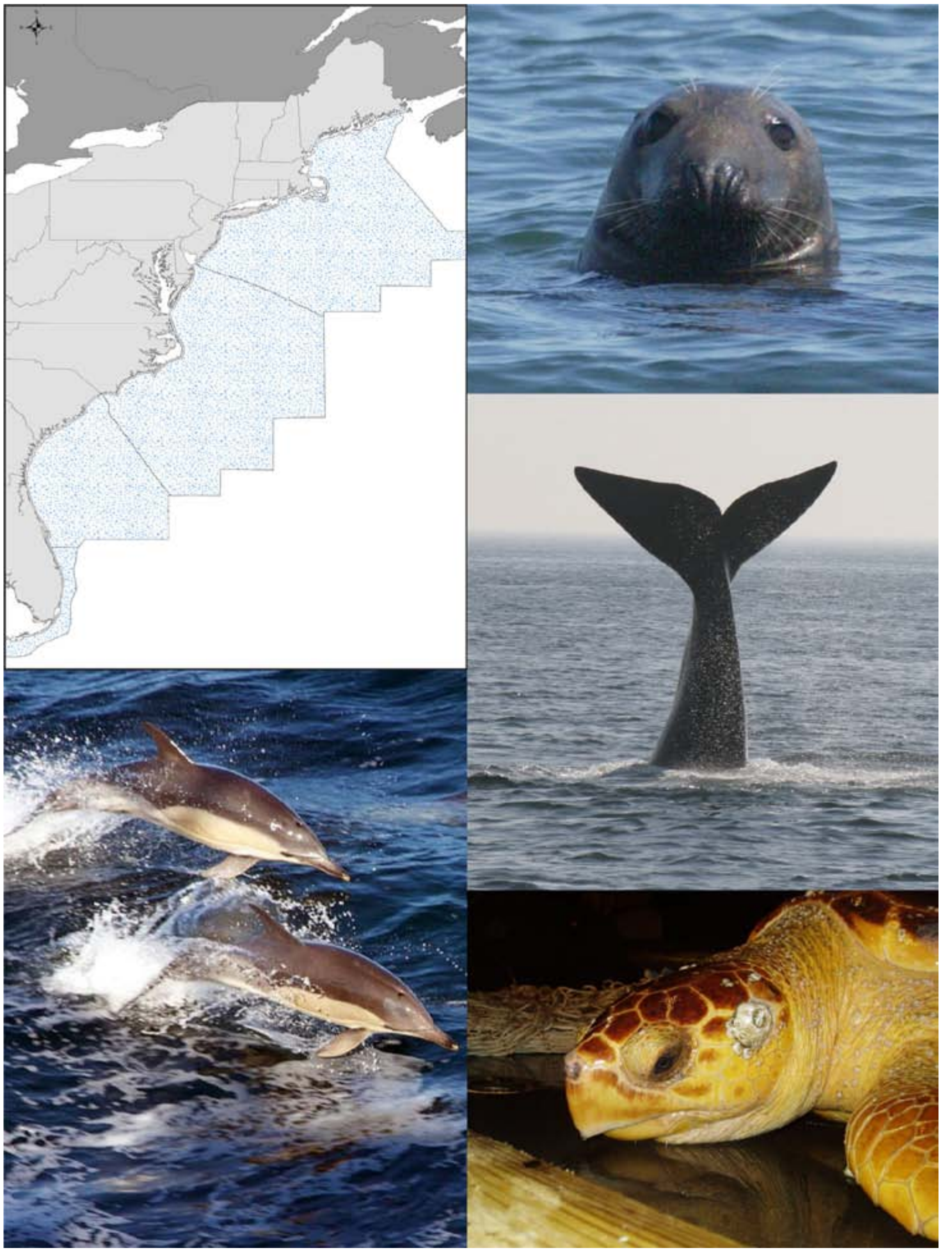
OCS Study
BOEM 2012-109

Literature Search and Data Synthesis for Marine Mammals and Sea Turtles in the U.S. Atlantic from Maine to the Florida Keys



U.S. Department of the Interior
Bureau of Ocean Energy Management
Gulf of Mexico OCS Region

OCS Study
BOEM 2012-109

Literature Search and Data Synthesis for Marine Mammals and Sea Turtles in the U.S. Atlantic from Maine to the Florida Keys

Authors

G.T. Waring
S.A. Wood
E. Josephson

Prepared under BOEM Contract
M09PG00018
by

NOAA Fisheries Service
Northeast Fisheries Science Center
Protected Species Branch
166 Water St.
Woods Hole, MA 02543-1026

Published by

U.S. Department of the Interior
Bureau of Ocean Energy Management
Gulf of Mexico OCS Region

New Orleans, LA
December 2012

DISCLAIMER

This report was prepared under contract between the Bureau of Ocean Energy Management (BOEM) and the National Oceanic and Atmospheric Administration (NOAA). This report has been technically reviewed by BOEM, and it has been approved for publication. Approval does not signify that the contents necessarily reflect the views and policies of BOEM, nor does mention of trade names or commercial products constitute endorsement or recommendation for use. It is, however, exempt from review and compliance with BOEM editorial standards.

REPORT AVAILABILITY

This report is available on CD from the Bureau of Ocean Energy Management, Gulf of Mexico OCS Region, for $15.00, and as a pdf file downloaded from the BOEM Web site. Copies can be viewed at selected Federal Depository Libraries.

This report is available on CD from the Bureau of Ocean Energy Management, Gulf of Mexico OCS Region, for $15.00, and free of charge as a pdf file downloaded from the BOEM Web site. Copies can also be viewed at selected Federal Depository Libraries. The addresses are listed below.

To order a CD, use the Gulf of Mexico OCS Region contact information below and reference OCS Study BOEM 2012-109. To download a pdf copy, use the Environmental Studies Program Information System (ESPIS) and search on the study report number. In the near future, you will also be able to get this report also from the National Technical Information Service.

U.S. Department of the Interior
Bureau of Ocean Energy Management
Gulf of Mexico OCS Region
Public Information Office (GM 217G)
1201 Elmwood Park Blvd.
New Orleans, Louisiana 70123-2394
Phone: (504) 736-2519, 1-800-200-GULF
Fax: (504) 736-2620

U.S. Department of Commerce
National Technical Information Service
5301 Shawnee Rd.
Springfield, Virginia 22312
Phone: (703) 605-6000, 1-800-553-6847
Fax: (703) 605-6900
Web site: http://www.ntis.gov/

CITATION

Waring, Gordon T., Stephanie A. Wood, and Elizabeth Josephson. **2012**. Literature search and data synthesis for marine mammals and sea turtles in the U.S. Atlantic from Maine to the Florida Keys. U.S. Dept. of the Interior, Bureau of Ocean Energy Management, Gulf of Mexico OCS Region, New Orleans, LA. OCS Study BOEM 2012-109. 456 pp.

CONTENTS

LIST OF FIGURES

LIST OF TABLES

ACRONYMS AND ABBREVIATIONS

AEP	Auditory evoked potentials
AMAPPS	Atlantic Marine Assessment Program for Protected Species
ATBA	Area to be avoided
BDE	Bromodiphenyl
BOEM	Bureau of Ocean Energy Management
BOEMRE	Bureau of Ocean Management, Regulation and Enforcement
BT	Butyltin
Cd	Cadmium
CETAP	Cetacean and Turtle Assessment Program
CI	Confidence interval
CITES	Convention on International Trade in Endangered Species of Wild Fauna and Flora
Cu	Copper
CV	Coefficient of variation
DDD	Dichlorodiphenyldichloroethane
DDE	Dichlorodiphenyldichloroethylene
DDT	Dichlorodiphenyltrichloroethane
DOD	Department of Defense
DOI	Department of the Interior
DTAG	Digital acoustic recording tag
EEZ	Exclusive Economic Zone
ESA	Endangered Species Act
Fe	Iron
HBCD	Hexabromocyclododecane
HCB	Hexachlorobenzene
HCH	Hexachlorocyclohexane
Hg	Mercury
IA	Interagency Agreement
ICES	International Council for the Exploration of the Sea
IUCN	International Union for Conservation of Nature
IWC	International Whaling Commission
MMPA	Marine Mammal Protection Act
Mn	Manganese
mtDNA	Mitochondrial DNA (deoxyribonucleic acid)
MW	Megawatt
NAFO	North Atlantic Fisheries Organization
NAO	North Atlantic Oscillation
NEFSC	Northeast Fisheries Science Center

NEPA	National Environmental Policy Act
Ni	Nickel
NMFS	National Marine Fisheries Service
NOAA	National Oceanographic and Atmospheric Administration
OBIS	Ocean Biogeographic Information System
OC	Organochlorine
OCS	Outer Continental Shelf
Pb	Lead
PBB	Polybrominated biphenyl
PBDPE	Polybrominated diphenyl ethers
PBR	Potential biological removal
PCB	Polychlorinated biphenyl
PDV	Phocine distemper virus
PFC	Perfluorinated compounds
PN2	Partial pressure nitrogen
POP	Persistent organic pollutant
PSB	Protected Species Branch (Northeast Fisheries Science Center)
SEFSC	Southeast Fisheries Science Center
SST	Sea surface temperature
TRT	Take Reduction Team
UK	United Kingdom
UME	Unusual Mortality Event
UNCW	University of North Carolina, Wilmington
Zn	Zinc

1. INTRODUCTION

In September 2009, the Minerals Management Service (renamed the Bureau of Ocean Energy Management, Regulation, and Enforcement [BOEMRE] on 21 June 2010, now the Bureau of Ocean Energy Management [BOEM]), Department of the Interior (DOI) and the Northeast Fisheries Science Center (NEFSC), National Marine Fisheries Service (NMFS), National Oceanic and Atmospheric Administration (NOAA), U.S. Department of Commerce established an inter-agency agreement (IA), through which NOAA was to provide services to the DOI, BOEMRE (now BOEM) in the form of literature review of the research on marine mammals and sea turtles in the U.S. Atlantic from Maine to the Florida Keys.

1.1 PROJECTS GOALS AND OBJECTIVES

The goals and objectives of this IA were: 1) to provide a reference document that includes the latest and best information on marine mammals and sea turtles in the U.S. Atlantic from Maine to the Florida Keys, 2) to identify data gaps in the knowledge of marine mammals and sea turtles in the Atlantic, and 3) to identify research priorities recommended by national/regional agencies and groups (e.g., Marine Mammal Commission, NMFS and FWS recovery plans, Sea Turtle Expert Working Group,). This study area corresponds to BOEM north-Atlantic, mid-Atlantic, south-Atlantic and Straits of Florida outer continental shelf (OCS) planning areas. The document provides a basis for analysis of potential impacts of BOEM-regulated activities on protected sea turtles and marine mammals as required by the National Environmental Policy Act (NEPA), for petitioning that might be required by the Marine Mammal Protection Act (MMPA), and for Section 7 consultations under the Endangered Species Act (ESA). The document can also serve as a guide to future decisions on planning and funding of needed research on sea turtles and marine mammals in the U.S. Atlantic.

Thirty-seven species of marine mammals seasonally occupy one or more of the BOEM OCS Planning Areas in U.S. Atlantic waters between the Florida Keys and Maine (Jefferson et al. 1994; Waring et al. 2011). The marine mammal community is diverse. Three of the five commonly recognized marine mammal taxa reside in this region: cetaceans (mysticetes [baleen whales] and odontocetes [toothed whales, dolphins, and porpoises]), sirenians (manatees), and pinnipeds (seals). Although marine mammals are broadly distributed throughout this region, individual species exhibit preferences for certain ecosystems (Bowen and Siniff 1999) based on ecological and life history requirements. Baleen whales have the widest distribution and make the most extensive seasonal migrations of all marine mammals in this region (Waring et al. 2011). For example, right (*Eubalaena glacialis*), humpback (*Megaptera novaeangliae*), fin (*Balaenoptera physalus*) and minke (*B. acutorostrata*) whales prefer continental shelf ecosystems, whereas blue (*B. musculus*), and sei (*B. borealis*), whales are associated with shelf-edge and deeper oceanic water. Humpback whales migrate from high latitude summer feeding grounds (i.e., Gulf of Maine) to winter breeding grounds in the Caribbean Sea (Katona and Beard 1990; Clapham 2000, 2009). Right whales exhibit a similar but less extensive migration; winter calving grounds are located in waters off the southeast U.S. coast (Kraus and Rolland 2007; Kenney 2009; Waring et al. 2011).

Odontocetes also exhibit preferences for particular ecosystems (Bowen and Siniff 1999; Reeves et al. 2002; Macleod et al. 2006a) . Harbor porpoise (*Phocoena phocoena*), Atlantic white-sided dolphins (*Lagenorhynchus acutus*) and long-finned pilot whales (*Globicephala melas*) are usually found in cool-temperate continental shelf waters off the northeast U.S. (Waring et al. 2011). Continental shelf break/pelagic species found in warm-temperate to cooler waters include bottlenose dolphins (*Tursiops truncatus*), short-beaked common dolphin (*Delphinus delphis*), Risso's dolphin (*Grampus griseus*) striped dolphin (*Stenella coeruleoalba*), spotted dolphins (*S. frontalis*), sperm (*Physeter macrocephalus*), and northern bottlenose (*Hyperoodon ampullatus*) whales, and Cuvier's (*Ziphius caviorstris*), Blainville's (*Mesoplodon densirostris*), Sowerby's (*M. bidens*), True's (*M. mirus*) beaked whales (Waring et al. 2011). Continental shelf break/pelagic species found in warm-temperate to tropical waters are pantropical spotted (*Stenella attenuata*), spinner (*S. longirostris*), Clymene (*S. clymene*), rough-toothed (*Steno bredanensis*), and Fraser's (*Lagenodelphis hosei*) dolphins, and melon-headed (*Peponocephala electra*), pygmy killer (*Feresa attenuata*), short-finned pilot (*G. macrorhynchus*), pygmy sperm (*Kogia breviceps*), dwarf sperm (*K. sima*) and Gervais' beaked (*M. europaeus*) whales. Within the odontocetes, only sperm and long-finned pilot whales are known to undergo long-range seasonal migrations (Bowen and Siniff 1999; Waring et al. 2011).

The four species of seals found in this area—harbor (*Phoca concolor vitulina*), gray (*Halichoerus grypus grypus*), harp (*Pagophilus groenlandicus*) and hooded (*Cystophora cristata*)—are primarily distributed in cool temperate waters off the northeast U.S. (New Jersey to Maine). Harp and hooded "ice seals" are transient in U.S. Atlantic waters. Stranding records extend as far south as Florida for all species, with most occurring in the North Atlantic Planning Area (Waring et al. 2011).

The Florida manatee (*Trichechus manatus latirostris*) is found in Atlantic coastal waters along the southeastern U.S., but extralimital movements northward have been recorded (Reynolds III and Wilcox 1994; Wright et al. 2002; Runge et al. 2007; FWS 2009).

Most marine mammal species in U.S. Atlantic waters are assumed to be trans-boundary stocks. Stock structure for most cetaceans is unknown, although significant progress has been made for North Atlantic right and humpback whales, bottlenose dolphins, and harbor porpoise (Waring et al. 2011). Harp and hooded seals found in U.S. waters are from Atlantic Canada breeding stocks, whereas gray seals from Atlantic Canada began recolonizing New England waters in the late 20th Century (Wood LaFond 2009). The putative stock structure for harbor seals spans the region between New England and the Canadian maritime providences of New Brunswick and southwest Nova Scotia (Stanley et al. 1996).

Marine mammal seasonal distribution and habitat use is closely associated with their prey and foraging ecology (Bowen and Siniff 1999). While cetaceans may occupy feeding areas for extended time periods (i.e., spring occurrence of right whales in the Great South Channel), pinnipeds forage both "locally" and at significant distances from haul-out sites.

Abundance estimates vary widely among the species, ranging from a few hundred (e.g., right whales) to around a hundred thousand (e.g., common dolphins). Based upon these estimates, the annual number of human-caused mortalities that can be sustained by the stocks without threatening its recovery (i.e. Potential Biological Removal or PBR) range from less than 1 (e.g., right whales) to 120,000 (common dolphins) (Waring et al. 2011).

Five species of sea turtles seasonally occupy one or more of the BOEM OCS Planning Areas in U.S. Atlantic waters between the Florida Keys and Maine. These species are protected under the Endangered Species Act of 1973. Although sea turtles live most of their lives in the ocean, adult females must return to beaches on land to lay their eggs. They often migrate long distances between foraging grounds and nesting beaches.

Kemp's ridleys (*Lepidochelys kempii*) are distributed throughout the Gulf of Mexico and U.S. Atlantic seaboard, from Florida to New England. Major nesting occurs in the Gulf of Mexico, but occasional nesting has been documented in North Carolina, South Carolina, and the Gulf and Atlantic coasts of Florida. The green turtle (*Chelonia mydas)* is globally distributed and generally found in tropical and subtropical waters along continental coasts and islands between 30° North and 30° South. In U.S. Atlantic and Gulf of Mexico waters, green turtles are found in inshore and nearshore waters from Texas to Massachusetts, the U.S. Virgin Islands, and Puerto Rico. Hawksbill turtles (*Eretmochelys imbricatea*) are circumtropical, usually occurring from 30° N to 30° S latitude in the Atlantic, Pacific, and Indian Oceans and associated bodies of water. Within the U.S., hawksbills are most common in Puerto Rico and its associated islands and in the U.S. Virgin Islands. In the continental U.S., the species is recorded from all the Gulf States and along the east coast as far north as Massachusetts, but sightings north of Florida are rare. Loggerhead turtles (*Caretta caretta*) are circumglobal, occurring throughout the temperate and tropical regions of the Atlantic, Pacific, and Indian Oceans. Loggerheads are the most abundant species of sea turtle found in U.S. coastal waters. In the Atlantic, the loggerhead turtle's range extends from Newfoundland to as far south as Argentina. During the summer, nesting occurs primarily in the subtropics. Although the major nesting concentrations in the U.S. are found from North Carolina through southwest Florida, minimal nesting occurs outside of this range westward to Texas and northward to southern Virginia. Leatherback turtle (*Dermochelys coriacea*) nesting grounds are located around the world, with the largest remaining nesting assemblages found on the coasts of northern South America and West Africa. The U.S. Caribbean, primarily Puerto Rico and the U.S. Virgin Islands, and southeast Florida support minor nesting colonies, but represent the most significant nesting activity within the United States. Adult leatherbacks are capable of tolerating a wide range of water temperatures, and have been sighted along the entire continental coast of the United States as far north as the Gulf of Maine and south to Puerto Rico, the U.S. Virgin Islands, and into the Gulf of Mexico.

Centuries of human activities have affected all North Atlantic marine mammal and sea turtle populations (Waring et al. 2011;). Although commercial harvesting ended decades ago, marine mammals and sea turtles in U.S. Atlantic waters are still affected by a variety of anthropogenic activities including: fishery bycatch, vessel strikes, acoustic activities, military activities, and energy production (Simmonds and Lopez-Jurado 1991; Croll et al. 2001; Laist et al. 2001; Lewison et al. 2004; Cox et al. 2006; Madsen et al. 2006a; Read et al. 2006; Merrick and Cole 2007; Vanderlaan and Taggart 2007; Waring et al. 2011).

1.2 REPORT ORGANIZATION

This report consists of four sections, including the introduction. Section two includes thirty-four (thirty-two species and two taxonomic groups) marine mammal and five sea turtle species summaries, representing all marine mammal and sea turtle species that occur in the BOEM Atlantic Planning Areas. The information presented here came from primary literature, NMFS stock assessment reports, species recovery plans and websites. The reports include information of the status, distribution, abundance, habitat preference, stock structure, life history traits, food habits, health (strandings, contaminants, disease), acoustics, fisheries interactions, vessel interactions, energy projects and data gaps and research recommendations. These summaries are meant to be comprehensive and current; therefore, they include information on these species from other parts of their range.

Section two also includes a collection of maps. These include sighting, stranding, tag, and fishery bycatch locations by species and well as other relevant information such as fishery management areas designated under the harbor porpoise regulations, and right whale critical habitat areas. Seasonal information is also presented. Sightings maps are not effort-corrected and as such should not be interpreted as complete representations of species distribution. A table of data sources utilized to create these maps is provided.

The third section of this report is a table identifying current marine mammal and sea turtle research occurring along the U.S. Atlantic Coast. This table was compiled from the abstracts of research presented at three important scientific conferences: The 18th Biennial Conference on the Biology of Marine Mammals, Quebec City, Canada (12-16 October 2009); the Annual Meeting of the Right Whale Consortium Meeting, New Bedford, Massachusetts, (3-4 November 2010); and the 31st Annual Symposium on Sea Turtle Biology and Conservation, San Diego, California (12-15 April 2011). The table includes research occurring in the BOEM Atlantic Planning Areas and contact information for the lead author.

The final section is a comprehensive bibliography of the literature included throughout the document. PDFs of open access literature have been compiled separately on a CD for BOEM.

2. SPECIES SUMMARIES

2.1 NORTH ATLANTIC RIGHT WHALE (*EUBALAENA GLACIALIS*)

2.1.1 Legal Status

The first legal protection of right whales, the 1931 Convention for the Regulation of Whaling, was put into effect in 1935 and was followed by the International Convention for the Regulation of Whaling in 1949. In the U.S., the right whale has been listed as endangered under the ESA (and its precursor) since 1970. The species is designated as depleted under the MMPA. In 2008, NMFS listed the northern right whale (*Eubalaena* spp.) as two separate, endangered species: the North Pacific right whale (*E. japonica*) and North Atlantic right whale (*E. glacialis*).

Critical habitat for the North Atlantic right whale was designated in 1994 (see Figure 3.1-2), and is currently under revision. Shipping regulations protecting the right whale were enacted in 2009, and the North-South lanes of the Boston Traffic Separation Scheme (TSS) were modified to reduce the interactions of vessels with whales. In 2009, a voluntary seasonal Area to be Avoided (ATBA) was also established from 1 April to 31 July for ships weighing more than 300 gross tons. Furthermore, since 2006, recommended routes were established in key right whale habitats in Florida, Georgia, and Massachusetts. In addition to vessel rerouting, speed restrictions are mandated, and vessels measuring at least 65 feet in length are required to travel at 10 knots or less in certain locations at certain times of the year (http://www.nmfs.noaa.gov/pr/shipstrike/).

To address fishing gear entanglement, under the auspice of the Atlantic Large Whale Take Reduction Team, NMFS published three rules in 2002 (parts of which have since been modified) directed at: seasonally closing certain areas to fishing, requiring sinking groundline at certain times, and requiring all buoys to have a weak link (http://www.nero.noaa.gov/whaletrp/). Despite initial resistance from fishermen, the team continues to push forward their efforts in hopes of reducing injuries to and deaths of large whales due to incidental entanglement in fishing gear.

2.1.2 General Distribution

The North Atlantic right whale is found seasonally in both coastal and shelf waters of the U.S. Atlantic coast. Individuals in the population range between winter calving and nursery grounds located off the coasts of Florida and Georgia (in the BOEM South Atlantic Planning Area) and summer feeding grounds from New England (BOEM North Atlantic Planning Area) to the Bay of Fundy and Scotian Shelf (National Marine Fisheries Service 2005). Longer distance movements are also reported (Knowlton et al. 1992), and the winter location of much of the population is unknown, so offshore distribution is uncertain (National Marine Fisheries Service 2005). Sighting and stranding locations of North Atlantic right whales are shown in Figure 2.1-1.

Patterns of occurrence and distribution of right whales have been studied using both aerial and shipboard surveys. Most of the population spends the spring and summer on feeding grounds off the northeastern U.S. and Nova Scotia (Kenney et al. 2001; Winn et al. 1986). In late fall, near-term females migrate to waters off the southeastern U.S. to give birth (Kraus et al. 1986).

2.1.3 General Abundance

An extensive photo-ID recapture database for the North Atlantic right whale (Hamilton et al. 2007) allows a near census of individual whales to be accumulated. The minimum population index (Minimum Number Alive or MNA), based on a review of the photo-ID recapture database as it existed on 6 July 2010, was 396 individually recognized whales known to be alive during 2007 (Waring et al. 2011). Mean growth rates of the MNA indices over the period 1990-2007 was 2.4% (Waring et al. 2011), which should mimic the population growth rate but is slightly biased high because the index does not include all whales.

An analysis of extinction time by Fujiwara and Caswell (2001) demonstrated that preventing the deaths of only two female right whales per year could increase the population growth rate to replacement level. This has recently been refuted in an updated analysis available from NEFSC (Pace, personal communication).

2.1.4 Habitat Preference

Brown and Winn (1989) examined patterns of right whale sighting in the Great South Channel feeding grounds and found that the distribution of sightings correlated with the 100-m isobath and with the thermal front. Using satellite telemetry on tagged whales, Baumgartner and Mate (2005) examined the summer and fall habitat of North Atlantic right whales in the Grand Manan Basin of the lower Bay of Fundy. Whale distribution and movements were correlated with environmental conditions such as depth, chlorophyll concentration, sea surface temperature and other hydrological variables. Whales that left the Bay of Fundy were found to visit areas characterized by low bottom water temperatures, high surface salinity, and high surface stratification and no correlation was found between whale distribution and oceanic fronts or areas with high phytoplankton concentrations. Whether or not it is frontal conditions that aggregate the copepods, right whale abundance and distribution on the feeding grounds is linked with copepod concentrations (Wishner et al. 1995; Pendleton et al. 2009; Pendleton 2010; Michaud and Taggart 2007). Michaud and Taggert (2007) estimated, based on energy density of available food in the Grand Manan feeding habitat at times when whales were present, that a minimum water column-integrated energy density of 3 kJ/m^3 could define a right whale feeding habitat. The absence of right whales from the Roseway Basin feeding ground between 1993 and 1999 was linked to the near absence of their prey species during that period (Patrician and Kenney 2010). Models have been developed that predict the developmental and reproductive rates of *Calanus finmarchicus* based on satellite-based measurements of sea surface temperature and chlorophyll and, from those, predict the arrival and abundance of right whales on their feeding grounds (Pershing et al. 2009; Record et al. 2008).

Right whale northerly migration patterns along the U.S. mid-Atlantic coast were modeled by Firestone et al. (2008). Regression analysis suggested an early to mid-March departure from Jacksonville, Florida, with a 30 day departure range, travel times of 21-24 days to the tip of Long Island, at a mean travel rate of 1.5-1.7 knots (2.8-3.1 km/h). Right whale habitat use in its migratory corridor is discussed by Schick et al (2009). These authors found that the range of habitat suitable for right whales exceeds previous estimates and argue for increased speed restriction buffer zones to protect more habitat.

Sighting distribution in the southeast U.S. right whale calving habitat was overlapped with sea surface temperatures by Keller et al. (2006a). Results suggested that the warm Gulf Stream waters may represent a thermal limit for right whales and within the area westward of the Gulf Stream, sea surface temperature (SST) greatly influenced whale distribution. Garrison (2007a) applied a similar analysis using bathymetry, modeled average wind data, and several spatial variables as well as satellite-derived sea surface temperature in a Generalized Additive Modeling approach. Peak sighting rates were found to occur at water depths between 10 and 20 m and temperatures between 13 and 15°C.

North Atlantic right whale habitat is likely vulnerable to climate change. Climate-driven changes in North Atlantic circulation patterns affect the composition and relative abundance of plankton in the feeding grounds of the Gulf of Maine and Western Scotian Shelf (Greene et al. 2003). Because calving rates appear grossly associated with food availability (Greene et al. 2003), climate change and circulation pattern fluctuations may be key factors in the reproductive success of the North Atlantic right whale population. Calving success was compared with indices of three major currents in the North Atlantic: the North Atlantic Oscillation, the Gulf Stream and the Southern Oscillation; the conclusion was that all three atmospheric cycles can be correlated with right whale reproduction rates (Kenney 2007).

2.1.5 Stock Structure

The split of *Eubalaena* into three species is supported by genetic work done by Rosenbaum et al. (2000) and by Gaines et al. (2005). Although genetic variability is low in the North Atlantic population, bottleneck analyses suggest that the primary loss of genetic diversity did not occur during the most recent population decline attributed to whaling during the 18th through the 20th centuries (Waldick et al. 2002). Microsatellite genetic analysis performed on the single right whale specimen found at Red Bay, Labrador, a 16th and 17th century Basque whaling site, supported the finding that low levels of right whale genetic diversity pre-date human exploitation (McLeod 2008; McLeod et al. 2010). Lower levels of genetic diversity were found in North Atlantic right whales compared with South Atlantic right whales (*Eubalaena australis*), and the divergence between the two species likely happened between 3 and 5 million years ago (Malik et al. 2000). Malik et al. (1999) identified five mitochondrial control region haplotypes in the North Atlantic right whale with significant genetic structuring. Using mtDNA analysis, Schaeff et al. (1992), suggested a division of the right whale population into two subgroups, defined by their use the Bay of Fundy as a nursery area. One of three mtDNA composite restriction morphs determined for 150 animals was not found in reproductive females which brought calves to the Bay of Fundy. Mcleod and White (2010) demonstrated evidence of heteroplasmy in the population resulting in a sixth haplotype.

2.1.6 Life History Traits

Kraus et al. (2001) assessed North Atlantic right whale reproductive parameters for the period 1980 through 1998. The number of calves expressed as a proportion of the total estimated population was estimated to be between 0.36 and 0.49, and the mean value for calves per mature female per year was 0.25. The mean age at first calving was 9.53 years, and the mean number of females recruited annually was 3.8. Calving intervals were found to have increased over the period, and calves per female decreased. In a more recent analysis, reproductive histories of whales in the right whale catalogue cover periods of up to 31 years and calving intervals seem to have decreased since the earlier report (Kraus et al. 2007).

Frasier et al. (2007) assessed male reproductive success in the North Atlantic right whale population. An uneven distribution of paternities was found, resulting in a reduced effective population size. There was a significant bias towards older males among assigned fathers, with average age at first paternity of approximately 15 years, an indication that mate competition could be preventing younger males from reproducing.

Right whale calf and perinatal mortality was examined by Browning et al. (2010). In the period between 1989 and 2003, 17 calves from 208 calving events were documented or presumed dead due to serious injury or disappearance from the sighting record. An additional 28 potential perinatal losses were presumed when female right whales were seen on the calving grounds without a calf.

Hlista et al. (2009) found a significant correlation between the number of right whale calves born and sea surface chlorophyll concentration averaged over the prior 2 years, supporting the idea that food availability during and just before the gestation period may be an important factor in the regulation of reproductive success. Calving intervals have varied from 3.5 years in 1990 to over 5 years between 1998 and 2003, and to just over 3 years in 2004 and 2005 (Kraus et al. 2007).

Analysis of fecal hormone metabolite levels has been demonstrated to be useful in determination of gender, detection of pregnancy and lactation, and in assessing age at sexual maturity (Rolland et al. 2007, 2005). Hunt et al. (2006) examined fecal glucocorticoids and concluded that such analysis could be a useful measure of adrenal activity and reproductive condition. Mating behavior was studied by Kraus and Hatch (2001). Most mating groups observed were composed of a single female and multiple, competing males.

Although work by Knowlton et al. (1994) indicated a slow rate of population recovery (ratio of first-year calves to the total non-calf population was estimated at 4.5%, population growth rate at 2.5% and mortality rate at 2.0% for the period 1980-1992), survival probability estimated by Caswell et al. (1999), showed that the population could face extinction within 191 years. Decreasing crude survival and decreasing reproductive rates were reported for the period 1980-1994. A workshop which focused on the causes of reproductive failure was convened in 2000 in Falmouth, Massachusetts (Reeves et al. 2001). The workshop concluded that if calf production and recruitment did not recover from the current levels, the population would be unlikely to recover. More recent calculations (Waring et al. 2011), suggest a positive trend, with a mean growth rate of 2.4%. A conservatively constructed population viability analysis (PVA) based on updated reproductive rates and survival estimates found none of 1000 simulated projected populations to end with a lower population than it began, strongly refuting the Fugiwara and Caswell (2001) analysis (Pace, personal communication).

2.1.7 Food Habits

The small-scale abundance of copepods near feeding right whales in the Great South Channel was measured by Beardsley et al. (1996). The whales were feeding on the copepod species

Calanus finmarchicus which was most abundant in the upper 10-20 m of the water column at concentrations measuring up to 3.3 x10^5 copepods per cubic meter. Total water column copepod biomass and *Calanus* biomass, measured by plankton tows at sites near feeding right whales, did not always differ from sites with no whales, but higher densities and relative proportions of older copepod lifestages were found close to right whales, leading researchers to suggest that the right whales were seeking out aggregations of older copepod lifestages (Wishner et al. 1995). The relationship between right whale sighting rates and late-stage *Calanus* was further explored by Baumgartner et al. (2003), who looked at the diel and tidal time-scale cycles of plankton availability. Baumgartner et al. (2011) investigated the effects of the diel vertical migration of *C. finmarchicus* on the occurrence of North Atlantic right whales and sei whales during spring in the southwestern Gulf of Maine. They found occurrence of right whales to be unrelated to variability in the migration behavior of *C. finmarchicus.* Dive patterns, however, measured by Winn et al. (1995) in a tagging study performed on right whales in the Great South Channel in 1988 and 1989, showed that the whales' diving behavior was closely correlated with the horizontal and vertical movements of the zooplankton on which they were feeding. Mayo and Marx (1990) observed that right whale surface feeding was rarely observed in Cape Cod Bay when zooplankton densities were less than 1,000 organisms/ m^3. Ingestion and defecation models were used to estimate that on average, a right whale ingests approximately 58,000 grams of wax ester lipids per day and defecates only 250 g of the wax esters, a rate of metabolism unusual for mammalian species (Swaim et al. 2009).

Kenney et al. (2001) proposed a conceptual model of right whale migratory and foraging strategies at varying scales, and presented a variety of speculations concerning the mechanisms involved. Pendleton et al. (2009) used aerial sighting data of right whales and measurements of copepod concentrations from vessel-based oceanographic sampling in Cape Cod Bay and the Great South Channel to conclude that regional-scale average copepod concentration is a good indicator of right whale presence in both habitats, with *C. finmarchicus* playing the most important role in the Great South Channel and other copepods being important in Cape Cod Bay. Particle transport and retention patterns in Cape Cod Bay, affected by the coastal current and prevailing winds, have significant correlation with *C. finmarchicus* abundance and thus with right whale sightings (Jiang et al. 2007). As mentioned above, the absence of right whales from the Roseway Basin, off southeastern Nova Scotia, for the period 1993-1999 can be tied to low abundance of *C. finmarchicus* during that time (Patrician and Kenney 2010).

Diving profiles of feeding right whales in the Great South Channel showed whales regularly dove steeply to a depth between 80 and 175 m, and then held that depth for 5 to 14 minutes before resurfacing (Baumgartner and Mate 2003). The average depth of dive was strongly correlated with the densest concentrations *C. finmarchicus* and with the average depth of the bottom mixed layer's upper surface. Right whale dives were classified into three dive shape groups by Nowacek (2008); one type was apparently a feeding behavior, another type was not associated with foraging, and a third was associated with low-quality foraging where animals may be searching for new prey aggregations.

Lysiak et al. (2008) examined long term trends in right whale migration behavior, health, and diet as expressed by incremental stable isotope ratios from 25 right whale baleen plates. The baleen was found not only to be a record of whale distribution, but also potentially useful for documenting ecosystem-level environmental changes. Variation in isotopic composition between individual right whales suggests differential habitat use, and may point to the use of alternative summer feeding grounds such as the Labrador Sea (Summers et al. 2006).

2.1.8 Health

2.1.8.1 Strandings

Between 1970 and 2001, 54 right whale mortalities were reported between Florida and the Canadian Maritimes (Moore et al. 2005). Of those 54 whales, 30 (18 adults and juveniles and 12 calves) were examined. Trauma presumed to be a result of vessel collision was a significant finding in 10 cases, and cause of death as a result of fishery entanglements was identified in 4 cases. The proportion of mortalities that are detected is unknown. Twenty confirmed North Atlantic right whale mortalities were reported in the mortality and serious injury report for the period 2005-2009 (Henry et al. 2011). This figure includes anthropogenic and non-anthropogenic mortalities.

2.1.8.2 Contaminants

North Atlantic right whale fecal samples and zooplankton prey from the area where the whales were feeding have been examined for paralytic shellfish poisoning toxins (Doucette et al. 2006; Durbin et al. 2002). Discovery of these toxins in both feed and feces suggested that trophic transfer of marine algal toxins could be a contributing factor in the right whale populations' failure to recover. The occurrence of marine biotoxin domoic acid was also assessed in right whale fecal samples and zooplankton prey (Leandro et al. 2010). Exposure to the biotoxin was confirmed, though concentrations measured in feces and estimated ingestion levels were lower than published levels for other marine mammals.

Right whale skin biopsy tissue samples were tested for exposure to chromium (Chen et al. 2009; Wise et al. 2008). Chromium accumulations were found in mean concentrations of 7.1 μg/g and proved to be both cytotoxic and genotoxic to North Atlantic right whale cells.

Brominated flame retardants and organochlorine contaminants in right whale blubber samples were analyzed by Montie et al. (2010). Of the organochlorine pesticides present in the right whales, c-hexachlorocyclohexane was found at the highest concentration, while Dieldrin, *bis*(*p*-chlorophenyl)-1,1-dichloroethane (*p*,*p*′-DDE), and *bis*(*p*-chlorophenyl)-1,1,1-trichloroethane (*p*,*p*′-DDT) were found at lower concentrations. Of the polybrominated diphenyl ethers measured in right whales, BDE (bromodiphenyl ether) 47 was found at the highest concentration. In the five right whales tested, 2,3,4,5,6-pentabromoethylbenzene (PBEB) was detected in four, hexabromobenzene (HBB) was detected in two, and pentabromotoluene (PBT) in four.

Organochlorines in North Atlantic right whale skin, feces, and prey were measured by Weisbrod et al (2000). Concentrations of 30 PCBs (polychlorinated biphenyl) and 20 pesticides in skin biopsies were consistent with other balenopterids. Patterns of contaminant burdens in blubber indicated that ingestion of different prey or prey from different areas and the release of stored organochlorines during lipid depletion in winter can cause the contaminant load to change annually.

Klansjcek et al. (2007) developed a dynamic energy budget model for marine mammals to demonstrate the relationships between energy intake and energetics and pharmacokinetics of environmental toxins. Using model parameters for the right whale, energy assimilation estimates were developed and the influences of energy availability on reproduction elucidated.

2.1.8.3 Disease

Fecal samples from 49 North Atlantic right whales were analyzed for *Cryptosporidium* spp. and *Giardia* spp. by Hughes-Hanks et al. (2005). Prevalence of *Cryptosporidium* was found to be 24.5% and prevalence of *Giardia* to be 71.4%. Techniques for making health assessments from photographs of right whales have been developed (Pettis et al. 2004). Photographic evidence of skin lesions on right whales was examined by Hamilton and Marx (2005). Two main types of lesions were identified: white lesions and blister lesions. The white lesions were detected primarily in the Bay of Fundy and peaked in occurrence in the late 1990s. Blister lesions appear to be a more chronic condition affecting a small percent of the population. The parasitic sea lamprey *Petromyzon marinus* is known to use right whales as hosts, with 35 photo-documented cases reported in the Bay of Fundy and Cape Cod Bay (Nichols and Hamilton 2004). It is not known what effect lamprey attachment has on right whales.

2.1.9 Acoustics

Morphometric analyses of inner ears from 13 stranded North Atlantic right whales were used for the development of a model of the frequency range of hearing this species (Parks et al. 2007).

Sound production on surface active groups of North Atlantic right whales was studied by Parks and Tyack (2005). The most common call recorded during this behavior was the scream call produced by the focal female. Other sounds recorded included gunshot and upcalls produced by males in the groups, and warble calls produced by female calves. Rates of right whale vocalizations from recordings made in 1999 and 2000 in the Great South Channel in spring and in the Bay of Fundy in summer were analyzed by Matthews et al. (2001). Calls of the moan type were correlated with whale group size and proximity to the surface. Autonomous hydrophone arrays collected right whale acoustical data from the Scotian Shelf from July 2004 to August 2005 (Mellinger et al. 2007). These data demonstrated seasonal and diel patterns of right whale calling activity, with most calls recorded in June through December.

The autonomous moored buoy acoustic array system for automatic detection of right whale calls is described in Spaulding et al. (2009). The system continuously monitors the shipping lanes off Boston to meet ship strike mitigation requirements.

Ambient noise levels and right whale upcall parameters were measured in three right whale habitat areas by Parks et al. (2009). Call parameters varied between habitats and between years within the same habitat area. However, there was evidence that right whales may be responding to noise frequencies in their environment, rather than overall levels. Increases in call amplitude by individual right whales was documented during periods of increased background noise (Parks et al. 2010), as were higher average frequencies and lower rates of call behavior (Parks et al. 2008). Brito et al. (2008), however, did not find increases in either vocal intensity or frequency shifts in North Atlantic right whale response to acoustic masking. A functional definition of communication masking and a metric to quantify its potential effects on marine mammals was presented by Clark et al. (2009). Spatial, spectral, and temporal maps of potential communication space were calculated for singing fin, singing humpback, and calling right whales. Communication masking by ship noise was found to be more severe for calling right whales.

Distortion of right whale acoustic signals by the shallow water environment of Cape Cod Bay was examined by Mohammed et al. (2008). Techniques for detecting right whale calls in the presence of ambient noise are discussed by Urazghildiiev (2009). Nonlinearities, including subharmonics, deterministic chaos, biphonation, and frequency jumps were analyzed in right whale and killer whale vocalizations by Tyson et al. (2007). The deterministic chaos type of nonlinear phenomena was detected in 87% of the Digital Acoustic Recording Tag (DTag) recordings of right whales in the Bay of Fundy analyzed in the study.

Visual aerial survey right whale sightings data were compared with acoustic detection data in Cape Cod Bay, Massachusetts (Clark et al. 2010). It was found that aerial surveys saw whales on only two-thirds of the days during which whales were detected acoustically.

2.1.10 Fisheries By-catch and Entanglement

Though ship strikes are more often immediately fatal, mortality by entanglement in fishing gear is a pervasive problem for many species of whales, and is less likely to be detected and reported, so estimations of fishery mortality based on observed mortalities are likely underestimations (Knowlton and Kraus 2001). Photographic analysis reveals that over 75% of well-photographed whales display entanglement scarring (Knowlton et al. 2005).

Pot and gillnet gear have been identified as the most common gear types involved in interactions with North Atlantic right whales (Johnson et al. 2005). Lobster trawl groundlines have been regulated for the protection of right whales since 1997. As of 1997, sinking groundlines were required in critical habitat areas between January and mid-May. In 2003, this restriction was imposed year-round in Cape Cod Bay critical habitat. In 2004, the restriction was extended to all waters west of the Cape Cod Bay Critical Habitat and south to 42°5´. Lobster pot groundlines in the Bay of Fundy were found to be generally lower than the 3 m elevation which has been hypothesized to be a threat to right whales (Brilliant and Trippel 2009). The pot fishery for black sea bass (*Centropristis striata*) has been shown to overlap in time and space with southeastern calving grounds (Levesque 2009a). Gillnet fishing was prohibited in the U.S. Southeast during the right whale calving season after the entanglement and death of a right whale calf in 2006 (Levesque 2009b).

2.1.11 Vessel Interactions

The slow-moving, surface-feeding and coastal habits of the right whale make it especially susceptible to threats from vessel traffic. Vessel traffic patterns within right whale habitat were characterized by Ward-Geiger et al. (2005). The authors found that 69% of vessel tracks in the Northeast transited right whale critical habitat and all but two vessel tracks in the Southeast transited critical habitat. High-use ship traffic corridors were also identified. The co-occurrence of whales and vessels in the southeastern U.S. right whale wintering habitat was modeled by Fonnesbeck et al. (2008), with estimates of risk associated with several hypothetical alternative shipping routes.

Threats and risks to right whales from vessel strikes and fishing gear have been modeled and quantified (Nichols and Kite-Powell 2005; Kite-Powell et al. 2007; Vanderlaan et al. 2009; 2010). Calculations that probability of ship strike lethality can drop to below 0.5% at vessel speeds below 12 knots, but increases to 100% when speeds are above 15 knots, contributed to the design and implementation of speed rule amendments in Canada.

To help protect right whales from vessel collisions, several regulatory measures have been enacted along the U.S. Atlantic seaboard. In 2006, NOAA established a set of recommended vessel routes through right whale habitat in Florida, Georgia, and Massachusetts. Seasonal management areas were established by NMFS in 2008 (73 Federal Register 60173, October 2008; Merrick 2005; Silber and Bettridge 2010). Under seasonal management area regulations, vessels 65 feet or longer must reduce their speed to 10 knots or less within these designated areas. On June 1, 2009, the north-south lanes of the Boston Traffic Separation Scheme were narrowed from 2 miles each to 1.5 miles each. It had been calculated that such a narrowing could reduce the shipping risk to right whales by 11% (Merrick and Cole 2007). Compliance rates with voluntary and mandatory speed restrictions in the southeast U.S. right whale critical habitat were measured by Lagueux et al. (2011), who found compliance rates on the mandatory measures to average 75% and on the voluntary restrictions to average 16%.

NMFS and other Federal and State agencies conduct extensive aerial surveys for right whales and issue alerts to mariners when right whales are spotted. Mandatory ship reporting systems have also been enacted, whereby ships over 300 gross tons are required to report to a shore station when they enter right whale habitats in the Southeast and in New England (Bettridge and Silber 2008).

In Jensen and Silber's database of large whale ship strikes worldwide in the period 1975 to 2002, there are 38 records of strikes involving North Atlantic right whales (Jensen and Silber 2003). Ten North Atlantic right whales are reported in NMFS records for the period 2004 through 2008 with sufficient information to confirm the cause of death as collisions with vessels (Glass et al. 2010). The resulting annual rate of serious injury or mortality was 2.0 right whales from vessel collisions.

2.1.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise by seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear. Low levels of localized avoidance have been observed in mysticetes exposed to seismic airgun noise in the UK and adjacent waters investigated 1997-2000 (Stone 2003; Stone and Tasker 2006). Work by McCauley et al. 2000 indicated that displacement of baleen whales by seismic activity could have serious effects on the animals and populations. Baleen whale reactions to low-frequency noise were also investigated by Croll et al. (2001), who found no obvious responses at the spatial and temporal scales examined, but cautioned that cumulative effects may be detrimental.

Shipping activities connected with energy projects increase risks to whales in the area, particularly to the strike-prone right whale. In a requirement to fulfill permitting requirements for the Northeast Gateway, a liquefied natural gas offloading facility, Excelerate Energy has been working with EOM Offshore to maintain a fleet of acoustic moorings near the Boston shipping lanes (http://www.eomoffshore.com/excelerate.php). These moorings continuously detect right whale sounds and have the capability to transmit data in real time directly to shore. Back on shore, at Cornell University, the clips are then confirmed and the Coast Guard is informed to make any necessary changes to shipping traffic and/or speed in hopes of reducing ship-whale collisions.

2.1.13 Data Gaps and Research Recommendations

Despite years of intensive study and many large-scaled survey efforts, there are modest gaps in our knowledge of North Atlantic right whale seasonal distribution. Variation by individual whales is quite high in the use patterns of the five best known habitats (southeastern U.S., Cape Cod Bay, Great South Channel/northern edge Georges Bank, Roseway Basin and the Bay of Fundy). Some reproductive females have been seen only in the southeastern U.S. or the southeast and the Great South Channel, so their principal summer and fall feeding grounds are unknown. Cause-specific mortality for large whales may be largely unknowable from the current survey approaches; however, estimating the fraction of the North Atlantic right whale population that annually succumbs to ship strike and entanglement-related mortality is of great importance, because a considerable U.S. government and private investment go into the reduction of these factors of unknown scale.

Research priorities identified in the right whale recovery plan (National Marine Fisheries Service 2005) include reducing the sources of human-caused death, injury and disturbance; developing demographically-based recovery criteria; to identify, characterize, protect and monitor important habitats; to monitor the status and trends of abundance and distribution of the western North Atlantic right whale population; and to coordinate Federal, State, local, international and private efforts to implement the recovery plan.

2.2 BLUE WHALE (*BALAENOPTERA MUSCULUS*)

2.2.1 Legal Status

The blue whale is listed as endangered throughout its range under the ESA and as depleted under the MMPA. The species has been protected from commercial whaling since 1966 under the International Convention for the Regulation of Whaling (Office of Protected Resources–NOAA Fisheries n.d.). The North Atlantic stock of *B. musculus musculus* is listed as endangered by the International Union for Conservation of Nature (IUCN) Red List of Threatened Species (Reilly et al. 2008a).

2.2.2 General Distribution

The blue whale is found in all oceans except the high Arctic (Reilly et al. 2008a). In the North Atlantic, blue whales are most frequently sighted from Iceland to West Greenland to waters off eastern Canada, especially in the Gulf of St. Lawrence (Sears 2002; Waring et al. 2011). Blue whales have been acoustically detected in deep waters east of the U.S. Exclusive Economic Zone (EEZ) (Clark 1995). 18th and 19th century whaling logbooks recorded blue whale sightings in U.S. waters off the mid-Atlantic and southeast coast (Reeves et al. 2004), but recent sightings in the U.S. EEZ are infrequent. Occasional sightings of blue whales have been made off Cape Cod, Massachusetts, in summer and fall (Wenzel et al. 1988). Sighting locations of blue whales are shown in Figure 2.2-1.

2.2.3 General Abundance

The global blue whale population size is uncertain but is thought to be between 10,000 and 25,000 animals (Reilly et al. 2008a). The western North Atlantic population is estimated to be in the low hundreds (Sears and Calambokidis 2002), based on photo-ID work done in the Gulf of St. Lawrence (Sears et al. 1990).

2.2.4 Habitat Preference

Coastal and pelagic waters are used by blue whales, and feeding aggregations are found primarily at the shelf edge (Sears and Calambokidis 2002). Acoustic detections of blue whales in the North Pacific were correlated with areas of sea surface temperature fronts and sea surface height elevations that indicated areas of upwelling (Moore et al. 2002). Those two variables appeared to be better indicators of good blue whale habitat than chlorophyll-a measurements. Stafford et al. (2009) also examined blue whale call detection rates in the North Pacific with respect to sea surface temperature, chlorophyll-a concentration, and mixed layer depth, finding that sea surface temperature was the best predictor of whale call detections. Distributions of blue whales were highly correlated with sea surface temperature fronts in the Gulf of St. Lawrence (Doniol-Valcroze et al. 2007).

2.2.5 Stock Structure

Blue whales in the North Atlantic belong to the subspecies *Balaenoptera musculus musculus*. Stock structure has not been defined, and while the species is sometimes considered to be divided into western and eastern North Atlantic stocks, acoustic tracking of blue whales ranging widely in the ocean basin (Clark 1995) supports a hypotheses of a single panmictic stock (National Marine Fisheries Service 1998).

2.2.6 Life History Traits

The blue whale gestation period is approximately 10-12 months, and calves are nursed for about 6-7 months (National Marine Fisheries Service 1998). Blue whales are thought to become sexually mature at 5-15 years of age and the calving interval is probably 2 to 3 years (National Marine Fisheries Service 1998).

Ramp et al. (2006) used sighting histories of photo-identified blue whales in the Gulf of St. Lawrence to estimate the adult blue whale survival rate to be 0.975 (95% CI 0.960 to 0.985).

The sighting histories also showed that while some blue whales have high site fidelity to the St. Lawrence area, others were only occasionally sighted there.

2.2.7 Food Habits

Blue whales feed largely on krill (euphausiids). In the Gulf of St. Lawrence, blue whales feed on two main euphausiid species, *Thysanoëssa racshii* and *Meganyctiphanes norvegica* (Sears et al. 1987).

2.2.8 Health

2.2.8.1 Strandings

Blue whales strand infrequently on the U.S. east coast, and no records of strandings exist in recent response organization databases. An 1891 stranding at Ocean City, Maryland, is the southernmost confirmed record on the U.S. east coast (National Marine Fisheries Service 1998).

2.2.8.2 Contaminants

Organochlorines contamination in blubber samples of blue whales in the St. Lawrence has been analyzed and compared to that in other balaenopterids in the area (Gauthier et al. 1997; Metcalfe et al. 2004). In blue whales, concentrations of some of the more persistent contaminants were present at higher concentrations in the blubber of males relative to females; indicating maternal transfer from females into their offspring (Metcalfe et al. 2004).

2.2.8.3 Disease

Crassicaudosis is a potentially lethal endemic disease caused by infection by the giant nematode *Crassicauda boopis*. It can affect blue, fin, and humpback whales, and occurs worldwide (Lambertsen 1992). Several types of skin lesions have been reported on blue whales seen off Chile (Brownell, Jr. et al. 2007).

2.2.9 Acoustics

Blue whale song can be divided into at least ten geographically distinct types, with one type unique to the North Atlantic (McDonald et al. 2006, 2009). Typically, blue whale vocalizations consist of repeated sequences of simple combinations of long-duration, very-low-frequency (15-20 Hz) sound units repeated every minute or two. The specific frequency, duration, and repetition intervals produced by blue whales in the North Atlantic, however, can be distinguished from blue whale recordings from other regions (Mellinger and Clark 2003). These song types retain consistent phrasing over time, although research by McDonald et al. (2009) demonstrated that all seven song types for which multiple years of data are available have shifted perceptibly lower in frequency. The mean frequency of the North Atlantic song type has declined from 23 Hz in 1959 to 17.6 Hz in 2004 (McDonald et al. 2009). The authors theorized that the frequency shifts may be related to changes in population densities as the blue whale populations recover from whaling.

Blue whales have been acoustically detected in the North Atlantic using the U.S. Navy underwater hydrophone arrays (Clark 1995; Mellinger and Clark 2003). Diel variation in blue whale calling was demonstrated in the eastern tropical Pacific (Stafford et al. 2005). The diel patterns of call production were correlated with patterns of prey distribution. Efforts have been made in waters to the north and west of Iceland to relate visual observations of blue and fin whales with continuous acoustic monitoring with the aim of further developing acoustic detection techniques for these species (Boisseau et al. 2008).

In the North Pacific, blue and fin whale call detections were correlated with sea-surface temperature, chlorophyll-a concentration and mixed layer depth (Stafford et al. 2009). Sea-surface temperature proved to be the best oceanographic variable for predicting call detections for both species.

2.2.10 Fisheries By-catch and Entanglement

There have been several documented cases of blue whale entanglements in fishing gear: one in 1987 off Stellwagen Bank, north of Cape Cod, Massachusetts, and at least three in the Gulf of St. Lawrence (National Marine Fisheries Service 1998; Sears and Calambokidis 2002).

2.2.11 Vessel Interactions

A blue whale ship strike mortality was reported in 1998 when a dead juvenile male was brought into Rhode Island waters on the bow of a tanker (National Marine Fisheries Service 1998; Waring et al. 2011). At least 9% of blue whales in the Gulf of St. Lawrence catalog show signs of ship strike scarring (Sears et al. 1990). Acoustic masking by vessel traffic in the Santa Barbara Channel off California, important blue whale habitat, was investigated by McKenna et al. (2009).

2.2.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise by seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear. Di Iorio et al. (2009) investigated blue whale calling behavior relative to seismic operations in the St. Lawrence Estuary. An increase in vocal behavior was observed, presumably in compensation for the masking effects of the seismic noise.

2.2.13 Data Gaps and Research Recommendations

The rarity of blue whale sightings in the U.S. Atlantic EEZ makes assumptions about the whales' distribution and habitat use in this area difficult. More data collection using acoustic detections could supplement the sightings data. More genetic analyses are necessary to better understand stock delineation.

2.3 FIN WHALE (*BALAENOPTERA PHYSALUS*)

2.3.1 Legal Status

The fin whale is classified as endangered under the ESA and depleted under the MMPA throughout its range. The species is classified in the “endangered” category under the IUCN (Reilly et al. 2008c).

2.3.2 General Distribution

Fin whales are widely distributed in all major oceans (National Marine Fisheries Service 2010a). In the North Atlantic, the fin whale can be found from the Gulf of Mexico to the edge of the Arctic pack ice (this range includes all BOEM North Atlantic planning areas). Sightings are more common north of 20° N (BOEM South Atlantic, Mid-Atlantic and North Atlantic planning areas) (National Marine Fisheries Service 2010a). Sighting, stranding and fishery bycatch locations of fin whales are shown in Figure 2.3-1.

Fidelity to feeding grounds and long-distance movement have been demonstrated by tagging and photo-identification studies (Edds and Macfarlane 1987; Agler et al. 1990; Seipt et al. 1990; Clapham and Seipt 1991; Agler et al. 1993; Coakes et al. 2005; Mizroch et al. 2009).

2.3.3 General Abundance

Worldwide, fin whale abundance is estimated to be in the tens of thousands (National Marine Fisheries Service 2010a). Hain et al. (1992) estimated a population size of about 5,000 fin whales in northeastern United States continental shelf waters in the spring and summer of 1978-1982, and approximately 1,500 in the fall and winter seasons. For the Western North Atlantic stock of fin whales, the best recent abundance estimate is 3,985 (CV=0.24). This is the sum of the estimate derived from an August 2006 Gulf of Maine survey and an estimate derived from a July-August 2007 northern Labrador to Scotian Shelf survey and includes a percentage of the estimate of animals identified as fin or sei whales (Waring et al. 2011). This does not include a correction for dive time bias.

2.3.4 Habitat Preference

Important habitat in the U.S. waters was described by Hain et al. (1992) as extending from the Great South Channel along the 50-m isobath past Cape Cod and over Stellwagen Bank, and northeast to Jeffreys Ledge. The mid-shelf region of the mid-Atlantic bight and the eastern edge of Georges Bank were also mentioned as important areas. Hamazaki’s cetacean habitat prediction model classifies the fin whale as a North Atlantic shelf species (Hamazaki 2002). Summer sightings were associated with mean water temperatures of 18.2°C, mean depths of 337 m, mean bottom slope of 1.1° and front probability of 6.3%.

In the Gulf of St. Lawrence, in an analysis of fin whale distribution during summer and autumn of 1989, sightings were found to occur at a mean depth of 128.7 m and mean surface temperature of 12.2°C (Woodley and Gaskin 1996). Areas with high topographic variation seemed to be favored. Echosounder measurements indicated that herring (*Clupea harengus*), and euphausiids were relatively abundant in areas where fin whales were found. Near Grand Manan Island, in the Bay of Fundy, Canada, fin and minke whales were observed to forage in the eddy region of an island wake, where they likely were able to exploit higher concentrations of prey (Johnston et al. 2005b).

In contrast to the shallow depths inhabited by fin whales in the Gulf of St. Lawrence, a study that correlated fin whale sightings with environmental variables in the Mediterranean indicated that over 90% of fin whale sights occurred in water depths of over 2,000 m, with water depth and distance from shore being the most significant variables describing fin whale distribution (Panigada et al. 2006, 2008). In the Mediterranean, fin whales were found in cold waters with steep bathymetry and temperature gradients (Cotté et al. 2009), and a predictive model was developed (Laran and Gannier 2008). Off Kodiak, Alaska, in 2002 and 2003, fin whales were found associated with deeper, cooler waters near areas of maximum slope, though association with shallow waters was noted for 2002 and not 2003 (Baraff 2006).

2.3.5 Stock Structure

The International Whaling Commission (IWC) defined seven management units for fin whales in the North Atlantic: Nova Scotia, Newfoundland-Labrador, West Greenland, East Greenland-Iceland, North Norway, West Norway-Faroe Islands, and British Isles-Spain-Portugal (Donovan 1991). Significant mtDNA heterogeneity has been found in North Atlantic fin whales between the Mediterranean Sea, the eastern (Spain) and the western (Gulf of Maine and Gulf of St. Lawrence) summer feeding areas (Bérubé et al. 1998). Variations in fin whale vocalizations have also indicated that Gulf of St. Lawrence fin whales are distinct from Gulf of Maine fin whales (Delarue et al. 2009). Fin whales off the U.S. Atlantic coast are presumed to be from a single stock, though stock substructure is unknown (Waring et al. 2011).

2.3.6 Life History Traits

From stranding records of neonates and calves, Hain et al. (1992) surmised that fin whale calving in U.S. waters takes place primarily between October and January at the latitude of the mid-Atlantic Bight. Annual rates of reproduction for fin whales in the Gulf of Maine in the 1980s were calculated to range from 0.03 to 0.12 (Agler et al. 1993). The birth rate was estimated at 0.37 young per mature female per year and a potential mean interval of birthing was estimated at 2.24 years (Agler et al. 1993). By the age of 30, most female fin whales caught off the coast of Iceland had begun reproductive senescence (Kjeld et al. 2006).

2.3.7 Food Habits

In the North Atlantic, fin whales are known to feed on pelagic crustaceans (mainly euphausiids or krill, including *Meganyctiphanes norvegica* and *Thysanoessa inermis*) and schooling fish such as capelin (*Mallotus villosus*), herring (*Clupea harengus*), and sand lance (*Ammodytes* spp.) (Mitchell 1974; Overholtz and Nicolas 1979; National Marine Fisheries Service 2010a). Fluctuations in the availability of prey, especially sand lance, are thought to have had a strong influence on the distribution and movements of fin whales in shelf waters of the northeastern U.S. (Kenney and Winn 1986; Payne et al. 1990; Hain et al. 1992). The diets of fin whales and humpback whales in the Gulf of St. Lawrence were compared using blubber fatty acid analysis (Borobia et al. 1995). The chemical and isotopic differences found indicated that humpbacks fed at a slightly lower trophic level than fin whales.

High-resolution digital tags attached to fin whales in the Southern California Bight were used to study the biomechanics of the lunge-feeding behavior (Goldbogen et al. 2006). The whales were found to glide during descent, execute a series of lunges at depth and then ascend to the surface by steady fluking. Larger fin whales have larger skulls and buccal cavities relative to body size, allowing for larger engulfment volumes during lunge-feeding (Goldbogen et al. 2010). A hydrodynamic model of fin whale lunge feeding was developed by Potvin et al. (2009) to test whether the mechanics of engulfment were passive or involved muscle action. Results suggested that adult rorquals actively push engulfed water forward from the onset of mouth opening, which involves a reflux of the engulfed water.

2.3.8 Health

2.3.8.1 Strandings

Hain et al. (1992) reported a total of 72 fin whale strandings along the U.S. Atlantic coast "so far this century." Strandings were relatively evenly distributed by month; however, a majority of the strandings occurred on Cape Cod, Massachusetts, or on Cape Hatteras, North Carolina. Thirty-four confirmed fin whale mortalities were reported in the mortality and serious injury report for the period 2005-2009 (Henry et al. 2011). This figure includes anthropogenic and non-anthropogenic mortalities.

2.3.8.2 Contaminants

Aguilar and Borrell (1994) estimated body loads of organochlorine pollutants in 169 fin whales caught off the northwestern coast of Spain. They found blubber to be the greatest reservoir for contaminants, although compared to other cetaceans its relative contribution to the total load was lower. Organochlorine body loads of males increased with age but leveled off once the whale was fully grown. In adult females, loads decreased with age due to fetal transfer but the transfer was lower than found in other cetaceans because of the shorter lactation period of fin whales. Blubber samples taken from minke whales, fin whales, blue whales, and humpback whales in the Gulf of St. Lawrence, Quebec in summer and fall of 1991 and 1992 were tested for chlorinated biphenyls and other persistent organochlorine compounds (Gauthier et al. 1997). Ratios of proportions of oxychlordane to trans-nonachlor were highest in fin whales. The fin whale was one of four marine mammal species chosen for a comparative study on temporal and geographic variation in levels of organochlorine contaminants in marine mammal blubber (Aguilar et al. 2002). Fin whale samples from western North America and Europe—especially the Mediterranean—were highest in organochlorine loads compared to those from other areas. Concentrations overall had decreased in areas where pollution was initially high but had increased in areas further from the pollution sources.

Bioaccumulation of flouride was demonstrated in bone samples from North Atlantic fin whales (Landy et al. 1991). Concentrations were found to be higher than in other mammals, and increased with age. The fact that krill from the stomachs of the whales also tested high for fluoride pointed to diet as the source.

2.3.8.3 Disease

Necropsies of two female immature fin whales that had stranded in France and Belgium revealed massive parasitic infestation and positive tests for morbillivirus exposure (Jauniaux et al. 2000).

Eighty-seven fin whales caught by commercial whalers in Icelandic waters were necropsied to determine cause of death (Lambertsen 1986). The giant nematode *Crassicauda boopis* was found in 82 of the whales. Crassicauda infection was so prevalent and severe that it was deemed capable of causing the death of the animals. In a study of serum chemistry of fin whales caught by commercial whalers in the Denmark Strait in 1984 and 1985, serum electrolytes, urea nitrogen, creatine, albumin, and globulin were measured (Lambertsen et al. 1986). While chase time was found to have no substantive effect on serum chemistry, one of the whales did exhibit serum chemistry consistent with renal failure.

2.3.9 Acoustics

Southall et al. (2007) classified fin whales and other balaenopterids in the “low-frequency cetaceans” functional hearing group, with an estimated hearing range of approximately 7 Hz to 22 kHz. Fin whale sounds recorded by a stationary hydrophone in the St. Lawrence Estuary consisted of vocalizations with frequencies below 120 Hz and impulsive sounds with frequencies up to 1 kHz (Edds 1988). Downsweeping calls were the most common, with frequency variations that were correlated with social context. Constant calls, upsweeps, wavers and a frequency and amplitude modulated call were also recorded but were uncommon.

The “20-Hz” call, which had been associated with the reproductive season (Watkins et al. 1987), was, through a combination of towed hydrophone recordings and biopsy genetic sampling, discovered to be made exclusively by male fin whales in the Gulf of California, Mexico (Croll et al. 2002). This sound, which can reach intensities of 184-186 decibels relative to 1 μPa of sound pressure, is thought to attract females from great distances to prey aggregations. These calls have also been recorded by autonomous hydrophones near the mid-Atlantic ridge (Nieukirk et al. 2004).

Interpulse intervals in fin whale calls vary between geographic areas so can be used to elucidate stock structure. Fin whale songs from the Gulf of St. Lawrence differ significantly from fin whale songs recorded in the Gulf of Maine (Delarue et al. 2009). Changes in song notes and features of fin whale vocalizations were observed in a comparison of recordings from Cape Cod Bay in 1961, 1978, 2001 and 2005 (Koltz 2007). Variation in seasonal and yearly patterns may confound comparison of these recordings with those collected from a fin whale population in Bermuda.

In the North Pacific, blue and fin whale call detections were correlated with sea-surface temperature, chlorophyll-a concentration and mixed layer depth (Stafford et al. 2009). Sea-surface temperature proved to be the best oceanographic variable for predicting call detections for both species.

Between 2006 and 2009 sea floor recorders monitored noise levels and fin whale presence in the western Mediterranean and adjacent NE Atlantic waters (Castellote et al. 2009, 2010). This period included a 10-day seismic survey in the region. Fin whales were found to decrease temporal and spectral parameters of their vocalizations when ambient noise levels rose. Airgun noise from the seismic survey caused singing fin whales to move away from the study area within 24 hours and to stay out of the area until 14 days after the seismic activity had ceased.

Airgun noise appeared to exceed the fin whales' tolerance threshold even though received levels were less than 120 dB during the singing overlap period.

Analysis of fin whale calls recorded by bottom-moored acoustic recorders in Davis Strait, between Greenland and Canada, has resulted in new understandings of habitat use and migration of fin whales in that area (Simon et al. 2010). Fin whales were shown to be present in Davis Strait much later in the year than previously expected and peaks in song activity implied mating activity.

A method for estimating fin whale population density was described by McDonald and Fox (1999). From one bottom-fixed hydrophone north of Oahu, Hawaii, an average calling whale density of 0.027 animals/1000 km^2 was derived, with a seasonal maximum calling whale density of 0.081 animals/1000 km^2. Fin whale call types detected were "doublet 20-Hz" calls, "20- to 35-Hz irregular repetition interval" calls, and "30- to 90-Hz shorter and more irregular repetition intervals" calls.

2.3.10 Fisheries By-catch and Entanglement

Two Western North Atlantic stock fin whales are reported in NMFS records for 2005 through 2009 with sufficient information to confirm the cause of death as fishery interaction, and two with fishery interaction serious injuries, resulting in an annual rate of serious injury or mortality from fishery interactions of 0.8 fin whales (Waring et al. 2011). The threat posed by fishery entanglement to the Western North Atlantic stock of fin whales is characterized as "low" in the Recovery Plan (National Marine Fisheries Service 2010a).

2.3.11 Vessel Interactions

Fin whales were the most often reported species involved with ship strikes worldwide, with 75 records of strike (out of a total of 292) in the period 1975 to 2002 (Jensen and Silber 2003). Between 1993 and 2002 at least 15 fin whales were ship strike mortalities off the U.S. east coast (Jensen and Silber 2003). Nine fin whales were reported in NMFS records for the Western North Atlantic stock during 2005 through 2009 with sufficient information to confirm the cause of death as collisions with vessels (Henry et al. 2011). The resulting annual rate of serious injury or mortality from vessel collisions was 1.8 fin whales.

Fin whale respiration data collected from Mt. Desert Rock, Maine (land-based survey) between 1983 and 1986 were examined to determine whether proximity to whale-watching boats altered the fin whales' breathing patterns (Stone et al. 1988). The presence or absence of vessels was found to have no significant effect on respiration characteristics. A study on reactions of Mediterranean fin whales to small vessel approaches, however, found significant behavior changes (Jahoda et al. 2003). Suspension of feeding behavior, increased travel velocity and reduction of time spent at the surface were observed.

2.3.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise by seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear. MacLeod et al. (2006b) suggested that an absence of fin and sei whale sightings to the west of the Outer Hebrides may be correlated with increased seismic activity in the area. Low levels of localized avoidance were found in mysticetes exposed to seismic airgun noise in the UK and adjacent waters investigated in 1997-2000 (Stone and Tasker 2006). Avoidance behavior and changes in vocalization parameters were observed in fin whales in the Mediterranean when exposed to seismic activity (Castellote et al. 2009, 2010).

2.3.11 Data Gaps and Research Recommendations

The Fin Whale Recovery Plan (National Marine Fisheries Service 2010a) identifies key actions that should be undertaken to further understand and protect fin whales. Priorities include better understanding of fin whale population structure; improvement of abundance estimates and trends of abundance; improvement of knowledge of fin whale habitat and feeding ecology; research on the impacts of climate change on fin whales; steps to minimize fin whale mortality from anthropogenic sources such as ship strike, fishery interaction and pollution; steps to determine and minimize detrimental anthropogenic noise; and effective response to strandings of fin whales.

2.4 SEI WHALE (*BALAENOPTERA BOREALIS*)

2.4.1 Legal Status

The sei whale is classified as endangered under the ESA and depleted under the MMPA throughout its range. The species is classified in the “endangered” category under the IUCN (Reilly et al. 2008b).

2.4.2 General Distribution

Sei whales have a wide, but primarily offshore, distribution in the North Atlantic, the North Pacific and in the southern hemisphere (Reilly et al. 2008b). The subspecies *B. b. borealis* inhabits the northern hemisphere and the subspecies *B. b. schlegellii* is found in the southern hemisphere (Rice 1998). Seasonal migration patterns are described between tropical and sub-tropical breeding grounds and high-latitude feeding grounds (Reeves et al. 1998; Kanda et al. 2006; Reilly et al. 2008b; Horwood 2009). Eight sei whales satellite-tagged in the Azores have been tracked to the Labrador Sea (1 by Olsen et al. 2009; 9 by Prieto et al. 2010). Sighting and stranding locations of sei whales are shown in figure 2.4-1.

Off the U.S. Atlantic coast, sei whales are found in the deeper waters east and south of Georges Bank in the spring through fall, occasionally making incursions into shallower waters including the Great South Channel, Stellwagen Bank, and the southern Gulf of Maine (Payne et al. 1990; Schilling et al. 1992; Waring et al. 2011). Sightings are primarily in the BOEM North Atlantic Planning Area.

2.4.3 General Abundance

An estimate of 386 (CV=0.85) sei whales was derived from a summer 2004 aerial and shipboard survey conducted in waters from Maryland to the lower Bay of Fundy (Waring et al. 2011). No population estimate for the entire North Atlantic is available.

2.4.4 Habitat Preference

Analysis of satellite-tracked movements of eight sei whales tagged in the Azores was evidence that hydrological structures such as sea surface temperature fronts and eddies create good foraging habitat for sei whales in the Labrador Sea (Prieto et al. 2010). In the Mid-Atlantic ridge region, sei whale distribution was correlated with cross-seamount or cross-frontal structures (Skov et al. 2008). While sei whale aggregations have been reported associated with submarine canyons, Kenney and Winn (1987) found no differential sighting rates for sei whales in the northeastern U.S. shelf region between canyon areas and neighboring shelf/slope areas.

2.4.5 Stock Structure

There are two stocks of sei whales recognized in the western North Atlantic—the Labrador Sea stock, whose range extends from the southeast coast of Newfoundland northward, and the Nova Scotia stock, found south of Newfoundland to at least North Carolina (Reeves et al. 1998).

2.4.6 Life History Traits

Age at maturity is approximately 10 years (Horwood 2009). Most conceptions occur in December in the northern hemisphere, with young carried for almost a year and nursed for 7 months (Horwood 2009). Natural mortality rates are not well known, but are thought to be around 5-10% per year (Horwood 2009).

2.4.7 Food Habits

Sei whales use both skim and lunge-feeding techniques to feed on calanoid copepods and euphausiids in the North Atlantic (Horwood 2009). Shifts in abundance of copepods and of sand lance (*Ammodytes* spp.) during the 1980s were studied by Payne et al. (1990). Sei whales were only common in the Gulf of Maine study area when copepod abundance reached a regional maximum and sand lance abundance was at it regional lowest. Sei whale and right whale occurrence patterns were significantly related. Sei whales were shown to be significantly less abundant in the Gulf of Maine during periods when diel vertical migration patterns of *Calanus finmarchicus* were strong (Baumgartner et al. 2011).

2.4.8 Health

2.4.8.1 Strandings

Sei whales strand infrequently on the U.S. east coast. Ten confirmed sei whale mortalities were reported in the mortality and serious injury report for the period 2005-2009 (Henry et al. 2011). This figure includes anthropogenic and non-anthropogenic mortalities.

2.4.8.2 Contaminants

Annual changes in mercury in western North Pacific zooplankton, pelagic fishes, and baleen whales, including sei whales, were examined by Yasunaga et al. (2009). Variation in total Hg levels in sei whale muscle samples was shown to be insignificant.

O'Shea and Brownell (1994), in their review of studies of the effects of contaminants on baleen whale populations, concluded that there is no firm evidence for direct impacts on baleen whale populations from organochlorine and metal contamination.

2.4.8.3 Disease

Lambertsen (1990) found that eastern North Atlantic sei whales did not seem to be infected with crassicaudiasis to the same extent as fin whales. However, 92% of the examined sei whales (N=24) had invasive infections of *Bolbosoma* spp. in the colon and 18% showed an inflammation in the lungs that appeared consistent with viral or mycoplasma pathogen. The sei whales were often scarred with bite wounds and unidentified external lesions.

2.4.9 Acoustics

Southall et al. (2007) classified sei whales and other balaenopterids in the "low-frequency cetaceans" functional hearing group, with an estimated hearing range of approximately 7 Hz to 22 kHz.

Baumgartner and Frantantoni (2008) examined associations between sei whale vocalization rates, oceanographic conditions, and the vertical distribution of the whales' prey, *C. finmarchicus*, during May 2005 in the southwestern Gulf of Maine using an array of autonomous ocean gliders. The diel vertical migration of *C. finmarchicus* was confirmed using acoustic backscatter measurements. Sei whale vocalization rates showed a corresponding diel periodicity, with more calls detected during the daytime when *C. finmarchicus* was observed at depth. It was hypothesized that calling rates were reduced at night while the whales were feeding, but increased with social activity during the day when copepods are either more difficult or less efficient to capture at depth. Low frequency downsweep vocalizations recorded by the autonomous gliders, were subsequently confirmed to be produced by sei whales by visual survey combined with acoustic recorder array deployments (Baumgartner et al. 2008). The New Jersey Shallow Water 2006 experiment recorded a number of sei whale low-frequency downsweep chirps in mid-Atlantic continental shelf waters off New Jersey (Newhall et al. 2009). Two variations of low-frequency downsweeps, as well as 105 other vocalizations, have been attributed to sei whales near the Hawaiian Islands (Rankin and Barlow 2007b).

2.4.10 Fisheries By-catch and Entanglement

Three Nova Scotia stock sei whales were reported in NMFS records for 2005 through 2005 with fishery interaction serious injuries, resulting in an annual rate of serious injury or mortality from fishery interactions of 0.6 sei whales (Waring et al. 2011).

2.4.11 Vessel Interactions

Only 3 records of sei whales were reported involved with ship strikes worldwide (out of a total of 292) in the period 1975 to 2002 (Jensen and Silber 2003). Three sei whales were reported in

NMFS records for the Nova Scotia stock during 2005 through 2009 with sufficient information to confirm the cause of death as collisions with vessels (Waring et al. 2011). The resulting annual rate of serious injury or mortality from vessel collisions was 0.4 sei whales.

2.4.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise by seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). MacLeod et al. (2006b) suggested that an absence of fin and sei whale sightings to the west of the Outer Hebrides may be correlated with increased seismic activity in the area. Low levels of localized avoidance were found in mysticetes exposed to seismic airgun noise in the UK and adjacent waters investigated in 1997-2000 (Stone 2003; Stone and Tasker 2006).

2.4.13 Data Gaps and Research Recommendations

The Draft Recovery Plan for the fin whale and sei whale (Reeves et al. 1998) identified key actions that should be undertaken to further understand and protect sei whales. Priorities include better understanding of sei whale population structure; improvement of abundance estimates and trends of abundance; identification and protection of essential sei whale habitats; research on the impacts of climate change on sei whales; reduction of sei whale mortality from anthropogenic sources such as ship strike, fishery interaction and pollution; steps to determine and minimize detrimental anthropogenic noise; and effective response to strandings of sei whales.

2.5 MINKE WHALE (*BALAENOPTERA ACUTOROSTRATA*)

2.5.1 Legal Status

The minke whale is protected under the MMPA and is classified in the "Least Concern" category under the IUCN (Reilly et al. 2008d). The species is not listed under the ESA.

2.5.2 General Distribution

Minke whales occur in coastal and offshore waters of all oceans of the world, in latitudes from 65°S to 80°N (Reilly et al. 2008d). In the western North Atlantic, minke whales are common in summer north to Baffin Bay and south to New Jersey—in the BOEM North Atlantic planning area (Reilly et al. 2008d). A study of minke whale occurrence in Massachusetts Bay and Cape Cod Bay showed that minkes generally arrive in these waters in early spring and were most abundant in late summer (Murphy 1995). Sighting, stranding and fishery bycatch locations of minke whales are shown in Figure 2.5-1.

Satellite-tracking data collected from two minke whales off northern Norway showed that, while the whales travelled an average of 78 and 79 km per day, they remained resident in the general area while transmitting (31 days and 19 days, respectively) (Heide-Jorgensen et al. 2001). Small-scale site fidelity has also been shown off Scotland (Gill and Fairbairns 1995;

Baumgartner 2008) and off the Pacific Northwest coast of the U.S. (Dorsey 1983; Dorsey et al. 1990).

2.5.3 General Abundance

Worldwide, minke whales are thought to number approximately 182,000 (Reilly et al. 2008d). Minke whale abundance estimates for northeast Atlantic populations are reported by Skaug et al. (107,205 CV=0.14; 2004) and Bøthun et al. (108,000 CV=0.23; 2009). For the Canadian East Coast stock of the western Atlantic, the best recent abundance estimate is 8,987 (CV=0.32), which is the sum of an August 2006 U.S. survey (3,312 CV=0.74) and a July-August 2007 Canadian survey (5,675 CV=0.25) (Waring et al. 2011).

2.5.4 Habitat Preference

A 12-year sighting history of minke whales in the Mingan Islands (Canada) was analyzed with respect to bottom topography, geomorphology, and depth (Naud et al. 2003). Distribution of sightings was found to be positively correlated with sandy bottom substrates, a good habitat for sand lance (*Ammodytidae*), the whales' principal prey in that area. In the Moray Firth, off Scotland, areas of highest minke whale encounter frequency were characterized by shallow depths (10-14 m), steep slopes (70-74 degrees) and sandy gravel bottom substrate (Robinson and Tetley 2005; Robinson et al. 2009), though age-class differences were seen in distribution relative to bottom substrate (Robinson et al. 2009). A strong positive linear relationship between tidal speed and whale occurrence was also shown for this region (Baumgartner 2008). Seasonal changes in whale distribution were demonstrated to show response to shifting prey preferences (Macleod et al. 2004) as were seasonal and diel changes in dive intervals (Stockin et al. 2001). High densities of phytoplankton during warm water plume events in the Moray Firth were associated with higher minke whale sighting rates, presumably due to their effect in attracting minke whale prey (Tetley et al. 2008). On the basis of multiple logistic regression analysis which modeled sightings of minke whales in the western North Atlantic with oceanographic and topographic variables, Hamazaki (2002) classified the species as a nearshore northern Atlantic species, with sightings concentrated in waters with sea surface temperatures of 10° - 15° C and depths less than 500 m. In the Bay of Fundy, minke whale sighting rates showed a non-linear relationship with increasing benthic slopes and a linear relationship with increasing water depth (Ingram et al. 2007). Distributions of minke whales were highly correlated with sea surface temperature fronts in the Gulf of St. Lawrence (Doniol-Valcroze et al. 2007).

2.5.5 Stock Structure

Four stocks of minke whales in the North Atlantic are recognized by the IWC—the Northeast Atlantic, Central North Atlantic, West Greenland, and Canadian East Coast stocks (Reilly et al. 2008d). Animals found along the U.S. Atlantic coast are considered members of the Canadian East Coast stock. The north Atlantic summer range has been split into ten management areas (Donovan 1991). Regional variations in organochlorine, fatty acid, elemental and stable isotope compositions (Born et al. 2002, 2003, 2007; Møller et al. 2003) and genetics (Andersen et al. 2003) have suggested population substructure in the eastern North Atlantic, but similar work has not been done in the western North Atlantic.

2.5.6 Life History Traits

Minke whales become sexually mature at about 6 to 7 years of age and observed pregnancy rates of up to 100% suggest an annual reproductive cycle (Perrin and Brownell Jr 2009). Gestation is approximately 10 months, and calves, which are 2.4-2.7 m at birth, are nursed for 4-5 months (Perrin and Brownell, Jr. 2009). Genetic research indicates that female minke whales mate promiscuously across seasons (Skaug et al. 2008).

2.5.7 Food Habits

Minke whales are opportunistic feeders, feeding on crustaceans, plankton, and schooling fish (Horwood 1990; Perrin and Brownell Jr 2009). Individual minke whales off Scotland were found to specialize in lunge feeding or bird-associated feeding strategies (Hoelzel et al. 1989). Auks in particular, (common guillemots *Uria aalge* and razorbills *Alca torda*) had a strong association with minkes, with the minkes taking advantage of the prey-aggregating behavior of the auks (Anderwald et al. 2011). Schooling mackerel (*Scomber scombrus*) created aggregations of sand lance used by minke whales (Robinson and Tetley 2007). In the mouth of the Saguenay River, Quebec, minkes were observed to exhibit novel behaviors for aggregating prey at the surface (Kuker et al. 2010), including "head slaps," "chin-up blows" and "exhales on the dive." Western North Pacific minke whale stomach contents examined by Tamura et al. (2009) contained one species of copepod, two krill species, two squid species, and eight fish species. Geographic and seasonal variability of prey selection that reflected changes in availability in the prey species was demonstrated for North Pacific minke whales (Tamura and Fujise 2002). Atlantic herring (*Clupea harengus*) and capelin (*Mallotus villosus*) were the primary prey species found in minke whale stomachs from the southern Barents Sea and coast of northern Norway (Lindstrøm et al. 2000, 2002; Sivertsen et al. 2006). The degree of variation in observed weights of forestomach contents from minke whales off Norway suggested that the whales were feeding in well-defined bouts separated by periods of non-feeding (Haug et al. 1997). Multispecies functional response modeling showed that minkes may deplete local prey aggregations at small spatial scales (Smout and Lindstrøm 2007). Stomach content analysis of 10 stranded minke whales in Scotland showed that sand eels made up about 2/3 of the prey consumed, with herring and sprat (*Sprattus sprattus*) accounting for most of the remainder (Pierce et al. 2004).

2.5.8 Health

2.5.8.1 Strandings

Minke whale strandings occur on a regular basis along the U.S. Atlantic coastline. Eighty-two confirmed minke whale mortalities were reported in the mortality and serious injury report for the period 2005-2009 (Henry et al. 2011). This figure includes anthropogenic and non-anthropogenic mortalities.

A minke whale calf was associated with a mass stranding event in North Carolina in 2005. The animal was emaciated and presumed to be a calf that had become separated from its mother. The mass stranding was temporally and spatially associated with a period of active sonar transmissions, which could not conclusively be ruled out as a causative factor (Hohn et al. 2006).

2.5.8.2 Contaminants

Organochlorine contaminants in minke whale samples from the northeast Atlantic have been studied by Kleivane and Skaare (1998), Hobbs (2002), Born (2007), and Gouteaux (2008). Blubber samples taken from minke, fin, blue, and humpback whales in the Gulf of St. Lawrence, Quebec, in summer and fall of 1991 and 1992 were tested for chlorinated biphenyls and other persistent organochlorine compounds (Gauthier et al. 1997). Concentrations of PCBs (polychlorinated biphenyls) and most OCs (organochlorine) were similar between the four species, but minke whales had lower levels of DDTs than the fin and blue whales. Mirex concentrations in the minke samples were double those found in the other balaenopterids.

Radioactive caesium-137 concentrations in minke whales from the northeast Atlantic, West Greenland, and the North Sea were studied by Born et al. (2002). The highest caesium concentrations were found in whales from the North Sea. Significant differences in mean caesium concentrations were found when samples from minke whales in Svalbard and the North Sea were compared with samples from other areas. Selenium metabolites in minke whale urine were examined by Hasunuma et al. (1993), who found a mean level of 1500 ng/ml—about 30 times the mean level in humans. Heavy metal and trace element concentrations have been examined in Antarctic minkes (Honda et al. 1987; Kunito et al. 2002), and in northeast Atlantic minkes (Born et al. 2007). The multi-elemental analysis was useful in identifying population substructure in the northeast Atlantic (Born et al. 2007).

2.5.8.3 Disease

Serum chemistry of minke whales caught off the coast of Norway was analyzed in order to obtain reference values for this species (Tryland and Brun 2001). Parasitic infections of minke whales from the Pacific were reported by Uchida et al. (1998).

2.5.9 Acoustics

Minke whale vocalizations include “clicks,” “grunts,” “pulse trains,” “ratchets,” “thumps,” and “boings” (Beamish and Mitchell 1973; Winn and Perkins 1976; Edds-Walton 2000; Mellinger et al. 2000; Rankin and Barlow 2005).

Work with dwarf minke whales at the Great Barrier Reef off Australia, has contributed to a large database collection of minke whale sounds and increased understanding of the acoustic ecology of this species (Costa 1998; Costa and Gedamke 2000; Gedamke 2004). Sound playback experiments showed that song could be important regulating spatial interaction (Costa 2003; Gedamke 2004).

Once the “boing” sound was confirmed to be produced by minke whales in the North Pacific (Rankin and Barlow 2005), attention focused on this distinctive sound. Acoustic and visual survey data collected in the Pacific Missile Range off the Hawaiian Islands was used to develop methods of detection, localization and characterization of minke whale calls (Martin et al. 2011; Nosal 2011). Boings recorded in the Hawaiian Islands exhibited significant differences in pulse repetition rates compared with boings recorded in the Marianas Islands (Norris et al. 2010), although boings recorded in the Marianas are more similar to Hawaiian boings than to those recorded east of 138°W (Oswald et al. 2008). In Hawaii, boings were detected from October until May, with a peak in March, and absent from June to September (Oswald et al. 2011). Data collected from combined visual and acoustic surveys around the Hawaiian Islands has suggested that minke whales are more common in that area than previously reported, or than would be assumed from visual data alone (Rankin et al. 2007) and is being used to develop a spatially explicit capture-recapture method of estimating density (Marques et al. 2010; Martin et al. 2010).

In 1973, Beamish and Mitchell recorded narrow-band pulses in the presence of minke whales in the North Atlantic (Beamish and Mitchell 1973). Edds-Walton (2000) studied minke whale acoustics in the Mingan Islands (Canada) and recorded frequency modulated downsweeps in the 80-120 Hz frequency range. In the West Indies, the “speed-up” pulse train—an accelerating series of pulses in the 200-400 Hz band, with individual pulses lasting 40-60 msec. and a “slow-down” pulse train—a decelerating series of pulses in the 250-350 Hz band have been attributed to minke whales (Mellinger et al. 2000).

2.5.10 Fisheries By-catch and Entanglement

Minke whale interactions have been reported in trawls, driftnet, gillnet, weir, pot/trap, and purse seine fisheries along the U.S. and Canadian Atlantic coast (Waring et al. 2011). The annual average fishery mortality for this stock in the period 2005-2009 was estimated to be 5.5 animals (Waring et al. 2011).

A minke whale with fresh entanglement-like injuries was observed surface-feeding on capelin in the Mingan Islands, Gulf of St. Lawrence, Canada (Kot et al. 2009). Feeding ability appeared somewhat impaired, though long-term health effects were not possible to assess.

2.5.11 Vessel Interactions

Two ship-strike mortalities of minke whales were reported between 2005 and 2009 along the U.S. Atlantic coast, one off Pleasant Point, New Jersey, and one off Port Elizabeth, New Jersey (Waring et al. 2011).

2.5.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise by seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear. A minke whale calf was part of a mass-stranding that was associated in time and space with a period of active sonar transmissions, but the sonar activity was not definitively implicated as a causative factor in the mass-stranding (Hohn et al. 2006). Low levels of localized avoidance were found in mysticetes exposed to seismic airgun noise in the UK and adjacent waters investigated in 1997-2000 (Stone and Tasker 2006).

2.5.13 Data Gaps and Research Recommendations

Much of minke whale research has been done in regions where data is available from whaling catches, such as off Norway and Japan, or in areas where there are resident or seasonal populations, such as off Scotland or in the Gulf of St Lawrence. Minke whale use of the remainder of their range is less well understood. Development of acoustic detection and acoustically-derived abundance estimates, such as the ones being developed in the Hawaiian Islands, would be useful if they could be adapted for use in the Atlantic.

2.6 HUMPBACK WHALE (*MEGAPTERA NOVAEANGLIAE*)

2.6.1 Legal Status

The humpback whale is classified as endangered under the ESA and depleted under the MMPA throughout its range. The species is classified in the “Least Concern” category under the IUCN (Reilly et al. 2008d).

2.6.2 General Distribution

Humpback whales are found in all major oceans from the equator to sub-polar latitudes and typically migrate between tropical mating and calving grounds and temperate feeding grounds. The summer range in the western North Atlantic extends from the Gulf of Maine to the Davis Strait (BOEM North Atlantic Planning Area) and the primary winter distribution is in the West Indies, from Cuba to Venezuela (Reilly et al. 2008d). High resighting rates of humpbacks in the southern Gulf of Maine as well as high rates of annual return to the area were measured by Clapham et al. (1993). Average occupancy in the area was 88.1 days, and the mean rate of return of individuals in consecutive years was 73.2%. Stevick et al. (2003a) found temporal differences in occupancy of the West Indies breeding grounds between humpbacks from different feeding aggregations, with individuals from the Gulf of Maine and eastern Canada arriving earlier than those from Greenland, Iceland, and Norway. Males were also seen earlier on the breeding grounds than females. Sighting and stranding locations of humpback whales are shown in Figure 2.6-1.

2.6.3 General Abundance

The abundance of humpback whales breeding in the West Indies was estimated at 10,752 for the years 1992-93 (CV=0.068); an estimated annual rate of increase at that time was 3.1% (SE 0.5%) (Stevick et al. 2003b). The best abundance estimate for the Gulf of Maine stock of humpback whales is 847 animals (CV=0.55), based on a NMFS sighting survey in 2006 that covered waters from the southern Gulf of Maine to the upper Bay of Fundy and the Gulf of St. Lawrence (Waring et al. 2011). North Atlantic humpback abundance data were analyzed using a spatially-explicit population dynamics model which confirmed the population increase (Punt et al. 2007).

2.6.4 Habitat Preference

Dalla Rosa (2010) modeled the feeding habitats of humpback whales in the waters of both British Columbia and Antarctica. Humpback whale distribution within the foraging habitat was shown to be influenced by physical and biological factors such as complex topography and temperature fronts that increased biological productivity. Distributions of humpback whales were highly correlated with sea surface temperature fronts in the Gulf of St. Lawrence (Doniol-Valcroze et al. 2007). Hamazaki’s cetacean habitat prediction model classifies the humpback as a North Atlantic shelf species (Hamazaki 2002). Summer sightings were associated with mean water temperatures of 17.2°C, mean depths of 261 m, mean bottom slope of 0.7° and front probability of 6.3%.

Warm and shallow waters are preferred habitat on the winter breeding grounds, and spatial habitat modeling has been used to predict habitat use in the Hawaiian Islands (Johnston et al. 2007). Depths less than 200 m and water temperatures greater than 21.1° C were identified as potential wintering habitat. Within a breeding site, variation in habitat use has been shown to be linked to behavioral class, with mother-calf pairs showing a preference for shallower water compared to other group types (Ersts and Rosenbaum 2003). Habitat use between sites in a breeding region has also been shown to vary with reproductive status (Craig and Herman 2000).

2.6.5 Stock Structure

Worldwide, approximately 13 stocks have been identified that winter in tropical and sub-tropical waters (National Marine Fisheries Service 1991). Six distinct feeding aggregations have been identified for North Atlantic humpbacks: Gulf of Maine, Gulf of St Lawrence, Newfoundland/Labrador, West Greenland, Iceland, and North Norway (Reilly et al. 2008d), although Stevick (2006) suggested, from examination of movement patterns, these may represent only four separate aggregations. Although these are relatively disjunct aggregations, whales from different feeding grounds mix on the North Atlantic breeding grounds (Reilly et al. 2008d). In other regions, analysis of migratory patterns and genetics has suggested the occasional overlap of individuals from populations in opposing hemispheres on the low-latitude breeding grounds (Rizzo and Schulte 2009), although high genetic heterogeneity suggests this gene flow is minimal (Valsecchi et al. 1997). While Palsbøll et al. (1995) observed apparent genetic homogeneity within the four western North Atlantic feeding aggregations, more recent studies mentioned in Clapham et al. (2003) found significant difference in mtDNA haplotype frequencies from these four feeding areas and spatial analysis of sighting data (Vigness-Raposa et al. 2010) supported the distinct nature of these units. Humpback whales observed in waters off the U.S. mid-Atlantic coast were compared to photos in catalogues of whales from the Gulf Maine and from other North Atlantic feeding areas (Barco et al. 2002). Most of the positive matches were made with Gulf of Maine whales; however, Newfoundland and Gulf of St. Lawrence whales were also represented.

2.6.6 Life History Traits

Data based on whaling records revealed that the average age of sexual maturity in females is approximately 5 years (Chittleborough 1965). Local sighting data collected for individually identified humpback whales were used to obtain an estimate of 5 years for youngest age at first parturition, 2.38 years for mean birth interval, 0.960 for noncalf survival rate and 0.875 for calf survival rate (Barlow and Clapham 1997). A significantly higher survival rate for females than for males was found by Ramp et al. (2010). Promiscuous mating by humpbacks was genetically confirmed by Clapham and Palsbøll (1997). Association patterns of mother/calf humpback whale pairs in the southern Gulf of Maine were examined by Sardi et al. (2005).

2.6.7 Food Habits

Humpback whales feed on a variety of schooling fish species and euphausiids (Robbins 2007), with sand lance (*Ammodytes* spp.) identified as the primary prey species in the southern Gulf of Maine and herring (*Clupea harengus*) in the northern part. Fluctuations in humpback abundance within the Stellwagen Bank National Marine Sanctuary appear closely tied to fluctuations in availability of sand lance (Payne et al. 1990). Diel patterns in humpback foraging behavior concurrent with changes in prey behavior were demonstrated by Friedlaender et al. (2009) on Stellwagen Bank, Massachusetts. Surface feeding occurred more during daylight hours and bottom feeding took place largely at night.

The diets of fin whales and humpback whales in the Gulf of S. Lawrence were compared using blubber fatty acid analysis (Borobia et al. 1995). The chemical and isotopic differences found indicated that humpbacks fed at a slightly lower trophic level than fin whales.

Two principal categories of feeding behaviors were described for humpback whales in the continental shelf waters of the northeastern U.S.: swimming/lunging behaviors and bubbling behaviors (Hain et al. 1982). Recent work off Stellwagen Bank, Massachusetts, has described in detail two general patterns of bubble-net feeding behavior (Wiley et al. 2011). Additionally, quantitative measures of humpback underwater behavior while foraging were obtained through the use of combining digital acoustic recording devices (DTAGs) on the whale with high-frequency acoustic sources deployed from small boats following the tagged whale (Ware et al. 2006; Schmidt et al. 2010).

In the North Pacific, acoustic time-depth transmitters on humpbacks off Kodiak, Alaska, revealed that the whales were foraging primarily between 92 m and 120 m and concentrating on areas with maximum capelin densities (Witteveen et al. 2008). The number of lunges executed per dive and the respiratory frequency between dives were examined used digital tags on humpback whales off California's central coast (Goldbogen et al. 2008). Maximum dive durations during foraging were found to be approximately half as long as those reported for non-feeding whales and respiratory rates much higher, suggesting that lunge feeding carries a high energetic cost for humpback whales.

2.6.8 Health

2.6.8.1 Strandings

Humpback whales strand fairly regularly along the U.S. Atlantic coast. Thirty-eight stranded humpbacks were recorded in the mid-Atlantic and southeast U.S. in the period 1985-1992 (Wiley et al. 1995). For 20 of these a cause of death was able to be determined and over half of those 20 indicated anthropogenic factors that contributed to their death. One hundred and fifteen confirmed humpback whale mortalities were reported in the mortality and serious injury report for the period 2005-2009 (Henry et al. 2011). This figure includes anthropogenic and non-anthropogenic mortalities.

Humpback whales were involved in Unusual Mortality Events (UMEs) in 2003 when 12-15 humpback whales died in the vicinity of the Northeast Peak of Georges Bank, in a 2005 large whale UME which involved seven humpbacks in New England, and in a 2006 UME which involved 21 humpbacks (Waring et al. 2011). Causes of these UMEs are unresolved.

2.6.8.2 Contaminants

Blubber samples taken from humpback whales in the Gulf of St. Lawrence, Quebec, in summer and fall of 1991 and 1992 were tested for chlorinated biphenyls and other persistent organochlorine compounds (Gauthier et al. 1997). Subsequent work confirmed those results and also found that there were no significant differences in the concentrations of contaminants between females and calves, suggesting that calves are susceptible to bioaccumulation of contaminants through placental and lactational transfer (Metcalfe et al. 2004). Comparisons of contaminant loads between North Pacific and North Atlantic humpback whales showed that North Atlantic (Gulf of Maine) whales had higher levels of PCBs, polybrominated diphenyl ethers (PBDPEs) and chlordanes, while North Pacific whales that fed off southern California had the highest levels of DDTs (Elfes 2008; Elfes et al. 2010).

2.6.8.3 Disease

Bacterial communities associated with the skin of both healthy and health-compromised North Pacific humpback whales were investigated by Apprill et al. (2011). The dinoflagellate neurotoxin, saxitoxin, was responsible for the deaths of 14 humpback whales in November/December 1987 (Geraci et al. 1989).

2.6.9 Acoustics

Humpback whales produce a variety of acoustic signals, both vocalizations and surface-generated signals, such as breaching or pectoral slapping. In some cases, humpbacks may switch from primarily vocal communication to surface-generated communication when wind speed or background noise levels increased (Dunlop et al. 2010).

Singing behavior by male humpbacks is well-studied but still only partially-understood. The function of humpback song is generally thought to serve to attract females (Tyack 1981) or to mediate interaction between males, either as a spacing mechanism (Frankel et al. 1995) or in the establishment of dominance hierarchies (Darling 1983; Darling and Bérubé 2001). Recent work has shown that males do use song to mediate intra-sexual interactions (Cholewiak 2008). Extensive work on song has both described song structure as well as considered other hypotheses as to the function of song (for example, Cerchio et al. 2001; Darling 1983; Frazer and Mercado 2000; Parsons et al. 2008). Singing in high-latitude feedings areas has also been documented. Off the coast of Massachusetts, Matilla et al. (1987) found that song may be common in the fall prior to migration, and Clark and Clapham (2004) documented song in spring and early summer post-migration. Singing in this context may include full renditions or abbreviated versions of song (McSweeney et al. 1989). The occurrence of song at these times of year is thought to be linked to seasonal elevations in hormone levels occurring near the breeding season.

Compared to song, non-song vocal activity has been relatively little studied. Research along the eastern Australian migratory routes described and quantified “social” vocalizations (Dunlop et al. 2007), and discussed their use relative to social context (Dunlop et al. 2008). Based on digital acoustic tag data collected in the Stellwagen Bank National Marine Sanctuary, Massachusetts, Stimpert et al. (2011) classified non-song humpback vocalizations into eight types with similar acoustic properties and identified two of these types (“wops” and “grunts”) as candidates for use in passive acoustic monitoring. Increased humpback call rates were observed associated with group-forming behavior in the Stellwagen Bank National Marine Sanctuary, leading researchers to believe that vocalization plays a role in the social cohesion of feeding groups (Palmer et al. 2010). Recordings of humpback click production associated with nighttime lunge feeding were reported by Stimpert et al. (2007). Humpback sounds recorded on the northwestern Atlantic feeding grounds were described and compared to those recorded on the Hawaiian breeding grounds by Stimpert et al. (2008a).

A predicted humpback audiogram and bandpass model of the humpback ear were developed by Hauser et al. (2001), with a zone of maximum sensitivity found to be in the 2-6 kHz range. When exposed to low-frequency active sonar, humpbacks were found to increase song length (Miller et al. 2000; Fristrup et al. 2003).

The effects of shallow-water propagation of humpback whale sounds were examined by Mercado and Frazer (1999). It was found that the whales would adjust their positions and sound production in response to environmental factors. The acoustic field in the vertical plane of singing humpback whales was measured by Au et al. (2006). It was found that sounds were projected in the horizontal direction regardless of the orientation of the singer. Source levels varied between 151 and 173 bd *re* 1 μPa and high-frequency harmonics extended beyond 24 kHz.

2.6.10 Fisheries By-catch and Entanglement

Humpback whale interactions have been reported for pot/trap, purse seine, trawl and gillnet fisheries along the U.S. Atlantic coast (Fertl and Leatherwood 1997; Waring et al. 2011). Analysis of Gulf of Maine humpback caudal peduncle photographs has yielded evidence that between 48% and 65% of each annually collected sample of photographs show evidence of entanglement scarring (Robbins and Mattila 2001). Twenty-five percent (5 out of 20) of humpback whales stranded along the mid-Atlantic and southeast Atlantic coast of the U.S. between 1985 and 1992 had injuries consistent with possible entanglement in fishing gear (Wiley et al. 1995). Volgenau et al. (1995) examined the impact of entanglement mortality on humpbacks and found that when entanglement losses were added to natural mortality estimates and subtracted from birth rate estimates, annual mortality rates were 5.4% for the Newfoundland and Labrador substock and 4.8% for the Gulf of Maine substock. Increases in fishery entanglement rates were linked to industrial activity in Trinity Bay, Newfoundland (Lien 1993; Todd et al. 1996).

2.6.11 Vessel Interactions

Analysis of humpback whales stranded along the mid-Atlantic and southeast Atlantic coast of the U.S. between 1985 and 1992 showed that 6 out of 20 of the animals examined had injuries likely attributable to ship strike (Wiley et al. 1995). The detected annual average mortality and serious injury rate due to ship strike for humpback whales from the Gulf of Maine stock for the years 2005-2009 was 1.4 (Henry et al. 2011).

The effect of whale watch vessel exposure on humpback calving rates and calf survival was investigated by Weinrich and Corbelli (2009). No direct evidence for negative effects was found. Behavioral differences were found, however, in humpbacks in Hervey Bay, Australia when in the presence of vessels (Corkeron 1995). For instance, whales were more likely to dive rather than slip underwater when within 300 m of a vessel. Au and Green (2000) characterized the underwater noise produced by five representative whale-watching boats used in the Hawaiian Islands and concluded that the levels of sound produced would not have deleterious effects on the auditory systems of humpback whales. Analysis of humpback song in the presence of boat noise revealed that while unit duration and phrase duration were shortened, frequency structure showed no response (Norris 1994). Measurement of the underwater noise production of the Hawaiian super ferry was conducted by Stimpert et al. (2008b) and the potential effects of the sounds on humpbacks discussed.

2.6.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear. Humpback response to the acoustic impacts of petroleum industry activities was tested in 1984 off southeast Alaska (Malme et al. 1985). Airgun sound was produced as well as playbacks of recorded drillship, drilling platform, production platform, semi-submersible drill rig, and helicopter fly-over noise and behavior of humpback whales in the area monitored. No clear avoidance to the area was observed. The fact that humpbacks are susceptible to severe blast trauma was demonstrated by examination of the temporal bones from two humpback whales which died following a 5,000-kg explosion in Trinity Bay, Newfoundland (Ketten et al. 1993). Long-term exposure to industrial activity in Trinity Bay was also linked to increased fishery entrapment rates (Lien 1993; Todd et al. 1996), and to decreases in return rates of photographically-identified whales (Borggaard et al. 1999). In 2002, an unusual increase in humpback whale strandings near Abrolhos Bank, Brazil coincided with 3D seismic surveys conducted in the area (Engel et al. 2004). Low levels of localized avoidance were found in mysticetes exposed to seismic airgun noise in the UK and adjacent waters investigated in 1997-2000 (Stone and Tasker 2006). A decrease in the detection of humpback whale singers was significantly correlated with seismic survey activity off the coast of Gabon (Cerchio et al. 2011), indicating that seismic activity may interrupt male breeding displays.

2.6.13 Data Gaps and Research Recommendations

The ongoing analysis of genetic data collected in the large-scale assessment called the More of North Atlantic Humpbacks (MoNAH) project, in which sampling was conducted on humpbacks in the Gulf of Maine/Scotian Shelf region and the primary wintering ground in the West Indies during 2004-2005 will refine knowledge of the North Atlantic humpback whales' population structure and help with updating abundance estimates. Further development of acoustic methods for estimating humpback abundance and distribution is also necessary to increase our knowledge of humpback habitat use and population dynamics.

Humpback whale research recommendations outlined in the recovery plan (National Marine Fisheries Service 1991) include maintaining and enhancing habitats used by humpback whales currently or historically; identifying and reducing direct human-related mortality, injury and disturbance; measuring and monitoring key population parameters; and improving administration and coordination of recovery programs.

2.7 SPERM WHALE (*PHYSETER MACROCEPHALUS*)

2.7.1 Legal Status

The sperm whale is classified as endangered under the ESA and depleted under the MMPA throughout its range. The species is classified in the “vulnerable” category under the IUCN (Taylor et al. 2008).

2.7.2 General Distribution

Sperm whales are widely distributed in all major oceans and are typically found in continental slope or deeper water (Taylor et al. 2008). Off the U.S. Atlantic coast, sightings of sperm whales are concentrated along the shelf break from Cape Hatteras to Georges Bank (Figure 2.7-1), primarily in BOEM mid-Atlantic and North Atlantic planning areas. Sighting and stranding locations of sperm whales are shown in Figure 2.7-1.

2.7.3 General Abundance

Worldwide sperm whale abundance is estimated to be approximately 360,000 (CV=0.36: Whitehead 2002). Pre-whaling abundance was estimated at 1,110,000 whales (95% CI: 672,000 to 1,512,000) in 1999, thus, ten years after the end of large-scale whaling for this species, the population size was at 32% of its original level (Whitehead 2002). For the Western North Atlantic stock of sperm whales, the best recent abundance estimate is 4,804 (CV=0.47). This is the sum of the estimate derived from a summer 2004 northern U.S. Atlantic survey (2,607, CV=0.57) and an estimate derived from a summer 2004 southern U.S. Atlantic survey (2,195, CV=0.47) (Waring et al. 2011).

2.7.4 Habitat Preference

Sighing data from summer shipboard sighting surveys between 1990 and 1998 were analyzed to determine habit use with respect to bathymetric and oceanographic features (Waring et al. 2001). Sperm whale sightings were associated with warm (mean of 23.9°C), deep (mean of 1942 m) water, with sighting rates slightly higher at canyon features. Warm core ring eddies from the Gulf Stream along the shelf-edge were also found to be significant habitat areas (Waring et al. 1993; Griffin 1999). Association of sperm whales and sea surface temperature fronts in the Mediterranean was noted by Gannier and Prace (2007). Along the Mid-Atlantic Ridge, sperm whales were found to be associated with cross-seamount or cross-frontal structures (Skov et al. 2008). Hamazaki’s cetacean habitat prediction model classifies the sperm whale as a mid-Atlantic offshore species (Hamazaki 2002). Summer sightings were associated with mean water temperatures of 22.5°C, mean depths of 2,011 m, mean bottom slope of 1.9° and front probability of 6.5%.

Scott and Sadove (1997) reported on sightings of sperm waters in shallow shelf waters off Long Island, New York between 1983 and 1984. These whales were seen mostly in late spring or early summer in waters that ranged from 41 m to 68 m in depth. It was speculated that the whales were taking up short-term residency in the area as they followed prey inshore.

2.7.5 Stock Structure

The International Whaling Commission recognizes one stock for the North Atlantic; whether the western North Atlantic population is discrete from the eastern North Atlantic population is unresolved (Waring et al. 2011). Low geographic structure and genetic diversity have been reported (Reeves and Whitehead 1997; Lyrholm and Gyllensten 1998; Dufault et al. 1999; Lyrholm et al. 1999; Engelhaupt 2004). However, strong genetic differentiation was found between social groups and mtDNA heterogeneity indicated moderate genetic differentiation on a global scale (Lyrholm and Gyllensten 1998; Lyrholm et al. 1999). Drouot (2003) reported genetic differentiation in mtDNA between samples from Mediterranean and North Atlantic sperm whales. Engelhaupt et al. (2009) demonstrated significant mtDNA subdivision between Gulf of Mexico, Mediterranean and North Atlantic samples, suggesting fidelity to coastal basins. Eighteen new single nucleotide polymorphism markers were characterized for the sperm whale by Morin et al. (2007). These are the first such markers designed for genotyping sperm whale populations.

Groups of female sperm whales have distinctive vocalization patterns or codas that may be more indicative of stock or substock structure than genetics. Coda repertoires are geographically specific at finer spatial scales than genetics (Whitehead 2003). Sperm whale populations are structured more along social lines than geographic ones, and culture may play an important evolutionary role (Whitehead 1998, 2003). The social organization of sperm whales is an important focus of current research (Whitehead et al. 1991; Whitehead 1996, 1998, 2003; Richard et al. 1996; Christal et al. 1998; Mesnick 2001; Christal and Whitehead 2001; Lettevall et al. 2002; Gero et al. 2007; Gero and Whitehead 2007; Pinela et al. 2009).

2.7.6 Life History Traits

Age at maturity for female sperm whales is approximately 9 years of age (Whitehead 2003). Females have one calf every 4-6 years, and will nurse the calf for approximately 2 years. Females and juveniles will stay together in social groups in tropical and temperate latitudes, while males leave the group at the age of 3-15 and migrate to higher latitudes, forming bachelor groups when young and travelling more singly as they age.

2.7.7 Food Habits

Sperm whales have a diverse diet which, in most areas, consists primarily of squid species (Whitehead 2003; Simon et al. 2003; Davis et al. 2007). Tagging of sperm whales and jumbo squid (*Dosidicus gigas*) in the Gulf of California showed that the dive depth of the sperm whales generally coincided with the depth range inhabited by the squid (Davis et al. 2007). Stomach contents of sperm whales stranded in Denmark, Scotland, Ireland and the Netherlands were all primarily composed of cephalopod beaks of the species *Gonatus fabricii* (Santos et al. 1999, 2002; Simon et al. 2003). Stomach samples from sperm whales caught off the west coast of Iceland, however, were predominantly composed of fish of the species *Cyclopterus lumpus*, *Sebastes sp.*, and *Lophius piscatorius* (Roe 1969). Sperm whales in the Gulf of California were shown to change their distribution in response to changes in distribution of their main prey in that area–jumbo squid (*Dosidicus gigas)* (Jaquet and Gendron 2002; Jaquet et al. 2003).

Sperm whale diet in the South Pacific was studied by Marcoux et al. (2007) by measuring carbon and nitrogen isotope ratios in sloughed skin samples and in squid beaks from fecal samples. Variation in diet was examined across region, year, social group and vocal clan, with significant differences found between groups and between clans, suggesting differential use of micro-habitats. Exploration of isotope ratios in Gulf of California sperm whales revealed that females and immature males had a predator-prey relationship with jumbo squid, but adult males did not (Ruiz-Cooley et al. 2004). It was theorized that the males' isotopic signature was indicative of an earlier high-latitude feeding ground. Isotope ratios in sperm whale teeth have also been examined, with differences in sex, age, and region indicating this as a useful technique in studying ontogenetic movements and dietary histories of this species (Mendes et al. 2007a, 2007b).

The hypothesis that sperm whales use echolocation for prey capture was supported by DTAG recordings of sperm whales in the Ligurian Sea and in the Gulf of Mexico (Miller et al. 2004). During foraging dives the whales produced regular clicks and rapid-click buzzes called "creaks." Creak production was highest at the bottom of the dive, when the animal was also noted to be actively maneuvering its body position. Palka et al. (2007), in an acoustic tagging study performed in 2003 off the northeast and mid-Atlantic shelf, also found that, while the majority of creaks occurred near the base of the dive, occasional shallow creaks and continued clicking through the ascent may indicate opportunistic foraging at shallower depths. An alternate theory of foraging behavior was presented by Fristup and Harbison (2002), who suggested that optical cues might play an important role.

Creak production and diving behavior were also observed in the northwestern Mediterranean, where creaks were assumed to correlate with feeding events and both number of creaks produced per dive and dive time were correlated with body size of the whale (Drouot et al. 2004). Number of creaks per dive was observed to be consistent between Atlantic Ocean, Gulf of Mexico, and Ligurian Sea sperm whales (Watwood et al. 2006). Duration of the foraging phase of dives, percentage of dive time in foraging phase, and average dive cycles were also consistent across regions, although maximum dive depth did vary. A sperm whale tagged with a suction-cup time depth recorder off Japan was shown to execute repeated dives to 400-1200 m with dive durations of 30-45 minutes (Amano and Yoshioka 2003). In the Gulf of California, in 10 instances sperm whales were observed diving to the depth of the acoustic backscatter layer and returning with jumbo squid in their jaws and in one instance returning with an unidentified fish (Gallo-Reynoso et al. 2009). Average depth of dive was 342 m, average dive time was 23 minutes, and average interval between dives was 6 minutes. Dive behavior of sperm whales off the coast of Japan was studied at two locations (Aoki et al. 2007). At the Ogasawara Islands, there was a distinct diel pattern of diving and whales dived deeper and swimming faster during the day, but along the Kumano Coast no diel rhythm was observed.

2.7.8 Health

2.7.8.1 Strandings

Sperm whale strandings, both mass- and single strandings, have been reported in many areas of the world (Evans 1997; Whitehead 2003; Pierce et al. 2007; Mazzariol et al. 2011). Various explanations have been suggested. Sperm whale stranding events along the North Sea coastline have been correlated with geomagnetic anomalies (Vanselow et al. 2009) and with positive temperature anomalies (Pierce et al. 2007). Lunar cycles have also been correlated with sperm whale strandings (Wright 2005). The stranding rate of sperm whales in the British Isles showed a marked increase since 1970 (Goold et al. 2002). Goold et al. (2002) also found that almost all of the sperm whales stranded on the British and eastern Canadian coasts were males. Eighteen sperm whales were reported stranded along the U.S. Atlantic coast and Puerto Rico during 1994-2000 and 15 were reported during 2001-2005 (Waring et al. 2011).

2.7.8.2 Contaminants

Aguilar (1983) measured residues of DDE, Dichlorodiphenyldichloroethane (DDD), DDT, and PCB in sperm whales caught off Spain's northwestern coast. Females were found to be more contaminated by organochlorine compounds than males, but overall levels were similar to levels found in sperm whales from other areas of the northern hemisphere and to be intermediate compared to other cetaceans. Global mean levels of chromium in sperm whale skin tissue were found to be 28 times higher than mean levels of chromium levels in skin of humans who had no occupational exposure (Wise et al. 2009). Levels ranged from 3.3 ± 0.4 µg/g in samples taken off the coast of Sri Lanka to 44.3 ± 14.4 µg/g in samples collected from sperm whales near the Islands of Kiribati in the Pacific Ocean. Elevated chromium levels were shown to induce cytotoxicity and genotoxicity in sperm whale skin fibroblasts (Wise et al. 2011). Heavy metals, organochlorines, and polycyclic aromatic hydrocarbons (PAHs) were measured in various tissues of sperm whales that had stranded on the southern coast of the North Sea in the winter of 1994/1995 (Holsbeek et al. 1999). High concentrations of cadmium (up to 300 µg/g), mercury (up to 130 µg/g), and PCBs (up to 5 µg/g) were detected but not considered to be the direct cause of mortality in any of the animals. Cytochrome P450 1A1 expression is a known biomarker of exposure to aryl hydrocarbon receptor agonists (Godard-Codding et al. 2010). Skin biopsies of sperm whales from various regions of the Pacific Ocean were analyzed for this biomarker, with the aim of establishing a baseline and to reveal regional differences within the Pacific Ocean.

2.7.8.3 Disease

Observations on natural disease in sperm whales, mostly derived from whaling activities, were reviewed by Lambertsen (1997). Osteonecrosis, likely caused by dysbarism ("the bends") was identified in sperm whales by Moore and Early (2004).

2.7.9 Acoustics

Sperm whales vocalize using click sounds for communication and echolocation and this behavior has received some important attention in recent publications (Møhl et al. 2000, 2003; Jaquet et al. 2001; Ridgway and Carder 2001; Thode et al. 2002; Madsen et al. 2002c, 2002b; Wahlberg 2002; Rendell and Whitehead 2003, 2004, 2005; Mellinger et al. 2004; Rhinelander and Dawson 2004; Laplanche et al. 2005; Kandia and Stylianou 2006; Morrissey et al. 2006; Schulz et al. 2008; Sidorovskaia et al. 2009; See, for example: André et al. 2008; Wahlberg et al. 2005; Hirotsu et al. 2009; Whitehead et al. 2006; Zaugg et al. 2010; Zimmer et al. 2005a, 2005b; Antunes et al. 2010, 2011).

The sperm whale click is the loudest sound recorded from a biological source (Møhl et al. 2000), and can be heard by other sperm whales at ranges up to 60 km (Madsen et al. 2002c). While traditionally the clicks had been thought to be multipulsed, recent data show that, in fact, a component of the click is monopulsed and highly directional and suitable for use as long-range sonar (Møhl et al. 2000, 2003; Zimmer et al. 2005a). Visualizing the beam pattern in three dimensions revealed that a component of the click is a narrow and forward-directed sonar signal, but less-directional click components may be used to communicate with conspecifics or for orientation (Zimmer et al. 2005b). Click production has been shown to happen simultaneously with breathing (Wahlberg et al. 2005). A classification system which can separate sperm whale clicks from impulsive ship noise during automated acoustic detection has been developed by Zaugg et al. (2010).

Acoustic techniques have been developed to estimate sperm whale abundance, or supplement visual sighting surveys. Acoustic and visual detection methods were found to result in similar abundance estimates in the eastern Pacific (Barlow and Taylor 2005). Due to the increased range of detection, and the ability to detect whales at night, more sperm whales were detected with acoustic methods; however, visual observations were necessary for group size estimation. Acoustic line-transect surveys have been carried out in the Ionian Sea and Mediterranean Sea (Lewis et al. 2007). Passive acoustic localization and tracking of sperm whales also has applications for minimizing negative interactions with human activities (André et al. 2008; André 2009).

Southall et al. (2007) classified sperm whales into the “mid-frequency cetaceans” functional hearing group, with an estimated hearing range of approximately 150 Hz to 160 kHz. A stranded sperm whale neonate examined during an attempt at rehabilitation made clicks at frequencies of 500 Hz to 12 kHz and an auditory brainstem response indicated that it was most sensitive to sounds in the 5 kHz-20 kHz range (Ridgway and Carder 2001).

2.7.10 Fisheries By-catch and Entanglement

Depredation of longline catches by sperm whales has been reported off the coast of Alaska (Sigler et al. 2008; Mathias et al. 2009; Mesnick et al. n.d.). Attraction to fishing vessels during net hauling was also reported for sperm whales around the Flemish Cap, in the North Atlantic Ocean (Karpouzli and Leaper 2004). Ingestion of floating fishing net debris was implicated in the deaths of two sperm whales stranded in Northern California in 2008 (Jacobsen et al. 2010). While fishery interactions have been reported in the now-closed pelagic drift gillnet fishery and sperm whales have been found entangled with longline and fine-mesh gillnet, no fishery interactions were reported for sperm whales in U.S. Atlantic water during the period 2001-2005 (Waring et al. 2011). The threat to sperm whales from fishery interactions in the North Atlantic is categorized in the recovery plan as low (National Marine Fisheries Service 2010b).

2.7.11 Vessel Interactions

The large whale ship strike database contains 17 records of sperm whales involved with ship strike worldwide (out of a total of 292) in the period from 1975 to 2002 (Jensen and Silber 2003). One sperm whale was reported in NMFS records for the Western North Atlantic stock during 2004 through 2008 with sufficient information to confirm the cause of death as a collision with a vessel (Waring et al. 2011). The resulting annual rate of serious injury or mortality from vessel collisions was 0.2 whales. The threat to sperm whales from ship strikes in the North

Atlantic is categorized in the recovery plan as unknown but potentially low (National Marine Fisheries Service 2010b).

The short-term reaction of sperm whales to whale-watching boats was investigated in the Azores by land-based and boat-based observers (Magalhães et al. 2002). Some indications of disturbance were observed, such as changes in frequency of aerial display, but no clear pattern was found. Richter et al. (2006) also found reactions of sperm whales to whale-watching boats off Kaikoura, New Zealand to vary by season and by individual, with some resident whales becoming habituated to contact.

2.7.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear. Madsen and Møhl (2000) recorded click rate measurements and behavioral observations of sperm whales during the discharge of detonators off northern Norway. No response was detected, perhaps due to the similarity between the detonations and sperm whale clicks. Madsen et al. (2002a) also found no response from male sperm whales in polar waters exposed to air-gun pulses from a seismic survey. Sound exposure levels of sperm whales in the Gulf of Mexico were measured using acoustic tags attached to sperm whales at ranges of 1.4-12.6 km from controlled air-gun array sources (Madsen et al. 2006a). Received levels varied with depth between 131-167 dB re. 1 µPa, including significant energy at high frequencies which could be detrimental to cetacean hearing and cause acoustic masking of sperm whale clicks. In a visual sighting survey conducted in the Gulf of Mexico in 1993, significantly fewer whales were observed in an area when seismic survey activity was taking place compared to the period just prior to the seismic operations (Mate et al. 1994); however, Jochens et al. (2008) found no horizontal avoidance to controlled exposure of seismic airgun noise in the Gulf of Mexico.

2.7.13 Data Gaps and Research Recommendations

The Sperm Whale Recovery Plan (National Marine Fisheries Service 2010b) identifies key actions that should be undertaken to further understand and protect sperm whales. Priorities include better understanding of sperm whale population structure; improvement of abundance estimates and trends of abundance; improvement of knowledge of sperm whale habitat and feeding ecology; steps to minimize sperm whale mortality from anthropogenic sources such as ship strike, fishery interaction and pollution; steps to determine and minimize detrimental anthropogenic noise; and effective response to strandings of sperm whales.

2.8 PYGMY & DWARF SPERM WHALE (*KOGIA* SPP.)

Two species of the genus *Kogia* are found in the northwest Atlantic. As it is difficult to differentiate species at sea, much of the information available and presented here is only to genus level.

2.8.1 Status

The pygmy and dwarf sperm whales are not listed under the Endangered Species Act. Both species are considered “Data Deficient” on the IUCN Red List. They are afforded protection in U.S. waters under the MMPA.

2.8.2 General Distribution

The *Kogia* spp. are found in temperate and tropical waters throughout the world although their at sea distribution is poorly understood (McAlpine 2009). Dwarf and pygmy sperm whales could potentially be observed in any of the BOEM planning areas but most observations are likely to be in the Mid-Atlantic, South Atlantic or Straits of Florida Planning Areas. Sighting, stranding and fishery bycatch locations of dwarf and pygmy sperm whales are shown in Figure 2.8-1.

2.8.3 General Abundance

Data is not available to estimate the total number of *Kogia* spp. along the U.S. Atlantic coast (Waring et al. 2011). The best estimate of *Kogia* spp. is 395 (CV=0.40), based on a combination of aerial and shipboard surveys that took place in 2004 from Florida to Maine. The estimate from Florida to Maryland is 37 (CV=0.75) and from Maryland to Bay of Fundy is 358 (CV=0.44) (Waring et al. 2011).

2.8.4 Habitat Preference

McAlpine et al. (1997) analyzed the stomach contents of pygmy sperm whales and reported that the squid prey species identified were typical of the mesopelagic slope-water community. McAlpine (2009) reported that prey analyses suggested these animals are generally found on the continental shelf and slope in the epi- and mesopelagic areas. Dunphy-Daly et al. (2008) studied dwarf sperm whale habitat use in the Bahamas. They found temporal and spatial variation in habitat type being used and group size. Generally, the whales were in deeper water during the summer. In winter, group size was bigger and the whales were over the slope where vertical relief was higher.

2.8.5 Stock Structure

Genetic variation of *Kogia* spp. was analyzed via mtDNA and cytochrome *b* markers by Chivers et al. (2005). They found the two species to be reciprocally monophyletic to each other. Within the *Kogia sima* samples, the authors identified two clades: one from samples collected in the Atlantic and the other from samples collected in the Indo-Pacific. The degree of differentiation between the two clades indicated they might be two species.

Arnason et al. (1993) included a pygmy sperm whale in their analysis of baleen whale mtDNA control region.

2.8.6 Life History Traits

Caldwell and Caldwell (1989) reported that *K. sima* becomes sexually mature at about 2.1 m. McAlpine (2009) reported that male *K. breviceps* become mature at about 2.7 m with females being slightly smaller at sexual maturity.

2.8.7 Food Habits

Dwarf and pygmy sperm whales are suction feeders (Bloodworth and Marshall 2005). McAlpine et al. (1997) presented information from stomach contents of pygmy sperm whales that stranded in Atlantic Canada. They found squid beaks from *Onychoteuthis banksi*, *Chiroteuthis sp*., *Lepidoteuthis grimaldii*, *Taonius pavo*, *Histioteuthis meleagroteuthis*, and *Histioteuthis corona corona*.

Santos et al. (2006) analyzed the stomach contents of pygmy sperm whales that stranded on the coasts of Spain and France. Thirteen of the fourteen stomachs contained virtually only cephalopod species. West et al. (2009) and Beatson (2007) analyzed the stomach contents of pygmy sperm whales that stranded, respectively in Hawaii and New Zealand. They also found cephalopods were the dominant prey species.

2.8.8 Health

2.8.8.1 Strandings

There were a total of 147 *Kogia* spp. strandings along the U.S. Atlantic coast from 2005-2009 as summarized in Table 2.8-1. The majority of these were identified as *Kogia breviceps* (n=111, 76%). North Carolina had the highest number of observed strandings (n=27, 18%) followed by, South Carolina and Florida.

In January 2005, 33 short-finned pilot whales, 1 minke whale and 2 dwarf sperm whales were part of an Unusual Mortality Event (Hohn et al. 2006) along the North Carolina coast. Gross necropsies and extensive tissue analyses (clinical pathology, parasitology, gross pathology, histopathology, microbiology, and serology) did not find a unifying cause for this event. No harmful algal blooms preceded it. The U.S. Navy had conducted routine, tactical, mid-frequency sonar operations in the area during the time period. Hohn et al. (2006) were not able to definitively link the UME to sonar operations or environmental variables and concluded that it was likely that the cause of the stranding event would remain unknown.

A pygmy sperm found stranded in New Jersey, that had ingested plastic was successfully rehabilitated and released (Stamper et al. 2006).

Table. 2.8.1. Stranded Pygmy and Dwarf Sperm Whales by Species and Year

State	2005			2006			2007			2008			2009			Total		
	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp
ME	0	0	0	0	1	0	0	2	0	0	0	0	0	1	0	0	4	0
MA	0	1	0	0	1	0	0	1	1	0	2	0	1	2	0	1	7	1
RI	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0
NY	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0	2	0
NJ	1	0	0	0	0	0	0	1	0	0	1	0	0	2	0	1	4	0
DE	0	0	0	0	2	0	0	2	0	0	0	0	0	0	0	0	4	0
VA	0	0	0	0	1	0	0	6	0	2	0	0	0	0	0	2	7	0
NC	4	5	0	8	7	0	7	5	0	1	4	1	1	6	0	21	27	1
SC	0	8	0	0	1	0	1	3	0	0	5	0	1	6	0	2	23	0
GA	2	3	0	0	2	0	0	1	0	0	3	0	0	2	0	2	11	0
FL	0	3	1	1	2	0	1	5	0	2	5	0	0	6	0	4	21	1
Tot	7	20	1	9	18	0	9	27	1	5	20	1	3	26	0	33	111	3

(Data from the Northeast and Southeast US Marine Mammal Stranding Networks)

2.8.8.2 Contaminants

Bustamante et al. (2003) measured 12 trace elements in the liver, muscle, and blubber of two pygmy sperm whales that stranded in New Caledonia. No other contaminant information was available.

2.8.8.3 Disease

Fire et al. (2009) tested stranded *Kogia* spp. from the mid- and southeastern-Atlantic U.S. for domoic acid exposure. Fifty-nine percent of the samples they analyzed tested positive. Cardiomyopathy and Myocardial degeneration are both conditions that have been documented in stranded *Kogia* spp. It has been documented more often in adults and in males but the etiology is not understood (Bossart et al. 2007b). Deardorff and Overstreet (1981) described *Terranova ceticola* from the stomachs of stranded dwarf sperm whales in the Gulf of Mexico. Otherwise, McAlpine (2009) wrote little is known about disease in *Kogia* spp.

2.8.9 Acoustics

Goold and Clarke (2000) described sound velocity in the head of a dwarf sperm whale across a range of temperature and pressure conditions. Marten (2000) described vocalization of a stranded male pygmy sperm whale that was in a holding tank. The whale emitted ultrasonic clicks peaking at 125 kHz.

2.8.10 Fisheries By-Catch/Entanglement

A pygmy sperm whale was seriously injured in a pelagic longline interaction in 2000. No other more recent records of fishery interactions exist in the Northeast Fisheries Observer database.

2.8.11 Vessel Interactions

A review of Northeast and Southeast U.S. marine mammal stranding data between 2005 and 2009 yielded no records of vessel interactions with *Kogia* spp.

2.8.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear. No specific information is available about the impact of energy projects on *Kogia* spp.

2.8.13 Data Gaps and Research Recommendations

In general, very little is known about pygmy and dwarf sperm whales. There is no abundance estimate for either species of whales in the U.S Atlantic. The limited amount of information available about pygmy and dwarf whales comes from parts of their range that is outside of the U.S. Atlantic coast (e.g., from the Pacific Ocean).

2.9 CUVIER'S BEAKED WHALE (*ZIPHIUS CAVIROSTRIS*)

2.9.1 Status

Cuvier’s beaked whale is not listed under the Endangered Species Act. The status on the IUCN Red List is “Least Concern.” It is afforded protection in U.S. waters under the MMPA.

2.9.2 General Distribution

Heyning (1989) reported that stranding records of Cuvier’s beaked whales show that it is found in most oceans and seas except in the polar regions. Leatherwood and Reeves (1983a) wrote that Cuvier’s beaked whales are one of the most widely distributed cetaceans. Along the U.S. Atlantic coast, they are sighted primarily along the continental shelf edge in the Mid-Atlantic area (Hamazaki 2002; Palka 2006; Waring et al. 2001). See also MacLeod et al. (2006a) for the most recent information on the distribution of beaked whales worldwide. Sighting and stranding locations of Cuvier’s beaked whales are shown in Figure 2.9-1.

2.9.3 General Abundance

Barlow et al. (2005) described the challenges (e.g., distribution, survey conditions, ability to identify whales to species) in obtaining solid abundance estimates of beaked whales. They suggested that passive acoustics could provide identification by vocalization–see Acoustics section for related studies. Data is not available to estimate the total number of beaked whales (*Mesoplodon* spp. and *Ziphius cavirostris*) along the U.S. Atlantic coast (Waring et al. 2011). The best estimate of undifferentiated beaked whales is 3,513 (CV=0.63) based on a combination of aerial and shipboard surveys that took place in 2004 from Florida to Maine (Waring et al. 2011). The estimate from Florida to Maryland is 674 (CV=0.36) and from Maryland to Bay of Fundy is 2,839 (CV=0.78).

2.9.4 Habitat Preference

Claridge (2006) reported on vessel surveys that were conducted from 1997 to 2002 around Great Abaco Island (Bahamas). She found that Cuvier's beaked whales were found at a mean depth of 1051 m (SD=111) and they shared offshore habitat with sperm whales (*Physeter macrocephalus*). In the Bay of Biscay, Cuvier's beaked whale's distribution was strongly correlated with water depths 1000 m to 4000 m and often associated with submarine canyons or the continental shelf slope (Smith 2010). Azzellino et al. (2011) used visual observations, passive acoustics and environmental data to develop a model that could predict Cuvier's beaked whale presence in the Mediterranean Sea. A study by Gannier (2011) also examined stranding and sighting data on Cuvier's beaked whales in the Mediterranean to characterize favorable beaked whale habitats.

Ferguson et al. (2006) used beaked whale sightings and environmental data to predict whale density in the Pacific Ocean. Their results for *Ziphius* spp. are summarized in Table 2.9-1.

Table 2.9.1.
Environmental Data for *Ziphius* spp.

Variable	Value
Beaufort	2.9
Distance offshore (km)	1097.8
Depth (m)	3445.8
Slope (°)	0.732
Sea Surface Temperature (°C)	26.8
Sea Surface Salinity (psu)	33.9
Surface Chlorophyll Concentration (mg/m^3)	0.203
Thermocline Depth (m)	53.9
Thermocline Strength (°/m)	0.367

2.9.5 Stock Structure

Dalebout et al. (2004) created a reference database for twenty beaked whale species (including Cuvier's). The database includes a 437 bp segment of the mtDNA control region, a 384bp cytochrome b segment. They compiled an additional database for 17 of the 21 beaked whales species of nuclear DNA (925bp actin intron sequences). Despite this study, Barlow et al. (2005) reported that almost nothing is known about beaked whale stock structure within species. Stock structure along the U.S. Atlantic coast is poorly understood (Waring et al. 2011).

2.9.6 Life History Traits

Average length of maturity for females is 580 cm and 550 cm for males (Heyning 1989; Heyning and Mead 2009). Mean length at birth is 270 cm (Heyning and Mead 2009).

2.9.7 Food Habits

Heyning and Mead (2009) reported that squid were the most common prey item in whales found in waters less than 1000 m but fish was the most common prey item in whales in deeper waters. They also reported that most prey items are open ocean, mesopelagic, or benthic organisms found in deep water.

MacLeod et al. (2003) reviewed the diets of three beaked whale genera (*Hyperoodon*, *Mesoplodon* and *Ziphius*). Their analyses showed that Cuvier's beaked whale stomachs contained cephalopods that were larger than those found in the other two species. Santos et al. (2001) found that Cuvier's beaked whale samples contained a larger range of species than that found in sperm whales or bottlenose whales in the northeast Atlantic. In their analyses of niche overlap by sperm whales, bottlenose whales, and Cuvier's beaked whales in the northern northeast Atlantic, Santos and Pierce (2005) found that *Gonatus sp*. and *Teuthowenia megalops* were the main prey of Cuvier's beaked whales. They also found that, of the three whale species, Cuvier's beaked whales fed on the widest size range of *Gonatus*.

2.9.8 Health

2.9.8.1 Strandings

A total of ten Cuvier's beaked whales stranded along the U.S. Atlantic coast from 2005 to 2009, as is summarized in Table 2.9-2. Georgia and Florida both reported the highest number of strandings (n=3 for both states). The whale that stranded in Georgia in 2005 had plastic debris in its stomach. The 2007 South Carolina whale showed evidence of a boat collision and the 2008 whale found in New Jersey had ingested fishing line.

Table. 2.9.2
Cuvier's Beaked Whale Strandings by Year and State

State	2005	2006	2007	2008	2009	Total
Massachusetts	0	1	0	0	0	1
New Jersey	1	0	0	1(1)	0	2
Georgia	1(1)	1	0	1	0	3
South Carolina	0	0	1(1)	0	0	1
Florida	0	0	2	1	0	3
Total	2	2	3	3	0	10

(The number of whales that showed evidence of human interaction is in parentheses.)

Podesta et al. (2005) analyzed Cuvier's beaked whale stranding data from the Mediterranean Sea. From these analyses they were able to identify geographic stranding areas.

2.9.8.2 Contaminants

Storelli et al. (1999) determined the concentration of heavy metals (Hg, Se, Cd, Pb and Cr) in a Cuvier's beaked whale that stranded on the Italian Coast in summer of 1996. Frodello et al. (2002) reported on total mercury, lead, cadmium, copper, and zinc that was measured in the organs (lung, liver, kidney, skin and bone) in a Cuvier's beaked whale that stranded in Corsica, France.

2.9.8.3 Disease

Several studies have focused on the existence of gas bubble lesions in stranded beaked whales (Fernandez et al. 2005; Hooker et al. 2009; Jepson et al. 2003). Jepson et al. (2003) and Fernandez et al. (2005) necropsied beaked whales (Cuvier's, Blainville's, Gervais), from a total of 14 animals that stranded in the Canary Islands in 2002 after being exposed to naval sonar activities. They found lesions related to bubble-associated tissue injury (similar to what is observed in decompression sickness) and fat embolisms. Rommel et al. (2006) reported that the most detrimental effect of sonar on beaked whales is the occurrence of gas bubbles which is related to an altered dive pattern or directly from ensonification. Hooker et al. (2009) modeled gas exchange and the effects of dive behavior and physiology in three beaked whale species (Cuvier's, Blainville's, and bottlenose whales). They reported that all three live with partial pressure nitrogen (PN2) tissue levels that would be toxic to terrestrial mammals. They also suggested that Cuvier's diving behavior may put them at particular risk of stranding after exposure to mid-frequency sonar.

2.9.9 Acoustics

Southall et al. (2007) classified *Ziphius* into a mid-frequency cetacean functional hearing group with an estimated auditory bandwidth of 150 Hz to 160 Hz. Cranford et al. (2008a) described Cuvier's beaked whale anatomy related to acoustic and diving function. Additionally, Cranford et al. (2008b) described sound pathways in Cuvier's beaked whales by using finite-element modeling (Vibro-acoustic Toolkit). Zimmer et al. (2005c) quantitatively described Cuvier's beaked whale clicks from the Ligurian Sea.

Zimmer et al. (2008) suggested that passive acoustic devices offer the only reliable way of detecting beaked whales given the difficulties associated with detecting them visually. Beaked whales appear to be particularly sensitive to underwater sonar. Numerous studies have sought to understand the relationship between anthropogenic activities and beaked whale mass strandings. The sound sources that have overlapped these stranding events are usually military mid-frequency sonar (2-10 kHz) and air gun arrays (used for geophysical exploration) (Barlow and Gisiner 2006). MacLeod and D'Amico (2006) reviewed five important areas of beaked whale ecology (social structure, life history, foraging/diving, acoustics and habitat) and how anthropogenic noise might impact each. D'Spain et al. (2005) described characteristics of the sound fields during multiple beaked whale mass strandings (Greece, Bahamas, Canary Islands). Similar features among all three events included water depth >1 km but close to land. Also, the noise source moved at speeds >5 knots and generated high amplitude pulses (1-10 kHz frequency band) (D'Spain et al. 2005).

D'Amico et al. (2009) analyzed 126 beaked whale mass stranding events that occurred between 1950 (time of introduction of powerful, mid-frequency sonar) and 2004 to elucidate the relationship between these events and naval exercises. They found that there was likely a high number of mid-frequency active sonar events worldwide but a relatively small number of beaked whale mass strandings. This led the authors to conclude that several risk factors must occur simultaneously in order for a mass stranding to occur.

Tyack et al. (2011) studied how whales responded to simulated (playback of simulated sonar, killer whale recordings and band-limited noise) and actual navy activity. They observed a disruption in foraging and a movement of whales out of the area (>10 km) in response to actual sonar activity. Their results suggest that beaked whales are more sensitive than other species to some anthropogenic noise and that a threshold of 140 dB SPL for behavioral response is more appropriate.

A Cuvier's beaked whale (tagged with an acoustic digital tag) exhibited an unusual foraging dive in conjunction with a large ship passing by (Aguilar Soto et al. 2006). Although a single observation, it led the authors to question the impact of ship noise on beaked whale behavior.

2.9.10 Fisheries By-Catch and Entanglement

In 2003, one undifferentiated beaked whale was seriously injured in the U.S. Atlantic pelagic long-line fishery. This event led to total annual fishery mortality of one for undifferentiated beaked from 2003-2007 (Waring et al. 2011).

2.9.11 Vessel Interactions

One 2007 South Carolina stranded Cuvier's beaked whale showed evidence of a boat collision. It should be noted that beaked whales do appear to be sensitive to sonar activities (see Acoustics section).

2.9.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear. Barlow and Gisiner (2006) reported that air gun activity (for geophysical exploration) has overlapped with beaked whale stranding events in space and time. These reports indicate that beaked whales are sensitive to anthropogenic noise and might be adversely impacted by energy projects.

2.9.13 Data Gaps and Research Recommendations

The low number of beaked whale sightings in U.S. waters has severely limited the information available about them. Most of the information (e.g., stock structure, life history traits, acoustics, food habits, or disease) comes from other parts of their range and often from stranded animals.

MacLeod and D'Amico (2006) suggested more cooperation between research groups and also using platforms of opportunity to fill in the data gaps that exist for beaked whales worldwide.

2.10 NORTHERN BOTTLENOSE WHALE (*HYPEROODON AMPULLATUS*)

2.10.1 Status

The northern bottlenose whale is not listed under the Endangered Species Act. Its status on the IUCN Red List is Data Deficient. It is afforded protection in U.S. waters under the MMPA. The Scotian shelf ("Gully") population is listed as endangered by the Committee on the Status of Endangered Wildlife in Canada.

2.10.2 General Distribution

Wimmer and Whitehead (2004) report that northern bottlenose whales are found in the North Atlantic from ice edge to the Azores. Reeves et al. (1993) report that there are two main areas of bottlenose whale distribution in the western north Atlantic: the Gully (east of Sable Island) and Davis Strait (off northern Labrador). Wimmer and Whitehead (2004) also sighted bottlenose whales in the Shortland (50 km east of Gully) and Haldimand (100 km east of Gully) canyons. Waring et al. (2010) reported sightings in 1993 (n=2) and 1996 (n=6) along the southern edge of Georges Bank. Mitchell and Kozicki (1975) reported strandings as far south as Rhode Island. Given these sighting records, bottlenose whales are likely to be found only in the BOEM North Atlantic Planning Area. Sighting and stranding locations of northern bottlenose whales are shown in figure 2.10-1.

2.10.3 General Abundance

Northern bottlenose whales were hunted from the 1850s to the 1970s, over 65,000 whales were reportedly killed, and many more were likely struck and lost (Reeves et al. 1993). Whitehead and Wimmer (2005) estimated the Gully population at 163 animals (95% CI 119-214). The number of northern bottlenose whales off the U.S. Atlantic coast is unknown (Waring et al. 2011).

2.10.4 Habitat Preference

Northern bottlenose whales are a deep water species that is rarely found in waters less than 2,000 m (Mead 1989b). They are most often associated with areas of high relief (e.g., submarine canyons or shelf edges) (Gowans 2009; Hooker et al. 2002b). They are common over a submarine canyon called "The Gully" approximately 300 km east of Halifax, Nova Scotia (Gowans 2009). In a long term study on distribution and individual range in the Gully, Hooker et al. (2002b) found changes in distribution to the north or south of the canyon which was likely due to shift in prey resources. Hooker et al. (2002a) provided information on the energy demands and ecological considerations of bottlenose whales in the Gully after this area was proposed as a marine protected area. The Gully was designated a marine protected area in May of 2004 by the Canadian Government (see http://www.dfo-mpo.gc.ca/oceans/marineareas-zonesmarines/mpa-zpm/atlantic-atlantique/factsheets-feuillets/gully-eng.htm).

2.10.5 Stock Structure

Whaling catch data suggested at least six centers of distribution that potentially represented different stocks: the Gully; Davis Strait; northern Iceland; off Andenes, Norway; off Møre, Norway; and Svalbard, Spitzbergen (Benjaminsen 1979). Gowans et al. (2001) studied the social organization of bottlenose whales in the Gully. They found that associations within age/sex classes were significantly higher than those between different classes but they did not find evidence of long-term female bonds.

Dalebout et al. (2001) analyzed a 434bp sequence of the mitochondrial control region in samples from the Gully (n=20 biopsy samples), Davis Strait (n=20 museum specimens) and northern Iceland (n=5 museum specimens). This analysis found very low genetic diversity among the 45 animals, suggesting a possible bottleneck event. They also found the distribution of mtDNA haplotypes suggested differentiation between the Gully and Davis Strait populations.

In a subsequent study, Dalebout et al. (2006) used both mtDNA control region sequences and 10 microsatellite markers to analyze samples from the Gully (n=34 biopsy samples) and Davis Strait (n=127; 124 museum specimens, 3 biopsy samples). This study further supported the separation of the Gully from the Davis Strait population. It also suggested that both males and females are philopatric. Finally, statistical analyses on the microsatellite data did not find evidence of a bottleneck. Stock structure is unknown for bottlenose whales in U.S. waters (Waring et al. 2011).

2.10.6 Life History Traits

Age at maturity is 7-11 years for males and 11 years for females (Benjaminsen and Christensen 1979). This data came from carcasses collected off Labrador. Gestation is at least 12 months and the oldest animal caught when they were still hunted was 37 years (Gowans 2009). In their study on the Gully population, Gowans et al. (2000) found that the sex ratio was approximately 1:1 and there were the same number of adult and sub-adult males in the population.

2.10.7 Food Habits

Hooker and Baird (1999) used time-depth recorder/VHF radio tags to learn about the diving behavior of bottlenose whales. They found that bottlenose whales dove to depths of 800 m or deeper and appeared to be foraging near or at the sea floor. Multiple sources report that the deep sea squid *Gonatus fabricii* is an important component of bottlenose whale diet throughout the North Atlantic (Benjaminsen and Christensen 1979; Clarke and Kristensen 1980; Lick and Piatkowski 1998). Hooker et al. (2001b) analyzed the stomach contents of two sub-adult males that stranded in eastern Canada. The squid *Gonatus steenstrupi* was the dominant prey species in both stomachs. In addition to these stomach content analyses, Hooker et al. (2001b) also analyzed fatty acid and stable isotope values from 21 whales off eastern Canada (n=3 stranded, n=18 biopsied in the Gully). The results of their study confirmed that squid of the genus *Gonatus* play a very important role in the food habits of northern bottlenose whales.

2.10.8 Health

2.10.8.1 Strandings

A mother-calf pair of bottlenose whales stranded in Delaware Bay in 2006. They first stranded in New Jersey and were returned to water. They subsequently re-stranded in Delaware; the mother was dead but the calf was returned to water again (Waring et al. 2011).

2.10.8.2 Contaminants

Hooker et al. (2008) analyzed contaminant concentrations and CYP1A protein expression in biopsy samples collected from bottlenose whales in the Gully and Davis Strait. These samples were collected before (1996-7) and after (2002-3) near oil and gas development sites. The contaminant levels (PCBs and OCs) found in these whales were similar to those in other odontocetes in the North Atlantic, although the Gully whales had higher concentrations than Davis Strait ones. CYP1A protein expression was higher in the 2003 Gully samples than in samples from previous years–this increase may be linked to oil spills related to exploration activities. Further, CYP1A protein expression was higher in Davis Strait animals.

2.10.8.3 Disease

Dagleigh et al. (2008) described the first case of fatal mycotic (*Aspergillus fumigates*) encephalitis in a juvenile male bottlenose whale that stranded along the coast of Scotland. They note this finding is important as mycotic infections can be indicative of immunosuppression. They also reported poxvirus-like lesions in the same animal for the first time in a bottlenose whale.

2.10.9 Acoustics

Southall et al. (2007) classified *Hyperoodon* into a mid-frequency cetacean functional hearing group with an estimated auditory bandwith of 150 Hz to 160 Hz. Winn et al. (1970) described a variety of sounds from an opportunistic recording of bottlenose whales in the Gully (e.g. single-pulse clicks, discrete-frequency whistles, sweep-frequency chirps). Hooker and Whitehead (2002) analyzed 428 minutes of bottlenose whale recordings from the Gully. They identified two types of clicks that differed in their amplitude. One type was heard at greater amplitude, in the form of “click trains” (speeding up and slowing down of the clicks). These were produced by whales at the surface that were socializing. The second clicks were low amplitude, regular clicks, likely produced by animals foraging in deep water.

2.10.10 Fisheries By-Catch and Entanglement

There are no records of fishery interactions for the most recent time period reported (2001-2005) in Waring et al. (2011).

2.10.11 Vessel Interactions

A review of Northeast and Southeast U.S. stranding data between 2005 and 2009 yielded no records of vessel interactions with bottlenose whales. Hooker et al. (2001a) found that whales did not avoid the research vessels from which researchers were making biopsy and tagging efforts; and often the whales would approach the vessel within minutes of a tag or biopsy attempt.

2.10.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear. Limited information is available about the impact of energy projects on northern bottlenose whales. For a discussion of CYP1A protein expression in relation to energy projects, see the Contaminants section above.

2.10.13 Data Gaps and Research Recommendations

The lack of northern bottlenose whale sightings in U.S. waters has severely limited the information available about them. Most of the information (e.g., stock structure, life history traits, acoustics, food habits, contaminant burden, or disease) on northern bottlenose whales in the western North Atlantic comes from outside U.S. waters in the Gully area in Canada.

2.11 BEAKED WHALES (*MESOPLODON SPP.*)

Four species of the genus *Mesoplodon* are found in the northwest Atlantic: True's beaked whale *M. mirus*; Gervais beaked whale *M. europaeus*; Blainville's beaked whale *M. densirostris*; and Sowerby's beaked whale *M. bidens* (Mead 1989a). As it is difficult to identify beaked whales to species at sea, much of the information available is only to genus level.

2.11.1 Status

None of the four species of beaked whales named above are listed under the Endangered Species Act. Their status on the IUCN Red List is Data Deficient. All *Mesoplodon* spp. are afforded protection in U.S. waters under the MMPA.

2.11.2 General Distribution

True's beaked whales are found from Cape Breton, Nova Scotia, to the Bahamas; Gervais' beaked whales are found from Cape Cod into the Gulf of Mexico; Blainville's beaked whales occur from southwest Nova Scotia to Florida; and Sowerby's beaked whales are reported from New England northward to the icepack (Leatherwood et al. 1976; Macleod et al. 2006a; Mead 1989a). Beaked whales can be found in all four of the BOEM Atlantic Planning Areas. See also MacLeod et al. (2006a) for the most recent information on the distribution of beaked whales worldwide. Sighting, stranding, and fishery bycatch locations of *Mesoplodon* beaked whales are shown in Figure 2.11-1.

2.11.3 General Abundance

Barlow et al. (2005) described the challenges (e.g., distribution, survey conditions, ability to identify whales to species) in obtaining solid abundance estimates of beaked whales. They suggested that passive acoustics could provide identification by vocalization–see Acoustics section for related studies. Data is not available to estimate the total number of beaked whales

(*Mesoplodon* spp.) along the U.S. Atlantic coast (Waring et al. 2011). The best estimate of undifferentiated beaked whales is 3,513 (CV=0.63) based on a combination of aerial and shipboard surveys that took place in 2004 from Florida to Maine (Waring et al. 2011). The estimate from Florida to Maryland is 674 (CV=0.36) and from Maryland to Bay of Fundy is 2,839 (CV=0.78).

2.11.4 Habitat Preference

Ferguson et al. (2006) used beaked whale sightings and environmental data to predict whale density in the Pacific Ocean. Their results for *Mesoplodon* spp. are summarized in Table 2.11-1.

Table 2.11.1.
Environmental Data for *Mesoplodon* spp.

Variable	Value
Beaufort	2.9
Distance offshore (km)	1052.0
Depth (m)	3513.6
Slope (°)	0.673
Sea Surface Temperature (°C)	26.3
Sea Surface Salinity (psu)	34
Surface Chlorophyll Concentration (mg/m^3)	0.255
Thermocline Depth (m)	56.2
Thermocline Strength (°/m)	0.391

MacLeod and Zuur (2005) analyzed fine scale habitat utilization by Blainville's beaked whales in the Bahamas. They found that aspect, gradient and depth were all related to occurrence. Claridge (2006) reported on vessel surveys that were conducted from 1997 to 2002 around Great Abaco Island (Bahamas). She found that Blainville's beaked whales were observed along the edge of the Great Bahama Canyon wall at a mean depth of 393 m (SD=283 m). She also found habitat partitioning between adult and sub-adult Blainville's whales with sub-adults occurring further offshore in deeper waters.

In the U.S. Atlantic, beaked whales (*Mesoplodon* spp.) have been sighted primarily along the shelf-edge, continental slope and deeper oceanic waters (Hamazaki 2002; Palka 2006; Tove 1995; Waring et al. 2001).

2.11.5 Stock Structure

Dalebout et al. (2004) created a genetic reference database for twenty beaked whale species. The data base includes a 437 bp segment of the mtDNA control region and a 384 bp cytochrome b segment. They compiled an additional database of nuclear DNA (925 bp actin intron sequences) for 17 of the 21 beaked whale species. Despite this study, Barlow et al. (2005) reported that almost nothing is known about beaked whale stock structure within species. Stock structure along the U.S. Atlantic coast is poorly understood (Waring et al. 2011).

2.11.6 Life History Traits

Limited information is available on the life history traits of beaked whales. Blainville's beaked whale age at sexual maturity is 9 years (Leatherwood and Reeves 1983a). Physically mature Gervais' beaked whales are 4.5-4.8 m long and their minimum longevity is estimated to be 27 years (Leatherwood and Reeves 1983a).

2.11.7 Food Habits

Santos et al. (1994) found mostly fish (whiting and sand eels) remains in a Sowerby's beaked whale that stranded in Scotland. MacLeod et al. (2003) reviewed the diets of three beaked whale genera (*Hyperoodon*, *Mesoplodon* and *Ziphius*). *Mesoplodon* spp. stomachs contained the most fish and the smallest prey compared to the other two genera.

Tyack et al. (2006) studied the dive behavior of three tagged Blainville's beaked whales. These tagged whales dove to depths between 222 and 1885 m and used echolocation to hunt their prey. Tyack et al. (2006) also estimated the whales targeted 30 individual prey per dive. Madsen et al. (2005) described the biosonar performance of beaked whales that were foraging using DTAGs. See the Acoustic section for other studies on how Blainville's beaked whales use echolocation to forage.

2.11.8 Health

2.11.8.1 Strandings

The most common species of *Mesoplodon* to strand along the U.S. Atlantic coast from 2005 to 2009 was the Gervais' beaked whale (n=12, 48%). Beaked whales appear to be very sensitive to underwater, anthropogenic noise; several mass strandings worldwide have overlapped in space and time with military activity. Often these stranded whales have gas emboli likely caused by decompression (Fernandez et al. 2005; Jepson et al. 2003). Tyack et al. (2006) studied deep diving behavior in beaked whales and suggest that these decompression problems are likely due to a behavioral response to sonar. See the Disease and the Acoustics sections below for details.

Table 2.11.1
Blainville's Beaked Whale Strandings by Year and State

State	2005	2006	2007	2008	2009	Total
North Carolina	1(1)	1	1	1	0	4
South Carolina	1	0	1(1)	0	0	2
Total	2	1	2	1	0	6

(The number of whales that showed evidence of human interaction is in parentheses; data from the NE and SE stranding network databases)

Table 2.11.2
Gervais' Beaked Whale Strandings by Year and State

State	2005	2006	2007	2008	2009	Total
New Jersey	0	0	0	1	0	1
Virginia	0	0	1	0	1	2
Maryland	0	0	0	0	1	1
North Carolina	2	0	0	0	1	3
Florida	0	0	1	2	2	5
Total	2	0	2	3	5	12

(The number of whales that showed evidence of human interaction is in parentheses; data from the NE and SE stranding network databases)

Table 2.11.3
Sowerby's Beaked Whale Strandings by Year and State

State	2005	2006	2007	2008	2009	Total
Virginia	0	0	0	0	2	2

(The number of whales that showed evidence of human interaction is in parenthese; data from the NE and SE stranding network databases)

Table 2.11.4
True's Beaked Whale Strandings by Year and State

State	2005	2006	2007	2008	2009	Total
New Jersey	0	0	1	0	0	1
New York	0	0	1	0	0	1
Total	0	0	2	0	0	2

(The number of whales that showed evidence of human interaction is in parentheses; data from the NE and SE stranding network databases)

Table 2.11.5
Unidentified Beaked Whale Strandings by Year and State

State	2005	2006	2007	2008	2009	Total
Rhode Island	0	0	1	0	0	1
North Carolina	0	0	1	0	0	1
Florida	1	0	0	0	0	1
Total	1	0	2	0	0	3

(Data from the NE and SE stranding network databases)

2.11.8.2 Contaminants

No information was found on contaminant levels in beaked whales.

2.11.8.3 Disease

Several studies have focused on the existence of gas bubble lesions in stranded beaked whales (Fernandez et al. 2005; Hooker et al. 2009; Jepson et al. 2003). Jepson et al. (2003) and Fernandez et al. (2005) necropsied beaked whales (Cuvier's, Blainville's, Gervais), that stranded in the Canary Islands in 2002 after being exposed to naval sonar activities. They found lesions related to bubble-associated tissue injury (similar to what is observed in decompression sickness) and fat embolisms. Hooker et al. (2009) modeled gas exchange and the effects of dive behavior and physiology in three beaked whale species (Cuvier's, Blainville's, and bottlenose whales). They reported that all three species live with PN2 tissue levels that would be toxic to terrestrial mammals. In their review of the impact of anthropogenic noise on beaked whales, Cox et al. (2006) concluded that gas-bubble disease (resulting from behavioral response to the noise) is a plausible explanation for the morbidity/mortality observed in beaked whale mass stranding events. Likewise, Rommel et al. (2006) reported that the most detrimental effect of sonar on beaked whales is the occurrence of gas bubbles which is related to an altered dive pattern or directly from ensonification.

2.11.9 Acoustics

Southall et al. (2007) classified *Mesoplodon* into a mid-frequency cetacean functional hearing group with an estimated auditory bandwith of 150 Hz to 160 Hz. Cook et al. (2006) used auditory evoked potentials to measure the hearing abilities of a juvenile, stranded Gervais' beaked whale. Rankin and Barlow (2007a) described four mid-frequency sounds that were recorded near Blainville's beaked whales in Hawaii.

Several studies on foraging Blainville's beaked whales have provided additional acoustic information. Jones et al. (2008) reported that Blainville's beaked whales use broadband, ultrasonic echolocation signals (-10 dB bandwidth from 26-51 kHz) to search for and catch their prey. Johnson et al. (2006) describe two clicks types that occur at different times of foraging. The first is "search" clicks (-10 dB bandwidth from 26-51 kHz) occur during prey location and "buzz" clicks (-10dB bandwidth from 25-80 kHz) that occur during capture. Johnson et al. (2008) suggested that these whales can adjust their echolocation behavior and movement in order to catch different types of prey.

Zimmer et al. (2008) suggested that passive acoustic devices offer the only reliable way of detecting beaked whales given the difficulties associated with detecting them visually. Advances in detection and localization of Blainville's beaked whale vocalizations in the Bahamas using a combination of passive, fixed, acoustic sensors and digital tags (DTags) (Moretti et al. 2006; Ward et al. 2008) has led to the development of acoustically-derived estimates of density (Marques et al. 2009; Moretti et al. 2006). Beaked whales appear to be particularly sensitive to underwater sonar. Numerous studies have sought to understand the relationship between anthropogenic activities and beaked whale mass strandings. The sound sources that have overlapped these stranding events are usually military mid-frequency sonar (2-10 kHz) and air gun arrays (used for geophysical exploration) (Barlow and Gisiner 2006). MacLeod and D'Amico (2006) reviewed five important areas of beaked whale ecology (social structure, life history, foraging/diving, acoustics and habitat) and how anthropogenic noise might impact each. D'Spain et al. (2005) described characteristics of the sound fields during multiple beaked whale mass strandings (Greece, Bahamas, Canary Islands). Similar features among all three events included water depth >1 km but close to land. Also, the noise source moved at speeds >5 knots and generated high amplitude pulses (1-10 kHz frequency band) (D'Spain et al. 2005).

D'Amico et al. (2009) analyzed 126 beaked whale mass stranding events that occurred between 1950 (time of introduction of powerful, mid-frequency sonar) and 2004 to elucidate the relationship between these events and naval exercises. They found that there was likely a high number of mid-frequency active sonar events worldwide but a relatively small number of beaked whale mass strandings. This led the authors to conclude that several risk factors must occur simultaneously in order for a mass stranding to occur; quantification of possible contributing factors (physical and biological) is necessary. They did find that most beaked whale strandings were less than 80 km from 1,000 m depth line.

Tyack et al. (2011) studied how whales responded to simulated (playback of simulated sonar, killer whale recordings and band-limited noise) and actual navy activity. They observed a disruption in foraging and a movement of whales out of the area (>10 km) in response to actual sonar activity. Their results suggest that beaked whales are more sensitive than other species to some anthropogenic noise and that a threshold of 140 dB SPL for behavioral response is more appropriate.

2.11.10 Fisheries By-Catch and Entanglement

In 2003 one undifferentiated beaked whale was seriously injured in the U.S. Atlantic pelagic longline fishery. This incident led to an estimated annual average fishery-related mortality/serious injury of 1 (CV=1.0) for 2003-2007, the most recent time period available (Waring et al. 2011).

2.11.11 Vessel Interactions

A review of Northeast and Southeast U.S. stranding records between 2005 and 2009 yielded no records of vessel interactions with beaked whales. It should be noted that beaked whales do appear to be sensitive to sonar activities (see the Acoustics section above).

2.11.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear. Barlow and Gisiner (2006) reported that air gun activity (for geophysical exploration) has overlapped in space and time with beaked whale stranding events. These reports indicate that beaked whales are sensitive to anthropogenic noise and might be adversely impacted by energy projects.

2.11.13 Data Gaps and Research Recommendations

The low number of beaked whale sightings in U.S. waters has severely limited the information available about them. Most of the information (e.g., stock structure, life history traits, acoustics, food habits, or disease) comes from other parts of their range and often from stranded animals. MacLeod and D'Amico (2006) suggested more cooperation between research groups and also using platforms of opportunity to fill in the data gaps that exist for beaked whales worldwide.

2.12 KILLER WHALE (*ORCINUS ORCA*)

2.12.1 Status

The killer whale is not listed under the Endangered Species Act. The species is considered "Data Deficient" on the IUCN Red List. It is afforded protection in U.S. waters under the MMPA.

2.12.2 General Distribution

Killer whales are truly cosmopolitan and are found in all of the world's oceans and most of the seas (Ford 2009). Leatherwood and Reeves (1983a) wrote that the only limits to killer whale distribution appear to be ice cover, prey and human hunting. Although killer whales are rare in the U.S. Atlantic Exclusive Economic Zone, Katona et al. (1988) suggested the possibility of a small, migrating population along the U.S. Atlantic coast and a year-round population south of 35°N. Given their cosmopolitan nature and widespread distribution, killer whales could potentially be observed in any of the four BOEM Atlantic Planning Areas. Sighting, stranding and fishery bycatch locations of killer whales are shown in Figure 2.12-1.

2.12.3 General Abundance

The killer whale's widespread distribution and scarceness in most areas make obtaining abundance estimates difficult (Ford (2009). The number of killer whales off the U.S. Atlantic coast is unknown. The available data are not adequate to calculate a minimum estimate (Waring et al. 2011).

2.12.4 Habitat Preference

Killer whales are most abundant in coastal habitats at high latitudes but can also be observed in tropical areas and open seas (Dahlheim and Heyning 1999). They are most abundant within 800 km of the continental coastline (Leatherwood and Reeves (1983a). In their analysis of whaling log books from the North Atlantic, Reeves and Mitchell (1988) found that 80% of killer whale observations were made on whaling grounds for sperm whales. These whaling logbooks also documented widespread offshore observations of killer whales in the North Atlantic.

2.12.5 Stock Structure

Stock structure for Western North Atlantic killer whales is unknown (Waring et al. 2011). Hoelzel et al. (2002) found low worldwide diversity in their analysis of mtDNA control region sequences and 7 microsatellite loci. They suggested that the lack of regional diversity is due to the matrilineal nature of killer whales and that the worldwide pattern might be due to a bottleneck. However, in their 2010 paper, Morin et al. (2010) used high-throughput sequencing to survey the entire mitochondrial genome. They suggested that the 3 named ecotypes (resident, transient and offshore) of killer whales be considered separate species and that two other types be considered subspecies. Dispersal from the natal pod is rare and gene flow between local populations is male-mediated (Pilot et al. 2010). Foote et al. (2009) used nitrogen isotopes, tooth wear and mtDNA to demonstrate ecological divergence of killer whales in the North Atlantic. Foote et al. (2010) analyzed 17 microsatellite loci and part of the mitochondrial control region in killer whales from the North Atlantic. They found three significantly differentiated populations.

2.12.6 Life History Traits

As summarized in Ford (2009), females give birth to their first calf at 12-14 years of age. The interval between viable calves is approximately 5 years and gestation is estimated to be 15-18 months. Average female life expectancy is 50 years but some individuals may live to 80-90 years of age. Killer whale females have a post-reproductive period. Males become sexually mature at approximately 15 years of age and their average life expectancy is 30 years with a maximum longevity of 50-60 years. Most of this information has come from the resident killer whale populations off the coast of Washington (U.S.) and British Columbia (Canada).

2.12.7 Food Habits

Over 140 prey species (e.g., mysticetes, odontocetes, pinnipeds, fish, and invertebrates) have been recorded for killer whales (Ford 2009). Although as a species they are considered generalists, certain localized populations have specialized feeding (Ford 2009). Ford et al. (2010) found that, although killer whales may consume a variety of species, the fish-eating populations they studied in the northeast Pacific are highly specialized and dependent on a single species of salmon. Katona et al. (1988) reported the following prey species for killer whales found in the western North Atlantic (Bay of Fundy to the Equator): finback, humpback, minke, and pilot whales, bluefin tuna, mackerel, squid, herring and sea turtles. Mehta et al. (2007) analyzed scars on baleen whales in high latitude regions. Their results implied that most killer whale attacks target young baleen whales (likely on their first trip from calving to feeding grounds) and that adult baleen whales are not an important food source for killer whales.

2.12.8 Health

2.12.8.1 Strandings

There are no records of killer whale strandings along the U.S. Atlantic coast (Waring et al. 2011).

2.12.8.2 Contaminants

Ford (2009) noted that because of their high place in the food web, killer whales are at risk for bioaccumulation of contaminants. Ikonomou et al. (2007) showed that accurate PCBs, PCDD/Fs and PBDEs levels could be obtained in killer whales using the relatively non-invasive microdart technique. Transient killer whales showed higher concentrations of organochlorines than resident killer whales in Kenai Fjords/Prince William Sound, AK (Ylitalo et al. 2001). This difference is likely due to the differences in food habits. Ylitalo et al. (2001) also found that female killer whale OC levels were lower than immature or adult males likely due to offloading through gestation and lactation. Biopsy samples from southern resident killer whales in Alaska were analyzed by Krahn et al. (2007) for persistent organic pollutants and stable isotopes. Rayne et al. (2004) analyzed samples from 3 groups in the northeastern Pacific. Their results suggest that PBDEs may be a contaminant of concern for these killer whale populations. Ross et al. (2000) analyzed blubber samples collected from both transient and resident killer whales off the coast of British Columbia, Canada. Their results indicate that these populations are among the most contaminated cetaceans. Although there is not a toxic threshold for PCBs in cetaceans, the levels in these whales greatly exceeds the adverse effect level established for harbor seals. McHugh et al. (2007) analyzed organochlorine and enantiomeric profiles in blubber samples collected from killer whales in British and Irish waters.

2.12.8.3 Disease

Bossart et al. (1996) documented the first case of cutaneous papillomaviral-like papillomatosis in a captive ~10-year old killer whale. In their literature review, Gaydos et al. (2003) identified 15 infectious agents (bacteria, viruses, or fungi) reported for free and captive killer whales. They were also able to identify another 28 agents that have been reported for sympatric odontocete species. Gaydos et al. (2003) suggested that *Brucella* spp., cetacean poxvirus, cetacean morbilliviruses, and herpesviruses should be studied further given their potential impact on killer whale populations.

2.12.9 Acoustics

Killer whales are vocal and produce a variety of clicks, whistles and pulsed calls for echolocation and communication purposes (Ford 2009). Southall et al. (2007) classified *Orcinus* sp. into a mid-frequency cetacean functional hearing group with an estimated auditory bandwidth of 150 kHz to 160 kHz. Szymanski et al. (1999) measured the audiograms of two trained, adult female killer whales using behavioral responses and auditory evoked potentials (AEPs). They found the most sensitive frequency was 20 kHz (36 dB) in the mean audiogram. In their analysis of acoustic behavior and surface activity, Simon et al. (2007a) found that feeding whales produced high rates of clicks and calls. Multiple studies by Simon et al. (Simon et al. 2006, 2007b) described the behavior and acoustics of Icelandic and Norwegian killer whales feeding on herring (*Clupea harrengus*). Tyson et al. (2007) documented nonlinearities (e.g., subharmonics, deterministic chaos, biphonation, and frequency jumps) in killer whale vocalizations. In their study on the acoustic behavior of killer whales observed carousel feeding, Van Opzeeland et al. (2005) found that the sequence of call types was more important than the use of isolated call types for the coordination of the group.

2.12.10 Fisheries By-Catch and Entanglement

There are no records of fishery interactions for the most recent time period reported (2001-2005) (Waring et al. 2011). Prior to that time, one killer whale was caught in the New England multispecies sink gillnet in 1994. The whale was released alive.

2.12.11 Vessel Interactions

Erbe (2002) documented the impacts from the underwater noise of whale-watching boats in southern British Columbia and northwestern Washington State (popular areas for killer whale watching). A software sound propagation and impact assessment model was used to estimate zones around the boats where killer whales could hear the boat noise, where the noise affected whale communication, where it caused behavioral avoidance, and where it possibly caused temporary threshold shift (TTS). Erbe (2002) suggests that these data could be used to develop whale-watching guidelines. Holt et al. (2009) studied the effects of anthropogenic sound exposure on the vocal behavior of free-ranging killer whales in Puget Sound, Washington, where there is a high level of motorized boat traffic. Their results showed that these killer whales increased their call amplitude by 1 dB for every 1 dB increase in background noise levels.

In the San Juan Islands, U.S., and Gulf Islands, Canada, Noren et al. (2009) documented killer whale surface active behaviors (SABs) in response to closely approaching vessels. Their study suggested that the minimum approach distance of 100 m in whale-watching guidelines may not be enough to prevent a behavioral response from whales.

Williams et al. (2002) studied the effect of "leap-frogging" (when whale-watch boats purposely cut off a whale in transit) on the behavior of 10 male killer whales in British Columbia. The whales responded to leap-frogging by adopting paths less straight and smooth compared to control conditions. The authors suggested that this change in path is likely due to the animals trying to avoid the boats and may have energetic costs. In another study on the impact of whale-watching boats in British Columbia, Williams et al. (2006b) found that in the presence of the boats, time spent foraging decreased, which could increase energetic costs.

2.12.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear. Stone (2003) studied the effects of seismic activity on cetaceans in UK waters. She reported that killer whales showed some avoidance of seismic activity.

2.12.13 Data Gaps and Research Recommendations

There is no abundance estimate for killer whales in the U.S Atlantic. Most of the information about the killer whales' life history traits, acoustics, food habits, contaminant burden, or disease comes from parts of their range that are outside of the U.S. Atlantic coast (e.g., from the Pacific Ocean).

2.13 LONG-FINNED PILOT WHALES (*GLOBICEPHALA MELAS*)

There are two species of pilot whales in the western Atlantic: the long-finned pilot whales and short-finned pilot whales. It is difficult to differentiate the two species at sea; therefore some of the information presented here is for *Glopicephala spp*.

2.13.1 Status

The long-fined pilot whale is not listed under the Endangered Species Act. The species is considered "Data Deficient" on the IUCN Red List. It is afforded protection in U.S. waters under the MMPA.

2.13.2 General Distribution

Long-finned pilot whales are found in the cold temperate waters of the North Atlantic (Olson 2009). As described in Waring et al. (2010), pilot whales off the U.S. Atlantic coast are found primarily along the continental shelf edge in the winter and early spring. They move onto Georges Bank and into the Gulf of Maine for the summer and much of fall. Their distribution is encompassed by the BOEM North and Mid-Atlantic Planning Areas. Sighting, stranding, and fishery bycatch locations of long-finned and undifferentiated pilot whales are shown in Figure 2.13-1.

2.13.3 General Abundance

The best estimate of long-finned pilot whales is 12,619 (CV=0.37) based on a combination of aerial and shipboard surveys that took place in 2004 and 2006 (Waring et al. 2011). An analysis of spatial distribution based on genetic studies and environmental variables such as SST and water depth was used to assign pilot whales to species where short and long-finned pilot whales overlap (Waring et al. 2011).

2.13.4 Habitat Preference

Off the U.S. Atlantic coast, short- and long-finned pilot whales are often found over areas of high relief (Waring et al. 2011) and are associated with the Gulf Stream wall and thermal fronts along the shelf edge (Waring et al. 1992). In his analysis of interaction of marine mammals and pelagic longline fishing gear in the U.S. Atlantic, Garrison (2007c) found that pilot whale interactions were correlated with warm water temperature and proximity to shelf break. Hamazaki (2002) used oceanographic and topographic factors to create a habitat prediction model for cetaceans in the western North Atlantic. He placed pilot whales (*Globicephala spp*.) in the Mid-Atlantic shelf category and noted that pilot whales tended to be in cool deep water. These studies analyzed two separate data sets and likely reflect differences between short and long-finned pilot whales.

2.13.5 Stock Structure

Pilot whales are social and swim in pods formed along the matriline (Amos et al. 1993a). Oremus et al. (2009) analyzed the mtDNA of both long-finned and short-finned pilot whales worldwide. They found that both species had low levels of genetic diversity compared to other widespread cetaceans. They did find structuring between oceanic basins and at the regional level. Fullard et al. (2000) used 8 highly polymorphic microsatellite loci to analyze the population structure of long-finned pilot whales in the North Atlantic. Their samples came from four sites: Cape Cod, U.S.; West Greenland; the Faeroe Islands; and the UK They documented substructure and suggested that SST plays a role in population isolation. The Greenland samples were the most different.

Andersen and Siegismund (1994) used three allozyme loci to assess paternity of fetuses caught in the Faroe Islands. Fathers of the fetuses were genetically different from males in the pods indicating males move into other pods for mating.

2.13.6 Life History Traits

As summarized in Olson (2009), females become sexually mature at 8 years and males at 12 years (although successful mating may not happen for several years after that). In the Northern Hemisphere, mating appears to occur in spring or early summer and calving in summer or fall. Lactation lasts for three years or longer. Although post reproductive females have been documented in pods of short-finned pilot whales, they have not been observed in long-finned pilot whales (Foote 2008).

2.13.7 Food Habits

Pilot whales mostly feed on squid (Olson 2009). Only cephalods were found in the stomach contents from stranded long-finned pilot whales in New Zealand (Beatson and Ogle 2007). In correlating whale occurrence and prey distribution (recorded acoustically), Doksaeter et al. (2008) found that cephalopods (especially *Gonatus* sp.) and Glacier lantern fish (*Benthosema glaciale*) were likely the most important prey species for long-finned pilot whales feeding along the Mid-Atlantic Ridge between Iceland and the Azores.

In their analysis of stable isotope ratios, Abend and Smith (1997), identified Atlantic mackerel (*Scomber scombrus*) as another important prey species for long-finned pilot whales in the western North Atlantic. Long-finned squid (*Loligo pealei*) was the most important prey species in the stomachs of long-finned pilot whales stranded (n=8) along the U.S. Mid-Atlantic and in whales (n=30) inadvertently caught in the Distant Water Fleet (DWF) mackerel fishery of the northeast U.S. (Gannon et al. 1997a, 1997b).

2.13.8 Health

2.13.8.1 Strandings

There are records of pilot whale mass strandings from the U.S. (Massachusetts, Florida, and North Carolina), the Faroe Islands, Ireland, Tasmania, Western Australia, and New Zealand. As described in Olson (2009), pilot whales tend to mass strand more frequently than most other cetacean species. The whales involved in mass strandings often do not show any sign of disease. There is a wide variety of hypotheses (e.g., animals become stuck in shallow water, geomagnetic abnormalities may confuse the whales, healthy animals follow a sick one ashore) but why healthy animals strand is not currently understood. In some instances, human activities such as sonar have overlapped pilot whales strandings in space and time (e.g., North Carolina). However, there are records of mass pilot whale strandings long before the invention of sonar (e.g. Cape Cod 1902, n=~100; New Zealand 1918, n=~1000). Along the U.S. Atlantic coast, Cape Cod appears to be a hot spot for long-finned pilot whale mass stranding events.

Table 2.13.1 summarizes the number of strandings of both long-finned pilot whales (LF) and pilot whales only identified to genus (Spp). Massachusetts had the highest number of stranded long-finned pilot whales during 2005-2009 (n=35, 51%). The stranded whale in Maine (2007) was released alive. Three of the stranded long-finned pilot whales showed signs of fishery interactions.

Table 2.13.1 Long-finned Pilot Whale Strandings by State and Year

State	2005		2006		2007		2008		2009		Total	
	LF	Spp	LF	Spp	LF	Spp	LF	Spp	Lf	Spp	LF	Spp
Maine	2	0	1	0	1	0	1	1	3	0	5	1
New Hampshire	0	0	1	0	0	0	0	0	0	0	1	0
Massachusetts	22	0	2	0	6	0	1	1	4	0	35	1
Rhode Island	0	0	0	0	0	0	2	0	2	0	4	0
New York	1	0	0	0	2	0	2	0	1	0	6	0
New Jersey	0	2	0	0	1	0	1	0	1	0	3	2
Maryland	4	0	0	0	0	0	0	0	0	0	4	0
Virginia	4	0	2	0	0	0	0	0	0	0	6	0
North Carolina	1	2	0	1	0	0	0	1	0	0	1	4
South Carolina	0	0	0	0	0	0	0	1	0	0	0	1
Total	35	4	6	1	10	0	7	4	11	0	69	9

(Data from the NE and SE marine mammal stranding response databases.)

2.13.8.2 Contaminants

Lindstroem et al. (1999) identified 19 tetra- to hexabromindated diphenyl ethers in the blubber of pilot whales caught in the Faroe Islands (1994 and 1996). They found that younger whales (pooled male and female samples) had higher concentrations of PDBEs than older whales (males or females). Muir et al. (1988) identified and measured organochlorines and heavy metals in tissue samples from pilot whales (n=41) stranded along the coast of Newfoundland, Canada. Age was correlated with mercury levels in liver and blubber, and Cadmium (Cd) in kidney. Cadmium levels were higher in these pilot whales than in other whale species from this area. Trace elements in the liver and kidney of pilot whales (from 7 different pods) caught around the Faroe Islands were measured by Caurant et al. (1994). They found that most metal concentrations were higher in females than males. This may be related to differences in feeding or metabolism (including hormones) between the two genders. Males are larger than females, so there may also be a dilution effect. Contaminant levels also appear to differ among the various schools tested.

Borrell et al. (1995) investigated reproductive transfer of organochlorine compounds (DDT, PCB) in long-finned pilot whales. Their study showed that mothers transfer 60-100% of their body load via lactation and only 4-10% during gestation. Tissues from long-finned pilot whales that stranded on the Massachusetts coast were analyzed for orchanochlorines by Tilbury et al. (1999). They found evidence of maternal transfer; that contaminant levels varied among animals; and that there was significant difference in certain analytes between males and females (probably due to maternal offloading through gestation and lactation). Alternatively, Weisbrod et al. (2001) measured organochlorines in pilot whales from the Gulf of Maine and found no gender difference in bioaccumulation.

2.13.8.3 Disease

Duigan et al. (1995a) looked for evidence of morbillivirus infection in long-finned pilot whales from the western Atlantic (stranded n=100, by-caught n=1). They found neutralizing antibody titers in 92% of the animals. The earliest sign of infection was found in a sample from an animal that stranded in 1982. The authors suggest that the social nature of the species, population size and migration patterns might facilitate infection.

2.13.9 Acoustics

Southall et al. (2007) classified *Globicephala* sp. into a mid-frequency cetacean functional hearing group with an estimated auditory bandwith of 150 kHz to 160 kHz. Weilgart and Whitehead (1990) described behaviors and related vocalizations made by long-finned pilot whales. Simple whistles were more often heard with resting behavior; whales that were active (possible feeding) made more complex whistles and pulsed sounds with greater frequency. In their analysis of a rehabilitated, stranded long-finned pilot whale, Pacini et al. (2010) found that the region of best hearing was between 11.2 and 50 kHz. The two-year old male had poor high frequency hearing compared to other odontocetes.

Rendell et al. (1999) analyzed tonal calls from five odontocete species. They found that long-finned and short-finned pilot whale calls were distinct. The mean frequency of long-finned pilot whales was 4,480 Hz (CV=0.52). Rendell and Gordon (1999) documented the reactions of a pod of long-finned pilot whales to military sonar in the Ligurian Sea.

2.13.10 Fisheries By-Catch and Entanglement

Total human-caused long-finned pilot whale mortality along the U.S. cannot be estimated. Historically, pilot whales were incidentally caught off the U.S. Atlantic coast in the Distant Water Fleet and other foreign fishing activities (Fairfield et al. 1993; Waring et al. 1990). Fairfield et al. (1993) described a high rate of takes in the DWF in 1988, which they attributed to a geographic concentration in fishing effort. Fertl and Leatherwood (1997) describe multiple observations of pilot whales actively feeding in and around trawl nets during haul back. Most incidental by-catch occurs along the mid-Atlantic coast where the two species of pilot whales overlap; this makes it difficult to assign the mortality to one species or the other (Waring et al. 2011). Total annual estimated average fishery-related mortality for 2005-2009 was 162 pilot whales (*Globicephala spp.*). The pelagic long-line, northeast midwater trawl and northeast groundfish fisheries have the strongest impact on pilot whales.

Table 2.13.2 Mean Annual Mortality of Undifferentiated Pilot Whales by Fishery 2005-2009

Fishery	Mean Annual Mortality
Mid-Atlantic Bottom Trawl	30 (0.16)
Northeast Bottom Trawl	12 (0.14)
Mid-Atlantic Mid-water Trawl	2.4 (0.99)
Northeast Mid-water Trawl	3.0 (0.61)
Pelagic Longline	114 (0.20)
2005 Pelagic Longline experimental fishery	1 (1.0)

(CV in parentheses) (Waring et al. 2011)

2.13.11 Vessel Interactions

A review of Northeast and Southeast U.S. stranding records between 2005 and 2009 yielded no records of vessel interactions with long-finned pilot whales.

2.13.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear. Stone (2003) studied the effects of seismic activity on cetaceans in UK waters. She reported that sighting rates of white-sided dolphins, white-beaked dolphins, *Lagenorhynchus* spp., all small odontocetes combined, and all cetaceans combined were significantly lower when large airgun arrays were being fired. The 2010 Report of the International Council for the Exploration of the Sea (ICES) Working Group on Marine Mammal Ecology (ICES 2010) stated that very little is known about the impact of the construction and operation of offshore wind-farms. Madsen et al. (2006b) reviewed available current information on wind farms and marine mammals and wrote that the noise impact is more severe during construction than operation. Weir (2008b) studied the reaction of long-finned pilot whales to an air-gun ramp up procedure off Gabon. She documented an avoidance response which was limited in time and space.

2.13.13 Data Gaps and Research Recommendations

The best abundance estimate is from aerial and ship surveys that took place in 2004–this data is 7 years old. There is very little recent information on long-finned pilot whale distribution and abundance outside of the summer months (June-September). Current research will continue to elucidate pilot whale stock structure in the northwest Atlantic.

2.14 SHORT-FINNED PILOT WHALE (*GLOBICEPHALA MACRORHYNCHUS*)

There are two species of pilot whales in the western Atlantic: the long-finned pilot whales and short-finned pilot whales. It is difficult to differentiate the two species at sea; therefore, some of the information presented here is for *Globicephala* spp.

2.14.1 Status

The short-fined pilot whale is not listed under the Endangered Species Act. The species is considered "Data Deficient" on the IUCN Red List. It is afforded protection in U.S. waters under the MMPA.

2.14.2 General Distribution

Short-finned pilot whales are found in tropical, subtropical and warm temperate waters throughout the world (Olson 2009). In the U.S. Atlantic, they are found from Florida to the Mid-Atlantic coast (Olson 2009). They overlap with long-finned pilot whales between Cape Hatteras (NC) and New Jersey (Waring et al. 2011). Short-finned pilot whale distribution along the U.S. Atlantic coast encompasses the BOEM North Atlantic (probably only the southern edge), Mid-Atlantic, South Atlantic and Straits of Florida Planning Areas. Sighting, stranding and fishery bycatch locations of short-finned and undifferentiated pilot whales are shown in Figure 2.14-1.

2.14.3 General Abundance

The best estimate of short-finned pilot whales is 24,674 (CV=0.45), based on a combination of aerial and shipboard surveys that took place in 2004 (Waring et al. 2011). An analysis of spatial distribution based on genetic studies and environmental variables such as SST and water depth was used to assign pilot whales to species where short and long-finned pilot whales overlap (Waring et al. 2011).

2.14.4 Habitat Preference

Off the U.S. Atlantic coast, pilot whales (*Glopicephala* spp.) are often found over areas of high relief (Waring et al. 2011) and are associated with the Gulf Stream wall and thermal fronts along the shelf edge (Waring et al. 1992). In his analysis of interactions between marine mammals and pelagic longline fishing gear in the U.S. Atlantic, Garrison (2007c) found that pilot whale interactions were correlated with warm water temperature and proximity to shelf break. Hamazaki (2002) used oceanographic and topographic factors to create a habitat prediction model for cetaceans in the western North Atlantic. He placed pilot whales (*Globicephala* spp.) in the Mid-Atlantic shelf category and noted that pilot whales tended to be in cool deep water. These studies analyzed two separate data sets and likely reflect differences between short and long-finned pilot whales.

2.14.5 Stock Structure

Pilot whales are social and swim in pods formed along the matriline (Amos et al. 1993a). Oremus et al. (2009) analyzed the mtDNA of both long-finned and short-finned pilot whales worldwide. They found that both species had low levels of genetic diversity compared to other widespread cetaceans. They did find structuring between oceanic basins and at the regional level. Analyses are currently underway at the Southeast Fisheries Science Center to elucidate the genetic relationship between short-finned pilot whales in the Western Atlantic, Gulf of Mexico and Caribbean Sea (Waring et al. 2011). Pending those results, short-finned pilot whales along the U.S. Atlantic coast are managed as a separate stock from the Gulf of Mexico and Caribbean.

2.14.6 Life History Traits

As summarized in Olson (2009), females become sexually mature at 9 years and males at 13-16 years (although successful mating may not happen for several years after that). In the Northern Hemisphere, mating appears to occur in spring or early summer and calving in summer or fall. Gestation is estimated to be 15-16 months in short-finned pilot whales. Lactation lasts for three years or longer. It has also been reported that female short-finned pilot whale can lactate after final ovulation. Complete reproductive senescence and a long post-reproductive life have been documented in short-finned pilot whales (Foote 2008; Olson 2009).

2.14.7 Food Habits

Pilot whales mostly feed on squid (Olson 2009). Overholtz and Waring (1991) analyzed the stomach contents of pilot whales (*Globicephala* spp.) caught incidentally in the Atlantic mackerel trawl operating off the mid-Atlantic U.S. east coast in 1989. They found that, on average, Atlantic mackerel was 71% of the wet weight of stomach and the remaining 29% was long-finned squid. Mintzer et al. (2008) analyzed the stomachs of 27 short-finned pilot whales that stranded on the North Carolina coast in January of 2005. They assessed prey importance by using two indices: frequency of occurrence and numerical abundance. These analyses showed that the most important species was *Brachioteuthis riisei*, followed by *Taonius pavo* and *Histioteuthis reversa* (all cephalopods). The most important fish species was *Scopelogadus beanii*. The presence of these prey species indicate these whales had likely been feeding off the continental shelf before they stranded.

2.14.8 Health

2.14.8.1 Strandings

There are records of pilot whale mass strandings from the U.S. (Massachusetts, Florida, and North Carolina), the Faroe Islands, Ireland, Tasmania, Western Australia, and New Zealand. As described in Olson (2009), pilot whales tend to mass strand more frequently than most other cetacean species. The whales involved in mass strandings often do not show any sign of disease. There is a wide variety of hypotheses (e.g., animals become stuck in shallow water, geomagnetic abnormalities may confuse the whales, healthy animals follow a sick one ashore) but it is not currently understood why healthy animals strand. In some instances, human activities, such as sonar, have overlapped in space and time with pilot whales strandings. However, there are records of mass pilot whale strandings long before the invention of sonar (e.g., Cape Cod 1902, n=~100; New Zealand 1918, n=~1000).

In January 2005, 33 short-finned pilots, 1 minke whale and 2 dwarf sperm whales were part of an Unusual Mortality Event (Hohn et al. 2006) along the North Carolina coast. Gross necropsies and extensive tissue analyses (clinical pathology, parasitology, gross pathology, histopathology, microbiology, and serology) did not find a unifying cause for this event. No harmful algal blooms preceded it. The Navy had conducted routine, tactical, mid-frequency sonar operations in the area during the time period. Hohn et al. (2006) were not able to definitively link the UME to sonar operations or environmental variables and concluded that it was likely that the cause of the stranding event would remain unknown.

Oremland et al. (2009) analyzed mandibular fractures in 50 male and female stranded short-finned pilot whales. They hypothesized that fractures would be more common in males given the species is polygynous and exhibits sexual dimorphism. They found that fractures were common (overall n=27, 54%; female n=17, 47%; male n=10, 71%), but suggest further research is needed to understand the causes and significance of these fractures.

Table 2.14.1 summarizes both short-finned pilot whales (SF) and pilot whales only identified to genus (Spp) (Waring et al. 2011). North Carolina had the highest number of recorded stranded short-finned pilot whales during 2005-2009 (n=40, 93%).

Table 2.14.1 Short-finned Pilot Whale (SF) and Pilot Whale, Identified to Genus (spp), Strandings by State and Year

State	2005		2006		2007		2008		2009		Total	
	SF	Spp	SF	Spp	SF	Spp	SF	Spp	SF	Spp	SF	Spp
Maine	0	0	0	0	0	0	0	1	0	0	0	1
Massachusetts	0	0	0	0	0	0	0	1	0	0	0	1
New Jersey	0	2	1	0	0	0	0	0	1	0	2	2
Delaware	0	0	0	0	0	0	0	0	1	0	1	0
North Carolina	35	2	0	1	0	0	3	1	2	0	40	4
South Carolina	0	0	0	0	0	0	0	1	0	0	0	1
Total	35	4	1	1	0	0	3	4	4	0	43	9

(Data from the NE and SE marine mammal stranding response databases)

2.14.8.2 Contaminants

Tanabe et al (1987) analyzed organochlorine concentrations (PCBs and DDE) in the blubber of short-finned pilot whales from the coast of Japan. Their analyses showed an increase in concentration up to 10 years of age, then a decrease from 10-25 years of age. Organochlorine concentrations appear to then increase again in whales over 25 years of age. This trend was particularly clear in females and may be due to offloading via gestation and lactation.

2.14.8.3 Disease

Duigan et al. (1995a) looked for evidence of morbillivirus infection in short-finned pilot whales from the western Atlantic (n=25). The whales had stranded between 1986 and 1994. They found neutralizing antibody titers in 64% of the animals. The authors suggest that the social nature of the species, population size and migration patterns might facilitate infection.

Five pilot whales that died on the North Carolina coast, part of the 2005 UME, were tested for evidence of morbillivirus. Three of these whales tested positive for morbillivirus titers (Hohn et al. 2006).

2.14.9 Acoustics

Southall et al. (2007) classified *Globicephala* spp. into a mid-frequency cetacean functional hearing group with an estimated auditory bandwith of 150 kHz to 160 kHz. Rendell et al. (1999) analyzed tonal calls from five odontocete species. They found that long-finned and short-finned pilot whale calls were distinct. The mean frequency of short-finned pilot whales was 7,870 Hz (CV=0.32).

Weir (2008b) studied the response of a pod of short-finned pilot whales to a seismic “ramp-up” procedure. (The ramp-up procedure is commonly used by the geophysical industry to mitigate the potential impact of seismic guns.) Initially, no change in behavior was observed in the whales. Ten minutes into the “ramp-up” one whale group turned away from the airguns and behavior such as milling and tail slapping was also observed, likely indicating the pilot whales were displaying an avoidance response.

2.14.10 Fisheries By-Catch and Entanglement

Total human-caused short-finned pilot whale mortality along the U.S. cannot be estimated. Historically, pilot whales were incidentally caught off the U.S. Atlantic coast in the DWF and other foreign fishing activities (Fairfield et al. 1993; Waring et al. 1990). Fairfield et al. (1993) described a high rate of takes in the DWF in 1988 that they attributed to a geographic concentration in fishing effort. Fertl and Leatherwood (1997) describe multiple observations of pilot whales actively feeding in and around trawl nets during haul back. Most incidental by-catch occurs along the mid-Atlantic coast where the two species of pilot whales overlap; this makes it difficult to assign the mortality to one species or the other (Waring et al. 2011). Total annual estimated average fishery-related mortality for 2005-2009 was 162 pilot whales (*Globicephala* spp.).

Table 2.14.2 Mean Annual Mortality of Undifferentiated Pilot Whales by Fishery 2004-2008

Fishery	Mean Annual Mortality
Mid-Atlantic Bottom Trawl	30 (0.16)
Northeast Bottom Trawl	12 (0.14)
Mid-Atlantic Mid-water Trawl	2.4 (0.99)
Northeast Mid-water Trawl	3.0 (0.61)
Pelagic Longline	114 (0.20)
2005 Pelagic Longline experimental fishery	1 (1.0)

(CV given in parentheses)

2.14.11 Vessel Interactions

A review of Northeast and Southeast U.S. stranding data between 2005 and 2009 yielded no records of vessel interactions with short-finned pilot whales.

2.14.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear. Stone (2003) studied the effects of seismic activity on cetaceans in UK waters. She reported that sighting rates of white-sided dolphins, white-beaked dolphins, Lagenorhynchus spp., all small odontocetes combined and all cetaceans combined were significantly lower when large airgun arrays were being fired. The 2010 Report of the Working Group on Marine Mammal Ecology (ICES 2010) stated that very little is known about the impact of the construction and operation of offshore wind-farms. Madsen et al. (2006b) reviewed available current information on wind farms and marine mammals and wrote that the noise impact is more severe during construction than operation. Although no information is available specific to short-finned pilot whales, Weir (2008b) studied the reaction of long-finned pilot whales to an air-gun ramp up procedure off Gabon. She documented an avoidance response although it was limited in time and space.

2.14.13 Data Gaps and Research Recommendations

The best abundance estimate is from summer 2004 aerial and ship surveys, this data is 7 years old. There is very little recent information on short-finned pilot whale distribution and abundance outside of the summer months (June-September). Current research will continue to elucidate pilot whale stock structure in the northwest Atlantic, Gulf of Mexico, and the Caribbean Sea. Contaminant loading information is from other populations of short-finned pilot whales.

2.15 PYGMY KILLER WHALE (*FERESA ATTENUATA*)

2.15.1 Status

The pygmy killer whale is not listed under the Endangered Species Act. The species is considered “Data Deficient” on the IUCN Red List. It is afforded protection in U.S. waters under the MMPA.

2.15.2 General Distribution

The pygmy killer whale is found in tropical and sub-tropical waters throughout the world (Jefferson et al. 1994). Despite their widespread distribution, they are poorly understood and one of the most rare whale species (McSweeney et al. 2009). Caldwell and Caldwell (1971a) documented the first pygmy killer whale in the western Atlantic in 1969 from an animal that stranded in the Lesser Antilles. In 1992, a group of 6 pygmy killer whales was sighted off Cape Hatteras, North Carolina; there have not been any sightings in subsequent surveys (Waring et al. 2011). Given this sighting and the species’ tendency to be in tropical/sub-tropical waters, pygmy killer whales could potentially be observed in the BOEM Mid-Atlantic, South Atlantic, or Straits of Florida Planning Areas. Sighting and stranding locations of pygmy killer whales are shown in Figure 2.15-1.

2.15.3 General Abundance

The number of pygmy killer whales off the U.S. Atlantic coast is unknown. The available data are not adequate to calculate a minimum estimate (Waring et al. 2011).

2.15.4 Habitat Preference

They are often found around volcanic islands such as Japan and Hawaii (Donahue and Perryman 2009). Zerbini and Cesar de Oliveira Santos (1997) reported that the stomach contents of a pygmy killer whale that stranded in Brazil indicated it had been feeding over the outer continental shelf and slope.

2.15.5 Stock Structure

McSweeney et al. (2009) analyzed 22 years of photo-identification data around the main Hawaiian Islands. Their analyses indicated stable long-term associations in mixed sex groups. Stock structure for Western North Atlantic pygmy killer whales is unknown (Waring et al. 2011).

2.15.6 Life History Traits

There is not a lot known about the life history traits of pygmy killer whales. Donahue and Perryman (2009) report a mean length at maturity of 2 m.

2.15.7 Food Habits

Pygmy whale food habits are not well understand but Donahue and Perryman (2009) report that remnants of cephalopods and small fish have been found in stranded and by-caught stomachs. Perryman and Foster (1980) suggested that pygmy killer whales sometimes attack and eat other dolphins (e.g., *Stenella spp*. and *Delphinus delphis*).

2.15.8 Health

2.15.8.1 Strandings

There are five pygmy whale stranding records along the U.S. Atlantic coast for the period 2005-2009 (NE and SE marine mammal stranding response databases): four in South Carolina and one in North Carolina. None of these animals showed signs of human interaction. Mignucci-Giannoni (1999) described the stranding of five (3 dead, 2 alive and returned to the water) pygmy killer whales in the British Virgin Islands one day after Hurricane Marilyn devastated the area. Wang and Yang (2006) described unusual stranding events that included pygmy killer whales, in Taiwan in 2004 and 2005 after naval exercises had occurred nearby.

2.15.8.2 Contaminants

Forrester et al. (1980) documented DDT and PCB levels in three pygmy killer whales that stranded in Florida in the 1970s. They noted that the two males had higher concentrations than the female and attributed this to different habitat. Given the tendency for female mammals to offload contaminants through gestation and lactation, it is likely this may also be the reason for the difference.

2.15.8.3 Disease

Forrester et al. (1980) describes a variety of parasites (e.g., Nematoda, Destoda, Trematoda) in three pygmy killer whales that stranded in Florida in the 1970s.

2.15.9 Acoustics

Madsen et al. (2004b) studied acoustic recordings of pygmy killer whales in the northern Indian Ocean. These recordings indicated that pygmy killer whales produce echolocation clicks and that they likely use these clicks (as other small delphinids do) to find food. Southall et al. (2007) classified *Feresa* into a mid-frequency cetacean functional hearing group with an estimated auditory bandwith of 150 Hz to 160 Hz. Montie et al. (2011) used CT (computed tomography**)** imaging and auditory evoked potentials (AEPs) procedures of live stranded pygmy killer whales to learn more about their auditory system. In a study using acoustic behavior to detect animals at sea, pygmy killer whales were detected acoustically 100% of the time that they were detected visually (Rankin et al. 2008).

Pygmy killer whales appear to be a species that is sensitive to anthropogenic noise. As reported above, Wang and Yang (2006) described unusual stranding events that included pygmy killer whales in Taiwan in 2004 and 2005 after naval exercises had occurred nearby.

2.15.10 Fisheries By-Catch and Entanglement

There are no records of fishery interactions for the most recent time period reported (2001-2005) (Waring et al. 2011).

2.15.11 Vessel Interactions

A review of Northeast and Southeast U.S. stranding data between 2005 and 2009 yielded no records of vessel interactions with pygmy killer whales.

2.15.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear. No information specific to pygmy killer whales is available.

2.15.13 Data Gaps and Research Recommendations

In general, very little is known about pygmy killer whales. There is no abundance estimate for pygmy killer whales in the U.S Atlantic. The limited amount of information available about pygmy killer whales comes from parts of their range that is outside of the U.S. Atlantic coast (e.g., from the Pacific Ocean).

2.16 MELON-HEADED WHALE (*PEPONOCEPHALA ELECTRA*)

2.16.1 Status

The melon-headed whale is not listed under the Endangered Species Act. The species is considered “Least Concern” on the IUCN Red List. It is afforded protection in U.S. waters under the MMPA.

2.16.2 General Distribution

Melon-headed whales are distributed throughout the world in both tropical and sub-tropical waters (Perryman 2009). There are two sighting records (1999 = 20 whales; 2002 = 80 whales) of melon-headed whales in the U.S. Atlantic–both off of Cape Hatteras, North Carolina (Waring et al. 2011). There are stranding records from New Jersey and Georgia (see Stranding section). Given sighting and stranding data, it is possible that melon-headed whales could be found in any of the BOEM Atlantic Planning Areas. Sighting and stranding locations of melon-headed whales are shown in Figure 2.16-1.

2.16.3 General Abundance

The number of melon-headed whales off the U.S. Atlantic is unknown (Waring et al. 2011). A best estimate is not available.

2.16.4 Habitat Preference

In general, melon-headed whales are found in deep, offshore waters (Perryman 2009). In the Gulf of Mexico, melon-headed whales exhibit an oceanic distribution (Mullin et al. 1994); all sightings reported in Mullin et al. (1994) were in waters deeper than 500 m. In the Hawaiin Islands, satellite-tagged melon-headed whales were shown to move offshore to occupy the edges of cold core cyclonic eddies and the centers of warm core anticyclonic eddies where prey could be concentrated (Woodworth et al. 2011)

Gross et al. (2009) used sighting and stable isotope data to estimate niche segregation among dolphins in the Indian Ocean. Their analyses showed that melon-headed whales prefer steep slopes and waters that are slightly deeper than those preferred by the spotted (*Stenella attenuata*) and spinner (*S. longirostris*) dolphins. Although melon-headed whales overlapped with the spotted and spinner dolphins in terms of habitat, the stable isotope analysis showed ecological differentiation between them.

2.16.5 Stock Structure

Stock structure in the northwest Atlantic is unknown. This population is considered separate from the Gulf of Mexico population for management purposes (Waring et al. 2011).

2.16.6 Life History Traits:

Females become sexually mature at 11.5 years and 235 cm; males mature around 15 years (Perryman 2009). Bryden et al. (1977) reported preliminary life history traits from stranded melon-headed whales in Australia. They found females probably become sexually mature when body length reaches 225 cm but before reaching 257 cm and after 4 laminations but before 12 laminations are deposited in dentine. For males, they found sexual maturity likely occurs about 260 cm and when 7 laminations have been put down. Gestation is approximately 1 year.

2.16.7 Food Habits

Squid, small fish, and shrimp have all been identified in melon-headed whale stomach contents (Clarke and Young 1998; Jefferson and Barros 1997). Bossart et al. (2007a) found 6-20 squid beaks in the stomachs of 5 stranded melon-headed whales on the Atlantic Coast of Florida in 2006. They did not find any other food items. Stable isotope analyses on melon-headed whales in the Indian Ocean suggested their diet included carnivorous fish and squid (Gross et al. 2009). These analyses also indicated that the carbon isotopic composition was influenced by benthic primary production.

2.16.8 Health

2.16.8.1 Strandings

Perryman et al. (1994) described mass strandings of melon-headed whales throughout their range. There have been 18 records of melon-headed whale strandings along the U.S. Atlantic coast, including two mass-strandings in Florida in 2006 (2 animals and 5 animals) and one mass-stranding in South Carolina in 2008 (3 animals). A melon-headed whale also stranded in Puerto Rico in 1999 (Waring et al. 2011). Gasparini and Sazima (1996) reported a stranded melon-headed whale along the coast of Brazil that had cookiecutter shark (*Isistius brasiliensis*) wounds.

Bossart et al. (2007a) described pathological findings of 5 melon-headed whales that stranded on the Atlantic coast of Florida in March 2006. As reported in Gasparini and Sazima (1996), all 5 showed signs of being attacked by cookiecutter sharks. They also found all of the whales were thin (postnuchal depression, pronounced scapula and peduncular vertebrae). The cause of this stranding was not identified but the authors noted that all 5 whales showed evidence of chronic disease (see disease section below) prior to this event.

Melon-headed whale stranding events have overlapped in space and time with human acoustic activities; for more details, see the acoustics section below.

2.16.8.2 Contaminants

Kajiwara et al. (2008) reported contaminant levels in melon-headed whales that stranded along the Japanese coast. A total of 55 samples were collected from 1982 to 2006. They found that DDT's and PCBs were predominant in all of the samples. They estimated total maternal transfer (gestation and lactation) was 85% of the mother's burden. PCBs, DDTs, and hexachlorobenzene (HCB) levels all decreased in samples collected after 2000. However, PBDE and chlordane levels showed an increase over the study time period (1982-2006).

Hart et al. (2008) analyzed the temporal trends of perfluorinated compounds (PFC) in melon-headed whales that had stranded along the Japanese coast (1982-2006). They also analyzed mother-fetus pairs and found that PFCs had higher transplacental transfer rates than PCBs or PBDEs.

2.16.8.3 Disease

Cannon (1977) reported on three different nematodes (*Anisakis simplex, A. typical, Stenurus globiocephalae*) and an immature tapeworm (*Phyllobothrium chamissonii*) in melon-headed whales that stranded in Australia. In the whales described by Bossart et al. (2007a), lesions were found through the gastro-intestinal tracts; the whales' peritoneum contained *Monorygma spp*. cysts; *Nasitrema*-associated sinusitis; and mild to moderate myocardial degeneration was found in all 5 whales.

2.16.9 Acoustics

Southall et al. (2007) classified *Peponocephala* sp. into a mid-frequency cetacean functional hearing group with an estimated auditory bandwidth of 150 Hz to 160 Hz. Watkins et al. (1997) observed melon-headed whales in the southeastern Caribbean. They found that maximum source levels were 155 dB for whistles and 165 dB for click-bursts. Clicks and click-bursts had a frequency emphasis between 20-24 kHz. The dominant frequencies for whistles were 8-12 kHz. The authors noted an increase in click bursts and whistles with increased activity. Baumann-Pickering et al. (2010) found that melon headed whales (n=50) near Palmyra Atoll produced median peak frequencies of averaged clicks of 24.4-29.7 kHz. Frankel and Yin (2010) made recordings of large groups of melon-headed whales around the Big Island of Hawaii. They described echolocation clicks, burst-pulse sounds and whistles.

Southall et al. (2006) described unusual behavior of melon-headed whales in Hanalei Bay, Kauai, Hawaii, July 3-4, 2004. A group of 150-200 whales "milled about" in the shallow area of the bay and eventually were herded out to sea by humans. Only one animal was known to have died (a calf found on 5 July). This event overlapped with Rim-of-the-Pacific naval exercises; ships converging for this activity had used active mid-frequency sonar intermittently July 2-3. Southall et al. (2006) analyzed environmental variables that could have contributed to this event and concluded that the sonar transmissions were likely a contributing factor to this event. Brownell et al. (2009) further investigated this event and other mass strandings and supported Southall et al.'s (2006) suggestion.

2.16.10 Fisheries By-Catch and Entanglement

There were no reports of mortality or serious injury to melon-headed whales during 2001-2005 (the most recent time frame available) (Waring et al. 2011). Therefore, the total annual estimated mortality for melon-headed whales was zero from 2001-2005 (Waring et al. 2011).

2.16.11 Vessel Interactions

A review of Northeast and Southeast U.S. stranding data between 2005 and 2009 yielded no records of vessel interactions with melon-headed whales. It should be noted that melon-headed whales do appear to be sensitive to sonar activities (see the Acoustics section above).

2.16.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear. No information specific to melon-headed whales is available.

2.16.13 Data Gaps and Research Recommendations

There is no abundance estimate for melon-headed whales in the U.S Atlantic. Most of the information about the melon-headed whales' life history traits, acoustics, food habits, contaminant burden, or disease comes from parts of their range that is outside of the U.S. Atlantic coast (e.g., from the Pacific Ocean).

2.17 WHITE-BEAKED DOLPHIN (*LAGENORHYNCHUS ALBIROSTRIS*)

2.17.1 Legal Status

The white-beaked dolphin is not listed under the Endangered Species Act and is considered a species of "Least Concern" on the IUCN Red List. It is afforded protection in U.S. waters under the MMPA.

2.17.2 General Distribution

White-beaked dolphins are found in the temperate and subarctic North Atlantic (Kinze 2009). They appear to be much more abundant in the eastern than in the western North Atlantic (Northridge et al. 1995). Along the U.S. east coast they are found from Cape Cod, Massachusetts, north into Canadian waters (Reeves et al. 1999b). Prior to the 1970s, white-sided dolphins in the U.S. Atlantic were found mostly off shore or on the continental slope while white-beaked dolphins were on the shelf but the two species switched distribution (Kenney et al. 1996). Most of the sightings have occurred in the BOEM North Atlantic Area. Sighting, stranding and fishery bycatch locations of white-beaked dolphins are shown in Figure 2.17-1.

2.17.3 General Abundance

The number of white-beaked dolphins in U.S. waters is unknown (Waring et al. 2011). Waring et al. (2011) report a best estimate of 2,003 (CV=0.94) animals in August 2006 for the northwestern Atlantic (U.S. and Canadian waters). This estimate is from aerial surveys flown along the 2000-m depth contour from southern Georges Bank to the upper Bay of Fundy and to the entrance of the Gulf of St. Lawrence. They caution that this number is probably low because the survey covered only part of the species range and only a small area of the BOEM North Atlantic Planning Area.

2.17.4 Habitat Preference

As described in the general distribution section, prior to the 1970s, white-beaked dolphins along the U.S. Atlantic coast were on the shelf. Since the 1970s they have been found mostly on the continental slope or off shore (Kenney et al. 1996). In other parts of their range, they are still most often found on the shelf Northridge et al. (1995) found that white-beaked dolphin distribution around the British Isles was limited to shelf areas. In their study on the co-occurrence of cetaceans and seabirds in the Northeast Atlantic, Skov et al.(1995) found that white-beaked dolphin and northern gannet (*Sula bassana*) distributions were highly correlated (r=0.54, P<0.001, df=49) and separated from the other cetacean/bird groupings, perhaps due to their reliance on shoaling fish (Evans 1980).

2.17.5 Stock Structure

In their analysis (principal component and partial least squares) on skull characteristics, Mikkelsen and Lund (1994) found significant differences between white-beaked dolphins from the eastern and western north Atlantic. Stock structure along the U.S. Atlantic coast has not been described.

2.17.6 Life History Traits

As is summarized in Kinze (2009), females become sexually mature at a mean age of 8.7 years (240 cm) and males at a mean age of 11.6 years (270 cm), gestation is approximately 11 months, and the maximum recorded age for females is 34 years. Young are born probably between June and September (Leatherwood and Reeves 1983c).

2.17.7 Food Habits

This species' diet includes squid, octopus, clupeids, gadids, hake, and some benthic crustaceans (Leatherwood and Reeves 1983c; Reeves et al. 1999b). De Pierrepont et al. (2005) analyzed the stomach contents of a single white-beaked dolphin that stranded on the coast of Normandy (France). Gadids comprised 64.7% of the numeric proportion of prey. Hai et al. (1996) analyzed the stomach contents of 20 white-beaked dolphins that were killed when entrapped by ice in Port-aux-Basques, Canada, in March of 1982. Ninety percent of the stomachs contained prey remains from Atlantic cod (*Gadus morhua*). Remnants of crab and seaweed were found in 10% of the stomachs.

2.17.8 Health

2.17.8.1 Strandings

A total of 5 white-beaked dolphins stranded along the U.S. Atlantic (Maine, Massachusetts and New Jersey) coast from 2005 to 2009. Two were in Maine, two in New Jersey and one in Massachusetts (Northeast and Southeast U.S. marine mammal stranding response databases). White-beaked dolphins have not been recorded as part of a UME.

2.17.8.2 Contaminants

Muir et al. (1988) documented PCB's, DDT's, chlordanes, toxaphene, chlorobenzenes, hexachlorocyclohexane isomers and seven elements in the tissue of white-beaked dolphins that died after being trapped in ice off Newfoundland, Canada. They found PCB levels in the dolphins were similar to that in harbor porpoise and beluga whales that were also sampled from what they refer to as "relatively contaminated Canadian and U.S. Atlantic waters." They also found high levels of lead (Pb) in the dolphin tissue.

2.17.8.3 Disease

Reeves et al. (1999b) report a high incidence of diseased jaws and teeth in white-beaked dolphins that stranded around the British coast. Hai et al. (1996) found worms (*Anisakis spp.*) in the stomachs of all the dolphins entrapped in ice in Port-aux-Basques. Buck and Spotte (2005) found enterobacteria (*Edwardsiella, Escherichia, Klebsiella*), *Plesiomonas, Aeromonas, Citrobacter, Pseudomonas,* and *Staphylococcus* in anal and blowhole swabs taken from 5 stranded white-beaked dolphins that were rescued from ice in Newfoundland and transported to the Mystic Marinelife Aquarium, Connecticut, in 1983. In their study of stranded marine mammals along Cape Cod and southeastern Massachusetts, Bogomolni et al. (2010) reported 2 stranded white-beaked dolphins. One had died of disease; the cause of death of the other dolphin could not be determined.

2.17.9 Acoustics

Schevill and Watkins (1962), as cited in Reeves et al. (1999b), described "squeals" in the range of 6.5-15 kHz. Mitson and Morris's (1988) recordings of white-beaked dolphins in the North Sea that showed the frequency of their emissions can increase to at least 305 kHz. It has been shown that white-beaked dolphins hear at very high frequencies–comparable to that heard by harbor porpoise (Nachtigall et al. 2008). However, Southall et al. (2007) classified *Lagenorhynchus spp*. into a mid-frequency cetacean functional hearing group with an estimated auditory bandwith of 150 Hz to 160 Hz.

2.17.10 Fisheries By-catch and Entanglement

No observations of fishery-related mortality have been reported for white-beaked dolphins in the U.S. EEZ (Waring et al. 2011).

2.17.11 Vessel Interactions

A review of Northeast and Southeast U.S. stranding data between 2005 and 2009 yielded no records of vessel interactions with white-beaked dolphins.

2.17.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear.

Stone (2003) studied the effects of seismic activity on cetaceans in UK waters. She reported that sighting rates of small odontocetes (including white-beaked dolphins) were significantly lower when large airgun arrays were being fired. She also found that in general, small odontocetes showed the strongest avoidance behavior to seismic noise (compared to baleen, killer, sperm and pilot whales). The 2010 Report of the Working Group on Marine Mammal Ecology (ICES 2010) stated that very little is known about the impact of the construction and operation of offshore wind-farms. Madsen et al. (2006b) reviewed available current information on wind farms and marine mammals and wrote that the noise impact is more severe during construction than operation. This was found to be the case for impacts to harbor porpoise (see Carstensen et al. [2006], Koschinski et al. [2003]). David (2006) wrote that pile driver-generated noise has the potential to impact dolphins by temporary displacement and interference with communication.

The 2011 Report of the Working Group of Marine Mammal Ecology (ICES 2011) reported that porpoise detection (T-POD data) was lower during construction of a tidal turbine compared to before and after construction.

2.17.13 Data Gaps and Research Recommendations

The best abundance estimate is from 2006 and is now 5 years old. There is no current information on white-beaked dolphin distribution and abundance outside of the summer months (June-August). There is limited contaminant information from animals trapped by ice off Newfoundland, Canada. There is no information on food habits, stock structure or the impacts of non-fishery related anthropogenic activities on white-beaked dolphin in the northwestern Atlantic.

2.18 ATLANTIC WHITE-SIDED DOLPHIN (*LAGENORHYNCHUS ACUTUS*)

2.18.1 Legal Status

The Atlantic white-sided dolphin is not listed under the Endangered Species Act and is considered a species of "Least Concern" on the IUCN Red List. It is afforded protection in U.S. waters under the MMPA.

2.18.2 General Distribution

Atlantic white-sided dolphins are found in the cold-temperate North Atlantic (Cipriano 2009). Before the 1970s, white-sided dolphins in the U.S. Atlantic were found mostly off shore or on the continental slope; white-beaked dolphins were on the shelf but the two species switched distribution (Kenney et al. 1996). Selzer and Payne (1988) found that, in U.S. waters, white-sided dolphins were common in the Gulf of Maine (40-44^{o}N). In the spring they are concentrated in the southwestern Gulf of Maine through the Great South Channel and along the western edge of Georges Bank with another concentration along the continental slope of the mid-Atlantic region (39^{o}N). In the fall they are found primarily in the Gulf of Maine Basin where most sightings occur between 42-44^{o}N. Weinrich et al. (2001) suggest that white-sided dolphins use coastal New England waters (referring primarily to Stellwagen Bank and Jeffereys Ledge) transitorily as part of a larger home range.

Seasonally, white-sided dolphins are found from Georges Bank to the lower Bay of Fundy in the summer (June to September), from Georges Bank to the southern Gulf of Maine in the fall (October to December) then densities of white-sided dolphins drop off in this area from January to May (Palka et al. 1997). White-sided dolphins are found south of Georges Bank (around Hudson Canyon) all year but at low densities (Palka et al. 1997). They are not found in the BOEM South Atlantic or Straits of Florida Planning Areas. Sighting, stranding, and fishery bycatch locations of white-sided dolphins are shown in Figure 2.18-1.

2.18.3 General Abundance

The total number of white-sided dolphins along the U.S. Atlantic coast is unknown (Waring et al. 2011). The best estimate of white-sided dolphin abundance in the western north Atlantic is 23,390 (CV=0.23), the sum of the 2006 and 2007 aerial surveys (Waring et al. 2011). While the combined estimate may include a certain amount of inter-annual redistribution, it is still felt to be more representative than either estimate alone. Because the estimated abundance of this species has large inter-annual variability (that is, the estimates were about 51,000 in 1999 and 109,000 in 2002 and about 24,000 recently), the spatial-temporal distribution is being investigated to more completely understand how this species utilizes US waters throughout the year (Waring et al. 2011).

2.18.4 Habitat Preference

As described in the general distribution section, white-sided dolphins in the U.S. Atlantic have been found mostly on the continental shelf since the 1970s (Kenney et al. 1996). Generally, white-sided dolphins are found in waters over the continental shelf and slope, and occasionally move into coastal or deeper waters (Cipriano 2009). In the northwestern Atlantic they prefer water 9-15° C over the continental slope (Leatherwood and Reeves 1983a). Selzer and Payne (1988) found that white-sided dolphins were most often found in waters with low SST and salinity and over areas with high topographical relief. Ninety-seven percent of their sightings occurred in water less than 12°C. They suggest that these factors primarily influence white-sided dolphin prey distribution which secondarily influences dolphin distribution. Hamazaki (2002) developed spatiotemporal prediction models of cetacean habitats off the northeast U.S. coast and found that white-sided dolphins preferred waters 400-500 m deep where SST was 10-15°C.

2.18.5 Stock Structure

In their analysis (principal component and partial least squares) on skull characteristics, Mikkelsen and Lund (1994) could not clearly separate white-sided dolphins from the eastern and western north Atlantic. Palka et al. (1997) reported that the combination of sightings, strandings, and by-catch data indicates there are three populations in the northwest Atlantic: Gulf of Maine, Gulf of St. Lawrence, and the Labrador Sea, but this has not been confirmed by genetic analyses. Amaral (2005) analyzed samples from stranded, by-caught, and biopsied white-sided dolphins from the Gulf of Maine, Cape Cod, Gulf of St. Lawrence, Sable Island, and the Faroe Islands. She found limited relatedness within some stranded groups but suggested this species has a fluid social structure.

2.18.6 Life History Traits

Females become sexually mature at 6-12 years; males mature at 7-11 years (Cipriano 2009). Sergeant et al. (1980) reported that in the northwest Atlantic gestation lasts approximately 11 months, most births occur in June and July, and lactation lasts about 18 months. The inter-calving interval is likely 2-3 years (Leatherwood and Reeves 1983a); however, some stranded females were observed to be both pregnant and lactating, indicating that some individuals may breed annually (Sergeant et al. 1980). In New England waters, Weinrich et al. (2001) found that early summer is an important calving period.

2.18.7 Food Habits

Craddock et al. (2009) analyzed 62 stomachs (28 from by-caught animals, 34 from stranded animals). Most of the by-caught animals were collected in the Gulf of Maine while the stranded animals were from Cape Cod, Massachusetts. They found silver hake (*Merluccius bilinearis*), spoonarm octopus (*Bathypolypus bairdii*), and haddock (*Melanogrammus aeglefinus*) were the most important prey species for the net-caught animals and sand lance (*Ammodytes spp*.) was the dominant prey of the stranded animals.

2.18.8 Health

One UME, which occured along the coast from New Jersey to North Carolina between 1 January and 26 April, 2008, involved white-sided dolphins (n=4 females). The cause of this UME is still under investigation.

2.18.8.1 Strandings

There were 245 white-sided dolphin strandings reported along the U.S. Atlantic coast from 2005-2009 (Northeast and Southeast U.S. marine mammal stranding response databases). Fourteen of these stranded dolphins showed signs of human interactions. Of these, a majority (n=182, 74%) occurred in Massachusetts, followed by New York (n=12, 5%). No strandings were reported south of South Carolina. Amaral (2005) analyzed stranding data 1973-1999 and found significantly more animals stranded along the U.S. Atlantic coast in the winter months. Bogomolni et al. (2010) analyzed the cause of death of marine mammals stranded along the coast of Cape Cod and southeastern Massachusetts. Most (69%) of the white-sided dolphins in this study were placed in the “Mass Stranding with No Significant Finding” category. White-sided dolphins had the highest number of individuals in this category of all species included in this study.

2.18.8.2 Contaminants

Weisbrod et al. (2001) documented PCB’s and DDE’s in white-sided dolphins sampled in the Gulf of Maine. Several of these samples contained organochlorine levels that were considered to have negatively impacted the health of other species of marine mammals. Tuerk et al. (2005) found that body length was a better predictor of contaminant burdens than age and that both male and female levels decreased with age. This decrease is probably due to lactation in females and either growth dilution or metabolic elimination in males. Montie et al. (2009) documented organochlorines, polychlorinated byphenyls, hydroxylated PCBs, methyl sulfone PCBs, flame retardants, and hydroxylated BDE’s in white-sided dolphins that stranded on Cape Cod, Massachusetts.

2.18.8.3 Disease

Bogomolni et al. (2010) found that disease was the cause of death in 21% of white-sided dolphins that stranded along the coast of Cape Cod and southeastern Massachusetts. Analyses of tissue from white-sided dolphins that stranded in Wellfleet, Massachusetts, in 1998 and 1999 found a high level (39%) of S*arcocystis spp*. infection (Ewing et al. 2002).

2.18.9 Acoustics

Southall et al. (2007) classified *Lagenorhynchus* sp. into a mid-frequency cetacean functional hearing group with an estimated auditory bandwith of 150 Hz to 160 Hz. Schevill and Watkins (1962), as described in Reeves et al. (1999a), documented “squeals” in the range of 1-24 kHz and clicks.

2.18.10 Fisheries By-catch and Entanglement

This species is caught in a variety of commercial fisheries; information is summarized below. The most recent mean annual mortality of all fisheries was estimated at 245 (CV=0.12) for the years 2005-2009 (Waring et al. 2011).

Table 2.18.1 Incidental White-sided Dolphin Mortality 2005-2009

Fishery	Mean Annual Mortality (CV)
Northeast Sink Gillnet	36 (0.34)
Northeast Bottom Trawl	160 (0.14)
Northeast Mid-water Trawl	1.9 (1.03)
Mid-Atlantic Mid-Water Trawl	24 (0.55)
Mid-Atlantic Bottom Trawl	23 (0.12)

The NMFS assembled the Atlantic Trawl Gear Take Reduction Team (ATGTRT) to address incidental mortality of white-sided dolphins (among several other species) in September 2006. This take reduction team focuses on the Mid-Atlantic Mid-Water Trawl, the Mid-Atlantic Bottom Trawl, Northeast Mid-Water Trawl and the Northeast Bottom Trawl fisheries. The Team found that mean annual incidental mortality (2001-2005) for white-sided dolphins was highest in the Northeast region in deep waters (>142.5 m) with low to mid SST (4.44-6.49^{o}C).

2.18.11 Vessel Interactions

In 2005, one white-sided dolphin stranding mortality in the Northeast and Southeast U.S. Marine Mammal Stranding Network databases exhibited signs of interaction with a boat propeller.

2.18.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear.

Stone (2003) studied the effects of seismic activity on cetaceans in UK waters. She reported that sighting rates of small odontocetes (including white-sided dolphins) were significantly lower when large airgun arrays were being fired. She also found that in general, small odontocetes showed the strongest avoidance behavior to seismic noise (compared to baleen, killer, sperm and pilot whales). The 2010 Report of the Working Group on Marine Mammal Ecology (ICES 2010) stated that very little is known about the impact of the construction and operation of offshore wind-farms. Madsen et al. (2006b) reviewed available current information on wind farms and marine mammals and wrote that the noise impact is more severe during construction than operation. This was found to be the case for impacts to harbor porpoise (see Carstensen et al. [2006], Koschinski et al. [2003]). David (2006) wrote that pile driver-generated noise has the potential to impact dolphins via temporary displacement and interference with communication. The 2011 Report of the Working Group of Marine Mammal Ecology (ICES 2011) reported that porpoise detection (T-POD data) was lower during construction of a tidal turbine compared to before and after construction.

2.18.13 Data Gaps and Research Recommendations

There is limited recent information on white-sided dolphin distribution and abundance outside of the summer months (June-September). Very little is understood about stock structure in the northwest Atlantic. No information on the impacts of non-fishery related anthropogenic activities on white-sided dolphin in the northwestern Atlantic has been published.

The ATGTRT recommended that there be an improved abundance estimate for all species (including white-sided dolphins) affected by the trawl fisheries including conducting more surveys, appropriately incorporating data from multiple years and by using appropriate stock structure boundaries. Additionally, they specifically recommended that environmental and biological factors that may be responsible for the large inter-annual variation in white-sided dolphin abundance estimates be investigated and modeled.

2.19 RISSO'S DOLPHIN (*GRAMPUS GRISEUS*)

2.19.1 Status

The Risso's dolphin is not listed under the Endangered Species Act and is considered a species of "Least Concern" on the IUCN Red List. It is afforded protection in U.S. waters under the MMPA.

2.19.2 General Distribution

Risso's dolphins are found worldwide in temperate and tropical oceans (Baird 2009). In the U.S. Atlantic they are found in the North Atlantic, Mid-Atlantic, South Atlantic and Straits of Florida Planning Areas (Kruse et al. 1999; Waring et al. 2011) although a majority of the sightings occur north of Maryland (38°N) (Waring et al. 2011). Sighting, stranding and fishery bycatch locations of Risso's dolphins are shown in Figure 2.19-1.

2.19.3 General Abundance

Waring et al. (2011) give a best estimate of 20,479 Risso's dolphins (CV=0.59), based on a combination of ship and aerial surveys conducted June-August 2004 from Florida to the Bay of Fundy. NMFS divided these surveys into two areas: the "North Atlantic" includes waters north of Maryland (38°N) to the Bay of Fundy (45°N) and the "South Atlantic" includes waters from Maryland (38°N) south to Florida (27.5°N). The 2004 Risso's dolphin estimate for the North Atlantic is 15,053 (CV=0.78). The 2004 estimate for the South Atlantic is 5,426 (CV=0.54). The NMFS North Atlantic Survey was conducted in part of the BOEM North Atlantic Planning Area while the NMFS South Atlantic Survey was conducted in parts of the BOEM Mid-Atlantic and South Atlantic Planning Areas. It is important to note that the surveys conducted did not necessarily cover the entire BOEM planning areas but likely sampled a subsection of these areas based on prior knowledge of species distribution and survey goals. Because the 2004 surveys occurred north of the Straits of Florida planning area, the 20,470 Risso’s dolphin abundance estimate does not include this area.

2.19.4 Habitat Preference

Risso’s dolphins prefer the steep shelf edge between 400-1000 m (Baird 2009). Generally they are found in waters between 15 °C and 20 °C and are rarely found in waters less than 10 °C (Baird 2009). Sightings from NMFS aerial and shipboard surveys along the U.S. Atlantic coast were mostly in the continental shelf edge and continental slope areas (Waring et al. 2011). Baumgartner (1997) reported that most Risso’s dolphins in the Gulf of Mexico are found over steep sections of the upper continental slope between the 350-975-m isobaths. This distribution is probably related to prey concentration. In the Azores, Risso’s dolphins prefer areas between 497 and 1,233 m in depth, with slopes between 27 and 35% (Pereira 2008). Hamazaki (2002) found that, compared to the other mid-Atlantic shelf species, Risso’s dolphins were found in warmer and deeper water. Dive and travel information was collected in a tagging study reported by Wells et al. (2009). A rehabilitated male Risso’s dolphin that had stranded on the Gulf coast of Florida was tagged and released in the Gulf of Mexico. The dolphin had travelled more than 3,300 km from the Gulf to the Atlantic Ocean off Delaware. More than 90% of the dolphin’s dives were within 50 m of the surface, with only a small percentage of dives to depths below 200 m. The deepest was a single dive to 400-500 m.

2.19.5 Stock Structure

Gaspari et al. (2007) reported on genetic differentiation between the Risso’s dolphin populations in the UK and the Mediterranean, but similar genetic work has not been done for the western Atlantic. There is no stock structure information available for Risso’s dolphins in the northwest Atlantic (Waring et al. 2011).

2.19.6 Life History Traits

Information on Risso’s dolphin life history traits is limited. Baird (2009) noted that age at first maturity is thought to be 8-10 years for females and 10-12 years for males. The inter-calving interval is 2.4 years. The composition of a school of 79 Risso’s dolphins killed in Taiji, Japan, in 1990 was examined by Anano and Miyazaki (2004). The group was composed of 49 females and 30 males, though only one of the males was fully mature. The oldest immature female was 10.5 years old and the youngest mature female was 8.5 years old. The average female body length was 270 cm at maturity. Ten to twelve years was estimated as the average age at maturity for males, and length at maturity was similar to that for females. Ten animals in the group were calves under a year of age, and two were between 1 and 1.5 years old. With the caveat that only a single school was studied, the authors made several inferences about Risso’s dolphin school organization, including the observation that young dolphins stay in the natal school until around puberty, mature females of similar reproductive status group together, and that mature males move among groups (Amano and Miyazaki 2004). Information on the social structure of free-ranging Risso’s dolphins off Pico Island in the Azores was gathered from field observations and photo-ID (Hartman et al. 2008). The authors found the basic units of Risso’s dolphin society to be clusters of 3-12 individuals grouped by age and sex classes. The dolphins exhibited stable long-term bonds and a high degree of site fidelity.

2.19.7 Food Habits

Risso's dolphins feed mostly on squid (neritic and oceanic species) (Baird 2009). Wurtz et al. (1992) analyzed the stomach contents of a Risso's dolphin that had died in a fishing net in the Mediterranean Sea. They found oceanic squid but no fish remains. Examinations of Risso's dolphin stomach contents stranded off the coast of England (Clarke and Pascoe 1985) and South Africa (Cockcroft et al. 1993) confirmed that cephalopods were the exclusive or primary prey. The study by Blanco et al. (2006) showed the main composition of Mediterranean stranded Risso's dolphins to be cephalopods, but also found several species of tunicates, which may be an overlooked component of the diet. Blanco et al. (2006) found evidence that Risso's dolphins in the Mediterranean feed preferentially in the middle slope (depths of 600-800 m).

2.19.8 Health

2.19.8.1 Strandings

Northeast and Southeast U.S. marine mammal stranding response databases contain records of 66 Risso's dolphin strandings along the U.S. Atlantic coast (Maine to Florida) from 2005-2009; the majority of the strandings took place along the Massachusetts coast. In their discussion of the possible connection between seismic activity and gas-bubble formation in cetacean tissue, Jepson et al. (2005a), observed that three out of 24 Risso's dolphins that had stranded off the British coasts between October 1992 and January 2003 exhibited gas-bubble-induced tissue damage. In a 2005 paper, Jepson et al. (2005a) present findings of gas-bubble formation and embolism in stranded Risso's and other dolphins, and suggest further research into the etiology of bubble formation and its association with anthropogenic noise.

2.19.8.2 Contaminants

Storelli and Marcotrigiano (2000) documented PCBs, DDT and HCB in the tissues of three Risso's dolphins that had stranded along the Italian coast. Concentrations of butyltins (BTs) in the liver and organochlorine compounds (OCs) in the blubber were looked at in Risso's dolphins caught off Taiji, Japan by Kim et al. (1996). They showed that while OC concentrations increased with age in males, they decreased with age in mature females, suggesting that these compounds were being transferred to the offspring. BT concentrations rose with age at the same rate in males and females.

2.19.8.3 Disease

Toxoplasmosis infection was documented in a Risso's dolphin and her fetus that had stranded on the coast of Spain (Resendes et al. 2002).

2.19.9 Acoustics

Madsen et al. (2004a) reported that Risso's dolphins produce short, broadband signals with peak frequencies around 50 kHz. They also found that Risso's dolphins use their echolocation to detect prey up to 80 m. Echolocation by a captive Risso's dolphin was also demonstrated by Philips et al. (2003). Corkeron and Van Parijs (2001) reported on recordings of Risso's dolphins made off the coast of Australia which included broadband clicks, barks, buzzes, grunts, chirps, whistles, and simultaneous whistle + burst-pulses in the 30 Hz and 22 kHz range. The hearing thresholds of Risso's dolphins have been studied by Nachtigall et al. (2005). An infant Risso's dolphin demonstrated very high-frequency sensitivity (Nachtigall et al. 2005). Response to sound stimuli was also tested in an infant Risso's dolphin, demonstrating that the response delay was among the fastest measured in marine mammals (Mooney et al. 2006)

2.19.10 Fisheries By-Catch and Entanglement

Waring et al. (2011) reported a mean annual fishery-related mortality or serious injury of 18 (CV=0.37) dolphins during the 5-year period (2005-2009). The 5-year mean annual mortalities by fishery are in Table 2.19-1.

Table 2.19.1 Fisheries Mortality of Risso's Dolphins

Fishery	Mean Annual Mortality (CV)
Northeast Sink Gillnet	3 (0.93)
Mid-Atlantic Gillnet	7 (0.73)
Pelagic Longline	8 (0.40)

2.19.11 Vessel Interactions

A review of Northeast and Southeast U.S. stranding data between 2005 and 2009 yielded one record of a Risso's dolphin stranding mortality with wounds consistent with propeller strike. The effects of whale watching activities on Risso's dolphins were studied in the Azores (Visser et al. 2011). When more than five whale watching vessels were present, Risso's dolphins spent significantly less time resting and socializing.

2.19.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear.

Stone (2003) studied the effects of seismic activity on cetaceans in UK waters. She reported that sighting rates of small odontocetes were significantly lower when large airgun arrays were being fired. She also found that in general, small odontocetes showed the strongest avoidance behavior to seismic noise (compared to baleen, killer, sperm and pilot whales). The 2010 Report of the Working Group on Marine Mammal Ecology (ICES 2010) stated that very little is known about the impact of the construction and operation of offshore wind-farms. Madsen et al. (2006b) reviewed available current information on wind farms and marine mammals and wrote that the noise impact is more severe during construction than operation. This was found to be the case for impacts to harbor porpoise (see Carstensen et al. (2006), Koschinski et al. (2003)). David (2006) wrote that pile driver-generated noise has the potential to impact dolphins via temporary displacement and interference with communication. The 2011 Report of the Working Group of Marine Mammal Ecology (ICES 2011) reported that porpoise detection (T-POD data) was lower during construction of a tidal turbine compared to before and after construction. No information specific to Risso's dolphins was found.

2.19.13 Data Gaps and Research Recommendations

The best abundance estimate is from 2004 and is now over 7 years old. There is no current information on Risso's dolphin distribution and abundance outside of the summer months (June-August). There is limited contaminant information, and what exists is from work done on dolphins from the Mediterranean Sea or off Japan. No information on food habits or stock structure of Risso's dolphins in the northwestern Atlantic has been published.

2.20 BOTTLENOSE DOLPHIN (*TURSIOPS TRUNCATUS*)

As of 2011, there are 15 recognized stocks of bottlenose dolphins along the U.S. Atlantic coast (see http://www.nmfs.noaa.gov/pr/pdfs/sars/).

2.20.1 Status

The bottlenose dolphin is not listed under the Endangered Species Act. The IUCN Red List lists its status as "Least Concern." The MMPA affords protection of all bottlenose dolphin stocks in U.S. waters and the coastal stocks are listed as "Depleted."

2.20.2 General Distribution

Bottlenose dolphins are found in temperate and tropical waters throughout the world (Wells and Scott 1999). Temperature appears to limit their distribution off North America either directly or by limiting prey distribution (Wells and Scott 2009). In the U.S. Atlantic, the offshore stock is found primarily along the outer continental shelf and continental slope from Georges Bank to Florida (Waring et al. 2011). The coastal stocks are found along the coast from south of Long Island Sound to Florida and are generally less than 7.5 km from shore while the estuarine stocks are found in bays and estuaries from North Carolina to Florida (Caldwell 2001; Garrison et al. 2003; Gubbins 2002a, 2002b, 2002c; Gubbins et al. 2003; Litz 2007; McLellan et al. 2003; Read et al. 2003; Rosel et al. 2009; Speakman et al. 2006; Stolen et al. 2007; Torres et al. 2003; Urian et al. 1999; Weller 1998). Bottlenose dolphins are found in the North Atlantic, Mid-Atlantic, South Atlantic and Straits of Florida BOEM Planning Areas. Sighting, stranding, and fishery bycatch locations of bottlenose dolphins are shown in Figure 2.20-1.

2.20.3 General Abundance

The largest population in U.S. Atlantic waters is the offshore stock (n=81,588) (Table 2.20.1). Population estimates are not available for all stocks, especially the recently defined estuarine populations.

Table 2.20.1 U.S. Atlantic Bottlenose Dolphin Population Estimates

Stock Area	Nbest	Year
Western North Atlantic, offshore	81,588 (0.17)	2002/2004
Western North Atlantic, coastal, northern migratory	9,604 (0.36)	2002
Western North Atlantic, coastal, southern migratory	12,482 (0.32)	2002
Western North Atlantic, coastal, S. Carolina/Georgia	7,738 (0.23)	2002/2004
Western North Atlantic, coastal, northern Florida	3,064 (024)	2002/2004
Western North Atlantic, coastal, central Florida	6,318 (0.26)	2002/2004
Northern North Carolina Estuarine System	Unk	Na
Southern North Carolina Estuarine System	2,454 (0.53)	2002
Charleston Estuarine System	Unk	Na
Northern Georgia/Southern South Carolina Estuarine System	Unk	Na
Southern Georgia Estuarine System	Unk	Na
Jacksonville Estuarine System	Unk	Na
Indian River Lagoon Estuarine System	Unk	Na
Biscayne Bay	Unk	na
Florida Bay	514 (0.17)	2003

(CV is reported in parentheses) (Waring et al. 2011)

2.20.4 Habitat Preference

Wells and Scott (1999) reported that bottlenose dolphins are a cosmopolitan species. As Wells and Scott (2005) summarized, bottlenose dolphins can be found in both coastal and pelagic waters, near oceanic islands over the continental shelf and break, and in estuarine habitats. Kenny (1990) used CETAP data to describe the association of bottlenose dolphins with certain environmental characteristics. He found that dolphins were sighted in depths from 0 to 4,932 m with a mean of 587.9 m and sea surface temperatures from 1.1° to 31.1°C with a mean of 19.7°C and a distinct mode at 10-22.5°C. The offshore stock was associated with a wider range of temperature and geography. Along the U.S. Atlantic coast, there are 15 stocks of dolphins inhabiting a variety of offshore, coastal and estuarine habitats which are detailed in the 2011 stock assessment reports (Waring et al. 2011).

2.20.5 Stock Structure

On-going research continues to elucidate and redefine bottlenose dolphin stock structure along the U.S. Atlantic coast. In their literature review of bottlenose dolphin research, Shane et al. (1986) found that many studies (behavioral, morphological, biochemical, etc.) indicated distinct stocks in some areas. Mead and Potter (1995) were able to identify two populations (offshore and coastal) of bottlenose dolphins along the U.S. Atlantic by analyzing distribution, size, skull morphology, food habits and parasite burden. Additionally, their analyses of skull morphology indicated that the coastal samples represented more than one population. Torres et al. analyzed 304 samples collected in the Atlantic from New York to central Florida up to 515 km from shore. Using CART (Classification and Regression Tree) analysis, they found a statistically significant break in the coastal compared with offshore ecotypes at 34 km offshore.

Rosel et al. (2009) identified a minimum of five populations in near-shore coastal and estuarine areas along the U.S. Atlantic coast from New York to Florida. They analyzed a mitochondrial control region sequence and 18 microsatellites in 404 samples. This study demonstrates that although bottlenose dolphins are a highly mobile animal living in a habitat with no obvious barriers to dispersal, they exhibit a significant amount of genetic structure.

As of 2011, a combination of genetic analyses, photo-ID studies and field work has identified 15 bottlenose dolphin stocks along the U.S. Atlantic coast (Maine to Florida Keys) (Caldwell 2001; Garrison et al. 2003; Gubbins 2002a, 2002b, 2002c; Gubbins et al. 2003; Litz 2007; Mazzoil et al. 2005; McLellan et al. 2003; Read et al. 2003; Rosel et al. 2009; Speakman et al. 2006; Stolen et al. 2007; Urian et al. 1999; Weller 1998). These 15 stocks can generally be classified into one of three morphotypes: offshore (n=1), coastal (n=5) or bay/estuarine (n=9). See also the 2011 bottlenose dolphin stock assessment reports for more detail on each stock (http://www.nmfs.noaa.gov/pr/pdfs/sars/).

2.20.6 Life History Traits

Mead and Potter (1990) present life history data based on bottlenose dolphins (n=172) that had stranded on the coasts of North Carolina, Virginia and Maryland between 1972 and 1980. They found that females attain sexual maturity at a mean age of 10.64 years. There were not enough males in their sample to estimate male maturity. The youngest mature male in their study was at least 8 years old. Sergeant et al. (1973) estimated that males from the northeast coast of Florida mature at 10-15 years. In summarizing available life history data for the Florida population, Odell (1975) found that females become sexually mature at 5-12 years (length = 220-235cm) and males at 10-13 years (length = 245-260cm). Mead and Potter (1990) documented a peak in calving in the spring (March) for the population along the central U.S. Atlantic coast. McFee et al. (2007) found two peaks in dolphin strandings along the coast of South Carolina: spring and autumn. These two seasons were highly significant for neonate strandings indicating a bimodal reproductive cycle.

2.20.7 Food Habits

Wells and Scott (1999) report bottlenose dolphins' diet includes a variety of fish and squid with a preference for sciaenids, scombrids, and mugilids. Bottom dwelling fish and noise producing fish are also both important.

Mead and Potter (1990) analyzed the stomach contents of 172 bottlenose dolphins that stranded along the coasts of North Carolina, Virginia, and Maryland from 1972-1980. They found that three prey species were dominant: weakfish or sea trout (*Cynoscion regalis*), croaker (*Micropogonias undulatus*), and spot (*Leiostomus xanthurus*). Gannon and Waples (2004) analyzed the stomach contents of 146 bottlenose dolphins (coastal estuary morphotype, no offshore samples were included) that had stranded along the Mid-Atlantic coast or were caught incidentally in fishing gear from 1993-2001. The most frequently occurring prey were sciaenid fishes. In terms of mass, croaker was highest in dolphins that had stranded in estuaries while weakfish was highest in the stomachs of dolphins stranded on ocean beaches. Inshore squid (*Loligo* sp.) were also found in the ocean dolphins' diet but not in that of estuarine dolphins.

Barros and Wells (1998) found only fish in the stomach contents of 19 dolphins that had stranded in Sarasota Bay, Florida. Many of the prey species identified were associated with sea grass beds to a certain extent and the main species were soniferous. Berens McCabe et al. (2010) also analyzed stomach content analysis of stranded dolphins (n=15) and prey availability in Sarasota Bay, Florida. The most important prey were: gulf toadfish (*Opsanus beta*), pinfish (*Lagodon rhomboides*), ladyfish (*Elops saurus*) and spotted seatrout (*Cynoscion nebulosus*). They also found that although soniferous fish were only 6.3% of the available prey, they made up 51.9% of the dolphins' diet. The dolphins selected against fish in the families *Gerridae*, *Clupeidae* and *Sparidae*.

In their study on group-hunting in Cedar Key, Florida, Gazda et al. (2005) found that individual dolphins have specific behavioral roles (e.g., driver vs. barrier dolphin). They observed two different groups that in each group an individual dolphin that was always responsible for herding the fish.

2.20.8 Health

2.20.8.1 Strandings

Bottlenose dolphin strandings have been documented along most of the U.S. Atlantic coast. The largest stranding event occurred from June of 1987 to May of 1988 when a total of 742 dolphins were found dead from New Jersey to Florida. (See the Disease section below for details.) It was estimated that over 50% of the inshore population had died in the affected areas (Lipscomb et al. 1994). This event led to the listing of coastal bottlenose dolphins as "Depleted" under the MMPA.

McFee and Hopkins-Murphy (2002) analyzed bottlenose dolphin strandings along the coast of South Carolina 1992-1996. They found that 49% of the strandings had occurred between April and July and the greatest number had occurred in July. There was an increase in strandings in the northern part of the state from November to March. Although neonate strandings occurred during every month except for March and October, 55% of the neonate strandings happened between May and July. Twenty-three percent of the dolphins had died as a result of human interaction (including fisheries interactions). McFee et al. (2007) analyzed South Carolina stranding data from 1997-2003. They found that the number of strandings peaked in the spring and autumn. Rope entanglements were the most common human interaction.

Given the complex population structure along the U.S. Atlantic coast, stranded dolphins were assigned to a stock based on genetics, location, and/or season as much as possible. These data are summarized in Table 2.20.2. More bottlenose strandings were attributed to the coastal southern migratory stock than any other (n=588, 38%). For some stocks only cumulative stranding data were available; these are summarized in Table 2.20.3. See Waring et al. 2011 for details.

Table 2.20.2 Bottlenose Dolphin Strandings Potentially Attributed to a Stock by Year

Stock Area	2004	2005	2006	2007	2008	Total
Western North Atlantic, offshore	Na	Na	Na	Na	Na	Na
Western North Atlantic, coastal northern migratory	121 (24)	85 (11)	98 (17)	94 (12)	86 (13)	484 (77)
Western North Atlantic, coastal southern migratory	134 (31)	124 (21)	104 (16)	118 (19)	108 (20)	588 (107)
Northern North Carolina Estuarine System	83 (25)	93 (16)	79 (14)	88 (13)	79 (13)	207 (81)
Southern North Carolina Estuarine System	27 (5)	12 (3)	13 (2)	12 (4)	14 (4)	78 (18)
Charleston Estuarine System	12 (2)	10 (3)	13 (2)	14 (3)	Na	49 (10)
Indian River Lagoon Estuarine System	32 (8)	35 (5)	38 (13)	51 (10)	Na	156 (36)
Total	409 (95)	359 (59)	345 (64)	377 (61)	287 (50)	1562 (329)

(The number of dolphins that showed evidence of human interaction is in parentheses) (Waring et al. 2011)

Table 2.20.3 Bottlenose Stocks for which Only Cumulative Data is Available

Stock Area	Years	# of Strandings
Western North Atlantic, coastal, S. Carolina/Georgia	2004-2008	128 (5)
Western North Atlantic, coastal northern Florida	2004-2008	78 (3)
Western North Atlantic, coastal, central Florida	2004-2008	82 (6)
Northern Georgia/Southern South Carolina Estuarine System	2003-2007	51 (0)
Southern Georgia Estuarine System	2003-2007	15 (0)
Jacksonville Estuarine Stock	2003-2007	16 (0)
Biscayne Bay	2003-2007	3* (0)
Florida Bay	2003-2007	7 (0)
Total		380 (14)

(The number of cases where evidence of human interaction was found is in parentheses)
*One of these animals was identified as an offshore morphotype.

2.20.8.2 Contaminants

Kuehl et al. (1991) analyzed chemical residues (e.g., pesticides, polychlorinated biphenyl (PCB), polybrominated biphenyl (PBB), and polybrominated diphenyl ether (PBDPE)) in blubber and/or organ tissue from bottlenose dolphins that had died during the 1987-88 epizootic. They found that, on average, levels were higher in adult males than females. They also found that the bottlenose dolphins analyzed had higher concentrations than either common or white-sided dolphins from the western North Atlantic.

Since that study, many studies have been performed on contaminant levels in U.S. Atlantic bottlenose dolphins (Fair et al. 2010, 2007; Houde et al. 2006; Lahvis et al. 1995; Wells et al. 2005; Yordy et al. 2010a, 2010b, 2010c). Fair et al. (2010, 2007) analyzed contaminant blubber burdens in wild bottlenose dolphins from the Indian River Lagoon and Charleston Estuarine populations collected 2003-2005. Dolphins from the two populations had accumulated a similar suite of contaminants. They found that both populations, but especially the animals from the Charleston Estuarine population, exhibited some of the highest levels of contaminants (PCB, DDT, PBDE) documented in marine mammals. The levels they reported are at or above levels at which adverse effects have been described. Fair et al. (2007) and Wells et al. (2005) also showed that adult males and juveniles had burdens that were higher than that of females (due to offloading via pregnancy and lactation). Houde et al. (2006) also analyzed samples from the Indian River Lagoon and Charleston Estuarine populations. They found that dolphins from the Charleston population had much higher levels of PCB and OH-PCB (measured in plasma) than the Indian River Lagoon dolphins but did not differ from animals sampled in Delaware Bay, New Jersey.

Schwacke et al. (2002) found that female dolphins (mostly primiparous) from populations near Beaufort, North Carolina, Sarasota, Florida, and Matagorda Bay, Texas all were at risk for reproductive effects from long term exposure to PCB’s.

Yordy et al. (2010a) analyzed 13 tissues from 4 bottlenose dolphins that had stranded or had been incidentally killed in fishing nets in North Carolina, Virginia, or Florida. Their goal was to provide a description of how POP’s were distributed in bottlenose dolphins. They found that over 90% of POP accumulation occurred in the blubber. Their study suggests that as blubber is metabolized, POP’s may redistribute to other tissues. Schwacke et al. (2011) studied PCB-related toxic effects in bottlenose dolphins in Georgia. Their results showed that anemia, hypothyroidism, and immune system suppression were all associated with the PCB exposure.

Schaefer et al. (2011) reported on the association of total mercury concentrations in blood and skin and endocrine, hepatic, renal and hematological parameters in dolphins that were sampled in Indian River Lagoon, Florida and Charleston Harbor, South Carolina. Stavros documented trace elements in stranded and wild-caught bottlenose dolphins from South Carolina to Indian River Lagoon, Florida, from 2000-2008.

2.20.8.3 Disease

As described in the Stranding section above, a large bottlenose dolphin die-off occurred June 1987 to May 1988 from New Jersey to Florida. Lipscomb et al. (1994) analyzed lung and lymph node samples from 79 of these dolphins. Their results showed that morbillivirus-induced disease was prevalent in dolphins that had died during this epizootic. The frequent occurrence of mobilliviral antigen in tissues and various bacterial and fungal infections indicated moribillivirus-mediated immunosuppression. Bossart et al. (2010) looked for morbillivirus antigens in samples taken in 2003-2007 from free-ranging bottlenose dolphins from the Indian River Lagoon and Charleston Estuarine populations (n=234). They found that 9.8% of the Indian River Lagoon animals sampled were positive for morbillivirus titers. Some of the animals sampled were old enough to have survived the 1987-1988 die-off, but others were not. The Charleston animals were seronegative. These results show that morbillivirus infections are occurring in the Indian River Lagoon without a large scale epizootic.

Fauquier et al. (2009) documented the prevalence of lungworm in live and dead stranded bottlenose dolphins in southwest Florida from 2003 to 2005. They found lungworm in 77% of the dead animals necropsied; 0% blowhole swabs and only 3% fecal cytology from live dolphins showed lungworm infection. Neonates and calves showed the highest prevalence of active infection. This study also provided evidence of transplacental infection (as demonstrated by an infected stillborn calf).

Murdoch et al. (2010) studied lacaziosis in bottlenose dolphins in Florida. They found that estuarine dolphins had a higher rate of infection than dolphins found in the Atlantic Ocean. In their study on papillomavirus antibody presence, Rehtanz et al. (2010) found that 90% of the wild dolphins they sampled from Indian River Lagoon, Florida, and estuarine waters near Charleston, South Carolina, were antibody positive.

2.20.9 Acoustics

Southall et al. (2007) classified *Tursiops* sp. into a mid-frequency cetacean functional hearing group with an estimated auditory bandwith of 150 Hz to 160 Hz. Bottlenose dolphins make three kinds of sounds: whistles, echolocation clicks, and burst-pulse sounds (Wells and Scott 2009). Wells and Scott (2009) summarized the purpose of each sound type: individual dolphins produce a unique “signature whistle” that is used for identity; echolocation clicks range from 40-130 kHz and are likely used to find prey, to navigate, and to spot predators; the burst-pulse sounds are likely used in social interactions.

Janik (2000) found that wild bottlenose dolphins in Moray Firth, Scotland, exhibit vocal matching–one dolphin responds to another by imitating the same whistle type. This matching occurred over distances up to 580 m. In their study of the resident population of bottlenose dolphins in Sarasota Bay, Florida, Cook et al. (2004) found that signature or probable signature whistles make up ~52% of all whistles produced by free-ranging dolphins.

Ibsen et al. (2009) studied the functional bandwidth of a 19-year-old captive female bottlenose dolphin. They found that during echolocation, the dolphin paid attention to frequencies 29-42 kHz (her functional bandwidth). The upper end of this range is close to the upper limit of her hearing range and the lower limit seemed to be a function of background noise, target reflection and frequency content of the clicks themselves (Ibsen et al. 2009).

There are many studies of temporary threshold shifts in bottlenose dolphins (Finneran et al. 2005, 2005, 2010, 2000; Mooney et al. 2009; Nachtigall et al. 2003; Schlundt et al. 2000). Sound exposure level and duration of exposure are both important factors. Finneran et al. (2000) used a behavioral response paradigm to examine masked underwater hearing thresholds before and after exposure to simulated underwater explosions. Although they did not find any change in masked-hearing thresholds, they did find a behavioral change in the two dolphins (5 kg at 9.3 km and 5 kg at 1.5 km).

Acevedo-Gutierrez and Stienessen (2000) analyzed the sounds groups of feeding and non-feeding bottlenose dolphins made around Isla del Coco, Costa Rica. They classified the sounds into three types: whistles, click trains, and pulse bursts. Their results showed that feeding dolphins produced more whistles per minute than non-feeding dolphins. Feeding dolphins produced more whistles than click trains or pulse bursts.

2.20.10 Fisheries By-Catch and Entanglement

The Bottlenose Dolphin Take Reduction Team (BDTRT) was created out of concern for the serious injury and deaths of coastal bottlenose dolphins interacting with 9 fisheries: the North Carolina inshore gillnet, Southeast Atlantic gillnet, Southeastern U.S. shark gillnet, U.S. Mid-Atlantic coastal gillnet, Atlantic blue crab trap/pot, Mid-Atlantic haul/beach seine, North Carolina long haul seine, North Carolina roe mullet stop net, and Virginia pound net. The Take Reduction Plan developed by the BDTRT included provisions for research and education, modifications of fishing practices for small, medium, and large-mesh gillnet fisheries from New York to Florida and seasonal closures for certain commercial fisheries in state waters.

Table 2.20.4 Most Recent Reported/Estimated Incidental Mortality of Bottlenose Dolphins by Stock

Stock	Fisheries Included	Total
Western North Atlantic, offshore (2006)	Northeast Sink Gillnet, Mid-Atlantic Gillnet	Min = 0 Max = 0
Western North Atlantic, coastal northern migratory (2008)	Mid-Atlantic Gillnet, Virginia Pound Net, Beach-based Gillnet Gear, Blue Crab Pot, Other Pot, Fishery Research	Min = 7.3 Max = 7.9
Western North Atlantic, coastal southern migratory (2008)	Mid-Atlantic Gillnet, Virginia Pound Net, Blue Crab Pot, Other Pot, Fishery Research	Min = 9.5 Max = 43.0
Western North Atlantic, coastal, S. Carolina/Georgia	Unknown	Unknown
Western North Atlantic, coastal northern Florida (2008)	No observer coverage in 2008.	Unknown
Western North Atlantic, coastal, central Florida (2008)	Southeast Gillnet	Min = 0 Max = 0
Northern North Carolina Estuarine System (2008)	Mid-Atlantic Gillnet, Virginia Pound Net, Beach-based Gillnet Gear, Blue Crab Pot, Other Pot, Fishery Research	Min = 2.7 Max = 14.9
Southern North Carolina Estuarine System (2008)	Mid-Atlantic Gillnet, Blue Crab Pot, Other Pot, Fishery Research	Min = 0.5 Max = 0.9
Charleston Estuarine System	Unknown	Unknown
Northern Georgia/Southern South Carolina Estuarine System	Unknown	Unknown
Southern Georgia Estuarine System	Unknown	Unknown
Jacksonville Estuarine System	Unknown	Unknown
Indian River Lagoon Estuarine System	Unknown	Unknown
Biscayne Bay	Unknown	Unknown
Florida Bay	Unknown	Unknown

(Waring et al. 2011)
(Minimums and maximums are presented, as there is uncertainty associated with assigning some by-caught dolphins to a specific stock given their spatial and temporal overlap. Year of estimate is in parentheses after stock name. Observer coverage is not available in all areas.)

2.20.11 Vessel Interactions

Bejder et al. (2006) compared dolphin abundance within adjacent tourism and control sites in Shark Bay, Australia, for three consecutive 4.5-year periods (total time period = 13.5 years). Tourism boats increased from 0 to 1 then to 2 dolphin-watching boats. The change from no tourism to one operator did not significantly impact dolphin abundance. The change from one to two operators caused a 14.9% (95% CI= -20.8 to -8.23) decrease in abundance. The authors conclude that this disturbance was probably not a major threat to the bottlenose dolphin population in Shark Bay because it is large and genetically diverse. They do point out that a similar decline in a smaller, more isolated population could be devastating. In a study on the impact of dolphin-watching tour boats on bottlenose dolphins in the Bay of Islands, New

Zealand, Constantine et al. (2004) found that bottlenose dolphin resting behavior decreased as boat number increased.

Buckstaff (2004) studied the effects of watercraft on bottlenose dolphin acoustic behavior. She found that whistle rate increased and suggested this either served to bring dolphins together or to compensate for signal masking.

2.20.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear.

Stone (2003) studied the effects of seismic activity on cetaceans in UK waters. She reported that sighting rates of small odontocetes were significantly lower when large airgun arrays were being fired. She also found that in general, small odontocetes showed the strongest avoidance behavior to seismic noise (compared to baleen, killer, sperm and pilot whales). The 2010 Report of the Working Group on Marine Mammal Ecology (ICES 2010) stated that very little is known about the impact of the construction and operation of offshore wind-farms. Madsen et al. (2006b) reviewed available current information on wind farms and marine mammals and wrote that the noise impact is more severe during construction than operation. This was found to be the case for impacts to harbor porpoise (see Carstensen et al. [2006], Koschinski et al. [2003]). David (2006) wrote that pile driver-generated noise has the potential to impact dolphins by temporary displacement and interference with communication. Pile-driving noise associated with the construction of two wind turbines off the coast of northeast Scotland was related to noise exposure criteria for bottlenose dolphins. It was concluded that auditory injury would have occurred within 100 m of the pile-driving and behavioral disturbance could have occurred up to 50 km away (Bailey et al. 2010). The 2011 Report of the Working Group of Marine Mammal Ecology (ICES 2011) reported that porpoise detection (T-POD data) was lower during construction of a tidal turbine compared to before and after construction.

2.20.13 Data Gaps and Research Recommendations

The best abundance estimates for bottlenose dolphins are now anywhere from 6 to 8 years old and should be updated. No abundance estimates are available for some of the populations (e.g., the newly defined estuarine stocks; see Table 2.20.1). Given the recent revision of bottlenose dolphin stock delineations, it is likely that our understanding of bottlenose dolphin stock structure will continue to evolve. Studies have shown differences in food habits (Gannon and Waples 2004) and contaminant burdens (Fair et al. 2010, 2007; Houde et al. 2006) between the various U.S. Atlantic populations. More specific information about each population (e.g., food habits, life history, contaminant burden, etc.) is required.

2.21 PANTROPICAL SPOTTED DOLPHIN (*STENELLA ATTENUATA*)

2.21.1 Legal Status

The pantropical spotted dolphin is not listed under the Endangered Species Act. The IUCN Red List lists its status as “Least Concern.” It is afforded protection in U.S. waters under the MMPA.

2.21.2 General Distribution

Pantropical spotted dolphins are found in tropical and subtropical waters around the world (Perrin 2009b; Perrin and Hohn 1994). There are observations of pantropical spotted dolphins along the U.S. Atlantic coast in the BOEM North Atlantic, Mid-Atlantic, South Atlantic and Straits of Florida Planning Areas. Sighting, stranding, and fishery bycatch locations of pantropical spotted dolphins are shown in Figure 2.21-1.

2.21.3 General Abundance

The total population of pantropical spotted dolphins along the U.S. Atlantic coast is unknown. Waring et al. (2011) report a best estimate of 4,439 (CV=0.49) based on a combination of ship and aerial surveys conducted June-August 2004 from Florida to the Bay of Fundy. NMFS divided these surveys into two areas: the “North Atlantic” includes waters north of Maryland (38°N) to the Bay of Fundy (45°N) and the “South Atlantic” includes waters from Maryland (38°N) south to Florida (27.5°N). The 2004 pantropical spotted dolphin estimate for the North Atlantic is 0 (CV=0). The 2004 estimate for the South Atlantic is 4,439 (CV=0.49). The NMFS North Atlantic Survey was conducted in part of the BOEM North Atlantic Planning Area; the NMFS South Atlantic Survey was conducted in parts of the BOEM Mid-Atlantic and South Atlantic Planning Areas. It is important to note that the surveys did not necessarily cover the entire BOEM planning areas but likely sampled a subsection of these areas based on prior knowledge of species distribution and survey goals. The 2004 survey did not cover waters within the Straits of Florida planning area.

2.21.4 Habitat Preference

In the Atlantic Ocean, pantropical spotted dolphins are a “dolphin of the high seas and oceanic islands” (Perrin and Hohn 1994). In the Pacific Ocean, the offshore form of the pantropical spotted dolphin is most abundant in waters that have a sharp thermocline, SST of 25°C or higher and salinity less than 34ppt (Perrin 2009b). The NMFS 2004 survey that provided the most recent abundance estimate (n=4,439, CV=0.49) occurred along the outer continental shelf and continental slope where water depth was equal or greater than 50 m (Waring et al. 2011).

2.21.5 Stock Structure

The western North Atlantic population is considered separate from the Gulf of Mexico population for management purposes but additional data are needed to elucidate stock structure along the U.S. Atlantic coast (Waring et al. 2011).

2.21.6 Life History Traits

Myrick et al. (1986) studied the reproductive biology of female spotted dolphins that had been incidentally killed in the yellowfin tuna fishery in the eastern tropical Pacific Ocean from 1973-1981. They found that average age at sexual maturity for all years ranged from 10.7-12.2 years (mean = 11.4). The calving interval was 3.03 years and lactation lasted 1.66 years. Age at first maturity for males killed in the same fishery (1973-1982) was 14.7 years (Hohn et al. 1985). In their summarization of data from the Pacific Ocean, Perrin and Hohn (1994) found that gestation is 11.2-11.5 months.

2.21.7 Food Habits

Perrin and Hohn (1994) report that the pantropical spotted dolphins' diet consists of small epipelagic fish, squid, and crustaceans. Robert and Chivers (1997) analyzed the contents of 428 stomachs of animals incidentally killed in tuna-purse seine nets in the eastern tropical Pacific from 1989-1991. Lantern fish (family *Myctophidae*) were the most frequently observed fish and flying squid (family *Ommastrephidae*) were the most frequently observed cephalopod. The high frequency of mesopelagic fish, coupled with the occurrence of fuller stomachs in dolphins that were killed in the morning, suggests that pantropical spotted dolphins feed at night when mesopelagic species are moving towards the surface.

Bernard and Hohn (1989) analyzed the stomach contents of 11 pregnant and 12 lactating spotted dolphins that were incidentally killed in the yellowfin tuna fishery in the eastern tropical Pacific from 1971-1985. They found that the diet of the two groups differed greatly: pregnant dolphins consumed more squid and lactating dolphins consumed more fish (by mass and proportion). Archer and Robertson (2004) reported that calves start taking food at 6 months but some nurse until 2 years.

2.21.8 Health

2.21.8.1 Strandings

Four pantropical spotted dolphins stranded in Florida in 1999 and 3 were reported stranded along the U.S. Atlantic coast from 2001 to 2005. One of these stranded in South Carolina and two in Florida (Waring et al. 2011). The pantropical dolphin that stranded in South Carolina was considered part of a Mid-Atlantic Small Cetacean UME declared from July 2004 to January 2005–no single cause was identified for this UME (Waring et al. 2011). (Waring et al. 2011). None of the stranded animals showed signs of fishery or human interaction. A review of Northeast and Southeast U.S. marine mammal stranding records for 2005-2009 yields no strandings of pantropical spotted dolphins along the Atlantic coast.

2.21.8.2 Contaminants

Andre et al. (1990a) found cadmium levels for most organs between 1 and 5 mg Cd/kg in pantropical spotted dolphins from the Pacific Ocean; age, body weight, and location of capture all influenced the levels. In a separate study by Andre et al. (1990b), mercury levels for most organs measured between 1 and 5 mg Hg/kg in pantropical spotted dolphins from the Pacific Ocean.

2.21.8.3 Disease

Perrin (2001) reports that parasitism may be a significant cause of natural mortality. Perrin and Powers (1980) found that high mortality appeared to be correlated with infestation by the nematode *Crassicauda* sp. or some related factor.

2.21.9 Acoustics

Perrin and Hohn (1994) summarized a report by Popper (1980) that pantropical spotted dolphins produce pulses ranging up to 150 kHz. Southall et al. (2007) classified *Stenella* spp. into a mid-frequency cetacean functional hearing group with an estimated auditory bandwith of 150 Hz to 160 Hz. No other specific information about pantropical spotted dolphin acoustics is available.

2.21.10 Fisheries By-catch and Entanglement

The only fishery in the U.S. Atlantic with reported incidental mortality is the pelagic long-line (Waring et al. 2011). The mean annual mortality of undifferentiated spotted dolphins (*Stenella frontalis, Stenella attenuata*) for this fishery was 6 (CV=1) for the years 2001-2005.

2.21.11 Vessel Interactions

A review of Northeast and Southeast U.S. stranding data between 2005 and 2009 yielded no records of vessel interactions with pantropical spotted dolphins.

2.21.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear.

Stone (2003) studied the effects of seismic activity on cetaceans in UK waters. She reported that sighting rates of small odontocetes were significantly lower when large airgun arrays were being fired. She also found that in general, small odontocetes showed the strongest avoidance behavior to seismic noise (compared to baleen, killer, sperm and pilot whales). The 2010 Report of the Working Group on Marine Mammal Ecology (ICES 2010) stated that very little is known about the impact of the construction and operation of offshore wind-farms. Madsen et al. (2006b) reviewed available current information on wind farms and marine mammals and wrote that the noise impact is more severe during construction than operation. This was found to be the case for impacts to harbor porpoise (see Carstensen et al. (2006), Koschinski et al. (2003)). David (2006) wrote that pile driver-generated noise has the potential to impact dolphins by temporary displacement and interference with communication. The 2011 Report of the Working Group of Marine Mammal Ecology (ICES 2011) reported that porpoise detection (T-POD data) was lower during construction of a tidal turbine compared to before and after construction. No information specific to pantropical spotted dolphins was found.

2.21.13 Data Gaps and Research Recommendations

The best abundance estimate is now 7 years old and needs to be updated. There is very little recent information on distribution outside of the summer months. Most of the information about the pantropical spotted dolphins' food habits, contaminant burden or stock structure comes from parts of their range that is outside of the U.S. Atlantic coast (e.g., from the Pacific Ocean).

2.22 ATLANTIC SPOTTED DOLPHIN (*STENELLA FRONTALIS*)

There are two forms of the Atlantic spotted dolphin. One is large and heavily spotted; the other is smaller and less spotted–see Habitat Preference and Stock Structure sections below.

2.22.1 Legal Status

The Atlantic spotted dolphin is not listed under the Endangered Species Act. The IUCN Red List lists its status as "Data Deficient." It is afforded protection in U.S. waters under the MMPA.

2.22.2 General Distribution

This species is found in the warm temperate and tropical Atlantic Ocean only (Perrin 2009). They are found in the BOEM North Atlantic, Mid-Atlantic, South Atlantic and Straits of Florida Planning Areas (Perrin et al. 1994a, Waring et al. 2011). Sighting, stranding and fishery bycatch locations of Atlantic spotted dolphins are shown in Figure 2.22-1.

2.22.3 General Abundance

Waring et al. (2011) provided a best estimate of 50,978 (CV=0.42) from Florida to the Bay of Fundy, June-August 2004. The portion of the NMFS June-August 2004 survey that was conducted from Maryland to the Bay of Fundy (encompassing the North and part of the Mid-Atlantic planning areas) was used for an abundance estimation of 3,578 (CV=0.48) Atlantic spotted dolphins (Palka 2006). A survey of the outer continental shelf between 27.5 ^{o}N and 38 ^{o}N in June-August 2004 produced an estimate of 47,400 (CV=0.45). Most of these animals were sighted north of Cape Hatteras, North Carolina, an area within the Mid-Atlantic Planning Area (Waring et al. 2011). No abundance estimates are available for the Straits of Florida Planning Area because the NMFS 2004 surveys occurred north of this area.

2.22.4 Habitat Preference

The large, heavily-spotted form prefers the continental shelf and shelf-break usually within the 200 m curve (Perrin 2009a). The smaller, less spotted version is found in pelagic, offshore waters and near oceanic islands (Perrin et al. 1994a). Hamazaki (2002) correlated environmental variables associated with spotted dolphin (Atlantic and pantropical) sightings and predicted a habitat ranging over the southern half of the mid-western Atlantic in June and expanding to the entire mid-western Atlantic in August. Of the mid-Atlantic offshore species studied, spotted dolphins were seen in the warmest waters and the mildest slopes. A satellite-tagging study done on an Atlantic spotted dolphin in the Gulf of Mexico by Davis et al. (1996) generated information on travel and dive patterns of the species. During the 24 days the tag was attached, mean minimum travel distance was 72 km/day. The animal ranged within the 20-60m isobaths at a mean depth of 32.6 m. The deepest dives made were 40-60 m deep and lasted up to 6 minutes, though most dives were less than 2 minutes in duration and to depths of less than 30 m (Davis et al. 1996). Griffin and Griffin (2003) examined habitat partitioning by Atlantic spotted dolphins, bottlenose dolphins, and loggerhead sea turtles in the Gulf of Mexico and found that, at depths of 20-180 m, the spotted dolphin was the most common species. Other variables, such as distance from shore, temperature, salinity, chlorophyll, and transmittance, were also examined. Griffin and Griffin (2004) also examined temporal patterns of Atlantic spotted dolphin distribution off the Florida Gulf of Mexico shelf and found consistent season variations, with higher densities found during the cooler months of November-May. Off the coast of Brazil, Atlantic spotted dolphins have been observed over the continental shelf and slope in waters up to 100 m of depth (Moreno et al. 2005). In southern Brazil, strandings occur predominantly in the summer months when water temperatures are influenced by the warm Brazil current (Moreno et al. 2005).

2.22.5 Stock Structure

Adams and Rosel (2006) used mitchondrial and nuclear DNA analyses to clarify Atlantic spotted dolphin stock structure in the Western North Atlantic and Gulf of Mexico. They identified three populations: the Gulf of Mexico, Mid-Atlantic Bight (north of 35°N), and the South Atlantic Bight (south of 35°N). Their results also indicate possible differentiation between coastal and offshore dolphins in the Mid-Atlantic Bight area.

2.22.6 Life History Traits

From a long-term study of a population of dolphins on Little Bahama Bank (east of Florida) Herzing (1997) estimated that age of maturity for females was 8-15 years and the inter-calving interval was 2.96 years (range 1-5 years). Herzing also found that lactation could last up to 5 years and visibly pregnant females were also observed lactating.

2.22.7 Food Habits

Perrin et al. (1994) reported a variety of prey, including clupeoid and carangid fishes, squid and halfbeaks (*Hemiramphus*). Aguiar dos Santos and Haimovici (2001) found *Loligo plei* in the stomach of six Atlantic spotted dolphins collected along the coast of Brazil. Bender et al. (2009) through analysis of video archives of Atlantic spotted dolphin foraging behavior, found evidence of maternal teaching. Coordinated feeding, where dolphins herded fish into dense balls, has been observed in the Gulf of Mexico (Fertl and Würsig 1995)

2.22.8 Health

2.22.8.1 Strandings

A total of 15 Atlantic spotted dolphins were reported stranded along the U.S. Atlantic coast (Massachusetts, North Carolina, Georgia, and Florida) from 2001-2005 (Waring et al. 2011). A spotted dolphin examined by Estep et al. (2005) that had stranded in 1993 on the Gulf Coast of Florida exhibited metastasized testicular tumors. Lesions caused by parasitic infections were found in skulls of bycaught spotted dolphins in the Eastern Tropical Pacific by Perrin and Powers (1980) in frequencies that suggested that parasitism is a major cause of natural mortality in the species. Parasitism was also examined in spotted dolphins that had stranded in the Canary Islands between 1992 and 2000 (Jaber et al. 2004, 2006). Eleven of 23 spotted dolphins examined in this study exhibited nonspecific chronic reactive hepatitis lesions (Jaber et al. 2004). A review of Northeast and Southeast U.S. marine mammal stranding records for 2005-2009 yielded 11 strandings of Atlantic spotted dolphins along the Atlantic coast, 4 in North Carolina, 4 in Florida, 2 in Georgia, and 1 in South Carolina.

2.22.8.2 Contaminants

Two Atlantic spotted dolphin calves were included in Watanabe et al.'s (2000) study of PCB's and organochlorine pesticides in small cetaceans stranded along Florida's Atlantic coast. The liver concentrations of several compounds were elevated compared to those in the blubber of marine mammals of various regions, even when normalized to lipid content, suggesting exposure to sources for these chemicals (Watanabe et al. 2000). In vitro studies showed that multiple organelle damage and nuclear budding were observed when spotted dolphin renal cells were exposed to mercury (Wang and Pfeiffer 2001) and that selenium had a possible protective effect on mercury toxicity (Wang et al. 2001). Pfeiffer et al. (2000) explored fuel oil toxicity in the spotted dolphin renal cell culture.

2.22.8.3 Disease

2.22.9 Acoustics

Atlantic spotted dolphins have been the subjects of acoustic research since Wood's work on captive dolphins at Marineland, Florida, in 1953 (Wood 1953). Acoustic research on Atlantic spotted dolphins continued in the 1960s and 1970s (see Caldwell and Caldwell 1966, 1971b; Caldwell et al. 1973, 1971). Steiner (1981) published a comparison of whistle vocalizations in five species of Atlantic dolphins. The recordings of Atlantic spotted dolphins Steiner used were made off the coast of Florida in 1966 and 1967. In their study on free-ranging dolphins, Au and Herzing (2003) found the dolphins' echolocation signals had bi-modal frequency spectra with a low-frequency peak between 40 and 50 kHz and a high-frequency peak between 110 and 130 kHz. Communication signals are discussed by Lammers et al. (2003). Their broad spectrum sampling found that the social signals produced by spotted dolphins span the full range of their hearing sensitivity and are spectrally quite varied. Burst pulses are predominately ultrasonic. Baron et al. (2008) discuss differences in spotted dolphin vocalizations between the Gulf of Mexico and the U.S. Atlantic coast.

2.22.10 Fisheries By-Catch and Entanglement

The only recent fishery bycatch information for Atlantic spotted dolphins in the Atlantic is for the pelagic longline fishery. One Atlantic spotted dolphin was reported by fishery observers as caught in a longline and released injured in 2003 (Garrison and Richards 2004). Waring et al. (2010) reported a mean annual mortality of six (CV=1) undifferentiated (pantropical and Atlantic) spotted dolphins in the pelagic longline fishery for the years 2001-2005.

2.22.11 Vessel Interactions

In 2007, one Atlantic spotted dolphin stranding mortality in the Northeast and Southeast U.S. Marine Mammal Stranding Network databases exhibited signs of interaction with a boat propeller.

2.22.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear.

Stone (2003) studied the effects of seismic activity on cetaceans in UK waters. She reported that sighting rates of small odontocetes were significantly lower when large airgun arrays were being fired. She also found that in general, small odontocetes showed the strongest avoidance behavior to seismic noise (compared to baleen, killer, sperm and pilot whales). The 2010 Report of the Working Group on Marine Mammal Ecology (ICES 2010) stated that very little is known about the impact of the construction and operation of offshore wind-farms. Madsen et al. (2006b) reviewed available current information on wind farms and marine mammals and wrote that the noise impact is more severe during construction than operation. This was found to be the case for impacts to harbor porpoise (see Carstensen et al. [2006], Koschinski et al. [2003]). David (2006) wrote that pile driver-generated noise has the potential to impact dolphins by temporary displacement and interference with communication. The 2011 Report of the Working Group of Marine Mammal Ecology (ICES 2011) reported that porpoise detection (T-POD data) was lower during construction of a tidal turbine compared to before and after construction.

Atlantic spotted dolphin reactions to seismic survey airgun operations were studied off the coast of Angola (Weir 2008a). Compared to humpbacks and sperm whales which were also studied, Atlantic spotted dolphins exhibited the most marked overt response to airgun sound. While there was no evidence for lasting or large scale displacement, the Atlantic spotted dolphin encounters occurred at a significantly greater distance when airguns were being fired compared to during periods of no seismic activity. Positive-approach behavior occurred only when guns were not in use.

2.22.13 Data Gaps and Research Recommendations

The best abundance estimate for Atlantic spotted dolphins in the Atlantic Ocean is 7 years old. There is limited information on Atlantic spotted dolphin distribution and abundance outside of the summer months (June-September). Information on contaminant exposure, life history, and diet in Atlantic spotted dolphins is sparse for the western Atlantic.

2.23 SPINNER DOLPHIN (*STENELLA LONGIROSTRIS*)

2.23.1 Status

The spinner dolphin is not listed under the Endangered Species Act. The species is considered “Data Deficient” on the IUCN Red List. It is afforded protection in U.S. waters under the MMPA.

2.23.2 General Distribution

The spinner dolphin is pantropical, found between approximately 30-40°N and 20-40°S (Jefferson et al. 2007) in tropical to warm temperate waters (Leatherwood and Reeves 1983a). Distribution in the Atlantic is not well known (Waring et al. 2011) but Leatherwood and Reeves (1983a) report they that the presence of spinner dolphins has been confirmed from Cape Hatteras, North Carolina, to Rio de Janeiro, Brazil. Stranding records exist from North Carolina and Florida. It appears that spinner dolphins are found in the BOEM Mid-Atlantic, South Atlantic and Straits of Florida Planning Areas. Sighting, stranding and fishery bycatch locations of spinner dolphins are shown in Figure 2.23-1.

2.23.3 General Abundance

The number of spinner dolphins off the U.S. Atlantic coast is unknown. The available data are not adequate to calculate a minimum estimate (Waring et al. 2011).

2.23.4 Habitat Preference

As summarized in Waring et al. (2011), spinner dolphins have only been sighted in deeper (>2000 m) oceanic waters along the U.S. coast. Leatherwood and Reeves (1983a) stated they are primarily in deep water/pelagic areas but move into shelf waters off the southeastern U.S.

2.23.5 Stock Structure

There are 4 subspecies of spinner dolphins: *S.l. longirostris* is globally distributed; *S.l. orientalis* is endemic to the eastern tropical pacific; *S.l. centroamericana* is located off the west coast of Central America; and *S.l. roseiventris* is a dwarf form located in central Southeast Asia. Farro et al. (2008) developed eight polymorphic microsatellite DNA markers from skin samples of 65 spinner dolphins. The tissue was collected from dolphins that stranded in the Fernando de Noronha Archipelago, Brazil. They found low observed heterozygosity, suggesting a high level of inbreeding in the southern Atlantic.

Andrews et al. (2010) studied the genetic relationship of spinner dolphins around the Hawaiian Islands that exhibit two types of social behavior: fluid and stable groups. They analyzed mtDNA control region sequences and 10 microsatellite loci. Their results showed that dolphins from the Southeast archipelago which form ever-changing groups had less genetic structuring than dolphins in the Northwest archipelago that form more stable groups. The western North Atlantic stock is managed separately from the Gulf of Mexico population but research is needed to confirm this separation (Waring et al. 2011).

2.23.6 Life History Traits

As summarized in Perrin (2009c), females become sexually mature at 4-7 years while males are 7-10 years at maturity. Gestation is approximately 10 months, lactation lasts 1-2 years and the calving interval is around 3 years.

2.23.7 Food Habits

Spinner dolphins feed on mesopelagic fish and squid in the Pacific (Dolar et al. 2003; Perrin and Gilpatrick 1994). A study off the coasts of Hawaii, Oahu, and Lana (Benoit-Bird and Au 2003) documented that spinner dolphins followed the horizontal and vertical migratory patterns of their prey. Pairs of dolphins were also seen foraging cooperatively within larger groups.

Benoit-Bird and Au (2009a, 2009b) used a multibeam echosounder to make observations of spinner dolphins and their prey along the coast of Oahu, Hawaii. They found groups of dolphins (n=16-28) working together to herd prey into higher densities, thus potentially allowing access to more prey than an individual dolphin foraging alone would have. Benoit-Bird and Au (2009b) have hypothesized that this cooperative feeding was maintained by acoustic communication (e.g., whistles). However, in their study, they recorded very few whistles when the dolphins appeared to be foraging. They suggest that spinner dolphins may communicate with only a few cues when cooperatively foraging.

2.23.8 Health

2.23.8.1 Strandings

Mead et al. (1980) reported on a stranding event on the West coast of Florida in 1976 (n=28). A review of Northeast and Southeast U.S. marine mammal stranding records for 2005-2009 yields 2 strandings of spinner dolphins along the Atlantic coast, both in Florida in 2008.

2.23.8.2 Contaminants

As described in Perrin and Gilpatrick (1994), relatively high levels of mercury and DDT have been documented in spinner dolphins. Tanabe et al. (1988) described how spinner dolphins metabolize PCB’s.

2.23.8.3 Disease

Perrin and Gilpatrick (1994) wrote that a variety of endoparasites have been documented in spinner dolphins (e.g., nematodes, trematodes, cestodes and acanthocephalans). Cowam and Walker (1979) necropsied 8 spinner dolphins killed in tuna purse-seines. They described a variety of naturally occurring diseases and various symptoms associated with capture. Migaki et al. (1990) reported a case of toxoplasmosis (*Toxoplasma gondii*) in a wild, free-living male spinner dolphin that was found beached in Oahu, Hawaii.

2.23.9 Acoustics

Southall et al. (2007) classified *Stenella* sp. into a mid-frequency cetacean functional hearing group with an estimated auditory bandwith of 150 kHz to 160 kHz. Spinner dolphins produce whistles and variable burst-pulse sounds, the function of which is only starting to be understood (Perrin 2009c).

2.23.10 Fisheries By-Catch and Entanglement

There are no records of fishery interactions for the most recent time period reported (2001-2005) (Waring et al. 2011), or for 2005-2009. Before 2001, by-catch was observed in the pelagic long-line and now-banned pelagic driftnet fisheries.

2.23.11 Vessel Interactions

A review of Northeast and Southeast U.S. stranding data between 2005 and 2009 yielded one record of a vessel interaction with a spinner dolphin. A 2008 Florida stranding mortality had wounds consistent with propeller strike. Courbis and Timmel (2009) studied spinner dolphin behavior in relation to vessel traffic and human swim activities in three Hawaiian bays in 2002. They used aerial behavior of the dolphins as a metric because it is noninvasive, has been shown to indicate changes in energy levels (Norris et al. 1985), and could be compared to other studies. They found changes in the aerial behavior of dolphins at Kealake'akua Bay but suggested further studies are needed to understand the impact of vessel/human swimmers on spinner dolphins (e.g., changes in resting periods). Timmel et al. (2008) studied the effects of human traffic on the movement patterns of spinner dolphins in Kealakekua Bay, Hawaii. They found that increasing levels of human activity did have a small but measurable impact on the dolphins at this site. Their study was complicated by the lack of a control (vessels/human were always present) and they suggested further research.

2.23.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear.

Stone (2003) studied the effects of seismic activity on cetaceans in UK waters. She reported that sighting rates of small odontocetes were significantly lower when large airgun arrays were being fired. She also found that, in general, small odontocetes showed the strongest avoidance behavior to seismic noise (compared to baleen, killer, sperm, and pilot whales). The 2010 Report of the Working Group on Marine Mammal Ecology (ICES 2010) stated that very little is known about the impact of the construction and operation of offshore wind-farms. Madsen et al. (2006b) reviewed available current information on wind farms and marine mammals and wrote that the noise impact is more severe during construction than during operation. This was found to be the case for impacts to harbor porpoise (see Carstensen et al. [2006], Koschinski et al. [2003]). David (2006) wrote that pile driver-generated noise has the potential to impact dolphins via temporary displacement and interference with communication. The 2011 Report of the Working Group of Marine Mammal Ecology (ICES 2011) reported that porpoise detection (T-POD data) was lower during construction of a tidal turbine compared to before and after construction.

No information specific to spinner dolphins was found.

2.23.13 Data Gaps and Research Recommendations

There is no abundance estimate for spinner dolphins in the U.S Atlantic. There is very little recent information on distribution outside of the summer months. Most of the information about the spinner dolphins' food habits, contaminant burden, disease or stock structure comes from parts of their range that are outside of the U.S. Atlantic coast (e.g., from the Pacific Ocean). Very little is known about spinner dolphins in the western North Atlantic.

2.24 ROUGH TOOTHED DOLPHIN (*STENO BREDANENSIS*)

2.24.1 Legal Status

The rough-toothed dolphin is not listed under the Endangered Species Act. Its status on the IUCN Red List is "Least Concern." It is afforded protection in U.S. waters under the MMPA.

2.24.2 General Distribution

Rough-toothed dolphins exhibit a wide spread distribution and are found in tropical and warm waters around the world (Leatherwood and Reeves 1983a). In the western Atlantic, they are found from the southeastern U.S. to southern Brazil (Jefferson 2009a). Rough-toothed dolphins are potentially found in the BOEM Mid-Atlantic, South Atlantic and Straits of Florida Planning Areas. Sighting and stranding locations of rough-toothed dolphins are shown in figure 2.24-1.

2.24.3 General Abundance

There are no abundance estimates for rough-toothed dolphin along the U.S. Atlantic coast (Waring et al. 2011). They have not been observed in any NMFS surveys along the U.S. Atlantic coast since 1999 (Waring et al. 2011).

2.24.4 Habitat Preference

Baird et al. (2008) reported that rough-toothed dolphins were cited most frequently in waters that are deeper than 1500 m around the Hawaiian Islands. They also found evidence of site fidelity.

2.24.5 Stock Structure

Almost nothing is known of rough-toothed dolphin stock structure world-wide (Jefferson 2009a). The western North Atlantic stock is considered separate from the Gulf of Mexico stock for management purposes, although there is no information to differentiate the two at this time (Waring et al. 2011).

2.24.6 Life History Traits

Very little is known about their life history traits. What is known comes from Japanese waters and is summarized in Jefferson (2009a): males reach sexual maturity at 14 years and 225 cm, females at 10 years and 210-220 cm. Longevity is approximately 32-36 years.

2.24.7 Food Habits

Jefferson (2009a) reported that rough-toothed dolphins feed on a variety of fish and cephalopods from both coastal and oceanic habitats. Pitman and Stinchcomb (2002) described four separate occasions of rough-toothed dolphins feeding on mahimahi (*Coryphaena hippurus*).

2.24.8 Health

2.24.8.1 Strandings

Approximately 72 rough-toothed dolphins stranded along the U.S. Atlantic coast from 2005-2009 (Southeast and Northeast U.S. marine mammal stranding response databases). This includes approximately 70 animals from a 2005 mass-stranding on the Atlantic side of Marathon Island in the Florida Keys, and 2 animals that stranded in North Carolina in 2006.

2.24.8.2 Contaminants

Struntz et al. (2004) measured levels of persistent organic pollutants in rough-toothed dolphins that stranded in the Gulf of Mexico. As with other mammals, they found evidence of maternal offloading to calves.

2.24.8.3 Disease

Jefferson (2009a) reported little is known about diseases in rough-toothed dolphins.

2.24.9 Acoustics

Southall et al. (2007) classified *Steno* spp. in a mid-frequency cetacean functional hearing group with an estimated auditory bandwith of 150 Hz to 160 Hz. Jefferson (2009a) described highly directional echolocation clicks.

2.24.10 Fisheries By-catch and Entanglement

There were no reports of fishery related mortality or serious injury of rough-toothed dolphins in the most recent report covering 2002-2006 along the U.S. Atlantic coast (Waring et al. 2011).

2.24.11 Vessel Interactions

A review of Northeast and Southeast U.S. stranding data between 2005 and 2009 yielded no records of vessel interactions with rough-toothed dolphins.

2.24.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation

(Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear.

Stone (2003) studied the effects of seismic activity on cetaceans in UK waters. She reported that sighting rates of small odontocetes were significantly lower when large airgun arrays were being fired. She also found that, in general, small odontocetes showed the strongest avoidance behavior to seismic noise (compared to baleen, killer, sperm, and pilot whales). The 2010 Report of the Working Group on Marine Mammal Ecology (ICES 2010) stated that very little is known about the impact of the construction and operation of offshore wind-farms. Madsen et al. (2006b) reviewed available current information on wind farms and marine mammals and wrote that the noise impact is more severe during construction than operation. This was found to be the case for impacts to harbor porpoise (see Carstensen et al. [2006], Koschinski et al. [2003]). David (2006) wrote that pile driver-generated noise has the potential to impact dolphins by temporary displacement and interference with communication. The 2011 Report of the Working Group of Marine Mammal Ecology (ICES 2011) reported that porpoise detection (T-POD data) was lower during construction of a tidal turbine compared with before and after construction. No information specific to rough-toothed dolphins was found.

2.24.13 Data Gaps and Research Recommendations

There is no abundance estimate or information on spatial/temporal habitat use or stock structure for rough-toothed dolphin along the U.S. Atlantic coast. Food habits and life history information are only available from other parts of the world (e.g., the Pacific Ocean). There is limited information on acoustics, disease and contaminant levels in rough-toothed dolphins.

2.25 CLYMENE DOLPHIN (*STENELLA CLYMENE*)

2.25.1 Status

The Clymene dolphin is not listed under the Endangered Species Act. The IUCN Red List lists its status as “Data Deficient.” It is afforded protection in U.S. waters under the MMPA.

2.25.2 General Distribution

Clymene dolphins are limited to the tropical and sub-tropical Atlantic (Perrin et al. 1981). The northernmost record of a Clymene dolphin along the U.S. Atlantic coast is in New Jersey (Perrin et al. 1981), which is located in the BOEM North Atlantic Planning Area. There are also records of sightings in Mid-Atlantic, South Atlantic, and Straits of Florida Planning Areas (Fertl et al. 2003; Perrin et al. 1981). Sighting, stranding, and fishery bycatch locations of Clymene dolphins are shown in Figure 2.25-1.

2.25.3 General Abundance

The best estimate for Clymene dolphins along the U.S. Atlantic coast is 6086 (CV=0.93) (Waring et al. 2011). This estimate is taken from shipboard surveys conducted from Maryland to central Florida during July-August 1998. No sightings of Clymene dolphins have been reported in subsequent NMFS surveys along the U.S Atlantic coast. See Figure 2.25-1.

2.25.4 Habitat Preference

The Clymene dolphin is an oceanic species (Fertl et al. 2003; Jefferson 2009b; Jefferson and Curry 2003) often observed in deep tropical and warm temperate waters (250-5000 m or deeper) (Fertl et al. 2003; Perrin et al. 1981). Records from the U.S. mid-Atlantic indicate that the Gulf Stream influences Clymene dolphin distribution in this area (Fertl et al. 2003). Waring et al. (2011) report that most of the sightings during the 1998 survey were on the continental slope east of Cape Hatteras, North Carolina.

2.25.5 Stock Structure

There is no information on stock structure of the Clymene dolphin. NMFS considers the Gulf of Mexico and northwest Atlantic populations as separate stocks for management purposes but this division has yet to be confirmed by morphological, genetic, or behavioral data (Waring et al. 2011).

2.25.6 Life History Traits

Very little is known of their life history traits but it is thought that they are generally similar to other species in the genus *Stenella* (Jefferson 2009b). Males and females appear to reach sexual maturity by the time they reach 180cm (Jefferson and Curry 2003).

2.25.7 Food Habits

Perrin et al. (1981) reported that it appears Clymene dolphins are mid-water or night feeders on small fish and squid. Stomach content analysis of a dolphin that stranded in New Jersey found otoliths from *Ceratoscopelus* sp., *Lampanyctus* sp., and *Symbolophorus* sp. and 1 pair of squid beaks (Perrin et al. 1981). Otherwise, Jefferson (2009b) stated that very little is known about their food habits as very few stomach have been studied.

2.25.8 Health

2.25.8.1 Stranding

One Clymene dolphin that stranded along the U.S. Atlantic coast (North Carolina) was part of a mid-Atlantic, small cetacean UME declared in August 2004 (Waring et al. 2011). Waring et al. (2011) reported that prior to the UME, one dolphin had stranded in Florida in 1999. Neither of the dolphins showed signs of human interaction.

2.25.8.2 Contaminants

Jefferson and Curry (2003) report that almost no work has been done on contaminants in this species.

2.25.8.3 Disease

No information on disease in Clymene dolphins is available.

2.25.10 Acoustics

Southall et al. (2007) classified *Stenella* spp. in a mid-frequency cetacean functional hearing group with an estimated auditory bandwith of 150 Hz to 160 Hz. More specific information about Clymene dolphin acoustics is not available.

2.25.11 Fisheries By-Catch and Entanglement

There were no reports of fishery-related mortality or serious injury to Clymene dolphins along the U.S. Atlantic coast from 2001-2005 (Waring et al. 2011), or from 2005-2009.

2.25.12 Ship Strikes

A review of Northeast and Southeast U.S. stranding data between 2005 and 2009 yielded no records of vessel interactions with Clymene dolphins.

2.25.13 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear.

Stone (2003) studied the effects of seismic activity on cetaceans in UK waters. She reported that sighting rates of small odontocetes were significantly lower when large airgun arrays were being fired. She also found that in general, small odontocetes showed the strongest avoidance behavior to seismic noise (compared to baleen, killer, sperm, and pilot whales). The 2010 Report of the Working Group on Marine Mammal Ecology (ICES 2010) stated that very little is known about the impact of the construction and operation of offshore wind-farms. Madsen et al. (2006b) reviewed available current information on wind farms and marine mammals and wrote that the noise impact is more severe during construction than operation. This was found to be the case for impacts to harbor porpoise (see Carstensen et al. (2006), Koschinski et al. (2003)). David (2006) wrote that pile driver-generated noise has the potential to impact dolphins by temporary displacement and interference with communication. The 2011 Report of the Working Group of Marine Mammal Ecology (ICES 2011) reported that porpoise detection (T-POD data) was lower during construction of a tidal turbine compared to before and after construction. No information specific to Clymene dolphins was found.

2.25.14 Data Gaps and Research Recommendations

The only abundance estimate for the U.S. Atlantic Clymene dolphin population is 13 years old (1998) and not suitable for determining stock status (Wade and Angliss 1997). There is very limited information on food habits, life history and acoustics. There is no information on stock structure, contaminants, or the impacts of non-fishery related anthropogenic activities on Clymene dolphins.

2.26 STRIPED DOLPHIN (*STENELLA COERULEOALBA*)

2.26.1 Status

The striped dolphin is not listed under the Endangered Species Act and is considered a species of "Least Concern" on the IUCN Red List. It is afforded protection in U.S. waters under the MMPA.

2.26.2 General Distribution

Striped dolphins are found in the warm-temperate to tropical waters worldwide (Archer 2009). In the western Atlantic, the species is found from the northern coast of South America to the coast of the northeast U.S.; the Gulf Stream limits its northern distribution (Archer 2009). The general distribution encompasses the BOEM North Atlantic, Mid-Atlantic, South Atlantic, and Straits of Florida Planning Areas. However, most observations of striped dolphins are in the North and Mid-Atlantic Planning Areas (Waring et al. 2011). Sighting, stranding, and fishery bycatch locations of striped dolphins are shown in Figure 2.26-1.

2.26.3 General Abundance

The total number of striped dolphins along the U.S. Atlantic coast is unknown (Waring et al. 2011). The best estimate of striped dolphin abundance in the western North Atlantic is 94,462 (CV=0.40) and is derived by combining two surveys that took place June-August 2004. The first survey used a combination of ship and aerial surveys and resulted in an estimate of 52,055 (CV=0.57) striped dolphins from Maryland to the Bay of Fundy. The second survey was ship-based only and provided an estimate of 42,407 (CV=0.53) dolphins between Florida and Maryland.

2.26.4 Habitat Preference

As Archer (2009) summarized, striped dolphins are generally found outside the continental shelf, on the continental slope and in oceanic waters. They are found in temperatures 10-26 ^{o}C with most records in 18-22 ^{o}C (Archer and Perrin 1999). Most sightings off the U.S. east coast are on the continental shelf edge and slope areas west of Georges Bank (Waring et al. 2011). In conducting cetacean habitat-use surveys off the U.S. Atlantic coast during 1991-2, Waring et al. (1992) found striped dolphins were associated with the Gulf Stream north wall and warm-core ring features. During a suvey of the New England seamounts, Palka (2006) found that striped dolphins were in waters 20-27 ^{o}C.

2.26.5 Stock Structure

The western North Atlantic population is considered separate from the Gulf of Mexico population for management purposes but additional data are needed to elucidate stock structure (Waring et al. 2011). In an analysis of mtDNA, Garcia-Martinez et al. (1999) found no shared haplotypes between striped dolphins from the Mediterranean and northeast Atlantic. However, this study did not rule out male gene flow. In an analysis of 5 microsatellite loci, Bourret et al. (2007) found higher genetic diversity in animals from the northeast Atlantic than in the Mediterranean.

2.26.6 Life History Traits

Archer (2009) reported age at sexual maturity is 5-13 years for females and 7-15 years for males, and gestation lasts 12-13 months. Calzada et al. (1996) reported that females in the western Mediterranean reach sexual maturity at 12 years and have a 4-year inter-calving interval.

2.26.7 Food Habits

The species feeds on pelagic or benthopelagic fish and squid (Archer 2009). Perrin et al. (1994b) report cephalopods and myctophid fishes being important prey species. Striped dolphin diet consists of fish (lanternfish most common) and cephalopods in the Bay of Biscay (Ringelstein et al. 2006). Spitz et al. (2006) found that individual striped dolphins in the Bay of Biscay were capable of exploiting prey in oceanic, neritic, and coastal habitats, displaying plasticity in their foraging.

2.26.8 Health

2.26.8.1 Strandings

A total of 66 striped dolphins stranded along the U.S. Atlantic coast (Maine to Florida) from 2005-2009 (Northeast and Southeast U.S. marine mammal stranding response databases). One live stranding in 2007 was classified as a human interaction because surfers pushed the animal back out to deep water. In 2008, two striped dolphins were found dead with net or line marks, prompting classification as human interactions.

2.26.8.2 Contaminants

Multiple studies have documented contaminants in striped dolphins in Europe and Japan (Aguilar and Borrell 2005; Isobe et al. 2009; Wafo et al. 2005). Isobe et al. (2009) analyzed contaminant levels from 1978 to 2003 and found that PCB's, DDT's, and hexachlorocyclohexane (HCH) levels were stable, but PDBE's and hexabromocyclododecanes (HBCD's) had increased in striped dolphins in Japan over this time period. Aguilar and Borrell (2005) documented a decline in total PCB and DDT levels in striped dolphins in the western Mediterranean 1987-2002. Honda et al. (1983) documented a number of heavy metals (Pb, Ni, Cd, Hg, Fe, Mn, Zn, Cu) in the tissue of live caught, presumably healthy, striped dolphins in Japan.

2.26.8.3 Disease

Domingo et al. (1992) showed that morbillivirus was the primary cause of the epizootic that killed thousands of striped dolphins in the Mediterranean in 1990. After the acute Mediterranean epizootic, Domingo et al. (1995) found evidence of chronic morbillivirus infection in the central nervous system of some striped dolphins. Di Guardo et al. (2010) reported *Toxoplasma gondii* had caused meningoencephalitis in stranded dolphins in Italy.

2.26.9 Acoustics

Southall et al. (2007) classified *Stenella* sp. into a mid-frequency cetacean functional hearing group with an estimated auditory bandwith of 150 Hz to 160 Hz. Kastelein et al. (2006) studied the response of a striped dolphin to an acoustic alarm. They found that despite the sound being within the dolphin's hearing range, it did not react to the alarm.

2.26.10 Fisheries By-Catch and Entanglement

There were 46 striped dolphins incidentally caught from 1989 through 2009. This includes two animals caught in trawl gear in December 1991, and 44 animals caught in pelagic drift gillnet gear between 1989 and 1998. The pelagic drift gillnet fishery was discontinued in 1998. There have been no observed interactions in other fisheries.

2.26.11 Vessel Interactions

A review of Northeast and Southeast U.S. stranding data between 2005 and 2009 yielded no records of vessel interactions with striped dolphins.

2.26.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear.

Stone (2003) studied the effects of seismic activity on cetaceans in UK waters. She reported that sighting rates of small odontocetes were significantly lower when large airgun arrays were being fired. She also found that in general, small odontocetes showed the strongest avoidance behavior to seismic noise (compared to baleen, killer, sperm, and pilot whales). The 2010 Report of the Working Group on Marine Mammal Ecology (ICES 2010) stated that very little is known about the impact of the construction and operation of offshore wind-farms. Madsen et al. (2006b) reviewed available current information on wind farms and marine mammals and wrote that the noise impact is more severe during construction than operation. This was found to be the case for impacts to harbor porpoise (see Carstensen et al. [2006], Koschinski et al. [2003]). David (2006) wrote that pile driver-generated noise has the potential to impact dolphins by temporary displacement and interference with communication. The 2011 Report of the Working Group of Marine Mammal Ecology (ICES 2011) reported that porpoise detection (T-POD data) was lower during construction of a tidal turbine compared to before and after construction. No information specific to striped dolphins was found.

2.26.13 Data Gaps and Research Recommendations

The best abundance estimate is from aerial and ship surveys that took place in 2004; this data is 7 years old. There is very little recent information on striped dolphin distribution and abundance outside of the summer months (June-September). Very little is understood about stock structure in the northwest Atlantic. Most of the food habits, life history, and contaminants data are from populations in Europe or Asia. No information on the impacts of non-fishery related anthropogenic activities on striped dolphin in the northwestern Atlantic has been published.

2.27 SHORT-BEAKED COMMON DOLPHIN (*DELPHINUS DELPHIS*)

2.27.1 Legal Status

The short-beaked common dolphin is not listed under the Endangered Species Act and is considered a species of "Least Concern" on the IUCN Red List. It is afforded protection in U.S. waters under the MMPA.

2.27.2 General Distribution

The short beaked common dolphin is one of the most widely distributed cetaceans and is abundant throughout the warm temperate, subtropical and tropical waters of the world (Evans 1994; Perrin 2009a). Common dolphins are found in the BOEM North Atlantic, Mid-Atlantic and South Atlantic Planning Areas. According to Jefferson et al. (2009) there are no records of common dolphins in Florida waters after the 1960s; therefore, it is unlikely that they are found in the BOEM Straits of Florida Planning Area. Sighting, stranding and fishery bycatch locations of short-beaked common dolphins are shown in Figure 2.27-1.

2.27.3 General Abundance

The best estimate available for the Northwest Atlantic stock of common dolphins is 120,743 (CV=0.23) (Waring et al. 2011). As described in Waring et al. (2011) this estimate is derived from a combination of aerial and shipboard surveys conducted from June to August 2004 in waters from Florida to the Bay of Fundy. Palka (2006) estimated 90,547 (CV=0.24) common dolphins in waters between Maryland and the Bay of Fundy in July-August 2004. Waring et al. (2011) reported 30,196 (CV=0.54) common dolphins in the waters between Florida and Maryland in July-August 2004. From these surveys it appears that common dolphins do not occur south of the South Carolina-Georgia border.

2.27.4 Habitat Preference

Selzer and Payne (1988) found that common dolphins off the northeastern U.S. were generally sighted in water with 25-32 ppt salinity and mean surface temperature of 11^{o}C. Selzer and Payne (1988) also cite the importance of sea floor relief and subsequently prey distribution in the distribution of common dolphins. The majority (66%) of common dolphin sightings in their study were in areas with the maximum sea floor relief. In his work developing spatiotemporal prediction models of cetacean habitats, Hamazaki (2002) found that common dolphins tend to be found on the continental shelf, offshore, and near seamounts. Generally they were in water that was 18 (+/- 5.7) ^{o}C and less than 1500 m deep. Morato et al. (2008) found that common dolphins were significantly more abundant in the vicinity of Azores seamounts (depth 200-400 m), probably because of localized prey.

2.27.5 Stock Structure

Luca et al. (2009) collected tissue samples from 424 common dolphins in the North Atlantic Ocean (western Atlantic N = 183, eastern Atlantic N = 205) from 1990 to 2005. They analyzed 14 microsatellite loci and a 360 bp segment of the mitochondrial control region. They found a high level of genetic diversity and no evidence of a bottleneck effect. Both nuclear and mitochondrial DNA analyses demonstrated a significant difference between the eastern and western Atlantic populations. The authors did not find genetic structure within the eastern and western Atlantic regions; this suggests that individual dolphins are mixing within these areas. Luca et al. (2009) concluded that there is a single stock of common dolphins off the U.S. Atlantic coast but leave open the possibility of discrete populations in areas that were not sampled. Westgate (2007) analysis of cranial morphology provides additional support for a single population off the U.S. Atlantic Coast. However, in their study on common dolphin feeding ecology, Pusineri et al. (2008) found that, although animals feed on similar prey type (small aggregating species in the epipelagic layer), groups of dolphins forage in either the neritic or open ocean habitat and do not move back and forth between these two areas. Additionally, a study using cadmium accumulation in tissue as an ecological tracer found the existence of long-term dietary segregation between neritic and oceanic common dolphins in the Bay of Biscay (Lahaye et al. 2005). Studies have also found bias in the sex ratio of by-caught animals (e.g., Westgate and Read 2007: 161 males vs. 74 females) which could indicate seasonal segregation by sex.

2.27.6 Life History Traits

Westgate and Read (2007) examined life history data for the western northern Atlantic population. They estimated age at first maturity as 8 and 9.5 years for females and males, respectively, and a minimum 2 year inter-calving interval (likely longer). Murphy et al. (2009) found the eastern north Atlantic common dolphin population had an inter-calving interval of 3.79 years and that maximum age was 29 years. Westgate and Read (2007) did not attempt to age animals older than 25 years.

2.27.7 Food Habits

Although common dolphins are often thought of as generalists that prey on common schooling fish, studies have shown that they actively select the most energy-rich prey available to them (e.g., Meynier et al. 2008). Waring et al. (1990) reported that long-finned squid was a major component of stomach samples (n=33) from common dolphins incidentally caught by distant-water vessels off the northeast U.S. Overholz and Waring (1991) analyzed the stomach contents of 4 common dolphins caught in the Atlantic mackerel (*Scomber scombrus*) trawl fishery between 19 March and 17 April 1989 in the Hudson Canyon area. Atlantic mackerel was the dominant prey but they also found remnants of long-finned squid (*Loligo pealei*) and hake (*Merluccius* sp.). Pusineri et al. (2008) examined the stomachs of common dolphins by-caught in drift nets in the oceanic areas of the Bay of Biscay. They found that the dolphins' diet was dominated by lantern fish (*Notoscopelus kroeyer, Benthosema glaciale, Myctophum punctatum*) and pearlsides (*Maurolicus muelleri*). Cephalopods (*Ancistrotheuthis lechtensteini, Gonatus steentrupi, Brachioteuthis riisei,* and *Teuthowenia megalops*) were also important in their diet, accounting for 47% of the mass of total diet. A study of the stomach contents of stranded common dolphins along the Bay of Biscay found that their diet was dominated by sardines (*Sardina pilchardus*), anchovy (*Engraulis encrasiocolus*), sprat (*Sprattus sprattus*) and horse mackerel (*Trachurus* spp.) (Meynier et al. 2008).

2.27.8 Health

2.27.8.1 Strandings

Three common dolphin stranding events have been classified as UME. Two occurred in 2004, the first off the coast of Virginia, the 2nd along the coast from Maryland to Georgia. In 2008 a

UME that included at least 31 common dolphins occurred along the coast from New Jersey to North Carolina. A total of 428 common dolphins stranded along the U.S. Atlantic coast from 2005 to 2009 (Waring et al. 2011). Human interaction was indicated in 16 of these strandings. A majority (70%) of the strandings occurred in Massachusetts (Waring et al. 2011). Bogomolni et al. (2010) found that the cause of death in 37% of common dolphins that had stranded along Cape Cod and southeastern Massachusetts was disease.

2.27.8.2 Contaminants

Pierce et al. (2008) found that diet, location, reproductive status, and season all affected the variation in persistent organic pollutants (POP: PCB's, PBDE's and HBCD's) in female common dolphins stranded and by-caught in European waters (Ireland, France, and Spain). They found high PCB levels in these animals; average summed PCB concentrations exceeded the threshold at which effects on reproduction could be expected. Despite these levels, they found no association between cause of death and PCB concentrations. There is no contaminant information available for the western north Atlantic common dolphin population.

2.27.8.3 Disease

2.27.9 Acoustics

Southall et al. (2007) classified *Delphinus* spp. in a mid-frequency cetacean functional hearing group with an estimated auditory bandwith of 150 Hz to 160 Hz. According to Evans (1994) and the references therein, common dolphins are a very vocal species. They produce the variety of sounds common to dolphins (e.g., clicks, squeaks, whistles and creaks). Barlow and Cameron (2003) found that acoustic deterrent devices (pingers) significantly reduced the number of common dolphins by-caught in California drift gillnet fishery.

2.27.10 Fisheries By-catch and Entanglement

Waring et al. (2011) reported that common dolphins are by-caught in a variety of fisheries along the U.S. Atlantic coast. The 5-year (2005-2009) mean annual mortality for these fisheries are listed in Table 2-27-1.

Table 2.27.1. Fisheries Mortality of Common Dolphins

Fishery	Mean Annual Mortality (CV)
Northeast Sink Gillnet	26 (0.39)
Northeast Bottom Trawl	23 (0.13)
Mid-Atlantic Gillnet	2.2 (1.03)
Mid-Atlantic Mid-Water Trawl	0.6 (0.70)
Mid-Atlantic Bottom Trawl	10 (0.13)
Pelagic Longline	1.7 (1.0)

The National Marine Fisheries Service assembled the Atlantic Trawl Gear Take Reduction Team (ATGTRT) to address incidental mortality of common dolphins (among several other species) in September 2006. This take reduction team focuses on are the Mid-Atlantic Mid-Water Trawl, the Mid-Atlantic Bottom Trawl, Northeast Mid-Water Trawl and the Northeast Bottom Trawl fisheries. The Team found that mean annual incidental mortality (2001-2005) for common dolphins was highest in the Mid-Atlantic region in waters with a slope greater than 0.965°. *Illex* and *Loligo* squid fisheries dominated this area.

2.27.11 Ship Strikes

A review of Northeast and Southeast U.S. stranding data between 2005 and 2009 yielded no records of vessel interactions with common dolphins.

2.27.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear.

Stone (2003) studied the effects of seismic activity on cetaceans in UK waters. She reported that sighting rates of small odontocetes were significantly lower when large airgun arrays were being fired. She also found that in general, small odontocetes showed the strongest avoidance behavior to seismic noise (compared to baleen, killer, sperm, and pilot whales). The 2010 Report of the Working Group on Marine Mammal Ecology (ICES 2010) stated that very little is known about the impact of the construction and operation of offshore wind-farms. Madsen et al. (2006b) reviewed available current information on wind farms and marine mammals and wrote that the noise impact is more severe during construction than operation. This was found to be the case for impacts to harbor porpoise (see Carstensen et al. [2006], Koschinski et al. [2003]). David (2006) wrote that pile driver-generated noise has the potential to impact dolphins by temporary displacement and interference with communication. The 2011 Report of the Working Group of Marine Mammal Ecology (ICES 2011) reported that porpoise detection (T-POD data) was lower during construction of a tidal turbine compared to before and after construction.

2.27.13 Data Gaps and Research Recommendations

The best abundance estimate of common dolphins is from 2004 and is now over 7 years old. The information on common dolphin distribution and abundance (other than stranding and by-caught observations) outside of the summer months is old (e.g., CETAP data from late 1970s and early 1980s). No information on food habits or contaminant levels in common dolphins in the western north Atlantic has been published.

The ATFTRT recommended that there be an improved abundance estimate for all species (including common dolphins) affected by the trawl fisheries including “conducting more surveys, appropriately incorporating data from multiple years and by using appropriate stock structure boundaries.” Additionally, they specifically recommended that morphometric and genetic research of common dolphins be performed to confirm previous work that indicates there is a single common dolphin stock in the U.S. Atlantic.

2.28 FRASER'S DOLPHIN (*LAGENODELPHIS HOSEI*)

2.28.1 Legal Status

The Fraser’s dolphin is not listed under the Endangered Species Act. Its status on the IUCN Red List is “Least Concern.” It is afforded protection in U.S. waters under the MMPA.

2.28.2 General Distribution

Fraser’s dolphins are found in tropical waters throughout the world (Perrin et al. 1994a). Dolar (2009) reported a distribution between 30^{o}N and 30^{o}S; strandings outside of this area should be considered unusual. The only live sighting of Fraser’s dolphins along the U.S. Atlantic coast occurred off of Cape Hatteras, North Carolina,, in 1999 (Waring et al. 2011). Hersh and Odell (1986) and Waring et al. (2011) both described records of Fraser’s dolphin standings in Florida. Waring et al. (2011) reported that the lack of sightings along the U.S. Atlantic coast is probably due to naturally low numbers. Sighting and stranding locations of Fraser’s dolphins are shown in Figure 2.28.1.

2.28.3 General Abundance

There are no abundance estimates for Fraser’s dolphin along the U.S. Atlantic coast (Waring et al. 2011). Two-hundred-fifty Fraser’s dolphins were sighted off the coast of North Carolina (in the BOEM Mid-Atlantic Planning Area) in 1999. This sighting could not be used for an abundance estimate because it was not made during line-transect sampling effort (Waring et al. 2011). Fraser’s dolphins have not been observed in any other NMFS survey along the U.S. Atlantic coast (Waring et al. 2011).

2.28.4 Habitat Preference

Perrin et al. (1994a) refer to the Fraser’s dolphin as a “high seas” animal. Generally, they are an oceanic species occurring in deep water beyond the continental shelf, but they are also observed in deep water that is near the coast (Dolar 2009; Wursig et al. 2000). In the Gulf of Mexico they are seen most commonly in waters around 1000 m (Wursig et al. 2000). The one live sighting of Fraser’s dolphins along the U.S. Atlantic coast was in waters 3300 m (Waring et al. 2011).

2.28.5 Stock Structure

Stock structure of the Fraser’s dolphin is unknown along the U.S. Atlantic coast. The western North Atlantic stock is considered separate from the Gulf of Mexico stock for management purposes although there is no information to differentiate the two at this time (Waring et al. 2011).

2.28.6 Life History Traits

Very little is known about Fraser’s dolphin life history traits. Amano and Miyazaki (1996) reported that females reach sexual maturity at 5-8 years (length = 210-220 cm) and males at 7-10 years (length = 220-230 cm). They estimated that gestation lasts 12.5 months and the inter-calving interval is 2 years. These data came from animals caught in a drive fishery in Japan.

2.28.7 Food Habits

Robison and Craddock (1983) analyzed the stomach contents of three female Fraser's dolphins that were incidentally caught in purse seine nets in the eastern tropical Pacific in May 1972. They found a mixed diet of mesopelagic fish, shrimps and squid, with fish being the most important. They also estimated that the 3 dolphins were feeding at depths of 250-500 m. Dolar et al. (2003) reported that fish (97.3% occurrence), cephalopods (89.2% occurrence), and crustaceans (67.0% occurrence) were all important in the diet of Fraser's dolphins (n=37) incidentally caught in the tuna drift net fishery in the eastern Sulu Sea (Philippines). Myctophids were the most commonly occurring fish family while Enoploteuthids were the most commonly occurring cephalopod family. Two families of crustaceans were identified in the dolphin stomachs: Oplophoridae and Sergestidae. Dolar et al. (2003) also reported that Fraser's dolphins likely forage in the vertical range from the surface to approximately 600 m.

2.28.8 Health

2.28.8.1 Strandings

Hersh and Odell (1986) identified the skeletons of 10 stranded Fraser's dolphins collected on Marquesas Key, Florida, from November 1981 to June 1982. The remains had been on the beach for several weeks at the time of discovery, so the cause of stranding could not be determined. Waring et al. (2011) reported a live mass stranding of ten Fraser's dolphins in Lee, Florida, in April 2003 and one stranding in Florida in 2004. There were no signs of human or fishery involvement in these strandings.

2.28.8.2 Contaminants

There is no information on contaminant load in Fraser's dolphins.

2.28.8.3 Disease

As is summarized in Dolar (2009), many parasites have been observed on and in Fraser's dolphins including: *Zenobalanus* sp., *Phullobothrium delphini*, *Monorhygma grimaldi, Anisakis simplex, Tetrabothrius* sp., *Bolbosoma* sp., *Strobicephalus triangularis, Campula* sp., and *Stenurus ovatus*. *Morbillivirus* was also found in Fraser's dolphins in the southwest Atlantic (Van Bressem et al. 2001, as cited in Dolar (2009).

2.28.9 Acoustics

Southall et al. (2007) classified *Lagenodelphis* spp. in a mid-frequency cetacean functional hearing group with an estimated auditory bandwith of 150 Hz to 160 Hz. No more specific information about Fraser's dolphin acoustics is available.

2.28.10 Fisheries By-catch and Entanglement

There were no reports of fishery related mortality or serious injury of Fraser's dolphins in the most recent report covering 2001-2005 along the U.S. Atlantic coast (Waring et al. 2011).

2.28.11 Vessel Interactions

A review of Northeast and Southeast U.S. stranding data between 2005 and 2009 yielded no records of vessel interactions with Fraser's dolphins.

2.28.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear.

Stone (2003) studied the effects of seismic activity on cetaceans in UK waters. She reported that sighting rates of small odontocetes were significantly lower when large airgun arrays were being fired. She also found that in general, small odontocetes showed the strongest avoidance behavior to seismic noise (compared to baleen, killer, sperm and pilot whales). The 2010 Report of the Working Group on Marine Mammal Ecology (ICES 2010) stated that very little is known about the impact of the construction and operation of offshore wind-farms. Madsen et al. (2006b) reviewed available current information on wind farms and marine mammals and wrote that the noise impact is more severe during construction than operation. This was found to be the case for impacts to harbor porpoise (see Carstensen et al. [2006], Koschinski et al. [2003]). David (2006) wrote that pile driver-generated noise has the potential to impact dolphins by temporary displacement and interference with communication. The 2011 Report of the Working Group of Marine Mammal Ecology (ICES 2011) reported that porpoise detection (T-POD data) was lower during construction of a tidal turbine compared to before and after construction. No information specific to Fraser's dolphins was found.

2.28.13 Data Gaps and Research Recommendations

There is no abundance estimate or information on spatial/temporal habitat use or stock structure for Fraser's dolphin along the U.S. Atlantic coast. Food habits and life history information are only available from other parts of the world (e.g. the Pacific Ocean). There is limited information on acoustics and no information on contaminant levels in Fraser's dolphins.

2.29 HARBOR PORPOISE (*PHOCOENA PHOCOENA*)

2.29.1 Status

The harbor porpoise's status is "undefined" under the Endangered Species Act. It is a species of "Least Concern" on the IUCN Red List. It is afforded protection in U.S. waters under the MMPA.

2.29.2 General Distribution

The harbor porpoise is found in the cold temperate and subarctic waters of the northern hemisphere (Leatherwood and Reeves 1983b). As described in Waring et al. (2011), harbor porpoise are found in the North Atlantic Planning Area year round with highest abundance during the summer months (July to September). They are also found in the Mid-Atlantic Planning Area during the winter (January to March). They are not found in the South Atlantic or Straits of Florida Planning Areas. Sighting, stranding and fishery bycatch locations of harbor porpoises are shown in Figure 2.29-1.

2.29.3 General Abundance

The best abundance estimate available for the harbor porpoise is 89,054 (CV=0.47) (Waring et al. 2011). This estimate was derived from an August 2006 aerial survey conducted from the 2000-m isobath on the southern edge of Georges Bank to the upper Bay of Fundy and to the entrance of the Gulf of St. Lawrence (Waring et al. 2011). This covers the presumed summer range of the Gulf of Maine/Bay of Fundy breeding population.

2.29.4 Habitat Preference

Generally, harbor porpoise are a coastal species (Leatherwood and Reeves 1983b). In the Gulf of Maine, Palka (1995) found that the highest harbor porpoise densities in the summers of 1991 and 1992 were associated with sea surface temperatures of 10-13.5 ^{o}C, fish densities of 1.5-11 fish caught per minute trawling and water depths of 50-130 m. Watts and Gaskin (1985) found that harbor porpoise in the Bay of Fundy tended to be found in areas with physiographic features (e.g. islands, shoals) that concentrated their preferred prey: Atlantic herring (*Clupea harengus*). This work in the Bay of Fundy was further supported by Johnston et al.'s (2005c) study. Gilles et al. (2011) used oceanographic parameters and generalized additive models to predict harbor porpoise density and distribution in the North Sea to assess the extent of potential conflicts with offshore energy development and to support conservation and management plans. Porpoises were found to prefer areas with stronger currents and areas where fronts were likely.

2.29.5 Stock Structure

Rosel et al. (1999) found very little evidence of transatlantic exchange of harbor porpoise when they compared mitochondrial control region sequences from the northwestern and northeastern Atlantic populations. They estimated an exchange of 2.7 females per generation between the northeastern and northwestern Atlantic populations. In a study on the genetic structure of harbor porpoise within the northwestern Atlantic population, (Rosel et al. 1999a) found evidence of three distinct summer breeding populations: the Gulf of Maine/Bay of Fundy population, the Newfoundland population and the Gulf of St. Lawrence/Western Greenland population. This structure was only apparent in the mitochondrial DNA, not in the microsatellite, indicating female philopatry in these populations. It also appears that the winter aggregation of harbor porpoise of the mid-Atlantic coast contains animals from more than one of the summer populations (Rosel et al. 1999a). Comparison of organochlorine contaminants in harbor porpoise in the northwest Atlantic (Westgate and Tolley 1999) provided further support for the three populations described by (1999a).

2.29.6 Life History Traits

Read and Hohn (1995), to determine vital rates, examined 239 harbor porpoises that had been killed in gill net fisheries in the Gulf of Maine from 1989 to 1993. They found that most females are sexually mature by age three, and are pregnant annually. From their limited sample set of males, they estimated that most male harbor porpoise become sexually mature at three to four years of age. Ovulation, conception, and parturition all occur in the spring and early summer. They are relatively short-lived; most animals examined were less than 10 years old.

2.29.7 Food Habits

Harbor porpoise generally feed on small, pelagic, schooling fish with high lipid content and some bottom-dwelling fish (Bjorge and Tolley 2009). Gannon et al. (1998) found Atlantic herring was the number one prey species of harbor porpoise in the Gulf of Maine during the fall. Smith and Read (1992) found that while clupeid and gadid fish were important in adults diets, calves (<1 year) fed predominantly on euphasiids (*Meganyctiphanes norvedica*). Food habit studies in Scottish waters found whiting (*Merlangius merlangus*) and sand eels (*Ammodytidae*) the most important prey; in Icelandic waters, capelin (*Mallotus villosus*) and sand eels were the most important (Santos et al. 2004). Spitz et al. (2006) reported that harbor porpoise that had stranded on the French Coast (Bay of Biscay) had fed on small schooling fish living close to the sea floor.

2.29.8 Health

2.29.8.1 Strandings

One harbor porpoise stranding event has been declared a UME. It involved the stranding of 38 animals along the North Carolina coast from 1 January to 28 March 2005. A total of 450 harbor porpoise have stranded along the U.S. Atlantic coast from 2005 to 2009 (Northeast and Southeast U.S. marine mammal stranding rersponse databases). A high percentage of these strandings occurred in Massachusetts (32%) or North Carolina (20%). Forty-eight of the stranded harbor porpoises showed signs of human interaction, although that was not always identified as the ultimate cause of death.

2.29.8.2 Contaminants

Westgate and Tolley (1999) documented multiple organochlorine (OC) contaminants (PCBs, DDTs, chlordanes, chlorinated bornanes, hexachlorocyclohexanes, and chlorobenzenes) in harbor porpoise in the northwestern Atlantic. Their results showed that the unique combination of OC levels could be used to distinguish three geographic groups: Bay of Fundy/Gulf of Maine, Newfoundland, and the Gulf of St. Lawrence. Westgate and Tolley (1999) also found significantly higher levels of PCBs in the Bay of Fundy/Gulf of Maine stock. When Jepson et al. (2005b) compared the PCB levels of UK harbor porpoises that had died of infectious disease with those that had died of physical trauma, they found the infectious disease group had significantly greater values. Hall et al. (2006) hypothesized that the risk of mortality from infectious disease was correlated with levels of PCB exposure for harbor porpoises in the UK . A case-control analysis supported the hypothesis. Trends in chlorobiphenyls in harbor porpoise blubber from the UK from 1991 to 2005 were described in Law et al. (2010). Weijs et al. (2011) created a physiologically based pharmacokinetic model to learn more about uptake, distribution, and elimination of PCB’s.

2.29.8.3 Disease

Neimanis et al. (2008) found that 2 out of 170 harbor porpoises incidentally caught in fishing nets in the Bay of Fundy had detectable antibodies for *Brucella* sp.

2.29.9 Acoustics

Southall et al. (2007) classified *Phocoena* spp. in a high-frequency cetacean functional hearing group with an estimated auditory bandwith of 200 Hz to 180 kHz. In their study on harbor porpoise sonar characteristics, Goodson and Sturtivant (1996) found that their sonar system developed as a short-range foraging tool. Kraus (1997) and Culik et al. (2001) found that harbor porpoises actively avoided fishing nets equipped with acoustic deterrents (pingers) in the Gulf of Maine and near Vancouver Island, Canada, respectively. As summarized in Lucke et al. (2008), the harbor porpoise hearing range is from 250 Hz to 160 kHz; the most sensitivity is between 100 to 140 kHz.

2.29.11 Fisheries By-Catch and Entanglement

Waring et al. (2011) reported that harbor porpoise are by-caught in a variety of fisheries along the U.S. Atlantic coast. The mean annual mortality estimates (2005-2009) for these fisheries are presented in Table 2-29-1. Cox et al. (1998) examined carcasses of harbor porpoise that had stranded in the Mid-Atlantic (North Carolina, Maryland and Virginia) from 1994 to 1996. Twenty-five of the forty carcasses that were fresh enough to assess exhibited clear signs of fisheries interactions. Because fisheries interactions continue to pose a threat, pingers are required on gillnets. Also, the Harbor Porpoise Take Reduction Team has identified management areas along the New England and Mid-Atlantic coasts that are closed at times through the year to prevent harbor porpoise entanglement (Figures 2.29-2 and 2.29-3).

Table 2.29.1. Fisheries Mortality of Harbor Porpoise, 2005-2009

Fishery	Mean Annual Mortality (CV)
Northeast Sink Gillnet	599 (0.16)
Northeast Bottom Trawl	6 (0.22)
Mid-Atlantic Gillnet	318 (0.26)

2.29.12 Ship Strikes

A review of Northeast and Southeast U.S. stranding data between 2005 and 2009 yielded three records of vessel interactions with harbor porpoises.

2.29.13 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise by seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear.

Stone (2003) studied the effects of seismic activity on cetaceans in UK waters. She reported that sighting rates of small odontocetes were significantly lower when large airgun arrays were being fired. She also found that, in general, small odontocetes showed the strongest avoidance behavior to seismic noise (compared to baleen, killer, sperm, and pilot whales). Harbor porpoise avoidance responses were detected at distances of over 70 km from seismic airguns in a study carried out in marine waters of British Columbia and Washington (Bain et al. 2010). Compared with other species examined in that study, the harbor porpoise appeared to be the species affected by the lowest level of noise. The 2010 Report of the Working Group on Marine Mammal Ecology (ICES 2010) stated that very little is known about the impact of the construction and operation of offshore wind-farms. Madsen et al. (2006b) reviewed available current information on wind farms and marine mammals and wrote that the noise impact is more severe during construction than operation. The 2011 Report of the Working Group of Marine Mammal Ecology (ICES 2011) reported that porpoise detection (T-POD data) was lower during construction of a tidal turbine compared to before and after construction.

Cartensen et al. (2006) monitored harbor porpoise echolocation activity in the area of the Nysten Offshore Winds Farm construction site in the Baltic Sea. They found that this construction significantly impacted harbor porpoise habitat-use; this was evidenced by harbor porpoises leaving the construction area (measured by decreased echolocation activity). In particular, the phase of construction involving the ramming and vibration of steel sheet piles into the sea bed displaced harbor porpoises dramatically. Pile driving activity associated with the construction of the Horns Rev II offshore wind farm in the Danish North Sea was related to changes in harbor porpoise acoustic activity by Brandt et al. (2011). In the first hour after pile driving porpoise acoustic activity was reduced 100% and remained at below normal levels for 24 to 72 hours at a distance of 2.6 km from the construction site.

Tougaard et al. (2009) studied underwater noise produced by three types of wind turbines. They concluded that it was unlikely that the noise was harmful to harbor porpoise and not capable of masking communication. Lucke (2007), however, pointed out that while a study conducted on captive harbor porpoises in the Netherlands showed that potential masking effects of wind turbine operational noise would be limited to short ranges in the open sea, existing sound measurements have been carried out only on relatively small turbines, and masking studies do not take into account the larger and potentially noisier turbines that may be developed in the future. In a study that played back wind farm operational noise to harbor porpoise off Vancouver, Canada, Koschinski et al. (2003) found that harbor porpoises showed a clear reaction to the wind turbine noise. Closest approaches increased from a median of 120 m to 182 m, and during exposure to the sound, the number of time intervals when porpoise echolocation clicks were detected doubled. Scheidat et al. (2011) studied harbor porpoise occurrence around the Dutch offshore wind farm Egmond aan Zee. They found that, compared to reference sites, harbor porpoise acoustic activity was significantly higher inside the wind farm. The cause is not clear but may be related to an increase of food availability or a decrease of boat traffic inside the wind farm.

2.29.14 Data Gaps and Research Recommendations

The best abundance estimate is from 2006 and is now 5 years old. The information on harbor porpoise distribution and abundance (other than stranding and by-catch observations) outside of the summer months is old (e.g., CETAP data from late 1970s and early 1980s). There is limited information on harbor porpoise food habits in the U.S. Atlantic.

2.30 WEST INDIAN MANATEE (*TRICHECHUS MANATUS LATIROSTRIS*)

2.30.1 Status

The Florida manatee is listed under the Endangered Species Act. The stock is protected by the State of Florida under the Florida Manatee Sanctuary Act of 1978, as amended (§ 379.2431(2), FS) (US Fish and Wildlife Service 2009). As an endangered species, manatees are considered by default to be a “strategic stock” and “depleted” under the MMPA (US Fish and Wildlife Service 2009). The IUCN Red List lists its status as “Vulnerable.”

2.30.2 General Distribution

Three species of manatees inhabit subtropical and tropical waters on both sides of the Atlantic Ocean (Reynolds III et al. 2009). The West Indian manatee is found in coastal waters from approximately the Chesapeake Bay south into the Gulf of Mexico, throughout the Caribbean and into northern South America. The range of the Florida subspecies (*T.m. latirostris*) is from eastern Texas to Virginia; individuals have been observed as far north as Massachusetts in summer months (Fertl et al. 2005; Reynolds III et al. 2009; Schwartz 1995; US Fish and Wildlife Service 2009). In winter, manatees are generally restricted to coastal and inland waters of peninsular Florida, where they seek shelter in and near warm-water springs, industrial effluents, and other warm water sites (Hartman 1979; Lefebvre et al. 2001; US Fish and Wildlife Service 2009).

Deutsch et al. (2003) radio-tagged 78 manatees along the coast of Georgia and Atlantic Florida from 1986-1998. They used these data to describe seasonal movements, migratory behavior, and site fidelity of manatees in this area. They found that most manatees migrated seasonally and individuals used 1-2 core areas 90% of the time in a given season. Information from tagged juvenile manatees provided evidence for strong philopatry.

In general, the distribution of the Florida manatee is influenced by temperature and possibly by access to fresh drinking water (Reynolds III et al. 2009). Most observations of manatees along the U.S. Atlantic coast occur in the BOEM Straits of Florida Planning Area; there have been seasonal observations in the South Atlantic Planning Areas. Extralimital occurrence has been documented in the North Atlantic and Mid-Atlantic Planning Areas. Manatee synoptic count data for the years 1991-2001 are shown in Figure 2.30-1.

2.30.3 General Abundance

The total population of Florida manatees along the U.S. Atlantic coast is unknown but it is believed to be increasing (Waring et al. 2011). The best count of manatees on the Florida Atlantic coast is 2,432 animals, based on a synoptic survey in January 2011 (http://myfwc.com/research/manatee/projects/population-monitoring/synoptic-surveys/). The survey was conducted in part of the BOEM Straits of Florida Planning Area.

Craig and Reynolds III (2004) presented a stochastic model that can be used to assess population trends and abundance in Florida manatees. Edwards et al. (2007) analyzed the probability of detecting manatees during aerial surveys and developed a correction factor. Kendall et al. (2004) created a multi-state mark-resighting model using photo-id sighting data for the Northwest and Atlantic manatee management units.

2.30.4 Habitat Preference

The Florida manatee lives in shallow, relatively calm, freshwater, brackish, and marine coastal habitats that are also used by humans (O'Shea and Kochman 1990). As stated in the General Distribution section above, Florida manatees are influenced by temperature; in cold weather, they will move south and seek out natural or artificially occurring areas of warmer water (Reynolds III et al. 2009). Groups of over 300 manatees have been observed in the warm water effluent of power plants along the Florida coast (often in response to cold weather) (Reynolds III et al. 2009).

Gannon et al. (2007) found that habitat selection was influenced by availability of food and thermal and freshwater sources. Females with calves were additionally influenced by ambient noise, currents and added nutritional requirements. Adimey et al. (2008) described the feasibility of using a "Crittercam" system on wild manatees. This technology can provide insight into habitat.

Manatees commonly use resting holes which are depression in the substrate. Bacchus et al. (2009) described the resting holes used by manatees in Belize. They found that resting hole sites were significantly deeper that non-resting hole sites and the water was calmer.

Miksis-Olds and Miller (2006) studied transmission loss in manatee habitats and found that the acoustic environment played an important role in habitat selection.

2.30.5 Stock Structure

Garcia-Rodriguez et al. (1998) analyzed a section of the mtDNA control region from 86 West Indian manatees that were sampled at 8 different geographic locations. They identified 15 haplotypes and found three distinct lineages that corresponded with three geographic areas: Florida and the West Indies; Gulf of Mexico to the Caribbean rivers of South America; northeast Atlantic coast of South America. Along the U.S. Atlantic coast of Florida, manatees are placed in one of two regional management units: Upper St. Johns River or Atlantic Coast (U.S. Fish and Wildlife Service 2001).

Carney et al. (2007) presented a minimally invasive technique for collecting manatee skin samples which can be performed from a boat, dock, etc., while leaving the animal in its habitat. Hunter et al. (2010) successfully used 17 Australian dugong (*Dugong dugon*) microsatellite primers to analyze the Florida manatee. This cross-species panel provided additional genetic information.

2.30.6 Life History Traits

Reynolds III et al. (2009) summarized available life history data. Mean age at first reproduction is 5 years and maximum age was reported as 60 years. Gestation is 11-13 months, lactation lasts 1-2 years and the inter-calving interval is 2.5 years. There are few details available about the social structure of manatees but they are thought to follow the fission-fusion model (Reynolds III et al. 2009).

Langtimm and Beck (2003) found adult manatees had lower survival rates during years with intense storms (e.g. hurricanes). Langtimm et al. (2003) use photo-id data to estimate survival probabilities of adults from the four regional populations of Florida manatees. They found adult survival ranged from 0.908-0.960.

2.30.7 Food Habits

Manatees are herbivores and hind gut digesters, like horses and elephants (Reynolds III et al. 2009). They feed on submerged, emergent, and floating vegetation (U.S. Fish and Wildlife Service 2001). Reynolds III et al. (2009) wrote that West Indian manatees have been reported to eat over 60 species of plants.

Alves-Stanley and Worthy (2009) pointed out that although manatees are endangered, relatively little is known about their feeding ecology. Their study provided baseline parameters for stable isotope analyses using skin samples. They found that turnover rates were slow in skin and thus might be useful in analyzing intake over a long period of time. Ames et al. (1996) also studied stable isotopes in the skin of captive manatees and in the internal organs of stranded manatees.

Lefebvre et al. (2000) compared manatee feeding patterns in Florida and Puerto Rico. Their results suggested that the manatees in Florida were less specialized than those in Puerto Rico.

In their analysis of manatee foraging and anthropogenic sound, Miksis-Olds et al. (2007) found that manatees selected grassbeds with lower ambient noise.

2.30.8 Health

Data on manatee mortality in the U.S. is collected by the Manatee Carcass Salvage Program. A total of 766 manatee deaths were documented in 2010. Eighty-nine (12%) of these were attributed to human interaction. (See http://myfwc.com/research/manatee/rescue-mortality-response/mortality-statistics/2010/). Potential sources of human-caused mortality and injury include vessel interactions, water control structures, commercial/recreational fishing gear, ingestion of marine debris, and others.

Bonde et al. (2004) suggested that manatees are a good candidate species to act as a sentinel to ocean health because they are generally resilient but are highly susceptible to harmful algal blooms. Additionally, when congregated in large groups, they are particularly susceptible to the impacts of epizootics and marine pollution.

2.30.8.1 Strandings

During 2003–2007, a total of 1805 stranded, dead manatees were collected. Four-hundred thirty three (24%) were attributed to human-caused mortality, 362 (20%) were perinatal, 169 (95%) were related to cold stress and the rest (841, 47%) were placed in the "Other" category which includes deaths attributed to red tide US Fish and Wildlife Service 2009). A total of 80 manatees were rescued or assisted and released from 2003-2007 US Fish and Wildlife Service 2009).

2.30.8.2 Contaminants

Ames and van Vleet (1996) analyzed 45 samples (liver, kidney, and blubber) from Florida manatees for organochlorines. The pesticide levels were low so they were unable to make any conclusions regarding age, gender or location. A captive-born young manatee had higher levels of pesticide concentrations indicating maternal transfer. Reynolds III et al. (2009) reported that the effects of contaminants on manatees are unknown.

2.30.8.3 Disease

There have been several epizootic events involving Florida manatees and dinoflagellate blooms (which produce red tides and poisonous neurotoxins). In 1982, 39 manatees were found dead (two were hit by boats) during a 10-week period. A die-off of double-crested cormorant (*Phalacrocorax auritus*) occurred simultaneously as did a bloom of red tide (produced by *Gymnodinium breve*).

Bossart et al. (1998) described brevitoxicosis in manatees that died as part of the 1996 epizootic in Florida. The authors suggested that the poisoning was a result of chronic exposure to neurotoxins via feeding and breathing. Flewelling et al. (2005) reported that fish and sea grass can accumulate high levels of the toxin and act as vectors. Bossart et al. (2002) documented viral papillomatosis in captive manatees.

Halvorsen and Keith (2008) described an immunosuppression cascade in manatees. They suggested that, although the manatee is relatively robust, a change in environmental conditions (e.g., red tide) can trigger an immunosuppressive cascade which includes interrelated diseases and conditions.

2.30.9 Acoustics

Underwater vocalizations of both captive and wild manatees were studied by O'Shea and Poche (2006). The found that the highest rate of vocalization occurred between mothers and calves. The manatees also used vocalization to stay in touch with the group. They measured individual fundamental frequencies from 1.75 to 3.90 kHz. Their study suggested that vocalizations were individually unique.

Colbert et al. (2009) studied sound localization abilities in two manatees using a four-choice discrimination paradigm. Their results suggested that manatees are capable of localizing manatee vocalizations and boat engine noises. Miksis-Olds and Tyack (2009) found that ambient noise had a detectable effect on manatee communication.

A number of papers by Gerstein et al. (1999, 2002, 2008, 2010) provided information about manatee hearing, communication, and acoustics. Gur and Niezrecki (2010, 2007) described experiments related to denoising manatee vocalizations in order to use passive acoustic detection. Phillips and Beusse (2006) described detection ranges of manatee vocalizations in various acoustic settings which can be used in creating manatee detection systems.

2.30.10 Fisheries By-Catch and Entanglement

At least ten manatees died as a result of entanglements or ingestion of marine debris from 2003-2007 (Waring et al. 2011). No known sources of commercial gear were identified in these cases.

Beck and Barros (1991) analyzed the stomach contents of 439 Florida manatees. They found debris in 63 of the gastrointestinal tracks; four other manatees died as a direct result of ingestion. Monofilament fishing line was the most commonly found debris.

2.30.11 Vessel Interactions

Watercraft collisions are a significant source of human-caused manatee mortality (Beck et al. 1982; U.S. Fish and Wildlife Service 2001, 2009). In 2010, 83 manatees were killed by watercraft (see http://myfwc.com/research/manatee/rescue-mortality-response/mortality-statistics/). Beck et al. (1982) analyzed propeller wounds to understand how propeller size and horsepower impact manatee mortality. They concluded that large motor boats with inboard engines likely caused many of the mortalities.

Buckingham et al. (1999) analyzed the relationship between manatees using warm water refuges, temperature and boating activity. They concluded that recreational boating influenced manatee distribution. Calleson and Frolich (2007) and Laist and Shaw (2006) found that a reduction in boat speed in areas commonly used by manatees appeared to reduce risk of injury to the animals. Manatees were found to respond to approaching vessels with a “fright” response and moved towards deeper water (Nowacek et al. 2004).

Aipanjiguly et al. (2003) surveyed boaters in Florida regarding manatee conservation. Results of this survey indicated that boaters preferred more education/public outreach than more stringent laws.

2.30.12 Energy Projects

Reynolds III and Wilcox (1994) described monitoring manatee numbers at five power plants. Their study suggested that up to 41% of the total Florida manatee population could be observed at power plants at one time. Deutsch et al. (2003) reported that many manatees are able to overwinter further north of their historic range because of the presence of industrial warm-water effluents. Laist and Reynolds III (2005) reported that many of the power plants manatees use might soon be retired. It is unclear if the manatees will move south into warmer areas or stay near the power plans and suffer an increase in the number of cold-related deaths. The authors suggested that in the absence of power plants, warm water springs in northern Florida would be the most important source of natural warm water habitat.

2.30.13 Data Gaps and Research Recommendations

The total population of manatees along the U.S. Atlantic is unknown. Research into the best sampling methodology for obtaining a total population estimate should continue. Stock structure along the Florida coast should be further clarified. Additionally, Alves-Stanley and Worthy (2009) stated that relatively little is known about manatee feeding ecology.

2.31 HARBOR SEAL (*PHOCA VITULINA CONCOLOR*)

2.31.1 Status

The harbor seal is not listed under the Endangered Species Act. The species is considered of "Least Concern" on the IUCN Red List. It is afforded protection in U.S. waters under the MMPA.

2.31.2 General Distribution

Harbor seals (*Phoca vitulina*) are found on either side of the North Atlantic and North Pacific Oceans (King 1983). Burns (2009) wrote that they have the widest distribution of any pinniped species. The western Atlantic stock (*P.v. concolor*) is found from approximately 40°N (New Jersey) to 73°N (Baffin Island, Canada) with individuals observed as far south as Florida (Burns 2009). There is a seasonal movement from Long Island Sound/Southern New England northeastward to the Maine coast (potentially further northeast) in late spring and then a southward reverse movement occurs in the fall (Barlas 1999; Jacobs and Terhune 2000; Rosenfeld et al. 1988; Whitman and Payne 1990; Waring et al. 2006). Harbor seals can be observed in all four BOEM Atlantic Planning Areas, although most of the population is concentrated in the North Atlantic Planning Area. Sighting, stranding and fishery bycatch locations of harbor seals are shown in Figure 2.31-1 and tracks of satellite-tagged harbor seals are shown in Figure 2.31-2.

2.31.3 General Abundance

There is no current abundance estimate for harbor seals along the U.S. Atlantic coast, although abundance is thought to be increasing (Waring et al. 2011). The most recent comprehensive survey took place May-June 2001, during which 38,014 (including 9,282 pups) were observed along the Maine coast (Gilbert et al. 2005). Radio tag data was used to correct this observed number for seals not hauled out (correction factor = 2.61). The corrected 2001 population estimate was 99,340 (CV=0.097).

Baird (2001) wrote that there was insufficient information available to evaluate the status of harbor seals on the east coast of Canada. Bowen et al. (2003) documented a decline in the number of harbor seals on Sable Island (reduced recruitment of breeding females and later implantation due to nutritional stress were observed). The authors suggested that the increasing Sable Island gray seal population and shark inflicted mortality may be factors in this decline. Lucas and Stobo (2000) also documented shark predation on harbor seals on Sable Island.

Thompson et al. (1990) evaluated the effectiveness of aerial vs. boat surveys to estimate population size in Orkney, UK. They used time-lapse photography to collect data on haul-out behavior in relation to time of day and tidal cycle and radio-tags to collect information on seal activity budgets. All of this data was used to correct raw counts. Additional studies throughout the UK have provided information useful for population estimates (e.g., seasonal distribution, activity patterns, use of photo-id, etc.) (Thompson 1989; Thompson et al. 1989; Thompson and Harwood 1990; Thompson and Wheeler 2008).

In a study similar to that of Thompson et al. (1990), Huber et al. (2001) used radio tags to correct aerial survey counts of harbor seals along the Washington and Oregon coasts. Their total population correction factor was 1.53. Ries et al. (1998) also used radio tags to provide an estimate of the proportion of seals hauled-out during aerial surveys in the Dutch Wadden Sea. A recent advance in technology has allowed the development of a tag that uses mobile phone systems to track the animals. This technology will likely add to knowledge of general distribution and haul-out behavior, both of which have implications for population estimates (Cronin et al. 2009). Bengston et al. (2007) compared the effectiveness of two techniques (shore based compared with large scale aerial photography) for surveying harbor seals in glacial fjords.

Harkonen et al. (1999) described the challenges of obtaining vital population information from a species that is structured by age and sex. They also wrote that age and sex specific behaviors need to be taken into account when estimating populations as they can influence population parameters, and described a methodology on how to correct for these sampling errors. Ver Hoef and Jansen (2007) presented models that took into account “zeros” in environmental data that are sometimes included in population models.

2.31.4 Habitat Preference

Globally, harbor seals can be found on intertidal ledges, mud flats, sand bars, sand or cobble beaches and occasionally ice (Reeves et al. 1992); they are found on all of these substrates along the U.S. Atlantic coast (Barlas 1999; Gilbert et al. 2005). They occur in more varied habitats than any other pinniped (Burns 2009). Renner (2005) and Murray (2008b) studied behavior and interspecies interactions of harbor and gray seals at Mt. Desert Rock, Maine, and Cape Cod, Massachusetts, respectively.

2.31.5 Stock Structure

Arnason and Johnsson (1992) sequenced the entire mtDNA of the harbor seal. They found the total length of the sequence was 16826bp but the control region was unusually long. Stanley et al. (1996) analyzed worldwide patterns of harbor seal mtDNA differentiation. They analyzed 227 samples from 24 locations using a 435bp segment of the control region. They found that populations in the eastern and western Atlantic and Pacific Oceans were separated into four regions. Within those four regions, seals that were further separated geographically were also further separated genetically.

Goodman (1998) used seven microsatellite DNA polymorphisms (n=1029 seals) to analyze differences between 12 geographic locations in the northeast Atlantic. He found that differentiation was greater over distances, especially where there was discontinuous distribution. These results suggested that harbor seal philopatry operates over a distance of 300-500 km.

In Alaska, O'Corry-Crowe et al. (2003) used mtDNA analyses to study population subdivision and dispersal of harbor seals. They collected samples from 881 seals in 180 geographic locations. Their results showed substantial population division along the Alaska coast and identified a minimum of 12 clusters that differed from each other significantly and could be considered separate stocks. Prior to this study the Alaskan harbor seal population was managed as only 3 stocks.

In a similar study, Huber et al. (2010) used mtDNA to analyze the population structure of harbor seals in Washington State. They analyzed samples from 552 harbor seals in nine different locations. They found a total of 73 haplotypes, and 37 individual seals had a unique haplotype. Their results identified four stocks and suggested that the inland waters should be managed as two stocks instead of one.

Coltman et al. (1998b) used six microsatellite markers to measure male mating success on Sable Island. Their results suggested that it was unlikely that most pups were fathered by a small group of successful males. They also found that females do not exhibit fidelity to the same male from year to year on Sable Island.

Herreman et al. (2009a) analyzed six microsatellite markers to estimate gene flow between Glacier Bay, Alaska, (declining population) and Prince William Sound, Alaska (stable population). They found these populations formed one panmitic population but there is higher gene flow from Glacier Bay to Prince William Sound (n=63/generation) than the reverse (n=22/generation). This gene flow was driven by male movement.

2.31.6 Life History Traits

As is summarized in Burns (2009), females and males become sexually mature at 3-4 and 4-5 years, respectively. Maximum life span is about 35 years. Females give birth to one pup per year and lactation last approximately four weeks. Reijnders et al. (2010) found that partuition date for harbor seals in the Wadden Sea has moved, which shortens the seals annual cycle. They deduced that this was most likely because of a shortening of embryonic diapauses due to an improved forage base.

Schreer et al. (2010) showed that harbor seal pups nurse primarily in the water. Skinner (2006) described the physical and behavioral development of harbor seal pups in mid-coast Maine.

2.31.7 Food Habits

Harbor seals are generalists and usually feed on the most abundant and easily caught prey (Burns 2009; Härkönen 1987; Pierce et al. 1991; Tollit et al. 1997). Tollit et al. (1997) found that the most abundant fish species in the Moray Firth area of Scotland contributed the most to the seals' diet. Harbor seal diets often show seasonal variation (Herreman et al. 2009b; Payne and Selzer 1989; Pierce et al. 1991). Prey species include: cod (*Gadus morhua*), hake (*Urophycis* spp.), mackerel (*Scomber scombrus),* herring (*Clupea* spp.), shad (*Alosa* spp.), capelin (*Mallotus villosus*), sand lance (*Ammodytes* spp.), sculpins (*Artediellus* spp.), and flat fish (Pleuronectiformes) (Brown and Mate 1982; Burns 2009; Hammill and Stenson 2000; Hauksson and Bogason 1997; Härkönen 1987; Pierce et al. 1991; Thompson et al. 1991; Thompson and Miller 1990). On Cape Cod, Massachusetts, Payne and Selzer (1989) found that sand lance was the single dominant prey, followed by herring. In another study on harbor seal food habits on Cape Cod, Ferland (1999) also found that sand lance was the most frequently occurring (85%) prey species followed by winter flounder. Williams (1999) analyzed the stomach contents of harbor seals that had been incidentally caught in sink gillnets in the Gulf of Maine. Silver hake was the most common prey she identified. Sand lance were not prevalent in these stomachs, likely reflecting the habitat in which the seals had been incidentally caught.

Radio tagging studies have showed that harbor seals feed within 30-45 km of haul-sites (Thompson et al. 1991; Tollit et al. 1998). Additionally, Tollit et al. (Tollit et al. 1998) found that most seals foraged in water that was 10-50 m in depth.

Herreman et al. (2009b) found evidence in Glacier Bay that competition led to poor quality prey; this has potentially led to emigration. Bowen et al. (1992) found that because female harbor seals do not have sufficient energy reserves to cover the joint costs of lactation and maternal metabolism, they must feed during lactation.

2.31.8 Health

2.31.8.1 Strandings

A total of 1,477 harbor seals have stranded along the U.S. Atlantic coast (Maine to Florida) from 2005 to 2009. They are the most frequently stranded pinniped in the U.S. Atlantic. Most of these were found along the Maine coast (n= 852, 59%), followed by Massachusetts (n= 370, 26%). These two states account for 84% of all stranding observations. There have been two unusual mortality events. The first was declared in the northern Gulf of Maine in 2003-2004; no common cause of death has been identified. A second UME was declared in the autumn of 2006 and was attributed to an infectious disease. Table 2.31-1 summarizes this stranding data (Waring et al. 2011). Bogomolni et al. (2010) analyzed stranding data from marine mammals and seabirds found along the coast of southern New England. They found that the most commonly identified cause of death in harbor seals was disease. .

Table 2.31.1 Harbor Seal Strandings by State and Year

State	2005	2006	2007	2008	2009	TOTAL
ME	121	371	106	178	76	852
NH	31	28	6	3	15	83
MA	101	94	51	50	74	370
RI	3	6	8	6	5	28
CT	2	1	3	0	0	6
NY	22	11	11	5	14	63
NJ	1	7	6	7	11	32
DE	3	2	0	0	0	5
MD	2	0	0	0	2	4
VA	3	2	0	1	3	9
NC	8	4	0	6	6	24
FL	0	1	0	0	0	1
TOTAL	297	527	191	256	206	1477

(Data from Waring et al. 2011)

2.31.8.2 Contaminants

Neale et al. (2009) analyzed PCB and DDE contamination in harbor seals from central California and Alaska. In Alaska, concentrations increased in males during their lives (likely due to continued exposure), but female levels lowered due to offloading via pregnancy and lactation. In California, pups showed the highest contaminant load, indicating stable to decreasing exposure. Hall et al. (1992) found that there was a significant difference in the OC concentration in tissues from seals that had died compared with seals that survived the 1988 European epizootic (phocine distemper virus - PDV) event.

Tabuchi et al. (2006) found evidence of contaminant-related disruption of TH (thyroid hormone) in harbor seals sampled in Washington state and British Columbia, Canada.

Several studies have used controlled feeding studies, during which captive seals were fed fish from contaminanted areas, such as the North Sea (Brouwer et al. 1989; Ross et al. 1996; Vost and Osterhaus 1995; de Swart et al. 1996, 1995). The goal of these studies was to analyze the health impacts of a diet of contaminanted fish. Brouwer et al. (1989) found that the seals eating fish contaminated with PCB exhibited both vitamin A and thyroid hormone deficiency. Ross et al. (1996) found a suppression of natural killer cell activity in seals that were fed fish from the Baltic Sea. de Swart (1995) documented impaired immune response in seals that were fed fish from the Baltic. In their review paper, de Swart et al. (1996) conclude that, given these feeding studies, it can be expected that seals in contaminated areas are likely to exhibit impacted immune function.

Anas (1974) analyzed heavy metals in Pacific harbor seals (*P.v. richardi*). Park (2009) documented PCB’s and OH-PCB’s in the livers of harbor seals from the Gulf of Maine and San Francisco Bay. Brookens et al. (2008) measured mercury body burdens in pups from central California. They analyzed THg (total mercury) concentrations in various organs and suggested that muscles sampled from specific sites is best for these analyses. Agusa et al. (2011b) analyzed the accumulation of trace metals in harbor seals that had stranded during the mass mortality in 2002 in the North Sea.

Shaw et al. (2009) measured perfluorochemicals in harbor seals samples from the northwest Atlantic. Their results showed concentrations in pups that were 2.6 times that of adult females indicating substantial maternal transfer.

2.31.8.3 Disease

There have been several documented harbor seal die offs along the New England coast. One occurred in 1979-1980 and was attributed to an avian virus, a second occurred in 1982 (Geraci et al. 1982; Hinshaw et al. 1984). A die off due to PDV occurred along the New England coast 1991-1992 (Duignan et al. 1995b). Several UME’s have been declared for harbor seals in New England; for more details, see the Strandings section above.

A longitudinal study (1980-1994) of harbor and gray seals on the Atlantic coast of North America showed that gray seals had a higher percentage of morbillivirus neutralizing antibodies

(73%) compared with harbor seals (37%) (Duignan et al. 1995b). The titers were significantly higher against PDV compare to other morbilliviruses. During winter 1991-1992 there was a harbor seal epizootic along the northeast U.S.; stranded animals tested during that event showed a significant increase in the presence of PDV antibody (83%). The authors suggested that PDV is enzootic in gray seals and that harbor seals become infected through casual contact with gray seals. Harkonen et al. (2006) reviewed how PDV outbreaks have spread in European waters. They also suggested that gray seals were potential vectors by acting as either reservoirs for the disease or by having sub clinical infections. Swinton et al. (1998) analyzed persistence thresholds for PDV infection in metapopulations of harbor seals.

Osterhaus et al. (1985) described the first isolation and partial characterization of a herpes virus from harbor seals in the Netherlands. Schumacher et al. (1993) found evidence of colloid depletion and fibrosis in the thyroids of harbor seals that died during the 1988-1999 epizootic in the North Sea.

Philippa et al. (2009) described neurological signs in harbor seals infected by PDV and those that had died during the 2002 epizootic in northern Europe. They found that neurological signs were one of the most common symptoms in pups with PDV. Nollens et al. (2010) found a respiratory coronavirus in one harbor seal that had died during a pneumonia epizootic in California. Cabezon et al. (2011) documented antibodies to *Toxoplasma gondii* in harbor seals sampled along the coast of the UK and France.

Moore et al. found gas bubbles in the tissues of seals incidentally killed in gillnet fishery.

In an effort to correlate genetic data with fitness, Rijks et al. (2008) analyzed heterozygosity and lung worm burden. They found that homozygosity predicted higher lung worm burdens in young animals (the age at which lung worms most impact fitness). To understand the relationship between genetic variation and birth weight and neonatal survival, Coltman et al. (1998a) used a measure of genomic diversity based on the mean squared difference between microsatellite alleles within an individual, called mean d^2. Using this technique, they found that birth weight was positively influenced by age of mother, gender and either heterozygosity or mean d^2.

2.31.9 Acoustics

Kastelein et al. (2009) quantified the underwater hearing capability of two captive one-year old female harbor seals. They found maximum sensitivity occurred at 1 kHz and the frequency range of best hearing was from 0.5 to 40 kHz. This study suggested that harbor seals hear over a greater range than previously thought. Kastelein et al. (2010) described the effect of signal duration on underwater hearing thresholds of two two-year old female harbor seals. Both Jacobs and Terhune (2002) and Nelson et al. (2006) studied the effectiveness of acoustic harassment devices on harbor seals. Jacobs and Terhune (2002) found no response or change in seal behavior (although sample size was small). Nelson et al.’s (2006) study showed that acoustic harassment was ineffective in deterring seal predation on salmon aquaculture farms in Maine. A study examining harbor seal haul-out numbers relative to time of day, ambient noise levels, and tide levels found a correlation between seal haul-out numbers during the night and noise levels

(Acevedo-Gutiérrez and Cendejas-Zarelli 2011). The authors were not able to conclude, however, that the nocturnal haul-out behavior was a related to human development.

2.31.10 Fisheries By-Catch/Entanglement

Total annual estimated average fishery-related mortality for 2004-2008 is 425 harbor seals. Most of those (n= 387, 91%) are attributed to the Northeast Sink Gillnet Fishery.

Table 2.31.2 Annual Estimated Fishery-related Mortality for Harbor Seals along the U.S. Atlantic coast, 2005-2009

Fishery (2004-8)	Mean Annual Mortality (CV)
Northeast Sink Gillnet	332 (0.14)
Mid-Atlantic Gillnet	45 (0.39)
Northeast Bottom Trawl	unk
Northeast Mid-Water Trawl	0.3 (0.81)

2.31.11 Vessel Interactions

Henry and Hammill (2001) studied the impact of small boats on the haul-out activity of harbor seals in the Saint Lawrence Estuary, Canada. They found that kayaks caused the most disturbances, followed by canoes, motor boats and sailboats. The number of seals hauled-out decreased after a disturbance except when the seals were molting and hesitant to go into the water. Jansen et al. (2006) studied the disturbance cruise ships have on harbor seals in Disenchantment Bay, Alaska. Koschinski (2008) reviewed available literature to assess the possible impact of personal watercraft (PWC) on harbor porpoise and harbor seals. The Northeast and Southeast U.S. Marine Mammal Stranding Network databases indicate that 13 stranding mortalities between 2005 and 2009 exhibited signs of boat strike interactions.

2.31.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear.

The 2010 Report of the Working Group on Marine Mammal Ecology (ICES 2010) stated that very little is known about the impact of the construction and operation of offshore wind-farms. Madsen et al. (2006b) reviewed available current information on wind farm and marine mammals and wrote that the noise impact is more severe during construction than operation. The 2011 Report of the Working Group of Marine Mammal Ecology (ICES 2011) reported that there was no change in seal haul-out behavior in the vicinity of a tidal turbine and shore-based surveys showed no evidence of disturbance during construction.

Koschinski et al. (2003) played back simulated 2 MW (Megawatt) wind-turbine noise in Fortune Island, Vancouver, Canada. They found that the seals showed a distinct reaction to the noise, including surfacing at a greater distance. They concluded that harbor seals are capable of hearing the low-frequency noise produced by the wind turbines.

The effect of a large offshore wind farm on harbor and gray seals within a nearby seal sanctuary (4 km) in Denmark was described by Edren et al. (2010). They used time lapse photography, visual and aerial surveys to document haul-out behavior during pre and post-construction. They found no long term impact on haul-out behavior. There was a significant short term decline in numbers during sheet pile driving at or near the wind farm.

Tougaard et al. (2009) studied underwater noise produced by three types of wind turbines. They concluded that it was unlikely that the noise was harmful to harbor seals (also harbor porpoise) and was incapable of masking communication.

2.31.13 Data Gaps and Research Recommendations

The most recent population estimate for the U.S. Atlantic coast is from 2001 and out of date. There is little known about local and regional movements of harbor seals along the northeast U.S. coast. Most food habits studies have been done in isolation and without understanding of foraging range. Very little life history data is available specific to this area. Most of the contaminant and disease research has been done in other parts of their range.

2.32 HARP SEAL (*PAGOPHILUS GROENLANDICUS*)

2.32.1 Status

The harp seal is not listed under the Endangered Species Act. The species is considered of “Least Concern” on the IUCN Red List. It is afforded protection in U.S. waters under the MMPA.

2.32.2 General Distribution

Harp seals are distributed throughout the North Atlantic and Arctic Oceans in three recognized stocks: East Ice (White Sea), West Ice (Greenland Sea), and the Northwest Atlantic (Lavigne 2009; Lavigne and Kovacs 1988). Its range is tied to the pack ice (2009). The Northwest Atlantic population has two breeding aggregations: The Front off the coast of Newfoundland and Labrador, and in the Gulf of St. Lawrence (Lavigne 2009). As is summarized in Lavigne (2009) and Lavigne and Kovacs (1988), harp seals are highly migratory (Lavigne 2009). After the breeding season, the Northwest Atlantic harp seals move north into feeding grounds. They move along the coast of Greenland and Baffin Island and into Hudson Bay. In the fall, they move south again.

The harp seals observed along the U.S. Atlantic Coast likely come from the northwestern Atlantic population. There is no resident population of harp seals in the U.S., but transient animals are observed on beaches and in stranding and by-catch data. Harp seals are most often seen in the months January-May (Harris et al. 2002), which corresponds with when they are at the southern end of their migration. Most of the animals observed in the U.S. are juveniles

(Harris et al. 2002) and most are in the BOEM North-Atlantic Planning Area. Sighting, stranding, and fishery bycatch locations of harp seals are shown in Figure 2.32-1 and tracks of satellite-tagged harp seals are shown in Figure 2.32-2.

2.32.3 General Abundance

The Northwest Atlantic population is the largest of the three (Lavigne 2009) and was estimated to be 6.5 million seals in 2008. This population has been increasing since the late 1970s (DFO 2010). There is no resident population of harp seals in the U.S. but see the Stranding and Fisheries By-catch sections below for observed numbers. Salberg et al. (2009) described the used of generalized additive models to estimate pup production. They found this technique decreased the coefficient of variation around the estimate.

There was a dramatic increase in the number of harp and hooded seals on Sable Island during the mid-1990s (Lucas and Daoust 2002). Likewise, there was an increase of harp and hooded seals throughout the northeast U.S. during this same time period (Harris et al. 2002; Harris and Gupta 2006; Stevick and Fernald 1998).

Harp seals are hunted throughout their range by both subsistence (2008= ~85,000 seals taken) and commercial (2008= 72,000 taken; has been over 300,000 in prior years) hunters (DFO 2010).

2.32.4 Habitat Preference

The harp seal’s range is generally linked to the southern and northern edge of the pack ice (Lavigne 2009). They use the ice for pupping (Lavigne 2009; Lavigne and Kovacs 1988); mating usually takes place in the water (Lavigne and Kovacs 1988). Schreer and Kovacs (1997) reported that they dive to a maximum depth of 370 m.

Johnston et al. (2005a) analyzed sea ice cover in eastern Canada 1969-2002. They found a high degree of variability and that heavy ice years corresponded with a positive spring North Atlantic Oscillation (NAO) conditions. They found that during light ice years, there was often a significant reduction of ice in the Gulf of St. Lawrence and Newfoundland by early March when harp seals pup. The authors point out that a reduction of ice could potentially lead to increased neonatal mortality, changes in available prey and increase in epizootics because of crowding at pupping sites. Friedlaender et al. (2010) also analyzed sea ice conditions in relation to the NAO and its impact on the habitat available for harp seals. They found a consistent positive relationship between sea ice in the western north Atlantic and the NAO index.

In their GIS-based analysis of harp and hooded seal strandings in the Gulf of Maine, Harris and Gupta (2006) found that being close to an offshore basin, deep water, public land and areas with lower human density were all predictors of where ice seals would strand.

2.32.5 Stock Structure

There are three recognized stocks of harp seals: East Ice (White Sea), West Ice (Greenland Sea), and the Northwest Atlantic (Lavigne 2009; Lavigne and Kovacs 1988). Perry et al. (2000) used mtDNA analyses to assess harp seal stock structure. They identified 13 haplotypes and found strong differentiation between seals from the northeast vs. northwest Atlantic. They did not find any differentiation between the two northwest Atlantic pupping sites (Front vs. Gulf).

Kretzman et al. (2006) analyzed samples from 65 harp seals that had stranded along Long Island, New York. They used between 9-12 microsatellite loci to measure d^2. Their results showed that animals that survived had a slightly higher d^2 than animals that died. Carr et al. (2008) included harp seals in their experiments utilizing DNA microarray analyses.

2.32.6 Life History Traits

Harp seals become sexually mature at approximately 5.7 years (DFO 2010), although this has varied through time (Sjare and Stenson 2009). As is summarized in Lavigne (2009), female harp seals give birth to one pup and nurse it for about 12 days.

Sjare and Stenson (2009) describe the long-term trends in female harp seal life history traits (1980-2004) in the Northwest Atlantic population. They documented changes in pregnancy rates and mean age at first reproduction that reflected density dependent responses. The changes in the population size during this time period (1980-2004), however, were likely influenced by other ecological or environmental factors.

2.32.7 Food Habits

Beck et al. (1993) analyzed the stomach contents of harp seals (n=140) taken from the Gulf of St. Lawrence and Hudson Strait. They found that stomach contents varied by season and location. Samples from the northern Gulf of St. Lawrence contained a variety of prey while samples from Hudson Bay and the Gulf contain mostly capelin (*Mallotus villosus*). Stenson et al. (1997) studied harp seal predation in eastern Canada. They found that both Arctic cod (*Boregadus saida*) and capelin were important prey off Newfoundland but also found capelin dominated in the Gulf of St. Lawrence. In their study on the estimated prey taken by seals in Atlantic Canada, Hammill and Stenson (2000) found that capelin was the most common prey eaten by harp seals. This was followed by sand lance (*Ammodytes* spp.), Arctic cod, Greenland halibut (*Reinhardtius hippoglossoides*), Atlantic cod (*Gadus morhua*), Atlantic herring (*Clupea harengus*), and redfish (Sebastes spp.*)*.

Tucker et al. (2009) used quantitative fatty acid signature analysis (QFASA) to investigate harp and hooded seal food habits. Their study included samples from 526 harp seals collected from 1994 to 2004. They found that harp seals fed primarily on amphipods, Arctic cod, capelin, herring, sand lance, and redfish. Marshall et al. (2010) developed molecular techniques to identify prey in harp seal stomach contents.

Harp seals dive to an average maximum depth of 370 m and mean length of dive is 16 minutes (Schreer and Kovacs 1997). Nordoy et al. (2008) satellite-tagged harp seals in the White and Barents Seas. They found that seals used the water column at 20-300 m and their movements/migration were closed linked to capelin migration.

2.32.8 Health

2.32.8.1 Strandings

A total of 511 harp seals have stranded along the U.S. Atlantic coast (Maine to North Carolina) from 2005 to 2009 (Northeast and Southeast U.S. marine mammal stranding response databases). Eighteen of these animals showed signs of human interaction. Massachusetts had the highest number of observed strandings (n= 255, 47%) followed by New York (n= 103, 19%) and Maine (n= 77, 14%). These three states account for 80% of all stranding observations. Moore et al. (2009) found gas bubbles in the tissues of seals that had been incidentally killed in gillnet fishery.

Table 2.32.1 Harp Seal Strandings by State and Year

State	2005	2006	2007	2008	2009	TOTAL
ME	10	14	8	15	9	56
NH	2	0	1	1	4	8
MA	44	24	51	51	59	229
RI	9	6	2	5	9	31
CT	3	4	1	2	3	13
NY	41	15	19	8	29	112
NJ	12	3	3	12	5	35
DE	2	0	2	0	0	4
MD	2	0	4	1	2	9
VA	4	0	5	3	1	13
NC	0	1	0	0	0	1
TOTAL	129	67	96	98	121	511

(NE and SE marine mammal stranding response databases)

2.32.8.2 Contaminants

Montie et al. (2010) documented brominated flame retardants (BFRs), including polybrominated diphenyl ethers (PBDEs) and current-use, non-PBDE BFRs, organochlorine (OC) pesticides and polychlorinated biphenyls (PCBs), in harp seals from the eastern U.S. and Canada.

Vetter et al. (2010) analyzed brominated flame retardant (DPTE) in blubber and brain tissues of harp seals from the Barents and Greenland Seas. Agusa et al. (2011a) reported the levels of 19 different trace metals in the liver, muscle, kidney, gonads and hair from 18 harp seals from Pangnirtung (Baffin Island).

2.32.8.3 Disease

Daoust et al. (1993) documented the first case of PDV in a harp seal in a two-month old female that had stranded on Prince Edward Island in 1991. Duignan et al. (1997) analyzed harp seal samples collected from the western Atlantic from 1988-1993 for PDV. Eighty-three percent of the seals they tested were positive for PDV antibodies. A novel *Heliobacter* spp. was isolated and characterized from two harp seals that had stranded along the Massachusetts coast (Harper et al. 2003).

Piche et al. (2010) experimentally infected harp seals with lungworm in order to characterize pathologic changes.

2.32.9 Acoustics

Rossong and Terhune (2009) recorded underwater calls of harp seal in the Gulf of St. Lawrence in March 2007. They found that communication ranges were impacted by call source levels, background noise, and patterns of sound transmission. Van Opzeeland et al. (2009) analyzed pup calls on the Front (Canada) and in the Greenland Sea. They found that at the Front, female pups had individually identifiable calls 38% of the time and males 42%. This is in contrast with the Greenland Sea animals: 55% of female pups had identifiable calls but only 8% of males. They also found that mother-pup attendance was different (85.1% of mother's time on Front; only 52.2% of mother's time in Greenland Sea). The authors suggest that these behavioral differences might be related to evolutionary change related to hunting or different environmental conditions at the two sites.

2.32.10 Fisheries By-Catch and Entanglement

Total annual estimated average U.S. fishery-related mortality for 2005-2009 is 231 for harp seals. Most of those (n=174, 75%) are attributed to the Northeast Sink Gillnet Fishery. See Table 2.32-2 for details. A little over 10,000 harp seals are estimated to be taken in Canadian fisheries annually (DFO 2010).

Table 2.32.2 Mean Annual Mortality by Fishery 2005-2009

Fishery (2004-8)	Mean Annual Mortality (CV)
Northeast Sink Gillnet	174 (0.18)
Mid-Atlantic Gillnet	57 (0.5)
Northeast Bottom Trawl	unk

(Waring et al. 2011) (CV in parentheses)

2.32.11 Vessel Interactions

In 2005 there is a record in the Northeast and Southeast U.S. Marine Mammal Stranding Network databases of a harp seal with wounding consistent with vessel interaction.

2.32.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear.

The 2010 Report of the Working Group on Marine Mammal Ecology (ICES 2010) stated that very little is known about the impact of the construction and operation of offshore wind-farms.

Madsen et al. (2006b) reviewed available current information on wind farm and marine mammals and wrote that the noise impact is more severe during construction than operation. The 2011 Report of the Working Group of Marine Mammal Ecology (ICES 2011) reported that there was no change in seal haul-out behavior in the vicinity of a tidal turbine and shore-based surveys showed no evidence of disturbance during construction. No information is available about the impact of energy projects on harp seals.

2.32.13 Data Gaps and Research Recommendations

Because harp seals are in U.S. waters only as transients, there is no abundance estimate for this part of their range. Most of the information about acoustics, food habits, contaminant burden, or disease comes from research conducted in Canadian Atlantic waters.

2.33 GRAY SEAL (*HALICHOERUS GRYPUS*)

2.33.1 Status

The gray seal is not listed under the Endangered Species Act. The species is considered of "Least Concern" on the IUCN Red List. It is afforded protection in U.S. waters under the MMPA.

2.33.2 General Distribution

Gray seals are found in the temperate and subarctic areas on both sides of the north Atlantic (King 1983). In the western North Atlantic, gray seals are found from Labrador (Canada) to New York (Davies 1957; Mansfield 1966; Lesage and Hammill 2001; Katona et al. 1993). Gray seals can be observed in all four BOEM Atlantic Planning Areas, although most of the population is concentrated in the North Atlantic Planning Area. There are three well-established pupping sites in the U.S.: Muskeget Island, Massachusetts; Green Island, Maine; and Seal Island, Maine (Wood LaFond 2009). Sighting, stranding, and fishery bycatch locations of gray seals are shown in Figure 2.33-1 and tracks of satellite-tagged gray seals are shown in F igure 2.33-2.

In their study on the comparative movement of harbor and gray seals in Moray Firth, Scotland, Thompson et al. (1996) found that gray seals potentially moved 125-365 km away from their haul-out site. Sjoberg and Ball (2000) found that most gray seals remained within 50 km of their haul-out in the Baltic Sea. Vincent et al. (2002) followed rehabilitated and released juvenile gray seals. They found that these animals were able to disperse across large areas quickly and also settled into known gray seal haul-out sites after a few months.

2.33.3 General Abundance

There is no current abundance estimate for the total number of gray seals along the U.S. Atlantic coast but it appears this population is increasing (Waring et al. 2011). Wood LaFond (2009) provided a single day pup count of 2,649 in 2008. This has increased since pups were first observed in the early 1990s (Rough 1991; Wood LaFond 2009). Muskeget Island, Massachusetts, is the largest of these three sites; over 80% of all U.S. gray seal pups are born there (Wood LaFond 2009).

The Canadian populations have increased since the 1960s. Pup production estimates from 2007 surveys estimate total pup production for the Gulf of St. Lawrence was 13,000 (SE=600) and Sable Island was 54,500 (SE=1288) (Bowen et al. 2007b; Hammill et al. 2007). Although these populations continue to grow, data from Sable Island has shown a change in vital population rates, which indicates that population growth is slowing there (Bowen et al. 2007a, 2007b). The annual rate of increase on Sable Island declined from 12.8% in 2004 to 7% in 2007 (Bowen et al. 2007a).

2.33.4 Habitat Preference

Katona et al. (1993) reported that gray seals are found on remote, exposed islands, reefs, and shoals. Throughout their range they haul-out, molt and give birth to their pups on ice or sandy or rocky beaches (King 1983). Renner (2005) and Murray (2008b) studied behavior and interspecies interactions of harbor and gray seals at Mt. Desert Rock, Maine and Cape Cod, Massachusetts, respectively.

On the Isle of May (Scotland), Pomeroy et al. (2000) documented changes in pupping sites as the colony expanded using fine scale digital elevation models. They found that in general the areas that were colonized first on the island were close to access points (to the sea) or near standing water. They also found that individually marked females showed strong site fidelity (median distance between sites used in consecutive years was 25 m). The authors suggested that the spatial scale of site fidelity is related to the scale of topographic variation within each site.

2.33.5 Stock Structure

Arnason et al. (1992) sequenced the entire mtDNA of the gray seal. They found the molecule was 16,797 base pairs. Allen et al. (1995) used eight microsatellite loci to detect genetic differentiation between two breeding sites in the UK. They found structure indicating site fidelity in adults and philopatry in pups. Boskovic et al. (1996) analyzed mtDNA from gray seals in the Gulf of St. Lawrence (Canada), Sable Island (Canada), Norway, and the Baltic Sea. There were no shared haplotypes between the eastern and western North Atlantic seals. However, when the two Canadian populations were compared, little or no separation was detected. Wood et al. (2011) used both nuclear (nine microsatellite loci) and mitochondrial (control region segment) DNA to analyze gray seal stock structure in the northwest Atlantic. Their study included samples from the Gulf of St. Lawrence; Sable Island; Green Island, Maine, and Muskeget Island, Massachusetts. Their results showed no genetic structure between these four sites.

In the UK, Gaggiotti et al. (2002) used genetic data to study metapopulation dynamics of gray seals. They found that density-dependent dispersal played an important role in the establishment of new gray seal pupping sites.

Walton et al. (2000) analyzed blubber biopsies from two breeding sites in Scotland to ascertain if fatty acid analysis could be used to distinguish between populations. They found that the two breeding sites could be identified as two distinct feeding groups. Reed et al. (1997) developed a technique to use genetic analyses to assign scat samples to species, sex, and individual identification. Assigning species and gender to scat samples can eliminate some of the variation surrounding seal diet.

Several studies have focused on the reproduction in the gray seal by combining behavioral observations with genetic analysis. Amos et al. (1993b) found that male reproductive success was not as high as behavioral observations (copulations) would suggest. In another study, Amos et al. (2001) reported that as few as 1% of males gain significantly higher reproductive success. They found that maternal half-siblings were more diverse than would be expected through random mating. The authors suggested that while behavioral choice is an unlikely mechanism, naturally occurring anti-sperm antibodies could prevent inbreeding in small, potentially closed populations.

2.33.6 Life History Traits

Hall and Thompson (2009) reported that females become sexually mature at 3-5 years old and males at 6 years old, although males probably don't breed until older.

Boness and James (1979) described the reproductive behavior of gray seals on Sable Island. They reported that the females are gregarious and probably return to the same part of the beach from year to year. Males do not maintain territories but rather compete for the right to stay near the group of females. Pomeroy et al. (1994) found that females on N. Rona (UK) exhibited pupping site fidelity.

In their study of the reproductive behavior of gray seals on North Rona Island, Anderson et al. (1974) concluded that males maintained a high level of sexual activity in order to mate with as many cows as possible, rather than spending energy through territorial fighting for exclusive access to females. Anderson and Harwood (1985) found that variations in the activity budgets of seals appeared to be determined by the type of habitat they were in.

Several studies have focused on gray seal breeding behavior on ice. Tinker et al. (1995) found that the average time budget of males on ice (Amet Island, Canada) was comparable to land-breeding males. Although body size was not correlated with success, reproductive effort (estimated by body mass loss over the season) and successful interactions with other males were both correlated with success. Haller et al. (1996) found that females breeding on ice transferred the same amount of energy to their pups as land-breeding animals but over a shorter period. This is likely due to the risk of being separated in an unstable habitat (shifting ice).

Boness et al. (1995) found that females were more frequently disturbed by males during the latter part of the pupping season. Mothers who pupped later in the season weaned pups that were 16% lighter than those weaned earlier. The authors suggested that this male harassment might contribute to reproductive synchrony by causing reduced maternal performance later on in the pupping season.

Evidence of non-filial nursing in gray seals was documented by Perry et al. (1998). They found frequency of fostering varied between three study sites. DNA fingerprinting showed that kin selection did not play a significant role in this behavior.

Twiss et al. (2007) studied the impact of climatic variation on effective population size. They found that during drier years at North Rona, the degree of polygyny decreased when females were more dispersed, moving around to find water. Their results showed that changes in local weather could alter the annual proportion of males who contributed to the effective population size by up to 61%.

There has also been a substantial amount of research on the reproductive behavior of gray seals. This research has often combined field observations with genetic analyses; see the Stock Structure section for additional details on those studies.

2.33.7 Food Habits

Gray seals feed on a variety of prey: sand lance (*Ammodytes* spp.), cod (*Gadus morhua*), flatfish (*Pleuronectiformes*), haddock (*Melanogrammus aeglefinus*), catfish (*Siluriformes*), scorpion fish (*Scorpaenidae*) and cephalopods are important prey species. Their diet shows geographical and seasonal variation (Bowen and Harrison 1994; Hammill and Stenson 2000; Hammond et al. 1994; Hauksson and Bogason 1997; Hauksson and Ólafsdóttir 1995; Mikkelsen et al. 2002; Murie and Lavigne 1992; Prime and Hammond 1990). Ampela (2009) analyzed scat samples collected in Nantucket Sound (Muskeget and Monomoy Islands) ~2004-2008 and stomach contents (from by-caught animals). She found that sand lance, winter flounder, red/white hake, and Atlantic cod together were 82% of diet by weight.

McConnell et al. (1999) reported that seals took either long (up to 2100 km) or local trips. In eighty-eight percent of trips to sea, the seals returned to the haul-out from which they left and the mean length of time for these trips was 2.33 days. Sjoberg and Ball (2000) reported that gray seals were mostly in water depths 11-40 m and avoided waters that were greater than 51 m deep. They also found that seals apparently selected foraging habitat because of bathymetric factors (water depth) rather than simply using the habitat surrounding the haul-out site.

Austin et al. (2006b) used stomach temperature telemetry to gain insight into gray seal feeding. They found that feeding intervals varied from one individual to the next and likely reflected differences in prey availability. They also found significant differences between males and females with the number of feeding events per day greater in males than females. Feeding event size differed by time of day and the length of time between meals increased with the size of the previous meal. In a related study, Austin et al. (2006a) deployed three different types of tags on the same animals (satellite tag, time depth recorder, and stomach temperature loggers) to better understand gray seal foraging success. Their results showed that several factors (e.g., mean bottom time, distance traveled, and bathymetry) can predict feeding but the importance of these factors varied with temporal scale.

Beck et al. (2007) used fatty acid analysis to examine differences in the diets of males and females. While sand lance and redfish were important prey, the males' diets were significantly more varied. They also found that females in the post-breeding season selected fewer prey but prey of higher quality. Tucker et al. (2007) analyzed carbon and nitrogen stable isotopes to understand diet segregation of gray seals on Sable Island. Their analyses showed the adults and juveniles could be identified from each other based on stable isotopes as could adult males from adult females.

2.33.8 Health

2.33.8.1 Strandings

There were 344 observed gray seal strandings along the U.S. Atlantic coast 2005-2009. The majority of these were in Massachusetts (n=210, 61%), followed by New York (n=52, 15%) (Northeast and Southeast U.S. Marine Mammal stranding response databases). Bogomolni et al. (2010) analyzed stranding data from marine mammals and sea birds found along the coast of Southern New England. They found that human interaction was the cause of death in 43% of the gray seals included in their study.

Table 2.33.1 Gray Seal Strandings by State and Year

State	2005	2006	2007	2008	2009	TOTAL
ME	4	3	5	6	3	21
NH	0	0	1	0	1	2
MA	26	29	50	53	52	210
RI	2	2	5	7	10	26
CT	0	0	0	0	1	1
NY	7	6	21	2	16	52
NJ	2	1	5	3	4	15
DE	0	0	0	1	0	1
MD	3	0	1	1	1	6
VA	1	0	1	1	2	5
NC	0	2	1	1	1	5
TOTAL	45	43	90	75	91	344

(Waring et al. 2011)

2.33.8.2 Contaminants

Jenssen (1996) estimated that approximately 50% of gray seals at the largest breeding site in Norway are contaminated by oil each year as a result of chronic, low level pollution from shipping traffic. Jenssen found indications that pups along the Norweigan coast are affected by this low-level exposure as exemplified by thyroid hormone and vitamin A status.

Nyman et al. (2002) measured DDT, PCB, and trace elements in the gray seals from Sable Island, Canada, and the Baltic Sea. They found that the Baltic population had a higher PCB and DDT loads than Sable Island population, although these levels were lower than observed in 1970s. They also found that on Sable Island only, males had higher contaminant loads than females (likely due to reproductive offloading). They documented residue amounts of trace metals (mercury, cadmium, lead, and selenium from various tissues) and did not see a decrease in metal burden in the Baltic animals. Bustamante et al. (2004) documented trace element accumulation in various tissue types of Faroese gray seals sampled from 1993-1995. This population exhibited higher levels of cadmium, indicating cephalpods are an important prey.

Blubber samples from pups born on Sable Island, Canada, were analyzed for lipid content and OC levels from 1988 to 1991 (Addison and Stobo 1993). These analyses showed that after weaning, there was a fairly constant OC burden around which lipid content changed. The authors concluded that there is very little a seal can do to degrade the contaminant level inherited from its mother. Debier et al. (2003) documented the dynamics of PCB transfer during lactation in UK gray seals. They found increased PCB concentration in mothers' serum and milk later in lactation which likely reflected changes in blubber. Additionally, they found that pups had significantly higher serum PCB levels than their mothers, which indicates that much of the transfer of these contaminants occurred in utero.

Harding and Harkonen (1999) used hunting records to model Baltic Sea gray seal population trends during the 20th century. They found that hunting had decreased this population during the first part of the 20th century, but then recovery was inhibited by reproductive sterility likely caused by organochlorines.

See the Disease section below, which describes symptoms associated with contaminant levels.

2.33.8.3 Disease

Baker (1984) studied gray seal pup mortality and morbidity at three sites in the Outer Hebrides. Mortality rates were generally similar (14.3-23.2%) but the leading cause of death varied from site to site and included starvation, infection, and being washing off of the island. Various species of *Streptococci* and *Corynebacterium* were documented and adult seals were found to be the source of the diseases.

The transfer of sealpox from captive gray seals to their handlers was documented by Hicks and Worthy (1987). Osterhaus et al. (1990) documented parapoxvius in five gray seals that were admitted to a rehab facility during a PDV outbreak. The authors suggested that the PDV infection potentially made the seals more susceptible to other infections. Anderson et al. (1974) documented a high incidence of skin lesions which were often associated with emaciation and nematode infection in the Dee Estuary (UK).

Bergman (1999) examined 159 gray seals (postmortem) from the Baltic Sea 1977-1996. A disease complex (lesions of claws, skull bone, intestines, kidneys, etc., and occlusions, stenoses, and tumors in the uterus) was reported early on. When the study period was divided into two decades (1977-1986 and 1987-1996), he found a decrease in uterine obstructions (42% to 11%), an increase in pregnancies (9% to 60%) and a decrease in uterine tumors (53% to 43%). However, an increase in the presence of colonic ulcers in juvenile seals was documented. Bergman et al. (2001) documented renal lesions in Baltic gray seals and suggest that the OC pollution in the Baltic Sea is likely a contributing factor.

A longitudinal study (1980-1994) of harbor and gray seals on the Atlantic coast of North America showed that gray seals showed a higher percentage of morbillivirus neutralizing antibodies (73%) compared to harbor seals (37%) (Duignan et al. 1995b). The titers were significantly higher against PDV compare to other morbilliviruses. During winter 1991-1992 there was a harbor seal epizootic along the northeast U.S.; stranded animals tested during that event showed a significant increase in the presence of PDV antibody (83%). The authors suggested that PDV is enzootic in gray seals and that harbor seals become infected though casual contact with gray seals. Harkonen et al. (2006) reviewed how PDV outbreaks have spread in

European waters. The also suggested gray seals were potential vectors by acting as a reservoir for the disease or by having sub clinical infections.

Bean et al. (2004) analyzed patterns of parental relatedness and pup survival. They found pups with higher internal relatedness (IR) had significantly lower survival rates. Cammen et al. (2011) studied the variation in the major histocompatibility complex (MHC) in gray seals in the eastern Atlantic. They found geographic variation in this gene complex with subpopulations breeding in similar habitats being more similar genetically, which suggests that habitat-specific pathogen could influence MHC evolution.

2.33.9 Acoustics

Caudron et al. (1998) analyzed pup calls to estimate call variation. They studied seven acoustic features and found three that showed the most individuality: location of maximum signal strength; the number of harmonics; and frequency and strength of the fundamental. Despite these findings, the authors suggest that vocal signals may not play a large role in mother/pup recognition, given the infrequent use of pup calls during reunification. A study in Scotland that used playback of pup calls found that individual identifiable pup calls failed to prevent allo-suckling (i.e., mothers failed to distinguish their pup’s call from that of a non-filial pup) (McCulloch et al. 1999). A similar study on Sable Island by McCulloch and Boness (2000) found that gray seals there reacted more strongly to their own pup’s call than to unfamiliar calls. The authors suggested that different selective pressure might be at work at these different sites.

2.33.10 Fisheries By-Catch and Entanglement

Total annual estimated average fishery-related mortality for 2005-2009 was 678 gray seals. Bogomolni et al. (2010) found conclusively that 43% of the gray seals included in their study died of human-related causes. The most common human interaction affecting gray seals was entanglement in fishing gear.

Table 2.31.2 Annual Estimated Fishery-related Mortality for Gray Seals along the U.S. Atlantic Coast 2005-2009

Fishery (2004-8)	Mean Annual Mortality (CV)
Northeast Sink Gillnet	678 (0.14)
Northeast Bottom Trawl	unk

2.33.11 Vessel Interactions

The impact of vessels on gray seals is unknown. In the Northeast and Southeast U.S. Marine Mammal Stranding Network databases two stranding mortalities in 2005 and two in 2008 exhibited signs of interaction with a boat propeller.

2.33.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise by seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear.

The 2010 Report of the Working Group on Marine Mammal Ecology (ICES 2010) stated that very little is known about the impact of the construction and operation of offshore wind-farms. Madsen et al. (2006b) reviewed available current information on wind farm and marine mammals and wrote that the noise impact is more severe during construction than operation. The 2011 Report of the Working Group of Marine Mammal Ecology (ICES 2011) reported that there was no change in seal haul-out behavior in the vicinity of a tidal turbine and shore-based surveys showed no evidence of disturbance during construction.

The effect of a large offshore wind farm on harbor and gray seals within a nearby seal sanctuary (4 km) in Denmark was described by Edren et al. (2010). They used time-lapse photography, visual and aerial surveys to document haul-out behavior during pre and post-construction. They found no long term impact on haul-out behavior. There was a significant short term decline in haul-out numbers during sheet pile driving at or near the wind farm.

2.33.13 Data Gaps and Research Recommendations

There is no population estimate for U.S. gray seals, only single day counts. There is little known about local and regional movements of gray seals off the northeast U.S. coast. Most food habits studies have been done in isolation and without understanding foraging range. Very little life history data is available specific to this area. Most of the contaminant and disease research has been done in other parts of their range.

2.34 HOODED SEAL (*CYSTOPHORA CRISTATA*)

2.34.1 Status

The hooded seal is not listed under the Endangered Species Act. The species is considered of "Vulnerable" on the IUCN Red List. It is afforded protection in U.S. waters under the MMPA.

2.34.2 General Distribution

Hooded seals are highly migratory and their range includes a large part of the North Atlantic Ocean (Kovacs 2009). It is not unusual for them to move out of this range; observations have been made as far south as Puerto Rico in the Atlantic Ocean and in California (Kovacs 2009). They form three breeding aggregations which are considered three separate stocks: one is off the east coast of Canada, and is further divided into two groups (The Front off Newfoundland and Labrador and the Gulf of St. Lawrence). The second breeding aggregation is found in Davis Straits and the third is located on the "West Ice" which is east of Greenland (Kovacs 2009; Lavigne and Kovacs 1988). After breeding, hooded seals move towards Denmark Strait for molting and then disperse widely from there (Lavigne and Kovacs 1988).

Hooded seals are transient in U.S. waters and are observed in the U.S. Atlantic from January to May in New England waters and off the southeast U.S. in the summer and fall (Harris et al. 2001; McAlpine et al. 1999; Mignucci-Giannoni and Odell 2001). Given their wide distribution, hooded seal could be observed in all BOEM Atlantic Planning Areas; however, they are most likely to be observed in the North Atlantic Planning Area. Sighting, stranding, and fishery bycatch locations of hooded seals are shown in Figure 2.34-1; tracks of satellite-tagged hooded seals are shown in Figure 2.34-2.

2.34.3 General Abundance

Hammill and Stenson (2006) estimated that the Northwest Atlantic hooded seal population (Front, Gulf, and Davis Strait) was 593,500 (SE=67,200) in 2005. There is a fair amount of uncertainty in this estimate due to low survey coverage in some areas/years, limited information on reproductive data, and uncertain catch statistics. Hammill and Stenson (2006) considered them "data poor" and recommended conservative catch limits. Hooded seals are hunted throughout their range by both subsistence hunters (Greenland 2008=6,397 seals taken) and in a much smaller Canadian commercial hunt (2006=0 taken but catch numbers have been much higher historically) (Stenson 2006).

2.34.4 Habitat Preference

Hooded seals are highly migratory and are usually found further offshore than harp seals (Lavigne and Kovacs 1988; Stenson et al. 1996). They are pack-ice seals and they spend most of their lives in and around sea ice (Kovacs 2009). They are found on the Newfoundland continental shelf for winter and spring (Stenson et al. 1996). Folkow and Blix (1999) studied hooded seal dive behavior using satellite- linked dive recorders. They found that most dives were to depths of 100-600 m, but hooded seals can dive to depths of over 1000 m.

2.34.5 Stock Structure

The three breeding populations have been considered panmitic (Kovacs 2009). Lavigne and Kovacs (1988) reported that all three stocks of hooded seals have their pups at the same time; this supports the theory of a single population. DNA fingerprinting (Sundt et al. 1994) and skull morphology (Wiig and Lie 1984) also supported a single hooded seal population.

Coltman et al. (2007) used 13 microsatellite loci and a portion of the mtDNA control region to assess hooded seal stock structure. They found no differentiation between the four breeding sites included: West Ice, the Gulf, the Front and Davis Strait.

2.34.6 Life History Traits

Kovacs (2009) summarized hooded seal life history traits. She reported that females reach sexual maturity at three years and males are a little bit older when they mature and are likely much older when they can actively compete for access to females. Hooded seals have the shortest lactation of any mammal, only four days (Bowen et al. 1985). Longevity is 25-35 years (Kovacs 2009). Hooded seals are sexually dimorphic; in other species this trait has resulted in females living longer than males but very little is known about hooded seal male survival (Hammill and Stenson 2006).

2.34.7 Food Habits

Hooded seals are pelagic feeders (Kovacs 2009). In their study on the estimated prey taken by seals in Atlantic Canada, Hammill and Stenson (2000) found that Greenland halibut

(*Reinhardtius hippoglossoides*) was the most common prey eaten by hooded seals. This was followed by Atlantic cod (*Gadus morhua*), redfish (*Sebastes* spp.), Arctic cod (*Boregadus saida*), Atlantic herring (*Clupea herengus*) and capelin (*Mallotus villosus*). Hauksson and Bogason (1997) reported that the main prey items of hooded seals in Icelandic waters were redfish and cod.

Tucker et al. (2009) used quantitative fatty acid signature analysis (QFASA) to investigate harp and hooded seal food habits. Their study included samples from 153 harp seals collected 1994-2004. They found that hooded seals fed primarily on amphipods, Atlantic argentine (*Argentina silus*), capelin, euphausiids, and redfish.

2.34.8 Health

2.34.8.1 Strandings

A total of 83 hooded seals stranded along the U.S. Atlantic coast (Maine to North Carolina) from 2005 to 2009 (Northeast and Southeast U.S. marine mammal stranding response databases). Nine of these animals showed signs of human interaction. Massachusetts had the highest number of observed strandings (n= 38, 46%), followed by Maine (n= 43, 20%). These two states account for 66% of all stranding observations. In their analysis of ice seal strandings, Harris and Gupta (2006) found that hooded seals tended to strand closer to deep water, farther north.

Table 2.34.1 Hooded Seal Strandings by State and Year

State	2005	2006	2007	2008	2009	TOTAL
ME	3	12	2	0	0	17
NH	0	2	0	0	1	3
MA	11	18	3	4	2	38
NY	4	6	1	1	1	13
NJ	0	3	0	1	0	4
DE	0	1	0	0	0	1
MD	0	1	0	0	0	1
VA	1	2	0	0	0	3
NC	0	2	0	0	0	2
TOTAL	19	47	7	6	4	83

(Data from the NE and SE stranding network databases)

2.34.8.2 Contaminants

Vetter et al. (2010) analyzed brominated flame retardant (DPTE) in blubber and brain tissues of hooded seals from the Barents and Greenland Seas. In hooded seals from the eastern U.S. and Canada, Montie et al. (2010) documented brominated flame retardants (BFRs), including polybrominated diphenyl ethers (PBDEs) and current-use, non-PBDE BFRs, as well as organochlorine (OC) pesticides and polychlorinated biphenyls (PCBs).

2.34.8.3 Disease

Duignan et al. (1997) analyzed hooded seal samples collected from the western Atlantic 1989-1994 for PDV. Twenty-four percent of the seals they tested were positive for PDV antibodies.

2.34.9 Acoustics

Ballard and Kovacs (1995) studied the airborne and waterborne vocalization of hooded seals in the Gulf of St. Lawrence during the breeding season. They identified three major classes of sounds containing five call types. They found that sounds that were created using the hood and septum were different from voiced calls. They recorded underwater clicks and knocks. Most of the hooded seal acoustics were used in agonistic (79%) or sexual (12%) situations.

2.34.10 Fisheries By-Catch and Entanglement

Total annual estimated average fishery-related mortality for 2001-2005 is 25 hooded seals, all of which were attributed to the Northeast Sink Gillnet Fishery. There are no records of fishery interactions involving hooded seals in the Northeast Fisheries Observer database between 2005 and 2009.

Table 2.34.2 Mean Annual Mortality by Fishery 2001-2005

Fishery (2001-2005)	Mean Annual Mortality (CV)
Northeast Sink Gillnet	25 (0.82)

(Waring et al. 2011) (CV in parentheses)

2.34.11 Vessel Interactions

There are records in the Northeast Stranding database of two hooded seals in 2006 with evidence of vessel interaction.

2.34.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on marine mammals is unclear.

The 2010 Report of the Working Group on Marine Mammal Ecology (ICES 2010) stated that very little is known about the impact of the construction and operation of offshore wind-farms. Madsen et al. (2006b) reviewed available current information on wind farm and marine mammals and wrote that the noise impact is more severe during construction than operation. The 2011 Report of the Working Group of Marine Mammal Ecology (ICES 2011) reported that there was no change in seal haul-out behavior in the vicinity of a tidal turbine and shore-based surveys showed no evidence of disturbance during construction. No information is available about the impact of energy projects on hooded seals.

2.34.13 Data Gaps and Research Recommendations

The abundance estimate for hooded seals is very uncertain due to limited surveys, limited reproductive data and uncertain catch statistics. There is very little information available on contaminant burdens or disease. Male life history data is limited.

2.35 KEMP'S RIDLEY TURTLE (*LEPIDOCHELYS KEMPII*)

2.35.1 Legal Status

The Kemp's ridley sea turtle was listed as an endangered species under the ESA in 1970 and has received Federal protection throughout its range since that time. All commercial international trade in this species has been banned since 1975 when it was listed on Appendix I by the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). The IUCN lists the Kemp's ridley as “Critically Endangered.”

2.35.2 General Distribution

Kemp's ridleys are distributed along the coastlines of the Gulf of Mexico and the northwestern Atlantic Ocean, as far north as the Grand Banks and Nova Scotia (National Marine Fisheries Service et al. 2010). They have also been reported on rare occasions in the Mediterranean (Tomás and Raga 2007), in the Azores (Bolton and Martins 1990), and in the eastern north Atlantic (Brongersma 1972). Nesting is predominantly restricted to beaches of the western Gulf of Mexico, most notably at one beach near Rancho Nuevo, Tamaulipas, Mexico. From 1978 through 1988 a total of 22,507 eggs were collected from Rancho Nuevo and incubated at South Padre Island, Texas in a successful experiment to establish a second nesting colony (Shaver and Wibbels 2007). Along the U.S. Atlantic coast, isolated nesting events have been reported in Florida (Johnson et al. 1999), Georgia (Williams et al. 2006a), and North and South Carolina (Anonymous 1992; Bowen et al. 1994). These areas fall mostly within the BOEM South Atlantic and Straits of Florida Planning Areas. The range of Kemp's ridley sea turtles is shown in Figure 2.35-1; nesting locations are shown in Figure 2.35-2. Tracks of satellite-tagged Kemp's ridleys are shown in Figure 2.35-3; sightings and fishery bycatch observations are shown in Figure 2.35-4.

2.35.3 General Abundance

Analysis of films of the nesting aggregations at Rancho Nuevo in 1947 have resulted in estimates of over 40,000 nesting females in the population at that time (Carr 1963; Hildebrand 1963). However, by 1985 the nest count reached its lowest level at 702 nests, representing a reproductive female population of fewer than 300 animals. Since the mid-1980s, nesting counts have increased by 14-16% each year (Heppell et al. 2005) and for the period 2005-2009 the number of nests on all monitored beaches indicate that approximately 5,500 females are nesting each season in the Gulf of Mexico (National Marine Fisheries Service et al. 2010). The population increase is likely due to a combination of management measures, including elimination of direct harvest, nest protection, the use of turtle excluder devices (TEDs), and reduced trawling effort in Mexico and the U.S. (National Marine Fisheries Service et al. 2010). The target criterion for downlisting Kemp's ridleys turtles from endangered to threatened is at least 10,000 nesting females in a season, a target which could be reached at current survival rates by 2015 (Heppell et al. 2005).

2.35.4 Habitat Preference

Kemp's ridley sea turtles use different habitats during different life stages. Terrestrial zones are used during oviposition and embryonic development. Figure 2.35-2 shows the distribution of Kemp's ridley nesting beaches. Patterns of nesting density were shown to be consistent with areas where oceanographic conditions are favorable for migration of hatchlings to foraging areas (Putman et al. 2010).

Hatchlings swim from the beach to the boundary current and float in the oceanic currents for the first two years of their lives. Some of the juveniles remain in the Gulf of Mexico and others are brought into the Atlantic on the Gulf Stream. The young turtles move into the nearshore waters of the Gulf of Mexico or the Atlantic coast from Florida to New England and spend the warmer months in shallow foraging areas, such as Pamlico Sound, North Carolina, Chesapeake Bay, Virginia, and Long Island Sound, New York (National Marine Fisheries Service et al. 2010). When water temperatures cool in the fall, the Kemp's ridleys join other species of sea turtles and migrate southward (Morreale and Standora 2005). By early November turtles from the northern areas of the Atlantic coast join individuals leaving mid-Atlantic areas, such as the Chesapeake Bay and North Carolina inshore waters. Coastal Florida waters south of Cape Canaveral have been demonstrated to be important wintering habitat, but Kemp's ridleys have also been shown to winter off North Carolina south of Cape Hatteras in eddies of the Gulf Stream (National Marine Fisheries Service et al. 2010). The northward migration begins again in the spring, and turtles arrive in Virginia by May and New York and New England by June (National Marine Fisheries Service et al. 2010).

Adult Kemp's ridleys are only occasionally found in the Atlantic. Their preferred habitat in the Gulf of Mexico is thought to be nearshore waters of 37 meters or less in the Gulf of Mexico (National Marine Fisheries Service et al. 2010; Shaver and Rubio 2008).

Most tagging studies using Kemp's ridley turtles have taken place in the Gulf of Mexico (Renaud and Williams 2005; Sasso and Witzell 2006; Schmid and Witzell 2006; Schmid 1998; Schmid et al. 2003, 2008, 2002; Seney and Landry 2008, 2010; Shaver and Rubio 2008; Shaver et al. 2005). However, Kemp's ridleys have also been tagged and tracked in the Long Island Sound area (Morreale and Standora 1998), and elsewhere along the U.S. Atlantic coast. Of 69 satellite tagged Kemp's ridleys displayed on SeaTurtle.org (http://www.seaturtle.org/tracking/), seven are in the Atlantic and one that was tagged at South Padre Island, Texas travelled into the Straits of Florida Planning Area. There are data from two tagged and released Kemp's ridley turtles in the Atlantic in the Ocean Biogeographic Information System (OBIS) dataset, one from 2007 and one from 2008 (Figure 2.35-3; http://www.iobis.org/). Seven Kemp's ridley turtles were tagged in Pamlico Sound, North Carolina, in 2002-2004 (McClellan et al. 2009). Gitschlag et al. (1996) tracked three Kemp's ridley turtles with satellite and radio tags in 1991 along the Southeast Atlantic coast. Percent submergence time of these turtles was 94-95%. These turtles overwintered within approximately 40 km off the coast between Cape Canaveral and Stuart, Florida and in late spring and early summer spent time in coastal South Carolina. Three Kemp's ridleys were included in Mansfield and Musick's 2003 tagging study in Chesapeake Bay, Virginia (Mansfield and Musick 2004).

Kemp's ridley sighting data in the OBIS dataset consists of six records from CETAP 1978-1980 (four aerial and two opportunistic), one record from a SEFSC 1992 cetacean aerial survey, eight records from a 1995 cetacean aerial survey, 16 records from SEFSC mid-Atlantic Tursiops surveys in 1995, 63 records from a NEFSC summer 1998 aerial survey, two 2 records from a NEFSC experimental aerial survey in 2002, 103 records from a UNCW marine mammal survey in 2002, five from UNCW marine mammal surveys 2006-2007, and 31 from UNCW right whale surveys 2005-2006. Kemp's ridley turtles are found as far north as the North Atlantic Planning Area in summer, and in the other planning areas in other seasons. The sighting data presented here is not effort-corrected density data and should be used only as an indication of where Kemp's ridleys have been seen and not a portrayal of their actual distribution.

A habitat suitability index (HSI) model has been developed for the Kemp's ridley sea turtle. A map of HIS model outputs by month is viewable at http://www.seaturtle.org/research/hsi.html. The model is based on assumptions of habitat preference of water depths less than 10 m and temperatures between 22°C and 32°C (Department of the Navy 2005).

2.35.5 Stock Structure

There are no stock divisions within the Kemp's ridley species. Dutton et al. (2006) compared mtDNA from Padre Island nesting turtles with that from animals from the Rancho Nuevo population and found genetic homogeneity.

2.35.6 Life History Traits

Nesting occurs April-July, often in synchronized emergences called *arribadas*. Each season nesting females lay an average of 2.5 clutches of about 100 eggs each (TEWG [Turtle Expert Working Group] 2000). Adult females nest, on average, every two years. Genetic work by Kichler et al. (2002) has demonstrated multiple paternity of Kemp's ridley clutches. Human protection of eggs by moving them to enclosed corrals has been implemented since 1978. Studies done on Rancho Nuevo nests left in situ have shown that while predator satiation is an effective natural survival strategy (LeBlanc et al. 2010), keeping eggs in restricted areas does enhance reproductive success (LeBlanc et al. 2008).

Threats to sea turtle nests include predation, pollution, construction, and other habitat modifications, such as beach nourishment, armoring, cleaning, vehicle driving, and erosion (National Marine Fisheries Service et al. 2010). Light pollution has been shown to cause disorientation of hatchlings (National Marine Fisheries Service et al. 2010).

The sex determination of Kemp's ridley sea turtles is temperature dependent (Wibbels 2007); females are produced at warm temperatures and males at cooler temperatures. Endochrinological examination of captive Kemp's ridleys has been performed to better understand seasonal reproductive cycles (Rostal et al. 1998).

Growth and aging research on Kemp's ridleys has been done by skeletochronological analysis (Avens and Hohn 2008; Snover and Hohn 2004; Snover et al. 2008; Zug et al. 1997) and mark-recapture studies (Schmid 1998). Age at maturity is estimated to be approximately 12 years (Snover et al. 2008).

2.35.7 Food Habits

Kemp's ridley sea turtles in their oceanic juvenile stage subsist on the macro algae *sargassum* and associated invertebrates. Stomach content analysis of several juvenile turtles that had stranded in Texas (Shaver 1991) supports this assumption.

Once recruitment to the nearshore neritic zone takes place, the juvenile turtles' diet shifts to mainly crabs (National Marine Fisheries Service et al. 2010), although the benthic tunicate (*Molgula occidentalis*) was found to be an important prey item in a 2005 study in the Ten Thousand Islands of southwest Florida (Witzell and Schmid 2005). A telemetric monitoring and GIS mapping study by Schmid et al. (2008) quantified habitat use by Kemp's ridleys in the developmental habitat in the Ten Thousand Islands area. Live bottom areas (sessile invertebrates on hard substrate) were preferred. Analysis of stomach contents of stranded Kemp's ridley in the Chesapeake Bay area showed dietary shifts from predominantly horseshoe crab (*Limulus polyphemus*) in the early to mid-1980s to blue crab (*Callinectes sapidus*) during the late 1980s and early 1990s, and then to menhaden (*Brevoortia tyrannus*) and croaker (*Micropogonias undulatus*) from the late 1990s to 2002. This suggests that population declines in crab populations may have led to turtles feeding on fish caught in nets or discarded (Seney and Musick 2008). Stomach content analysis by Morreale and Standora (1998) of juvenile Kemp's ridleys in the New York Sound area showed that relative abundance of available crab species was not the primary factor governing prey selection and species composition in the gut samples seemed to reflect the more easily-captured crabs. Burke et al. (1993) reported that the mostly commonly-found crab in Kemp's ridley diet in the Long Island area was the nine-spined spider crab (*Libnia emarginata*)

2.35.8 Health

2.35.8.1 Strandings

When water temperatures fall below 8-10°C, Kemp's ridleys are susceptible to becoming incapacitated in a natural phenomenon called cold stunning. This is common each year along the shores of Long Island Sound (Morreale et al. 1992) and Cape Cod Bay. From 1994 through 2006, 1,084 immature Kemp's ridleys were cold-stunned in the northeast U.S., with over half initially stranded alive. Some of the turtles were rehabilitated and released (National Marine Fisheries Service et al. 2010).

Hart et al. (2006) examined the spatial distribution of sea turtle strandings in North Carolina and concluded that strandings follow patterns that can be predicted from physical oceanography.

The Southeast Fisheries Science Center hosts a queryable dataset on Atlantic and Gulf of Mexico sea turtle strandings at http://www.sefsc.noaa.gov/seaturtleSTSSN.jsp. Figures 2.35-4 and 2.35-5 show density distributions of strandings along the Atlantic coast. Strandings are high in the

North Atlantic Planning Area in winter and fall; in spring and summer strandings are more prevalent in the Mid and South Atlantic Planning Areas.

2.35.8.2 Contaminants

Lake et al. (1994) studied PCB, DDE, and transnonachlor concentrations in cold-stunned Kemp's ridleys that had stranded in the Long Island Sound area between 1980 and 1989. While higher levels of PCBs were found in Kemp's ridleys than were reported in other species of sea turtles, levels were more than 20 times lower than thresholds found to cause reproductive problems in freshwater turtles. More recently, Innis et al. (2008) examined cold-stunned juvenile Kemp's ridleys that had stranded on Cape Cod between 2001 and 2006 and concluded that the turtles may be exposed to contaminants at a young age but that further investigation into sources of the contaminants and the effects on the health of the turtles is needed. Work by Keller et al. (2008) focused on PFC concentrations in loggerhead and Kemp's ridley sea turtles and demonstrated positive correlations between length and latitude of capture, suggesting that bio-accumulation of these compounds may be influenced by age and trophic level. Kemp's ridleys had significantly higher PFC concentrations than the loggerhead sea turtles tested (Keller et al. 2008) and turtles captured in near shore waters of North Carolina had higher concentrations than those captured offshore from South Carolina to Florida (Keller et al. 2005). Work done by Swarthout et al. (2010) has helped to establish baseline organohalogen contaminant concentrations for Kemp's ridleys and green turtles. Levels found in Kemp's ridleys from the southeastern U.S. coast were similar to those found in Kemp's ridleys from the Gulf of Mexico, although highly chlorinated PCBs were higher in Atlantic samples, indicating exposure to Aroclor 1268 (Swarthout et al. 2010).

2.35.8.3 Disease

Pathological and parasitological examination was performed on cold-stunned Kemp's ridleys that had stranded on Cape Cod between 2001 and 2006 (Innis et al. 2009). Necrotizing enterocolitis and bacterial or fungal pneumonia were the most frequently encountered lesions. While considered clinically significant, in many cases the lesions were not judged to have been the primary cause of death, suggesting that additional factors, such as hypothermia; drowning, and metabolic or respiratory problems, may be factors causing death in cold-stunned turtles.

2.35.9 Acoustics

Little has been done specifically on Kemp's ridleys and acoustics, though studies on loggerhead sea turtles have demonstrated sensitivity to low frequency noise (Bartol et al. 1999; Samuel et al. 2005). Underwater sound recordings from the Peconic Bay Estuary system in Long Island, New York demonstrated high levels of anthropogenic noise at frequencies in the range of the turtles' highest sensitivity that also overlapped spatially and temporally with high turtle use of the habitat (Samuel et al. 2005). Sea turtle exposure to seismic air gun noise was studied by Weir (2007), though no conclusions on the impacts to the turtles by the airguns were able to be drawn in that study. Viada et al. (2008) discussed sea turtle response to underwater explosions and mitigation scenarios developed for explosive removal of oil and gas structures. The authors concluded that, while existing regulations provide more protection for marine turtles than had previously existed, little is known about sub-lethal effects of underwater explosions on turtles, especially potential auditory impacts (Viada et al. 2008).

2.35.10 Fisheries By-Catch and Entanglement

Bycatch of Kemp's ridley sea turtles has been documented in several Atlantic fisheries, including mid-Atlantic gillnet (Murray 2009a), northeast bottom trawl, mid-Atlantic sea scallop dredge (Murray 2004a, 2004b), beach haul seine and pound nets (Epperley et al. 2007). Between 1998 and 2009, 23 Kemp's ridley interactions were reported in the NMFS fishery observer database. In addition, a dead Kemp's ridley was observed in a Massachusetts fishing weir in 2007 (National Marine Fisheries Service et al. 2010). Most of the observed fishery bycatch of Kemp's ridley turtles has been in the BOEM Mid-Atlantic Planning Area.

McLellan et al. (2009) used telemetry to assess the potential for interactions between turtles and gillnet fisheries in the Pamlico Sound area of North Carolina. Snoddy et al. (2009) studied the blood biochemistry of Kemp's ridley and green sea turtles incidentally captured in the coastal gillnet fishery of North Carolina and found that entanglement time, depth, and severity of entanglement had negative impacts on the selected blood markers associated with health status.

2.35.11 Vessel Interactions

Damage from collisions with boats and ships is commonly found in sea turtles. In the stranding data collected by the Sea Turtle Stranding and Salvage Network from 1997 to 2001, 12.7% of all stranded turtles were documented as having sustained injuries consistent with propeller wounds or collision, although some of these injuries may have occurred post-mortem (National Marine Fisheries Service et al. 2010).

2.35.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on turtles is unclear. McMichael and Wyneken (2010) reported on a research program aimed at monitoring and minimizing the impacts from interactions between sea turtles and ocean energy technology within the Florida Current.

Entrainment in nuclear power plant intake canals has been a source of Kemp's ridley morality along the U.S. Atlantic coast. Over 100 interactions with Kemp's ridley turtles have been documented at nuclear power plants since 1976 in Florida, North Carolina, and New Jersey, although many of these were non-lethal (National Marine Fisheries Service et al. 2010).

Effects of oil and oil spill response guidelines are presented in a report edited by Shigenaka (2003). The report concluded that not only do oil collection areas overlap with sensitive sea turtle habitat, but there is also strong, if limited and dated, information that indicates oil is harmful to turtles, particularly to dermal tissues and membranes. In addition, fresh oil has been found to harm sea turtle eggs. The relationship between offshore oil platforms and the distribution of sea turtles is discussed in a report made by Lohoefener et al. at the 1989 Ternary Studies meeting sponsored by the MMS (Lohoefener et al. 1989). That study found that in some areas of the Gulf of Mexico, chelonid sea turtles were positively correlated with platforms, and in others there was either a negative correlation or no correlation at all.

Under a plan developed by the U.S. Fish and Wildlife Service, NMFS, and the Florida Fish and Wildlife Conservation Commission, Kemp's Ridley, loggerhead, and green sea turtle nests from areas of the Florida panhandle and Alabama affected by the Deepwater Horizon oil spill were relocated to the Atlantic coast of Florida (for more information see http://www.fws.gov/home/dhoilspill/pdfs/TurtleNestHatchProgram.pdf**).**

2.35.13 Data Gaps and Research Recommendations

Hays (2008) reviewed recent research on sea turtles, pointing out topical areas and some remaining questions, and Heenehan (2010) summarized work on acoustic impacts to sea turtles. There have been no recent systematic abundance surveys for Kemp's ridley sea turtles, but the ongoing Atlantic Marine Assessment Program for Protected Species (AMAPPS) project is designed to address this shortfall. As stated in the draft recovery plan "Additional research and monitoring are needed to identify important marine foraging, breeding, and internesting habitats; determine migratory pathways among foraging grounds and between foraging grounds and nesting beaches; and collect data on interactions between Kemp's ridleys and recreational and commercial fisheries" (National Marine Fisheries Service et al. 2010). The National Research Council's *Assessment of Sea-turtle Status and Trends* (Committee on Sea Turtle Population Assessment Methods; National Research Council 2010) recommended that NMFS and the USFWS develop a coherent national strategy for sea-turtle assessments to improve collection, quality and availability of data and to develop a plan of external review of data and models used for population status estimation and trend analysis.

2.36 GREEN TURTLE (*CHELONIA MYDAS*)

2.36.1 Legal Status

The green turtle was listed under the ESA on July 28, 1978. The breeding populations in Florida and Mexico's Pacific coast are listed as endangered; elsewhere the species is listed as threatened. The IUCN lists green turtles as endangered globally, based on the 2004 global status assessment (Marine Turtle Specialist Group 2004) which discussed extensive subpopulation declines in the last three generations (generation length used for the Florida subpopulation was 35.5). International trade of green turtles is prohibited under CITES. Other international treaties and agreements protecting green turtles include the Memorandum of Understanding on the Conservation and Management of Marine Turtles and their Habitats of the Indian Ocean and South-East Asia (IOSEA) and the Memorandaum of Understanding Concerning Conservation Measures for Marine Turtles of the Atlantic Coast of Africa under the CMS, Annex II of the Specially Protected Areas and Wildlife (SPAW) Protocol of the Cartagena Convention, and the Inter-American Convention for the Protection and Conservation of Sea Turtles (IAC). Green turtles are also protected by the Memorandum of Understanding on ASEAN Sea Turtle Conservation and Protection and the Memorandum of Agreement on the Turtle Islands Heritage Protected Area (TIHPA).

2.36.2 General Distribution

Green turtles are found in tropical and subtropical water circumglobally. This highly migratory species inhabits the coastal waters of over 140 countries, and nesting occurrs in more than 80 countries. In the Atlantic U.S. there is a regionally significant nesting aggregation in Florida (the second largest colony in the western hemisphere). Green turtles also nest in small numbers in the U.S. Virgin Islands, Puerto Rico, Georgia, South Carolina, and North Carolina (see Table 2.36.3-1). The BOEM South Atlantic and Straits of Florida Planning Areas are the primary BOEM areas with nesting green turtles. The range of green sea turtles is shown in Figure 2.36-1 and nesting locations are shown in Figure 2.36-2. Tracks of satellite-tagged green turtles are shown in Figure 2.36-3; sightings and fishery bycatch observations are shown in Figure 2.36-4.

2.36.3 General Abundance

The 2004 IUCN Red List assessment of world green turtle populations evaluated nesting trends at 32 index sites, each presumed to be genetically distinct (Marine Turtle Specialist Group 2004). The Florida index site was shown to have an increasing population between 1980 and 2001. Nesting continued to rise between 2001 and 2005 (Meylan et al. 2006; National Marine Fisheries Service and U.S. Fish and Wildlife Service 2007b). Chaloupka et al. (2008) estimated that rates of green turtle nesting populations at six of the world's major rookeries increased 4-14% per year between 1980 and 2005. Highest population growth rates were found at the Archie Carr National Wildlife Refuge in Florida (Chaloupka et al. 2008). Table 2.36.3-1 shows U.S. nest counts from 2001 through 2010.

Table 2.36.3 Green Turtle Nest Counts

	Florida	Georgia	S. Carolina	N. Carolina
2001	581	-	1	-
2002	9,201	-	0	-
2003	2,262	-	1	-
2004	3,577	-	1	-
2005	9,644	-	0	-
2006	4,970	-	1	-
2007	12,751	-	1	-
2008	9,228	-	1	0
2009	4,462	0	1	3
2010	13,151	6	6	18

Data from http://research.myfwc.com/images/articles/2496/green_turtle_nesting_data__2005-2009.pdf, http://myfwc.com/research/wildlife/sea-turtles/nesting/statewide/ , http://www.seaturtle.org/nestdb/index.shtml, North Carolina Wildlife Resources Commission Sea Turtle Nest Monitoring System, South Carolina Department of Natural Resources Sea Turtle Nest Monitoring System, Georgia Department of Natural Resources Sea Turtle Nest Monitoring System, and National Marine Fisheries Service and U.S. Fish and Wildlife Service

The large inter-annual variation in nesting numbers has been attributed to synchrony in the intervals between nesting years of individuals, and hypothesized to be attributable to environmental variations that affect the feeding conditions at sea (Broderick et al. 2001). Solow et al. (2002) showed that sea surface temperature did have a significant effect on the re-migration interval in the largest Atlantic population of green turtles.

Bjorndal et al. (2005) generated annual abundance estimates of juvenile green turtles at two foraging grounds in the Bahamas, based on long-term capture-mark-recapture (CMR) studies. At one of the sites, abundance showed little variation over time, while at the other there were successive phases of increase, decrease, and stability, resulting from changes in immigration. These trends did not conform to the general increasing trend of the nesting population at Tortuguero, Costa Rica, and points out the difficulties in assessing population-wide trends in green turtles and other long-lived marine species.

In a 2000 paper, Bjorndal et al. determined that for green turtles on a feeding ground in the Bahamas, mean annual growth rate and condition index were inversely correlated with population density pointing to density dependent nutrient limitation (Bjorndal et al. 2000a). They estimated the carrying capacity of pastures of the seagrass *Thalassia testudium*, the major diet plant of the green turtle, to be in the range of 122 to 4439 kg green turtles/ha.

Chaloupka et al. (2002) developed a stochastic simulation model to provide better insight into the population dynamics of the southern Great Barrier Reef (sGBR) green sea turtle stock. Fertility and adult survival were the most important high level parameters affecting population growth. A model developed to describe Hawaiian green turtle population dynamics was presented by Chaloupka and Balazs (2007). Mazaris et al. (2006) developed an individual-based model to analyze the variability of extinction probabilities as a result of mortality factors at different life stages of a sea turtle population. The results of their analysis showed that the pelagic stage has a significant effect on population persistence. Increased survival of the first year cohort was also shown to be beneficial for the population.

2.36.4 Habitat Preference

Green turtles occupy three different habitat types, depending on life stage. Nesting takes place on high-energy oceanic beaches, hatchlings go through a pelagic period in open ocean convergence zones, and when juveniles reach a carapace length of approximately 20-25 cm they move into benthic feeding grounds (National Marine Fisheries Service and U.S. Fish and Wildlife Service 1991).

As with other marine turtles, hatching success and sex determination are influenced by nesting beach temperatures. As a result, global climate change poses a threat to these species. Temperatures at green turtle nesting sites on the Great Barrier Reef of Australia were modeled by Fuentes et al. (2009), who found that, under an extreme scenario of climate change, hatchlings would be all female by 2070, although other nesting grounds would still produce males under the same scenario. While loggerhead turtles seem to be adjusting the seasonality of their nesting in response to warming temperatures (Pike et al. 2006; Weishampel et al. 2004), green turtle nesting patterns studied at the Canaveral National Seashore, Florida, were unrelated to environmental temperatures (Pike 2009).

Time-depth recorders mounted on adult and immature green turtles in Moreton Bay, Australia, recorded the turtles' preferences for shallow water habitats (Hazel et al. 2009). The turtles studied spent 80% of their time at depths of 3 meters or less. Time-depth recordings of green turtles in the Cayman Islands also showed use of the shallow lagoon waters, but, in addition, recorded turtle excursions into the less-protected reef areas where deeper dives occurred (Blumenthal et al. 2010).

Work by Makowski et al. (2005) showed that juvenile green turtles occupy stable home ranges along the nearshore worm-rock reefs of southeast Florida, during the summer and fall.

Brooks et al. (2009) correlated turtle movements and tides in a feeding area on the Pacific coast of Baja California. They found turtles to be transported on continual tides, allowing them to exploit the patchy and seasonal distribution of the algae which forms their main diet component.

Sharks are a primary marine predator of adult and juvenile sea turtles. Heithaus et al. (2002) examined frequencies of shark-inflicted damage to green turtle and loggerhead turtles in Shark Bay, Western Australia. Green turtles were found in tiger shark habitats more frequently than were loggerheads, yet they exhibited less evidence of shark-inflicted injuries, and no sex difference in injury frequency.

In a summary of satellite-tracking studies of sea turtles by Godley et al. (2008), green turtles are one of the most studied species of marine turtle; tagging research helps to elucidate such issues as oceanic migrations, fidelity to foraging sites, and dive behavior. However, the U.S. Atlantic coast is absent from the list. Telemetry data in OBIS for the Atlantic region consists of data from 14 green turtles tagged by the Duke University Marine Lab (5 in 2003, 1 in 2005, 7 in 2006 and 1 in 2007), two rehabilitated turtles released by the Virginia Stranding Response Program (1 in 2007 and 1 in 2009), and one rehabilitated turtle released by the Cape Cod Sea Turtle Release Program.

Green turtle sighting data in the OBIS dataset consists of 7 records from CETAP (3 in 1972 and 4 in 1980), three records from SEAMAP South Atlantic, one record from a SEFSC 1992 cetacean aerial survey, 8 records from a 1995 cetacean aerial survey in 1991, 1992 and 1993, 8 records from SEFSC surveys in 1992, 5 from a SEFSC survey in 1995, and one from a SEFSC survey in 1999, 4 records from a NEFSC summer 2004 aerial survey, one record from a UNCW right whale aerial survey in 2006, and 15 records from the Bahamas Marine Mammal Research Organization between 1992 and 2005. While this evidence is sparse, it does show that green turtles are found as far north as BOEM's North Atlantic Planning Area in summer, and in the other planning areas in other seasons. The sighting data presented here is not effort-corrected density data and should be used only as an indication of where green turtles have been seen and not a portrayal of their full distribution.

2.36.5 Stock Structure

Bass and Witzell (2000) analyzed mtDNA markers in the juvenile green turtle population at Hutchinson Island, Florida, to determine the nesting origin. The composition of the population was determined to be 53% from Costa Rica nesting grounds, 42% from the U.S. and Mexico and 4% from Venezuela and Surinam (Bass and Witzell 2000). In a 2006 paper, Bass et al. presented a similar analysis of a North Carolina foraging aggregation of green turtles, indicating that 54% of the animals originated from the east coast of the U.S. and 27% originated in Mexico (Bass et al. 2006). They found significant nesting aggregation genetic structure, as well as regional population structure between northern and southern foraging aggregations in the Caribbean. Genetic diversity was highest at foraging aggregations in confluences of major current systems, suggesting that both currents and behavior influence the composition of these groups. Juvenile homing to regional foraging grounds was also shown.

2.36.6 Life History Traits

A skeletochronological analysis of juvenile green turtles in the developmental habitat of Indian River Lagoon, Florida, yielded age estimates for that population of 3-14 years (Zug and Glor 1998). Growth rate estimates ranged from 30-53 mm/year. This suggests that in the western Atlantic, green turtles leave the pelagic stage at 5 or 6 years of age, and spend 6 or more years in the developmental neritic zone (Zug and Glor 1998). Goshe (2009), also using skeletochronological techniques, estimated age at maturation to be 44 years for green turtles of the Florida population and 42.5 years for the Costa Rican population.

2.36.7 Food Habits

Green turtles are largely herbivorous in their benthic juvenile and adult stages, subsisting mainly on seagrass and macroalgae. In the Hawaiian Islands, green turtles have shifted dietary preferences to include non-native invasive algae species (Russell and Balazs 2009). In a study of juvenile green turtles in Florida, Gilbert et al. (2008) found that the turtles selectively foraged on chlorophytic and rhodophytic algae. Juvenile green turtles off Brevard County, Florida, were studied by Holloway-Adkins (2006), who found that in that shallow, high-energy environment, the turtles were primarily feeding on red algae. Hatase et al. (2006) used satellite telemetry and stable isotope analysis to determine that oceanic habitats were alternative feeding areas for adult post-nesting green turtle females off Japan. Parker and Balazs (2008) studied stomach contents of eleven oceanic-stage green turtles in the North Pacific. Results showed that these turtles were omnivorous, concentrating on prey items at or near the surface–mainly coelenterates and associated amphipods. Boyle and Limpus (2008) examined stomach contents of post-hatchling loggerhead and green turtles in the southwest Pacific and found similar results–the turtles at that stage were opportunistic feeders, consuming zooplankton and other floating prey, and that small oceanic green and loggerhead turtles had very similar diets. Stable isotope analysis of oceanic stage green turtles around the Azores revealed that these turtles were spending 3-5 years as carnivores before settling into the herbivorous benthic phase (Reich et al. 2007).

2.36.8 Health

2.36.8.1 Strandings

The Southeast Fisheries Science Center hosts a queryable dataset on Atlantic and Gulf of Mexico sea turtle strandings at http://www.sefsc.noaa.gov/seaturtleSTSSN.jsp. Figures 2.36-5 and 2.36-6 show density distributions of strandings along the Atlantic coast. Strandings are observed in all planning areas in the fall, but are limited to the Mid-Atlantic, South Atlantic and Straits of Florida Planning Areas in other seasons.

2.36.8.2 Contaminants

The use of blood samples to estimate green turtle tissue contamination by persistent organic pollutants (POPs) and heavy metals was demonstrated to be effective by van der Merwe et al. (2010b). Gas chromatography with tandem mass spectrometry was also proven effective at analyzing contaminants in green turtle eggs, blood, and tissue (van de Merwe et al. 2009). Van der Merwe et al. (2010a) looked at maternal transfer of persistent organic pollutants in green turtles in Malaysia and the effects of POPs on embryonic development. They found evidence of maternal transfer, as well as a significant inverse relationship between POP levels in eggs and mass/length ratios in hatchlings. Work done by Swarthout et al. (2010) has helped to establish baseline organohalogen contaminant concentrations for Kemp's ridleys and green turtles. PCBs, DDTs and chlordanes were one order of magnitude lower in green turtles then in the Kemp's ridley turtles examined, and PBDE concentrations were lower by half due to trophic level differences. Decreasing lipid contaminant burdens with turtle size were observed in green turtles by Mckenzie et al. (1999), which the authors postulated is attributable to a change in diet with age. Pattern differences between loggerhead and green turtles, elucidated by principal component analysis, confirmed bioaccumulation differences (McKenzie et al. 1999).

2.36.8.3 Disease

The tumor disease fibropapillomatosis (FP), while prevalent in all hard-shelled turtles, is most frequent in green turtles. Hirama and Ehrhart (2007) compared occurrence rates and severity of FP in green turtles from three sites on the Florida Atlantic coast: Indian River Lagoon, the nearshore reef, and the Trident Submarine Basin. FP prevalence was highest in the Lagoon system. McGarrity (2008) examined stress protein expression in both healthy and fibropapilloma afflicted green and loggerhead turtles from Indian River Lagoon and compared levels to those from turtles from more pristine offshore reef environments. The stability in the marine environment of the herpesvirus associated with FP, as well as several other green turtle diseases, including lung-eye-trachea disease and gray patch disease, was studied by Curry et al. (2000). They found that the virus remains infectious for extended periods of time in sea water. Herbst and Klein (1995) looked at the role of environmental contaminants in green turtle FP infection, and argued that, while occurrence seems to be higher in areas with high anthropogenic impact, there may be other factors, such as the prevalence of other stressors, density dependent relationships, or habitat favorability, that may be contributing to the rates of infection in these areas. However, Foley et al. (2005) in an examination of stranded turtles in southern Florida, demonstrate that FP was most prevalent in the area with the greatest degree of marine habitat degradation and pollution, largest extent of shallow-water area, and lowest wave-energy level; they also point out that a high prevalence of FP did not correspond to a high density of green turtles.

A high rate of infection by spirorchiid trematodes was found in Florida stranded green and loggerhead turtles, and in some cases caused or contributed to death. (Stacy et al. 2010).

2.36.9 Acoustics

Ridgway et al. (1969) measured the cochlear potentials in three specimens of the green sea turtle in response to both aerial and mechanical stimulation, demonstrating maximum sensitivity in the region of 300 to 400 Hz. They detected a rapid decline in sensitivity for lower and especially for higher tones. The upper limit for the observation of cochlear potentials without injury was 2000 Hz. Yudhana et al. (2010) studied Auditory Brainstem Responses of green turtles in Malaysia and also found maximum sensitivity levels at 300 Hz. Sea turtle exposure to seismic air gun noise was studied by Weir (2007), though no conclusions on the impacts to the turtles by the airguns were able to be drawn in that study. Viada et al. (2008) discussed sea turtle response to underwater explosions and mitigation scenarios developed for explosive removal of oil and gas structures. The authors concluded that while existing regulations provide more protection for marine turtles than had previously existed, little is known about sub-lethal effects of underwater explosions on turtles, especially potential auditory impacts (Viada et al. 2008).

2.36.10 Fisheries By-catch and Entanglement

A satellite-tracking study by McClellan and Read (2009) examined the fine-scale movements of juvenile green turtles in Pamlico Sound, North Carolina. The study showed that turtles and gill net fishers had similar habitat preferences, which resulted in a high potential for interaction and entanglement. Turtles in Pamlico Sound established summer foraging sites in seagrass habitat and demonstrated strong site-fidelity. Fisheries observers have also documented high interaction rates between green turtles and large mesh sink gillnet gear (Murray 2009; Byrd et al. 2011) and pound nets (Epperly et al. 2007) in North Carolina waters.

2.36.11 Vessel Interactions

Damage from collisions with from boats and ships is commonly found in sea turtles. In the stranding data collected by the Sea Turtle Stranding and Salvage Network from 1997 to 2001, 12.7% of all stranded turtles were documented as having sustained injuries consistent with propeller wounds or collision, although some of these injuries may have occurred post-mortem (National Marine Fisheries Service et al. 2010). The correlation between internesting loggerhead and green turtle dive behavior at Casey Key, Florida, and boat traffic in the area was assessed by Sobin and Tucker (2010) to determine if there were periods of increased surface activity that put turtles at greater risk for boat strike injury. The proportion of green turtles that fled to avoid an oncoming vessel was found to decrease as the vessel speed increased and turtles that fled from faster approaching vessels did so at significantly shorter distances (Hazel et al. 2007).

2.36.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on turtles is unclear. McMichael and Wyneken (2010) reported on a research program aimed at monitoring and minimizing the impacts from interactions between sea turtles and ocean energy technology within the Florida Current.

Effects of oil and oil spill response guidelines are presented in a report edited by Shigenaka (2003). The report concluded that not only do oil collection areas overlap with sensitive sea turtle habitat, but there is also strong, if limited and dated, information that indicates oil is harmful to turtles, particularly to dermal tissues and membranes. In addition, fresh oil has been found to harm sea turtle eggs. The relationship between offshore oil platforms and the distribution of sea turtles is discussed in a report made by Lohoefener et al. at the 1989 Ternary Studies meeting sponsored by the MMS (Lohoefener et al. 1989). That study found that in some areas of the Gulf of Mexico, chelonid sea turtles were positively correlated with platforms, while in others there was either a negative correlation or no correlation at all. High amplitude simulated pile driving vibrations were found to decrease sea turtle hatching success (Ripcke 2011).

Under a plan developed by the U.S. Fish and Wildlife Servece, NMFS, and the Florida Fish and Wildlife Conservation Commission, Kemp's Ridley, loggerhead, and green sea turtle nests from areas of the Florida panhandle and Alabama affected by the Deepwater Horizon oil spill were relocated to the Atlantic coast of Florida (see http://www.fws.gov/home/dhoilspill/pdfs/TurtleNestHatchProgram.pdf**).**

2.36.13 Data Gaps and Research Recommendations

Hays (2008) reviewed recent research on sea turtles, pointing out topical areas and some remaining questions, and Heenehan (2010) summarized work on acoustic impacts to sea turtles. There have been no recent systematic abundance surveys for green sea turtles but the ongoing AMAPPS project is designed to address this shortfall. Data on fisheries interactions with green turtles in the western Atlantic is sparse, as is work on the contaminant burdens of western Atlantic green turtles. More information on the abundance and dispersion of the oceanic phase juveniles would help improve understanding of the U.S. Atlantic population.

2.37 HAWKSBILL TURTLE (*ERETMOCHELYS IMBRICATA*)

2.37.1 Legal Status

The hawksbill sea turtle was listed as an endangered species in 1970. Coastal waters surrounding Mona and Monito Islands, Puerto Rico, were designated as critical habitat for hawksbill turtles by NMFS in 1998. All commercial international trade in this species has been banned since it was listed on Appendix I by the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) in 1975. The IUCN lists the hawksbill as "Critically Endangered."

2.37.2 General Distribution

Hawksbill turtles are circumtropical, found in the Atlantic, Pacific, and Indian Oceans and associated water bodies between 30°N and 30°S. Hawksbills are widely distributed throughout the Caribbean Sea and western Atlantic Ocean, regularly occurring in southern Florida and the Gulf of Mexico, in the Greater and Lesser Antilles, and along the Central American mainland south to Brazil. On the Atlantic coast of the U.S., the species is recorded as far north as Massachusetts, but sightings north of Florida are rare (NMFS Office of Protected Resources n.d. http://www.nmfs.noaa.gov/pr/species/turtles/hawksbill.htm). In Florida, hawksbills are regularly seen in the waters near the Florida Keys and on the reefs off Palm Beach County (U.S. Fish and Wildlife Service 1999). Nesting along the Atlantic coast is rare, and is restricted to the southeast coast of Florida and the Florida Keys (NMFS Office of Protected Resources n.d. http://www.nmfs.noaa.gov/pr/species/turtles/hawksbill.htm), thus primarily in the BOEM Straits of Florida Planning Area. From 1979 to 1992, 11 nests were reported in the state of Florida (U.S. Fish and Wildlife Service 1999). The range of hawksbill sea turtles is shown in Figure 2.37-1 and nesting locations are shown in Figure 2.37-2. Tracks of satellite-tagged hawksbills are shown in Figure 2.37-3; sightings and fishery bycatch observations are shown in Figure 2.37-4.

2.37.3 General Abundance

Globally, most populations of hawksbill turtles are declining, depleted, or are remnants of larger populations (Meylan and Donnelly 1999). There are only five regional populations with more than 1,000 females nesting annually (Meylan and Donnelly 1999). In the Caribbean, hawksbill populations are reported to be declining or depleted in 22 of the 26 geopolitical units reviewed by Meylan (1999). The number of females nesting annually in the region was estimated to be 5000. The only increasing populations were those of Mexico and Mona Island, Puerto Rico. However, recent monitoring of populations on Guadeloupe, French West Indies (Kamel and Delcroix 2009), Barbados, West Indies (Beggs et al. 2007), Antigua, West Indies (Richardson et al. 2006), and Panama (Ordonez et al. 2008) show encouraging increases. McClenachan et al. (2008) used historical sources to calculate that 20% of historic nesting sites in the Caribbean have been lost entirely and 50% of the remaining nesting sites have been reduced to dangerously low populations.

2.37.4 Habitat Preference

Hawksbills nest on insular and mainland beaches of tropical and subtropical oceans of the world, exhibiting a wide tolerance for nesting substrate type (US Fish and Wildlife Service 1999). In the Jumby Bay, Antigua nesting aggregation in the West Indies, hawksbills tend to nest in vegetation (Ballentine et al. 2008). Caribbean region hawksbill hatchlings go through a pelagic period in open ocean convergence zones, and when juveniles reach a carapace length of approximately 20-25 cm, they move into benthic feeding grounds (National Marine Fisheries Service and U.S. Fish and Wildlife Service 2007a; U.S. Fish and Wildlife Service 1999).

As juveniles, subadults, and adults, hawksbill turtles are closely associated with coral reef and other hard-bottom habitats. Seagrass pastures, mangroves, algal beds, and mudflats can be peripheral habitats (Mortimer and Donnelly 2008). Addressing the concern that dwindling healthy coral reef resources may have profound effects on the viability of hawksbill populations, Bjorndal and Bolten (2009) used data from a 30-year mark-recapture study of hawksbills and green turtles in the southern Bahamas to assess the quality of a seagrass habitat for hawksbills; they determined that the seagrass pastures can support healthy, productive hawksbill aggregations.

Some individuals exhibit strong fidelity to certain foraging localities, but others will migrate extensively (National Marine Fisheries Service and U.S. Fish and Wildlife Service 2007a). Strong site-fidelity and close association with both natural and artificial coral reef structures was observed in a tagging and mark-recapture study carried out in Palm Beach County, Florida (Wood and Meylan 2008). A mark-recapture study undertaken at Mona and Monito Islands, Puerto Rico, demonstrated limited home-ranges for hawksbill turtles in that area; turtles maintained residency for at least several years (van Dam and Diez 1998). Flipper tag returns, satellite telemetry, and genetics were analyzed in Tortuguero, Costa Rica, to determine movements and habitat use (Troëng et al. 2005). Starbird (1993) used radio and ultra-sonic telemetry to track seven hawksbill turtles around Buck Island, St. Croix, Virgin Islands. Between nesting events in the season, most of the turtles maintained residency within 3 km of Buck Island, in water depths between 9 and 20 m. The mean dive duration was 56.2 minutes and mean surface time was 1.6 minutes. Three of the turtles left the St. Croix region immediately following their last nesting event. In another study at Buck Island, Storch et al. (2005) found an increase in dive duration with decreasing water temperatures in winter. Blumenthal et al. (2009c) deployed time depth recorders and ultrasonic tags on 21 hawksbills in the Cayman Islands. Pronounced patterns of diurnal activity and nocturnal resting were found. The mean diurnal dive depth (±SD) was 8±5 m, range was 2–20 m, mean nocturnal dive depth was 5±5 m, range 1-14 m, and maximum diurnal dive depth was 43±27 m, range 7-91 m. Maximum diurnal dive depth was significantly correlated with body mass, suggesting partitioning of vertical habitat by size. Direct in-water observations were made of hawksbills at D'Arros Island in the Seychelles (von Brandis et al. 2010). Mean dive depth and duration were found to be 8.2 m and 27.4 minutes, respectively. Surfacing intervals lasted on average 81.5 seconds and the mean number of breaths was 6.6 per surface interval. As in the Caymans, longer dives were correlated with larger body size.

Harewood and Horrocks (2008) examined impacts of coastal development on survival and swimming success for hatchling hawksbill turtles in Barbados. Predation rates were not significantly affected by offshore substrate type or beachfront lighting. However, a smaller percentage of the hatchlings leaving beaches that were lit with artificial lighting were able to swim the prescribed distance seawards during the observation period. It was postulated that artificial light may override the effects of wave cues in the low wave energy conditions characteristic of leeward Caribbean beaches, causing misorientation in the hatchlings. Kamel and Mrosovsky (2006), in examination of the thermal profile of a relatively pristine hawksbill nesting beach in Guadeloupe, French West Indies, found that temperatures in the forested areas were significantly cooler than temperatures in the more open, deforested areas. Because beach temperatures affect sex ratios of sea turtles, coastal forests are important male producing areas for the hawksbill sea turtle, and human alteration of this habitat could have important population consequences.

The only record of a hawksbill sighting, on NEFSC or SEFSC surveys or in the OBIS dataset, in the BOEM planning areas was in August of 1995 off of the Chesapeake Bay in the mid-Atlantic BOEM planning area.

2.37.5 Stock Structure

Bass et al. (1996) found that mtDNA lineages were highly structured among western Atlantic nesting colonies, with at least 6 female breeding stocks in the Caribbean and western Atlantic. This supported a natal homing model for recruitment of breeding females. Blumenthal et al. (2009a) performed mixed-stock analysis on samples collected from 92 neritic juvenile hawksbills in the Cayman Islands and found a diverse mixed stock with recruitment on a scale of 200-2500 km. A significant correlation was found between genetic profiles of foraging aggregations and oceanic drift models. Bowen et al. (2007a) compiled previously published and new mtDNA haplotype data for 10 nesting colonies in the western Atlantic and compared the profiles to those from feeding populations. Nesting colonies differed significantly in mtDNA haplotype, supporting theories of nesting site fidelity. Feeding aggregations showed lower but significant structure, indicating that foraging populations are not homogenous across the Caribbean Sea. The influence of proximity on recruitment to feeding areas was also demonstrated. LeRoux and Dutton (2008) revised population sub-structuring from earlier mixed stock analysis by amplifying mtDNA haplotypes from the St. Croix and Costa Rican nesting populations. Browne et al. (2008) performed an mtDNA d-loop analysis of the hawksbill at rookeries and foraging areas in Grenada and the southern Grenadine Islands, and found recruitment to foraging sites from various regional rookeries, some as far as Mexico.

2.37.6 Life History Traits

The nesting season of the hawksbill is six months long; the peak season in the Caribbean is between July and October. Nesting is principally nocturnal. Hawksbills nest an average of 4.5 times per season at intervals of approximately 14 days. In Florida and the U.S. Caribbean, clutch size is approximately 140 eggs, and several records exist of over 200 eggs per nest (U.S. Fish and Wildlife Service 1999). Carapace length growth data obtained from 197 hawksbill turtles which had been marked and recaptured from 1992 to 2000 at feeding grounds of Mona and Monito Islands, Puerto Rico (Diez and van Dam 2002), ranged from –0.59 to 9.08 cm/yr. Growth rates measured at the Breakers Reef, in Palm Beach, Florida have averaged 2.5 cm/year (Wood, n.d. http://www.floridahawksbills.com/Site/What_Weve_Learned.html). Sex ratios in this Palm Beach population showed a significant female bias (2.37:1) (Blanvillain et al. 2007). Age-to-maturity has been estimated at 20 or more years in the Caribbean (National Marine Fisheries Service and U.S. Fish and Wildlife Service 2007a). Hawksbill hatchlings have been shown to be less active than other species of marine turtles (Chung et al. 2009a, 2009b). Chung et al. (2009b) hypothesized that the smaller and slower hatchlings of this species avoid predators by hiding in flotsam.

2.37.7 Food Habits

Eggs of pelagic fish, *Sargassum*, and floating debris, such as tar droplets, styrofoam, and plastic, were identified in the stomach contents of posthatchling hawksbills in the pelagic environment (Meylan 1984b).

Juvenile and adult hawksbill turtles feed primarily on sponges. Leon and Bjorndal (2002) compared prey species ingested by hawksbill turtles with their availability at two sites in the Southwest Dominican Republic. Six benthic species were found in the hawksbill diet: 5 demosponges (*Chondrilla nucula*, *Geodia neptuni*, *Myriastra kalitetilla*, *Spirastrella coccinea*, and *Tethya crypta*) and one corallimorpharian (*Ricordea florida*). Diet choice was found to be based on a combination of selectivity for certain species and local abundance. The dominance of *R. florida* in the diet challenged the view that Caribbean hawksbills are strict spongivores (see, for example, Meylan 1988). Underwater photography taken at a Cayman Island foraging ground has shown that hawksbills forage occasionally on jellyfish, as well as on sponges (Blumenthal et al. 2009b). Observations have been made of hawksbill turtles feeding on the bubble coral *Physogyra lichtensteinii* in the Indian Ocean (Obura et al. 2010). However, the hawksbill's dependence on sponges as its principal prey, and its dependence on filter feeding, hard-bottom communities, make it vulnerable to deteriorating conditions on coral reefs (U.S. Fish and Wildlife Service 1999).

2.37.8 Health

2.37.8.1 Strandings

Hart et al. (2006) examined the spatial distribution of sea turtle strandings in North Carolina and concluded that strandings follow patterns that can be predicted from physical oceanography.

The Southeast Fisheries Science Center hosts a queryable dataset on Atlantic and Gulf of Mexico sea turtle strandings at http://www.sefsc.noaa.gov/seaturtleSTSSN.jsp. Figures 2.37-4 and 2.37-5 show density distributions of strandings along the Atlantic coast.

2.37.8.2 Contaminants

In Japan, hawksbills were shown to accumulate arsenic at rates higher than green turtles (Agusa et al. 2008).

2.37.8.3 Disease

Fibropapillomatosis is a disease characterized by internal and external tumors in sea turtles. While it has been reported in all species of sea turtles, it is infrequent in hawksbills and is not considered to be a major threat (National Marine Fisheries Service and U.S. Fish and Wildlife Service 2007a).

2.37.9 Acoustics

Little has been done specifically on hawksbill turtles and acoustics, though studies on loggerhead sea turtles have demonstrated sensitivity to low frequency noise (Bartol et al. 1999; Samuel et al. 2005). Sea turtle exposure to seismic air gun noise was studied by Weir (2007), though no conclusions on the impacts to the turtles by the airguns were able to be drawn in that study. Viada et al. (2008) discussed sea turtle response to underwater explosions and mitigation scenarios developed for explosive removal of oil and gas structures. The authors concluded that, while existing regulations provide more protection for marine turtles than had previously existed, little is known about sub-lethal effects of underwater explosions on turtles, especially potential auditory impacts (Viada et al. 2008).

2.37.10 Fisheries By-Catch and Entanglement

There are no records of fishery interactions with hawksbill turtles in the Northeast Fisheries Observer database. The Southeast Fisheries database has three records of observed interactions with hawksbill turtles in pelagic longline fisheries, one in 1992, one in 1997 and one in 1998 (see Figure 2.37-3).

2.37.11 Vessel Interactions

Damage from propeller and collision injuries from boats and ships is commonly found in sea turtles. In the stranding data collected by the Sea Turtle Stranding and Salvage Network from 1997 to 2001, 12.7% of all stranded turtles were documented as having sustained injuries consistent with propeller wounds or collision, although some of these injuries may have occurred post-mortem (National Marine Fisheries Service et al. 2010).

2.37.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on turtles is unclear. McMichael and Wyneken (2010) reported on a research program aimed at monitoring and minimizing the impacts from interactions between sea turtles and ocean energy technology within the Florida Current.

Hawksbills are occasionally captured (and released alive) in the cooling water intakes of industrial facilities, such as Florida Power and Light Company’s St. Lucie Power Plant on Hutchinson Island. Between March 1976 (when the St. Lucie Plant opened) and June 1992, nine hawksbills were captured (Ernest et al. 1989; National Marine Fisheries Service and U.S. Fish & Wildlife Service 1993).

Effects of oil and oil spill response guidelines are presented in a report edited by Shigenaka (2003). The report concluded that not only do oil collection areas overlap with sensitive sea turtle habitat, but there is also strong, if limited and dated, information that indicates oil is harmful to turtles, particularly to dermal tissues and membranes. In addition, fresh oil has been found to harm sea turtle eggs (Shigenaka 2003). The relationship between offshore oil platforms and the distribution of sea turtles are discussed in a report made by Lohoefener et al. at the 1989 Ternary Studies meeting sponsored by the MMS (Lohoefener et al. 1989). That study found that in some areas of the Gulf of Mexico, chelonid sea turtles were positively correlated with platforms, but in others there was either a negative correlation or no correlation at all. High amplitude simulated pile driving vibrations were found to decrease sea turtle hatching success (Ripcke 2011).

2.37.13 Data Gaps and Research Recommendations

Hays (2008) reviewed recent research on sea turtles, pointing out topical areas and some remaining questions. Heenehan (2010) summarized work on acoustic impacts to sea turtles. Information on growth, age-to-maturity, and annual reproductive output is scarce for many subpopulations of this species. The oceanic phase of post-hatchlings is poorly understood (National Marine Fisheries Service and U.S. Fish and Wildlife Service 2007a). Response of hawksbill populations to loss of habitat is an important issue, given global threats to coral reef ecosystems. Long-term trend data at foraging sites is lacking for hawksbill turtles (National Marine Fisheries Service and U.S. Fish and Wildlife Service 2007a). Filling the large gaps in available data for population assessments and developing a rigorous process for the assessments have been identified as high priorities by the National Research Council (Committee on Sea Turtle Population Assessment Methods; National Research Council 2010).

2.38 LOGGERHEAD TURTLE (*CARETTA CARETTA*)

2.38.1 Legal Status

The loggerhead sea turtle was listed as a threatened species throughout its range under the ESA in 1978. In 2010, NMFS and USFWS proposed to list 9 Distinct Population Segments (DPSs) of loggerhead sea turtles under the ESA (U.S. Fish and Wildlife Service and NOAA 2010). All commercial international trade in this species has been banned since it was listed on Appendix I by the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) in 1975. The IUCN lists the loggerhead as Endangered (IUCN 2010).

2.38.2 General Distribution

Loggerhead turtles are circumglobal, found in the Atlantic, Pacific, and Indian Oceans in temperate and tropical regions. They are the most abundant species of marine turtle found in U.S. waters; their range includes all BOEM Atlantic planning areas. Nesting occurs from Florida to North Carolina. The Peninsular Florida aggregation is the largest nesting aggregation in the Atlantic (Turtle Expert Working Group 2009) and is one of only two worldwide nesting aggregations that number more than 10,000 females nesting annually (Conant et al. 2009). The range of loggerhead sea turtles is shown in Figure 2.38-1 and nesting locations are shown in Figure 2.38-2. Tracks of satellite-tagged loggerheads are shown in Figure 2.38-3; sightings and fishery bycatch observations are shown in Figure 2.38-4.

2.38.3 General Abundance

According to the latest Turtle Expert Working Group assessment (2009), annual nest counts of all Western Atlantic subpopulations of loggerhead turtles have been decreasing during the past decade. Of special concern is the Peninsular Florida subpopulation, which showed a 49% decrease between 1998 and 2007, a decline which has been attributed to decreasing numbers of adult females in the population (Witherington et al. 2009). An updated analysis of Florida's long-term loggerhead sea turtle nesting data by the Florida Fish and Wildlife Institute (http://myfwc.com/research/wildlife/sea-turtles/nesting/loggerhead-trends/), shows that nest counts declined 16 percent between 1998 and 2011, but that the trend since 1989 may be stabilizing (trend was increasing between 1989 and 1998). Table 2.38-1 shows Atlantic Florida nest counts by county from 2005 through 2010.

Table 2.38.1 Annual Loggerhead Nest Counts in Atlantic Florida by County

County	2005	2006	2007	2008	2009	2010
Nassau	89	97	63	132	92	199
Duval	67	103	36	99	81	154
St. Johns	208	205	245	313	236	825
Flagler	212	220	219	352	226	458
Volusia	1,375	1,350	1,495	2,239	1,343	2,270
Brevard	19,339	18,089	14,829	21,242	17,194	25,741
Indian River	3,781	3,272	2,905	3,738	3,390	5,147
St. Lucie	4,073	3,204	3,589	4,523	3,936	5,459
Martin	5,822	5,532	5,210	7,356	6,643	9,120
Palm Beach	10,791	11,196	10,559	12,704	11,565	15,775
Broward	1,819	1,740	1,593	1,929	1,808	2,283
Miami-Dade	301	302	295	323	358	352
Monroe	77	66	85	73	199	254
Totals	47,954	45,376	41,123	55,023	47,071	68,037

(Adapted from the FWC Fish and Wildlife Research Institute Statewide Nesting Beach Survey Program, at http://research.myfwc.com/images/articles/2411/loggerhead_nesting_data__2005-2009.pdf and 2010 figures from http://myfwc.com/research/wildlife/sea-turtles/nesting/statewide/)

The 2008 Loggerhead Recovery Plan (2008) summarizes recent regional in-water population studies. This includes work by Morreale et al. (2005) on New York inshore waters; Mansfield (2006) on Chesapeake Bay, Virginia; Epperly et al. (2007) on Pamlico Sound, North Carolina; NMFS (2001) on South U.S. Atlantic – SEAMAP; Maier et al. (2004) on Southeast U.S. Atlantic; and Ehrhart et al. (2007) on Indian River Lagoon, Florida. The Turtle Expert Working Group Assessment (2009) also summarizes these regional in-water studies, concluding that though most of the southern areas show increasing population trends, the Chesapeake Bay and Long Island Sound areas show apparent declines. There was also a general increasing trend in median size of the neritic juveniles.

2.38.4 Habitat Preference

Loggerhead sea turtles use different habitats during different life stages. Terrestrial zones are use during oviposition and embryonic development. Nesting loggerheads along the U.S. Atlantic coast generally prefer high energy, relatively narrow, steeply sloped, coarse-grained ocean beaches (NMFS Office of Protected Resources n.d., http://www.nmfs.noaa.gov/pr/species/turtles/loggerhead.htm). Figure 2.38-2 shows the distribution of loggerhead nesting beaches in Florida.

Over 90% of U.S. Atlantic loggerhead nesting takes place in Florida. Annual nest totals from the Northern Recovery Unit, which includes Georgia, North and South Carolina, and Virginia, averaged 5,215 nests from 1989 to 2008, compared with a mean of 64,513 loggerhead nests per year in the Peninsular Florida Recovery Unit from 1989 to 2007 (National Marine Fisheries Service and U.S. Fish and Wildlife Service 2008).

Site fidelity ranging from 1.9 km to 109.1 km for all nests deposited by a female loggerhead within a season was measured by Tucker (2010). Bowen and Karl (2007) showed that nesting populations of loggerheads in south and northeast Florida, separated only by 50-100 km, are distinct in terms of mtDNA haplotype frequencies, indicating that loggerhead females are capable of relatively fine-scale homing.

Bell et al. (2007), in research on Cayman Islands loggerhead and green turtle nesting sites, observed no significant correlation between density of coastal development and clutch density, adult emergence success, or hatching success. However, they did find a significant relationship between density of coastal development and incidence of misorientation events in loggerhead hatchlings. Misorientation has been shown to occur in areas with high levels of artificial lighting (Sella et al. 2006; Witherington and Martin 1996). Whelan and Wyneken (2007) compared loggerhead hatchling survival rates at three locations in southern Florida. The predation rate observed during the first 15 minutes of the migration away from the beach for the three study sites combined was 4.6%. Though the predation rate was low during the brief swims observed, the authors concluded that predation levels may become higher when misoriented hatchlings fail to quickly get into deeper waters. Antworth et al. (2006) looked at nesting patterns and hatching success of loggerhead (and leatherback and green) turtles on Canaveral National Seashore. This site provided a rare opportunity to collect baseline data at an undisturbed/uninhabitated natural beach which could be compared to those at more disturbed localities. Brinn et al. (2010) analyzed compaction, bulk density, water content, color, and grain size on seven pairs of nourished and natural beaches along Florida panhandle with the aim of determining how such parameters affect sea turtle nesting success. Baseline data on temperatures of 26 loggerhead nesting beaches from South Carolina to Alabama were collected by Estes et al. (2010). Wright et al. (2010) looked at hatchling production rates before and after a beach construction and nourishment project in Broward County, Florida, and found no significant impacts on hatchling production.

After hatchling loggerheads leave the natal beach, they spend more than a decade in an oceanic phase before returning to nearshore neritic habitats. It is assumed that this oceanic phase is primarily spent drifting in association with *Sargassum* and with lines of floating material at areas of downwelling, but turtles are able to engage in directed swimming in order to feed opportunistically when they are not in areas with densities of floating material (Witherington 2002). They may also take part in extensive oriented swimming when currents are taking them onshore or into cold waters. At this oceanic stage turtles spend roughly 75% of their time in the upper 5 m of the water column but occasionally dive to depths greater than 200 m (Bolten 2003).

Work by McClellan and Read (2007) has shown that the shift from the oceanic phase to the neritic developmental phase is complex and reversible. One example of this is seen in a tagged juvenile loggerhead that, following a summer foraging in the Chesapeake Bay and a winter off Cape Hatteras, spent over three years in the north Atlantic gyre south of the Grand Banks (Mansfield et al. 2010).

Neritic juvenile loggerheads in the Northwest Atlantic inhabit continental shelf waters from Cape Cod Bay, Massachusetts, south through Florida, the Bahamas, Cuba, and the Gulf of Mexico. Important inshore habitat areas of the U.S. Atlantic coast include Long Island Sound, Chesapeake Bay, Pamlico and Core Sounds, Mosquito and Indian River Lagoons, Biscayne Bay, and Florida Bay (Conant et al. 2009). Coles and Musick (2000) looked at juvenile loggerhead turtle distribution along the North Carolina coastline from aerial survey data collected during 1991 and 1992 and correlated it with sea surface temperatures to arrive at a preferred temperature range of 13.3 to 28°C. Avens et al. (2003) reported on a survey of Cape Lookout Bight, North Carolina, where a seasonal aggregation of loggerheads and other turtles has been observed. On two survey days observers recorded 28 and 43 sightings of turtles per hour, many of them loggerheads. Mud observed on the backs of many of the turtles could have been from emergence after dormancy but also may have been evidence of benthic foraging. One loggerhead was satellite tagged during this survey, and after release it travelled north to approximately 30 km east of Virginia Beach where it remained through the summer. Bowen et al. (2004) offered evidence of natal homing in neritic loggerheads. Haplotype frequency differences were significantly correlated between coastal feeding populations and adjacent nesting populations, which demonstrates that, at the end of their oceanic phase, juvenile loggerheads are returning to feeding habitats that are near their natal rookeries. A mark-recapture study of juvenile loggerheads in Core Sound, North Carolina, showed that loggerheads exhibit fidelity to certain areas during summer months and are able to navigate back to those areas after being relocated or after long migrations (Avens et al. 2003). The Turtle Expert Working Group report (2009) presents an analysis of loggerhead distribution in U.S. waters using conventional tagging, satellite tagging, and sighting survey data. Areas of high use for satellite-tagged juvenile turtles were nearshore coastal waters between the North Carolina-South Carolina border into Chesapeake Bay and coastal waters of the mid-Atlantic Bight from spring through fall. In the winter, the turtles spent more time off Cape Hatteras. Observed seasonal sightings showed similar patterns to the satellite data, allowing the Group to conclude that: 1) the mid-Atlantic Bight is important seasonal habitat for juveniles and adult females from the Northern U.S. subpopulation; 2) the shelf waters of the eastern U.S. seaboard are important migratory habitat, particularly around Cape Hatteras, and 3) that the shelf off southwestern Florida is important habitat for female loggerheads (Turtle Expert Working Group 2009).

Hawkes et al. (2007a) combined satellite-tag data with remotely-sensed oceanographic data to summarize migratory strategies used by loggerheads. Twelve adult female turtles were tagged following nesting in North Carolina, and, though, most of them travelled north to forage at higher latitudes during summer before migrating south in the fall, others went south after nesting and didn’t make a fall migration. Both groups used warm waters at the edge of the Gulf Stream during winter, and made long resting dives. McCarthy et al. (2010) also combined satellite tag data with oceanographic variables, but analyzed the straightness index of the satellite tracks, to understand the relationship between the turtle movements and the oceanography. A negative relationship found between straightness and chlorophyll indicated that the turtle was spending more time searching and foraging and the positive relationship observed between straightness and ocean depth and SST meant that the turtle was travelling quicker through deeper or warmer areas.

Stable isotope confirmed that benthic invertebrates dominated the recent diet of neritic turtles and pelagic prey was dominant for oceanic animals (McClellan et al. 2010). However, when temporally-protracted diet composition was analyzed, results indicated that all turtles had fed in the pelagic zone during overwintering periods. Reich et al. (2009) used stable isotope analysis and examination of the epibionts, or community of organisms, using the turtle's carapace as substrate, to further demonstrate the bimodal, or even polymodal, foraging strategies of adult female loggerheads, where some turtles return to oceanic habitats and some remain in neritic areas. Preliminary analysis of male loggerheads using these same techniques (Reich et al. 2010) does not show the bimodal or polymodal pattern.

Sharks are a primary marine predator of adult and juvenile sea turtles. Heithaus et al. (2002) examined frequencies of shark-inflicted damage to green turtle and loggerhead turtles in Shark Bay, Western Australia. Green turtles were found in tiger shark habitats more frequently than loggerheads, yet exhibited less evidence of shark-inflicted injuries, with no sex difference in injury frequency.

2.38.5 Stock Structure

Five subpopulations, or recovery units, were identified for the Western North Atlantic in the 2008 Loggerhead Recovery Plan (National Marine Fisheries Service and U.S. Fish and Wildlife Service 2008). The separations are based on a combination of geographic distribution of nesting densities, spatial separation, and genetic differences. The first two of the subpopulations fall within the BOEM Mid-Atlantic, South Atlantic and Straits of Florida Planning Areas. The five subpopulations (units) are described below.

1. **Northern Recovery Unit:** Loggerheads originating from nesting beaches from the Florida-Georgia border through southern Virginia (the northern extent of the nesting range).

2. **Peninsular Florida Recovery Unit:** Loggerheads originating from nesting beaches at the Florida-Georgia border through Pinellas County on the west coast of Florida, excluding the islands west of Key West, Florida.

3. **Dry Tortugas Recovery Unit:** Loggerheads originating from nesting beaches throughout the islands located west of Key West, Florida, because these islands are geographically separated from other recovery units.

4. **Northern Gulf of Mexico Recovery Unit:** Loggerheads originating from nesting beaches from Franklin County on the northwest Gulf coast of Florida through Texas (the western extent of U.S. nesting range).

5. **Greater Caribbean Recovery Unit:** Loggerheads originating from all other nesting assemblages within the Greater Caribbean (Mexico through French Guiana, the Bahamas, Lesser Antilles, and Greater Antilles).

Genetic work done by Shamblin et al. (2010) suggests that the peninsular Florida subpopulation should, in fact, be divided into central and southern subpopulations. They saw a demographic break in haplotype frequencies in the vicinity of the Canaveral National Seashore.

Bowen et al. (2005) surveyed maternally inherited mtDNA in loggerhead turtles and found that the level of population structure increased as the turtles progressed through life history stages. Pelagic post-hatchlings showed no population structure across the northern Atlantic, subadults in coastal habitat showed low structure, and nesting colonies along the southeastern coast of the U.S. showed strong structure. However, a survey of biparentally inherited microsatellite DNA showed no significant population structure across the same nesting colonies (Bowen et al. 2005).

Bass et al. (2004) performed a mixed-stock analysis of juvenile loggerheads on the feeding ground at Pamlico-Albemarle Estuarine Complex, North Carolina, to determine the rookery origins of that feeding cohort. They determined that 80% of the turtles originated from the South Florida population, 12% were from the northeast Florida to North Carolina nesting population, 6% were from the Yucatan, Mexico, and 2% were from other rookeries. Juvenile loggerhead sea turtles in pelagic habitats of the Azores and Madeira were shown to be derived from nesting populations in the western Atlantic by Bolten et al. (1998). Maximum likelihood analyses were used to estimate that 100% of the turtles were from the nesting populations in the southeastern United States and the Yucatan Peninsula, Mexico. Roberts et al. (2005) performed a mixed stock analysis on seasonal juvenile feeding grounds from North Carolina to northern Florida to determine the rookery origins of that feeding population and found that the majority originated at nearby rookeries.

2.38.6 Life History Traits

In their 2009 assessment, the Turtle Expert Working Group recognized 5 life stages of loggerhead turtles (Turtle Expert Working Group 2009). Table 2.38-2 shows the life stages and their definitions.

Table 2.38.2 Life Stages of the Loggerhead Turtle

Life Stage	Habitat	Straight Carapace Length (SCL)
Year One	terrestrial to oceanic	≤ 15 cm
Juvenile (1)	exclusively oceanic	15-63 cm
Juvenile (2)	oceanic or neritic	41-82 cm
Juvenile (3)	oceanic or neritic	63-100 cm
Adult	neritic or oceanic	≥ 82 cm

A summary of recently published information on loggerhead life stages, presenting data from Stewart and Wyneken (2004), Whelan and Wyneken (2007), Bjorndal et al. (2000b, 2003a, 2001, 2003b), Snover (2002), Sasso et al. (2006), Braun-McNeill (2008, 2007a, 2007b), Hedges (2007), Scott (2006), Byrd (2005), and Dahlen et al. (2000) can be found in Table 2 of the TEWG (2009 p. 7).

Age at sexual maturity for loggerhead turtles is estimated to be about 35 years. Mating occurs in late March to early June in the southeastern U.S. and females lay eggs between late April and early September. Females lay between three and five nests (occasionally more) during a single nesting season. The eggs incubate for approximately two months and hatchlings emerge between late June and mid-November (NMFS Office of Protected Resources n.d., http://www.nmfs.noaa.gov/pr/species/turtles/loggerhead.htm).

The timing of loggerhead nesting on the Florida coast was correlated with sea surface temperatures by Weishampel et al. (2004). The study found that between 1989 and 2003 the median nesting date became earlier by approximately 10 days and sea surface temperatures warmed an average of 0.8 °C. Similarly, Bowers (2010) looked at first nesting dates from ninety beaches in the Southeast U.S. over a 30-year period. Loggerhead sea turtles arrived 0.2 days earlier every year during this period, 1.4 days earlier for every point increase in the North Atlantic Oscillation index, and 3.6 days later for every degree increase in latitude; this suggests that the turtles are responding to climate variability and long-term trends. Hawkes et al. (2007b) investigated the potential impacts of climate change on sex ratios of loggerheads, because, as in other sea turtles, the sex determination in this species is temperature dependent. Recorded sand temperatures and historical air temperatures at Bald Head Island, North Carolina, were used to examine past and predict future sex ratios under scenarios of warming. There were no significant trends in primary sex ratio evident in recent years and estimated mean annual sex ratio was 58% female. Earlier nesting and longer nesting seasons were correlated with warmer sea surface temperature. Populations of turtles in more southern parts of the U.S. are currently highly female biased and are likely to become even more so with as little as 1 °C of warming and to experience extreme levels of mortality if warming exceeds 3 °C (Hawkes et al. 2007b).

2.38.7 Food Habits

A listing of known loggerhead food items was compiled by Dodd (1988). The diet is predominatly carnivorous, consisting mostly benthic invertebrates and coelenterates.

Diet analysis of bycaught oceanic loggerhead turtles in the central North Pacific was carried out by Parker et al. (2005). That study demonstrated that the turtles fed mostly on surface prey species such as *Janthina* spp. (*Gastropoda*); *Carinaria cithara* (*Heteropoda*); a chondrophore, *Velella velella* (*Hydrodia*); *Lepas* spp. (*Cirripedia*); *Planes* spp. (*Decapoda: Grapsidae*); and *pyrosomas* (*Pyrosoma* spp.).

Digestive tract contents of neonate loggerhead turtles associated with downwelling areas of the slope water near the Gulf Stream front off east-central Florida were examined in 1997 by Witherington (2002). Hydroids, copepods, and pleuston, such as *Janthina*, *Creseis*, *Porpita*, and *Halobates*, were the predominant animal prey species. Ingested plants were largely *Sargassum* fragments. In addition, 20% of the turtles sampled had ingested tar and 15% had ingested plastic. Ingestion of anthropogenic debris by loggerheads is common and can have serious lethal and sublethal effects (Bjorndal et al. 1994; McCauley and Bjorndal 1999). Boyle and Limpus (2008) examined stomach contents of post-hatchling loggerhead and green turtles in the southwest Pacific and found that the turtles at that stage were opportunistic feeders, consuming zooplankton and other floating prey, and that small oceanic green and loggerhead turtles had very similar diets (Boyle and Limpus 2008). Frick et al. (2009) examined stomach contents and fecal samples in an analysis of diet composition of oceanic-stage loggerhead turtles from the North Atlantic Ocean. They found that loggerheads from the Azores are opportunistic carnivores that feed upon a variety of oceanic and pelagic organisms, as well as novel food resources, such as non-indigenous species.

Seney and Musick (2008, 2008) examined digestive tract contents from 297 loggerhead turtles in Virginia during 1983-2002. Analyses showed a changing diet, from predominantly Atlantic horseshoe crab (*Limulus polyphemus*) during the early to mid-1980s, to predominantly common blue crab (*Callinectes sapidus*) during the late 1980s and early 1990s, and then to predominantly finfish, particularly Atlantic menhaden (*Brevoortia tyrannus*) and Atlantic croaker (*Micropogonias undulatus*), in the mid-1990s and in 2000–2002. The authors postulated that crab population declines have led the turtles to forage in nets or on discarded fishery bycatch. Immature loggerheads in the Core Sound, North Carolina, area did not appear to feed on fish or fish discards, as evidenced by stable isotope analysis (Wallace et al. 2009). However, the turtles of Core Sound did show dietary preferences for species, such as blue crab and whelk (*Busycon* spp.), that are valuable or are commonly taken in commercial fisheries, which suggests possible competitive interactions. Fecal sample analysis of juvenile loggerheads and Kemp's ridley sea turtles in Long Island Sound showed a predominance of nine-spined spider crabs (*Libinia emarginata*) (Burke et al. 1993).

Gut contents and feces of 95 turtles caught in both neritic and pelagic areas of the central Mediterranean Sea were examined by Casale et al. (2008). Findings suggested highly opportunistic foraging; benthic organisms found in small and larger turtles challenge assumptions of strict habitat segrations by life stage. The authors proposed an alternative or "relaxed" model, in which temporary or permanent fidelity to specific oceanic or neritic zones might vary among individuals or populations according to food availability and oceanographic features in the foraging or migratory areas.

2.38.8 Health

2.38.8.1 Strandings

Hart et al. (2006) examined the spatial distribution of sea turtle strandings in North Carolina and concluded that strandings follow patterns that can be predicted from physical oceanography.

Mazzarella (2007) compared haplotype frequencies of stranded loggerheads in North and South Carolina and compared them with those from loggerheads captured in nearby waters. The stranded animals were not significantly different; this suggests that strandings are representative of the nearshore aggregations.

The Southeast Fisheries Science Center hosts a queryable dataset on Atlantic and Gulf of Mexico sea turtle strandings at http://www.sefsc.noaa.gov/seaturtleSTSSN.jsp. Figures 2.38-5 and 2.38-6 show density distributions of strandings along the Atlantic coast.

2.38.8.2 Contaminants

Keller et al. (2004a) collected blood samples from juvenile loggerhead sea turtles captured in Core Sound, North Carolina, and analyzed them for 55 polychlorinated biphenyl (PCB) congeners and 24 OC pesticides by gas chromatography with electron capture detection and mass spectrometry. Concentrations were found to be similar to previously reported levels in blood components of humans and reptiles from relatively clean sites, but lower than those measured in blood of fish-eating birds and marine mammals. Carlson et al. (2010) analyzed blood samples of five juvenile loggerheads captured in Core Sound, North Carolina, and coastal South Carolina for PBDE, PCB, and OCP levels. No significant temporal trends were found. Possible health effects of OC contaminants on loggerheads were investigated (Keller et al. 2004b). Anemia, modulation of the immune system, and possible hepatocellular damage were correlated with OC levels, showing that contamination may be affecting the health of loggerhead sea turtles. Comparisons with in-vitro studies also suggested that OC exposure modulates immunity in loggerheads (Keller et al. 2006b). Keller et al. (2008, 2005) also measured perfluorinated compounds (PFCs) in loggerhead turtles, and found that total PFC concentrations, though not influenced by sex, were higher in turtles captured from inshore waters of North Carolina than in turtles from waters offshore from the South Carolina−Florida coast. POPs measured in loggerhead eggs from three areas of the U.S.Atlantic coast showed that levels were significantly higher in the northern nests, where elevations in embryonic abnormalities were also observed (Alava et al. 2008). Metal concentrations have also been evaluated in loggerhead samples from the U.S. Atlantic and Gulf coasts (Day 2003; Day et al. 2005, 2008, 2007; White et al. 2008), with some evidence that elevation in blood mercury levels is correlated with proximity to major industrial river mouths (Day et al. 2005), and that most metal concentrations were higher in eggs collected on Florida's Gulf coast than on its Atlantic coast (White et al. 2008).

Blood samples taken from stranded, nesting, and foraging loggerhead turtles along the coast of Georgia were analyzed using physical examinations, hematology, plasmabiochemistry, plasma protein electrophoresis, and toxicologic parameters (Deem et al. 2009). Significant differences in many blood parameters were found between stranded animals, nesting females, and turtles captured by trawl.

Sea turtles are adversely impacted by brevetoxin exposure, which is caused by blooms of the toxic dinoflagelate *Karenia brevis*. Significant brevetoxin effects on loggerhead turtle immune function were demonstrated by Walsh et al. (2010) in rescued loggerhead sea turtles, and by in vitro experiments using peripheral blood leukocytes (PBL) collected from captive sea turtles.

2.38.8.3 Disease

A high rate of infection by spirorchiid trematodes was found in Florida stranded green and loggerhead turtles, and in some cases caused or contributed to death (Stacy et al. 2010). McGarrity (2008) examined stress protein expression in both healthy and fibropapilloma afflicted green and loggerhead turtles from Indian River Lagoon and compared levels to those from turtles from more pristine offshore reef environments.

2.38.9 Acoustics

Studies on loggerhead sea turtles have demonstrated sensitivity to low frequency noise (Bartol et al. 1999; Samuel et al. 2005). Underwater sound recordings from the Peconic Bay Estuary system in Long Island, New York, demonstrated that high levels of anthropogenic noise at frequencies in the range of the turtles' highest sensitivity also overlapped spatially and temporally with high turtle use of the habitat (Samuel et al. 2005). Sea turtle exposure to seismic air gun noise was studied by Weir (2007), though no conclusions on the impacts to the turtles by the airguns were able to be drawn in that study. Viada et al. (2008) discussed sea turtle response to underwater explosions and mitigation scenarios developed for explosive removal of oil and gas structures. The authors concluded that, though existing regulations provide more protection for marine turtles than had previously existed, little is known about sub-lethal effects of underwater explosions on turtles, especially potential auditory impacts (Viada et al. 2008).

2.38.10 Fisheries By-Catch and Entanglement

Wallace et al. (2010) compiled a database of reported marine turtle bycatch in gillnet, longline, and trawl fisheries worldwide from 1990 to 2008. The total reported global marine turtle bycatch was approximately 85,000 turtles, though the authors say this is likely an underestimate by at least two orders of magnitude, and they do not say what percentage of these are loggerheads. In the U.S., shrimp trawling is the fishery with the highest impacts to loggerhead turtles (National Marine Fisheries Service and U.S. Fish and Wildlife Service 2008). Turtle Excluder Devices (TEDs) are net modifications designed to allow turtles to escape from shrimp nets. Since December 1994, TED use has been mandated in shrimp trawls. Bycatch of loggerhead sea turtles has been documented in several other Atlantic fisheries, including mid-Atlantic gillnet (Murray 2009a, 2009b), southeast shark gillnet (Garrison 2007a), northeast bottom trawl (Murray 2006, 2007, 2008a), mid-Atlantic sea scallop dredge (Haas et al. 2008; Murray 2004a, 2004b, 2005), and pelagic longline (Gilman et al. 2006; Lewison et al. 2004; Lewison and Crowder 2007; Watson et al. 2005). Loggerhead bycatch in the Canadian pelagic longline fishery has been estimated to be substantial and of similar magnitude to that of the U.S. fleet in the North Atlantic (Brazner and McMillan 2008). Gardner et al. (2008) modeled loggerhead bycatch in the pelagic longline fishery and found that water temperature, region, bottom depth, and target species are all significant predictors of the number of loggerhead sea turtles captured.

Merrick and Haas (2008) discussed impacts to loggerhead populations by scallop dredge and trawl fisheries and concluded that, though impacts are detectable, they do not significantly change the calculated risk of extinction of the population of adult female Western North Atlantic loggerheads over the next 100 years. Sasso and Epperly (2007) reported on a study of satellite-tagged loggerheads, some of which had been lightly hooked in the longline fishery and some had been dip-netted. No difference in survival rates was detected between turtles which had been caught by the longline gear and those in the control group which had not.

Mitigation measures to reduce fishery impacts on loggerhead turtles have been discussed by several authors. Gilman et al. (2010) looked at mitigation measures employed in small-scale, coastal, passive net fisheries which, the authors argue, may be the largest single threat to some sea turtle populations. Lewison et al. (2003) looked at the impact of TEDs on loggerheads and found that low compliance was correlated with increased rates of loggerhead strandings. Lawson et al. (2007) reported on field research that had been conducted on effects that several TED configurations had on the catch of the target species in the mid-Atlantic, inshore, summer flounder trawl fishery. Smolowitz et al. (2010) reported on the development of a modified scallop dredge which is designed to reduce the probability of a turtle going under the dredge frame. Watson et al. (2005) examined changes in bait type and hook design in the longline fishery to reduce turtle mortality and found that the use of circle hooks and mackerel bait was effective. The use of circle hooks in the pelagic longline fishery was also evaluated by Read (2007), who cautioned that their use may not reduce turtle mortality in every longline fishery and that more field testing is required. Haas (2010) used four conservation metrics (magnitude of adult-equivalent bycatch mortalities, magnitude of bycatch mortalities, magnitude of bycatch, and percent of encountered turtles captured) to evaluate various TED configurations. The most informative metric was adult-equivalent bycatch mortality. A TED with a large escape opening had the highest estimated conservation value. Up to 66 more adult-equivalent loggerheads were estimated to be protected by the large opening than by the standard opening. Extending TED use north of 37°N could protect a similar number. This builds on a paper by Wallace et al. (2008) which discusses the use of reproductive value indices to evaluate fishery impacts on a population, with focus on loggerhead turtle interactions with trawl gear.

2.38.11 Vessel Interactions

Damage from propeller and collision injuries from boats and ships is commonly found in sea turtles. In the stranding data collected by the Sea Turtle Stranding and Salvage Network from 1997 to 2001, 12.7% of all stranded turtles were documented as having sustained injuries consistent with propeller wounds or collision, although some of these injuries may have occurred post-mortem (National Marine Fisheries Service et al. 2010). The correlation between internesting loggerhead and green turtle dive behavior at Casey Key, Florida, and boat traffic in the area was assessed by Sobin and Tucker (2010). Work et al. (2010) ran trials of different vessel propulsion systems and operational characteristics and their impacts of sea turtles and found that vessel speed did significantly influence the likelihood of catastrophic damage to turtles, whereas depth in the water column did not. They found that propeller guards were ineffective at planing speed and only slightly helpful at idle speed. Jet propulsion systems conferred dramatic improvements in animal safety compared to traditional outboards.

Hopper dredge operations have also been identified as a cause of loggerhead mortality (National Marine Fisheries Service and U.S. Fish and Wildlife Service 2008). Relocation of turtles prior to dredging as well as seasonal restrictions on dredge activities have been employed to mitigate turtle interactions.

2.38.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways, including habitat alteration and the introduction of noise via seismic exploration activities (e.g., airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on turtles is unclear. McMichael and Wyneken (2010) reported on a research program aimed at monitoring and minimizing the impacts from interactions between sea turtles and ocean energy technology within the Florida Current.

Entrainment in nuclear power plant intake canals has been a source of loggerhead mortality along the U.S. Atlantic coast. In the St. Lucie Nuclear Power plant in St. Lucie, Florida 6,482 loggerhead have been captured and released between 1976 and 2004.

Effects of oil and oil spill response guidelines are presented in a report edited by Shigenaka (2003). The report concluded that not only do oil collection areas overlap with sensitive sea turtle habitat, but there is also strong, if limited and dated, information that indicates oil is harmful to turtles, particularly to dermal tissues and membranes. In addition, fresh oil has been found to harm sea turtle eggs. The relationship between offshore oil platforms and the distribution of sea turtles is discussed in a report made by Lohoefener et al. at the 1989 Ternary Studies meeting sponsored by the MMS (Lohoefener et al. 1989). That study found that in some areas of the Gulf of Mexico, chelonid sea turtles were positively correlated with platforms, while in others there was either a negative correlation or no correlation at all. The explosive removal of offshore petroleum platforms is known to have negative impacts on sea turtles (Klima et al. 1988). Five of eight loggerheads and Kemp's ridley turtles experimentally subjected to blasts off Texas in 1986 were rendered unconscious (Klima et al. 1988). In addition, increases in turtle stranding mortalities were correlated with periods of high numbers of offshore explosions (see also Acoustics 3.38.9). High amplitude simulated pile driving vibrations were found to decrease sea turtle hatching success (Ripcke 2011).

Under a plan developed by the U.S. Fish and Wildlife Servece, NMFS, and the Florida Fish and Wildlife Conservation Commission, Kemp's Ridley, loggerhead, and green sea turtle nests from areas of the Florida panhandle and Alabama affected by the Deepwater Horizon oil spill were relocated to the Atlantic coast of Florida (for more information, see http://www.fws.gov/home/dhoilspill/pdfs/TurtleNestHatchProgram.pdf**).**

2.38.13 Data Gaps and Research Recommendations

Hays (2008), who reviewed recent research on sea turtles, pointed out topical areas and some remaining questions. Heenehan (2010) summarized work on acoustic impacts to sea turtles. Detailed research recommendations were presented in the TEWG report (Turtle Expert Working Group 2009). That document called for better estimates of population parameters, improved understanding of spatial and temporal distribution, better definition of population structure, additional research on the effects of fishery bycatch on survival rate and population growth, and increased information on trophic changes. Post-hatchlings and neonates, juveniles in Florida, males, and oceanic loggerheads were all identified as needing additional tracking data in order to fully elucidate the life history and spatial representation of loggerhead turtles. Currently, data from the Atlantic Marine Assessment Program for Protected Species (AMAPPS) loggerhead turtle satellite tracking program are being generated and analyzed. The National Research Council's *Assessment of Sea-turtle Status and Trends* (Committee on Sea Turtle Population Assessment Methods; National Research Council 2010) recommended that NMFS and the USFWS develop a coherent national strategy for sea-turtle assessments in order to improve collection, quality, and availability of data and also to develop a plan of external review of data and models used for population status estimation and trend analysis. The Florida Fish and Wildlife Conservation Commission (Eaton et al. 2008) has assembled an overview of in-water research projects on sea turtles in Florida and identified geographical gaps and subjects that could benefit from further collaboration and standardization.

2.39 LEATHERBACK TURTLE (*DERMOCHELYS CORIACEA*)

2.39.1 Legal Status

The leatherback sea turtle was listed as an endangered species under the ESA in 1970. In 1998, NMFS designated critical habitat for leatherback turtles; it includes the coastal waters adjacent to Sandy Point, St. Croix, U.S. Virgin Islands. In 2009, NMFS proposed to revise the critical habitat to include areas off of the U.S. west coast. All commercial international trade in this species has been banned since it was listed on Appendix I by the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) in 1975. The IUCN lists the leatherback as “Critically Endangered.”

2.39.2 General Distribution

The leatherback turtle is found worldwide in tropical to sub-polar oceans (NMFS Office of Protected Resources n.d.) Leatherbacks with a curved carapace length smaller than 100 cm appear to be limited to regions warmer than 26°C (Eckert 2002) but adults have a broad thermal tolerance and forage from 71°N to 47°S (National Marine Fisheries Service 2001). Though juveniles have been reported near the coastlines in some regions, the species is considered primarily pelagic. Occurrence of leatherbacks is documented in all BOEM planning areas in summer and fall, and primarily south of Cape Hatteras, in the mid-Atlantic and South Atlantic planning areas in winter and spring. The range of leatherback sea turtles is shown in figure 2.39-1. Tracks of satellite-tagged leatherbacks are shown in Figure 2.39-2 and sightings and fishery bycatch observations are shown in Figure 2.39-3.

Leatherback turtle nesting grounds are located on tropical (and rarely subtropical) beaches around the world. The largest remaining nesting colonies are located on the coasts of northern South America and West Africa (NMFS Office of Protected Resources n.d.,. In the Atlantic U.S., leatherback nesting is concentrated in southeast Florida (Turtle Expert Working Group 2007). Nesting in Florida was first documented in 1947 and is now a regular seasonal occurrence. Nesting is rarer north of Florida. Georgia reported seven leatherback nests in 2009 and four in 2010 (see http://www.seaturtle.org/nestdb/index.shtml?view=2&year=2010). South Carolina reported one leatherback nest in 2006 and three each year in 2008-2010. At least ten leatherback nests were reported in North Carolina in 2007 (Stewart 2007). One nest was reported in North Carolina in 2009 and one in 2010 (data from North Carolina Wildlife Resources Commission, available at http://www.seaturtle.org/nestdb/index.shtml?view=1&year=2010).

2.39.3 General Abundance

Although leatherback nesting in Florida is increasing, worldwide the population is in decline. Collapses have been recorded in some previously important sites (Sarti Martinez 2000). A global estimate of 115,000 adult females calculated in 1982 (Pritchard 1982) fell to 20,000-30,000 in 1996 (Spotila et al. 1996).

The Turtle Expert Working Group (2007) assessed abundance of leatherbacks in several Atlantic populations. Using estimates of the number of nests in 2004-2005, nests per female, remigration interval, and sex ratio, the TEWG point estimate for adult turtles in the Florida stock was 580 (CV=0.33) animals. The Northern Caribbean stock was estimated at 2,700, the Western Caribbean stock at 8,100, the Southern Caribbean stock at 20,000 and the West African stock at 28,000. Trends for six management units (Florida, Northern Caribbean, Western Caribbean, Southern Caribbean/Guianas, South Africa and Brazil) of leatherback turtles were also estimated (Turtle Expert Working Group 2007). All stocks but one (Western Caribbean) appeared to be increasing over the time period examined, although the authors cautioned that increases could be an artifact of increases in monitoring.

In Florida, Stewart (2007) estimated leatherback nesting increases at 11.3% per year between 1979 and 2005. Contributing factors postulated for this increase were improved monitoring and protection, and changes in ocean climate and food availability. Table 2.31-2 shows Florida nest counts by county from 2005 through 2010. Figure 2.41-3 shows the nesting count trend between 1988 and 2010.

Table 2.39.1 Annual Leatherback Nest Counts in Florida by County

County	2005	2006	2007	2008	2009	2010
Nassau	0	0	0	0	0	1
Duval	0	0	2	1	5	2
St. Johns	5	0	5	1	20	4
Flagler	8	1	1	0	8	0
Volusia	8	2	17	3	19	15
Brevard	67	16	105	33	70	77
Indian River	50	16	75	27	61	87
St. Lucie	94	57	197	116	235	203
Martin	230	205	494	274	663	561
Palm Beach	284	225	490	243	615	368
Broward	25	15	41	14	45	14
Miami-Dade	9	3	8	10	5	2
Monroe	0	0	0	0	0	0
Totals	780	540	1,435	722	1,746	1,334

(Adapted from the FWC Fish and Wildlife Research Institute Statewide Nesting Beach Survey Program at http://research.myfwc.com/images/articles/2479/leatherback_nesting_data__2005-2009.pdf with 2010 data from http://myfwc.com/research/wildlife/sea-turtles/nesting/statewide/)

Shoop and Kenney's (1992) effort-corrected analysis of over three years of aerial and shipboard surveys conducted in northeastern U.S. continental shelf waters showed scattered areas of high relative abundance of leatherbacks. Clusters of animals were found south of Long Island and in the eastern Gulf of Maine. The leatherback population of the total study area (Cape Hatteras, North Carolina, to Nova Scotia, Canada, from the coast out to 5 nm past the 1,000 fathom iosbath) was estimated at 100-900 individuals.

Murphy et al. (2006) discussed the abundance and distribution of leatherbacks in South Carolina waters. While strandings and sightings of this species were rare prior to 1989, increases have been observed in more recent years. Between 1994 and 2003, during April-June, 1131 live leatherbacks (0.04 per km) were observed during 50 nearshore aerial surveys flown parallel to the South Carolina coast. The highest concentration during a single flight was in May 2002, when 175 leatherbacks were seen over 605 km of transect line or 0.29 per km. Numbers observed varied significantly between inner and outer transect lines, among years, and among flights within a year.

2.39.4 Habitat Preference

Leatherback turtles use the terrestrial zone for oviposition and embryonic development and may use shallower waters to feed and reproduce, especially during the nesting season (Georges et al. 2007). They are otherwise a pelagic species, inhabiting the open ocean from hatchling through adulthood (National Marine Fisheries Service 2001). Leatherback nesting habitat consists primarily of high energy beaches with either a deep water oceanic or shallow water mud bank approach (Turtle Expert Working Group 2007). The spatial patterns of leatherback nest distributions along the Florida coastline were quantified by Weishampel et al. (2003), who found that, unlike those of loggerhead and green turtles, leatherback nest distributions were not significantly different from random.

Data from tagged male leatherbacks has suggested that mating occurs in waters adjacent to nesting beaches (James et al. 2005a). Males were found to migrate to and stay in these areas for up to 96 days. Return migrations indicated breeding site fidelity.

Female leatherbacks are known to embark on trans-oceanic migrations after nesting (National Marine Fisheries Service 2001). Tag recovery studies (e.g., Pritchard 1976) and long-distance tracking studies such as those by Duron-Dufrenne (1987), Hughes et al. (1998), and Moreale et al. (1996) have demonstrated migrations of more than 7,000 km.

Fossett et al. (2010) analyzed long-term tracking data from 21 leatherback turtles to assess spatio-temporal foraging patterns during their migrations in the Northern Atlantic. Fifteen were females tagged at nesting beaches in Grenada, French Guiana, and Suriname, and the remaining six were captured and tagged off Ireland and Nova Scotia. Current-corrected movement patterns and diving behavior were analyzed to identify areas of high and low foraging success. High foraging success, when turtles travelled more slowly and performed shallower dives, occurred in a sub-equatorial zone spanning the width of the Atlantic and at high (>30°N) latitudes. Turtles transited quickly between these zones in the center of the North Atlantic Gyre. Ferraroli et al. (2004) also reconstructed movements of satellite-tagged leatherbacks and found that, unlike in the Pacific where leatherbacks follow narrow migration corridors, in the Atlantic the turtles dispersed widely. They also found that turtles travelled more slowly along productive oceanic front areas where gelatinous plankton are concentrated; this suggests that feeding was occurring in these areas.

State-space modeling was performed on satellite tracking data from Pacific leatherbacks by Bailey et al. (2008). This study provided estimates of behavioral modes during movement and enabled the internesting period to be described based on a shift between transiting and foraging modes. They found that the turtles did not immediately move offshore, indicating that the length of time and area that turtles utilize during internesting could previously have been underestimated. Internesting movements were also studied by Byrne et al. (2009). Two leatherback turtles were tracked through three consecutive internesting intervals in the Commonwealth of Dominica, West Indies. Near-shore residence, as well as extensive inter-nesting movements, was recorded, with movements away from the nesting beach increasing towards the end of the nesting season.

Eckert et al. (2006) satellite-tracked ten leatherback turtles from two Florida Atlantic Coast nesting beaches. Movement and foraging areas of these turtles, like the leatherbacks tracked in 2000 from the Archie Carr National Wildlife Refuge of Eastern Florida (Bagley et al. 2001), was more coastal than previous leatherback tracking studies had demonstrated. The primary internesting habitat was centered east-southeast of Cape Canaveral, Florida, from 2 to 60 km offshore and extending 215 km along the coast. Atlantic foraging areas were located primarily on the continental shelf from 30° to 50°N between March and November, and in an offshore area centered at 42°N, 65°W from December through February, as well as off Africa in the Mauritania upwelling.

The movements and behavior of nine female leatherbacks tagged at their nesting beaches on the Caribbean island of Trinidad were monitored for up to 370 days (Eckert 2006). The three longest tracked turtles moved from Trinidad to foraging areas at higher latitudes—Northeast of the Flemish Cap, along the continental shelf of the Iberian peninsula to the Bay of Biscay, and along the North Atlantic subtropical front—where they remained until the end of November; none of them transiting through U.S. Atlantic waters. Between mid-October and mid-November, the turtles left high latitudes for a presumed foraging area in the Mauritania upwelling, where they stayed until their tags stopped transmitting. Diving depths were generally less than 51 m during the internesting period. As the turtles migrated out of the Caribbean, dives initially were to depths of 100-300 meters and relatively long (>26 mins) but became shallower and shorter at high latitudes.

Sherrill-Mix et al. (2008) used telemetry data obtained from 27 leatherback turtles tagged off Nova Scotia and Cape Breton Island and satellite-derived environmental variables to investigate the role of environmental correlates, latitudinal and longitudinal position, and turtle body size and sex on the timing of southward migration. They found that the turtles' departure rates did not increase with decreasing temperatures; in fact, they were more likely to depart from warmer and more chlorophyll-rich waters. Departure rates increased with latitude. A diel pattern in travel rates of leatherbacks was found by Jonsen et al. (2006) to change over different phases of their migratory cycle. While foraging in northern waters, the turtles had similar travel rates during day and night, but on their southward migration to tropical waters, travel rates were faster during the day. Changes in diel patterns of diving activity were similar, with less variation when on the northern feeding grounds, but longer, deeper diving occurring during the night on the southward migration (James et al. 2006). State-space modeling was also applied to the Nova Scotia tagged leatherback data by Jonsen et al. (2006) to show that transiting leatherbacks make longer, deeper dives than foraging ones.

Witt et al. (2007) identified probable foraging grounds for leatherback turtles in the Northeast Atlantic using monthly landscapes of gelatinous organism distribution constructed from Continuous Plankton Recorder Survey data integrated with thermal tolerance parameters.

McMahon and Hays (2006) looked at the impacts that climate change may have on leatherback distribution. They used long-term satellite telemetry to define the habitat use of the turtles and showed that the northerly distribution limit of this species is at the 15 °C isotherm. The summer position of this isotherm has moved north by 330 km in the North Atlantic in the last 17 years.

Population characteristics of leatherbacks captured during eight seasons of fieldwork off Nova Scotia were summarized by James et al. (2007). Animals captured at these high latitudes were mainly large sub-adults and adults, and had a mean curved carapace length of 148.1 cm and mean body mass of 392.6 kg.

Leatherback sighting data in the OBIS dataset consists of 129 records from CETAP 1966-1981 and 246 records from NEFSC, SEFSC, and UNCW surveys between 1991 and 2006. There is no tagging data in the OBIS collection at date of last access. However, the Sea Turtle Conservancy, through the Caribbean Leatherback Tracking and Conservation Project, has made several tagged leatherback tracklines available online (http://conserveturtles.org/seaturtletracking.php?page=sat-leatherback). Several of the turtles tagged in the Caribbean have made extensive use of the U.S. Atlantic coastal waters.

2.39.5 Stock Structure

The Turtle Expert Working Group (2007), delineated seven global stocks of the leatherback turtle based on genetics: Florida, Northern Caribbean, Western Caribbean, Southern Caribbean/Guyana Shield/Trinidad, West Africa, South Africa, and Brazil. A conceptual model developed by the Working Group proposed that hatchlings move from these seven nesting stock areas to an unknown pelagic developmental habitat, and then travel to one of five foraging grounds: the Gulf of Mexico, North/Central Atlantic, Northwestern shelf, Southeastern U.S. coast, or the Eastern shelf (Turtle Expert Working Group 2007).

Dutton et al. (1999) performed analyses of mtDNA control region sequences from 175 leatherback turtles from 10 nesting colonies; this revealed shallow phylogenetic structuring of maternal lineages on a global scale. Nesting populations were found to be strongly subdivided globally and within ocean basins, despite the leatherback's highly migratory nature. Within the Atlantic, significant differences in haplotype frequency distributions, with some exceptions, provisionally support the natal homing hypothesis for leatherback turtles.

2.39.6 Life History Traits

Leatherbacks may reach sexual maturity faster than the hard-shelled sea turtles (Stewart et al. 2007). Small reproductive female leatherbacks have been recorded on nesting beaches since the 1930s; reproductive females as small as 105-125 cm curved carapace length have been observed at most nesting sites and their nests have produced viable hatchlings. Sizes of nesting female leatherbacks analyzed from Indian, Atlantic, and Pacific Ocean populations were found to vary by location and population (Stewart et al. 2007). Avens and Goshe (2008), using skeletochronological analysis of leatherbacks in the Atlantic and Gulf of Mexico, estimated age at maturity to be 29 years of age.

Crim et al. (2002) investigated the paternity of nearly one thousand leatherback hatchlings in Costa Rica. They compared hatchling genotypes with DNA collected from 32 adult females and came to the conclusion that the leatherbacks of this population were both polyandrous and polygynous.

The low reproductive success rate of leatherbacks nesting at Playa Grande, Costa Rica was examined by Bell et al. (2004) and determined to be a factor of embryonic mortality, rather than low fertility. Santidrián Tomillo et al. (2009), also at the Playa Grande nesting site, found that while hatchling production was variable from year to year, high temperatures reduced hatching success and emergence rate. The temporal effect on emergence success meant that more hatchlings were produced at the beginning of the season. Predation on hatchlings at Playa Grande was primarily by ghost crabs, great blue herons, and yellow-crowned night herons (Santidrián Tomillo et al. 2010). Hatchling emergence in Gabon was studied for the effect of artificial lighting (Bourgeois et al. 2009). It was found that the attraction to artificial lighting, which causes misorientation in hatchlings, was stronger than the effect of landward silhouette cues, but was influenced by the brightness of the moon.

Rivalan et al. (2005) used capture-recapture models to investigate intermittent breeding and reproductive effort in leatherback turtles nesting in French Guiana. Using the multistate capture-recapture model developed, the mean survival rate was estimated at 0.91 and the average resighting probability estimated at 0.58 (with a range of 0.30 to 0.99). The breeding cycle was found to be limited to three years.

Georges and Fossette (2006) developed a general linear model to predict body mass from morphometric measurements obtained from leatherback turtles nesting in French Guiana.

Wallace et al. (2007) looked at variations in maternal investment by leatherback turtles. High phenotypic variation in reproductive traits was observed in female leatherbacks, and was postulated to be a response to environmental variability and/or an expression of genotypic variability within the population.

Leatherback turtle vital rates were summarized in the stock assessments of loggerhead and leatherback turtles (National Marine Fisheries Service 2001). Multiple studies were examined for parameters, such as remigration rates, nests/year, yolked eggs/nest, hatch success, size of nesters, and adult mortality (Binckley et al. 1998; Boulon et al. 1996; Campbell et al. 1996; Chevalier et al. 1999; Dutton et al. 2000, 1995; Eckert 2001, 1987; Girondot and Fretey 1996; Godfrey et al. 1996; Hoekert et al. 2000; Hughes 1996; Leslie et al. 1996; McDonald and Dutton 1996; Mrosovsky et al. 1984; Steyermark et al. 1996). Mickelson and Downie (2010) looked at how incubation temperature profiles in Tobago, West Indies, leatherback nests influenced hatchling morphology and locomotor performance. They found that lower incubation temperatures produced hatchlings with traits that were advantageous to terrestrial locomotion.

2.39.7 Food Habits

Based on stomach content analyses, leatherback diet has been determined to be mainly cnidarians (jellyfish, siphonophores) and tunicates (salps, pyrosomas) (Eckert 2001). James and Herman (2001) reported on observations of leatherbacks feeding on jellyfish in waters off Nova Scotia and Newfoundland, Canada. The connection between jellyfish aggregations and leatherback foraging patterns was explored by Houghton et al. (2006) using aerial survey data of the Irish Sea. Surveys discovered consistent aggregations of *Rhizostoma octopus* which were correlated geographically with 22.5% of the anecdotal sightings and strandings records from around the United Kingdom and Ireland.

The leatherback diet of jellyfish puts it at high risk for ingestion of plastic. Autopsy records of 408 leatherback turtles, spanning 123 years (1885-2007), were examined by Mrosovskyet al. (2009) for the presence or absence of plastic in the GI tract. Plastic was reported in 34% of these cases.

Meyers and Hays (2006) used time-depth recorders in conjunction with mouth-opening sensors on leatherbacks in the southern Caribbean to learn if leatherbacks feed during the breeding season. Patterns of diving and mouth-opening activities implied that the leatherbacks were attempting to forage and that gustatory cues may be important. Constantino and Salmon (2003) performed experiments on captive leatherback hatchlings to study the cues used in location and of recognition prey. They concluded that the turtles used both visual and chemical cues to search for and locate food, but that visual stimuli may be of primary importance.

The overlap in food niche exploitation between leatherbacks and another large consumer of gelatinous zooplankton, the ocean sunfish (*Mola mola*), was explored by Hays et al. (2009). While both species foraged in the upper 200 m of waters off South Africa, sunfish were also found to feed on deeper prey beyond the normal diving range of the leatherback turtles. Hays et al. (2004a) had previously looked at the dive profiles of leatherbacks moving from the Caribbean to the Atlantic after nesting. Diving was generally confined to the upper 200 m but varied with foraging success, increasing in depth in the Atlantic where foraging success was expected to be greater. In a study examining dive profiles of leatherbacks in the North Atlantic by Houghton et al. (2008), while the mean maximum dive depth was 59.3 m, dives were recorded as deep as 1250 m. The authors hypothesized that the infrequent deep dives were to survey the water column for diurnally descending prey.

Bradshaw et al. (2007) examined dive duration and depth data collected for nine free-swimming leatherback turtles to infer aerobic dive limits and diving metabolic rates. These findings supported the notion that diving leatherback turtles are ectothermic. Their capacity to have a warm body core even in cold water, therefore, seems to derive from their large size, heat exchangers, thermal inertia, and insulating fat layers and not from an elevated metabolic rate. James at el. (2004) captured foraging leatherbacks off Nova Scotia and recorded their body temperatures. Mean excess temperature over that of the sea surface (15.0-16.7 °C) averaged 8.2 °C. Bostrom et al. (2010) demonstrated juvenile leatherbacks increase activity to control heat gain. Heat loss is regulated physiologically, presumably by regulation of blood flow distribution. Several physiological adaptations for temperature regulation in leatherbacks were examined by Davenport et al. (2009a, 2009b). Thick blubber layers in the head and neck help maintain core body temperatures even when turtles are foraging in cold waters on cold gelatinous prey (Davenport et al. 2009b). Major ontogenetic changes in tracheal structure related to diving and temperature regulation take place in leatherbacks (Davenport et al. 2009a).

Stable isotope analysis has been done by Seminoff et al. (2009) and by Wallace et al. (2006). The Seminoff study looked at stable carbon and stable nitrogen diet-tissue discrimination in captive leatherbacks that were fed a controlled diet. Wallace et al. used leatherback stable isotope analysis to compare nitrogen cycling regimes in the Pacific and Atlantic Oceans.

2.39.8 Health

2.39.8.1 Strandings

The Southeast Fisheries Science Center hosts a queryable dataset on Atlantic and Gulf of Mexico sea turtle strandings at http://www.sefsc.noaa.gov/seaturtleSTSSN.jsp. Figures 2.39-4 and 2.39-5 show density distributions of strandings along the Atlantic coast.

Murphy et al. (2006) reported on the rise in leatherback strandings on South Carolina beaches since 1989. Between 1980 and 2003, 141 leatherback carcasses stranded. The seasonal peak for strandings occurred in the spring and was followed by a lesser peak in the fall. Based on 23 necropsies, there were 7 males and 16 females.

2.39.8.2 Contaminants

Orós et al. (2009) examined tissue samples from 30 loggerheads, one leatherback, and one green turtle that had stranded on the coasts of the Canary Islands. While concentrations of PCB 209 were high in the liver of the loggerheads and in the liver and adipose tissue of the leatherback, it was not possible to draw a clear correlation between the PCB concentrations and cause of death. Edmonds (1994) found arsenic in tissues of a leatherback turtle. Low exposure to OC contamination by leatherbacks was measured by McKenzie et al. (1999). Sample size was too low to draw conclusions from, but it was postulated that the leatherback's diet of pelagic jellyfish could be an explanation for its lower evidence of exposure.

Concentrations of essential (copper, zinc, selenium) and non-essential elements (cadmium, lead, mercury) were determined in the blood and eggs of 46 free-ranging leatherback females sampled in French Guiana (Guirlet et al. 2008). All trace elements were detectable in both tissues. Levels of toxic metals were lower than essential elements, likely because of the high pelagic nature of leatherbacks that seems to limit exposure to toxic elements. Over the course of the nesting season, copper levels decreased while blood lead levels increased. The high demand on the body during the breeding season could affect blood copper concentrations. The calcium requirement for egg production with concomitant lead mobilization could explain the increase in blood lead concentrations. The authors also examined maternal transfer of organochlorine contaminants (OCs), pesticides (DDTS and HCHs) and polychlorinated biphenyls (PCBs) (Guirlet et al. 2010). PCBs were the dominant OCs found. OC concentrations were lower than concentrations measured in other marine turtles; this might be explained by the lower trophic position (diet based on jellyfish) and by the location of their foraging and nesting grounds. All OCs detected in leatherback blood were detected in eggs; this suggests a maternal transfer of OCs. During the nesting season, OC concentrations in eggs and the percentage of lipid in eggs were found to decline in successive clutches, highlighting a process of offloading from females to their eggs and a decreasing investment of lipid from females into their clutches. OCs in eggs tended to be higher in females spending three years on the foraging grounds between nesting seasons than in those spending two years.

2.39.8.3 Disease

Santoro et al. (2008) tested leatherback turtles from the nesting population of Pacuare Nature Reserve in Caribbean Costa Rica for pathogenic bacteria. A total of 189 isolates, including 113 gram-negative and 76 gram-positive bacteria, most of which may be considered as potential pathogens, were identified in samples from 70 nesting females,.

2.39.9 Acoustics

Little has been done specifically on leatherbacks and acoustics, though studies on loggerhead sea turtles have demonstrated sensitivity to low frequency noise (Bartol et al. 1999; Samuel et al. 2005). Underwater sound recordings from the Peconic Bay Estuary system in Long Island, New York demonstrated high levels of anthropogenic noise at frequencies in the range of the turtles' highest sensitivity; these also overlapped spatially and temporally with high turtle use of the habitat (Samuel et al. 2005). Sea turtle exposure to seismic air gun noise was studied by Weir (2007), though no conclusions on the impacts to the turtles by the airguns were able to be drawn in that study. Viada et al. (2008) discussed sea turtle response to underwater explosions and mitigation scenarios developed for explosive removal of oil and gas structures. The authors concluded that though existing regulations provide more protection for marine turtles than had previously existed, little is known about sub-lethal effects of underwater explosions on turtles, especially potential auditory impacts (Viada et al. 2008).

2.39.10 Fisheries By-Catch and Entanglement

Globally, leatherback bycatch in pelagic longline fisheries was estimated to have been between 50,000 and 60,000 in 2000 (Lewison et al. 2004).

Bycatch of leatherback sea turtles has been documented in several Atlantic fisheries, including pelagic longline, shark gillnet, sink gillnet, and bottom otter trawl. Between 2005 and 2009, in the Florida East Coast Pelagic longline fishing area, which corresponds roughly to the BOEM Straits of Florida Planning Area, leatherback bycatch was estimated to range between 6.6 (2007) and 62.3 (2005). During the same period, in the mid-Atlantic Bight, which falls within the mid-Atlantic and North Atlantic planning areas, estimated leatherback bycatch in the longline fishery ranged from 30 (2006) to 114.1 (2007). In the Northeast Coastal Area, part of which falls within the boundaries of the BOEM North Atlantic Planning Area, between 2005 and 2009 the estimated bycatch of leatherbacks ranged from 0 (2008) to 116 (2006). The South Atlantic Bight Area, falling in the mid- and South Atlantic Planning Areas, saw bycatch ranging from 0 (2007 and 2008) to 39 (2006) during the same period (Fairfield Walsh and Garrison 2006, 2007; Fairfield and Garrison 2008; Garrison and Stokes 2010; Garrison et al. 2009). In the shark gillnet fishery between 2000 and 2006, eight loggerheads were reported taken in drift nets, four in strike nets, and one in a sink net (Garrison 2007a). Five leatherback turtles were observed in mid-Atlantic gillnet gear between 1995 and 2006 (Murray 2009a).

Work by Ferarroli et al. (2004), showed that for leatherbacks nesting in French Guinea and Suriname, foraging patterns overlap with both pelagic longline fisheries along oceanic fronts and with coastal shrimp and red snapper fisheries near the nesting grounds. Hays et al. (2004b) also discussed the overlap between leatherback foraging and longline fisheries in the Atlantic. They pointed out that the potential for interaction with longlines was increased because, in addition to the pan-oceanic movements which put the turtles into contact with fished areas, the turtles also spent more time in shallow dive patterns when they were in the feeding areas. Kot et al. (2010) looked at temporal patterns of sea turtle bycatch using logbook data from U.S. Atlantic longline fisheries. Most regions that were examined exhibited a strong annual/seasonal periodicity in turtle bycatch rates.

Satellite telemetry data of leatherbacks tagged off Nova Scotia, Canada, and morphometrics from foraging animals and entanglement records were examined by James et al. (2005b), and demonstrated that, because leatherbacks migrate annually to key feeding areas in northern latitudes, they are vulnerable to entanglement in northern coastal and shelf waters.

2.39.11 Vessel Interactions

Injuries indicative of collisions with vessels are commonly found in sea turtles. In the stranding data collected from 1997 to 2001 by the Sea Turtle Stranding and Salvage Network, 12.7% of all stranded turtles were documented as having sustained injuries consistent with propeller wounds or collision, although some of these injuries may have occurred post-mortem (National Marine Fisheries Service et al. 2010).

2.39.12 Energy Projects

Both extractive and renewable offshore energy projects have the potential to impact the marine environment in a variety of ways. including habitat alteration and the introduction of noise via seismic exploration activities (e.g.. airguns), vessel/aircraft activity, construction and operation (Geraci and St. Aubin 1980, Bain and Williams 2006, Thompsen et al. 2006, Cada et al. 2007, Compton 2008, Evans 2008, MMS 2008, Burgman 2010, Boehlert and Gill 2010, Dolman and Simmonds 2010, Edrén et al. 2010, Bush 2011, ICES 2011). The impact of these activities on turtles is unclear. McMichael and Wyneken (2010) reported on a research program aimed at monitoring and minimizing the impacts from interactions between sea turtles and ocean energy technology within the Florida Current.

Leatherbacks are occasionally captured in the cooling water intakes of industrial facilities, such as Florida Power and Light Company’s St. Lucie Power Plant on Hutchinson Island. Between March 1976 (when the St. Lucie Plant opened) and November 1988, eight leatherbacks were captured (Ernest et al. 1989).

Effects of oil and oil spill response guidelines are presented in a report edited by Shigenaka (2003). The report concluded that not only do oil collection areas overlap with sensitive sea turtle habitat, but there is also strong, if limited and dated, information that indicates oil is harmful to turtles, particularly to dermal tissues and membranes. In addition, fresh oil has been found to harm sea turtle eggs. The relationship between offshore oil platforms and the distribution of sea turtles is discussed in a report made by Lohoefener et al. at the 1989 Ternary Studies meeting sponsored by the MMS (Lohoefener et al. 1989). That study found that in some areas of the Gulf of Mexico, chelonid sea turtles were positively correlated with platforms, but in others there was either a negative correlation or no correlation at all. High amplitude simulated pile driving vibrations were found to decrease sea turtle hatching success (Ripcke 2011).

2.39.13 Data Gaps and Research Recommendations

Hays (2008) reviewed recent research on sea turtles, and pointed out topical areas and some remaining questions; Heenehan (2010) summarized work on acoustic impacts to sea turtles. Research recommendations outlined in the Leatherback Assessment by the Turtle Expert

Working Group (2007) include increased collection and estimation of demographic parameters, bycatch, and mortality; continued genetic sampling; and increased research on the effects of pollutants, disease, strandings and other threats to leatherback populations. The National Research Council’s *Assessment of Sea-turtle Status and Trends* (Committee on Sea Turtle Population Assessment Methods; National Research Council 2010) recommended that NMFS and the USFWS develop a coherent national strategy for sea-turtle assessments to improve collection, quality and availability of data, and to develop a plan of external review of data and models used for population status estimation and trend analysis.

Table 2.39.13 Data Sources

(Unless otherwise specified, data presented in the maps comes from the following sources. Sightings maps are not effort-corrected and as such should not be interpreted as complete representations of species distribution.)

Data Set	Description	Website/Contact
Aerial survey of upper trophic level predators on Platts Bank, Gulf of Maine (Gulf of Maine Census of Marine Life Program)	For description see: http://gcmd.nasa.gov/KeywordSearch/Metadata.do?Portal=gommp&KeywordPath=&NumericId=27198&MetadataView=Full&MetadataType=0&lbnode=mdlb2	nwolff@usm.maine.edu
Bald Head Island 2004: Loggerhead Turtles	Coyne, M. S., and B. J. Godley. 2005. Satellite Tracking and Analysis Tool (STAT): an integrated system for archiving, analyzing and mapping animal tracking data. Marine Ecology Progress Series. Vol. 301:1-7.	B.J.Godley@exeter.ac.uk Data retrieved from: http://www.iobis.org
Bald Head Island 2005: Loggerhead Turtles	Coyne, M. S., and B. J. Godley. 2005. Satellite Tracking and Analysis Tool (STAT): an integrated system for archiving, analyzing and mapping animal tracking data. Marine Ecology Progress Series. Vol. 301:1-7.	mcoyne@seaturtle.org Data retrieved from http://www.iobis.org
BLM Cetacean and Turtle Assessment Program (CETAP) SHIP Sightings	University of Rhode Island BLM Cetacean and Turtle Assessment Program (CETAP) SHIP Sightings. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2005-08-02 15:09:16.272989-04, vector digital data.	rkenney@gso.uri.edu Data retrieved from http://www.iobis.org
BLM CETAP AIR Sightings	University of Rhode Island BLM Cetacean and Turtle Assessment Program (CETAP) AIR Sightings. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2005-08-02 15:08:23.268819-04, vector digital data.	rkenney@gso.uri.edu Data retrieved from http://www.iobis.org
BLM CETAP OPP Sightings	University of Rhode Island BLM Cetacean and Turtle Assessment Program (CETAP) OPP Sightings. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2005-08-02 15:08:47.889912-04, vector digital data.	rkenney@gso.uri.edu Data retrieved from http://www.iobis.org
Cape Cod Sea Turtle Release 2007	Coyne, M. S., and B. J. Godley. 2005. Satellite Tracking and Analysis Tool (STAT): an integrated system for archiving, analyzing and mapping animal tracking data. Marine Ecology Progress Series. Vol. 301:1-7.	cmerigo@neaq.org Data retrieved from http://www.iobis.org

Data Set	Description	Website/Contact
Casey Key Loggerheads - 2009	Coyne, M. S., and B. J. Godley. 2005. Satellite Tracking and Analysis Tool (STAT): an integrated system for archiving, analyzing and mapping animal tracking data. Marine Ecology Progress Series. Vol. 301:1-7. Girard, C., A. D. Tucker, and B. Calmettes. 2009. Post-nesting migrations of loggerhead sea turtles in the Gulf of Mexico: dispersal in highly dynamic conditions. Marine Biology. 156:1827-1839. DOI:10.1007/s00227-009-1216-z. Tucker, A. D. 2009. Eight nests recorded for a loggerhead turtle within one season. Marine Turtle Newsletter. 124:16-17.	tucker@mote.org Data retrieved from http://www.iobis.org
Casey Key Loggerheads 2005-2006 - Southwest Florida	Mote Marine Laboratory Casey Key Loggerheads - Southwest Florida. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu, 2006-11-02 05:35:48-05, vector digital data.	tucker@mote.org Data retrieved from http://www.iobis.org
Casey Key Loggerheads-2007	Coyne, M. S., and B. J. Godley. 2005. Satellite Tracking and Analysis Tool (STAT): an integrated system for archiving, analyzing and mapping animal tracking data. Marine Ecology Progress Series. Vol. 301:1-7. Girard, C., A. D. Tucker, and B. Calmettes. 2009. Post-nesting migrations of loggerhead sea turtles in the Gulf of Mexico: dispersal in highly dynamic conditions. Marine Biology. 156:1827-1839. DOI:10.1007/s00227-009-1216-z Tucker, A. D. 2009. Eight nests recorded for a loggerhead turtle within one season. Marine Turtle Newsletter. 124:16-17	tucker@mote.org Data retrieved from http://www.iobis.org

Data Set	Description	Website/Contact
Casey Key Loggerheads-2008	Coyne, M. S., and B. J. Godley. 2005. Satellite Tracking and Analysis Tool (STAT): an integrated system for archiving, analyzing and mapping animal tracking data. Marine Ecology Progress Series. Vol. 301:1-7. Girard, C., A. D. Tucker, and B. Calmettes. 2009. Post-nesting migrations of loggerhead sea turtles in the Gulf of Mexico: dispersal in highly dynamic conditions. Marine Biology. 156:1827-1839. DOI:10.1007/s00227-009-1216-z. Tucker, A. D. 2009. Eight nests recorded for a loggerhead turtle within one season. Marine Turtle Newsletter. 124:16-17 Retrieved from http://www.iobis.org	tucker@mote.org Data retrieved from http://www.iobis.org
Duke North Atlantic Turtle Tracking	Duke University Marine Laboratory Duke North Atlantic Turtle Tracking. In: OBIS-SEAMAP. OBIS-SEAMAP, http://seamap.env.duke.edu, 2006-11-02 05:36:26-05, vector digital data.	catherin@duke.edu Data retrieved from http://www.iobis.org
DUML Vessel-Based Surveys for USWTR site 2009-2010	Nilsson, P.B., Foley, H. J., Hardee, R.E., Holt, R. C., McAlarney, R.J., Cummings, E. W., Johnston, D.W., McLellan, W. A., Pabst, D. A. and Read, A.J. 2010. Protected species monitoring in the proposed Under Sea Warfare Training Range Off-Shore of Jacksonville, FL: Jan - Dec 2009. SEAMAMMS 2010 Conference Abstract, Poster	david.johnston@duke.edu Presentation retrieved from http://www.iobis.org
Fishery bycatch - NE	1989-2010 Northeast Fishery Observer Program data	http://www.nefsc.noaa.gov/fsb/
Hatteras Eddy Cruise 2004	Duke / UNC Oceanographic Consortium Hatteras Eddy Cruise 2004. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2006-03-29 13:15:44.214284-05, vector digital data.	khyrenba@u.washington.edu or khyrenba@duke.edu Data retrieved from http://www.iobis.org
Joint Deepwater Systematics and Marine Mammal Survey	NOAA NEFSC Joint Deepwater Systematics and Marine Mammal Survey. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2005-08-17 19:49:42.240994-04, vector digital data.	Debra.Palka@noaa.gov Data retrieved from http://www.iobis.org

Data Set	Description	Website/Contact
Loggerhead Turtles: Bald Head Island 2003	Marine Turtle Research Group Loggerhead Turtles: Bald Head Island 2003. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu, 2008-01-11 05:04:01-05, vector digital data.	Data retrieved from http://www.iobis.org
Mammal Strandings	NOAA National Marine Mammal Health and Stranding Response Database and the NOAA SER Marine Mammal Stranding Database, Northeast and Southeast US Marine Mammal Stranding Networks 2005-2009	
Mote Marine Laboratory - Sea Turtle Rehabilitation Hospital	Coyne, M. S., and B. J. Godley. 2005. Satellite Tracking and Analysis Tool (STAT): an integrated system for archiving, analyzing and mapping animal tracking data. Marine Ecology Progress Series. Vol. 301:1-7.	tucker@mote.org Data retrieved from http://www.iobis.org
National Marine Life Center	Coyne, M. S., and B. J. Godley. 2005. Satellite Tracking and Analysis Tool (STAT): an integrated system for archiving, analyzing and mapping animal tracking data. Marine Ecology Progress Series. Vol. 301:1-7.	bmoore@nmlc.org Data retrieved from http://www.iobis.org
NEFSC 1995 AJ9501 (Part I)	Debra Palka, David Potter, National Marine Fisheries Service. 1991. Cruise report of the harbor porpoise survey - 1991 AJ91-02	Debra.Palka@noaa.gov
NEFSC Aerial Circle-Back Abundance Survey 2004	NOAA Northeast Fisheries Science Center. 2004. Aerial survey results; NOAA Twin Otter aircraft; Circle-Back Abundance Survey.	Debra.Palka@noaa.gov Data retrieved from http://www.iobis.org
NEFSC Aerial Survey - Experimental 2002	NOAA Northeast Fisheries Science Center (NEFSC) NEFSC Aerial Survey - Experimental 2002. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2006-03-08 11:43:55-05, vector digital data.	Debra.Palka@noaa.gov Data retrieved from http://www.iobis.org
NEFSC Aerial Survey - Summer 1995	NOAA Northeast Fisheries Science Center (NEFSC) NEFSC Aerial Survey - Summer 1995. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2006-03-08 11:43:17-05, vector digital data.	Debra.Palka@noaa.gov Data retrieved from http://www.iobis.org
NEFSC Aerial Survey - Summer 1998	NOAA Northeast Fisheries Science Center (NEFSC) NEFSC Aerial Survey - Summer 1998. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2006-05-12 11:33:36-04, vector digital data.	Debra.Palka@noaa.gov Data retrieved from http://www.iobis.org

Data Set	Description	Website/Contact
NEFSC Marine Mammal Abundance Survey - Leg 1	NOAA Northeast Fisheries Science Center (NEFSC) Marine Mammal Abundance Survey - Leg 1. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2005-08-17 19:25:23.992499-04, vector digital data.	Debra.Palka@noaa.gov Data retrieved from http://www.iobis.org
NEFSC Marine Mammal Survey PE 95-02	NOAA Northeast Fisheries Science Center (NEFSC) Marine Mammal Survey PE 95-02. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2005-08-17 19:24:07.349686-04, vector digital data.	Debra.Palka@noaa.gov Data retrieved from http://www.iobis.org
NEFSC Mid-Atlantic Marine Mammal Abundance Survey 2004	NOAA Northeast Fisheries Science Center. 2004. Cruise Results; R/V Endeavor; Cruise No. EN 04-395/396; Mid-Atlantic Marine Mammal Shipboard Abundance Survey.	Debra.Palka@noaa.gov Data retrieved from http://www.iobis.org
NEFSC Survey 1998 1	NOAA Northeast Fisheries Science Center (NEFSC) NEFSC Survey 1998 1. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2004-02-13 15:42:07.2188-05, vector digital data.	Debra.Palka@noaa.gov Data retrieved from http://www.iobis.org
NEFSC Survey 1998 2	NOAA Northeast Fisheries Science Center (NEFSC) NEFSC Survey 1998 2. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2004-02-13 15:43:01.279903-05, vector digital data.	Debra.Palka@noaa.gov Data retrieved from http://www.iobis.org
NEFSC Aerial Survey - Summer 2010	Survey report available at: http://www.nefsc.noaa.gov/psb/surveys/documents/air2010.pdf	Debra.Palka@noaa.gov
NEFSC Aerial Survey - Summer 2008	Survey report available at: http://www.nefsc.noaa.gov/psb/surveys/documents/air2008.pdf	Debra.Palka@noaa.gov
NEFSC Aerial Survey - Summer 2007	Survey report available at: http://www.nefsc.noaa.gov/psb/surveys/documents/air2007.pdf	Debra.Palka@noaa.gov
NEFSC Aerial Survey - Summer 2006	Survey report available at: http://www.nefsc.noaa.gov/psb/surveys/documents/CruiseReport.aerial2006.full.pdf	Debra.Palka@noaa.gov
Newport Aquarium 2004: Loggerhead Turtle	Coyne, M. S., and B. J. Godley. 2005. Satellite Tracking and Analysis Tool (STAT): an integrated system for archiving, analyzing and mapping animal tracking data. Marine Ecology Progress Series. Vol. 301:1-7.	B.J.Godley@exeter.ac.uk Data retrieved from http://www.iobis.org

Data Set	Description	Website/Contact
NOAA Atlantic Cetacean Survey 1999; Sightings	NOAA Southeast Fisheries Science Center (SEFSC) NOAA Atlantic Cetacean Survey 1999; Sightings. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2005-06-10 07:55:07.256674-04, vector digital data.	Lance.Garrison@noaa.gov Data retrieved from http://www.iobis.org
NOAA Southeast Cetacean Aerial Survey 1995; Sightings	NOAA Southeast Fisheries Science Center (SEFSC) NOAA Southeast Cetacean Aerial Survey 1995; Sightings. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2005-06-28 21:14:28.534611-04, vector digital data.	Lance.Garrison@noaa.gov Data retrieved from http://www.iobis.org
North Carolina Aquarium at Pine Knoll Shores Sea Turtle Awareness (OBIS-SEAMAP)	Coyne, M. S., and B. J. Godley. 2005. Satellite Tracking and Analysis Tool (STAT): an integrated system for archiving, analyzing and mapping animal tracking data. Marine Ecology Progress Series. Vol. 301:1-7.	heather.broadhurst@ncaquariums Data retrieved from http://www.iobis.org
Northeast Fisheries Science Center Bottom Trawl Survey Data	NOAA's National Marine Fisheries Service (NMFS) Northeast Fisheries Science Center Northeast Fisheries Science Center Bottom Trawl Survey Data. NOAA's National Marine Fisheries Service (NMFS) Northeast Fisheries Science Center, Ecosystems Survey Branch, Woods Hole, Massachusetts, United States of America, 2005-04-30, Tabular Digital Data. Additional information can be found at http://www.nefsc.noaa.gov.	Linda.Despres@noaa.gov Data retrieved from http://www.iobis.org
Pelagic longline bycatch	1992-2010	Lance.Garrison@noaa.gov
PIROP Northwest Atlantic 1965-1992	See description at: http://seamap.env.duke.edu/datasets/detail/280	khyrenba@u.washington.edu Data retrieved from http://www.iobis.org
Sea Turtle Strandings	Sea Turtle Stranding and Salvage Network	http://www.sefsc.noaa.gov/species/turtles/strandings.htm
Sargasso 2005 - cetacean sightings	See description at: http://seamap.env.duke.edu/dataset/332	hwhitehe@dal.ca Data retrieved from http://www.iobis.org
SEFSC Mid-Atlantic Tursiops Survey, 1995 2	NOAA Southeast Fisheries Science Center (SEFSC) NOAA Mid Atlantic Tursiops Surveys 1995 (2); Sightings. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2005-06-28 21:40:00.121113-04, vector digital data.	Lance.Garrison@noaa.gov Data retrieved from http://www.iobis.org

Data Set	Description	Website/Contact
SEFSC Mid-Atlantic Tursiops Survey, 1995 3	NOAA Southeast Fisheries Science Center (SEFSC) NOAA Mid Atlantic Tursiops Surveys 1995 (3); Sightings. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2005-06-28 21:37:27.211758-04, vector digital data.	Lance.Garrison@noaa.gov Data retrieved from http://www.iobis.org
SEFSC Southeast Cetacean Aerial Survey 1992	NOAA Southeast Fisheries Science Center (SEFSC) NOAA Southeast Cetacean Aerial Survey 1992; Sightings. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2005-06-28 21:18:36.46644-04, vector digital data.	Lance.Garrison@noaa.gov Data retrieved from http://www.iobis.org
SEFSC sightings	2004, 2005, 2005 SEFSC shipboard surveys	Lance.Garrison@noaa.gov
Southeast Area Monitoring and Assessment Program (SEAMAP) South Atlantic (USOBIS)	For description see: http://www.seamap.org/	Data retrieved from http://www.iobis.org
Summer 2004 Cape Hatteras	Duke University Marine Laboratory Summer 2004 Cape Hatteras. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2006-05-25 16:05:08.17444-04, vector digital data.	eal12@duke.edu Data retrieved from http://www.iobis.org
Turtle Hospital, Marathon Florida	Coyne, M. S., and B. J. Godley. 2005. Satellite Tracking and Analysis Tool (STAT): an integrated system for archiving, analyzing and mapping animal tracking data. Marine Ecology Progress Series. Vol. 301:1-7.	turtlehosp@aol.com, tucker@mote.org Data retrieved from http://www.iobis.org
UNCW Aerial Survey 98-99	University of North Carolina at Wilmington UNCW Aerial Survey 98-99. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2005-03-11 18:52:29.590331-05, vector digital data.	mclellanw@uncw.edu Data retrieved from http://www.iobis.org
UNCW Aerial Surveys for monitoring of proposed Oslow Bay USWTR site - Left side - (OBIS-SEAMAP) (OBIS-SEAMAP)	For description see: http://www.seamap.org/	mclellanw@uncw.edu Data retrieved from http://www.iobis.org
UNCW Aerial Surveys for monitoring of proposed Oslow Bay USWTR site - Right side - (OBIS-SEAMAP) (OBIS-SEAMAP)	For description see: http://www.seamap.org/	mclellanw@uncw.edu Data retrieved from http://www.iobis.org
UNCW Marine Mammal Aerial Surveys 2006-2007	For description see: http://seamap.env.duke.edu/dataset/400/html	mclellanw@uncw.edu Data retrieved from http://www.iobis.org

Data Set	Description	Website/Contact
UNCW Marine Mammal Sightings 2002	University of North Carolina at Wilmington UNCW Marine Mammal Sightings, Southeastern US 2002. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2007-03-26 16:44:01-05, vector digital data.	mclellanw@uncw.edu Data retrieved from http://www.iobis.org
UNCW Marine Mammal Sightings, Southeastern US 1998-1999	University of North Carolina at Wilmington UNCW Marine Mammal Sightings, Southeastern US 1998-1999. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/ , 2006-03-21 17:53:37-05, vector digital data.	mclellanw@uncw.edu Data retrieved from http://www.iobis.org
UNCW Marine Mammal Sightings, Southeastern US 2001	University of North Carolina at Wilmington UNCW Marine Mammal Sightings, Southeastern US 2001. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2007-03-26 16:44:57-05, vector digital data.	mclellanw@uncw.edu Data retrieved from http://www.iobis.org
UNCW Right Whale Aerial Survey 05-06	University of North Carolina at Wilmington UNCW Right Whale Aerial Survey 05-06. In: OBIS-SEAMAP . OBIS-SEAMAP, http://seamap.env.duke.edu/, 2006-07-11 14:26:21.835383-04, vector digital data.	mclellanw@uncw.edu Data retrieved from http://www.iobis.org
UNCW Right Whale Aerial Surveys 2008	For description see: http://seamap.env.duke.edu/dataset/464	mclellanw@uncw.edu Data retrieved from http://www.iobis.org
USWTR JAX Aerial Survey -Left side- 2009-2010 (OBIS-SEAMAP) (OBIS-SEAMAP)	William McLellan. 2010. UNCW USWTR JAX Aerial Survey -Left side- 2009-2010.	mclellanw@uncw.edu Data retrieved from http://www.iobis.org
USWTR Onslow Bay Aerial Survey -Right side- 2008-2010 (OBIS-SEAMAP) (OBIS-SEAMAP)	William McLellan. 2010. UNCW USWTR Onslow Bay Aerial Survey -Right side- 2008-2010.	mclellanw@uncw.edu Data retrieved from http://www.iobis.org

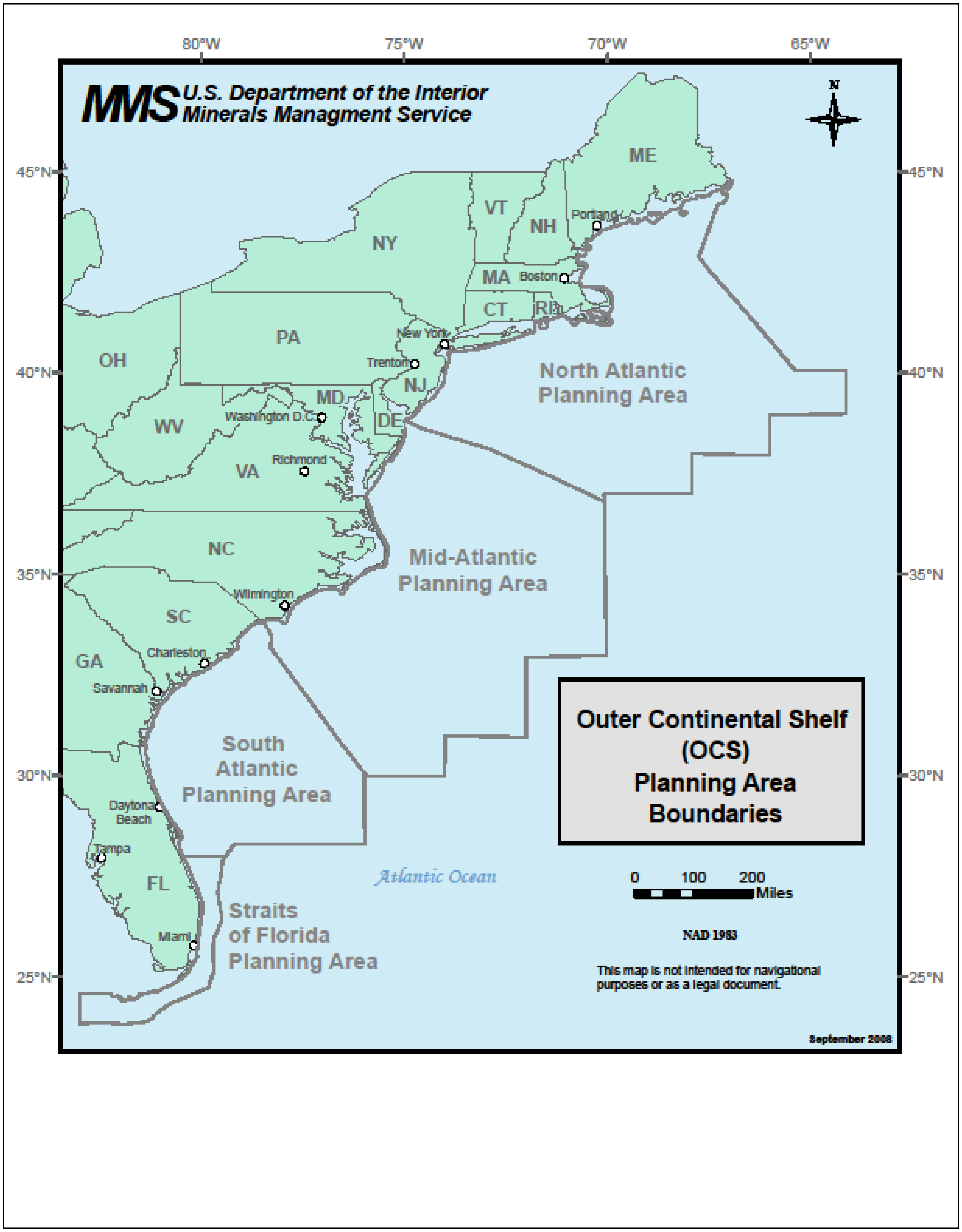


Figure 1.1.1. BOEM Atlantic OCS Planning Areas.

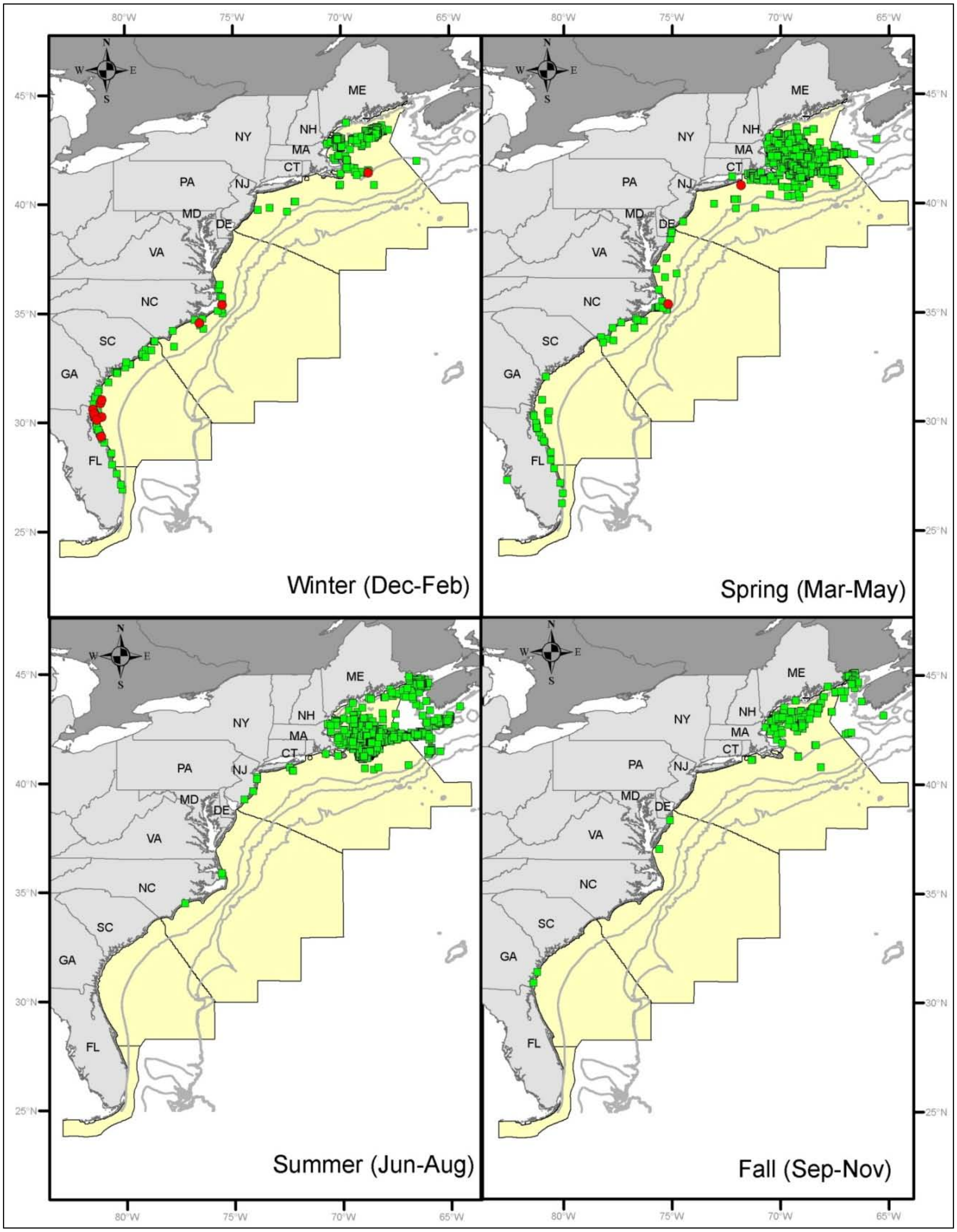


Figure 2.1.1. North Atlantic right whale sighting (green squares) and stranding (red dots) observations by season.

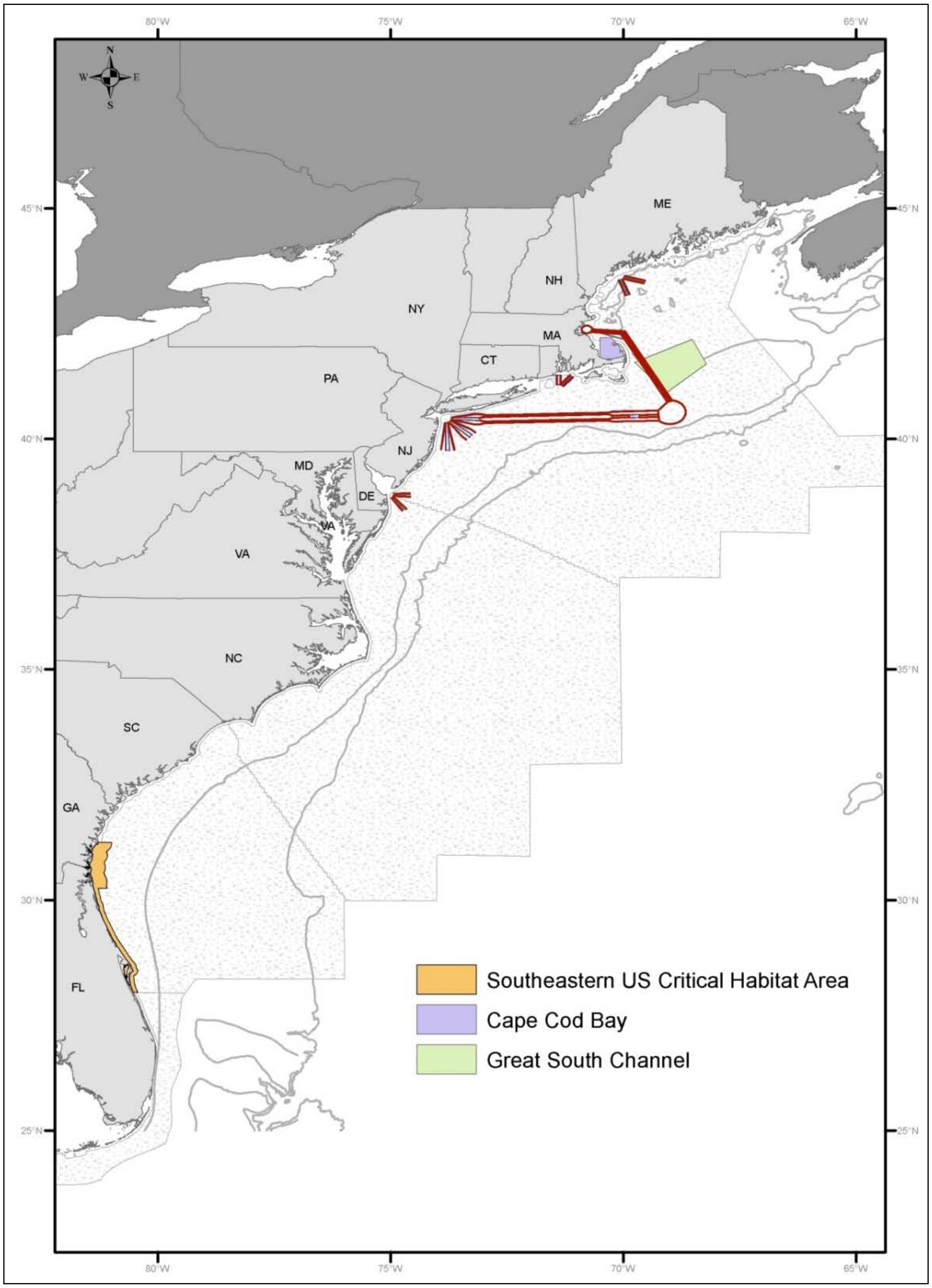


Figure 2.1.2. North Atlantic right whale critical habitat areas and northeastern shipping lanes.

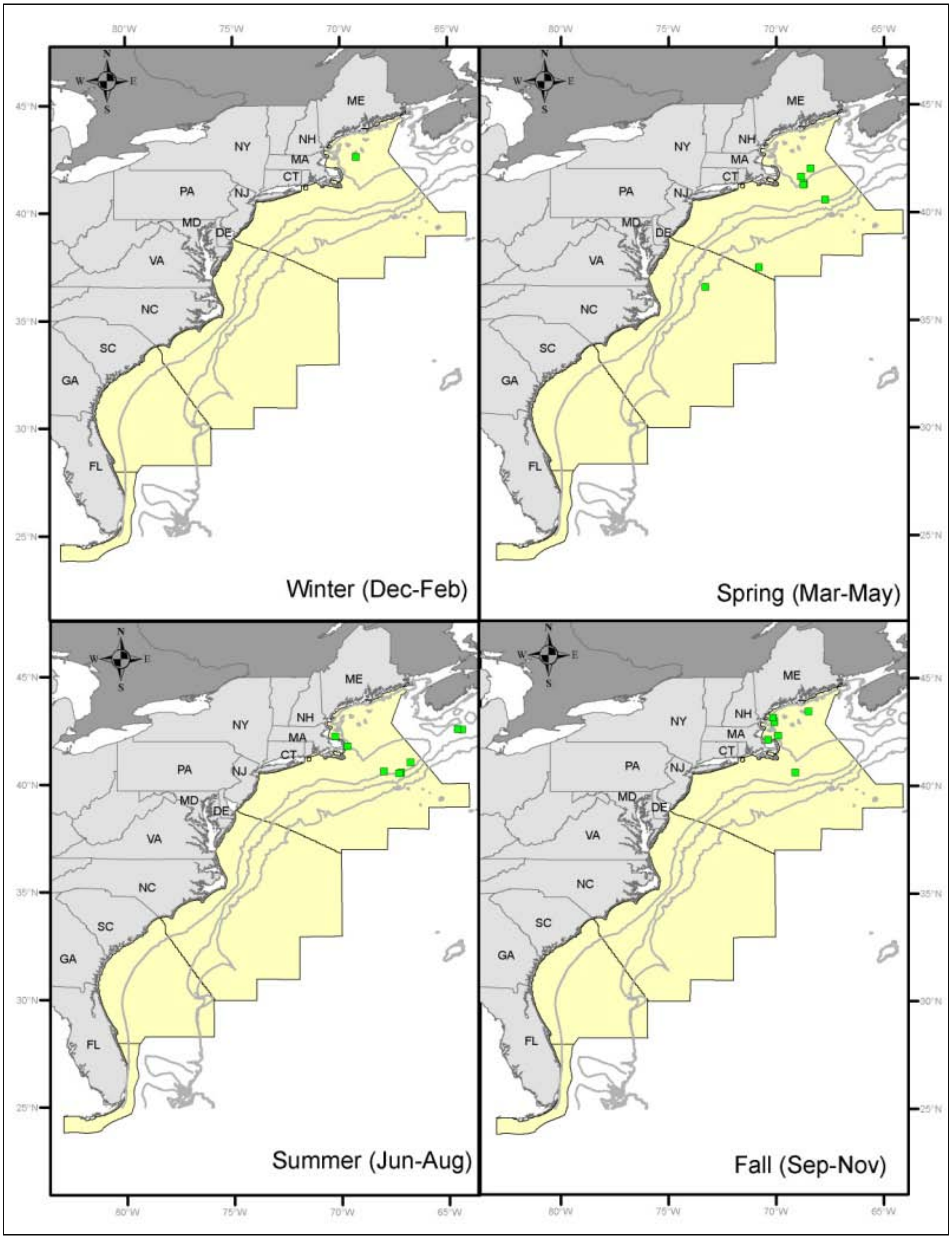


Figure 2.2.1. Blue whales sighting (green squares) observations by season.

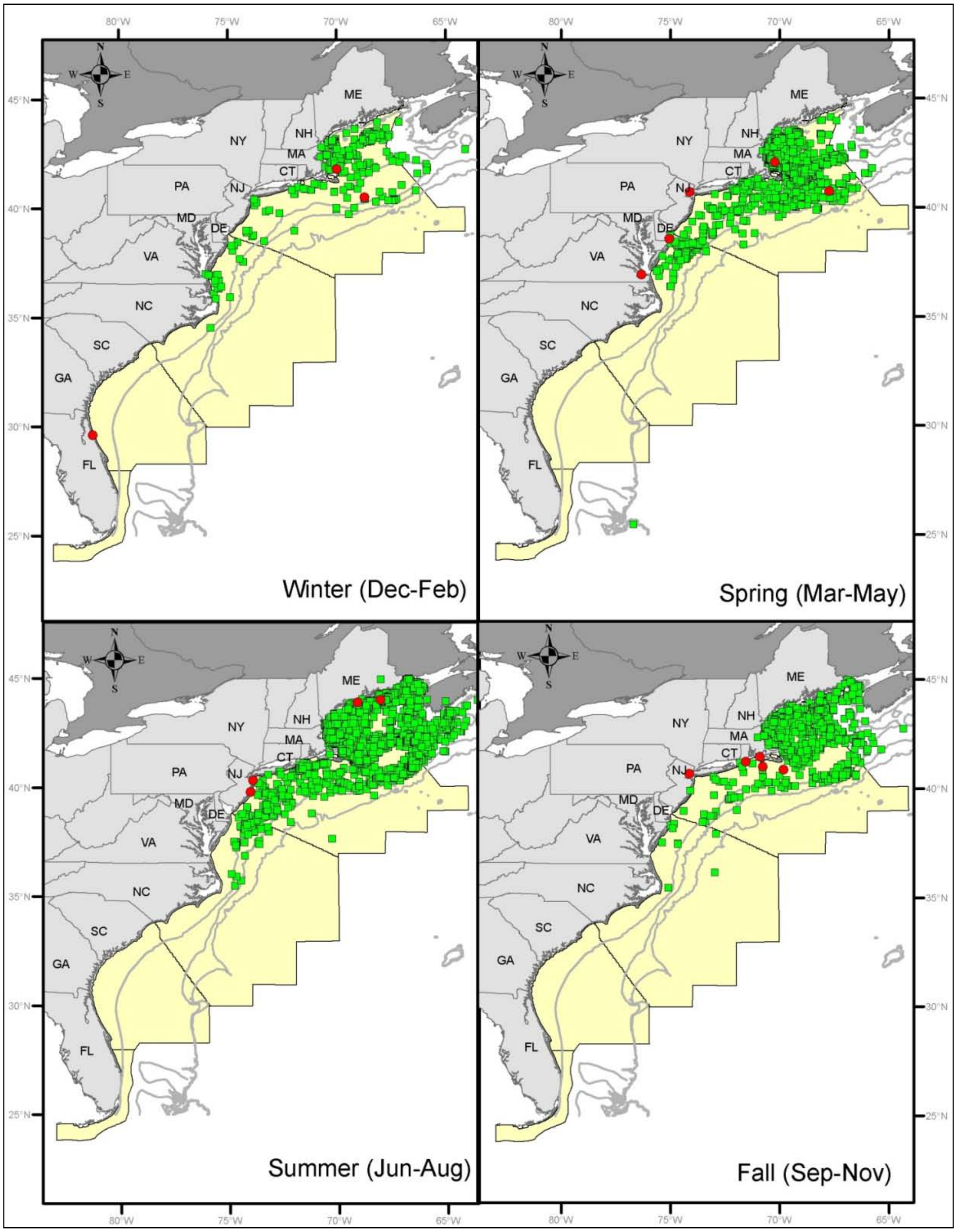


Figure 2.3.1. Fin whale sighting (green squares), stranding (red dots) and fishery by-catch (black stars) observations by season.

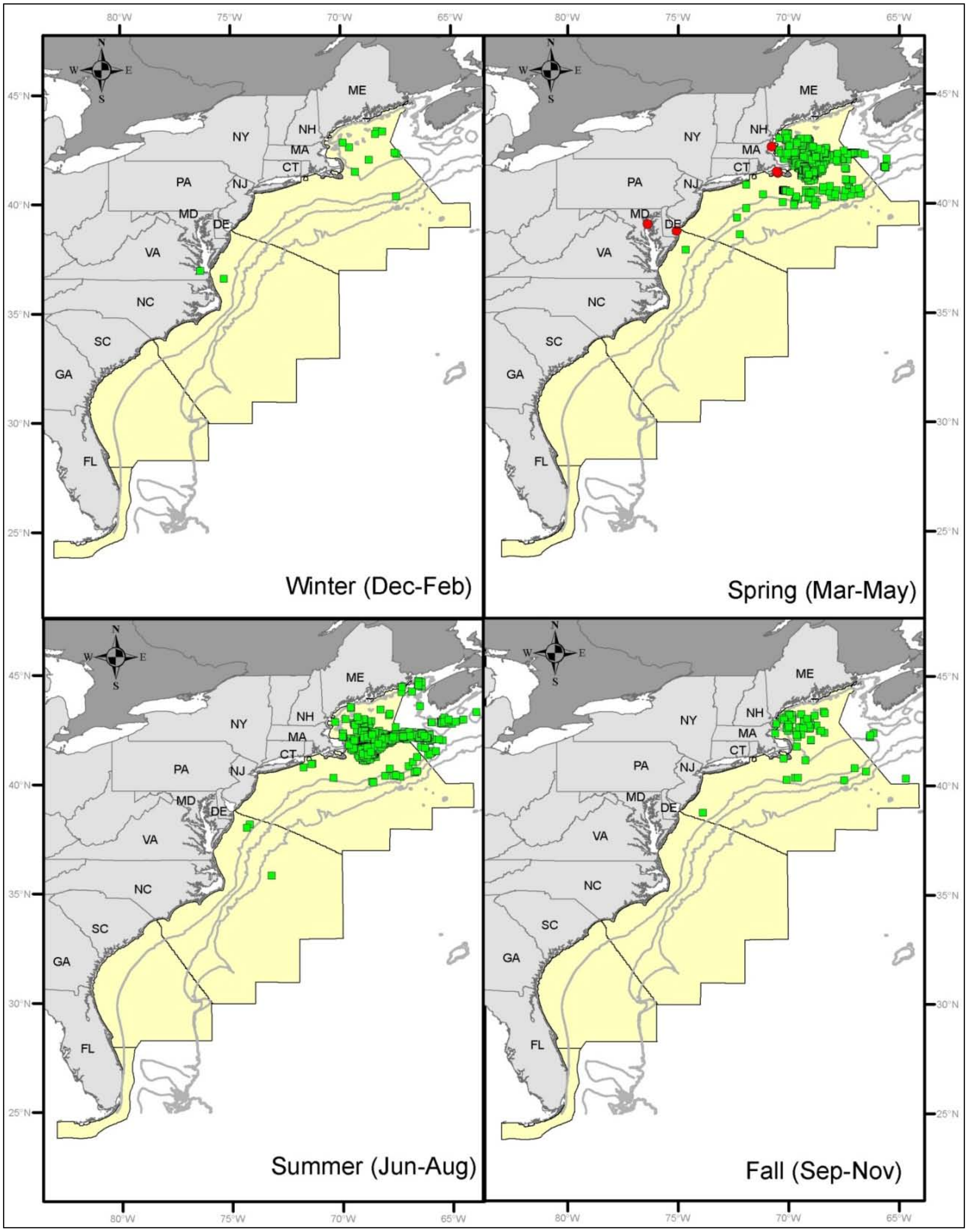


Figure 2.4.1. Sei whale sighting (green squares) and stranding (red dots) observations of by season.

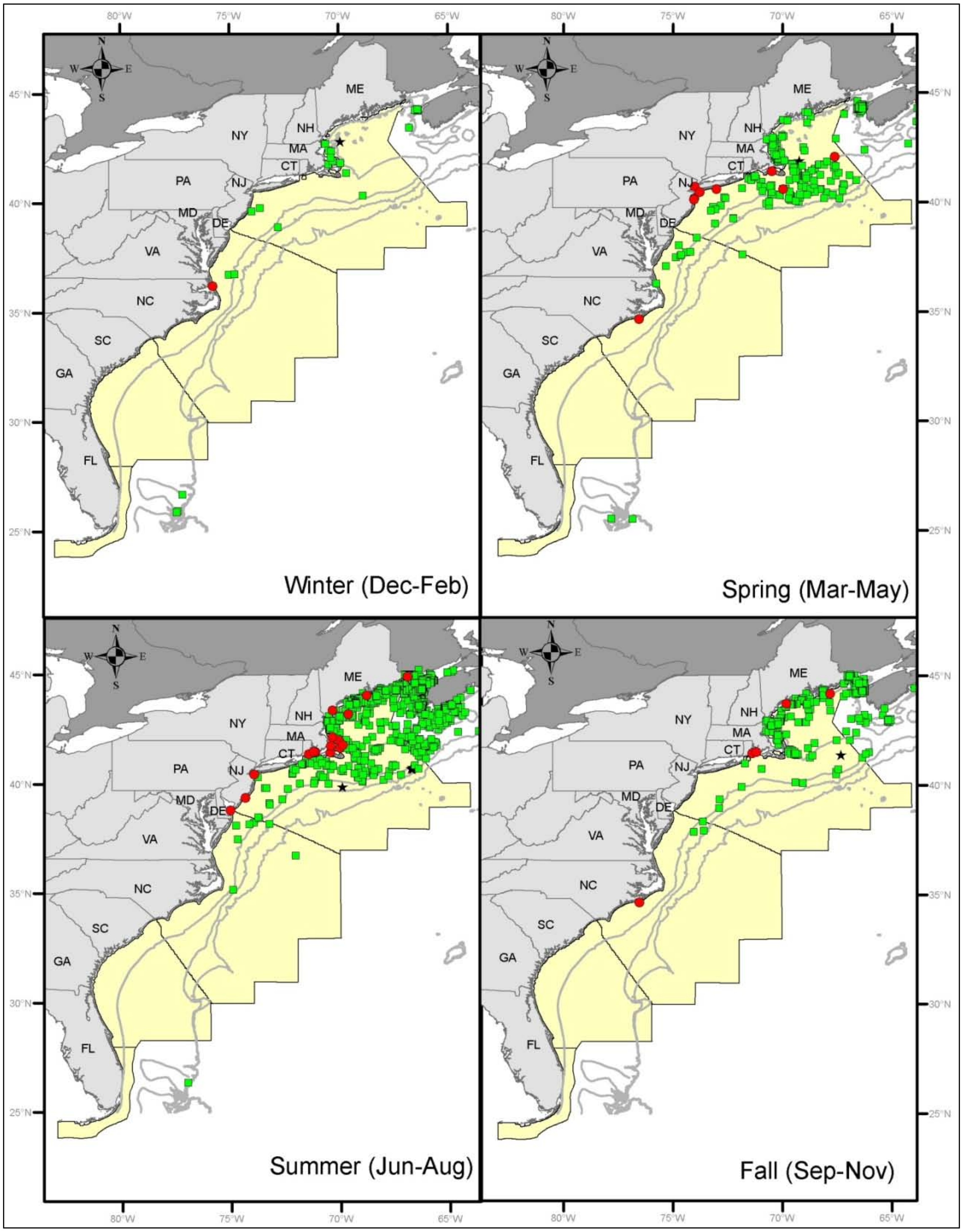


Figure 2.5.1. Minke whale sighting (green squares), stranding (red dots) and fishery by-catch (black stars) observations by season.

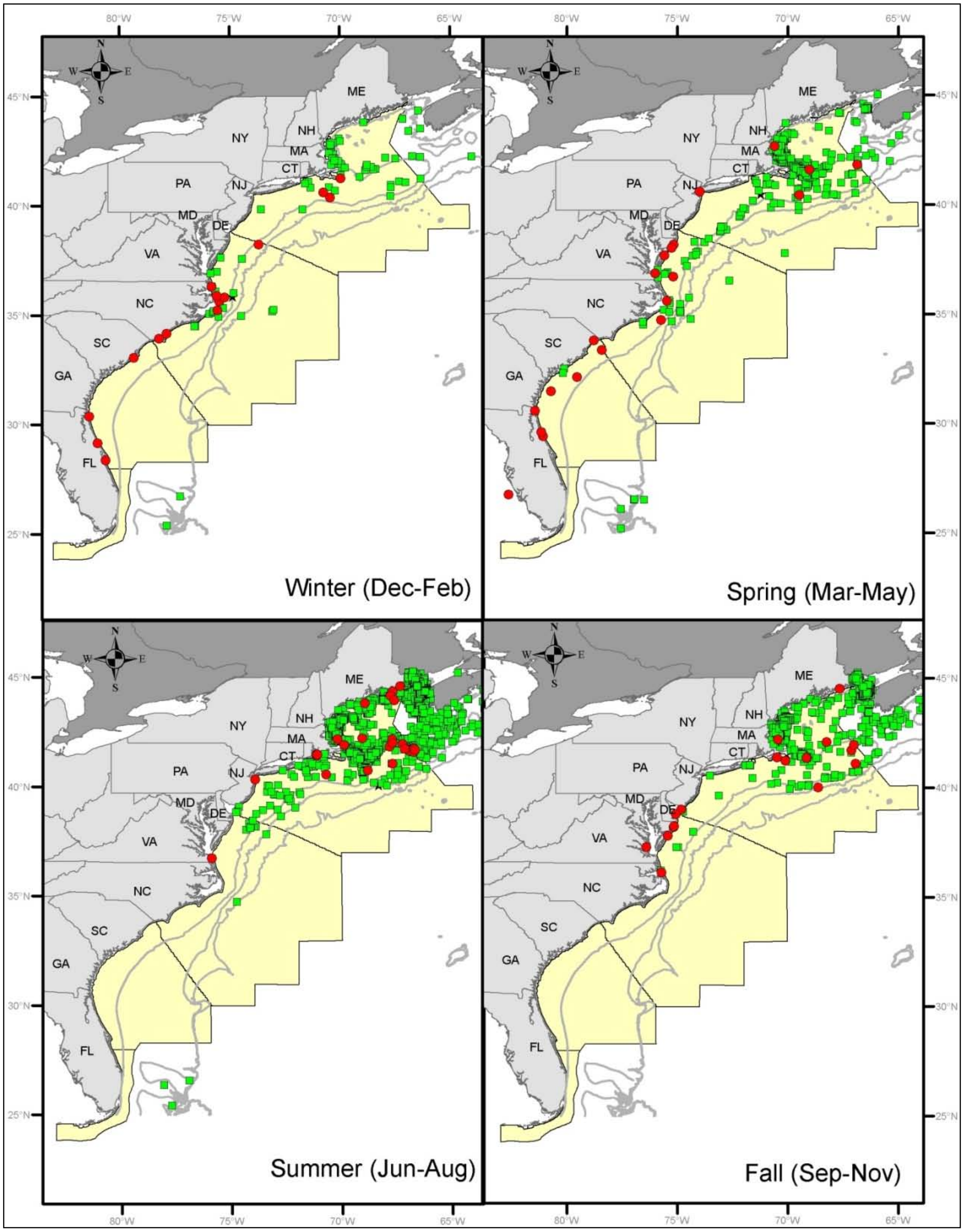


Figure 2.6.1. Humpback whale sighting (green squares) and stranding (red dots) observations by season.

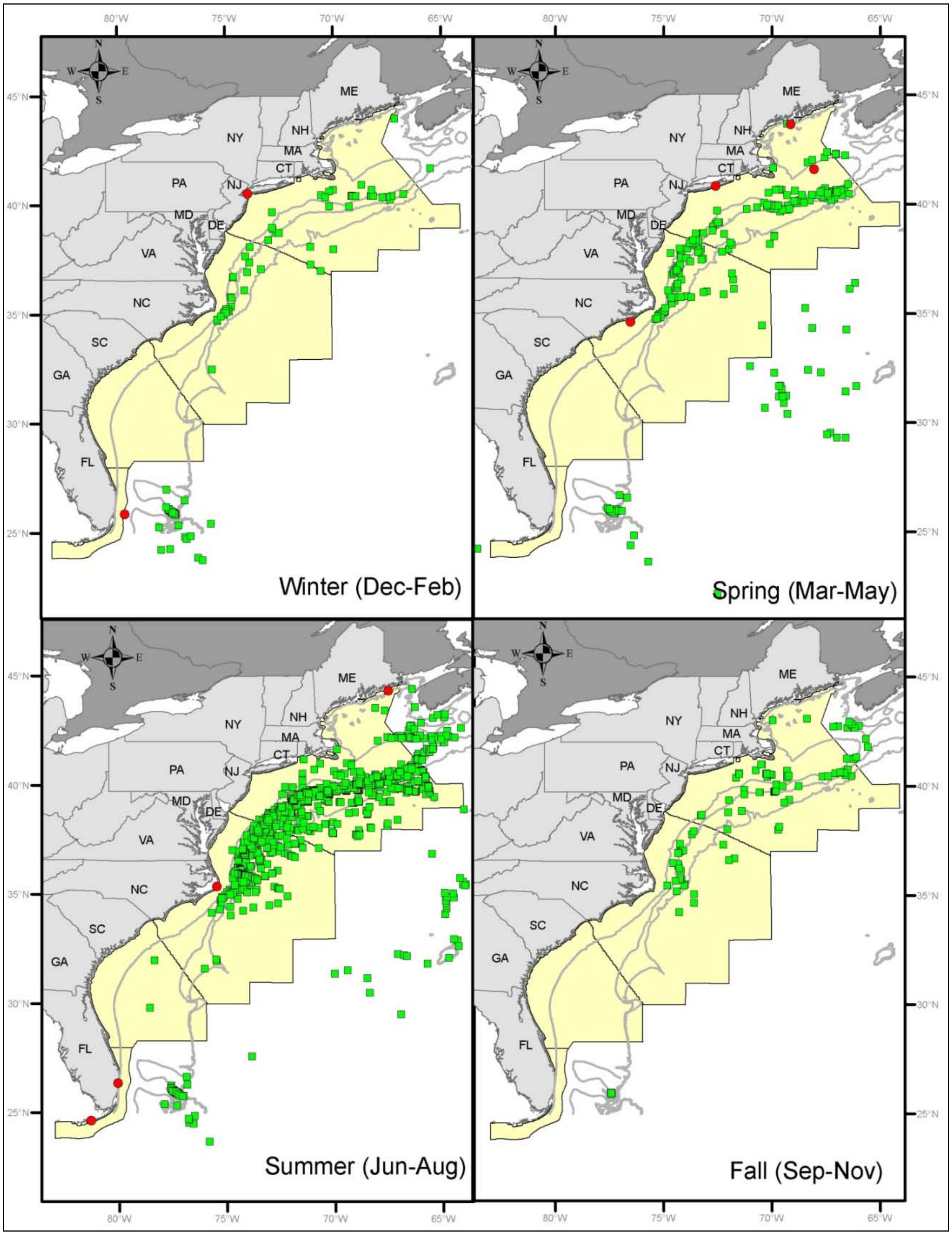


Figure 2.7.1. Sperm whale sighting (green squares) and stranding (red dots) observations by season.

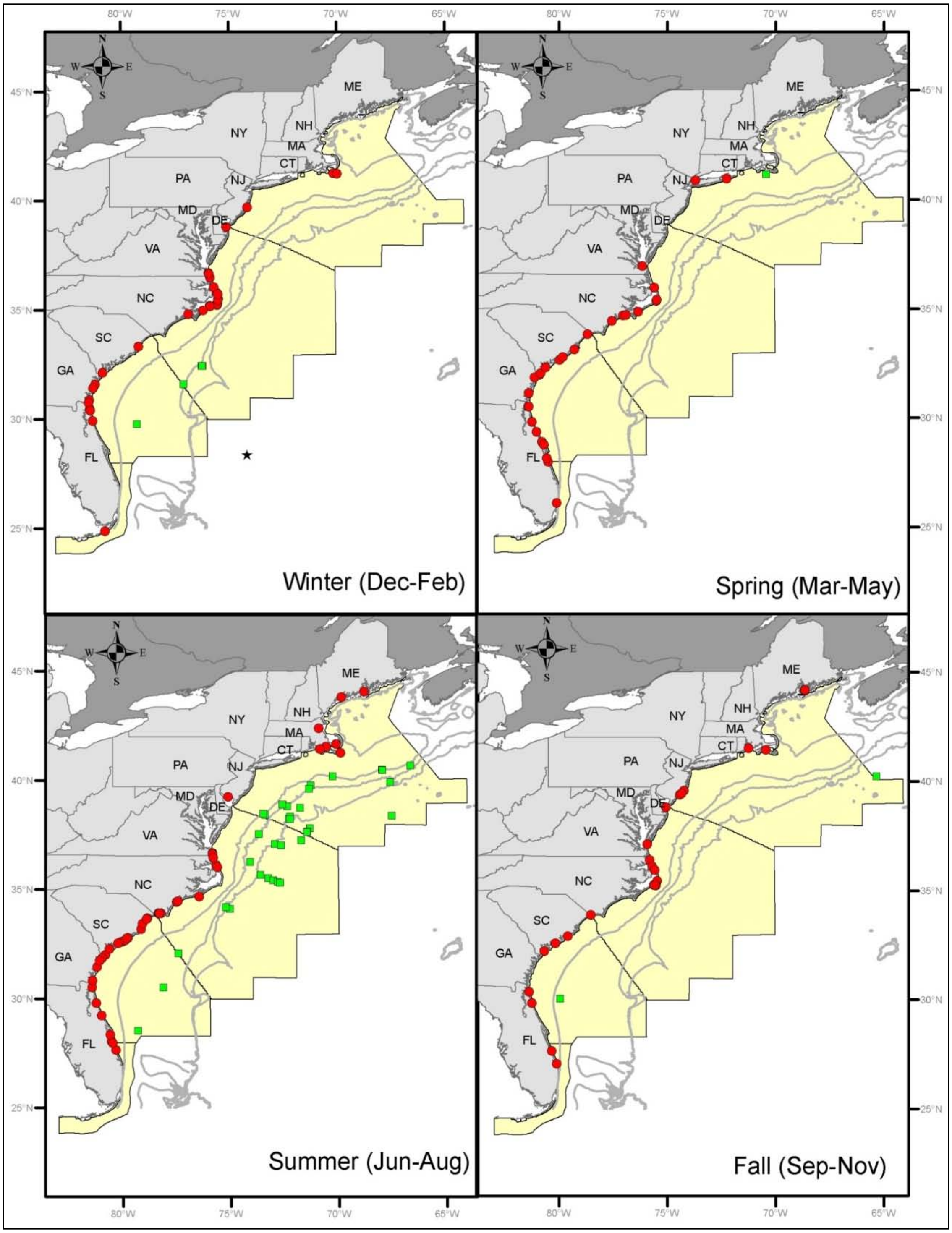


Figure 2.8.1. Dwarf and pygmy sperm whale sighting (green squares), stranding (red dots) and fishery and fishery by-catch (black stars) observations by season.

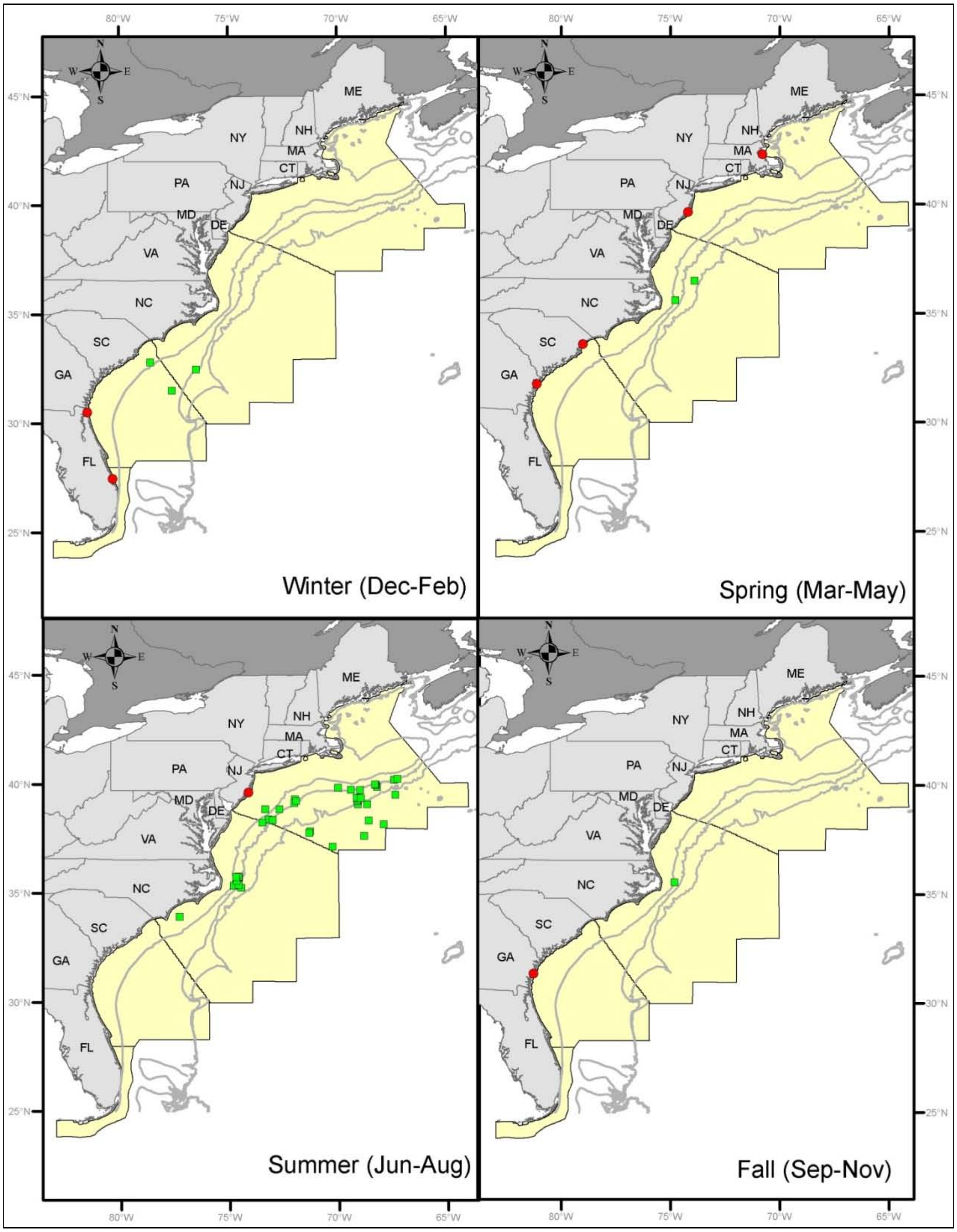


Figure 2.9.1. Cuvier's beaked whale sighting (green squares) and stranding (red dots) observations by season.

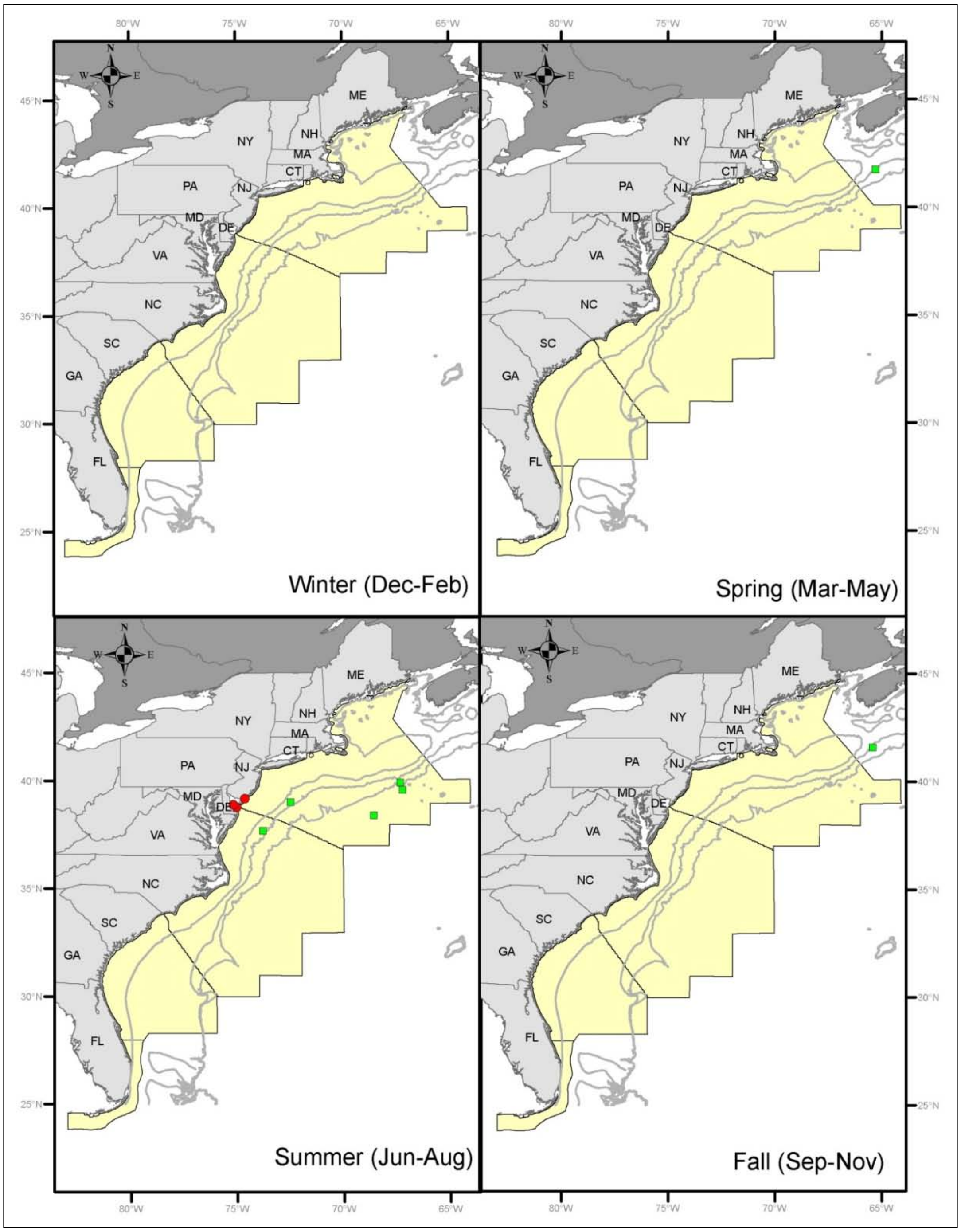


Figure 2.10.1. Northern bottlenose whale sighting (green squares) and stranding (red dots) by season.

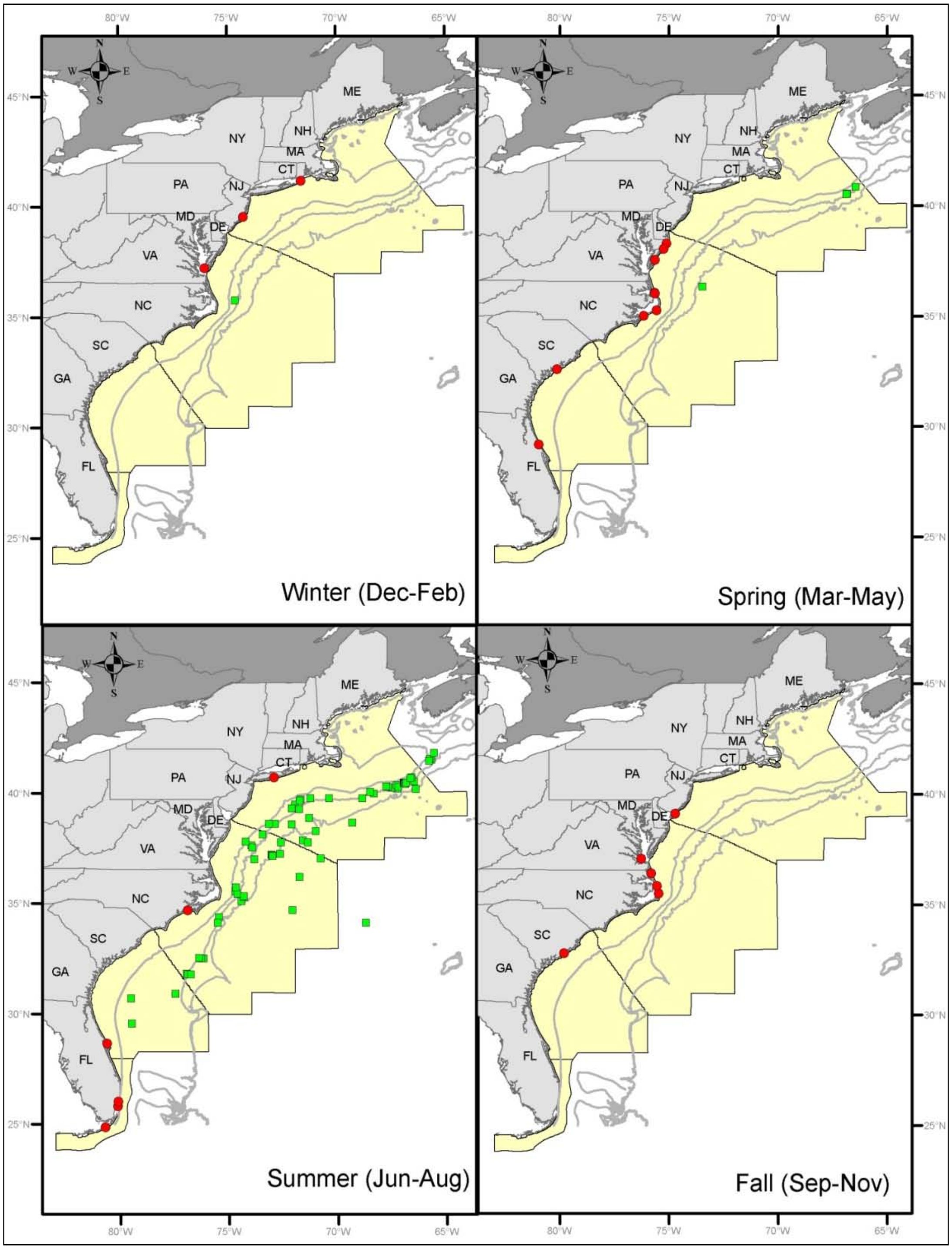


Figure 2.11.1. Mesoplodon beaked whale sighting (green squares), stranding (red dots) and fishery by-catch (black stars) observations by season.

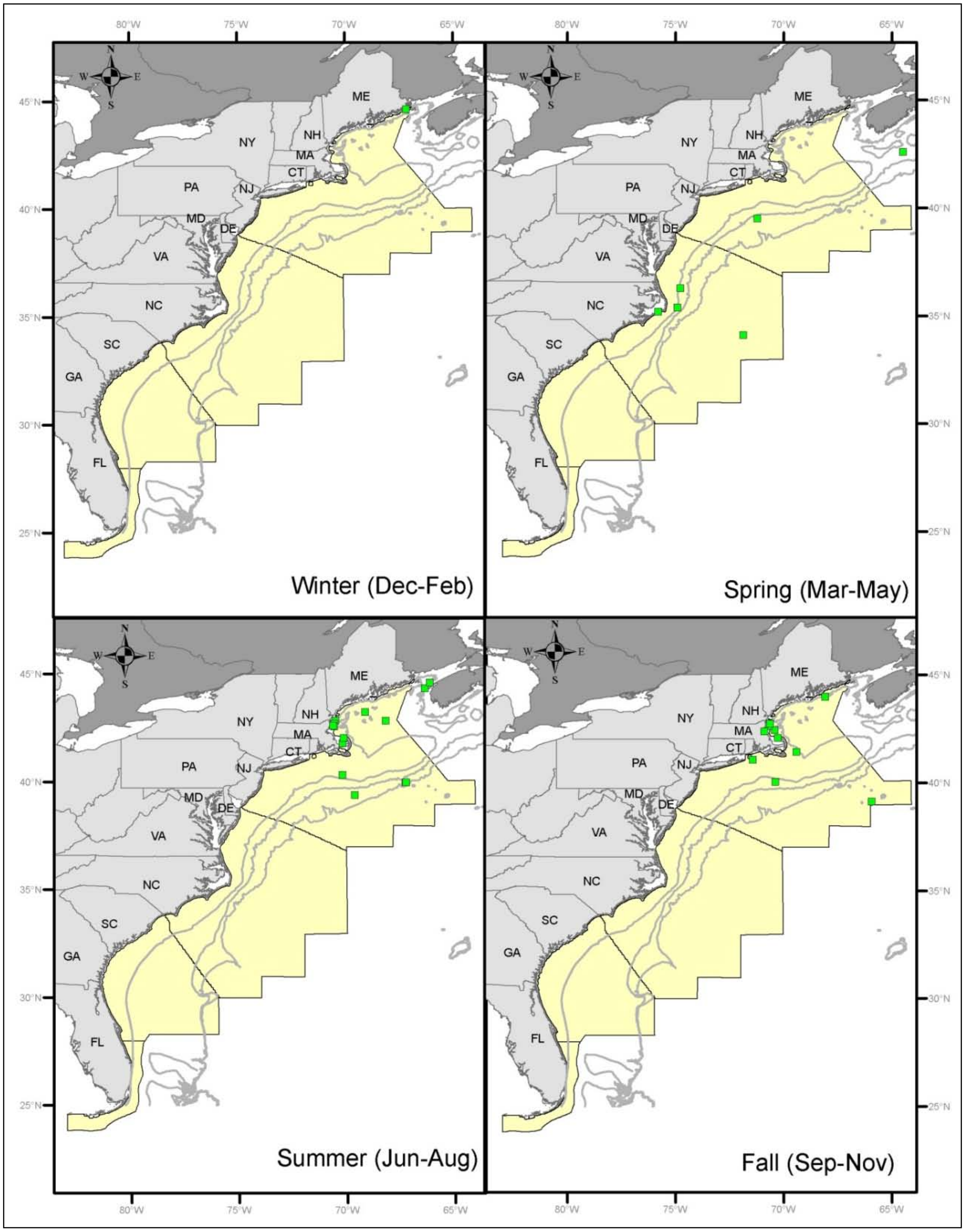


Figure 2.12.1. Killer whale sighting (green squares) observations by season.

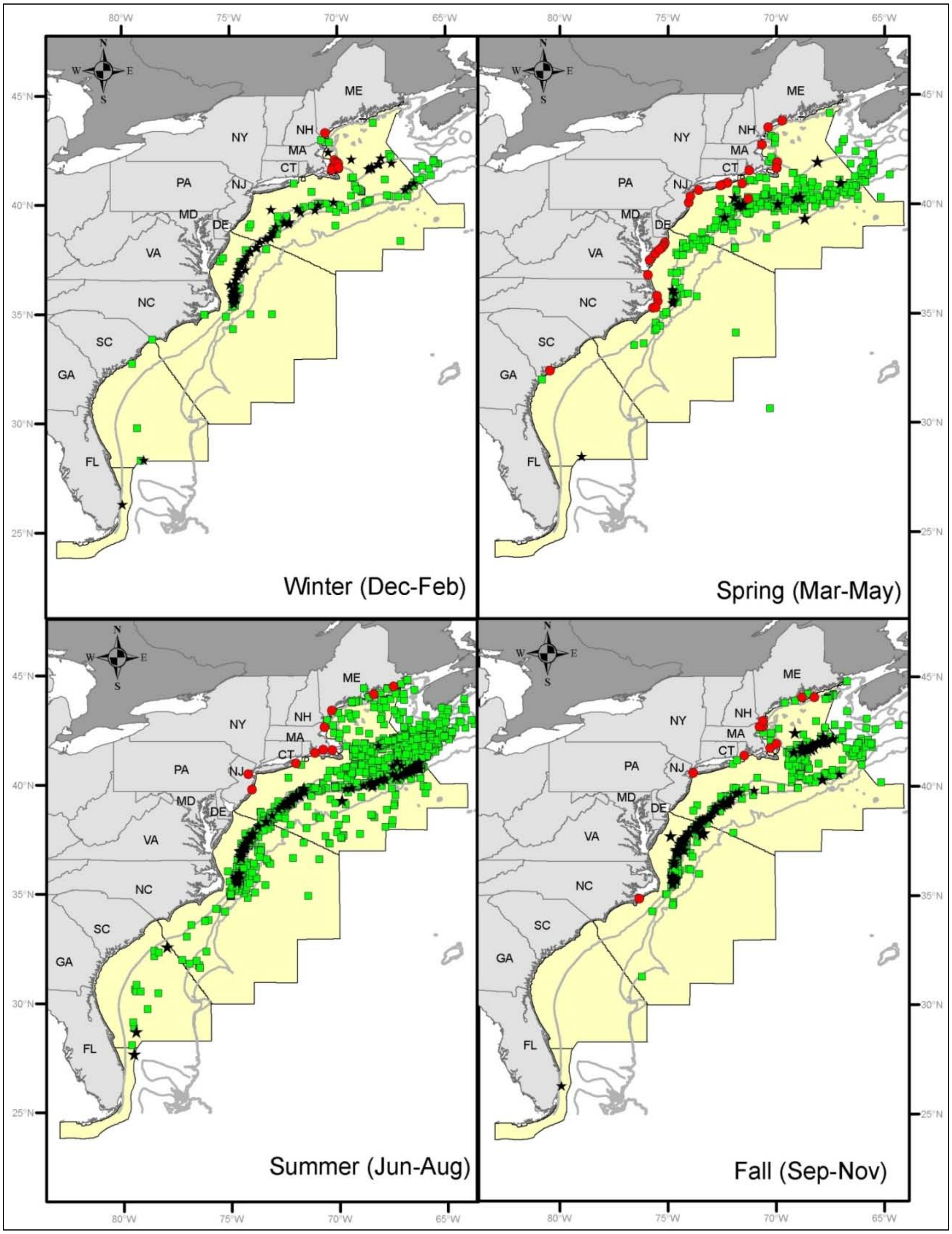


Figure 2.13.1. Long-finned and undifferentiated pilot whale sighting (green squares), stranding (red dots) and fishery by-catch (black stars) observations by season.

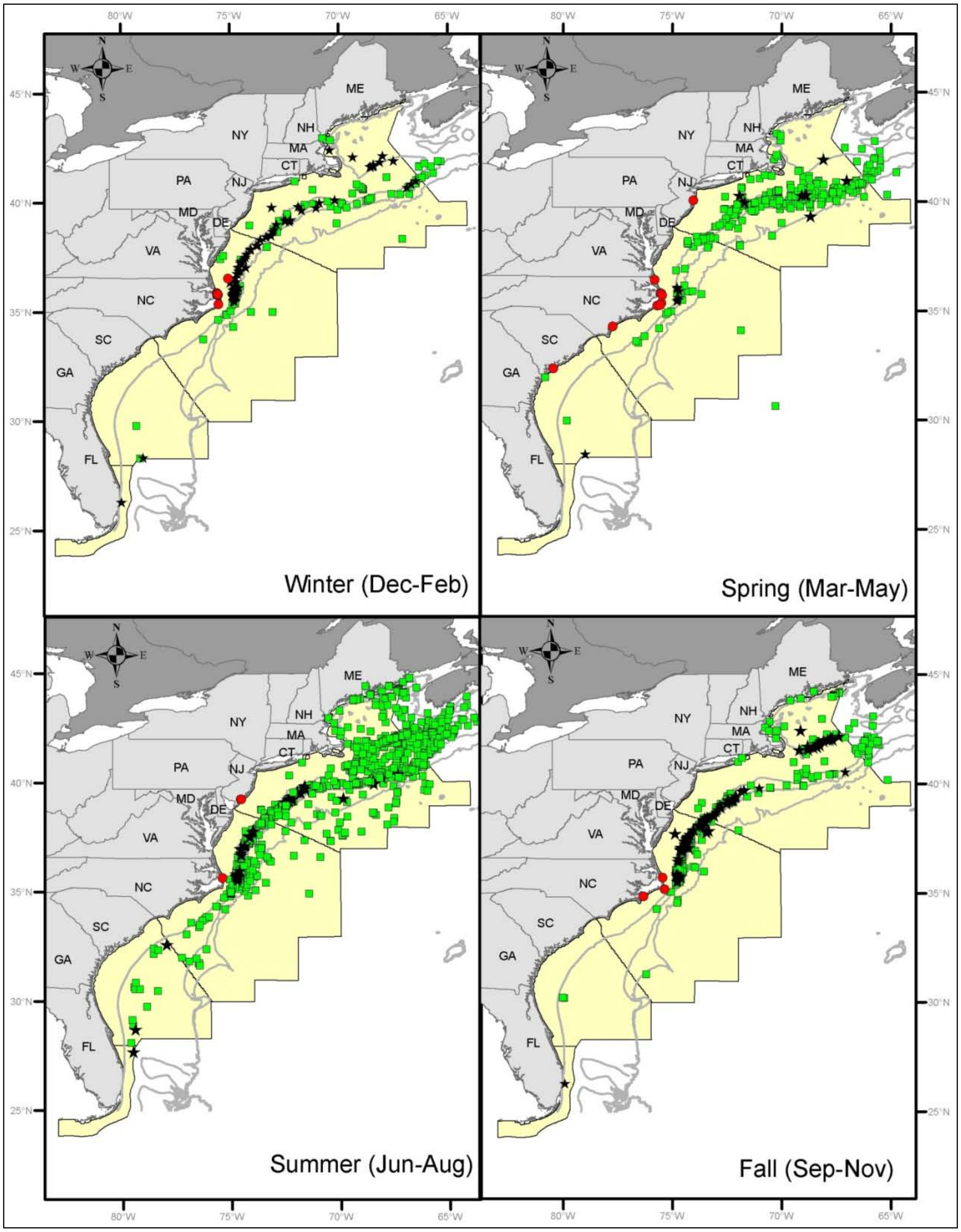


Figure 2.14.1. Short-finned and undifferentiated pilot whale sighting (green squares), stranding (red dots) and fishery by-catch (black stars) observations by season.

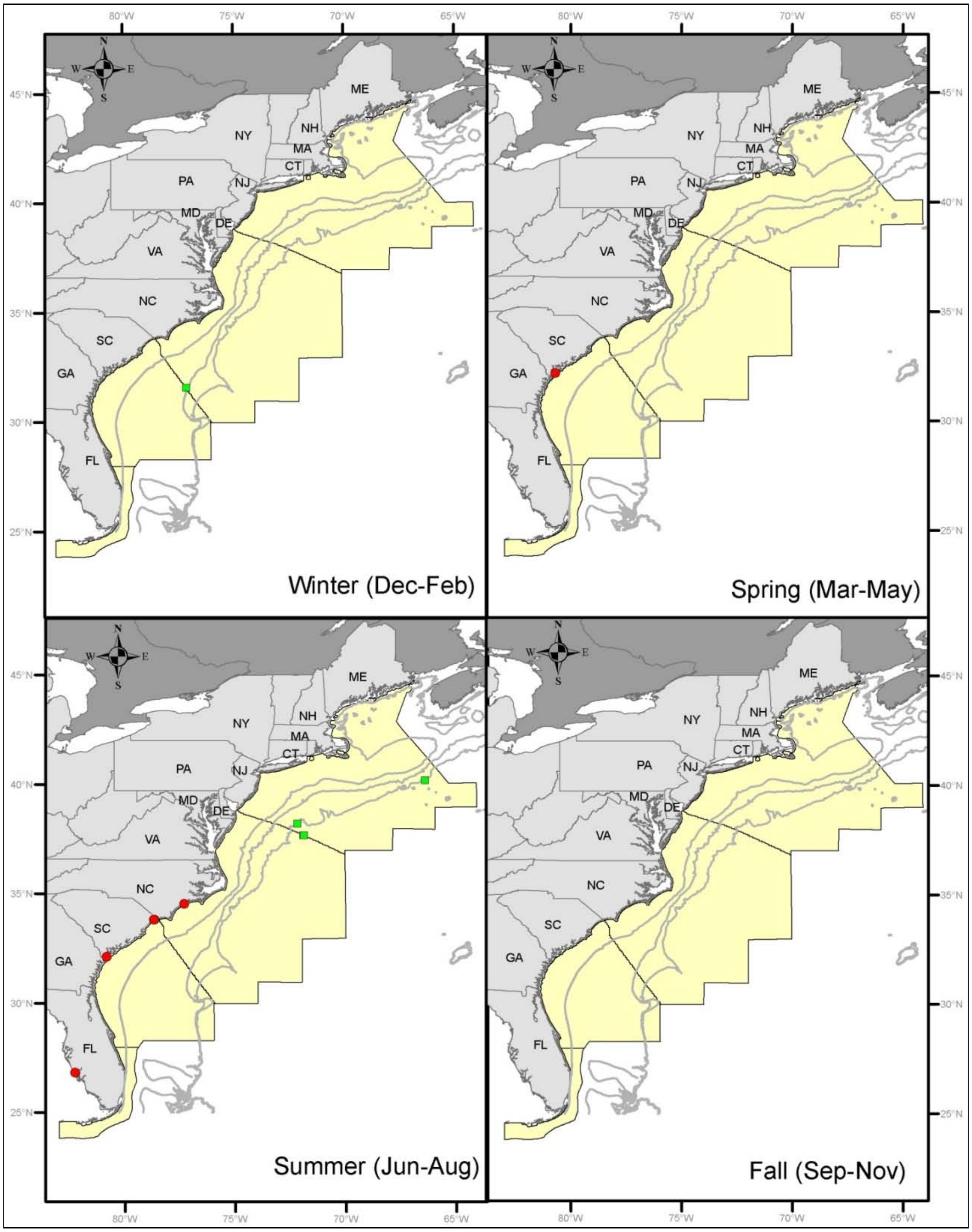


Figure 2.15.1. Pygmy killer whale sighting (green squares) and stranding (red dots) observations by season.

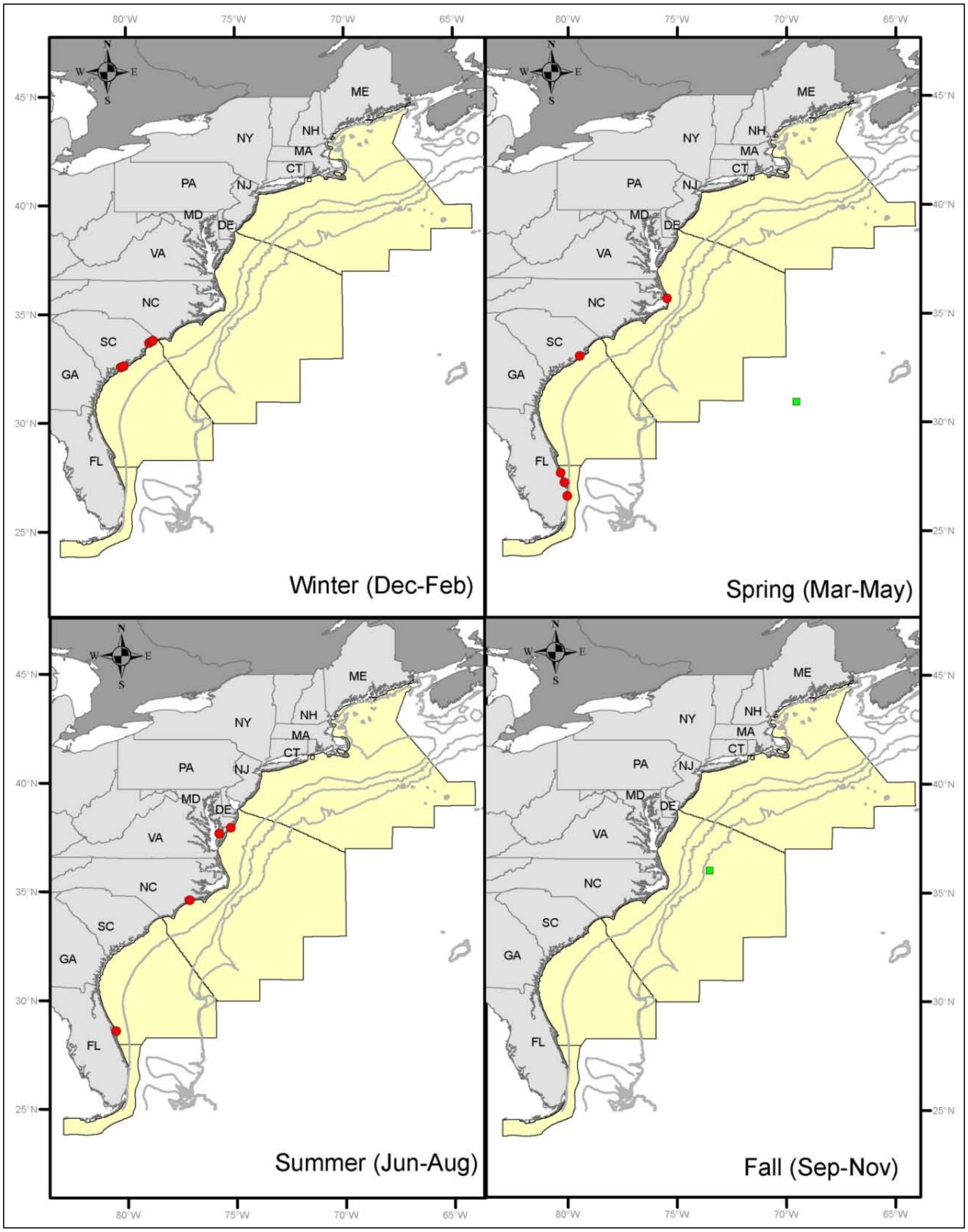


Figure 2.16.1. Melon-headed whale sighting (green squares) and stranding (red dots) observations by season.

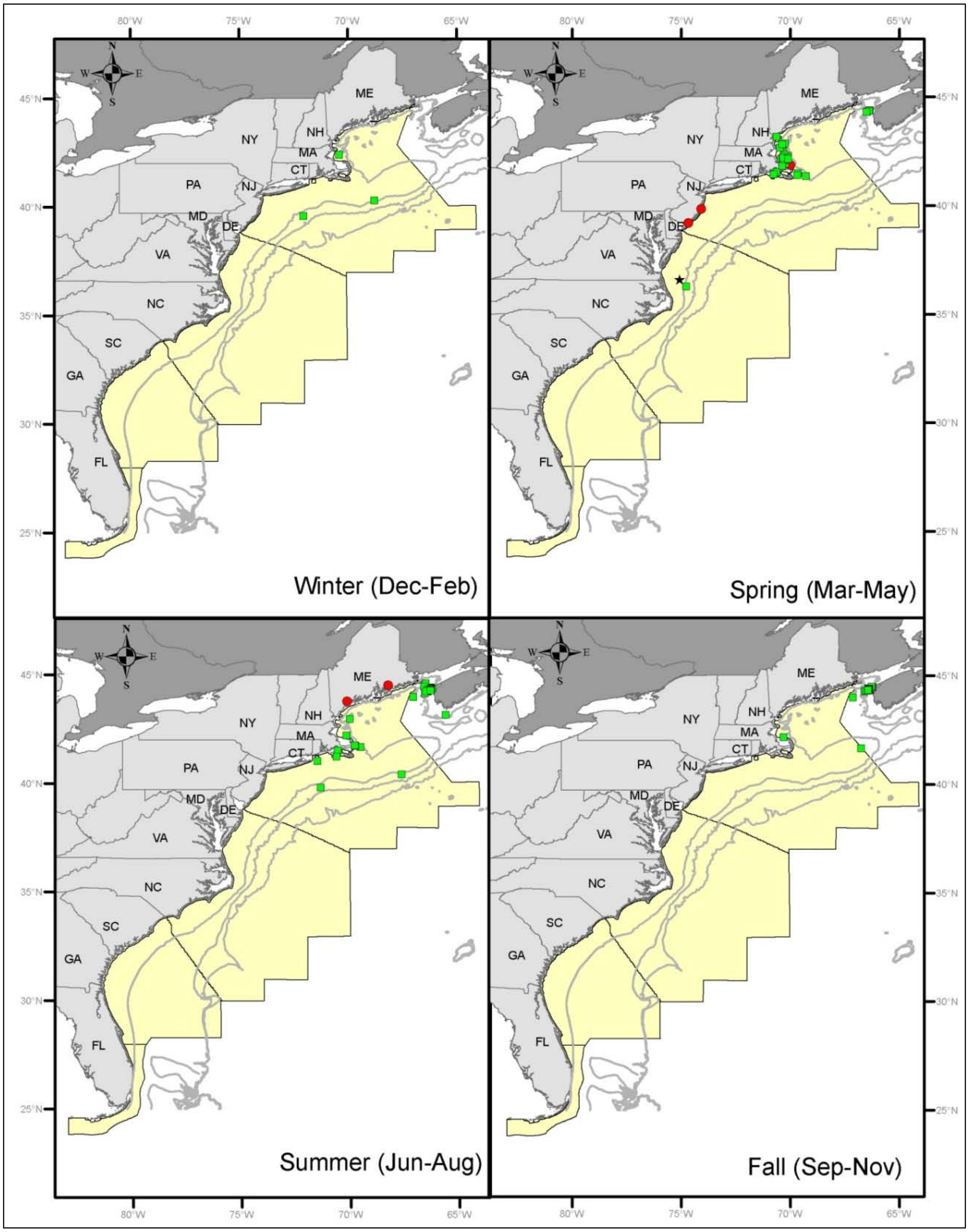


Figure 2.17.1. White-beaked dolphin sighting (green squares), stranding (red dots) and fishery by-catch (black stars) observations by season.

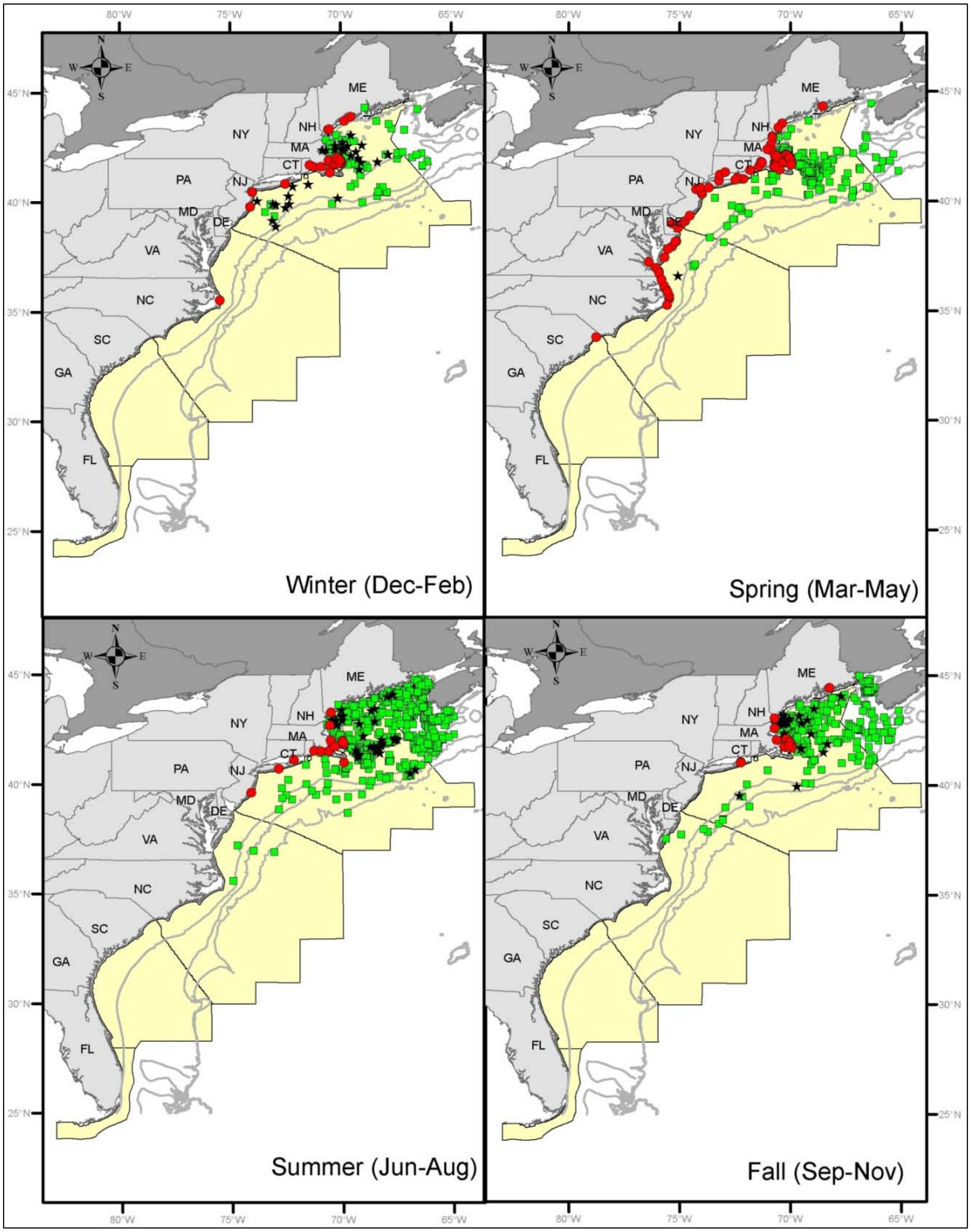


Figure 2.18.1. White-sided dolphin sighting (green squares), stranding (red dots) and fishery by-catch (black stars) observations by season.

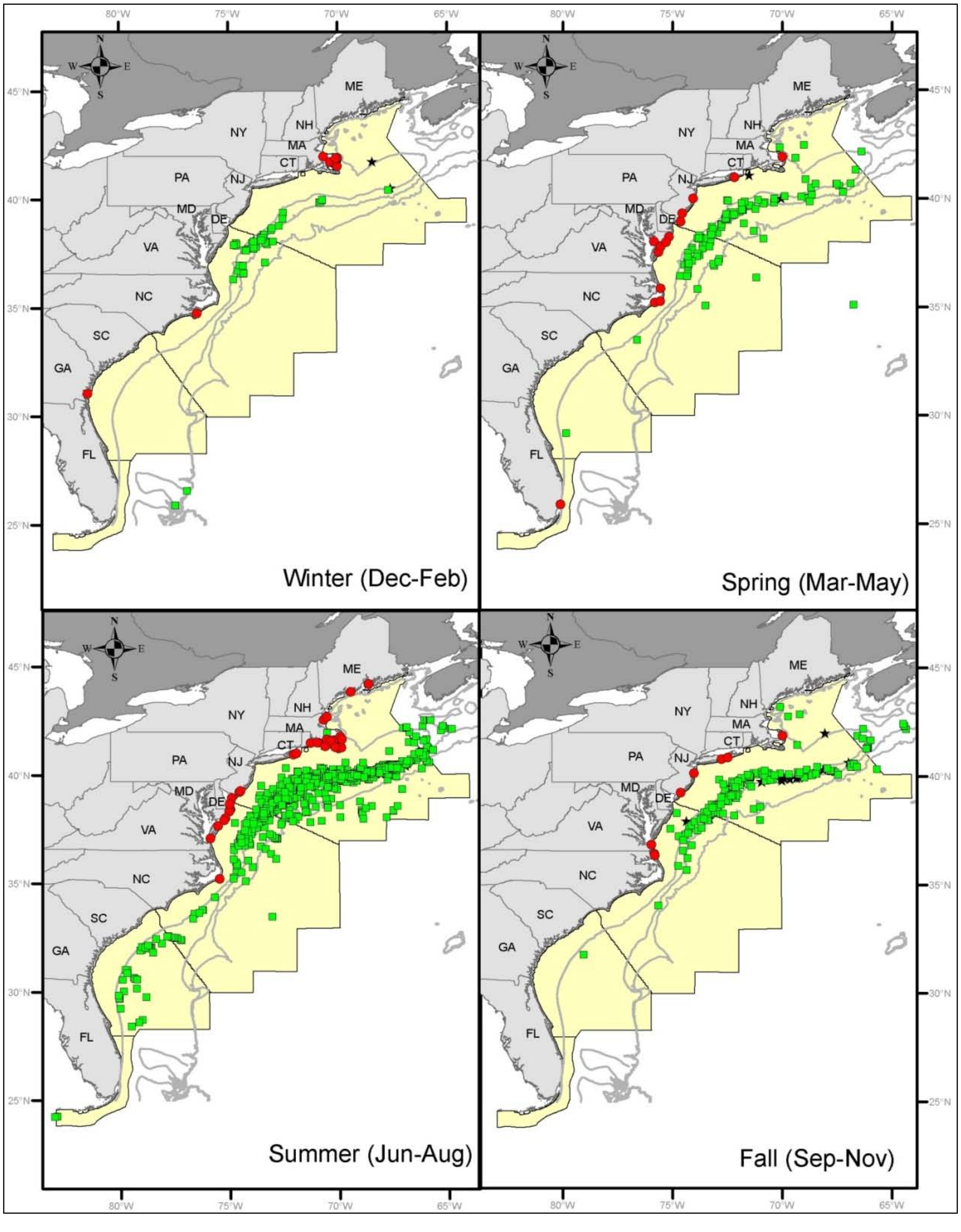


Figure 2.19.1. Risso's dolphin sighting (green squares), stranding (red dots) and fishery by-catch (black stars) observations by season.

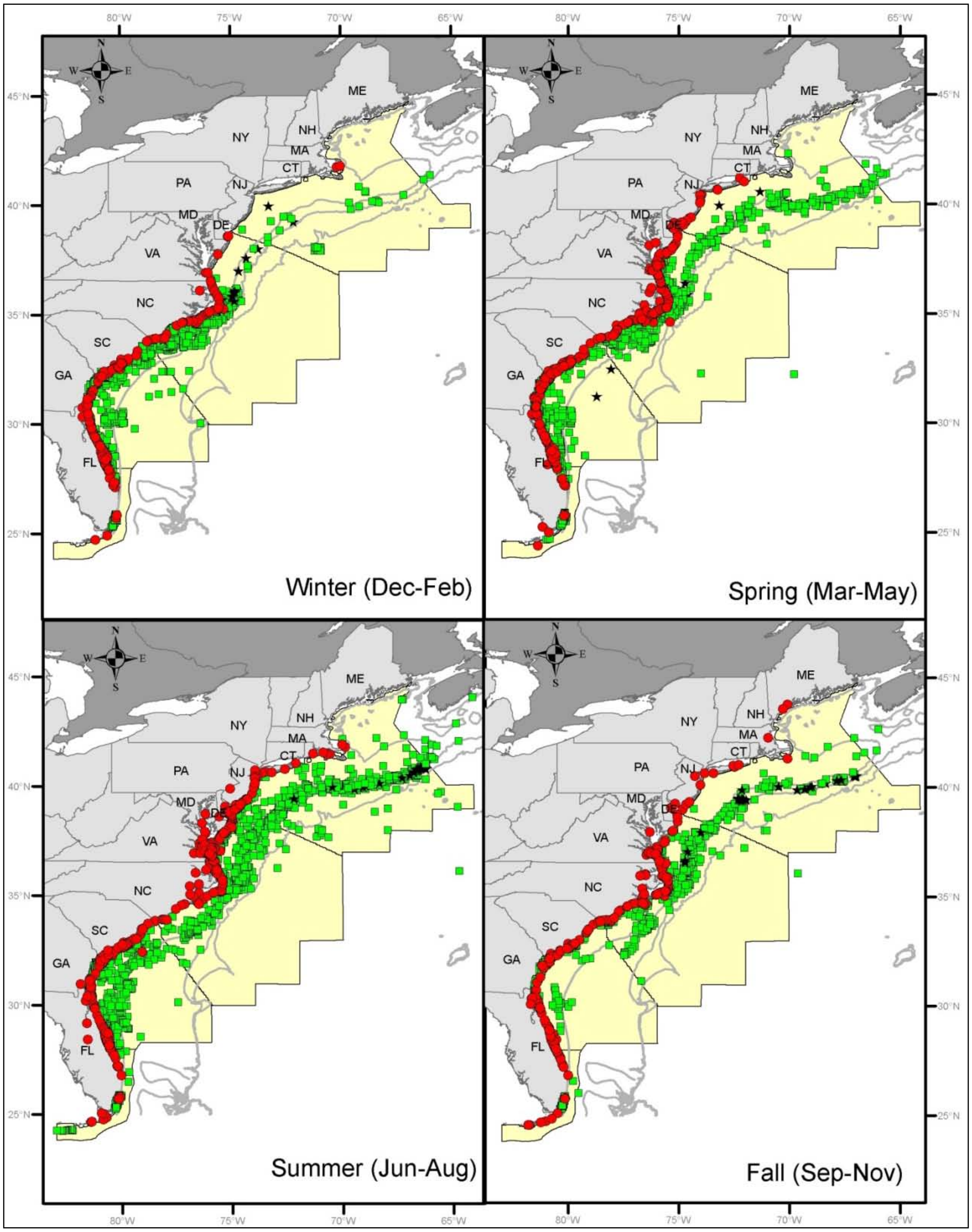


Figure 2.20.2. Bottlenose dolphin sighting (green squares), stranding (red dots) and fishery by-catch (black stars) observations by season.

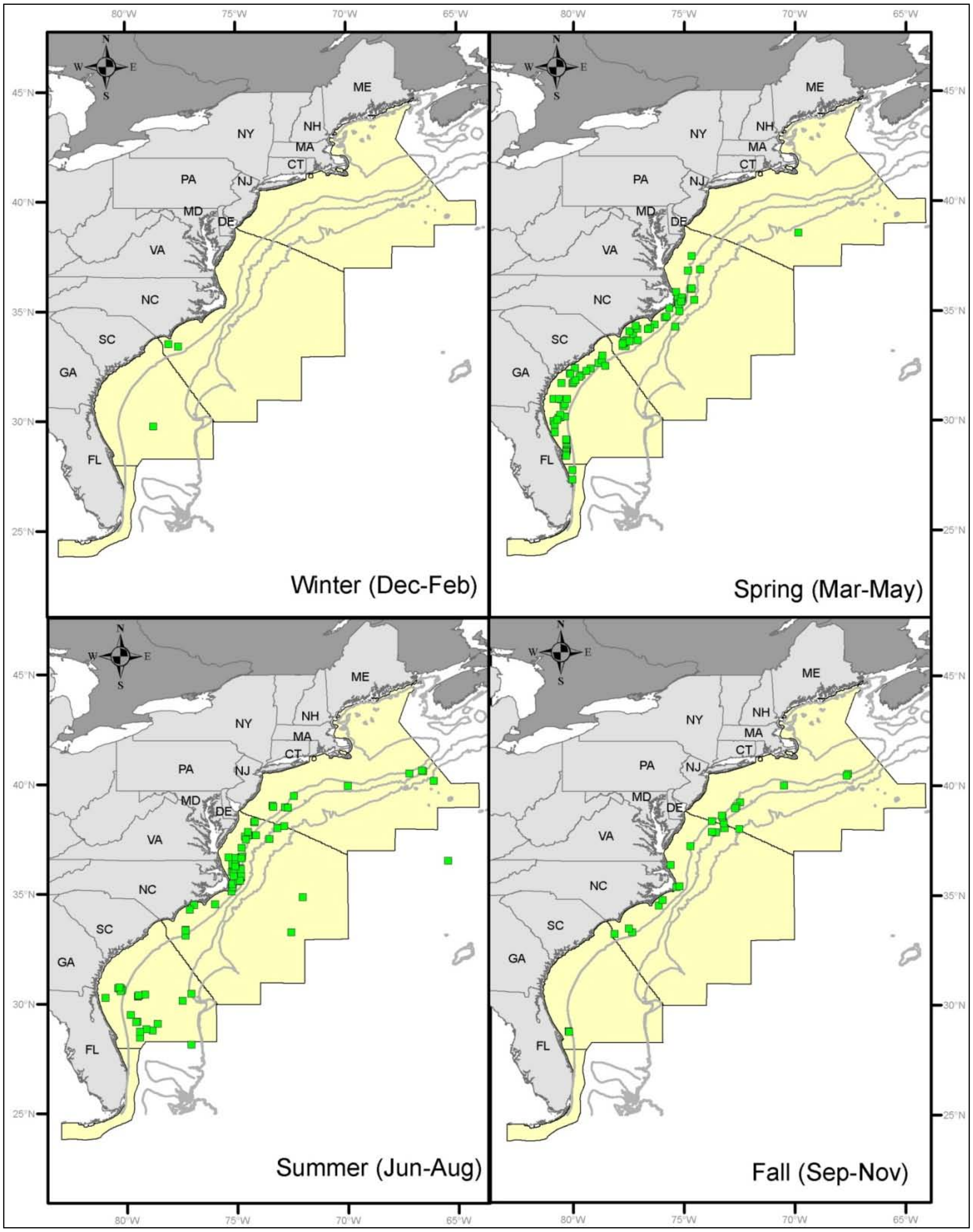


Figure 2.21.1. Pantropical spotted dolphin sighting (green squares), stranding (red dots) and fishery by-catch (black stars) observations by season.

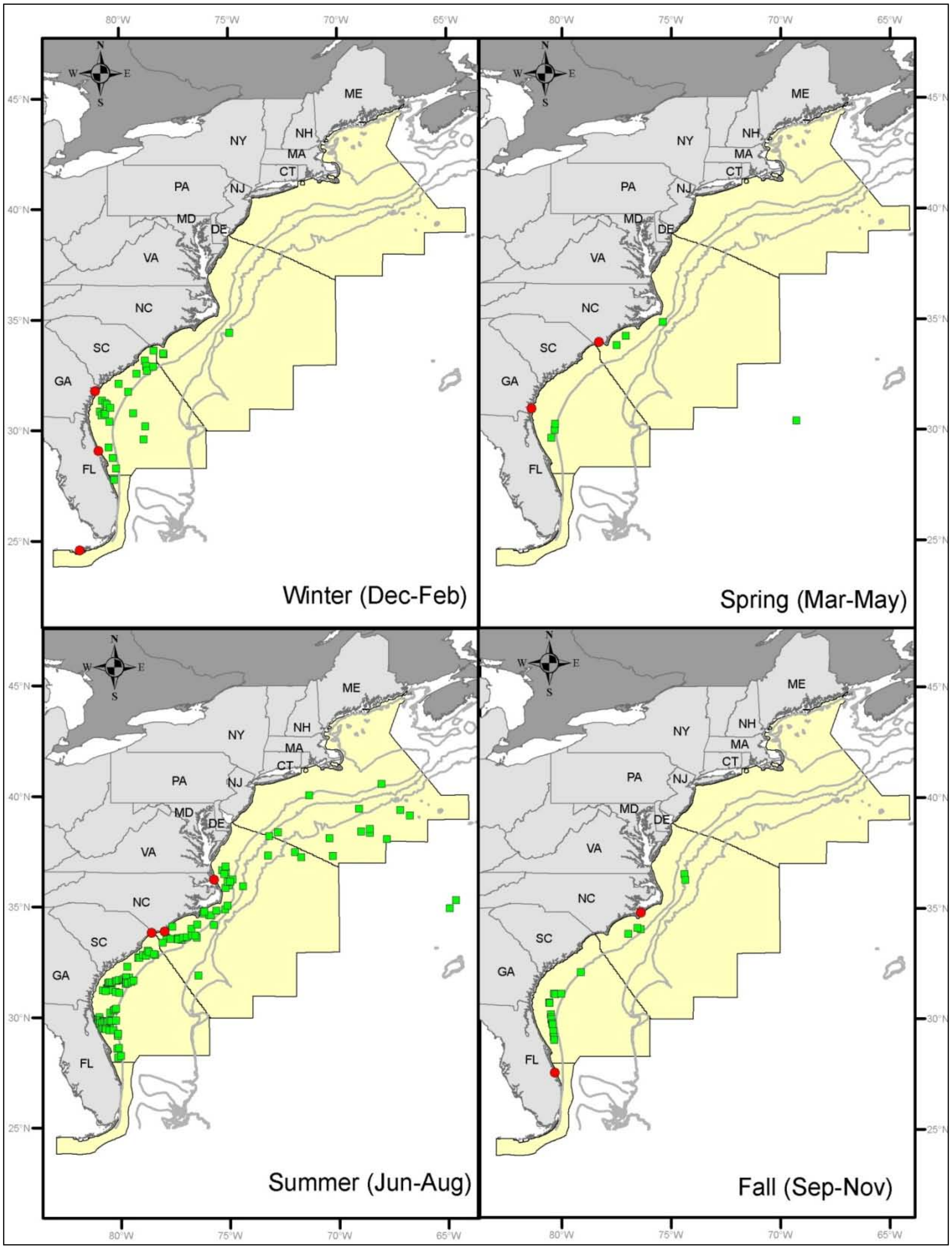


Figure 2.22.1. Atlantic spotted dolphin sighting (green squares) and stranding (red dots) observations by season.

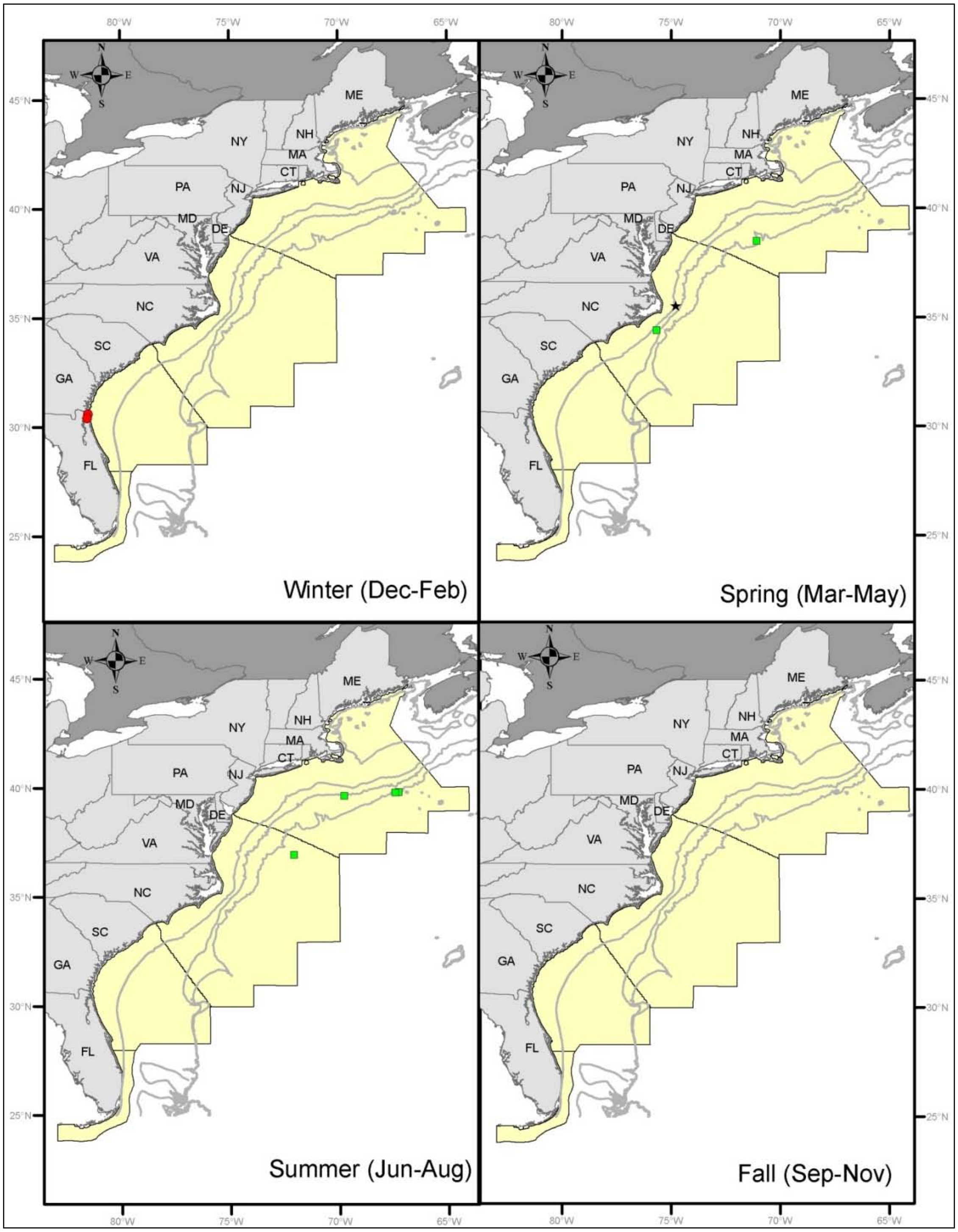


Figure 2.23.1. Spinner dolphin sighting (green squares), stranding (red dots) and fishery by-catch (black stars) observations by season.

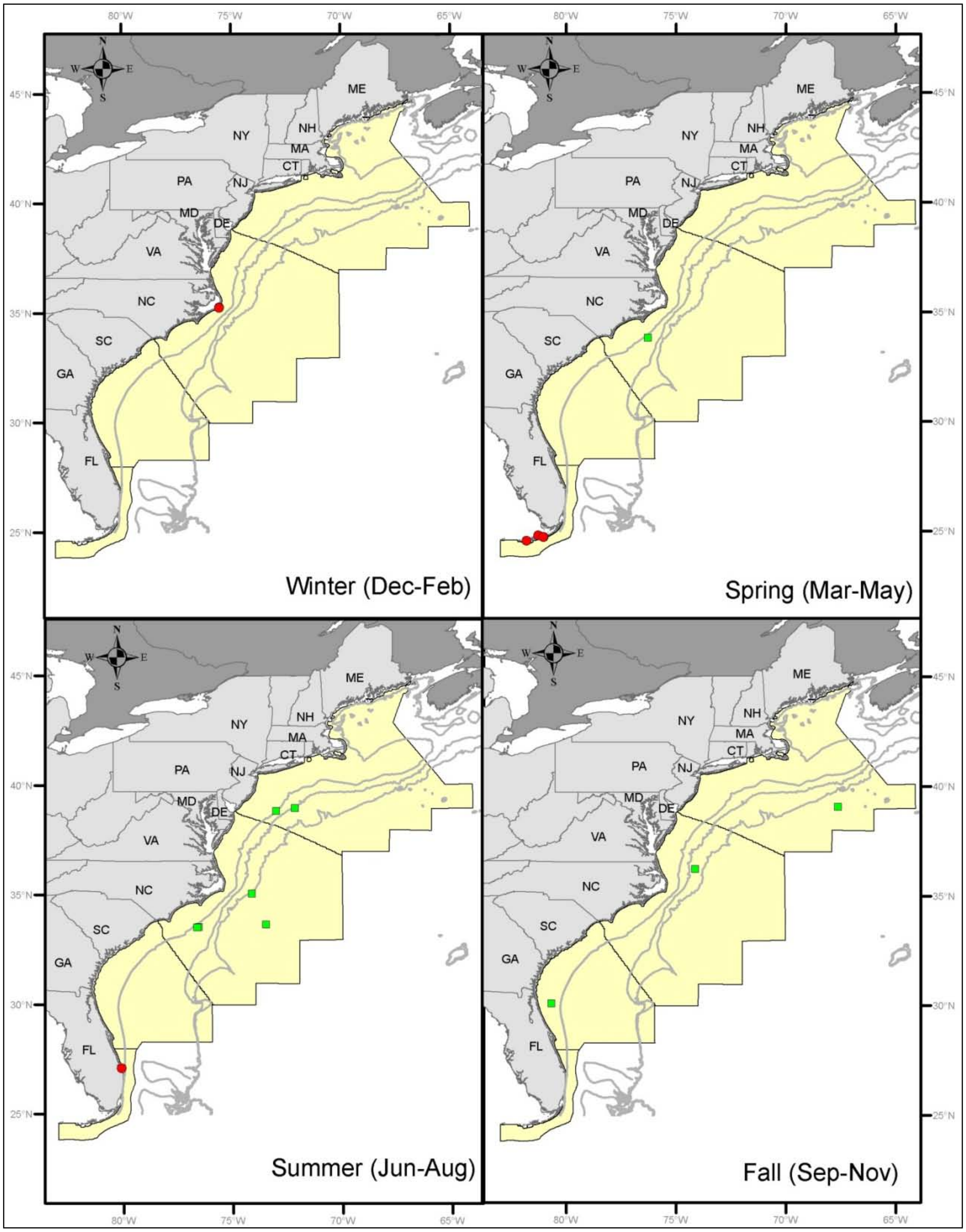


Figure 2.24.1. Rough-toothed dolphin sighting (green squares) and stranding (red dots) observations by season.

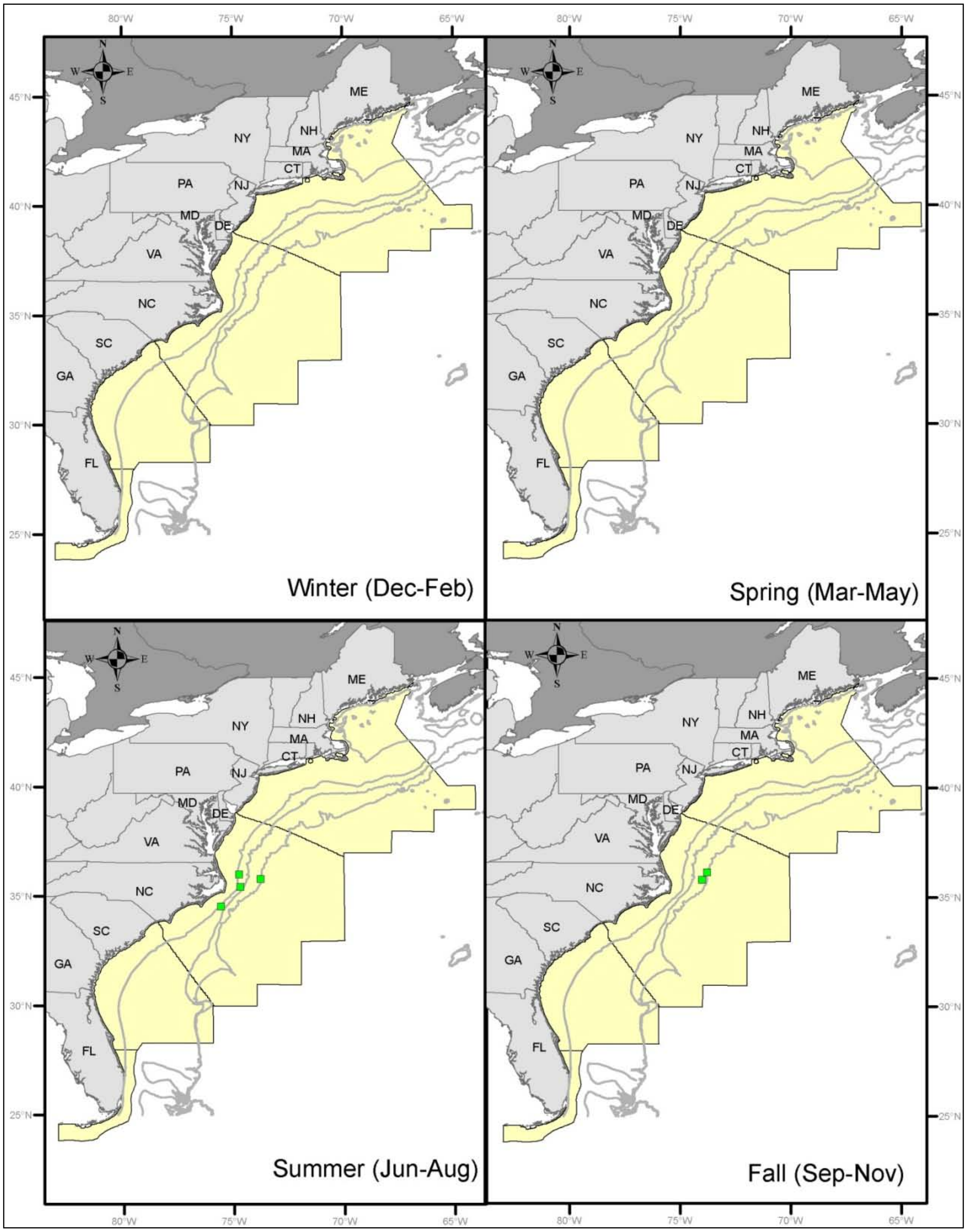


Figure 2.25.1. Clymene dolphin sighting (green squares), stranding (red dots) and fishery by-catch (black stars) observations by season.

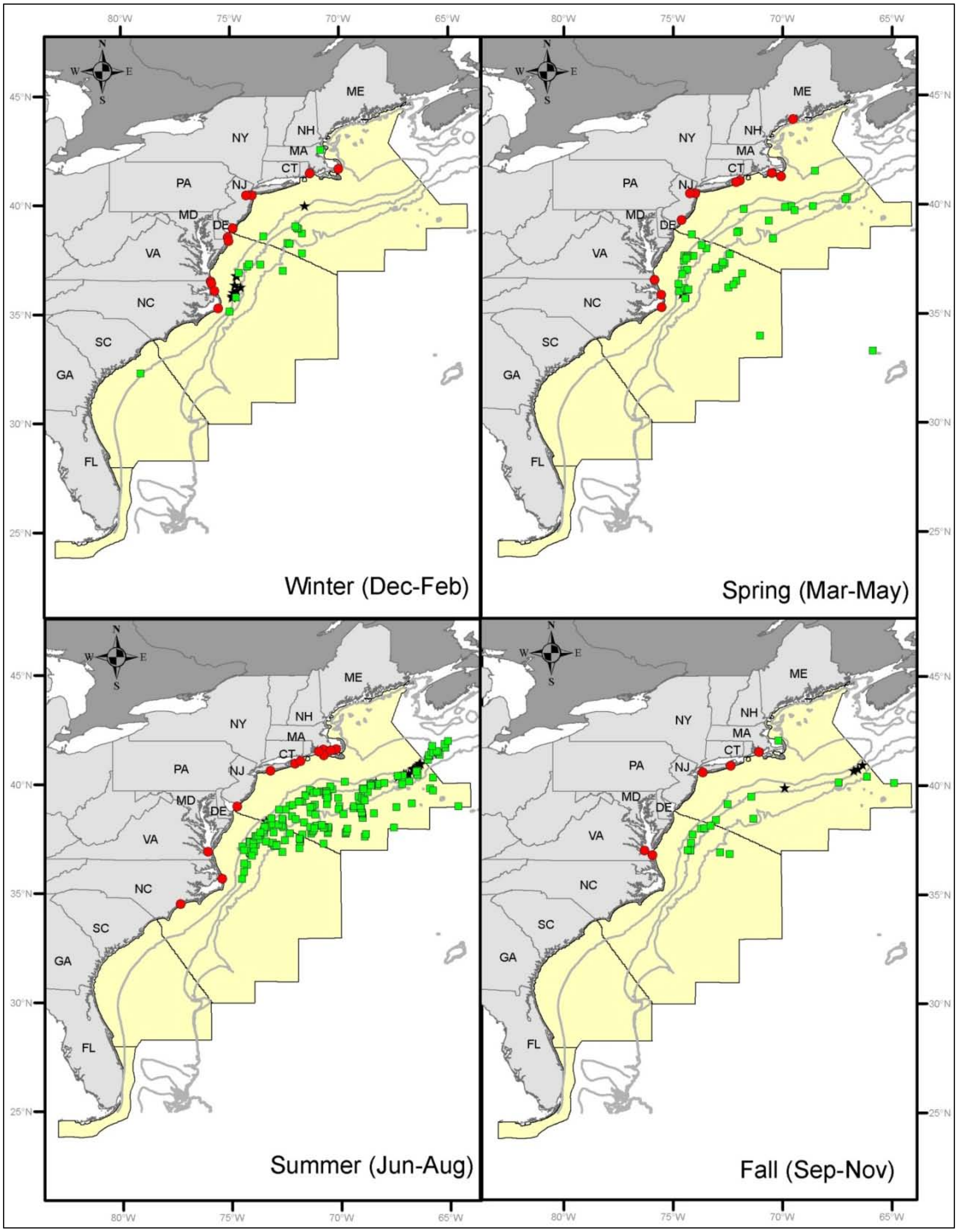


Figure 2.26.1. Striped dolphin sighting (green squares), stranding (red dots) and fishery by-catch (black stars) observations by season.

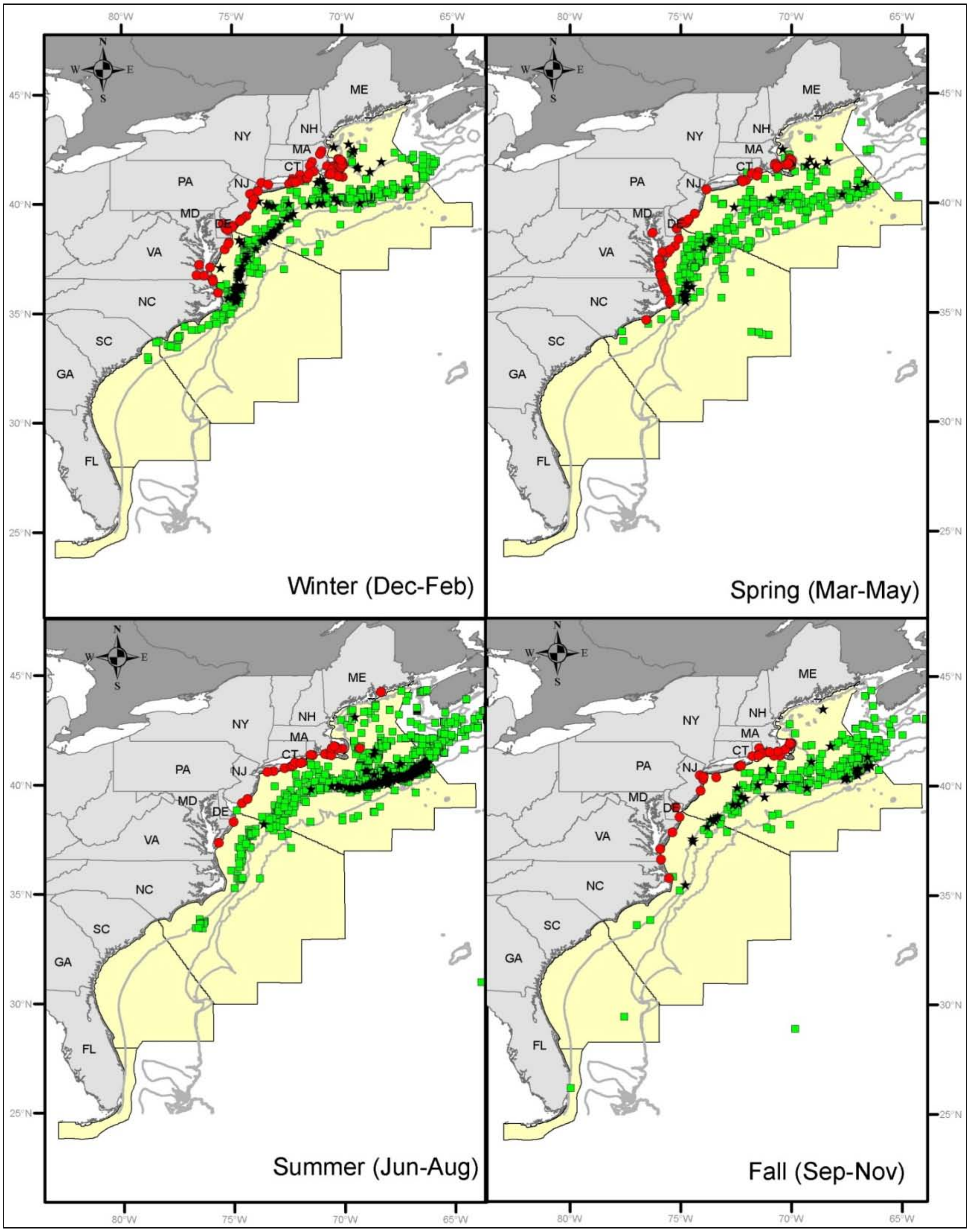


Figure 2.27.1. Common dolphin sighting (green squares), stranding (red dots) and fishery by-catch (black stars) observations by season.

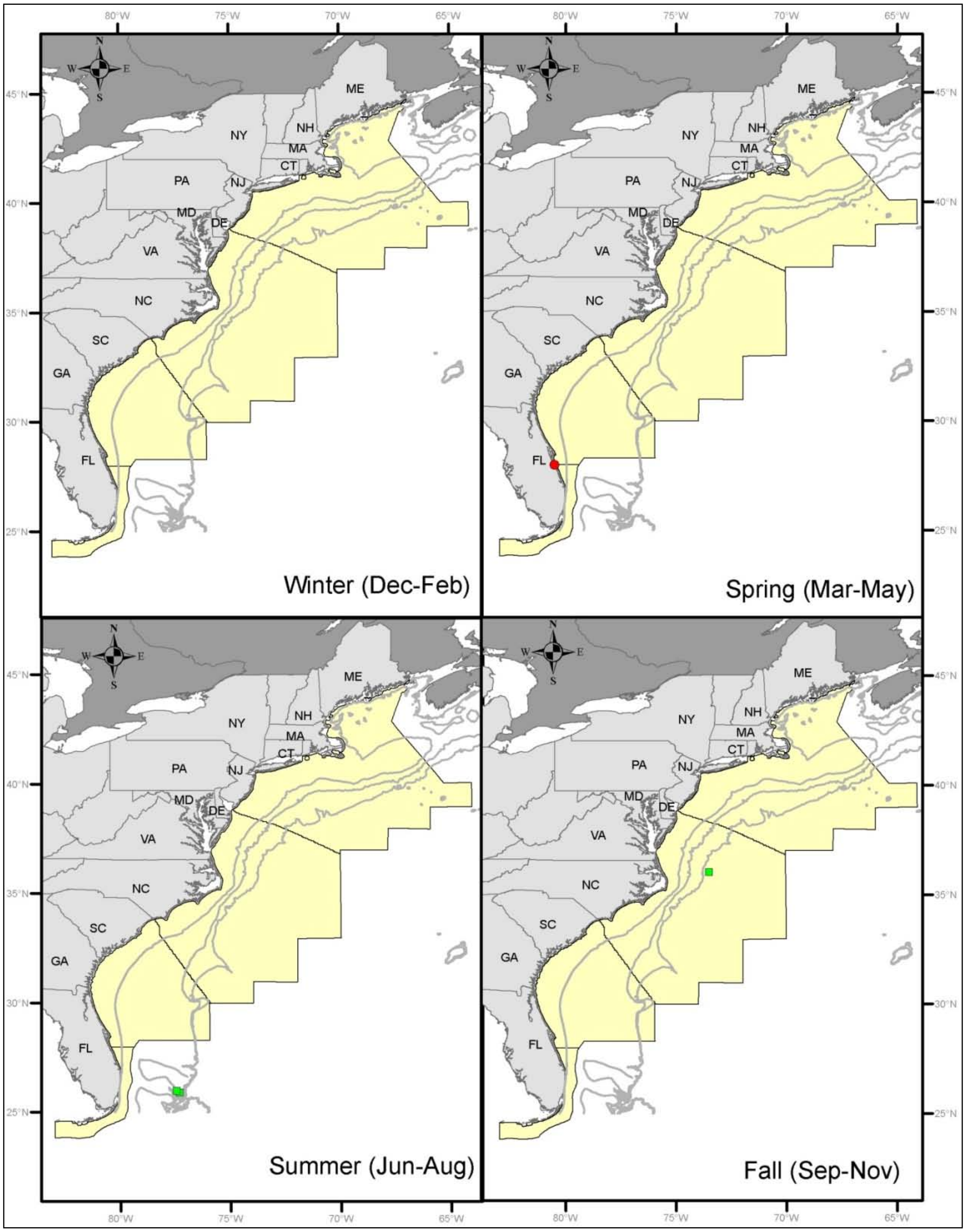


Figure 2.28.2. Fraser's dolphin sighting (green squares) and stranding (red dots) observations by season.

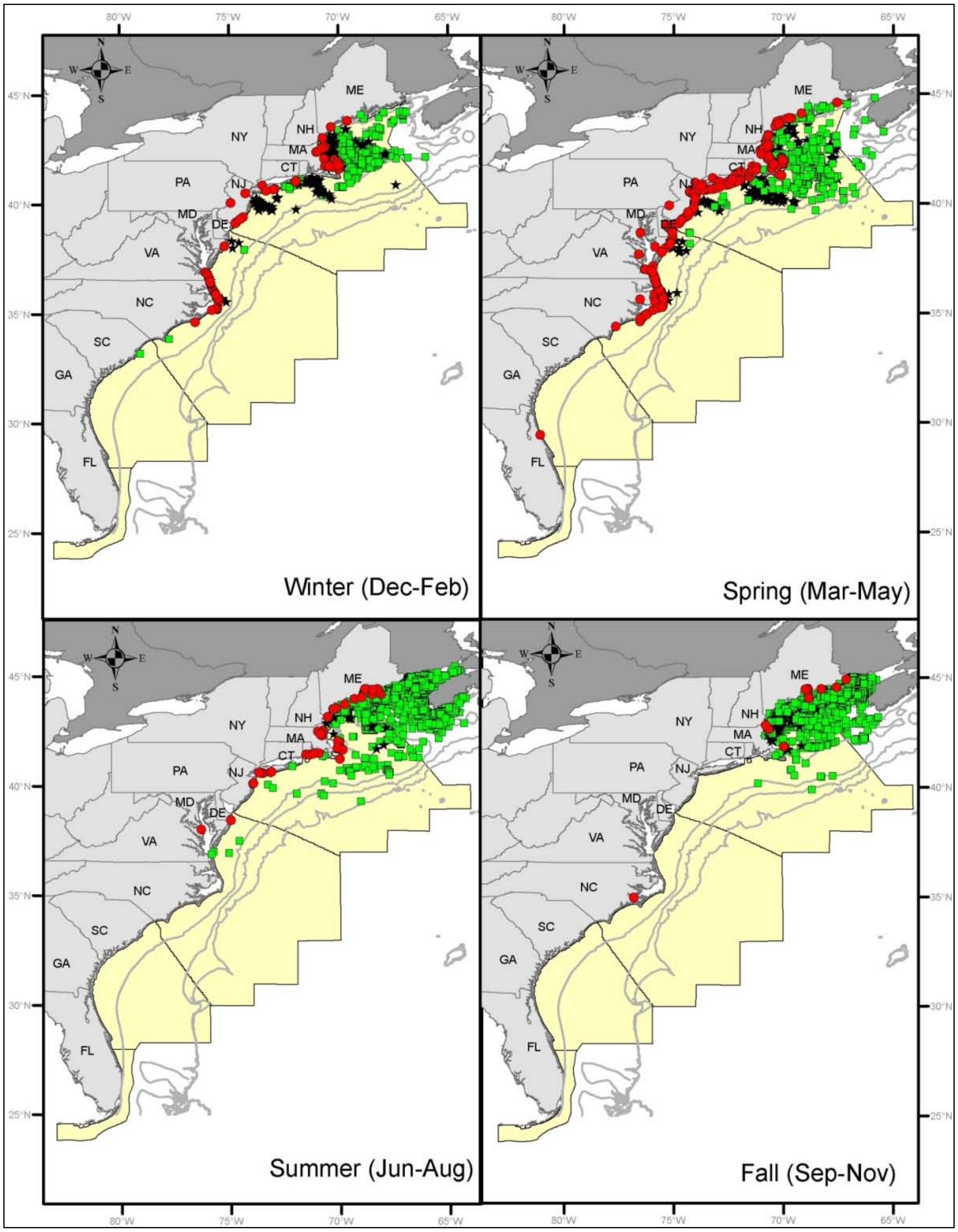


Figure 2.29.1. Harbor porpoise sighting (green squares), stranding (red dots) and fishery by-catch (black stars) observations by season.

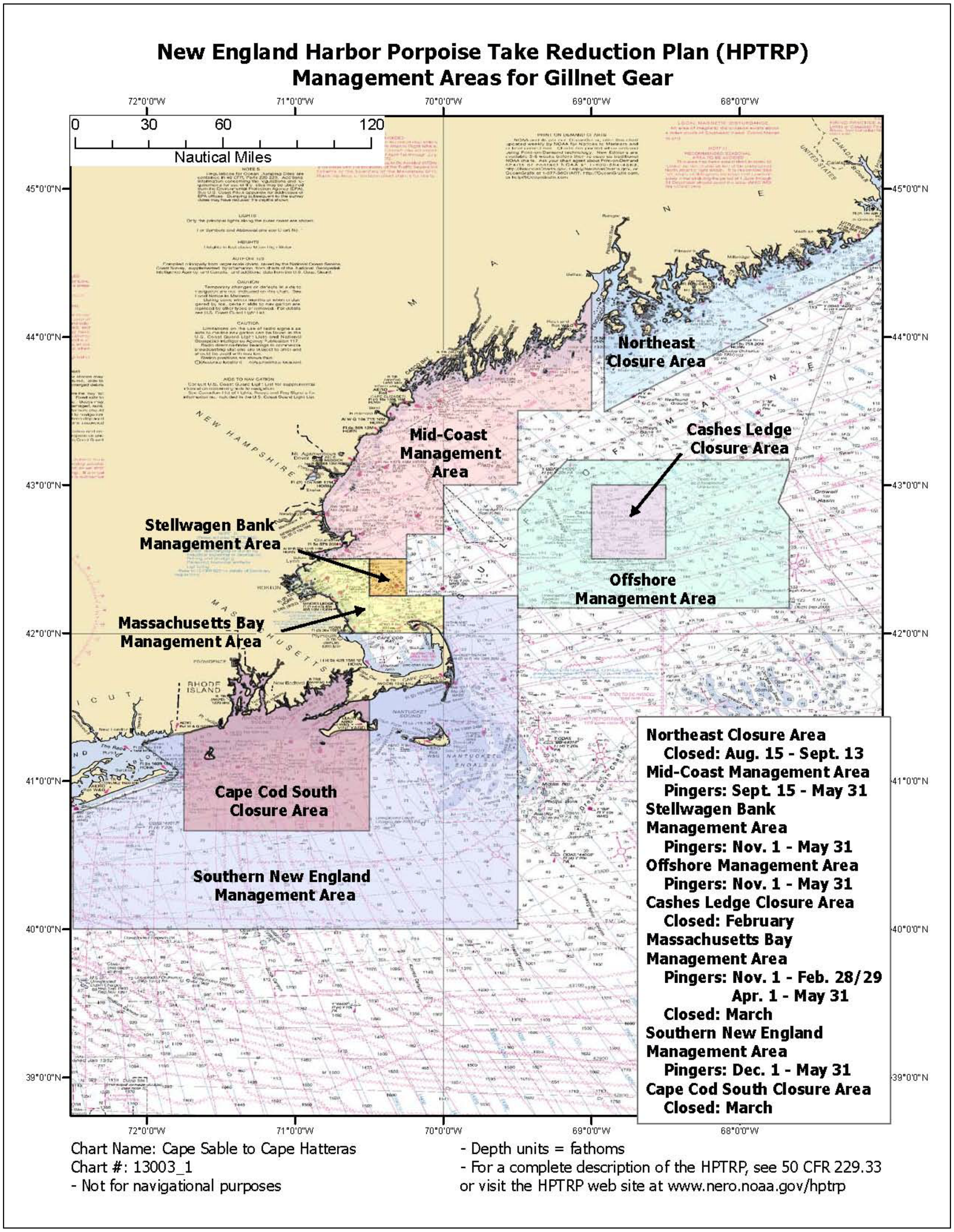
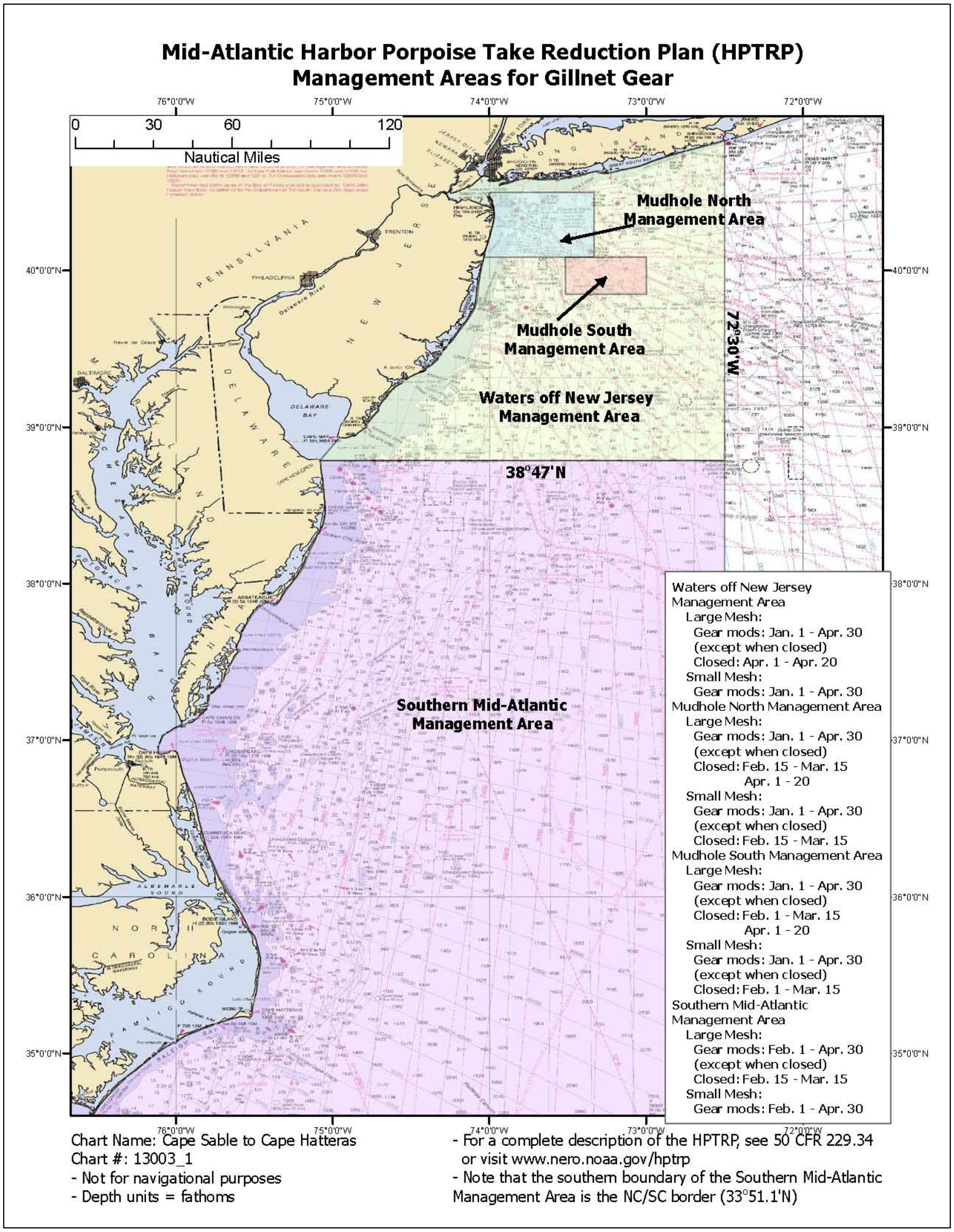


Figure 2.29.2. Map of New England closure areas according to the harbor porpoise Take Reduction Team.



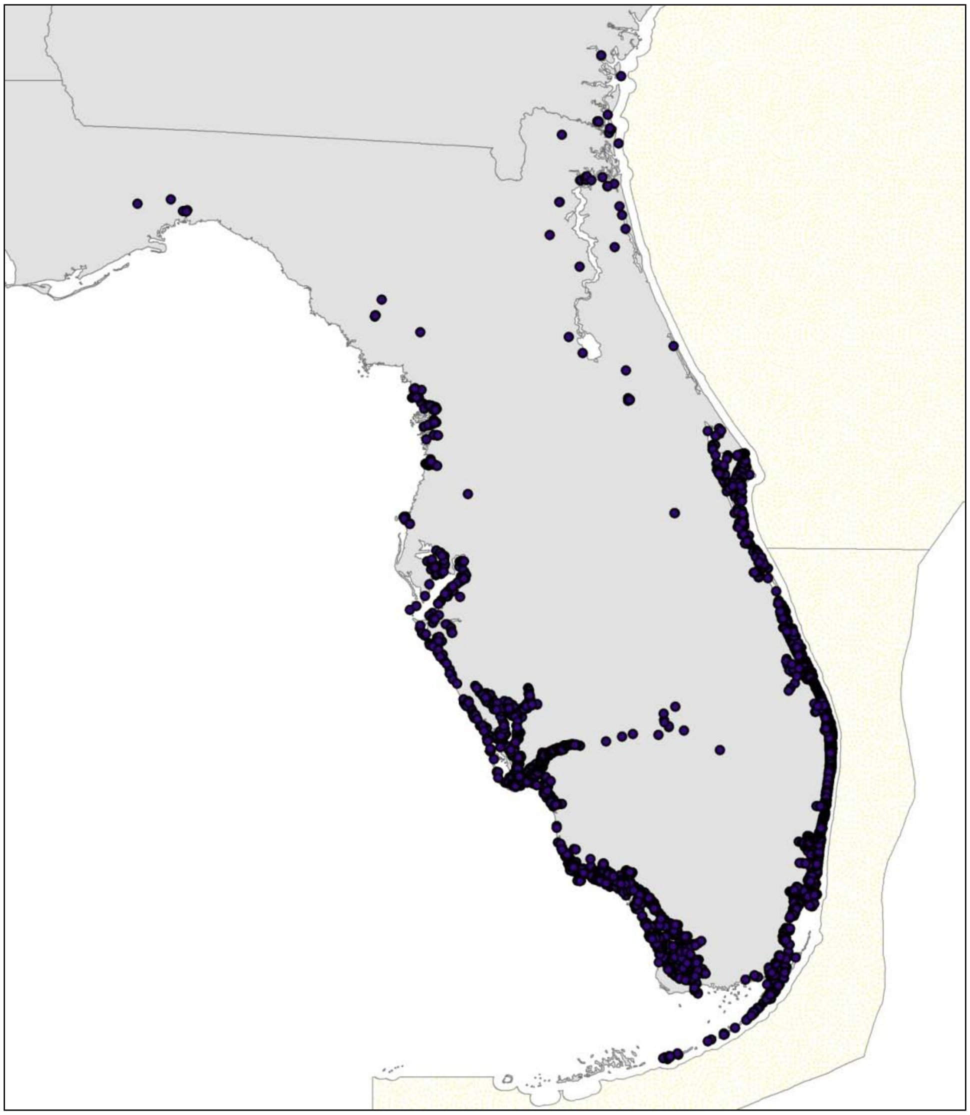
Mid-Atlantic Harbor Porpoise Take Reduction Plan (HPTRP) Management Areas for Gillnet Gear

Waters off New Jersey Management Area
  Large Mesh:
    Gear mods: Jan. 1 - Apr. 30 (except when closed)
    Closed: Apr. 1 - Apr. 20
  Small Mesh:
    Gear mods: Jan. 1 - Apr. 30
Mudhole North Management Area
  Large Mesh:
    Gear mods: Jan. 1 - Apr. 30 (except when closed)
    Closed: Feb. 15 - Mar. 15
    Apr. 1 - 20
  Small Mesh:
    Gear mods: Jan. 1 - Apr. 30 (except when closed)
    Closed: Feb. 15 - Mar. 15
Mudhole South Management Area
  Large Mesh:
    Gear mods: Jan. 1 - Apr. 30 (except when closed)
    Closed: Feb. 1 - Mar. 15
    Apr. 1 - 20
  Small Mesh:
    Gear mods: Jan. 1 - Apr. 30 (except when closed)
    Closed: Feb. 1 - Mar. 15
Southern Mid-Atlantic Management Area
  Large Mesh:
    Gear mods: Feb. 1 - Apr. 30 (except when closed)
    Closed: Feb. 15 - Mar. 15
  Small Mesh:
    Gear mods: Feb. 1 - Apr. 30

Chart Name: Cape Sable to Cape Hatteras
Chart #: 13003_1
- Not for navigational purposes
- Depth units = fathoms

- For a complete description of the HPTRP, see 50 CFR 229.34 or visit www.nero.noaa.gov/hptrp
- Note that the southern boundary of the Southern Mid-Atlantic Management Area is the NC/SC border (33°51.1'N)

Figure 2.29.3 Map of Mid-Atlantic closure areas according to the harbor porpoise Take Reduction Team.



Data from http://ocean.floridamarine.org/mrgis/Description_Layers_Marine.htm#marmam

Figure 2.30.1. Manatee Synoptic Count Data 1991-2011.

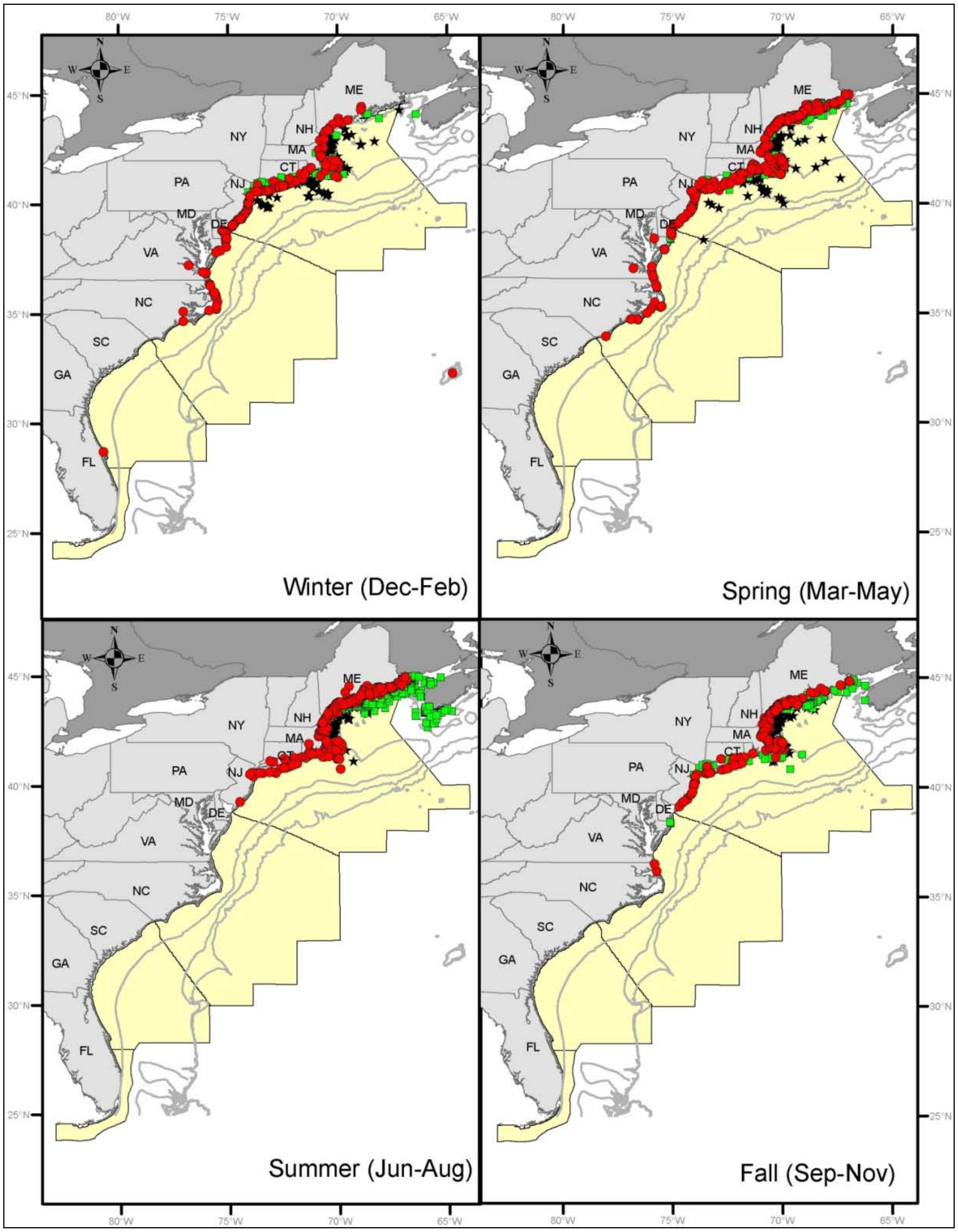
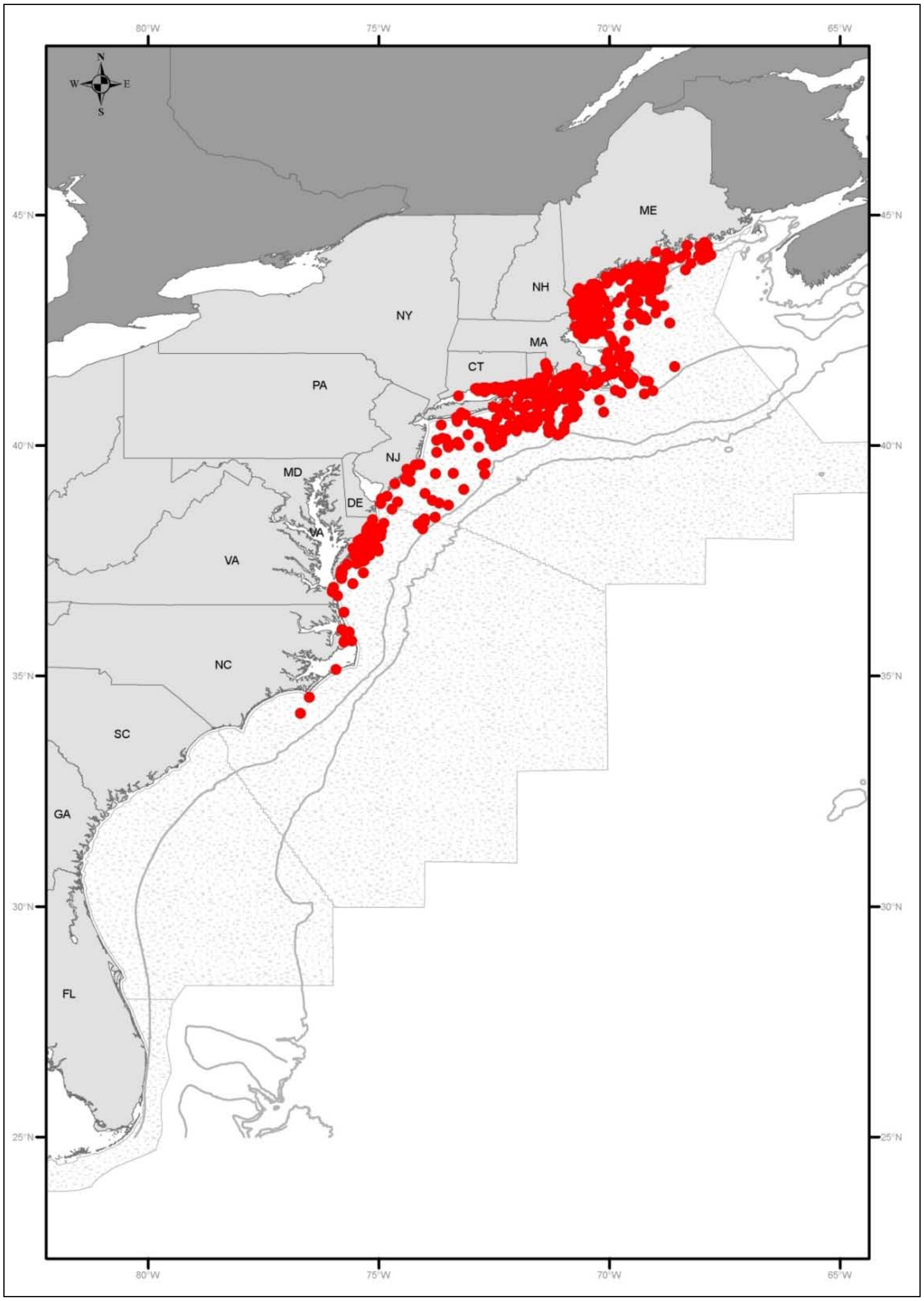


Figure 2.31.1. Harbor seal sighting (green squares), stranding (red dots) and fishery by-catch (black stars) observations by season.



Data from Whalenet.

Figure 2.31.2. Positions of satellite-tagged harbor seals.

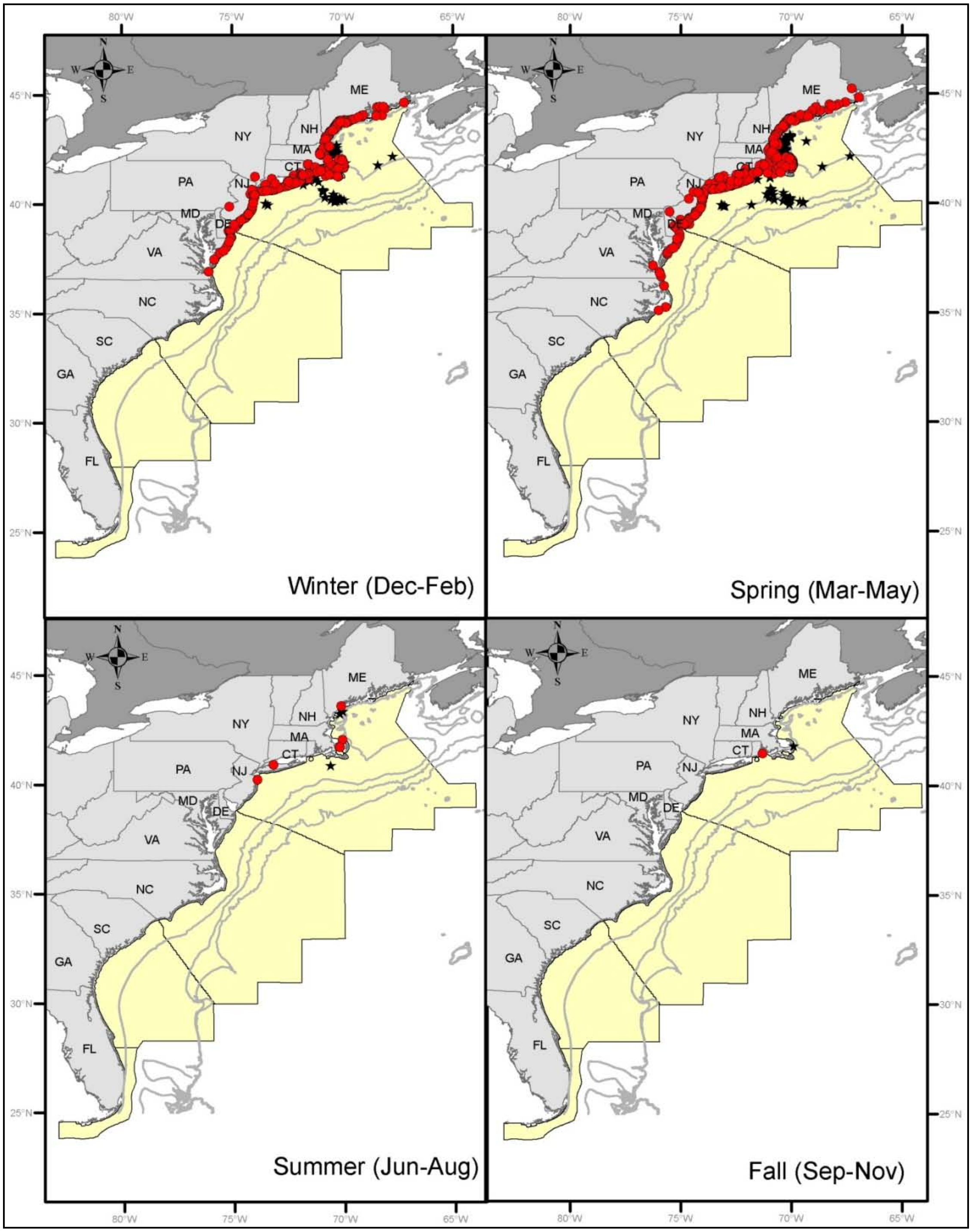
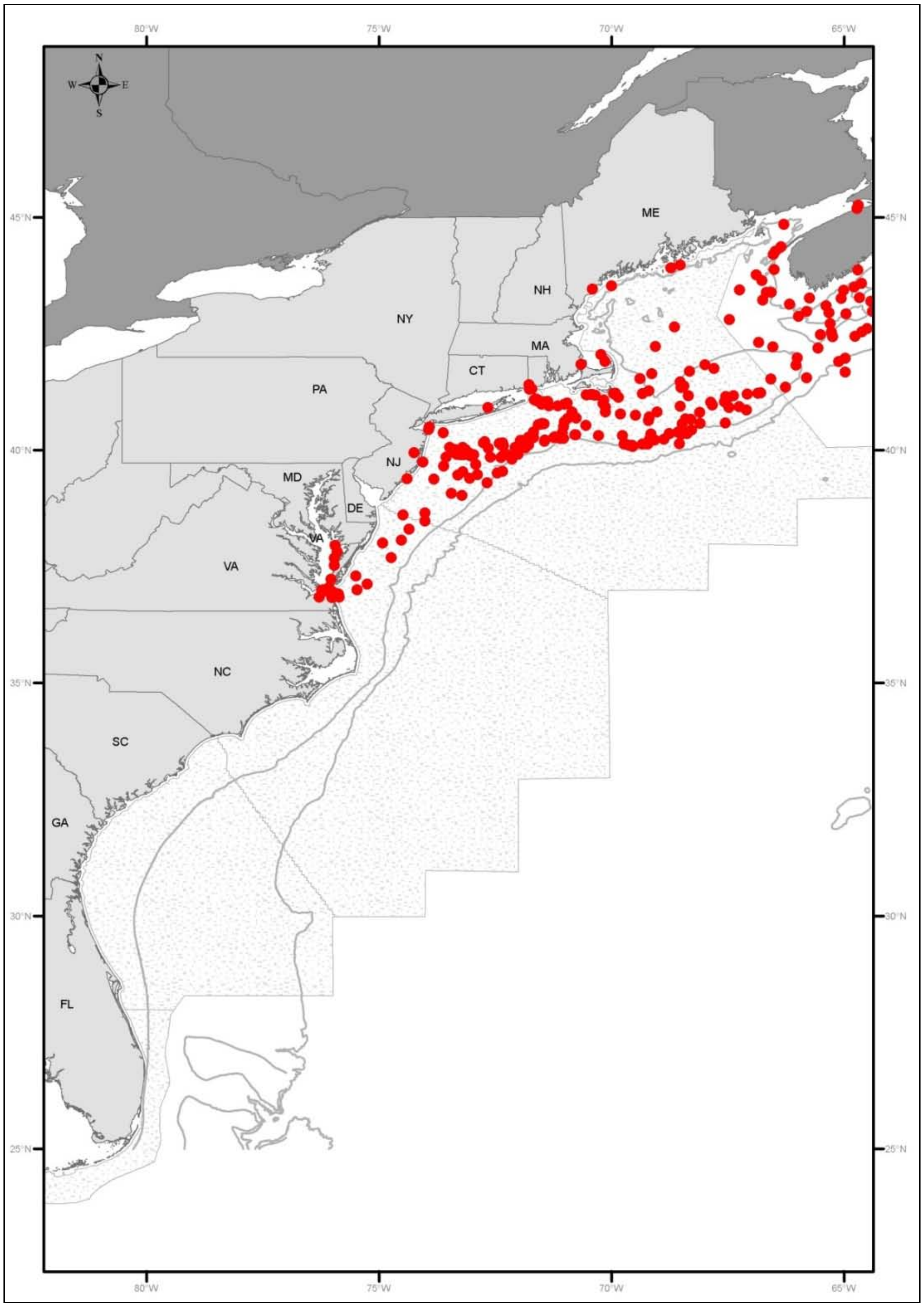


Figure 2.32.1. Harp seal stranding (red dots) and fishery by-catch (black stars) observations by season.



Data from Whalenet.

Figure 2.32.2. Positions of satellite-tagged harp seals.

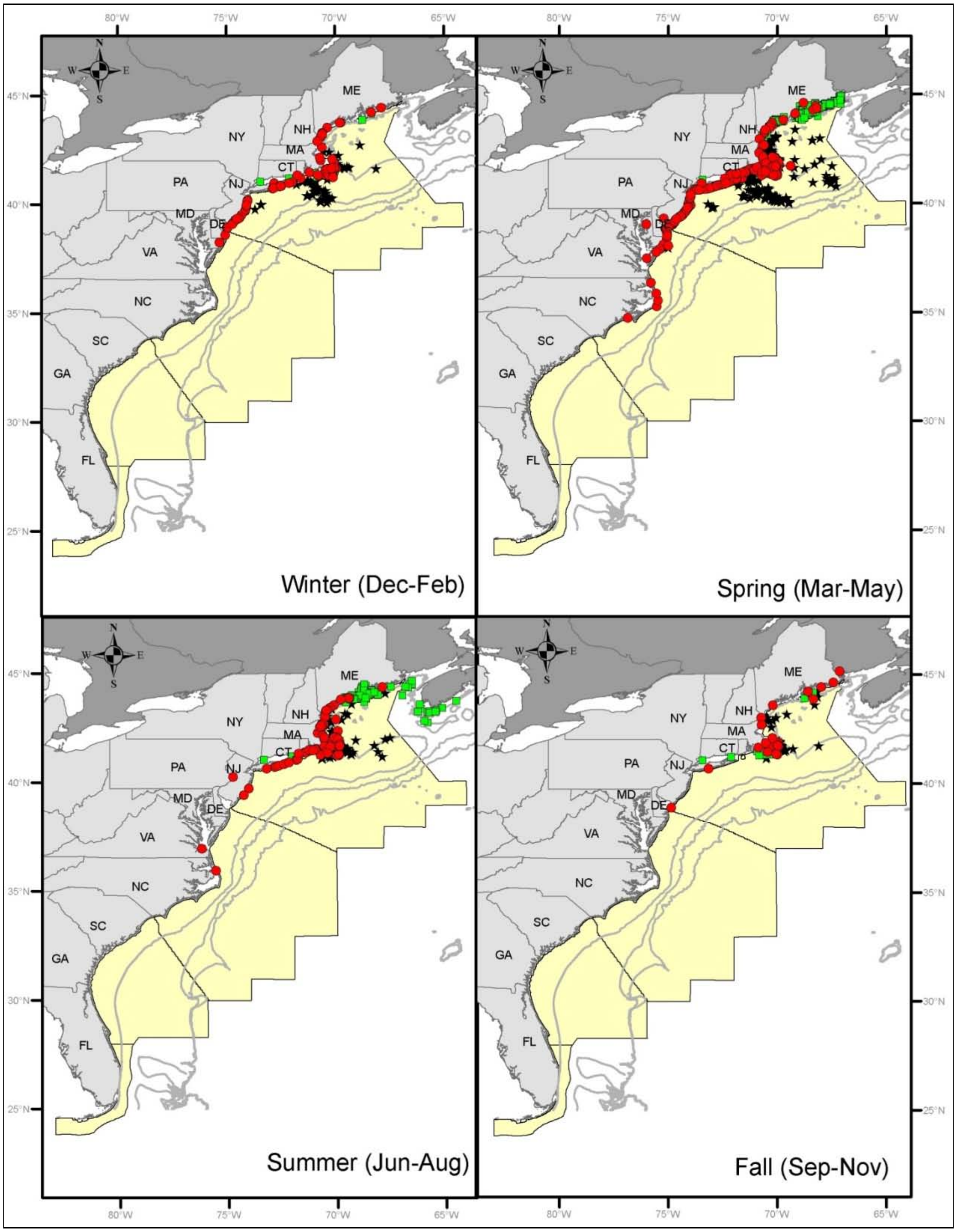
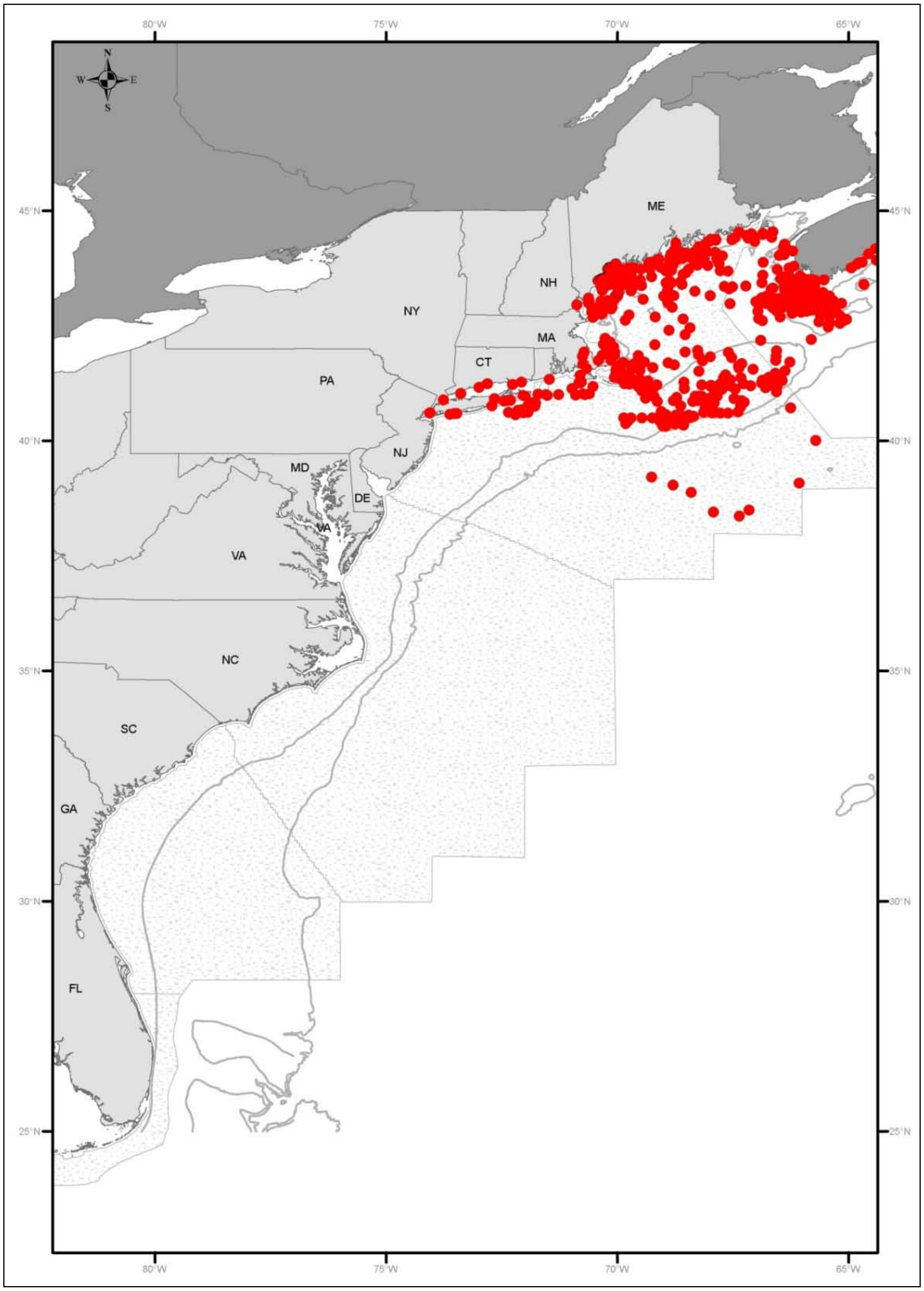


Figure 2.33.1. Gray seal sighting (green squares), stranding (red dots) and fishery by-catch (black stars) observations by season.



Data from Whalenet.

Figure 2.33.2. Positions of satellite-tagged gray seals.

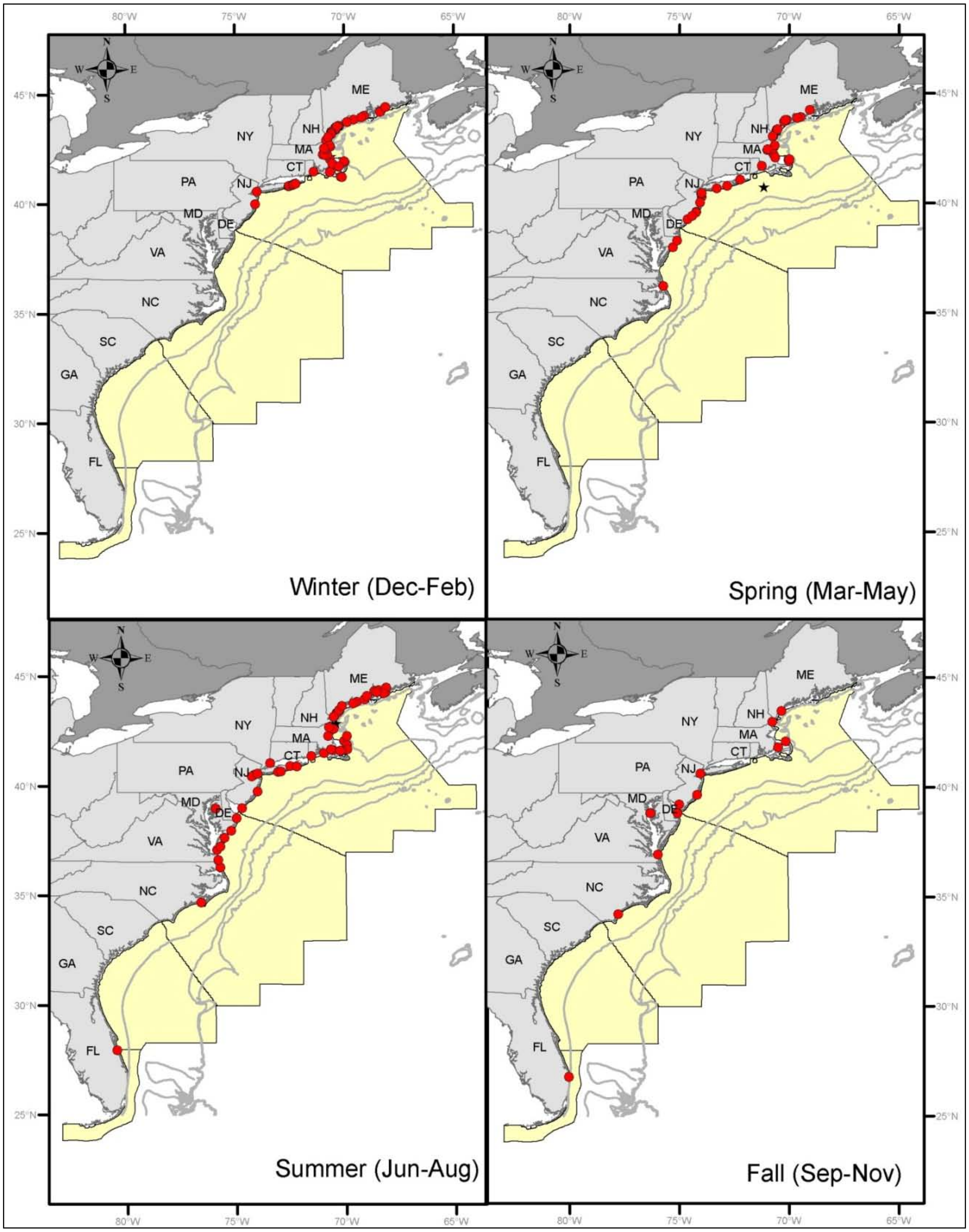
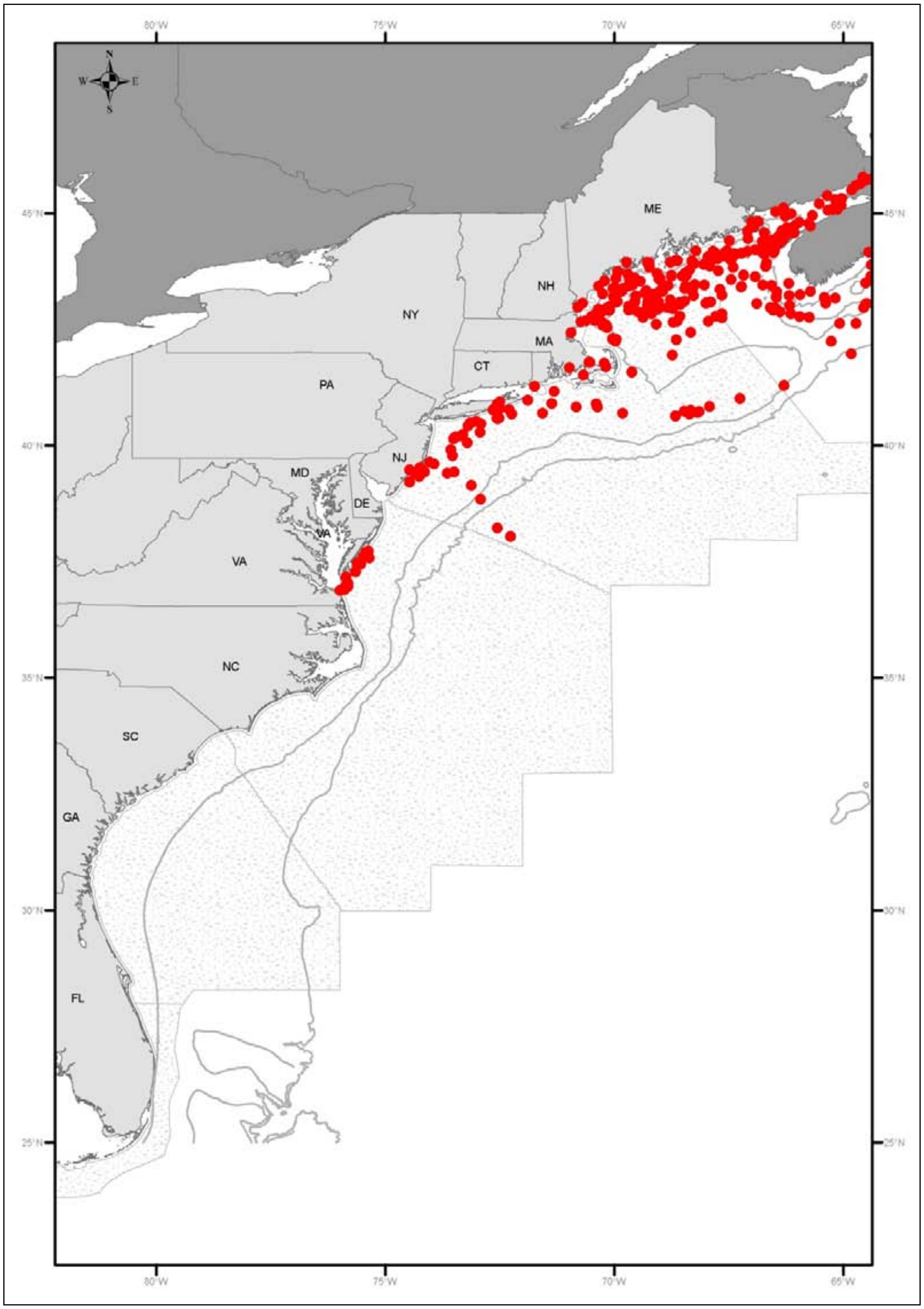
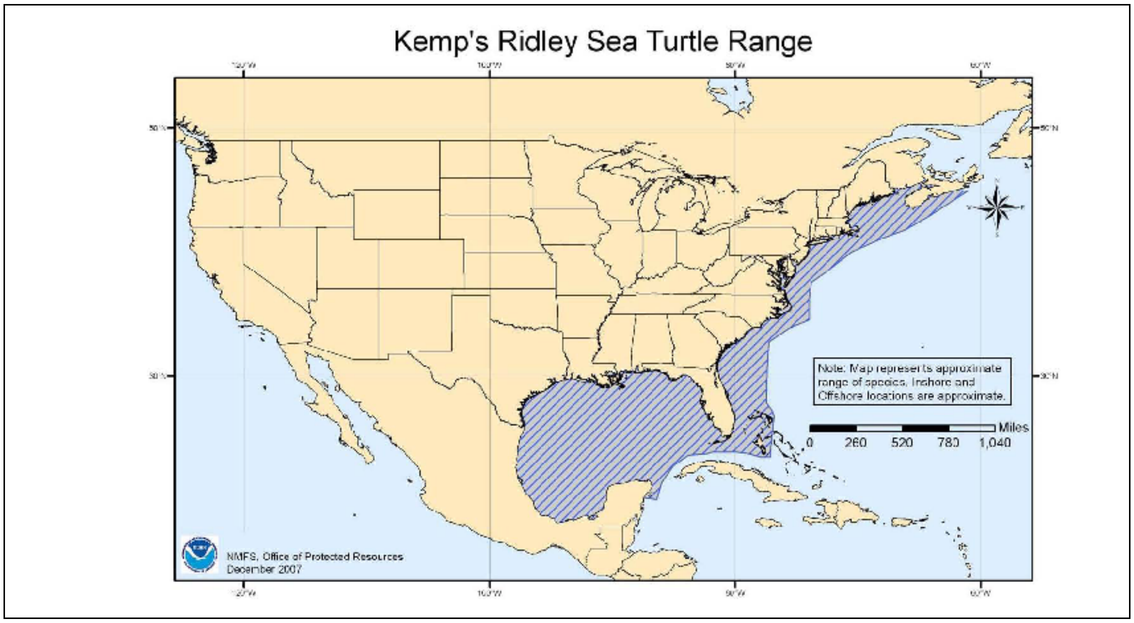


Figure 2.34.1. Hooded seal stranding (red dots) and fishery by-catch (black stars) observations by season.



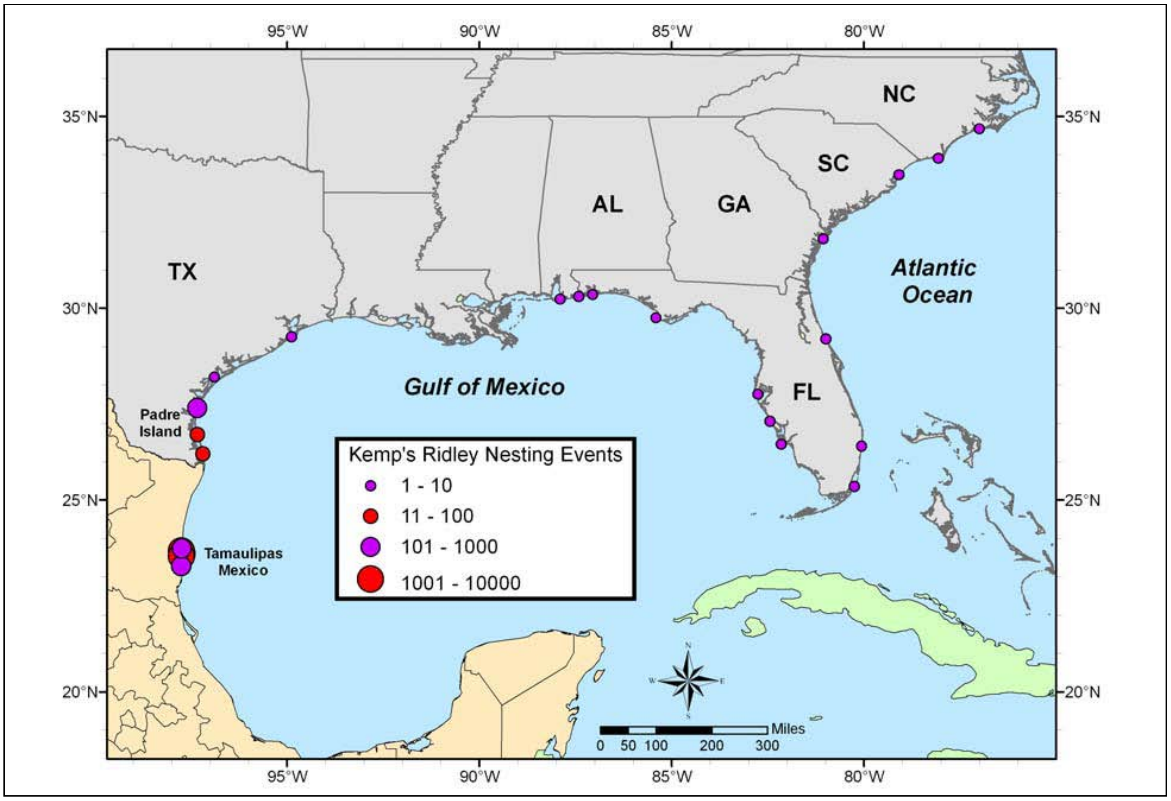
Data from Whalenet.

Figure 2.34.2. Positions of satellite-tagged hooded seals.



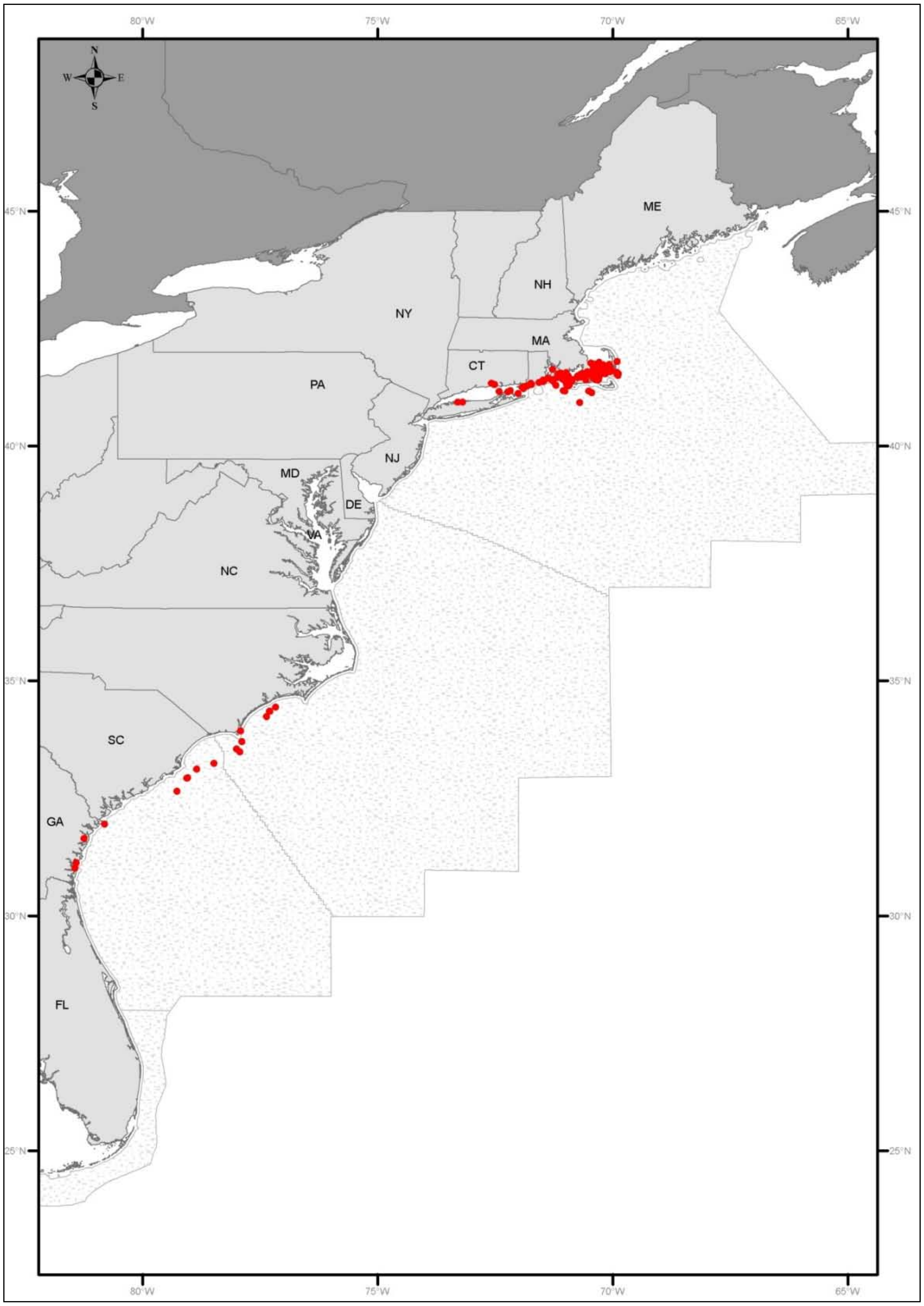
Map from http://www.nmfs.noaa.gov/pr/pdfs/rangemaps/kemps_ridley_turtle.pdf.

Figure 2.35.1. Kemp's ridley range.



Map from National Marine Fisheries Service et al. (2010).

Figure 2.35.2. Kemp's ridley nesting locations in 2007.



Data from OBIS.

Figure 2.35.3. Positions of satellite-tagged Kemp’s ridley sea turtles.

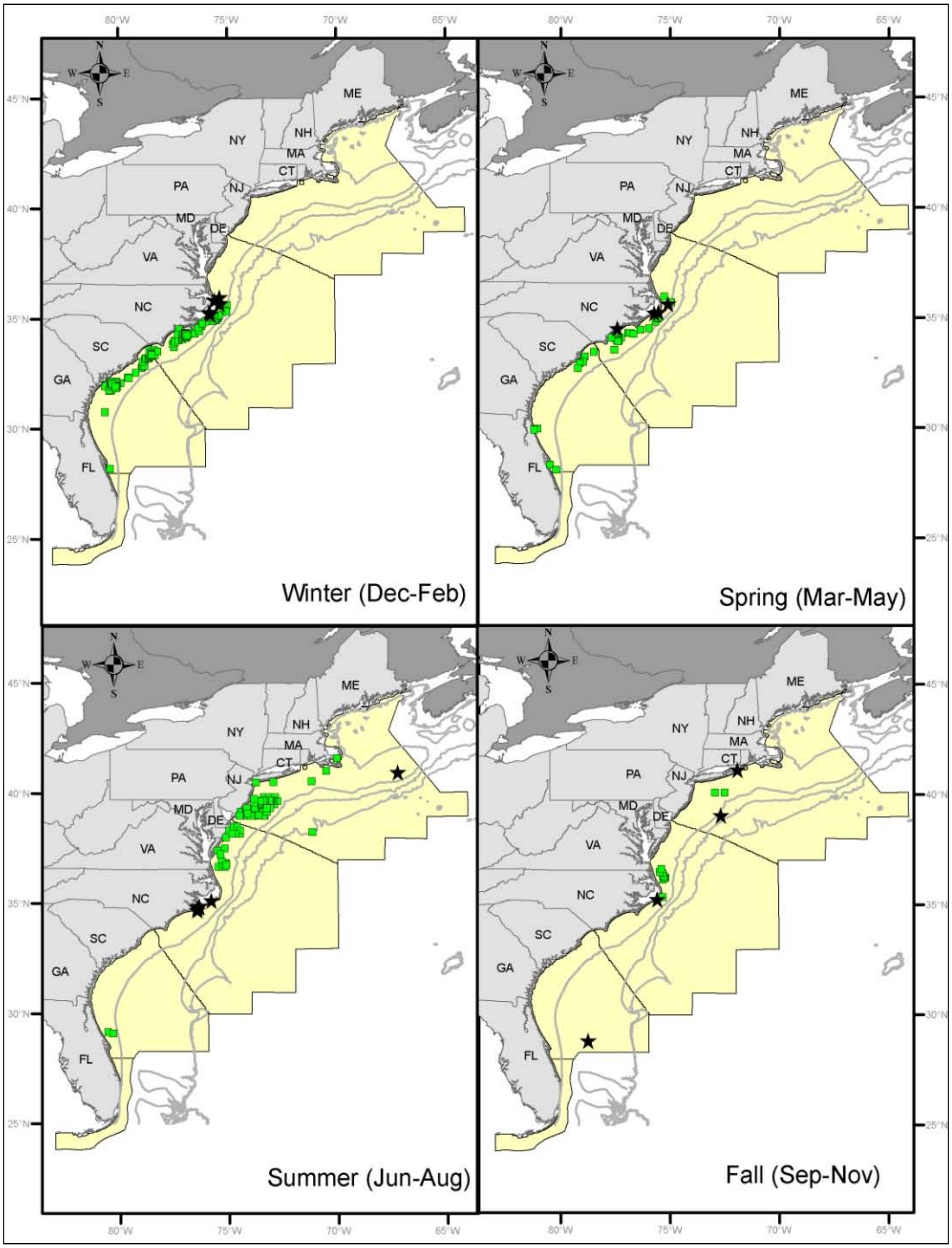
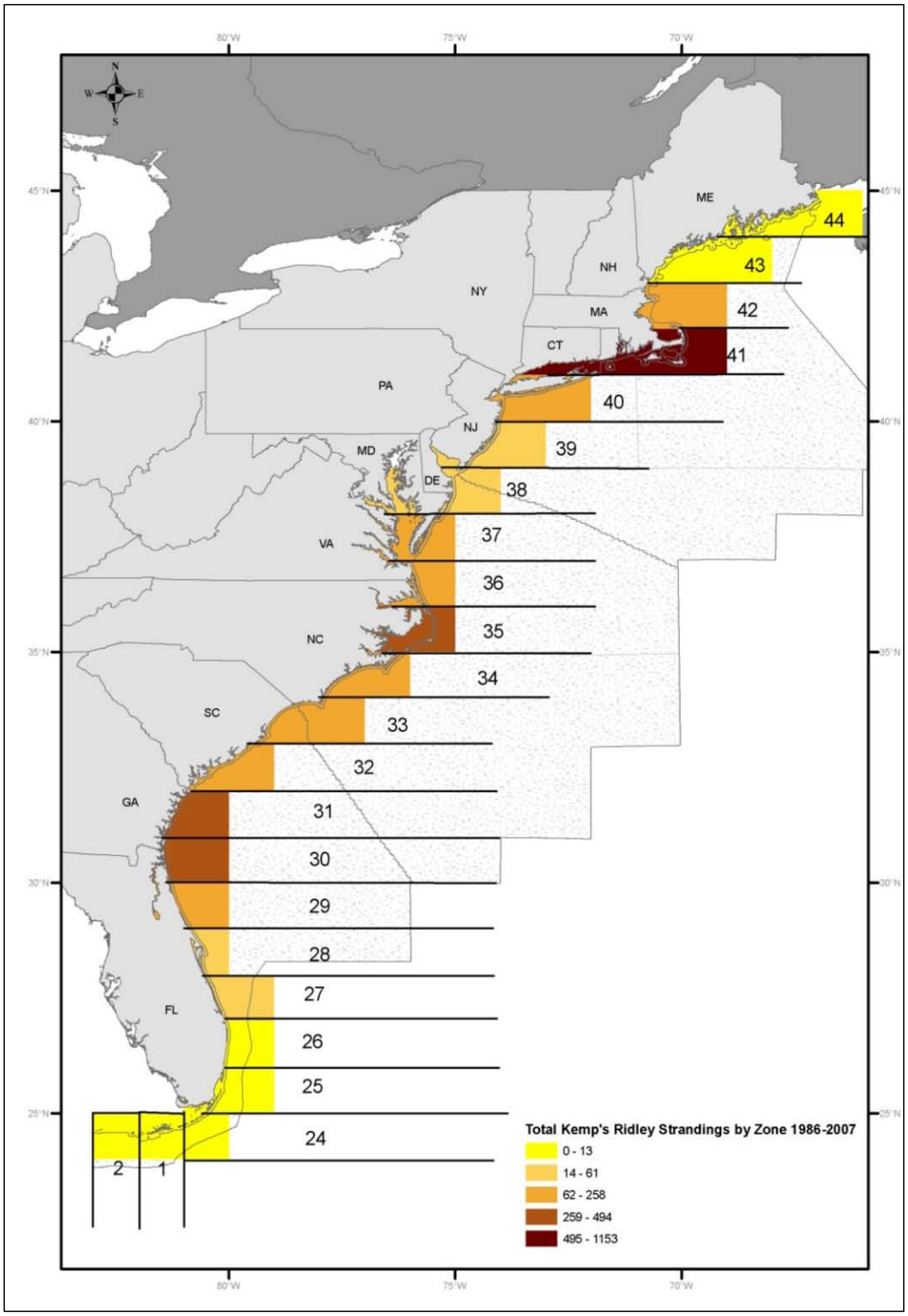
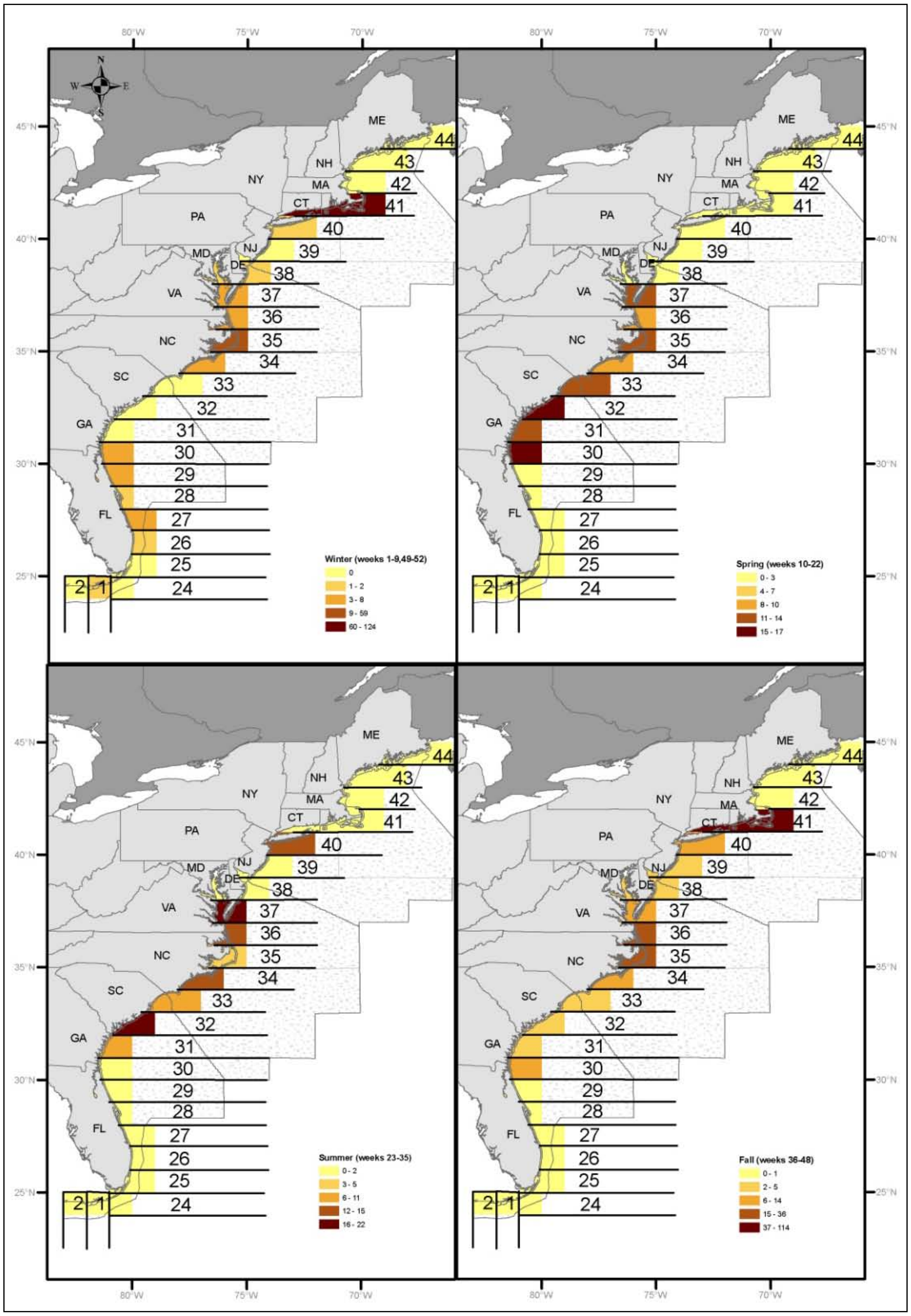


Figure 2.35.4. Kemp's ridley sea turtle sighting (green squares) and fishery by-catch (black stars) observations by season.



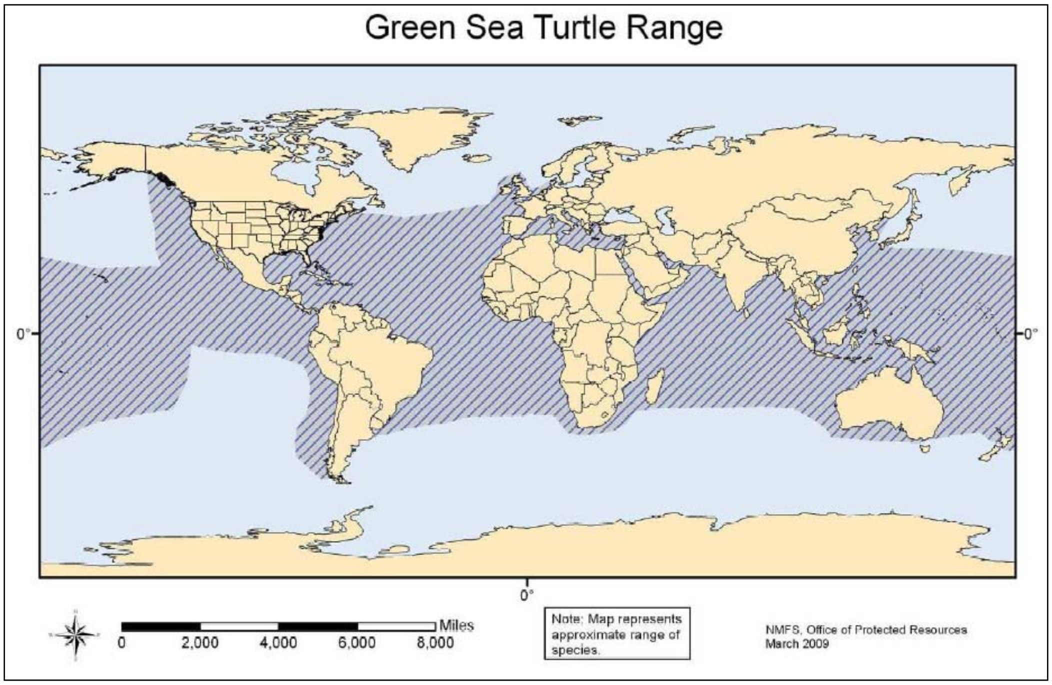
Data from the Sea Turtle Stranding and Salvage Network.

Figure 2.35.5. Kemp's ridley strandings by zone, 1986-2007.

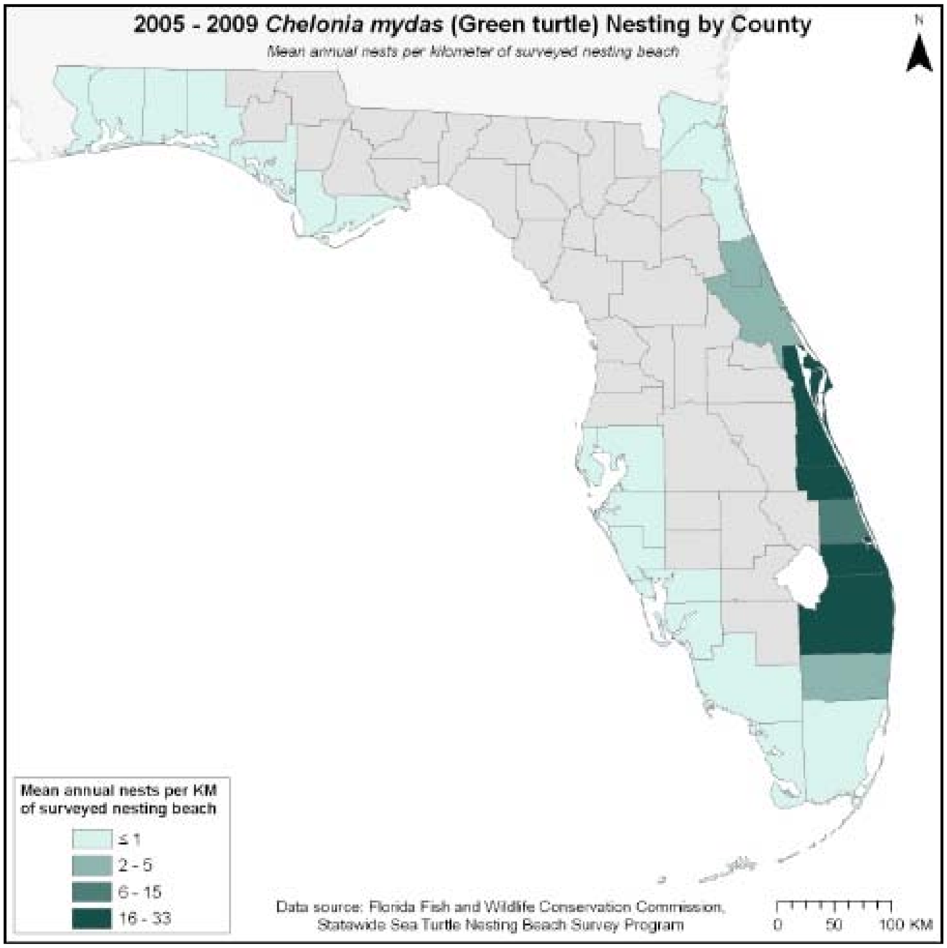


Data from the Sea Turtle Stranding and Salvage Network.

Figure 2.35.6. Seasonal Kemp's ridley stranding data for years 2008 and 2009.

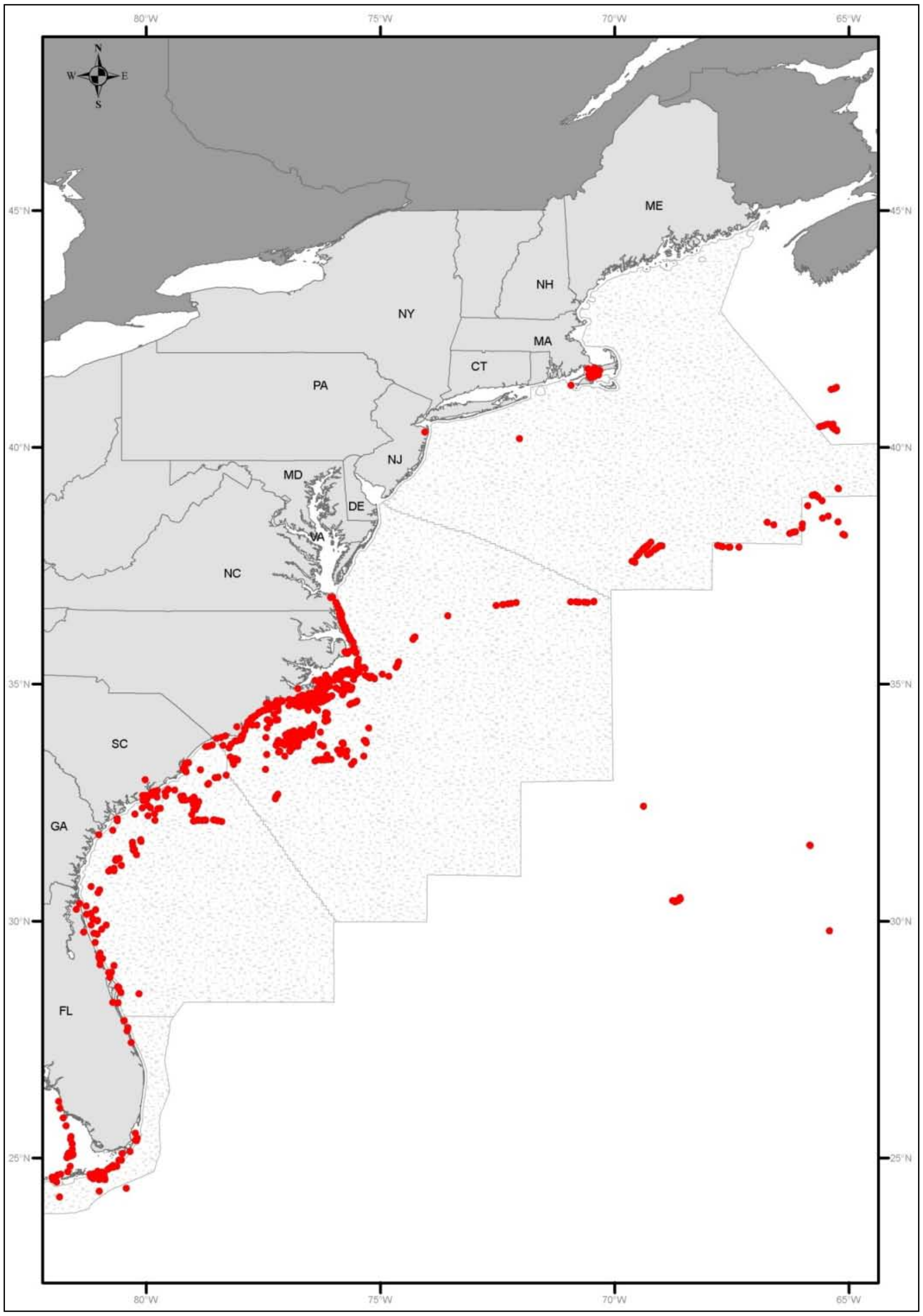


Map from http://www.nmfs.noaa.gov/pr/species/turtles/green.htm#regulations
Figure 2.36.1. Green turtle distribution.



Map from http://research.myfwc.com/features/view_article.asp?id=2496.

Figure 2.36.2. Green turtle nesting densities by county in Florida.



Data from OBIS.

Figure 2.36.3. Positions of satellite-tagged green turtles.

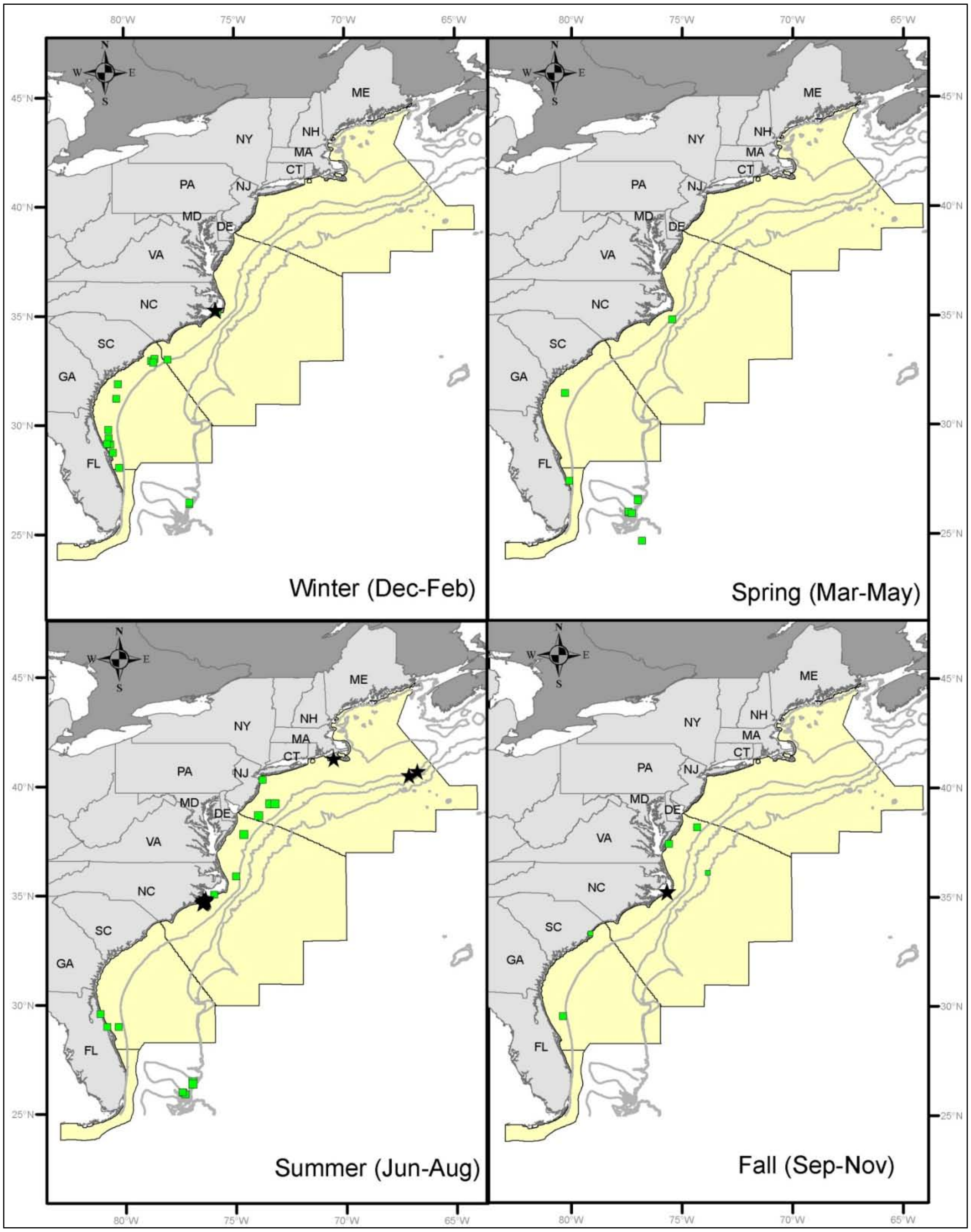
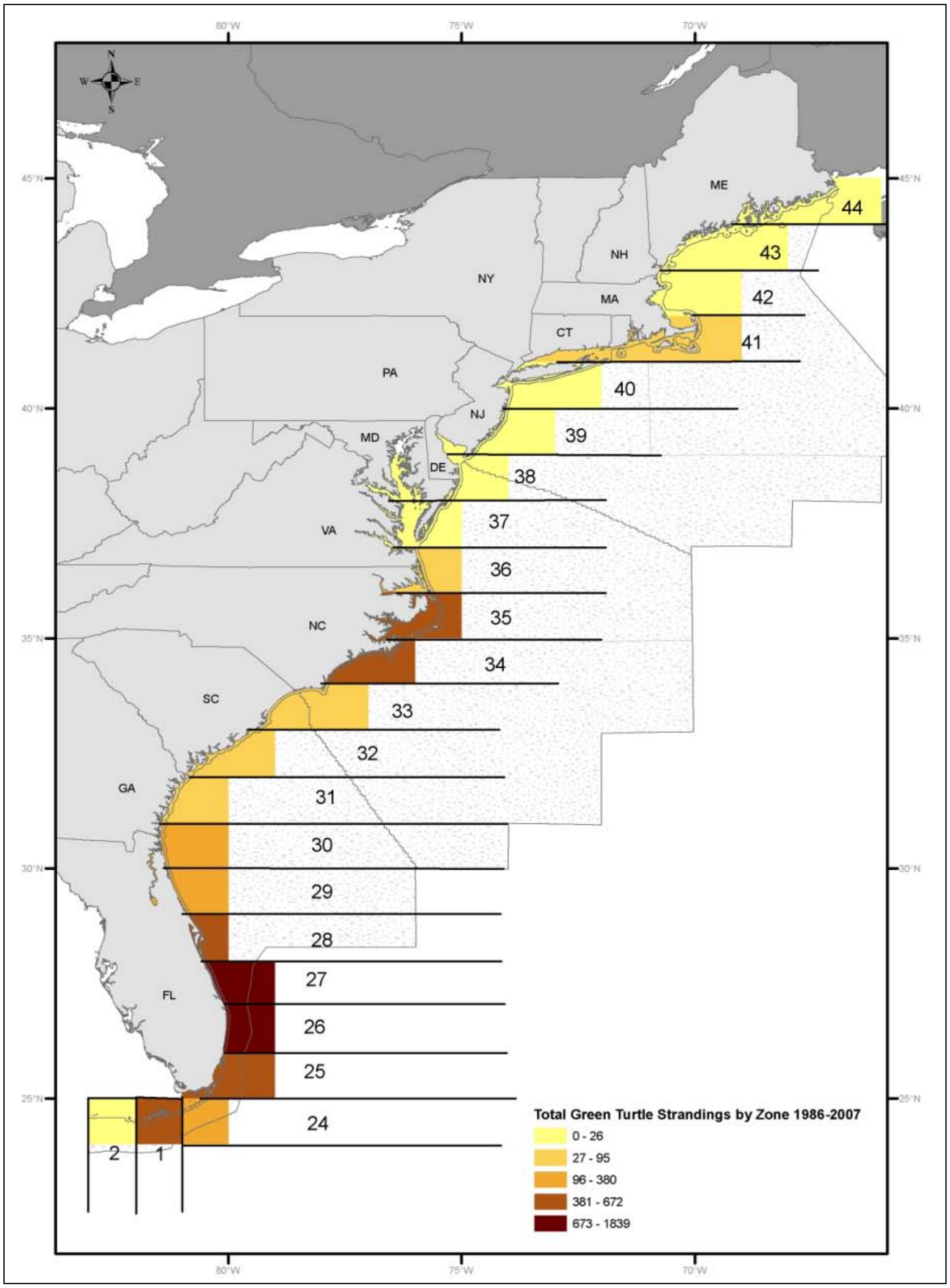
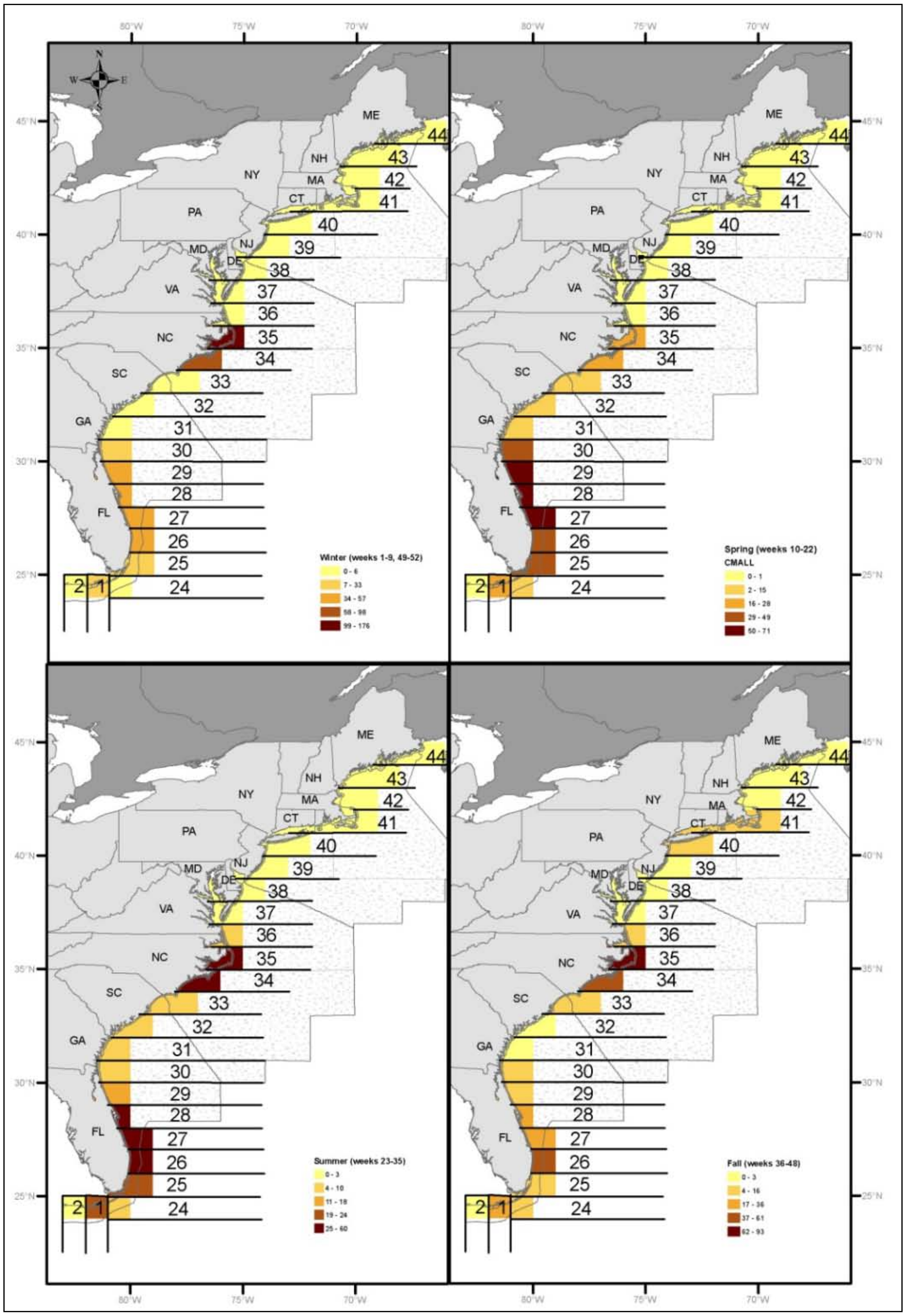


Figure 2.36.4. Green sea turtle sighting (green squares) and fishery by-catch (black stars) observations by season.



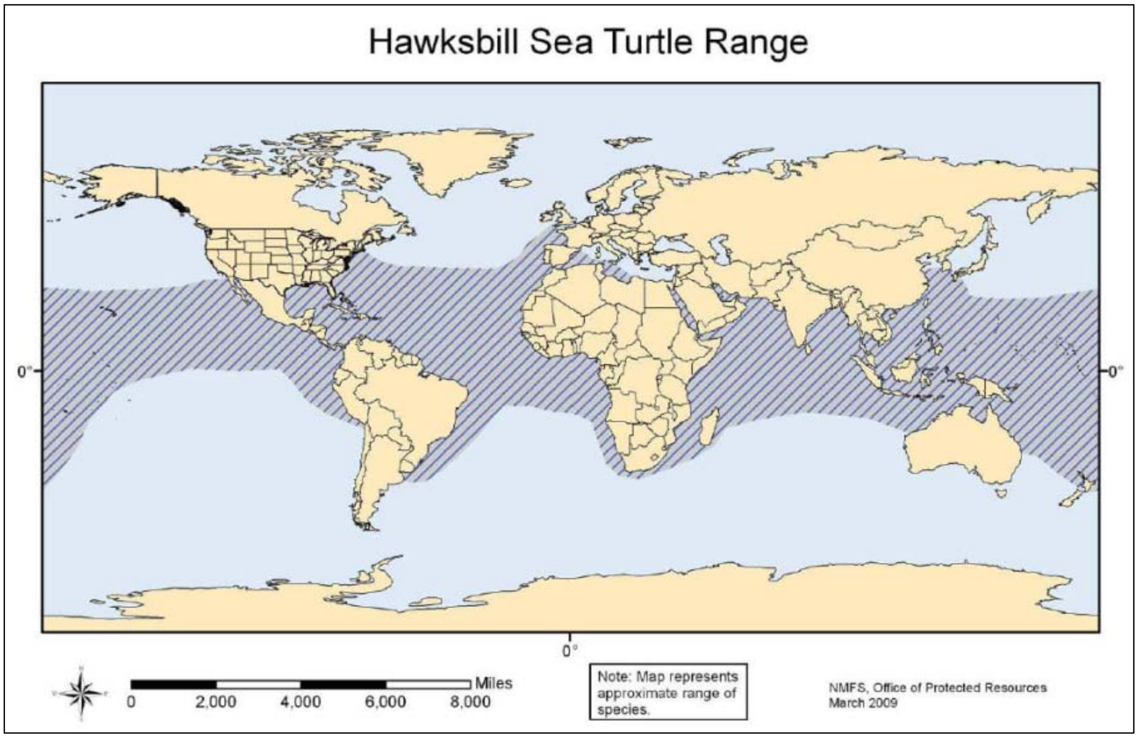
Data from the Sea Turtle Stranding and Salvage Network.

Figure 2.36.5. Green turtle strandings by zone, 1986-2007.



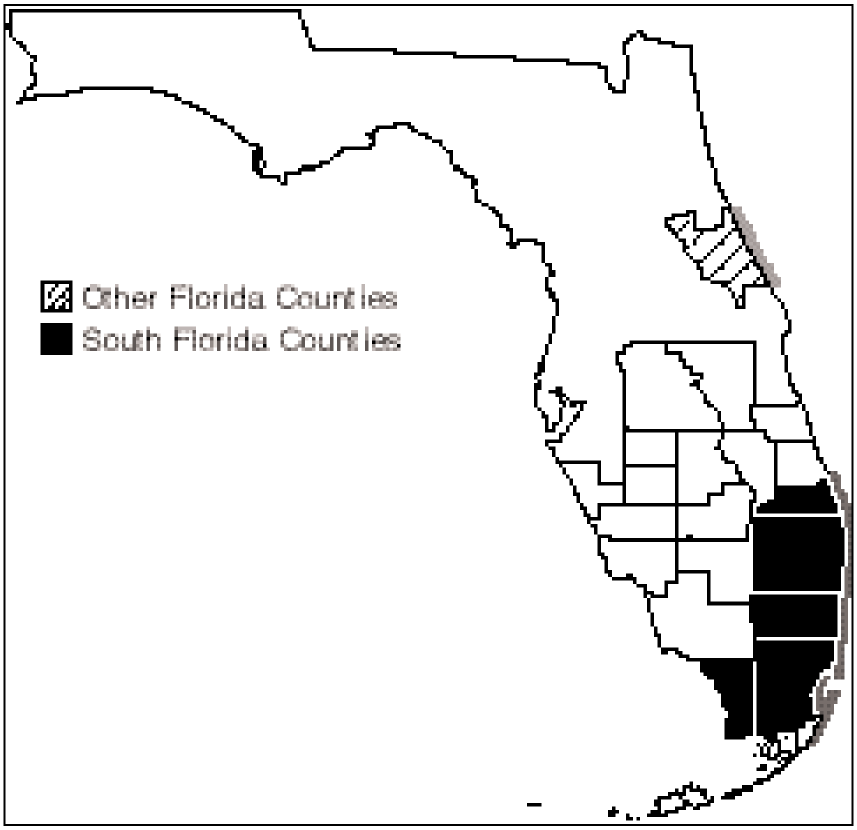
Data from the Sea Turtle Stranding and Salvage Network.

Figure 2.36.6. Seasonal green turtle stranding data for years 2008 and 2009.



Map from http://www.nmfs.noaa.gov/pr/pdfs/rangemaps/hawksbill_turtle.pdf.

Figure 2.37.1. Hawksbill global distribution.



Map from http://www.fws.gov/southeast/vbpdfs/species/reptiles/hstu.pdf
Figure 2.37.2. Florida nesting distribution of the hawksbill turtle.

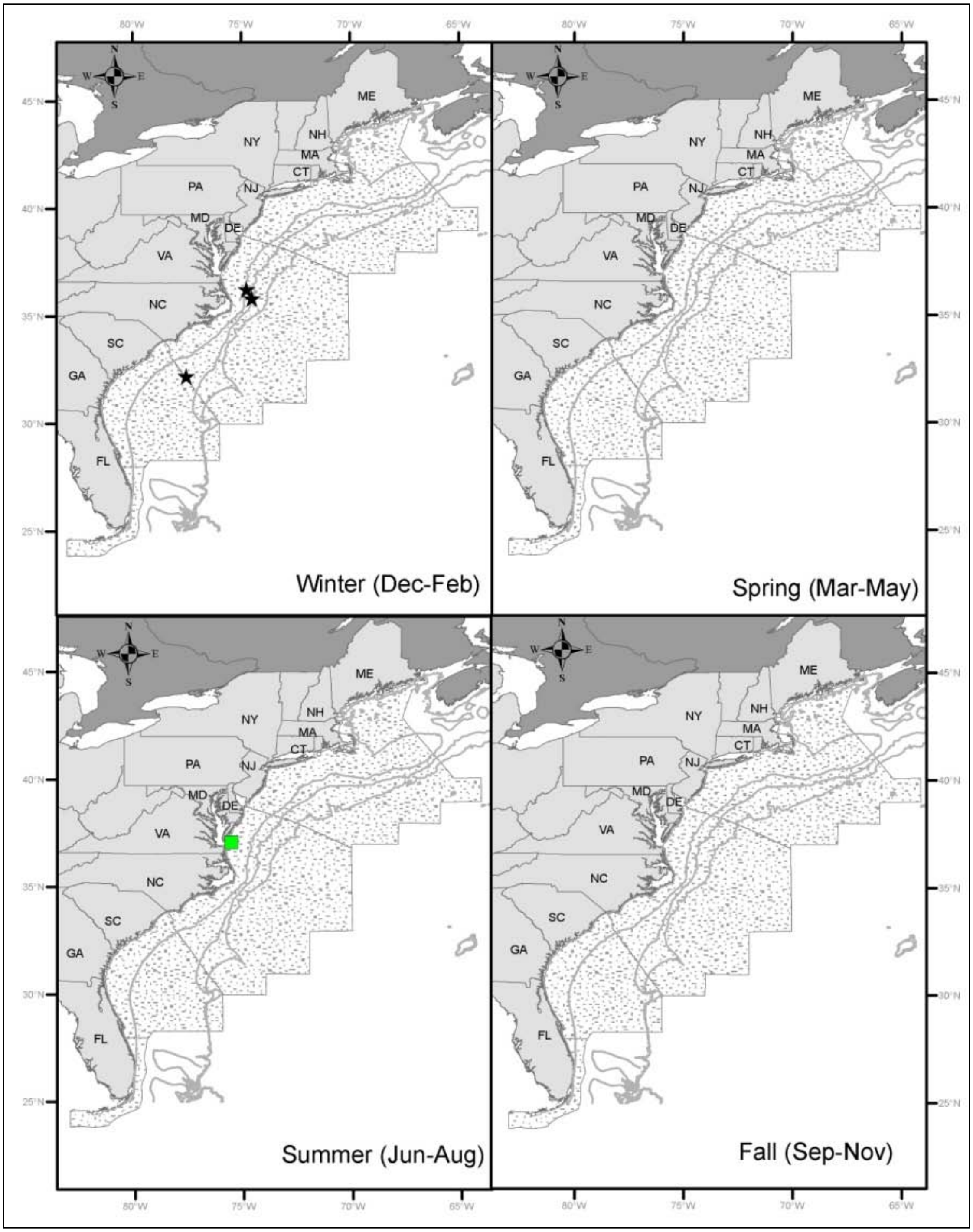
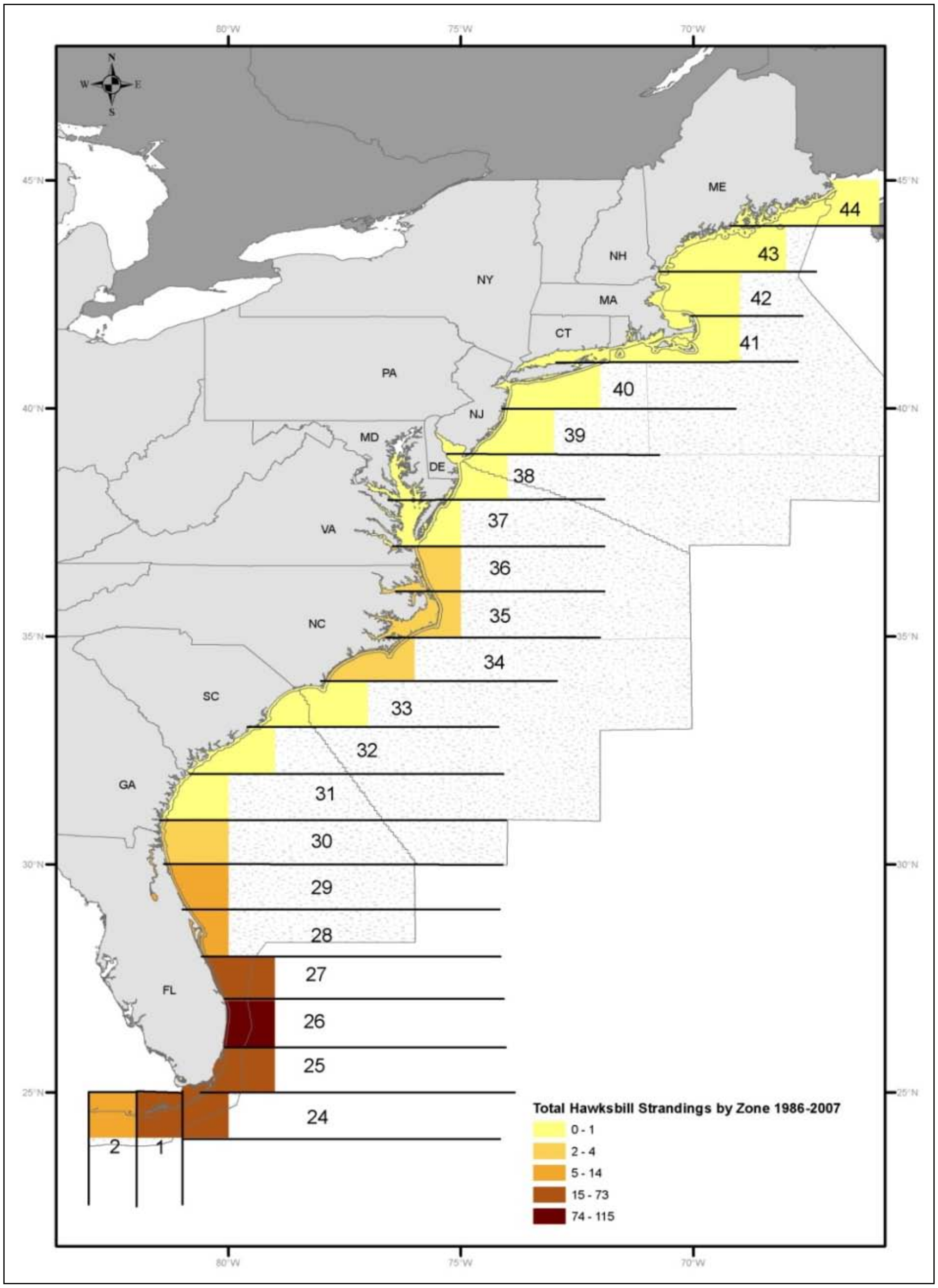
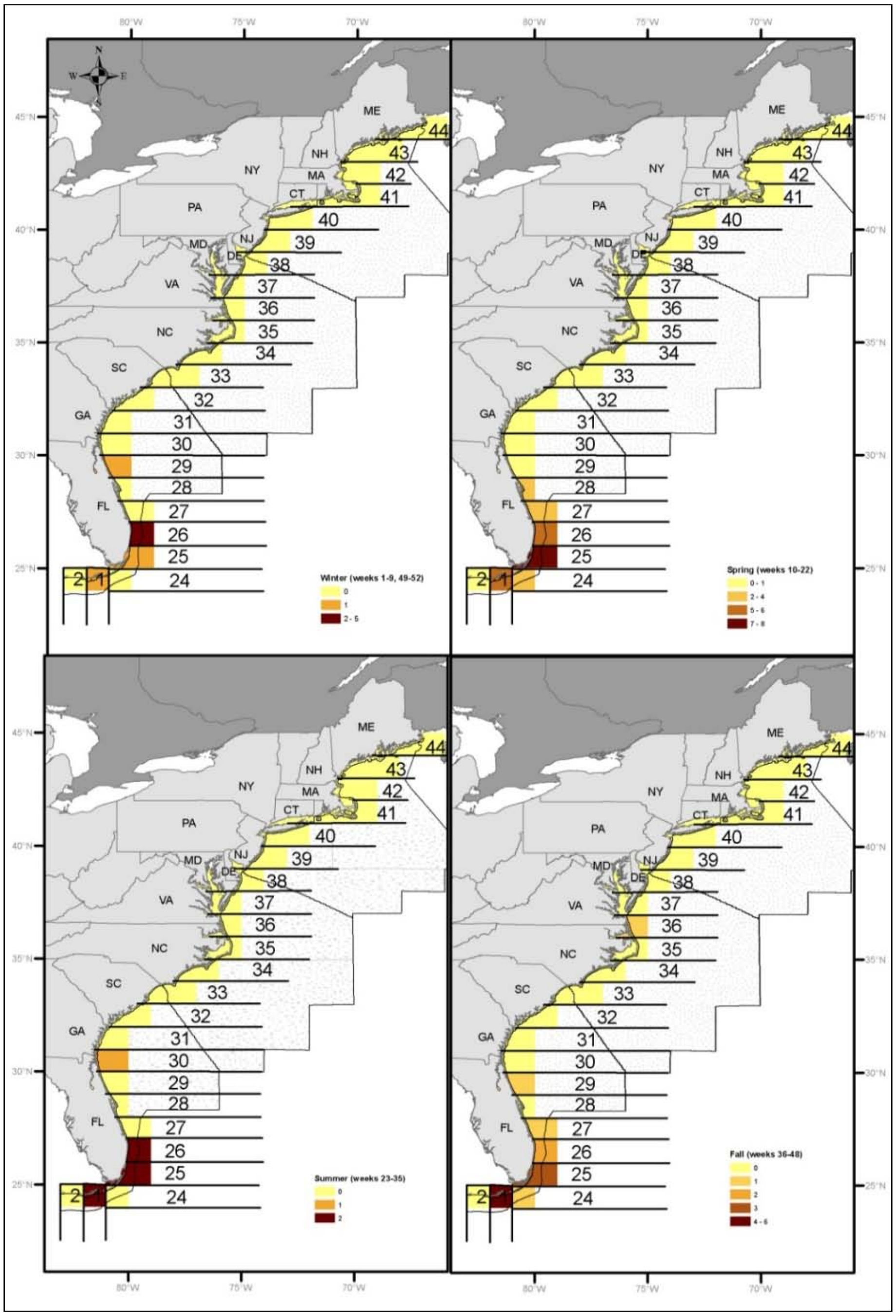


Figure 2.37.3. Hawksbill sea turtle sighting (green squares) and fishery by-catch (black stars) observations by season.



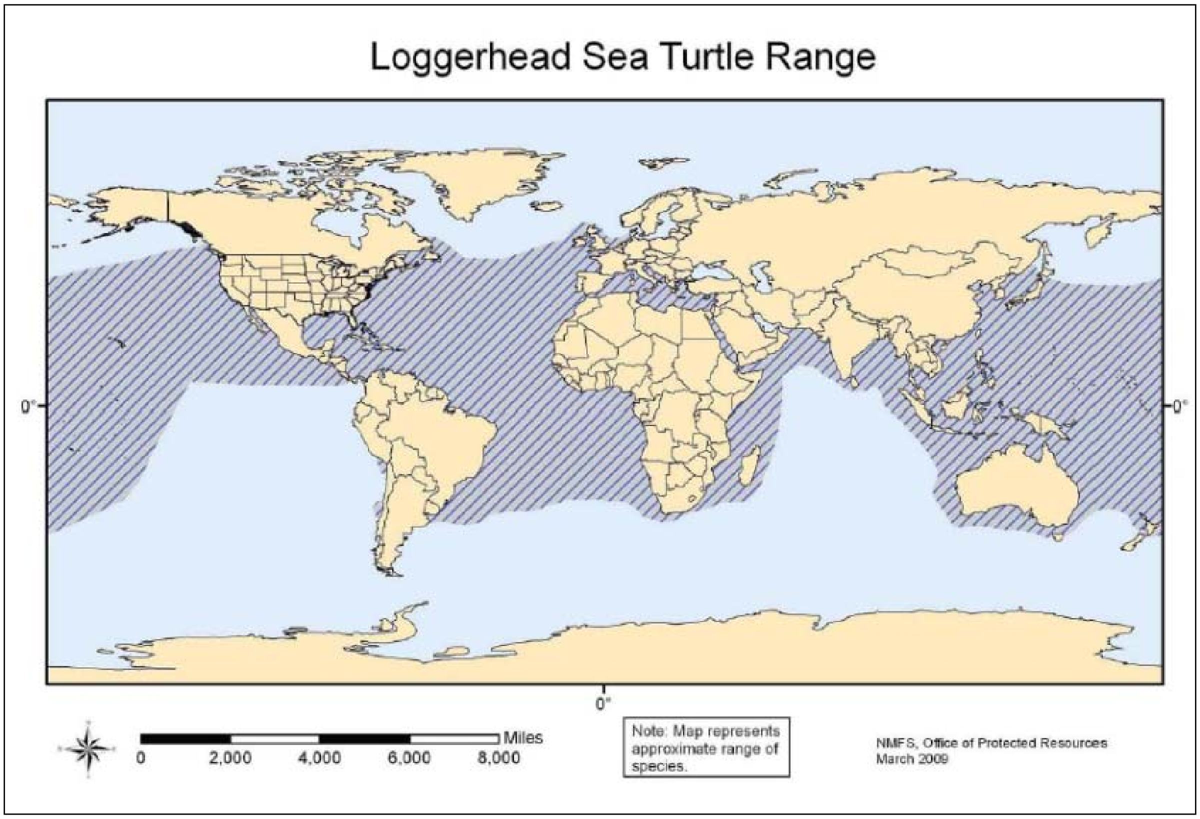
Data from the Sea Turtle Stranding and Salvage Network.

Figure 2.37.4. Hawksbill strandings by zone, 1986-2007.



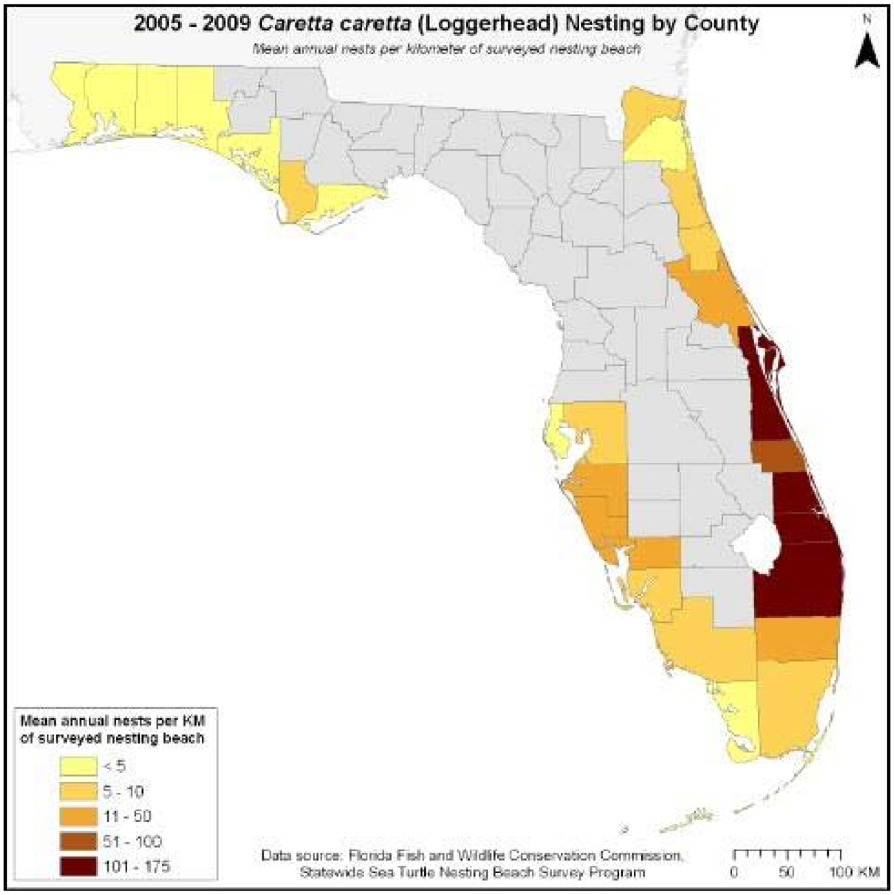
Data from the Sea Turtle Stranding and Salvage Network.

Figure 2.37.5. Seasonal hawksbill stranding data for years 2008 and 2009.



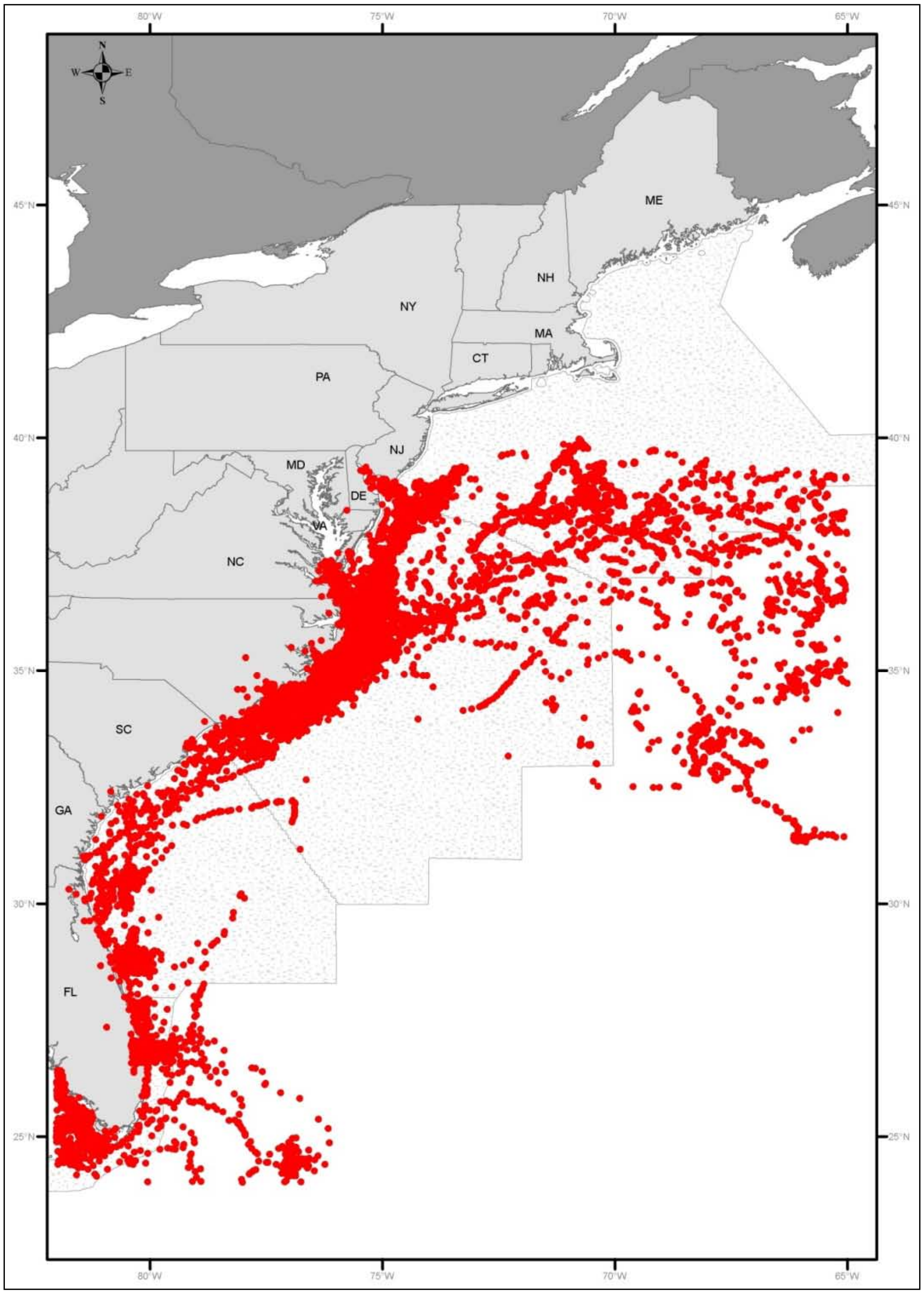
Map from http://www.nmfs.noaa.gov/pr/pdfs/rangemaps/loggerhead_turtle.pdf

Figure 2.38.1. Loggerhead global distribution.



Map from http://research.myfwc.com/features/view_article.asp?id=2411

Figure 2.38.2. Distribution of loggerhead nesting sites in Florida.



Data from OBIS.

Figure 2.38.3. Positions of satellite-tagged loggerhead turtles.

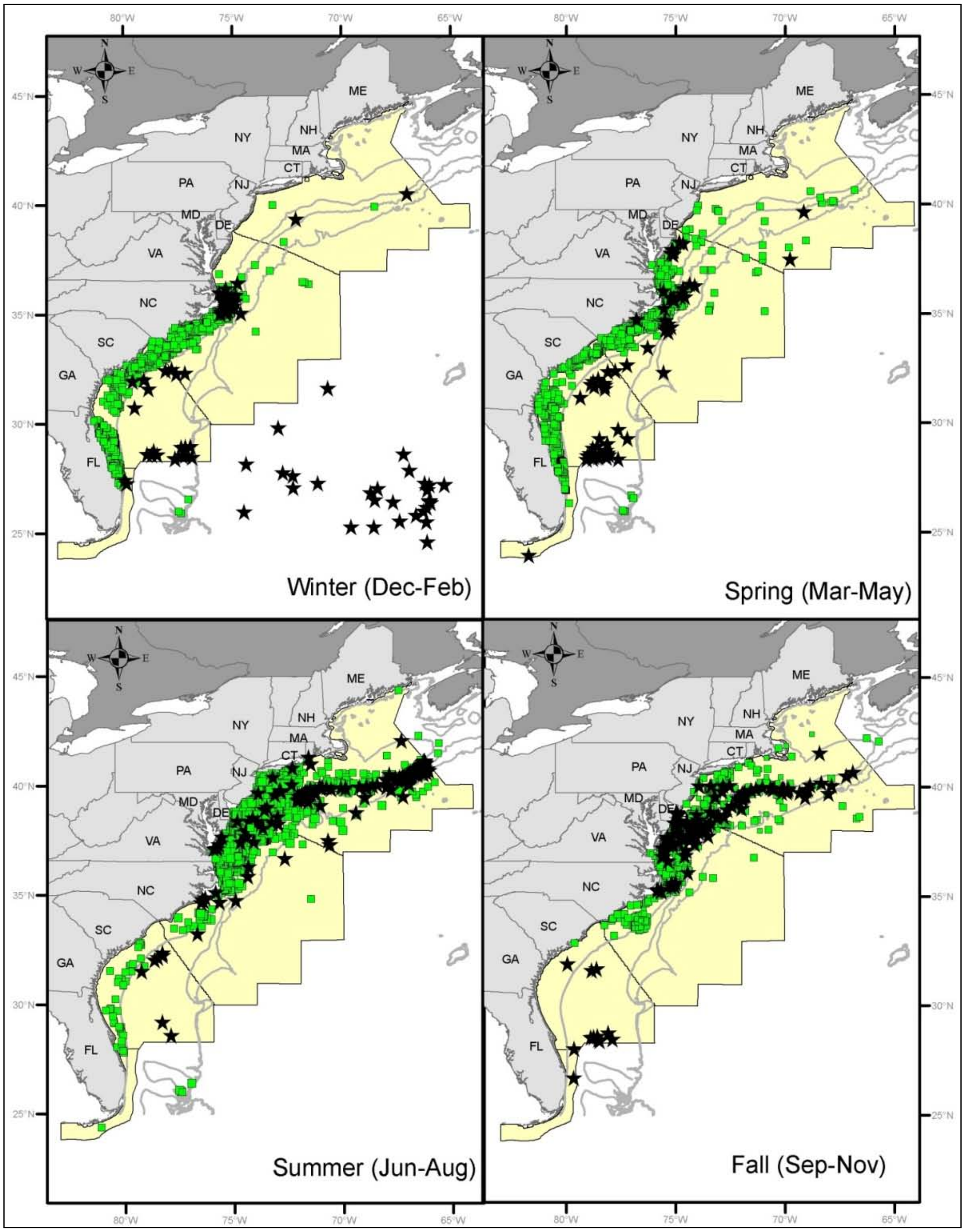
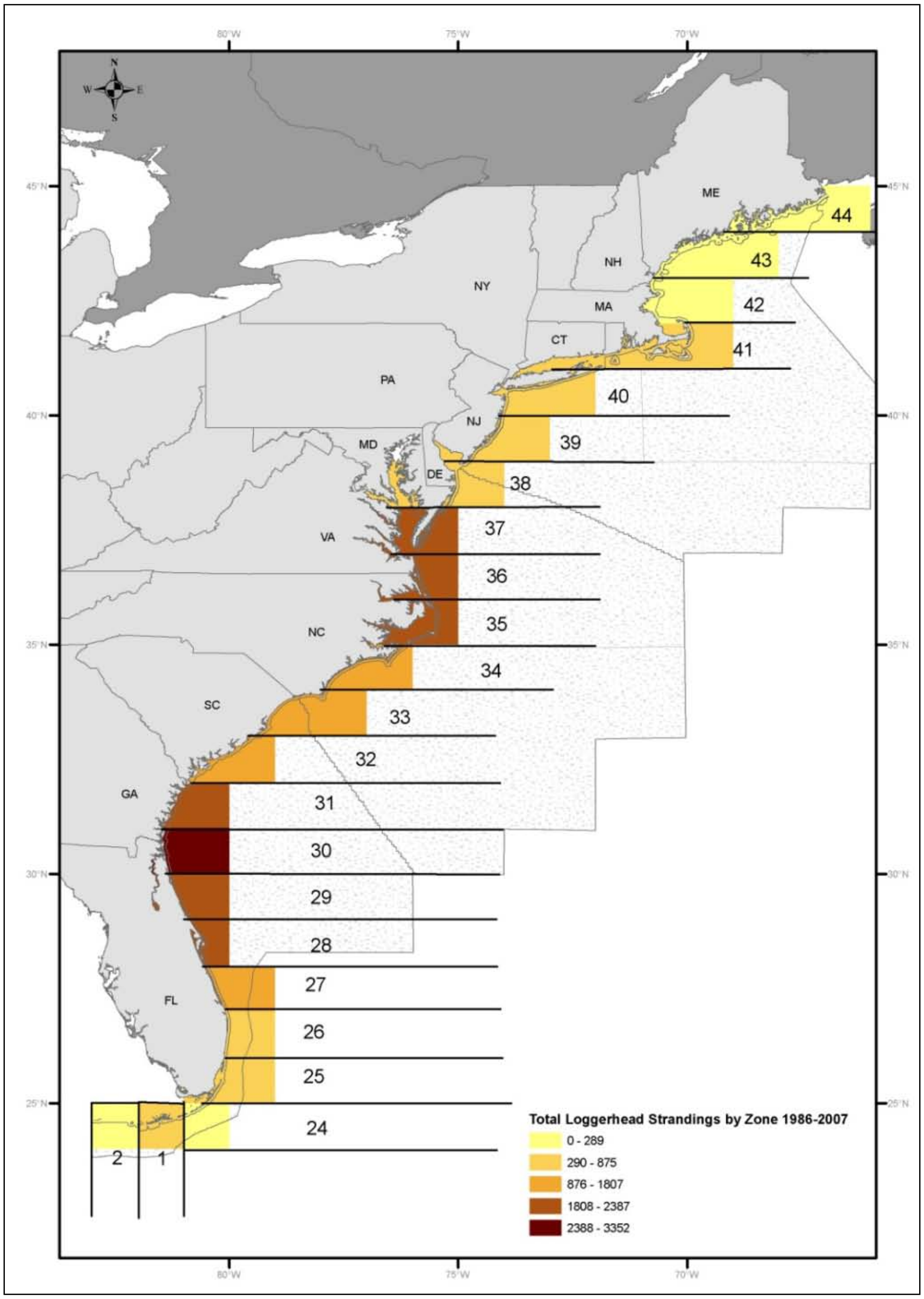
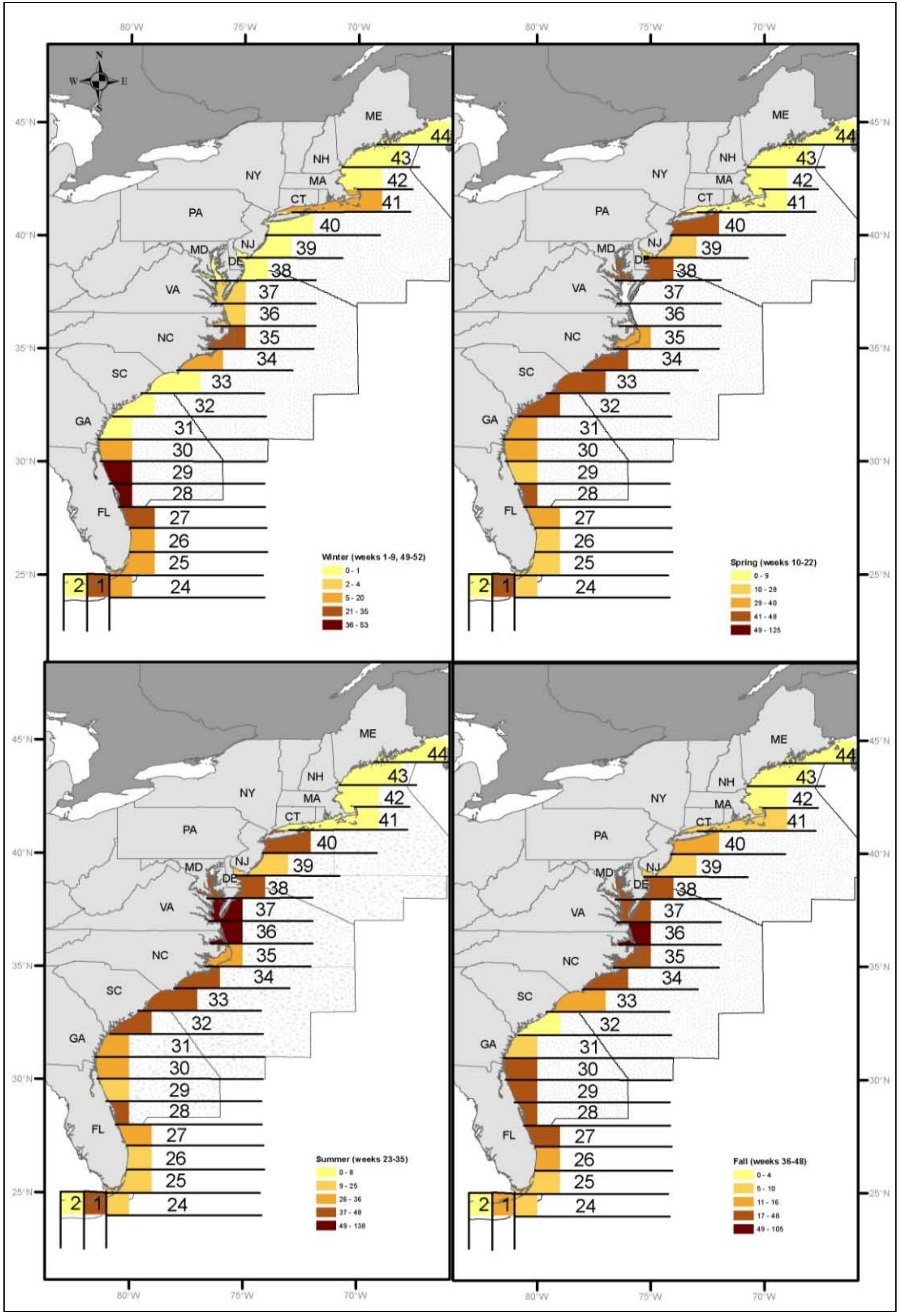


Figure 2.38.4. Loggerhead sea turtle sighting (green squares) and fishery by-catch (black stars) observations by season.



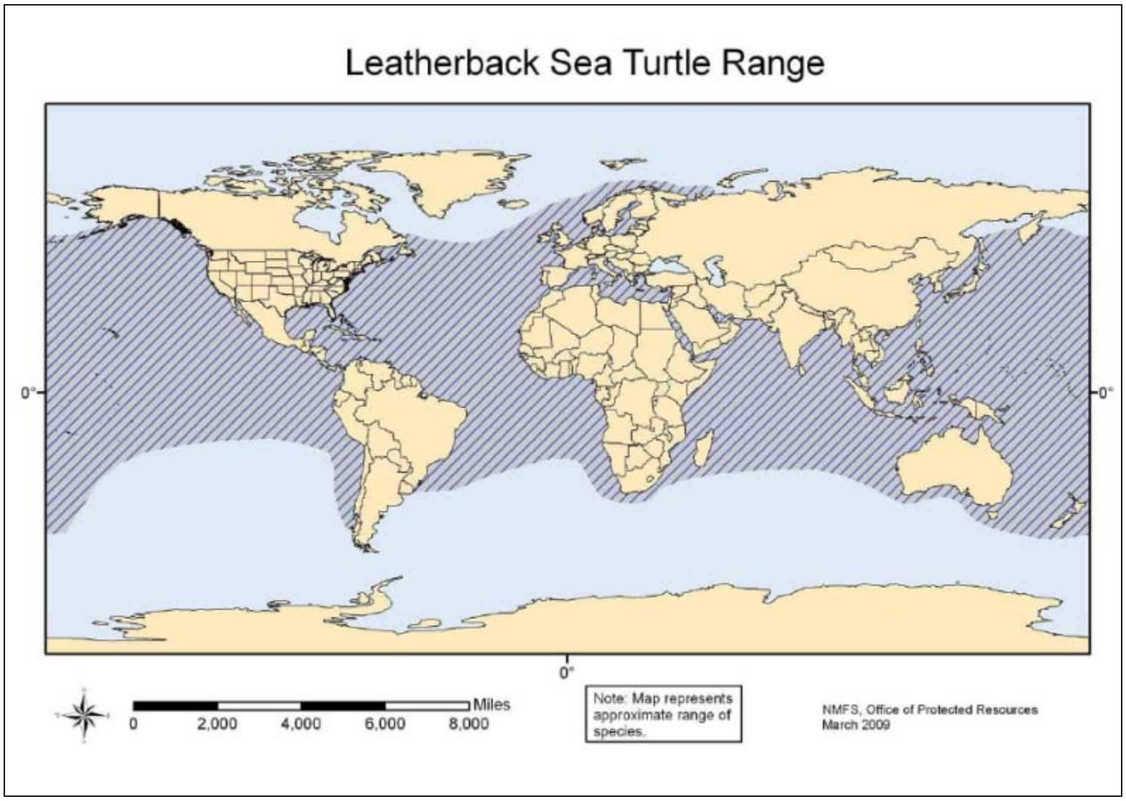
Data from the Sea Turtle Stranding and Salvage Network.

Figure 2.38.5. Loggerhead strandings by zone, 1986-2007.



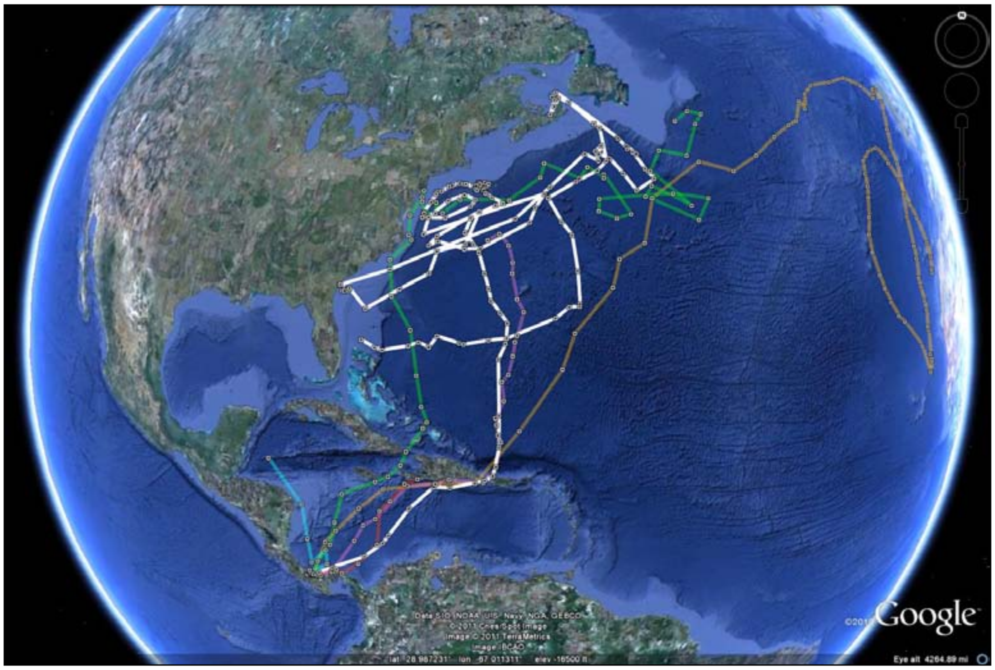
Data from the Sea Turtle Stranding and Salvage Network.

Figure 2.38.6. Seasonal loggerhead stranding data for years 2008 and 2009.



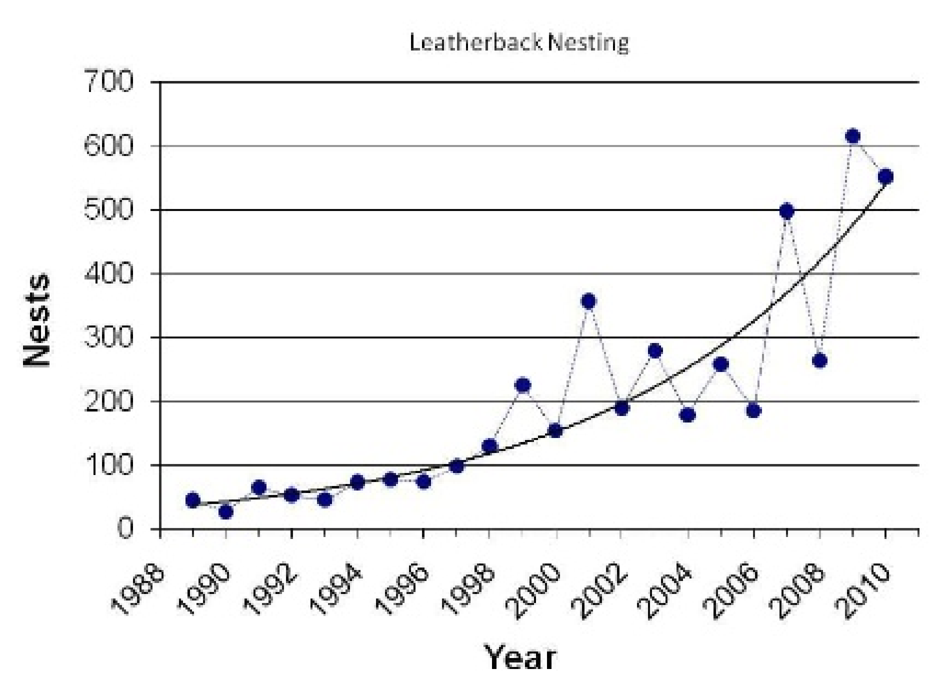
Map from http://www.nmfs.noaa.gov/pr/pdfs/rangemaps/leatherback_turtle.pdf

Figure 2.39.1. Leatherback global distribution.



Map from the Sea Turtle Conservancy (STC). Data and map © STC.

Figure 2.39.2. Selected leatherback tag location data 2005-2009.



Graph from http://research.myfwc.com/images/articles/10690/20100928_143948_17893.jpg

Figure 2.39.3. Leatherback nesting in Florida.

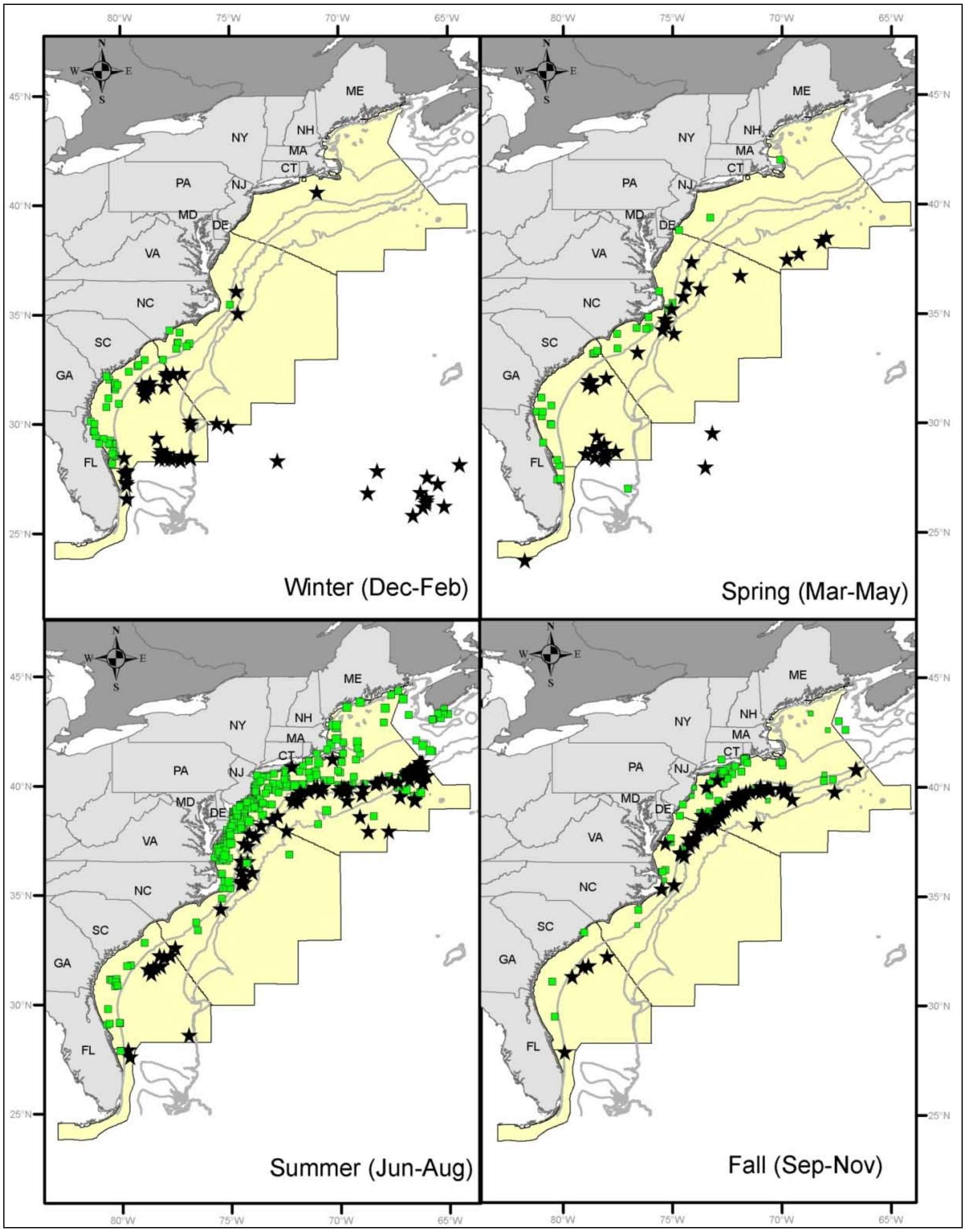
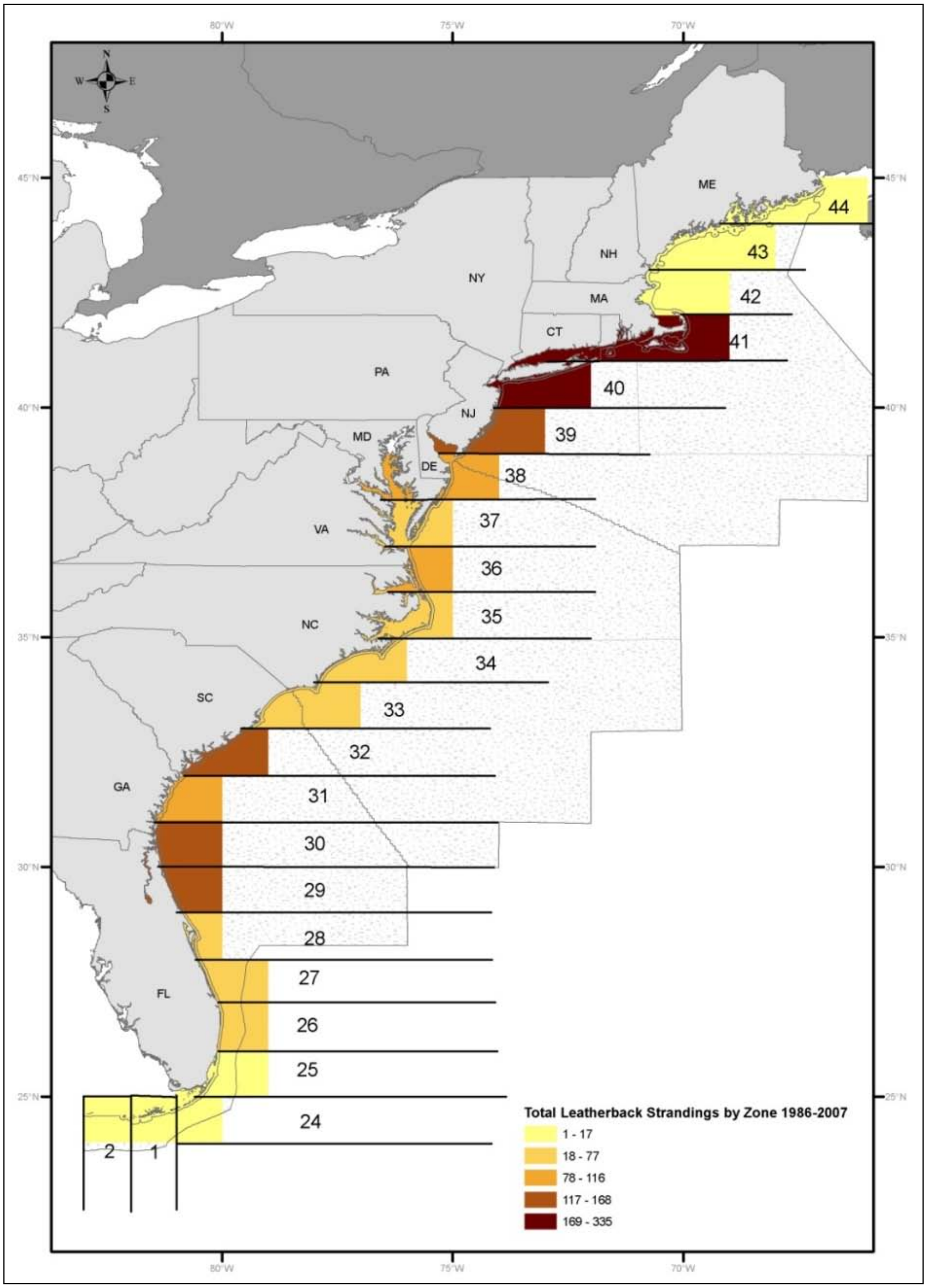
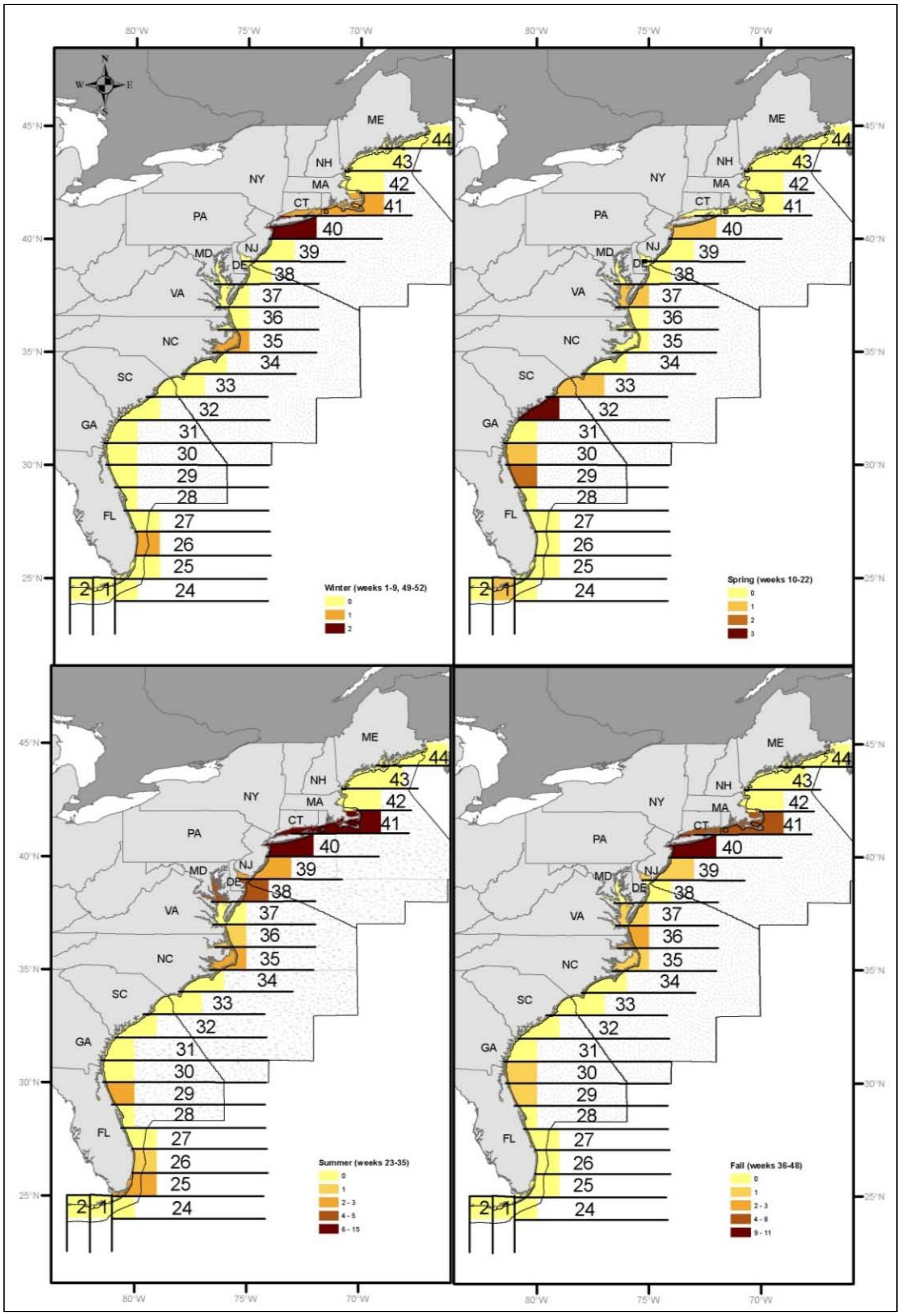


Figure 2.39.4. Leatherback sea turtle sighting (green squares) and fishery by-catch (black stars) observations by season.



Data from the Sea Turtle Stranding and Salvage Network.

Figure 2.39.5. Leatherback strandings by zone, 1986-2007.



Data from the Sea Turtle Stranding and Salvage Network.

Figure 2.39.6. Seasonal leatherback stranding data for years 2008 and 2009.

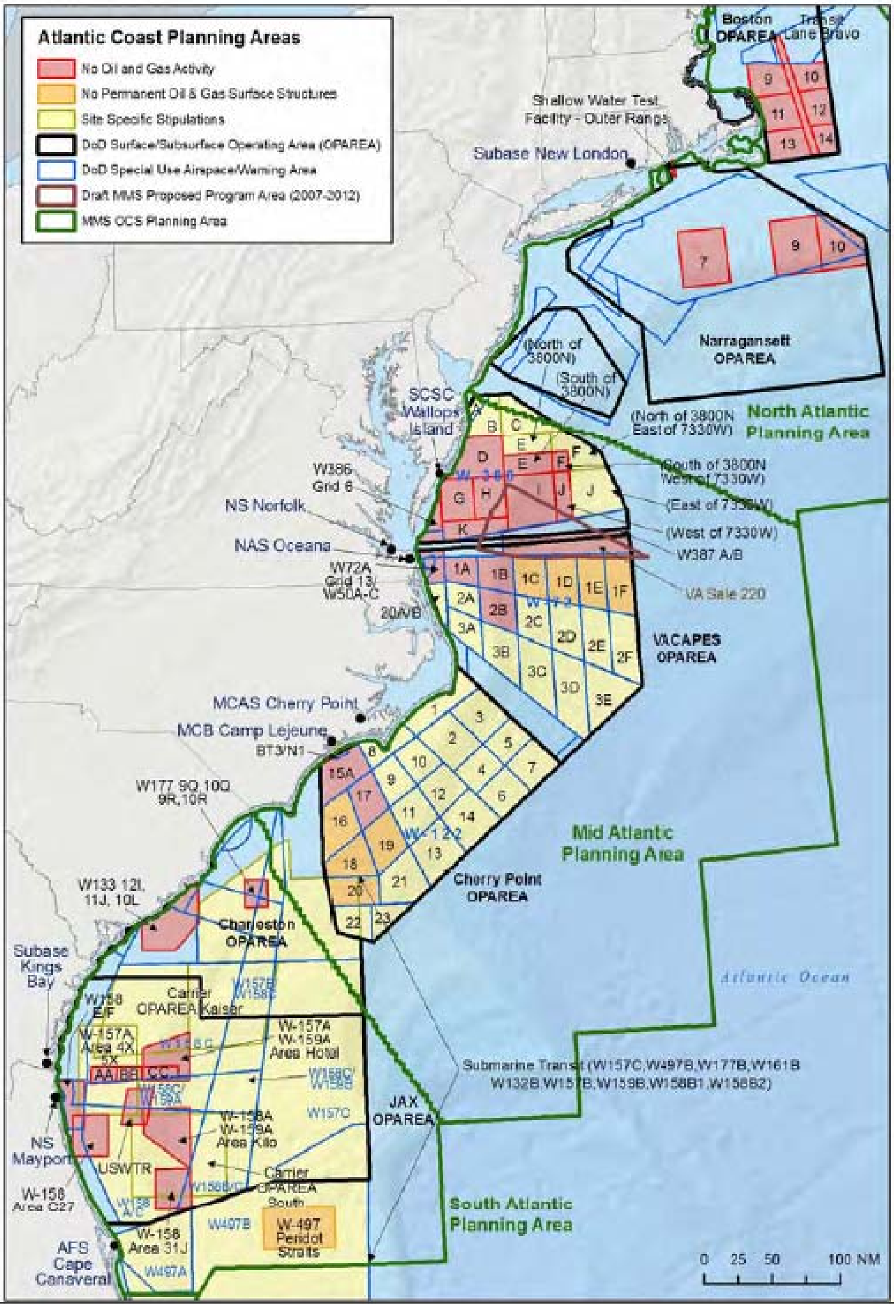


Figure 2.40.1. BOEM OCS Planning Areas and DoD OPAREAs 3 and Warning Areas.

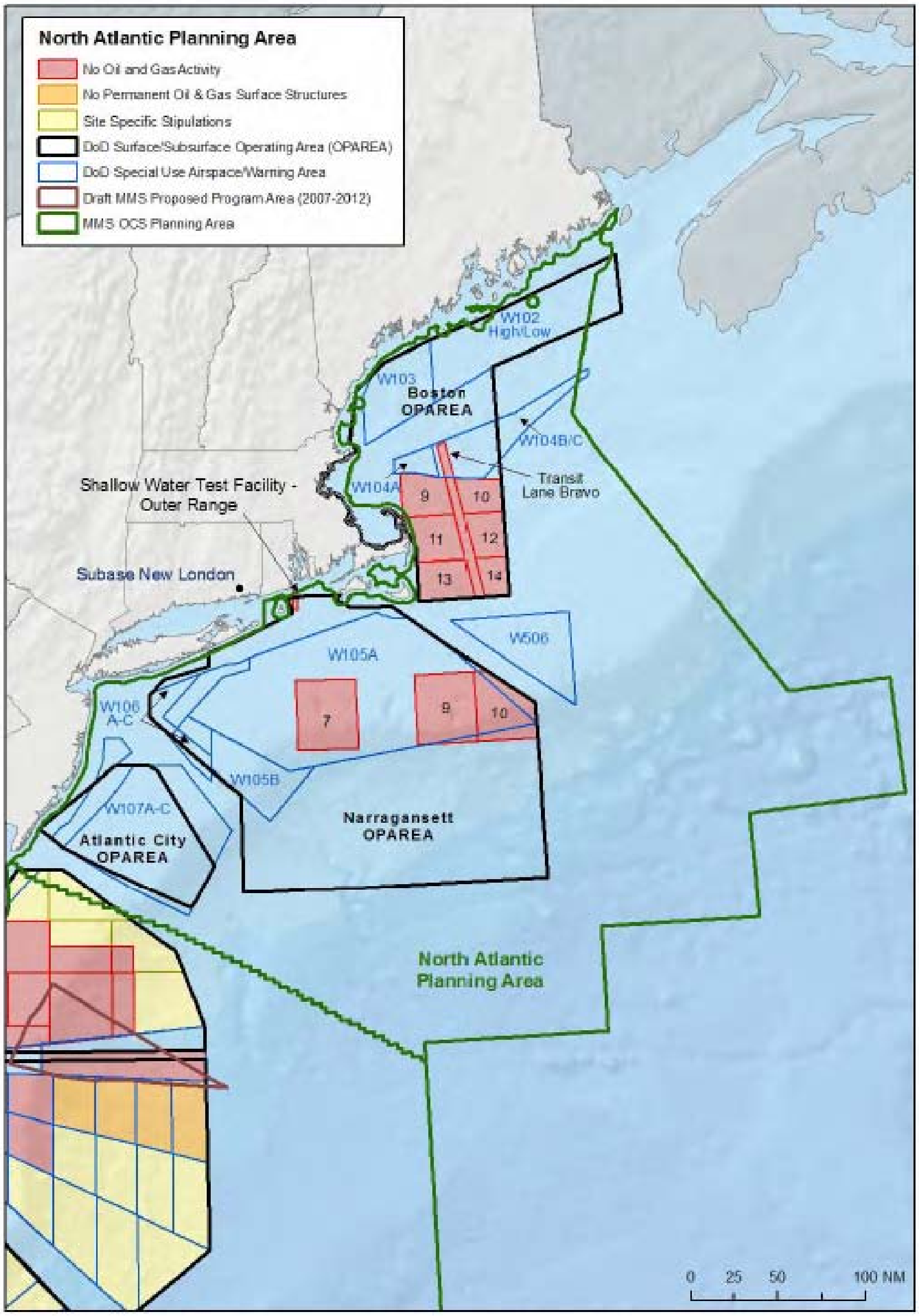


Figure 2.40.2. BOEM OCS North Atlantic Planning Area and DoD OPAREAs 3 and Warning Areas.

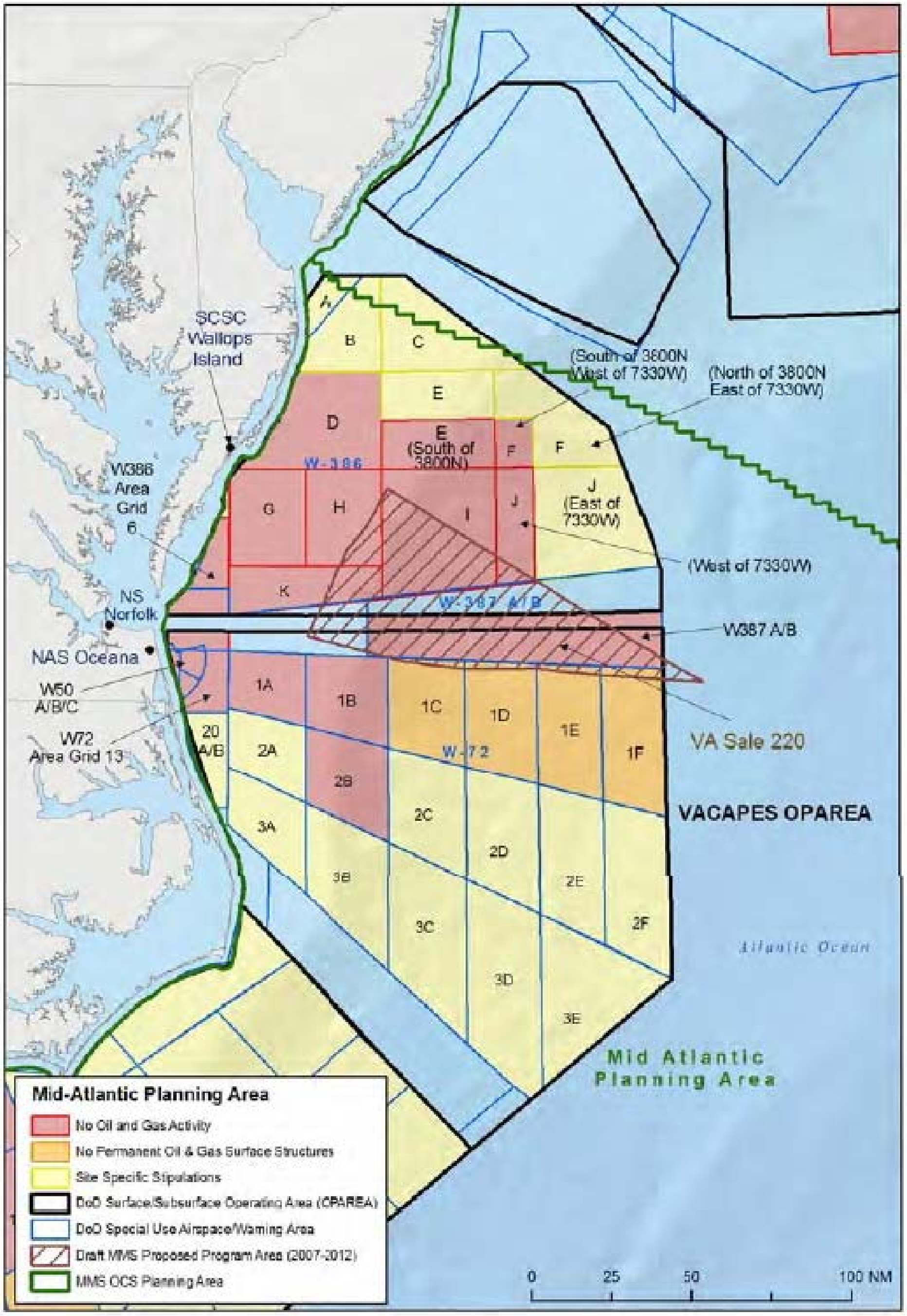


Figure 2.40.3. BOEM OCS Mid-Atlantic Planning Area and DoD OPAREAs 3 and Warning Areas.

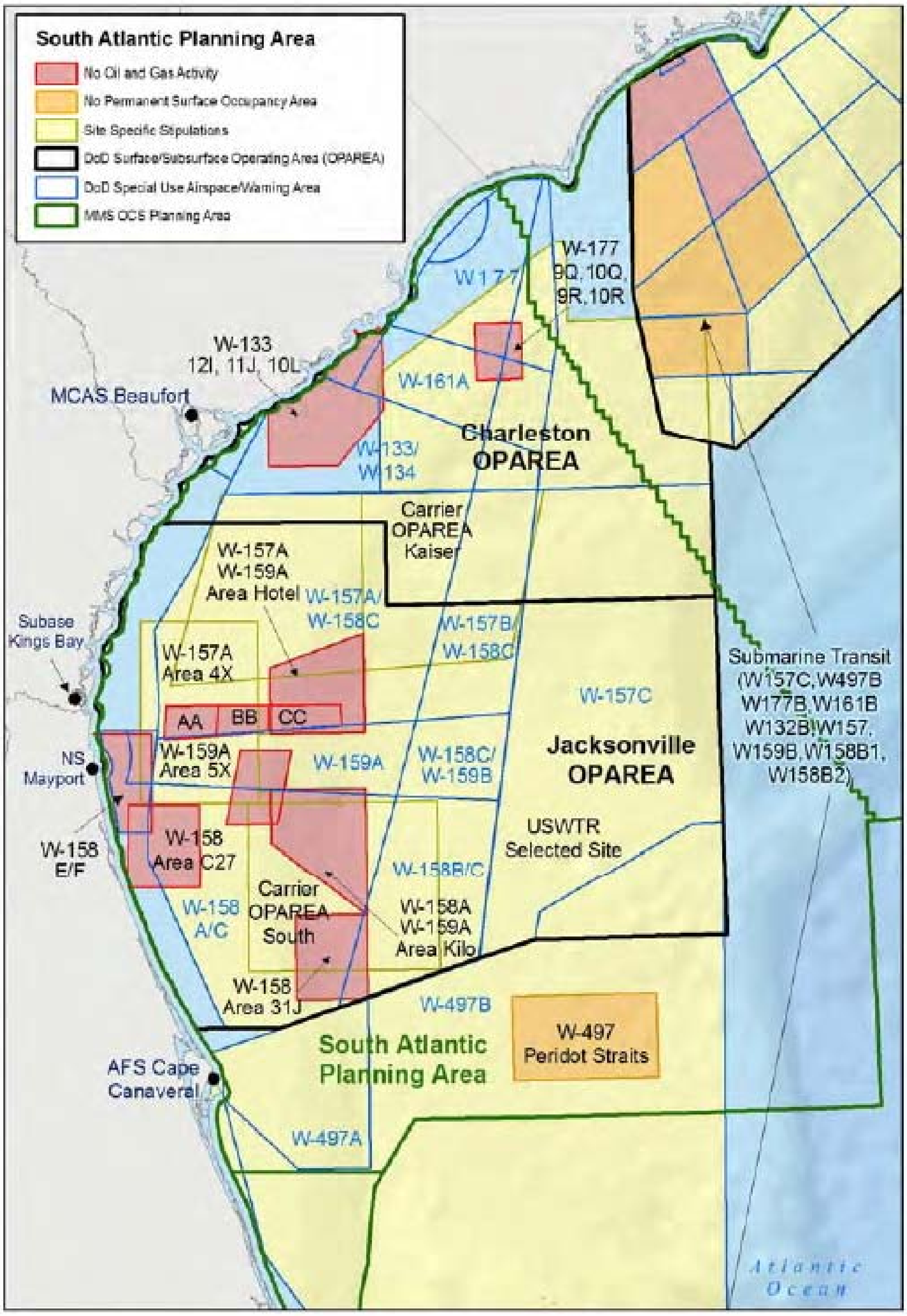


Figure 2.40.4. BOEM OCS South Atlantic Planning Area and DoD OPAREAs 3 and Warning Areas.

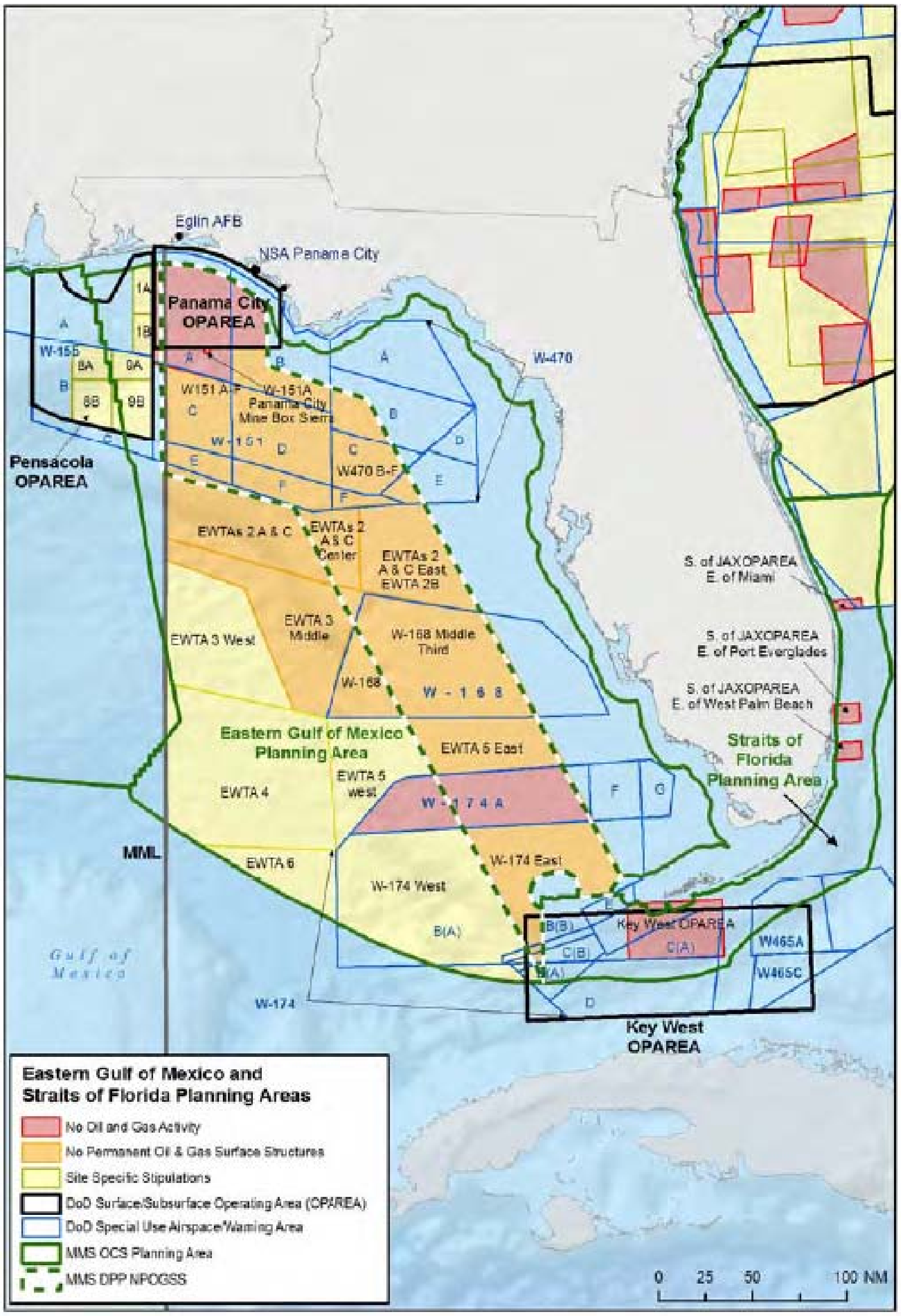


Figure 2.40.5. BOEM OCS Eastern Gulf of Mexico and Straits of Florida Planning Area and DoD OPAREAs 3 and Warning Areas.

3.0 CURRENT RESEARCH

Table 3.0

Current Research

(ASSTB&C 2011=31st Annual Symposium on Sea Turtle Biology and Conservation, SMM 2009 = Society for Marine Mammology Biennial Conference 2010, RWC2010= Right Whale Consortium Meeting 2010)

Species	**NAPA**	**MAPA**	**SAPA**	**SFPA**	**Title**	**Author**	**Institution**	**Email**	**Citation**
Balaenoptera acutorostrata	X				ABUNDANCE, DENSITY, AND DISTRIBUTION OF MARINE MAMMALS IN NEW JERSEY'S NEARSHORE WATERS	Whitt, Amy	Geo-Marine, Inc.	awhitt@geo-marine.com	SMM 2009
Balaenoptera acutorostrata	X				DETERMINING THE SEASONAL DISTRIBUTION OF CETACEANS IN NEW YORK COASTAL WATERS USING PASSIVE ACOUSTIC MONITORING	Biedron, Ingrid	Cornell University	ib49@cornell.edu	SMM 2009
Balaenoptera acutorostrata					MEASUREMENTS OF MECHANICAL PROPERTIES OF CETACEAN EARS	Zosuls, Aleks	Boston University	azosuls@bu.edu	SMM 2009
Balaenoptera acutorostrata					WAITING TIMES TO MONITOR MANAGEMENT ACTIONS: ATLANTIC LARGE WHALES SHOW NO RESPONSE TO GEAR RULES	Pace, Richard	NMFS-NEFSC	Richard.Pace@noaa.gov	SMM 2009
Balaenoptera acutorostrata	X				WANDERING WHALE WATCHES: AN ANALYSIS OF DATA COLLECTED ON WHALE WATCHES TRAVELING TO STELLWAGEN BANK, MA FROM 1994-1997	Koslovsky, Stacie	Duke Marine Lab	staciekoslovsky@gmail.com	SMM 2009

Species	NAPA	MAPA	SAPA	SFPA	Title	Author	Institution	Email	Citation
Balaenoptera borealis	X				DETERMINING THE SEASONAL DISTRIBUTION OF CETACEANS IN NEW YORK COASTAL WATERS USING PASSIVE ACOUSTIC MONITORING	Biedron, Ingrid	Cornell University	ib49@cornell.edu	SMM 2009
Balaenoptera musculus	X				DETERMINING THE SEASONAL DISTRIBUTION OF CETACEANS IN NEW YORK COASTAL WATERS USING PASSIVE ACOUSTIC MONITORING	Biedron, Ingrid	Cornell University	ib49@cornell.edu	SMM 2009
Balaenoptera physalus	X				ABUNDANCE, DENSITY, AND DISTRIBUTION OF MARINE MAMMALS IN NEW JERSEY'S NEARSHORE WATERS	Whitt, Amy	Geo-Marine, Inc.	awhitt@geo-marine.com	SMM 2009
Balaenoptera physalus	X				ANALYSIS OF MATERNAL AND PATERNAL CO-ANCESTRY OF PHOTOGRAPHICALLY-IDENTIFIED FIN WHALES, BALAENOPTERA PHYSALUS, IN THE GULF OF MAINE.	DenDanto, Dan	Univ. of Maine - Orono	dan.dendanto@umit.maine.edu	SMM 2009
Balaenoptera physalus	X				DETERMINING THE SEASONAL DISTRIBUTION OF CETACEANS IN NEW YORK COASTAL WATERS USING PASSIVE ACOUSTIC MONITORING	Biedron, Ingrid	Cornell University	ib49@cornell.edu	SMM 2009
Balaenoptera physalus					WAITING TIMES TO MONITOR MANAGEMENT ACTIONS: ATLANTIC LARGE WHALES SHOW NO RESPONSE TO GEAR RULES	Pace, Richard	NMFS-NEFSC	Richard.Pace@noaa.gov	SMM 2009
Balaenoptera physalus	X				WANDERING WHALE WATCHES: AN ANALYSIS OF DATA COLLECTED ON WHALE WATCHES TRAVELING TO STELLWAGEN BANK, MA FROM 1994-1997	Koslovsky, Stacie	Duke Marine Lab	staciekoslovsky@gmail.com	SMM 2009

Species	NAPA	MAPA	SAPA	SFPA	Title	Author	Institution	Email	Citation
Baleanopter musculus					MULTIMODAL SOMATOSENSORY SYSTEM IN MYSTICETI EXPLAINS EXTENDED BODY LENGTH AND LONG-DISTANCE SOURCE SIGNAL TRIANGULATION	Eldridge, Sherri	Univ. of Maine	newblueview@roadrunner.com	SMM 2009
Baleanopter physalus	X				ASSESSING GEOGRAPHIC AND ACOUSTIC MYSTICETE-SHIPSTRIKE RISK IN THE GULF OF MAINE	Peterson, Michael	University of Maine	Michael.Peterson@maine.edu	SMM 2009
Baleanopter physalus	X				DETERMINATION OF SPATIO-TEMPORAL HABITAT USE BY MYSTICETES IN THE NORTHEAST GULF OF MAINE USING A 20-YEAR OPPORTUNISTIC SIGHTING DATABASE	Todd, Sean	College of the Atlantic	stodd@coa.edu	SMM 2009
Baleanopter physalus	X				EVIDENCE FOR SPATIAL ASSOCIATIONS OF SYMPATRIC HUMPBACK AND FIN WHALES IN THE GULF OF MAINE USING MONTE CARLO SIMULATIONS	Lubansky, Tanya	NJIT	tml5@njit.edu	SMM 2009
Baleanopter physalus					MULTIMODAL SOMATOSENSORY SYSTEM IN MYSTICETI EXPLAINS EXTENDED BODY LENGTH AND LONG-DISTANCE SOURCE SIGNAL TRIANGULATION	Eldridge, Sherri	Univ. of Maine	newblueview@roadrunner.com	SMM 2009
Baleanopter physalus	X				UTILIZING THREE DECADES OF INDIVIDUAL PHOTOGRAPHIC-IDENTIFICATION DATA OF FINBACK WHALES (BALAENOPTERA PHYSALUS) IN THE WESTERN NORTH ATLANTIC	Vashro, Courtney	College of the Atlantic	vashrocourtney@yahoo.com	SMM 2009

Species	**NAPA**	**MAPA**	**SAPA**	**SFPA**	**Title**	**Author**	**Institution**	**Email**	**Citation**
Caretta caretta		X			A FISHERIES RELATED SEA TURTLE UNUSUAL MORTALITY EVENT IN NORTHAMPTON COUNTY, VIRGINIA, USA: A REPORT OF STRANDING ACTIVITY, NECROPSY FINDINGS AND FISHING EFFORTS	Christina Trapani	Virginia Aquarium Stranding Response Program, Virginia Beach, Virginia	mytwodolphins@hotmail.com	ASSTB&C 2011
Caretta caretta				X	BEACH RESTORATION AND ITS EFFECT ON LOGGERHEAD SEA TURTLE HATCHLING FITNESS IN FLORIDA	Mario Mota	University of Central Florida, Orlando	mario.mota@yahoo.com	ASSTB&C 2011
Caretta caretta			X		BEST MANAGEMENT PRACTICE DESIGN CRITERIA FOR HOPPER DREDGE/SEA TURTLE FRIENDLY BORROW SITES	Bates Phillip	US Army Corps of Engineers, Jacksonville, FL	phillip.c.bates@usace.army.mil	ASSTB&C 2011
Caretta caretta		X			CORRECTED LOGGERHEAD (CARETTA CARETTA) SEA TURTLE POPULATION DENSITY AND ABUNDANCE ESTIMATES IN ESTUARINE AND COASTAL WATERS OF NORTH CAROLINA, USA	Joanne Braun-McNeill	SEFSC Beaufort Laboratory	joanne.b.mcneill@noaa.gov	ASSTB&C 2011
Caretta caretta				X	DETERMINING THE IMPACTS OF BEACH RESTORATION ON LOGGERHEAD (CARETTA CARETTA) AND GREEN TURTLE (CHELONIA MYDAS) NESTING AND REPRODUCTIVE SUCCESS ALONG FLORIDA’S ATLANTIC COAST	Allison Hays	University of Central Florida, Orlando, Florida	allison.w.hays@gmail.com	ASSTB&C 2011

Species	**NAPA**	**MAPA**	**SAPA**	**SFPA**	**Title**	**Author**	**Institution**	**Email**	**Citation**
Caretta caretta			X		EFFECTS OF ROCK ARMORING STRUCTURES AND DUNE QUALITY ON THE NESTING PATTERNS OF THE LOGGERHEAD SEA TURTLE ON JEKYLL ISLAND, GEORGIA	Christina Martin	Georgia Sea Turtle Center, Jekyll Island, Georgia	christina_martin1@live.com	ASSTB&C 2011
Caretta caretta	X				ENTANGLEMENT OF SEA TURTLES IN VERTICAL LINES OF FIXED GEAR FISHERIES: A SUMMARY OF THE GEOGRAPHIC RANGE, SEASONALITY, INVOLVED SPECIES AND NATURE OF REPORTED ENTANGLEMENT EVENTS IN THE NORTHEAST REGION OF THE UNITED STATES	Kate Sampson	NOAA Fisheries Service Northeast Region, Gloucester, MA,	kate.sampson@noaa.gov	ASSTB&C 2011
Caretta caretta	X				EVALUATING SEA TURTLE INJURIES IN NORTHEAST REGION FISHING GEAR, USA	Carrie Upite	NMFS, Northeast Regional Office, Gloucester, Massachusetts	carrie.upite@noaa.gov	ASSTB&C 2011
Caretta caretta		X			EVALUATION OF A TURTLE EXCLUDER DEVICE (TED) DESIGNED FOR USE IN THE U.S. MID-ATLANTIC ATLANTIC CROAKER FISHERY	Jeff Gearhart	SEFSC	jeff.gearhart@noaa.gov	ASSTB&C 2011
Caretta caretta					EXPLORING DENSITY-DEPENDENCE IN LOGGERHEAD, CARETTA CARETTA, POPULATION VIABILITY ANALYSIS	Elena Finkbeiner	Duke University Marine Lab, Beaufort, North Carolina	efinkbeiner@seaturtle.org	ASSTB&C 2011

Species	**NAPA**	**MAPA**	**SAPA**	**SFPA**	**Title**	**Author**	**Institution**	**Email**	**Citation**
Caretta caretta					EXPONENTIAL GROWTH OF NEST PRODUCTION ON A "LOGGERHEAD BEACH": GREEN TURTLES AND LEATHERBACKS ON THE ARCHIE CARR NATIONAL WILDLIFE REFUGE, FL, USA	Christopher Long	University of Central Florida	clong@knights.ucf.edu	ASSTB&C 2011
Caretta caretta	X				FORAGING ECOLOGY OF LEATHERBACK SEA TURTLES IN THE WESTERN NORTH ATLANTIC DETERMINED THROUGH MULTI-TISSUE STABLE ISOTOPE ANALYSES	Kara Dwyer Dodge	Large Pelagics Research Center, University of New Hampshire	kara.dodge@unh.edu	ASSTB&C 2011
Caretta caretta			X	X	FORAGING HABITAT USE BY MALE LOGGERHEAD TURTLES AS REVEALED BY STABLE ISOTOPES	Mariela Pajuelo	Department of Biology and Archie Carr Center for Sea Turtle Research. University of Florida	mariela.pajuelo@gmail.com	ASSTB&C 2011
Caretta caretta			X		GENETIC MARK-RECAPTURE OF THE FEMALE LOGGERHEAD POPULATION NESTING IN GEORGIA, 2008-2009: ABUNDANCE, NEST SITE FIDELITY AND RELATEDNESS	Brian Shamblin	Warnell School of Forestry and Natural Resources, University of Georgia	brianshm@uga.edu	ASSTB&C 2011
Caretta caretta			X	X	HARDWIRED FOR NAVIGATION: HOW AN INHERITED MAGNETIC MAP GUIDES YOUNG LOGGERHEADS ON THEIR FIRST TRANSOCEANIC MIGRATION	Kenneth Lohmann	University of North Carolina, Chapel Hill	klohmann@email.unc.edu	ASSTB&C 2011

Species	NAPA	MAPA	SAPA	SFPA	Title	Author	Institution	Email	Citation
Caretta caretta		X	X	X	HOOKING POSITION AS A FUNCTION OF HOOK SIZE, BAIT TYPE, AND TURTLE SIZE IN LOGGERHEAD TURTLES (CARETTA CARETTA) INCIDENTALLY CAPTURED IN THE U.S. ATLANTIC PELAGIC LONGLINE FISHERY	Lesley Stokes	SEFSC	lesley.stokes@noaa.gov	ASSTB&C 2011
Caretta caretta		X	X	X	IDENTIFYING MIGRATORY PATHWAYS AND FORAGING HABITAT USE BY LOGGERHEAD TURTLES (CARETTA CARETTA) NESTING ON FLORIDA’S EAST COAST	Dan Evans	Sea Turtle Conservancy, Gainesville, Florida	drevans@conserveturtles.org	ASSTB&C 2011
Caretta caretta		X			INFERRING VESSEL CHARACTERISTICS FROM WOUNDS ON STRANDED SEA TURTLES: CAN WE APPLY THE MANATEE METHOD?	Susan Barco	Virginia Aquarium & Marine Science Center Foundation Research & Conservation Division	sgbarco@virginiaaquarium.com	ASSTB&C 2011
Caretta caretta			X		INITIAL INVESTIGATIONS OF VITAMIN D3 IN DISPLAY AND REHABILITATION SEA TURTLES	Shane Boylan	South Carolina Aquarium	drshaneboylan@gmail.com	ASSTB&C 2011
Caretta caretta		X			INTERACTIONS BETWEEN SEA TURTLES AND DREDGE GEAR IN THE U.S. SEA SCALLOP FISHERY, 2001-2008	Kimberly Murray	Northeast Fisheries Science Center	kimberly.murray@noaa.gov	ASSTB&C 2011
Caretta caretta			X		INVESTIGATING THE UTILITY OF GROUND PENETRATING RADAR FOR SEA-TURTLE NEST DETECTION	Christopher Hintz	Savannah State University, Savannah	hintzc@savannahstate.edu	ASSTB&C 2011

Species	**NAPA**	**MAPA**	**SAPA**	**SFPA**	**Title**	**Author**	**Institution**	**Email**	**Citation**
Caretta caretta			X	X	INVESTIGATION OF SEA TURTLE UNUSUAL MORTALITY EVENTS IN THE EASTERN UNITED STATES: 2001-2010	Brian Stacy	Marine Animal Disease Laboratory, University of Florida	stacyb@vetmed.ufl.edu	ASSTB&C 2011
Caretta caretta			X		JANUARY 2010 COLD STUN: TURTLES IN THE NORTH CENTRAL INDIAN RIVER LAGOON AND FATE OF 60 TURTLES SUBSEQUENTLY TRACKED	Jane Provancha	Kennedy Space Center	jane.a.provancha@nasa.gov	ASSTB&C 2011
Caretta caretta		X			LOGGERHEAD SEA TURTLE (CARETTA CARETTA) INTERACTIONS WITH U.S. MID-ATLANTIC BOTTOM TRAWL GEAR FOR FISH AND SCALLOPS, 2005-2008	Melissa Warden	NEFSC	melissa.warden@noaa.gov	ASSTB&C 2011
Caretta caretta			X	X	LOGGERHEAD SPECIALIZATION REVEALED BY STABLE ISOTOPE ANALYSIS	Hannah Vander Zanden	Department of Biology and Archie Carr Center for Sea Turtle Research. University of Florida	hvz@ufl.edu	ASSTB&C 2011
Caretta caretta			X		LOGGERHEADS (CARETTA CARETTA) ON THE RISE? NESTING IN THE ARCHIE CARR NWR, FLORIDA, USA IN 2010	Cheryl Sanchez	University of Central Florida, Orlando, Florida	cheryl.lynn.sanchez@gmail.com	ASSTB&C 2011
Caretta caretta			X		LONG-TERM STUDY OF LOGGERHEAD SEA TURTLE HATCHLING SEX RATIOS ON TWO GEORGIA BARRIER ISLANDS (2000-2010)	Kris Williams	Caretta Research Project, Savannah, GA	wassawcrp@aol.com	ASSTB&C 2011
Caretta caretta				X	MAMMALIAN PREDATION OF SEA TURTLE NESTS IN BOCA RATON, FL- FIFTEEN YEARS OF STUDY AND THE EFFECTIVENESS OF USING HABENERO PEPPER POWDER FOR CONTROL	Kirt Rusenko	Gumbo Limbo Naure Center	rusenkoki@aol.com	ASSTB&C 2011

Species	NAPA	MAPA	SAPA	SFPA	Title	Author	Institution	Email	Citation
Caretta caretta		X			MARYLAND, USA: A 20 YEAR SUMMARY OF DEAD SEA TURTLE STRANDINGS	Jamie Testa	Maryland Department of Natural Resources, Oxford, Maryland	jschofield@dnr.state.md.us	ASSTB&C 2011
Caretta caretta			X		MEDICAL AND SURGICAL MANAGEMENT OF WOUNDS IN MARINE TURTLES	Terry Norton	Georgia Sea Turtle Center, Jekyll Island	tnmynahvet@aol.com	ASSTB&C 2011
Caretta caretta			X		MULTIPLE PATERNITY WITHIN THE NORTHERN SUBPOPULATION OF LOGGERHEAD SEA TURTLE (CARETTA CARETTA)	Jacob Lasala	Georgia Southern University,	jlasala321@gmail.com	ASSTB&C 2011
Caretta caretta			X		NEST TEMPERATURES AND HATCHLING SEX RATIOS FROM LOGGERHEAD TURTLE NESTS INCUBATED UNDER NATURAL FIELD CONDITIONS IN GEORGIA, USA	Anne Marie LeBlanc	Georgia Southern University, Statesboro, Georgia	annemarielb@juno.com	ASSTB&C 2011
Caretta caretta	X	X	X	X	ORGANOHALOGEN CONTAMINANT CONCENTRATIONS VARY IN ADULT MALE LOGGERHEADS BASED ON MIGRATION PATTERNS	Jennifer Keller	College of Charleston	jennifer.keller@noaa.gov	ASSTB&C 2011
Caretta caretta		X			PHOTOGRAPHIC IDENTIFICATION OF SEA TURTLES IN THE ESTUARINE WATERS OF NORTH CAROLINA, USA	Mary Goodman	SEFSC Beaufort Laboratory, Beaufort, NC	april.goodman@noaa.gov	ASSTB&C 2011
Caretta caretta					QUANTUM LIGHT MEASUREMENT TO AID ASSESSMENT OF ANTHROPOGENIC AND NATURAL LIGHT INFLUENCES ON SEA TURTLE NESTING, TYBEE ISLAND, GA	Ana Reyes	Savannah State University, Savannah, Georgia	areyes@student.savannahstate.edu	ASSTB&C 2011

Species	**NAPA**	**MAPA**	**SAPA**	**SFPA**	**Title**	**Author**	**Institution**	**Email**	**Citation**
Caretta caretta				X	RECORD HIGH BEACH TEMPERATURES: IMPLICATIONS FOR LOGGERHEAD TURTLE (CARETTA CARETTA) HATCH AND EMERGENCE SUCCESS IN BOCA RATON, FLORIDA, USA	Micah Rogers	Florida Atlantic University, Boca Raton, Florida	micahm.rogers@gmail.com	ASSTB&C 2011
Caretta caretta			X		RELATING NESTING FEMALE BODY SIZE TO CLUTCH SIZE, NESTING FREQUENCY, AND HATCHING SUCCESS IN LOGGERHEAD SEA TURTLES, CARETTA CARETTA, AT JEKYLL ISLAND, GEORGIA, USA	Joseph Scarola	Georgia Sea Turtle Center, Jekyll Island, Georgia	jscarola8@hotmail.com	ASSTB&C 2011
Caretta caretta			X	X	REPORT ON A MASSIVE HYPOTHERMIC STUNNING EVENT OF SEA TURTLES IN FLORIDA DURING JANUARY 2010	Allen Foley	Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute	allen.foley@myfwc.com	ASSTB&C 2011
Caretta caretta			X	X	SATELLITE TRACKING CONFIRMS THE USE OF STABLE ISOTOPES TO INFER FORAGING GROUNDS OF LOGGERHEAD TURTLES (CARETTA CARETTA) NESTING ON FLORIDA'S EAST COAST	Simona Ceriani	University of Central Florida, Orlando	simona.ceriani@gmail.com	ASSTB&C 2011
Caretta caretta			X	X	SATELLITE TRACKING THE SEA TURTLE "LOST YEARS"	Kate Mansfield	Southeast Fisheries Science Center	kate.mansfield@noaa.gov	ASSTB&C 2011
Caretta caretta			X		SEA TURTLE NEST MANAGEMENT: EXAMINING THE USE OF RELOCATION AS A MANAGEMENT TOOL ON THREE SOUTH CAROLINA BEACHES, USA	Gretchen Coll	College of Charleston, Charleston, South Carolina	babyruth2414@yahoo.com	ASSTB&C 2011

Species	**NAPA**	**MAPA**	**SAPA**	**SFPA**	**Title**	**Author**	**Institution**	**Email**	**Citation**
Caretta caretta				X	SEA TURTLES AND OCEAN ENERGY TECHNOLOGY	Erin McMichael	Florida Atlantic University, Boca Raton, Florida	emcmich@hotmail.com	ASSTB&C 2011
Caretta caretta			X		SEASONAL VARIATION IN EGG SIZE IN THE LOGGERHEAD SEA TURTLE: RESOURCE PARTITIONING IN THE NESTING FEMALE	Ketan Patel	Georgia Southern University	patelkv@hendrix.edu	ASSTB&C 2011
Caretta caretta				X	SURVIVAL OF IMMATURE LOGGERHEAD TURTLES (CARETTA CARETTA) IN THE INDIAN RIVER LAGOON, FL, USA	Andrew Sterner	University of Central Florida, Orlando, FL	asterner816@yahoo.com	ASSTB&C 2011
Caretta caretta				X	THE EFFECT OF AIR TEMPERATURE ON THE INCUBATION PERIOD AND HATCHING SUCCESS OF LOGGERHEAD SEA TURTLE (CARETTA CARETTA) CLUTCHES IN BROWARD COUNTY, FLORIDA	L. Teal Kawana	Nova Southeastern University Oceanographic Center	ltkawana@gmail.com	ASSTB&C 2011
Caretta caretta		X	X		TWO CASES OF REHABILITATED LOGGERHEAD (CARETTA CARETTA) SEA TURTLES SURVIVING IN THE WILD AFTER RELEASE	Kelly Thorvalson	South Carolina Aquarium, Charleston, SC	kthorvalson@scaquarium.org	ASSTB&C 2011
Caretta caretta				X	USING GIS TO DETERMINE THE EFFECT OF SKYGLOW ON NESTING SEA TURTLES OVER A TEN YEAR PERIOD	Kirt Rusenko	Gumbo Limbo Nature Center, Boca Raton	rusenkoki@aol.com	ASSTB&C 2011
Caretta caretta				X	UTILIZATION OF TESTOSTERONE RIA TO EVALUATE THE SEX RATIO OF JUVENILE LOGGERHEADS INHABITING THE ATLANTIC COASTAL WATERS OF FLORIDA	Jennifer Estes Layton	Samford University, Birmingham, AL	jestes@uab.edu	ASSTB&C 2011

Species	NAPA	MAPA	SAPA	SFPA	Title	Author	Institution	Email	Citation
Caretta caretta		X			VERTICAL HABITAT UTILIZATION OF IMMATURE LOGGERHEAD SEA TURTLES (CARETTA CARETTA) IN U.S. MID-ATLANTIC SHELF WATERS	Heather Haas	Northeast Fisheries Science Center	heather.haas@noaa.gov	ASSTB&C 2011
Chelonia mydas			X	X	448 TURTLES IN THE FREEZER: NECROPSY AND POPULATION ASSESSMENT OF GREEN SEA TURTLES STRANDED DEAD IN ST. JOSEPH BAY, FLORIDA, USA, DURING THE JANUARY 2010 MASS COLD-STUNNING	Larisa Avens	SEFSC Beaufort Laboratory	larisa.avens@noaa.gov	ASSTB&C 2011
Chelonia mydas				X	DETERMINING THE IMPACTS OF BEACH RESTORATION ON LOGGERHEAD (CARETTA CARETTA) AND GREEN TURTLE (CHELONIA MYDAS) NESTING AND REPRODUCTIVE SUCCESS ALONG FLORIDA'S ATLANTIC COAST	Allison Hays	University of Central Florida, Orlando, Florida	allison.w.hays@gmail.com	ASSTB&C 2011
Chelonia mydas	X				ENTANGLEMENT OF SEA TURTLES IN VERTICAL LINES OF FIXED GEAR FISHERIES: A SUMMARY OF THE GEOGRAPHIC RANGE, SEASONALITY, INVOLVED SPECIES AND NATURE OF REPORTED ENTANGLEMENT EVENTS IN THE NORTHEAST REGION OF THE UNITED STATES	Kate Sampson	NOAA Fisheries Service Northeast Region, Gloucester, MA,	kate.sampson@noaa.gov	ASSTB&C 2011
Chelonia mydas			X	X	INVESTIGATION OF SEA TURTLE UNUSUAL MORTALITY EVENTS IN THE EASTERN UNITED STATES: 2001-2010	Brian Stacy	Marine Animal Disease Laboratory, University of Florida	stacyb@vetmed.ufl.edu	ASSTB&C 2011

Species	NAPA	MAPA	SAPA	SFPA	Title	Author	Institution	Email	Citation
Chelonia mydas			X		JANUARY 2010 COLD STUN: TURTLES IN THE NORTH CENTRAL INDIAN RIVER LAGOON AND FATE OF 60 TURTLES SUBSEQUENTLY TRACKED	Jane Provancha	Kennedy Space Center	jane.a.provancha@nasa.gov	ASSTB&C 2011
Chelonia mydas		X			MARYLAND, USA: A 20 YEAR SUMMARY OF DEAD SEA TURTLE STRANDINGS	Jamie Testa	Maryland Department of Natural Resources, Oxford, Maryland	jschofield@dnr.state.md.us	ASSTB&C 2011
Chelonia mydas			X		MEDICAL AND SURGICAL MANAGEMENT OF WOUNDS IN MARINE TURTLES	Terry Norton	Georgia Sea Turtle Center, Jekyll Island	tnmynahvet@aol.com	ASSTB&C 2011
Chelonia mydas		X			PHOTOGRAPHIC IDENTIFICATION OF SEA TURTLES IN THE ESTUARINE WATERS OF NORTH CAROLINA, USA	Mary Goodman	SEFSC Beaufort Laboratory, Beaufort, NC	april.goodman@noaa.gov	ASSTB&C 2011
Chelonia mydas			X	X	REPORT ON A MASSIVE HYPOTHERMIC STUNNING EVENT OF SEA TURTLES IN FLORIDA DURING JANUARY 2010	Allen Foley	Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute	allen.foley@myfwc.com	ASSTB&C 2011
Chelonia mydas				X	SEA TURTLES AND OCEAN ENERGY TECHNOLOGY	Erin McMichael	Florida Atlantic University, Boca Raton, Florida	emcmich@hotmail.com	ASSTB&C 2011
Chelonia mydas				X	SUN COMPASS ORIENTATION BY JUVENILE GREEN SEA TURTLES (CHELONIA MYDAS)	Cody Mott	Gumbo Limbo Nature Center, Boca Raton	codyrmott@hotmail.com	ASSTB&C 2011
Chelonia mydas		X			TIMING AND PATHWAYS OF FALL MIGRATION FOR JUVENILE GREEN SEA TURTLES IN BACK AND CORE SOUNDS, NORTH CAROLINA	Amanda Southwood	University of North Carolina Wilmington	southwooda@uncw.edu	ASSTB&C 2011

Species	NAPA	MAPA	SAPA	SFPA	Title	Author	Institution	Email	Citation
Chelonia mydas			X	X	USE OF TRACE ELEMENTS TO INFER POPULATION CONNECTIVITY IN MARINE ENVIRONMENTS: GETTING THE GOOD FROM THE BAD	Melania Cecilia Lopez Castro	Department of Biology and Archie Carr Center for Sea Turtle Research. University of Florida	melania_lopez@yahoo.com.mx	ASSTB&C 2011
Chelonia mydas				X	USING GIS TO DETERMINE THE EFFECT OF SKYGLOW ON NESTING SEA TURTLES OVER A TEN YEAR PERIOD	Kirt Rusenko	Gumbo Limbo Nature Center, Boca Raton	rusenkoki@aol.com	ASSTB&C 2011
Cystophora cristata	X				CHARACTERISTICS OF PHOCID SEAL BYCATCH IN NEW ENGLAND FISHERIES.	Belden, Dana	Office of Naval Research	dana.belden.ctr@navy.mil	SMM 2009
Cystophora cristata	X				HARP SEAL (PHOCA GROENLANDICA) AND HOODED SEAL (CYSTOPHORA CRISTATA) TRENDS IN ABUNDANCE IN THE NORTHERN GULF OF MAINE, USA FROM 1994-2009.	Seton, Rosemary	College of the Atlantic	rseton@coa.edu	SMM 2009
Delphinus delphis	X				ABUNDANCE, DENSITY, AND DISTRIBUTION OF MARINE MAMMALS IN NEW JERSEY'S NEARSHORE WATERS	Whitt, Amy	Geo-Marine, Inc.	awhitt@geo-marine.com	SMM 2009
Delphinus delphis					ACOUSTIC ANALYSIS OF WHISTLE CHARACTERISTICS OF DELPHINID SPECIES IN THE NW ATLANTIC.	Schiebel, Hayley	NOVA Southeastern University	hayhay121@hotmail.com	SMM 2009
Dermochleys coriacea	X	X	X	X	UNDERWATER HEARING SENSITIVITY IN THE LEATHERBACK SEA TURTLE (DERMOCHELYS CORIACEA): ASSESSING THE POTENTIAL EFFECT OF ANTHROPOGENIC NOISE (NT-10-X33)	Michael Rasser	Wider Caribbean Sea Turtle Conservation Network (Widecast)	Michael.Rasser@boemre.gov?subject=Underwater%20Hearing%20Sensitivity%20in%20the%20Leatherback%20Sea%20Turtle	http://www.boemre.gov/eppd/PDF/EPPDStudies/Leatherback_Sea_Turtle_Audiogram.pdf

Species	NAPA	MAPA	SAPA	SFPA	Title	Author	Institution	Email	Citation
Dermochleys coriacea	X				ENTANGLEMENT OF SEA TURTLES IN VERTICAL LINES OF FIXED GEAR FISHERIES: A SUMMARY OF THE GEOGRAPHIC RANGE, SEASONALITY, INVOLVED SPECIES AND NATURE OF REPORTED ENTANGLEMENT EVENTS IN THE NORTHEAST REGION OF THE UNITED STATES	Kate Sampson	NOAA Fisheries Service Northeast Region, Gloucester, MA,	kate.sampson@noaa.gov	ASSTB&C 2011
Dermochleys coriacea		X			INFERRING VESSEL CHARACTERISTICS FROM WOUNDS ON STRANDED SEA TURTLES: CAN WE APPLY THE MANATEE METHOD?	Susan Barco	Virginia Aquarium & Marine Science Center Foundation Research & Conservation Division	sgbarco@virginiaaquarium.com	ASSTB&C 2011
Dermochleys coriacea			X		MEDICAL AND SURGICAL MANAGEMENT OF WOUNDS IN MARINE TURTLES	Terry Norton	Georgia Sea Turtle Center, Jekyll Island	tnmynahvet@aol.com	ASSTB&C 2011
Dermochleys coriacea				X	SEA TURTLES AND OCEAN ENERGY TECHNOLOGY	Erin McMichael	Florida Atlantic University, Boca Raton, Florida	emcmich@hotmail.com	ASSTB&C 2011
Dermochlys coriacea		X			MARYLAND, USA: A 20 YEAR SUMMARY OF DEAD SEA TURTLE STRANDINGS	Jamie Testa	Maryland Department of Natural Resources, Oxford, Maryland	jschofield@dnr.state.md.us	ASSTB&C 2011
Dermochlys coriacea			X	X	SELENIUM SAVES THE DAY: FIRST EXPLANATION FOR DECREASED HATCH AND EMERGENCE SUCCESS IN LEATHERBACK SEA TURTLES (DERMOCHELYS CORIACEA)-LESSONS LEARNED FROM TWO POPULATIONS	Justin Perrault	Florida Atlantic University	jperrau2@fau.edu	ASSTB&C 2011

Species	**NAPA**	**MAPA**	**SAPA**	**SFPA**	**Title**	**Author**	**Institution**	**Email**	**Citation**
Eretmochleys imbricata				X	A COMPARISON OF HAWKSBILL TURTLE SITE OCCUPANCY BETWEEN NATURAL AND ARTIFICIAL REEFS IN PALM BEACH COUNTY, FL, USA	Suzanne Nuttall	Green Mountain College, Poultney, VT	suzannenuttall@hotmail.com	ASSTB&C 2011
Eretmochleys imbricata					AMPHIBIOUS HEARING IN HATCHLING HAWKSBILL SEA TURTLES (ERETMOCHELYS IMBRICATA)	Wendy Dow Piniak	Duke University Marine Lab	wed3@duke.edu	ASSTB&C 2011
Eretmochleys imbricata				X	GPS- LINKED SATELLITE TELEMETRY OF HAWKSBILL TURTLES (ERETMOCHELYS IMBRICATA) IN PALM BEACH COUNTY, FLORIDA, USA	Lawrence Wood	Zoological Society of the Palm Beaches	wood9794@bellsouth.net	ASSTB&C 2011
Eretmochleys imbricata			X		JANUARY 2010 COLD STUN: TURTLES IN THE NORTH CENTRAL INDIAN RIVER LAGOON AND FATE OF 60 TURTLES SUBSEQUENTLY TRACKED	Jane Provancha	Kennedy Space Center	jane.a.provancha@nasa.gov	ASSTB&C 2011
Eretmochleys imbricata			X	X	REPORT ON A MASSIVE HYPOTHERMIC STUNNING EVENT OF SEA TURTLES IN FLORIDA DURING JANUARY 2010	Allen Foley	Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute	allen.foley@myfwc.com	ASSTB&C 2011
Eretmochleys imbricata				X	SEA TURTLES AND OCEAN ENERGY TECHNOLOGY	Erin McMichael	Florida Atlantic University, Boca Raton, Florida	emcmich@hotmail.com	ASSTB&C 2011
Eretmochleys imbricata				X	USING GIS TO DETERMINE THE EFFECT OF SKYGLOW ON NESTING SEA TURTLES OVER A TEN YEAR PERIOD	Kirt Rusenko	Gumbo Limbo Nature Center, Boca Raton	rusenkoki@aol.com	ASSTB&C 2011

Species	NAPA	MAPA	SAPA	SFPA	Title	Author	Institution	Email	Citation
Eubalaena glacialis	X				A MODEL OF ENTANGLEMENT RISK FOR LOBSTER FISHING GEAR OFF THE COAST OF MAINE	Chris Brehme	Keene State College	cbrehme@keene.edu	RWC2010
Eubalaena glacialis	X				A NEW ANALYSIS TESTING THE LIKELIHOOD OF JORDAN BASIN BEING A MATING GROUND FOR RIGHT WHALES	Tim Cole	Northeast Fisheries Science Center	Tim.Cole@noaa.gov	RWC2010
Eubalaena glacialis	X		X		AMBIENT NOISE LEVELS AND ACOUSTIC ECOLOGY OF NORTH ATLANTIC RIGHT WHALE HABITATS	Charles A. Muirhead	Cornell Bioacoustics Research Program	cam236@cornell.edu	RWC2010
Eubalaena glacialis	X				BEHAVIORAL DIFFERENCES OF FORAGING NORTH ATLANTIC RIGHT AND HUMPBACK WHALES IN APRIL 2009 AND 2010	Susan E. Parks	Appl. Res. Lab., Penn State, State College, PA	sep20@psu.edu	RWC2010
Eubalaena glacialis	X				COMPARATIVE ANALYSIS OF THE OCCURRENCE AND SEASONALITY PATTERNS OF NORTH ATLANTIC RIGHT WHALE CALL TYPES IN THE NORTHWEST ATLANTIC	Sarah Mussoline	WHOI/ Northeast Fisheries Science Center	sarah.mussoline@noaa.gov	RWC2010
Eubalaena glacialis	X				COMPUTER MODELING OF WHALE ENTANGLEMENT	Laurens E. Howle	Duke University Department of Mechanical Engineering and Materials Science	laurens.howle@duke.edu	RWC2010
Eubalaena glacialis	X				CORRELATION OF ZOOPLANKTON PREY ABUNDANCE AND DISTRIBUTION WITH THE BEHAVIOR OF TAGGED RIGHT WHALES IN CAPE COD BAY DURING SPRING 2010	Joseph D. Warren	School of Marine and Atmospheric Sciences, Stonybrook University	joe.warren@stonybrook.edu	RWC2010

Species	**NAPA**	**MAPA**	**SAPA**	**SFPA**	**Title**	**Author**	**Institution**	**Email**	**Citation**
Eubalaena glacialis	X				CURRENT GEAR RESEARCH AND COMPLIANCE RATES WITHIN THE MAINE COASTAL LOBSTER FISHERY	Erin Summers	Maine Division of Marine Resources	erin.l.summers@maine.gov	RWC2010
Eubalaena glacialis	X				DETERMINING THE ABSOLUTE PROBABILITY OF LETHAL VESSEL-STRIKES TO NORTH ATLANTIC RIGHT WHALES, EUBALAENA GLACIALIS, IN THE ROSEWAY BASIN REGION OF THE SCOTIAN SHELF	Julie van der Hoop	WHOI	jvanderhoop@whoi.edu	RWC2010
Eubalaena glacialis	X				ENTANGLEMENT SEVERITY RELATED TO ROPE CHARACTERISTICS FOR NORTH ATLANTIC RIGHT WHALES	Amy Knowlton	New England Aquarium	aknowlton@neaq.org	RWC2010
Eubalaena glacialis	X				ESTIMATED ABSORBANCE SPECTRA OF THE VISUAL PIGMENTS OF THE NORTH ATLANTIC RIGHT WHALE (EUBALAENA GLACIALIS) AND THEIR POTENTIAL ROLE IN DETECTING CONCENTRATIONS OF THE CALANOID COPEPOD CALANUS FINMARCHICUS	Jeffry I. Fasick	Kean University	jfasick@kean.edu	RWC2010
Eubalaena glacialis	X				FINE-SCALE SWIMMING KINEMATICS OF NORTH ATLANTIC RIGHT WHALES: USING LOCOMOTOR GAITS TO DETERMINE VARIATIONS IN ENERGY EXPENDITURE	Anna E. McGregor	Duke University	anna.mcgregor@alumni.duke.edu	RWC2010

Species	**NAPA**	**MAPA**	**SAPA**	**SFPA**	**Title**	**Author**	**Institution**	**Email**	**Citation**
Eubalaena glacialis	X				FIRST USE OF A CUTTING BROADHEAD TO RESOLVE A RIGHT WHALE ENTANGLEMENT: A NOVEL TOOL TO DIVERSIFY DISENTANGLEMENT OPTIONS AND ENHANCE RESPONDER SAFETY	Scott Landry	PCCS	sclandry@coastalstudies.org	RWC2010
Eubalaena glacialis	X				GAUGING EFFECTS OF ANTHROPOGENIC MORTALITY REDUCTION ON NORTH ATLANTIC RIGHT WHALE RECOVERY: A WHAT IF PVA	Richard M. Pace, III.	Northeast Fisheries Science Center	Richard.Pace@noaa.gov	RWC2010
Eubalaena glacialis	X				HOW MUCH IS ENOUGH? ESTIMATING THE NUTRITIONAL REQUIREMENTS OF NORTH ATLANTIC RIGHT WHALES WITH A GENERALIZED BIOENERGETICS MODEL	Sarah Fortune	Marine Mammal Research Unit, University of British Columbia	s.fortune@fisheries.ubc.ca	RWC2010
Eubalaena glacialis	X				METHODOLOGY OF COASTAL MAINE LOBSTER FISHERY GEAR CONFIGURATION DATA COLLECTION	Heather Tetreault	Maine Lobstermens Association	heather@mainelobstermen.org	RWC2010
Eubalaena glacialis	X				MONITORING RIGHT WHALE DISTRIBUTION AND HABITAT IN THE OUTER FALL REGION WEST OF JORDAN BASIN DURING LATE AUTUMN WITH AUTONOMOUS VEHICLES	Mark Baumgartner	WHOI	mbaumgartner@whoi.edu	RWC2010

Species	**NAPA**	**MAPA**	**SAPA**	**SFPA**	**Title**	**Author**	**Institution**	**Email**	**Citation**
Eubalaena glacialis	X				NORTHERN ATLANTIC RIGHT WHALE UPCALLS: UNDERSTANDING CALL CHARACTERISTICS, PATTERNS AND BEHAVIORAL ECOLOGY AT DIFFERENT SPATIAL AND TEMPORAL SCALES	Sofie Van Parijs	Northeast Fisheries Science Center	sofie.vanparijs@noaa.gov	RWC2010
Eubalaena glacialis			X		OBSERVATIONS OF A NORTH ATLANTIC RIGHT WHALE (EUBALAENA GLACIALIS) BIRTH OFFSHORE OF THE PROTECTED SOUTHEAST U.S. CRITICAL HABITAT	Heather Foley	Duke University	Heather.Foley@duke.edu	RWC2010
Eubalaena glacialis	X				OCEAN FLOOR ENCOUNTERS BY RIGHT WHALES: IMPLICATIONS FOR ENTANGLEMENT RISK FROM FISHING GEAR	Scott D. Kraus	New England Aquarium	skraus@neaq	RWC2010
Eubalaena glacialis	X				OUTREACH AND EDUCATION TO INCREASE VESSEL COMPLIANCE AND COMMITMENT IN THE CAPE COD BAY AND OFF RACE POINT RIGHT WHALE SEASONAL MANAGEMENT AREAS	David Wiley	Stellwagen Bank Natl. Marine Sanctuary	David.Wiley@noaa.gov	RWC2010
Eubalaena glacialis			X		PASSIVE ACOUSTIC MONITORING: MOTHER-CALF PAIRS IN THE SOUTHEAST UNITED STATES UNLIKELY TO BE DETECTED	James H.W. Hain	Associated Scientists at Woods Hole	jhain@earthlink.net	RWC2010
Eubalaena glacialis	X				ROSEWAY BASIN RIGHT WHALE CRITICAL HABITAT: SPACE-TIME VARIATION IN THE PREY-FIELD AND ASSOCIATED WATER MASS CHARACTERISTICS	Kimberley Davies	Dalhousie University	kim.davies@dal.ca	RWC2010

Species	**NAPA**	**MAPA**	**SAPA**	**SFPA**	**Title**	**Author**	**Institution**	**Email**	**Citation**
Eubalaena glacialis	X				SINKING GROUNDLINE REPERCUSSIONS IN MAINE’S LOBSTER INDUSTRY	Laura Ludwig	Maine Division of Marine Resources	laura.ludwig@maine.gov	RWC2010
Eubalaena glacialis	X				THE GROSS MORPHOLOGY OF THE MELON IN A NEONATE RIGHT WHALE (EUBALAENA GLACIALIS)	Cally Harper	Brown University	caroline_harper@brown.edu	RWC2010
Eubalaena glacialis	X				THE INFLUENCE OF COPEPOD DIEL VERTICAL MIGRATION ON COMPETITION BETWEEN RIGHT WHALES AND ZOOPLANKTIVOROUS FISH	Mark Baumgartner	WHOI	mbaumgartner@whoi.edu	RWC2010
Eubalaena glacialis	X				THE RIGHT (AND WRONG) WAYS TO PROTECT RIGHT WHALES	Sierra Weaver	Defenders of Wildlife	sweaver@defenders.org	RWC2010
Eubalaena glacialis	X				VESSEL COMPLIANCE AND COMMITMENT IN THE CAPE COD BAY AND OFF RACE POINT RIGHT WHALE SEASONAL MANAGEMENT AREAS	Michael Thompson	Stellwagen Bank Natl. Marine Sanctuary	michael.a.thompson@noaa.gov	RWC2010
Eubalaena glacialis			X		VESSEL COMPLIANCE WITH RIGHT WHALE PROTECTION MEASURES IN THE SOUTHEASTERN U.S. SEASONAL MANAGEMENT AREA DURING THE 2009-2010 CALVING SEASON	Joel G. Ortega-Ortiz	Florida Wildlife Research Institute	joelgortega@gmail.com	RWC2010
Eubalaena glacialis	X				ABUNDANCE, DENSITY, AND DISTRIBUTION OF MARINE MAMMALS IN NEW JERSEY'S NEARSHORE WATERS	Whitt, Amy	Geo-Marine, Inc.	awhitt@geo-marine.com	SMM 2009

Species	**NAPA**	**MAPA**	**SAPA**	**SFPA**	**Title**	**Author**	**Institution**	**Email**	**Citation**
Eubalaena glacialis	X				ASSESSING GEOGRAPHIC AND ACOUSTIC MYSTICETE-SHIPSTRIKE RISK IN THE GULF OF MAINE	Peterson, Michael	University of Maine	Michael.Peterson@maine.edu	SMM 2009
Eubalaena glacialis	X				ASSESSING LONG-TERM FORAGING IN NORTH ATLANTIC RIGHT WHALES WITH BIOGEOCHEMICAL TRACERS.	Lysiak, Nadine	WHOI	nlysiak@whoi.edu	SMM 2009
Eubalaena glacialis					CHEMICAL SEDATION OF ENTANGLED RIGHT WHALES TO ENHANCE DISENTANGLEMENT EFFORTS	Smith, Jamison	NMFS-NERO	Jamison.Smith.noaa.gov	SMM 2009
Eubalaena glacialis	X				DETERMINATION OF SPATIO-TEMPORAL HABITAT USE BY MYSTICETES IN THE NORTHEAST GULF OF MAINE USING A 20-YEAR OPPORTUNISTIC SIGHTING DATABASE	Todd, Sean	College of the Atlantic	stodd@coa.edu	SMM 2009
Eubalaena glacialis	X				DETERMINING THE SEASONAL DISTRIBUTION OF CETACEANS IN NEW YORK COASTAL WATERS USING PASSIVE ACOUSTIC MONITORING	Biedron, Ingrid	Cornell University	ib49@cornell.edu	SMM 2009
Eubalaena glacialis	X	X	X	X	GIARDIA AND CRYPTOSPORIDIUM IN NORTH ATLANTIC AND SOUTHERN RIGHT WHALES (EUBALAENA GLACIALIS AND E. AUSTRALIS)	Rolland, Roz	NEAq	rrolland@neaq.org	SMM 2009

Species	NAPA	MAPA	SAPA	SFPA	Title	Author	Institution	Email	Citation
Eubalaena glacialis		X			LIVE STRANDING OF A SUB-ADULT NORTH ATLANTIC RIGHT WHALE (EUBALAENA GLACIALIS) WITH EVIDENCE OF CHRONIC VERTEBRAL COLUMN RESPONSE TO LINE ENTANGLEMENT	McLellan, William	UNC Wilmington	mclellanw@uncw.edu	SMM 2009
Eubalaena glacialis	X				NEAR REALTIME SPECIES DISTRIBUTION MODELING OF NORTH ATLANTIC RIGHT WHALE HABITAT	Pendleton, Daniel	Cornell University	dep22@cornell.edu	SMM 2009
Eubalaena glacialis		X	X		NORTH ATLANTIC RIGHT WHALE (EUBALAENA GLACIALIS) SIGHTINGS IN THE US MID-ATLANTIC AND SOUTHEAST ATLANTIC BIGHT (VIRGINIA THROUGH SOUTH CAROLINA) FROM 2001-2008	Pabst, D. Ann	UNC Wilmington	pabsta@uncw.edu	SMM 2009
Eubalaena glacialis	X				POPULATION TRENDS IN NORTH ATLANTIC RIGHT WHALES: WHY HAVEN'T THEY INCREASED LIKE EUBALAENA AUSTRALIS?.	Kraus, Scott	NEAq	skraus@neaq.org	SMM 2009
Eubalaena glacialis	X				POPULATION TRENDS IN NORTH ATLANTIC RIGHT WHALES: WHY HAVEN'T THEY INCREASED LIKE EUBALAENA AUSTRALIS?.	Good, Caroline Paddock	Duke Marine Lab	cpg2@duke.edu	SMM 2009
Eubalaena glacialis	X	X	X	X	REDUCING NORTH ATLANTIC RIGHT WHALE SHIP STRIKES—THE US EXPERIENCE	Merrick, Richard	NMFS-NEFSC	Richard.Merrick@noaa.gov	SMM 2009

Species	**NAPA**	**MAPA**	**SAPA**	**SFPA**	**Title**	**Author**	**Institution**	**Email**	**Citation**
Eubalaena glacialis					WAITING TIMES TO MONITOR MANAGEMENT ACTIONS: ATLANTIC LARGE WHALES SHOW NO RESPONSE TO GEAR RULES	Pace, Richard	NMFS-NEFSC	Richard.Pace@noaa.gov	SMM 2009
Globicephala macrorhynchus					ACOUSTIC ANALYSIS OF WHISTLE CHARACTERISTICS OF DELPHINID SPECIES IN THE NW ATLANTIC.	Schiebel, Hayley	Nova Southeastern U.	hayhay121@hotmail.com	SMM 2009
Globicephala macrorhynchus		X			DISCRIMINATING BETWEEN FREE-RANGING LONG-FINNED (GLOBICEPHALA MELAS) AND SHORT-FINNED (GLOBICEPHALAMACRORHYNCHUS) PILOT WHALES OFF THE EAST COAST OF THE UNITED STATES	Rone, Brenda	NOAA-NEFSC	Brenda.Rone@noaa.gov	SMM 2009
Globicephala macrorhynchus		X			FORAGING ECOLOGY OF SHORT-FINNED PILOT WHALES OFF CAPE HATTERAS, NORTH CAROLINA, USA AND OBSERVATIONS OF INTERACTIONS WITH THE PELAGIC LONGLINE FISHERY	Urian, Kim	Duke Marine Lab	kurian@ec.rr.com	SMM 2009
Globicephala macrorhynchus		X			ODONTOCETE VOCALIZATIONS IN ONSLOW BAY, NORTH CAROLINA: INTEGRATING DATA FROM TWO PASSIVE ACOUSTIC TECHNIQUES	Williams, Lynne	Duke Marine Lab	lw32@duke.edu	SMM 2009
Globicephala melas		X			DISCRIMINATING BETWEEN FREE-RANGING LONG-FINNED (GLOBICEPHALA MELAS) AND SHORT-FINNED (GLOBICEPHALAMACRORHYNCHUS) PILOT WHALES OFF THE EAST COAST OF THE UNITED STATES	Rone, Brenda	NOAA-NEFSC	Brenda.Rone@noaa.gov	SMM 2009

Species	**NAPA**	**MAPA**	**SAPA**	**SFPA**	**Title**	**Author**	**Institution**	**Email**	**Citation**
Grampus griseus					ACOUSTIC ANALYSIS OF WHISTLE CHARACTERISTICS OF DELPHINID SPECIES IN THE NW ATLANTIC.	Schiebel, Hayley	Nova Southeastern U.	hayhay121@hotmail.com	SMM 2009
Grampus griseus		X			AERIAL AND VESSEL SURVEYS OF THE UNDERSEA WARFARE TRAINING RANGE SITE ALTERNATIVE IN ONSLOW BAY, NC, USA	Nilsson, Peter	UNC Wilmington	nilssonp@uncw.edu	SMM 2009
Halichoerus grypus	X				AN INCREASE IN GRAY SEAL (HALICHOERUS GRYPUS) SIGHTINGS AND STRANDINGS IN NEW YORK WATERS.	DiGiovanni, Robert	Riverhead Fdn for Marine Res & Pres	rdigiovanni@riverheadfoundation.org	SMM 2009
Halichoerus grypus	X				AN INCREASE IN THE NUMBER OF GRAY SEAL (HALICHOERUS GRYPUS) PUPS OBSERVED IN U.S. WATERS BETWEEN 1994 AND 2008: RE-ESTABLISHMENT OF FORMER PUPPING SITES	Wood, Stephanie	Umass. Boston	Stephanie.Wood@noaa.gov	SMM 2009
Halichoerus grypus	X				CHARACTERISTICS OF PHOCID SEAL BYCATCH IN NEW ENGLAND FISHERIES.	Belden, Dana	Office of Naval Research	dana.belden.ctr@navy.mil	SMM 2009
Halichoerus grypus	X				ESTIMATING MARINE MAMMAL CONSUMPTION OF COMMERCIALLY AND ECOLOGICALLY IMPORTANT PREY SPECIES ON THE NORTHEAST US CONTINENTAL SHELF	Col, Laurel	NOAA-NEFSC	Laurel.Col@noaa.gov	SMM 2009

Species	NAPA	MAPA	SAPA	SFPA	Title	Author	Institution	Email	Citation
Halichoerus grypus	X				PHOTOGRAPHIC SURVEYS OF ENTANGLEMENT OCCURRENCE AT GRAY SEAL (HALICHOERUS GRYPUS) AND HARBOR SEAL (PHOCA VITULINA) HAUL-OUTS ON CAPE COD, MASSACHUSETTS	Sette, Lisa	PCCS	sette@coastalstudies.org	SMM 2009
Halichoerus grypus	X				THE DIET OF GRAY SEALS (HALICHOERUS GRYPUS) IN UNITED STATES WATERS, ESTIMATED FROM SCATS, STOMACH SAMPLES AND FATTY ACID PROFILES	Ampela, Kristen	CUNY-Staen Island	krs10a@gmail.com	SMM 2009
Kogia spp.	X	X	X	X	ANALYSIS AND REVIEW OF KOGIA STRANDING DATA FROM THE EAST COAST OF THE UNITED STATES (1994-2008): WHAT CAN IT TELL US AND WHAT ARE WE MISSING?	Litz, Jenny	NOAA-SEFSC	Jenny.Litz@noaa.gov	SMM 2009
Kogia spp.				X	CORRELATIONS BETWEEN OCEANOGRAPHIC VARIABLES AND KOGIA STRANDINGS IN FLORIDA	Cordero, Vincent	Nova Southeastern U.	vcordero@nova.edu,	SMM 2009
Kogia spp.		X	X	X	DOMOIC ACID EXPOSURE IN PYGMY AND DWARF SPERM WHALES (KOGIA SPP.) FROM SOUTHEASTERN AND MID-ATLANTIC U.S. WATERS	Fire, Spencer	NOAA-NOS	Spencer.Fire@noaa.gov	SMM 2009
Kogia spp.					FIRST ASSESSMENT OF PYGMY AND DWARF SPERM WHALES (KOGIA SPP.) STOCK STRUCTURE IN THE NORTHWESTERN ATLANTIC AND NORTHERN GULF OF MEXICO	Viricel, Amelia	University of Louisiana	viricel@louisiana.edu	SMM 2009

Species	NAPA	MAPA	SAPA	SFPA	Title	Author	Institution	Email	Citation
Lagenorhynchus acutus	X				HABITAT PREFERENCES OF WHITE-SIDED DOLPHINS IN THE NORTHWEST ATLANTIC	Palka, Deborah	NOAA-NEFSC	Debra.Palka@noaa.gov	SMM 2009
Lepidochelys kempii		X			A FISHERIES RELATED SEA TURTLE UNUSUAL MORTALITY EVENT IN NORTHAMPTON COUNTY, VIRGINIA, USA: A REPORT OF STRANDING ACTIVITY, NECROPSY FINDINGS AND FISHING EFFORTS	Christina Trapani	Virginia Aquarium Stranding Response Program, Virginia Beach, Virginia	mytwodolphins@hotmail.com	ASSTB&C 2011
Lepidochelys kempii				X	DETERMINING THE IMPACTS OF BEACH RESTORATION ON LOGGERHEAD (CARETTA CARETTA) AND GREEN TURTLE (CHELONIA MYDAS) NESTING AND REPRODUCTIVE SUCCESS ALONG FLORIDA’S ATLANTIC COAST	Allison Hays	University of Central Florida, Orlando, Florida	allison.w.hays@gmail.com	ASSTB&C 2011
Lepidochelys kempii		X			INFERRING VESSEL CHARACTERISTICS FROM WOUNDS ON STRANDED SEA TURTLES: CAN WE APPLY THE MANATEE METHOD?	Susan Barco	Virginia Aquarium & Marine Science Center Foundation Research & Conservation Division	sgbarco@virginiaaquarium.com	ASSTB&C 2011
Lepidochelys kempii			X	X	INVESTIGATION OF SEA TURTLE UNUSUAL MORTALITY EVENTS IN THE EASTERN UNITED STATES: 2001-2010	Brian Stacy	Marine Animal Disease Laboratory, University of Florida	stacyb@vetmed.ufl.edu	ASSTB&C 2011
Lepidochelys kempii		X			MARYLAND, USA: A 20 YEAR SUMMARY OF DEAD SEA TURTLE STRANDINGS	Jamie Testa	Maryland Department of Natural Resources, Oxford, Maryland	jschofield@dnr.state.md.us	ASSTB&C 2011
Lepidochelys kempii			X		MEDICAL AND SURGICAL MANAGEMENT OF WOUNDS IN MARINE TURTLES	Terry Norton	Georgia Sea Turtle Center, Jekyll Island	tnmynahvet@aol.com	ASSTB&C 2011

Species	**NAPA**	**MAPA**	**SAPA**	**SFPA**	**Title**	**Author**	**Institution**	**Email**	**Citation**
Lepidochelys kempii		X			PHOTOGRAPHIC IDENTIFICATION OF SEA TURTLES IN THE ESTUARINE WATERS OF NORTH CAROLINA, USA	Mary Goodman	SEFSC Beaufort Laboratory, Beaufort, NC	april.goodman@noaa.gov	ASSTB&C 2011
Lepidochelys kempii			X	X	REPORT ON A MASSIVE HYPOTHERMIC STUNNING EVENT OF SEA TURTLES IN FLORIDA DURING JANUARY 2010	Allen Foley	Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute	allen.foley@myfwc.com	ASSTB&C 2011
Lepidochelys kempii				X	SEA TURTLES AND OCEAN ENERGY TECHNOLOGY	Erin McMichael	Florida Atlantic University, Boca Raton, Florida	emcmich@hotmail.com	ASSTB&C 2011
Megaptera novaeangliea					A NOVEL ANALYSIS OF HIERACHICAL STRUCTURE IN HUMPBACK WHALE (MEGAPTERA NOVAEANGLIAE) SONG	Handel, Stephen	Univ. of Tenessee, Knoxville	stephen.handel@gmail.com	SMM 2009
Megaptera novaeangliea	X				ABUNDANCE, DENSITY, AND DISTRIBUTION OF MARINE MAMMALS IN NEW JERSEY'S NEARSHORE WATERS	Whitt, Amy	Geo-Marine, Inc.	awhitt@geo-marine.com	SMM 2009
Megaptera novaeangliea	X				ASSESSING GEOGRAPHIC AND ACOUSTIC MYSTICETE-SHIPSTRIKE RISK IN THE GULF OF MAINE	Peterson, Michael	University of Maine	Michael.Peterson@maine.edu	SMM 2009
Megaptera novaeangliea	X				DETERMINATION OF DIVING BEHAVIORS OF HUMPBACK WHALES (MEGAPTERA NOVAEANGLIAE), OVER ROCKY BOTTOM HABITATS USING DTAGS	Summers, Erin	Maine DMR	erin.l.summers@maine.gov	SMM 2009

Species	**NAPA**	**MAPA**	**SAPA**	**SFPA**	**Title**	**Author**	**Institution**	**Email**	**Citation**
Megaptera novaeangliea	X				DETERMINATION OF SPATIO-TEMPORAL HABITAT USE BY MYSTICETES IN THE NORTHEAST GULF OF MAINE USING A 20-YEAR OPPORTUNISTIC SIGHTING DATABASE	Todd, Sean	College of the Atlantic	stodd@coa.edu	SMM 2009
Megaptera novaeangliea	X				DETERMINING THE SEASONAL DISTRIBUTION OF CETACEANS IN NEW YORK COASTAL WATERS USING PASSIVE ACOUSTIC MONITORING	Biedron, Ingrid	Cornell University	ib49@cornell.edu	SMM 2009
Megaptera novaeangliea	X				EVIDENCE FOR SPATIAL ASSOCIATIONS OF SYMPATRIC HUMPBACK AND FIN WHALES IN THE GULF OF MAINE USING MONTE CARLO SIMULATIONS	Lubansky, Tanya	NJIT	tml5@njit.edu	SMM 2009
Megaptera novaeangliea	X				HUMPBACK WHALE (MEGAPTERA NOVAEANGLIAE) SONG OCCURS EXTENSIVELY ON A FEEDING GROUND IN THE NORTHWESTERN ATLANTIC OCEAN	Vu, Elizabeth Tram-Anh	NMFS-NEFSC	jeepurs@gmail.com	SMM 2009
Megaptera novaeangliea	X				HUMPBACK WHALE (MEGAPTERA NOVAEANGLIEA) CALF AGE- AND SEX-RELATED BEHAVIORAL PATTERNS: IMPLICATIONS FOR SPECIES MANAGEMENT	Zoidis, Ann Margaret	CETOS Research Org.	ann@cetosresearch.org	SMM 2009
Megaptera novaeangliea	X	X	X	X	MOVEMENT PATTERNS OF NORTH ATLANTIC HUMPBACK WHALES (MEGAPTERA NOVAEANGLIAE) IDENTIFIED AT BERMUDA	Beaudette, Adrianna	College of the Atlantic	abeaudette@coa.edu	SMM 2009

Species	**NAPA**	**MAPA**	**SAPA**	**SFPA**	**Title**	**Author**	**Institution**	**Email**	**Citation**
Megaptera novaeangliea					MULTIMODAL SOMATOSENSORY SYSTEM IN MYSTICETI EXPLAINS EXTENDED BODY LENGTH AND LONG-DISTANCE SOURCE SIGNAL TRIANGULATION	Eldridge, Sherri	Univ. of Maine	newblueview@roadrunner.com	SMM 2009
Megaptera novaeangliea	X				NEITHER CONSORT NOR KIN: MOLECULAR ECOLOGY OF A LONG-TERM ASSOCIATION BETWEEN FEEDING HUMPBACK WHALES (MEGAPTERA NOVAEANGLIAE)	Tackaberry, Jennifer	Whale Center of N.E.	jenn@whalecenter.org	SMM 2009
Megaptera novaeangliea	X				NOT ALL ENTANGLEMENTS ARE CREATED EQUAL	Landry, Scott	PCCS	sclandry@coastalstudies.org	SMM 2009
Megaptera novaeangliea	X				SUMMER HABITAT OF HUMPBACK WHALES (MEGAPTERA NOVAEANGLIAE) THROUGHOUT THE NORTH ATLANTIC OCEAN	Vigness-Raposa, Kathleen	University of Rhode Island	kvigness@aol.com	SMM 2009
Megaptera novaeangliea	X				UNDERWATER KINEMATICS OF BUBBLE FEEDING HUMPBACK WHALES (MEGAPTERA NOVAEANGLIAE)	Wiley, David	Stellwagen Bank NMS	David.Wiley@noaa.gov	SMM 2009
Megaptera novaeangliea					WAITING TIMES TO MONITOR MANAGEMENT ACTIONS: ATLANTIC LARGE WHALES SHOW NO RESPONSE TO GEAR RULES	Pace, Richard	NMFS-NEFSC	Richard.Pace@noaa.gov	SMM 2009
Megaptera novaeangliea	X				WANDERING WHALE WATCHES: AN ANALYSIS OF DATA COLLECTED ON WHALE WATCHES TRAVELING TO STELLWAGEN BANK, MA FROM 1994-1997	Koslovsky, Stacie	Duke Marine Lab	staciekoslovsky@gmail.com	SMM 2009

Species	NAPA	MAPA	SAPA	SFPA	Title	Author	Institution	Email	Citation
Pagophilus groenlandicus	X				CHARACTERISTICS OF PHOCID SEAL BYCATCH IN NEW ENGLAND FISHERIES.	Belden, Dana	Office of Naval Research	dana.belden.ctr@navy.mil	SMM 2009
Phoca vitulina	X				ABUNDANCE, DENSITY, AND DISTRIBUTION OF MARINE MAMMALS IN NEW JERSEY'S NEARSHORE WATERS	Whitt, Amy	Geo-Marine, Inc.	awhitt@geo-marine.com	SMM 2009
Phoca vitulina	X				CHARACTERISTICS OF PHOCID SEAL BYCATCH IN NEW ENGLAND FISHERIES.	Belden, Dana	Office of Naval Research	dana.belden.ctr@navy.mil	SMM 2009
Phoca vitulina	X				MARINE CONSERVATION: ASSESSING THREATS, AND CHARACTERIZING HABITATS OF HARBOR SEALS (PHOCA VITULINA CONCOLOR; PHOCIDAE) IN SOUTHERN NEW JERSEY (USA).	Slocum, Carol*	The Richard Stockton College of New Jersey	*Michael Davenport	SMM 2009
Phoca vitulina	X				PHOTOGRAPHIC SURVEYS OF ENTANGLEMENT OCCURRENCE AT GRAY SEAL (HALICHOERUS GRYPUS) AND HARBOR SEAL (PHOCA VITULINA) HAUL-OUTS ON CAPE COD, MASSACHUSETTS	Sette, Lisa	PCCS	sette@coastalstudies.org	SMM 2009
Phocoena phocoena	X				ABUNDANCE, DENSITY, AND DISTRIBUTION OF MARINE MAMMALS IN NEW JERSEY'S NEARSHORE WATERS	Whitt, Amy	Geo-Marine, Inc.	awhitt@geo-marine.com	SMM 2009
Phocoena phocoena	X				EVALUATING BIAS WHEN ESTIMATING SMALL CETACEAN BYCATCH MORTALITY IN GILLNET FISHERIES USING DIFFERENT UNITS OF EFFORT	Rossman, Marjorie	NOAA-NEFSC	Marjorie.Rossman@noaa.gov	SMM 2009

Species	**NAPA**	**MAPA**	**SAPA**	**SFPA**	**Title**	**Author**	**Institution**	**Email**	**Citation**
Phocoena phocoena	X				HARBOUR PORPOISE (PHOCOENA PHOCOENA) PRESENCE PATTERNS AT AN AQUACULTURE CAGE SITE IN THE BAY OF FUNDY, CANADA	Haar, Marthe	University of New Brunswick	mhaarr@hotmail.com	SMM 2009
Physeter macrocephalus					BIG BANGS FROM A BIG NOSE: BIOMECHANICS OF AIR DRIVEN SOUND PRODUCTION IN SPERM WHALES	Madsen, Peter	Aarhus U., Denmark	peter.madsen@biology.au.dk	SMM 2009
Physeter macrocephalus	X	X	X	x	DEVELOPMENT OF A PREDICTIVE HORIZONTAL HABITAT MODEL FOR SPERM AND PILOT WHALES IN THE NORTHWEST ATLANTIC OCEAN	Cross, Cheryl Lynne	Nova Southeastern U.	ccross@nova.edu	SMM 2009
Physeter macrocephalus		X			ODONTOCETE VOCALIZATIONS IN ONSLOW BAY, NORTH CAROLINA: INTEGRATING DATA FROM TWO PASSIVE ACOUSTIC TECHNIQUES	Williams, Lynne	Duke Marine Lab	lw32@duke.edu	SMM 2009
Physeter macrocephalus		X			USING PASSIVE ACOUSTICS TO MONITOR THE PRESENCE OF ODONTOCETE CETACEANS DURING NAVAL EXERCISES IN ONSLOW BAY, NC	Kumar, Anurag	Naval Facilities Engineering Command Atlantic	anurag.kumar@navy.mil	SMM 2009
Stenella coeruleoalba					ACOUSTIC ANALYSIS OF WHISTLE CHARACTERISTICS OF DELPHINID SPECIES IN THE NW ATLANTIC.	Schiebel, Hayley	Nova Southeastern University	hayhay121@hotmail.com	SMM 2009
Stenella frontalis					ACOUSTIC ANALYSIS OF WHISTLE CHARACTERISTICS OF DELPHINID SPECIES IN THE NW ATLANTIC.	Schiebel, Hayley	Nova Southeastern U.	hayhay121@hotmail.com	SMM 2009

Species	NAPA	MAPA	SAPA	SFPA	Title	Author	Institution	Email	Citation
Stenella frontalis		X			AERIAL AND VESSEL SURVEYS OF THE UNDERSEA WARFARE TRAINING RANGE SITE ALTERNATIVE IN ONSLOW BAY, NC, USA	Nilsson, Peter	UNC Wilmington	nilssonp@uncw.edu	SMM 2009
Steno bredanensis		X			AERIAL AND VESSEL SURVEYS OF THE UNDERSEA WARFARE TRAINING RANGE SITE ALTERNATIVE IN ONSLOW BAY, NC, USA	Nilsson, Peter	UNC Wilmington	nilssonp@uncw.edu	SMM 2009
Trichechus manatus latirostris				X	EVALUATION OF TRACE METALS IN THE FLORIDA MANATEE (TRICHECHUS MANATUS LATIROSTRIS) AND THE EXPRESSION OF METALLOTHIONEIN AS A POTENTIAL BIOMARKER OF METAL EXPOSURE	Takeuchi, Noel Y.	University of Florida	ntakeuchi@vetmed.ufl.edu	SMM 2009
Trichechus manatus latirostris				X	FLORIDA MANATEE BEHAVIOR DURING VESSEL APPROACHES	Rycyk, Athena	Florida State University	rycyk@ocean.fsu.edu	SMM 2009
Trichechus manatus latirostris				X	RESCUE, REHABILITATION AND RELEASE OF FLORIDA MANATEES: ANALYSIS OF FACTORS AFFECTING SURVIVAL	Adimey, Nicole	US Fish and Wildlife Service	nicole_adimey@fws.gov	SMM 2009
Trichechus manatus latirostris				X	SPATIAL DISTRIBUTION OF THE FLORIDA MANATEE (TRICHECHUS MANATUS LATIROSTRIS) AROUND POWER PLANTS IN BROWARD COUNTY, FLORIDA	Eldredge, Laura F.	Nova Southeastern U.	leldredg@nova.edu	SMM 2009
Trichechus manatus latirostris				X	THE MANATEE AUDIOGRAM AND AUDITORY CRITICAL RATIOS	Bauer, Gordon	New College of Florida	bauer@ncf.edu	SMM 2009

Species	**NAPA**	**MAPA**	**SAPA**	**SFPA**	**Title**	**Author**	**Institution**	**Email**	**Citation**
Trichechus manatus latirostris				X	THIS WON'T HURT A BIT: NEW MOLECULAR TOOLS FOR POPULATION ASSESSMENT OF THE FLORIDA MANATEE	Davis, Michelle	FL Fish & Wildlife Cons. Com.	Michelle.Davis@myfwc.com	SMM 2009

4.0 LITERATURE CITED

Abend, A. G., and T. D. Smith. **1997**. Differences in ratios of stable isotope ratios of carbon and nitrogen between long-finned pilot whales (*Globicephala melas*), and their primary prey in the western North Atlantic. ICES Journal of Marine Science 54(3):500-503.

Acevedo-Gutiérrez, A. and Cendejas-Zarelli .**2011**. Nocturnal haul-out patterns of harbor seals (*Phoca vitulina*) related to airborne noise levels in Bellingham, Washington, USA. Aquatic Mammals 37(2):167-174.

Acevedo-Gutierrez, A., and N. Parker. **2000**. Surface behavior of bottlenose dolphins is related to spatial arrangement of prey. Marine Mammal Science 16(2):287-298.

Addison, R. F., and W. T. Stobo. **1993**. Organochlorine residue concentrations and burdens in grey seal (*Halichoerus grypus*) blubber during the first year of life. Journal of Zoology 230(3):443-450.

Adimey, N. M., K. Abernathy, J. I. Gaspard, and G. Marshall. **2008**. Meeting the manatee challenge: The feasibility of using crittercam on wild manatees. Marine Technology Society Journal 41(4):14-18.

Agler, B. A., R. L. Schooley, S. E. Frohock, S. K. Katona, and I. E. Seipt. **1993**. Reproduction of photographically identified fin whales, *Balaenoptera physalus*, from the Gulf of Maine. Journal of Mammalogy 577-587.

Agler, B. A., J. A. Beard, R. S. Bowman, H. D. Corbett, S. E. Frohock, M. P. Hawvermale, S. K. Katona, S. S. Sadove, and I. E. Seipt. **1990**. Fin whale (*Balaenoptera physalus*) photographic identification: methodology and preliminary results from the western North Atlantic. Report of the International Whaling Commission (Special Issue 12) 349-356.

Aguilar Soto, N., M. Johnson, P. T. Madsen, P. L. Tyack, A. Bocconcelli, and J. Fabrizio Borsani. **2006**. Does intense ship noise disrupt foraging in deep-diving Cuvier's beaked whales (*Ziphius cavirostris*)? Marine Mammal Science 22(3):690-699.

Aguilar, A. **1983**. Organochlorine pollution in sperm whales, *Physeter macrocephalus*, from the temperate waters of the Eastern North Atlantic. Marine Pollution Bulletin 14(9):349-352.

Aguilar, A., A. Borrell, and P. J. H. Reijnders. **2002**. Geographical and temporal variation in levels of organochlorine contaminants in marine mammals. Marine Environmental Research 53(5):425-452.

Aguilar, A., and A. Borrell. **1994**. Reproductive transfer and variation of body load of organochlorine pollutants with age in fin whales (*Balaenoptera physalus*). Archives of Environmental Contamination and Toxicology 27(4):546-554.

Aguilar, A., and A. Borrell. **2005**. DDT and PCB reduction in the western Mediterranean from 1987 to 2002, as shown by levels in striped dolphins (*Stenella coeruleoalba*). Marine Environmental Research 59:391-404.

Agusa, T., K. Nomura, Y. Anan, H. Iwata, and S. Tanabe. **2011a**. Accumulation of trace elements in harp seals (*Phoca groenlandica*) from Pangnirtung in the Baffin Island, Canada. Marine Pollution Bulletin 63:489-499.

Agusa, T., K. Takagi, R. Kubota, Y. Anan, H. Iwata, and S. Tanabe. **2008**. Specific accumulation of arsenic compounds in green turtles (*Chelonia mydas*) and hawksbill turtles (*Eretmochelys imbricata*) from Ishigaki Island, Japan. Environmental Pollution 153(1):127-136.

Agusa, T., S. Yasugi, A. Iida, T. Ikemoto, Y. Anan, T. Kuiken, A. D. M. E. Osterhaus, S. Tanabe, and H. Iwata. **2011b**. Accumulation features of trace elements in mass-stranded harbor seals (*Phoca vitulina*) in the North Sea coast in 2002: The body distribution and association with growth and nutrition status. Marine Pollution Bulletin 62:963-975.

Aipanjiguly, S., S. K. Jacobson, and R. Flamm. **2003**. Conserving manatees: knowledge, attitudes, and intentions of boaters in Tampa Bay, Florida. Conservation Biology 17(4):1098-1105.

Alava, J. J., J. M. Keller, J. Wyneken, L. Crowder, G. Scott, and J. Kucklick. **2008**. Geographical comparison of persistent organochlorine pollutant levels in loggerhead eggs (*Caretta caretta*) collected along the southeast coast of USA, In: Proceedings of the 25th annual symposium on sea turtle biology and conservation, p. 34, NOAA Tech Memo NMFS-SEFSC-582.

Allen, P. J., W. Amos, P. P. Pomeroy, and S. D. Twiss. **1995**. Microsatellite variation in grey seals (*Halichoerus grypus*) shows evidence of genetic differentiation between two British breeding colonies. Molecular Ecology 4(6):653-662.

Alves-Stanley, C. D., and G. A. J. Worthy. **2009**. Carbon and nitrogen stable isotope turnover rates and diet-tissue discrimination in Florida manatees (*Trichechus manatus latirostris*). Journal of Experimental Biology 212(15):2349-2355.

Amano, M., N. Miyazaki, and F. Yanagisawa. **1996**. Life history of Fraser’s dolphin, *Lagenodelphis hosei*, based on a school capture off Pacific coast of Japan. Marine Mammal Science 12:199-214.

Amano, M., and M. Yoshioka. **2003**. Sperm whale diving behavior monitored using a suction-cup-attached TDR tag. Marine Ecology Progress Series 258:291-295.

Amano, M., and N. Miyazaki. **2004**. Composition of a school of Risso’s dolphins, *Grampus griseus*. Marine Mammal Science 20(1):152-160.

Amaral, K. A. **2005**. Atlantic white-sided dolphin (*Lagenorhynchus acutus*) social structure based on stranding trends and genetics. MS thesis. University of Massachusetts, Dartmouth. 78 p.

Ames, A. L., E. S. Vleet, and W. M. Sackett. **1996**. The use of stable carbon isotope analysis for determining the dietary habits of the Florida manatee, *Trichechus manatus latirostris*. Marine Mammal Science 12(4):555-563.

Ames, A. L., and E. S. Van Vleet. **1996**. Organochlorine residues in the Florida manatee, *Trichechus manatus latirostris*. Marine Pollution Bulletin 32(4):374-377.

Amos, B., C. Schlotterer, and D. Tautz. **1993a**. Social structure of pilot whales revealed by analytical DNA profiling. Science 260:670-672.

Amos, W., S. Twiss, P. P. Pomeroy, and S. S. Anderson. **1993b**. Male mating success and paternity in the grey seal, *Halichoerus grypus*: a study using DNA fingerprinting. Proceedings: Biological Sciences 252(1335):199-207.

Amos, W., J. Worthington Wilmer, and H. Kokko. **2001**. Do female grey seals select genetically diverse mates? Animal Behaviour 62(1):157-164.

Ampela, K. **2009**. The diet and foraging ecology of gray seals (*Halichoerus grypus*) in United States waters. Ph.D. thesis. The City University of New York, Steten Island, NY. 176 p.

Anas, R. E. **1974**. Heavy metals in the northern fur seal, *Callorhinus ursinus*, and harbor seal, *Phoca vitulina richardi*. Fishery Bulletin 72(1):133-137.

Andersen, L. W., E. W. Born, R. Dietz, T. Haug, N. Øien, and C. Bendixen. **2003**. Genetic population structure of minke whales *Balaenoptera acutorostrata* from Greenland, the North East Atlantic and the North Sea probably reflects different ecological regions. Marine Ecology Progress Series 247:263-280.

Andersen, L. W., and H. R. Siegismund. **1994**. Genetic evidence for migration of males between schools of long-finned pilot whales, *Globicephala melas*. Canadian Journal of Zoology 105:1-7.

Anderson, S. S., and J. Harwood. **1985**. Time budgets and topography: how energy reserves and terrain determine the breeding behaviour of grey seals. Animal Behaviour 33(4):1343-1348.

Anderson, S., W. N. Bonner, J. R. Baker, and R. Richards .**1974**. Grey seals, *Halichoerus grypus*, of the Dee Estuary and observations on a characteristic skin lesion in British seals. Journal of Zoology 174(3):429-440.

Anderwald, P., P. G. H. Evans, L. Gygax, and A. R. Hoelzel. **2011**. Role of feeding strategies in seabird-minke whale associations. Mar Ecol Prog Ser 424:219-227.

Andre, J. M., J. C. Amiard, C. Amiard-Triquet, A. Boudou, and F. Ribeyre. **1990a**. Cadmium contamination of tissues and organs of delphinids species (*Stenella attenuata*) - Influence of biological and ecological factors. Ecotoxicology and Environmental Safety 20:290-306.

Andre, J. M., F. Ribeyre, and A. Boudou. **1990b**. Mercury contamination levels and distribution in tissues and organs of Delphinids (*Stenella attenuata*) from the eastern tropical Pacific, in relation to biological and ecological factors. Marine Environmental Research 30:43-72.

Andrews, K. R., L. Karczmarksi, W. W. L. Au, S. H. Rickards, C. A. Vanderlip, B. W. Bowen, E. G. Grau, and R. J. Toonen. **2010**. Rolling stones and stable homes: social structure, habitat diversity and population genetics of the Hawaiian spinner dolphin (*Stenella longirostris*). Molecular Ecology 19(4):732-748.

André, M. **2009**. The sperm whale sonar: Monitoring and use in mitigation of anthropogenic noise effects in the marine environment. Nuclear Instruments and Methods in Physics Research Section A: Accelerators, Spectrometers, Detectors and Associated Equipment 602(1):262-267.

André, M., M. Van Der Schaar, E. Delory, J. Castell, and A. Mas. **2008**. 3D tracking and monitoring of sperm whales with passive acoustics and ambient noise imaging techniques, In: SARTI (Technological Development Centre of Remote Acquisition and Data processing Systems).

Anonymous. **1992**. First Kemp’s ridley nesting in South Carolina. Marine Turtle Newsletter 59:23.

Antunes, R., L. Rendell, and J. Gordon. **2010**. Measuring inter-pulse intervals in sperm whale clicks: Consistency of automatic estimation methods. The Journal of the Acoustical Society of America 127:3239-3247.

Antunes, R., T. M. Schulz, S. Gero, H. Whitehead, J. Gordon, and L. Rendell. **2011**. Individually distinctive acoustic features in sperm whale codas. Animal Behaviour 81(4):723-730.

Antworth, R. L., D. A. Pike, and J. C. Stiner. **2006**. Nesting ecology, current status, and conservation of sea turtles on an uninhabited beach in Florida, USA. Biological Conservation 130(1):10-15.

Aoki, K., M. Amano, M. Yoshioka, K. Mori, D. Tokuda, and N. Miyazaki. **2007**. Diel diving behavior of sperm whales off Japan. Marine Ecology Progress Series 349:277-287.

Apprill, A., T. A. Mooney, E. Lyman, A. K. Stimpert, and M. S. Rappé. **2011**. Humpback whales harbour a combination of specific and variable skin bacteria. Environmental Microbiology Reports 3(2):223-232.

Archer, F. I., and W. F. Perrin. **1999**. *Stenella coeruleoalba*. Mammalian Species 603:1-9.

Archer, F. I. **2009**. Striped dolphin *Stenella coeruleoalba*, In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, pp. 1127-1129, Academic Press, San Diego.

Archer, F. I., and K. M. Robertson. **2004**. Age and length at weaning and development of diet of pantropical spotted dolphins, *Stenella attenuata*, from the eastern tropical Pacific. Marine Mammal Science 20(2):232-245.

Arnason, U., A. Gullberg, E. Johnsson, and C. Ledje. **1992**. The nucleotide sequence of the mitochondrial DNA molecule of the grey seal, *Halichoerus grypus*, and a comparison with mitochondrial sequences of other true seals. Journal of Molecular Evolution 37(4):323-390.

Arnason, U., A. Gullberg, and B. Widegren.**1993**. Cetacean mitochondrial DNA control region: sequences of all extant baleen whales and two sperm whale species. Molecular Biology and Evolution 10(5):960-970.

Arnason, U., and E. Johnsson. **1992**. The complete mitochondrial DNA sequence of the harbor seal, *Phoca vitulina*. Journal of Molecular Evolution 34(6):493-505.

Au, W. W. L., and M. Green. **2000**. Acoustic interaction of humpback whales and whale-watching boats. Marine Environmental Research 49(5):469-481.

Au, W. W. L., A. A. Pack, M. O. Lammers, L. M. Herman, M. H. Deakos, and K. Andrews. **2006**. Acoustic properties of humpback whale songs. The Journal of the Acoustical Society of America 120:1103-1110.

Au, W. W. L., and D. L. Herzing. **2003**. Echolocation signals of wild Atlantic spotted dolphin (*Stenella frontalis*). Journal of the Acoustical Society of America 113(1):598-604.

Austin, D., W. D. Bowen, J. I. McMillan, and S. J. Iverson. **2006a**. Linking movement, diving, and habitat to foraging success in a large marine predator. Ecology 87(12):3095-3108.

Austin, D., W. D. Bowen, J. I. Mcmillan, and D. J. Boness. **2006b**. Stomach temperature telemetry reveals temporal patterns of foraging success in a free-ranging marine mammal. Journal of Animal Ecology 75(2):408-420.

Avens, L., J. Braun-McNeill, S. P. Epperly, and K. Lohmann. **2003**. Site fidelity and homing behavior in juvenile loggerhead sea turtles, *Caretta caretta*. Marine Biology 143:211-220.

Avens, L., and L. Goshe. **2008**. Skeletochronological analysis of age and growth for leatherback sea turtles in the western north Atlantic, In: Proceedings of the 27th annual symposium on sea turtle biology and conservation, p. 201, NOAA Tech Memo NMFS-SEFSC-569.

Avens, L., and A. Hohn. **2008**. Skeletochronological analysis of Kemp’s ridley and loggerhead scleral ossicles, In: Proceedings of the 24th annual symposium on sea turtle biology and conservation, p. 113, NOAA Tech Memo NMFS-SEFSC-567.

Azzellino, A., C. Lanfredi, A. D’Amico, G. Pavan, M. Podestà, and J. Haun. **2011**. Risk mapping for sensitive species to underwater anthropogenic sound emissions: Model development and validation in two Mediterranean areas. Marine Pollution Bulletin 63:56-70.

Bacchus, M.-L. C., S. G. Dunbar, and C. Self-Sullivan. **2009**. Characterization of resting holes and their use by the Antillean Manatee (*Trichechus manatus manatus*) in the Drowned Cayes, Belize. Aquatic Mammals 35(1):62-71.

Bagley, D. A., S. A. Eckert, and L. M. Ehrhart .**2001**. Satellite tracking leatherbacks from the Archie Carr National Wildlife Refuge on the east coast of Florida. Florida Scientist 64(1):45-46.

Bailey, H., G. Shillinger, D. Palacios, S. Bograd, J. Spotila, F. Paladino, and B. Block **2008**. Identifying and comparing phases of movement by leatherback turtles using state-space models. Journal of Experimental Marine Biology and Ecology 356(1-2):128-135.

Bailey, H., B. Senior, M. Simmons, J. Rusin, G. Picken, and P. M. Thomp. **2010**. Assessing underwater noise levels during pile-driving at an offshore windfarm and its potential effects on marine mammals. Marine Pollution Bulletin 60:888-897.

Bain, D. E., and R. Williams. **2006**. Long-range effects of airgun noise on marine mammals: responses as a function of received sound level and distance. International Whaling Commission Scientific Committee Paper SC/58/ E35.

Baird, R. W. **2001**. Status of harbour seals, *Phoca vitulina*, in Canada. The Canadian Field-Naturalist 115:663-675.

Baird, R. W. **2009**. Risso's dolphin *Grampus griseus*, In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, pp. 975-976, Academic Press, San Diego.

Baird, R. W., D. L. Webster, S. D. Mahaffy, D. J. McSweeney, G. S. Schorr, and A. D. Ligon. **2008**. Site fidelity and association patterns in a deep-water dolphin: Rough-toothed dolphins (*Steno bredanensis*) in the Hawaiian Archipelago. Marine Mammal Science 24(3):535-553.

Baker, J. R. **1984**. Mortality and morbidity in grey seal pups (*Halichoerus grypus*). Studies on its causes, effects of environment, the nature and sources of infectious agents and the immunological status of pups. Journal of Zoology 203(1):23-48.

Ballard, K. A., and K. M. Kovacs. **1995**. The acoustic repotoire of hooded seals (*Cystophora cristata*). Canadian Journal of Zoology 73:1362-1374.

Ballentine, A., P. Mason, and J. I. Richardson. **2008**. Nesting preferences of hawksbill turtles (*Eretmochelys imbricata*) on Jumby Bay, Long Island, Antigua, West Indies, In: Proceedings of the 25th annual symposium on sea turtle biology and conservation, p. 72, NOAA Tech Memo NMFS-SEFSC-582.

Baraff, L. S. **2006**. Summer distribution and habitat characteristics of fin whales (*Balaenoptera physalus*) and humpback whales (*Megaptera novaeangliae*) off Northeast Kodiak Island, Alaska. MS thesis. University of Alaska, Fairbanks. 131 p.

Barco, S. G., W. A. McLellan, J. M. Allen, R. A. Asmutis-Silvia, R. Mallon-Day, E. M. Meacher, D. A. Pabst, J. Robbins, R. E. Seton, and W. M. Swingle. **2002**. Population identity of humpback whales (*Megaptera novaengliae*) in the waters of the US mid-Atlantic states. Journal of Cetacean Research and Management 4(2):135-142.

Barlas, M. E. **1999**. The distribution and abundance of harbor seals (*Phoca vitulina concolor*) and gray seals (*Halichoerus grypus*) in southern New England winter 1998-summer 1999. MA thesis. Boston University, Boston, MA. 52 p.

Barlow, J., M. C. Ferguson, W. F. Perrin, L. Ballance, T. Gerrodette, G. Joyce, C. D. Macleod, K. Mullin, D. L. Palka, and G. Waring. **2005**. Abundance and densities of beaked and bottlenose whales (family Ziphiidae). Journal of Cetacean Research and Management 7(3):263-270.

Barlow, J., and P. J. Clapham. **1997**. A new birth-interval approach to estimating demographic parameters of humpback whales. Ecology 78(2):535-546.

Barlow, J., and R. Gisiner. **2006**. Mitigating, monitoring and assessing the effects of anthropogenic sound on beaked whales. Journal of Cetacean Research and Management 7(3):239-249.

Barlow, J., and B. L. Taylor. **2005**. Estimates of sperm whale abundance in the northeastern temperate Pacific from a combined acoustic and visual survey. Marine Mammal Science 21(3):429-445.

Barlow, J., and G. A. Cameron. **2003**. Field experiments show that acoustic pingers reduce marine mammal bycatch in the California drift gill net fishery. Marine Mammal Science 19(2):265-283.

Barros, N. B., and R. S. Wells. **1998**. Prey and feeding patterns of resident bottlenose dolphins (*Tursiops truncatus*) in Sarasota Bay, Florida. Journal of Mammalogy 79(3):1045-1059.

Bartol, S. M., J. A. Musick, and M. L. Lenhardt. **1999**. Auditory evoked potentials of the loggerhead sea turtle (*Caretta caretta*). Copeia 1999(3):836-840.

Bass, A. L., S. P. Epperly, and J. Braun-McNeill. **2006**. Green turtle (*Chelonia mydas*) foraging and nesting aggregations in the Caribbean and Atlantic: impact of currents and behavior on dispersal. Journal of Heredity 97(4):346-354.

Bass, A. L., S. P. Epperly, and J. Braun-McNeill. **2004**. Multi-year analysis of stock composition of a loggerhead turtle (*Caretta caretta*) foraging habitat using maximum likelihood and Bayesian methods. Conservation Genetics 5:783-796.

Bass, A. L., D. A. Good, K. A. Bjorndal, J. I. Richardson, Z.-M. Hillis, J. A. Horrocks, and B. W. Bowen. **1996**. Testing models of female reproductive migratory behaviour and population structure in the Caribbean hawksbill turtle, *Eretmochleys inbricata*, with mtDNA sequences. Molecular Ecology 5:321-328.

Bass, A. L., and W. N. Witzell. **2000**. Demographic composition of immature green turtles (*Chelonia mydas*) from the east central Florida coast. Herpetologica 56(3):357-367.

Baumann-Pickering, S., S. M. Wiggins, and J. A. Hildebrand. **2010**. Discriminating features of echolocation clicks of melon-headed whales (*Peponocephala electra*), bottlenose dolphins (*Tursiops truncatus*), and Gray's spinner dolphins (*Stenella longirostris longirostris*). Journal of the Acoustical Society of America 128(4):2212-2224.

Baumgartner, M. F., T. V. N. Cole, R. G. Campbell, G. J. Teegarden, and E. G. Durbin. **2003**. Associations between North Atlantic right whales and their prey, *Calanus finmarchicus*, over diel and tidal time scales. Marine Ecology Progress Series 264:155-166-.

Baumgartner, M. F., S. M. Van Parijs, F. W. Wenzel, C. J. Tremblay, H. C. Esch, and A. M. Warde. **2008**. Low frequency vocalizations attributed to sei whales (*Balaenoptera borealis*). The Journal of the Acoustical Society of America 124:1339.

Baumgartner, M. F., and D. M. Fratantoni. **2008**. Diel periodicity in both sei whale vocalization rates and the vertical migration of their copepod prey observed from ocean gliders. Limnology and Oceanography 53(5):2197-2209.

Baumgartner, M. F., and B. R. Mate. **2005**. Summer and fall habitat of North Atlantic right whales (*Eubalaena glacialis*) inferred from satellite telemetry. Canadian Journal of Fisheries and Aquatic Sciences 62(3):527-543.

Baumgartner, M. F., and B. R. Mate. **2003**. Summertime foraging ecology of North Atlantic right whales. Marine Ecology Progress Series 264:123-135.

Baumgartner, M. F. **1997**. The distribution of Risso's dolphin (*Grampus griseus*) with respect to the physiography of the Northern Gulf of Mexico. Marine Mammal Science 13(4):614-638.

Baumgartner, M., N. Lysiak, C. Schuman, J. Urban-Rich, and F. Wenzel. **2011**. Diel vertical migration behavior of *Calanus finmarchicus* in the southwestern Gulf of Maine and its influence on right and sei whale occurrence. Marine Ecology Progress Series 423:167-184.

Baumgartner, N. **2008**. Distribution, diving behaviour and identification of the North Atlantic minke whale in northeast Scotland. Ph.D. thesis. University of Aberdeen, Aberdeen, Scotland. 114 p.

Beamish, P., and E. Mitchell. **1973**. Short pulse length audio frequency sounds recorded in the presence of a minke whale (*Balaenoptera acutorostrata*), In: Deep Sea Research and Oceanographic Abstracts, vol. 20, pp. 375-380.

Bean, K., W. Amos, P. P. Pomeroy, S. D. Twiss, T. N. Coulson, and I. L. Boyd. **2004**. Patterns of parental relatedness and pup survival in the grey seal (*Halichoerus grypus*). Molecular Ecology 13(8):2365-2370.

Beardsley, R. C., A. W. Epstein, C. Chen, K. F. Wishner, M. C. Macaulay, and R. D. Kenney. **1996**. Spatial variability in zooplankton abundance near feeding right whales in the Great

South Channel. Deep Sea Research Part II: Topical Studies in Oceanography 43(7-8):1601-1625.

Beatson, E., and M. Ogle. **2007**. First report on the stomach contents of long-finned pilot whales, *Globicephala melas*, stranded in New Zealand. New Zealand Journal of Zoology 34(1):51-56.

Beck, C. A., R. K. Bonde, and G. B. Rathbun .**1982**. Analyses of propeller wounds on manatees in Florida. The Journal of Wildlife Management 46(2):531-535.

Beck, C. A., and N. B. Barros. **1991**. The impact of debris on the Florida manatee. Marine Pollution Bulletin 22(10):508-510.

Beck, C. A., S. J. Iverson, W. D. Bowen, and W. Blanchard. **2007**. Sex differences in grey seal diet reflect seasonal variation in foraging behaviour and reproductive expenditure: evidence from quantitative fatty acid signature analysis. Journal of Animal Ecology 76(3):490-502.

Beck, G. G., M. O. Hammill, and T. G. Smith .**1993**. Seasonal variation in the diet of harp seals (*Phoca groenlandica*) from the Gulf of St. Lawrence and western Hudson Strait. Canadian Journal of Fisheries and Aquatic Sciences 50(7):1363-1371.

Beggs, J. A., J. A. Horrocks, and B. H. Krueger. **2007**. Increase in hawksbill sea turtle *Eretmochelys imbricata* nesting in Barbados, West Indies. Endangered Species Research 3(2):159-168.

Bejder, L., A. Samu, H. Whitehead, N. Gales, J. Mann, R. Connor, M. Heithaus, J. Watson-Capps, C. Flaherty, and M. Krutzen. **2006**. Decline in relative abundance of bottlenose dolphins exposed to long-term disturbance. Conservation Biology 20(6):1791-1798.

Bell, B. A., J. R. Spotila, F. V. Paladino, and R. D. Reina. **2004**. Low reproductive success of leatherback turtles, *Dermochelys coriacea*, is due to high embryonic mortality. Biological Conservation 115(1):131-138.

Bell, C. D., J. L. Solomon, J. M. Blumenthal, T. J. Austin, G. Ebanks-Petrie, A. C. Broderick, and B. J. Godley. **2007**. Monitoring and conservation of critically reduced marine turtle nesting populations: lessons from the Cayman Islands. Animal Conservation 10(1):39-47.

Bender, C. E., D. L. Herzing, and D. F. Bjorklund .**2009**. Evidence of teaching in Atlantic spotted dolphins (*Stenella frontalis*) by mother dolphins foraging in the presence of their calves. Animal Cognition 12(1):43-53.

Bengtson, J. L., A. V. Phillips, E. A. Mathews, and M. A. Simpkins. **2007**. Comparison of survey methods for estimating abundance of harbor seals (*Phoca vitulina*) in glacial fjords. Fishery Bulletin 105(3):348.

Benjaminsen, T. **1979**. On the biology of the bottlenose whale, *Hyperoodon ampullatus* (Forster). Norwegian Journal of Zoology 20:233-241.

Benjaminsen, T., and I. Christensen. **1979**. The natural history of the bottlenose whale, *Hyperoodon ampullatus*(Forster)., In: Behavior of Marine Animals, vol. 3, pp. 143-164, Plenum Press, New York.

Benoit-Bird, K. J., and W. W. L. Au. **2003**. Prey dynamics affect foraging by a pelagic predator (*Stenella longirostris*) over a range of spatial and temporal scales. Behavioral Ecology and Sociobiology 53(6):364-373.

Benoit-Bird, K. J., and W. W. L. Au. **2009a**. Cooperative prey herding by the pelagic dolphin, *Stenella longirostris*. The Journal of the Acoustical Society of America 125(1):125-137.

Benoit-Bird, K. J., and W. W. L. Au. **2009b**. Phonation behavior of cooperatively foraging spinner dolphins. The Journal of the Acoustical Society of America 125(1):539-546.

Berens McCabe, E., D. P. Gannon, N. B. Barros, and R. S. Wells. **2010**. Prey selection by resident common bottlenose dolphins (*Tursiops truncatus*) in Sarasota Bay, Florida. Marine Biology 157:931-942.

Bergman, A. **1999**. Health condition of the Baltic grey seal (*Halichoerus grypus*) during two decades. APMIS 107(1-6):270-282.

Bergman, A., A. Bergstrand, and A. Bignert. **2001**. Renal Lesions in Baltic Grey Seals (*Halichoerus grypus*) and Ringed Seals (*Phoca hispida botnica*). Ambio 30(7):397-409.

Bernard, H. J., and A. A. Hohn. **1989**. Differences in feeding habits between pregnant and lactating spotted dolphins (*Stenella attenuata*). Journal of Mammalogy 70(1):211-215.

Bettridge, S., and G. K. Silber .**2008**. Update on the United States' actions to reduce the threat of ship collisions with large whales. Prepared for the International Whaling Commission's Working Group on Ship Strikes and Presented at the International Whaling Commission's Conservation Committee, Santiago, Chile.

Binckley, C. A., J. R. Spotila, K. S. Wilson, and F. V. Paladino. **1998**. Sex determination and sex ratios of Pacific leatherback turtles, Dermochelys coriacea. Copeia 1998(2):291-300.

Bjorndal, K. A., A. B. Bolten, T. Dellinger, C. Delgado, and H. R. Martins. **2003a**. Compensatory growth in oceanic loggerhead sea turtles: response to a stochastic environment. Ecology 84(5):1237-1249.

Bjorndal, K. A., A. B. Bolten, and M. Y. Chaloupka. **2000a**. Green turtle somatic growth model: evidence for density dependence. Ecological Applications 10(1):269-282.

Bjorndal, K. A., A. B. Bolten, and M. Y. Chaloupka. **2005**. Evaluating trends in abundance of immature green turtles, *Chelonia mydas*, in the greater Caribbean. Ecological Applications 15(1):304-314.

Bjorndal, K. A., A. B. Bolten, and H. R. Martins. **2000b**. Somatic growth model of juvenile loggerhead sea turtles *Caretta caretta*: Duration of pelagic stage. Marine Ecology Progress Series 202:265-272.

Bjorndal, K. A., A. B. Bolten, B. A. Schroeder, D. J. Shaver, W. G. Teas, and W. N. Witzell. **2001**. Somatic growth function for immature loggerhead sea turtles, *Caretta caretta*, in southeastern US waters. Fishery Bulletin 99:240-246.

Bjorndal, K. A., A. B. Bolten, and H. R. Martins. **2003b**. Estimates of survival probabilities for oceanic-stage loggerhead sea turtles (*Caretta caretta*) in the North Atlantic. Fishery Bulletin 101(4):732-736.

Bjorndal, K. A., A. B. Bolten, and C. J. Lagueux. **1994**. Ingestion of marine debris by juvenile sea turtles in coastal Florida habitats. Marine Pollution Bulletin 28(3):154-158.

Bjorndal, K. A., and A. B. Bolten. **2009**. Hawksbill sea turtles in seagrass pastures: success in a peripheral habitat. Marine Biology 157(1):135-145.

Blanco, C., M. Á. Raduán, and J. A. Raga. **2006**. Diet of Risso's dolphin (*Grampus griseus*) in the western Mediterranean Sea. Scientia Marina 70(3):407-411.

Blanvillain, G., L. D. Wood, A. B. Meylan, and P. A. Meylan. **2007**. Sex ratio prediction of juvenile hawksbill sea turtles (*Eretmochelys imbricata*) from South Florida, USA. Herpetological Conservation and Biology 3:21-27.

Bloodworth, B., and C. D. Marshall. **2005**. Feeding kinematics of *Kogia* and *Tursiops* (Odontocete: Cetacea): characterization of suction and ram feeding. Journal of Experimental Biology 208(19):3721-3730.

Blumenthal, J. M. et al. **2009a**. Turtle groups or turtle soup: dispersal patterns of hawksbill turtles in the Caribbean. Molecular Ecology 18(23):4841-4853.

Blumenthal, J. M., T. J. Austin, C. D. L. Bell, and J. B. Bothwell. **2009b**. Ecology of hawksbill turtles, *Eretmochelys imbricata*, on a Western Caribbean foraging ground. Chelonian Conservation and Biology 8(1):1-10.

Blumenthal, J. M., T. J. Austin, J. B. Bothwell, A. C. Broderick, G. Ebanks-Petrie, J. R. Olynik, M. F. Orr, J. L. Solomon, M. J. Witt, and B. J. Godley **2009c**. Diving behavior and movements of juvenile hawksbill turtles *Eretmochelys imbricata* on a Caribbean coral reef. Coral Reefs 28(1):55-65.

Blumenthal, J., T. Austin, J. Bothwell, A. Broderick, G. Ebanks-Petrie, J. Olynik, M. Orr, J. Solomon, M. Witt, and B. Godley. **2010**. Life in (and out of) the lagoon: fine-scale movements of green turtles tracked using time-depth recorders. Aquatic Biology 9:113-121.

Boehlert, G. W., and A. B. Gill. **2010**. Environmental and ecological effects of ocean renewable energy development: A current synthesis. Oceanography 23(2):68-81.

Boehlert, G. W., G. R. McMurray, and C. E. Tortorici. **2007**. Ecological effects of wave energy development in the Pacific Northwest. US Department of Commerce, NOAA Technical Memorandum 11-12.

Bogomolni, A. L., K. R. Pugliares, S. M. Sharp, K. Patchett, C. T. Harry, J. M. LaRocque, K. M. Touhey, and M. Moore. **2010**. Mortality trends of stranded marine mammals on Cape Cod and southeastern Massachusetts, USA, 2000 to 2006. Diseases of Aquatic Organisms 88:143-155.

Boisseau, O., D. Gillespie, R. Leaper, and A. Moscrop. **2008**. Blue (*Balaenoptera musculus*) and fin (*B. physalus*) whale vocalisations measured from northern latitudes of the Atlantic Ocean. Journal of Cetacean Research and Management 10(1):23-30.

Bolten, A. B. **2003**. Active swimmers–passive drifters: the oceanic juvenile stage of loggerheads in the Atlantic system, In: AB Bolten and BE Witherington (editors), Loggerhead Sea Turtles, pp. 63-78, Smithsonian Institution Press, Washington, DC.

Bolten, A. B., K. A. Bjorndal, H. R. Martins, T. Dellinger, M. Biscoito, S. E. Encalada, and B. W. Bowen. **1998**. Transatlantic developmental migrations of loggerhead sea turtles demonstrated by mtDNA sequence analysis. Ecological Applications 8(1):1-7.

Bolton, A. B., and H. R. Martins. **1990**. Kemp's Ridley captured in the Azores. Marine Turtle Newsletter 48(23).

Bonde, R. K., A. A. Aguirre, and J. Powell. **2004**. Manatees as sentinels of marine ecosystem health: are they the 2000-pound canaries? EcoHealth 1(3):255-262.

Boness, D. J., W. D. Bowen, and S. J. Iverson. **1995**. Does male harassment of females contribute to reproductive synchrony in the grey seal by affecting maternal performance? Behavioral Ecology and Sociobiology 36(1):1-10.

Boness, D. J., and H. James. **1979**. Reproductive behaviour of the grey seal (*Halichoerus grypus*) on Sable Island, Nova Scotia. Journal of Zoology 188(4):477-500.

Borggaard, D., J. Lien, and P. Stevick. **1999**. Assessing the effects of industrial activity on large cetaceans in Trinity Bay, Newfoundland (1992-1995). Aquatic Mammals 25(3):149-161.

Born, E. W., H. Dahlgaard, F. F. Riget, R. Dietz, N. Øien, and T. Haug. **2002**. Regional variation of caesium-137 in minke whales *Balaenoptera acutorostrata* from West Greenland, the Northeast Atlantic and the North Sea. Polar Biology 25(12):907-913.

Born, E. W., P. Outridge, F. F. Riget, K. A. Hobson, R. Dietz, N. Řien, and T. Haug. **2003**. Population substructure of North Atlantic minke whales (*Balaenoptera acutorostrata*) inferred from regional variation of elemental and stable isotopic signatures in tissues. Journal of Marine Systems 43(1-2):1-17.

Born, E. W., M. C. Kingsley, F. F. Rigét, R. Dietz, P. Møller, T. Haug, D. C. Muir, P. Outridge, and N. Øien. **2007**. A multi-elemental approach to identification of subpopulations of North Atlantic minke whales *Balaenoptera acutorostrata*. Wildlife Biology 13(1):84-97.

Borobia, M., P. J. Gearing, Y. Simard, J. N. Gearing, and P. Béland. **1995**. Blubber fatty acids of finback and humpback whales from the Gulf of St. Lawrence. Marine Biology 122(3):341-353.

Borrell, A., D. Bloch, and G. Desportes. **1995**. Age trends and reproductive transfer of organochlorine compounds in long-finned pilot whales from the Faroe Islands. Environmental Pollution 88(3):283-292.

Boskovic, R., K. M. Kovacs, M. O. Hammill, and B. N. White. **1996**. Geographic distribution of mitochondrial DNA haplotypes in grey seals (*Halichoerus grypus*). Canadian Journal of Zoology 74(10):1787-1796.

Bossart, G. D., D. G. Baden, R. Y. Ewing, B. Roberts, and S. D. Wright. **1998**. Brevetoxicosis in manatees (*Trichechus manatus latirostris*) from the 1996 epizootic: Gross, histologic, and immunohistochemical features. Toxicologic Pathology 26(2):276-282.

Bossart, G. D., R. Y. Ewing, M. Lowe, M. Sweat, S. J. Decker, C. J. Walsh, S.-je Ghim, and A. Bennett Jenson. **2002**. Viral papillomatosis in Florida manatees (*Trichechus manatus latirostris*). Experimental and Molecular Pathology 72(1):37-48.

Bossart, G. D. et al. **2007a**. Pathological findings in a rare mass stranding of melon-headed whales (*Peponocephala electra*) in Florida. Aquatic Mammals 33(2):235-240.

Bossart, G. D., G. Hensley, J. D. Goldstein, K. Kroell, C. A. Manire, R. H. Defran, and J. S. Reif. **2007b**. Cardiomyopathy and myocardial degeneration in stranded pygmy (*Kogia breviceps*) and dwarf (*Kogia sima*) sperm whales. Aquatic Mammals 33(2):214-222.

Bossart, G. D., C. Cray, J. L. Solorzano, S. J. Decker, L. H. Cornell, and N. H. Altman. **1996**. Cutaneous papillomaviral-like papillomatosis in a killer whale (*Orcinus orca*). Marine Mammal Science 12(2):274-281.

Bossart, G. D., J. S. Reif, A. M. Schaefer, J. Goldstein, P. A. Fair, and J. Saliki. **2010**. Morbillivirus infection in free-ranging Atlantic bottlenose dolphins (*Tursiops truncatus*) from the southeastern United States: Seroepidemiologic and pathologic evidence of subclinical infection. Veterinary Microbiology 143:160-166.

Bostrom, B. L., T. T. Jones, M. Hastings, and D. R. Jones. **2010**. Behaviour and physiology: The thermal strategy of leatherback turtles. PLoS ONE 5(11):e13925.

Boulon, R. H., Jr., P. H. Dutton, and D. L. McDonald. **1996**. Leatherback turtles (*Dermochelys coriacea*) on St. Croix, U. S. Virgin Islands: Fifteen years of conservation. Chelonian Conservation and Biology 2(2):141-147.

Bourgeois, S., E. Gilot-Fromont, A. Viallefont, F. Boussamba, and S. L. Deem. **2009**. Influence of artificial lights, logs and erosion on leatherback sea turtle hatchling orientation at Pongara National Park, Gabon. Biological Conservation 142(1):85-93.

Bowen, B. W. et al. **2004**. Natal homing in juvenile loggerhead turtles (*Caretta caretta*). Molecular Ecology 13(12):3797-3808.

Bowen, B. W., A. L. Bass, L. Soares, and R. J. Toonen. **2005**. Conservation implications of complex population structure: lessons from the loggerhead turtle (*Caretta caretta*). Molecular Ecology 14(8):2389-2402.

Bowen, B. W., W. S. Grant, Z. Hillis-Starr, D. J. Shaver, K. A. Bjorndal, A. B. Bolten, and A. L. Bass. **2007a**. Mixed-stock analysis reveals the migrations of juvenile hawksbill turtles (*Eretmochelys imbricata*) in the Caribbean Sea. Molecular Ecology 16(1):49-60.

Bowen, B. W., and S. A. Karl. **2007**. Population genetics and phylogeography of sea turtles. Molecular Ecology 16(23):4886-4907.

Bowen, M. W., T. A. Conant, and S. R. Hopkins-Murphy. **1994**. Where are they now? The Kemp's ridley headstart project. Conservation Biology 8(3):853-856.

Bowen, W. D., J. I. McMillan, D. Ligard, and W. Blanchard. **2007b**. Continued reduction in population growth rate of grey seals at Sable Island., Canadian Science Advisory Secretariat (CSAS) Res. Doc. 2007/087.

Bowen, W. D., J. I. McMillan, and W. Blanchard. **2007c**. Reduced population growth of grey seals at Sable Island: evidence from pup production and age of primiparity. Marine Mammal Science 23(1):48-64.

Bowen, W. D., O. T. Oftedal, and D. J. Boness. **1985**. Birth to weaning in 4 days: remarkable growth in the hooded, *Cystophora* cristata. Canadian Journal of Zoology 63:2841-2842.

Bowen, W. D., and G. D. Harrison. **1994**. Offshore diet of grey seals *Halichoerus grypus* near Sable Island, Canada. Marine Ecology Progress Series 112(1):1-11.

Bowen, W. D., and D. B. Siniff. **1999**. Distribution, population biology, and feeding ecology of marine mammals, In: Pages 423-84 in: W. D. Bowen, D. B. Siniff, J. E. Reynolds and S. A. Rommel, (eds.) Biology of Marine Mammals, Smithsonian Institution Press.

Bowen, W. D., O. T. Oftedal, and D. J. Boness .**1992**. Mass and energy transfer during lactation in a small phocid, the harbor seal (*Phoca vitulina*). Physiological Zoology 65(4):844-866.

Bowen, W., S. L. Ellis, S. J. Iverson, and D. J. Boness. **2003**. Maternal and newborn life-history traits during periods of contrasting population trends: implications for explaining the decline of harbour seals (*Phoca vitulina*), on Sable Island. Journal of Zoology 261(2):155-163.

Bowers, M. **2010**. Phenological shifts in loggerhead sea turtle nesting dates. MS thesis. Duke University. 16 p.

Boyle, M. C., and C. J. Limpus. **2008**. The stomach contents of post-hatchling green and loggerhead sea turtles in the southwest Pacific: an insight into habitat association. Marine Biology 155(2):233-241.

Bradshaw, C. J. A., C. R. McMahon, and G. C. Hays. **2007**. Behavioral inference of diving metabolic rate in free-ranging leatherback turtles. Physiological and Biochemical Zoology 80(2):209-219.

Brandt, M., A. Diederichs, K. Betke, and G. Nehls. **2011**. Responses of harbour porpoises to pile driving at the Horns Rev II offshore wind farm in the Danish North Sea. Marine Ecology Progress Series 421:205-216.

Braun-McNeill, J., S. P. Epperly, L. Avens, M. L. Snover, and J. C. Taylor. **2008**. Growth rates of loggerhead sea turtles (*Caretta caretta*) from the western North Atlantic. Herpetological Conservation and Biology 3(2):273-281.

Braun-McNeill, J., S. P. Epperly, D. W. Owens, L. Avens, E. Williams, and C. A. Harms. **2007a**. Seasonal reliability of testosterone radioimmunoassay (RIA) for predicting sex ratios of juvenile loggerhead (*Caretta caretta*) turtles. Herpetologica 63(3):275-284.

Braun-McNeill, J., C. R. Sasso, and L. Avens. **2007b**. Estimates of realized survival for juvenile loggerhead sea turtles (*Caretta caretta*) in the United States. Herpetological Conservation and Biology 2:100-105.

Brazner, J. C., and J. McMillan. **2008**. Loggerhead turtle (*Caretta caretta*) bycatch in Canadian pelagic longline fisheries: Relative importance in the western North Atlantic and opportunities for mitigation. Fisheries Research 91(2-3):310-324.

Brilliant, S. W., and E. A. Trippel. **2009**. Elevations of lobster fishery groundlines in relation to their potential to entangle endangered North Atlantic right whales in the Bay of Fundy, Canada. ICES Journal of Marine Science 67:355-364.

Brinn, L. A., R. R. Carthy, and L. Patrick. **2010**. Assessement of beach compaction and associated effects on loggerhead sea turtle (*Caretta caretta*) nesting on natural and nourished beaches in northwest Florida, In: Proceedings of the 28th annual symposium on sea turtle biology and conservation, p. 76, NOAA Tech Memo NMFS-SEFSC-602.

Brito, M., P. L. Tyack, and W. X. Zimmer. **2008**. Do North Atlantic right whales compensate for increases in short term ambient noise?, American Geophysical Union, 2000 Florida Ave., N. W. Washington DC 20009 USA.

Broderick, A. C., B. J. Godley, and G. C. Hays. **2001**. Trophic status drives interannual variability in nesting numbers of marine turtles. Proceedings of the Royal Society of London. Series B: Biological Sciences 268(1475):1481-1487.

Brongersma, L. D. **1972**. European Atlantic Turtles. Zoologische Verhandelingen 121(1):1-318.

Brookens, T. J., T. M. O'Hara, R. J. Taylor, G. R. Bratton, and J. T. Harvey .**2008**. Total mercury body burden in Pacific harbor seal, *Phoca vitulina richardii*, pups from central California. Marine Pollution Bulletin 56(1):27-41.

Brooks, L., J. Harvey, and W. Nichols. **2009**. Tidal movements of East Pacific green turtle *Chelonia mydas* at a foraging area in Baja California Sur, México. Marine Ecology Progress Series 386:263-274.

Brouwer, A., P. J. H. Reijnders, and J. H. Koeman. **1989**. Polychlorinated biphenyl (PCB)-contaminated fish induces vitamin A and thyroid hormone deficiency in the common seal (*Phoca vitulina*). Aquatic Toxicology 15(1):99-105.

Brown, C. W., and H. E. Winn. **1989**. Relationship between the distribution pattern of right whales, *Eubalaena glacialis*, and satellite-derived sea surface thermal structure in the Great South Channel. Continental Shelf Research 9(3):247-260.

Brown, R. R., and B. R. Mate. **1982**. Abundance, movements, and feeding habits of harbor seals, *Phoca vitulina*, at Netarts and Tillamook Bays, Oregon. Fishery Bulletin 81(2):291-302.

Browne, D. C., F. A. Abreu-Grobois, R. King, C. Lloyd, C. A. Isaac, and J. A. Horrocks. **2008**. Population structure of hawksbill turtles (*Eretmochelys Imbricata*) at rookeries and foraging areas in Grenada, West Indies, based on mitochondrial dna sequences, In: Proceedings of the 27th annual symposium on sea turtle biology and conservation, p. 119, NOAA Tech Memo NMFS-SEFSC-569.

Brownell Jr, R. L., C. A. Carlson, B. Galletti Vernazzani, and E. Cabrera. **2007**. Skin lesions on blue whales off southern Chile: Possible conservation implications?, International Whaling Commission Report SC/59/SH21.

Brownell Jr., R. L., K. Ralls, S. Baumann-Pickering, and M. M. Poole. **2009**. Behavior of melon-headed whales, *Peponocephala electra*, near oceanic islands. Marine Mammal Science 25(3):639-658.

Browning, C. L., R. M. Rolland, and S. D. Kraus. **2010**. Estimated calf and perinatal mortality in western North Atlantic right whales (*Eubalaena glacialis*). Marine Mammal Science 26(3):648-662.

Bryden, M. M., R. J. Harrison, and R. J. Lear. **1977**. Some aspects of the biology of *Peponocephala electra* (Cetacea: Delphinidae). I. General and reproductive biology. Marine and Freshwater Research 28(6):703-715.

Buck, J. D., and S. Spotte. **2005**. Microbiology of captive white-beaked dolphins (*Lagenorhynchus albirostris*) with comments on epizootics. Zoo Biology 5(4):321-329.

Buckingham, C. A., L. W. Lefebvre, J. M. Schaefer, and H. I. Kochman. **1999**. Manatee response to boating activity in a thermal refuge. Wildlife Society Bulletin 27(2):514-522.

Buckstaff, S. **2004**. Effects of watercraft noise on the acoustic behavior of bottlenose dolphins, *Tursiops truncatus*, in Sarasota Bay, Florida. Marine Mammal Science 20(4):709-725.

Burgman, J. **2010**. An analysis of the potential acoustic effects of Cape Wind's offshore wind farm on marine mammal populations. MA thesis. Duke University. 84 p.

Burke, V. J., E. A. Standora, and S. J. Morreale. **1993**. Diet of juvenile Kemp's ridley and loggerhead sea turtles from Long Island, New York. Copeia 1993(4):1176-1180.

Burns, J. J. **2009**. Harbor seal and spotted seal (*Phoca vitulina* and *P. largha*)., In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, pp. 533-542, Academic Press Inc.

Bush, D. **2011**. Uncertainty and the impacts of offshore wind. MS thesis. Duke University, Durham, NC.

Bustamante, P., C. Garrigue, L. Breau, F. Caurant, W. Dabin, J. Greaves, and R. Dodemont. **2003**. Trace elements in two odontocete species (*Kogia breviceps* and *Globicephala macrorhynchus*) stranded in New Caledonia (South Pacific). Environmental Pollution 124(2):263-271.

Bustamante, P., C. Morales, B. Mikkelsen, M. Dam, and F. Caurant. **2004**. Trace element bioaccumulation in grey seals *Halichoerus grypus* from the Faroe Islands. Marine Ecology Progress Series 267:291-301.

Byrd, B. L., A. Hohn, and M. H. Godfrey. **2011**. Emerging fisheries, emerging fishery interactions with sea turtles: A case study of the large-mesh gillnet fishery for flounder in Pamlico Sound, North Carolina, USA. Marine Policy 271-285.

Byrd, J., S. Murphy, and A. Von Harten. **2005**. Morphometric analysis of the northern subpopulation of *Caretta caretta* in South Carolina, USA. Marine Turtle Newsletter 107:1–4.

Byrne, R., J. Fish, T. K. Doyle, and J. D. R. Houghton. **2009**. Tracking leatherback turtles (*Dermochelys coriacea*) during consecutive inter-nesting intervals: Further support for direct transmitter attachment. Journal of Experimental Marine Biology and Ecology 377(2):68–75.

Bérubé, M., A. Aguilar, D. Dendanto, G. Notabartolo Di Sciara, R. Sears, J. Sigurjónsson, J. Urban-R, and P. J. Palsbøll. **1998**. Population genetic structure of North Atlantic, Mediterranean Sea and Sea of Cortez fin whales, *Balaenoptera physalis* (Linnaeus 1758): analysis of mitochondrial and nuclear loci. Molecular Ecology 7:585-599.

Bøthun, G., H. J. Skaug, and N. Øien. **2009**. Abundance of minke whales in the Northeast Atlantic based on survey data collected over the period 2002-2007. International Whaling Commission Scientific Committee Paper IWC SC/61/RMP2.

Cabezon, O., A. J. Hall, C. Vincent, M. Pabon, I. Garcia-Bocanegra, J. P. Dubey, and S. Almeria. **2011**. Seroprevalence of *Toxoplasma gondii* in North-eastern Atlantic harbor seal

(*Phoca vitulina vitulina*) and grey seal (*Halichoerus grypus*). Veterinary Parasitology 179:253-256.

Cada, G., J. Ahlgrimm, M. Bahleda, T. Bigford, S. D. Stavrakas, D. Hall, R. Moursund, and M. Sale .**2007**. Potential impacts of hydrokinetic and wave energy conversion technologies on aquatic environments. Fisheries 32(4):174-181.

Caldwell, D. K., and M. C. Caldwell. **1966**. Observations on the distribution, coloration, behavior and audible sound production of the spotted dolphin, *Stenella plagiodon* (Cope). Los Angeles County Museum Contribution to Science 104:1-28.

Caldwell, D. K., and M. C. Caldwell. **1989**. Pygmy sperm whale, *Kogia breviceps*, dwarf sperm whale *Kogia simus*., In: Handbook of Marine Mammals, vol. 4, pp. 235-260, Academic Press Inc., San Diego.

Caldwell, D. K., and M. Caldwell. **1971a**. The pygmy killer whale, *Feresa attenuata*, in the western Atlantic, with a summary of world records. Journal of Mammalogy 52:206-206.

Caldwell, D. K., and M. C. Caldwell. **1971b**. Underwater pulsed sounds produced by captive spotted dolphins, *Stenella plagiodon*. Cetology 1:1-17.

Caldwell, M. **2001**. Social and genetic strucutre of bottlenose dolphin (*Tursiops truncatus*) in Jacksonville, Florida. Ph.D. thesis. University of Miami, Miami, Florida. 143 p.

Caldwell, M. C., D. K. Caldwell, and J. F. Miller. **1973**. Statistical evidence for individual signature whistles in the spotted dolphin, *Stenella plagiodon*. Cetology 16:1-21.

Caldwell, M. C., N. R. Hall, and D. K. Caldwell. **1971**. Ability of an Atlantic bottlenosed dolphin (*Tursiops truncatus*) to discriminate the vocalizations of the Atlantic spotted dolphin (*Stenella plagiodon*). Cetology 6:1-16.

Calleson, C. S., and R. K. Frolich. **2007**. Slower boat speeds reduce risks to manatees. Endangered Species Research 3(3):295-304.

Cammen, K., J. I. Hoffman, L. A. Knapp, J. Harwood, and W. Amos. **2011**. Geographic variation of the major histocompatibility complex in Eastern Atlantic grey seals (*Halichoerus grypus*). Molecular Ecology 20(4):740-752.

Campbell, C. L., C. J. Lagueux, and J. A. Mortimer. **1996**. Leatherback turtle, *Dermochelys coriacea*, nesting at Tortuguero, Costa Rica, in 1995. Chelonian Conservation and Biology 2(2):169-172.

Cannon, L. R. G. **1977**. Some aspects of the biology of *Peponocephala electra* (Cetacea: Delphinidae). II. Parasites. Marine and Freshwater Research 28(6):717–722.

Carlson, B. K. R., J. Braun-McNeill, L. Avens, A. L. Segars, J. R. Kucklick, and J. M. Keller. **2010**. Polybrominated diphenyl ethers and organochlorine contaminants in loggerhead sea turtles (*Caretta caretta*) distribution among blood components and temporal trends, In:

Proceedings of the 28th annual symposium on sea turtle biology and conservation, p. 8, NOAA Tech Memo NMFS-SEFSC-602.

Carney, S. L., E. E. Bolen, S. L. Barton, K. M. Scolardi, C. C. Englund, M. D. Tringali, and J. E. Reynolds. **2007**. A minimally invasive method of field sampling for genetic analysis of the Florida manatee (*Trichechus manatus latirostris*). Marine Mammal Science 23(4):967-975.

Carr, A. **1963**. Panspecific reproductive convergence in *Lepidochelys kempii*. Ergebnisse der Biologye 26:298-303.

Carr, S. M., H. D. Marshall, A. T. Duggan, S. Flynn, K. A. Johnstone, A. M. Pope, and C. D. Wilkerson. **2008**. Phylogeographic genomics of mitochondrial DNA: highly-resolved patterns of intraspecific evolution and a multi-species, microarray-based DNA sequencing strategy for biodiversity studies. Comparative Biochemistry and Physiology Part D: Genomics and Proteomics 3(1):1–11.

Carstensen, J., O. D. Henriksen, and J. Teilmann. **2006**. Impacts of offshore wind farm construction on harbour porpoises: acoustic monitoring of echolocation activity using porpoise detectors (T-PODs). Marine Ecology Progress Series 312:295-308.

Casale, P., G. Abbate, D. Freggi, N. Conte, M. Oliverio, and R. Argano. **2008**. Foraging ecology of loggerhead sea turtles Caretta caretta in the central Mediterranean Sea: evidence for a relaxed life history model. Marine Ecology Progress Series 372:265–276.

Castellote, M., C. W. Clark, and M. O. Lammers. **2010**. Potential negative effects in the reproduction and survival on fin whales (*Balaenoptera physalus*) by shipping and airgun noise. International Whaling Commission Scientific Committee Paper IWC SC/62/E3.

Castellote, M., C. W. Clark, F. Colmenares, and J. A. Esteban. **2009**. Mediterranean fin whale migration movements altered by seismic exploration noise. Journal of the Acoustical Society of America 125(4):2519-2519.

Caswell, H., M. Fujiwara, and S. Brault. **1999**. Declining survival probability threatens the North Atlantic right whale. Proceedings of the National Academy of Sciences of the United States of America 96:3308-3313.

Caudron, A. K., A. A. Kondakov, and S. V. Siryanov. **1998**. Acoustic structure and individual variation of grey seal (*Halichoerus grypus*) pup calls. Journal of the Marine Biological Association of the United Kingdom 78(2):651-658.

Caurant, F., J. C. Amiard, C. Amiard-Triquet, and P. G. Sauriau. **1994**. Ecological and biological factors controlling the concentrations of trace elements (As, Cd, Cu, Hg, Se, Zn) in delphinids *Globicephala melas* from the North Atlantic Ocean. Marine Ecology Progress Series 103:207-219.

Cerchio, S., T. Collins, S. Strindberg, C. Bennett, and H. Rosenbaum. **2011**. Humpback whale singing activity off northern Angola: an indication of the migratory cycle, breeding habitat and impact of seismic surveys on singer number in Breeding Stock B1. International Whaling Commission Scientific Committee Paper SC/62/SH12.

Cerchio, S., J. K. Jacobsen, and T. F. Norris. **2001**. Temporal and geographical variation in songs of humpback whales, *Megaptera novaeangliae*: synchronous change in Hawaiian and Mexican breeding assemblages. Animal Behaviour 62(2):313–329.

Chaloupka, M. **2002**. Stochastic simulation modelling of southern Great Barrier Reef green turtle population dynamics. Ecological Modelling 148(1):79–109.

Chaloupka, M., K. A. Bjorndal, G. H. Balazs, A. B. Bolten, L. M. Ehrhart, C. J. Limpus, H. Suganuma, S. Troëng, and M. Yamaguchi. **2008**. Encouraging outlook for recovery of a once severely exploited marine megaherbivore. Global Ecology and Biogeography 17(2):297–304.

Chaloupka, M., and G. Balazs. **2007**. Using Bayesian state-space modelling to assess the recovery and harvest potential of the Hawaiian green sea turtle stock. Ecological Modelling 205(1-2):93–109.

Chen, T. L., S. S. Wise, A. Holmes, F. Shaffiey, J. P. Wise Jr, W. D. Thompson, S. Kraus, and J. P. Wise Sr. **2009**. Cytotoxicity and genotoxicity of hexavalent chromium in human and North Atlantic right whale (*Eubalaena glacialis*) lung cells. Comparative Biochemistry and Physiology Part C: Toxicology & Pharmacology 150(4):487-494.

Chevalier, J., X. Desbois, and M. Girondot. **1999**. The reason of decline of leatherback turtles (*Dermochelys coriacea*) in French Guiana: an hypothesis, In: 9th Extraordinary Meeting of the Societas Europaea Herpetologica, R. Guyétant and C. Miaud, eds.(Université de Savoie, Le Bourget du Lac: Université de Savoie).

Chittleborough, R. G. **1965**. Dynamics of two populations of the humpback whale, *Megaptera novaeangliae* (Bonnaterre). Australian Journal of Marine and Freshwater Research 16:33-128.

Chivers, S. J., R. G. Leduc, K. M. Robertson, N. B. Barros, and A. E. Dizon. **2005**. Genetic variation of *Kogia* spp. with preliminary evidence for two species of *Kogia sima*. Marine Mammal Science 21(4):619–634.

Cholewiak, D. **2008**. Evaluating the role of song in the humpback whale (*Megaptera novaeangliae*) breeding system with respect to intra-sexual interactions. Ph. D. thesis. Cornell University, Ithaca, NY. 174 p.

Christal, J., H. Whitehead, and E. Lettevall. **1998**. Sperm whale social units: variation and change. Canadian Journal of Zoology 76(8):1431–1440.

Christal, J., and H. Whitehead. **2001**. Social affiliations within sperm whale (*Physeter macrocephalus*) groups. Ethology 107(4):323–340.

Chung, F. C., N. J. Pilcher, M. Salmon, and J. Wyneken. **2009a**. Offshore migratory activity of hawksbill turtle (*Eretmochelys imbricata*) hatchlings, I. Quantitative analysis of activity, with comparisons to green turtles (*Chelonia mydas*). Chelonian Conservation and Biology 8(1):28-34.

Chung, F. C., N. J. Pilcher, M. Salmon, and J. Wyneken. **2009b**. Offshore migratory activity of hawksbill turtle (*Eretmochelys imbricata*) hatchlings, II. Swimming gaits, swimming speed, and morphological comparisons. Chelonian Conservation and Biology 8(1):35-42.

Cipriano, F. **2009**. Atlantic white-sided dolphin *Lagenorhynchus acutus*, In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, pp. 56-58, Academic Press, San Diego.

Clapham, P. J., and P. J. Palsbøll. **1997**. Molecular analysis of paternity shows promiscuous mating in female humpback whales (*Megaptera novaeangliae*, Borowski). Proceedings of the Royal Society B: Biological Sciences 264:95-98.

Clapham, P. J. **2009**. Humpback Whale, In: Pages 582-584 in: W. Perrin, B. Wursig and J. G. M. Thewissen, (eds.) Encyclopedia of Marine Mammals, 2nd ed, Academic Press, New York.

Clapham, P., J. Barlow, M. Bessinger, T. Cole, D. Mattila, R. Pace, D. Palka, J. Robbins, and R. Seton. **2003**. Abundance and demographic parameters of humpback whales from the Gulf of Maine, and stock definition relative to the Scotian Shelf. Journal of Cetacean Research and Management 5(1):13-22.

Clapham, P. J. **2000**. The humpback whale: seasonal feeding and breeding in a baleen whale, In: in: J. Mann, P. L. Tyack, R. Connor and H. Whitehead, (eds.) Cetacean Societies, pp. 173-196, University of Chicago Press, Chicago, IL.

Clapham, P. J., and I. E. Seipt. **1991**. Resightings of independent fin whales, *Balaenoptera physalus*, on maternal summer ranges. Journal of Mammology 72:788-790.

Clapham, P. J., L. S. Baraff, C. A. Carlson, M. A. Christian, D. K. Mattila, C. A. Mayo, M. A. Murphy, and S. Pittman. **1993**. Seasonal occurrence and annual return of humpback whales, *Megaptera novaeangliae*, in the southern Gulf of Maine. Canadian Journal of Zoology 71(2):440-443.

Claridge, D. E. **2006**. Fine-scale distribution and habitat selection of beaked whales. MSc thesis. University of Abredeen, Scotland, U.K. 127 p.

Clark, C. W. **1995**. Application of US Navy underwater hydrophone arrays for scientific research on whales. Reports of the International Whaling Commission 45:210–212.

Clark, C. W., W. T. Ellison, B. L. Southall, L. Hatch, S. M. Van Parijs, A. Frankel, and D. Ponirakis. **2009**. Acoustic masking in marine ecosystems: intuitions, analysis, and implication. Marine Ecology Progress Series 395:201–222.

Clark, C. W., and P. J. Clapham. **2004**. Acoustic monitoring on a humpback whale (*Megaptera novaeangliae*) feeding ground shows continual singing into late spring. Proceedings of the Royal Society B: Biological Sciences 271(1543):1051-1057.

Clark, C. W., M. W. Brown, and P. Corkeron. **2010**. Visual and acoustic surveys for North Atlantic right whales, *Eubalaena glacialis*, in Cape Cod Bay, Massachusetts, 2001-2005: Management implications. Marine Mammal Science 26(4):837-854.

Clarke, M. R., and T. K. Kristensen. **1980**. Cephalopod beaks from the stomachs of two northern bottlenosed whales (*Hyperoodon ampullatus*). Journal of the Marine Biological Association of the United Kingdom 60:151-156.

Clarke, M. R., and R. Young. **1998**. Description and analysis of cephalopod beaks from stomachs of six species of odontocete cetaceans stranded on Hawaiian shores. Journal of the Marine Biological Association of the United Kingdom 78:623-641.

Coakes, A., S. Gowans, P. Simard, J. Giard, C. Vashro, and R. Sears. **2005**. Photographic identification of fin whales (*Balaenoptera physalus*) off the Atlantic coast of Nova Scotia, Canada. Marine Mammal Science 21(2):323–326.

Colbert, D. E., J. C. Gaspard 3rd, R. Reep, D. A. Mann, and G. B. Bauer .**2009**. Four-choice sound localization abilities of two Florida manatees, *Trichechus manatus latirostris*. The Journal of Experimental Biology 212(Pt 13):2105.

Coles, W. C., and J. A. Musick. **2000**. Satellite sea surface temperature analysis and correlation with sea turtle distribution off North Carolina. Copeia 2000(2):551-554.

Coltman, D. W., W. D. Bowen, and J. M. Wright. **1998a**. Birth weight and neonatal survival of harbour seal pups are positively correlated with genetic variation measured by microsatellites. Proceedings of the Royal Society of London. Series B: Biological Sciences 265(1398):803.

Coltman, D. W., W. D. Bowen, and J. M. Wright. **1998b**. Male mating success in an aquatically mating pinniped, the harbour seal (*Phoca vitulina*), assessed by microsatellite DNA markers. Molecular Ecology 7(5):627-638.

Coltman, D. W., G. B. Stenson, M. O. Hammill, T. Haug, C. S. Davis, and T. L. Fulton. **2007**. Panmictic population structure in the hooded seal (*Cystophora cristata*). Molecular Ecology 16(8):1639-1648.

Committee on Sea Turtle Population Assessment Methods; National Research Council. **2010**. Assessment of sea turtle status and trends: Integrating demography and abundance, The National Academies Press, Washington, D.C.

Compton, R., L. Goodwin, R. Handy, and V. Abbott. **2008**. A critical examination of worldwide guidelines for minimising the disturbance to marine mammals during seismic surveys. Marine Policy 32(3):255-262.

Conant, T. A. et al. **2009**. Loggerhead sea turtle (*Caretta caretta*) 2009 status review under the U.S. Endangered Species Act. Report of the loggerhead biological review team to the National Marine Fisheries Service, August 2009.

Constantine, R., D. Brunton, and T. Dennis. **2004**. Dolphin-watching tour boats change bottlenose dolphin (*Tursiops truncatus*) behaviour. Biological Conservation 117:299-307.

Constantino, M. A., and M. Salmon. **2003**. Role of chemical and visual cues in food recognition by leatherback posthatchlings (*Dermochelys coriacea* L). Zoology 106(3):173.

Cook, M. L. H., R. A. Varela, J. D. Goldstein, S. D. McCulloch, G. D. Bossart, J. J. Finneran, D. Houser, and D. A. Mann. **2006**. Beaked whale auditory evoked potential hearing measurements. Journal of Comparative Physiology A: Neuroethology, Sensory, Neural, and Behavioral Physiology 192(5):489–495.

Cook, M. L. H., L. S. Sayigh, J. E. Blum, and R. S. Wells. **2004**. Signature-whistle production in undisturbed free-ranging bottlenose dolphins (*Tursiops truncatus*). Proceeding of the Royal Society B 271:1043-1049.

Corkeron, P. J., and S. M. Van Parijs. **2001**. Vocalizations of eastern Australian Risso’s dolphins, *Grampus griseus*. Canadian Journal of Zoology 79(1):160-164.

Corkeron, P. J. **1995**. Humpback whales (*Megaptera novaeangliae*) in Hervey Bay, Queensland: behaviour and responses to whale-watching vessels. Canadian Journal of Zoology 73(7):1290-1299.

Costa, D. P. **1998**. Acoustic ecology of minke whales, , Office of Naval Research contract number N00014-97-1-1005, University of California, Santa Cruz.

Costa, D. P. **2003**. Minke whale (*Balaenoptera acutorostrata*) response to a sound playback experiment, Office of Naval Research contract number N00014-02-0-848, University of California, Santa Cruz.

Costa, D. P., and J. Gedamke. **2000**. Acoustic ecology and remote acoustic monitoring of a minke whale population, , Office of Naval Research contract number N00014-98-1-0859, University of California, Santa Cruz.

Cotté, C., C. Guinet, I. Taupier-Letage, B. Mate, and E. Petiau. **2009**. Scale-dependent habitat use by a large free-ranging predator, the Mediterranean fin whale. Deep Sea Research Part I: Oceanographic Research Papers 56(5):801–811.

Courbis, S., and G. Timmel. **2009**. Effects of vessels and swimmers on behavior of Hawaiian spinner dolphins (*Stenella longirostris*) in Kealake’akua, Honaunau, and Kauhako bays, Hawai’i. Marine Mammal Science 25(2):430-440.

Cowan, D. F., and W. A. Walker. **1979**. Disease factors in *Stenella attenuata* and *Stenella longirostris* taken in the eastern tropical yellowfin purse seine fishery, Southwest Fisheries Center Admin. Rep., Southwest Fisheries Science Center, La Jolla, CA.

Cox, T. M. et al. **2006**. Understanding the impacts of anthropogenic sound on beaked whales. Journal of Cetacean Research and Management 7(3):177–187.

Cox, T. M. et al. **1998**. Documenting the bycatch of harbor porpoises, *Phocoena phocoena*, in coastal gillnet figheries from stranded carcasses. Fishery Bulletin 96:727-734.

Craddock, J. E., P. T. Polloni, B. Hayward, and F. Wenzel. **2009**. Food habits of Atlantic white-sided dolphins (*Lagenorhynchus acutus*) off the coast of New England. Fishery Bulletin 107(3):384-394.

Craig, A. S., and L. M. Herman. **2000**. Habitat preferences of female humpback whales *Megaptera novaeangliae* in the Hawaiian Islands are associated with reproductive status. Mar Ecol Prog Ser 193:209-216.

Craig, B. A., and J. Reynolds III. **2004**. Determination of manatee population trends along the Atlantic coast of Florida using a bayesian approach with temperature-adjusted aerial survey data. Marine Mammal Science 20(3):386-400.

Cranford, T. W., M. F. Mckenna, M. S. Soldevilla, S. M. Wiggins, J. A. Goldbogen, R. E. Shadwick, P. Krysl, J. A. St Leger, and J. A. Hildebrand. **2008a**. Anatomic geometry of sound transmission and reception in Cuvier's beaked whale (*Ziphius cavirostris*). The Anatomical Record: Advances in Integrative Anatomy and Evolutionary Biology 291(4):353–378.

Cranford, T. W., P. Krysl, and J. A. Hildebrand. **2008b**. Sound pathways revealed: simulated sound transmission and reception in Cuvier's beaked whale *Ziphius cavirostris* using the vibro-acoustic toolkit. Bioacoustics 17(1-3):68–70.

Crim, J. L., L. D. Spotila, J. R. Spotila, M. O'connor, R. Reina, C. J. Williams, and F. V. Paladino. **2002**. The leatherback turtle, *Dermochelys coriacea*, exhibits both polyandry and polygyny. Molecular Ecology 11(10):2097–2106.

Croll, D. A., C. W. Clark, J. Calambokidis, W. T. Ellison, and B. R. Tershy. **2001**. Effect of anthropogenic low-frequency noise on the foraging ecology of *Balaenoptera* whales. Animal Conservation 4(1):13–27.

Croll, D. A., C. W. Clark, A. Acevedo, B. Tershy, S. Flores, J. Gedamke, and J. Urban. **2002**. Only male fin whale sing loud songs. Nature 417:809.

Cronin, M., A. Zuur, E. Rogan, and B. McConnell. **2009**. Using mobile phone telemetry to investigate the haul-out behaviour of harbour seals *Phoca vitulina vitulina*. Endangered Species Research 10:255-267.

Culik, B. M., S. Koschinski, N. Tregenza, and G. M. Ellis. **2001**. Reactions of harbor porpoises *Phocoena phocoena* and herring *Clupea harengus* to acoustic alarms. Marine Ecology Progress Series 211:255-260.

Curry, S. S., D. R. Brown, J. M. Gaskin, E. R. Jacobson, L. M. Ehrhart, S. Blahak, L. H. Herbst, and P. A. Klein. **2000**. Persistent infectivity of a disease-associated herpesvirus in green turtles after exposure to seawater. Journal of Wildlife Diseases 36(4):792.

DFO. **2010**. Current status of Northwest Atlantic harp seals, (*Pagophilus groenlandicus*), Canadian Science Advisory Secretariat Science Advisory Report 2009/074.

Dagleish, M. P., G. Foster, F. E. Howie, R. J. Reid, and J. Barley. **2008**. Fatal mycotic encephalitis caused by *Aspergillus fumigatus* in a northern bottlenose whale (*Hyperoodon ampullatus*). Veterinary Record 163(20):602.

Dahlen, M. K., R. Bell, J. I. Richardson, and T. H. Richardson. **2000**. Beyond D-0004: Thirty-four years of loggerhead (*Caretta caretta*) research on Little Cumberland Island, Georgia, 1964-1997, In: Proceedings of the 18th annual symposium on sea turtle biology and conservation, pp. 60-62, NOAA Tech Memo NMFS-SEFSC-436.

Dahlheim, M. E., and J. E. Heyning. **1999**. Killer whale, (*Orcinus orca*), In: Ridgway S. H.. and R. Harrison (eds.) Handbook of Marine Mammals Vol. 6: The Second Book of Dolphins and Porpoises, pp. 281-322, Academic Press, San Diego.

Dalebout, M. L., D. E. Ruzzante, H. Whitehead, and N. I. Oeien. **2006**. Nuclear and mitochondrial markers reveal distinctiveness of a small population of bottlenose whales (*Hyperoodon ampullatus*) in the western North Atlantic. Molecular Ecology 15(11):3115–3129.

Dalebout, M. L., C. S. Baker, J. G. Mead, V. G. Cockcroft, and T. K. Yamada. **2004**. A comprehensive and validated molecular taxonomy of beaked whales, Family Ziphiidae. Journal of Heredity 95(6):459-473.

Dalebout, M. L., S. K. Hooker, and I. Christensen. **2001**. Genetic diversity and population structure among northern bottlenose whales, *Hyperoodon ampullatus*, in the western North Atlantic Ocean. Canadian Journal of Zoology 79(3):478-484.

Dalla Rosa, L. **2010**. Modeling the foraging habitat of humpback whales. Ph.D. thesis. University of British Columbia, Vancouver. 185 p.

D'Amico, A., R. Gisiner, D. R. Ketten, J. A. Hammock, C. Johnson, P. L. Tyack, and J. G. Mead. **2009**. Beaked whale strandings and naval air exercises. Aquatic Mammals 35(4):452-472.

Daoust, P.-Y., D. M. Haines, R. D. Terry, J. Thorsen, P. J. Duignan, and J. R. Geraci. **1993**. Phocine distemper in a harp seal (*Phoca groenlandica*) from the Gulf of St. Lawrence, Canada. Journal of Wildlife Diseases 29(1):114-117.

Darling, J. D. **1983**. Migrations, abundance and behavior of Hawaiian humback whales, *Megaptera novaeangliae* (Borowski). Ph. D. thesis. University of California, Santa Cruz.

Darling, J. D., and M. Bérubé. **2001**. Interactions of singing humpback whales with other males. Marine Mammal Science 17(3):570-584.

Davenport, J., J. Fraher, E. Fitzgerald, P. McLaughlin, T. Doyle, L. Harman, T. Cuffe, and P. Dockery. **2009a**. Ontogenetic changes in tracheal structure facilitate deep dives and cold water foraging in adult leatherback sea turtles. Journal of Experimental Biology 212(21):3440-3447.

Davenport, J., J. Fraher, E. Fitzgerald, P. McLaughlin, T. Doyle, L. Harman, and T. Cuffe. **2009b**. Fat head: an analysis of head and neck insulation in the leatherback turtle (Dermochelys coriacea). Journal of Experimental Biology 212(17):2753-2759.

David, J. A. **2006**. Likely sensitivity of bottlenose dolphins to pile-driving noise. Water and Environment Journal 20(1):48–54.

Davies, J. L. **1957**. The geography of the gray seal. Journal of Mammalogy 38:297-310.

Davis, A. E. **2010**. Potential impacts of ocean energy development on marine mammals in Oregon. College of Oceanic and Atmospheric Sciences, Oregon State University, Corvallis, OR 97331. Available at: http://ir.library.oregonstate.edu/xmlui/bitstream/handle/1957/19586/Amy%20Davis%27%20review%20paper.pdf?sequence=1.

Davis, R. W., N. Jaquet, D. Gendron, U. Markaida, G. Bazzino, and W. Gilly. **2007**. Diving behavior of sperm whales in relation to behavior of a major prey species, the jumbo squid, in the Gulf of California, Mexico. Marine Ecology Progress Series 333:291-302.

Davis, R. W., G. A. J. Worthy, B. Würsig, and S. K. Lynn. **1996**. Diving behavior and at-sea movements of an Atlantic spotted dolphin in the Gulf of Mexico. Marine Mammal Science 12(4):569-581.

Day, R. D. **2003**. Mercury in loggerhead sea turtles, *Caretta caretta*: Developing monitoring strategies, investigating factors affecting contamination, and assessing health impacts. MS thesis. College of Charleston, Charleston, SC. 145 p.

Day, R. D., S. J. Christopher, P. R. Becker, and D. W. Whitaker. **2005**. Monitoring mercury in the loggerhead sea turtle, *Caretta caretta*. Environmental Science & Technology 39(2):437-446.

Day, R. D., J. M. Keller, T. M. Norton, and M. Dodd. **2008**. Mercury and health in loggerhead sea turtles, In: Proceedings of the 25th annual symposium on sea turtle biology and conservation, p. 157, NOAA Tech Memo NMFS-SEFSC-582.

Day, R. D., A. L. Segars, M. D. Arendt, A. M. Lee, and M. M. Peden-Adams. **2007**. Relationship of blood mercury levels to health parameters in the loggerhead sea turtle (*Caretta caretta*). Environmental Health Perspectives 115(10):1421.

Deardorff, T. L., and R. M. Overstreet. **1981**. *Terranova ceticola* n. sp. (Nematoda: Anisakidae) from the dwarf sperm whale; *Kogia simus* (Owen), in the Gulf of Mexico. Systematic Parasitology 3(1):25-28.

Debier, C., P. P. Pomeroy, C. Dupont, C. Joiris, V. Comblin, E. Le Boulengé, Y. Larondelle, and J. P. Thomé. **2003**. Quantitative dynamics of PCB transfer from mother to pup during lactation in UK grey seals *Halichoerus grypus*. Marine Ecology Progress Series 247:237–248.

Deem, S. L., T. M. Norton, M. Mitchell, A. Segars, A. R. Alleman, C. Cray, R. H. Poppenga, M. Dodd, and W. B. Karesh. **2009**. Comparison of blood values in foraging, nesting, and stranded loggerhead turtles (*Caretta caretta*) along the coast of Georgia, USA. Journal of wildlife diseases 45(1):41.

Delarue, J., S. K. Todd, S. M. Van Parijs, and L. Di Iorio. **2009**. Geographic variation in Northwest Atlantic fin whale (*Balaenoptera physalus*) song: Implications for stock structure assessment. The Journal of the Acoustical Society of America 125(3):1774-1782.

Department of the Navy. **2005**. Marine Resource Assessment for the Northeast Operating Areas, Contract Number N62470-02-D-9997, Naval Facilities Engineering Command Atlantic; Norfolk, VA.

De Pierrepont, J. F., B. Dubois, S. Desormonts, M. B. Santos, and J. P. Robin. **2005**. Stomach contents of English Channel cetaceans stranded on the coast of Normandy. Journal of the Marine Biological Association of the United Kingdom 85:1539-1546.

de Swart, R. L., P. S. Ross, J. G. Vos, and A. D. Osterhaus. **1996**. Impaired immunity in harbour seals (*Phoca vitulina*) exposed to bioaccumulated environmental contaminants: review of a long-term feeding study. Environmental Health Perspectives 104(Suppl 4):823-828.

de Swart, R. L., P. S. Ross, H. H. Timmerman, H. W. Vos, P. J. H. Reijnders, J. G. Vos, and A. D. M. E. Osterhaus. **1995**. Impaired cellular immune response in harbour seals (Phoca vitulina) feeding on environmentally contaminated herring. Clinical and Experimantal Immunology 101:480-486.

Deutsch, C. J., J. P. Reid, R. K. Bonde, D. E. Easton, H. I. Kochman, and T. J. O’Shea. **2003**. Seasonal movements, migratory behavior, and site fidelity of West Indian manatees along the Atlantic Coast of the United States. Wildlife Monographs (151):1-77.

Diez, C. E., and R. P. van Dam. **2002**. Habitat effect on hawksbill turtle growth rates on feeding grounds at Mona and Monito Islands, Puerto Rico. Marine Ecology Progress Series 234:301–309.

Di Guardo, G. et al. **2010**. Cerebral Toxoplasmosis in striped dolphins (*Stenella coeruleoalba*) stranded along the Ligurian sea coast of Italy. Veterinary Pathology 47(2):245-253.

Di Iorio, L., and C. W. Clark. **2009**. Exposure to seismic survey alters blue whale acoustic communication. Biology Letters 6(1):51-54.

Dodd Jr, C. K. **1988**. Synopsis of the biological data on the loggerhead sea turtle *Caretta caretta* (Linnaeus 1758)., U.S. Fish and Wildlife Service Biological Report 88(10). 110 p.

Doksaeter, L., E. Olsen, L. Nottestad, and A. Ferno. **2008**. Distribution and feeding ecology of dolphins along the Mid-Atlantic Ridge between Iceland and the Azores. Deep Sea Research Part II: Topical Studies in Oceanography 55(1-2):243-253.

Dolar, M. L. L., W. A. Walker, G. L. Kooyman, and W. F. Perrin. **2003**. Comparative feeding ecology of spinner dolphins (*Stenella longirostris*) and Fraser's dolphins (*Lagenodelphis hosei*) in the Sulu Sea. Marine Mammal Science 19:1-19.

Dolar, M. L. L. **2009**. Fraser's dolphin, In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, pp. 469-471, Academic Press, San Diego.

Dolman, S. M., and M. Simmonds. **2010**. Towards best environmental practice for cetacean conservation in developing Scotland's marine renewable energy. Marine Policy 34:1021-1027.

Domingo, M., J. Visa, M. Pumarola, A. J. Marco, L. Ferrer, R. Rabanal, and S. Kennedy. **1992**. Pathologic and immunocytochemical studies of morbillivirus infection in striped dolphins (*Stenella coeruleoalba*). Veterinary Pathology 29(1):1-10.

Domingo, M., M. Vilafranca, J. Visa, N. Prats, A. Trudgett, and I. Visser. **1995**. Evidence for chronic morbillivirus infection in the Mediterranean striped dolphin (*Stenella coeruleoalba*). Veterinary Microbiology 44:229-239.

Donahue, M. A., and W. L. Perryman. **2009**. Pygmy killer whale, *Feresa attenuata*, In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, pp. 938-939, Academic Press.

Doniol-Valcroze, T., D. Berteaux, P. Larouche, and R. Sears. **2007**. Influence of thermal fronts on habitat selection by four rorqual whale species in the Gulf of St. Lawrence. Marine Ecology Progress Series 335:207–216.

Donovan, G. P. **1991**. A review of IWC stock boundaries. Report of the International Whaling Commission (Special Issue 13):39-68.

Dorsey, E. M. **1983**. Exclusive adjoining ranges in individually identified minke whales (*Balaenoptera acutorostrata*) in Washington state. Canadian Journal of Zoology 61(1):174–181.

Dorsey, E. M., S. J. Stern, A. R. Hoelzel, and J. Jacobsen. **1990**. Minke whale (*Balaenoptera acutorostrata*) from the west coast of North America: individual recognition and small-scale site fidelity. Rept. Int. Whal. Commn., Special 357–368.

Doucette, G. J., A. D. Cembella, J. L. Martin, J. Michaud, T. V. N. Cole, and R. M. Rolland. **2006**. Paralytic shellfish poisoning (PSP) toxins in North Atlantic right whales *Eubalaena glacialis* and their zooplankton prey in the Bay of Fundy, Canada. Marine Ecology Progress Series 306:303–313.

Drouot, V. **2003**. Ecology of sperm whale (*Physeter macrocephalus*) in the Mediterranean Sea. Ph.D. thesis. University of Wales, Bangor.

Drouot, V., A. Gannier, and J. C. Goold. **2004**. Diving and feeding behaviour of sperm whales (*Physeter macrocephalus*) in the northwestern Mediterranean Sea. Aquatic Mammals 30(3):419–426.

D'Spain, G. L., A. D'Amico, and D. M. Fromm. **2005**. Properties of the underwater sound fields during some well-documented beaked whale mass stranding events. Journal of Cetacean Research and Management 7(3):223.

Dufault, S., H. Whitehead, and M. Dillon. **1999**. An examination of the current knowledge on the stock structure of sperm whales (*Physeter macrocephalus*) worldwide. Journal of Cetacean Research and Management 1(1):1–10.

Duignan, P. J. et al. **1995a**. Morbillibirus infection in two species of pilot whale (*Globicephala* sp.) from the western Atlantic. Marine Mammal Science 11(2):150-162.

Duignan, P. J., O. Nielsen, C. House, K. M. Kovacs, N. Duffy, G. A. Early, S. Sadove, D. J. St. Aubin, B. K. Rima, and J. R. Geraci. **1997**. Epizootiology of morbillivirus infection in harp, hooded, and ringed seals from the Canadian arctic and western Atlantic. Journal of Wildlife Diseases 33(1):7-19.

Duignan, P. J., J. Saliki, D. J. St. Aubin, G. Early, S. Sadove, J. A. House, K. Kovacs, and J. R. Geraci. **1995b**. Epizootiology of morbillivirus infection in North American harbor seals (*Phoca vitulina*) and gray seals (*Halichoerus grypus*). Journal of Wildlife Diseases 31(4):491-501.

Dunlop, R. A., D. H. Cato, and M. J. Noad. **2008**. Non-song acoustic communication in migrating humpback whales (*Megaptera novaeangliae*). Marine Mammal Science 24(3):613–629.

Dunlop, R. A., D. H. Cato, and M. J. Noad. **2010**. Your attention please: increasing ambient noise levels elicits a change in communication behaviour in humpback whales (*Megaptera novaeangliae*). Proceedings of the Royal Society B: Biological Sciences 277(1693):2521.

Dunlop, R. A., M. J. Noad, D. H. Cato, and D. Stokes. **2007**. The social vocalization repertoire of east Australian migrating humpback whales (*Megaptera novaeangliae*). The Journal of the Acoustical Society of America 122:2893.

Dunphy-Daly, M. M., M. R. Heithaus, and D. E. Claridge. **2008**. Temporal variation in dwarf sperm whale (*Kogia sima*) habitat use and group size off Great Abaco Island, Bahamas. Marine Mammal Science 24(1):171-182.

Durbin, E., G. Teegarden, R. Campbell, A. Cembella, M. F. Baumgartner, and B. R. Mate. **2002**. North Atlantic right whales, *Eubalaena glacialis*, exposed to paralytic shellfish poisoning (PSP) toxins via a zooplankton vector, *Calanus finmarchicus*. Harmful Algae 1(3):243–251.

Duron-Dufrenne, M. **1987**. First satellite-based tracking in the Atlantic Ocean of a leatherback turtle *Dermochelys coriacea*. Comptes Rendue Academie des Sciences Paris 304:399-402.

Dutton, D. L., P. H. Dutton, R. Boulon, and T. Wibbels. **2000**. Recruitment and mortality estimates for female leatherbacks nesting in St. Croix, U. S. Virgin Islands. NOAA Tech Memo NMFS-SEFSC-443 268.

Dutton, P. H., B. W. Bowen, D. W. Owens, A. Barragan, and S. K. Davis. **1999**. Global phylogeography of the leatherback turtle (*Dermochelys coriacea*). Journal of Zoology 248:397-409.

Dutton, P., D. McDonald, J. I. Richardson, and T. H. Richardson. **1995**. Hatch rates of leatherback (*Dermochelys coriacea*) clutches reburied with and without yolkless eggs. NOAA Tech Memo NMFS-SEFSC-361 177 p.

Dutton, P., V. Pease, and D. Shaver. **2006**. Characterization of MtDNA variation among Kemp's ridleys nesting on Padre Island with reference to Rancho Nuevo genetic stock, In: Proceedings of the 26th annual symposium on sea turtle biology and conservation, p. 189, International Sea Turtle Society, Athens, Greece.

Eaton, C., E. McMichael, B. Witherington, A. Foley, R. Hardy, and A. Meylan. **2008**. In-water sea turtle monitoring and research in Florida: review and recommendations, NOAA Tech Memo, NMFS-OPR-38.

Eckert, K. L. **2001**. Status and distribution of the leatherback turtle, *Dermochelys coriacea*, in the Wider Caribbean Region, In: Proceedings of the regional meeting:Marine turtle conservation in the Wider Caribbean Region: A dialogue for effective regional management, pp. 24-31, WIDECAST, IUCN-MTSG, WFF and UNEP-CEP.

Eckert, K. L. **1987**. Environmental unpredictability and leatherback sea turtle (*Dermochelys coriacea*) nest loss. Herpetologica 43(3):315-323.

Eckert, S. A. **2002**. Distribution of juvenile leatherback sea turtle *Dermochelys coriacea* sightings. Marine Ecology Progress Series 230:289–293.

Eckert, S. A., D. Bagley, S. Kubis, L. Ehrhart, C. Johnson, K. Stewart, and D. DeFreese. **2006**. Internesting and postnesting movements and foraging habitats of leatherback sea turtles (*Dermochelys coriacea*) nesting in Florida. Chelonian Conservation and Biology 5(2):239-248.

Eckert, S. A. **2006**. High-use oceanic areas for Atlantic leatherback sea turtles (*Dermochelys coriacea*) as identified using satellite telemetered location and dive information. Marine Biology 149(5):1257-1267.

Edds, P. L. **1988**. Characteristics of finback *Balaenoptera physalus* vocalizations in the St. Lawrence Estuary. Bioacoustics 1(2/3):131–149.

Edds, P. L., and J. A. F. Macfarlane. **1987**. Occurrence and general behavior of balaenopterid cetaceans summering in the St. Lawrence Estuary, Canada. Canadian Journal of Zoology 65(6):1363-1376.

Edds-Walton, P. L. **2000**. Vocalizations of minke whales *Balaenoptera acutorostrata* in the St. Lawrence Estuary. Bioacoustics 11(1):31–50.

Edmonds, J. S., Y. Shibata, R. I. T. Prince, K. A. Francesconi, and M. Morita. **1994**. Arsenic compounds in tissues of the leatherback turtle, *Dermochelys coriacea*. Journal of the Marine Biological Association of the United Kingdom 74:463-466.

Edrén, S. M. C., S. M. Andersen, J. Teilmann, J. Carstensen, P. B. Harders, R. Dietz, and L. A. Miller. **2010**. The effect of a large Danish offshore wind farm on harbor and gray seal haul-out behavior. Marine Mammal Science 26(3):614-634.

Edwards, H. H., K. H. Pollock, B. B. Ackerman, J. E. Reynolds, and J. A. Powell. **2007**. Estimation of detection probability in manatee aerial surveys at a winter aggregation site. Journal of Wildlife Management (6):2052-2060.

Ehrhart, L. M., W. E. Redfoot, and D. A. Bagley. **2007**. Marine turtles of the central region of the Indian River Lagoon system, Florida. Florida Scientist 70(4):415-434.

Elfes, C. T. **2008**. Persistent organic pollutant levels in North Pacific and North Atlantic humpback whales (*Megaptera novaeangliae*). MS thesis. University of Washington. 98 p.

Elfes, C. T. et al. **2010**. Geographic variation of persistent organic pollutant levels in humpback whale (*Megaptera novaeangliae*) feeding areas of the North Pacific and North Atlantic. Environmental Toxicology and Chemistry 29(4):824-834.

Engel, M. H., M. C. C. Marcondes, C. C. A. Martins, F. O. Luna, R. P. Lima, and A. Campos. **2004**. Are seismic surveys responsible for cetacean strandings? An unusual mortality of adult humpback whales in Abrolhos Bank, northeastern coast of Brazil, IWC SC/56/E28.

Engelhaupt, D. T. **2004**. Phylogeography, kinship and molecular ecology of sperm whales (*Physeter macrocephalus*). Ph.D. thesis. University of Durham.

Engelhaupt, D. et al. **2009**. Female philopatry in coastal basins and male dispersion across the North Atlantic in a highly mobile marine species, the sperm whale (*Physeter macrocephalus*). Molecular Ecology 18(20):4193–4205.

Epperly, S., J. Braun-McNeill, and P. Richards. **2007**. Trends in catch rates of sea turtles in North Carolina, USA. Endangered Species Research 3:283-293.

Erbe, C. **2002**. Underwater noise of whale-watching boats and potential effects on killer whales (*Orcinus orca*), based on an acoustic impact model. Marine Mammal Science 18(2):394–418.

Ernest, R. G., R. E. Martin, N. William-Walls, and J. R. Wilcox. **1989**. Population dynamics of sea turtles utilizing shallow waters off Hutchinson Island, Florida, In: Proceedings of the 9th annual workshop on sea turtle conservation and biology, pp. 57-59, NOAA Tech Memo NMFS SEFSC- 232.

Ersts, P. J., and H. C. Rosenbaum. **2003**. Habitat preference reflects social organization of humpback whales (Megaptera novaeangliae) on a wintering ground. Journal of Zoology 260(4):337–345.

Estep, J. S., R. E. Baumgartner, F. Townsend, D. A. Pabst, W. A. McLellan, A. Friedlaender, D. G. Dunn, and T. P. Lipscomb. **2005**. Malignant seminoma with metastasis, sertoli cell tumor, and pheochromocytoma in a spotted dolphin (*Stenella frontalis*) and malignant seminoma with metastasis in a bottlenose dolphin (*Tursiops truncatus*). Veterinary Pathology 42:357-359.

Estes, J., T. Wibbels, T. Tucker, J. Wyneken, L. M. Ehrhart, R. R. Carthy, R. E. Martin, R. Ernest, M. Bresette, C. Johnson, B. Ball, J. Schmid, J. Phillips, S. Dawsey, B. Drye, and K. Watson. **2010**. Evaluation of loggerhead nesting beach temperatures throughout the southeastern Unites States, In: Proceedings of the 28th annual symposium on sea turtle biology and conservation, p. 81, NOAA Tech Memo NMFS-SEFSC-602.

Evans, P. G. H. **2008**. Proceedings of the ASCOBANS/ECS workshop offshore wind farms and marine mammals: impacts & methodologies for assessing impacts, vol. ECS special publication series no. 49, The Aquarium, San Sebastian, Spain.

Evans, P. G. H. **1997**. Ecology of sperm whales (*Physeter macrocephalus*) in the eastern North Atlantic, with special reference to sightings and strandings records from the British Isles. Bulletin van het Koninklijk Belgisch Instituut voor Natuurwetenschappen. Biologie= Bulletin de l'Institut Royal des Sciences Naturelles de Belgique. Biologie.

Evans, P. **1980**. Associations between seabirds and cetaceans: a review. Mammal Review 12:187-206.

Evans, W. E. **1994**. Common Dolphin, White-bellied Porpoise, In: Ridgway, S.H. and R. Harrison (eds.) Handbook of Marine Mammals. Vol. 5: The First Book of Dolphins, pp. 191-224, Academic Press, New York.

Ewing, R., J. Zaias, M. A. Stamper, G. D. Bossart, and J. P. Dubey. **2002**. Prevalence of Sarcocystis sp. in stranded Atlantic white-sided dolphins (*Lagenorhynchus acutus*). Journal of Wildlife Diseases 38(2):291-295.

Fair, P. A. et al. **2010**. Contaminant blubber burdens in Atlantic bottlenose dolphins (*Tursiops truncatus*) from two southeastern US estuarine areas: Concentrations and patterns of PCBs, pesticides, PBDEs, PFCs, and PAHs. Science of the Total Environment 408:1577-1597.

Fair, P. A., G. Mitchum, T. C. Hulsey, J. Adams, E. Zolman, W. McFee, E. Wirth, and G. D. Bossart. **2007**. Polybrominated diphenyl ethers (PBDEs) in blubber of free-ranging bottlenose dolphins (*Tursiops truncatus*) from two southeast Atlantic estuarine areas. Archives of Environmental Contamination and Toxicology 53:483-494.

Fairfield Walsh, C., and L. P. Garrison. **2006**. Estimated bycatch of marine mammals and turtles in the U.S. Atlantic pelagic longline fleet during 2005. NOAA Tech Memo NMFS-SEFSC-539 52.

Fairfield Walsh, C., and L. P. Garrison. **2007**. Estimated bycatch of marine mammals and turtles in the U.S. Atlantic pelagic longline fleet during 2006. NOAA Tech Memo NMFS-SEFSC-560 58.

Fairfield, C. P., G. T. Waring, and M. H. Sano. **1993**. Pilot whales incidentally taken during the distant water fleet Altantic mackeral fishery in the Mid-Atlantic Bight, 1984-1988, Biology of Northern Hemisphere Pilot Whales, Report of the International Whaling Commission, International Whaling Commission, Cambridge.

Fairfield, C. P., and L. P. Garrison. **2008**. Estimated bycatch of marine mammals and sea turtles in the US Atlantic pelagic longline fleet during 2007. NOAA Tech Memo NMFS-SEFSC-572 63.

Farro, A. P. C., M. M. Rollo, J. M. Silva, and C. L. Marino. **2008**. Isolation and characterization of microsatellite DNA markers for spinner dolphin (*Stenella longirostris*). Conservation Genetics 9:1319-1321.

Fauquier, D. A., M. J. Kinsel, M. D. Dailey, G. E. Sutton, M. K. Stolen, R. S. Wells, and F. M. D. Gulland. **2009**. Prevalence and pathology of lungworm infection in bottlenose dolphins dolphins *Tursiops truncatus* from southwest Florida. Diseases of Aquatic Organisms 88:85-90.

Ferguson, M. C., J. Barlow, S. B. Reilly, and T. Gerrodette. **2006**. Predicting Cuvier's (*Ziphius cavirostris*) and *Mesoplodon* beaked whale population density from habitat characteristics in the eastern tropical Pacific Ocean. Journal of Cetacean Research and Management 7(3):287-299.

Ferland, A. **1999**. Feeding habits of harbor seals (*Phoca vitulina concolor*) along the coast of Cape Cod, Massachusetts and the northern Gulf of Maine. MA thesis. Boston University, Boston, MA. 46 p.

Fernandez, A., J. F. Edwards, F. Rodriguez, A. E. de los Monteros, P. Herraez, P. Castro, J. R. Jaber, V. Martin, and M. Arbelo. **2005**. "Gas and Fat Embolic Syndrome" involving a mass stranding of beaked whales (Family Ziphiidae) exposed to anthropogenic sonar signals. Veterinary Pathology 42(4):446-457.

Ferraroli, S., J. Y. Georges, P. Gaspar, and Y. L. Maho **2004**. Where leatherback turtles meet fisheries. Nature 429:521–522.

Fertl, D., A. J. Schiro, G. T. Regan, C. A. Beck, N. Adimey, L. Price-May, A. Amos, G. A. J. Worthy, and R. Crossland. **2005**. Manatee occurrence in the northern Gulf of Mexico, west of Florida. Gulf and Caribbean Research 17:69–94.

Fertl, D., and S. Leatherwood. **1997**. Cetacean interactions with trawls: A preliminary review. Journal of the Northwest Atlantic Fisheries Society 22:219-248.

Fertl, D., and B. Würsig. **1995**. Coordinated feeding by Atlantic spotted dolphins (*Stenella frontalis*) in the Gulf of Mexico. Aquatic Mammals 31:3-5.

Fertl, D., T. A. Jefferson, I. B. Moreno, A. N. Zerbini, and K. D. Mullin. **2003**. Distribution of the Clymene dolphin *Stenella clymene*. Mammal Review 33(3):253-271.

Finneran, J. J., D. A. Carder, C. E. Schlundt, and S. H. Ridgway. **2005**. Temporary threshold shift in bottlenose dolphins (*Tursiops truncatus*) exposed to mid-frequency tones. Journal of the Acoustical Society of America 118(4):2696-2705.

Finneran, J. J., D. Carder, C. E. Schlundt, and R. Dear. **2010**. Growth and recovery of temporary threshold shift at 3 kHz in bottlenose dolphins: experimental data and mathematical models. Journal of the Acoustical Society of America 127(5):3256-3266.

Finneran, J. J., C. E. Schlundt, D. A. Carder, J. A. Clark, J. A. Young, J. B. Gaspin, and S. H. Ridgway. **2000**. Auditory and behavioral responses of bottlenose dolphins (*Tursiops truncatus*) and a beluga whale (*Delphinapterus leucas*) to impulsive sounds resembling distant signatures of underwater explosions. Journal of the Acoustical Society of America 108(1):417-431.

Fire, S. E. et al. **2009**. Domoic acid exposure in pygmy and dwarf sperm whales (*Kogia* spp.) from southeastern and mid-Atlantic US waters. Harmful Algae 8(5):658–664.

Firestone, J., S. B. Lyons, C. Wang, and J. J. Corbett. **2008**. Statistical modeling of North Atlantic right whale migration along the mid-Atlantic region of the eastern seaboard of the United States. Biological Conservation 141(1):221–232.

Flewelling, L. J. et al. **2005**. Brevetoxicosis: Red tides and marine mammal mortalities. Nature 435(7043):755–756.

Florida Fish and Wildlife Conservation Commission **n.d.** Florida marine resources GIS. http://ocean.floridamarine.org/mrgis/. Accessed 24 August 2010.

Foley, A. M., B. A. Schroeder, A. E. Redlow, K. J. Fick-Child, and W. G. Teas. **2005**. Fibropapillomatosis in stranded green turtles (*Chelonia mydas*) from the eastern United States (1980-98): trends and associations with environmental factors. Journal of Wildlife Diseases 41(1):29-41.

Folkow, L. P., and A. S. Blix. **1999**. Diving behaviour of hooded seals (*Cystophora cristata*) in the Greenland and Norwegian Seas. Polar Biology 22(1):61-74.

Fonnesbeck, C. J., L. P. Garrison, L. I. Ward-Geiger, and R. D. Baumstark. **2008**. Bayesian hierarchichal model for evaluating the risk of vessel strikes on North Atlantic right whales in the SE United States. Endangered Species Research 6:87-94.

Foote, A. D., J. Newton, S. B. Piertney, E. Willerslev, and M. T. P. Gilbert. **2009**. Ecological, morphological and genetic divergence of sympatric North Atlantic killer whale populations. Molecular Ecology 18(24):5207–5217.

Foote, A. D. et al. **2010**. Genetic differentiation among North Atlantic killer whale populations. Molecular Ecology 20(3):629-641.

Foote, A. D. **2008**. Mortality rate acceleration and post-reproductive lifespan in matrilineal whale species. Biology Letters 4(2):189 -191.

Ford, J. K. B. **2009**. Killer whale *Orcinus orca*, In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, pp. 650-657, Academic Press.

Ford, J. K. B., G. M. Ellis, P. Olesiuk, and K. C. Balcomb. **2010**. Linking killer whale survival and prey abundance: food limitation in the oceans' apex predator? Biology Letters 6(1):139-142.

Forrester, D., D. K. Odell, N. Thompson, and J. White **1980**. Morphometrics, parasites, and cholorinated hydrocarbon residues of pygmy killer whales from Florida. Journal of Mammalogy 61(2):356-360.

Fossette, S., V. J. Hobson, C. Girard, B. Calmettes, P. Gaspar, J. Y. Georges, and G. C. Hays. **2010**. Spatio-temporal foraging patterns of a giant zooplanktivore, the leatherback turtle. Journal of Marine Systems 81(3):225–234.

Frankel, A. S., C. W. Clark, L. M. Herman, and C. M. Gabriele. **1995**. Spatial distribution, habitat utilization, and social interactions of humpback whales, *Megaptera novaeangliae*, off Hawai'i, determined using acoustic and visual techniques. Canadian Journal of Zoology 73(6):1134–1146.

Frankel, A., and S. Yin. **2010**. A description of sounds recorded from melon-headed whales (*Peponocephala electra*) off Hawai'i. Journal of the Acoustical Society of America 127(5):3248-3255.

Frasier, T. R., P. K. Hamilton, M. W. Brown, L. A. Conger, A. R. Knowlton, M. K. Marx, C. K. Slay, S. D. Kraus, and B. N. White. **2007**. Patterns of male reproductive success in a highly promiscuous whale species: the endangered North Atlantic right whale. Molecular Ecology 16(24):5277–5293.

Frazer, L. N., and E. Mercado. **2000**. A sonar model for humpback whale song. IEEE Journal of Oceanic Engineering 25(1):160-182.

Frick, M., K. Williams, A. Bolten, K. Bjorndal, and H. Martins. **2009**. Foraging ecology of oceanic-stage loggerhead turtles *Caretta caretta*. Endangered Species Research 9:91-97.

Friedlaender, A. S., D. W. Johnston, and P. N. Halpin. **2010**. Effects of the North Atlantic Oscillation on sea ice breeding habitats of harp seals (*Pagophilus groenlandicus*) across the North Atlantic. Progress In Oceanography 86(1-2):261-266.

Friedlaender, A., E. Hazen, D. Nowacek, P. Halpin, C. Ware, M. Weinrich, T. Hurst, and D. Wiley. **2009**. Diel changes in humpback whale *Megaptera novaeangliae* feeding behavior in response to sand lance *Ammodytes* spp. behavior and distribution. Marine Ecology Progress Series 395:91-100.

Fristrup, K. M., L. T. Hatch, and C. W. Clark. **2003**. Variation in humpback whale (*Megaptera novaeangliae*) song length in relation to low-frequency sound broadcasts. The Journal of the Acoustical Society of America 113:3411-3424.

Fristrup, K. M., and G. R. Harbison. **2002**. How do sperm whales catch squids? Marine Mammal Science 18(1):42-54.

Frodello, J. P., D. Viale, and B. Marchand. **2002**. Metal levels in a Cuvier's beaked whale (*Ziphius cavirostris*) found stranded on a Mediterranean Coast, Corsica. Bulletin of Environmental Contamination and Toxicology 69:662-666.

Fuentes, M., M. Hamann, and C. J. Limpus. **2009**. Past, current and future thermal profiles of green turtle nesting grounds: Implications from climate change. Journal of Experimental Marine Biology and Ecology 383(1):56-64.

Fujiwara, M., and H. Caswell. **2001**. Demography of the endangered North Atlantic right whale. Nature 414(6863):537-541.

Fullard, K. J., G. Early, M. P. Heide-Jørgensen, D. Bloch, A. Rosing-Asvid, and W. Amos. **2000**. Population structure of long-finned pilot whales in the North Atlantic: a correlation with sea surface temperature? Molecular Ecology 9(7):949-958.

Gaggiotti, O., F. Jones, W. M. Lee, W. Amos, J. Harwood, and R. A. Nichols. **2002**. Patterns of colonization in a metapopulation of grey seals. Nature 416:424-427.

Gaines, C. A., M. P. Hare, S. E. Beck, and H. C. Rosenbaum. **2005**. Nuclear markers confirm taxonomic status and relationships among highly endangered and closely related right whale species. Proceedings of the Royal Society B: Biological Sciences 272(1562):533-542.

Gallo-Reynoso, J. P., J. Égido-Villarreal, and E. M. Coria-Galindo. **2009**. Sperm whale distribution and diving behaviour in relation to presence of jumbo squid in Guaymas Basin, Mexico. Marine Biodiversity Records 2:e139.

Gannier, A., and E. Praca. **2007**. SST fronts and the summer sperm whale distribution in the north-west Mediterranean Sea. Journal of the Marine Biological Association of the United Kingdom 87(01):187–193.

Gannier, A. **2011**. Using existing data and focused surveys to highlight Cuvier's beaked whales favourable areas: A case study in the central Tyrrhenian Sea. Marine Pollution Bulletin 63(1-4):10-17.

Gannon, D. P., A. J. Read, J. E. Craddock, and J. G. Mead. **1997a**. Stomach contents of long-finned pilot whales (*Globicephala melas*) stranded on the US mid-Atlantic coast. Marine Mammal Science 13(3):405–418.

Gannon, D. P., and D. M. Waples. **2004**. Diets of coastal bottlenose dolphins from the US mid-Atlantic coast differ by habitat. Marine Mammal Science 20(3):527–545.

Gannon, D. P., J. E. Craddock, and A. J. Read. **1998**. Autumn food habits of harbor porpoises, *Phocoena phocoena*, in the Gulf of Maine. Fishery Bulletin 96(3):428-437.

Gannon, D. P., A. J. Read, J. E. Craddock, K. M. Fristrup, and J. R. Nicolas. **1997b**. Feeding ecology of long-finned pilot whales *Globicephala melas* in the western North Atlantic. Marine Ecology Progress Series 148(1):1-10.

Gannon, J. G., K. M. Scolardi, J. E. Reynolds III, J. K. Koelsch, and T. J. Kessenich. **2007**. Habitat selection by manatees in Sarasota Bay, Florida. Marine Mammal Science 23(1):133–143.

Garcia-Rodriguez, A. I., B. W. Bowen, D. Domning, A. A. Mignucci-Giannoni, M. Marmontel, R. A. Montoya-Ospina, B. Morales-Vela, M. Rudin, R. K. Bonde, and P. M. McGuire. **1998**. Phylogeography of the West Indian manatee (*Trichechus manatus*): how many populations and how many taxa? Molecular Ecology 7(9):1137-1149.

Gardner, B., P. J. Sullivan, S. Epperly, and S. J. Morreale. **2008**. Hierarchical modeling of bycatch rates of sea turtles in the western North Atlantic. Endangered Species Research 5:279-289.

Garrison, L. **2007a**. Estimated marine mammal and turtle bycatch in shark gillnet fisheries along the southeast U.S. Atlantic coast: 2000-2006. SEFSC PRD Contribution #PRD--7/08-02.

Garrison, L. P. **2007b**. Defining the North Atlantic right whale calving habitat in the southeastern United States: An application of a habitat model. NOAA Tech Memo NMFS-SEFSC-553 63 p.

Garrison, L. P. **2007c**. Interactions between marine mammals and pelagic longline fishing gear in the U.S. Atlantic Ocean between 1992 and 2004. Fishery Bulletin 105(3):408-417.

Garrison, L. P., P. E. Rosel, A. A. Hohn, R. Baird, and W. Hoggard. **2003**. Abundance of the coastal morphotype of bottlenose dolphin *Tursiops truncatus*, in U.S. continental shelf waters between New Jersey and Florida during winter and summer 2002., NMFS/SEFSC report prepared and reviewed for the Bottlenose Dolphin Take Reduction Team, NMFS, Southeast Fisheries Science Center.

Garrison, L. P., L. Stokes, and C. Fairfield. **2009**. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2008. NOAA Tech Memo NMFS-SEFSC-591. 63p.

Garrison, L. P., and P. M. Richards. **2004**. Estimated bycatch of marine mammals and turtles in the U.S. Atlantic pelagic longline fleet during 2003. NOAA Tech Memo NMFS-SEFSC-527. 57 p.

Garrison, L. P., and L. Stokes .**2010**. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2009. NOAA Tech Memo NMFS-SEFSC-607. 63 p.

Gaspari, S., S. Airoldi, and A. R. Hoelzel. **2007**. Risso's dolphins (*Grampus griseus*) in UK waters are differentiated from a population in the Mediterranean Sea and genetically less diverse. Conservation Genetics 8(3):727-732.

Gasparini, J. L., and I. Sazima. **1996**. A stranded melon-headed whale, *Peponocephala electra*, in southeastern Brazil, with comments on wounds from the cookiecutter shark, Isistius brasiliensis. Marine Mammal Science 12(2):308-312.

Gauthier, J. M., C. D. Metcalfe, and R. Sears. **1997**. Chlorinated organic contaminants in blubber biopsies from northwestern Atlantic balaenopterid whales summering in the Gulf of St Lawrence. Marine Environmental Research 44(2):201-223.

Gaydos, J. K., K. C. I. Balcomb, R. W. Osborne, and L. Dierauf. **2003**. Evaluating potential infectious disease threats for southern resident killer whales, *Orcinus orca*: a model for endangered species. Biological Conservation 117(3):253-262.

Gazda, S. K., R. C. Connor, R. K. Edgar, and F. Cox. **2005**. A division of labour with role specialization in group–hunting bottlenose dolphins (*Tursiops truncatus*) off Cedar Key, Florida. Proceeding of the Royal Society B 272:135-140.

Gedamke, J. **2004**. Minke whale song, spacing, and acoustic communication on the Great Barrier Reef, Australia. Ph.D. thesis. University of California, Santa Cruz.

Georges, J.-Y., S. Fossette, A. Billes, S. Ferraroli, J. Fretey, D. Gremillet, Y. Le Maho, A. E. Myers, H. Tanaka, and G. C. Hays. **2007**. Meta-analysis of movements in Atlantic leatherback turtles during the nesting season: conservation implications. Marine Ecology Progress Series 338:225-232.

Georges, J.-Y., and S. Fossette. **2006**. Estimating body mass in leatherback turtles *Dermochelys coriacea*. Marine Ecology Progress Series 318:255-262.

Georgia DNR Sea Turtle Conservation Program. **n.d.** Sea Turtle Nest Monitoring System. http://www.seaturtle.org/nestdb/index.shtml?view=3&year=2010. Accessed 26 August 2010.

Geraci, J. R., and D. J. St. Aubin. **1980**. Offshore petroleum resource development and marine mammals: a review and research recommendations. Marine Fisheries Review 42:1-12.

Geraci, J. R., D. M. Anderson, R. J. Timperi, D. J. S. Aubin, G. A. Early, J. H. Prescott, and C. A. Mayo. **1989**. Humpback whales (*Megaptera novaeangliae*) fatally poisoned by dinoflagellate toxins. Canadian Journal of Fisheries and Aquatic Sciences 46:1895-1898.

Gero, S., J. Gordon, C. Carlson, P. Evans, and H. Whitehead. **2007**. Population estimate and inter-island movement of sperm whales, *Physeter macrocephalus*, in the Eastern Caribbean Sea. Journal of Cetacean Research and Management 9(2):143-150.

Gero, S., and H. Whitehead. **2007**. Suckling behavior in sperm whale calves: Observations and hypotheses. Marine Mammal Science 23(2):398-413.

Gerstein, E. **2002**. Manatees, bioacoustics and boats: Hearing tests, environmental measurements and acoustic phenomena may together explain why boats and animals collide. American Scientist 90(2):154-163.

Gerstein, E. R., L. Gerstein, S. E. Forsythe, and J. E. Blue. **1999**. The underwater audiogram of the West Indian manatee (*Trichechus manatus*). The Journal of the Acoustical Society of America 105:3575-3583.

Gerstein, E. R., and L. A. Gerstein. **2010**. Manatee hearing and communication shaped by boundaries of their shallow water habitats. The Journal of the Acoustical Society of America 128:2468-2468.

Gerstein, E., L. Gerstein, J. Blue, and S. Forsythe. **2008**. Ultrasonic hearing and vocalizations are used in communication by West Indian manatee mothers and calves. Journal of the Acoustical Society of America 124(4):2549-2549.

Gilbert, E. I., L. M. Ehrhart, E. V. Valdes, and L. J. Walters. **2008**. Juvenile green turtle (*Chelonia mydas*) foraging ecology: feeding selectivity and forage nutrient analysis. Proceedings of the 25th annual symposium on sea turtle biology and conservation NOAA Tech Memo NMFS-SEFSC-582. 89 p.

Gilbert, J. R., G. T. Waring, K. M. Wynne, and N. Guldager. **2005**. Changes in abundance of harbor seals in Maine, 1981-2001. Marine Mammal Science 21(3):519-535.

Gill, A., and R. S. Fairbairns. **1995**. Photo-identification of the minke whale *Balaenoptera acutorostrata* off the Isle of Mull, Scotland, In: Whales, seals, fish and man, Proceedings of the International Symposium on the Biology of Marine Mammals in the North East Atlantic , 29 November-1 December 1994, vol. Volume 4, pp. 129-132, Elsevier Science.

Gilles, A., S. Adler, K. Kaschner, M. Scheidat, and U. Siebert. **2011**. Modelling harbour porpoise seasonal density as a function of the German Bight environment: implications for management. Endangered Species Research 14(2):157-169.

Gilman, E. et al. **2010**. Mitigating sea turtle by-catch in coastal passive net fisheries. Fish and Fisheries 11(1):57-88.

Gilman, E., E. Zollett, S. Beverly, H. Nakano, K. Davis, D. Shiode, P. Dalzell, and I. Kinan. **2006**. Reducing by-catch of sea turtles in pelagic longline fisheries. Fish and Fisheries 7:2-23.

Girondot, M., and J. Fretey. **1996**. Leatherback turtles, *Dermochelys coriacea*, nesting in French Guiana, 1978-1995. Chelonian Conservation and Biology 2(2):204–208.

Gitschlag, G. R. **1996**. Migration and diving behavior of Kemp's ridley (Garman) sea turtles along the US southeastern Atlantic coast. Journal of Experimental Marine Biology and Ecology 205(1-2):115–135.

Glass, A., T. V. N. Cole, and M. Garron. **2010**. Mortality and serious injury determinations for baleen whale stocks along the United States eastern seaboard and adjacent Canadian Maritimes, 2004-2008. NOAA Tech Memo NMFS-NE-214.

Godard-Codding, C. A. J. et al. **2010**. Pacific Ocean–wide profile of CYP1A1 expression, stable carbon and nitrogen isotope ratios, and organic contaminant burden in sperm whale skin biopsies. Environmental Health Perspectives 119(3):337-343.

Godfrey, M. H., R. Barreto, and N. Mrosovsky. **1996**. Estimating past and present sex ratios of sea turtles in Suriname. Canadian Journal of Zoology 74:267-277.

Goldbogen, J. A., J. Calambokidis, D. Croll, J. Harvey, K. Newton, E. Oleson, G. Schorr, and R. E. Shadwick. **2008**. Foraging behavior of humpback whales: Kinematic and respiratory patterns suggest a high cost for a lunge. Journal of Experimental Biology 211:3712-3719.

Goldbogen, J. A., J. Calambokidis, R. E. Shadwick, E. M. Oleson, M. A. McDonald, and J. A. Hildebrand. **2006**. Kinematics of foraging dives and lunge-feeding in fin whales. Journal of Experimental Biology 209(7):1231.

Goldbogen, J. A., J. Potvin, and R. E. Shadwick. **2010**. Skull and buccal cavity allometry increase mass-specific engulfment capacity in fin whales. Proceedings of the Royal Society B: Biological Sciences 277(1683):861-868.

Goodman, S. J. **1998**. Patterns of extensive genetic differentiation and variation among European harbor seals (*Phoca vitulina vitulina*) revealed using microsatellite DNA polymorphisms. Molecular Biology and Evolution 15(2):104-118.

Goodson, A. D., and C. R. Sturtivant. **1996**. Sonar characteristics of the harbour porpoise (*Phocoena phocoena*): source levels and spectrum. ICES Journal of Marine Science 53:465-472.

Goold, J. C., H. Whitehead, and R. J. Reid. **2002**. North Atlantic sperm whale, *Physeter macrocephalus*, strandings on the coastlines of the British Isles and eastern Canada. Canadian Field-Naturalist 116(3):371-388.

Goold, J. C., and M. C. Clarke. **2000**. Sound velocity in the head of the dwarf sperm whale, *Kogia sima*, with anatomical and functional discussion. Journal of the Marine Biological Association of the United Kingdom 80:535-542.

Goshe, L. **2009**. Age at maturation and growth rates of green sea turtles (*Chelonia mydas*) along the southeastern U.S. Atlantic coast using skeletochronology. MS thesis. University of North Carolina, Wilmington. 78 p.

Gouteux, B., D. C. G. Muir, S. Backus, E. W. Born, R. Dietz, T. Haug, T. Metcalfe, C. Metcalfe, and N. Øien. **2008**. Toxaphene in minke whales (*Balaenoptera acutorostrata*) from the North Atlantic. Environmental Pollution 153(1):71-83.

Gowans, S. **2009**. Bottlenose whales, In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, pp. 129-131, Academic Press, San Diego.

Gowans, S., H. Whitehead, J. K. Arch, and S. K. Hooker. **2000**. Population size and residency patterns of northern bottlenose whales (*Hyperoodon ampullatus*) using the Gully, Nova Scotia. Journal of Cetacean Research and Management 2(3):201-210.

Gowans, S., H. Whitehead, and S. K. Hooker. **2001**. Social organization in northern bottlenose whales, *Hyperoodon ampullatus*: Not driven by deep-water foraging? Animal Behaviour 62(2):369–377.

Greene, C. H., A. J. Pershing, R. D. Kenney, and J. W. Jossi. **2003**. Impact of climate variability on the recovery of endangered North Atlantic right whales. Oceanography 16(4):98-103.

Griffin, R. B. **1999**. Sperm whale distributions and community ecology associated with a warm-core ring off Georges Bank. Marine Mammal Science 15(1):33–51.

Griffin, R. B., and N. J. Griffin. **2003**. Distribution, habitat partitioning, and abundance of Atlantic spotted dolphins, bottlenose dolphins, and loggerhead sea turtles on the eastern Gulf of Mexico continental shelf. Gulf of Mexico Science 21(1):23–34.

Griffin, R. B., and N. J. Griffin. **2004**. Temporal variation in Atlantic spotted dolphin (*Stenella frontalis*) and bottlenose dolphin (*Tursiops truncatus*) densities on the West Florida continental shelf. Aquatic Mammals 30(3):380-390.

Gubbins, C. **2002a**. Association patterns of resident bottlenose dolphins (*Tursiops truncatus*) in a South Carolina estuary. Aquatic Mammals 28:24-31.

Gubbins, C. **2002b**. Use of home ranges by resident bottlenose dolphins (*Tursiops truncatus*) in a South Carolina estuary. Journal of Mammalogy 83(1):178-187.

Gubbins, C. M. **2002c**. The dolphins of Hilton Head: Their natural history, University of South Carolina Press, Coumbian, S.C.

Gubbins, C. M., M. Caldwell, S. Barco, K. Rittmaster, N. Bowles, and V. Thayer .**2003**. Abundance and sighting patterns of bottlenose dolphins (*Tursiops truncatus*) at four northwest Atlantic coastal sites. Journal of Cetacean Research and Management 5(2):141-147.

Guirlet, E., K. Das, J.-P. Thomé, and M. Girondot. **2010**. Maternal transfer of chlorinated contaminants in the leatherback turtles, *Dermochelys coriacea*, nesting in French Guiana. Chemosphere 79(7):720-726.

Guirlet, E., K. Das, and M. Girondot. **2008**. Maternal transfer of trace elements in leatherback turtles (*Dermochelys coriacea*) of French Guiana. Aquatic Toxicology 88(4):267-276.

Gur, B. M., and C. Niezrecki. **2010**. Shallow water acoustic propagation in West Indian manatee habitats. The Journal of the Acoustical Society of America 128:2466-2466.

Gur, B. M., and C. Niezrecki. **2007**. Autocorrelation based denoising of manatee vocalizations using the undecimated discrete wavelet transform. The Journal of the Acoustical Society of America 122(1):188-199.

Haas, H. L., E. LaCasella, R. LeRoux, H. Milliken, and B. Hayward. **2008**. Characteristics of sea turtles incidentally captured in the US Atlantic sea scallop dredge fishery. Fisheries Research 93(3):289-295.

Haas, H. L. **2010**. Using observed interactions between sea turtles and commercial bottom-trawling vessels to evaluate the conservation value of trawl gear modifications. Marine and Coastal Fisheries 2:263-276.

Hai, D. J., J. Lien, D. Nelson, and K. Curren. **1996**. A contribution to the biology of the white-beaked dolphin, *Lagenorhynchus albirostris*, in waters off Newfoundland. Canadian Field-Naturalist 110(2):278-287.

Hain, J. H. W., M. J. Ratnaswamy, R. D. Kenney, and H. E. Winn. **1992**. The fin whale, *Balaenoptera physalus*, in waters of the northeastern United States continental shelf. Reports of the International Whaling Commission 42:653-669.

Hain, J. H. W., G. R. Carter, S. D. Kraus, C. A. Mayo, and H. E. Winn. **1982**. Feeding behavior of the humpback whale, *Megaptera novaeangliae*, in the western North Atlantic. Fishery Bulletin 80(2):259-268.

Hall, A. J., R. J. Law, D. E. Wells, J. Harwood, H. M. Ross, S. Kennedy, C. R. Allchin, L. A. Campbell, and P. P. Pomeroy. **1992**. Organochlorine levels in common seals (*Phoca vitulina*) which were victims and survivors of the 1988 phocine distemper epizootic. Science of the Total Environment 115(1):1-2.

Hall, A. J., R. Deaville, R. J. Law, C. R. Allchin, and P. D. Jepson. **2006**. The risk of infection from polychlorinated biphenyl exposure in the harbor porpoise (*Phocoena phocoena*): A case–control approach. Environmental Health Perspectives 114(5):704-711.

Haller, M. A., K. Kovacs, and M. O. Hammill. **1996**. Maternal behaviour and energy investment by grey seals (*Halichoerus grypus*) breeding on land-fast ice. Canadian Journal of Zoology 74(8):1531-1541.

Halvorsen, K. M., and E. O. Keith. **2008**. Immunosuppression cascade in the Florida manatee (*Trichechus manatus latirostris*). Aquatic Mammals 34(4):412-419.

Hamazaki, T. **2002**. Spatiotemporal prediction models of cetacean habitats in the Mid-Western North Atlantic Ocean (from Cape Hatteras, North Carolina, U.S.A. to Nova Scotia, Canada). Marine Mammal Science 18(4):920-939.

Hamilton, P. K., and M. K. Marx. **2005**. Skin lesions on North Atlantic right whales: categories, prevalence and change in occurance in the 1990s. Diseases of Aquatic Organisms 68:71-82.

Hamilton, P. K., A. R. Knowlton, and M. K. Marx. **2007**. Right whales tell their own stories: The photo-identification catalog, In: Kraus S.D. and R.M. Rolland (eds.) The Urban Whale: North Atlantic Right Whales at the Crossroads, pp. 75-104, Harvard University Press, Cambridge, MA.

Hammill, M. O., J. W. Lawson, G. B. Stenson, and D. Ligard. **2007**. Pup production of Northwest Atlantic grey seals in the gulf of St.Lawrence and along the Nova Scotia Eastern Shore., Canadian Science Advisory Secretariat (CSAS) Res. Doc.

Hammill, M. O., and G. Stenson. **2006**. Abundance of northwest Atlantic hooded seals (1960-2005), Research Document, Department of Fisheries and Oceans. 23 p.

Hammill, M. O., and G. B. Stenson. **2000**. Estimated prey consumption by harp seals (*Phoca groenlandica*), hooded seals (*Cystophora cristata*), grey seals (*Halichoerus grypus*) and harbour seals (*Phoca vitulina*) in Atlantic Canada. Journal of Northwest Atlantic Fishery Science 26:1-24.

Hammond, P. S., A. J. Hall, and J. H. Prime. **1994**. The diet of grey seals around Orkney and other island and mainland sites in north-eastern Scotland. Journal of Applied Ecology 31:340-350.

Harding, K., and T. Härkönen. **1999**. Development in the Baltic grey seal (*Halichoerus grypus*) and Ringed Seal (*Phoca hispida*) populations during the 20th century. Ambio 28(7):619-627.

Harewood, A., and J. Horrocks. **2008**. Impacts of coastal development on hawksbill hatchling survival and swimming success during the initial offshore migration. Biological Conservation 141(2):394-401.

Harper, C. G. et al. **2003**. Isolation and characterization of novel *Helicobacter* spp. from the gastric mucosa of harp seals *Phoca groenlandica*. Diseases of Aquatic Organisms 57(1/2):1-9.

Harris, D. E., B. Lelli, and G. Jakush. **2002**. Harp seal records from the southern Gulf of Maine: 1997-2001. Northeastern Naturalist 9(3):331–340.

Harris, D. E., and S. Gupta. **2006**. GIS-based analysis of ice-breeding seal strandings in the Gulf of Maine. Northeastern Naturalist 13:403-420.

Harris, D. E., B. Lelli, G. Jakush, and G. Early. **2001**. Hooded Seal (*Cystophora cristata*) records from the southern Gulf of Maine. Northeastern Naturalist 8(4):427-434.

Hart, K. M., P. Mooreside, and L. B. Crowder. **2006**. Interpreting the spatio-temporal patterns of sea turtle strandings: Going with the flow. Biological Conservation 129(2):283–290.

Hart, K., K. Kannan, T. Isobe, S. Takahashi, T. K. Yamada, N. Miyazaki, and S. Tanabe. **2008**. Time trends and transplacental transfer of perfluorinated compounds in melon-headed whales stranded along the Japanese coast in 1982, 2001/2002, and 2006. Environmental Science & Technology 42(19):7132-7137.

Hartman, D. S. **1979**. Ecology and behavior of the manatee (*Trichechus manatus*) in Florida. American Society of Mammalogists, Special Publication 5,153 p.

Hartman, K. L., F. Visser, and A. J. E. Hendriks. **2008**. Social structure of Risso's dolphins (*Grampus griseus*) at the Azores: a stratified community based on highly associated social units. Canadian Journal of Zoology 86:294-306.

Hasunuma, R., T. Ogawa, Y. Fujise, and Y. Kawanishi. **1993**. Analysis of selenium metabolites in urine samples of minke whale (*Balaenoptera acutorostrata*) using ion exchange chromatography. Comparative Biochemistry and Physiology Part C: Comparative Pharmacology 104(1):87-89.

Hatase, H., K. Sato, M. Yamaguchi, K. Takahashi, and K. Tsukamoto. **2006**. Individual variation in feeding habitat use by adult female green sea turtles (*Chelonia mydas*): are they obligately neritic herbivores? Oecologia 149(1):52–64.

Haug, T., K. T. Nilssen, U. Lindstrøm, and H. J. Skaug. **1997**. On the variation in size and composition of minke whale (*Balaenoptera acutorostrata*) forestomach contents. Journal of Northwest Atlantic Fishery Science 22:105–114.

Hauksson, E., and V. Bogason. **1997**. Comparative feeding of grey (*Halichoerus grypus*) and common seals (*Phoca vitulina*) in coastal waters of Iceland, with a note on the diet of hooded (*Cystophora cristata*) and harp seals (*Phoca groenlandica*). Journal of the Northwest Atlantic Fisheries Society 22:125–135.

Hauksson, E., and D. Ólafsdóttir. **1995**. Grey seal (*Halichoerus grypus* Fabr.), population biology, food and feeding habits, and importance as a final host for the lifecycle of sealworm (*Pseudoterranova decipiens* Krabbe) in Icelandic Waters. Developments in Marine Biology 4:565-572.

Hawkes, L. A., A. C. Broderick, M. S. Coyne, M. H. Godfrey, and B. J. Godley .**2007a**. Only some like it hot-quantifying the environmental niche of the loggerhead sea turtle. Diversity and Distributions 13(4):447–457.

Hawkes, L. A., A. C. Broderick, M. H. Godfrey, and B. J. Godley. **2007b**. Investigating the potential impacts of climate change on a marine turtle population. Global Change Biology 13(5):923-932.

Hays, G. C., M. R. Farquhar, P. Luschi, S. L. H. Teo, and T. M. Thys. **2009**. Vertical niche overlap by two ocean giants with similar diets: Ocean sunfish and leatherback turtles. Journal of Experimental Marine Biology and Ecology 370(1-2):134–143.

Hays, G. C. **2008**. Sea turtles: a review of some key recent discoveries and remaining questions. Journal of Experimental Marine Biology and Ecology 356(1-2):1-7.

Hays, G. C., J. D. R. Houghton, C. Isaacs, R. S. King, C. Lloyd, and P. Lovell. **2004a**. First records of oceanic dive profiles for leatherback turtles, *Dermochelys coriacea*, indicate behavioural plasticity associated with long-distance migration. Animal Behaviour 67(4):733–743.

Hays, G. C., J. D. R. Houghton, and A. E. Myers. **2004b**. Pan-Atlantic leatherback turtle movements. Nature 4(429):522-522.

Hazel, J., I. R. Lawler, and M. Hamann .**2009**. Diving at the shallow end: Green turtle behaviour in near-shore foraging habitat. Journal of Experimental Marine Biology and Ecology 371(1):84–92.

Hazel, J., I. R. Lawler, H. Marsh, and S. Robson. **2007**. Vessel speed increases collision risk for the green turtle *Chelonia mydas*. Endangered Species Research 3(2):105-113.

Hedges, M. E. **2007**. Development and application of a multistate model to the northern subpopulation of loggerhead sea turtles. MS thesis. Virginia Polytechnic Institute and State University. 155 p.

Heide-Jorgensen, M. P., E. S. Nordoy, N. Oien, L. P. Folkow, L. Kleivane, A. S. Blix, M. V. Jensen, and K. L. Laidre. **2001**. Satellite tracking of minke whales (*Balaenoptera acutorostrata*) off the coast of northern Norway. Journal of Cetacean Research and Management 3(2):175-178.

Heenehan, K. **2010**. Overview of acoustic impacts to sea turtles. Report to NOAA NE Regional Office, Protected Resources Division as part of the NOAA Ernest F. Hollings Scholarship Program. 42 p.

Heithaus, M., A. Frid, and L. M. Dill. **2002**. Shark-inflicted injury frequencies, escape ability, and habitat use of green and loggerhead turtles. Marine Biology 140(2):229-236.

Henry, A., T. V. N. Cole, and M. Garron. **2011**. Mortality and serious injury determinations for baleen whale stocks along the United States eastern seaboard and adjacent Canadian Maritimes, 2005-2009. NOAA Tech Memo NMFS-NE-11-18.

Henry, E., and M. O. Hammill. **2001**. Impact of small boats on the haulout activity of harbour seals (*Phoca vitulina*) in Metis Bay, Saint Lawrence Estuary, Quebec, Canada. Aquatic Mammals 27(2):140-148.

Heppell, S. S., D. T. Crouse, S. P. Epperly, W. Gabriel, T. Henwood, R. Marquez, and N. B. Thompson. **2005**. A population model to estimate recovery time, population size, and management impacts on Kemp's ridley sea turtles. Chelonian Conservation and Biology 7(4):767-773.

Herbst, L. H., and P. A. Klein. **1995**. Green turtle fibropapillomatosis: challenges to assessing the role of environmental cofactors. Environmental Health Perspectives 103(Suppl 4):27-30.

Herreman, J. K., G. M. Blundell, D. B. McDonald, and M. Ben-David. **2009a**. Asymmetrical male-mediated gene flow between harbor seal (*Phoca vitulina*) populations in Alaska. Canadian Journal of Zoology 87(6):498-507.

Herreman, J., G. Blundell, and M. Ben-David. **2009b**. Evidence of bottom-up control of diet driven by top-down processes in a declining harbor seal *Phoca vitulina richardsi* population. Marine Ecology Progress Series 374:287-300.

Hersh, S. L., and D. K. Odell. **1986**. Mass stranding of Fraser's dolphin, *Lagenodelphis hosei*, in the western North Atlantic. Marine Mammal Science 2(1):73-76.

Heyning, J. E. **1989**. Cuvier's beaked whales, *Ziphius cavirostris*, In: Handbook of Marine Mammals, vol. 4, pp. 289-308, Academic Press Inc., London.

Heyning, J. E., and J. G. Mead. **2009**. Cuvier's Beaked Whale *Ziphius cavirostris*, In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, pp. 294-245, Academic Press, San Diego.

Hicks, B. D., and G. A. Worthy. **1987**. Sealpox in captive grey seals (*Halichoerus grypus*) and their handlers. Journal of Wildlife Diseases 23(1):1-6.

Hildebrand, H. H. **1963**. Hallazgo del area de anidacion de la tortuga "lora" *Lepidochelys kempii* (Garman), en la costa occidental del Golfo de Mexico (Rept., Chel.). Ciencia Mexico 22(4):105-112.

Hirama, S., and L. M. Ehrhart. **2007**. Description, prevalence and severity of green turtle fibropapillomatosis in three developmental habitats on the east coast of Florida. Florida Scientist 70(4):435-448.

Hirotsu, R., T. Ura, J. Kojima, H. Sugimatsu, R. Bahl, and M. Yanagisawa. **2009**. Classification of sperm whale clicks and triangulation for real-time localization with SBL arrays, In: OCEANS 2008, pp. 1-7.

Hlista, B., H. Sosik, L. Martin Traykovski, R. Kenney, and M. Moore. **2009**. Seasonal and interannual correlations between right-whale distribution and calving success and chlorophyll concentrations in the Gulf of Maine, USA. Marine Ecology Progress Series 394:289-302.

Hobbs, K. E., D. C. G. Muir, E. W. Born, R. Dietz, T. Haug, T. Metcalf, C. Metcalfe, and N. Øien. **2003**. Levels and patterns of persistent organochlorines in minke whale (*Balaenoptera acutorostrata*) stocks from the North Atlantic and European Arctic. Environmental Pollution 121(2):239-252.

Hoekert, W. E. J., L. H. G. Van Tienan, P. Van Nugteren, and S. Dench. **2000**. The sea turtles of Suriname 1997—project: comparing relocated nests to undisturbed nests, In: Proceedings of the 18th annual symposium on sea turtle biology and conservation, pp. 192-194, NOAA Tech Memo NMFS-SEFSC-436.

Hoelzel, A. R., A. Natoli, M. E. Dahlheim, C. Olavarria, R. W. Baird, and N. A. Black. **2002**. Low worldwide genetic diversity in the killer whale (*Orcinus orca*): implications for demographic history. Proceedings of the Royal Society B: Biological Sciences 269(1499):1467-1473.

Hoelzel, A. R., E. M. Dorsey, and S. J. Stern. **1989**. The foraging specializations of individual minke whales. Animal Behaviour 38(5):786-794.

Hohn, A. A., S. J. Chivers, and J. Barlow. **1985**. Reproductive maturity and seasonality of male spotted dolphins, *Stenella attenuata*, in the eastern tropical Pacific. Marine Mammal Science 1(4):273-293.

Hohn, A. A., D. S. Rotstein, C. A. Harms, and B. L. Southall. **2006**. Report on Marine Mammal Unusual Mortality Event UMESE0501Sp: Multispecies Mass Stranding of Pilot Whales (*Globicephala macrorhynchus*), Minke Whale (*Balaenoptera acutorostrata*), and Dwarf Sperm Whales (*Kogia sima*) in North Carolina on 15-16 January 2005, NOAA Tech MemoNMFS-SEFSC-537. 222 p.

Holloway-Adkins, K. G. **2006**. Juvenile green turtles (*Chelonia mydas*) foraging on a high-energy, shallow reef on the east coast of Florida, USA, p 221 in: Proceedings of the 26th annual symposium on sea turtle biology and conservation, International Sea Turtle Society, Crete, Greece.

Holsbeek, L., C. R. Joiris, V. Debacker, I. B. Ali, P. Roose, J. P. Nwllissen, S. Gobert, J. Bouquegneau, and M. Bossicart. **1999**. Heavy metals, organochlorines and polycyclic aromatic hydrocarbons in sperm whales stranded in the southern North Sea during the 1994/1995 winter. Marine Pollution Bulletin 38(4):304–313.

Holt, M. M., D. P. Noren, V. Veirs, C. K. Emmons, and S. Veirs. **2009**. Speaking up: Killer whales (*Orcinus orca*) increase their call amplitude in response to vessel noise. Journal of the Acoustical Society of America 125(1):EL27-EL32.

Honda, K., R. Tatsukawa, K. Itano, N. Miyazaki, and T. Fujiyama. **1983**. Heavy metal concentrations in muscle, liver and kidney tissue of striped dolphin, *Stenella coeruleoalba*, and their variations with body length, weight, age and sex. Agriculture, Biology, and Chemistry 47(6):1219-1228.

Honda, K., Y. Yamamoto, H. Kato, and R. Tatsukawa. **1987**. Heavy metal accumulations and their recent changes in southern minke whales *Balaenoptera acutorostrata*. Archives of Environmental Contamination and Toxicology 16(2):209–216.

Hooker, S. K., H. Whitehead, and S. Gowans. **2002a**. Ecosystem consideration in conservation planning: energy demand of foraging bottlenose whales (*Hyperoodon ampullatus*) in a marine protected area. Biological Conservation 104(1):51–58.

Hooker, S. K., and R. W. Baird. **1999**. Deep-diving behaviour of the northern bottlenose whale, *Hyperoodon ampullatus* (Cetacea: Ziphiidae). Proceedings of the Royal Society B: Biological Sciences 266(1420):671-676.

Hooker, S. K., and H. Whitehead. **2002**. Click characteristics of northern bottlenose whales (*Hyperoodon ampullatus*). Marine Mammal Science 18(1):69–80.

Hooker, S. K., H. Whitehead, S. Gowans, and R. W. Baird. **2002b**. Fluctuations in distribution and patterns of individual range use of northern bottlenose whales. Marine Ecology Progress Series 225:287-297.

Hooker, S. K., R. W. Baird, S. Al-Omari, S. Gowans, and H. Whitehead. **2001a**. Behavioral reactions of northern bottlenose whales (*Hyperoodon ampullatus*) to biopsy darting and tag attachment procedures. Fishery Bulletin 99(2):303-308.

Hooker, S. K., S. J. Iverson, P. Ostrom, and S. C. Smith. **2001b**. Diet of northern bottlenose whales inferred from fatty-acid and stable-isotope analyses of biopsy samples. Canadian Journal of Zoology 79(8):1442-1454.

Hooker, S. K., T. Metcalfe, C. Metcalfe, C. M. Angell, J. Y. Wilson, M. Moore, and H. Whitehead. **2008**. Changes in persistent contaminant concentration and CYP1A1 protein expression in biopsy samples from northern bottlenose whales, *Hyperoodon ampullatus*, following the onset of nearby oil and gas development. Environmental Pollution 152(1):205-216.

Hooker, S. K., R. W. Baird, and A. Fahlman. **2009**. Could beaked whales get the bends?: Effect of diving behaviour and physiology on modelled gas exchange for three species: *Ziphius cavirostris*, *Mesoplodon densirostris* and *Hyperoodon ampullatus*. Respiratory Physiology & Neurobiology 167(3):235-246.

Horwood, J. **1990**. Biology and exploitation of the minke whale, CRC Press, Boca Raton, FL.

Horwood, J. **2009**. Sei whale (*Balaenoptera borealis*), In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, pp. 1001-1003, Academic Press, San Diego.

Houde, M. et al. **2006**. Polychlorinated biphenyls and hydrozylated polychlorinated biphenyls in plasma of bottlenose dolphins (*Tursiops truncatus*) from the western Atlantic and the Gulf of Mexico. Environmental Science & Technology 40:5860-5866.

Houghton, J. D. R., T. K. Doyle, M. W. Wilson, J. Davenport, and G. C. Hays. **2006**. Jellyfish aggregations and leatherback turtle foraging patterns in a temperate coastal environment. Ecology 87(8):1967-1972.

Houghton, J. D. R., T. K. Doyle, J. Davenport, R. P. Wilson, and G. C. Hays. **2008**. The role of infrequent and extraordinary deep dives in leatherback turtles (*Dermochelys coriacea*). Journal of Experimental Biology 211(16):2566-2575.

Houser, D. S., D. A. Helweg, and P. W. B. Moore. **2001**. A bandpass filter-bank model of auditory sensitivity in the humpback whale. Aquatic Mammals 27(2):82-91.

Huber, H. R., S. J. Jeffries, R. F. Brown, R. L. DeLong, and G. R. VanBlaricom. **2001**. Correcting aerial survey counts of harbor seals (*Phoca vitulina richardsi*) in Washington and Oregon. Marine Mammal Science 17(2):276-293.

Huber, H. R., S. J. Jeffries, D. M. Lambourn, and B. R. Dickerson. **2010**. Population substructure of harbor seals (*Phoca vitulina richardsi*) in Washington State using mtDNA. Canadian Journal of Zoology 88(3):280-288.

Hughes, G. R. **1996**. Nesting of the leatherback turtle (*Dermochelys coriacea*) in Tongaland, KwaZulu-Natal, South Africa, 1963-1995. Chelonian Conservation and Biology 2(2):153–158.

Hughes, G. R., P. Luschi, R. Mencacci, and F. Papi. **1998**. The 7000-km oceanic journey of a leatherback turtle tracked by satellite. Journal of Experimental Marine Biology and Ecology 229(2):209-217.

Hughes-Hanks, J. M., L. G. Rickard, C. Panuska, J. R. Saucier, T. M. O’Hara, L. Dehn, and R. M. Rolland. **2005**. Prevalence of *Cryptosporidium* spp. and *Giardia* spp. in five marine mammal species. The Journal of Parasitology 91(5):1225–1228.

Hunt, K. E., R. M. Rolland, S. D. Kraus, and S. K. Wasser. **2006**. Analysis of fecal glucocorticoids in the North Atlantic right whale (*Eubalaena glacialis*). General and Comparative Endocrinology 148(2):260–272.

Hunter, M. K., D. BroderickJ. Ovendon, K. P. Tucker, R. K. Bode, P. M. McGuire, and J. Lanyon. **2010**. Characterization of highly informative cross-species microsatellite panels for the Australian dugong (*Dugong dugon*) and Florida manatee (*Trichechus manatus latirostris*) including five novel primers. Molecular Ecology Resources 10(2):368-377.

Härkönen, T., R. Dietz, P. Reijnders, J. Teilmann, K. Harding, A. Hall, S. Brasseur, U. Siebert, S. J. Goodman, P. D. Jepson, T. D. Rasmussen and P. Thompson. A review of the 1988 and 2002 phocine distemper virus epidemics in European harbour seals. Diseases of Aquatic Organisms 68:115–130.

Härkönen, T. **1987**. Seasonal and regional variations in the feeding habits of the harbour seal, *Phoca vitulina*, in the Skagerrak and the Kattegat. Journal of Zoology 213(3):535-543.

Härkönen, T., K. Harding, and S. G. Lunneryd. **1999**. Age- and sex-specific behaviour in harbour seals *Phoca vitulina* leads to biased estimates of vital population parameters. Journal of Applied Ecology 36:825-841.

ICES 2010. Report of the Working Group on Marine Mammal Ecology (WGMME), 12–15 April **2010**, Horta, The Azores., ICES.

ICES 2011. Report of the Working Group on Marine Mammal Ecology (WGMME), 21–24 February, **2011**, Berlin, Germany., ICES.

IUCN **2010**. *Caretta caretta* (Loggerhead). The IUCN Red List of Threatened Species. http://www.iucnredlist.org/apps/redlist/details/3897/0. Accessed 6 October 2010.

Ibsen, S. D., W. W. L. Au, P. E. Nachtigall, and M. Breese. **2009**. Functional bandwidth of an echolocating Atlantic bottlenose dolphin (*Tursiops truncatus*). Journal of the Acoustical Society of America 125(2):1214-1221.

Ikonomou, M. G., S. Rayne, and N. F. Crewe. **2007**. Concentrations of Polychlorinated Biphenyls (PCBs), Polychlorinated Dibenzo-p-dioxins and Furans (PCDD/Fs), and Polybrominated Diphenyl Ethers (PDVEs) as functions of sample depth in killer whale (*Orcinus orca*) blubber. Nature Precedings Available from Nature Precedings http://hdl.handle.net/10101/npre.2007.638.1.

Ingram, S. N., L. Walshe, D. Johnston, and E. Rogan. **2007**. Habitat partitioning and the influence of benthic topography and oceanography on the distribution of fin and minke whales in the Bay of Fundy, Canada. Journal of the Marine Biological Association of the United Kingdom 87(01):149–156.

Innis, C., A. Nyaike, C. R. Williams III, B. Dunnigan, C. Merigo, D. Woodward, E. S. Weber, and S. Frasca Jr. **2009**. Pathologic and parasitological findings of cold-stunned Kemp's ridley sea turtles (*Lepidochelys kempii*) stranded on Cape Cod, Massachusetts 2001-2006. Journal of Wildlife Diseases 45(3):594-610.

Innis, C., M. Tlusty, C. Perkins, S. Holladay, C. Merigo, and Es. Weber. **2008**. Trace metal and organochlorine pesticide concentrations in cold-stunned juvenile Kemp's ridley turtles (*Lepidochelys kempii*) from Cape Cod, Massachusetts. Chelonian Conservation and Biology 7(2):230-239.

Isobe, T., Y. Ochi, K. Ramu, T. Yamamoto, Y. Tajima, T. K. Yamada, M. Amano, N. Miyazaki, S. Takahashi, and S. Tanabe. **2009**. Organohalogen contaminants in striped dolphins (*Stenella coeruleoalba*) from Japan: Present contamination status, body distribution and temporal trends (1978–2003). Marine Pollution Bulletin 58(3):396–401.

Jaber, J. R., J. Perez, M. Arbelo, M. Andrada, M. Hidalgo, J. C. Gomez-Villamandos, T. Van Den Ingh, and A. Fernandez. **2004**. Hepatic lesions in cetaceans stranded in the Canary Islands. Veterinary Pathology 41(2):147-153.

Jaber, J. R., J. Pérez, M. Arbelo, R. Zafra, and A. Fernández. **2006**. Pathological and immunohistochemical study of gastrointestinal lesions in dolphins stranded in the Canary Islands. The Veterinary Record 159(13):410-414.

Jacobs, S. R., and J. M. Terhune. **2000**. Harbor seal (*Phoca vitulina*) numbers along the New Brunswick coast of the Bay of Fundy in autumn in relation to aquaculture. Northeastern Naturalist 7(3):289-296.

Jacobs, S. R., and J. M. Terhune. **2002**. The effectiveness of acoustic harassment devices in the Bay of Fundy, Canada: seal reactions and a noise exposure model. Aquatic Mammals 28(2):147–158.

Jacobsen, J. K., L. Massey, and F. Gulland. **2010**. Fatal ingestion of floating net debris by two sperm whales (*Physeter macrocephalus*). Marine Pollution Bulletin 60:765-767.

Jahoda, M., C. L. Lafortuna, N. Biassoni, C. Almirante, A. Azzellino, S. Panigada, M. Zanardelli, and G. N. Sciara. **2003**. Mediterranean fin whale's (*Balaenoptera physalus*) response to small vessels and biopsy sampling assessed through passive tracking and timing of respiration. Marine Mammal Science 19(1):96–110.

James, M. C., S. A. Eckert, and R. A. Myers. **2005a**. Migratory and reproductive movements of male leatherback turtles (*Dermochelys coriacea*). Marine Biology 147(4):845–853.

James, M. C., and N. Mrosovsky. **2004**. Body temperatures of leatherback turtles (*Dermochelys coriacea*) in temperate waters off Nova Scotia, Canada. Canadian Journal of Zoology 82(8):1302–1306.

James, M. C., C. A. Ottensmeyer, S. A. Eckert, and R. A. Myers. **2006**. Changes in diel diving patterns accompany shifts between northern foraging and southward migration in leatherback turtles. Canadian Journal of Zoology 84(5):754-765.

James, M. C., C. A. Ottensmeyer, and R. A. Meyers. **2005b**. Identification of high-use habitat and threats to leatherback sea turtles in northern waters: new directions for conservation. Ecology Letters 8:195-201.

James, M. C., S. A. Sherrill-Mix, and R. A. Myers. **2007**. Population characteristics and seasonal migrations of leatherback sea turtles at high latitudes. Marine Ecology Progress Series 337:245-254.

James, M. C., and T. B. Herman. **2001**. Feeding of *Dermochelys coriacea* on Medusae in the northwest Atlantic. Chelonian Conservation and Biology 4(1):2002-2005.

Janik, V. **2000**. Whistle matching in wild bottlenose dolphins (*Tursiops truncatus*). Science 289:1355-1357.

Jansen, J. K., J. L. Bengtson, P. L. Boveng, S. P. Dahle, and J. Ver Hoef. **2006**. Disturbance of harbor seals by cruise ships in Disenchantment Bay, Alaska: an investigation at three spatial and temporal scales. AFSC Processed Rep. 2006-02. Alaska Fish. Sci. Cent., Seattle, WA: National Marine Mammal Laboratory.

Jaquet, N., S. Dawson, and L. Douglas. **2001**. Vocal behavior of male sperm whales: Why do they click? The Journal of the Acoustical Society of America 109:2254-2259.

Jaquet, N., D. Gendron, and A. Coakes. **2003**. Sperm whales in the Gulf of California: Residency, movements, behavior, and the possible influence of variation in food supply. Marine Mammal Science 19(3):545-562.

Jaquet, N., and D. Gendron. **2002**. Distribution and relative abundance of sperm whales in relation to key environmental features, squid landings and the distribution of other cetacean species in the Gulf of California, Mexico. Marine Biology 141(3):591-601.

Jauniaux, T., G. Charlier, M. Desmecht, J. Haelters, T. Jacques, B. Losson, J. Van Gompel, J. Tavernier, and F. Coignoul. **2000**. Pathological findings in two fin whales (*Balaenoptera physalus*) with evidence of morbillivirus infection. Journal of Comparative Pathology 123(2-3):198-201.

Jefferson, T. A. **2009a**. Rough-toothed dolphin *Steno bredanensis*, In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, pp. 990-992, Academic Press, San Diego.

Jefferson, T. A., S. Leatherwood, and M. A. Weber. **1994**. Marine Mammals of the World., FAO, Rome.

Jefferson, T. A., M. A. Webber, and R. L. Pitman. **2007**. Marine Mammals of the World: A Comprehensive Guide to Their Identification., Academic Press, San Diego.

Jefferson, T. A., and N. B. Barros. **1997**. *Peponocephala electra*. Mammalian Species 553:1-6.

Jefferson, T. A. **2009b**. Clymene dolphin *Stenella clymene*, In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, pp. 241-243, Academic Press, San Diego.

Jefferson, T. A., D. Fertl, J. Bolanos-Jimenez, and A. N. Zerbini. **2009**. Distribution of common dolphins (*Delphinus* spp.) in the western Atlantic Ocean: a critical re-examination. Marine Biology 156:1109-1124.

Jefferson, T. A., and B. E. Curry. **2003**. *Stenella clymene*. Mammalian Species 726:1-5.

Jensen, A. S., and G. K. Silber. **2003**. Large whale ship strike database, NOAA Tech Memo NMFS-OPR-25.

Jenssen, B. M. **1996**. An overview of exposure to, and effects of, petroleum oil and organochlorine pollution in Grey seals (*Halichoerus grypus*). Science of the Total Environment 186(1-2):109-118.

Jepson, P. D. et al. **2003**. Gas-bubble lesions in stranded cetaceans. Nature 425(6958):575-576.

Jepson, P. D., R. Deaville, I. A. P. Patterson, A. M. Pocknell, H. M. Ross, J. R. Baker, F. E. Howie, R. J. Reid, A. Colloff, and A. A. Cunningham. **2005a**. Acute and chronic gas bubble lesions in cetaceans stranded in the United Kingdom. Veterinary Pathology 42(3):291-305.

Jepson, P. D., P. M. Bennet, R. Deaville, C. R. Allchin, J. R. Baker, and R. J. Law. **2005b**. Relationships between polychlorinated byphenyls and health status in harbor porpoises (*Phocoena phocoena*) stranded in the United Kingdom. Environmental Toxicology and Chemistry 24(1):238-248.

Jiang, M., M. W. Brown, J. T. Turner, R. D. Kenney, C. A. Mayo, Z. Zhang, and M. Zhou. **2007**. Springtime transport and retention of *Calanus finmarchicus* in Massachusetts and Cape Cod Bays, USA, and implications for right whale foraging. Marine Ecology Progress Series 349:183-197.

Jochens, A. et al. **2008**. Sperm whale seismic study in the Gulf of Mexico: Synthesis report, MMS Contract 1435-01-02-CA-85186, U. S. Department of the Interior, Minerals Management Service Gulf of Mexico OCS Regional Office New Orleans LA USA. 7 p.

Johnson, A., G. Salvador, J. Kenney, J. Robbins, S. Kraus, S. Landry, and P. Clapham. **2005**. Fishing gear involved in entanglements of right and humpback whales. Marine Mammal Science 21(4):635-645.

Johnson, M., L. S. Hickmott, N. Aguilar Soto, and P. T. Madsen. **2008**. Echolocation behaviour adapted to prey in foraging Blainville's beaked whale (*Mesoplodon densirostris*). Proceedings of the Royal Society B: Biological Sciences 275(1631):133-139.

Johnson, M., P. T. Madsen, W. M. X. Zimmer, N. Aguilar de Soto, and P. L. Tyack. **2006**. Foraging Blainville's beaked whales (*Mesoplodon densirostris*) produce distinct click types matched to different phases of echolocation. Journal of Experimental Biology 209(24):5038-5050.

Johnson, S. A., A. L. Bass, B. Libert, M. Marshall, and D. Fulk. **1999**. Kemp's ridley (*Lepidochelys kempii*) nesting in Florida. Florida Scientist 62(3/4):194–204.

Johnston, D. W., M. E. Chapla, L. E. Williams, and D. K. Mattila. **2007**. Identification of humpback whale *Megaptera novaeangliae* wintering habitat in the Northwestern Hawaiian Islands using spatial habitat modeling. Endangered Species Research 3(3):249-257.

Johnston, D. W., A. S. Friedlaender, L. G. Torres, and D. M. Lavigne. **2005a**. Variation in sea ice cover on the east coast of Canada from 1969-2002: climate variability and implications for harp and hooded seals. Climate Research 29:209-222.

Johnston, D. W., L. H. Thorne, and A. J. Read. **2005b**. Fin whales *Balaenoptera physalus* and minke whales *Balaenoptera acutorostrata* exploit a tidally driven island wake ecosystem in the Bay of Fundy. Marine Ecology Progress Series 305:287-295.

Johnston, D. W., A. J. Westgate, and A. J. Read. **2005c**. Effects of fine-scale oceanographic features on the distribution and movements of harbour porpoises *Phocoena phocoena* in the Bay of Fundy. Marine Ecology Progress Series 295:279-293.

Jones, B. A., T. K. Stanton, A. C. Lavery, M. P. Johnson, P. T. Madsen, and P. L. Tyack. **2008**. Classification of broadband echoes from prey of a foraging Blainville's beaked whale. The Journal of the Acoustical Society of America 123(3):1753-1762.

Jonsen, I. D., R. A. Myers, and M. C. James. **2006**. Robust hierarchical state-space models reveal diel variation in travel rates of migrating leatherback turtles. Journal of Animal Ecology 75(5):1046-1057.

Kajiwara, N., S. Kamikawa, M. Amano, A. Hayano, T. K. Yamada, N. Miyazaki, and S. Tanabe. **2008**. Polybrominated diphenyl ethers (PBDEs) and organochlorines in melon-headed whales, *Peponocephala electra*, mass stranded along the Japanese coasts: Maternal transfer and temporal trend. Environmental Pollution 156(1):106-114.

Kamel, S. J., and E. Delcroix. **2009**. Nesting ecology of the hawksbill turtle, *Eretmochelys imbricata*, in Guadeloupe, French West Indies from 2000-07. Journal of Herpetology 43(3):367-376.

Kamel, S. J., and N. Mrosovsky. **2006**. Deforestation: risk of sex ratio distortion in hawksbill sea turtles. Ecological Applications 16(3):923-931.

Kanda, N., M. Goto, and L. A. Pastene. **2006**. Genetic characteristics of western North Pacific sei whales, *Balaenoptera borealis*, as revealed by microsatellites. Marine Biotechnology 8(1):86-93.

Kandia, V., and Y. Stylianou. **2006**. Detection of sperm whale clicks based on the Teager-Kaiser energy operator. Applied Acoustics 67(11-12):1144-1163.

Karpouzli, E., and R. Leaper. **2004**. Opportunistic observations of interactions between sperm whales and deep-water trawlers based on sightings from fisheries observers in the northwest Atlantic. Aquatic Conservation: Marine and Freshwater Ecosystems 14(1):95-103.

Kastelein, R. A., N. Jennings, W. C. Verboom, D. de Haan, and N. M. Schooneman. **2006**. Differences in the response of a striped dolphin (*Stenella coeruleoalba*) and a harbor porpoise (*Phocoena phocoena*) to an acoustic alarm. Marine Environmental Research 61:363-378.

Kastelein, R. A., P. Wensveen, L. Hoek, W. C. Verboom, and J. M. Terhune. **2009**. Underwater detection of tonal signals between 0.125 kHz and 100 kHz by harbor seals (*Phoca vitulina*). Journal of the Acoustical Society of America 125(2):1222-1229.

Kastelein, R. A., L. Hoek, P. J. Wensveen, J. M. Terhune, and C. A. F. de Jong. **2010**. The effect of signal duration on the underwater hearing thresholds of two harbor seals (*Phoca vitulina*) for single tonal signals between 0.2 and 40 kHz. The Journal of the Acoustical Society of America 127(2):1135-1145.

Katona, S. K., J. A. Beard, P. E. Girton, and F. W. Wenzel. **1988**. Killer whales (*Orcinus orca*) from the Bay of Fundy to the Equator, including the Gulf of Mexico. Rit Fiskideildar 9:205-224.

Katona, S. K., and J. A. Beard. **1990**. Population size, migrations, and feeding aggregations of the humpback whale (*Megaptera novaeangliae*) in the western North Atlantic Ocean. Reports of the International Whaling Commission (Special Issue) 12:295-306.

Katona, S. K., V. Rough, and D. J. Richardson. **1993**. A Field Guide to the Whales, Porpoises and Seals of the Gulf of Maine and Eastern Canada., 3rd ed., Charles Scribner's Sons, New York.

Keller, C. A., L. I. Ward-Geiger, W. B. Brooks, C. K. Slay, C. R. Taylor, and B. J. Zoodsma. **2006a**. North Atlantic right whale distribution in relation to sea-surface temperature in the southeastern United States calving grounds. Marine Mammal Science 22(2):426-445.

Keller, J. M. et al. **2008**. Emerging contaminants in loggerhead and Kemp's ridley sea turtles from the southeastern coast of the U.S., In: Proceedings of the 25th annual symposium on sea turtle biology and conservation, p. 158, NOAA Tech Memo NMFS-SEFSC-582.

Keller, J. M., K. Kannan, S. Taniyasu, N. Yamashita, R. D. Day, M. D. Arendt, A. L. Segars, and J. R. Kucklick. **2005**. Perfluorinated compounds in the plasma of loggerhead and Kemp's ridley sea turtles from the southeastern coast of the United States. Environmental Science & Technology 39(23):9101-9108.

Keller, J. M., J. R. Kucklick, and P. D. McClellan-Green. **2004a**. Organochlorine contaminants in loggerhead sea turtle blood: extraction techniques and distribution among plasma and red blood cells. Archives of Environmental Contamination and Toxicology 46(2):254-264.

Keller, J. M., J. R. Kucklick, M. A. Stamper, C. A. Harms, and P. D. McClellan-Green. **2004b**. Associations between organochlorine contaminant concentrations and clinical health parameters in loggerhead sea turtles from North Carolina, USA. Environmental Health Perspectives 112(10):1074-1079.

Keller, J. M., P. D. McClellan-Green, J. R. Kucklick, D. E. Keil, and M. M. Peden-Adams. **2006b**. Effects of organochlorine contaminants on loggerhead sea turtle immunity: Comparison of a correlative field study and in vitro exposure experiments. Environmental Health Perspectives 114(1):70-76.

Kendall, W. L., C. A. Langtimm, C. A. Beck, and M. C. Runge. **2004**. Capture-recapture analysis for estimating manatee reproductive rates. Marine Mammal Science 20(3):424–437.

Kenney, R. D. **2007**. Right whales and climate change: Facing the prospect of a greenhouse future, In: Kraus S.D. and R.M. Rolland (eds.) The Urban Whale: North Atlantic Right Whales at the Crossroads. p 436-439, Harvard University Press, Cambridge, MA.

Kenney, R. D. **2009**. Right whales, In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, 2nd edition, pp. 962-971. Academic Press, New York.

Kenney, R. D., C. A. Mayo, and H. E. Winn. **2001**. Migration and foraging strategies at varying spatial scales in western North Atlantic right whales: a review of hypotheses. Journal of Cetacean Research and Management 2:251-260.

Kenney, R. D., P. M. Payne, D. W. Heinemann, and H. E. Winn. **1996**. Shifts in northeast shelf cetacean distributions relative to trends in Gulf of Maine/Georges Bank finfish abundance, In: Sherman, K., N.A. Jaworski, and T.J. Smayda (eds.) The Northeast Shelf Ecosystem: Assessment, Sustainability, and Management, pp. 169-196, Blackwell Science, Cambridge, MA.

Kenney, R. D., and H. E. Winn. **1987**. Cetacean biomass densities near submarine canyons compared to adjacent shelf/slope areas. Continental Shelf Research 7(2):107–114.

Kenney, R. D., and H. E. Winn. **1986**. Cetacean high-use habitats of the northeast United States continental shelf. Fishery Bulletin 84:345-357.

Kenney, R. D. **1990**. Bottlenose dolphins off the northeastern United States., In: The bottlenose dolphin., pp. 369-386, Academic Press Inc., San Diego.

Ketten, D. R., J. Lien, and S. Todd. **1993**. Blast injury in humpback whale ears: evidence and implications. The Journal of the Acoustical Society of America 94:1849-1850.

Kichler, K., M. T. Holder, S. K. Davis, R. Márquez-M, and D. W. Owens. **2002**. Detection of multiple paternity in the Kemp's Ridley sea turtle with limited sampling. Molecular Ecology 8(5):819-830.

Kim, G. B., S. Tanabe, R. Iwakiri, R. Tatsukawa, M. Amano, N. Miyazaki, and H. Tanaka. **1996**. Accumulation of butyltin compounds in Risso's dolphin (*Grampus griseus*) from the Pacific coast of Japan: comparison with organochlorine residue pattern. Environmental Science & Technology 30(8):2620-2625.

King, J. E. **1983**. Seals of the World, 2nd ed., Comstock Publishing Associates, Ithaca, NY.

Kinze, C. C. **2009**. White-beaked dolphin *Lagenorhynchus albirostris*, In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, pp. 1255-1258, Academic Press, San Diego.

Kite-Powell, H. K., A. Knowlton, and M. Brown. **2007**. Modeling the effect of vessel speed on right whale ship strike risk. Project report for NOAA/NMFS Project NA04NMF47202394.

Kjeld, M., O. Olafsson, G. A. Vikingsson, and J. Sigurjonsson. **2006**. Sex hormones and reproductive status of the North Atlantic fin whales (*Balaenoptera physalus*) during the feeding season. Aquatic Mammals 32(1):75–84.

Klanjscek, T., R. M. Nisbet, H. Caswell, and M. G. Neubert. **2007**. A model for energetics and bioaccumulation in marine mammals with applications to the right whale. Ecological Applications 17(8):2233-2250.

Kleivane, L., and J. U. Skaare. **1998**. Organochlorine contaminants in northeast Atlantic minke whales (*Balaenoptera acutorostrata*). Environmental Pollution 101(2):231-239.

Klima, E. F., G. R. Gitschlag, and M. L. Renaud. **1988**. Impacts of the explosive removal of offshore petroleum platforms on sea turtles and dolphins. Marine Fisheries Review 50(3):33-42.

Knowlton, A. R., S. D. Kraus, and R. D. Kenney. **1994**. Reproduction in North Atlantic right whales (*Eubalaena glacialis*). Canadian Journal of Zoology 72(7):1297-1305.

Knowlton, A. R., and S. D. Kraus. **2001**. Mortality and serious injury of northern right whales (*Eubalaena glacialis*) in the western North Atlantic Ocean. Journal of Cetacean Research and Management (special issue) 2:193-208.

Knowlton, A. R., J. Sigukjósson, J. N. Ciano, and S. D. Kraus. **1992**. Long-distance movements of North Atlantic right whales (*Eubalaena glacialis*). Marine Mammal Science 8(4):397-405.

Knowlton, A. R., M. K. Marx, H. M. Pettis, P. K. Hamilton, and S. D. Kraus. **2005**. Analysis of scarring on North Atlantic right whales (*Eubalaena glacialis*): monitoring rates of entanglement interaction 1980-2002, Final report to NMFS Contract # 43EANF030107.

Koltz, A. **2007**. Changes in fin whale (*Balaenoptera physalus*) song over a forty-four year period in New England waters. Honors thesis in Biological Sciences. Cornell University, Ithaca, NY. 33 p.

Koschinski, S. **2008**. Possible impact of Personal Watercraft (PWC) on harbor porpoises (*Phocoena phocoena*) and harbor seals (*Phoca vitulina*), 15th ASCONBANS Advisory Committee Meeting, Document AC15/Doc.45(O), Society for the Conservation of

Koschinski, S., B. M. Culik, O. D. Henriksen, N. Tregenza, G. M. Ellis, C. Jansen, and G. Kathe. **2003**. Behavioural reactions of free-ranging porpoises and seals to the noise of a simulated 2 MW windpower generator. Marine Ecology Progress Series 265:263-273.

Kot, B. W., C. Ramp, and R. Sears. **2009**. Decreased feeding ability of a minke whale (*Balaenoptera acutorostrata*) with entanglement-like injuries. Marine Mammal Science 25(3):706-713.

Kot, C. Y., A. M. Boustany, and P. N. Halpin. **2010**. Temporal patterns of target catch and sea turtle bycatch in the US Atlantic pelagic longline fishing fleet. Canadian Journal of Fisheries and Aquatic Sciences 67(1):42-57.

Kovacs, K. M. **2009**. Hooded Seal *Cystophora cristata*, In: Encyclopedia of Marine Mammals, pp. 569-573, Academic Press Inc.

Krahn, M. M. et al. **2007**. Persistent organic pollutants and stable isotopes in biopsy samples (2004/2006) from Southern Resident killer whales. Marine Pollution Bulletin 54(12):1903-1911.

Kraus, S. D., J. H. Prescott, A. R. Knowlton, and G. S. Stone. **1986**. Migration and calving of right whales (*Eubalaena glacialis*) in the western North Atlantic. Report of the International Whaling Commission (Special Issue 10):139-144.

Kraus, S. D., and J. J. Hatch. **2001**. Mating strategies in the North Atlantic right whale (*Eubalaena glacialis*). Journal of Cetacean Research and Management (special issue) 237-244.

Kraus, S. D., and R. Rolland. **2007**. The Urban whale: North Atlantic Right Whales at the Crossroads, Harvard Univ Press. Cambridge, MA.

Kraus, S. D., P. K. Hamilton, R. D. Kenney, A. Knowlton, and C. K. Slay. **2001**. Reproductive parameters of the North Atlantic right whale. Journal of Cetacean Research and Management (Special Issue) 2:231-236.

Kraus, S. D., R. M. Pace, and T. R. Frasier. **2007**. High investment, low return: the strange case of reproduction in *Eubalaena glacialis*, In: Kraus S.D. and R.M. Rolland (eds.) The Urban Whale: North Atlantic Right Whales at the Crossroads, pp. 172-199, Harvard University Press, Cambridge, MA.

Kraus, S. D., A. J. Read, A. Solow, K. Baldwin, T. Spradlin, E. Anderson, and J. Williamson. **1997**. Acoustic alarms reduce porpoise mortality. Nature 388:525-525.

Kretzmann, M., L. Mentzer, R. A. J. DiGiovanni, M. S. Leslie, and G. Amato. **2006**. Microsatellite diversity and fitness in stranded juvenile harp seals (*Phoca groenlandica*). Journal of Heredity 97(6):555-560.

Kruse, S., D. K. Caldwell, and M. C. Caldwell. **1999**. Risso's dolphin *Grampus griseus* (G. Cuvier, 1812), In: Ridgway S. H.. and R. Harrison (eds.) Handbook of Marine Mammals Vol. 6: The Second Book of Dolphins and Porpoises, pp. 183-212, Academic Press, San Diego.

Kuehl, D. W., R. Haebler, and C. W. Potter. **1991**. Chemical residues in dolphins from the U.S. Atlantic coast including Atlantic bottlenose obtained during the 1987-88 mass mortality. Chemosphere 22(11):1071-1084.

Kuker, K. J., J. A. Thomson, and U. Tscherter. **2010**. Novel surface feeding tactics of minke whales, *Balaenoptera acutorostrata*, in the Saguenay-St. Lawrence National Marine Park. The Canadian Field-Naturalist 119(2):214–218.

Kunito, T., I. Watanabe, G. Yasunaga, Y. Fujise, and S. Tanabe. **2002**. Using trace elements in skin to discriminate the populations of minke whales in southern hemisphere. Marine Environmental Research 53(2):175-197.

Lagueux, K. M., M. Zani, A. R. Knowlton, and S. D. Kraus. **2011**. Response by vessel operators to protection measures for right whales Eubalaena glacialis in the southeast US calving ground. Endangered Species Research 14:69-77.

Lahaye, V., P. Bustamante, J. Spitz, W. Dabin, K. Das, G. J. Pierce, and F. Caurant. **2005**. Long-term dietary segregation of common dolphins *Delphinus delphis* in the Bay of Biscay, determined using cadmium as an ecological tracer. Marine Ecology Progress Series 305:275-285.

Lahvis, G. P., R. S. Wells, D. W. Kuehl, J. L. Stewart, H. L. Rhinehart, and C. S. Via. **1995**. Decreased lymphocyte responses in free-ranging bottlenose dolphins (*Tursiops truncatus*) are associated with increased concentrations of PCBs and DDT in peripheral blood. 103 (Supplement 4):67-72.

Laist, D. W., A. R. Knowlton, J. G. Mead, A. S. Collet, and M. Podesta. **2001**. Collisions between ships and whales. Marine Mammal Science 17(1):35-75.

Laist, D. W., and J. E. Reynolds III. **2005**. Influence of power plants and other warm-water refuges on Florida manatees. Marine Mammal Science 21(4):739-764.

Laist, D. W., and C. Shaw. **2006**. Preliminary evidence that boat speed restrictions reduce deaths of Florida manatees. Marine Mammal Science 22(2):472-479.

Lake, J. L., R. Haebler, R. McKinney, C. A. Lake, and S. S. Sadove. **1994**. PCBs and other chlorinated organic contaminants in tissues of juvenile Kemp's Ridley turtles (*Lepidochelys kempii*). Marine Environmental Research 38(4):313-327.

Lambertsen, R. H. **1986**. Disease of the common fin whale (*Balaenoptera physalus*): Crassicaudiosis of the urinary system. Journal of Mammalogy 67(2):353-366.

Lambertsen, R. H. **1997**. Natural disease problems of the sperm whale. Bulletin van het Koninklijk Belgisch Instituut voor Natuurwetenschappen. Biologie= Bulletin de l'Institut Royal des Sciences Naturelles de Belgique. Biologie.

Lambertsen, R. H. **1990**. Disease biomarkers in large whale populations of the North Atlantic and other oceans. Biomarkers of environmental contamination (J. McCarthy and L. Shugart, eds.). Lewis Publishers, Chelsea, Michigan 395-417.

Lambertsen, R. H. **1992**. Crassicaudosis: a parasitic disease threatening the health and population recovery of large baleen whales. Review of Science and Technology, Office of International Epizoology 11(4):1131-1141.

Lambertsen, R. H., B. Birnir, and J. E. Bauer. **1986**. Serum chemistry and evidence of renal failure in the North Atlantic fin whale population. Journal of Wildlife Diseases 22(3):389-396.

Landy, R. B., R. H. Lambertsen, P. A. Palsson, L. Krook, A. Nevius, and R. Eckerlin. **1991**. Fluoride in the bone and diet of fin whales, *Balaenoptera physalus*. Marine Environmental Research 31(3):241-247.

Langtimm, C. A., and C. A. Beck. **2003**. Lower survival probabilities for adult Florida manatees in years with intense coastal storms. Ecological Applications 13(1):257-268.

Laplanche, C., O. Adam, M. Lopatka, and J. F. Motsch. **2005**. Male sperm whale acoustic behavior observed from multipaths at a single hydrophone. The Journal of the Acoustical Society of America 118:2677-2687.

Laran, S., and A. Gannier. **2008**. Spatial and temporal prediction of fin whale distribution in the northwestern Mediterranean Sea. ICES Journal of Marine Science 65(7):1260-1269.

Law, R. J., P. Bersuder, J. Barry, R. Deaville, R. J. Reid, and P. D. Jepson. **2010**. Chlorobiphenyls in the blubber of harbour porpoises (*Phocoena phocoena*) from the UK: levels and trends 1991-2005. Marine Pollution Bulletin 60:470-473.

Lavigne, D. M. **2009**. Harp Seal *Pagophilus groenlandicus*, In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, pp. 542-546, Academic Press Inc.

Lavigne, D. M., and K. M. Kovacs. **1988**. Harps and hoods: ice breeding seals of the Northweast Atlantic., 1st ed., University of Waterloo Press, Waterloo, Ontario, Canada. 174 p.

Lawson, D., J. DeAlteris, C. Parkins, and W. Hole. **2007**. An evaluation of the catch efficiency of a NMFS certified, standard turtle excluder device (TED) required in the mid-Atlantic summer flounder fishery. Contract EA133F-05-SE-6561, Woods Hole, Massachusetts.

LeBlanc, A. M., T. Wibbels, M. Antonio-P., G. Tavera, L. Vega-M., H. J. Martinez-O., J. Pena-V., P. M. Burchfield, E. Possardt, and B. Schroeder. **2010**. The arribada phenomenon and predator satiation: Evaluation of natural predation on a Kemp’s ridley arribada, In: Proceedings of the 28th annual symposium on sea turtle biology and conservation, p. 91. NOAA Tech Memo NMFS-SEFSC-602.

LeBlanc, A. M., T. Wibbels, H. J. Martinez-O., M. Antonio-P., L. Vega-M., D. J. Lira-R., J. Pena-V., P. M. Burchfield, and B. Schroeder. **2008**. Back to the natural nesting beach: A transition from egg corrals to in situ in the Kemp’s ridley sea turtle, In: Proceedings of the 27th annual symposium on sea turtle biology and conservation, p. 94, NOAA Tech Memo NMFS-SEFSC-569.

LeRoux, R. A., and P. H. Dutton. **2008**. Stock structure of hawksbill turtles, *Eretmochelys imbricata*, in the Caribbean: re-examination with additional mtDNA sequences, In: Proceedings of the 25th annual symposium on sea turtle biology and conservation, p. 114, NOAA Tech Memo NMFS-SEFSC-582.

Leandro, L., R. Rolland, P. Roth, N. Lundholm, Z. Wang, and G. Doucette. **2010**. Exposure of the North Atlantic right whale Eubalaena glacialis to the marine algal biotoxin, domoic acid. Marine Ecology Progress Series 398:287-303.

Leatherwood, S., D. K. Caldwell, and H. E. Winn. **1976**. Whales, dolphins, and porpoises of the western North Atlantic., NMFS Circ. 396 NOAA Tech. Report. 176 p.

Leatherwood, S., and R. R. Reeves. **1983a**. Rough-toothed Dolphin, In: The Sierra Club Handbook of Whales and Dolphins, pp. 180-183, Sierra Club Books, San Francisco.

Leatherwood, S., and R. R. Reeves. **1983b**. Harbor Porpoise, In: The Sierra Club handbook of whales and dolphins, pp. 262-266, Sierra Club Books, San Francisco.

Leatherwood, S., and R. R. Reeves. **1983c**. White-beaked Dolphin, In: The Sierra Club Handbook of Whales and Dolphins, pp. 194-196, Sierra Club Books, San Francisco.

Lefebvre, L. W., M. Marmontel, J. P. Reid, G. B. Rathbun, and D. P. Domning. **2001**. Status and biogeography of the West Indian manatee. Biogeography of the West Indies: Patterns and perspectives. CRC Press, Boca Raton, FL 425-474.

Lefebvre, L. W., J. P. Reid, W. J. Kenworthy, and J. A. Powell. **2000**. Characterizing manatee habitat use and seagrass grazing in Florida and Puerto Rico: implications for conservation and management. Pacific Conservation Biology 5(4):289-298.

Leon, Y. M., and K. A. Bjorndal. **2002**. Selective feeding in the hawksbill turtle, an important predator in coral reef ecosystems. Marine Ecology Progress Series 245:249-258.

Lesage, V., and M. O. Hammill. **2001**. The status of the grey seal, *Halichoerus grypus*, in the Northwest Atlantic. Canadian Field-Naturalist 115(4):653-662.

Leslie, A. J., D. N. Penick, J. R. Spotila, and F. V. Paladino. **1996**. Leatherback turtle, *Dermochelys coriacea*, nesting and nest success at Tortuguero, Costa Rica, in 1990-1991. Chelonian Conservation and Biology 2(2):159-168.

Lettevall, E., C. Richter, N. Jaquet, E. Slooten, S. Dawson, H. Whitehead, J. Christal, and P. M. C. Howard. **2002**. Social structure and residency in aggregations of male sperm whales. Canadian Journal of Zoology 80(7):1189-1196.

Levesque, J. **2009a**. Characterization of the southeastern US black sea bass (*Centropristis striata*) pot commercial fishery and implications for western North Atlantic right whale (*Eubalaena glacialis*) management and policy. Marine Policy 33(1):40-48.

Levesque, J. **2009b**. Establishment and closure of the directed Florida whiting (*Menticirrhus americanus*) fishery and implications for North Atlantic right whale (*Eubalaena glacialis*) conservation and management. Marine Policy 33(2):233-247.

Lewis, T., D. Gillespie, C. Lacey, J. Matthews, M. Danbolt, R. Leaper, R. McLanaghan, and A. Moscrop. **2007**. Sperm whale abundance estimates from acoustic surveys of the Ionian Sea and Straits of Sicily in 2003. Journal of the Marine Biological Association of the United Kingdom 87(01):353-357.

Lewison, R. L., L. B. Crowder, and D. J. Shaver. **2003**. The impact of turtle excluder devices and fisheries closures on loggerhead and Kemp’s ridley strandings in the western Gulf of Mexico. Conservation Biology 17(4):1089-1097.

Lewison, R. L., S. A. Freeman, and L. B. Crowder. **2004**. Quantifying the effects of fisheries on threatened species: the impact of pelagic longlines on loggerhead and leatherback sea turtles. Ecology Letters 7(3):221-231.

Lewison, R. L., and L. B. Crowder. **2007**. Putting longline bycatch of sea turtles into perspective. Conservation Biology 21(1):79-86.

Lick, R., and U. Piatkowski. **1998**. Stomach contents of a northern bottlenose whale (*Hyperoodon ampullatus*) stranded at Hiddensee, Baltic Sea. Journal of the Marine Biological Association of the United Kingdom 78:643-650.

Lien, J. **1993**. The reaction of humpback whales to underwater explosions: Orientation, movements, and behavior. The Journal of the Acoustical Society of America 94(3):1849-1849.

Lindstrøm, G., H. Wingfors, M. Dam, and B. v Bavel .**1999**. Identification of 19 Polybrominated Diphenyl Ethers (PBDE's) in long-finned pilot whale (*Globicephala melas*) from the Atlantic. Achives of Environmental Contamination and Toxicology 36(3):355-363.

Lindstrøm, U., A. Harbitz, T. Haug, and T. Pedersen **2000**. Foraging behaviour of minke whales (*Balaenoptera acutorostrata*) in the southern Barents Sea, ICES CM 2000/Q:07.

Lindstrøm, U., T. Haug, and I. Røttingen. **2002**. Predation on herring, *Clupea harengus*, by minke whales, *Balaenoptera acutorostrata*, in the Barents Sea. ICES Journal of Marine Science 59:58-70.

Lipscomb, T. P., F. Y. Schulman, D. Moffett, and S. Kennedy. **1994**. Morbilliviral disease in Atlantic bottlenose dolphins (*Tursiops truncatus*) from the 1987-1988 epizootic. Journal of Wildlife Diseases 30(4):567-571.

Litz, J. A. **2007**. Social structure, genetic structure, and persistnet organohalogen pollutants in bottlenose dolphins (*Tursiops truncatus*) in Biscayne Bay, Florida. Ph.D. thesis. University of Miami, Miami, Florida. 140 p.

Lohoefener, R. R., W. Hoggard, K. Mullin, C. Roden, and C. Rogers .**1989**. Petroleum structures and the distribution of sea turtles, In: Proceedings of the Spring Ternary Gulf of Mexico Studies Meeting, Minerals Management Service, New Orleans, LA. OCS Study MMS 89-0062.

Lucas, Z., and P.-Y. Daoust. **2002**. Large increases of harp seals (Phoca groenlandica) and hooded seals (Cystophora cristata) on Sable Island, Nova Scotia, since 1995. Polar Biology 25(8):562-568.

Lucas, Z., and W. T. Stobo. **2000**. Shark-inflicted mortality on a population of harbour seals (*Phoca vitulina*) at Sable Island, Nova Scotia. Journal of Zoology 252(03):405-414.

Lucke, K., P. A. Lepper, M.-A. Blanchet, B. Hoeve, E. Everatts, N. van Elk, and U. Siebert. **2008**. Auditory studies on harbour porpoises in relation to offshore wind turbines, Document AC15/Doc.42 (C), ASCOBANS, Bonn, Germany.

Lucke, K., P. A. Lepper, B. Hoeve, E. Everaarts, N. van Elk, and U. Siebert .**2007**. Perception of low-frequency acoustic signals by a harbour porpoise (*Phocoena phocoena*) in the presence of simulated offshore wind turbine noise. aquatic mammals 33(1):55-68.

Lyrholm, T., O. Leimar, B. Johanneson, and U. Gyllensten. **1999**. Sex–biased dispersal in sperm whales: contrasting mitochondrial and nuclear genetic structure of global populations. Proceedings of the Royal Society of London. Series B: Biological Sciences 266(1417):347-354.

Lyrholm, T., and U. Gyllensten. **1998**. Global matrilineal population structure in sperm whales as indicated by mitochondrial DNA sequences. Proceedings of the Royal Society of London. Series B: Biological Sciences 265(1406):1679-1684.

Lysiak, N. S., M. J. Moore, A. R. Knowlton, and I. Valiela. **2008**. Interpreting a long-term stable isotope record derived from North Atlantic right whale baleen: implications for ecosystem-level changes in the Gulf of Maine?, American Geophysical Union, 2000 Florida Ave., N. W. Washington DC 20009 USA,.

MacLeod, C. D., and A. D’Amico. **2006**. A review of beaked whale behaviour and ecology in relation to assessing and mitigating impacts of anthropogenic noise. Journal of Cetacean Research and Management 7(3):211-221.

MacLeod, C. D., M. B. Santos, and G. J. Pierce. **2003**. Review of data on diets of beaked whales: evidence of niche separation and geographic segregation. Journal of the Marine Biological Association of the United Kingdom 83(3):651-665.

MacLeod, C. D., and A. F. Zuur. **2005**. Habitat utilization by Blainville’s beaked whales off Great Abaco, northern Bahamas, in relation to seabed topography. Marine Biology 147(1):1-11.

Macleod, C. D. et al. **2006a**. Known and inferred distributions of beaked whale species (Cetacea: Ziphiidae). Journal of Cetacean Research and Management 7(3):271-286.

Macleod, K., R. Fairbairns, A. Gill, B. Fairbairns, J. Gordon, C. Blair-Myers, and E. C. M. Parsons. **2004**. Seasonal distribution of minke whales *Balaenoptera acutorostrata* in relation to physiography and prey off the Isle of Mull, Scotland. Marine Ecology Progress Series 277(263-274):74.

Macleod, K., M. P. Simmonds, and E. Murray. **2006b**. Abundance of fin (*Balaenoptera physalus*) and sei whales (*B. borealis*) amid oil exploration and development off northwest Scotland. Journal of Cetacean Research and Management 8(3):247-254.

Madsen, P. T., M. Johnson, P. J. O. Miller, N. Aguilar Soto, J. Lynch, and P. Tyack. **2006a**. Quantitative measures of air-gun pulses recorded on sperm whales (*Physeter macrocephalus*) using acoustic tags during controlled exposure experiments. The Journal of the Acoustical Society of America 120(4):2366-2379.

Madsen, P. T., M. Johnson, N. A. de Soto, W. M. X. Zimmer, and P. Tyack. **2005**. Biosonar performance of foraging beaked whales (*Mesoplodon densirostris*). Journal of Experimental Biology 208(2):181-194.

Madsen, P. T., I. Kerr, and R. Payne. **2004a**. Echolocation clicks of two free-ranging, oceanic delphinids with different food preferences: false killer whales *Pseudorca crassidens* and Risso's dolphins *Grampus griseus*. The Journal of Experimental Biology 207:1811-1823.

Madsen, P. T., I. Kerr, and R. Payne. **2004b**. Source parameter estimates of echolocation clicks from wild pygmy killer whales (*Feresa attenuata*) (L). The Journal of the Acoustical Society of America 116(4):1909-1912.

Madsen, P. T., B. Møhl, B. K. Nielsen, and M. Wahlberg 2002a. Male sperm whale behaviour during exposures to distant seismic survey pulses. Aquatic Mammals 28(3):231-240.

Madsen, P. T., R. Payne, N. U. Kristiansen, M. Wahlberg, I. Kerr, and B. Mohl. **2002b**. Sperm whale sound production studied with ultrasound time/depth-recording tags. Journal of Experimental Biology 205(13):1899-1906.

Madsen, P. T., M. Wahlberg, J. Tougaard, K. Lucke, and P. Tyack. **2006b**. Wind turbine underwater noise and marine mammals: implications of current knowledge and data needs. Marine Ecology Progress Series 309:279-295.

Madsen, P. T., and B. Møhl. **2000**. Sperm whales (*Physeter catodon* L. 1758) do not react to sounds from detonators. The Journal of the Acoustical Society of America 107:668-671.

Madsen, P., M. Wahlberg, and B. Møhl. **2002c**. Male sperm whale (*Physeter macrocephalus*) acoustics in a high-latitude habitat: implications for echolocation and communication. Behavioral Ecology and Sociobiology 53(1):31-41.

Magalhães, S., R. Prieto, M. A. Silva, J. Gonçalves, M. Afonso-Dias, and R. S. Santos. **2002**. Short-term reactions of sperm whales (*Physeter macrocephalus*) to whale-watching vessels in the Azores. Aquatic Mammals 28(3):267–274.

Maier, P. P., A. L. Segars, M. D. Arendt, J. D. Whitaker, B. W. Stender, L. Parker, R. Vendetti, D. W. Owens, J. K. Quattro, and S. R. Murphy. **2004**. Development of an index of sea turtle abundance based upon in-water sampling with trawl gear. Final project report to NMFS/NOAA, Grant Number NA07FL0499 86 p.

Makowski, C., J. A. Seminoff, and M. Salmon. **2005**. Home range and habitat use of juvenile Atlantic green turtles (*Chelonia mydas* L.) on shallow reef habitats in Palm Beach, Florida, USA. Marine Biology 148(5):1167-1179.

Malik, S., M. W. Brown, S. D. Kraus, A. R. Knowlton, P. K. Hamilton, and B. N. White. **1999**. Assessment of mitochondrial DNA structuring and nursery use in the North Atlantic right whale (*Eubalaena glacialis*). Canadian Journal of Zoology 77(8):1217–1222.

Malik, S., M. W. Brown, S. D. Kraus, and B. N. White. **2000**. Analysis of mitochondrial DNA diversity within and between North and South Atlantic right whales. Marine Mammal Science 16(3):545–558.

Malme, C. I., P. R. Miles, P. Tyack, C. W. Clark, and J. E. Bird. **1985**. Investigation of the potential effects of underwater noise from petroleum-industry activities on feeding humpback whale behavior. Final report, Bolt, Beranek and Newman, Inc., Cambridge, MA (USA).

Mansfield, A. W. 1966. The grey seal in eastern Canadian waters. Canadian Audubon Magazine 28:161-166.

Mansfield, K. L. **2006**. Sources of mortality, movements and behavior of sea turtles in Virginia. Ph.D. thesis. The College of William and Mary. 343 p.

Mansfield, K. L., V. S. Saba, and J. A. Musick. **2010**. Movements, migrations and wintering habitat of Virginia's (USA) immature loggerhead sea turtles, In: Proceedings of the 28th annual symposium on sea turtle biology and conservation, p. 46, NOAA Tech Memo NMFS-SEFSC-602.

Mansfield, K. L., and J. A. Musick. **2004**. Sea turtle diving behavior in Virginia, Final report submitted to the US Army Corps of Engineers, Norfolk,.

Marcoux, M., H. Whitehead, and L. Rendell. **2007**. Sperm whale feeding variation by location, year, social group and clan: evidence from stable isotopes. Marine Ecology Progress Series 333:309-314.

Marine Turtle Specialist Group. **2004**. 2004 Global Status Assessment - Green turtle (*Chelonia mydas*), Marine Turtle Specialist Group. The World Conservation Union (IUCN) Species Survival Commission Red List Programme.

Marques, T. A., L. Thomas, S. W. Martin, D. K. Mellinger, S. Jarvis, R. P. Morrissey, C. A. Ciminello, and N. DiMarzio. **2010**. Spatially explicit capture–recapture methods to estimate minke whale density from data collected at bottom-mounted hydrophones. Journal of Ornithology 1–11.

Marques, T. A., L. Thomas, J. Ward, N. DiMarzio, and P. L. Tyack. **2009**. Estimating cetacean population density using fixed passive acoustic sensors: An example with Blainville's beaked whales. The Journal of the Acoustical Society of America 125(4):1982-1994.

Marshall, H. D., K. A. Hart, E. S. Yaskowiak, G. B. Stenson, D. McKinnon, and E. A. Perry. **2010**. Molecular identification of prey in the stomach contents of Harp Seals (Pagophilus groenlandicus) using species-specific oligonucleotides. Molecular Ecology Resources 10(1):181-189.

Marten, K. **2000**. Ultrasonic analysis of pygmy sperm whale (*Kogia breviceps*) and Hubbs' beaked whale (*Mesoplodon carlhubbsi*) clicks. Aquatic Mammals 26(1):45-48.

Martin, S. W., T. Norris, E. M. Nosal, D. K. Mellinger, R. P. Morrissey, and S. Jarvis. **2011**. Automatic localization of individual Hawaiian minke whales from boing vocalizations. The Journal of the Acoustical Society of America 129:2506-2506.

Martin, S. W., L. Thomas, T. A. Marques, R. P. Morrissey, S. Jarvis, N. DiMarzio, and D. K. Mellinger. **2010**. Minke whale boing vocalization density estimation at the Pacific Missile Range Facility, Hawaii. The Journal of the Acoustical Society of America 127:1824-1824.

Mate, B. R., K. M. Stafford, and D. K. Ljungblad. **1994**. A change in sperm whale (*Physeter macroephalus*) distribution correlated to seismic surveys in the Gulf of Mexico. The Journal of the Acoustical Society of America 96:3268-3269.

Mathias, D., A. Thode, J. Straley, and K. Folkert. **2009**. Relationship between sperm whale (*Physeter macrocephalus*) click structure and size derived from videocamera images of a depredating whale (sperm whale prey acquisition). The Journal of the Acoustical Society of America 125:3444-3453.

Matthews, J. N., S. Brown, D. Gillespie, M. Johnson, R. McLanaghan, A. Moscrop, D. Nowacek, R. Leaper, T. Lewis, and P. Tyack. **2001**. Vocalisation rates of the North Atlantic right whale (*Eubalaena glacialis*). Journal of Cetacean Research and Management 3(3):271-282.

Mattila, D. K., L. N. Guinee, and C. A. Mayo. **1987**. Humpback whale songs on a North Atlantic feeding ground. Journal of Mammalogy 68(4):880–883.

Mayo, C. A., and M. K. Marx. **1990**. Surface foraging behaviour of the North Atlantic right whale, *Eubalaena glacialis*, and associated zooplankton characteristics. Canadian Journal of Zoology 68(10):2214–2220.

Mazaris, A. D., B. Broder, and Y. G. Matsinos. **2006**. An individual based model of a sea turtle population to analyze effects of age dependent mortality. Ecological Modelling 198(1-2):174-182.

Mazzarella, K. T. **2007**. Analysis of stranded loggerhead sea turtles (*Caretta caretta*) in North and South Carolina: Genetic composition and the effectiveness of newly implemented TED regulations. MS thesis. College of Charleston. 101 p.

Mazzariol, S. et al. **2011**. Sometimes sperm whales (*Physeter macrocephalus*) cannot find their way back to the high seas: A multidisciplinary study on a mass stranding. PLoS ONE 6(5):e19417.

Mazzoil, M., S. D. McCulloch, and R. H. Defran. **2005**. Observations on the site fidelity of bottlnose dolphins (*Tursiops truncatus*) in the Indian River Lagoon. Florida Scientist 68(4):217-226.

McAlpine, D. F. **2009**. Pygmy and Dwarf Sperm Whales *Kogia breviceps* and *K. sima*., In: Encyclopedia of Marine Mammals, pp. 936-938, Academic Press Inc.

McAlpine, D. F., P. T. Stevick, L. D. Murison, and S. D. Turnball. **1999**. Extralimital records of hooded seals (*Cystphora cristata*) from the Bay of Fundy and northern Gulf of Maine. Northeastern Naturalist 6:225-230.

McCarthy, A. L., S. Heppell, F. Royer, C. Freitas, and T. Dellinger. **2010**. Identification of likely foraging habitat of pelagic loggerhead sea turtles (*Caretta caretta*) in the North Atlantic through analysis of telemetry track sinuosity. Progress In Oceanography 86(1-2):224-231.

McCauley, R. D., J. Fewtrell, A. J. Duncan, C. Jenner, M. N. Jenner, J. D. Penrose, R. I. T. Prince, A. Adhitya, J. Murdoch, and K. McCabe. **2000**. Marine seismic surveys: a study of environmental implications. Prepared for the Australian Petroleum Production Exploration Association from the Centre for Marine Science and Technology, Curtin University. CMST R 99.

McCauley, S. J., and K. A. Bjorndal. **1999**. Conservation implications of dietary dilution from debris ingestion: Sublethal effects in post-hatchling loggerhead sea turtles. Conservation Biology 13(4):925-929.

McClellan, C. M., J. Braun-McNeill, L. Avens, B. P. Wallace, and A. J. Read. **2010**. Stable isotopes confirm a foraging dichotomy in juvenile loggerhead sea turtles. Journal of Experimental Marine Biology and Ecology 387:44-51.

McClellan, C. M., A. J. Read, B. A. Price, W. M. Cluse, and M. H. Godfrey. **2009**. Using telemetry to mitigate the bycatch of long-lived marine vertebrates. Ecological Applications 19(6):1660–1671.

McClellan, C. M., and A. J. Read. **2007**. Complexity and variation in loggerhead sea turtle life history. Biology Letters 3(6):592-594.

McClellan, C. M., and A. J. Read. **2009**. Confronting the gauntlet: understanding incidental capture of green turtles through fine-scale movement studies. Endangered Species Research 10:165-179.

McClenachan, L., J. B. C. Jackson, and M. J. H. Newman. **2008**. Conservation implications of historic sea turtle nesting beach loss. Frontiers in Ecology and the Environment 4(6):290-296.

McConnell, B. J., M. A. Fedak, P. Lovell, and P. S. Hammond. **1999**. Movements and foraging areas of grey seals in the North Sea. Journal of Applied Ecology 36:573-590.

McCulloch, S., P. P. Pomeroy, and J. B. Slater. **1999**. Individually distinctive pup vocalizations fail to prevent allo-suckling in grey seals. Canadian Journal of Zoology 77:716-723.

McCulloch, S., and D. J. Boness. **2000**. Mother–pup vocal recognition in the grey seal (*Halichoerus grypus*) of Sable Island, Nova Scotia, Canada. Journal of Zoology 251(4):449-455.

McDonald, D. L., and P. H. Dutton. **1996**. Use of PIT tags and photoidentification to revise remigration estimates of leatherback turtles (*Dermochelys coriacea*) nesting in St. Croix, U. S. Virgin Islands, 1979-1995. Chelonian Conservation and Biology 2(2):148-152.

McDonald, M. A., S. L. Mesnick, and J. A. Hildebrand. **2006**. Biogeographic characterisation of blue whale song worldwide: using song to identify populations, IWC SC/55/SH7.

McDonald, M. A., and C. G. Fox. **1999**. Passive acoustic methods applied to fin whale population density estimation. The Journal of the Acoustical Society of America 105:2643-2651.

McDonald, M., J. Hildebrand, and S. Mesnick. **2009**. Worldwide decline in tonal frequencies of blue whale songs. Endangered Species Research 9:13-21.

McFee, W. E., S. R. Hopkins-Murphy, and L. H. Schwacke. **2007**. Trends in bottlenose dolphin (*Tursiops truncatus*) strandings in South Carolina, USA, 1997-2003: Implications for the southern North Carolina and South Carolina management units. Journal of Cetacean Research and Management 8(2):195-201.

McFee, W. E., and S. R. Hopkins-Murphy. **2002**. Bottlenose dolphin (*Tursiops truncatus*) strandings in South Carolina, 1992-1996. Fishery Bulletin 100:258-265.

McGarrity, M. E., S. E. Milton, and P. L. Lutz. **2008**. Heat shock protein expression and fibropapillomatosis: Novel use of molecular techniques to evaluate health and stress levels in marine turtles. Proceedings of the 25th annual symposium on sea turtle biology and conservation NOAA Tech Memo NMFS-SEFSC-582. 64 p.

McHugh, B., R. J. Law, C. R. Allchin, E. Rogan, S. Murphy, M. B. Foley, D. Glynn, and E. McGovern. **2007**. Bioaccumulation and enantiomeric profiling of organochlorine pesticides and persistent organic pollutants in the killer whale (*Orcinus orca*) from British and Irish waters. Marine Pollution Bulletin 54(11):1724-1731.

McKenna, M. F., M. Soldevilla, E. Oleson, S. Wiggins, J. A. Hildebrand, C. C. Damiani, and D. K. Garcelon. **2009**. Increased underwater noise levels in the Santa Barbara Channel from commercial ship traffic and the potential impact on blue whales (*Balaenoptera musculus*), In: Proceedings of the Seventh California Islands Symposium, Oxnard, California, February 5-8, 2008.

McKenzie, C., B. J. Godley, R. W. Furness, and D. E. Wells. **1999**. Concentrations and patterns of organochlorine contaminants in marine turtles from Mediterranean and Atlantic waters. Marine Environmental Research 47:117-135.

McLellan, W. M., A. S. Friedlaender, J. G. Mead, C. W. Potter, and D. A. Pabst. **2003**. Analysing 25 years ofbottlenose dolphin (*Tursiops truncatus*) strandings along the Atlantic coast of the USA: Do historic records support the coastal migratory stock hypothesis? Journal of Cetacean Research and Management 4:297-304.

McLeod, B. A. **2008**. Historic levels of genetic diversity in the North Atlantic right (*Eubalaena glacialis*) and bowhead whale (*Balaena mysticetus*). Ph.D. thesis. Trent University, Peterborough, Ontario. 185 p.

McLeod, B. A., M. W. Brown, T. R. Frasier, and B. N. White. **2010**. DNA profile of a sixteenth century western North Atlantic right whale (*Eubalaena glacialis*). Conservation Genetics 11(1):339–345.

McLeod, B. A., and B. N. White. **2010**. Tracking mtDNA heteroplasmy through multiple generations in the North Atlantic right whale (*Eubalaena glacialis*). Journal of Heredity 101(2):235-239.

McMahon, C. R., and G. C. Hays. **2006**. Thermal niche, large-scale movements and implications of climate change for a critically endangered marine vertebrate. Global Change Biology 12(7):1330-1338.

McMichael, E., and J. Wyneken. **2010**. Sea turtles and ocean energy technology, Paper presented at the 31st Annual Symposium on Sea Turtle Biology and Conservation, San Diego, CA.

McSweeney, D. J., K. C. Chu, W. F. Dolphin, and L. N. Guinee. **1989**. North Pacific humpback whale songs: A comparison of southeast Alaskan feeding ground songs with Hawaiian wintering ground songs. Marine Mammal Science 5(2):139–148.

McSweeney, D. J., R. W. Baird, S. D. Mahaffy, D. L. Webster, and G. S. Schorr. **2009**. Site fidelity and association patterns of a rare species: Pygmy killer whales (*Feresa attenuata*) in the main Hawaiian Islands. Marine Mammal Science 25(3):557-572.

McAlpine, D. F., L. D. Murison, and E. P. Hoberg. **1997**. New records for the pygmy sperm whale, *Kogia breviceps* (Physeteridae) from Atlantic Canada with notes on diet and parasites. Marine Mammal Science 13(4):701–704.

Mead, J. G., and C. W. Potter. **1995**. Recognizing two populations of the bottlenose dolphin (*Tursiops truncatus*) off the Atlantic coast of North America: morphologic and ecologic considerations. International Biological Research Institute Reports 5:31–43.

Mead, J. G. **1989a**. Beaked whales of the genus *Mesoplodon*, In: Handbook of Marine Mammals, Vol. 4: River dolphins and toothed whales, pp. 349-430, Academic Press, San Diego.

Mead, J. G. **1989b**. Bottlenose whales., In: Handbook of Marine Mammals, vol. 4: River dolphins and toothed whales,, pp. 321-348, Academic Press Inc., San Diego.

Mead, J. G., D. K. Odell, R. S. Wells, and M. D. Scott. **1980**. Observations on a mass stranding of spinner dolphin, *Stenella longirostris*, from the west coast of Florida. Fishery Bulletin 78(2):353-360.

Mead, J. G., and C. W. Potter. **1990**. Natural history of bottlenose dolphins along the central Atlantic coast of the United States., In: The Bottlenose Dolphin, pp. 165-198, Academic Press Inc., San Diego.

Mehta, A. et al. **2007**. Baleen whales are not important as prey for killer whales *Orcinus orca* in high-latitude regions. Marine Ecology Progress Series 348:297-307.

Mellinger, D. K., C. D. Carson, and C. W. Clark. **2000**. Characteristics of minke whale (*Balaenoptera acutorostrata*) pulse trains recorded near Puerto Rico. Marine Mammal Science 16(4):739-756.

Mellinger, D. K., K. M. Stafford, and C. G. Fox. **2004**. Seasonal occurrence of sperm whale (*Physeter macrocephalus*) sounds in the Gulf of Alaska, 1999–2001. Marine Mammal Science 20(1):48-62.

Mellinger, D. K., S. L. Nieukirk, H. Matsumoto, S. L. Heimlich, R. P. Dziak, J. Haxel, M. Fowler, C. Meinig, and H. V. Miller. **2007**. Seasonal occurrence of North Atlantic right whale (*Eubalaena glacialis*) vocalizations at two sites on the Scotian Shelf. Marine Mammal Science 23(4):856-867.

Mellinger, D. K., and C. W. Clark. **2003**. Blue whale (*Balaenoptera musculus*) sounds from the North Atlantic. The Journal of the Acoustical Society of America 114(2):1108-1119.

Mendes, S., J. Newton, R. J. Reid, A. Frantzis, and G. J. Pierce. **2007a**. Stable isotope profiles in sperm whale teeth: variations between areas and sexes. Journal of the Marine Biological Association of the United Kingdom 87(02):621-627.

Mendes, S., J. Newton, R. J. Reid, A. F. Zuur, and G. J. Pierce. **2007b**. Stable carbon and nitrogen isotope ratio profiling of sperm whale teeth reveals ontogenetic movements and trophic ecology. Oecologia 151(4):605-615.

Mercado III, E., and L. N. Frazer. **1999**. Environmental constraints on sound transmission by humpback whales. The Journal of the Acoustical Society of America 106:3004-3016.

Merrick, R. L. **2005**. Identification of seasonal area management zones for North Atlantic right whale conservation. NEFSC Reference Document 05-09, 28 p.

Merrick, R., and H. Haas. **2008**. Analysis of Atlantic sea scallop (*Placopecten magellanicus*) fishery impacts on the North Atlantic population of loggerhead sea turtles (*Caretta caretta*). NOAA Tech Memo NMFS-NE-207.

Merrick, R. L., and T. V. N. Cole. **2007**. Evaluation of northern right whale ship strike reduction measures in the Great South Channel of Massachusetts. NOAA Tech Memo NMFS-NE-202 12.

Mesnick, S. L. **2001**. Genetic relatedness in sperm whales: evidence and cultural implications. Behavioral and Brain Sciences 24(02):346–347.

Mesnick, S., N. Warner, J. Straley, V. O'Connell, M. Purves, C. Guinet, J. E. Dyb, C. Lunsford, C. Roche, and N. Gasco. **n.d.** Global sperm whale (*Physeter macrocephalus*) depredation of demersal longlines. Available at: http://www.vanaqua.org/depredation/Documents/MesnicketalFinalExtendedAbstract.kh.pdf

Metcalfe, C., B. Koenig, T. Metcalfe, G. Paterson, and R. Sears. **2004**. Intra-and inter-species differences in persistent organic contaminants in the blubber of blue whales and humpback whales from the Gulf of St. Lawrence, Canada. Marine Environmental Research 57(4):245–260.

Meylan, A. B. **1999**. Status of the hawksbill turtle (*Eretmochelys imbricata*) in the Caribbean region. Chelonian Conservation and Biology 3(2):177–184.

Meylan, A. B., and M. Donnelly. **1999**. Status justification for listing the hawksbill turtle (*Eretmochelys imbricata*) as Critically Endangered on the 1996 IUCN Red List of Threatened Animals. Chelonian Conservation and Biology 3(2):200–224.

Meylan, A. B., B. E. Witherington, B. Brost, R. Rivero, and P. S. Kubilis. **2006**. Sea turtle nesting in Florida, USA: Assessments of abundance and trends for regionally significant populations of *Caretta*, *Chelonia,* and *Dermochelys*, In: Book of Abstracts. Twenty-sixth Annual Symposium on Sea Turtle Biology and Conservation, pp. 306-307, International Sea Turtle Society, Atherns, Greece.

Meylan, A. **1988**. Spongivory in hawksbill turtles: A diet of glass. Science, New Series 239(4838):393-395.

Meynier, L., C. Pusineri, J. Spitz, M. B. Santos, G. J. Pierce, and V. Ridoux. **2008**. Intraspecific dietary variation in the short-beaked common dolphin *Delphinus delphis* in the Bay of Biscay: importance of fat fish. Marine Ecology Progress Series 354:277-287.

Michaud, J., and C. T. Taggart. **2007**. Lipid and gross energy content of North Atlantic right whale food, *Calanus finmarchicus*, in the Bay of Fundy. Endangered Species Research 3(1):77.

Mickelson, L. E., and J. R. Downie. **2010**. Influence of incubation temperature on morphology and locomotion performance of leatherback (*Dermochelys coriacea*) hatchlings. Canadian Journal of Zoology 88(4):359–368.

Migaki, G., T. R. Sawa, and J. P. Dubey. **1990**. Fatal disseminated toxoplasmosis in a spinner dolphin (*Stenella longirostris*). Veterinary Pathology 27(6):463-464.

Mignucci-Giannoni, A. A., and D. K. Odell. **2001**. Tropical and subtropical records of hooded seals (*Cystophora cristata*) dispel the myth of extant Caribbean monk seals. Caribbean Bulletin of Marine Science 64:47-58.

Mignucci-Giannoni, A. A., G. M. Toyos-Gonzalez, J. Perez-Padilla, M. A. Rodriguez-Lopez, and J. Overing. **1999**. Mass stranding of pygmy killer whales (*Feresa attenuata*) in the British Virgin Islands. Journal of the Marine Biological Association of the United Kingdom 80:759-760.

Mikkelsen, A. M. H., and A. Lund. **1994**. Intraspecific variation in the dolphins *Lagenorhynchus albirostris* and *L. acutus* in metrical and non-metrical skull characters, with remarks on occurrence. Journal of Zoology 234:289-299.

Mikkelsen, B., T. Haug, and K. T. Nilssen. **2002**. Summer diet of grey seals (*Halichoerus grypus*) in Faroese waters. Sarsia: North Atlantic Marine Science 87(6):462-471.

Miksis-Olds, J. L., P. L. Donaghay, J. H. Miller, P. L. Tyack, and J. A. Nystuen. **2007**. Noise level correlates with manatee use of foraging habitats. The Journal of the Acoustical Society of America 121(5 Pt1):3011-3020.

Miksis-Olds, J. L., and J. H. Miller. **2006**. Transmission loss in manatee habitats. The Journal of the Acoustical Society of America 120(4):2320-2327.

Miksis-Olds, J. L., and P. L. Tyack. **2009**. Manatee (*Trichechus manatus*) vocalization usage in relation to environmental noise levels. The Journal of the Acoustical Society of America 125(3):1806-1815.

Miller, P. J. O., M. P. Johnson, and P. L. Tyack. **2004**. Sperm whale behaviour indicates the use of echolocation click buzzes “creaks” in prey capture. Proceedings of the Royal Society of London. Series B: Biological Sciences 271(1554):2239-2247.

Miller, P. J. O., N. Biassoni, A. Samuels, and P. L. Tyack. **2000**. Whale songs lengthen in response to sonar. Nature 405(6789):903. doi:10.1038/35016148.

Mintzer, V. J., D. P. Gannon, N. B. Barros, and A. J. Read. **2008**. Stomach contents of mass-stranded short-finned pilot whales (*Globicephala macrorhynchus*) from North Carolina. Marine Mammal Science 24(2):290-302.

Mitchell, E. **1974**. Present status of northwest Atlantic fin and other whale stocks, In: W.E. Scheville (ed.) The whale problem: A status report, pp. 108-169, Harvard University Press, Cambridge, MA.

Mitchell, E., and V. M. Kozicki. **1975**. Autumn stranding of a northern bottlenose whale (*Hyperoodon ampullatus*) in the Bay of Fundy, Nova Scotia. Journal of the Fisheries Research Board of Canada 32:1019-1040.

Mitson, R. B., and R. J. Morris. **1988**. Evidence of high-frequency acoustic emissions from the white-beaked dolphin (*Lagenorhynchus albirostris*). Journal of the Acoustical Society of America 83(2):825-826.

Mizroch, S. A., D. W. Rice, D. Zweifelhofer, J. Waite, and W. L. Perryman. **2009**. Distribution and movements of fin whales in the North Pacific Ocean. Mammal Review 39(3):193-227.

MMS (Minerals Management Service). **2008**. Cape Wind Energy Project, Nantucket Sound: Biological Assessment, Cape Wind Energy Project, Nantucket Sound: Biological Assessment. 286 p. Available at: http://www.nero.noaa.gov/prot_res/section7/MMS-signedBOs/CapeWindBiOpSignedNov132008.pdf.

Mohammad, B., R. McHugh, and J. Bell. **2008**. The effects of a shallow-water acoustic channel on right whale vocalisations. Paper presented at Acoustics ’08. 29 June-4 July 2008, Paris, France. 6 p.

Montie, E. W., R. J. Letcher, C. M. Reddy, M. J. Moore, B. Rubinstein, and M. E. Hahn. **2010**. Brominated flame retardants and organochlorine contaminants in winter flounder, harp and hooded seals, and North Atlantic right whales from the Northwest Atlantic Ocean. Marine Pollution Bulletin 60(8):1160-1169.

Montie, E. W., C. A. Manire, and D. A. Mann. **2011**. Live CT imaging of sound reception anatomy and hearing measurements in the pygmy killer whale, *Feresa attenuata*. Journal of Experimental Biology 214(6):945-955.

Montie, E. W., C. M. Reddy, W. A. Gebbink, and K. M. Touhey. **2009**. Organohalogen contaminants and metabolites in cerebrospinal fluid and cerebellum gray matter in short-beaked common dolphins and Atlantic white-sided dolphins from the western North Atlantic. Environmental Pollution 157:2345-2358.

Mooney, T. A., P. E. Nachtigall, M. Breese, S. Vlachos, and W. W. L. Au. **2009**. Predicting temporar threshold shifts in a bottlenose dolphin (*Tursiops truncatus*): the effects of noise level and duration. Journal of the Acoustical Society of America 125(3):1816-1826.

Moore, M. J., A. R. Knowlton, S. D. Kraus, W. A. Mc Lellan, and R. K. Bonde. **2005**. Morphometry, gross morphology and available histopathology in North Atlantic right whale (*Eubalaena glacialis*) mortalities. Journal of Cetacean Research and Management 6(3):199-214.

Moore, M. J., and G. A. Early. **2004**. Cumulative sperm whale bone damage and the bends. Science 306(5705):2215-2215.

Moore, M. J., A. L. Bogomolni, S. E. Dennison, G. Early, M. M. Garner, B. A. Hayward, B. J. Lentell, and D. S. Rotstein. **2009**. Gas bubbles in seals, dolphins, and porpoises entangled and drowned at depth in gillnets. Veterinary Pathology Online 46(3):536-547.

Moore, S. E., W. A. Watkins, M. A. Daher, J. R. Davies, and M. E. Dalheim. **2002**. Blue whale habitat associations in the Northwest. Oceanography 15(3):20-25.

Morato, T., D. A. Varkey, C. Damaso, M. Machete, M. Santos, R. Prieto, R. S. Santos, and T. J. Pitcher. **2008**. Evidence of a seamount effect on aggregating visitors. Marine Ecology Progress Series 357:23-32.

Moreno, I. B., A. N. Zerbini, D. Danilewicz, M. C. O. Santos, P. C. Simões-Lopes, J. Lailson-Brito, and A. F. Azevedo. **2005**. Distribution and habitat characteristics of dolphins of the genus *Stenella* (Cetacea: Delphinidae) in the southwest Atlantic Ocean. Marine Ecology Progress Series 300:229-240.

Moretti, D., N. DiMarzio, R. Morrissey, J. Ward, and S. Jarvis. **2006**. Estimating the density of Blainville's beaked whale (*Mesoplodon densirostris*) in the Tongue of the Ocean (TOTO) using passive acoustics, Naval Undersea Warfare Center Division Code 71, Bldg. 1351 Newport, RI 02841 USA.

Morin, P. A., N. C. Aitken, N. Rubio-Cisneros, A. E. Dizon, and S. Mesnick. **2007**. Characterization of 18 SNP markers for sperm whale (*Physeter macrocephalus*). Molecular Ecology Notes 7(4):626-630.

Morin, P. A. et al. **2010**. Complete mitochondrial genome phylogeographic analysis of killer whales (*Orcinus orca*) indicates multiple species. Genome Research 20(7):908-916.

Morreale, S. J., and E. A. Standora. **1998**. Early life stage ecology of sea turtles in northeastern U.S. waters. NOAA Tech Memo NMFS-SEFSC-413 49.

Morreale, S. J., C. F. Smith, K. Durham, R. A. J. DiGiovanni, and A. A. Aguirre. **2005**. Assessing health, status and trends in northeastern sea turtle populations Year End Report - September 2002-November 2004. Interim report to the National Marine Fisheries Service, Northeast Regional Office Contract Number EA133F-02-SE-0191. 42 p.

Morreale, S. J., and E. A. Standora. **2005**. Western North Atlantic waters: Crucial developmental habitat for Kemp's ridley and loggerhead sea turtles. Chelonian Conservation and Biology 4(4):872–882.

Morreale, S. J., E. A. Standora, J. R. Spotila, and F. V. Paladino. **1996**. Migration corridor for sea turtles. Nature 384(6607):319-320.

Morreale, S. J., A. B. Meylan, S. S. Sadove, and E. A. Standora. **1992**. Annual occurrence and winter mortality of marine turtles in New York waters. Journal of Herpetology 301–308.

Morrissey, R. P., J. Ward, N. DiMarzio, S. Jarvis, and D. J. Moretti. **2006**. Passive acoustic detection and localization of sperm whales (*Physeter macrocephalus*) in the Tongue of the Ocean. Applied Acoustics 67(11-12):1091-1105.

Mortimer, J. A., and M. Donnelly. **2008**. *Eretmochleys imbricata*. In: IUCN 2010 Red List of Threatened Species, Version 2010.4. www.iucnredlist.org. Accessed 16 September 2010.

Mrosovsky, N., P. H. Dutton, and C. P. Whitmore. **1984**. Sex ratios of two species of sea turtle nesting in Suriname. Canadian Journal of Zoology 62(11):2227-2239.

Mrosovsky, N., G. D. Ryan, and M. C. James. **2009**. Leatherback turtles: The menace of plastic. Marine Pollution Bulletin 58(2):287-289.

Muir, D. C. G., R. Wagemann, N. P. Grift, R. J. Norstom, M. Simon, and J. Lien. **1988**. Organochlorine chemical and heavy metal contaminants in white-beaked dolphins (*Lagenorhynchus*) and pilot whales (*Globicephala melaena*) from the coast of Newfoundland, Canada. Achives of Environmental Contamination and Toxicology 17:613-629.

Mullin, K. D., T. A. Jefferson, L. J. Hansen, and W. Hoggard. **1994**. First sightings of melon-headed whales (*Peponocephala electra*) in the Gulf of Mexico. Marine Mammal Science 10(3):342-348.

Murdoch, M., M. Mazzoil, S. McCulloch, S. Bechdel, G. O’Corry-Crowe, G. Bossart, and J. Reif 2010. Lacaziosis in bottlenose dolphins *Tursiops truncatus* along the coastal Atlantic Ocean, Florida, USA. Diseases of. Aquatic Organisms 92(1):69-73.

Murie, D. J., and D. M. Lavigne. **1992**. Growth and feeding habits of grey seals (*Halichoerus grypus*) in the northwestern Gulf of St. Lawrence, Canada. Canadian Journal of Zoology 70(8):1604-1613.

Murphy, M. A. **1995**. Occurrence and group characteristics of minke whales, *Balaenoptera acutorostrata*, in Massachusetts Bay and Cape Cod Bay. Fishery Bulletin 93(3):577-585.

Murphy, S. et al. **2009**. Importance of biological parameters in assessing the status of *Delphinus delphis*. Marine Ecology Progress Series 388:273-291.

Murphy, T. M., S. R. Murphy, D. B. B. Griffin, and C. P. Hope. **2006**. Recent occurrence, spatial distribution, and temporal variability of leatherback turtles (*Dermochelys coriacea*) in nearshore waters of South Carolina, USA. Chelonian Conservation and Biology 5(2):216-224.

Murray, K. T. **2004a**. Bycatch of sea turtles in the Mid-Atlantic sea scallop (*Placopecten magellanicus*) dredge fishery during 2003. NEFSC Reference Document 04-11.

Murray, K. T. **2004b**. Magnitude and distribution of sea turtle bycatch in the sea scallop (*Placopecten magellanicus*) dredge fishery in two areas of the northwestern Atlantic Ocean, 2001–2002. Fishery Bulletin 102:671-681.

Murray, K. T. **2005**. Total bycatch estimate of loggerhead turtles (*Caretta caretta*) in the 2004 Atlantic sea scallop (*Placopecten magellanicus*) dredge fishery. NEFSC Reference Document 05-12.

Murray, K. T. **2006**. Estimated average annual bycatch of loggerhead sea turtles (*Caretta caretta*) in U.S. mid-Atlantic bottom otter trawl gear, 1996-2004. NEFSC Reference Document 06-19.

Murray, K. T. **2007**. Estimated bycatch of loggerhead sea turtles (*Caretta caretta*) in U.S. Mid-Atlantic scallop trawl gear, 2004-2005, and in sea scallop dredge gear, 2005. NEFSC Reference Document 07-04.

Murray, K. T. **2008a**. Estimated average annual bycatch of loggerhead sea turtles (*Caretta caretta*) in US Mid-Atlantic bottom otter trawl gear, 1996-2004. NEFSC Reference Document 08-20.

Murray, K. T. **2009a**. Characteristics and magnitude of sea turtle bycatch in U.S. mid-Atlantic gillnet gear. Endangered Species Research 8:211-224.

Murray, K. T. **2009b**. Proration of estimated bycatch of loggerhead sea turtles in U.S. mid-Atlantic sink gillnet gear to vessel trip report landed catch, 2002-2006. NEFSC Reference Document 09-19. 7 p.

Murray, M. J. **2008b**. Behavioral interactions between harbor seals (*Phoca vitulina*) and gray seals (*Halichoerus grypus*) on Cape Cod, Massachusetts. MS thesis. Northeastern University, Boston, MA. 56 p.

Myers, A. E., and G. C. Hays. **2006**. Do leatherback turtles *Dermochelys coriacea* forage during the breeding season? A combination of data-logging devices provide new insights. Marine Ecology Progress Series 322:259–267.

Myrick, A. C., A. A. Hohn, J. Barlow, and P. A. Sloan. **1986**. Reproductive biology of female spotted dolphins, *Stenella attenuata*, from the eastern tropical Pacific. Fishery Bulletin 84(2):247-259.

Møhl, B., M. Wahlberg, P. T. Madsen, A. Heerfordt, and A. Lund. **2003**. The monopulsed nature of sperm whale clicks. The Journal of the Acoustical Society of America 114:1143-1154.

Møhl, B., M. Wahlberg, P. T. Madsen, L. A. Miller, and A. Surlykke. **2000**. Sperm whale clicks: Directionality and source level revisited. The Journal of the Acoustical Society of America 107:638-648.

Møller, P., E. W. Born, R. Dietz, T. Haug, D. E. Ruzzante, and N. Øien. **2003**. Regional differences in fatty acid composition in common minke whales (*Balaenoptera acutorostrata*) from the North Atlantic. Lipids and Stable Isotopes in Marine Food Webs in West Greenland 5(2). 91 p.

NMFS Office of Protected Resources **n.d.** Hawksbill Turtle (*Eretmochelys imbricata*). http://www.nmfs.noaa.gov/pr/species/turtles/hawksbill.htm. Accessed 22 December 2010.

NMFS Office of Protected Resources **n.d.** Leatherback Turtle (*Dermochelys coriacea*). http://www.nmfs.noaa.gov/pr/species/turtles/leatherback.htm. Accessed 28 March 2011.

NMFS Office of Protected Resources. **n.d.** Loggerhead Turtle (*Caretta caretta*). http://www.nmfs.noaa.gov/pr/species/turtles/loggerhead.htm. Accessed 21 October 2010.

Nachtigall, P. E., M. M. L. Yuen, T. A. Mooney, and K. A. Taylor. **2005**. Hearing measurements from a stranded infant Risso's dolphin, *Grampus griseus*. Journal of Experimental Biology 208(21):4181-4188.

Nachtigall, P. E., T. A. Mooney, K. A. Taylor, L. A. Miller, M. H. Rasmussen, T. Akamatsu, J. Teilmann, M. Linnenschmidt, and G. A. Víkingsson. **2008**. Shipboard measurements of the hearing of the white-beaked dolphin *Lagenorhynchus albirostris*. The Journal of Experimental Biology 211:642-647.

Nachtigall, P. E., J. L. Pawloski, and W. W. L. Au. **2003**. Temporary threshold shifts and recovery frollowing noise exposure in the Atlantic bottlenosed dolphin (*Tursiops truncatus*). Journal of the Acoustical Society of America 113(6):3425-3429.

National Marine Fisheries Service. **1991**. Final recovery plan for the humpback whale *Megaptera novaeangliae*., Prepared by the Humpback Whale Recovery Team for the

National Marine Fisheries Service, Silver Spring, Maryland. Availble at: http://www.nmfs.noaa.gov/pr/pdfs/recovery/whale_humpback.pdf.

National Marine Fisheries Service. **1998**. Recovery plan for the blue whale (*Balaenoptera musculus*), Prepared by Reeves R.R., P.J. Clapham, R.L. Brownell, Jr., and G.K. Silber for the National Marine Fisheries Service, Silver Spring, MD, 42 pp. Available at: http://www.nmfs.noaa.gov/pr/pdfs/recovery/whale_blue.pdf.

National Marine Fisheries Service **2001**. Stock assessments of loggerhead and leatherback seaturtles and an assessment of the impact of the pelagic longline fishery on the loggerhead and leatherback sea turtles of the Western North Atlantic. NOAA Tech Memo NMFS-SEFC-455.

National Marine Fisheries Service. **2005**. Recovery plan for the North Atlantic right whale (*Eubalaena glacialis*), National Marine Fisheries Service, Silver Spring, MD. Available at: http://www.nmfs.noaa.gov/pr/pdfs/recovery/whale_right_northatlantic.pdf.

National Marine Fisheries Service **2010a**. Recovery plan for the fin whale (*Balaenoptera physalus*), National Marine Fisheries Service, Silver Spring, MD. Available at: http://www.nmfs.noaa.gov/pr/pdfs/recovery/finwhale.pdf.

National Marine Fisheries Service. **2010b**. Recovery plan for the sperm whale (*Physeter macrocephalus*), National Marine Fisheries Service, Silver Spring, MD. Available at: http://www.nmfs.noaa.gov/pr/pdfs/recovery/final_sperm_whale_recovery_plan_21dec.pdf.

National Marine Fisheries Service, US Fish and Wildlife Service, and SEMARNAT. **2010**. Draft Bi-National Recovery Plan for the Kemp's Ridley sea turtle, second revision. Availble at: http://www.nmfs.noaa.gov/pr/pdfs/recovery/turtle_kempsridley_draft2.pdf.

National Marine Fisheries Service, and U.S. Fish & Wildlife Service. **1993**. Recovery plan for the hawksbill turtle in U.S. Carribbean, Atlantic Ocean and Gulf of Mexico, St. Petersburg, Florida. Available at: http://www.nmfs.noaa.gov/pr/pdfs/recovery/turtle_hawksbill_atlantic.pdf.

National Marine Fisheries Service, and U.S. Fish and Wildlife Service. **1991**. Recovery plan for U.S. population of Atlantic green turtle, NMFS, Washington, DC. Available at: http://www.nmfs.noaa.gov/pr/pdfs/recovery/turtle_green_atlantic.pdf.

National Marine Fisheries Service, and U.S. Fish and Wildlife Service. **2007a**. Five year review Hawksbill sea turtle. Available at: http://www.nmfs.noaa.gov/pr/pdfs/species/hawksbill_5yearreview.pdf.

National Marine Fisheries Service, and U.S. Fish and Wildlife Service. **2007b**. Green sea turtle (*Chelonia mydas*) 5-year review: Summary and evaluation. Available at http://www.nmfs.noaa.gov/pr/pdfs/species/greenturtle_5yearreview.pdf.

National Marine Fisheries Service, and U.S. Fish and Wildlife Service 2008. Recovery plan for the Northwest Atlantic population of the loggerhead sea turtle (*Caretta caretta*), second revision. Available at: http://www.nmfs.noaa.gov/pr/pdfs/recovery/turtle_loggerhead_atlantic.pdf.

Naud, M.-J., B. Long, J.-C. Brêthes, and R. Sears. **2003**. Influences of underwater bottom topography and geomorphology on minke whale (*Balaenoptera acutorostrata*) distribution in the Mingan Islands (Canada). Journal of the Marine Biological Association of the United Kingdom 83(04):889-896.

Neale, J. C. C., K. R. Schmelzer, J. T. Harvey, E. A. Berg, R. J. Small, E. K. Grigg, S. G. Allen, and R. S. Tjeerdema. **2009**. PCB and DDE contamination in harbor seals (*Phoca vitulina*) from North-Central California and Bristol Bay, Alaska. Aquatic Mammals 35(1):1-11.

Neimanis, A. S., H. N. Koopman, A. J. Westgate, and F. A. Leighton. **2008**. Evidence of exposure to *Brucella* sp. in harbor porpoises (*Phocoena phocoena*) from the Bay of Fundy, Canada. Journal of Wildlife Diseases 44(2):480-485.

Nelson, M. L., J. R. Gilbert, and K. J. Boyle. **2006**. The influence of siting and deterrence methods on seal predation at Atlantic salmon (*Salmo salar*) farms in Maine, 2001-2003. Canadian Journal of Fisheries and Aquatic Sciences 63:1710-1721.

Newhall, A. E., Y. T. Lin, J. F. Lynch, and M. F. Baumgartner. **2009**. Sei whale localization and vocalization frequency sweep rate estimation during the New Jersey Shallow Water 2006 experiment. The Journal of the Acoustical Society of America 125:2738-2738.

Nichols, O. C., and P. K. Hamilton. **2004**. Occurrence of the parasitic sea lamprey *Petromyzon marinus* on western North Atlantic right whales *Eubalaena glacialis*. Environmental Biology of Fishes 71(4):413-417.

Nichols, O. C., and H. L. Kite-Powell. **2005**. Analysis of risk to North Atlantic right whales (*Eubalaena glacialis*) from shipping traffic in Cape Cod Bay, NEFSC contract report NA03NMF4720489, Woods Hole, MA.

Nieukirk, S. L., K. M. Stafford, D. K. Mellinger, R. P. Dziak, and C. G. Fox. **2004**. Low-frequency whale and seismic airgun sounds recorded in the mid-Atlantic Ocean. The Journal of the Acoustical Society of America 115:1832-1843.

Nollens, H., J. Wellehan, L. Archer, L. Lowenstine, and F. Gullande. **2010**. Detection of a respiratory coronavirus from tissues archived during a pneumonia epizootic in free-ranging Pacific harbor seals *Phoca vitulina richardsii*. Diseases of Aquatic Organisms 90(2):113-120.

Nordøy, E. S., L. P. Folkow, V. Potelov, V. Prischemikhin, and A. S. Blix. **2008**. Seasonal distribution and dive behaviour of harp seals (*Pagophilus groenlandicus*) of the White Sea–Barents Sea stock. Polar Biology 31(9):1119-1135.

Noren, D., A. Johnson, D. Rehder, and A. Larson. **2009**. Close approaches by vessels elicit surface active behaviors by southern resident killer whales. Endangered Species Research 8:179-192.

Norris, K., B. Wursig, R. Wells, M. Wursig, S. Brownlee, C. Johnson, and J. Solow. **1985**. The behavior of the Hawaiian spinner dolphin, *Stenella longirostris*., Southwest Fisheries Science Center Administrative Report, Southwest Fisheries Science Center, La Jolla, CA.

Norris, T. F. **1994**. Effects of boat noise on the acoustic behavior of humpback whales. The Journal of the Acoustical Society of America 96:3251-3251.

Norris, T., T. M. Yack, S. Martin, J. N. Oswald, A. J. Cummins, and L. Thomas. **2010**. The acoustic ecology and behavior of minke whales (*Balaenoptera acutorostrata*) near tropical and subtropical North Pacific Islands. The Journal of the Acoustical Society of America 128:2414-2414.

Northridge, S. P., M. L. Tasker, A. Webb, and J. M. Williams. **1995**. Distribution and relative abundance of harbour porpoises (*Phocoena phocoena*), white-beaked dolphins (*Lagenorhynchus albirostris*), and minke whales (*Balaenoptera acutorostrata*) around the British Isles. ICES Journal of Marine Science 52:55-66.

Nosal, E. M. **2011**. Passive acoustic tracking of minke whales, Final report, Grant number N00014101352, University of Hawaii, Honolulu, Hawaii.

Nowacek, D. **2008**. A bio-energetic model for North Atlantic right whales: Locomotion, anatomy and diving behavior. ONR Contract number N000140610576/N000140811222. Avalible at http://www.onr.navy.mil/en/Science-Technology/Departments/Code-32/All-Programs/Atmosphere-Research-322/Marine-Mammals-Biology/~/media/Files/Annual%20Reports/MMB/FY08/mbnowac1.ashx.

Nowacek, S. M., R. S. Wells, E. C. G. Owen, T. R. Speakman, R. O. Flamm, and D. P. Nowacek. **2004**. Florida manatees, *Trichechus manatus latirostris*, respond to approaching vessels. Biological Conservation 119(4):517-523.

Nyman, M., J. Koistinen, M. L. Fant, T. Vartiainen, and E. Helle. **2002**. Current levels of DDT, PCB and trace elements in the Baltic ringed seals (*Phoca hispida baltica*) and grey seals (*Halichoerus grypus*). Environmental Science & Technology 119(3):399-412.

Obura, D. O., A. Harvey, T. Young, M. M. Eltayeb, and R. von Brandis. **2010**. Hawksbill turtles as significant predators on hard coral. Coral Reefs 29(3):759-759.

O'Corry-Crowe, G. M., K. K. Martien, and B. L. Taylor. **2003**. The analysis of population genetic structure in Alaskan harbor seals, *Phoca vitulina*, as a framework for the identification of management stocks., Administrative Report, Southwest Fisheries Science Center, La Jolla, CA.

Odell, D. K. 1975. Status of aspects of the life hisotry of the bottlenose dolphin, *Tursiops truncatus*, in Flordia. Journal of the Fisheries Research Board of Canada 32(7):1055-1058.

Office of Protected Resources-NOAA Fisheries **n.d.** Blue Whale (*Balaenoptera musculus*). http://www.nmfs.noaa.gov/pr/species/mammals/cetaceans/bluewhale.htm. Accessed 12 April 2011.

Olsen, E., W. P. Budgell, E. Head, L. Kleivane, L. Nøttestad, R. Prieto, M. A. Silva, H. Skov, G. A. Víkingsson, and N. Øien. **2009**. First satellite-tracked long-distance movement of a sei whale (*Balaenoptera borealis*) in the North Atlantic. Aquatic Mammals 35:313-318.

Olson, P. **2009**. Pilot whales *Globicephala melas* and *G. macrorhynchus*, In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, pp. 847-852, Academic Press, San Diego.

Ordonez, C., S. Troeng, A. Ruiz, E. Possardt, D. Godfrey, P. Meylan, A. Meylan, and N. Decastro. **2008**. Hawksbill (*Eretmochelys imbricata*) and leatherback (*Dermochelys coriacea*) turtle nesting at Comarca Ngoebe-Bugle Chiriqui Beach, Escudo de Veraguas and Bastimentos Island National Marine Park, Panama, In: Proceedings of the 25th annual symposium on sea turtle biology and conservation, p. 32, NOAA Tech Memo NMFS-SEFSC-582.

Oremland, M. S., B. M. Allen, P. J. Clapham, M. J. Moore, C. Potter, and J. G. Mead. **2009**. Mandibular fractures in short-finned pilot whales, *Globicephala macrorhynchus*. Marine Mammal Science 26(1):1-16.

Oremus, M., R. Gales, M. L. Dalebout, N. Funahashi, T. Endo, T. Kage, D. Steel, and S. C. Baker. **2009**. Worldwide mitochondrial DNA diversity and phylogeography of pilot whales (*Globicephala* spp.). Biological Journal of the Linnean Society 98:729-744.

Orós, J., O. M. González-Díaz, and P. Monagas. **2009**. High levels of polychlorinated biphenyls in tissues of Atlantic turtles stranded in the Canary Islands, Spain. Chemosphere 74(3):473-478.

O'Shea, T. J., and L. B. Poché Jr. **2006**. Aspects of underwater sound communication in Florida manatees (*Trichechus manatus latirostris*). Journal of Mammalogy 87(6):1061–1071.

O'Shea, T. J., and H. I. Kochman. **1990**. Florida manatees: Distribution, geographically referenced data sets, and ecological and behavioral aspects of habitat use. Florida Marine Research Publications. St. Petersburg FL.

Osterhaus, A. D., H. Yang, H. E. M. Spijkers, J. Groen, J. S. Teppema, and G. van Steenis. **1985**. The isolation and partial characterization of a highly pathogenic herpesvirus from the harbor seal (*Phoca vitulina*). Archives of Virology 86(3-4):239-251.

Osterhaus, A. D. M. E., H. W. Broeders, I. K. G. Visser, J. S. Teppema, and E. J. Vedder 1990. Isolation of an orthopoxvirus from pox-like lesions of a grey seal (*Halichoerus grypus*). Veterinary Record 127(4):91-92.

Oswald, J. N., W. W. L. Au, and F. Duennebier. **2011**. Minke whale (*Balaenoptera acutorostrata*) boings detected at the Station ALOHA Cabled Observatory. The Journal of the Acoustical Society of America 129:3353-3360.

Oswald, J., T. Norris, W. Au, and F. Duennebier 2008. Temporal and geographic patterns in the occurrence and distribution of minke whale (*Balaenoptera acutorostrata*) boings in the central and western North Pacific. The Journal of the Acoustical Society of America 124:2484-2484.

Overholtz, W. J., and G. T. Waring. **1991**. Diet composition of pilot whales *Globicephala* sp. and common dolphins *Delphinus delphis* in the Mid-Atlantic Bight during spring 1989. Fishery Bulletin 89(4):723–728.

Overholtz, W. J., and J. R. Nicolas. **1979**. Apparent feeding by the fin whale (*Balaenoptera physalus*) and humpback whale (*Megaptera novaeangliae*), on the American sand lance (*Ammodytes americanus*), in the Northwest Atlantic. Fishery Bulletin US 77:285-287.

O'Shea, T. J., and R. L. Brownell Jr. **1994**. Organochlorine and metal contaminants in baleen whales: a review and evaluation of conservation implications. Science of the Total Environment 154(2-3):179-200.

Pace, R. M., III. Research Wildlife Biologist, NMFS Northeast Fisheries Science Center, August 4, **2011**, Personal communication, email.

Pacini, A. F., P. E. Nachtigall, L. N. Kloepper, M. Linnenschmidt, A. Sogorb, and S. Matias. **2010**. Audiogram of a formerly stranded long-finned pilot whale (*Globicephala melas*) measured using auditory evoked potentials. The Journal of Experimental Biology 213:3138–3143.

Palka, D. **1995**. Influences on spatial patterns of Gulf of Maine harbor porpoises, In: Blix, A.S., L. Walløe, and Ø. Ulltang (eds.) Whales, Seals, Fish and Man, pp. 69-75, Elsevier Science, Amsterdam.

Palka, D., A. Read, and C. Potter. **1997**. Summary of knowledge of white-sided dolphins (*Lagenorhynchus acutus*) from US and Canadian Atlantic waters. Reports of the International Whaling Commission 47:729-734.

Palka, D., M. Johnson, and D. Palka. **2007**. Cooperative research to study dive patterns of sperm whales in the Atlantic Ocean, U. S. Department of the Interior, Minerals Management Service Gulf of Mexico OCS Regional Office New Orleans LA USA. 40 p.

Palka, D. L. **2006**. Summer abundance estimates of cetaceans in the US North Atlantic Navy Operating Areas, NEFSC Reference Document 06-03. 41 p.

Palmer, K., S. E. Parks, D. Wiley, M. Weinrich, J. Tackaberry, A. Bocconcelli, and A. Friedlaender. **2010**. Vocal behavior of North Atlantic humpback whales during reunion events on Stellwagen Bank. The Journal of the Acoustical Society of America 127:1894-1894.

Palsbøll, P. J., P. J. Clapham, D. K. Mattila, F. Larsen, R. Sears, H. R. Siegismund, J. Sigurjónsson, O. Vasquez, and P. Arctander. **1995**. Distribution of mtDNA haplotypes in North Atlantic humpback whales: the influence of behaviour on population structure. Marine Ecology Progress Series 116(1):1-10.

Panigada, S., G. N. D. Sciara, and M. Z. Panigada. **2006**. Fin whales summering in the Pelagos Sanctuary (Mediterranean Sea): Overview of studies on habitat use and diving behaviour. Chemistry and Ecology 22:255–263.

Panigada, S., M. Zanardelli, M. MacKenzie, C. Donovan, F. Mélin, and P. S. Hammond. **2008**. Modelling habitat preferences for fin whales and striped dolphins in the Pelagos Sanctuary (Western Mediterranean Sea) with physiographic and remote sensing variables. Remote Sensing of Environment 112(8):3400–3412.

Park, J.-S., O. I. Kalantzi, D. Kopec, and M. Petreas. **2009**. Polychlorinated biphenyls (PCBs) and their hydroxylated metabolites (OH-PCBs) in livers of harbor seals (*Phoca vitulina*) from San Francisco Bay, California and Gulf of Maine. Marine Environmental Research 67(3):129-135.

Parker, D. M., W. J. Cooke, and G. H. Balazs. **2005**. Diet of oceanic loggerhead sea turtles (*Caretta caretta*) in the central North Pacific. Fishery Bulletin 103:142-152.

Parker, D. M., and G. H. Balazs. **2008**. Diet of the oceanic green turtle, *Chelonia mydas*, in the North Pacific, In: Proceedings of the 25th annual symposium on sea turtle biology and conservation, pp. 94-95, NOAA Tech Memo NMFS-SEFSC-582.

Parks, S. E., D. R. Ketten, J. T. O’Malley, and J. Arruda. **2007**. Anatomical predictions of hearing in the North Atlantic right whale. The Anatomical Record: Advances in Integrative Anatomy and Evolutionary Biology 290(6):734–744.

Parks, S. E., C. W. Clark, and P. L. Tyack. **2008**. Long- and short-term changes in right whale acoustic behavior in increased low-frequency noise. Bioacoustics 17(1-3):179-180.

Parks, S. E., M. Johnson, D. Nowacek, and P. L. Tyack. **2010**. Individual right whales call louder in increased environmental noise. Biology Letters 7(1):33-35.

Parks, S. E., I. Urazghildiiev, and C. W. Clark. **2009**. Variability in ambient noise levels and call parameters of North Atlantic right whales in three habitat areas. The Journal of the Acoustical Society of America 125:1230.

Parks, S. E., and P. L. Tyack. **2005**. Sound production by North Atlantic right whales (Eubalaena glacialis) in surface active groups. The Journal of the Acoustical Society of America 117(5):3297-3306.

Parsons, E. C. M., A. J. Wright, and M. Gore. **2008**. The nature of humpback whale *Megaptera novaeangliae* song. Journal of Marine Animals and Their Ecology 1(1):22–31.

Patrician, M. R., and R. D. Kenney. **2010**. Using the Continuous Plankton Recorder to investigate the absence of North Atlantic right whales (*Eubalaena glacialis*) from the Roseway Basin foraging ground. Journal of Plankton Research 32(12):1685-1695.

Payne, P. M., and L. A. Selzer. **1989**. The distribution, abundance and selected prey of the harbor seal, *Phoca vitulina concolor*, in southern New England. Marine Mammal Science 5(2):173-192.

Payne, P. M., D. N. Wiley, S. B. Young, S. Pittman, P. J. Clapham, and J. W. Jossi. **1990**. Recent fluctuations in the abundance of baleen whales in the southern Gulf of Maine in relation to changes in selected prey. Fishery Bulletin 88(4):687–696.

Pendleton, D. E. **2010**. Models of North Atlantic right whale habitat: Tools for science and conservation. Ph.D. thesis. Cornell University, Ithaca, NY.

Pendleton, D., A. Pershing, M. Brown, C. Mayo, R. Kenney, N. Record, and T. Cole. **2009**. Regional-scale mean copepod concentration indicates relative abundance of North Atlantic right whales. Marine Ecology Progress Series 378:211-225.

Pereira, J. N. D. S. G. **2008**. Field notes on Risso's dolphin (*Grampus griseus*) distribution, social ecology, behaviour, and occurrence in the Azores. Aquatic Mammals 34(4):426-435.

Perrin, W. F., E. D. Mitchell, J. G. Mead, D. K. Caldwell, and P. J. H. van Bree. **1981**. *Stenella clymene*, a rediscovered tropical dolphin of the Atlantic. Journal of Mammalogy 62(3):582-598.

Perrin, W. F., and R. L. Brownell Jr. **2009**. Minke Whales *Balaenoptera acutorostrata* and *B. bonaerensis*, In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, pp. 733-735, Academic Press, San Diego.

Perrin, W. F., and J. E. Powers. **1980**. Role of a nematode in natural mortality of spotted dolphins. The Journal of Wildlife Management 44(4):960–963.

Perrin, W. F., S. Leatherwood, and A. Collet. **1994a**. Fraser's dolphin *Lagenodelphis hosei* (Fraser 1956), In: Ridgway, S.H. and R. Harrison (eds.) Handbook of Marine Mammals. Vol. 5: The First Book of Dolphins, pp. 225- 240, Academic Press, New York.

Perrin, W. F., and J. W. Gilpatrick. **1994**. Spinner dolphin, *Stenella longirostris*, In: Ridgway, S.H. and R. Harrison (eds.) Handbook of Marine Mammals. Vol. 5: The First Book of Dolphins, pp. 99-128, Academic Press, New York, NY.

Perrin, W. F. **2001**. *Stella attenuata*. Mammalian Species 683:1-8.

Perrin, W. F. **2009a**. Common dolphins, In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, p. 1316, Academic Press, San Diego.

Perrin, W. F. **2009b**. Pantropical spotted dolphin *Stenella attenuata*, In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, pp. 819-821, Academic Press, San Diego.

Perrin, W. F. **2009c**. Spinner Dolphin *Stenella longirostris*, In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, pp. 1100-1103, Academic Press, San Diego.

Perrin, W. F., D. K. Caldwell, and M. C. Caldwell. **1994b**. Atlantic spotted dolphin *Stenella frontalis* (G. Cuvier, 1829), In: Ridgway, S.H. and R. Harrison (eds.) Handbook of Marine Mammals. Vol. 5: The First Book of Dolphins, p. 418, Academic Press, New York.

Perrin, W. F., C. E. Wilson, and F. I. Archer. **1994c**. Striped Dolphin, *Stenella coeruleoalba*, In: Ridgway, S.H. and R. Harrison (eds.) Handbook of Marine Mammals. Vol. 5: The First Book of Dolphins, pp. 129-159, Academic Press, New York.

Perrin, W. F., and A. A. Hohn. **1994**. Pantropical Spotted Dolphin, *Stenella attenuata*, In: Ridgway, S.H. and R. Harrison (eds.) Handbook of Marine Mammals. Vol. 5: The First Book of Dolphins, Academic Press, New York.

Perry, E. A., D. J. Boness, and R. C. Fleischer. **1998**. DNA fingerprinting evidence of nonfilial nursing in grey seals. Molecular Ecology 7:81-85.

Perry, E. A., G. B. Stenson, S. E. Bartlett, W. S. Davidson, and S. M. Carr. **2000**. DNA sequence analysis identifies genetically distinguishable populations of harp seals (Pagophilus groenlandicus) in the northwest and northeast Atlantic. Marine Biology 137(1):53–58.

Perryman, W. L. **2009**. Melon-headed Whale, In: Encyclopedia of Marine Mammals, pp. 719-721, Academic Press, Burlington, MA.

Perryman, W. L., D. W. K. Au, S. Leatherwood, and T. A. Jefferson. **1994**. Melon-headed Whale, *Peponocephala electra*, In: Handbook of Marine Mammals, Vol. 5: The first book of dolphins, vol. 5, pp. 363-386, Academic Press, New York.

Perryman, W. L., and T. C. Foster. **1980**. Preliminary report on predation by small whales , mainly the false killer whale, *Pseudorca crassidens*, on dolphins (*Stenella* spp. and *Delphinus delphis*) in the eastern tropical Pacific., NOAA SWFSC Admin. Rep., NOAA.

Pershing, A., N. Record, B. Monger, C. Mayo, M. Brown, T. Cole, R. Kenney, D. Pendleton, and L. Woodard. **2009**. Model-based estimates of right whale habitat use in the Gulf of Maine. Marine Ecology Progress Series 378:245-257.

Pettis, H. M., R. M. Rolland, P. K. Hamilton, S. Brault, A. R. Knowlton, and S. D. Kraus. **2004**. Visual health assessment of North Atlantic right whales (*Eubalaena glacialis*) using photographs. Canadian Journal of Zoology 82(1):8-19.

Pfeiffer, C. J., L. V. Sharova, and L. Gray. **2000**. Functional and ultrastructural cell pathology induced by fuel oil in cultured dolphin renal cells. Ecotoxicology and Environmental Safety 47:210-217.

Philippa, J. D. W., M. W. G. van de Bildt, T. Kuiken, P. ’t Hart, and A. D. M. E. Osterhaus. **2009**. Neurological signs in juvenile harbour seals (*Phoca vitulina*) with fatal phocine distemper. Veterinary Record 164:327-331.

Phillips, R., C. Niezrecki, and D. O. Beusse. **2006**. Theoretical detection ranges for acoustic based manatee avoidance technology. The Journal of the Acoustical Society of America 120:153-163.

Piche, C., L. Measures, C. Bedard, and S. Lair. **2010**. Bronchoalveolar lavage and pulmonary histopathology in harp seals (Phoca groenlandica) experimentally infected with Otostrongylus circumlitus. Journal of Wildlife Diseases 46(2):409-421.

Pierce, G. J. et al. **2008**. Bioaccumulation of persistent organic pollutants in female common dolphins (*Delphinus delphis*) and harbour porpoises (*Phocoena phocoena*) from western European seas: Geographical trends, causal factors and effects on reproduction and mortality. Environmental Pollution 153(2):401-415.

Pierce, G. J., M. B. Santos, R. J. Reid, I. A. P. Patterson, and H. M. Ross. **2004**. Diet of minke whales *Balaenoptera acutorostrata* in Scottish (UK) waters with notes on strandings of this species in Scotland 1992–2002. Journal of the Marine Biological Association of the United Kingdom 84(06):1241-1244.

Pierce, G. J., P. M. Thompson, A. Miller, J. S. W. Diack, D. Miller, and P. R. Boyle. **1991**. Seasonal variation in the diet of common seals (*Phoca vitulina*) in the Moray Firth area of Scotland. Journal of Zoology 223(4):641-652.

Pierce, G. J., M. B. Santos, C. Smeenk, A. Saveliev, and A. F. Zuur. **2007**. Historical trends in the incidence of strandings of sperm whales (*Physeter macrocephalus*) on North Sea coasts: An association with positive temperature anomalies. Fisheries Research 87(2-3):219-228.

Pike, D. A. **2009**. Do green turtles modify their nesting seasons in response to environmental temperatures? Chelonian Conservation and Biology 8(1):43-47.

Pike, D. A., R. L. Antworth, and J. C. Stiner. **2006**. Earlier nesting contributes to shorter nesting seasons for the loggerhead seaturtle, *Caretta caretta*. Journal of Herpetology 40(1):91-94.

Pilot, M., M. E. Dahlheim, and A. R. Hoelzel. **2010**. Social cohesion among kin, gene flow without dispersal and the evolution of population genetic structure in the killer whale (*Orcinus orca*). Journal of Evolutionary Biology 23(1):20-31.

Pinela, A. M., S. Quérouil, S. Magalhaes, M. A. Silva, R. Prieto, J. A. Matos, and R. S. Santos. **2009**. Population genetics and social organization of the sperm whale (*Physeter macrocephalus*) in the Azores inferred by microsatellite analyses. Canadian Journal of Zoology 87(9):802-813.

Pitman, R. L., and C. Stinchcomb. **2002**. Rough toothed dolphins (*Steno bredanensis*) as predators of Mahimahi (*Coryphaena hippurus*.). Pacific Science 56(4):447-450.

Podesta, M., A. D Amico, G. Pavan, A. Drougas, A. Komnenou, and N. Portunato. **2005**. A review of Cuvier's beaked whale strandings in the Mediterranean Sea. Journal of Cetacean Research and Management 7(3):251-261.

Pomeroy, P. P., S. S. Anderson, S. D. Twiss, and B. J. McConnell. **1994**. Dispersion and site fidelity of breeding female grey seals (*Halichoerus grypus*) on North Rona, Scotland. Journal of Zoology 233(3):429-447.

Pomeroy, P. P., S. D. Twiss, and C. D. Duck. **2000**. Expansion of a grey seal (*Halichoerus grypus*) breeding colony: changes in pupping site use at the Isle of May, Scotland. Journal of Zoology 250(1):1-12.

Potvin, J., J. A. Goldbogen, and R. E. Shadwick. **2009**. Passive versus active engulfment: verdict from trajectory simulations of lunge-feeding fin whales *Balaenoptera physalus*. Journal of the Royal Society Interface 6(40):1005-1025.

Prieto, R., M. A. Silva, I. Cascão, M. J. Cruz, C. I. B. Oliveira, G. Waring, and J. Gonçalves. **2010**. The importance of oceanic fronts in the Labrador Sea to North Atlantic sei whales (*Balaenoptera borealis*). Clues from satellite telemetry, In: Proceedings from the Arctic Frontiers Conference, Trømso 24-29 January 2010.

Prime, J. H., and P. S. Hammond. **1990**. The diet of grey seals from the south-western North Sea assessed from analyses of hard parts found in faeces. Journal of Applied Ecology 27(2):435-447.

Pritchard, P. C. H. **1982**. Nesting of the leatherback turtle, *Dermochelys coriacea* in Pacific Mexico, with a new estimate of the world population status. Copeia 741-747.

Pritchard, P. C. H. **1976**. Post-nesting movements of marine turtles (Cheloniidae and Dermochelyidae) tagged in the Guianas. Copeia 1976(4):749-754.

Punt, A. E., N. A. Friday, and T. D. Smith. **2007**. Reconciling data on the trends and abundance of North Atlantic humpback whales within a population modelling framework. Journal of Cetacean Research and Management 8(2):145-159.

Pusineri, C., O. Chancollon, J. Ringelstein, and V. Ridoux. **2008**. Feeding niche segregation among the Northeast Atlantic community of oceanic top predators. Marine Ecology Progress Series 361:21-34.

Putman, N. F., T. J. Shay, and K. J. Lohmann. **2010**. Is the geographic distribution of nesting in the Kemp's ridley turtle shaped by the migratory needs of offspring?, In: From the symposium "Integrative Migration Biology" presented at the annual meeting of the Society for Integrative and Comparative Biology, January 3–7, 2010, at Seattle, Washington.

Ramp, C., M. Berube, W. Hagen, and R. Sears. **2006**. Survival of adult blue whales *Balaenoptera musculus* in the Gulf of St. Lawrence, Canada. Marine Ecology Progress Series 319:287-295.

Ramp, C., M. Bérubé, P. Palsbøll, W. Hagen, and R. Sears. **2010**. Sex-specific survival in the humpback whale *Megaptera novaeangliae* in the Gulf of St. Lawrence, Canada. Marine Ecology Progress Series 400:267–276.

Rankin, S., T. F. Norris, M. A. Smultea, C. Oedekoven, A. M. Zoidis, E. Silva, and J. Rivers. **2007**. A visual sighting and acoustic detections of minke whales, *Balaenoptera acutorostrata* (Cetacea: Balaenopteridae), in nearshore Hawaiian waters. Pacific Science 61(3):395-398.

Rankin, S., J. N. Oswald, and J. Barlow. **2008**. Acoustic behavior of dolphins in the Pacific ocean: implications for using passive acoustic methods for population studies. Canadian Acoustics 36:88-92.

Rankin, S., and J. Barlow. **2007a**. Sounds recorded in the presence of Blainville's beaked whales, *Mesoplodon densirostris*, near Hawai'i. Journal of the Acoustical Society of America 122(1):42-45.

Rankin, S., and J. Barlow. **2007b**. Vocalizations of the sei whale *Balaenoptera borealis* off the Hawai'ian Islands. Bioacoustics 16:137-145.

Rankin, S., and J. Barlow. **2005**. Source of the North Pacific "boing" sound attributed to minke whales. The Journal of the Acoustical Society of America 118(5):3346-3351.

Rayne, S., M. G. Ikonomou, P. S. Ross, G. M. Ellis, and L. G. Barrett-Lennard. **2004**. PBDEs, PBBs, and PCNs in three communities of free-ranging killer whales (*Orcinus orca*) from the Northeastern Pacific Ocean. Environmental Science & Technology 38(16):4293-4299.

Read, A. J. **2007**. Do circle hooks reduce the mortality of sea turtles in pelagic longlines? A review of recent experiments. Biological Conservation 135(2):155-169.

Read, A. J., P. Drinker, and S. Northridge. **2006**. Bycatch of marine mammals in US and global fisheries. Conservation Biology 20(1):163-169.

Read, A. J., and A. A. Hohn. **1995**. Life in the fast lane: the life history of harbor porpoises from the Gulf of Maine. Marine Mammal Science 11(4):423-440.

Read, A., K. W. Urian, B. Wilson, and D. M. Waples. **2003**. Abundance of bottlenose dolphins in the bays, sounds, and estuaries of North Carolina. Marine Mammal Science 19(1):59-73.

Record, N. R., A. J. Pershing, B. Monger, D. E. Pendleton, C. Mayo, and C. Chen. **2008**. Operational forecasts of right whale distributions from satellite and oceanographic data using dynamical models, American Geophysical Union, 2000 Florida Ave., N. W. Washington DC 20009 USA.

Reed, J. Z., D. J. Tollit, P. M. Thompson, and W. Amos. **1997**. Molecular scatology: the use of molecular genetic analysis to assign species, sex and individual identity to seal faeces. Molecular Ecology 6:225-234.

Reeves, R. R., G. K. Silber, and P. M. Payne. **1998**. Draft recovery plan for the fin whale *Balaenoptera physalus* and sei whale *Balaenoptera borealis*., NMFS Office of Protected Resources. 66 p.

Reeves, R. R., T. D. Smith, E. A. Josephson, P. J. Clapham, and G. Woolmer. **2004**. Historical observations of humpback and blue whales in the North Atlantic Ocean: clues to migratory routes and possibly additional feeding grounds. Marine Mammal Science 20(4):774-786.

Reeves, R. R., B. S. Stewart, P. J. Clapham, and J. A. Powell. **2002**. Sea Mammals of the World, A. C. Black and Sons, New York.

Reeves, R. R., and H. Whitehead. **1997**. Status of the sperm whale, *Physeter macrocephalus*, in Canada. Canadian Field-Naturalist 111(2):293-307.

Reeves, R. R., E. Mitchell, and H. Whitehead. **1993**. Status of northern bottlenose whale, *Hyperoodon ampullatus*. Canadian Field-Naturalist 107:490-508.

Reeves, R. R., R. Rolland, and P. J. Clapham (eds.). **2001**. Causes of reproductive failure in North Atlantic right whales: new avenues of research. Report of a workshop held 26-28 April 2000, Falmouth, Massachusetts. NEFSC Reference Document 01-16. 46 p.

Reeves, R. R., C. Smeenk, R. L. Brownell, and C. C. Kinze. **1999a**. Atlantic White-sided Dolphin, *Lagenorhynchus acutus*, In: Ridgway S. H.. and R. Harrison (eds.) Handbook of Marine Mammals Vol. 6: The Second Book of Dolphins and Porpoises, pp. 31-56, Academic Press, San Diego.

Reeves, R. R., C. Smeenk, C. C. Kinze, R. L. Brownell, and J. Lien. **1999b**. White-beaked Dolphin *Lagenorhynchus albirostris*, In: Ridgway S. H. and R. Harrison (eds.) Handbook of Marine Mammals Vol. 6: The Second Book of Dolphins and Porpoises, pp. 1-30, Academic Press, San Diego.

Reeves, R. R., B. S. Stewart, and S. Leatherwood. **1992**. Harbor Seal *Phoca vitulina*, In: The Sierra Club Handbook of Seals and Sirenians, pp. 130-142, Sierra Club Books, San Francisco.

Reeves, R. R., and E. Mitchell. **1988**. Killer whale sightings and takes by American pelagic whalers in the North Atlantic. Rit Fiskideildar 11:7-23.

Rehtanz, M., S.-je Ghim, W. E. McFee, B. Doescher, G. Lacave, P. A. Fair, J. S. Reif, G. D. Bossart, and A. B. Jenson. **2010**. Papillomavirus antibody prevalence in free-ranging and captive bottlenose dolphins (*Tursiops truncatus*). Journal of Wildlife Diseases 46(1):136-145.

Reich, K. J., K. A. Bjorndal, A. B. Bolten, M. G. Frick, B. E. Witherington, M. D. Arendt, and A. L. Segars. **2010**. Oceanic and neritic foraging strategies of adult loggerhead turtles: A sexually dimorphic feeding strategy?, In: Proceedings of the 28th annual symposium on sea turtle biology and conservation, p. 138, NOAA Tech Memo NMFS-SEFSC-602.

Reich, K. J., K. A. Bjorndal, and A. B. Bolten. **2007**. The "lost years" of green turtles: using stable isotopes to study cryptic lifestages. Biology Letters 3(6):712-714.

Reich, K. J., K. A. Bjorndal, M. G. Frick, B. E. Witherington, C. Johnson, and A. B. Bolten. **2009**. Polymodal foraging in adult female loggerheads (*Caretta caretta*). Marine Biology 157(1):113-121.

Reijnders, P. J. H., S. M. J. M. Brasseur, and E. H. W. G. Meesters. **2010**. Earlier pupping in harbour seals, *Phoca vitulina*. Biology Letters 6(6):854-857.

Reilly, S. B. et al. **2008a**. *Balaenoptera musculus* (Blue Whale, Pygmy Blue Whale, Sibbald's Rorqual, Sulphur-bottom Whale). In: IUCN 2010. IUCN Red List of Threatened Species. Version 2010.4. www.iucnredlist.org. Accessed 2 April 2011.

Reilly, S. B. et al. **2008b**. *Balaenoptera borealis* In: IUCN 2010. IUCN Red List of Threatened Species. Version 2010.4. www.iucnredlist.org. Accessed 11 April 2011.

Reilly, S. B. et al. **2008c**. *Balaenopteraera physalus*. In: IUCN 2010. IUCN Red List of Threatened Species. Version 2010.4. www.iucnredlist.org. Accessed 10 June 2010.

Reilly, S. B. et al. **2008d**. *Megaptera novaeangliae*. In: IUCN 2010. IUCN Red List of Threatened Species. Version 2010.4. www.iucnredlist.org. Accessed 20 November 2010.

Renaud, M. L., and J. A. Williams. **2005**. Kemp's ridley sea turtle movements and migrations. Chelonian Conservation and Biology 4:808-816.

Rendell, L. E., and H. Whitehead. **2003**. Vocal clans in sperm whales (*Physeter macrocephalus*). Proceedings of the Royal Society B: Biological Sciences 270(1512):225-231.

Rendell, L., and H. Whitehead. **2004**. Do sperm whales share coda vocalizations? Insights into coda usage from acoustic size measurement. Animal Behaviour 67(5):865-874.

Rendell, L., and H. Whitehead. **2005**. Spatial and temporal variation in sperm whale coda vocalizations: stable usage and local dialects. Animal Behaviour 70(1):191-198.

Rendell, L. E., J. N. Matthews, A. Gill, J. C. D. Gordon, and D. W. Macdonald 1999. Quantitative analysis of tonal calls from five odontocete species, examining interspecific and intraspecific variation. Journal of Zoology 249:403-410.

Rendell, L. E., and J. C. D. Gordon. **1999**. Vocal response of long-finned pilot whales (*Globicephala melas*) to military sonar in the Ligurian Sea. Marine Mammal Science 15(1):198-204.

Renner, S. C. **2005**. An analysis of harbor seal (*Phoca Vitulina*) and gray seal (*Halichoerus Grypus*) haul-out patterns, behavior budgets, and aggressive interactions on Mount Desert Rock, Maine. MS thesis. University of Maine, Orono, ME. 90 p.

Resendes, A., S. Almeria, J. P. Dubey, E. Obon, C. Juan-Salles, E. Degollada, F. Alegre, O. Cabezon, S. Pont, and M. Domingo. **2002**. Disseminated Toxoplasmosis in a Mediterranean pregnant Risso's dolphin (*Grampus griseus*) with tranplacental fetal infection. Journal of Parasitology 88(5):1029-1032.

Reynolds III, J. E., and J. R. Wilcox. **1994**. Observations of Florida manatees (*Trichechus manatus latirostris*) around selected power plants in winter. Marine Mammal Science 10(2):163-177.

Reynolds III, J. E., J. A. Powell, and C. R. Taylor. **2009**. Manatees *Trichechus manatus, T. senegalensis*, and *T. inunguis*., In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, pp. 682-691, Academic Press Inc.

Rhinelander, M. Q., and S. M. Dawson. **2004**. Measuring sperm whales from their clicks: Stability of interpulse intervals and validation that they indicate whale length. The Journal of the Acoustical Society of America 115:1826-1831.

Rice, D. W. **1998**. Marine mammals of the world: Systematics and distribution, Special Publication Number 4. The Society for Marine Mammology, Lawrence, KS. 231 pp.

Richard, K. R., M. C. Dillon, H. Whitehead, and J. M. Wright. **1996**. Patterns of kinship in groups of free-living sperm whales (*Physeter macrocephalus*) revealed by multiple molecular genetic analyses. Proceedings of the National Academy of Sciences of the United States of America 93(16):8792-8795.

Richardson, J. I., D. B. Hall, P. A. Mason, K. M. Andrews, R. Bjorkland, Y. Cai, and R. Bell. **2006**. Eighteen years of saturation tagging data reveal a significant increase in nesting hawksbill sea turtles (*Eretmochelys imbricata*) on Long Island, Antigua. Animal Conservation 9(3):302–307.

Richter, C., S. Dawson, and E. Slooten. **2006**. Impacts of commercial whale watching on male sperm whales at Kaikoura, New Zealand. Marine Mammal Science 22(1):46-63.

Ridgway, S. H., E. G. Wever, J. G. McCormick, J. Palin, and J. H. Anderson. **1969**. Hearing in the giant sea turtle, *Chelonia mydas*. Proceedings of the National Academy of Sciences of the United States of America 64(3):884-890.

Ridgway, S. H., and D. A. Carder. **2001**. Assessing hearing and sound production in cetaceans not available for behavioral audiograms: Experiences with sperm, pygmy sperm, and gray whales. Aquatic Mammals 27(3):267-276.

Ries, E. H., L. A. Hiby, and P. J. H. Reijnders. **1998**. Maximum likelihood population size estimation of harbour seals in the Dutch Wadden Sea based on a mark-recapture experiment. Journal of Applied Ecology 35:332-339.

Rijks, J. M., J. I. Hoffman, T. Kuiken, A. Osterhaus, and W. Amos. **2008**. Heterozygosity and lungworm burden in harbour seals (*Phoca vitulina*). Heredity 100(6):587–593.

Ringelstein, J., C. Pusineri, S. Hassani, L. Meynier, R. Nicol, and V. Ridoux. **2006**. Food and feeding ecology of the striped dolphin, *Stenella coeruleoalba*, in the oceanic waters of the north-east Atlantic. Journal of the Marine Biological Association of the United Kingdom 86(4):909-918.

Ripcke, M. **2011**. Preliminary results of the effects of simulated pile driving vibrations on sea turtle embryogenesis, Conference paper presented at the 31st Annual Sea Turltle Symposium, San Diego, CA.

Rivalan, P., A.-C. Prévot-Julliard, R. Choquet, R. Pradel, B. Jacquemin, and M. Girondot. **2005**. Trade-off between current reproductive effort and delay to next reproduction in the leatherback sea turtle. Oecologia 145(4):564-574.

Rizzo, L. Y., and D. Schulte. **2009**. A review of humpback whales’ migration patterns worldwide and their consequences to gene flow. Journal of the Marine Biological Association of the United Kingdom 89(05):995-1002.

Robbins, J. **2007**. Structure and dynamics of the Gulf of Maine humpback whale population. Ph.D. thesis. University of St. Andrews, Aberdeen, Scotland. 179 p.

Robbins, J., and D. Mattila. **2001**. Monitoring entanglements of humpback whales (*Megaptera novaeangliae*) in the Gulf of Maine on the basis of caudal peduncle scarring, International Whaling Commission Scientific Committee, 135 Station Road, Impington, Cambridge, UK.

Roberts, M. A., C. J. Andersen, B. Stender, A. Segars, D. Whittaker, J. M. Grady, and J. M. Quattro. **2005**. Estimated contribution of Atlantic coastal loggerhead turtle nesting populations to offshore feeding aggregations. Conservation Genetics 6(1):133-139.

Robertson, K. M., and S. J. Chivers. **1997**. Prey occurence in pantropical spotted dolphins, *Stenella attenuata*, from the eastern tropical Pacific. Fishery Bulletin 95:334-348.

Robinson, B. H., and J. E. Craddock. **1983**. Mesopelagic fishes eaten by Fraser’s dolphin, *Lagenodelphis hosei*. Fishery Bulletin 81(2):283-289.

Robinson, K. P., M. J. Tetley, and E. G. Mitchelson-Jacob. **2009**. The distribution and habitat preference of coastally occurring minke whales (*Balaenoptera acutorostrata*) in the outer southern Moray Firth, northeast Scotland. Journal of Coastal Conservation 13(1):39-48.

Robinson, K. P., and M. J. Tetley. **2005**. Environmental factors affecting the fine-scale distribution of minke whales (*Balaenoptera acutorostrata*) in a dynamic coastal ecosystem. Poster presentation, ICES CM.

Robinson, K. P., and M. J. Tetley. **2007**. Behavioural observations of foraging minke whales (*Balaenoptera acutorostrata*) in the outer Moray Firth, north-east Scotland. Journal of the Marine Biological Association of the United Kingdom 87(01):85-86.

Roe, H. S. J. **1969**. The food and feeding habits of the sperm whales (*Physeter catodon* L.) taken off the west coast of Iceland. ICES Journal of Marine Science 33(1):93-102.

Rolland, R. M., P. K. Hamilton, S. D. Kraus, B. Davenport, R. M. Gillett, and S. K. Wasser. **2007**. Faecal sampling using detection dogs to study reproduction and health in North Atlantic right whales (*Eubalaena glacialis*). Journal of Cetacean Research and Management 8(2):121-125.

Rolland, R. M., K. E. Hunt, S. D. Kraus, and S. K. Wasser. **2005**. Assessing reproductive status of right whales (*Eubalaena glacialis*) using fecal hormone metabolites. General and Comparative Endocrinology 142(3):308-317.

Rommel, S. A. et al. **2006**. Elements of beaked whale anatomy and diving physiology and some hypothetical causes of sonar-related stranding. Journal of Cetacean Research and Management 7(3):189-209.

Rosel, P. E., L. Hansen, and A. A. Hohn. **2009**. Restricted dispersal in a continuously distributed marine species: common bottlenose dolphins *Tursiops truncatus* in coastal waters of the western North Atlantic. Molecular Ecology 18:5030-5045.

Rosel, P. E., R. Tiedemann, and M. Walton. **1999**. Genetic evidence for limited trans-Atlantic movements of the harbor porpoise Phocoena phocoena. Marine Biology 133(4):583-291.

Rosenbaum, H. C. et al. **2000**. World-wide genetic differentiation of *Eubalaena*: questioning the number of right whale species. Molecular Ecology 9(11):1793-1802.

Rosenfeld, M., M. George, and J. M. Terhune. **1988**. Evidence of autumnal harbour seal, *Phoca vitulina*, movement from Canada to the United States. Canadian Field-Naturalist 102(3):527-529.

Ross, P. S., R. L. DeSwart, H. H. Timmerman, P. J. H. Reijnders, J. G. Vos, H. Van Loveren, and Osterhaus. **1996**. Suppression of natural killer cell activity in harbour seals (*Phoca vitulina*) fed Baltic Sea herring. Aquatic Toxicology 34(1):71-84.

Ross, P. S., G. M. Ellis, M. G. Ikonomou, L. G. Barrett-Lennard, and R. F. Addison. **2000**. High PCB concentrations in free-ranging Pacific killer whales, *Orcinus orca*: effects of age, sex and dietary preference. Marine Pollution Bulletin 40(6):504-515.

Rossong, M. A., and J. M. Terhune. **2009**. Source levels and communication-range models for harp seal (Pagophilus groenlandicus) underwater calls in the Gulf of St. Lawrence, Canada. Canadian Journal of Zoology 87:609-617.

Rostal, D. C., D. W. Owens, J. S. Grumbles, D. S. MacKenzie, and M. S. J. Amoss. **1998**. Seasonal reproductive cycle of the Kemp's ridley sea turtle (*Lepidochelys kempii*). General and Comparative Endocrinology 109:232-243.

Rough, V. **1991**. Muskeget gray seals, winter and spring, 1991., Report to the Marine Mammal Commission. Contract T10155615 28 p.

Ruiz-Cooley, R. I., D. Gendron, S. Aguíñiga, S. Mesnick, and J. D. Carriquiry. **2004**. Trophic relationships between sperm whales and jumbo squid using stable isotopes of C and N. Marine Ecology Progress Series 277:275-283.

Runge, M. C., C. A. Sanders-Reed, C. A. Langtimm, and C. J. Fonnesbeck. **2007**. A quantitative threats analysis for the Florida manatee (*Trichechus manatus latirostris*). U.S. Geological SurveyOpen-File Report 2007-1086. 34 pp.

Russell, D. J., and G. H. Balazs. **2009**. Dietary shifts by green turtles (*Chelonia mydas*) in the Kāne ‘ohe Bay region of the Hawaiian Islands: A 28-year study. Pacific Science 63(2):181-192.

Salberg, A. B., T. A. Oigard, G. B. Stenson, T. Haug, and K. T. Nilssen. **2009**. Estimation of seal pup production from aerial surveys using generalized additive models. Canadian Journal of Fisheries and Aquatic Sciences 66(5):847-858.

Samuel, Y., S. J. Morreale, C. W. Clark, C. H. Greene, and M. E. Richmond. **2005**. Underwater, low-frequency noise in a coastal sea turtle habitat. The Journal of the Acoustical Society of America 117(3):1465-1472.

Santidrián Tomillo, P., F. V. Paladino, J. S. Suss, and J. R. Spotila. **2010**. Predation of leatherback turtle hatchlings during the crawl to the water. Chelonian Conservation and Biology 9(1):18-25.

Santidrián Tomillo, P., J. S. Suss, B. P. Wallace, K. D. Magrini, G. Blanco, F. V. Paladino, and J. R. Spotila. **2009**. Influence of emergence success on the annual reproductive output of leatherback turtles. Marine Biology 156(10):2021-2031.

Santoro, M., G. Hernandéz, M. Caballero, and F. García. **2008**. Potential bacterial pathogens carried by nesting leatherback turtles (*Dermochelys coriacea*) in Costa Rica. Chelonian Conservation and Biology 7(1):104-108.

Santos, B. M., and G. J. Pierce. **2005**. A note on niche overlap in teuthophagous whales in the northern northeast Atlantic, In: Cephalopod biology, recruitment and culture: International Cephalopod Symposium and Workshop, 17-21 February 2003, p. 291.

Santos, M. B. et al. **1999**. Stomach contents of sperm whales *Physeter macrocephalus* stranded in the North Sea 1990-1996. Marine Ecology Progress Series 183:281-294.

Santos, M. B. et al. **2002**. Additional notes on stomach contents of sperm whales *Physeter macrocephalus* stranded in the north-east Atlantic. Journal of the Marine Biological Association of the United Kingdom 82(03):501-507.

Santos, M. B., G. J. Pierce, J. Herman, A. Lopez, A. Guerra, E. Mente, and M. R. Clarke. **2001**. Feeding ecology of Cuvier’s beaked whale (*Ziphius cavirostris*): a review with new information on the diet of this species. Journal of the Marine Biological Association of the United Kingdom 81(04):687-694.

Santos, M. B., G. J. Pierce, J. A. Learmonth, R. J. Reid, H. M. Ross, I. A. P. Patterson, D. G. Reid, and D. Beare. **2004**. Variability in the diet of harbor porpoises (*Phocoena phocoena*) in Scottish waters 1992-2003. Marine Mammal Science 20(1):1-27.

Santos, M. B., G. J. Pierce, A. Lopez, R. J. Reid, V. Ridoux, and E. Mente. **2006**. Pygmy sperm whales *Kogia breviceps* in the northeast Atlantic: New information on stomach contents and strandings. Marine Mammal Science 22(3):600-616.

Santos, M. B., G. J. Pierce, H. M. Ross, R. J. Reid, and B. Wilson. **1994**. Diets of small cetaceans from the Scottish coast. ICES CM 1994/N:11.

Sardi, K. A., M. T. Weinrich, and R. C. Connor. **2005**. Social interactions of humpback whale (*Megaptera novaeangliae*) mother/calf pairs on a North Atlantic feeding ground. Behaviour 142(6):731-750.

Sarti Martinez, A. L. **2000**. IUCN Red List Assessment- *Dermochelys coriacea*. in: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.1. www.iucnredlist.org. Accessed 2 Nov 2010.

Sasso, C. R., J. Braun-McNeill, L. Avens, and S. P. Epperly. **2006**. Effects of transients on estimating survival and population growth in juvenile loggerhead turtles. Marine Ecology Progress Series 324:287-292.

Sasso, C. R., and W. N. Witzell. **2006**. Diving behaviour of an immmature Kemp’s ridley turtle (*Lepidochelys kempii*) from Gullivan Bay, Ten Thousand islands, south-west Florida. Journal of the Marine Biological Association of the United Kingdom 86(4):919-925.

Sasso, C. R., and S. P. Epperly. **2007**. Survival of pelagic juvenile loggerhead turtles in the open ocean. Journal of Wildlife Management 71(6):1830-1835.

Schaefer, A. M., H.-C. W. Stavros, G. D. Bossart, P. A. Fair, J. D. Goldstein, and J. S. Reif. **2011**. Associations between mercury and hepatic, renal, endocrine, and hematological parameters in Atlantic bottlenose dolphins (*Tursiops truncatus*) along the eastern coast of Florida and South Carolina. Archives of Environmental Contamination and Toxicology. DOI 10.1007/s00244-011-9651-5.

Schaeff, C. M., S. D. Kraus, M. W. Brown, and B. N. White. **1992**. Assessment of the population structure of western North Atlantic right whales (*Eubalaena glacialis*) based on sighting and mtDNA data. Canadian Journal of Zoology 71:339-345.

Scheidat, M., J. Tougaard, S. Brasseur, J. Carstensen, T. van Polanen Petel, J. Teilmann, and P. Reijnders. **2011**. Harbour porpoises (*Phocoena phocoena*) and wind farms: a case study in the Dutch North Sea. Environmental Research Letters 6:1-10.

Schick, R. S., P. N. Halpin, A. J. Read, C. K. Slay, S. D. Kraus, B. R. Mate, M. F. Baumgartner, J. J. Roberts, B. D. Best, and C. P. Good. **2009**. Striking the right balance in right whale conservation. Canadian Journal of Fisheries and Aquatic Sciences 66(9):1399-1403.

Schilling, M. R., I. Seipt, M. T. Weinrich, S. E. Frohock, A. E. Kuhlberg, and P. J. Clapham. **1992**. Behavior of individually identified sei whales, *Balaenoptera borealis*, during an episodic influx into the southern Gulf of Maine in 1986. Fishery Bulletin 90(4):749-755.

Schlundt, C. E., J. J. Finneran, D. A. Carder, and S. H. Ridgway. **2000**. Temporary shift in masked hearing thresholds of bottlenose dolphins, *Tursiops truncatus*, and white whales, *Delphinapterus leucas*, after exposure to intense tones. Journal of the Acoustical Society of America 107(6):3496-3508.

Schmid, J. R., and W. N. Witzell. **2006**. Seasonal migrations of immature Kemp's ridley turtles (*Lepidochelys kempii* Garman) along the west coast of Florida. Gulf of Mexico Science 24(1/2):28-40.

Schmid, J. R. **1998**. Marine turtle populations on the west-central coast of Florida: results of tagging studies at Cedar Keys, Florida, 1986-1995. Fishery Bulletin 96(3):589-602.

Schmid, J. R., A. B. Bolten, K. A. Bjorndal, W. J. Lindberg, H. F. Percival, and P. D. Zwick. **2003**. Home range and habitat use by Kemp's ridley turtles in west central Florida. Journal of Wildlife Management 67(1):196-206.

Schmid, J. R., W. N. Witzell, D. S. Addison, and D. W. Ceilley. **2008**. Essential habitat for Kemp's ridley turtles in Western Florida, In: Proceedings of the 25th annual symposium on sea turtle biology and conservation, p. 20, NOAA Tech Memo NMFS-SEFSC-582.

Schmid, J. R., A. B. Bolten, K. A. Bjorndal, and W. J. Lindberg. **2002**. Activity patterns of Kemp's ridley turtles, *Lepidochelys kempii* , in the coastal waters of the Cedar Keys, Florida. Marine Biology 140(2):215-228.

Schmidt, V., T. C. Weber, D. N. Wiley, and M. P. Johnson. **2010**. Underwater tracking of humpback whales (*Megaptera novaeangliae*) with high-frequency pingers and acoustic recording tags. Oceanic Engineering, IEEE Journal of 35(4):821-836.

Schreer, J. F., J. L. Lapierre, and M. O. Hammill. **2010**. Stomach temperature telemetry reveals that harbor seal (*Phoca vitulina*) pups primarily nurse in the water. Aquatic Mammals 36(3):270-277.

Schreer, J. F., and K. M. Kovacs. **1997**. Allometry of diving capacity in air-breathing vertebrates. Canadian Journal of Zoology 75:339-358.

Schulz, T. M., H. Whitehead, and L. Rendell. **2008**. Off-axis effects on the multi-pulse structure of sperm whale coda clicks. The Journal of the Acoustical Society of America 125(3):1768-1773.

Schumacher, U., S. Zahler, H. P. Horny, G. Heidemann, K. Skirnisson, and U. Welsch. **1993**. Histological investigations on the thyroid glands of marine mammals (*Phoca vitulina*, *Phocoena phocoena*) and the possible implications of marine pollution. Journal of Wildlife Diseases 29(1):103-108.

Schwacke, L. H. et al. **2011**. Anaemia, hypothyroidism and immune suppression associated with polychlorinated biphenyl exposure in bottlenose dolphins (*Tursiops truncatus*). Proceedings of the Royal Society B: Biological Sciences. doi: 10.1098/rspb.2011.0665.

Schwacke, L. H., E. O. Voit, L. J. Hansen, R. S. Wells, G. B. Mitchum, A. A. Hohn, and P. A. Fair. **2002**. Probabilistic risk assessment of reproductive effects of polychlorinated biphenyls on bottlenose dolphins (*Tursiops truncatus*) from the southeast United States coast. Environmental Toxicology and Chemistry 21(12):2752-2764.

Schwartz, F. J. **1995**. Florida manatees, *Trichechus manatus* (Sirenia: Trichechidae), in North Carolina 1919-1994. Brimleyana 22:53-60.

Scott, J. A. **2006**. Use of satellite telemetry to determine ecology and management of loggerhead turtle (*Caretta caretta*) during the nesting season in Georgia. MS thesis. University of Georgia. 162 p.

Scott, T. M., and S. S. Sadove. **1997**. Sperm whale, *Physeter macrocephalus* sightings in the shallow shelf waters off Long Island, New York. Marine Mammal Science 13(2):317-321.

Sears, R. **2002**. Long range movements of a blue whale (*Balaenoptera musculus*) between the Gulf of St. Lawrence and West Greenland. Marine Mammal Science 18(1):281-285.

Sears, R., F. W. Wenzel, and J. M. Williamson. **1987**. The blue whale: A catalogue of individuals from the western North Atlantic (Gulf of St. Lawrence), Mingan Island Cetacean Study, St. Lambert, Quebec. 27 p.

Sears, R., J. M. Williamson, F. W. Wenzel, M. Bérubé, D. Gendron, and P. Jones. **1990**. Photographic identification of the blue whale (*Balaenoptera musculus*) in the Gulf of St. Lawrence, Canada. Report of the International Whaling Commission (Special Issue 12) 335-342.

Sears, R., and J. Calambokidis. **2002**. COSEWIC Assessment and Update Status Report on the Blue Whale (*Balaenoptera musculus*) in Canada, Committee on the Status of Endangered Wildlife in Canada. Ottawa. 32 p.

Seipt, I. E., P. J. Clapham, C. A. Mayo, and M. P. Hawvermale. **1990**. Population characteristics of individually identified fin whales *Balaenoptera physalus* in Massachusetts Bay. Fishery Bulletin 88(2):271-278.

Sella, K. N., M. Salmon, and B. E. Witherington. **2006**. Filtered streetlights attract hatchling marine turtles. Chelonian Conservation and Biology 5(2):255-261.

Selzer, L. A., and P. M. Payne. **1988**. The distribution of white-sided (*Lagenorhynchus acutus*) and common dolphins (*Delphinus delphis*) vs. environmental features of the continental shelf of the northeastern United States. Marine Mammal Science 4(3):141-153.

Seminoff, J. A., T. T. Jones, T. Eguchi, M. Hastings, and D. R. Jones. **2009**. Stable carbon and nitrogen isotope discrimination in soft tissues of the leatherback turtle (*Dermochelys* *coriacea*): Insights for trophic studies of marine turtles. Journal of Experimental Marine Biology & Ecology 381(1):33-41.

Seney, E. E., and J. A. Musick. **2008**. Loggerhead and Kemp’s ridley diet in Virginia, U.S.A. during 1980 to 2002: Changing with the times?, In: Proceedings of the 24th annual symposium on sea turtle biology and conservation, p. 153, NOAA Tech Memo NMFS-SEFSC-567.

Seney, E. E., and A. M. Landry. **2008**. Movements of Kemp’s ridley sea turtles nesting on the upper Texas coast: implications for management. Endangered Species Research 4(1-2):73-84.

Seney, E. E., and A. M. J. Landry. **2010**. Kemp’s ridley migratory paths and foraging areas in the northwestern Gulf of Mexico, In: Proceedings of the 28th annual symposium on sea turtle biology and conservation, NOAA Tech Memo NMFS-SEFSC-602.

Sergeant, D. E., D. K. Caldwell, and M. C. Caldwell. **1973**. Age, growth and maturity of bottlenosed dolphin (*Tursiops truncatus*) from northeast Florida. Journal of the Fisheries Research Board of Canada 30:1009-1011.

Sergeant, D. E., D. J. St. Aubin, and J. R. Geraci. **1980**. Life history and northwest Atlantic status of the Atlantic white-sided dolphin, *Lagenorhynchus acutus*. Cetology 37:1-12.

Shamblin, B. M., M. G. Dodd, A. B. Meylan, D. A. Bagley, L. M. Ehrhart, C. Johnson, M. Blackburn, R. E. Martin, B. Libert, and C. J. Nairn. **2010**. Fine scale mitochondrial DNA population structure in loggerhead turtles (*Caretta caretta*) nesting on Florida’s Atlantic coast: The clinal shift revisited, In: Proceedings of the 28th annual symposium on sea turtle biology and conservation, p. 62, NOAA Tech Memo NMFS-SEFSC-602.

Shane, S. H., R. S. Wells, and B. Wursig. **1986**. Ecology, behavior and social organization of the bottlenose dolphin: a review. Marine Mammal Science 2(1):34-63.

Shaver, D. J., and T. Wibbels. **2007**. Head-starting Kemp’s ridleys, In: Plotkin, P. T. (editor), Biology and Conservation of Ridley Sea Turtles, Johns Hopkins University Press, Baltimore, MD. p. 297-323.

Shaver, D. J. **1991**. Feeding ecology of wild and head-started Kemp’s ridley sea turtles in south Texas waters. Journal of Herpetology 25(3):327-334.

Shaver, D. J., B. A. Schroeder, R. A. Byles, P. M. Burchfield, J. Pena, R. Marquez, and H. J. Martinez. **2005**. Movements and home ranges of adult male Kemp’s ridley sea turtles (*Lepidochelys kempii*) in the Gulf of Mexico investigated by satellite telemetry. Chelonian Conservation and Biology 4(4):817-827.

Shaver, D. J., and C. Rubio. **2008**. Post-nesting movement of wild and head-started Kemp’s ridley sea turtles *Lepidochelys kempii* in the Gulf of Mexico. Endangered Species Research 4:43-55.

Shaw, S., M. L. Berger, D. Brenner, L. Tao, Q. Wu, and K. Kannan. **2009**. Specific accumulation of perfluorochemicals in harbor seals (*Phoca vitulina concolor*) from the northwest Atlantic. Chemosphere 74(8):1037-1043.

Sherrill-Mix, S.A., M.C. James, and R.A. Myers. **2008**. Migration cues and timing in leatherback sea turtles. Behavioral Ecology 19(2):231-236.

Shigenaka, G. **2003**. Oil and sea turtles: Biology, planning and response, NOAA National Ocean Service Office of Response and Restoration. 116 p. Availble online at http://response.restoration.noaa.gov/book_shelf/35_turtle_complete.pdf.

Shoop, C.R., and R.D. Kenney **1992**. Seasonal distributions and abundances of loggerhead and leatherback sea turtles in waters of the northeastern United States. Herpetological Monographs 6:43-67.

Sidorovskaia, N., P. Schexnayder, A. Ekimov, J. Sabatier, G. E. Ioup, and J. W. Ioup. **2009**. Rhythmic analysis of sperm whale broadband acoustic signals. The Journal of the Acoustical Society of America 125:2738-2738.

Sigler, M. F., C. R. Lunsford, J. M. Straley, and J. B. Liddle. **2008**. Sperm whale depredation of sablefish longline gear in the northeast Pacific Ocean. Marine Mammal Science 24(1):16–27.

Silber, G. K., and S. Bettridge. **2010**. Vessel operations in right whale protection areas in 2009. NOAA Tech Memo NMFS-OPR-44. 52p.

Simmonds, M. P., and L. F. Lopez-Jurado. **1991**. Whales and the military. Nature 351(6326):448-448.

Simon, M. J., T. K. Kristensen, O. S. Tendal, C. C. Kinze, and S. Tougaard. **2003**. *Gonatus fabricii* (Mollusca, Theuthida) as an important food source for sperm whales (*Physeter macrocephalus*) in the Northeast Atlantic. Sarsia 88(3):244-246.

Simon, M. J., F. Ugarte, M. Wahlberg, and L. A. Miller. **2006**. Icelandic killer whales Orcinus orca use a pulsed call suitable for manipulating the schooling behaviour of herring Clupea harengus. Bioacoustics 16:57-74.

Simon, M., P. McGregor, and F. Ugarte. **2007a**. The relationship between the acoustic behaviour and surface activity of killer whales (*Orcinus orca*) that feed on herring (*Clupea harengus*). Acta Ethologica 10(2):47-53.

Simon, M., K. M. Stafford, K. Beedholm, C. M. Lee, and P. T. Madsen. **2010**. Singing behavior of fin whales in the Davis Strait with implications for mating, migration and foraging. The Journal of the Acoustical Society of America 128(5):3200-3210.

Simon, M., M. Wahlberg, and L. A. Miller. **2007b**. Echolocation clicks from killer whales (*Orcinus orca*) feeding on herring (*Clupea harengus*). Journal of the Acoustical Society of America 121(2):749-752.

Sivertsen, S. P., T. Pedersen, U. Lindstrøm, and T. Haug. **2006**. Prey partitioning between cod (*Gadus morhua*) and minke whale (*Balaenoptera acutorostrata*) in the Barents Sea. Marine Biology Research 2(2):89-99.

Sjare, B., and G. B. Stenson. **2009**. Changes in the reproductive parameters of female harp seals (*Pagophilus groenlandicus*) in the Northwest Atlantic. ICES Journal of Marine Science 67:304-315.

Sjöberg, M., and J. P. Ball. **2000**. Grey seal, *Halichoerus grypus*, habitat selection around haulout sites in the Baltic Sea: bathymetry or central-place foraging? Canadian Journal of Zoology 78:1661-1667.

Skaug, H.J., M. Bérubé, M.B. Rew, and P.J. Palsbøll **2008**. Genetic analyses reveal promiscuous mating in female minke whales, *Balaenoptera acutorostrata*. Journal of Cetacean Research and Management 9:249-251.

Skaug, H.J., N. Øien, T. Schweder, and G. Bøthun 2004. Abundance of minke whales (*Balaenoptera acutorostrata*) in the Northeast Atlantic: variability in time and space. Canadian Journal of Fisheries and Aquatic Sciences 61:870-886.

Skinner, J. P. **2006**. Physical and behavioral development of nursing harbor seal (*Phoca vitulina*) pups in Maine. MS thesis. The University of Maine, Orono, ME. 140 p.

Skov, H., T. Gunnlaugsson, W.P. Budgell, J. Horne, L. Nřttestad, E. Olsen, H. Sřiland, G. Víkingsson, and G. Waring. **2008**. Small-scale spatial variability of sperm and sei whales in relation to oceanographic and topographic features along the Mid-Atlantic Ridge. Deep Sea Research Part II: Topical Studies in Oceanography 55(1-2):254-268.

Skov, H., J. Durinck, F. Danielsen, and D. Bloch. **1995**. Co-occurrence of cetaceans and sea birds in the northeast Atlantic. Journal of Biogeography 22:71-88.

Smith, J. **2010**. The ecology of Cuvier’s beaked whale, *Ziphius cavirostris* (Cetacea: Ziphiidae), in the Bay of Biscay. Ph. D. thesis. University of Southampton, School of Ocean and Earth Science, South Hapmton, UK. 229 p.

Smolowitz, R., H. Haas, H. O. Milliken, M. Weeks, and E. Matzen. **2010**. Using sea turtle carcasses to assess the conservation potential of a Turtle Excluder Dredge. North American Journal of Fisheries Management 30(4):993-1000.

Smout, S., and U. Lindstrøm. **2007**. Multispecies functional response of the minke whale *Balaenoptera acutorostrata* based on small-scale foraging studies. Marine Ecology Progress Series 341:277-291.

Snoddy, J. E., M. Landon, G. Blanvillain, and A. Southwood. **2009**. Blood biochemistry of sea turtles captured in gillnets in the lower Cape Fear River, North Carolina, USA. Journal of Wildlife Management 73(8):1394-1401.

Snover, M. L. **2002**. Growth and ontogeny of sea turtles using skeletochronology: Methods, validation and application to conservation. PhD thesis. Duke University. 156 p.

Snover, M. L., A. A. Hohn, and L. B. Crowder. **2008**. Age and growth in juvenile Kemp's ridley (Lepidochelys kempii) sea turtles using skeletochronology, In: Proceedings of the 24th annual symposium on sea turtle biology and conservation, vol. NOAA Tech Memo NMFS-SEFSC-567, p. 43.

Snover, M. L., and A. A. Hohn. **2004**. Validation and interpretation of annual skeletal marks in loggerhead (*Caretta caretta*) and Kemp's ridley (*Lepidochelys kempii*) sea turtles. Fishery Bulletin 102(4):682-692.

Sobin, J. M., and A. D. Tucker. **2010**. Dive behavior of internesting loggerhead turtles (*Caretta caretta*) and green turtles (*Chelonia mydas*) and risks of boat impact, In: Proceedings of the 28th annual symposium on sea turtle biology and conservation, p. 53, NOAA Tech Memo NMFS-SEFSC-602.

Solow, A. R., K. A. Bjorndal, and A. B. Bolten. **2002**. Annual variation in nesting numbers of marine turtles: the effect of sea surface temperature on re-migration intervals. Ecology Letters 5(6):742-746.

Southall, B. L., R. Braun, F. M. D. Gulland, A. D. Heard, R. W. Baird, S. M. Wilkin, and T. K. Rowles. **2006**. Hawaiian melon-headed whale (*Peponacephala electra*) mass stranding event of July 3-4, 2004. NOAA Tech Memo NMFS-OPR-31.

Southall, B. L. et al. **2007**. Marine mammal noise exposure criteria: Initial scientific recommendations. Aquatic Mammals 33(4):411-521.

Spaulding, E., M. Robbins, T. Calupca, C. Clark, T. Tremblay, A. Waack, A. Warde, J. Kemp, and K. Newhall. **2009**. An autonomous, near real time buoy system for automatic detection of North Atlantic right whale calls. The Journal of the Acoustical Society of America 125:2615-2615.

Speakman, T., E. Zolman, J. Adams, R. H. Defran, D. Laska, L. Schwacke, J. Craigie, and P. Fair. **2006**. Temporal and spatial aspects of bottlenose dolphin occurrence in coastal and estuarine waters near Charleston, South Carolina., NOAA Tech Memo NOS-NCCOS-37. 56 p.

Spitz, J., E. Richard, L. Meynier, C. Pusineri, and V. Ridoux. **2006**. Dietary plasticity of the oceanic striped dolphin, *Stenella coeruleoalba*, in the neritic waters of the Bay of Biscay. Journal of Sea Research 55:309-320.

Spotila, J. R., A. E. Dunham, A. J. Leslie, A. C. Steyermark, P. T. Plotkin, and F. V. Paladino. **1996**. Worldwide population decline of *Dermochelys coriacea*: are leatherback turtles going extinct?. Chelonian Conservation and Biology 2(2):209-222.

Stacy, B. A., A. M. Foley, E. Greiner, L. H. Herbst, A. Bolten, P. Klein, C. A. Manire, and E. R. Jacobson. **2010**. Spirorchiidiasis in stranded loggerhead *Caretta caretta* and green turtles

Chelonia mydas in Florida (USA): host pathology and significance. Diseases of Aquatic Organisms 89(3):237-259.

Stafford, K. M., J. J. Citta, S. E. Moore, M. Daher, and J. George. **2009**. Environmental correlates of blue and fin whale call detections in the North Pacific Ocean from 1997 to 2002. Marine Ecology Progress Series 395:37-53.

Stafford, K. M., S. E. Moore, and C. G. Fox. **2005**. Diel variation in blue whale calls recorded in the eastern tropical Pacific. Animal Behaviour 69(4):951-958.

Stamper, M. A., B. R. Whitaker, and T. D. Schofield. **2006**. Case study: morbidity in a pygmy sperm whale *Kogia breviceps* due to ocean-bourne plastic. Marine Mammal Science 22(3):719-722.

Stanley, H. F., S. Casey, J. M. Carnahan, S. Goodman, J. Harwood, and R. K. Wayne. **1996**. Worldwide patterns of mitochondrial DNA differentiation in the harbor seal (*Phoca vitulina*). Molecular Biology and Evolution 13(2):368-382.

Starbird, C. H. **1993**. Internesting movements and behavior of hawksbill sea turtles (*Eretmochelys imbricata*) around Buck Island Reef National Monument, St. Croix, United States Virgin Islands. MS. thesis San Jose State University, San Jose, California.

Stavros, H. C. W., G. D. Bossart, T. C. Hulsey, and P. A. Fair. **2007**. Trace element concentrations in skin of free-ranging bottlenose dolphins (*Tursiops truncatus*) from the southeast Atlantic coast. Science of the Total Environment 388(1-3):300-315.

Steiner, W. W. **1981**. Species-specific differences in pure tonal whistle vocalizations of five western North Atlantic dolphin species. Behavioral Ecology and Sociobiology 9:241-246.

Stenson, G. B. **2006**. Estimates of human induced mortality in Northwest Atlantic Harp Seals, 1952-2004, Fisheries and Oceans Canada, Science Advisory Research Document. 36 p. http://www.dfo-mpo.gc.ca/CSAS/Csas/DocREC/2005/RES2005_050_e.pdf

Stenson, G. B., M. O. Hammill, and J. W. Lawson. **1997**. Predation by harp seals in Atlantic Canada: preliminary consumption estimates for Arctic cod, capelin and Atlantic cod. Journal of the Northwest Atlantic Fisheries Society 22:137-154.

Stenson, G. B., R. A. Myers, I. H. Ni, and W. G. Warren. **1996**. Pup production of hooded seals (*Cystophora cristata*) in the northwest Atlantic. NAFO Sci. Coun. Studies 26: 105-114.

Stevick, P. T. et al. **2003a**. Segregation of migration by feeding ground origin in North Atlantic humpback whales (*Megaptera novaeangliae*). Journal of Zoology 259(03):231-237.

Stevick, P. T. et al. **2003b**. North Atlantic humpback whale abundance and rate of increase four decades after protection from whaling. Marine Ecology Progress Series 258:263-273.

Stevick, P. T., and T. W. Fernald. **1998**. Increase in extralimital records of harp seals in Maine. Northeastern Naturalist 5(1):75-82.

Stevick, P. T. et al. **2006**. Population spatial structuring on the feeding grounds in North Atlantic humpback whales (*Megaptera novaeangliae*). Journal of Zoology 270(2):244-255.

Stewart, K. R., and J. Wyneken. **2004**. Predation risk to loggerhead hatchlings at a high-density nesting beach in southeast Florida. Bulletin of Marine Science 74(2):325-335.

Stewart, K. R. **2007**. Establishment and growth of a sea turtle rookery: the population biology of the leatherback in Florida. Ph.D.thesis Duke University. 143 p.

Stewart, K., C. Johnson, and M. H. Godfrey. **2007**. The minimum size of leatherbacks at reproductive maturity, with a review of sizes for nesting females from the Indian, Atlantic and Pacific Ocean basins. Herpetological Journal 17(2):123-128.

Steyermark, A. C., K. Williams, J. R. Spotila, F. V. Paladino, D. C. Rostal, S. J. Morreale, M. T. Koberg, and R. Arauz-Vargas. **1996**. Nesting leatherback turtles at Las Baulas National Park, Costa Rica. Anidamiento de las tortugas baula en el Parque Nacional Marino Las Baulas, Costa Rica. Chelonian Conservation and Biology 2(2):173-183.

Stimpert, A. K., W. W. L. Au, S. E. Parks, and T. Hurst. **2011**. Common humpback whale (*Megaptera novaeangliae*) sound types for passive acoustic monitoring. Journal of the Acoustical Society of America 129(1):476-482.

Stimpert, A. K., W. W. L. Au, D. N. Wiley, and D. Mattila. **2008a**. Contextual sound production by tagged humpback whales (*Megaptera novaeangliae*) on a feeding and breeding ground. The Journal of the Acoustical Society of America 124:2484-2484.

Stimpert, A. K., A. Rudd, and W. W. L. Au. **2008b**. Underwater sound measurements of the Hawaii super ferry and the potential impact to humpback whales in the main Hawaiian Islands. The Journal of the Acoustical Society of America 124:2483-2483.

Stimpert, A. K., D. N. Wiley, W. W. L. Au, M. P. Johnson, and R. Arsenault. **2007**. "Megapclicks": acoustic click trains and buzzes produced during night-time foraging of humpback whales (*Megaptera novaeangliae*). Biology Letters 3(5):467-470.

Stockin, K. A., R. S. Fairbairns, E. C. M. Parsons, and D. W. Sims. **2001**. Effects of diel and seasonal cycles on the dive duration of the minke whale (*Balaenoptera acutorostrata*). Journal of the Marine Biological Association of the United Kingdom 81(01):189–190.

Stolen, M. K., W. N. Durden, and D. K. Odell. **2007**. Historical synthesis of bottlenose dolphin (*Tursiops truncatus*) stranding data in the Indian River Lagoon system, Florida, from 1977-2005. Florida Scientist 70:45-54.

Stone, C. J. **2003**. The effects of seismic activity on marine mammals in UK waters, 1998-2000, JNCC Report No. 323. 78 p. http://jncc.defra.gov.uk/pdf/jncc323.pdf.

Stone, C. J., and M. L. Tasker. **2006**. The effects of seismic airguns on cetaceans in UK waters. Journal of Cetacean Research and Management 8(3):255-263.

Stone, G. S., A. Mainwaring, S. K. Katona, J. A. Beard, and H. D. Corbett. **1988**. Surfacing, respiration, and dive characteristics of finback whales (*Balaenoptera physalus*) observed from Mount Desert Rock, Maine, Final report to NMFS, contract 40EANF501300 and 40EANF601426.

Storch, S., R. P. Wilson, Z. Hillis-Starr, and D. Adelung. **2005**. Cold-blooded divers: temperature-dependent dive performance in the wild hawksbill turtle *Eretmochelys imbricata*. Marine Ecology Progress Series 293:263-271.

Storelli, M. M., N. Zizzo, and G. O. Marcotrigiano. **1999**. Heavy metals and methylmercury in tissues of Risso’s dolphin (*Grampus griseus*) and Cuvier’s beaked whale (*Ziphius cavirostris*) stranded in Italy (South Adriatic Sea). Bulletin of Environmental Contamination and Toxicology 63(6):703-710.

Storelli, M. M., and G. O. Marcotrigiano. **2000**. Persistent organochlorine residues in Risso’s dolphins (*Grampus griseus*) from the Mediterranean Sea (Italy). Marine Pollution Bulletin 40(6):555-558.

Struntz, W. D. J., J. R. Kucklick, M. M. Schantz, P. R. Becker, W. E. McFee, and M. K. Stolen. **2004**. Persistent organic pollutants in rough-toothed dolphins (*Steno bredanensis*) sampled during an unusual mass stranding event. Marine Pollution Bulletin 48(1):164-173.

Summers, E., J. A. Estrada, and S. Zeeman. **2006**. A note on geographical and seasonal fluctuations in the isotopic composition of baleen in four North Atlantic right whales (*Eubaelaena glacialis*). Journal of Cetacean Research and Management 8(3):241-245.

Sundt, R. C., G. Dahle, and G. Nevdal. **1994**. Genetic variation in the hooded seal, (*Cystophora cristata*), based on enzyme polymorphism and multi-locus DNA fingerprinting. Hereditas 121:147-155.

Swaim, Z., A. Westgate, H. Koopman, R. Rolland, and S. Kraus. **2009**. Metabolism of ingested lipids by North Atlantic right whales. Endangered Species Research 6:259-271.

Swarthout, R. F., J. M. Keller, M. Peden-Adams, A. M. Landry, P. A. Fair, and J. R. Kucklick. **2010**. Organohalogen contaminants in blood of Kemp’s ridley (*Lepidochelys kempii*) and green sea turtles (*Chelonia mydas*) from the Gulf of Mexico. Chemosphere 78(6):731-741.

Swinton, J., J. Harwood, B. T. Grenfell, and C. A. Gilligan. **1998**. Persistence thresholds for phocine distemper virus infection in harbour seal *Phoca vitulina* metapopulations. Journal of Animal Ecology 67(1):54-68.

Szymanski, M. D., D. E. Bain, K. Kiehl, S. Pennington, S. Wong, and K. R. Henry. **1999**. Killer whale (*Orcinus orca*) hearing: auditory brainstem response and behavioral audiograms. The Journal of the Acoustical Society of America 106:1134-1141.

Tabuchi, M., N. Veldhoen, N. Dangerfield, S. Jeffries, C. C. Helbing, and P. S. Ross. **2006**. PCB-related alteration of thyroid hormones and thyroid hormone receptor gene expression in free-ranging harbor seals (*Phoca vitulina*). Environmental Health Perspectives 114(7):1024-1031.

Tamura, T., K. Konishi, T. Isoda, R. Okamoto, and T. Bando. **2009**. Prey consumption and feeding habits of common minke, sei and Bryde's whales in the western North Pacific, IWC SC/J09/JR16.

Tamura, T., and Y. Fujise. **2002**. Geographical and seasonal changes of the prey species of minke whale in the Northwestern Pacific. ICES Journal of Marine Science 59(3):516.

Tanabe, S., B. G. Loganathan, A. Subramanian, and R. Tatsukawa. **1987**. Organochlorine residues in short-finned pilot whale possible use as tracers of biological parameters. Marine Pollution Bulletin 18(10):561-563.

Tanabe, S., S. Watanabe, H. Kan, and R. Tatsukawa. **1988**. Capacity and mode of PCB metasolism in small cetceans. Marine Mammal Science 4(2):103-124.

Taylor, B. L., R. Baird, J. Barlow, S. M. Dawson, J. Ford, J. G. Mead, G. Notarbartolo di Sciara, P. Wade, and B. L. Taylor. **2008**. *Physeter macrocephalus*. In: IUCN 2011. IUCN Red List of Threatened Species. Version 2011.1. www.iucnredlist.org. Accessed on 19 June 2011.

Tetley, M. J., E. G. Mitchelson-Jacob, and K. P. Robinson. **2008**. The summer distribution of coastal minke whales (*Balaenoptera acutorostrata*) in the southern outer Moray Firth, NE Scotland, in relation to co-occurring mesoscale oceanographic features. Remote Sensing of Environment 112(8):3449-3454.

Thode, A., D. K. Mellinger, S. Stienessen, A. Martinez, and K. Mullin. **2002**. Depth-dependent acoustic features of diving sperm whales (*Physeter macrocephalus*) in the Gulf of Mexico. The Journal of the Acoustical Society of America 112:308.

Thompsen, F., K. Ludemann, R. Kafemann, and W. Piper. **2006**. Effects of offshore wind farm noise on marine mammals and fish, Biola, Hamburg, Germany on behalf of COWRIE Ltd. 62 p.

Thompson, P. M. **1989**. Seasonal changes in the distribution and composition of common seal (*Phoca vitulina*) haul-out groups. Journal of Zoology. Vol. 217 (2):281-294.

Thompson, P. M., M. A. Fedak, B. J. McConnell, and K. S. Nicholas. **1989**. Seasonal and sex-related variation in the activity patterns of common seals (*Phoca vitulina*). Journal of Applied Ecology 26(2):521-535.

Thompson, P. M., B. J. McConnell, D. J. Tollit, A. Mackay, C. Hunter, and P. A. Racey. **1996**. Comparative distribution, movements and diet of harbour and grey seals from the Moray Firth, N.E. Scotland. Journal of Applied Ecology 33:1572-1584.

Thompson, P. M., G. J. Pierce, J. R. G. Hislop, and J. S. W. Diack. **1991**. Winter foraging by common seals (*Phoca vitulina*) in relation to food availability in the inner Moray Firth, N.E. Scotland. Journal of Animal Ecology 60:283-294.

Thompson, P. M., and J. Harwood. **1990**. Methods for estimating the population size of common seals, Phoca vitulina. Journal of Applied Ecology 27:924-938.

Thompson, P. M., and D. Miller. **1990**. Summer foraging activity and movements of radio-tagged common seals (*Phoca vitulina.* L.) in the Moray Firth, Scotland. Journal of Applied Ecology 27:492-501.

Thompson, P. M., and H. Wheeler. **2008**. Photo-ID-based estimates of reproductive patterns in female harbor seals. Marine Mammal Science 24(1):138-146.

Tilbury, K. L., N. G. Adams, C. A. Krone, J. P. Meador, G. Early, and U. Varanasi 1999. Organochlorines in stranded pilot whales (*Globicephala melaena*) from the coast of Massachusetts. Archives of Environmental Contamination and Toxicology 37:125-134.

Timmel, G., S. Courbis, H. Sargeant-Green, and H. Markowitz. **2008**. Effects of human traffic on the movement patterns of Hawaiian spinner dolphins (*Stenella longirostris*) in Kealakekua Bay, Hawaii. Aquatic Mammals 34(4):402-411.

Tinker, M. T., K. Kovacs, and M. O. Hammill. **1995**. The reproductive behavior and energetics of male gray seals (*Halichoerus grypus*) breeding on a land-fast ice substrate. Behavioral Ecology and Sociobiology 36:159-170.

Todd, S., P. Stevick, J. Lien, F. Marques, and D. Ketten. **1996**. Behavioural effects of exposure to underwater explosions in humpback whales (*Megaptera novaeangliae*). Canadian Journal of Zoology 74(9):1661-1672.

Tollit, D. J., A. D. Black, P. M. Thompson, A. Mackay, H. M. Corpe, B. Wilson, S. M. Parijs, K. Grellier, and S. Parlane. **1998**. Variations in harbour seal *Phoca vitulina* diet and dive-depths in relation to foraging habitat. Journal of Zoology 244(2):209-222.

Tollit, D. J., P. M. Thomp, and S. P. R. Greenstreet. **1997**. Prey selection by harbour seals, *Phoca vitulina*, in relation to variations in prey abundance. Canadian Journal of Zoology 75:1508-1518.

Tomás, J., and J. A. Raga. **2007**. Occurrence of Kemp’s ridley sea turtle (*Lepidochelys kempii*) in the Mediterranean. Journal of the Marine Biological Association of the United Kingdom 1-3.

Torres, L. G., P. E. Rosel, C. D’Agrosa, and A. J. Read. **2003**. Improving management of overlapping bottlenose dolphin ecotypes through spatial analysis and genetics. Marine Mammal Science 19(3):502-514.

Tougaard, J., O. D. Henriksen, and L. A. Miller. **2009**. Underwater noise from three types of offshore wind turbines: Estimation of impact zones for harbor porpoises and harbor seals. The Journal of the Acoustical Society of America 125:3766-3773.

Tove, M. **1995**. Live sighting of *Mesoplodon CF. M. Mirus*, True’s beaked whale. Marine Mammal Science 11(1):80-85.

Troëng, S., P. H. Dutton, and D. Evans. **2005**. Migration of hawksbill turtles *Eretmochelys imbricata* from Tortuguero, Costa Rica. Ecography 28(3):394-402.

Tryland, M., and E. Brun. **2001**. Serum chemistry of the minke whale from the northeastern Atlantic. Journal of Wildlife Diseases 37(2):332-341.

Tucker, A. D. **2010**. Nest site fidelity and clutch frequency of loggerhead turtles are better elucidated by satellite telemetry than by nocturnal tagging efforts: Implications for stock estimation. Journal of Experimental Marine Biology and Ecology 383(1):48-55.

Tucker, S., W. Bowen, S. Iverson, W. Blanchard, and G. Stenson. **2009**. Sources of variation in diets of harp and hooded seals estimated from quantitative fatty acid signature analysis (QFASA). Marine Ecology Progress Series 384:287-302.

Tucker, S., W. D. Bowen, and S. J. Iverson. **2007**. Dimensions of diet segregation in grey seals *Halichoerus grypus* revealed through stable isotopes of carbon (delta^{13}C) and nitrogen (delta^{15}N). Marine Ecology Progress Series 339:271-282.

Tuerk, K. J. S., J. Kuck, W. E. McFee, R. S. Pugh, and P. R. Becker. **2005**. Factors influencing persistent organic pollutant concentrations in the Atlantic white-sided dolphin (*Lagenorhynchus acutus*). Environmental Toxicology and Chemistry 24(5):1079-1087.

[TEWG] Turtle Expert Working Group. **2000**. Assessment update for the Kemp’s ridley and loggerhead sea turtle populations in the western North Atlantic. NOAA Tech Memo NMFS-SEFSC-444.

Turtle Expert Working Group [TWEG]. **2007**. An assessment of the leatherback turtle population in the Atlantic Ocean, NOAA Tech Memo NMFS-SEFSC-555. 124 p

Turtle Expert Working Group [TWEG]. **2009**. An assessment of the loggerhead turtle population in the Western North Atlantic Ocean. NOAA Tech Memo NMFS-SEFC-575. 131 p.

Twiss, S. D., C. Thomas, V. Poland, J. A. Graves, and P. P. Pomeroy. **2007**. The impact of climatic variation on the opporunity for sexual section. Biology Letters 3:12-15.

Tyack, P. **1981**. Interactions between singing Hawaiian humpback whales and conspecifics nearby. Behavioral Ecology and Sociobiology 8(2):105-116.

Tyack, P. L., M. Johnson, N. Aguilar Soto, A. Sturlese, and P. T. Madsen. **2006**. Extreme diving of beaked whales. Journal of Experimental Biology 209(21):4238-4253.

Tyack, P. L. et al. **2011**. Beaked whales respond to simulated and actual navy sonar. PLoS ONE 6(3):e17009.

Tyson, R. B., D. P. Nowacek, and P. J. O. Miller. **2007**. Nonlinear phenomena in the vocalizations of North Atlantic right whales (*Eubalaena glacialis*) and killer whales (*Orcinus orca*). The Journal of the Acoustical Society of America 122(3):1365-1373.

U.S. Fish and Wildlife Service. **2001**. Florida manatee recovery plan, (*Trichechus manatus latirostris*), Third Revision., U.S. Fish and Wildlife Service, Atlanta, Georgia. 144 p. http://www.fws.gov/northflorida/Manatee/Documents/Recovery%20Plan/MRP-start.pdf.

U.S. Fish and Wildlife Service. **1999**. Hawksbill sea turtle: Multi-species recovery plan for South Florida. Available at: http://wwf.panda.org/who_we_are/wwf_offices/central_america/?96260/Hawksbill-Sea-Turtle-Multi-Species-Recovery-Plan-for-South-Florida.

U.S. Fish and Wildlife Service. **2009**. West Indian manatee (*Trichechus manatus*) Florida stock. Available at: http://www.fws.gov/northflorida/Manatee/SARS/20091230_rpt_Final_Florida_Manatee_SAR.pdf.

U.S. Fish and Wildlife Service, and NOAA. **2010**. Endangered and Threatened Species; Proposed Listing of Nine Distinct Population Segments of Loggerhead Sea Turtles as Endangered or Threatened; Proposed Rule, 50 CFR Parts 17, 223, and 224.

Uchida, A., J. Araki, S. Yuzu, K. Nagasawa, S. Kishikawa, T. Kuramochi, and M. Machida. **1998**. Histopathology of parasitic infections minke whales (*Balaenoptera acutorostrata*) from the Western North Pacific Ocean and Southern Sea of Okhotsk. Parasitology International 47(Supplement 1):354-354.

Urazghildiiev, I. R., C. W. Clark, T. P. Krein, and S. E. Parks. **2009**. Detection and recognition of North Atlantic right whale contact calls in the presence of ambient noise. Oceanic Engineering, IEEE Journal of 34(3):358-368.

Urian, K. W., A. A. Hohn, and L. J. Hansen. **1999**. Status of the photoidentification catalog of coastal bottlenose dolphins of the western North Atlantic: Report of a workshop of catalog contributors. NOAA Tech Memo NMFS-SEFSC-425.

Valsecchi, E. et al. **1997**. Microsatellite genetic distances between oceanic populations of the humpback whale (*Megaptera novaeangliae*). Molecular Biology and Evolution 14(4):355-362.

van Dam, R. P., and C. E. Diez. **1998**. Home range of immature hawksbill turtles (*Eretmochelys imbricata* (Linnaeus) at two Caribbean islands. Journal of Experimental Marine Biology and Ecology 220(1):15–24.

Vanderlaan, A. S. M. **2010**. Estimating risk to the North Atlantic right whale (*Eubalaena glacialis*) from ocean-going vessels and fishing gear. PhD thesis. Dalhousie University, Halifax, Nova Scotia, Canada. 197 p.

Vanderlaan, A. S. M., and C. T. Taggart. **2007**. Vessel collisions with whales: the probability of lethal injury based on vessel speed. Marine Mammal Science 23(1):144-156.

Vanderlaan, A. S. M., J. J. Corbett, S. L. Green, J. A. Callahan, C. Wang, R. D. Kenney, C. T. Taggart, and J. Firestone. **2009**. Probability and mitigation of vessel encounters with North Atlantic right whales. Endangered Species Research 6:273-285.

Vanderlaan, A. S. M., and C. T. Taggart. **2009**. Efficacy of a voluntary area to be avoided to reduce risk of lethal vessel strikes to endangered whales. Conservation Biology 23(6):1467-1474.

van de Merwe, J. P., M. Hodge, J. M. Whittier, K. Ibrahim, and S. Lee. **2010a**. Persistent organic pollutants in the green sea turtle *Chelonia mydas*: nesting population variation, maternal transfer, and effects on development. Marine Ecology Progress Series 403:269-278.

van de Merwe, J. P., M. Hodge, H. A. Olszowy, J. M. Whittier, and S. Y. Lee. **2010b**. Using blood samples to estimate persistent organic pollutants and metals in green sea turtles (*Chelonia mydas*). Marine Pollution Bulletin 60(4):579-588.

van de Merwe, J. P., M. Hodge, J. M. Whittier, and S. Y. Lee. **2009**. Analysing persistent organic pollutants in eggs, blood and tissue of the green sea turtle (*Chelonia mydas*) using gas chromatography with tandem mass spectrometry (GC-MS/MS). Analytical and Bioanalytical Chemistry 393(6-7):1719-1731.

Van Opzeeland, I., P. Corkeron, D. Risch, G. Stenson, and S. Van Parijs. **2009**. Geographic variation in vocalizations of pups and mother-pup behavior of harp seals *Pagophilus groenlandicus*. Aquatic Biology 6:109-120.

Van Opzeeland, I. C., P. J. Corkeron, T. Leyssen, T. Similä, and S. M. Van Parijs. **2005**. Acoustic behaviour of Norwegian killer whales, *Orcinus orca*, during carousel and seiner foraging on spring-spawning herring. Aquatic Mammals 31(1):110-119.

Vanselow, K. H., K. Ricklefs, and F. Colijn. **2009**. Solar driven geomagnetic anomalies and sperm whale (*Physeter macrocephalus*) strandings around the North Sea: An analysis of long term datasets. Open Marine Biology Journal 3:89-94.

Ver Hoef, J. M., and J. K. Jansen. **2007**. Space-time zero-inflated count models of harbor seals. Environmetrics 18(7):697-712.

Vetter, W., R. von der Recke, P. Ostrowicz, and N. Rosenfelder. **2010**. Liquid chtomatographic enantioseparation of the brominated flame retardant 2,3-dibromopropyl-2,4,6-tribromophenyls ether (DPTE) and enantiomer fractions in seal blubber. Chemosphere 78(2):134-138.

Viada, S. T., R. M. Hammer, R. Racca, D. Hannay, M. J. Thompson, B. J. Balcom, and N. W. Phillips. **2008**. Review of potential impacts to sea turtles from underwater explosive removal of offshore structures. Environmental Impact Assessment Review 28(4-5):267-285.

Vigness-Raposa, K. J., R. D. Kenney, M. L. Gonzalez, and P. V. August. **2010**. Spatial patterns of humpback whale (*Megaptera novaeangliae*) sightings and survey effort: Insight into North Atlantic population structure. Marine Mammal Science 26(1):161-175.

Vincent, C., V. Ridoux, M. A. Fedak, and S. Hassani. **2002**. Mark-recapture and satellite tracking in rehabilitated juvenile grey scals (*Halichoerus grypus*): dispersal and potential effects on wild populations. Aquatic Mammals 28(2):121–130.

Visser, F., K. L. Hartman, E. J. J. Rood, A. J. E. Hendriks, and W. J. Wolff. **2011**. Risso's dolphins alter daily resting pattern in response to whale watching at the Azores. Marine Mammal Science 27(2):366-381

Volgenau, L., S. D. Kraus, and J. Lien 1995. The impact of entanglements on two substocks of the western North Atlantic humpback whale, *Megaptera novaeangliae*. Canadian Journal of Zoology 73(9):1689-1698.

von Brandis, R. G., J. A. Mortimer, and B. K. Reilly. **2010**. In-water observations of the diving behaviour of immature hawksbill turtles, *Eretmochelys imbricata*, on a coral reef at D'Arros Island, Republic of Seychelles. Chelonian Conservation and Biology 9(1):26-32.

Vost, G., and A. Osterhaus. **1995**. Impaired cellular immune response in harbour seals (*Phoca vitulina*) feeding on environmentally contaminated herring. Clinical and Experimantal Immunology 101:480-486.

Wafo, E., L. Sarrazin, C. Diana, F. Dhermain, T. Schembri, V. Lagadec, M. Pecchia, and P. Rebouillon. **2005**. Accumulation and distribution of organochlorines (PCBs and DDTs) in various organs of *Stenella coeruleoalba* and a *Tursiops truncatus* from Mediterranean littoral environment (France). The Science of the Total Environment 348(1-3):115-127.

Wahlberg, M. **2002**. The acoustic behaviour of diving sperm whales observed with a hydrophone array. Journal of Experimental Marine Biology and Ecology 281(1-2):53-62.

Wahlberg, M., A. Frantzis, P. Alexiadou, P. T. Madsen, and B. Møhl. **2005**. Click production during breathing in a sperm whale (*Physeter macrocephalus*). The Journal of the Acoustical Society of America 118(6):3404-3407.

Waldick, R. C., S. D. Kraus, M. Brown, and B. N. White. **2002**. Evaluating the effects of historic bottleneck events: An assessment of microsatellite variability in the endangered, North Atlantic right whale. Molecular Ecology 11(11):2241-2250.

Wallace, B. P., L. Avens, J. Braun-McNeill, and C. M. McClellan. **2009**. The diet composition of immature loggerheads: Insights on trophic niche, growth rates, and fisheries interactions. Journal of Experimental Marine Biology and Ecology 373(1):50-57.

Wallace, B. P., S. S. Heppell, R. L. Lewison, S. Kelez, and L. B. Crowder. **2008**. Impacts of fisheries bycatch on loggerhead turtles worldwide inferred from reproductive value analyses. Journal of Applied Ecology 45(4):1076-1085.

Wallace, B. P., R. L. Lewison, S. L. McDonald, R. K. McDonald, C. Y. Kot, S. Kelez, R. K. Bjorkland, E. M. Finkbeiner, S. Helmbrecht, and L. B. Crowder. **2010**. Global patterns of marine turtle bycatch. Conservation Letters 3(3):131-142.

Wallace, B. P., J. A. Seminoff, S. S. Kilham, J. R. Spotila, and P. H. Dutton. **2006**. Leatherback turtles as oceanographic indicators: stable isotope analyses reveal a trophic dichotomy between ocean basins. Marine Biology 149(4):953-960.

Wallace, B. P., P. R. Sotherland, P. Santidrian Tomillo, R. D. Reina, J. R. Spotila, and F. V. Paladino. **2007**. Maternal investment in reproduction and its consequences in leatherback turtles. Oecologia 152(1):37-47.

Walsh, C. J., S. R. Leggett, B. J. Carter, and C. Colle. **2010**. Effects of brevetoxin exposure on the immune system of loggerhead sea turtles. Aquatic Toxicology 97(4):293-303.

Walton, M., R. J. Henderson, and P. P. Pomeroy. **2000**. Use of blubber fatty acid profiles to distinguish dietary differences between grey seals *Halichoerus grypus* from two UK breeding colonies. Marine Ecology Progress Series 193:201-208.

Wang, A., D. Barber, and C. J. Pfeiffer. **2001**. Protective effects of selenium against mercury toxicity in cultured Atlantic spotted dolphin (*Stenella plagiodon*) renal cells. Archives of Environmental Contamination and Toxicology 41(4):403-409.

Wang, A., and C. J. Pfeiffer. **2001**. Cytopathology induced by mercuric chloride and methylmercury in cultured renal cells of the Atlantic spotted dolphin (*Stenella plagiodon*). Journal of Submicroscopic Cytology and Pathology 33(1-2):7-16.

Wang, J. Y., and S.-C. Yang. **2006**. Unusual cetacean stranding events of Taiwan in 2004 and 2005. Journal of Cetacean Research and Management 8(3):283-292.

Ward, J., R. Morrissey, D. Moretti, N. DiMarzio, S. Jarvis, M. Johnson, P. Tyack, and C. White. **2008**. Passive acoustic detection and localization of *Mesoplodon densirostris* (Blainville's beaked whale) vocalization using distributed bottom-mounted hydrophones in conjunction with a digital tag (DTAG) recording. Canadian Acoustics 36(1):60-66.

Ward-Geiger, L., G. Silber, R. Baumstark, and T. Pulfer. **2005**. Characterization of ship traffic in right whale critical habitat. Coastal Management 33(3):263-278.

Ware, C., R. Arsenault, M. Plumlee, and D. Wiley. **2006**. Visualizing the underwater behavior of humpback whales. IEEE Computer Graphics and Applications 14-18.

Waring, G. T., C. P. Fairfield, C. M. Ruhsam, and M. Sano. **1993**. Sperm whales associated with Gulf Stream features off the north-eastern USA shelf. Fisheries Oceanography 2(2):101-105.

Waring, G. T., J. R. Gilbert, J. Loftin, and N. Cabana. **2006**. Short-term movements of radio-tagged harbor seals in New England. Northeastern Naturalist 13(1):1-14.

Waring, G. T., C. P. Fairfield, C. M. Ruhsam, and M. Sano. **1992**. Cetaceans associated with Gulf Stream features off the northeastern USA shelf., CM 1992/N:12, ICES Marine Mammals Comm.

Waring, G. T., P. T. Gerrior, P. M. Payne, and J. R. Nicolas. **1990**. Incidental takes of marine mammals in foreign fishery activities off the northeast United States, 1977-1988. Fishery Bulletin 88:347-360.

Waring, G. T., T. Hamazaki, D. Sheehan, G. Wood, and S. Baker. **2001**. Characterization of beaked whale (Ziphiidae) and sperm whale (*Physeter macrocephalus*) summer habitat in shelf-edge and deeper waters off the northeast U.S. Marine Mammal Science 17(4):703-717.

Waring, G., E. Josephson, K. Maze-Foley, and P. E. Rosel. **2011**. U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments -- 2011, NOAA Tech Memo NMFS NE, in press.

Watanabe, M., K. Kannan, A. Takahashi, B. G. Loganathan, D. K. Odell, S. Tanabe, and J. P. Giesy. **2000**. Polychlorinated biphenyls, organochlorine pesticides, tris (4-chlorophenyl) methane, and tris (4-chlorophenyl) methanol in livers of small cetaceans stranded along Florida coastal waters, USA. Environmental Toxicology and Chemistry 19(6):1566-1574.

Watkins, W. A., M. A. Daher, A. Samuels, and D. P. Gannon. **1997**. Observations of *Peponocephala electra*, the melon-headed whale, in the southeastern Caribbean. Caribbean Journal of Science 33:34-40.

Watkins, W. A., P. Tyack, K. E. Moore, and J. E. Bird. **1987**. The 20-Hz signals of finback whales (*Balaenoptera physalus*). The Journal of the Acoustical Society of America 82:1901-1912.

Watson, J. W., S. P. Epperly, A. K. Shah, and D. G. Foster. **2005**. Fishing methods to reduce sea turtle mortality associated with pelagic longlines. Canadian Journal of Fisheries and Aquatic Sciences 62(5):965-981.

Watts, P., and D. Gaskin. **1985**. Habitat index analysis of the harbor porpoise (*Phocoena phocoena*) in the southern coastal Bay of Fundy, Canada. Journal of Mammalogy 66(4):733-744.

Watwood, S. L., P. J. O. Miller, M. Johnson, P. T. Madsen, and P. L. Tyack. **2006**. Deep-diving foraging behaviour of sperm whales (*Physeter macrocephalus*). Journal of Animal Ecology 75(3):814-825.

Weijs, L., R. S. H. Yang, A. Covaci, K. Das, and R. Blust. **2010**. Physiologically based pharmacokinetic (PBPK) models for lifetime exposure to PCB 153 in male and female harbor porpoises (*Phocoena phocoena*): Model development and evaluation. Environmental Science & Technology 44:7023-7030.

Weilgart, L. S., and H. Whitehead. **1990**. Vocalizations of the North Atlantic pilot whale (*Globicephala melas*) as related to behavioral contexts. Behavioral Ecology and Sociobiology 26:399-402.

Weinrich, M., and C. Corbelli. **2009**. Does whale watching in southern New England impact humpback whale (*Megaptera novaeangliae*) calf production or calf survival? Biological Conservation 142(12):2931-2940.

Weinrich, M. T., C. R. Belt, and D. Morin. **2001**. Behavior and ecology of the Atlantic white-sided dolphin (*Lagenorhynchus acutus*) in coastal New England waters. Marine Mammal Science 17(2):231-248.

Weir, C. A. **2007**. Observations of marine turtles in relation to seismic airgun sound off Angola. Marine Turtle Newsletter 116:17-20.

Weir, C. A. **2008a**. Overt responses of humpback whales (*Megaptera novaeangliae*), sperm whales (*Physeter macrocephalus*), and Atlantic spotted dolphins (*Stenella frontalis*) to seismic exploration off Angola. Aquatic Mammals 34(1):71-83.

Weir, C. A. **2008b**. Short-finned pilot whales (*Globicephala macrorhynchus*) respond to an airgun ramp-up procedure off Gabon. Aquatic Mammals 34(3):349-354.

Weisbrod, A. V., D. Shea, M. J. Moore, and J. J. Stegeman. **2000**. Organochlorine exposure and bioaccumulation in the endangered northwest Atlantic right whale (*Eubalaena glacialis*) population. Environmental Toxicology and Chemistry 19(3):654-666.

Weisbrod, A. V., D. Shea, M. J. Moore, and J. J. Stegeman. **2001**. Species, tissue, and gender-related organochlorine bioaccumulation in white-sided dolphins, pilot whales and their common prety in the Northwest Atlantic. Marine Environmental Research 51:29-50.

Weishampel, J. F., D. A. Bagley, L. M. Ehrhart, and B. L. Rodenbeck. **2003**. Spatiotemporal patterns of annual sea turtle nesting behaviors along an East Central Florida beach. Biological Conservation 110(2):295-303.

Weishampel, J. F., D. A. Bagley, and L. M. Ehrhart. **2004**. Earlier nesting by loggerhead sea turtles following sea surface warming. Global Change Biology 10(8):1424-1427.

Weller, D. W. **1998**. Global and regional variation in the biology and behavior of bottlenose dolphins. Ph.D. Texas A & M University, College Station, Texas. 142 p.

Wells, R. S., V. Tornero, A. Borrell, A. Aguilar, T. K. Rowles, H. L. Rhinehart, S. Hofmann, W. M. Jarman, A. A. Hohn, and J. C. Sweeney. **2005**. Integrating life-history and reproductive success data to examine potential relationships with organochlorine compounds for bottlenose dolphins (*Tursiops truncatus*) in Sarasota Bay, Florida. Science of the Total Environment 349:106-119.

Wells, R. S., and M. D. Scott. **1999**. Bottlenose Dolphin *Tursiops truncatus*, In: Ridgway S. H.. and R. Harrison (eds.) Handbook of Marine Mammals Vol. 6: The Second Book of Dolphins and Porpoises, pp. 137-182, Academic Press, San Diego.

Wells, R. S., and M. D. Scott. **2009**. Common bottlenose dolphin, In: Perrin W.F., B. Wursig, and J.G.M. Thewissen (eds.) Encyclopedia of Marine Mammals, second edition, pp. 249-255, Academic Press, San Diego.

Wenzel, F., D. K. Mattila, and P. J. Clapham. **1988**. *Balaenoptera musculus* in the Gulf of Maine. Marine Mammal Science 4(2):172-175.

West, K. L., W. A. Walker, R. W. Baird, W. White, G. Levine, E. Brown, and D. Schofield 2009. Diet of pygmy sperm whales (*Kogia breviceps*) in the Hawaiian Archipelago. Marine Mammal Science 25(4):931-943.

Westgate, A. J. **2007**. Geographic variation in cranial morphology of short-beaked common dolphins (*Delphinus delphis*) from the North Atlantic. Journal of Mammology 88(3):678-688.

Westgate, A. J., and A. J. Read. **2007**. Reproduction in short-beaked common dolphins (*Delphinus delphis*) from the western North Atlantic. Marine Biology 150:1011-1024.

Westgate, A. J., and K. A. Tolley. **1999**. Geographical differences in organochlorine contaminants in harbour porpoises *Phocoena phocoena* from western North Atlantic. Marine Ecology Progress Series 177:255-268.

Whelan, C. L., and J. Wyneken. **2007**. Estimating predation levels and site-specific survival of hatchling loggerhead seaturtles (*Caretta caretta*) from South Florida beaches. Copeia 2007(3):745-754.

White, A. J., M. Harwell, and D. Marcovich. **2008**. Metal concentrations in loggerhead sea turtle eggs from the Florida Gulf and Atlantic coasts, In: Proceedings of the 25th annual symposium on sea turtle biology and conservation, p. 163, NOAA Tech Memo NMFS-SEFSC-582.

Whitehead, H. **1996**. Babysitting, dive synchrony, and indications of alloparental care in sperm whales. Behavioral Ecology and Sociobiology 38(4):237-244.

Whitehead, H. **2002**. Estimates of the current global population size and historical trajectory for sperm whales. Marine Ecology Progress Series 242:295-304.

Whitehead, H. **2003**. Sperm whales: social evolution in the ocean, University of Chicago Press.

Whitehead, H., L. Rendell, and M. Marcoux. **2006**. Coda vocalizations recorded in breeding areas are almost entirely produced by mature female sperm whales (*Physeter macrocephalus*). Canadian Journal of Zoology 84(4):609-614.

Whitehead, H., S. Waters, and T. Lyrholm. **1991**. Social organization of female sperm whales and their offspring: constant companions and casual acquaintances. Behavioral Ecology and Sociobiology 29(5):385-389.

Whitehead, H., and T. Wimmer. **2005**. Heterogeneity and the mark–recapture assessment of the Scotian Shelf population of northern bottlenose whales (*Hyperoodon ampullatus*). Canadian Journal of Fisheries and Aquatic Sciences 62(11):2573-2585.

Whitehead, H. **1998**. Cultural selection and genetic diversity in matrilineal whales. Science 282(5394):1708-1711.

Whitman, A. A., and P. M. Payne. **1990**. Age of harbour seals, *Phoca vitulina concolor* , wintering in southern New England. Canadian Field-Naturalist 104(4):579-582.

Wibbels, T. **2007**. Sex determination and sex ratios in ridley turtles, In: Biology and Conservation of Ridley Sea Turtles, pp. 167-189, Johns Hopkins University Press.

Wiig, O., and R. W. Lie. **1984**. Analysis of the morphological relationship between the hooded seals (*Cystophora cristata*) of Newfoundland, the Denmark Strait and Jan Mayen. Journal of Zoology 203:227-220.

Wiley, D., C. Ware, A. Bocconcelli, D. Cholewiak, A. Friedlaender, and M. Thompson. **2011**. Underwater components of humpback whale bubble-net feeding behaviour. Behaviour 148:575-602.

Wiley, D. N., R. A. Asmutis, T. D. Pitchford, and D. P. Gannon. **1995**. Stranding and mortality of humpback whales, *Megaptera novaeangliae*, in the mid-Atlantic and southeast United States, 1985-1992. Fishery Bulletin 93(1):196–205.

Williams, A. S. **1999**. Prey selection by harbor seals in relation to fish taken by the Gulf of Maine sink gillnet fishery. MS thesis. University of Maine, Orono, ME. 36 p.

Williams, K. L., M. G. Frick, and J. B. Pfaller. **2006a**. First report of green, *Chelonia mydas*, and Kemp's ridley, *Lepidochelys kempii* turtle nesting on Wassaw Island, Georgia, USA. Marine Turtle Newsletter 113:8.

Williams, R., D. E. Bain, J. K. B. Ford, and A. W. Trites. **2002**. Behavioural responses of male killer whales to a leapfrogging vessel. Journal of Cetacean Research and Management 4(3):305–310.

Williams, R., D. Lusseau, and P. S. Hammond. **2006b**. Estimating relative energetic costs of human disturbance to killer whales (*Orcinus orca*). Biological Conservation 133(3):301–311.

Wimmer, T., and H. Whitehead. **2004**. Movements and distribution of northern bottlenose whales, *Hyperoodon ampullatus*, on the Scotian Slope and in adjacent waters. Canadian Journal of Zoology 82(11):1782-1794.

Winn, H. E., J. D. Goodyear, R. D. Kenney, and R. O. Petricig. **1995**. Dive patterns of tagged right whales in the Great South Channel. Continental Shelf Research 15(4-5):593-611.

Winn, H. E., C. A. Price, and P. W. Sorensen. **1986**. The distributional biology of the right whale (Eubalaena glacialis) in the western North Atlantic. Reports of the International Whaling Commission Special Issue 10:129-138.

Winn, H. E., and P. J. Perkins. **1976**. Distribution and sounds of the minke whale, with a review of mysticete sounds. Cetology 19:1-12.

Winn, H. E., P. J. Perkinds, and L. Winn. **1970**. Sounds and behavior of the northern bottle-nosed whale., In: 7th Annual Conference on Biological Sonar and Diving Mammals, pp. 53-59, Stanford Research Institute, Menlo Park, CA.

Wise, J. P., Sr. et al. **2009**. A global assessment of chromium pollution using sperm whales (*Physeter macrocephalus*) as an indicator species. Chemosphere 75(11):1461-1467.

Wise, J. P., Sr, S. S. Wise, C. LaCerte, J. P. Wise Jr, and A. E.-M. Aboueissa. **2011**. The genotoxicity of particulate and soluble chromate in sperm whale (*Physeter macrocephalus*) skin fibroblasts. Environmental and Molecular Mutagenesis 52(1):43-49.

Wise, J. P. et al. **2008**. Hexavalent chromium is cytotoxic and genotoxic to the North Atlantic right whale (*Eubalaena glacialis*) lung and testes fibroblasts. Mutation Research/Genetic Toxicology and Environmental Mutagenesis 650(1):30-38.

Wishner, K. F., J. R. Schoenherr, R. Beardsley, and C. Chen. **1995**. Abundance, distribution and population structure of the copepod *Calanus finmarchicus* in a springtime right whale feeding area in the southwestern Gulf of Maine. Continental Shelf Research 15(4-5):475–507.

Witherington, B. E. **2002**. Ecology of neonate loggerhead turtles inhabiting lines of downwelling near a Gulf Stream front. Marine Biology 140(4):843-853.

Witherington, B., P. Kubilis, B. Brost, and A. Meylan. **2009**. Decreasing annual nest counts in a globally important loggerhead sea turtle population. Ecological Applications 19(1):30-54.

Witherington, B. E., and R. E. Martin. **1996**. Understanding, assessing, and resolving light-pollution problems on sea turtle nesting beaches. Florida Marine Research Institute Technical Report TR-2 73.

Witt, M. J., A. C. Broderick, D. J. Johns, C. Martin, R. Penrose, M. S. Hoogmoed, and B. J. Godley. **2007**. Prey landscapes help identify potential foraging habitats for leatherback turtles in the NE Atlantic. Marine Ecology Progress Series 337:231-244.

Witteveen, B. H., R. J. Foy, K. M. Wynne, and Y. Tremblay. **2008**. Investigation of foraging habits and prey selection by humpback whales (*Megaptera novaeangliae*) using acoustic tags and concurrent fish surveys. Marine Mammal Science 24(3):516-534.

Witzell, W. N., and J. R. Schmid. **2005**. Diet of immature Kemp's ridley turtles (*Lepidochelys kempii*) from Gullivan Bay, Ten Thousand Islands, southwest Florida. Bulletin of Marine Science 77(2):191-200.

Wood LaFond, S. A. **2009**. The dynamics of recolonization: a study of the gray seal in the northeast U.S. Ph.D. thesis. University of Massachusetts, Boston, Boston, MA. 166 p.

Wood, F. G. **1953**. Underwater sound production and concurrent behavior of captive porpoises, *Tursiops truncatus*, and *Stenella plagiodon*. Bulletin of Marine Science 3(2):120-133.

Wood, L. D., and A. Meylan. **2008**. A preliminary assessment of hawksbill turtles (*Eretmochelys imbricata*) in Palm Beach County waters, Florida, USA, In: Proceedings of the 25th annual symposium on sea turtle biology and conservation, p. 22, NOAA Tech Memo NMFS-SEFSC-582.

Wood, L. **n.d.** The Florida Hawksbill Project: What We've Learned. http://www.floridahawksbills.com/Site/What_Weve_Learned.html. Accessed 19 December 2010.

Wood, S. A., T. R. Frasier, B. A. McLeod, J. R. Gilbert, B. N. White, W. D. Bowen, M. O. Hammill, G. T. Waring, and S. Brault. **2011**. The genetics of recolonization: an analysis of the stock structure of grey seals (*Halichoerus grypus*) in the northwest Atlantic. Canadian Journal of Zoology 89:490-497.

Woodley, T. H., and D. E. Gaskin. **1996**. Environmental characteristics of North Atlantic right and fin whale habitat in the lower Bay of Fundy, Canada. Canadian Journal of Zoology 74(1):75-84.

Woodworth, P. A., G. S. Schorr, R. W. Baird, D. L. Webster, D. J. McSweeney, M. B. Hanson, R. D. Andrews, and J. J. Polovina. **2011**. Eddies as offshore foraging grounds for melon-headed whales (*Peponocephala electra*). Marine Mammal Science DOI:10.1111/j.1748-7692.2011.00509.x.

Work, P. A., A. L. Sapp, D. W. Scott, and M. G. Dodd. **2010**. Influence of small vessel operation and propulsion system on loggerhead sea turtle injuries. Journal of Experimental Marine Biology and Ecology 393:168-175.

Wright, A. J. **2005**. Lunar cycles and sperm whales (*Physeter macrocephalus*) strandings on the North Atlantic coastlines of the British isles and eastern Canada. Marine Mammal Science 21(1):145-149.

Wright, I. E., J. E. Reynolds III, B. B. Ackerman, L. I. Ward, B. L. Weigle, and W. A. Szelistowski. **2002**. Trends in manatee (*Trichechus manatus latirostris*) counts and habitat use in Tampa Bay, 1987–1994: implications for conservation. Marine Mammal Science 18(1):259-274.

Wright, L. J., L. Fisher, and C. M. Burney. **2010**. Beach nourishment impacts of sea turtle nesting and live hatchling production rates in Broward County, Florida, USA, In: Proceedings of the 28th annual symposium on sea turtle biology and conservation, p. 192, NOAA Tech Memo NMFS-SEFSC-602.

Wursig, B., T. A. Jefferson, and D. J. Schmidly. **2000**. The Marine Mammals of the Gulf of Mexico., 1st ed., Texas A&M University Press, College Station, Texas.

Wurtz, M., R. Poggi, and M. R. Clarke. **1992**. Cephalopods from the stomachs of a Risso's dolphin (*Grampus griseus*) from the Mediterranean. Journal of the Marine Biological Association of the United Kingdom 72:861-867.

Yasunaga, G., and Y. Fujise. **2009**. Temporal trends and factors affecting mercury levels in common minke, Bryde's and sei whales and their prey species in the western North Pacific, IWC SC/J09/JR23.

Ylitalo, G. M., C. O. Matkin, J. Buzitis, M. M. Krahn, L. L. Jones, and T. K. Rowles. **2001**. Influence of life-history parameters on organochlorine concentrations in free-ranging killer whales (*Orcinus orca*) from Prince William Sound, AK. The Science of the Total Environment 281:183-203.

Yordy, J. E., D. A. Pabst, W. A. McLellan, R. S. Wells, T. K. Rowles, and J. R. Kucklick. **2010a**. Tissue-specific distribution and whole-body burden estimates of persistent organic pollutants in the bottlenose dolphin (*Tursiops truncatus*). Environmental Toxicology and Chemistry 29(6):1263-1273.

Yordy, J. E., R. S. Wells, B. C. Balmer, L. H. Schwacke, T. K. Rowles, and J. R. Kucklick. **2010b**. Life history as a source of variation for persistent organic pollutant (POP) patterns in a community of common bottlenose dolphins (*Tursiops truncatus*) resident to Sarasota Bay, FL. Science of the Total Environment 408:2163-2172.

Yordy, J. E., R. S. Wells, B. C. Balmer, L. H. Schwacke, T. K. Rowles, and J. R. Kucklick. **2010c**. Partitioning of persistent organic pollutants between blubber and blood of wild bottlenose dolphins: implications for biomonitoring and health. Environmental Science & Technology 44(12):4789-4795.

Yudhana, A., J. Din, Sunardi, S. Abdullah, and R. B. R. Hassan. **2010**. Green turtle hearing identification based on frequency spectral anaylsis. Applied Physics Research 2(1):125-134.

Zaugg, S., M. Schaar, L. Houegnigan, and M. André. **2010**. Real time classification of sperm whale clicks and shipping impulses from fixed ocean observatories. Applied Acoustics 71(11):1011-1019.

Zerbini, A. N., and M. Cesar de Oliveira Santos. **1997**. First record of the pygmy killer whale *Feresa attenuata* (Gray, 1874) for the Brazillian coast. Aquatic Mammals 23(2):105-109.

Zimmer, W. M. X., P. T. Madsen, V. Teloni, M. P. Johnson, and P. L. Tyack. **2005a**. Off-axis effects on the multipulse structure of sperm whale usual clicks with implications for sound production. The Journal of the Acoustical Society of America 118:3337-3345.

Zimmer, W. M. X., P. L. Tyack, M. P. Johnson, and P. T. Madsen. **2005b**. Three-dimensional beam pattern of regular sperm whale clicks confirms bent-horn hypothesis. The Journal of the Acoustical Society of America 117:1473-1485.

Zimmer, W. M. X., J. Harwood, P. L. Tyack, M. P. Johnson, and P. T. Madsen. **2008**. Passive acoustic detection of deep-diving beaked whales. The Journal of the Acoustical Society of America 124(5):2823-2832.

Zimmer, W. M. X., M. P. Johnson, P. T. Madsen, and P. L. Tyack. **2005c**. Echolocation clicks of free-ranging Cuvier's beaked whales (*Ziphius cavirostris*). The Journal of the Acoustical Society of America 117(6):3919-3927.

Zug, G. R., H. J. Kalb, and S. J. Luzar. **1997**. Age and growth in wild Kemp's ridley seaturtles *Lepidochelys kempii* from skeletochronological data. Biological Conservation 80(3):261-268.

Zug, G. R., and R. E. Glor. **1998**. Estimates of age and growth in a population of green sea turtles (*Chelonia mydas*) from the Indian River lagoon system, Florida: a skeletochronological analysis. Canadian Journal of Zoology 76(8):1497-1506.

MARCH 3, 1849

The Department of the Interior Mission

As the Nation's principal conservation agency, the Department of the Interior has responsibility for most of our nationally owned public lands and natural resources. This includes fostering the sound use of our land and water resources; protecting our fish, wildlife, and biological diversity; preserving the environmental and cultural values of our national parks and historical places; and providing for the enjoyment of life through outdoor recreation. The Department assesses our energy and mineral resources and works to ensure that their development is in the best interests of all our people by encouraging stewardship and citizen participation in their care. The Department also has a major responsibility for American Indian reservation communities and for people who live in island communities.

The Bureau of Ocean Energy Management Mission

The Bureau of Ocean Energy Management (BOEM) promotes energy independence, environmental protection, and economic development through responsible, science-based management of offshore conventional and renewable energy.