelf waters throughout the year, with concentrations in the Northern Gulf of Maine in summer, dispersion throughout the region in spring and fall and southern movement in winter. The best estimate of abundance for the Gulf of Maine harbor porpoise stock is 89,054 (CV = 0.47; Waring *et al.,* 2009). This species is known to use coastal waters.

Pilot whales are mostly found toward the edges of the continental shelf but are known to occur in Cape Cod waters, often as part of mass strandings (Pittman *et al.,* 2006; Bogomolni *et al.,* 2010). Due to confusion between the short-finned and long-finned pilot whale, population abundance indices for each species are difficult to establish (Waring *et al.,* 2009).

Pygmy sperm whales, Risso's dolphins, sperm whales and striped dolphins all occur in warmer, lower-latitude, offshore waters and only rarely occur in the waters of Northeastern northeast U.S. (Katona *et al.,* 1993). Population information for these species is lacking (Waring *et al.,* 2009). White-beaked dolphins are similarly rare in the area, although they may have been displaced inshore by white-sided dolphins in response to shifts in prey of both species (Kenney *et al.,* 1996).

2.1.3 Distribution & Abundance in the Nantucket Sound – Muskeget Channel Area

2.1.3.1 Mysticetes

In the study area south of Cape Cod, including Muskeget Channel, SPUE of fin, sei, minke and humpback whales was very low or zero (Pittman *et al.,* 2006). Given the low survey effort in the area, all sightings, including those recorded opportunistically, warrant further discussion. Sightings of fin (n = 141), sei (n = 1), humpback (n = 27), and minke (n = 23) whales archived in the North Atlantic Right Whale Consortium sightings database (Right Whale Consortium 2010) were generally distributed to the south of Nantucket Sound (Figure 2.1). However, this does not necessarily reflect spatial distribution patterns, as systematic survey effort in the study area was distributed in a similar manner (See Data Summary). Stranding data obtained from NOAA Fisheries[1] for animals that stranded on or in the vicinity of Martha's Vineyard, Nantucket and nearby islands included records of fin (n = 4), sei (n = 1), humpback (n = 7) and minke (n = 9) whales. Stranding data must be interpreted with caution, as unhealthy or otherwise compromised animals may not ordinarily occur in the area, and carcasses can drift from distant locations.

1. Data courtesy of Tracy Bowen and Mendy Garron (NOAA Fisheries) and contributing stranding networks spanning 1988-2009. These data should not be used out of context or without verification.

2.1.3.2 Odontocetes

Pittman *et al.* (2006) analyzed survey data for the following species as well as unidentified toothed whales and noted low or zero SPUE throughout the area south of Cape Cod, including Muskeget Channel: Atlantic white-side dolphin, bottlenose dolphin, common dolphin, harbor porpoise, pilot whales, Risso's dolphin and white-beaked dolphin. Sightings of toothed whales archived in the North Atlantic Right Whale Consortium sightings database (Right Whale Consortium 2010) were generally distributed to the south of Nantucket Sound (Figure 2.1). However, this does not necessarily reflect spatial distribution patterns, as systematic survey effort in the study area was distributed in a similar manner (See Data Summary, Appendix I). Stranding data obtained from NOAA Fisheries[2] for animals that stranded on or in the vicinity of Martha's Vineyard, Nantucket and nearby islands are listed below and compared to the Right Whale Consortium sightings data from the broader area including the waters to the south as defined in the Data Summary. The stranding data included the above species as well as the pygmy sperm whale, sperm whale and striped dolphin. The strandings of sperm whales are noteworthy, as the species is listed as endangered under the U.S. Endangered Species Act.

Table 1:

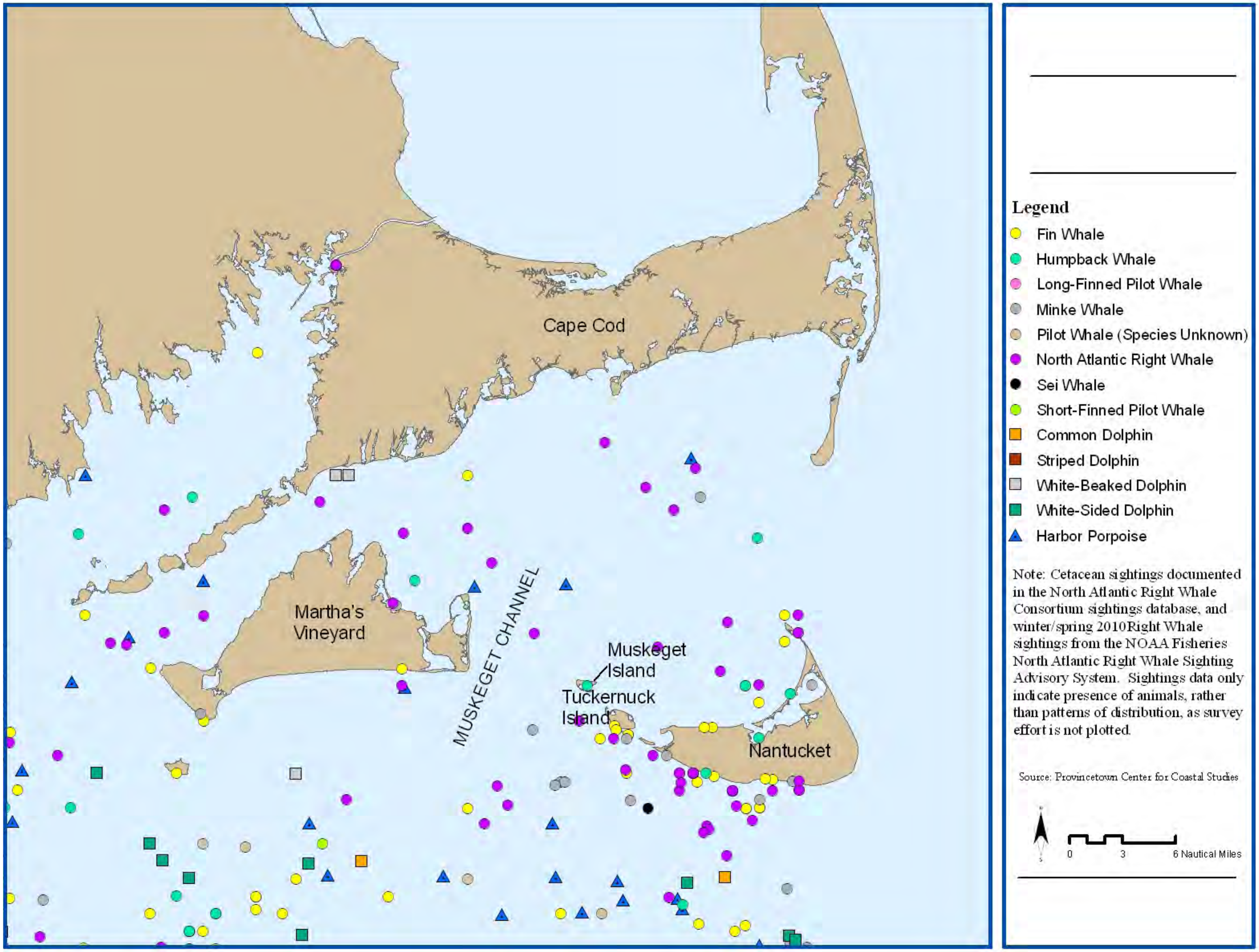
Comparison of strandings data from NOAA Fisheries for animals that stranded on or in the vicinity of Martha's Vineyard, Nantucket, and nearby islands with Right Whale Consortium sightings data for the same region

Species	Number of animals by data source	
	Right Whale Consortium	**NOAA Fisheries Strandings**
Atlantic white-sided dolphin	759	4
Bottlenose dolphin	-	10
Common dolphin	65	31
Harbor porpoise	66	8
Pilot whale (all spp.)	133	22
Pygmy sperm whale	-	7
Risso's dolphin	-	8
Sperm whale	-	4
Striped dolphin	1	8
White-beaked dolphin	50	-

Although extreme caution is necessary when interpreting stranding data as well as sightings data due to lack of systematic effort, it is noteworthy that so few Atlantic white-sided dolphins strand on the islands around the study area when compared to the number of animals documented in the Right Whale Consortium sightings data. Such discrepancies may reflect offshore distributions of this and other species (Figure 2.1).

2. Data courtesy of Tracy Bowen and Mendy Garron (NOAA Fisheries) and contributing stranding networks spanning 1988-2009. These data should not be used out of context or without verification.

Figure 2.1: Cetacean sightings in Southern Massachusetts.



2.2 Pinnipeds

2.2.1 Introduction

Nantucket Sound is home to a resident gray seal (*Halichoerus grypus*) population and a seasonal harbor seal (*Phoca vitulina concolor*) population. Gray seals utilize areas of Nantucket Sound for pupping, molting, foraging and hauling out. Harbor seals are found in Nantucket Sound during the winter months (~September to April) and utilize the Sound for foraging and hauling out. Harbor seals move north of the Massachusetts/New Hampshire border for pupping and molting. Harp (*Pagophilus groendlandicus*) and hooded (*Cystophora cristata)* seals also occur sporadically in Nantucket Sound (see Table A1 for strandings data).

All marine mammals, including gray seals and harbor seals, are protected by the Marine Mammal Protection Act of 1972 as amended.

2.2.2 Species Descriptions

2.2.2.1 Harbor seal (*Phoca vitulina*)

Harbor seals are widely distributed, occurring in the North Atlantic and North Pacific Oceans. There are four sub-populations: *P.v. vitulina* in the Eastern Atlantic, *P.v. concolor* in the Western Atlantic, *P.v. richardsi* in the Eastern Pacific and *P.v. stejnegeri* in the Western Pacific (King, 1980).

Adult males and females can measure up to 1.5 m and weigh 110 kg and 90 kg respectively. Generally, males mature at 4-6 years while females mature slightly younger, at 3-4 years of age (Katona *et al.,* 1993; Burns, 2009). Pups are often born in the inter-tidal zone and therefore can swim minutes after birth (Reeves *et al.,* 1992). In the U.S. Atlantic, harbor seals pups are born in May and June from the Isle of Shoals, New Hampshire northwards along the Maine coast (Gilbert *et al.,* 2005). Harbor seals remain in this region through July and August to molt. In September, a subset of the population moves south into Southern New England and west into Long Island Sound (Schneider and Payne, 1983).

The stock structure of the U.S. Atlantic harbor seal population is not understood. Harbor seals are a non-migratory marine mammal. A subset of Atlantic Coast harbor seals moves to Southern New England from September to April (Schneider and Payne, 1983). Waring *et al.* (2006) showed that at least some of the seals return to the Maine coast for pupping and molting (May-August). The relationship between the U.S. and Canadian Atlantic harbor seals in unclear, but Rosenfeld *et al.* (1988) suggested that some Canadian seals over-wintered in the U.S. and thus would be a trans-boundary stock.

2.2.2.2 Gray seal (*Halichoerus grypus*)

Gray seals are found throughout the cold temperate waters of the North Atlantic (King, 1980). The species is generally divided into three distinct populations based on cranial differences (Rice, 1998) and mtDNA studies (Boskovic *et al.,* 1996): the Baltic Sea, the Northeast Atlantic (U.K.) and the Northwest Atlantic (Canada & U.S.) population. The time of breeding varies geographically, with seals in the Baltic Sea pupping in March, in the Northeastern Atlantic in September-November and in the Western North Atlantic in December-February (King, 1983).

The gray seal is a large, sexually dimorphic species. Males reach a size of up to 2.3 m and 300-350 kg, while females reach a maximum size of 2.0 m and 150-200 kg (Hannah, 1998). Gray seals are gregarious and gather in large groups during the pupping/breeding and molting seasons. Gray seals are unique in that they can breed on sandy beaches, rocky ledges, ice (Reeves *et al.,* 1992) or in caves (Hewer, 1974). In the U.S. Atlantic, gray seals can be found on a year-round basis in Nantucket Sound and Mid-coast Maine (Wood LaFond, 2009). There is no seasonal movement similar to that observed in the harbor seal.

2.2.2.3 Diet

Harbor and gray seals are thought to be generalists that forage on available prey. There have been several seal food habits studies conducted in the Nantucket Sound/Cape Cod area which provide a general idea of which prey species are important in this region. Payne and Selzer (1989) analyzed 248 harbor seal scat samples. Ninety-five percent of these samples came from Race Point (Provincetown), Jeremy Point (Wellfleet) and Monomoy Island and were collected from 1984-1987. Sand lance (*Ammodytes americanus*) was the single dominant prey found in the Cape Cod samples.

In another study on harbor seal food habits, Ferland (1999) analyzed 172 scat samples collected from December 1998 to May 1999 at three sites on Cape Cod: Jeremy Point (Wellfleet), Chatham Harbor and Gull Island (Elizabeth Islands). Thirty-one of the samples were collected at Jeremy Point where both harbor and gray seals were hauled out. These samples therefore could not be assigned to a seal species. Sand lance was the most frequently occurring (85%) prey species and also provided the largest percentage of wet mass (50%) in the seals' diet. This was followed by winter flounder (*Pseudopleuronectes americanus*) – 32% wet mass. Ampela (2009) analyzed 305 gray seal scat samples collected on Monomoy and Muskeget Islands from winter 2004 through winter 2008. Sand lance provided the largest percent wet weight (53%) in this study as well, followed by winter flounder (19%) and Atlantic Cod (*Gadus morhua*) (6.4%).

2.2.3 Distribution & Abundance in the Nantucket Sound – Muskeget Channel Area

2.2.3.1 Distribution & Abundance: Harbor Seal

In order to understand the abundance and seasonal distribution of harbor seals in Nantucket Sound, it is necessary to consider the U.S. Atlantic harbor seal population as a whole. Harbor seals use Southern New England (including Nantucket Sound) for hauling out and foraging during the fall, winter and spring but return to Maine (or possibly Canada) for pupping, mating and molting (Waring *et al.,* 2006; Figure 2.2). Waring *et al.* (2006) reported that 75% of the harbor seals radio-tagged in Chatham, Massachusetts during the month of March relocated to Maine later in the spring. Gilbert *et al.* (2005) describes a 6.6% rate of increase in the number of harbor seals hauled out during the pupping season from 1981 to 2001 along the Maine coast (Figure 2.3). The corrected count for 2001 was 99,340 seals and is an estimate of the total U.S. Atlantic harbor seal population. Although not a current estimate, this data set demonstrates a steady increase in the number of harbor seals in U.S. Atlantic waters.

Payne and Selzer (1989) documented winter harbor seal abundance in Southern New England from 1983 through 1987 (Figure 2.4). As with Gilbert's data set, these data provide evidence of an increase in the number of harbor seals in the Atlantic U.S. This trend is even more apparent when Payne and Selzer's counts are compared to Barlas's (1999; Figure 2.4). Barlas (1999) collected aerial survey data in the Plymouth to Woods Hole region between 1998 and 1999. This study provides the most recent harbor seal abundance estimates for Southern Massachusetts including Nantucket Sound (Figure 2.5), and also shows a winter peak in harbor seal abundance. Barlas also surveyed west of Martha's Vineyard and counted 198 harbor seals in March of 1999 on Nomans Land (a National Wildlife Refuge).

deHart (2002) documented peak harbor seal abundance in Woods Hole in the February to April time period. He also found a slight increase in the number of harbor seals hauled out in Woods Hole from 2001 (n = 164) to 2002 (n = 184; Figure 2.6). Counts of harbor seals at the Nantucket jetties (NMFS unpub. data) show presence there during the winter months from 2004 to 2008 (Figure 2.7). Finally, a study of harbor seal abundance and seasonal distribution in Narragansett Bay, Rhode Island (Schroeder, 2000) provides additional evidence of an increase in the number of harbor seals in Southern New England and a seasonal peak during the winter in this region (Figure 2.8).

From these data sets, Monomoy Island (a National Wildlife Refuge) is the only location in Nantucket Sound where there has been a documented decline in the number of harbor seals (Figure 2.9). This decline has occurred as the Nantucket Sound gray seal has population has been growing (Figure 2.10).

2.2.3.2 Distribution & Abundance: Gray Seal

Muskeget Island is the largest gray seal pupping colony in the U.S. Pup counts from aerial survey data are available in Rough (1995, 2000) and Wood LaFond (2009) from 1991 through 2008 (Figure 2.10). No data is available for 2000. The number of pups born on Muskeget has increased dramatically over this time period. Only 6 pups were born in 1991. Seventeen years later, on 15 January 2008, a minimum of 2,090 pups were counted.

The data available outside of the pupping season is older and not as continuous. Reports by Rough (1995, 2000) and Barlas (1999) contain gray seal counts during the spring molt season at Muskeget and Monomoy Islands for several years in the 1990s (Figures 2.11 & 2.12). Although out of date, these counts also show an increase in the number of gray seals in Nantucket Sound during the months of April and May. Ampela (2009) collected scat samples at Muskeget and/or Monomoy Islands during every season from winter 2004 to winter 2008 and thereby documented a continued presence of gray seals in Nantucket Sound. In addition to these sites, when Wasque Shoal is available, gray seals utilize it (Wood LaFond, *pers. obs.*). Wasque Shoal is located between Nantucket and Martha's Vineyard Islands and periodically appears due to strong currents and storms. Sette (*unpublished data*) has also documented gray seals on tidal haul-outs near Gull and Penikese Islands (Elizabeth Islands). These studies together provide evidence for an increasing, permanent gray seal population in Nantucket Sound.

Whalenet (http://whale.wheelock.edu/), an educational program at Wheelock College funded by the National Science Foundation, has worked with scientists to deploy numerous satellite tags on harbor and gray seals (Table A4). Thirteen of the tagged seals spent time in Nantucket Sound or around Cape Cod.

Results of genetic analyses have shown that U.S. gray seals constitute a trans-boundary stock. To identify the source population for the recovering U.S. gray seal population and to assess the stock structure of gray seals in the Northwestern Atlantic, Wood LaFond (2009) collected a total of 231 tissue samples from both Canadian and U.S. populations for genetic analyses. Samples were collected (mostly from pups) at three sites during the pupping season: Sable Island (Canada), the Gulf of St. Lawrence (Canada) and Muskeget Island (Massachusetts). These analyses showed that there was no significant difference between the three sites sampled, demonstrating that an adequate number of individual gray seals are moving between these pupping sites for the sites to be indistinguishable from each other. See Wood LaFond (2009) for more detail.

2.2.3.3 Historic Presence of Seals in Nantucket Sound

As reported in Ritchie (1969), harbor and gray seal remains were found in archeological sites on Martha's Vineyard. The most extensive sources of information on gray seal sightings throughout the Northeastern U.S. during the 20th century were the reports written to U.S. Federal and State agencies by Valerie Rough, who documented the re-colonization of Muskeget Island in Nantucket Sound by gray seals. Her accounts are summarized in Wood LaFond (2009) and are useful in understanding the status of gray seals in the early to late 20th century in two ways: they document that people were looking for them, and the sparseness of their sightings shows that gray seals were probably truly rare throughout the U.S. during most of the 20th century. Table A3 summarizes gray seal observations on Cape Cod, Martha's Vineyard Island and Nantucket Island. Unfortunately, such reports do not exist for harbor seals during this time period.

In addition to literature, museum collections were searched for evidence of harbor and gray seals in the Nantucket Sound/Cape Cod area. These collections contained records from 1632 through the present day (Tables A2 & A3).

These records demonstrate that both harbor and gray seals had a historic presence in Nantucket Sound.

As summarized in Lelli *et al.* (2009), seal bounties existed in the states of Massachusetts and Maine during the late 19th and early 20th century. These bounties were not species-specific and likely targeted both harbor and gray seals. Under the bounty systems, hunters were paid $1-5 U.S. for each seal killed. The Massachusetts bounty existed from years 1888 to 1908 and from 1919 to 1962. The statewide Maine bounty was briefer, only lasting 10 years, from 1895 to 1905. Through an extensive search of state and county records, (Lelli *et al.,* 2009) found records of 15,690 seal bounties paid in Massachusetts and 24,831 seal bounties paid in Maine during the time of their respective bounties. There is evidence of cheating (*e.g*. a hunter would turn one seal pelt into multiple noses or tails), so the bounty records probably do not accurately reflect the actual number of seals killed. These records do, however, demonstrate that there was hunting pressure on seals in the Northeastern U.S. well into the middle of the 20th century. In 1965 the state of Massachusetts enacted a law to protect the gray seal, and in 1972 the U.S. government passed the Marine Mammal Protection Act, which provided blanket protection in all states. These two laws acted to protect seals in the U.S.

Figure 2.2: Seal pupping colonies and haul-out sites in the Nantucket Sound – Muskeget Channel area.

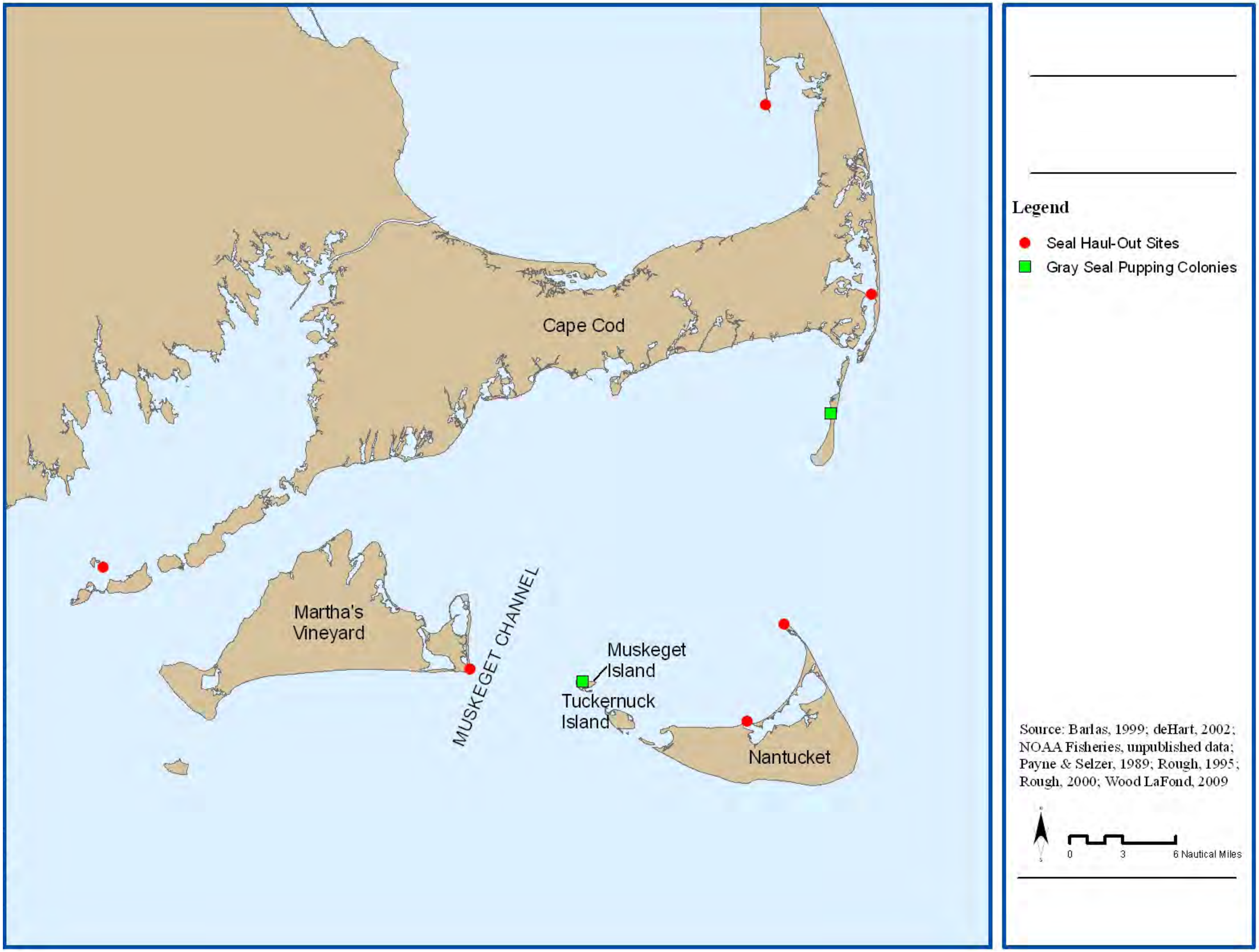


Figure 2.3: Harbor seal abundance in Maine, 1981-2001 (Gilbert et al., 2005).

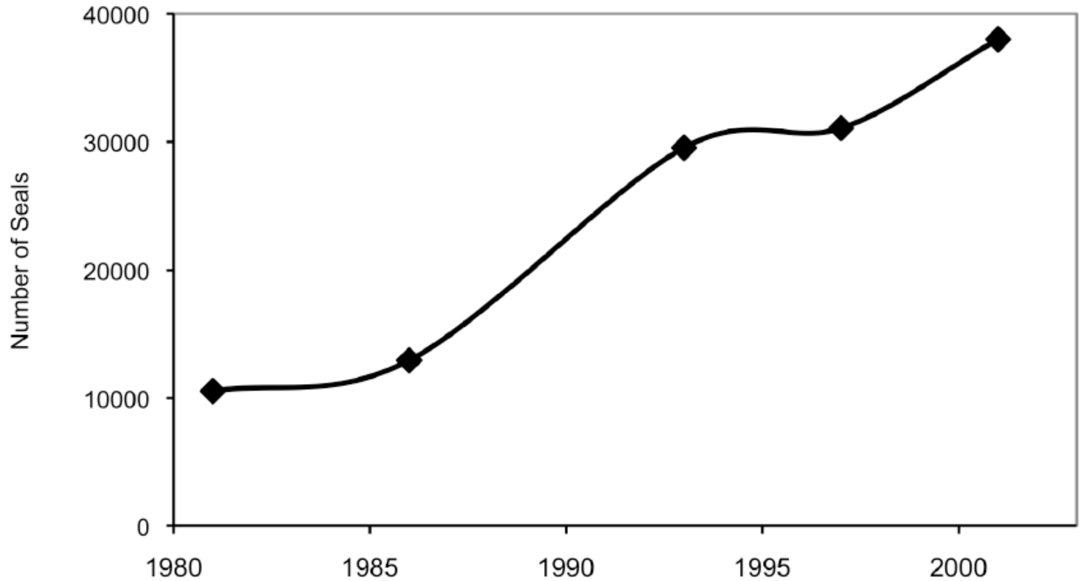


Figure 2.4: Harbor seal abundance from Plymouth to Woods Hole, Massachusetts (Payne & Selzer, 1989; Barlas 1999).

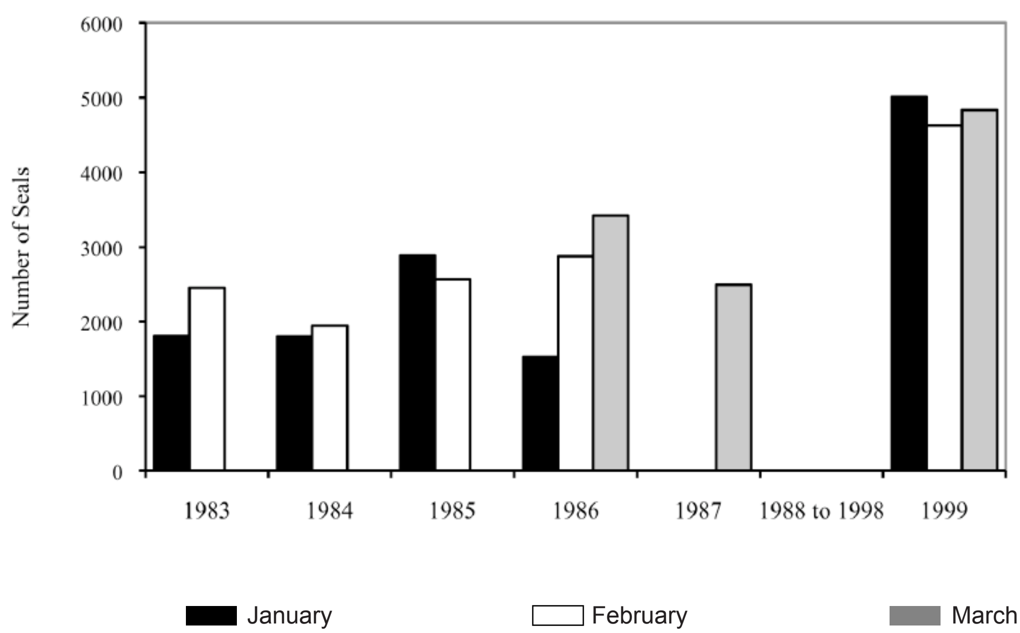


Figure 2.5: Harbor seal abundance 1998-99: Plymouth to Woods Hole (Barlas, 1999).

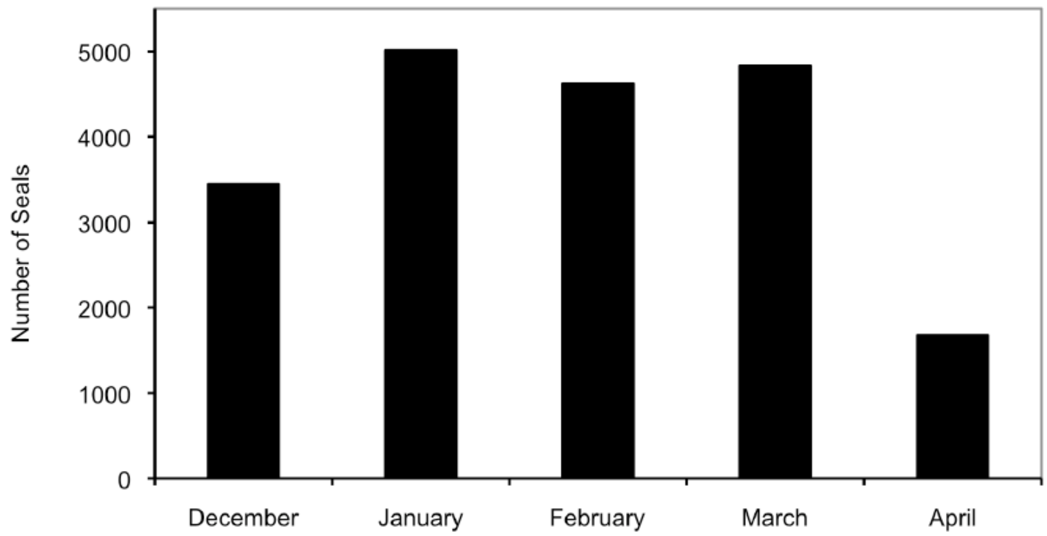


Figure 2.6: Harbor seal abundance in Woods Hole, Massachustts (deHart, 2002).

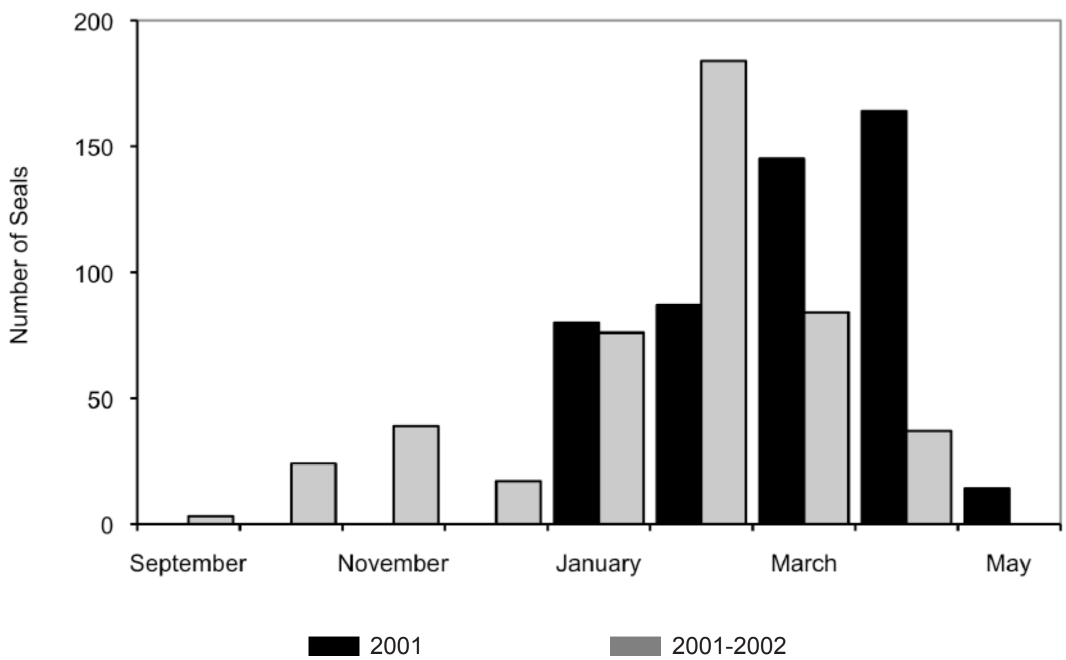


Figure 2.7: Harbor seal abundance at the Nantucket, Massachusetts jetties, 2004-08 (NMFS unpublished data).

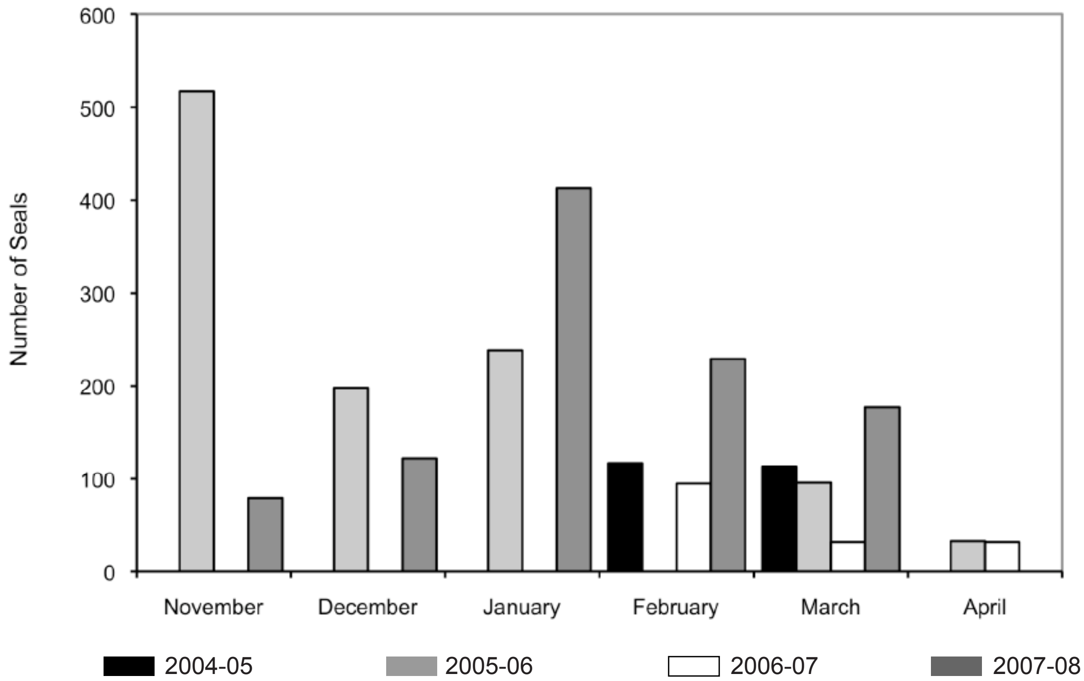


Figure 2.8: Seasonal and annual trends in harbor seal abundance in Narragansett Bay, Rhode Island (Schroder 2000).

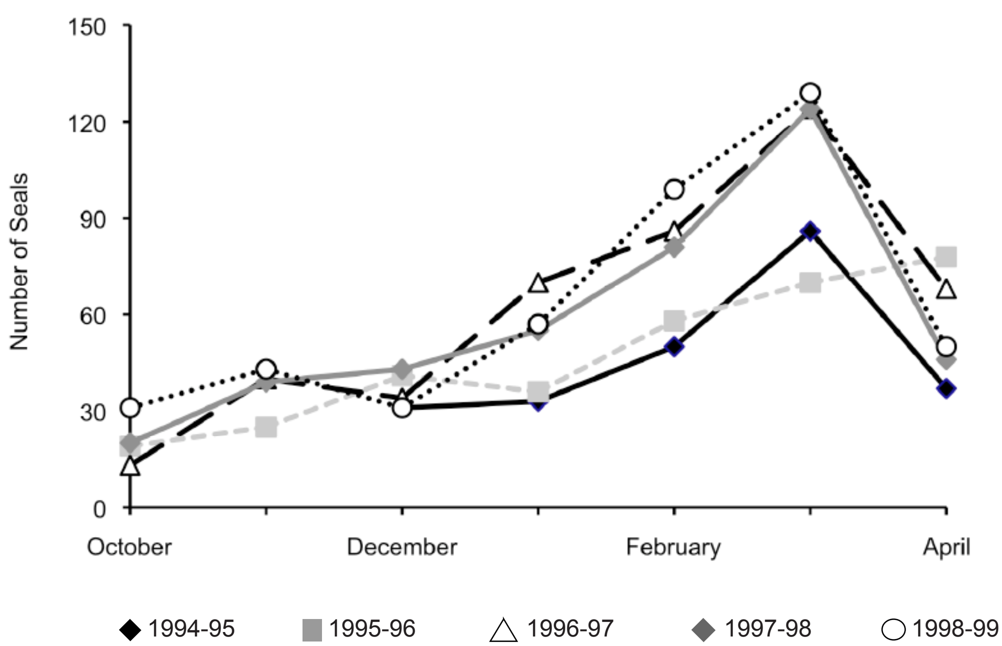


Figure 2.9: Harbor seal abundance on Monomoy Island, Massachusetts (Payne & Selzer, 1989; Barlas, 1999).

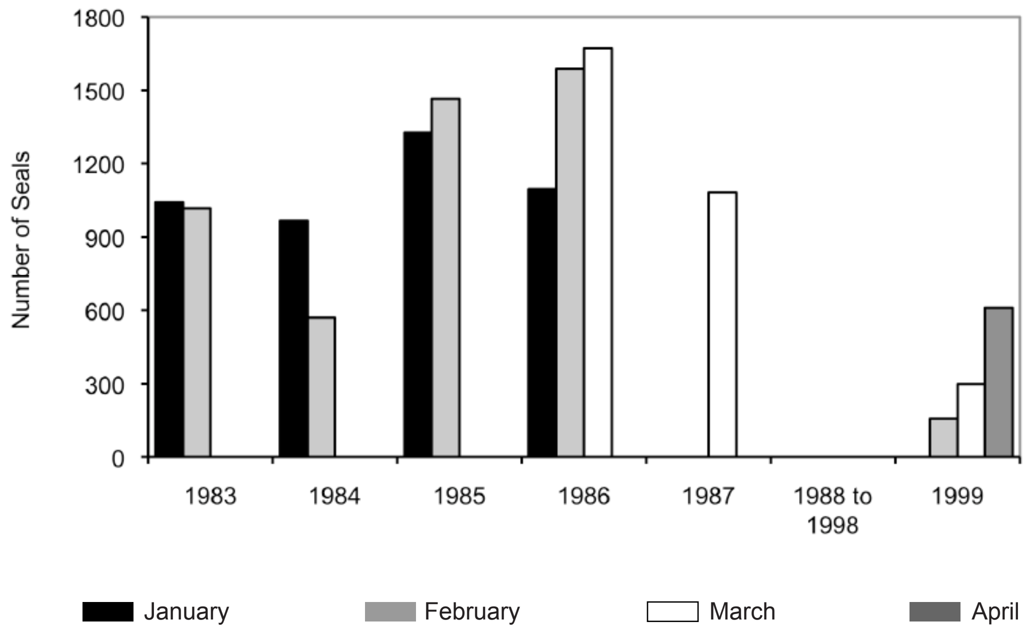


Figure 2.10: Gray seal pup counts on Muskeget Island, Massachusetts, 1991-2008 (Rough, 1995, 2000; Wood LaFond, 2009).

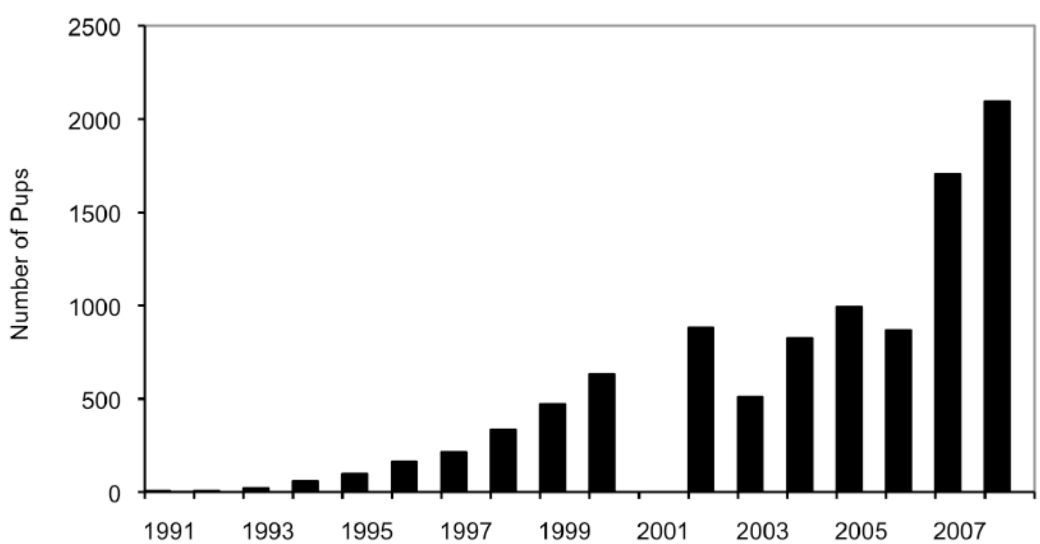


Figure 2.11: Gray seal molt counts for Muskeget Island, Massachusetts, 1992-99 (Rough, 2000; Barlas, 1999).

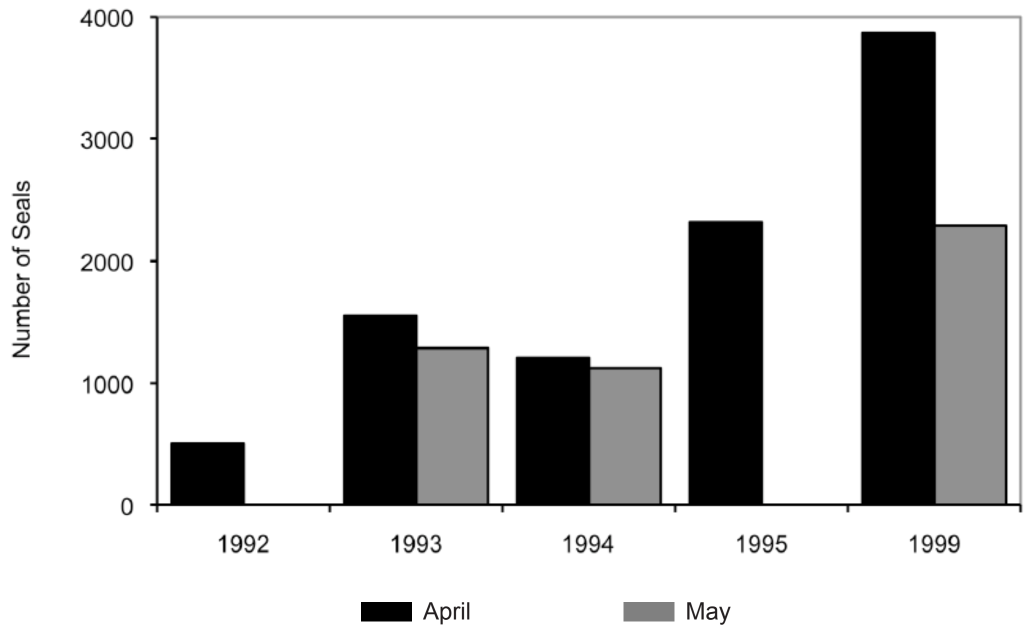
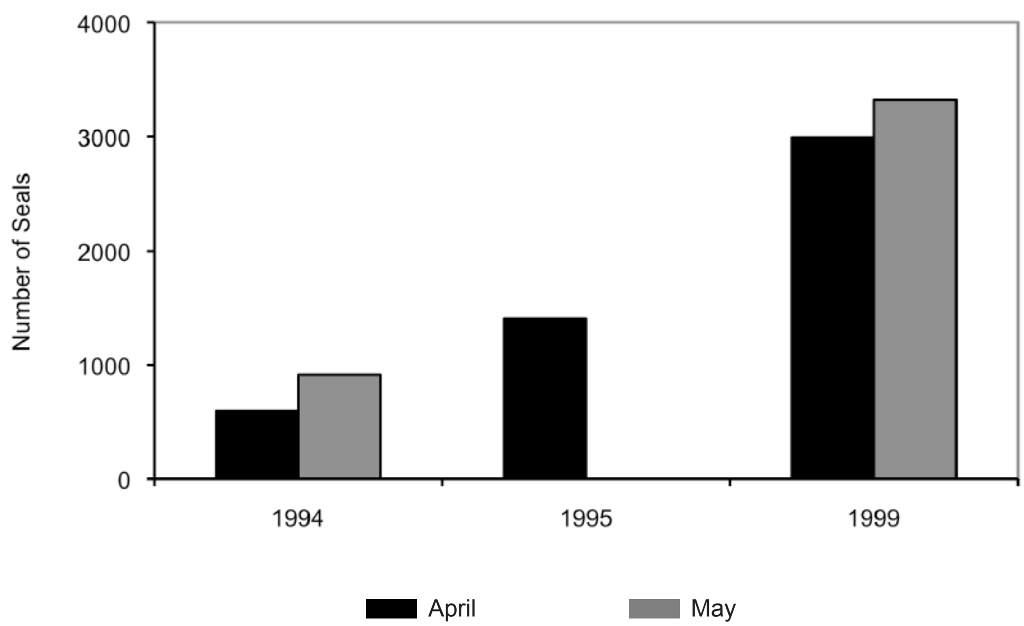


Figure 2.12: Gray seal molt counts for Monomoy Island, Massachusetts in 1994, 1995 and 1999 (Rough, 2000; Barlas, 1999).



2.3 Dermochelid & Chelonids

2.3.1 Introduction

There are four species of sea turtles that have been recorded in Nantucket Sound either seasonally foraging or transiting the waters south of Cape Cod: leatherback *(Dermochelys coriacea),* loggerhead *(Caretta caretta)*, green *(Chelonia mydas)* and Kemp's ridley *(Lepidochelys kempi)* (Jones, 1886; Lazell, 1976; Lazell, 1980; USFWS and NMFS, 1992; Prescott, 1988; Dwyer *et al.,* 2003; Perkins *et al.,* 2003; Morreale and Standora, 2005; Sadoti *et al.,* 2005; Ernst and Lovich, 2009; http://www.seaturtlesightings.org/, 2010). A fifth species, the hawksbill *(Eretmochelys imbricate),* is considered to be a "rare" vagrant to New England (Lazell, 1976). Depending on the age and species, sea turtles will typically migrate offshore or south to their nesting beaches in fall as local water temperatures decrease (Bleakney, 1965; Lazell, 1976; Shoop and Kenney, 1992; Wynne and Schwartz, 1999; Ernst and Lovich, 2009).

Another marine reptile, the northern diamond-backed terrapin *(Malaclemys terrapin),* is a salt marsh turtle and year round resident of Massachusetts. While not considered a sea turtle, the northern diamond-backed terrapin inhabits estuaries, rivers, creeks, salt marshes and mud and is known to nest in dry, sandy uplands near its foraging areas (Babcock, 1926; Lazell, 1976; Lewis, 2002; Brennessel, 2007). There are records of northern diamond-backed terrapin from areas along the southwest coastal region of Cape Cod (Babcock, 1926; Lazell, 1976; Brennessel, 2007).

All sea turtles included in this report are listed under the Endangered Species Act (ESA) of 1973 except for the diamond-backed terrapin, which is listed by Massachusetts as threatened. The leatherback, Kemp's ridley and hawksbill are listed as endangered at the federal and state level; the loggerhead and green are listed as threatened at the federal and state level (NMFS and USFWS, 1991a; NMFS and USFWS, 1991b; USFWS and NMFS, 1992; NMFS and USFWS, 1995; NMFS and USFWS, 1998b; NMFS and USFWS, 2007a, 2007b, 2007c, 2007d; http://www.mass.gov/dfwele/dfw/nhesp/species_info/mesa_list/mesa_list.htm, 2010). The International Union for Conservation of Nature (IUCN) Red List categorizes loggerhead and green sea turtles as "endangered" (Marine Turtle Specialist Group, 1996a; Seminoff, 2004), while the leatherback, Kemp's ridley and hawksbill are listed as "critically endangered" (Marine Turtle Specialist Group, 1996b; Martinez, 2000; Mortimer and Donnelly, 2008).

2.3.2 Species Descriptions

2.3.2.1 Dermochelid

Leatherback turtles are the largest of all sea turtle species and the only living species in the genus *Dermochelys* (Lazell, 1976; Ernst and Lovich, 2009). Leatherbacks can be distinguished from all other sea turtles found in New England by their large size and ridged carapace. Leatherbacks lack the hard keratin scutes covering the carapace of other sea turtles. Instead of scutes, they have a thick, leathery skin that covers their carapace (Lazell, 1976; Wynn and Schwartz, 1999; Ernst and Lovich, 2009). The surface of the leatherback's carapace is colored dark grey to black and has pink and white blotches and spots. An adult leatherback carapace can measure up to 1.8 m in length and it typically weighs 727 kg to 1 ton (Wynn and Schwartz, 1999; Ernst and Lovich, 2009). Leatherbacks feed almost exclusively on gelatinous animals such as jellyfish and salps (Lazell, 1976; Bjorndal, 1997; Wynn and Schwartz, 1999; Ernst and Lovich, 2009). Leatherback turtle nesting grounds have been documented around the world (Ernst and Lovich, 2009). Adult leatherbacks have been sighted along the entire continental coast of the United States up into the Gulf of Maine and south to Puerto Rico, the U.S. Virgin Islands and into the Gulf of Mexico (Wynne and Schwartz, 1999; Ernst and Lovich, 2009). Recorded sightings of leatherback turtles (Figures 2.13-2.15) suggest that they are typically in New England, including Nantucket Sound, between May and October (Bleakney, 1965; Lazell, 1976; Prescott, 1988; Shoop and Kenney, 1992; Wynne and Schwartz, 1999; NMFS and USFWS, 2001; Sadoti *et al.,* 2005; http://seaturtlesightings.org/speciesmap.html).

2.3.2.2 Chelonids

Loggerhead turtles, named for their proportionally large heads, are characterized by their heart-shaped carapace and the brown coloration of adults and subadults. The plastron, or ventral surface of the shell, is generally a pale yellow. The carapace length of adults in the U.S. is approximately 0.92 m and the average weight of an adult is about 115 kg (Wynn and Schwartz, 1999; Ernst and Lovich, 2009). Subadult and adult loggerheads feed mainly upon benthic invertebrates such as whelks and conch (Bjorndal, 1997; Wynn and Schwartz, 1999; NMFS and USFWS, 2001; Ernst and Lovich, 2009). Loggerheads are found throughout the temperate and tropical regions of the Atlantic, Pacific and Indian Oceans. Loggerhead turtles have been observed in the Northeast, including Nantucket Sound, as early as June (http://seaturtlesightings.org/speciesmap.html), and the majority leave the Northeast by late fall (Figure 2.14; Shoop and Kenney, 1992; Sadoti *et al.,* 2005; http://seaturtlesightings.org/speciesmap.html). As summarized in Morreale and Standora (2005), the Western North Atlantic is considered to be an important developmental habitat for loggerhead turtles.

Green turtles are hard-shelled sea turtles named for the greenish color of the cartilage and fat deposits that surround their internal organs. However, the carapace of a green sea turtle is typically dark black, brown or greenish yellow with a yellowish white plastron ventrally. Hatchlings are just 50 mm long, while adults can grow to 1 m long and an average weight of 150 kg (Wynne and Schwartz, 1999; Wynne and Schwartz, 1999; Ernst and Lovich, 2009). Adult green sea turtles feed mainly on algae and seagrasses (Bjorndal, 1997; Ernst and Lovich, 2009). Green sea turtles are globally distributed and are generally found in tropical and subtropical waters along continental coasts and islands (Wynn and Schwartz, 1999; Ernst and Lovich, 2009). Green sea turtles are seasonal visitors to the Northeastern waters of Massachusetts typically between May and October (Morreale and Standora, 1998; Wynne and Schwartz, 1999; http://seaturtlesightings.org/speciesmap.html, 2010).

Kemp's ridley turtles are the smallest known sea turtle species in the world (Marquez et al., 2005) with adults generally weighing less than 40-50 kg and measuring approximately 0.58 to 0.80 m. The color of the carapace changes significantly as they age. The carapace of a hatchling can be grayish black, while adults have a lighter grayish or olive-colored carapace and a creamy white or yellowish plastron (Wynne and Schwartz, 1999; Ernst and Lovich, 2009). Their diet is comprised mainly of crabs but can also include shrimp and mollusks (Bjorndal, 1997; Wynne and Schwartz, 1999; Ernst and Lovich, 2009). Kemp's ridley turtles are known to range from Nova Scotia to Mexico and have been documented in Nantucket Sound and Vineyard Sound (Figure 2.14; Lazell, 1976; Musick and Limpus, 1997; Sadoti et al., 2005; http://seaturtlesightings.org/speciesmap.html). As reported in Morreale and Standora (2005), the Western North Atlantic is considered to be important developmental habitat for Kemp's ridley turtles.

The **hawksbill turtle** is a small- to medium-sized sea turtle with a narrow pointed beak and small head. The carapace of the hawksbill is uniquely characterized by scutes that overlap with a streaked or marbled yellow or brown coloration. The edge of the carapace is often serrated in younger animals. Hawksbill turtles are typically less than 1 m in length with an average weight of 82 kg (Ernst and Lovich, 2009; Wynne and Schwartz, 1999). Hawksbills utilize different habitats at different stages of their life cycle. It is believed that post-hatchling hawksbills are pelagic (Wynne and Schwartz, 1999; NMFS and USFWS, 1993) and then subadults and adults reenter coastal areas and feed primarily on sponges (Bjorndal, 1997; Wynne and Schwartz, 1999; Ernst and Lovich, 2009). The hawksbill is considered to be "rare" in New England waters (Lazell, 1976; http://www.mass.gov/dfwele/dfw/wildlife/facts/.../herp_list.html), with only three records from Massachusetts (B. Prescott, *pers. comm.*, 19 September 2010).

The **northern diamond-backed terrapin** is a medium-sized salt marsh turtle (Lewis, 2002; Ernst and Lovich, 2009). The carapace can be grayish, green, black and/or light brown. Northern diamond-backed terrapins have concentric ring patterns on their carapace and a ridged mid-line keel. Adult females range from 15-23 cm in length and are typically larger than adult males, which range from 10-15 cm. Hatchlings look very similar to adults and are approximately 2.6 cm length (Lazell, 1976; Lewis, 2002; Ernst and Lovich, 2009). Salt marshes are very important foraging areas for northern diamond-backed terrapins (Lazell, 1976; Lewis, 2002; Ernst and Lovich, 2009). Their diet includes gastropods, crabs, mollusks, insects, fish and carrion (Lazell, 1976; Lewis, 2002; Brennessel, 2007; Ernst and Lovich, 2009).

During the spring, male and female diamond-backed terrapins come together in coves or small bays to mate (Lewis, 2002; Ernst and Lovich, 2009). Once mated, the females will travel upland as far as 0.4 km to prepare a nest for her eggs (Lewis, 2002; Brennessel, 2007). Yearicks *et al.* (1981) reported that northern diamond-backed terrapins hibernate in winter under water, either singly or in groups on the bottom, buried in mud or in the side of mud banks. Diamond-backed terrapins are the only species of chelonid included in this report that overwinter in Massachusetts (Lazell, 1976; Ernst and Lovich, 2009). The northern diamond-backed terrapin's range includes the Atlantic and Gulf Coasts from Cape Cod, Massachusetts to southern Texas and the Florida Keys (Lazell, 1976; Ernst and Lovich, 2009). There are records of northern diamond-backed terrapins from areas along the southwest coastal region of Cape Cod (Babcock, 1926; Lazell, 1976; Brennessel, 2007).

2.3.3 Distribution & Abundance in the Nantucket Sound – Muskeget Channel Area

Data on sea turtle distribution and abundance has been collated from a number of sources.

The Sea Turtle Sighting Hotline for Southern New England Boaters was initiated in 2002. Its primary goals are to document where and when sea turtles are seen in Southern New England waters and to alert boaters to the presence of sea turtles in the summer and fall. Data points included in the hotline database do not represent a systematic survey, nor do they represent an accurate count of sea turtles since multiple calls may report the same individual turtle. The majority of hotline reports are from waters around Cape Cod, including Buzzards Bay, Vineyard Sound, Nantucket Sound and Cape Cod Bay. Hotspots have been noted off Sakonnet Point (Rhode Island) and near Lucas Shoal in Vineyard Sound. Many of the August sightings are from the recreational fishing areas south of Martha's Vineyard and Nantucket Islands. Sightings are plotted on maps posted on the hotline's website: www.seaturtlesightings.org (K. Moore Dourdeville, *pers. comm.*, 24 August 2010).

Since initiating **satellite tagging** of leatherback turtles in Nantucket Sound, researcher Kara Dodge from the University of New Hampshire Large Pelagics Research Center has tagged twenty leatherbacks off Massachusetts. Based on her track analysis, three of the twenty turtles may have navigated through Muskeget Channel during the monitoring period. No turtles in her study took up residence in Muskeget Channel for any period of time, primarily using it to move between Nantucket Sound and regions south of Martha's Vineyard and Nantucket Islands. In 2008, George Breen, a spotter pilot utilized by the research team, reported seeing three leatherbacks using Muskeget Channel. Based on her work to date, Dodge suggests that leatherbacks appear to favor areas where tidal fronts may entrain and aggregate gelatinous zooplankton, thus forming dense prey patches and enabling leatherbacks to forage efficiently (K. Dodge, *pers. comm.,* 26 August 2010).

In late fall and winter when the ocean environment cools, sea turtles remaining in Massachusetts waters can become "cold stunned," a form of hypothermic reaction caused by prolonged exposure to cold water temperatures (http://www.nero.noaa.gov/prot_res/stranding/cold.html). Severely cold-stunned turtles become lethargic and drift helplessly, resulting in animals coming ashore alive (Lazell, 1976). As summarized in Dodge *et al.* (2008) from 1979 to 2002, 1,289 sub-adult and adult cold-stunned marine turtles were discovered stranded on Cape Cod beaches. Of those turtles stranded, 76.6% were Kemp's ridley, 21.1% loggerhead, 2.3% green and 0.03% hybrid. These data and other reports suggest that the northeast coast might be an important foraging area for these species (Lazell, 1976; Lazell, 1980; Burke *et al.,* 1991; Morreale and Sandora, 2005).

The Massachusetts Sea Turtle Disentanglement Network (MASTDN) was formed to respond to and document bycatch issues related to sea turtles in and around the state waters of Massachusetts. From its inception in 2005 to the present (12 September 2010), MASTDN has received 77 confirmed entanglement reports. Of those, 46 reports are from the waters of Nantucket Sound, Vineyard Sound and Buzzards Bay (Fig. 2.13). Entanglement reports are received seasonally from May to October with a peak during August. A majority of the reports in the study area involve leatherback sea turtles (n = 44, 96%), with only two (n = 2, 4%) involving species other than leatherbacks: one loggerhead and one turtle unidentifiable due to decomposition. Support for this work is provided by ESA Section 6 in conjunction with Massachusetts Division of Marine Fisheries. Data can only be used for the purpose of this literature review and should not be used for any other reason or application without the express written consent of PCCS.

Figure 2.13: Confirmed entangled sea turtle sightings in waters south of Cape Cod as reported to the Massachusetts Sea Turtle Disentanglement Network (MASTDN). Support for this work is provided by ESA Section 6 in conjunction with Massachusetts Division of Marine Fisheries. Data can only be used for the purpose of this literature review and should not be used for any other reason or application without the express written consent of the PCCS.

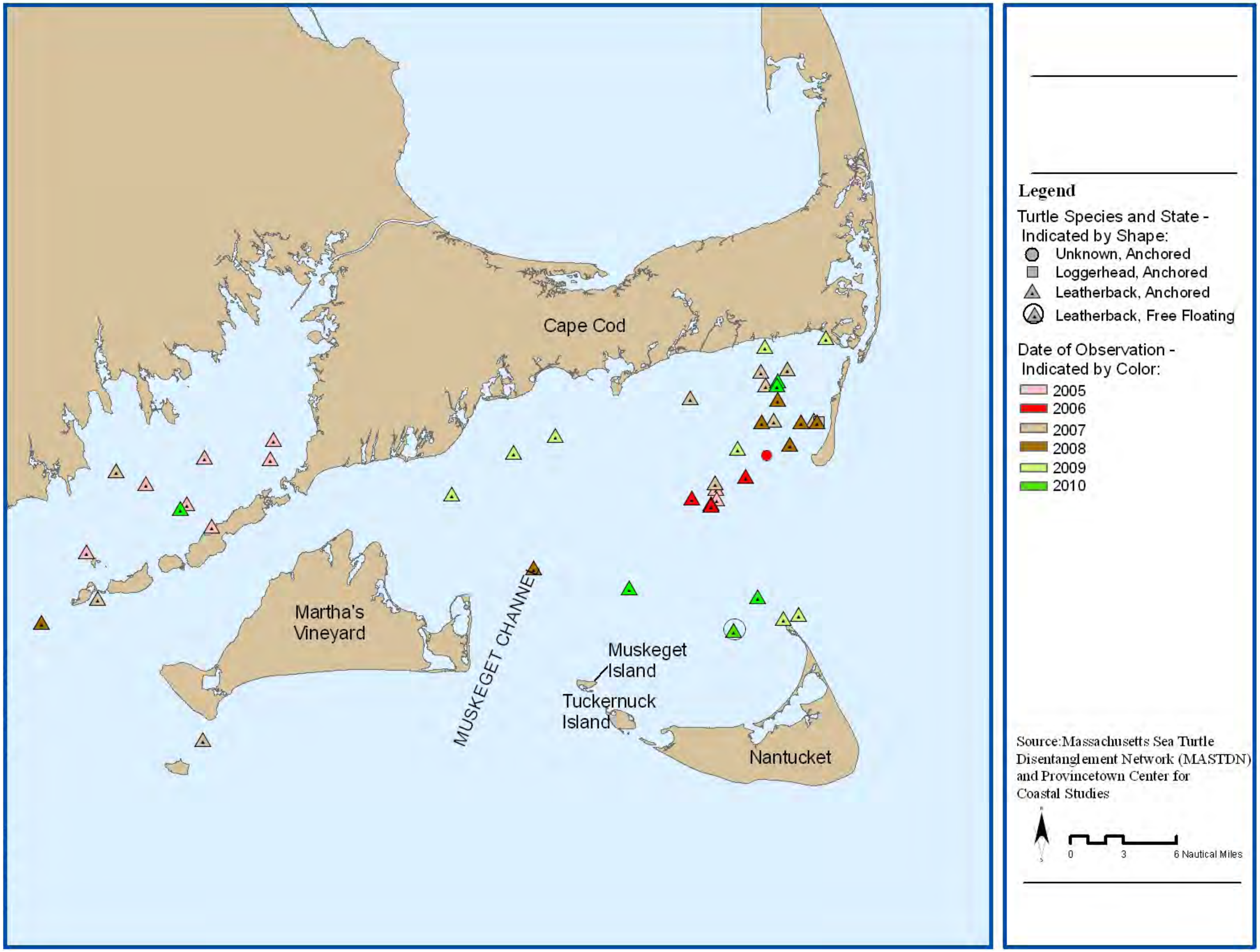


Figure 2.14: Opportunistic sea turtle sightings recorded by Massachusetts Audubon during aerial surveys of tern activity in Nantucket Sound as part of an assessment for the Cape Wind energy project. Surveys were completed in August and September from 2002-2004.

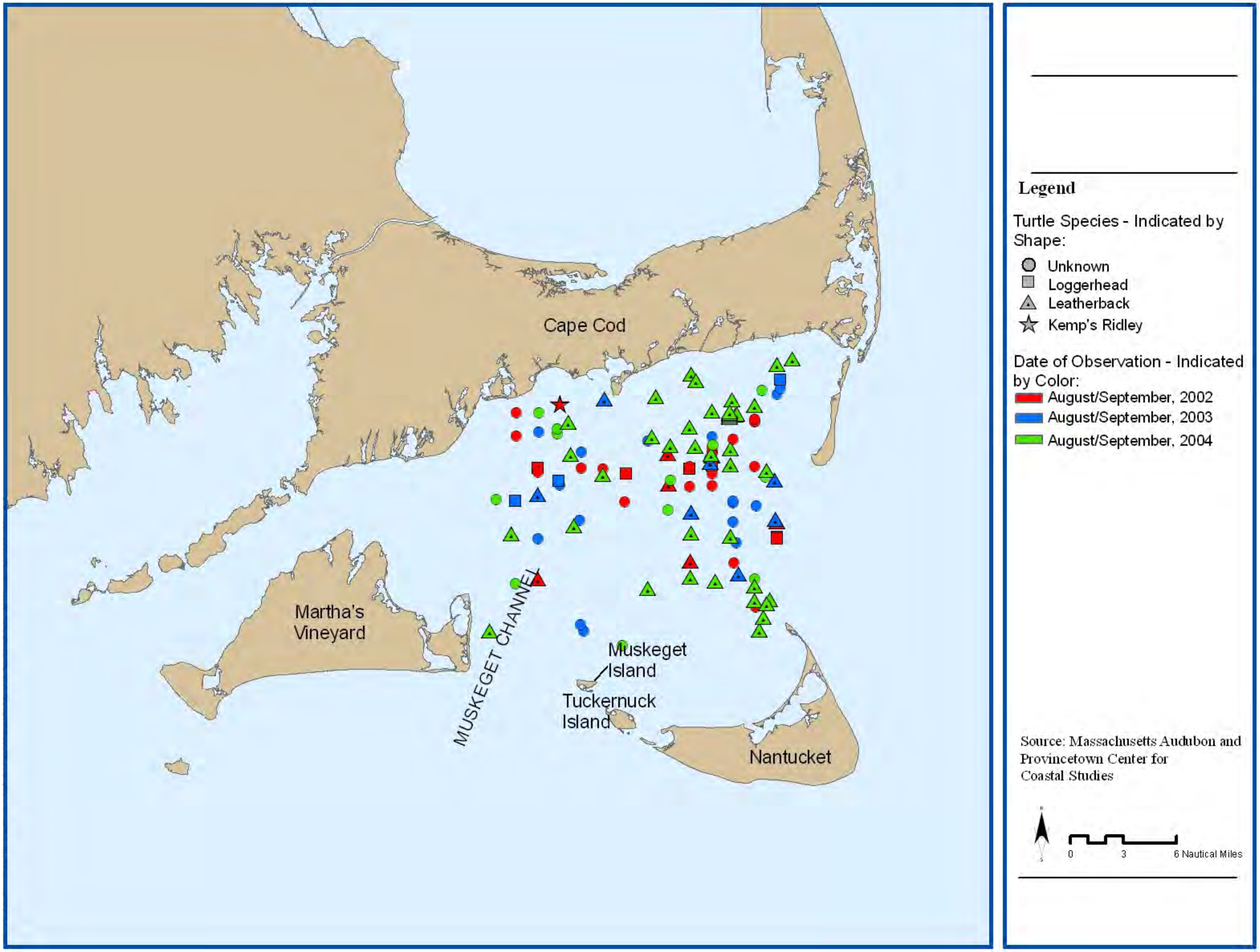
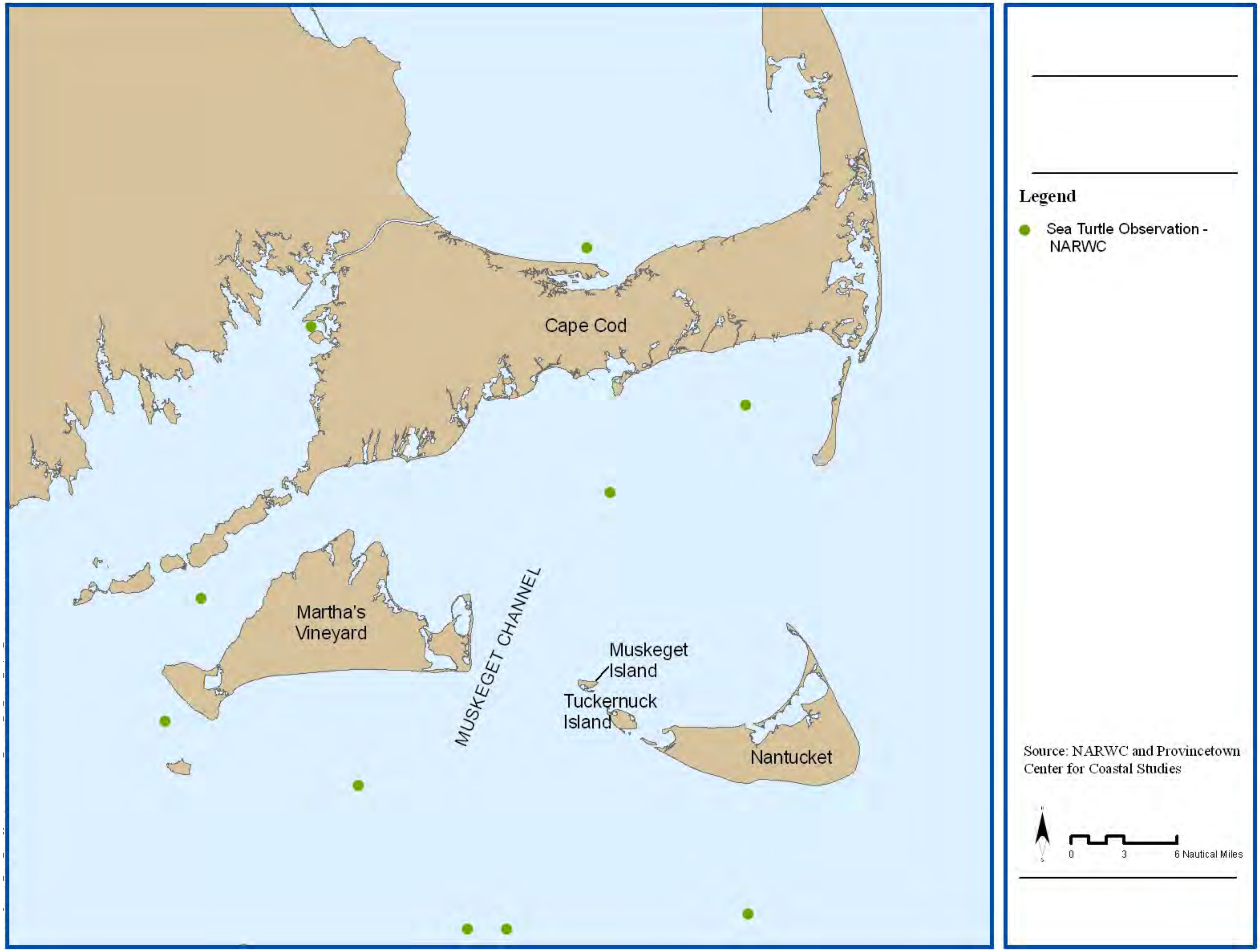


Figure 2.15: Sea turtle sightings documented in the North Atlantic Right Whale Consortium sightings database for Nantucket Sound. Sightings data only indicate presence of animals, rather than patterns of distribution, as survey effort is not plotted.



2.4 Basking Shark

2.4.1 Introduction

The common name of the basking shark, *Cetorhinus maximus*, refers to its appearance of "basking" while feeding at the surface. The basking shark is the second largest fish in the world (over 9 m total length), second in size only to the whale shark. Basking sharks are filter feeders, straining zooplankton from the water using gill rakers inside their gill slits, which extend almost completely around the head and are located behind their conical snout and large mouth (Martin and Harvey-Clark, 2004). The aforementioned features render the basking shark easily identifiable.

The basking shark is distributed circumglobally, occurring in the North and South Atlantic Oceans, Mediterranean Sea, North and South Pacific Oceans, Sea of Japan, off southern Australia and around New Zealand (Compagno, 2001). Canadian records from both Atlantic and Pacific waters indicate *C. maximus* occurs in most coastal temperate waters where temperatures exceed 6-7 °C (Campana *et al.,* 2008), and recent tagging efforts indicate that migrations to tropical waters also occur (Skomal *et al.,* 2009).

The life history of basking sharks is poorly understood; however, long lifespan, slow growth and low fecundity likely render this species vulnerable to reductions in population (Martin and Harvey-Clark, 2004). Despite advances in understanding of the species' distributional ecology, data are lacking on population structure and size with which to assess conservation status (Sims *et al.,* 2008). Relative abundance indices in U.S. waters have exhibited little variation since 1979 (Campana *et al.,* 2008). Basking sharks are listed under Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) and listed as "vulnerable" globally and "endangered" in the Northeastern Atlantic and in the North Pacific by the International Union for Conservation of Nature (IUCN; Sims, 2008). In U.S. waters, federal regulations prohibit fishermen from possessing basking sharks.

2.4.2 Distribution & Abundance in the Nantucket Sound – Muskeget Channel Area

Sighting frequency of basking sharks off the northeast U.S. is highest from May-August (Kenney *et al.,* 1985; Campana *et al.,* 2008). Sightings in the vicinity of the study area in the North Atlantic Right Whale Consortium sightings database (n = 104) reflected a similar temporal distribution and generally occurred south of Martha's Vineyard and Nantucket Islands (Figure 2.16; Right Whale Consortium, 2010). However, this does not necessarily reflect spatial distribution patterns, as systematic survey effort in the study area was distributed in a similar manner (See Data Summary). Two additional sightings recorded during 2003-2004 aerial seabird surveys conducted by the Massachusetts Audubon Society in Nantucket Sound (See Data Summary) are included in Figure 2.16. Skomal (2007) summarized opportunistic examinations of stranded basking sharks in Massachusetts coastal waters, noting that six of seven fish examined (one of which was stranded on Martha's Vineyard) were immature, suggesting that study area waters may serve as secondary nursery habitat for the species.

2.5 Ocean Sunfish

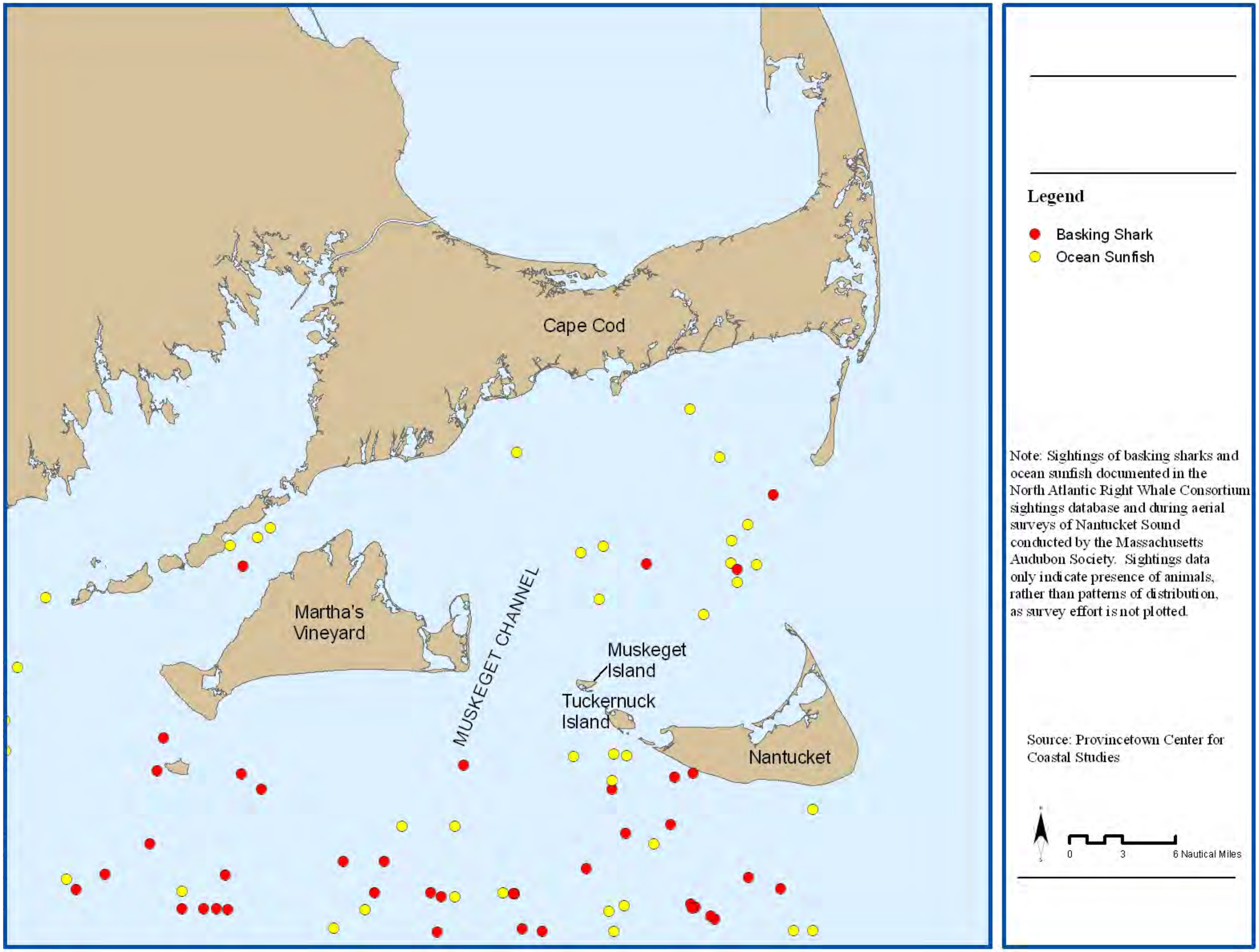
2.5.1 Introduction

The ocean sunfish (*Mola mola*) is the largest bony fish in mass – a 2.7 m record-length specimen weighed 2.3 mt (Pope *et al.,* 2010). There are virtually no fisheries for *M. mola*, although they are frequently bycaught in other fisheries (e.g. Silvani *et al.,* 1999), and much of the species' biology and ecology remains unknown. Distribution is worldwide in temperate and tropical seas, but an accurate accounting of range or abundance is nonexistent due to the lack of fisheries and the associated data collection. No quantitative information exists on diet or habitat requirements, and while many observations indicate near-surface feeding on gelatinous zooplankton, stomach contents and recent telemetry studies indicate than ocean sunfish may be omnivorous and feed throughout the water column (Pope *et al.,* 2010). Bigelow and Schroeder (1953) reported that stomachs of all specimens brought to the Bureau of Fisheries in Woods Hole appeared to contain remnants of jellies, ctenophores,or salps. Due to the lack of data, conservation status of this species is difficult to assess.

2.5.2 Distribution & Abundance in the Nantucket Sound – Muskeget Channel Area

Kenney (1995) estimated ocean sunfish abundance from aerial surveys in the shelf waters from Cape Hatteras north to the Gulf of Maine, noting that abundance in Southern New England waters peaked in summer and declined to zero in winter and distribution patterns were similar to those of leatherback turtles. Sightings in the vicinity of the study area in the North Atlantic Right Whale Consortium sightings database (n = 37) reflected a similar temporal distribution and generally occurred south of Martha's Vineyard and Nantucket Islands (Figure 2.16; Right Whale Consortium, 2010). However, this does not necessarily reflect spatial distribution patterns, as systematic survey effort in the study area was distributed in a similar manner (See Data Summary). Sadoti *et al.* (2005) noted 17 sightings of ocean sunfish in August and September of 2002-2004 during aerial surveys for seabirds in Nantucket Sound, but did not plot sighting locations. Sighting locations from 2003-2004 surveys were obtained from the Massachusetts Audubon Society (See Data Summary) and are included in Figure 2.16.

Figure 2.16: Basking sharks and ocean sunfish sightings data for Southern Massachusetts.



3

Review of the Distribution of Fishery Resources and Habitats and Commercial and Recreational Fishing Activity in the Nantucket Sound – Muskeget Channel Area

Photo: E. Bradfield

3.1 Introduction

There is little readily available data with which to evaluate the specific importance of the Muskeget Channel study area to commercial and recreational fisheries (DT&A, 2006). During the Massachusetts Ocean Management Plan development process, the Muskeget Channel area was designated as an area of "medium importance" to fisheries resources based on analysis of 30 years of trawl survey data (Commonwealth of Massachusetts, 2009). Mapping of commercial fisheries activity indicated that "low" to "medium" levels of commercial fishing activity occur in Muskeget Channel. The Channel and surrounding waters are considered to be of "high importance" to recreational fisheries based primarily on landings data and interview-based surveys (Commonwealth of Massachusetts, 2009).

While the above analyses used a spatially-explicit approach to identify areas of importance to fisheries[1], the trawl surveys were designed to measure relative abundance of species rather than fine-scale distribution patterns, and effort is scarce in the Muskeget Channel area (King *et al.,* 2010). Further, many species, including pelagics, shellfish and forage fish, are not vulnerable to capture during the surveys, which occur only in spring and fall (Commonwealth of Massachusetts, 2009; King *et al.,* 2010). The maps of fishing effort were also interpolated from fisheries-dependent data collected at coarser spatial scales. While the Ocean Management Plan process incorporated a detailed spatial analysis, it is impossible to make species- and fishery-specific interpretations of the maps at the scale of an area the size of Muskeget Channel. Therefore, the following section will focus on fisheries activity and resources within the larger Nantucket Sound area, with specific reference to Muskeget Channel when possible.

3.1.1 Commercial Fisheries

Recent attempts to characterize the fisheries of Nantucket Sound have been hampered by absent or overlapping data on effort and landings (MMS, 2009). Fisheries-dependent data are generally binned into either state or federal statistical reporting areas. The Muskeget Channel study area falls within Massachusetts Division of Marine Fisheries (DMF) Area 10 (Nantucket Sound) and Area 12 which includes state waters (3 nm from shore) to the south of Martha's Vineyard and Nantucket (http://www.mass.gov/dfwele/dmf/commercialfishing/inshore_areas.htm). Most of the project falls within federal NOAA Fisheries statistical Area 538, which includes Nantucket and Vineyard Sounds as well as Buzzards Bay, although the much larger Area 537 borders the study area to the south. In studies of the fisheries of Nantucket Sound for the Cape Wind Energy Project, a subarea of Area 538 that roughly overlaps DMF Area 10, called Area 075, was used to define federal landings within the Sound (ESS, 2006a). The coarse spatial scale of the publicly available data from Nantucket Sound as assembled by ESS (2006a) and

1 Detailed descriptions of the analytical approach used in the fisheries component of the Massachusetts Ocean Management Plan can be found in the report of the Fisheries Workgroup: http://www.env.state.ma.us/eea/mop/tech_reports/112608_ocean_mgt_fish_wkgp.pdf

reviewed by MMS (2009) renders it difficult to make conclusions about specific gears or species within the Muskeget Channel study area. The following is a summary of available information on commercial fisheries within Nantucket Sound, based largely on the review conducted by MMS (2009), except where otherwise cited.

Commercial fisheries in Nantucket Sound are diverse, targeting many species of fish and invertebrates, including squid, conch, quahogs, fluke, sea bass, bluefish, striped bass, Atlantic mackerel and lobster. Fishing gears employed in the Sound include otter trawls, dredges, weirs, seines, traps, pots and hand lines. The dominant gear type in the Sound (Area 538/075) reported via federal Vessel Trip Reports (VTRs) is the otter trawl. Interpretation of landings data even at this large scale must be done with caution due to the overlap between state- and federally-reported fisheries, as well as gaps in federally-reported landings due to vessels with state-only permits (e.g. Massachusetts Coastal Access Permits for squid and fluke; Wiersma, 2008). The top ten federally-reported species of finfish (including squid; annual average catch in weight) from 1998-2007, in decreasing order of percent total catch, were squid, fluke, Atlantic mackerel, black sea bass, scup, bluefish, menhaden, butterfish, winter flounder and king whiting, together comprising 99% of all landings in Nantucket Sound. Squid accounted for 50% of total annual average catch, while the second largest component (fluke) was 14% of the total. Within Massachusetts waters, virtually all squid landings occur within Nantucket Sound and neighboring Vineyard Sound in spring and summer (McKiernan and Pierce, 1995). Federally-reported landings of shellfish are dominated by conch (88%) and include ocean quahogs, surf clams, hard clams and horseshoe crabs, comprising 99% of 1998-2007 VTR catches (MMS, 2009).

State-reported landings in Nantucket Sound (DMF Area 10) primarily include squid and finfish catches from hook and line, fish weirs, gillnets lobster and fish pots, as well shellfish landings collected by municipalities. Weir fishing effort occurs primarily in the Northeastern Sound. The top ten state-reported species of finfish (including squid; annual average catch in weight) from 1998-2007, in decreasing order of percent total catch, were black sea bass, Atlantic mackerel, squid, fluke, scup, striped bass, menhaden, bluefish, butterfish and bonito, together comprising 99% of all landings in Nantucket Sound. State-reported landings of shellfish are dominated by conch (72%) and include hard clams and lobsters, comprising 99% of 1998-2007 DMF catches (MMS, 2009).

Distribution of fisheries effort in state waters around the boundaries of the Sound is mapped in the Massachusetts Ocean Management Plan, but is not specific to species or gear type. Federal VTR data were mapped by MMS (2009), indicating that squid catches were concentrated in the central portion of the Sound north of Muskeget Channel, fluke catches were primarily located on the eastern side of the Sound with a small concentration northwest of Muskeget Channel and shellfish landings were concentrated on the eastern side of the Sound. Cape Poge Bay, which lies immediately west of Muskeget Channel, contains eelgrass habitat which supports a variable but productive bay scallop fishery, which contributed 57% of Martha's Vineyard's total 1991-2004 bay scallop landings (MacKenzie, 2008).

Surveys of commercial fishermen fishing in the Sound indicated that mobile gear fishing effort followed the above patterns, with minimal effort in the Muskeget Channel area (off Cape Poge). Summer hook-and-line fishing for bluefish and striped bass, as well as fall trawling for fluke and hook-and-line fishing for black sea bass and tautog, were among the fishing activities undertaken at a "medium" activity level (15-30% of active vessels fishing); no activity in the Channel was listed as greater than 30% of active fishing effort (ESS, 2006b). Hall-Arber *et al*. (2004) interviewed commercial fishermen who fished in the Sound and noted that fishing primarily occurs during spring, summer and fall, with little winter effort. Participating fishermen mapped their knowledge of fishing effort, indicating that mobile gear effort was concentrated in the central and eastern portions of the Sound, while "other" gears were used in the remainder of the Sound, including the Muskeget Channel area. No mobile gear (e.g. otter trawl) fishing effort was indicated in the Channel. Limited sample sizes and a focus on the area of the proposed Cape Wind energy project indicate that the results of the Hall-Arber *et al.* (2004) and ESS (2006b) studies should be interpreted cautiously.

3.1.2 Recreational Fisheries

Attempts to assess the extent of recreational fisheries in Nantucket Sound have encountered similar challenges to studies of commercial fishing due to lack of data or absence of spatially-explicit information. In order to examine the potential effects of the Cape Wind project on recreational fisheries, MMS (2009) summarized NOAA Fisheries Marine Recreational Fisheries Statistics Survey (MRFSS) 2005-2007 data and noted that the top eight species, representing 99% of the catch by weight, were bluefish, scup, striped bass, fluke, black sea bass, little tunny, bonito and tautog. Highest recreational fishing pressure occurs during the summer months, during the seasonal peak of tourism. Shore-based fishing accounted for 73% of average annual effort, while private/rental vessels represented 25% and party/charter vessels the remainder (MMS, 2009). Data collected in 1998-2007 from federally-permitted charter vessels subject to VTR reporting requirements indicated that the top species landed were scup (74%), squid, black sea bass, fluke, bluefish, tautog, striped bass and sea robin, together comprising nearly 100% of the total catch. Surveys targeting recreational fishing charter/party vessel operators indicated that preferred target species included striped bass, scup and tunas, with other target species including bluefish, bonito, black sea bass and fluke (Battelle, 2003).

Federally-reporting (VTR) charter vessel landings primarily were recorded in the northern portion of the Sound (MMS, 2009). Survey respondents noted that during half-day charters, Muskeget Channel was among the top 40% of sites fished and the Tuckernuck area to the east was targeted by 24% of trips, while 9% of full-day trips targeted shoals around Tuckernuck Island (Battelle, 2003). One charter fisherman from a small sample surveyed by ESS (2006b) noted that he fished 50% of the time in Nantucket Sound, off Falmouth and off Cape Poge (western side of Muskeget Channel). Surf casting for bluefish and striped bass has been reported to be popular off Wasque Point, on the western side of Muskeget Channel (DT&A, 2006). As is the case with the surveys of commercial fishermen, the above studies by Battelle (2003) and ESS (2006b) need to be interpreted with caution due to small sample sizes and their focus on the Cape Wind site.

3.1.3 Fisheries Resources

The Muskeget Channel study area straddles the boundary between two ten-minute squares within which Essential Fish Habitats (EFH) are designated under the 1996 amendments to the Magnuson-Stevens Fishery Conservation and Management Act (MSFCMA). Most of the study area falls within the 10-minute square between 41° 20' – 41° 30' N and 70° 20' – 70° 30' W. Discussion will be limited to those species with EFH designations found within the above boundaries because the adjacent square to the south encompasses only a small portion of the study area and a larger area of other habitat types south of the Sound. This discussion is intended to highlight species of potential importance should a formal EFH assessment be conducted and is not an exhaustive summary of species for which EFH assessment may be necessary. A formal EFH consultation process coupled with an understanding of the potential project impacts will better inform this discussion and the list of species for which EFH may need to be considered. The table below includes 18 species (16 fish, 2 invertebrates) for which EFH has been designated between 41° 20' – 41° 30' N and 70° 20' – 70° 30' W (Table 2), and does not include additional EFH-designated species in the ten-minute square to the south.

3.2 Species Descriptions

The following section is not intended to be an exhaustive list of species found in the study area or potentially affected by the project. In the absence of spatially-explicit data on species distribution, knowledge of the potential extent of project impacts and a formal EFH consultation, the following accounts are intended to summarize information on fish and invertebrate species known to be of importance to commercial and recreational fisheries in the vicinity of the study area. They are loosely ordered according to fishery and taxonomy. Some species are listed despite a lack of EFH designation in the quadrant that encompasses the majority of the study area due to their importance to local fisheries. Conversely, some EFH-designated species are not listed, as they are not principal fishery resources in the area. Discussion of ranges is generally confined to Western North Atlantic populations.

Table 2:

Life Stages of 18 Species (16 fish, 2 invertebrates) for which EFH has been Designated Between 41° 20' – 41° 30' N and 70° 20' – 70° 30' W

Species	Eggs	Larvae	Juveniles	Adults
Atlantic cod *(Gadus morhua)*				X
Winter flounder *(Pleuronectes americanus)*	X	X	X	X
Yellowtail flounder *(Pleuronectes ferruginea)*			X	
Long finned squid *(Loligo pealei)*	n/a	n/a	X	X
Atlantic butterfish *(Peprilus triacanthus)*	X	X	X	X
Atlantic mackerel *(Scomber scombrus)*	X	X	X	X
Summer flounder *(Paralicthys dentatus)*	X	X	X	X
Scup *(Stenotomus chrysops)*	n/a	n/a	X	X
Black sea bass *(Centropristus striata)*	n/a	X	X	X
Surf clam *(Spisula solidissima)*	n/a	n/a	X	X
King mackerel *(Scomberomorus cavalla)*	X	X	X	X
Spanish mackerel *(Scomberomorus maculatus)*	X	X	X	X
Cobia *(Rachycentron canadum)*	X	X	X	X
Blue shark (*Prionace glauca)*				X
Bluefin tuna *(Thunnus thynnus)*			X	X
Shortfin mako shark *(Isurus oxyrhyncus)*			X	
Little skate *(Leucoraja erinacea)*			X	X
Winter skate *(Leucoraja ocellata)*			X	X

Sources: http://www.nero.noaa.gov/hcd/STATES4/cape_cod/41207020.html and http://www.nero.noaa.gov/hcd/skateefhmaps.htm

The **longfin inshore squid** (*Loligo pealeii*) is distributed in continental shelf and slope waters of the Northwestern Atlantic Ocean from Newfoundland south to the Gulf of Venezuela (Roper *et al.,* 1984). The species is considered a single unit stock within its range of commercial exploitation from Cape Hatteras north to Georges Bank (Hendrickson and Jacobson, 2006). Longfin inshore squid support a highly variable, "boom-or-bust" fishery, particularly in the inshore waters of Nantucket Sound (Brodziak and Rosenberg, 1993). Virtually all Massachusetts squid landings (including a small proportion of shortfin squid, *Illex illecebrosus*) occur within Nantucket and Vineyard Sounds in spring and summer (McKiernan and Pierce, 1995). Longfin squid were the most frequently captured species during DMF trawl surveys in Nantucket Sound (1978-2007), occurring in 90.5% and 99.9% of spring and fall tows, respectively (King *et al.,* 2010). In Nantucket Sound, at least two cohorts of squid arrive in spring and summer: larger animals in late April and early May that spawn in late spring, followed by smaller individuals that spawn in early fall (Brodziak and Rosenberg, 1993; McKiernan and Pierce, 1995). Spawning in Nantucket Sound occurs primarily from May-July (Hatfield and Cadrin, 2002), during which females deposit clusters of egg capsules on the bottom (Arnold and Williams-Arnold, 1977). Arrival and distribution in the Sound are likely correlated with environmental variables, including wind forcing and water temperature, but confirmatory studies have yet to be completed. The species' stock status is undetermined; overfishing is not considered to be occurring (MAFMC, 2010).

The **fluke** (*Paralicthys dentatus*), also known as summer flounder, is a demersal flatfish that is distributed from the

Southern Gulf of Maine to South Carolina. Commercial and recreational fisheries occur from Cape Cod south to Cape Hatteras. Fluke are managed as a single unit stock from North Carolina to Maine (Terceiro, 2006a). Fluke are concentrated in shallow bays and estuaries from late spring through early fall, when an offshore migration to the outer continental shelf occurs. Spawning occurs during fall and early winter, followed by inshore larval transport via prevailing currents. Post-larval and juvenile development primarily occurs within bays and estuaries. Fluke arrive inshore in Massachusetts waters in early May and occur in shallow waters south of Cape Cod and Buzzards Bay, Vineyard Sound, Nantucket Sound and coastal waters around Martha's Vineyard. Offshore migration of fluke from Massachusetts waters begins in late September and October (Packer *et al., 1999). Fluke* occurred in 55.4% of spring and 64.6% of fall tows during DMF trawl surveys in Nantucket Sound from 1978-2007 (King *et al.,* 2010). Fluke are not considered to be overfished, and overfishing is not occurring (MAFMC, 2010).

The **Atlantic mackerel** (*Scomber scombrus*) is a pelagic, schooling species distributed in the Northwestern Atlantic from Labrador south to North Carolina. There are two major components of the population: a southern group that spawns primarily in the Mid-Atlantic Bight during April and May and a northern group that spawns in the Gulf of St. Lawrence in June and July. Both groups overlap slightly in winter between Sable Island (off Nova Scotia) and Cape Hatteras, with extensive northerly (spring) and southerly (autumn) migrations to and from spawning and summering areas. Both groups are managed as a unit stock. Atlantic mackerel are targeted by seasonal commercial and recreational fisheries throughout most of their range. Commercial landings are caught primarily between January and May in southern New England and Mid-Atlantic coastal waters and between May and December in the Gulf of Maine, while recreational landings are caught mainly between April and October (Studholme *et al.,* 1999; Overholtz, 2006a). Based on 1978-1996 DMF bottom trawl data summarized by Studholme et al. (1999), juveniles were most common in Vineyard Sound in spring, and adults were most common in Nantucket Sound in spring. A more recent summary of the DMF trawl survey data indicates a relatively low occurrence in Nantucket Sound when averaged across 1978-2007 (King *et al.,* 2010), likely reflecting a decline in landings in recent years reported by many commercial fishermen working in the Sound. The species' stock status is undetermined and overfishing is not considered to be occurring (MAFMC, 2010).

Black sea bass (*Centropristis striata*) are distributed from Nova Scotia to Florida, with Cape Hatteras serving as a geographic boundary between northern and southern stocks. Structures such as reefs, wrecks or oyster beds form preferred habitats. Spawning in the northern stock primarily occurs from April to June following migration into coastal habitats. Larvae and juveniles develop and grow in inshore habitats. Sea bass remain in coastal habitats until water temperatures decrease in fall into early winter, and then migrate to deeper offshore water along the edge of the continental shelf. In spring, most fish return to the area in which they were distributed the previous fall (Shepherd, 2006a; Drohan *et al.,* 2007). Black sea bass occurred in 30% of spring and 81% of fall tows during DMF trawl surveys in Nantucket Sound from 1978-2007 (King *et al.,* 2010). Black sea bass are not considered to be overfished, and overfishing is not occurring (MAFMC, 2010).

The **scup** (*Stenotomus chrysops*) is a schooling species found primarily from Massachusetts south to Cape Hatteras. Spring and fall scup migrations are associated with seasonal changes in water temperature. When seawater temperature rises in spring, scup move north and inshore to spawn. Large adult fish arrive off southern New England by early May, followed by schools of sub-adults. Larger scup are found during summer near bay mouths and in the ocean within the 20-fathom contour while smaller fish are found in shallower habitats (Steimle *et al.,* 1999). Scup are managed as a single stock, despite limited evidence from tagging studies for two stocks: one in Southern New England waters and one ranging south from New Jersey (Terceiro, 2006b). Scup occurred in 47.9% of spring and 99.7% of fall tows during DMF trawl surveys in Nantucket Sound from 1978-2007 (King *et al.,* 2010). Scup are not considered to be overfished, and overfishing is not occurring (MAFMC, 2010).

Butterfish (*Peprilus triacanthus*) range from Newfoundland and the Gulf of St. Lawrence south to the Atlantic and Gulf Coasts of Florida, with peak abundance from the Gulf of Maine to Cape Hatteras, where the species is considered a single unit stock (Overholtz, 2006b). The butterfish is a fast-growing, short-lived, pelagic species that forms loose schools, often near the surface. Butterfish winter in Mid-Atlantic Bight outer shelf waters and migrate inshore in the spring into Southern New England and Gulf of Maine waters. In summer, butterfish occur over the entire Mid-Atlantic shelf in nearshore waters, bays and estuaries. In late fall, butterfish migrate southward and offshore as seawater temperatures decrease (Cross *et al.,* 2009). Butterfish occurred in 24.7% of spring and 91.8% of fall tows during DMF trawl surveys in Nantucket Sound from 1978-2007 (King *et al.,* 2010). The stock status of butterfish is unknown, and overfishing is not considered to be occurring (MAFMC, 2010).

Bluefish (*Pomatomus saltatrix*) are distributed in the Western Atlantic from Nova Scotia and Bermuda south to Argentina, with greatest occurrence between Florida and the Gulf of Maine. Bluefish are found in schools of similarly sized fish and undertake seasonal migrations, moving into the Mid-Atlantic Bight during spring and south or farther offshore during fall. Within Mid-Atlantic waters, bluefish occur in large bays and estuaries as well as across the extent of the continental shelf (Shepherd and Packer, 2006). Bluefish are considered a single unit stock (Shepherd, 2006b). Bluefish were caught far more frequently in fall (22.9%) than spring (0.8%) during DMF trawl survey tows in Nantucket Sound (King *et al.,* 2010). Bluefish are not considered to be overfished, and overfishing is not occurring (MAFMC, 2010).

The **striped bass** (*Morone saxatilis*) spends the majority of its adult life in coastal estuaries or the ocean, undertaking north (summer) and south (winter) seasonal migrations, and ascending rivers to spawn in the spring. After larvae arrive in the riverine and estuarine nursery areas, they mature into juveniles, remaining in coastal sounds and estuaries for two to four years before migration to the North Atlantic. Important wintering grounds are located from offshore New Jersey as far south as Cape Hatteras. With warming water temperatures in the spring, mature adult fish migrate to the riverine spawning areas to complete their life cycle. The Chesapeake Bay spawning area produces the majority of coastal migratory striped bass (ASMFC, 2003). Striped bass are not considered to be overfished, and overfishing is not occurring (ASMFC, 2009).

The **Atlantic menhaden** (*Brevoortia tyrannus*) is a euryhaline species that inhabits nearshore and inland tidal waters and is found in large, dense schools from Florida to Nova Scotia. Spawning primarily occurs at sea with some activity in bays and sounds in the northern portion of its range. Eggs hatch at sea and larvae are transported inshore by ocean currents to estuaries, where juvenile development occurs. Distribution of adults occurs by size during the summer, with older, larger individuals found farther north. In fall, Atlantic menhaden migrate south and disperse from nearshore surface waters off North Carolina by late January or early February. Schools of adult fish reassemble in late March or early April and migrate north, redistributed from Florida to Maine by June. Atlantic menhaden are an important forage species for numerous commercially and recreationally sought finfish as well as other piscivores (Ahrenholz, 1991). The species is managed as a single unit stock and is not considered to be overfished, nor is overfishing currently occurring (ASFMC, 2010).

Several other species of interest also occur in the region. The Sound's conch fishery is supported by the channeled whelk (*Busycotypus canaliculatus*) and knobbed whelk (*Busycon carica*), large gastropods that feed on bivalve molluscs and other benthic prey (Gosner, 1978). Little information is available on the local distributional ecology of either species, although an expanding fishery is prompting the development of research projects. King *et al.* (2010) reported the occurrence of channeled whelks (54.7% spring, 48.7% fall) and knobbed whelks (26.4% spring, 53.8% fall) in survey trawls in Nantucket Sound. Atlantic cod (*Gadus morhua*) do not support fisheries in Nantucket Sound, however the species' importance to commercial fisheries in other areas and the high occurrence of juveniles in the Sound (53.4% of spring survey tows; King *et al.,* 2010) warrant consideration. Similarly, winter flounder (*Pseudopleuronectes americanus*) are not heavily fished in the Sound, but occur at high frequency in spring surveys (87.9% of tows; King *et al.,* 2010), indicating that the Sound may be important winter habitat for the species. Little skates (*Leucoraja erinacea*) and winter skates (*Leucoraja ocellata*) also occur at high frequency during surveys in the Sound (King *et al.,* 2010). Bonito (*Sarda sarda*) and tautog (*Tautoga onitis*) are other species that historically have supported commercial and recreational fisheries in the Sound (Hall-Arber *et al.,* 2004) but are not currently reported among the more heavily fished species. Skomal (2007) noted that study area waters (just off Cape Poge) may serve as nursery habitat for several recreationally and commercially important shark species, including smooth dogfish (*Mustelis canis*), sandbar sharks (*Carcharinus plumbeus*) and sand tiger sharks (*Carcharias taurus*). Forty sightings of blue sharks (*Prionace glauca*) were documented in the region between 1979 and 1992 (Right Whale Consortium, 2010).

4 Data Gaps & Concerns

Photo: Gray seal, Provincetown Center for Coastal Studies image taken under NOAA permit 775-1875

4.1 Introduction

Sections 2 and 3 identified the lack of baseline data on marine megavertebrates, fishery resources and fishing activity for the Nantucket Sound – Muskeget Channel area. This lack of data presents significant challenges for an assessment of the potential for environmental impacts of the proposed tidal turbine project in Muskeget Channel and of future marine renewable energy projects in this area.

The following section lists the gaps uncovered during data mining for this report. It also highlights the species likely to be encountered in the Muskeget region for which there are conservation concerns and highlights potential species-specific issues regarding tidal energy technologies. Section 5 provides a more detailed discussion of the known and suggested effects of MREIs on marine megavertebrates. Section 6 provides a more detailed discussion of monitoring techniques and recommends a monitoring program specific to the tidal energy project proposed by the Town of Edgartown for Muskeget Channel.

4.2 Cetaceans

Data Gaps

The lack of systematic survey effort on all species sighted in the study area precludes an accurate assessment of the abundance and distribution of cetaceans in the region.

Concerns

Many species of cetaceans have experienced severe population decline in recent decades, and still face numerous threats such as bycatch (e.g. Read *et al.,* 2006; Leeney *et al.,* 2008), ship strike (e.g. Cole *et al.,* 2005; Panigada *et al.,* 2006) and habitat degradation (e.g. Bearzi *et al.,* 2008a, b). One species which deserves special consideration with respect to the proposed project in Muskeget Channel is the North Atlantic right whale, *Eubalaena glacialis*. As detailed in the cetacean discussion in Section 2.1, this species numbers less than 450 individuals and is listed as endangered under the Endangered Species Act (ESA) and depleted under the Marine Mammal Protection Act (MMPA). It is also classified as "endangered" by the IUCN Red List. This species' range is restricted to the east coast of North America, with concentrations occurring fairly predictably in several key habitat areas, one of which

(Cape Cod Bay) borders on the planned construction area for the Muskeget Channel project.

The main threats to right whales are ship strike and entanglement in fishing gear (e.g. Johnson *et al.,* 2005; Elvin & Taggart 2008). The risk of ship strike to right whales will be increased with any increase in vessel traffic, such as that which may be associated with maintenance activity around any MREI. Likewise subsurface lines, cables or other non-solid structures in the water column may pose a collision or entanglement risk to subsurface feeding right whales, which often become entangled in fishing gear of various types (Johnson *et al.,* 2005; Cole *et al.,* 2006).

4.3 Pinnipeds

Data Gaps

All of the harbor seal population data available at the time of this report are out of date. Wade and Angliss (1997) recommended that population estimates older than eight years should not be used to calculate the potential biological removal (PBR – a management tool used to estimate how many individuals can be removed without impacting the population). The 2001 estimate by Gilbert *et al.* (2005) of the U.S. Atlantic harbor seal population is nine years old and outside of this recommended time limit. Barlas' 1999 study was the last comprehensive survey of Southern New England and even older than Gilbert's work. The studies since then (deHart, 2002; NMFS data on Nantucket jetties) are more recent but only cover a very small area of Nantucket Sound.

The stock structure of U.S. Atlantic harbor seals is poorly understood. Waring *et al.* (2006) provided evidence of individual seals moving from Nantucket Sound to Maine just before the breeding season. The relationship between U.S. and Canadian harbor seals is unknown.

The data on gray seal numbers and seasonal distribution outside of the pupping season (December-February) are also out of date; the most recent counts are from 1999. An accurate determination of the increase in the size of the gray seal population in Nantucket Sound is required. The pup counts from Muskeget Island are recent, continuous and can be used as a proxy for the increase in the Nantucket Sound gray seal population. However, it must be noted that these numbers are single day counts and not estimates of total pup production. The data available outside of the pupping season is over ten years old and out of date. In addition, local movements and habitat use by gray seals is poorly understood. Very little is known about local gray seal movements in Nantucket Sound (around Cape Cod and the Islands).

Concerns

Pinnipeds face a number of threats throughout their range. Significant levels of mortality due to anthropogenic activities or unusual events can place a population under pressure, making it more vulnerable to other, existing pressures.

An Unusual Mortality Event is defined as a stranding that is unexpected, involves a significant die-off of any marine mammal population and demands immediate response (http://www.nmfs.noaa.gov/pr/health/mmume/). There have been two Unusual Mortality Events in the Gulf of Maine (GoM; 2003 and 2006) of undetermined cause. Disease events occurring in the GoM are a threat to the Nantucket Sound harbor seals, as these animals are known to move throughout the GoM region. In their study of stranded marine mammals along Cape Cod and Southeastern Massachusetts, Bogomolni *et al.* (2010) found that 60% of harbor seals in their data set died of disease. Stranding data on harbor, gray, harp and hooded seals in Nantucket Sound and along outer Cape Cod are summarized in Table A1.

Pinnipeds often interact with fishing gear, and in some regions fisheries bycatch can have a negative impact on pinniped populations. In their study on mortality in stranded animals on Cape Cod and Southeastern Massachusetts, Bogomolni *et al.* (2010) found that 43% of the gray seals included in this study conclusively died of human-related causes. The most common human interaction affecting gray seals was entanglement in fishing gear. Waring *et al*. (2009) estimated the mean annual mortality in the commercial fisheries as 611 (cv = 0.15) for harbor seals and 331 (cv = 0.21) for gray seals. Additional lines or structures in the water column, particularly if such structures attract fish, may potentially pose an entanglement risk to seals.

4.4 Dermochelid & Chelonids

Data Gaps

There is little directed research on sea turtle seasonal distribution and abundance, foraging behavior and diet in the Nantucket Sound – Muskeget Channel area.

There is a lack of data on Northern diamond-backed terrapin foraging habitat in the waters of Nantucket Sound region.

Concerns

Several anthropogenic factors continue to threaten sea turtle populations. Entanglement in fishing gear (National Research Council, 1990; Lutcavage *et al.,* 1997; Dwyer *et al.,* 2003), incidental catches in fisheries (NRC, 1990; Lutcavage *et al.,* 1997; Witzell, 1999; James *et al.,* 2005), vessel strike (NRC, 1990; Lutcavage *et al.,* 1997), ingestion of marine debris (Carr, 1987; Lutz & Alfaro-Shulman, 1991; Lutcavage *et al.,* 1997), pollution (NRC, 1990; Lutcavage *et al.,* 1997), decline of habitat along the Western Atlantic coast (NRC, 1990; Lutcavage *et al.,* 1997; Witherington & Martin, 2000) and loss of nesting habitat (NRC, 1990; Lutcavage *et al.,* 1997) are some of the documented anthropogenic impacts that have led to declines in sea turtle populations. Leatherback turtles are listed as "critically endangered" by the IUCN Red List, with a "decreasing" population trend, and are federally- and state-listed as endangered species. In fact, several species of sea turtles face extinction from unsustainable bycatch in fisheries (NRC, 1990). Anthropogenic noise is thought to be detrimental to sea turtles (Samuel *et al.,* 2005), with likely effects on their behavior and ecology; however, no studies (to our knowledge) have been done specifically addressing the effects on this species group of noise sources generated by MREI construction and operation.

4.5 Basking Shark & Ocean Sunfish

Data Gaps

Systematic survey effort in the study area is lacking for both basking sharks and ocean sunfish.

Concerns

The basking shark (*Cetorhinus maximus*) is listed in Appendix 3 of the Convention on International Trade in Endangered Species (CITES). There remain numerous targeted fisheries for basking sharks, and the IUCN Red

List lists basking sharks as “vulnerable,” with a “decreasing” population trend. Satellite tagging studies in recent years suggest that basking sharks cross the oceans (Gore *et al.,* 2008) and even the equator (Skomal *et al.,* 2009), therefore the world’s population is likely smaller than previously thought. Any threat to basking sharks in the Nantucket Sound – Muskeget Channel area should thus be considered a threat to the Atlantic population.

The ocean sunfish is not Red Listed, and its biology and status remain poorly understood. However, any suggested amelioration of conditions for sunfish due to population reductions in predatory species in the world’s oceans (Myers & Worm, 2003) and increases in gelatinous prey (Mills, 2001) may be offset by anthropogenic mortality due to bycatch (Pope *et al.,* 2010; Cartamil & Lowe, 2004; Silvani *et al.,* 1999). Sunfish engage in much smaller “migration” patterns than basking sharks, which are usually linked to water temperature, and populations appear to be more regionally confined. Any impact on a local population, which is unlikely to be re-stocked by individuals from other regions, would be more detrimental to that particular population than for a wide-ranging species.

Basking sharks and sunfish are probably the most difficult megavertebrate species to assess. In New England waters, both species are most commonly sighted at the surface of the water during summer months, when the water temperatures are warm. But neither speices has to surface to breathe, and thus neither is always detectable using visual survey methods. Both basking sharks and ocean sunfish are known to dive to considerable depths and to move extensively through the water column (Pope *et al.,* 2010; Sims, 2008; Sims *et al.,* 2005; Skomal *et al.,* 2009), limiting the efficacy of visual survey for detection. The behavioral responses of basking sharks and sunfish to moving objects at depth are difficult to predict, and little information exists on the sensory capabilities of either species.

4.6 Fishery Resources and Fishing Activity

Data Gaps

There is a lack of spatially-specific data on the distribution of species and fishing effort within the Muskeget Channel area and the surrounding waters. Existing trawl survey data are insufficient to assess impacts due to low effort in the immediate vicinity of the project area. Landings data do not reflect effort controls or other management measures intended to aid in the rebuilding of fishery resources. Socio-economic data on local commercial and recreational fisheries are also lacking.

Concerns

Given the limited study on the topic, it is difficult to define the nature and spatio-temporal extent of potential project-specific impacts. Many of the region’s fishery resources and fishing communities have experienced recent declines and may be especially sensitive to ecological or socio-economic impacts (Buchsbaum *et al.,* 2005).

During an informal interview, one of the participants interviewed in the Hall-Arber *et al.* (2004) study noted that mobile gear fishing activity in the Muskeget Channel area was minimal and also commented that fishing vessels frequently transit the Channel and often deploy their stabilizers or “birds” during passage, which draw as much as 25 feet when fully extended in a rolling sea (Capt. Philip Michaud, *F/V Susan C III*, *pers. comm.*, 13 September 2010). The latter issue should be considered in any assessment of the area’s importance to fishing or navigation.

5 The Effects of Marine Renewable Energy Installations on Marine Megavertebrates: A Review

Photo: Leatherback sea turtle, Provincetown Center for Coastal Studies image taken during turtle disentanglement activities conducted under the U.S. Endangered Species Act and Final Rule 50 CFR Part 222

5.1 Introduction

At present, the main form of renewable energy generation in the marine environment is wind power. Wave and tidal energy conversion devices have been in development for many years, with numerous pilot projects for these devices now underway in areas such as Canada, the U.K. and the U.S. While some of the effects of introducing marine renewable energy installations (MREIs) to the marine environment may be the same regardless of the installation involved, other effects will be device-specific. Effects will vary with the stage (construction, operation and decommissioning) and scale of the project and will depend on location and the ecosystem in that area.

This section provides a summary of the existing literature on and knowledge of the effects of MREIs on marine megavertebrates. Here, the term "marine megavertebrates" encompasses all of the larger marine vertebrate species commonly encountered in coastal and offshore habitats; e.g. cetaceans (whales, dolphins and porpoises), pinnipeds (seals and sea lions), marine turtles and large fish including basking sharks and sunfish. Seabirds are often considered to be part of this group and thus are also discussed.

5.2 General Effects of Offshore Construction and Operation

The initial means by which any renewable energy device can affect the marine environment is during the period when it is being put in place. Not only does such work involve added vessel traffic in and around the planned location, but it may also involve blasting or drilling of the seafloor in order to attach or anchor the device to the seabed and further disturbances associated with the laying of power transmission cables. The construction phase of such projects will likely cause physical disturbance (e.g. presence of additional vessels and structures in the marine environment, disturbance of sediment) and acoustic disturbance (noise from engines of additional vessels in the area, drilling, pile driving and other construction methods). Many of these activities and their effects will be similar, regardless of the type of installation involved.

5.2.1 Underwater Noise

Underwater anthropogenic noise in the oceans is increasing due to activities such as commercial shipping, seismic exploration, marine construction and sonar technology (e.g. NRC 2003, McKenna and IFAW 2008). This is a growing cause for concern as our understanding develops about the negative effects, both immediate and long-term, of noise on marine life. Underwater noise is especially relevant for cetaceans, as they rely on sound as their main form of communication, often over distances of tens or hundreds of kilometers (e.g. Weilgart, 2007 and references therein). Depending on the context in which the often-complex vocalizations of cetaceans are produced, their masking by anthropogenic noise could affect foraging efficiency and the ability of conspecifics to maintain group coherence for functions such as reproduction (Croll *et al.,* 2001). Any such disturbances could lead to reduced fitness in a local population. For example, Payne and Webb (1971; referred to in Croll *et al.,* 2001) estimated that low frequency noise from shipping traffic may have reduced the area over which blue and fin whales can communicate by several orders of magnitude; from an estimated 2.1 x 10^6 km^2 (6 x 10^5 nm^2) in pre-shipping times to about 2.1 x 10^4 km^2 (6 x 10^3 nmi^2) in present-day conditions. Nowacek *et al.* (2007) provide a comprehensive review of the behavioral and acoustic responses of cetaceans to anthropogenic noise. In response to the recent need for systematic, objective and science-based interpretation of the available data on the effects of anthropogenic noise on protected species, Southall *et al.* (2007), utilizing the full body of scientific data on marine mammal hearing and the effects of noise on hearing and behavior, developed recommendations regarding noise exposure criteria for marine mammals.

5.2.1.1 Underwater Noise Caused by Vessel Traffic

Noise from vessel traffic is the dominant source of anthropogenic noise in the marine environment, and it is on the increase. Both the physical presence of vessels and the noise generated by their engines can influence the behavior of marine megavertebrates. Parks *et al.* (2007) reported both short and long-term changes in the calling behavior of North Atlantic right whales that were correlated with increased underwater noise levels. Similar findings have been made for blue whales (Di Iorio and Clark, 2010). Au and Green (2000) observed changes in the behavior of humpback whales in response to vessel noise from whale watch boats. Killer whales have been documented to produce longer calls (Foote *et al.,* 2001) and to increase the amplitude of their calls (Holt *et al.,* 2008) in response to increased vessel noise, and they also appear to change their movement patterns in the presence of "leapfrogging" whale watch vessels (Williams *et al.,* 2002). Lemon *et al.* (2006) measured changes in visible surface behavior, but not acoustic behavior, of Australian bottlenose dolphins in response to experimental powerboat approaches. In Florida, however, Buckstaff (2004) did detect effects of watercraft noise on the rate of whistle production in bottlenose dolphins. Lesage *et al.* (1999) observed reduced calling rates and a shift in the frequency band of calls from beluga whales (*Delphinapterus leucas*) when approached by vessels. Forced changes in behavioral state or a forced change in vocal output to compensate for more noise in an animal's environment may have energetic costs for affected individuals (Oberweger and Goller, 1991). Forced changes could also cause increased stress levels or degradation in communication among individuals, which has potential consequences at the population level of the species.

5.2.1.2 Underwater Noise Caused by Construction

Construction may involve activities such as drilling, controlled explosions, pile driving and the use of sonar to assess the seabed. Blasting of the seabed may be required as part of the construction phase, for example in order to lay cables, and has the potential to cause serious injury to the ears of cetaceans and pinnipeds (Ketten, 1995). There is a considerable gap in our knowledge of the effects of noise on marine mammals, which makes the management and mitigation of noise disturbances difficult. The effects of these noises will vary with environmental conditions such as water depth and propagation conditions as well as the depth of the animals receiving the sound. Controlled exposure experiments (CEEs) have been proposed (Gordon *et al.,* 2003; Tyack *et al.,* 2003) as a means of addressing questions relating to the effects of anthropogenic noise on marine mammals.

Several recent environmental assessments for offshore wind farms have identified pile driving as the activity that has the greatest potential to impact local cetacean populations. Several studies in European waters have used static acoustic monitoring in and around areas of construction of offshore wind farms to examine the effects of such activities on odontocetes. Edrén *et al.* (2010) reported lower numbers of gray and harbor seals hauling out in a nearby (4 km from construction) seal sanctuary during pile driving activities for a wind farm in Danish waters.

Brandt *et al.* (2009) reported that harbor porpoises in Danish waters appeared to leave the area of pile driving activity during and immediately after a pile driving event. The effect appeared to be lessened at greater distances from the activity. Tougaard *et al.* (2009a) documented longer "waiting times" between acoustic detections of porpoises in the period immediately after a pile driving event when compared with the wind farm construction period as a whole, and they were able to infer that the "zone of responsiveness" within which porpoises were reacting to the noise was greater than 21 km (the range to which this study was able to detect impacts). Carstensen *et al.* (2006) reported similar effects of pile driving on acoustically-detected habitat-use by harbor porpoises. It is unclear whether the "displacement" effect, when animals leave areas during periods of intense disturbance, has any longer-term costs for the animals in terms of fitness. However, since increasing the duration of construction phase will likely have increasing ecological impacts for certain marine vertebrate species, this should be taken into consideration at the planning stage of any such project. In addition to the direct effects of noise on marine mammals, there may be effects on fish populations that could have indirect effects for their predators.

There is a huge diversity in hearing capabilities among fish species (Thomsen *et al.,* 2006). Popper and Hastings (2009) detail the range of potential effects that sound could have on fish, from little or no effect through medium-level effects such as tissue damage and reduced fitness, behavioral changes and temporary hearing loss to immediate death. The effect will depend on both the species and the nature of the sound source – the levels of intensity and duration being key factors. As well as the physiological effects of noise, it may affect intra-specific communication, which could lead to stress, lowered fitness or changes in behavior. According to Thomsen *et al.* (2006, and references therein) anthropogenic underwater noise, including sources such as shipping, seismic airguns, pile driving and operational noise from wind turbines, exhibits major energy below 1,000 Hz and is, therefore, within the frequency range of hearing of most fishes.

In an assessment of the effects of offshore wind farm related noise on selected marine mammal and fish species, Thomsen *et al.* (2006) suggest that cod (*Gadus morhua*) and herring (*Clupea harengus*) may be able to perceive piling noise at distances of up to 80 km from the sound source and that behavioral effects at this scale may thus be possible. It has been argued that fish are killed if they are sufficiently close to pile driving activities, but data on the percentage of fish killed, differences in susceptibility of various species and variability of effect with distance are limited (Popper & Hastings, 2009 reviewed in Hastings and Popper, 2005). Furthermore, information on damage to fish outside the "kill zone," which may later die from injuries, does not exist. Additionally, there are numerous complexities within pile driving activities that might affect the effects on fish. Different types of piles (steel or concrete), for example, have different response characteristics and sound spectra. It is also not known whether there is a cumulative effect from being exposed to multiple pile strikes or whether each strike can be considered as an independent effect. Despite the lack of data in this area, it is evident that consideration must be given to the potential impacts of noise on fishes and any indirect effects this may have on their marine predators.

5.2.2 Increased Vessel Traffic

The response of small cetaceans to motorized vessels may be a reaction to noise, visual cues or a combination of both (e.g. Richardson *et al.,* 1995; Bejder *et al.,* 1999; Lesage *et al.,* 1999). In addition to affecting cetacean behavior, vessel traffic can be a cause of direct mortality. Collisions between vessels and cetaceans, termed "ship strikes," have been documented in many areas around the world and for numerous species of whale (e.g. Panigada *et al.,* 2006; Douglas *et al.,* 2008; Elvin and Taggart 2008) and dolphin (e.g. Bloom and Jager, 1994; Elwen and Leeney, 2010).

Issues which may arise from the physical presence of vessels include immediate effects such as animal-vessel collisions, medium-term effects such as evasive behavior by animals experiencing stress and longer-term effects such as decreased fitness or even habitual avoidance of areas where disturbance is common (e.g. Constantine, 2001; Hastie *et al.,* 2003; Lusseau, 2004; Lusseau, 2005; Bejder *et al.,* 2006; Lusseau *et al.,* 2009). Vessel traffic will invariably increase in offshore areas where MREIs are planned and located, not only during the construction phase but on an ongoing basis thereafter as maintenance and, eventually, decommissioning and removal of these structures will be required.

5.2.3 Electromagnetic Fields

Electromagnetic fields (EMF) can be emitted from undersea power transmission cables such as those associated with offshore wind farm developments and, possibly, tidal power generators. The magnetic component of EMF has the potential to affect magnetosensitive species such as bony fish, elasmobranchs, marine mammals and sea turtles (Wiltschko and Wiltschko, 2005; Luschi *et al.,* 2007; Gould, 2008). According to Gill *et al.* (2005), there are many fish species within the U.K. waters which are potentially sensitive to EMF given that these EMF components appear to be within their range of detection. The consequences for the fish, however, are unknown. It is also possible that animals using geomagnetic cues as navigation aids during migration, such as turtles and baleen whales, may be affected by magnetic fields, although the role of such cues for various species remains poorly understood (Lohmann *et al.,* 2008). Overall, the potential effects of EMF are difficult to predict and at present, and much more research is required (Gill, 2005; Gill *et al.,* 2005; Öhman *et al.,* 2007).

5.2.4 Secondary Impacts

5.2.4.1 Artificial Reefs & Additional In-water Structures

The placement of fixed structures on the seabed can have an "artificial reef" effect on the area. An artificial reef is defined as one or more objects of natural or human origin deployed purposefully on the sea floor, usually used to enhance recreational fishing and diving opportunities in the marine environment (Sutton and Bushnell, 2007). Adding vertical profile and surface area to the marine environment allows for growth of sedentary organisms, which in turn support other species. In a study of offshore wind farms in Danish waters, Maar et al. (2009) reported considerable aggregations of blue mussels on turbine pillars which created local hotspots of biological activity and changed ecosystem dynamics in the area. Petersen and Malm (2006) likewise suggest that the reef effect of offshore wind farms can have a significant effect on local species assemblages and biological structure, and that the importance of this impact may have been overlooked in many environmental impact assessments (EIAs) to date.[1]

An increase in the productivity of an area may actually attract marine vertebrates by providing a food resource. While fixed submerged structures are likely to pose little collision risk, cables and chain (which may be used for anchoring submerged structures such as tidal turbines), power lines and free-moving components on the surface or in the water column can present a hazard to some submarine species. Both large and small cetaceans as well as basking sharks and turtles are frequently entangled in fixed fishing gear (e.g. Julian and Beeson, 1998; Berrow, 2004; Garrison, 2005; Read et al., 2006). Seabed-to-surface lines, such as the end-lines of lobster fishing gear, are a well-known entanglement risk for humpback whales, right whales and numerous other species (e.g. Volgenau et al., 1995; Moore et al., 2004; Brillant and Trippel, 2010). Similar structures used in MREI developments may present the same risks to such species.

5.2.4.2 Fisheries Exclusion Zones

The introduction of artificial structures into the marine environment presents an immediate navigational hazard and the risk of fishing gear entanglement. Thus, even without enforced exclusion, the waters inside the boundary of most MREIs will become inaccessible to many fisheries. Extensive installations with numerous devices, in particular tidal stream and wave energy devices, will likely have enforced exclusion zones surrounding them to protect the installations as well as for navigational safety. These exclusion zones will become *de facto* no-take zones (NTZ).

A growing body of evidence suggests that NTZs and other forms of highly-protected Marine Protected Areas

1 The reef effect and the resulting accumulation of marine life in areas where artificial structures are added to the marine environment also highlights the potential for biofouling of tidal turbine and WEC structures themselves, which in turn would likely affect the operational efficiency of these devices.

(MPAs) are ecologically beneficial to both the protected area itself and to nearby areas. Benefits include enhanced stocks and individual fish or shellfish size (e.g. Cole *et al.,* 1990; Babcock *et al.,* 1999; Beukers-Stewart *et al.,* 2005; Blyth-Skyrme *et al.,* 2006) due to recovery from overfishing (Thurstan and Roberts, 2010) and protection of benthic environments from damaging fishing techniques such as bottom trawling (e.g. Thrush *et al.,* 1998; Blyth *et al.,* 2004). The changes in community structure that can result from the designation of protected areas can also show higher trophic complexity as well as increased primary and secondary productivity (Babcock *et al.,* 1999).

Although MREIs may act as NTZs, the habitats protected by these installations may not be priority habitats for conservation, fisheries management or restoration, and while protected from fishing, these habitats may be impacted by the MREI itself. MREI sites will be selected primarily based on the quality of the renewable energy resource, suitability of the seabed in respect of construction considerations, location relative to a mainland grid connection and the requirements of other marine stakeholders. Nevertheless, if sites are appropriately managed and designed, they might increase local biodiversity and benefit the wider marine environment, both by protecting living marine resources within their boundaries (Friedlander *et al.,* 2007) and by providing "recruitment subsidy" (Gerber *et al.,* 2003; Sale *et al.,* 2005) and "spillover effects" (DeMartini, 1993), whereby larvae, juveniles and adults produced in or utilizing the protected area will later move to adjacent areas, potentially bolstering fisheries surrounding the MPA.

However, Blyth-Skyrme (2010) highlighted the importance of recognizing the potential disruption to commercial fishing activities, through loss of fishing grounds or gear restrictions, posed by the establishment of MREIs such as offshore wind farms. As the number of these developments increase, support for commercial fishermen and dependent fishing communities may become necessary as well as recognition of the possible displacement of local fishing industry and an assessment of the socio-economic value of that loss.

5.2.4.3 Impacts Caused by Construction Activities

In addition to underwater noise, a number of secondary effects caused by construction activities should be considered. Such impacts are difficult to predict, but could include increased levels of suspended sediment in the water column, which might impair echolocation in odontocetes; avoidance of the area by fish and other prey species and/or perhaps attraction of marine predators into the area in response to large numbers of disoriented prey species.

5.2.5 Decommissioning

If located in Masschusetts state waters, MREI structures are licensed under the state's tidelands law and regula¬tions (301CMR 9.27). These regulations require the removal of structures "upon nullification, expiration or revocation" of the license. U.S. federal regulations also require that structures be removed and the seafloor cleared of all obstructions (30CFR Chapter II, Part 285.90). A set of impacts similar to those associated with construction are likely during this phase. For structures based on pilings, noise levels during decommissioning could be lower than during construction, as the pilings will likely be cut to below seabed level rather than being fully removed. There will also be no pile driving associated with this phase.

5.3 Device-specific Effects

5.3.1 Offshore Wind Turbines

Wind power has rapidly increased in capacity in recent years (Herbert *et al.,* 2007). High demand for space on land and aesthetic concerns about terrestrial wind farms (Taylor, 2004), combined with the better wind conditions in offshore areas, has resulted in an escalation in the development of offshore wind farms (Michel *et al.,* 2007).

Some offshore wind turbine sites have been in place for many years in areas such as the Baltic and North Seas. As a result, of all the categories of MREIs, the greatest knowledge base regarding effects on the marine environment comes from offshore wind turbine developments (e.g. Evans, 2008; COWRIE [2]). In a recent review of the environmental effects of coastal and offshore wind energy generation, Wilson *et al.* (2010) conclude that, while not environmentally benign, the environmental impacts of these developments are comparatively minor and can be mitigated through good siting practices. The authors also suggest that such MREIs provide the opportunity for environmental benefits through habitat creation and protected or inaccessible areas.

Because the operational portion of these devices is above the surface of the water, the only known potential means by which the operation of these devices might affect the marine environment are via the noise of the turbines, which can be transmitted through the base of the turbine to the underwater environment, and the "artificial reef" effects of the bases of the turbines (see Section 2.3). The known underwater noise levels emitted from operating offshore wind farms have been assessed as low relative to any standard (Madsen *et al.,* 2006), but they nonetheless constitute another source of anthropogenic noise in the marine environment. Tougaard *et al.* (2009b) investigated the operating sounds of three different types of wind turbine and estimated their likely effects on the behavior of harbor porpoises and harbor seals. The authors concluded that, due to the low noise levels, behavioral reactions of porpoises were unlikely except when immediately adjacent to the turbine foundations, while behavioral reactions from seals might occur up to distances of a few hundred meters. In all cases the noise was considered incapable of masking acoustic communication by seals and porpoises. Diederichs *et al.* (2008) documented no difference between harbor porpoise activity inside and outside two offshore wind farm areas in Danish waters and surmised that the presence of the wind turbines and their operational noise was unlikely to be affecting porpoise activity. Edrén *et al.* (2010) detected no long-term effects of wind farm construction and operation on the haul-out behavior of either gray or harbor seals at a site in Danish waters. Likewise, Madsen *et al.* (2006) suggest that noise impact on marine mammals is more severe during the construction of wind farms than during their operation.

Some fish species have been shown to react to the noise generated by wind farms (Andersson *et al.,* 2007), but Wahlberg and Westerberg (2005) suggest that while such noise levels may mask communication and orientation signals, they are unlikely to have destructive hearing effects. Nonetheless, any effect of wind farm noise on prey species may have a secondary effect on predators such as marine mammals.

A final, but significant, concern with regards to offshore wind turbines is the effects of these structures on birds and bats. Barrier effects due to flight avoidance, displacement resulting in habitat loss and fatalities resulting from collisions with turbine blades are the three primary threats to birds (Allison et al., 2008). Erickson et al. (2001) suggested that, relative to other human-made structures such as power lines, buildings and windows, the per-structure rate of avian collision with wind turbines is low. Despite a decade of study on turbine effects on birds, the impacts of terrestrial wind farms on birds at the population level are poorly understood. All that is clear is that the potential for bird impacts depends on the region and the local species complement.

Some studies suggest negative impacts (e.g. Barrios and Rodriguez, 2004; Garthe and Huppop, 2004), but Stewart *et al.* (2007), reviewing studies of this topic, suggested that the short time frame and poor design of many studies, which often lack good baseline data, make it difficult to truly assess of the effects of wind farms on avian fauna. There are even fewer data available for offshore wind farms. Blew *et al.* (2007) observed several seabird species using the area inside wind farms in Danish waters, and reported their increased risk of collision with wind turbines. Not all bird species will use areas occupied by wind farms, however, or fly at altitudes which place them at risk of collision with the turbine blades. Even if actual mortality levels due to collisions are low (Drewitt and Langston, 2006), reductions in local abundance may be observed due to avoidance of the area by certain species (Desholm and Kahlert, 2005).

Several studies have reported that migratory species appear to avoid wind farm areas, whereas resident species or those spending extended periods in the area did not (e.g. Krijgsveld *et al.,* 2005; Blew *et al.,* 2007). Non-lethal effects of wind turbines, including disturbance and reduced habitat quality, are at present poorly understood, but initial studies suggest that birds can habituate to these changes (Madsen and Boertmann, 2008). The risk to seabirds and other birds with migratory pathways through areas suitable for wind farms could thus be considerable.

2 Collaborative Offshore Wind Research Into The Environment. Numerous reports available online - http://www.offshorewindfarms.co.uk/Pages/COWRIE/

5.3.2 Tidal Turbines

More recently, tidally-dynamic areas have become the focus of the renewable energy sector, with projects shifting from barrage systems (e.g. Larsen, 1981; Rulifson and Dadswell, 1987; Fry, 2005) to capturing coastal tidal streams (e.g. Bahaj and Myers, 2003; Myers and Bahaj, 2005; Bryden and Couch, 2006; Sutherland et al., 2007; Block, 2008; Denny, 2009). The development of tidal turbines is increasingly seen as a more predictable alternative to wind generation. Tidal stream energy is derived from the kinetic energy of the moving flow of high velocity sea currents created by the movement of the tides; this is analogous to the way a wind turbine operates in air.

Fine-scale oceanographic features can be of great importance to pelagic predators (Wolanski and Hamner, 1988), providing enhanced concentrations of prey species which can be easily exploited by cetaceans, seabirds and large fishes. Many marine predators are known to forage in tidally driven oceanographic features, where they exploit predictable aggregations of prey. For example, bottlenose dolphins (Tursiops truncates; Mendes et al., 2002), harbor porpoises (Phocoena phocoena; Pierpoint, 2008), foraging seabirds (Hunt and Schneider, 1987) and basking sharks (Cetorhinus maximus; Sims and Quayle, 1998) have been associated with tidal intrusion fronts or tide "races." Harbor porpoises, fin whales (Balaenoptera physalus) and minke whales (Balaenoptera acutorostrata) congregate to feed within localized upwellings and fronts in the Bay of Fundy (Gaskin and Smith, 1979; Watts and Gaskin, 1985; Johnston et al., 2005a, b). Several species of tuna (e.g. albacore, Thunnus alalunga) have also been documented to forage at oceanic fronts (Fiedler and Bernard, 1987). Generally, the energy in marine tidal currents is diffuse, but it may be concentrated at a certain sites where the sea is channeled through restrictive topographies such as straits and between islands, making the use of marine currents attractive (Myers and Bahaj, 2005). The fact that marine megavertebrates and seabirds in coastal environments associate spatially with such areas of high tidal flow highlights a concern for sites where tidal turbine developments are planned.

Outside of the general construction, maintenance and decommissioning phases (which will be common to all devices) and the effects of the presence of large structures on the seafloor (covered in Section 5.2, above), there are several additional means by which tidal turbines may affect marine animals and their environment. The effects of the actual tidal turbines themselves, once they are running, on marine organisms and particularly on marine megavertebrates remain unknown at present.

The disruption and reduction of the net flow of water may affect the distribution of prey species (Parker, 1993; Fry, 2005), water turbidity or the ability of predators to hunt efficiently in these areas. We do not currently have a good understanding of the level to which these effects will occur and how that in turn will impact the predatory species which utilize these areas. Watts and Gaskin (1985) found a decline in the number of porpoises sighted on transects in the Bay of Fundy with increasing current speed, which they suggest is due to avoidance of shallow, turbulent areas which are energetically expensive for the animals to occupy. Gaskin and Watson (1985) also documented greater relative abundance of harbor porpoises in Fish Harbor, Canada during neap tides than during stronger spring tides.

These observations of the effects of natural fluctuations in tidal energy on cetacean habitat use suggest that tidal stream speed is a factor affecting porpoise habitat. This is an important consideration, since tide turbines extract a considerable amount of energy from the tidal flow (Sutherland *et al.,* 2007), thus net flow will be reduced at sites where tide turbines are in operation. A reduction in tidal flow may imply a reduction in feeding efficiency for small cetaceans or a loss of feeding habitat, forcing a local population to shift its range. This could have a significant effect on a given local population of marine megavertebrate in light of the importance of localized regions for feeding as "critical habitats."

Wilson *et al.* (2007) point out that rotating underwater turbines (models with open blades) present the most likely circumstance for collisions with marine vertebrates. The blade tips of these devices will likely move at speeds of about 12 ms^{-1}, or 23 knots. In collision terms, blades rotating at this speed are akin to the bow of a ship or keel of a yacht, both of which are involved in cetacean-ship strikes, a major cause of mortality for some cetacean species (e.g. IWC, 2001; Knowlton and Kraus, 2001). Given how few tidal turbines are operational and how little data are available to date on their actual effects in relation to marine megavertebrates, it is difficult to evaluate vertebrate collision risks.

Man-made collision risks are more common than is generally supposed, and behavior in the face of collision risk will vary with age. Juveniles of a given species are at greater risk due to lack of prior experience. The potential for animals to escape collisions with marine renewable devices will vary from species to species and will depend on factors such as body size, social behavior (especially schooling or group structure), foraging tactics, curiosity, underwater agility and sensory abilities (Wilson *et al.,* 2007). Fully aquatic species will of course be at greater risk than those such as diving birds, which only spend a small proportion of their time underwater.

Wilson *et al.* (2007) developed a model to investigate the potential encounter rate between 100 tidal turbines off the Scottish coast and local populations of harbor porpoises and herring of well-documented abundance. In one year of operation, the model predicted that 2% of the local herring population and 3.6 to 10.7% of the porpoise population would encounter an operational turbine. While encounters do not equate to collisions, there is no information at present on how marine organisms will react to such an encounter. If a large proportion of encounters were to result in collisions, the authors concluded that such levels of injury to the porpoise population would have a severe impact at the population level. The findings also show that encounter rate and thus collision risk increases with body size – herring have a lower likelihood of encountering a turbine than do porpoises – thus, animals such as whales and basking sharks will have greater still encounter rates. A detailed and comprehensive description of the collision hazards presented by the variety of tidal turbine technologies and associated mooring equipment, to fish, marine mammals and birds is available in Wilson *et al.* (2007).

Fraenkel (2006) suggested that collisions between marine wildlife and tidal turbine blades would be unlikely and, if they occurred, probably not fatal due to the smooth and "not very fast moving" surface. The author points out that ship propellers interact with the water at far greater power densities and apply energy to the water rather than removing it, thus posing a more serious risk to wildlife. However, Wilson *et al.* (2007) highlight the fact that the turbine blades are operating at speeds more similar to the movement of a ship's hull. Marine megavertebrates, especially large whales, often collide with moving vessels (see Section 2.2), therefore, at this point, the presence of moving turbine blades underwater should be considered a possible risk to at least some species.

The tidal turbine in Strangford Loch, Northern Ireland has been in place since 2007 (Bedford and Fortune, 2010; Davison and Mallows, 2005). The evidence so far from environmental impact assessment studies suggests no fatal interactions between seals and the turbine blades (from examination of dead stranded seal carcasses), nor does the turbine appear to present a barrier to harbor porpoise movement (from analysis of acoustic monitoring data).

It has been suggested that tidal flow installations could lead to changes in tidal level, turbidity and sedimentation, which could impact estuary ecosystems (Gordon, 1994). Changes in sediment transport around an installation may particularly affect salt marsh habitats, which in turn could impact species such as the Northern diamond-backed terrapin.

5.3.3 Wave Power Devices

The potential to capture wave energy has seen increasing interest, with pilot projects in a number of countries (Dal Ferro, 2006; Cada *et al.,* 2007; Boehlert *et al.,* 2008; Nelson *et al.,* 2008). The technology lags behind that of offshore wind power, but it could, potentially, provide a significant contribution to renewable energy production in areas with suitable wave conditions (Carbon Trust, 2006; Kerr, 2007). Wave energy converter devices, or WECs, tend to involve less rigid structure in the water column, but do consist of significant components at the water surface. There is therefore a risk of collision between marine animals and WECs, especially for species which regularly cross the air-water interface or spend a significant proportion of their time on the surface.

Pinnipeds (seals and sea lions) may be likely to use floating devices as haul-out sites, and birds may use them as landing or roosting areas; thus there may be risks to these animals as they get onto or off of the structures and potentially come into contact with exposed moving or articulated parts (Wilson *et al.,* 2007). Cetaceans are regularly at the water surface to breathe, while basking sharks and sunfish can, in certain seasons, spend extended periods at or very close to the water surface. These species are at risk either of swimming directly into the structures or of being hit if the structures were to pitch down on an animal in rough sea conditions (Wilson *et al.,* 2007). It is unclear how aware cetaceans and large fish will be of the presence of such structures and thus how capable they will be of avoiding them.

It is also thought that WECs may act as fish aggregating devices (FADs), a technique used in fisheries whereby floating material is placed in the water to attract fish. In a study of offshore wind farms in Swedish waters, Wilhelmsson *et al.* (2006) suggested that these structures were functioning as combined artificial reefs and FADs for small demersal fish. Fayram and deRisi (2007) suggest that floating offshore wind turbines (and thus other structures such as WECs) could positively affect multiple stakeholder groups and potentially support higher recreational fish catch. However, any FAD effect will then likely also attract predators (such as marine mammals and birds) to these areas, which in turn may increase collision risk to these species.

5.4 The Future

5.4.1 Possible Long-term Effects

In anticipating what effect various MREIs have on the marine ecosystems into which they are placed, the immediate and short-term effects of installation and operation at an individual and community level as well as longer-term effects at the population level need to be considered. It is possible that some species will develop avoidance skills to deal with circumstances which may otherwise cause them harm or may become habituated to impacts such as noise and turbidity. They may exhibit short-term changes in behavior in response to anthropogenic disturbance (e.g. Bejder *et al.,* 1999; Hastie *et al.,* 2003; Lusseau, 2003).

Alternatively, certain areas may be abandoned by species whose environment has become compromised through the introduction of MREIs; short-term avoidance strategies may lead to long-term displacement (Lusseau, 2004). Abandonment of otherwise favourable habitats by cetaceans due to anthropogenic disturbance has been observed in the past (e.g. Bryant *et al.,* 1984; Jefferson, 2000). Lusseau (2005) suggested that avoidance of a key habitat area by bottlenose dolphins, as a result of pressure from boat traffic, could have demographic impacts at a population level.

It is also important to recognize that the response of one species of marine megavertebrate to any given source of disturbance will not be indicative of responses by other species. Watkins (1986) documented species-specific changes in behavior, both positive and negative, in relation to vessels over a 25-year period. Clearly, considerable research effort will be necessary in this field and will require the support and cooperation of the MREI industry. It will also be essential to consider the effects of any planned MREI not in isolation but rather in combination with other MREIs and other anthropogenic impacts in the region, bearing in mind the scales of habitat use relevant to marine megavertebrates.

5.4.2 Mitigation

Much in the same way as pingers, seal-scarers and other Acoustic Deterrent Devices (ADDs) have been developed to deter cetaceans and seals away from trawl nets and fish farms (with varying degrees of success; e.g. Hodgson *et al.,* 2007; Berrow *et al.,* 2008; Caretta *et al.,* 2008; Gazo *et al.,* 2008; Leeney *et al.,* 2008), it may be possible to develop new technologies to alert animals to the presence of tidal turbines or other MREI structures which pose a risk. However, even if such devices are initially effective, animals can also become habituated to these devices, making them less effective over time (Dawson *et al.,* 1998; Cox *et al.,* 2001). Deterrents will not work for all species, as different species have different primary senses and different visual, olfactory and auditory capabilities.

Vocalizations and echolocation are essential to communication and environmental exploration for cetaceans, so deterrents using noise work well for this group of species. Sea turtles and pelagic fishes are highly visual predators, thus visual cues most likely play an important role (Southwood *et al.,* 2008). The reliance of some species on visual cues may also suggest that detection of MREI devices at night may be compromised; directed research will be required to address whether this will be a concern.

Since many species of megavertebrate are known to exhibit diel patterns of habitat use (e.g. Goold, 2000; Elwen *et al.,* 2006), it will be essential to use acoustic monitoring to at least provide data on cetacean habitat use around MREIs at night as well as during the day, in order to assess risk levels outside of daylight hours. The use of sonar devices to detect approaches by marine megavertebrates, as utilized in the SeaGen tidal turbine project in Stangford Lough, may also be beneficial in addressing this issue (Bedford and Fortune, 2010).

6 References

Photo: E. Bradfield

Section 2.1

Data Summary

Massachusetts Audubon Society Aerial Surveys:

Aerial surveys for seabirds were conducted in Nantucket Sound by the Massachusetts Audubon Society in summer and fall 2002-2004 as part of the assessment of the Cape Wind offshore wind energy project (Sadoti *et al.,* 2005). Sightings of sea turtles, ocean sunfish and basking sharks were recorded opportunistically during the surveys (T. Allison, Massachusetts Audubon Society, *pers. comm.*, 7 September 2010).

North Atlantic Right Whale Consortium Sightings Database:

The North Atlantic Right Whale Consortium (NARWC) sightings database contains records of thousands of sightings of right whales in the North Atlantic Ocean as well as sightings of many other species of whales, dolphins, sea turtles, seals and large fishes (Kenney, 2001). Most sightings in the sightings database are from aerial and shipboard surveys conducted from the late 1970s to the present. The sightings contained in the database come from a wide variety of contributors, both Consortium members and others. For this report, the database was queried (Right Whale Consortium, 2010) for sightings within the area encompassing Nantucket and Vineyard Sounds and the waters south of Muskeget Channel and Martha's Vineyard and Nantucket Islands (41° 00' N north to 41° 45' N, and 71° 10' W east to 70° 00' W). Sightings data are not effort corrected and purely reflect presence of animals, rather than patterns of distribution.

Approximately 11,000 km of survey effort was conducted in the above area from 1979-2005, most of which was south of the Sounds (R. Kenney, University of Rhode Island/North Atlantic Right Whale Consortium, *pers. comm.*, 2 September 2010). For comparison, 66,466 km of aerial survey effort was conducted during winter and spring 1998-2002 in Cape Cod Bay, a smaller area of known importance to North Atlantic right whales (Nichols *et al*., 2008).

Massachusetts Sea Turtle Disentanglement Network

The Massachusetts Sea Turtle Disentanglement Network (MASTDN) was formed in 2005 to respond to and document bycatch issues related to sea turtles in and around the state waters of Massachusetts. MASTDN receives reports from federal, state and municipal agencies as well as the commercial and recreational boating public through a dedicated marine animal reporting hotline. Support for this work is provided by ESA Section 6 in conjunction with Massachusetts Division of Marine Fisheries. Data can only be used for the purpose of this literature review and should not be used for any other reason or application without the express written consent of PCCS.

Section 2.2

Museum Collections:

American Museum of Natural History (AMNH): http://entheros.amnh.org/db/emuwebamnh/logon.php.

Harvard Museum of Comparative Zoology (MCZ): http://mczbase.mcz.harvard.edu/SpecimenSearch.cfm.

Smithsonian Institute (SI): http://collections.nmnh.si.edu/vzmammals/pages/nmnh/vz/DtlQueryMammals.php.

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APPENDIX I:
Supplementary Tables

Table A1:

Seals Observed Stranded in Nantucket Sound and along Outer Cape Cod by Species and Year. Data courtesy of the National Marine Fisheries Services (NMFS)

	2005	**2006**	**2007**	**2008**	**2009**	**Species total**
Harbor	35	27	15	35	17	129
Harp	38	14	30	24	30	136
Gray	31	27	50	50	45	203
Hooded	3	8	2	2	0	15
Yearly Total	107	76	97	111	92	483

Table A2:

Historic and Recent Observations of Harbor Seals in Southern New England

Decade	Observation	Source
1860-1869	Allen describes harbor seals occurring in Wellfleet and Provincetown (rare). He also describes hundreds of seals on the Boston Harbor Islands in the summer.	Allen (1869)
1870-1879	A harbor seal was collected on Penikese Island (Elizabeth Islands) in 1873	MCZ
1890-1899	A harbor seal was collected in Chatham in 1893	MCZ
1900-1929	No observations	
1930-1939	Harbor seals were permanent residents on Cape Cod and pupping occurred through out Massachusetts.	Prescott (1981) as reported in Payne & Schneider (1984)
1940-1949	Harbor seals were permanent residents on Cape Cod and pupping occurred through out Massachusetts.	Prescott (1981) as reported in Payne & Schneider (1984)
1950-1959	No observations	
1960-1969	A harbor seal was collected on Muskeget Island in the late spring or early summer 1960. Another harbor seal was collected in Cape Cod Bay in May 1962.	MCZ
1970-1979	There are 15 harbor seal records in the Smithsonian Institute's collections and 8 in the American Museum of Natural History's collections. All seals were collected on Cape Cod or Islands.	SI, AMNH
1980-1989	There are 28 harbor seal records in the American Museum of Natural History collected on Cape Cod or the Islands. The MCZ holdings contain 3 harbor seals.	AMNH, MCZ

Table A3:

Historic and Recent Observations of the Gray Seal in Southern New England

Decade	Observation	Source
1920-29	Two adult males were killed on Muskeget Island (MA).	Andrews & Mott (1967)
1930-39	No observations available	
1940-49	4 mummified pups observed on Muskeget (MA) in 1948.	Rough (1981)
	Interviews with local Nantucket (MA) residents indicate that bounties were paid on approximately 40 gray seals (mostly mothers and pups) in the late 1940's and early 1950's.	Andrews & Mott (1967)
1950-59	Mr. Clint Andrews brings the skull of a large skeleton and that of a pup to the MCZ for identification in 1958 (pup was collected on Muskeget Island, MA prior to 1958). According to the MCZ's records, the seals were collected in 1948, MCZ51282 & MCZ51283.	Andrews & Mott (1967) & MCZ
	Massachusetts paid bounties on approximately 25 gray seals from 1958-1962 (likely but the latter date is unclear).	Andrews & Mott (1967)
1960-69	Massachusetts bounty is repealed in 1962.	Massachusetts Acts & Resolves
	Three pups were born at Muskeget (MA) in 1963.	Rough (1983)
	An adult female was shot for bounty (despite its repeal) at the Elizabeth Islands (MA) in 1964. This is probably MCZ51488, collected in Lackey's Bay (Elizabeth Islands) in 1964.	Andrews & Mott (1967) MCZ
	Massachusetts legislation passed in 1965 protects the gray seal.	Massachusetts Acts & Resolves
	Less than 1 pup per year was observed in Nantucket Sound (MA) 1964-1970.	Rough (1983)
1970-79	A white coat pup was observed on Nantucket (MA) in March 1973.	Gilbert *et al.* (1977)
	Aerial surveys flown during January-April, 1977, found no pups in Nantucket Sound.	
	A white coat pup was observed on 10 February 1978 in Provincetown, MA.	Rough (1981)
1980-89	MCZ58032, a gray seal that stranded on a beach in Orleans, MA in 1980 and later died at the New England Aquarium, Boston, MA.	MCZ
	A juvenile gray seal stranded and died on Block Island in 1980.	Kenney (2005)
	MCZ60654, a juvenile gray seal that stranded and died on Martha's Vineyard in 1987.	MCZ

Table A4:

Summary of Whalenet Satellite Tags Deployed on Harbor and Gray Seals in Nantucket Sound and Cape Cod Waters

Seal ID	Species	Release Date	Important Locations
91088	Gray	3 May 2009	Muskeget Channel
39387	Gray	23 June 2009	Chatham Harbor N. Monomoy I.
39392	Gray	15 April 2008	South of Nantucket I.
39391	Gray	20 June 2007	Monomoy I.
39389	Gray	5 August 2007	Monomoy I. Outer Cape Cod
47823	Harbor	26 April 2005	Muskeget Channel
47822	Harbor	29 December 2004	Monomoy I. Outer Cape Cod
44861	Harbor	15 March 2005	Monomoy I. Nantucket I. Muskeget Channel?
Solange	Gray	7 February 2004	Nantucket I. Monomoy I.
Hopper	Harbor	17 June 2004	Buzzard's Bay Cape Cod Canal
Jersey Girl	Harbor	10 January 2003	Buzzard's Bay Cape Cod Canal
Bristol	Harbor	22 October 1998	Monomoy I.
McHenry	Gray	23 November 1998	Monomoy I. Muskeget I. Tuckernuck I. Elizabeth Islands

Appendix II:

Table of Online Sites Relating to Marine Renewable Energy Developments and Monitoring Methods

website	wind	wave	tidal	industry	research	MREI site	other	notes
http://www.seageneration.co.uk/			x	x	x	x		SeaGen tidal tubine in Strangford Lough, Northern Ireland
http://www.wavec.org/				x	x			Wave Energy Centre - development and promotion of ocean wave energy - Portugal
http://www.sams.ac.uk/research/departments/research/research-themes/theme-3-marine-renewable-energy	x	x	x		x			Scottish Association of Marine Science (SAMS)
http://www.mrec.umassd.edu/	x	x	x		x			New England Marine Renewable Energy Center - USA
http://www.emec.org.uk/		x	x		x			European Marine Energy Centre. Test site in Orkney, Scotland
http://www.seai.ie/Renewables/Ocean_Energy/AMETS/		x	x		x	x		Atlantic Marine Energy Test Site - Ireland
http://nnmrec.oregonstate.edu/		x	x		x			Northwest National Marine Renewable Energy Center - USA
http://www.oregonwave.org/		x			x		non-profit	Oregon Wave Energy Trust (OWET) - supports responsible development of wave energy in Oregon. Environmental research at http://www.oregonwave.org/our-work-overview/environmental-research/
http://www.pge.com/about/environment/pge/cleanenergy/waveconnect/		x			x			Humboldt WaveConnect project - test site for multiple devices in California
http://www.smru.co.uk/renewable-energy.aspx			x		x			Sea Mammal Research Unit Ltd - EIA services & research
http://www.nrel.gov/					x			Research, development, commercialization and deployment of renewable energy - USA

website	wind	wave	tidal	industry	research	MREI site	other	notes
http://www.bwea.com/	x	x	x	x				Renewable UK - trade and professional body for the UK wind and marine renewables industries
http://www.fredolsen-renewables.com/	x	x	x	x				Fred Olsen Renewables - Norway & UK
http://www.vattenfall.com/en	x			x				Vattenfall - UK & Scandinavia
http://www.dongenergy.co.uk/Pages/	x			x				Dong Energy - offshore wind developments, Europe
http://www.aquamarinepower.com/		x		x				Aquamarine Power – "the oyster;" Scotland
http://www.aegirwave.com/		x		x		x		Wave power test sites - Shetland (UK)
http://www.pelamiswave.com/		x		x				Pelamis - wave power technology developer, UK
http://www.wavebob.com/		x		x				Wavebob - Ireland
http://www.seapower.ie/		x		x				Sea Power Ltd - Ireland
http://www.oceanpowertechnologies.com/		x		x				Ocean Power Technologies - USA - includes projects at Kanoehe Bay, Hawaii; Coos Bay, Oregon & the Wavehub, UK
http://www.marineturbines.com/			x	x				Marine Current Turbines - includes projects at the Skerries, North Wales & Bay of Fundy, Canada
http://www.oceanrenewablepower.com/ocgenproject_alaska.htm			x	x		x		Ocean Renewable Power Company - Alaska projects including Cook Inlet tidal energy project
http://www.atlantisresourcescorporation.com/			x	x				Atlantis Resources Corporation - technology (turbine) development, resource assessment, project management. London & Singapore
http://www.renewableenergyworld.com/rea/home	x	x	x					Renewable Energy World - online news re renewable energy
http://www.offshorewindfarms.co.uk/Pages/COWRIE/	x							COWRIE (Collaborative Offshore Wind Research Into The Environment)
http://www.hornsrev.dk/Engelsk/default_ie.htm	x					x		Horns Rev offshore wind farm (EIA reports at http://www.hornsrev.dk/Engelsk/default_ie.htm)

website	wind	wave	tidal	industry	research	MREI site	other	notes
http://www.dongenergy.com/Nysted/EN/Pages/	x					x		Nysted offshore wind farm - Denmark
http://www.renewableenergyfocus.com/		x	x				magazine & online resource	Renewable Energy Focus - forum covering renewable energy industry topics
http://www.wavehub.co.uk/		x				x		Wave Hub - test site for multiple devices in Cornwall, UK
http://www.racerocks.com/racerock/energy/tidalenergy/tidalenergy2.htm			x					Race Rocks tidal energy project - Canada
http://www.environmentalexchange.info/Links/							list of links	Environmental Impacts of Offshore Renewable Energy Developments for the Exchange of Information (on behalf of OSPAR)
http://www.peventuresllc.com/								Consulting and business development firm - services include project development, regulatory coordination, stakeholder engagement & project financing
http://www.oreg.ca/							activities unclear	Ocean Renewable Energy Group - Canada
http://www.iea-eces.org/					x			International Energy Agency - development of alternative energy sources, energy research and development

Marine Megavertebrates and Fishery Resources in the Nantucket Sound – Muskeget Channel Area

Figure 2.1: Cetacean sightings in Southern Massachusetts.

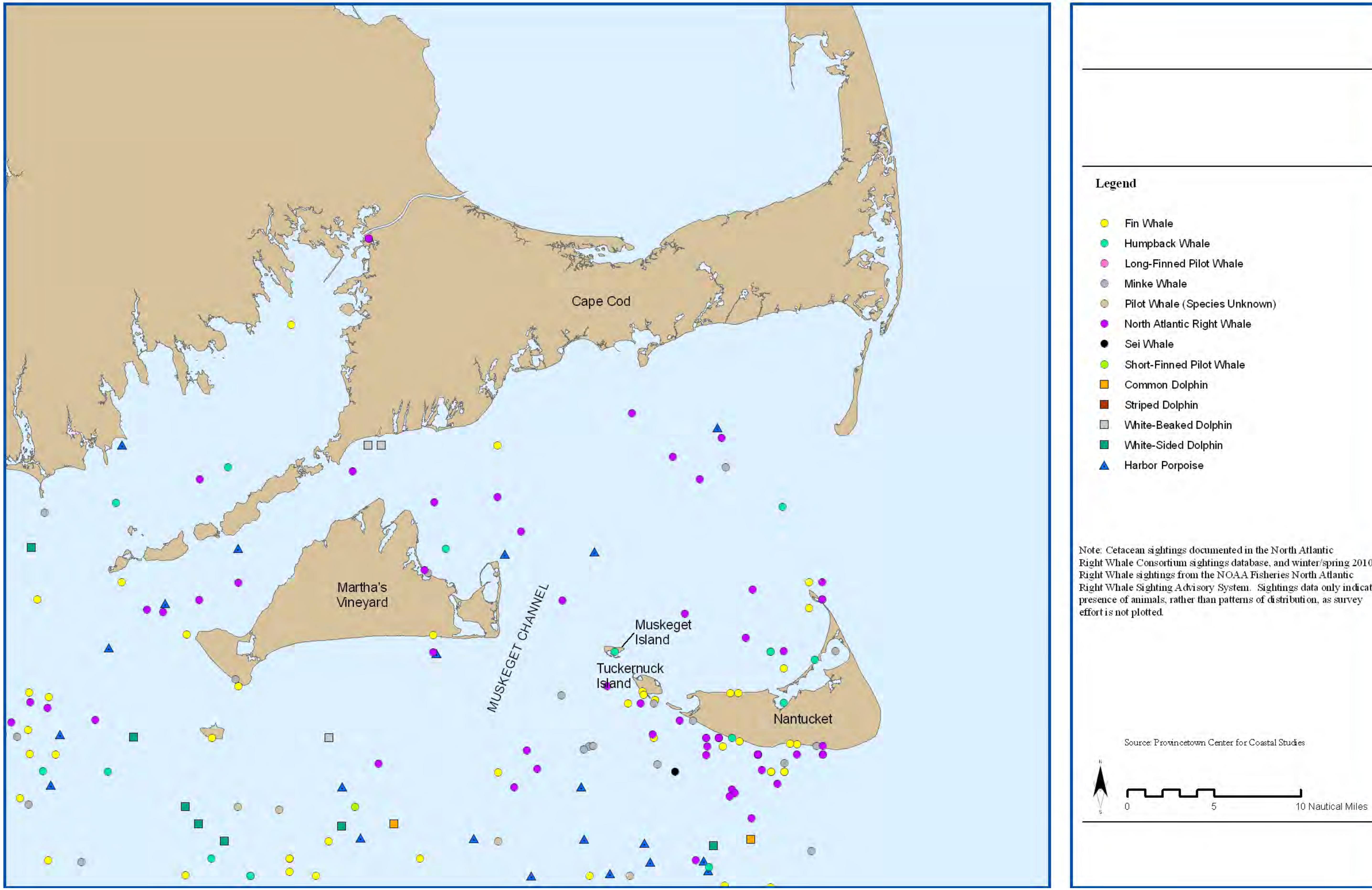

Figure 2.2: Seal pupping colonies and haul-out sites in the Nantucket Sound – Muskeget Channel area.

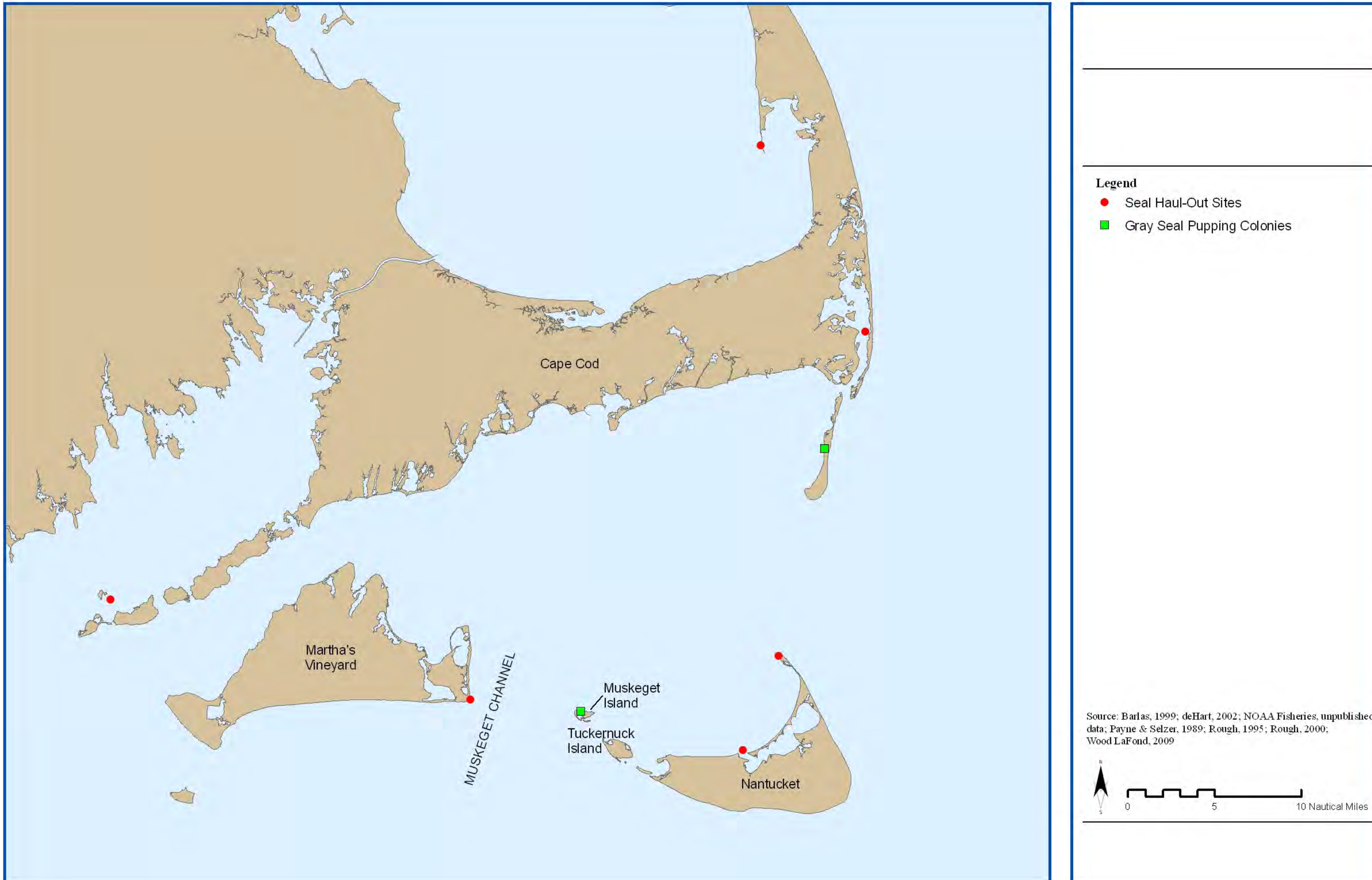

Figure 2.13: Confirmed entangled sea turtle sightings in waters south of Cape Cod as reported to the Massachusetts Sea Turtle Disentanglement Network (MASTDN). Support for this work is provided by ESA Section 6 in conjunction with Massachusetts Division of Marine Fisheries. Data can only be used for the purpose of this literature review and should not be used for any other reason or application without the express written consent of the PCCS.

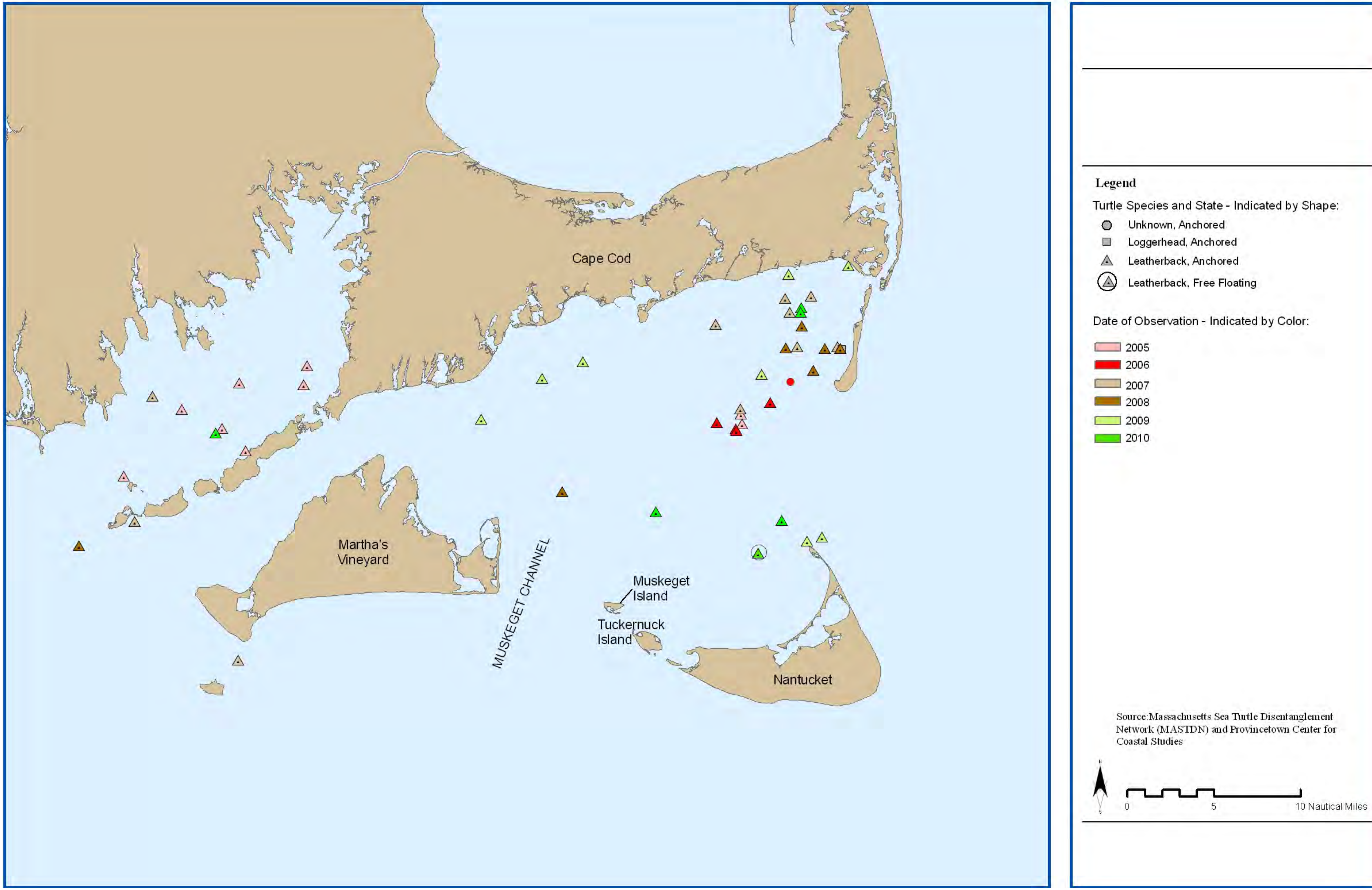

Figure 2.14: Opportunistic sea turtle sightings recorded by Massachusetts Audubon during aerial surveys of tern activity in Nantucket Sound as part of an assessment for the Cape Wind energy project. Surveys were completed in August and September from 2002-2004.

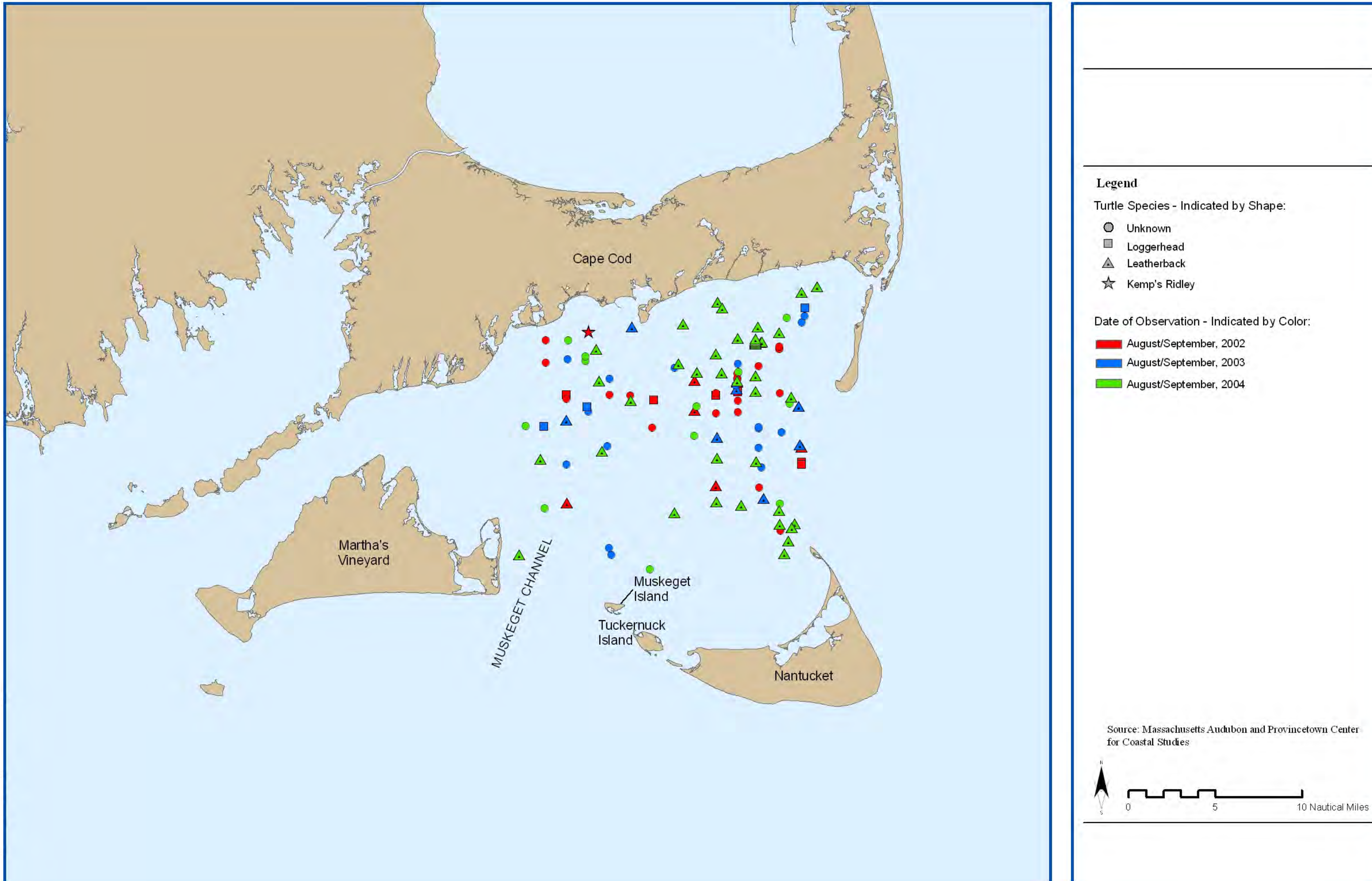

Figure 2.15: Sea turtle sightings documented in the North Atlantic Right Whale Consortium sightings database for Nantucket Sound. Sightings data only indicate presence of animals, rather than patterns of distribution, as survey effort is not plotted.

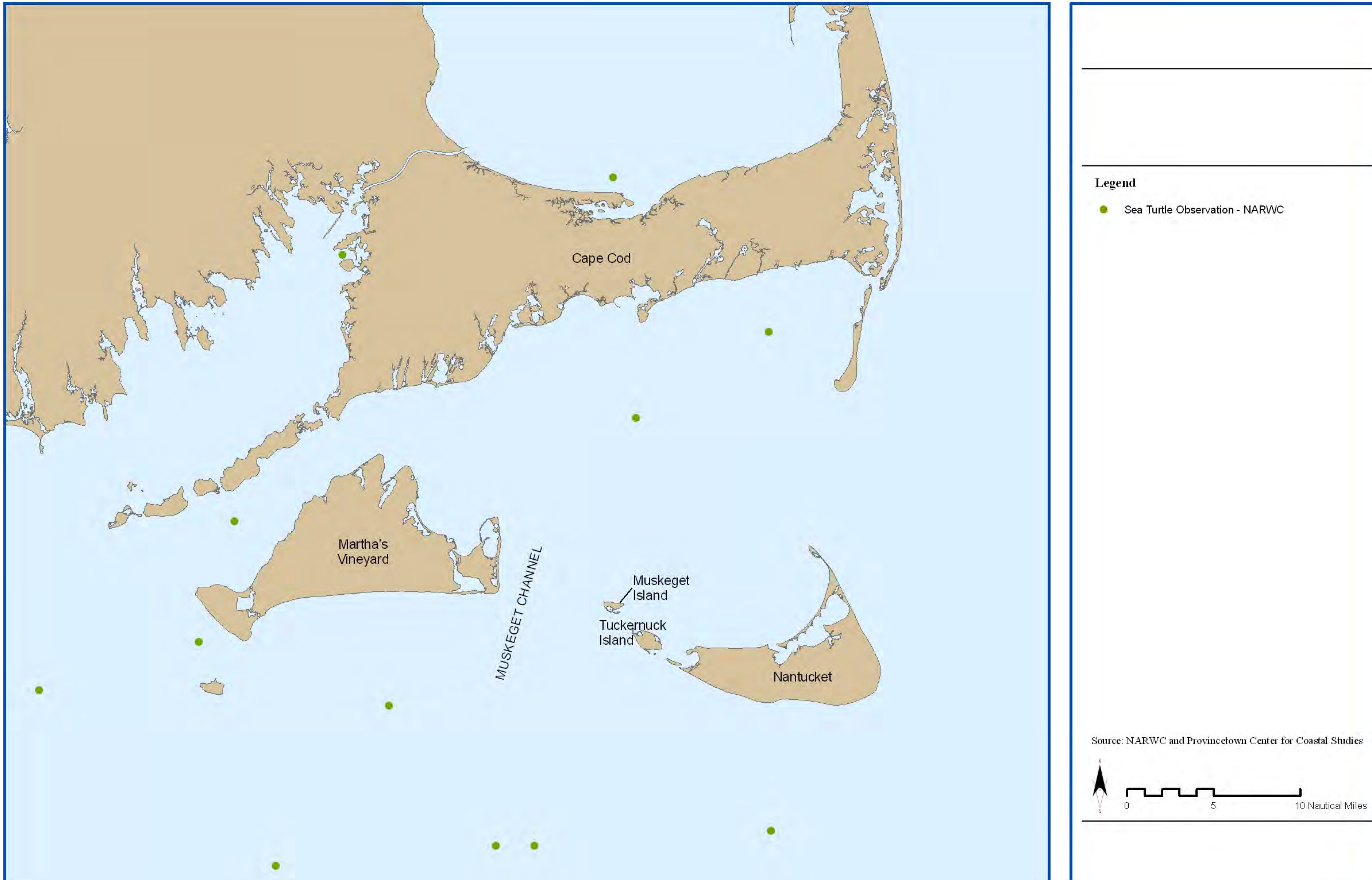

Figure 2.16: Basking sharks and ocean sunfish sightings data for Southern Massachusetts.