

3.2.34. Gray Seal *Halichoerus grypus* (Fabricius, 1791)

Description

Gray seals are sexually dimorphic, with adult males up to 2.3 m long and females up to 2.0 m (Jefferson et al., 1993; Wynne and Schwartz, 1999). Sexes also differ in color—males mainly dark with irregular light patches and females light with dark spots. Pups are born with a solid white or yellowish coat, and molt to a spotted coat in 2–4 weeks. Gray seals (including pups) are distinguished from harbor and harp seals by the distinctive shape of the head. Gray seals have an elongate snout with a flat or slightly convex profile. The distance between the eyes and nose is about twice the distance between the eyes and the ear openings. The neck and chest of males may be wrinkled, scarred, and often devoid of fur. The latter is believed to result from male-male fights over access to females. Females are sleeker and lack scarring (Hall, 2002). The nostrils are widely separated and from the front look like the letter “M” or “W.”

Status

Gray seals are not listed under the U.S. Endangered Species Act or on the Rhode Island state list, and are classified as Least Concern on the IUCN Red List. Gray seal populations in both the northeastern U.S. and eastern Canada have grown significantly since low points in the 1960s. Starting from a handful of animals and no pupping, the Massachusetts colony now has an annual pup production of over 1,000 and >5,600 seals total. There are >1,700 animals present in Maine (Waring et al., 2008). The eastern Canadian population was estimated at only 5,600 seals in the 1960s (Mansfield, 1966), but grew to 144,000 in 1993, 195,000 in 1997, and 209,000–223,000 in 2004 (Lesage and Hammill, 2001; Hammill, 2005).

Gray seals were hunted by Native Americans for subsistence. They were hunted by European settlers, for oil, meat, and leather, to the point where abundance was extremely low from the mid-19th to mid-20th centuries (Lavigne and Kovacs, 1988). In the modern era, commercial hunting has been relatively limited because of low abundance and relatively low pelt value. Most modern hunting has been primarily for population control to reduce sealworm infestation and minimize damage to commercial fishery gear and seal consumption of

commercial fish stocks (Bonner, 1981). Bounties paid by state authorities in both Maine and Massachusetts were one factor leading to the near extirpation in the 1960s of gray seals in the northeastern U.S. (Andrews and Mott, 1967; Rough, 1995). In Canada, gray seal stocks were also greatly reduced (Mansfield, 1966). There were culls at Sable Island averaging over 1,700 per year from the late 1960s to the early 1980s (Waring et al., 2008). At present there is a small commercial hunt in the Gulf of St Lawrence (few hundred per year), and hunting is not permitted at Sable Island (Waring et al., 2008). In addition, a personal hunting license in Canada allows killing up to six gray seals (Lesage and Hammill, 2001). The 2001–2005 annual average bycatch mortality of gray seals from entanglement in the northeastern and mid-Atlantic U.S. sink gillnet fisheries was 304 animals, with unknown levels of mortality in the bottom trawl fishery and some Canadian fisheries (Waring et al., 2008).

Ecology and life history

Like harbor seals, but unlike harp and hooded seals, gray seals haul out routinely for resting and not only for breeding or molting. They appear to be flexible in selection of haul-out substrates, utilizing rocky ledges, sandy beaches, and sea ice.

After the winter breeding season, there is a post-breeding pelagic feeding period in February–April. This is followed by a haul-out for molting in May or June, then another dispersed feeding period until the next winter's pupping season begins (Lesage and Hammill, 2001). Juveniles disperse more widely than adults during feeding phases of the annual cycle (Ronald and Gots, 2003). Three gray seals were taken in 1996 by Spanish trawlers on southern edge of the Grand Banks (Lens, 1997), suggesting they are capable of moving long distances and far offshore during pelagic feeding. Recent satellite-linked tagging studies have confirmed that Canadian gray seals commonly travel long distances far from their breeding sites (Beck et al., 2002; Austin et al., 2003).

Gray seals feed on a variety of fish species and cephalopods, with no evidence for significant dietary differences between first-year juveniles and adults (Bonner, 1981). Scat samples from Muskeget Island, Massachusetts, included flounder, silver hake, sand lance, skates, and gadids (Rough, 1995). Species identified from scats collected from Sable Island, Grand Manan Island, and eastern Nova Scotia include sand lance, herring, silver hake, cod, pollack,

capelin, flounders, mackerel, and squid (W. D. Bowen et al., 1993; Bowen and Harrison, 1994). In New York waters, stomach contents of stranded gray seals show herring to be the predominant prey, as well as mackerel, gadids, and flounders (S. S. Sadove, pers. comm.).

Gray seals give birth to single pups in January or February (Bonner, 1981; Riedman, 1990; Nowak, 1999; Hall, 2002). Adult females attend their pups continuously from birth to weaning and do not feed at all during that time. The breeding fast is even longer for adult males, since they arrive first to stake out and defend territories. Pups are weaned and abandoned in about 18 days, followed by a post-weaning fast of 10–28 days. Pups are born with a white lanugo coat that is molted around the time of weaning. Ovulation and mating take place late in lactation, and implantation is delayed for about 3.4 months.

Age at sexual maturity differs between sexes (Bonner, 1981; Hall, 2002). Most females mature at 4 or 5 years. Males mature at 6 years, but do not begin to breed until 8 years. Most breeding bulls are 12 to 18 years old.

Sharks prey on gray seals around Sable Island (Brodie and Beck, 1983; Stobo and Lucas, 2000). A variety of different shark species has been implicated, but Greenland sharks are suspected as a principal predator.

Bonner (1981) reviewed the occurrence of disease and parasites in gray seals. Most disease incidences are known from pups where the immune system has been compromised by starvation, rendering them subject to a variety of opportunistic infections. Common infections include pneumonia, conjunctivitis, and septicemia. External parasites include seal lice (*Echinophthirius horridus*) and nasal mites (*Halarachne halichoeri*). Internal parasites include a variety of nematodes, acanthocephalans, cestodes, and trematodes in the gut, lungs, liver, and kidneys. Of particular interest is the anisakine nematode *Pseudoterranova decipiens*, the sealworm or codworm (Templeman, 1990). The penultimate phase of the parasite's life cycle is as a large juvenile encysted in the muscle tissue of a fish like cod or haddock, greatly reducing the palatability and marketability of the fillets. Piscivorous seals are the final host in the life cycle of the worms, which mature and reproduce in the seal's gut. Sealworms infect other seal species, but are most commonly found in gray seals in most areas, which has led to seal reduction programs such as bounties or culls. Disease and parasites are better known in harbor seals, and it is likely that many of the same organisms affect gray seals.

General distribution

Gray seals are found only in the North Atlantic (Bonner 1981; Riedman 1990; Nowak 1999; Hall 2002; Ronald and Gots 2003). There are three separate populations: a Canadian stock that occurs from Massachusetts to Labrador, a European stock that occurs from France to Russia and west to Iceland, and a third stock in the Baltic Sea. There are two principal pupping concentrations of the Canadian stock: one in the Gulf of St. Lawrence and the other on Sable Island off the southern coast of Nova Scotia. The Massachusetts population has grown substantially, and at least two pupping colonies are now established in Maine (Waring et al. 2008).

Historical occurrence

Gray seals were largely absent from Rhode Island and nearby waters until recently. Cronan and Brooks (1968) reported that the species was unknown from Rhode Island, but said that there was one record to the south. That surely referred to Goodwin's (1933) report of a juvenile male taken in a net at Young's Million Dollar Pier in Atlantic City, New Jersey in 1931. Archaeological finds indicate that Native Americans utilized gray seals on Block Island and along the Connecticut coast (Waters, 1967), however, the number of individuals was apparently relatively small. It is quite possible that the Indians simply made opportunistic use of stranded animals at no greater frequency than current stranding rates. Neither De Kay (1842) nor Connor (1971) knew of any occurrences in New York. Similarly, Linsley (1842) did not mention gray seals for Connecticut, and Goodwin (1935) stated that the species had not been recorded in Connecticut.

Waters and Rivard (1962) said that gray seals might occur in low numbers in winter off Massachusetts to as far south as Block Island. There was a small breeding colony of gray seals in Massachusetts during the first half of the 20th Century (Andrews and Mott, 1967; Rough, 1995). They pupped on Muskeget Island, a low sandy island off the west end of Nantucket. They had been nearly extirpated by the 1960s due to hunting, primarily for bounties paid by state authorities in both Maine and Massachusetts. Annual pup production of the Massachusetts colony declined from 14–19 in the early 1950s to only 1 by the end of 1960s. No pups were

observed and adults were scarce in 1971–1979, but the number of seals increased during the 1980s and pupping resumed by 1988 (Rough, 1995).

Recent occurrence

The recovery of the Massachusetts and Canadian populations led to an increased occurrence in southern New England and mid-Atlantic waters. There are gray seal specimens in the Smithsonian collection from strandings in New Jersey in 1973 and 1978. These were the first records west of Massachusetts after the 1931 Atlantic City animal. The three earliest strandings in Rhode Island, all from Block Island, were in 1980, 1986, and 1988 (Nawojchik, 2002), although the 1980 specimen was misidentified and labeled as a harbor seal and then lost in a freezer for 24 years (Kenney, 2005). The first sighting of a gray seal in eastern Long Island was in about 1980 (S. S. Sadove, pers. comm.). Strandings and occasional sightings throughout the region have become common beginning in the 1990s.

Gray seal occurrences in the Rhode Island study area are mostly represented by stranding records—155 of 193 total records (80%). Gray seal records in the region are primarily from the spring (87.1%), with much smaller numbers in all other seasons—5.7% in winter, 5.2% in summer, and 2.1% in fall. Strandings were broadly distributed along ocean-facing beaches in Long Island and Rhode Island, with a few spring records in Connecticut (Fig. 54). There were no strandings on the north shore of Long Island.

As with other seals, habitat use by gray seals in the Rhode Island study area is poorly known. They are seen mainly when stranded or hauled out and infrequently at sea. No definitive conclusions about habitat preferences should be drawn from strandings. Gray seals are frequently observed mixed in with groups of harbor seals at haul-out sites in Massachusetts and northward. There are very few observations of gray seals in Rhode Island other than strandings. In New York, apparently healthy gray seals are similarly seen at harbor seal haul-outs, usually only one or two animals but in larger numbers on a few occasions (S. S. Sadove, pers. comm.). The most regular occurrences are at the haul-outs on Great Gull Island and Fisher's Island.

The annual numbers of gray seal strandings in the Rhode Island study area since 1993 have fluctuated markedly, from a low of 1 in 1999 to a high of 23 in 2004 (Fig. 55). There is some suggestion of a 3-4 year periodicity, but any underlying factors are not understood.

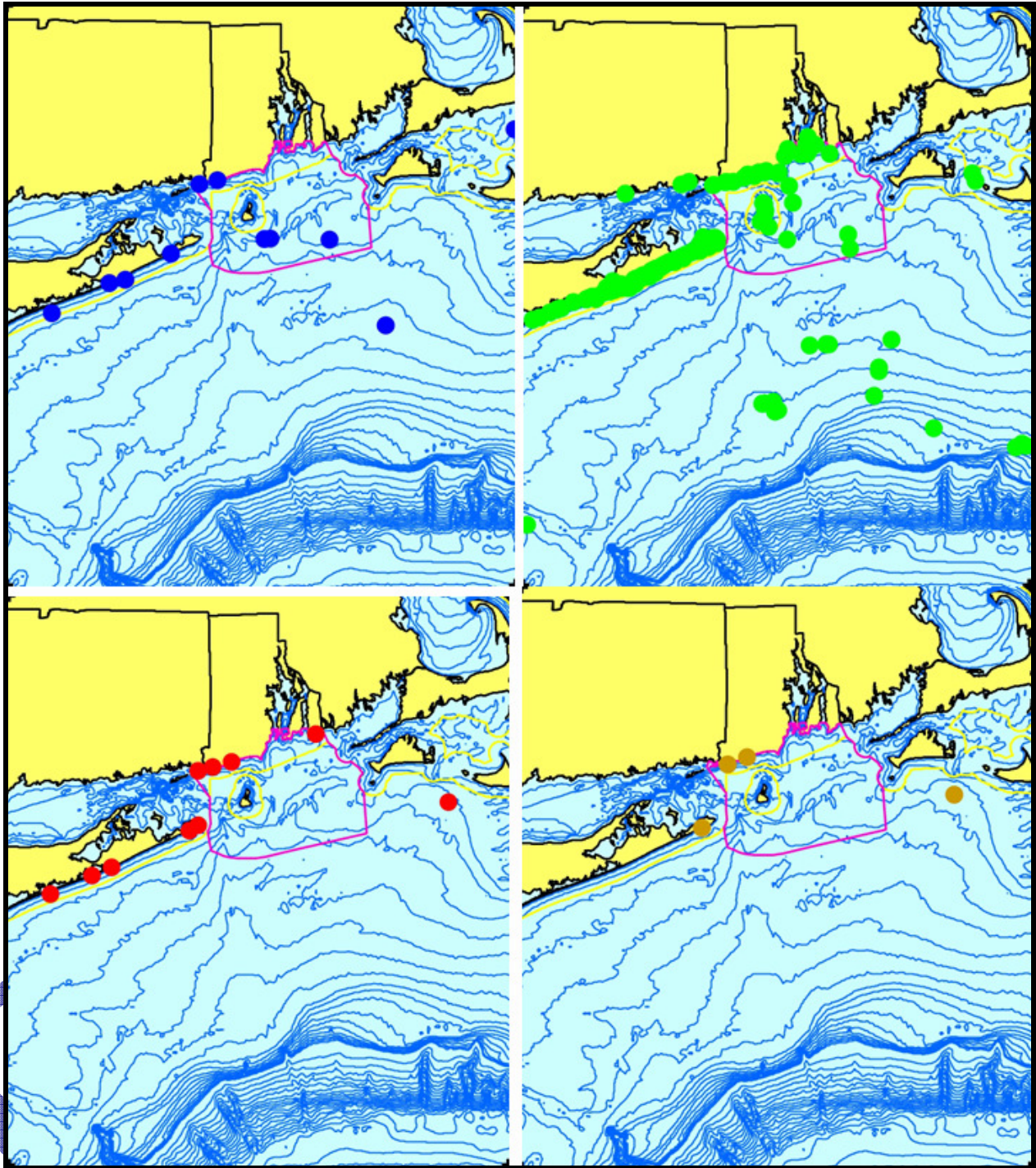


Figure 54. Aggregated sighting, stranding, and bycatch records of gray seals in the Rhode Island study area, 1986–2008 (n = 193: winter = 11, spring = 168, summer = 10, fall = 4).

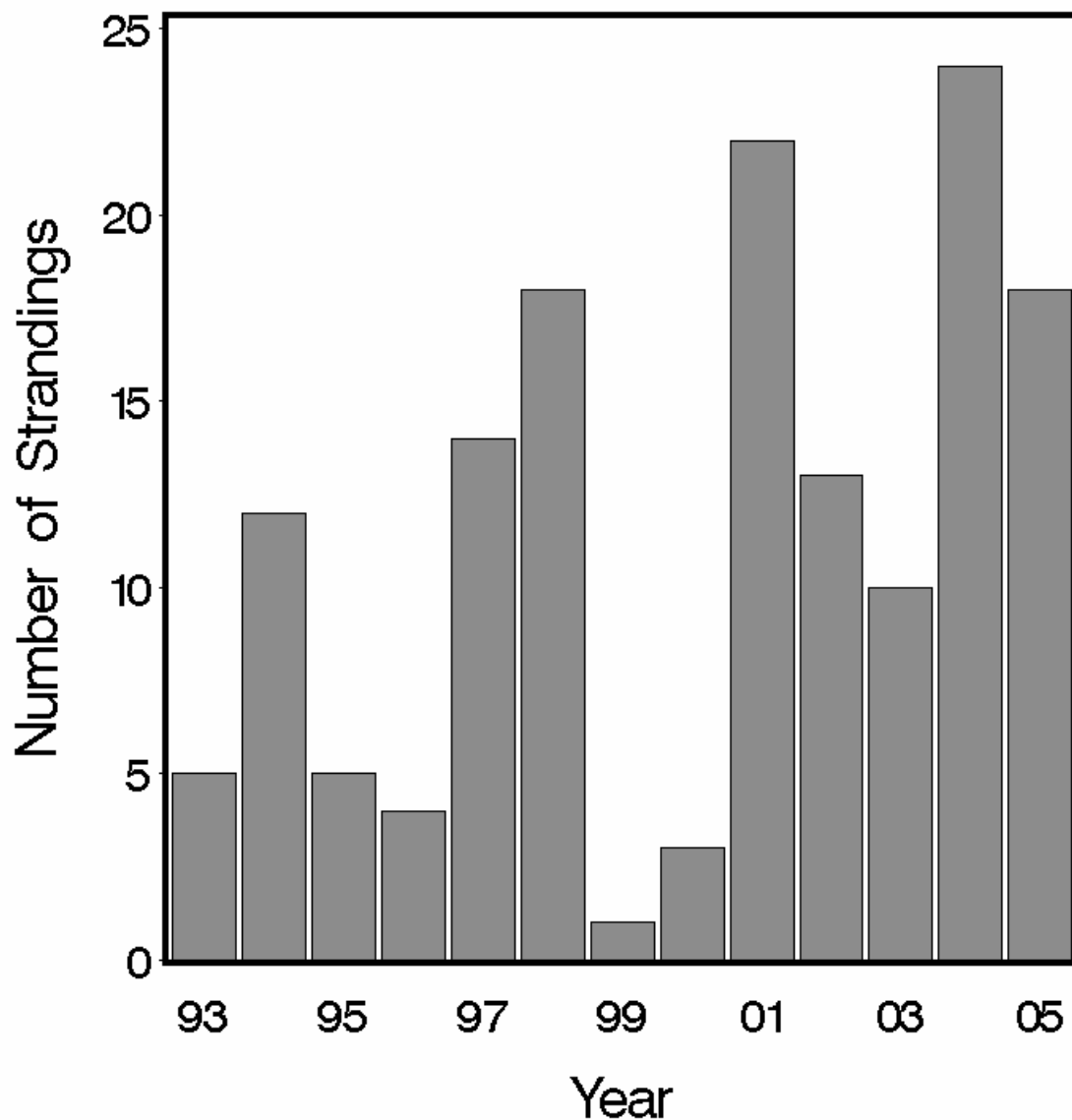


Figure 55. Annual stranding frequencies for gray seals in the Rhode Island study area, 1993–2005.

The very strong seasonality observed in gray seal occurrence in the study area is clearly related to the timing of pupping in January–February. The majority of individuals in the study area appear to be post-weaning juveniles, and starved or starving juveniles are the most common stranded individuals encountered (Nawojchik, 2002; Kenney, 2005). The expected period of feeding dispersal by newly weaned pups that have just completed their post-weaning fast and

molt would be in March and April. A peak in gray seal stranding frequency in the study area occurs in April (n = 82, 43%), followed by March (61, 32%) and May (25, 13%) (Fig. 56).

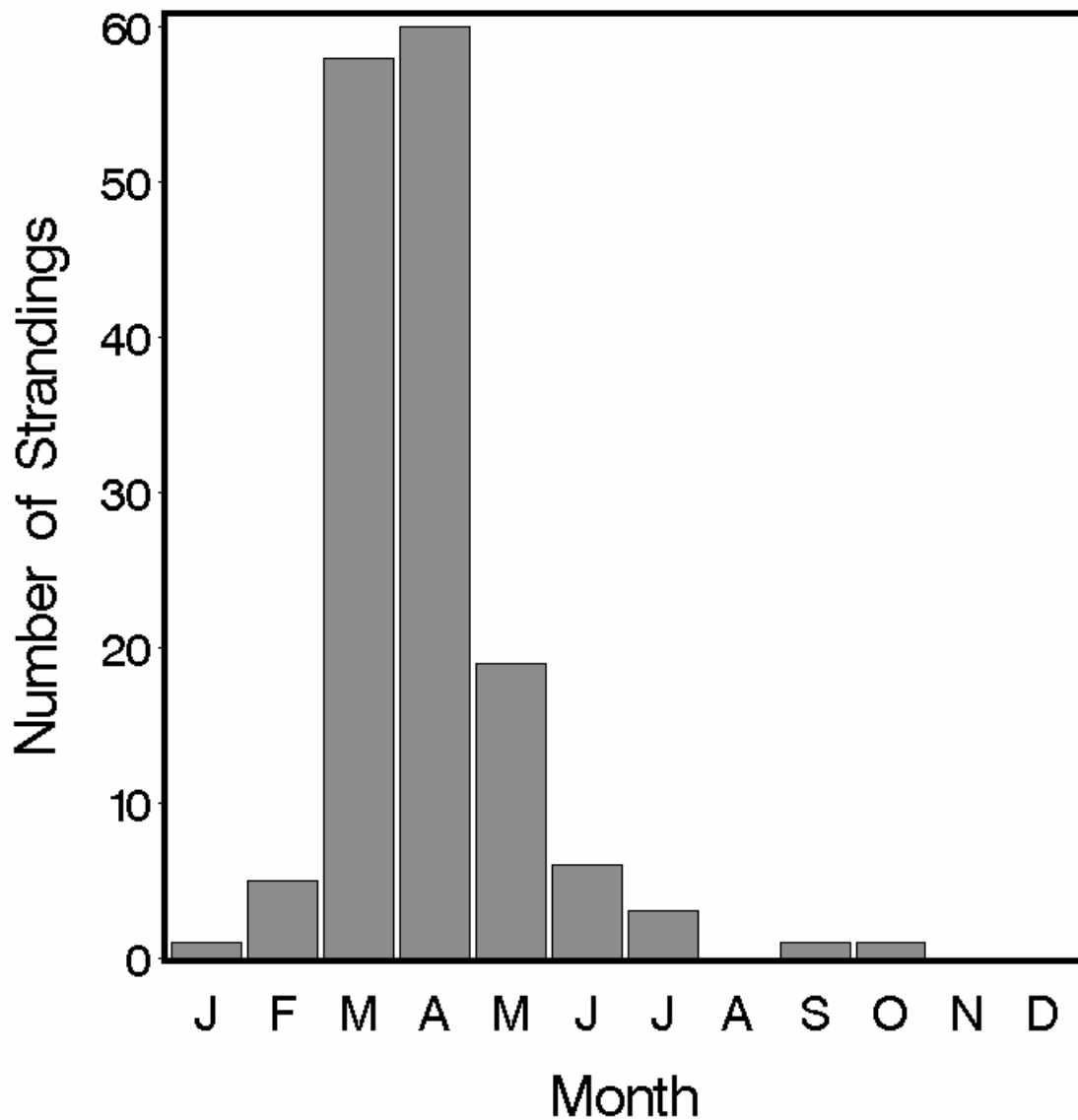


Figure 56. Monthly stranding frequencies for gray seals in the Rhode Island study area.

Including six pre-1993 stranding records provided by Mystic Aquarium, gray seal strandings in Rhode Island alone have been relatively uncommon (Fig. 57). Most years had 0–3 strandings, but there was a short-term spike with 7 in 2003 and 8 in 2004.

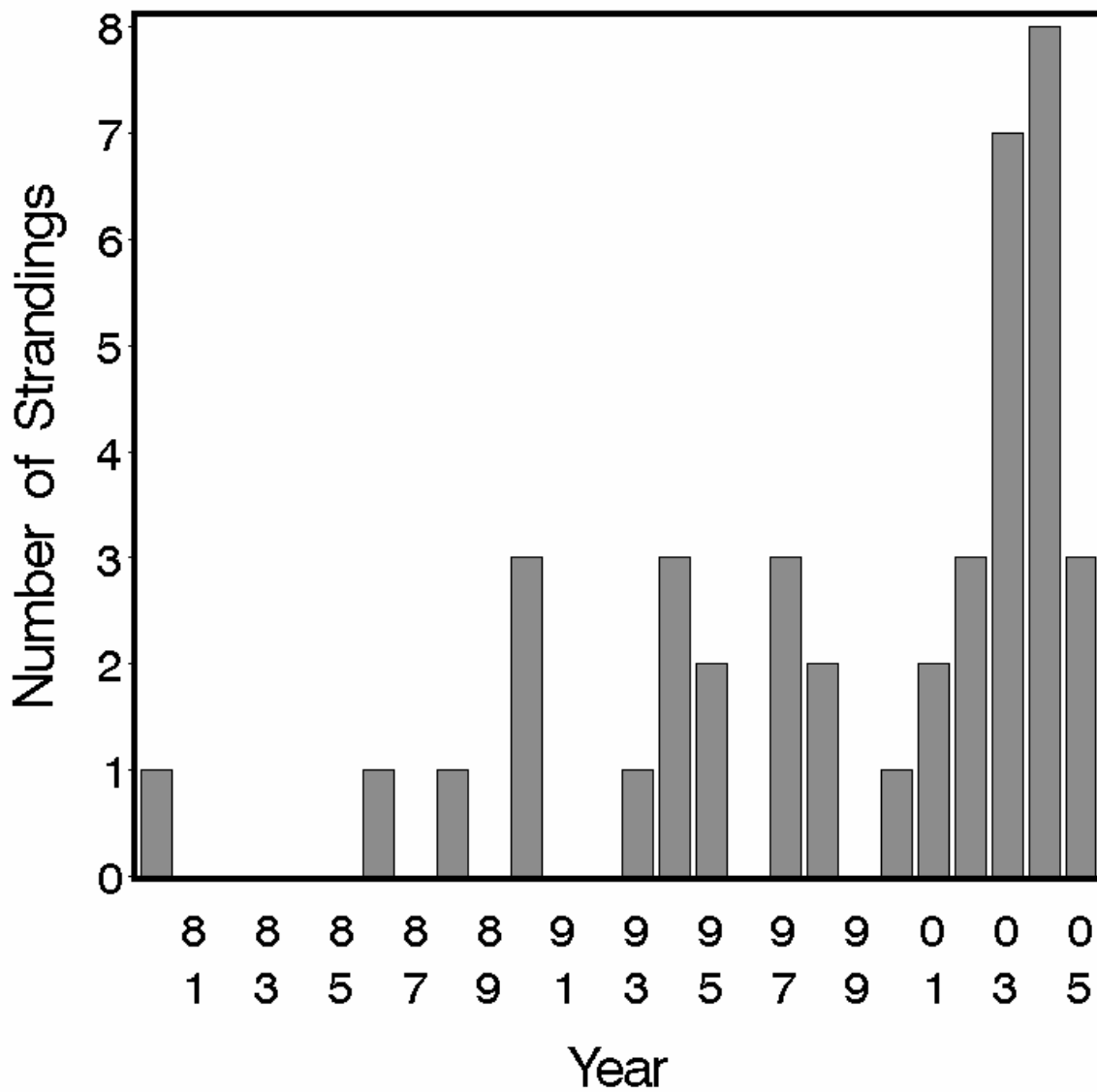


Figure 57. Annual stranding frequencies for gray seals in Rhode Island alone, 1980–2005.

Conclusions

The occurrence of gray seals in the Rhode Island study area appears to be increasing over time, but the seals present are mostly dispersing juveniles in the spring. There are no consistent haul-out locations in the study area except for the sandy shoals around Nantucket and Monomoy in Massachusetts. Consequently, gray seals are not a significant concern relative to the SAMP. Over the longer term, one might speculate that continued expansion of the breeding colony in Massachusetts could lead to establishment of pupping at Sandy Point at the northern end of Block Island, which might have the right combination of habitat and low disturbance.

3.2.25. Harp Seal *Pagophilus groenlandicus* (Erxleben, 1777)

Description

Adult harp seals are relatively distinctive and easily recognized. While they are roughly the same size (1.7–1.9 m) and shape as harbor seals, with heads that appear slightly smaller, their color pattern is distinctive (Jefferson et al., 1993; Wynne and Schwartz, 1999). An adult has a pale white to silvery-gray body with a black face and a black inverted V- or harp-shaped marking on the back. Juveniles are silvery gray with scattered large black blotches, and are much less spotted than similar-sized harbor seals. Harp seals go through a sequence of pelages from neonate to adult (Lavigne, 2002). Pups known as “thin white-coats” are born covered in a fine, white fetal fur or lanugo. They become “fat white-coats” as they gain weight during nursing. At weaning, the juvenile coat has filled in and is visible under the white lanugo. The pup is now known as a “gray-coat.” The lanugo is shed after weaning, and the pup then has a silvery juvenile coat with scattered dark blotches. At this stage young seals are referred to as “beaters” because of their awkward, splashing manner of swimming on the surface. The second molt occurs at 13 to 14 months into a similar “bedlamer” pelage, with somewhat more dark patches. Juvenile and adult seals molt annually, hauling out in dense aggregations on the pack ice north of the breeding areas in April and May (Ronald and Gots, 2003). Adult pelage is attained at the time of sexual maturity. The transition tends to be much quicker in males than females, with some females never completely developing the harp pattern. Adults with the intermediate pattern of both a partially developed harp marking and typical juvenile dark blotches are known as “spotted harps.”

Status

Harp seals are not listed under the U.S. Endangered Species Act or on the Rhode Island state list, and are classified as Least Concern on the IUCN Red List. Despite substantial annual harvests by commercial and subsistence hunters, the abundance of harp seals in the eastern Canadian populations appears to have increased steadily (Waring et al., 2008). Abundance is estimated using production models based on pup counts. The total Canadian population was

estimated at 3.1 million in 1990, 4.8 million in 1994, 5.2 million in 1999, 5.5 million in 2000, and 5.9 million in 2004. The other two populations are substantially smaller—0.3 million near Jan Mayen and 1.5–2.0 million in the White Sea (Lavigne, 2002). There are no estimates for the numbers of harp seals off the northeastern U.S. or in the Rhode Island study area.

Harp seals have traditionally been hunted for subsistence use by the Inuit in Greenland and eastern Canada (MacLean et al., 2002). They still are hunted in Greenland; one of the first returns of a flipper tag from a live-stranded harp seal that had been rehabilitated and released by Mystic Aquarium came from an Inuit hunter in Greenland (R. Nawojchik, pers. comm.). Lavigne and Kovacs (1988) extensively reviewed the history of the eastern Canadian seal hunt. Early European settlers did not immediately exploit harp seals, since other species were more accessible. The walrus was the first species hunted for ivory, oil, and leather, but it was extirpated in the Gulf of St. Lawrence and off Nova Scotia by the early 18th century. Hunters also took gray seals and, to a lesser extent, harbor seals for oil, meat, and skins. Winter harp seal hunting began in the St. Lawrence River in the mid-17th century. Hunting methods quickly shifted from shooting seals on the ice from boats to the use of nets, adopting the Inuit methods. By the mid-18th century, harp sealing spread throughout the Gulf and along the northeastern coast of Newfoundland. Total annual takes ranged from 7,000 to 128,000 seals. It was also in the 18th century that the early spring hunt for white-coat pups began. During those years whelping patches, on the pack ice, were easily accessible from shore. In the 19th century, technological advances such as steam-powered ships enabled additional expansion of the hunt. Annual takes ranged from more than 500,000 to 740,000 seals. Hooded seals were also taken. Oil rendered from the blubber layer was the main product of the seal hunt, until tanning methods (developed in the 1940s and 1950s) made the pelts of white-coat harp seals and, especially, blue-back hooded seals extremely valuable. Beginning in the 1960s, opposition to the white-coat hunt became a major campaign of environmental organizations. Because of widespread opposition and a European ban on importation of white-coat pelts, commercial hunting of seal pups was banned in Canada in 1987. Hunting is now restricted to non-breeding adults, juveniles, and independent, post-weaning pups. The harp seal hunt is currently managed under quotas set by the Canadian Dept. of Fisheries and Oceans (Waring et al., 2008). Total annual take in Canada and Greenland, by commercial and subsistence hunters, including animals struck and lost, is about 440,000 harp seals. There is also substantial mortality caused by entanglement in gillnets in the

Canadian lumpfish fishery, varying between 5,000 and 19,000 annually. Entanglement mortality in U.S. fisheries is lower, averaging 73 per year in 2001–2005 in the sink gillnet fishery plus an undetermined number in the bottom trawl fishery.

Starvation is by far the most frequent cause of mortality and morbidity for harp seals in the Rhode Island study area. The most common harp seal encountered is a stranded, starved or starving juvenile in winter or early spring. The timing coincides with the feeding transition period, when 1-year-olds must switch from near-surface feeding on krill to diving deeper for fish, and some proportion of animals simply do not seem to make that transition successfully. Lucas et al. (2003) reported the same phenomenon at Sable Island, where three-quarters of the harp and hooded seals encountered were starved or emaciated juveniles. They also reported on the prevalence of gravel in the stomachs, and concluded that juveniles were often unable to feed successfully. Disease and parasites are much better known in harbor seals, and it is likely that many of the same organisms affect harp seals.

Ecology and life history

Harp seals are gregarious in their northern range, hauling out for pupping and molting in large aggregations. In the Rhode Island study area, however, they are most often solitary. Nearly all individuals observed are juveniles. Three adults (one stranded dead, one photographed alive but extremely emaciated, and one apparently healthy) have been reported in Rhode Island, and Sadove and Cardinale (1993) reported one stranded adult in New York. An adult was captured in 1945 in Virginia (McAlpine and Walker, 1990), and adult markings were described for a harp seal in New Jersey by Allen (1880) (see Historical Occurrence below). The increase in juvenile harp seal occurrences in the Rhode Island study area in the 1990s coincided with growth of the seal population in Canada and declines in fish stocks. One might speculate that juveniles are forced to disperse more widely because of competition for prey (McAlpine et al., 1999a). However, complicating factors such as changes in climatic and oceanographic conditions (Frank, 2003) prevent taking the idea much beyond speculation.

Harp seals in their usual range are associated with sea ice, with an annual migration following the annual cycle of pack ice, moving north in summer and south in winter (Ronald and Healey, 1981; Lavigne, 2002; Ronald and Gots, 2003). Off the northeastern U.S., almost nothing

is known of their habitat preferences except for stranded individuals. Like hooded seals, they are most likely to occur on relatively flat, sandy beaches.

Adult harp seals feed on a wide variety of small pelagic and demersal fishes, squid, and crustaceans, especially on capelin and Arctic cod (Wallace and Lawson, 1997). Pups undergo a transition in prey type and feeding depth during their first year (Ronald and Healey, 1981). After the post-weaning fast, pups first feed mainly on euphausiid crustaceans (“krill”) in near-surface waters. At about one year of age, they make a transition to diving to intermediate depths and feeding on pelagic fishes. Stomach contents of harp seals stranded in New York sometimes include herring or similar fishes (S. S. Sadove, pers. comm.). Often, stomachs are empty, or at times filled with stones and shells, leading to serious medical complications or death (Medic, 2005). No reason for the pathologic ingestion of stones has been determined, but it is speculated that it is a consequence of their habit of eating ice as a source of fresh water. Stranding response protocols for harp and hooded seals have been modified in an attempt to recover starving juveniles as soon as possible before they have a chance to start eating stones.

Female harp seals give birth to single pups on the dense pack ice (Ronald and Healey, 1981; Lavigne and Kovacs, 1988; Lavigne, 2002; Ronald and Gots, 2003). Females select areas of thick, hummocky ice that provides protection for pups. These locations are some distance from the ice margin but where open water is still accessible. Females gather in aggregations separated only by a couple of meters from one another. The timing differs slightly among breeding populations. Most pups in the Gulf herd are born between 20 February and 10 March, while births are slightly later in the Front herd.

Pups average a meter in length, weigh 11–12 kg at birth, and have little blubber. They nurse for 10–12 days on milk that is up to 43% fat and 10% protein, gaining 2.2 kg per day. Females fast entirely, or feed little, during lactation. They abandon the pups immediately after weaning. At weaning the pups have a 5-cm thick layer of blubber and weigh ca. 36 kg. Pups then remain on the ice for a post-weaning fasting period as long as 6 weeks, during which they can lose up to half of their body mass.

Mating occurs just after the pup is weaned. It usually takes place in the water, though there have been observations of mating on the ice. Implantation of the embryo is delayed about three months. Adult females breed annually, and both males and females can remain reproductively active into their twenties (Ronald and Healey, 1981). Both males and females

reach sexual maturity at an average age of 5.5 years, but males generally are not reproductively active and successful until age 8 (Ronald and Healey, 1981).

General distribution

Harp seals are found only in the North Atlantic and Arctic, from eastern Canada east to northwestern Siberia (Ronald and Healey, 1981; Lavigne and Kovacs, 1988; Riedman, 1990; Nowak, 1999; Lavigne, 2002; Ronald and Gots, 2003). Their breeding distribution is similar to that of hooded seals. There are three breeding populations—in the White Sea north of Russia, in the Greenland Sea near Jan Mayen, and in two locations near Newfoundland—the “Front herd” to the northeast and the “Gulf herd” to the west. Harp seal breeding patches are located somewhat inshore of those of hooded seals. Their distribution during the remainder of their annual cycle is poorly known.

Historical occurrence

Until recently harp seals were very rare in the Rhode Island study area and nearly as rare from Massachusetts to Maine (McAlpine and Walker, 1990). Cronan and Brooks (1968) cited one earlier report of a harp seal in Connecticut (but see the following), but knew of no records from Rhode Island. Waters and Rivard (1962) described harp seals as rare winter visitors to New England, but gave no specific records. Linsley (1842) reported a single occurrence in Connecticut: “The white seal, commonly called the harp seal, is very rare, and has been seen only at Stonington a few times on the rocks. During the past winter, attempts were made to take him, but unfortunately the hunters went to the windward side of him, and though they came so near as to shoot at him while sliding off, he escaped. I have information from J. H. Trumbull, Esq., of Stonington, who says ‘his color was a dusky white throughout.’ I conclude, therefore, it must be the *groenlandica*.” Given that (1) the identification was based only on color from a second-hand report, (2) the report said nothing about markings, (3) harp seals in southern New England are more likely to haul out on flat sand than rocks, and (4) some harbor seals, especially when dry, appear very pale-colored, it seems that Linsley’s seal was more likely a harbor seal. Goodwin (1935) and Connor (1971) repeated Linsley’s account, and Connor added an

unsubstantiated report from Kieran (1959) of harp seals offshore at Coney Island in winter. Allen (1880) reported a harp seal at Trenton, New Jersey and did include reasonable identifying details, but it was a third-hand report without documentation. Goodwin (1954) reported an adult male captured at Cape Henry, Virginia, in March 1945, documented by a newspaper photograph that was reprinted by McAlpine and Walker (1990), making it the only well-documented historical record south of Massachusetts.

Recent occurrence

Harp seals in the Rhode Island study area are known almost exclusively from strandings (688 of 703 records = 98%). Strandings are widespread on ocean-facing beaches throughout Long Island and Rhode Island (Fig. 58). The apparent absence in Massachusetts is due only to the geographic scope of the stranding dataset we had acquired. The records are almost entirely from spring (68.3%) and winter (30.4%). Harp seals are nearly absent in summer and fall. Strandings are common on both sides of Long Island Sound, more than any other species of seal. Harp seals also make occasional appearances well inland up rivers.

Beginning in the late 1980s harp seal occurrences began to increase in the Gulf of Maine (McAlpine and Walker, 1990; Stevick and Fernald, 1998; McAlpine et al., 1999a; Harris et al., 2002). The available regional stranding dataset for the Rhode Island study area begins in 1993, when there were 9 harp seal strandings (Fig. 59). Harp seal records in the region more than quadrupled to 38 in 1994, then increased to 55 in 1995 and to a peak of 67 in 1996. In 1995, for the first time harp seals exceeded the total for harbor seals. They have been the most common stranded seal in the region since, with the exception of 2003. Strandings spiked sharply in 2001 at 150% higher than the average annual rate in the other years from 1994 to 2005.

Monthly stranding frequencies provide a clearer view of the trend in harp seal strandings over the year (Fig. 60). Strandings peak in late winter-early spring, with very few outside of January–May. Peak strandings are in March (42%), with 22% in both February and April. The timing is too late for the strandings to be pups born in late February–early March in Newfoundland, confirming that strandings in the region are primarily yearlings.

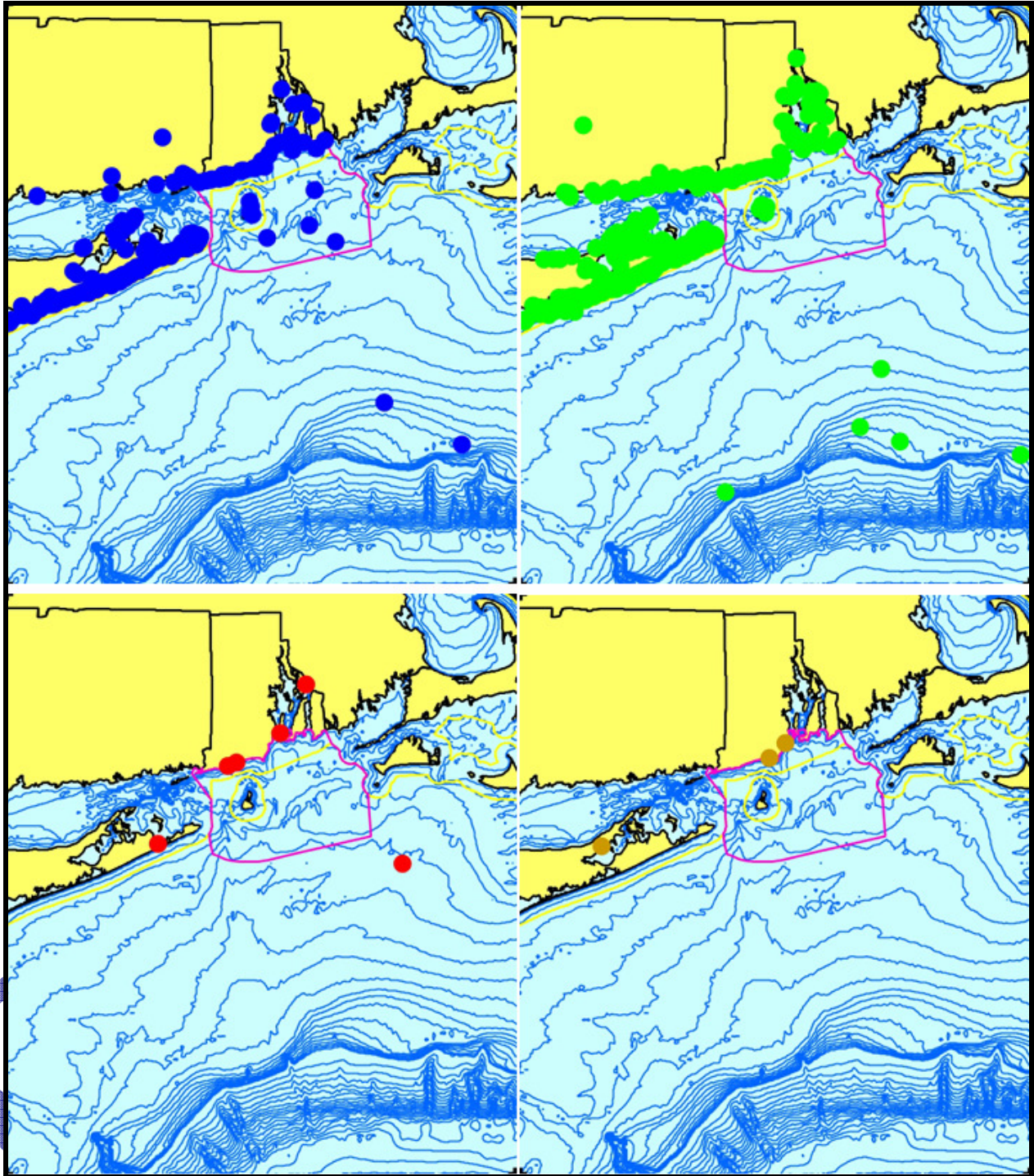


Figure 59. Aggregated stranding, sighting, and bycatch records of harp seals in the Rhode Island study area, 1989–2007 (n = 703: winter = 214, spring = 480, summer = 6, fall = 3).

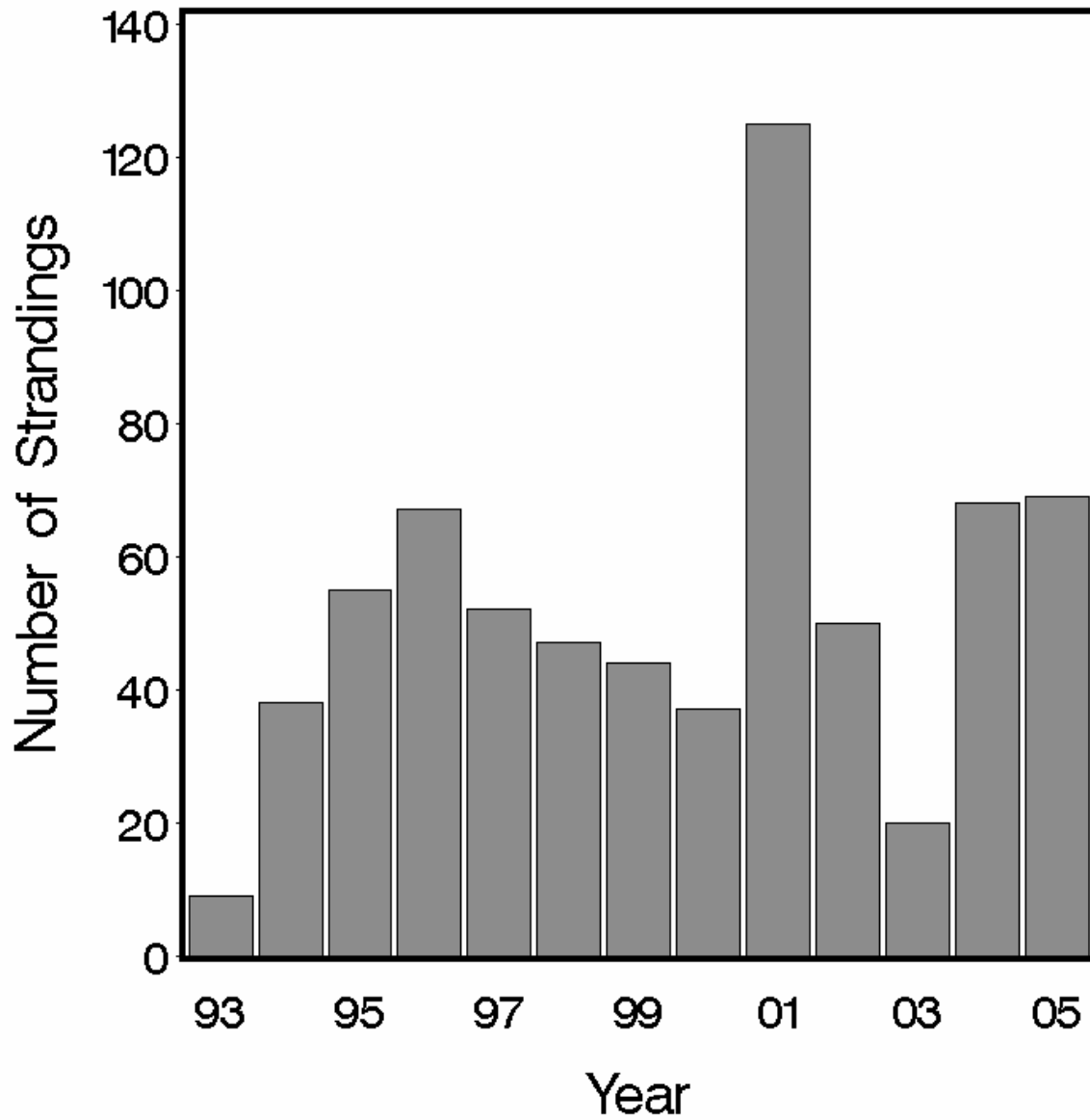


Figure 59. Annual stranding frequencies for harp seals in the Rhode Island study area, 1993–2005.

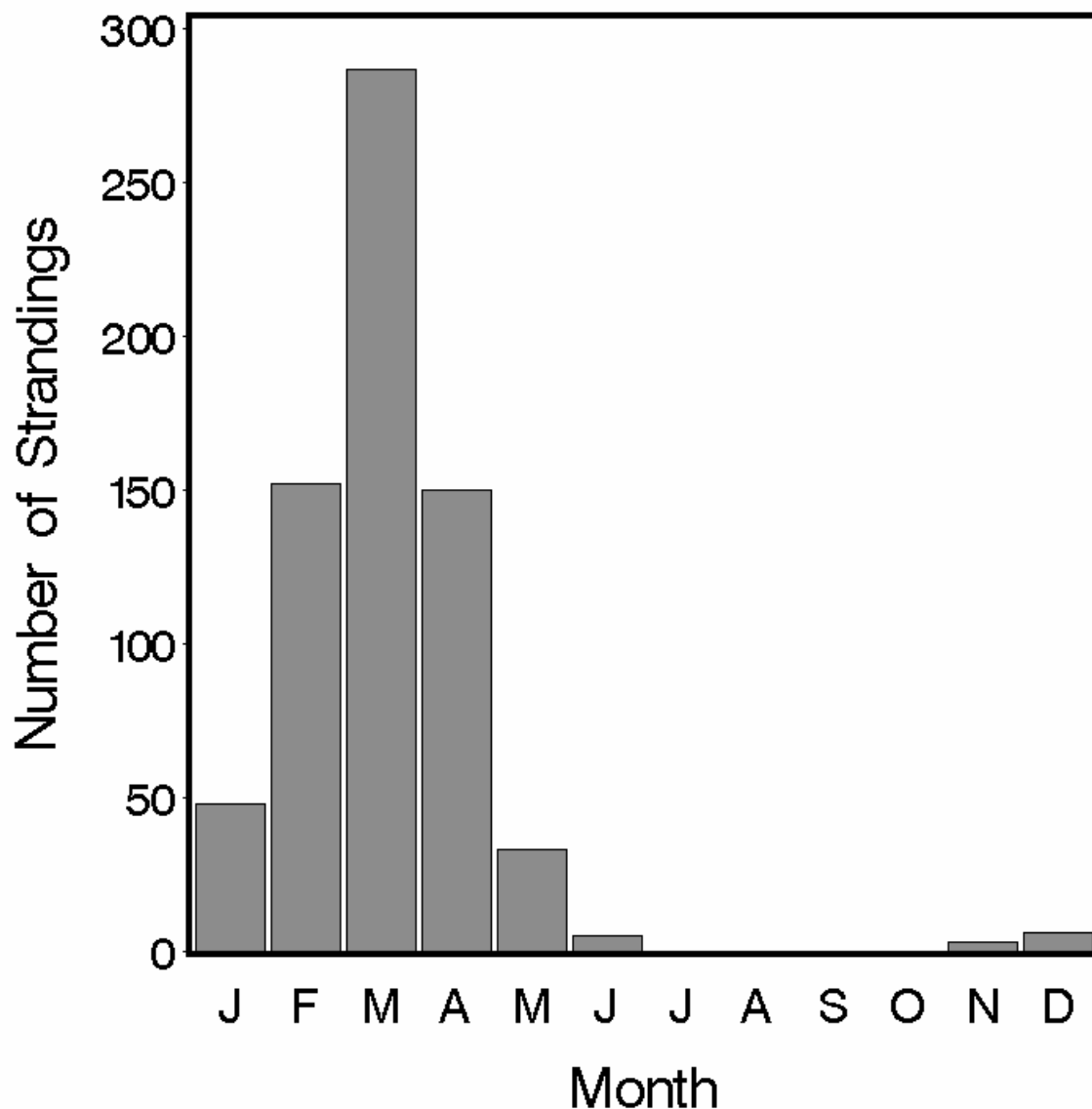


Figure 60. Monthly stranding frequencies of harp seals in the Rhode Island study area.

It appears from the study area stranding trend that harp seal occurrence increased sharply in 1994 (Fig. 59), but that dataset doesn't quite capture the beginning of their presence in the region. There were three earlier strandings in Rhode Island—near the Quonochontaug Breachway in Charlestown in May 1989, on Napatree Point in Westerly in April 1990, and at Mackerel Cove in Jamestown in January 1992. There were also strandings before 1993 in New

York (Sadove and Cardinale, 1993), but we don't have those records. Looking only at Rhode Island (Fig. 61), the stranding trend closely matches that for the entire study area, confirming that 1994 was the year when their presence really began to increase. The spike in 2001 is even higher in Rhode Island, at 284% above the 1994–2005 average background rate.

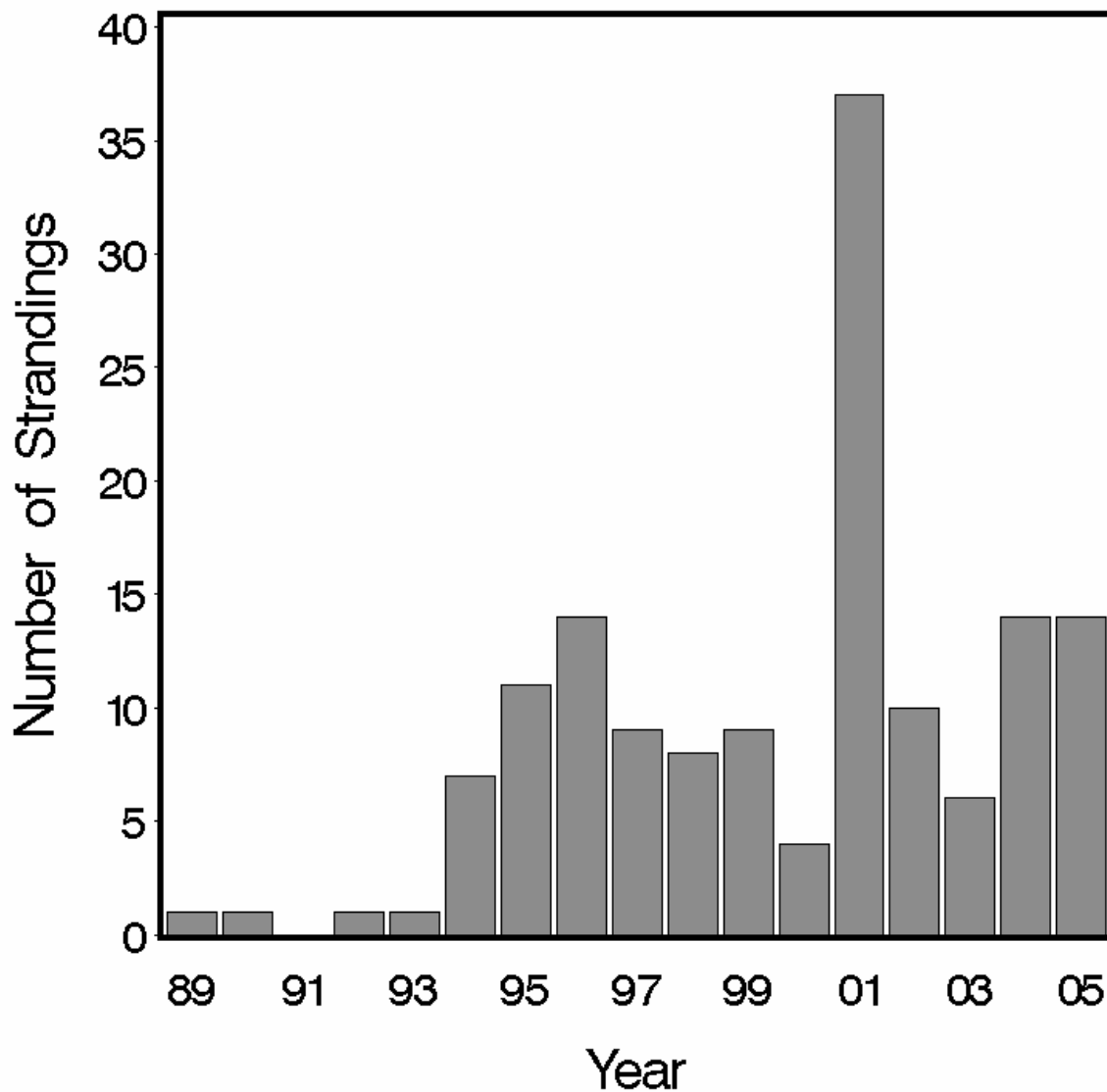


Figure 61. Annual stranding frequencies for harp seals in Rhode Island alone, 1989–2005.

Conclusions

While harp seals may be relatively abundant in the Rhode Island study area, they are predominantly juveniles dispersed from a population center far to the north in eastern Canada. They are not of concern relative to the SAMP.

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3.2.36. Hooded Seal *Cystophora cristata* (Erxleben, 1777)

Description

Hooded seals are moderately sexually dimorphic, with males 2.3–2.7 m long and females 2.0–2.2 m (Jefferson et al. 1993; Wynne and Schwartz 1999). Adult males have a two-lobed, inflatable proboscis or hood on the top of the snout. They can also inflate the nasal septum out of one nostril like a red balloon. Adults are silvery blue-gray with a black face, irregular black blotches, and a lighter belly. Pups, known as “blue-backs,” are solid dark blue-gray above, with a creamy whitish belly clearly demarcated from the dark back. The head is broader, flatter, and rounder, with noticeably larger eyes, than other Atlantic seals.

Status

Hooded seals are not listed under the U.S. Endangered Species Act or on the Rhode Island state list, and are classified as Vulnerable on the IUCN Red List. There is no current, reliable estimate of abundance for the entire hooded seal population in the North Atlantic or for the animals within U.S. waters or in the Rhode Island study area (Waring et al., 2008). Breeding herd abundance estimates are extrapolated from pup counts assuming a ratio of 1:5 (pups:total population). The most recent estimates were from counts in 2005 (IUCN, 2008). The total abundance of the Northwest Atlantic stock (in eastern Canada and western Greenland) was estimated at 592,000, which represents a moderate increase since 1980. The West Ice stock (east of Greenland) was estimated at about 70,000, which is a decline of 85–90% in the last 60 years, and pup production declined from 24,000 in 1997 to 15,250 in 2005. The cause of the decline is not understood, but it is the reason for the Vulnerable classification on the Red List.

Hooded seals have long been hunted both by subsistence hunters (MacLean et al., 2002) and commercial sealers (Lavigne and Kovacs, 1988; Waring et al., 2008). There is no longer any hunting of blue-backs for their pelts. The annual commercial quota for the Front herd off eastern Newfoundland has been set at 10,000 since 1998, but recent catches have been low, and no commercial hunting is allowed in the Gulf of St. Lawrence or Davis Strait (Waring et al., 2008). The West Ice commercial hunt is jointly managed by Norway and Russia, with several thousand

taken each year (NMFCA, 2006). An average of 16 hooded seals per year have been killed in recent years in U.S. waters by entanglement in the sink gillnet fishery, and others are taken as bycatch in Canadian fisheries (Waring et al., 2008). The total incidental take from all sources is believed to be low relative to the population's total abundance.

Ecology and life history

Hooded seals are solitary and aggressive (Kovacs, 2002). Most of the year they are widely dispersed and asocial. Even when aggregating during the breeding and molting season, they are aggressive with one another. Adult males fight for prime space near a mother and pup, but a female will keep a larger male at a distance from her pup. Even newly weaned pups have a reputation for aggressiveness.

Almost nothing is known of habitat use by healthy hooded seals in the Rhode Island study area. The species is known exclusively from strandings, which are nearly all recently weaned blue-back pups, many of which are under-nourished or even starving. A subadult male that live-stranded in Westerly in February 1999 was an exception. It was rehabilitated and released (see below). In their normal range, hooded seals are most often associated with sea ice. As with harp seals, the other ice seal that occurs in the Rhode Island study area, hooded seals are most often observed on relatively flat sandy beaches.

Outside of the breeding season, hooded seals are most likely highly pelagic. Scholander (1940) recorded a month-old hooded seal pup diving to a depth of 75 m on its first dive. Based on telemetry tagging, hooded seals are capable of dives deeper than 1000 m and lasting almost an hour (Folkow and Blix, 1995; Kovacs, 2002).

In their normal Arctic range, adult hooded seals feed on deepwater fish species such as Greenland halibut, redfish, and a variety of other fishes and squids (Reeves and Ling, 1981; Kovacs and Lavigne, 1986; Kovacs, 2002; Ronald and Gots, 2003), while pups feed more on crustaceans at shallower depths. Their prey preferences in the Rhode Island study area are poorly known. Stomachs of stranded animals contain a variety of prey items, probably reflecting local prey availability.

Hooded seal reproduction was reviewed by Reeves and Ling (1981), Kovacs and Lavigne (1986), Lavigne and Kovacs (1988), and Kovacs (2002). Single pups are born in late March,

with pupping in all the stocks occurring synchronously. Pupping takes place on loose pack ice, with females at least 50 m apart. Hooded seals tend to pup farther offshore than harp seals in all areas except the Gulf of St. Lawrence. Pups are about 1 m in length and weigh 20–25 kg at birth. They shed the gray lanugo in utero and are born in a relatively advanced state in their juvenile blue-back coats. They are nursed on milk that averages 60% fat content, and weaned at 50–60 kg in only four days, the shortest known lactation period of any mammal (Bowen et al., 1985).

Each female-pup pair is usually guarded by a single male. Males compete vigorously for the opportunity via displays at first, but competition frequently escalates to violent, bloody fights. After weaning, the female abandons the pup and returns to the water, where mating takes place. At an earlier time, these mother-pup-male triads were anthropomorphically interpreted as families, and hooded seals were presumed to have a monogamous mating system. However, after mating with one female the male is free to move to another, resulting in a polygynous mating system (Boness et al., 1988). The most successful males may mate with up to 8 females in one breeding season. Implantation of the embryo is delayed for about four months, extending gestation to match a tightly synchronized annual cycle. Adults then disperse until aggregating, along with juveniles, for molting in June and July.

Pups remain alone on the ice for a post-weaning fast period of at least several days. They then disperse widely. They skip the molt during their first year and undergo the first post-natal molt at 14 months of age.

Females mature at age 3. Males mature at 4–6 years, but probably need to be older in order to successfully compete for mating opportunities (Lavigne and Kovacs, 1988).

General distribution

Like most pinnipeds, the distribution of hooded seals is well-known only for the portion of their annual cycle when they haul out for pupping (“whelping”). Hooded seals occur only in two separate breeding stocks in the North Atlantic (Reeves and Ling, 1981; Kovacs, 2002). The Northwest Atlantic stock pups in three areas, two in eastern Canada—the Gulf herd in the Gulf of St. Lawrence west of Newfoundland and the Front herd northeast of Newfoundland and east of Labrador, and in the Davis Strait between eastern Canada and Greenland. The West Ice stock pups in the Greenland Sea, east of Greenland and near Jan Mayen. After the breeding season,

adults and pups disperse, then seals from all areas, except pups, re-aggregate in the Denmark Strait between Greenland and Iceland to molt, with a second molting area farther north off the east coast of Greenland for some of the West Ice animals (Nowak, 1999). Their distribution at sea is poorly known, but they apparently disperse widely through much of the northwestern North Atlantic and into the Arctic Ocean (Lavigne and Kovacs, 1988). A few hooded seals, particularly pups and juveniles, have been known to disperse surprisingly far from their breeding areas, including the Caribbean and the North Pacific. Strandings have recently increased in frequency in New England, primarily between January and May coinciding with the breeding season (McAlpine et al., 1999b; Harris et al., 2001).

Historical occurrence

Historical literature confirms both the presence and extreme rarity of hooded seals in the Rhode Island study area or in southern New England more generally. Cronan and Brooks (1968) reported a single record in the Providence River, but the date was unknown to them. Waters and Rivard (1962) also mentioned the Providence occurrence and one other at Newburyport on the Massachusetts north shore. De Kay (1824) reported that an adult male was killed in Westchester County, New York. Linsley (1842) and Goodwin (1935) stated that they were not known from Connecticut. Connor (1971) added a second New York record—an anecdotal report of a hooded seal in a New York Harbor tributary “within just the last few years.”

Recent occurrence

Hooded seal occurrences in the Rhode Island study area are almost entirely strandings (96 of 97 records, 99%). The first confirmed strandings were recorded in 1993, though there were scattered anecdotal reports earlier than that (Sadove and Cardinale, 1993). They have been relatively common since. Hooded seal strandings are broadly distributed across ocean-facing beaches in the region, with only rare occurrences in Long Island Sound (Fig. 62). Strandings are most common in spring and winter (45% and 36% of all records, respectively), and rare in summer and fall. They occasionally occur well up rivers—for example, in southeastern Connecticut in spring, but less often than harp seals.

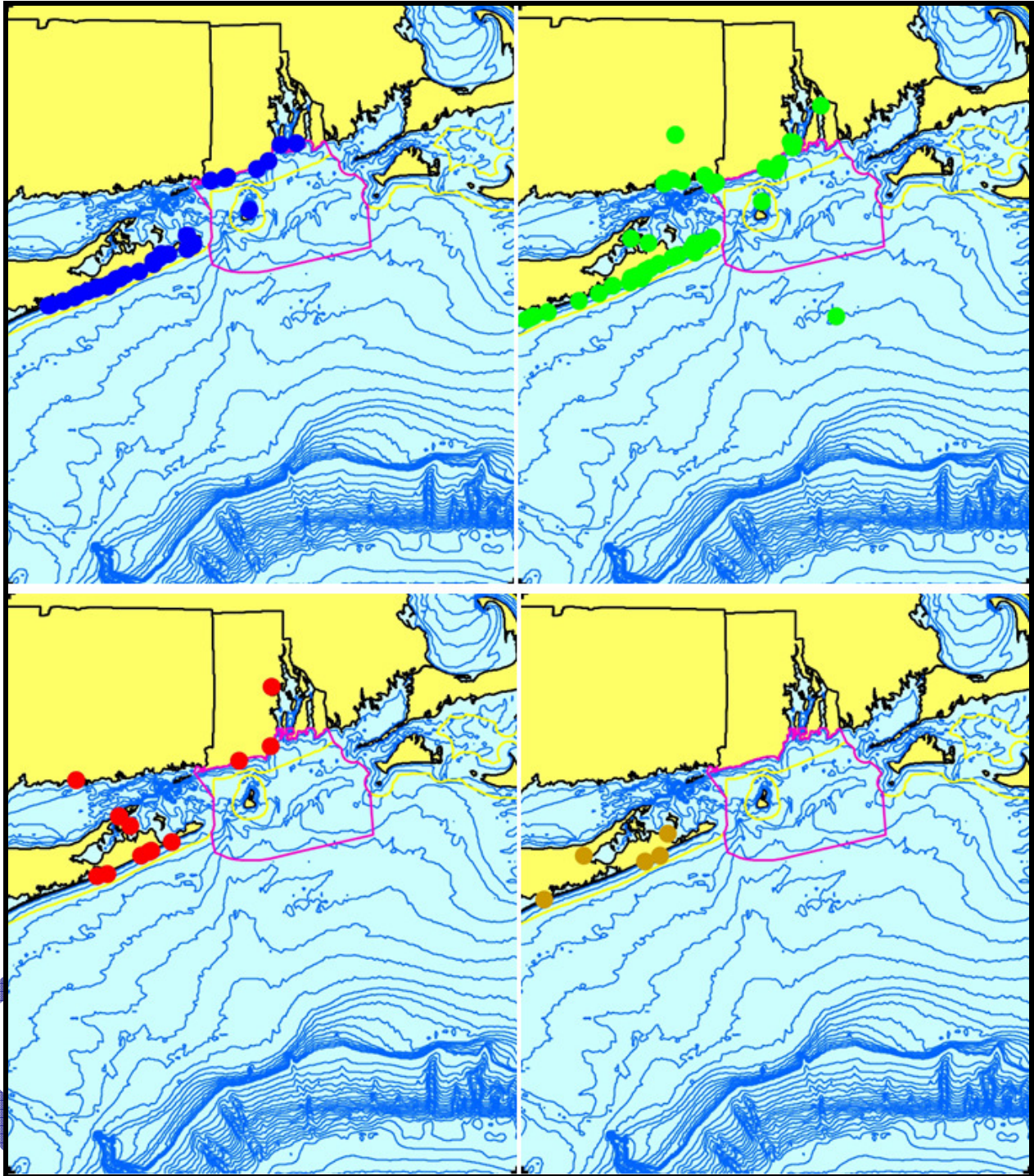


Figure 63. Aggregated sighting, stranding, and bycatch records of hooded seals in the Rhode Island study area, 1993–2005 (n = 97: winter = 36, spring = 43, summer = 13, fall = 5).

The time-series of strandings in the study area showed a marked spike in 1998 (Fig. 63). The same pattern is seen if only Rhode Island strandings are considered, where the 1993–2005 background level was 0–2 strandings per year with 3 in 1996, but there were 9 in 1998 (Fig. 64). There were no hooded seal strandings in Rhode Island before 1993. The reason for the sharp short-term increase is not known, but it may be related to hydrographic patterns in the region. In 1998 a cold mass of Labrador Subarctic Slope Water just offshore of the continental shelf extended much farther south than normal, reaching the latitude of southern New Jersey by February (Greene et al., 2003). The phenomenon was linked to a sharp decrease in the North Atlantic Oscillation Index in 1996.

Monthly stranding frequencies show maximum values in February (31%) and March (29%), but the occurrence is more spread out than in either gray seals or harbor seals (Fig. 65). As with harp seals, the peak in strandings is too early in the year to be pups. Most strandings are therefore yearlings, although summer and fall blueback strandings may be pups of the year. There was one interesting stranding event in 1999. On the 8th of February there was a report from Block Island of a live seal, possibly in distress, that sounded from the description like an adult hooded seal. The next day a subadult male hooded seal stranded on Misquamicut Beach in Westerly. It was in very poor condition and was not expected to survive overnight (R. Nawojchik, pers. comm.). Contrary to expectations, it gained over 100 kg over the next month and was released at Monahan's Dock in Narragansett Pier.

Conclusions

As with harp seals, hooded seals in the Rhode Island study area are predominantly juveniles dispersed from a population center far to the north in eastern Canada. They are not of concern relative to the SAMP.

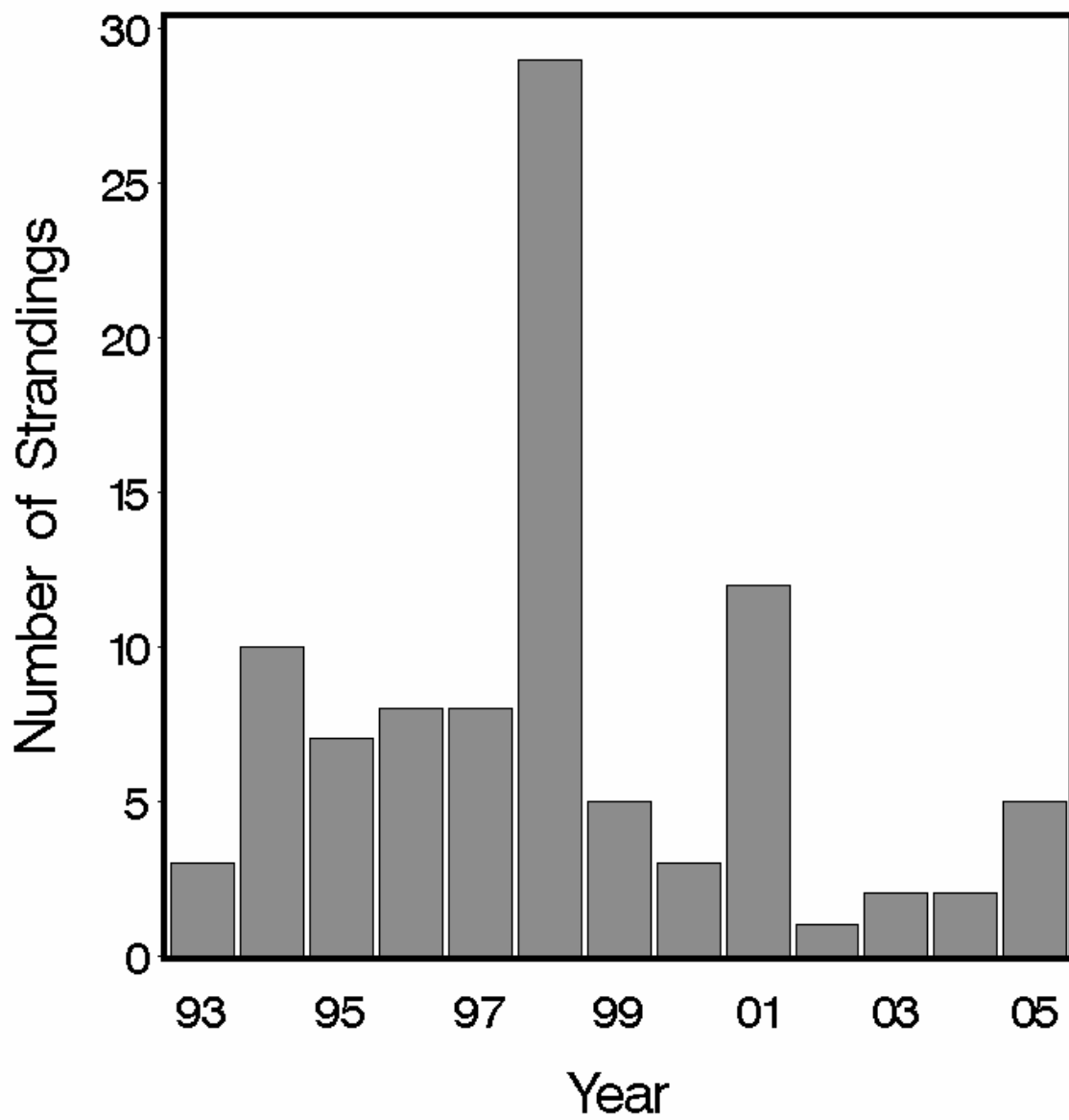


Figure 63. Annual stranding frequencies for hooded seals in the Rhode Island study area, 1993–2005.

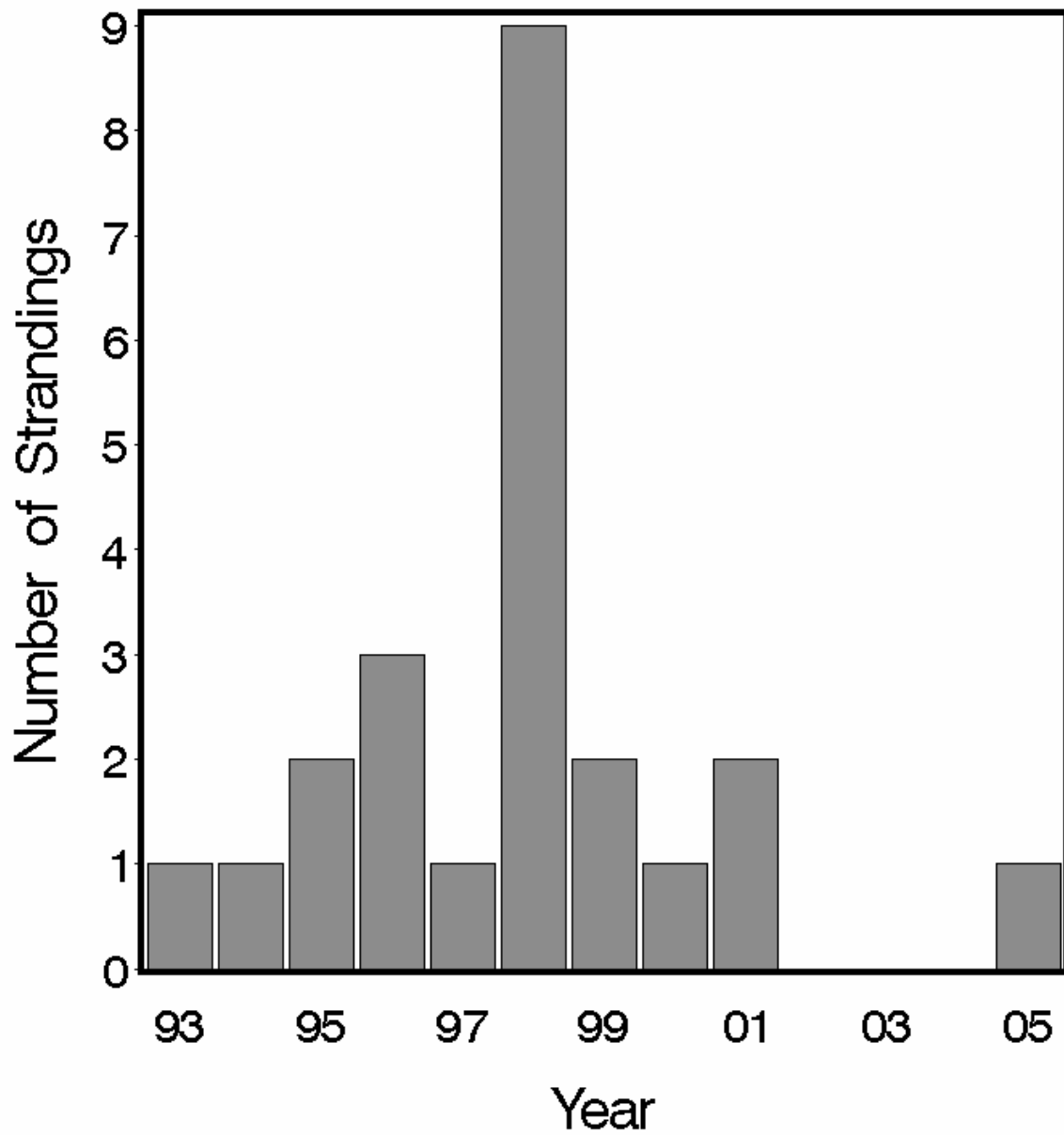


Figure 64. Annual stranding frequencies for hooded seals in Rhode Island alone, 1993–2005.

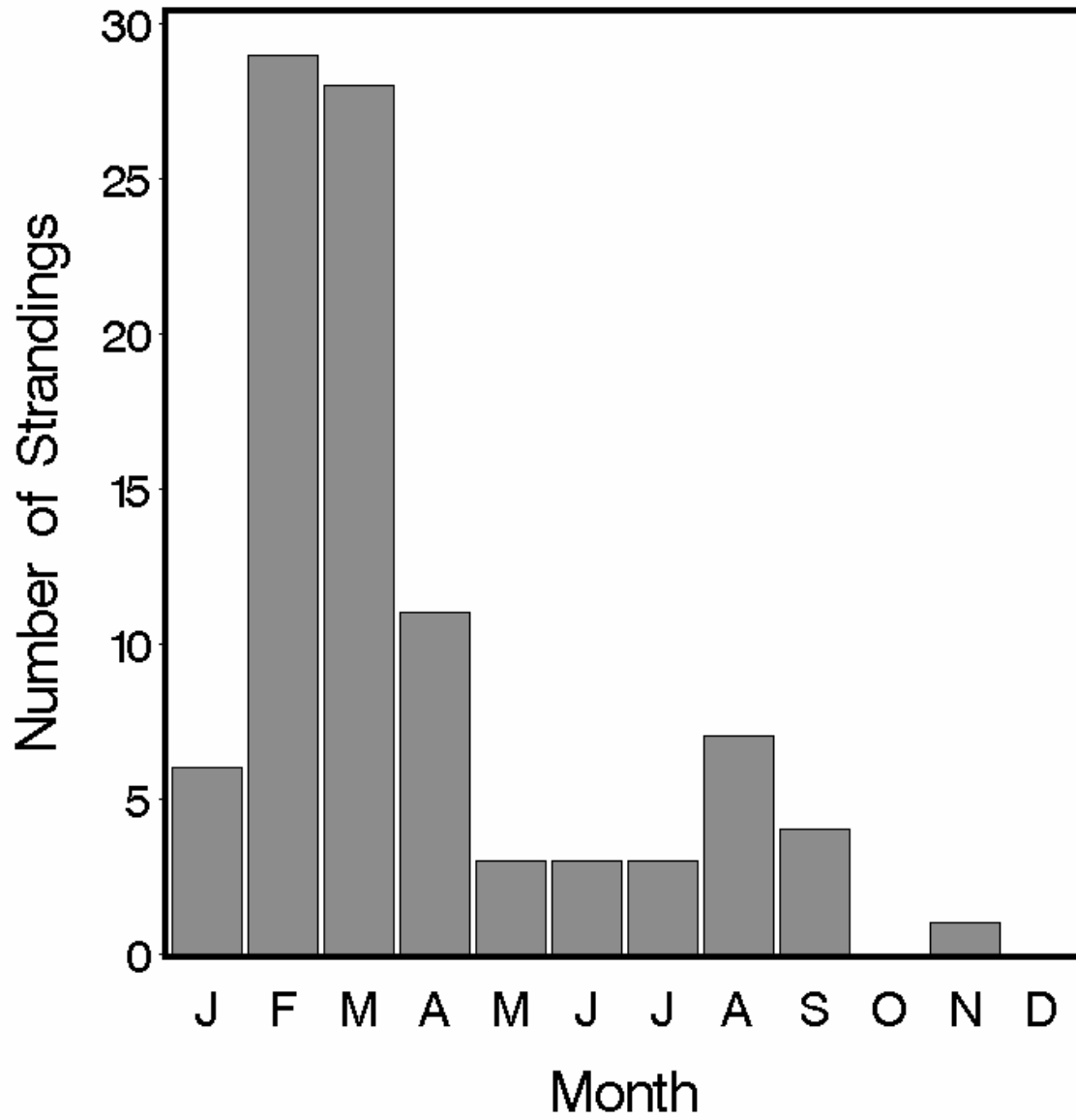


Figure 65. Monthly stranding frequencies of hooded seals in the Rhode Island study area.

DRAFT

3.2.27. Ringed Seal *Pusa hispida* (Schreber, 1775)

Description

Ringed seals are the smallest pinnipeds of the North Atlantic, with average adult lengths of 1.2–1.4 m (Frost and Lowry, 1981) and maximum length of about 1.6 m (Jefferson et al., 1993). They are plumper than harbor seals (maximum girth up to 80% of total length), with a shorter, almost cat-like snout. The ventral aspect of the adult coat is a solid light silver; dorsally it is dark gray with oval spots that are about the same or slightly darker than the background and surrounded by pale rings. Pups are born covered in a fine white lanugo that is shed between 2 and 8 weeks after birth; juveniles are colored like adults but without the spots.

Status

Ringed seals are not listed under the U.S. Endangered Species Act or on the Rhode Island state list, and are classified as Least Concern on the IUCN Red List. There is no reliable estimate of ringed seal abundance in the North Atlantic. World-wide abundance has been estimated as high as 7 million (Kelly, 1988) and is at least 2.5 million (Miyazaki, 2002). Ringed seals are taken in subsistence hunts by natives around the Arctic (MacLean et al., 2002), but are not hunted commercially.

Ecology and life history

Ringed seals feed on a variety of small fishes and crustaceans (Frost and Lowry, 1981; Miyazaki, 2002), and they are known to dive to depths of at least 90 m (Lavigne and Kovacs, 1988).

Ringed seal pups are born in late March or early April in birth lairs constructed under the snow on stable, shore-fast ice (Frost and Lowry, 1981; Miyazaki, 2002). Pups are weaned in 5–7 weeks and then abandoned, at just around the time of ice break-up. As is typical of all phocids, mating takes place just around weaning (Riedman, 1990), therefore in late April–early May in ringed seals. Implantation of the embryo is then delayed for some time, synchronizing pupping to

a tight annual cycle. That delay varies with the length of lactation, and is about three and a half months in ringed seals. Both sexes mature at 5–7 years of age, and record longevity is 43 years (Frost and Lowry, 1981; Miyazaki, 2002).

General distribution

Ringed seals are widely distributed around the Arctic (Miyazaki, 2002). In the North Atlantic they occur from Labrador, Iceland, and Norway to the North Pole, with isolated populations (recognized as three separate subspecies) in the Baltic Sea, Lake Ladoga, and Lake Saimaa (Frost and Lowry, 1981). They are associated with sea ice most of the year. Their distribution both during the pupping season and in the remainder of the year is extremely dispersed, likely driven by polar bear predation.

Historical and recent occurrence

There are no ringed seal records in Rhode Island. In New York waters, ringed seals are known only from very rare strandings and opportunistic sightings. There is one confirmed ringed seal stranding record in the Northeast regional dataset—in Easthampton in eastern Long Island in February 1998, and several other earlier anecdotal observations (Sadove and Cardinale, 1993). There was also a live-stranded sub-adult male ringed seal on the north shore of Long Island in 2006, which was rehabilitated and released (RFMRP, 2006).

Conclusions

Ringed seals are clearly rare, accidental visitors to the Rhode Island study area and are not recorded from the SAMP area. They are not a concern relative to the SAMP.

3.2.38. West Indian manatee *Trichechus manatus* Linnaeus, 1758

Sirenia includes the marine and aquatic species known collectively as “sea cows” (Reynolds and Odell, 1991; Shoshani, 2005). There are four extant species in two families—three manatees of the tropical Atlantic (*Trichechus* spp.: Trichechidae) and the dugong (*Dugong dugon*: Dugongidae) of the tropical Indo-Pacific. A fifth species, Steller’s sea cow (*Hydrodamalis gigas*), a sub-Arctic dugongid found only around the Commander Islands in the western Bering Sea, was both discovered and extirpated in the 18th Century.

Sirenians are fully aquatic, with many adaptations similar to those seen in the cetaceans, including a more or less fusiform body, absence of hair except for well-developed vibrissae on the muzzle, loss of the hind limbs, forelimbs modified into paddle-like flippers, and swimming powered by a horizontally flattened tail. They were long considered to be herbivorous cetaceans (e.g., Hamilton, 1839) and De Kay (1842) included the “Manatidae” as family I in the Cetacea, but sirenians are not closely related to the other marine mammals in the Cetacea and Carnivora. All sirenians are obligate herbivores, feeding primarily on seagrasses and also on submerged and floating aquatic vegetation.

Description

West Indian manatees are large, rotund, docile, and slow-moving, ranging in length from 2.5 to 4.5 m (Jefferson et al., 1993; Wynne and Schwartz, 1999). The body is tapered and somewhat streamlined, with a relatively small head and a large, rounded tail. The skin is relatively smooth, hairless, and uniformly gray or gray-brown, often with distinctive scars from boat collisions. The eyes are small and deep-set, and the fleshy muzzle is covered with stiff vibrissae. The only teeth present, except for vestigial incisors that are resorbed soon after birth, are 5–7 molars in each upper and lower jaw, which are replaced from the rear and drop out at the front of the row when worn (Husar, 1978; Caldwell and Caldwell, 1985). The skull and post-cranial bones are very dense, perhaps adapted to serve as internal “dive weights.” The forelimbs are relatively long and flexible, with blunt, rounded ends and elephant-like nails. The forelimbs are often used in feeding, in conjunction with the nearly prehensile upper lips, for manipulating vegetation into the mouth.

Status

West Indian manatees are classified as Endangered under the U.S. Endangered Species Act, are not included on the Rhode Island state list, and are classified as Vulnerable on the IUCN Red List, although both the Florida population and the population in the West Indies are classified as Endangered. Florida manatee numbers have been assessed since 1991 by aerial surveys following winter cold fronts, which concentrate the animals into the available warm-water refuges (FFWCC, 2006). The highest count was 3,807 in January 2009, more than 500 higher than the previous high of 3,300 in 2001 (FFWCC, 2009). Mortality is high, averaging 183 deaths annually since 1974 and more than 300 per year in the last decade or so. About 30% of the mortality can be attributed as human-related mortalities, primarily collisions with watercraft (24%) but also including crushing in floodgates and canal locks, poaching, ingestion of persistent debris, and drowning or entanglement in fishing gear. Categories of natural mortalities include perinatal, cold stress, and biotoxins from “red tides.”

Ecology and life history

Manatees feed on a wide variety of marine, estuarine, and aquatic vegetation, including seagrasses, algae, mangrove leaves and seedlings, floating aquatic plants, overhanging and streamside terrestrial plants, and even acorns (Reynolds and Odell, 1991). Manatees typically spend 6–8 hours a day feeding. They are not deep divers, but are capable of remaining submerged for as long as 20 minutes.

Manatees become sexually mature at 6–10 years old and about 2.7 m in length (Reynolds and Odell, 1991). Gestation is believed to be about 12–13 months. Calves are born at about 1.2 m and 60 kg. In Florida, births can occur at any time of year, most are in spring and summer. Lactation lasts for about a year, although a calf may remain with its mother for another year. Intervals between births range from 2 to 5 years.

General distribution

West Indian manatees occur in warm subtropical and tropical waters of the western North Atlantic (Husar, 1978; Caldwell and Caldwell, 1985; Reynolds and Powell, 2002). They are primarily found in freshwater systems, estuaries, and shallow, nearshore, coastal waters. The species ranges from the southeastern U.S. to Central and northern South America, the Caribbean, and the West Indies. Florida manatees disperse in summer to feeding grounds as far north as the Chesapeake (Reynolds and Odell, 1991; Reynolds and Powell, 2002).

Historical occurrence

There are no historical records of manatees in the Rhode Island study area.

Recent occurrence

One individual (an adult male known as “Chessie”) was the first manatee confirmed to occur in Rhode Island waters. He was captured in a Chesapeake tributary as winter approached in 1994 (ORG, 2003). He was transported to Florida, equipped with a radio transmitter that could be tracked by satellite, and released. When the weather warmed the following spring, he departed from Florida and headed north along the coast. Chessie did not make the expected left turn into Chesapeake Bay, but continued north past New Jersey into New York Harbor and then into Long Island Sound. He traveled the entire length of the Sound before finally reaching Point Judith on the 16th of August. Then he turned around and went back. He eventually lost the tag near New Haven, Connecticut, but was sighted in Virginia on 23 September and recognized back in his normal winter habitat in Florida in November. Chessie was re-sighted in August 2001 in Virginia (USGS, 2006).

Three other manatees have since visited the study area (Fig. 66). A manatee was seen in Montauk Harbor for about a week in late July of 1998 (Kimberly Durham, Riverhead Foundation, pers. comm.). Another wayward manatee visited southern New England in the summer of 2006, leaving an extensive trail of sighting reports (Hamilton and Puckett, 2006). It was first reported in Ocean City, Maryland on 11 July. It was then seen in Delaware Bay on 14

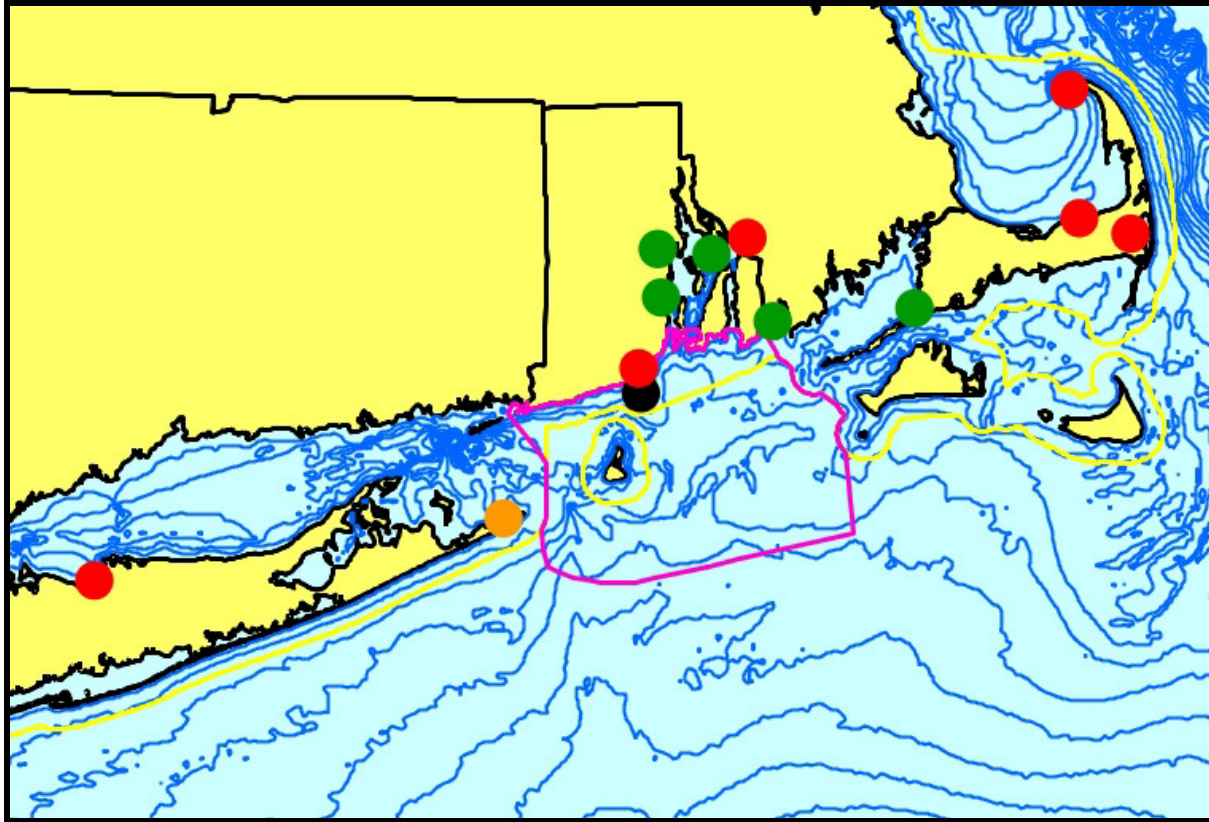


Figure 66. Sightings of four individual manatees in and near the Rhode Island study area, in 1995 (black), 1998 (orange), 2006 (green), and 2008 (red).

July and at Barnegat Inlet, New Jersey on 22–23 July. Next it lingered for about a week in the Hudson River, from the 1st to the 8th of August, and was sighted off Manhattan and Harlem and more than 40 km upriver north of the Tappan Zee Bridge in Westchester County. The next sighting was far to the east, in Quissett Harbor near Woods Hole, Massachusetts, on 17 August, before it turned around and started on the return trip. It was seen on the 19th in Westport, Massachusetts, and then caused a brief media furor in Rhode Island—drinking from a storm drain for the Channel 10 television cameras in a marina in Greenwich Bay on 20 August, and making brief appearances in Wickford harbor on the 22nd and Bristol harbor on the 27th or 28th. It has not been seen since (though there was an undocumented report of a manatee in Barnegat Bay, New Jersey in September), and is assumed to have returned home.

The last manatee to visit Rhode Island was in 2008; its locations and movements were extracted from a series of media reports. It first was seen on 11 August off Crown Point, on the South Kingstown side of Point Judith Pond and near Skip's Dock in Snug Harbor. The next

report, on 21 August, came from a family fishing from a dock in Stony Brook Harbor on the north shore of Long Island. Then it laid low for almost a month, until the Massachusetts Division of Marine Fisheries reported on 19 September that a manatee had been seen for a couple of days under the Braga Bridge in Fall River. It showed up five days later on the 24th in a cove off Pleasant Bay in Harwich, Massachusetts—on the outside of Cape Cod. It apparently then went around the outer Cape, showing up on the 29th near the whale-watching boats in Provincetown harbor. The next day it was seen in Sesuit Harbor in Dennis, in the southeast corner of Cape Cod Bay, where it remained until 11 October. On that day it was captured for relocation to Florida, however it died in transit from cold stress.

Conclusions

Florida manatees are clearly accidental visitors to the SAMP study area, and are most likely to occur on those occasions in shallow waters very close to shore, where there are sea grass beds. They can safely be ignored in planning for any developments in the SAMP area.

DRAFT

3.2.39. Leatherback sea turtle *Dermochelys coriacea* (Vandelli, 1761)

A turtle, encased within its shell (comprised of an upper carapace and a lower plastron), is something that is instantly recognizable to most people. The sea turtles include seven or eight species in two closely related families. Sea turtles spend their entire lives at sea except for nesting; adult females deposit their eggs in nests dug above the high-tide mark on sandy beaches in the tropics and sub-tropics. Their limbs are adapted for swimming—modified into simplified, flattened flippers. Sexes are generally indistinguishable, except that adult males usually can be identified by their very long tails. Only five species typically occur in the North Atlantic, although one other may occur accidentally in the West Indies (Ernst et al., 1994). Four species are known from the Rhode Island study area—leatherback, loggerhead, Kemp’s ridley, and green sea turtles (Table 1). The hawksbill sea turtle is known from single historical stranding records in Massachusetts in 1968 (Lazell, 1980; McAlpine et al. 2007) and New York in 1938 (Morreale et al., 1992), and is considered to be hypothetical for this analysis.

Description

The leatherback sea turtle is one of the largest living reptiles, and is the only living species in its family, Dermochelyidae (Ernst et al., 1994). Leatherbacks differ from all other sea turtles in lacking the outer layer of keratin plates or scutes on the shell. The bony shell, composed of a mosaic of thousands of tiny dermal bones, is covered by a layer of soft, leathery skin. Carapace lengths (the standard for measuring a turtle is to measure the length and width of the carapace without including the head, tail, or limbs) of adults are up to 1.8–2 m or more, and large leatherbacks can reach weights of 1,000 kg (Wynne and Schwartz, 1999). The carapace tapers from front to back, and there are seven longitudinal ridges. The overall color is black, and there are usually white or pinkish spots, especially underneath. The front flippers are very long and flexible; both front and rear flippers lack claws.

Status

Leatherback sea turtles are classified as Endangered under the U.S. Endangered Species Act, as Federally Endangered on the Rhode Island state list, and as Critically Endangered on the IUCN Red List. The status of populations in the North Atlantic does not seem to be as precarious as it is for those in the Pacific, where nesting populations have declined by more than 80%. Estimates of the total number of adult females in the world declined from 115,000 in 1982 to 20–30,000 in 1996 (IUCN, 2008).

Estimates of sea turtle population abundance for any region are rare or non-existent. Sea turtles are wide-ranging, difficult to detect at sea, and capable of long submergences; in addition, aerial surveys detect only individuals above a certain size threshold—about 75 cm carapace lengths (Shoop and Kenney, 1992). The northeastern U.S. is one of a few locations where there have been published estimates of abundance of pelagic sea turtle populations, based on line-transect aerial surveys (CETAP, 1982). Shoop and Kenney (1992) summarized the CETAP estimates, which showed that 100–900 leatherbacks occurred off the northeastern U.S. in the summer. Those numbers are minimum values, since they do not account for animals missed because they were below the surface and not visible when the survey aircraft passed.

Abundance is more typically indexed by counts of nesting adult females. There are seven known leatherback nesting populations in the Atlantic (reviewed in TEWG, 2007; NMFS & USFWS, 2007c), with the total number of adults estimated at 34,000–94,000. The Florida population grew from 98 nests in 1988 to 800–900 per year in the early 2000s, with a 17% increase rate on index beaches. The Northern Caribbean population nests on Puerto Rico and the Virgin Islands. Nests in Puerto Rico increased at 10% annually, from 9 in 1978 to 469–882 in 2000–2005. Nesting in the U.S. Virgin Islands increased at 10% from 1986 to 2004, and at 13% from 1994 to 2001. There were 143 nests in 1990 and 1008 in 2001. The number of nests in the British Virgin Islands increased from a few in the late 1980s to 35–65 in the 2000s, at a rate of 20% in 1994–2004. The Western Caribbean populations nests from Honduras to Colombia, especially in Costa Rica, Panama, and Colombia, and shows declining trends. At the major nesting beach in Tortuguero, Costa Rica, nesting declined by 68% between 1995 and 2006. The Southern Caribbean population nests in Guyana, Suriname, French Guiana, Trinidad, Dominica, and Venezuela, with perhaps 40% of the world's leatherback nesting in Suriname and French

Guyana. The trend is generally stable to a slight increase. The other three populations are in the South Atlantic—Brazil, West Africa, and South Africa.

All sea turtle species share a nearly identical suite of survival threats (reviews in NRC, 1990; Lutcavage et al., 1997; NMFS & USFWS, 2007a, 2007b, 2007c, 2007d). Harvesting of adults and eggs depleted populations in many areas of the world. Predators, both natural and introduced, take significant numbers of eggs, hatchlings, and juveniles. There are two major anthropogenic impacts on sea turtles—loss or degradation of nesting habitat and incidental capture in fisheries. While there are natural sources of habitat loss (e.g., beach erosion, hurricanes), development of beachfronts for residences or tourism, beach armoring, disorientation of hatchlings by artificial lighting, sand mining, beach replenishment, and spread of non-native vegetation are much more serious. Sea turtles are captured frequently in many fisheries, including pelagic longlines, high-sea driftnets, sink gillnets, pound nets, trap and pots, and trawls; turtles can also be entangled in other types of persistent debris. Other anthropogenic impacts include boat strikes and plastic ingestion.

Lewison et al. (2004) estimated that 50,000 leatherbacks were killed in pelagic longline fisheries worldwide in 2005, mainly in the Pacific. About 3,000 a year were killed in the U.S. Atlantic and Gulf of Mexico shrimp fishery; leading NMFS to require a larger escape opening in Turtle Excluder Devices (TEDs) beginning in 2003 (NMFS & USFWS, 2007c). Morreale and Standora (1998) reported 8 leatherback turtles that were entangled in fishing gear near Long Island during 1987–1992 and released after tagging. In Rhode Island waters, a leatherback entangled in buoy lines for lobster traps is the most common sea turtle entanglement.

Ecology and Life History

The basic picture of sea turtle life history has long been known, is essentially the same across all species, and has been well-described in the works of Archie Carr and his colleagues (Carr, 1967, 1980, 1986, 1987, 1995; Carr and Meylan, 1980; Hamner, 1988; Musick and Limpus, 1997). An adult female crawls up onto a sandy beach, digs a nest hole, deposits a clutch of eggs, covers it over, and returns to the sea. About two months later a batch of hatchlings emerges from the nest and scrambles down the beach and into the ocean. The hatchlings swim straight out to sea and disappear until they next show up as small juveniles—long termed the

“lost year.” Carr theorized, which was later confirmed, that hatchlings get passively carried in ocean current systems and collect in sargassum patches and other surface convergences, where they feed on a wide variety of plant parts and invertebrates. Pelagic post-hatchlings grow into small juveniles, who move into developmental habitats, usually in coastal waters. Larger juveniles move into the same foraging habitats as the adults.

Sea turtles are very difficult to age, so that the durations of the various life-stages were not known. For leatherbacks, growth seemed to be relatively fast, and the age at maturity had been estimated from as short as 2–3 years to as long as 13–14 years (Pritchard and Trebbau, 1984; Rhodin, 1985; Zug and Parham, 1996; Dutton et al., 2005). More recent work, however, suggests that it may average 29 years (Avens and Goshe, 2008).

Adult leatherback sea turtles feed mainly on jellyfish and other gelatinous invertebrates, especially the lion’s mane jelly *Cyanea capillata* (Bleakney, 1965; Lazell, 1980; Bjorndal, 1985; Mortimer, 1995).

General distribution

The leatherback sea turtle has the widest distribution of any species of sea turtle, extending worldwide from tropical and subtropical at least into cold-temperate waters and sometimes even more poleward (Ernst et al., 1994; NMFS & USFWS, 2007c). In the North Atlantic, leatherbacks have been observed in waters of the U.S., Nova Scotia, Europe, the eastern Mediterranean, Newfoundland and Labrador, Greenland, the North Sea, and the Barents Sea (Bleakney, 1965; Brongersma, 1972, 1995; Threlfall, 1978; Goff and Lein, 1988; Marquez, 1990; Casale et al., 2003; Hays et al., 2004, 2006; James et al., 2005; McAlpine et al., 2007). They are capable of maintaining a body temperature well above ambient through a combination of anatomy, physiology, and behavior (Frair et al., 1972; Greer et al., 1973).

Off the northeastern U.S., leatherbacks were sighted commonly in summer in shelf waters from North Carolina to Maine, and in much lower numbers in spring and fall (Shoop and Kenney, 1992). The densest aggregation of sightings was in the nearshore waters south of central Long Island. Despite being present in much lower numbers than loggerheads (less than 5% of the number of sightings) leatherbacks were far more likely to occur within the Gulf of Maine north of Cape Cod—consistent with their known tolerance for colder water.

Historical occurrence

Lazell (1980) reported that the first recorded occurrence of a leatherback turtle in New England was in 1886 by the Monomoy lighthouse keeper. However, Babcock (1919) reported that the first New England occurrence was in Massachusetts Bay in 1824, and that the specimen was in the collection of the Boston Society of Natural History. He reported two earlier records—in 1811 at an unknown locality and in 1816 at Sandy Hook, New Jersey. He listed a total of 31 known records between 1811 and 1917, ranging from New Jersey to Maine, including three from Rhode Island and seven others from the Rhode Island study area. The Rhode Island records included one in Narragansett Bay in 1878 and two off Southeast Point, Block Island around 30 July 1886. The plate illustrating leatherbacks in his monograph included two photos of one of the Block Island specimens. The other leatherback records in the study area included: 1826—Long Island Sound; 1875—one at Stonington, Connecticut, another between New London and Montauk; 1876—Stonington; 1879—Buzzards Bay at Marion, Massachusetts; 1891—caught in a fish trap in Buzzards Bay near Woods Hole; 1907—fouled in an anchor rope a few miles south of Noman's Land.

Lazell (1980) also mapped a number of leatherback sightings from Brongersma (1972) and Lazell (1976) along the New England coast from Rhode Island to Downeast Maine, concluding that “the greatest concentrations of non-nesting leatherback records in the Atlantic are around the Gulf of Maine.”

Recent occurrence

Leatherback turtles occur relatively commonly in the Rhode Island study area (Fig. 67), and are almost entirely limited to summer (57.7%) and fall (41.6%). Leatherbacks occurred over much of the continental shelf in the study area. There is an aggregation of occurrences in the SAMP area, but 20 of those are strandings on Block Island extracted from Nawojchik (2002). There is also somewhat of an aggregation south of central Long Island, in the same area noted by Shoop and Kenney (1992) as a leatherback concentration area. There were 24 sightings in summer and 5 in fall from the whale-watching boats.

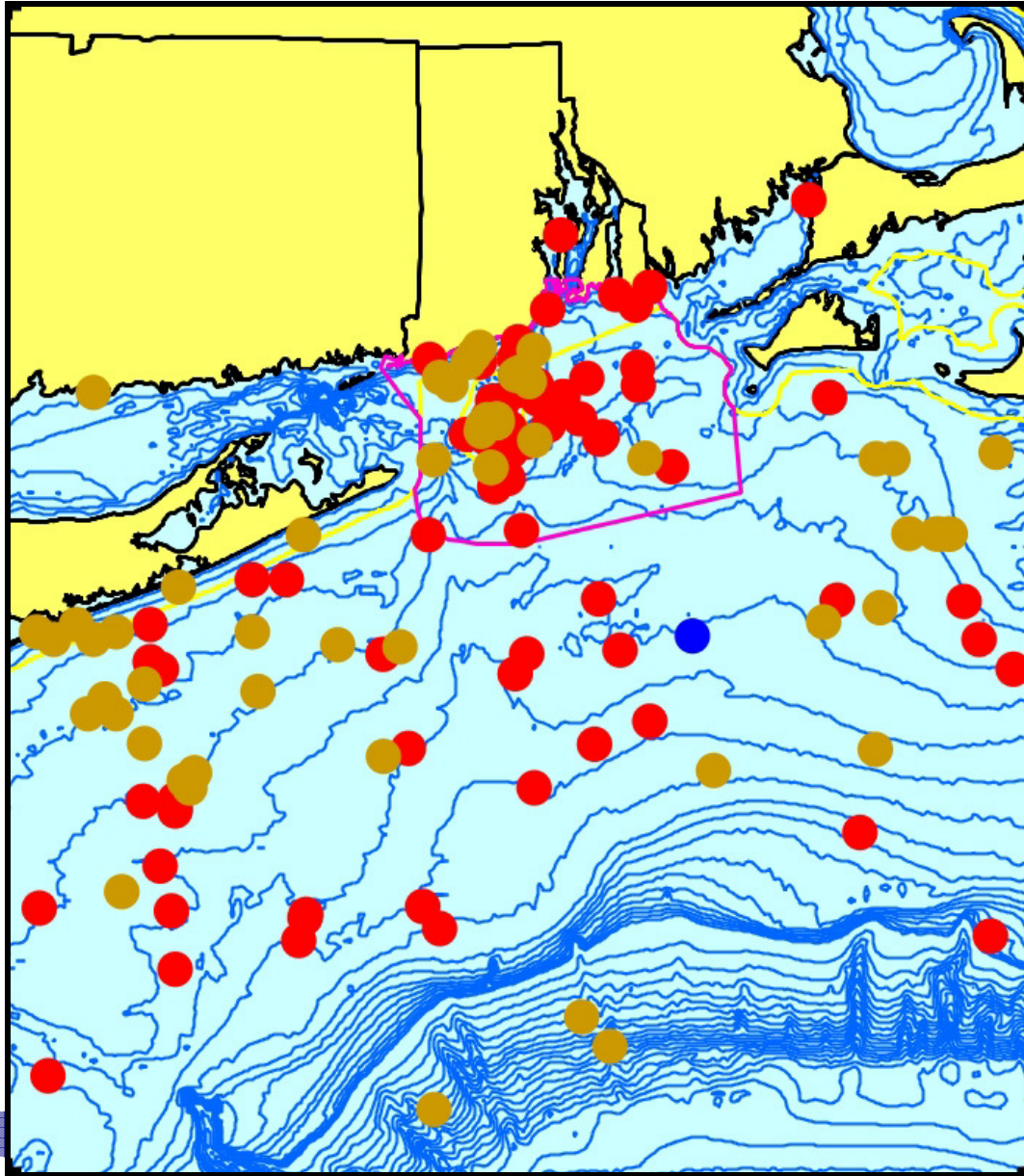


Figure 67. Aggregated sighting, stranding, and bycatch records of leatherback sea turtles in the Rhode Island study area, 1974–2008 ($n = 142$: winter = 1, spring = 0, summer = 82, fall = 59).

The relative abundance patterns (Fig. 68) show leatherbacks to be relatively dispersed and not particularly abundant. The areas of higher abundance are all beyond the boundary of the mapped area, and the model output does not predict occurrence within the SAMP area.

Leatherback strandings are relatively common in Rhode Island, however we did not have access to most of those records. Nawojchik and St. Aubin (2003) reported that, of the 146 sea turtle

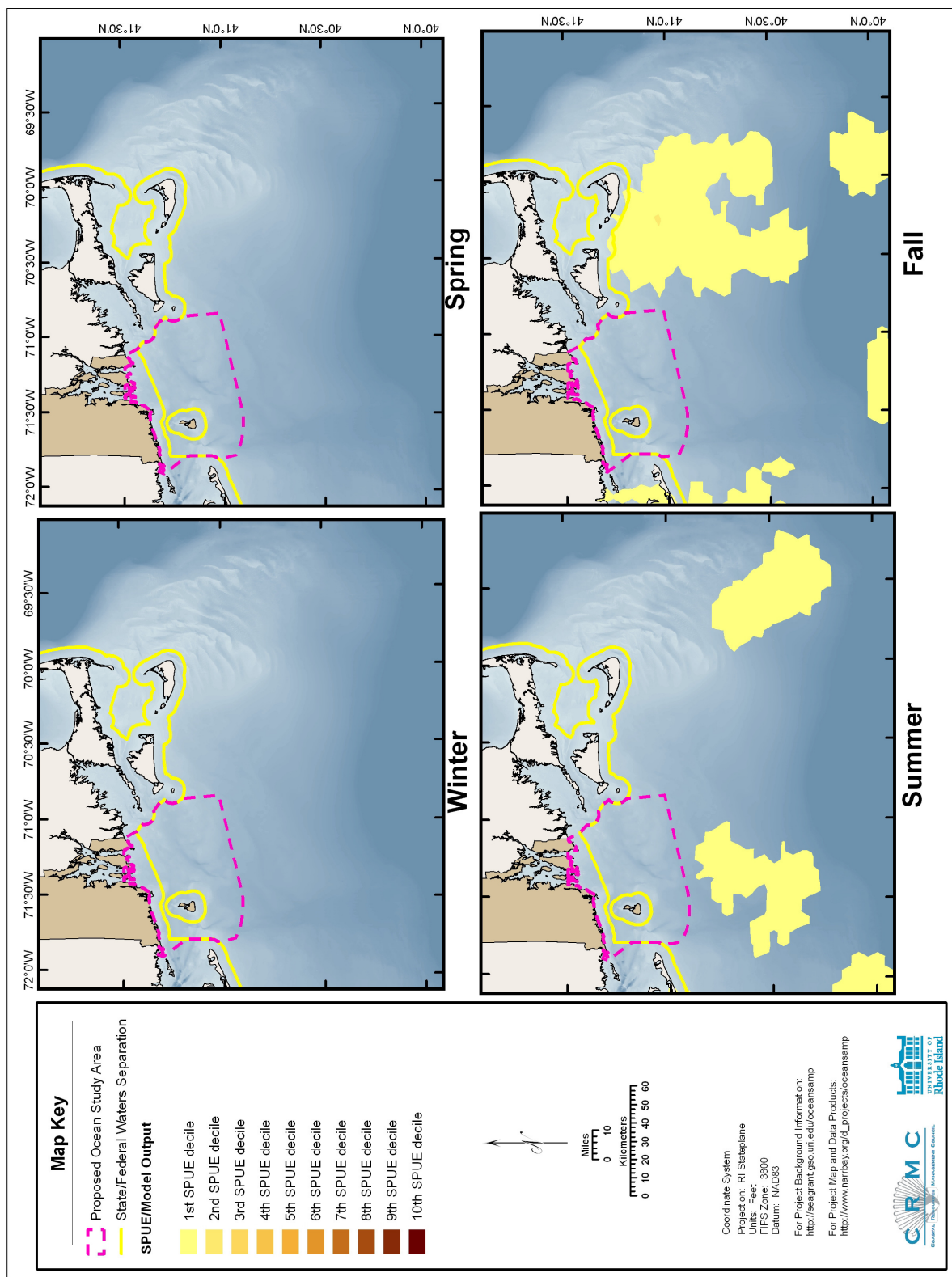


Figure 68. Modeled seasonal relative abundance patterns of leatherback sea turtles in the Rhode Island study area, corrected for uneven survey effort.

strandings responded to by Mystic Aquarium from 1987 to 2001, 124 (84.9%) were in Rhode Island, and 120 of the 146 were leatherbacks. All strandings occurred during June through November, with the biggest numbers in August and September. This is fully consistent with the sighting data. Leatherbacks were the only sea turtle species to strand on Block Island.

Conclusions

The relative abundance analysis does not predict that leatherback sea turtles will occur in the Ocean SAMP area (Fig. 68), however the more extensive data, including sightings from the whale-watching boats, show that leatherbacks do occur in the SAMP area. The lower survey effort in summer and fall may explain some of the difference. Given the leatherback's status as an Endangered species, they should be considered in any planning process.

3.2.40. Loggerhead sea turtle *Caretta caretta* (Linnaeus, 1758)

All of the sea turtles other than leatherbacks belong to a separate family—Cheloniidae, the so-called “shelled” sea turtles. The bony shell is much thicker and heavier than in leatherbacks, and it is covered by a layer of keratin plates or scutes. The arrangements and numbers of scutes are important characters used to identify species, especially small individuals or decomposed carcasses.

Description

The loggerhead sea turtle is one of the two species of larger shelled turtles found in the North Atlantic, with adult carapace lengths of 85-120 cm (Wynne and Schwartz, 1999), although the maximum known length was 213 cm (Ernst et al., 1994). The shell is shaped like a broad oval, tapering toward the rear. The head is much larger relative to body size than in the other sea turtle species, with broad crushing surfaces on both the upper and lower jaws. The color is a distinctive yellowish- to reddish-brown.

Status

Loggerhead sea turtles are classified as Threatened under the U.S. Endangered Species Act, as Federally Threatened on the Rhode Island state list, and as Endangered on the IUCN Red List. Shoop and Kenney (1992) estimated the summer pelagic population off the northeastern U.S. of large juveniles and adults detectable from aerial surveys at 2,200–11,000, not accounting for diving behavior, with less than half as many in spring and fall. There are no more recent comparable estimates.

TEWG (2000), Ehrhart et al. (2003), and NMFS & USFWS (2007d) reviewed the status of loggerhead nesting populations in the North Atlantic. The largest is in the southeastern U.S. and Gulf of Mexico, which is the second largest loggerhead nesting population in the world after the one in the eastern Indian state of Orissa. The total numbers of nests and nesting females per year are estimated at 53,000–92,000 and 32,000–56,000, respectively. The population is divided into five sub-populations. The Northern sub-population nests in Georgia and the Carolinas. The

average nests per year is 5,151, with a 1.9% declining trend over 1989–2005. The largest sub-population is South Florida, with an average of 65,460 nests and 15,966 females and a declining trend of 22.3% in 1989–2005. That decline may be accelerating. The Dry Tortugas sub-population shows no detectable trend and has annual averages of 246 nests and 60 females. The averages for the Florida Panhandle sub-population in 1995–2005 were 910 nests, 222 females, and a declining trend of 6.8%. The Yucatan sub-population increased from 903 nests in 1987 to 2,331 in 2001, but may currently be decreasing.

Other western North Atlantic populations include the eastern Bahamas, with 500–600 nests per year, and Cuba, with 250–300. Loggerheads formerly nested on Jamaica, Haiti, the Dominican Republic, and Puerto Rico, but no longer do so. The nesting population in northeastern Brazil has shown a long-term increase, with 4,837 nests in 2004. The only nesting population in the eastern North Atlantic is in the Cape Verde islands, with several thousand nests per year. Loggerheads also nest in the eastern Mediterranean, where nest counts can exceed 7,000 per year, although monitoring is incomplete.

Impacts on loggerheads are the same as for other sea turtles. Lewison et al. (2004) estimated that 60,000–80,000 loggerheads were killed annually by incidental capture in Atlantic pelagic longline fisheries, primarily in the western Mediterranean, and 200,000 globally. NRC (1990) estimated that, prior to regulations requiring TEDs, 5,000–50,000 loggerheads were killed each year in the southeastern U.S. and Gulf of Mexico shrimp trawl fishery.

In southern New England, juvenile sea turtles sometimes strand dead, comatose, or seemingly paralyzed. The event happens in the fall of the year, when water temperatures decline, and is referred to as “cold-stunning.” In 1985, 56 cold-stunned turtles stranded in eastern Long Island (Meylan, 1986), sparking the establishment of a monitoring, research, and rehabilitation program. A similar program exists in Cape Cod Bay.

Ecology and Life History

Loggerheads follow the typical sea turtle life history pattern. Post-hatchlings disperse and are entrained in ocean currents (Carr, 1986). Small juveniles are present in high abundance around the Azores (Bolten, 2003), where they remain resident for extended periods and feed on pelagic invertebrates such as siphonophores, jellies, salps, gastropods, barnacles, and isopods.

Small juveniles may also congregate on the Grand Banks off Newfoundland. In the Mediterranean, genetic profiling has shown that small and medium juvenile loggerheads come from both the eastern Mediterranean nesting population and from western North Atlantic populations (B. W. Bowen et al., 1993). Eventually juveniles reach the size where they return to coastal waters, first into shallower developmental habitats in bays and estuaries and then into adult foraging habitats. The diet of juveniles in developmental habitats is dominated by crabs (Burke et al., 1993). Adults feed on a wide variety of benthic prey, including bivalves, gastropods, crabs, sea pens, anemones, and seaweeds (reviewed by Bjorndal, 1997).

General distribution

Loggerhead sea turtles are distributed worldwide in subtropical and temperate waters (Ernst et al., 1994; Ehrhart et al., 2003). In the western North Atlantic, they are common off the southeastern U.S. and in the Gulf of Mexico. Off the northeastern U.S., there are few sightings north of the latitude of Long Island, with only one in the northern Gulf of Maine (CETAP, 1982; Shoop and Kenney, 1992), although there are inshore records from Nova Scotia and juveniles are commonly taken as bycatch in fisheries on the Newfoundland Grand Banks (Bleakney, 1965; Brongersma, 1972, 1995; Bolten, 2003; McAlpine et al., 2007).

From Long Island south to North Carolina, loggerhead occurrence is strongly seasonal (CETAP, 1982; Shoop and Kenney, 1992). They are nearly absent in winter. In spring they spread northward from south of Cape Hatteras. The distribution is most extensive in summer—from the shore to the mid-shelf area and also along the outer shelf. The distribution then contracts southward in the fall.

Historical occurrence

Babcock (1919) stated that loggerhead turtles “not uncommonly visit Long Island Sound and the Massachusetts coast.” He reported that “a number of specimens usually about two feet in length [were] taken every year” in fish traps in Menemsha Bight of the northwestern side of Martha’s Vineyard. He also included an interesting report that small loggerheads were “taken in Long Island Sound in a benumbed condition as late as December 4,” possibly one of the first

reports of cold-stunning from the region.

Lazell (1980) wrote that loggerhead turtles were “common in New England waters and the Canadian portions of the Gulf of Maine.” However, McAlpine et al. (2007) suggested that Lazell was going beyond the limits of his available data in trying to make his point, and that loggerheads were rare north of Cape Cod.

Recent occurrence

The occurrence of loggerhead sea turtles in the Rhode Island study area (Fig. 69) is fully consistent with the reports of CETAP (1982) and Shoop and Kenney (1992). Sightings are strongly concentrated in the summer (73.4%), and then the fall (26.2%). The concentration of sightings is highest in the western half of the study area, and sightings in the eastern half are more on the outer part of the shelf. There is one cluster of sightings in the southwestern quarter of the SAMP area, which includes the majority of the 10 summer sightings and 1 fall sighting from the whale-watching boats.

As with leatherbacks, the areas of high relative abundance were to the west of the area mapped (Fig. 70). Within the study area, most areas of predicted loggerhead summer and fall occurrence were offshore of the SAMP area. One area of lowest abundance extended into the SAMP area's southwest corner in the fall, and there was an area of moderate occurrence on the outer shelf southeast of Nantucket in summer.

We did not have access to sea turtle stranding data for the study area. Nawojchik and St. Aubin (2003) reported that 23 of the 146 sea turtle strandings in Rhode Island and Connecticut (15.8%) were loggerheads—many fewer than leatherbacks even though the population in shelf waters is estimated to be an order of magnitude larger.

Many of the loggerheads that occur in coastal embayments such as Peconic Bay in eastern Long Island or Cape Cod Bay are juveniles that are too small to be detected during surveys. Morreale et al. (1992) reported that 28 juvenile loggerheads collected in eastern Long Island in 1986–1988 ranged from 36.6 to 59.6 cm, with a mean of 49.5. Over a longer period from 1984 to 1998, the mean size of 298 juvenile loggerheads in Long Island was smaller at 45.5 cm (SD = 18.0; Saari et al., 2000). Shoop et al. (1999) considered that 45 cm was the lower end of the 45–85 cm size range of large, benthic-

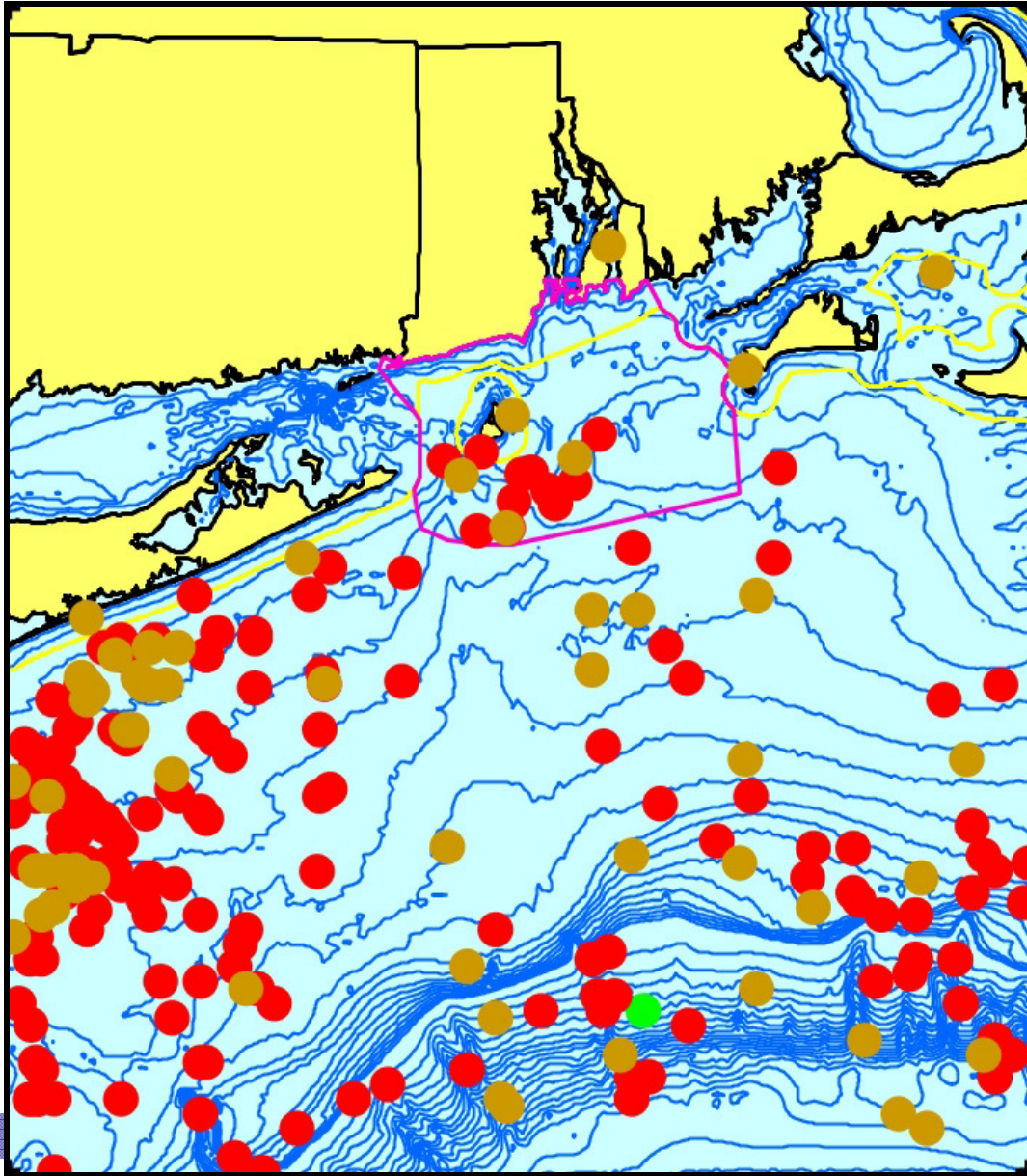


Figure 69. Aggregated sighting, stranding, and bycatch records of loggerhead sea turtles in the Rhode Island study area, 1963–2006 (n = 233: winter = 0, spring = 1, summer = 171, fall = 61).

feeding juveniles found off Georgia. Assuming a normal distribution, that would suggest that loggerheads around Long Island are about half and half small juveniles <45 cm and large juveniles >45 cm.

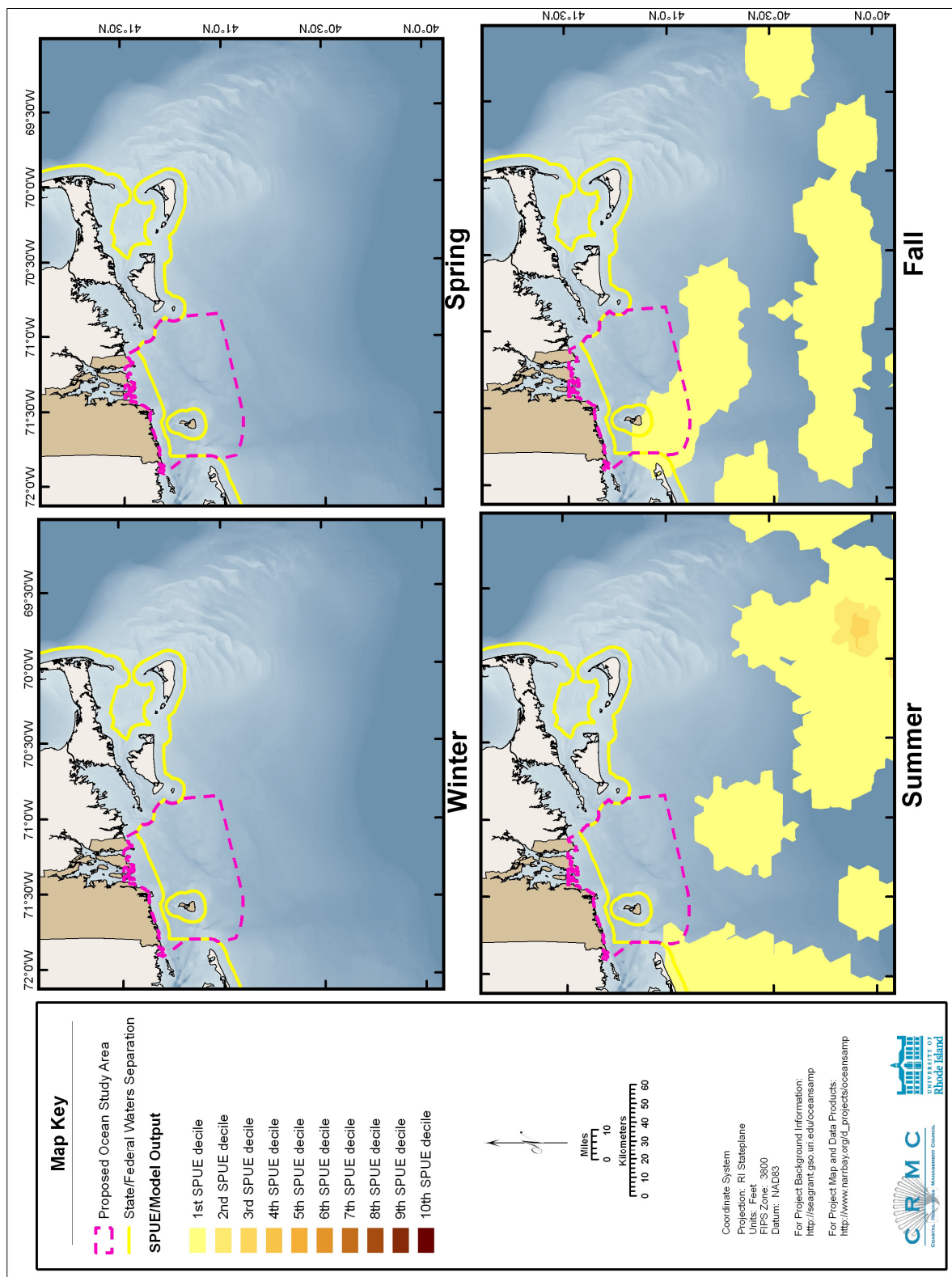


Figure 70. Modeled seasonal relative abundance patterns of loggerhead sea turtles in the Rhode Island study area, corrected for uneven survey effort.

The species proportions differ between areas and between collection methods (Table 6). Collecting turtles for measurement, sampling, and tagging from those caught in fishing gear should sample more across the available size range than collecting individuals debilitated by declining temperatures in the fall (“cold-stunned”), which effects juveniles more than adults. Of 519 turtles live-captured around eastern Long Island in 1984–1998, 298 (57.4%) were loggerheads (Saari et al., 2000). However, of 130 cold-stunned turtles in 1986–1988, only 28 (21.5%) were loggerheads (Morreale et al., 1992). In Rhode Island stranding records, loggerheads are far less frequent than leatherbacks (Nawojchick and St. Aubin, 2003).

Table 6. Comparisons of relative frequencies and percentages of leatherback (LeTu), loggerhead (LoTu), Kemps’ ridley (KRTu) and green (GrTu) sea turtles in different collections from southern New England localities.

Collection and Source	LeTu	LoTu	KRTu	GrTu
RI (85%) & CT (15%) strandings, 1987–2001 (Nawojchik and St. Aubin, 2003)	120 82.2%	23 15.8%	1 0.7%	2 1.4%
Long Island live captures, 1984–1998 (Saari et al., 2000)	0 –	298 57.4%	120 23.1%	101 19.5%
Long Island cold-stunned, 1986–1988 (Morreale et al., 1992)	0 –	28 21.5%	97 74.6%	5 3.8%
Peconic Bay live captures, 2002–2003 (Aguirre et al., 2008)	0 –	2 6.9%	11 37.9%	16 55.2%
Cape Cod Bay cold-stunned, 1979–2003 (Dodge et al., 2008)	0 –	272 21.1%	983 76.3%	30 2.3%

Conclusions

Although loggerhead turtles are much more abundant off southern New England than leatherbacks, they are less likely to occur in nearshore waters or in the SAMP area. Even though they are listed as a Threatened species, they can probably be discounted in planning for any

development in the SAMP area, since mitigation taken for leatherbacks will also benefit loggerheads.

DRAFT

3.2.41. Kemp's ridley sea turtle *Lepidochelys kempii* (Garman, 1880)

Description

Kemp's ridleys are smaller shelled sea turtles, with adult carapace lengths of 60-80 cm (Ernst et al., 1994; Wynne and Schwartz, 1999). Individuals encountered off the northeastern U.S. are mostly juveniles. The shell is slightly heart-shaped to nearly circular, and is usually gray.

Status

Kemp's ridley sea turtles are classified as Endangered under the U.S. Endangered Species Act, are not included on the Rhode Island state list, and are classified as Critically Endangered on the IUCN Red List.

At least 60% of all Kemp's ridley nesting takes place on one 40-km stretch of beach near Rancho Nuevo, Tamaulipas, Mexico (Ernst et al., 1994; TEWG, 2000; NMFS & USFWS, 2007b). As many as 40,000 females nested there on a single night in 1947 (Carr, 1963). By 1985, the total number of nests per year had declined to 740, and nesting females to about 250 (TEWG, 2000). Nesting increased through the 1990s. In 2002 there were over 4,000 nests at Rancho Nuevo and 6,000 in all of Mexico. In 2006 the respective counts were 7,866 and 12,143, with about 100 nests in the U.S., mainly at Padre Island, Texas. Given average estimates of nests per female per season and years between nesting years, the total number of adult females in the population is estimated at 7,000–8,000 (TEWG, 2000; NMFS & USFWS, 2007b).

There are no estimates of the number of Kemp's ridleys off the northeastern U.S. (Shoop and Kenney, 1992). Even most adults are too small to be sighted during aerial surveys, so the numbers of sightings are far too few to calculate densities.

Ecology and Life History

Kemp's ridley sea turtles follow the typical sea turtle life history pattern (reviewed in TEWG, 2000). Hatchlings are entrained in oceanic current patterns and passively drift about in

the Gulf of Mexico and North Atlantic until they reach about 20 cm in carapace length (Collard and Ogren, 1990). At that point, which takes 1–4 years, they transition from a pelagic existence to a benthic-feeding juvenile stage and migrate into shallow developmental habitats. They reach sexual maturity at about 60 cm, by which time they have moved into typical adult foraging habitats and migratory patterns (Morreale et al., 2007). The total time from hatching to maturity is 10–17 years. The typical re-migration interval for adult females (i.e., years between nesting years) is 2 years; 60% of females are on 2-year cycles, 20% on annual cycles, 15% on 3-year cycles, and 5% on 4-year cycles.

Pelagic post-hatchlings and small juveniles probably feed on the same types of prey as loggerheads of the same life-stage, but are poorly known (Bjorndal, 1997). Benthic juveniles and adults feed primarily on crabs (Shaver, 1991; Burke et al., 1994; Bjorndal, 1997; Morreale and Standora, 1998).

General distribution

Kemp's ridley sea turtles occur only in the North Atlantic (Ernst et al., 1994) and nearly all nesting is in the western Gulf of Mexico. Sighting and stranding records are concentrated heavily in the Gulf of Mexico and southeastern U.S. Atlantic (TEWG, 2000). Juveniles are dispersed about the Gulf of Mexico and North Atlantic. Brongersma (1972) pointed out the very interesting phenomenon that the smallest known Kemp's ridleys outside of hatchlings leaving the nesting beach were strandings in western Europe, and the second smallest were strandings in New England. Carr (1967) wrote that "The greatest concentration of positively identified Atlantic ridleys that I ever heard of (away from Tamaulipas) occurred in just about the most unlikely place that anybody could imagine. It was Martha's Vineyard, Massachusetts." Until relatively recently, it was often assumed that small ridleys in the temperate North Atlantic represented "waifs" that were lost to the population, however it now appears well established that they are a normal component of the species life history.

Historical occurrence

Babcock (1919) did not include Kemp's ridley sea turtles as occurring in New England,

however Shoop et al. (1981) suggested that Babcock had incorrectly included many Kemp's ridley records as hawksbills. At that time, many did not accept that Kemp's ridleys were a valid species, and instead believed them to be hybrids of other species ("bastard turtles") (Carr, 1967). In addition, juvenile ridleys have a sharp, beak-like mouth similar to a hawksbill's. Babcock quoted several sources who said that small hawksbills were occasionally taken in fish traps in Massachusetts, and wrote that they were "reported to be more common in Buzzard's Bay than loggerheads." However only one or two specimens were ever collected.

Lazell (1980) summarized the substantial numbers of records of Kemp's ridley sea turtles in southern New England that had been collected to that time. He argued that New England waters constituted normal and important habitat for the species, and should be protected by designation as "Critical Habitat" under the ESA.

Recent occurrence

We had only 14 records of Kemp's ridley sea turtles in the Rhode Island study area—12 (85.7%) in summer and 2 (14.3%) in fall (Fig. 71). Four of the summer records came from whale-watching boats. Kemp's ridleys occurred either in the southwestern corner of the study area, or in or near the SAMP area. The sightings were far too few to generate relative abundances. There was one very recent stranding in Rhode Island—a live juvenile that was caught in a fisherman's net in Greenwich Bay in late October of 2004 (Wyman et al., 2004); its photo graced the front page of the Fall/Winter 2004 issue of *Narragansett Bay Journal*. Only one other Kemp's ridley stranding from Rhode Island and Connecticut was recorded in 1987–2001 (Nawojchik and St. Aubin, 2003), although the exact year and location are not known to us.

The occurrence record for Kemp's ridleys in the study area is biased due to two factors. Most are simply too small to be detected from surveys. Morreale et al. (1992) reported that the carapace lengths of cold-stunned Kemp's ridleys in eastern Long Island in 1986–1988 ranged from 22.5 to 37.6 cm ($N = 97$, mean = 29.4). Dodge et al. (2008) reported that the typical cold-stunned Kemp's ridley in Cape Cod Bay was the size of a 2-year-old juvenile, based on sizes of some known-age individuals that had been tagged as hatchlings. The second factor is that the shallow bays and estuaries utilized by ridleys within the study area are usually excluded from survey designs. It is very clear that juvenile Kemp's ridleys are relatively common both around eastern

Long Island and in Cape Cod Bay (Table 6). It is likely that Rhode Island simply does not have equivalent environments that would constitute good habitat for juvenile ridleys or other juvenile sea turtles. Given that they are common both east and west of Rhode Island, it is possible, however, that small ridleys regularly transit the Rhode Island and SAMP study areas.

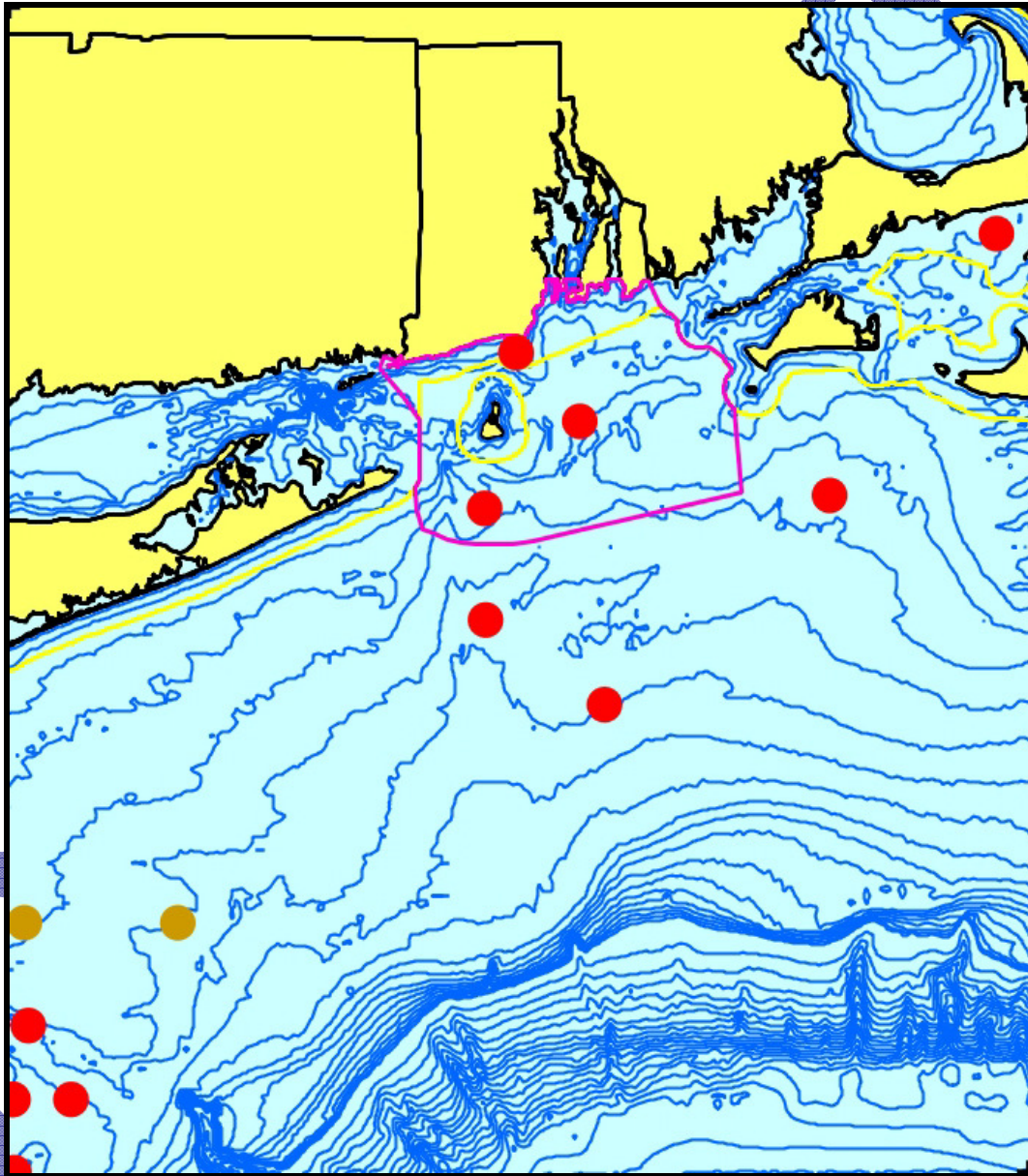


Figure 71. Aggregated sighting, stranding, and bycatch records of Kemp's ridley sea turtles in the Rhode Island study area, 1979–2002 ($n = 14$: winter = 0, spring = 0, summer [red] = 12, fall [brown] = 2).

Conclusions

Kemp's ridley sea turtles have occurred in the SAMP area, but they are much rarer in the study area than leatherbacks or loggerheads. There is some small concern that the small juvenile ridleys that are found around eastern Long Island or Cape Cod might transit through the SAMP area during their migrations. Any mitigation relative to the SAMP or development activities for leatherbacks would also benefit Kemp's ridleys, so it does not seem necessary to consider them separately.

DRAFT

3.2.42. Green sea turtle *Chelonia mydas* (Linnaeus, 1758)

Description

Green turtle adults are usually about the same size as or slightly larger than loggerheads, although the largest adults reach only about 150 cm (Ernst et al., 1994; Wynne and Schwartz, 1999). The shell is not as broad as in the loggerhead, and is more oval and less tapered. The color can be extremely variable, from pale olive to dark brown, with distinctive mottling or radiating patterns on the scutes. The head is much narrower than in loggerheads and lacks the broad crushing plates on the jaws.

Status

At the species level, green sea turtles are classified as Threatened under the U.S. Endangered Species Act, however the Florida nesting population is listed as Endangered. Since the population identity of any individual green turtle encountered off the northeastern U.S. is impossible to determine, the risk-averse strategy would be to consider them as Endangered. Green turtles are not included on the Rhode Island state list, and are classified as Endangered on the IUCN Red List.

There are 46 identified nesting concentrations of green turtles in the world, including 13 in the Atlantic (reviewed in NMFS & USFWS, 2007a): five in the western North Atlantic, four in the eastern Mediterranean, one in Brazil, two in western Africa, and one on Ascension Island. The total number of nesting adult females worldwide is estimated between 110,000 and 150,000. The five western North Atlantic nesting populations include: Florida, with an average of 5,055 nests per year, mostly in Brevard and Palm Beach Counties, and with an increasing trend in 2001–2005; Yucatan, with 1,500 nests in the 2000s and an increasing trend; Costa Rica, with 17,402–37,290 nesting females in a year during 1999–2003, and with an increasing trend; Venezuela, with 335–443 nesting females and no detectable trend; and Suriname, with 1,803 nesting females in 1995 and an increasing trend.

There are no estimates of the number of green sea turtles off the northeastern U.S. (Shoop and Kenney, 1992). The numbers of sightings are far too few to calculate densities, and

many individuals are too small to be sighted during aerial surveys.

Ecology and Life History

Green sea turtles follow the typical sea turtle life history pattern (reviewed in NMFS and USFWS, 2007a). Hatchlings are entrained in oceanic current patterns and passively drift about in association with sargassum patches (Carr, 1987). After 5–6 years of pelagic existence, they reach 20–25 cm in carapace length and move into developmental habitats in shallow coastal waters. They spend about 6 years in these habitats, then move into typical adult foraging habitats. The total time from hatching to maturity may be as long as 40 years (Limpus and Chaloupka, 1997). Adult females exhibit remigration intervals of 2–5 years, on average deposit three nests per breeding year, and have a reproductive lifetime of 17–23 years.

Green sea turtle adults and benthic feeding juveniles are herbivores, feeding on a variety of sea grasses and algae (Bjorndal, 1985, 1995, 1997; Mortimer, 1995). They also consume small amounts of animal material, including jellyfish, salps, and sponges (Bjorndal, 1997). Juvenile green turtles from Long Island were recorded as feeding on eel-grass, three species of green algae, and two species of brown algae (Burke et al., 1991). Pelagic post-hatchlings and small juveniles are not herbivorous, but are probably omnivores feeding more on animal food than on plant material (Bjorndal, 1985, 1997).

General distribution

Green sea turtles are globally distributed in tropical and sub-tropical regions, with some individuals occurring in cooler, temperate regions (Ernst et al., 1994; NMFS & USFWS, 2007a). In the western North Atlantic, they are most common in the Gulf of Mexico and Caribbean. Because of their herbivorous diet, green turtles are most likely to occur in shallow, nearshore habitats with extensive sea grass meadows.

Historical occurrence

Babcock (1919) wrote that green sea turtles were occasionally recorded in southern New

England. He reported that one was captured in New Bedford harbor in September 1878, but that there were no other records from Buzzards Bay, where he expected they should occur. He also said that there were numerous records from Long Island Sound back to 1840, including two captured in the Housatonic River in Connecticut. Lazell (1980) recounted anecdotal evidence for a resident population of juvenile green turtles in Nantucket Sound, where they were regularly caught in pound nets and were often sold as exhibit specimens to commercial aquaria.

Recent occurrence

There has been only one confirmed green turtle sighting in the Rhode Island study area—on 25 March 2005 south of Long Island between the 40- and 50-m isobaths (Fig. 72). The sighting was made during an aerial survey for right whale monitoring, and was assigned an identification reliability of “probable.” Nawojchik and St. Aubin (2003) reported only two strandings in Connecticut and Rhode Island during 1987–2001, but the dates and locations are not known to us. However, like Kemp’s ridleys, juvenile green turtles are known to be present in shallow waters around eastern Long Island and Cape Cod (Table 6). Those data suggest that green turtles are relatively more common around Long Island than they are in Massachusetts.

Conclusions

Green sea turtles have never been recorded in the SAMP area, and they are much rarer in the study area than leatherbacks or loggerheads. There is some small concern that the juvenile green turtles that are found around eastern Long Island or Cape Cod might transit through the SAMP area during their migrations. Any mitigation relative to the SAMP or development activities for leatherbacks would also benefit green turtles, so it does not seem necessary to consider them separately.

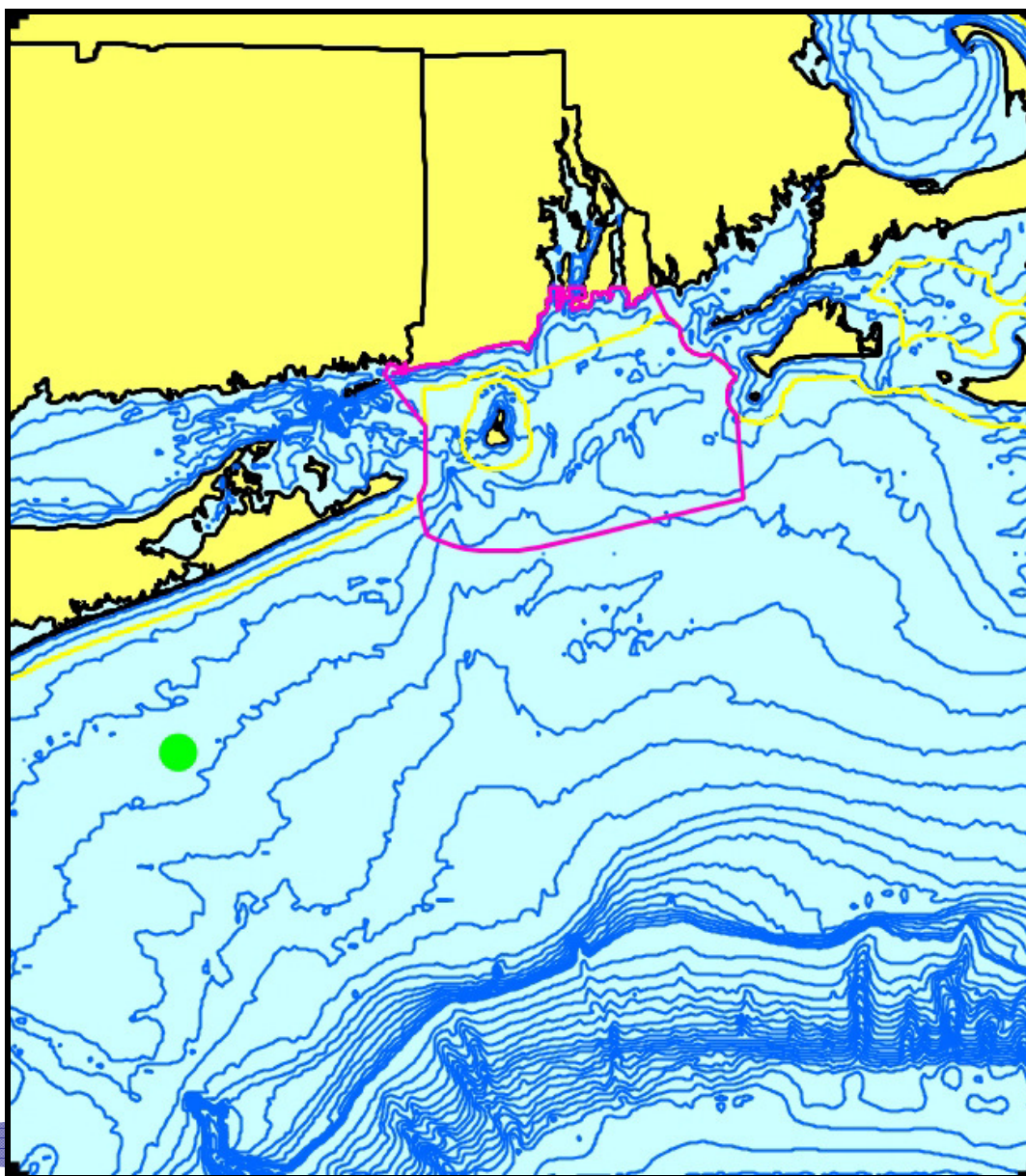


Figure 72. Aggregated sighting, stranding, and bycatch records of green sea turtles in the Rhode Island study area, 2005 (n = 1: winter = 0, spring [green] = 1, summer = 0, fall = 0).

4. CONCLUSIONS AND RECOMMENDATIONS

Forty species of marine mammals and sea turtles are known to occur in the waters of the Rhode Island study area—encompassing Narragansett Bay, Block Island Sound, Rhode Island Sound, and nearby coastal and continental shelf areas. Sixteen are categorized as common to abundant, six as regular, and eighteen as rare to accidental. Eleven of those species—six whales, the manatee, and four sea turtles—are listed as Endangered or Threatened under the U.S. Endangered Species Act. One other species was present historically but is now extinct in the North Atlantic. Eight other species, including one Endangered sea turtle, are considered to be hypothetical in the study area—with one or more records nearby.

Every species does not pose the same level of conservation concern relative to the Rhode Island Ocean SAMP or to the development of alternative energy projects or other industrial/commercial projects in our coastal waters. Factors that influence the level of concern include overall abundance of the population, abundance in the study area, seasonal distribution patterns and likelihood of occurrence in or near any area of development, ESA-listing status, sensitivity to specific anthropogenic activities, and existence of other known threats to the population.

The 40 marine mammal and sea turtle species known to occur in the study area have been ranked into five levels of conservation priority relative to the SAMP (Table 7). The ranking has been done using the factors outlined above. Below we have summarized the general characteristics of each priority level, as well as the species included.

Priority 1

The highest priority level includes species that are common in the Rhode Island study area, that are known to occur in the SAMP area at least seasonally, and that are listed as Endangered under the U.S. Endangered Species Act. The North Atlantic right whale almost deserves to be in a category by itself. The species is one of the rarest mammals in the world, there is serious concern about long-term population viability, and there is known anthropogenic mortality from ship collisions, as well as from entanglement in commercial fishing gear. The populations of humpback whales and fin whales are at least an order of magnitude larger than the right whale's, but they can be abundant in or near the SAMP area, and they are also subject to

Table 7. Prioritized conservation rankings of 49 species of marine mammals and sea turtles relative to the Rhode Island Ocean Special Area Management Plan. Species listed as Endangered or Threatened under the U.S. Endangered Species Act are identified by *E* or *T*, respectively.*

Rank	Species included
1a	North Atlantic right whale (<i>E</i>)
1b	humpback whale (<i>E</i>), fin whale (<i>E</i>), leatherback sea turtle (<i>E</i>)
2	sperm whale (<i>E</i>), harbor porpoise, white-sided dolphin, short-beaked common dolphin, harbor seal, loggerhead sea turtle (<i>T</i>)
3	sei whale (<i>E</i>), common minke whale, long-finned pilot whale, Risso's dolphin, bottlenose dolphin
4	blue whale (<i>E</i>), pygmy sperm whale, dwarf sperm whale, Cuvier's beaked whale, Blainville's beaked whale, Gervais' beaked whale, Sowerby's beaked whale, True's beaked whale, striped dolphin, gray seal, harp seal, hooded seal, Kemp's ridley sea turtle (<i>E</i>), green sea turtle (<i>T & E</i>)
5	Bryde's whale, northern bottlenose whale, beluga, short-finned pilot whale, killer whale, false killer whale, white-beaked dolphin, Atlantic spotted dolphin, pantropical spotted dolphin, ringed seal, West Indian manatee (<i>E</i>)
na	Atlantic gray whale, pygmy killer whale, melon-headed whale, rough-toothed dolphin, spinner dolphin, Clymene dolphin, bearded seal, walrus, hawksbill sea turtle (<i>E</i>) [1 extirpated and 8 hypothetical species]

* the order of species within any ranking category does not imply any priority within that category; it is simply the order in which the species appear in this report.

human-caused mortality. Leatherback sea turtles are a global conservation priority, and are the most likely sea turtle species to be encountered in the SAMP area.

Priority 2

The second level of priority includes species in two different classes. The first would include sperm whales and loggerhead sea turtles. Both are ESA-listed, sperm whales as Endangered and loggerheads as Threatened, and both are common in the Rhode Island study area. However, neither species is likely to occur more than rarely to occasionally in the SAMP area. The other group includes four very abundant marine mammals. Harbor porpoises, Atlantic white-sided dolphins, and short-beaked common dolphin are probably the most abundant marine mammals in the study area, and all are likely to occur in significant numbers within the SAMP area at least seasonally. Harbor seals are the most common seal species in the study area, and are the only marine mammal that can be considered as resident in Rhode Island. They are known to occupy haul-out sites on the periphery of Block Island, where they could be subject to disturbance from development activities.

Priority 3

The third level of priority includes five marine cetacean species. Sei whales are considered regular in the Rhode Island study area and are ESA-listed as Endangered, however they are not likely to occur in the SAMP area, typically occur beyond the study area, and are only likely to visit the study area irregularly in a small number of years. Minke whales, long-finned pilot whales, Risso's dolphins, and bottlenose dolphins are all common species in the study area, but likely to occur in the SAMP area only infrequently.

There could be an argument made to shift loggerheads to this class from #2. On the other hand, one could also argue that Kemp's ridleys and/or green turtles should be moved here from #4 because of concerns about undetected little turtles that could be transiting through the SAMP area and study area. The second argument is probably the stronger of the two.

Priority 4

The fourth level of priority includes mostly rare species with known centers of occurrence outside of the SAMP area, or even beyond the Rhode Island study area. Blue whales

are endangered, but occur accidentally in southern New England. Kemp's ridley and green sea turtles are ESA-listed, but have mainly tropical and sub-tropical distributions and are not known to occur in the SAMP area. However juveniles of both species are known to utilize shallow developmental habitats around eastern Long Island and Cape Cod. Pygmy sperm whales, dwarf sperm whales, Cuvier's beaked whales, Blainville's beaked whales, Gervais' beaked whales, Sowerby's beaked whales, True's beaked whales, and striped dolphins are all regular or rare species in the study area with distributions that are primarily offshore—at the shelf edge and beyond. Gray seals, harp seals, and hooded seals are all common species in the study area and very frequently stranded. However, for all three species the majority of individuals in the study area appear to be dispersing juveniles, the main centers of the adult populations are elsewhere in the western North Atlantic, and the strandings appear to be simply a component of natural juvenile mortality.

Should blue whales be moved to Priority 5? They are accidental in the area. One could argue that only the one whale seen three times in 1990 is a reliable record for the area.

Priority 5

The lowest level of priority includes all rare species, with many occurring only accidentally in the study area—Bryde's whale, northern bottlenose whale, beluga whale, short-finned pilot whale, killer whale, false killer whale, white-beaked dolphin, Atlantic spotted dolphin, pantropical spotted dolphin, ringed seal, and West Indian manatee.

Recommendations

Again—should mitigation be addressed here, or is that beyond the scope of a data review?

In the event that a full EIS is required at some point in the future for a wind farm, other alternative energy project, or other commercial/industrial development, there will be a requirement to estimate the levels of “take” of protected species. Calculating take estimates will require estimating the densities (animals per km²) of each species present. The relative

abundances generated in this technical report will not be sufficient. There are old seasonal, stratified density estimates from CETAP based on aerial line-transect surveys in 1979–1981 (CETAP, 1982; Kenney et al., 1985a). The sampling design was year-round, and there were or two survey strata that could approximate the SAMP area. However, the sampling coverage was low and was reduced each year of the project. Later NMFS aerial and shipboard surveys were designed to estimate stock abundances for the annual stock assessment reports (e.g., Waring et al., 2008) that were required under the 1994 amendments to the MMPA. However, the surveys were almost all in summer, the coverage was low with surveys spaced several years apart, many of the surveys concentrated on offshore waters coverage, the objective was to generate one abundance estimate and not densities by local sampling areas, and the actual densities are not published in the stock assessments. The most recent attempt to generate density estimates was for the Navy (DoN, 2007), done by . That project tried to break down the estimates into smaller areas, however they were based on only a subset of the NMFS aerial surveys, and again were mainly from summer (exclusively so in the vicinity of the SAMP study area.) It is possible that those estimates have never been externally reviewed outside of the agencies involved in generating them.

New surveys might be necessary to generate regionally specific seasonal density estimates of protected species. They could be either aerial or shipboard surveys, however a combination of both would be ideal to capture species that are best sampled by only one or the other. An ideal survey program would need to run year-round to adequately capture seasonal variability, for at least 2–3 years to capture interannual variability, and at sufficient intensity and sample sizes to generate reliable estimates with reasonably low CVs. It would not be a simple or inexpensive undertaking.

Tagging studies might provide additional data on habitat use and movement patterns of animals of concern for specific projects, for example small sea turtles or resident seals.

Relatively low-cost, long-term monitoring during the summer could be accomplished by supporting student interns who would ride on the whale-watching boat from Galilee and collect sighting records in standardized form. Such an intern could also assist the company in computerizing their previous logbooks for additional trend analyses.

5. ACKNOWLEDGEMENTS

TO COME LAST

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