CHAPTER 11

SEA TURTLES OF THE GULF OF MEXICO

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11.1 INTRODUCTION

Five species of sea turtles are found in the Gulf of Mexico: the Kemp's ridley (*Lepidochelys kempii*), loggerhead (*Caretta caretta*), green (*Chelonia mydas*), leatherback (*Dermochelys coriacea*), and hawksbill (*Eretmochelys imbricata*). While individuals of some species of sea turtles may nest on beaches and spend nearly their entire lives in the Gulf of Mexico, such as the Kemp's ridley, others may only use the Gulf to nest, as a foraging area, or as part of their migration routes. The Gulf of Mexico provides important sea turtle nesting habitat, and many Gulf of Mexico beaches where sea turtles nest have been protected as refuges and parks. For example, protected beaches in Rancho Nuevo, Mexico and at Padre Island National Seashore (PAIS), Texas, are the major nesting beaches for the Kemp's ridley, and loggerheads and green sea turtles nest on beaches at Dry Tortugas and Everglades National Parks in Florida. Sea turtles often spend their post-hatchling and early juvenile years in the pelagic Gulf of Mexico (Witherington et al. 2012), and the nearshore waters of the Gulf provide critical foraging habitat for juvenile and adult sea turtles, as well as important mating and internesting habitat (Musick and Limpus 1997; Bolten et al. 1998; Hopkins-Murphy et al. 2003). The northern Gulf of Mexico can be divided oceanographically into the eastern half and the western half. The eastern half is influenced strongly by Caribbean inflow and has relatively clear water, while the western half is influenced by the turbid Mississippi River, incurs significant shrimp trawling pressure, and has thousands of oil and gas production structures (Hopkins-Murphy et al. 2003).

This chapter describes Gulf of Mexico sea turtle populations prior to the Deepwater Horizon incident that occurred on April 20, 2010, especially with regard to abundance and distribution. The types of information summarized for sea turtles that use the Gulf of Mexico for at least some portion of their life cycle includes life history, distribution, and abundance; location of nesting beaches and nesting numbers; and habitat use and foraging area locations for the various life stages. The natural and anthropogenic threats that affect sea turtles in the Gulf of Mexico are also discussed, including sea turtle stranding, fisheries bycatch, and other types of less common but important impacts, such as boat strikes.

Sea turtles that occur in the Gulf of Mexico are not randomly or evenly distributed spatially or temporally. They are difficult to study and monitor since they are broadly distributed, with nesting aggregations in various locations of the Gulf, have wide-ranging migrations, have long generation times and long life spans, and spend the majority of their lives at sea (Holder and Holder 2007; Witherington et al. 2009; NRC 2010). In addition, the oceanic habitat of juveniles is a major obstacle to studying immature stages of sea turtles (NRC 2010). For these reasons, the most common sea turtle population assessments have been made at nesting beaches (Schroeder and Murphy 1999). Counts of sea turtle nests provide an index of annual population productivity and an approximate index of abundance for adult females (Karnauskas

C.H. Ward (ed.), *Habitats and Biota of the Gulf of Mexico: Before the Deepwater Horizon Oil Spill*,
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et al. 2013). However, this segment of the population represents a very small portion of the sea turtle population, and determining population sizes of sea turtles, as well as quantifying impacts on juvenile and adult males and females, is extremely challenging (NRC 2010). This type of population assessment is similar to estimating human population trends by counting women in maternity wards: while useful information is obtained, if the children were decimated from an impact, the mortality would not be detected in the adult population for decades (Bjorndal et al. 2011). The need for assessing populations of both juvenile and adult sea turtles in the water to complement assessments of nesting beaches has been widely recognized (Magnuson et al. 1990; TEWG 1998, 2000). Due to advances in genetic analyses, satellite telemetry technology and the development of new methodologies and technologies over the past 20 years, as well as the growing long-term monitoring and tagging sea turtle datasets, much has been learned regarding where sea turtles go and what they do when they are not on the nesting beach. Nevertheless, significant data gaps remain.

11.1.1 Generalized Life History of Gulf of Mexico Sea Turtles

Three basic ecosystem zones characterize the life history patterns of the five species of sea turtles that occur in the Gulf of Mexico (Figure 11.1) (NMFS et al. 2011).

1. Terrestrial Zone: the nesting beach where females lay eggs and where embryos develop.
2. Neritic Zone: the inshore marine environment from the surface to the seafloor, including bays, sounds, and estuaries, as well as the continental shelf, where water depths do not exceed 200 meters (m) (656.2 feet [ft]).
3. Oceanic Zone: the open ocean environment from the surface to the seafloor where water depths are greater than 200 m (656.2 ft).

On the nesting beach sea turtle eggs require a high-humidity environment, an incubation temperature between 25 and 35 degrees Celsius (°C), and adequate conditions for gas exchange for proper development (Ackerman 1997). The length of the incubation period varies and is inversely related to nest temperature; the warmer the sand surrounding the egg chamber, the

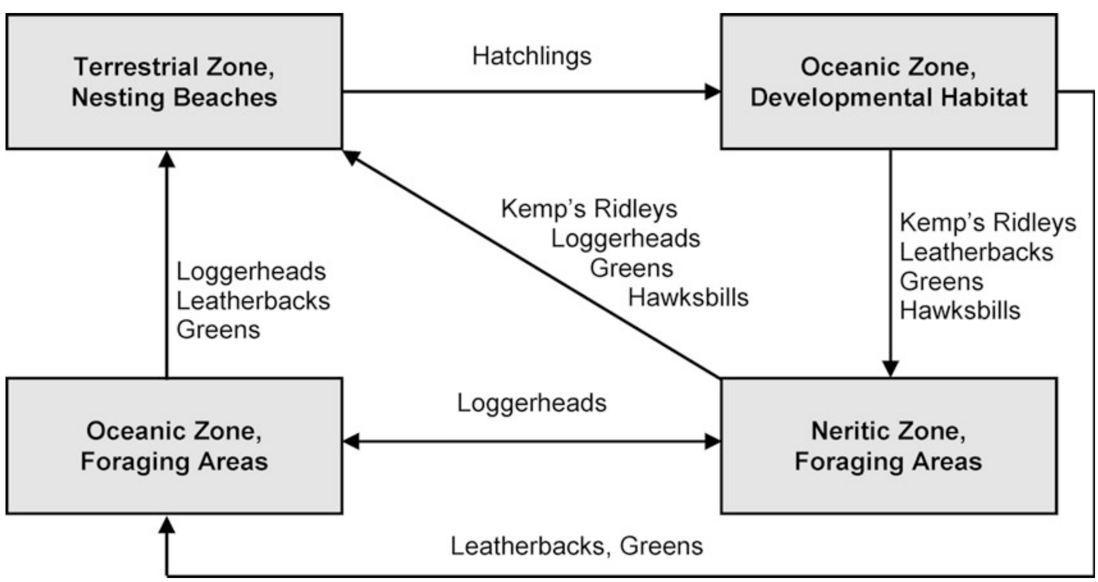

Figure 11.1. Generalized life cycle of sea turtle species that occur in the Gulf of Mexico.

faster the embryos develop (Mrosovsky and Yntema 1980; Ackerman 1997). Sex is determined by incubation temperatures prevailing during the middle third of the incubation period for all sea turtle species (Mrosovsky and Yntema 1980; Wibbels 2007). Each species has a pivotal temperature, which is the temperature at which the sex ratio is one to one. Nest temperatures higher than the pivotal temperature produce mostly females, and lower temperatures produce mostly males (Witherington et al. 2006a).

Immediately after emerging from the nest, sea turtle hatchlings begin a period of frenzied activity. During this period, they move from their nest to the surf, swim, and are swept through the surf zone (Witherington 1995; Conant et al. 2009). The hatchlings use a progression of orientation cues as they crawl to the water, swim through the surf, and migrate offshore (Lohmann and Lohmann 2003; Lohmann et al. 2012). Once they reach the oceanic zone, hatchlings spend a number of years growing and developing.

Some sea turtle species return to the neritic zone as juveniles (e.g., Kemp's ridleys, loggerheads, greens, hawksbills), while others stay in the oceanic zone (e.g., leatherbacks) (Figure 11.1). Adults of some species remain in the neritic zone their entire lives (e.g., Kemp's ridleys), and some move back and forth between the neritic and oceanic zones (e.g., loggerheads). Some sea turtles spend their entire lives in the oceanic zone (e.g., leatherbacks), with the exception of females nesting in the terrestrial zone.

While differences exist between and within species, adult females typically return to nest in the general vicinity of the beach where they hatched from eggs many years earlier and often nest at the same beach throughout their reproductive years. For example, while most loggerheads return to nest at the same beach from which they were hatched, individual loggerhead sea turtles have been known to nest on both the Atlantic and Gulf coasts of Florida (LeBuff 1974). Green turtles typically nest on the same beach where they hatched (Bowen et al. 1989; Allard et al. 1994).

11.1.2 Historical Abundance of Gulf of Mexico Sea Turtles

Sea turtles were once highly abundant throughout the Gulf of Mexico and the Caribbean. By some estimates, they may have numbered in the millions (Jackson 1997). However, since the discovery of the New World, their rookeries in the region have decreased significantly, mainly due to overexploitation of these reptiles for their meat, shell, and eggs. Because of the high quality of its meat, the impact of the direct take of juvenile and adult green turtles has historically been more pronounced than for any other sea turtle species. It has been suggested that Caribbean green turtle populations have declined as much as 99 percent (%) since the arrival of Christopher Columbus (Bowen and Avise 1995; Jackson 1997). There are numerous accounts of consumption of this species throughout the region. In fact, the name of the turtle does not derive from the outer coloration of this animal but from the color of the green fat found under the shell, for which this turtle was considered a delicacy by British royalty in the eighteenth and nineteenth centuries (Witzell 1994). Consumption of this species fueled early exploitation by local artisanal and industrial fisheries, leading to the exploitation of other sea turtle species. To appreciate the magnitude of the impacts, it is important to examine the history of green turtle exploitation in the greater Caribbean.

In his book *The Green Turtle and Man*, James Parsons (1962) documents the use of green turtles and their eggs in the region by Europeans and New World settlers. Green turtles were sought after for their calipee and calipash, the cartilage associated with the plastron and the carapace, respectively. These products were used by the English aristocracy to make turtle soup, which became a staple after the discovery of the New World. Besides supplying the English kitchens with gourmet soup, green turtle oil was also used as a substitute for butter, lamp fuel, and as a lubricant. The green turtle trade between London and the West Indies began in the mid-eighteenth century. The calipee was more abundant, with a large turtle producing between 1.1 and 1.6 kilograms (kg) (2.5 and 3.5 pounds [lb]). The English used green turtle soup as a cure for scurvy in the long transatlantic voyages and as a substitute for, or in addition to, salted beef, since turtles could be kept alive below deck on their backs for weeks. Green sea turtles were abundant and supported the exploration and settlement of the greater Caribbean, providing sailors and pioneer settlers with fresh meat. By 1878, it was estimated that some 15,000 Cayman Islands green turtles had been landed in London, ranging from 11 to 136 kg (25 to 300 lb). In 1880, imports of preserved turtles (sun-dried meat and calipee) amounted to 4,899 kg (10,800 lb). That year, a Key West factory had an estimated production of 200,000 cases of calipee. Although London was the main market, New York also constituted an important market for these products. In 1883, the largest factory of green turtle soup was Moore & Company Soups, Inc. of Newark, New Jersey. By that year, imports of live green turtles into the United States amounted to 468,646 kg (1,033,187 lb), mainly from Mexican and Nicaraguan waters. At an estimated 73 kg (160 lb) per turtle, this would be equivalent to 6,457 turtles, which did not include turtles caught in Florida and those sacrificed for calipee.

The high demand for green turtle meat and soup contributed significantly not only to the demise of many Caribbean populations, but also to those in the Gulf of Mexico. In the western Gulf, green turtles were once abundant enough to support meat and soup canneries in Texas (Groombridge and Luxmoore 1989). At least five green turtle canneries existed in the late 1800s along the coast of the northwestern Gulf of Mexico. Green and loggerhead turtle fisheries supplied these canneries (Figure 11.2). To give an idea of the magnitude of this fishery, it was reported that in 1890, a total of 265,000 kg (584,225 lb) of green turtles were caught in Texas (Hildebrand 1982). The canneries were located in Fulton, Rockport, Indianola, Point Isabel, and

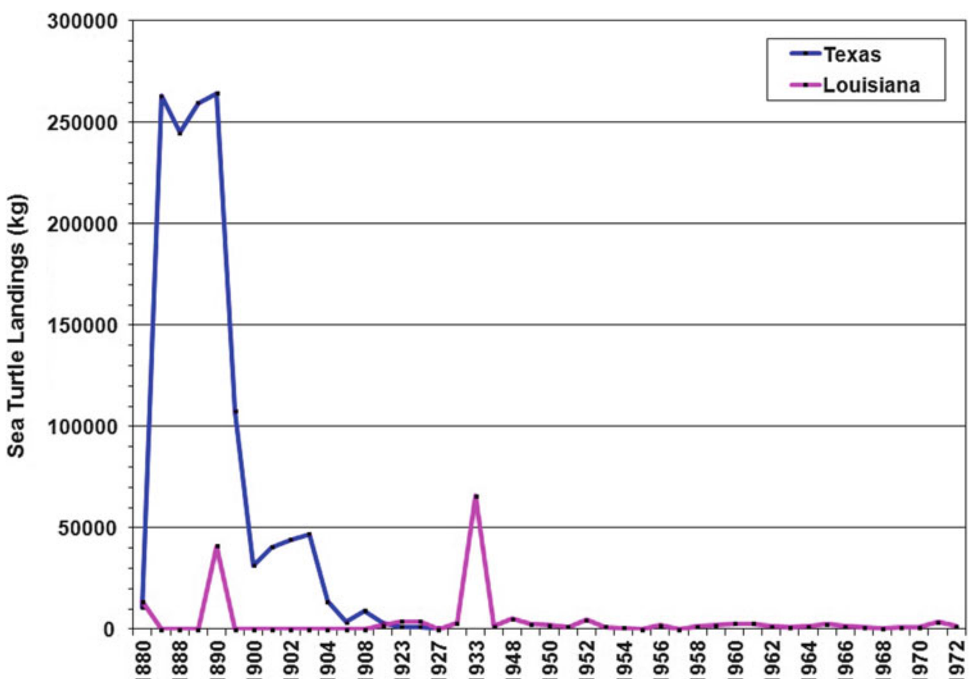

Figure 11.2. Sea turtle landings in Texas and Louisiana for available years from 1880 through 1972. The records include mainly green and loggerhead sea turtles (Rebel 1974; Cato et al. 1978; Doughty 1984).

Corpus Christi, Texas (True 1887; Doughty 1984). The Fulton cannery was the largest and operated between 1881 and 1896; in 1890, this cannery alone processed 900 green turtles (41.7 % of the turtle catch in the state) totaling 110,223 kg (243,000 lb) and produced about 40,000 0.9-kg (2-lb) cans (Doughty 1984). The green turtle cannery located in Rockport was founded around 1886. During its first 6 months of operation, the cannery processed about 3,856 kg (8,500 lb) of green turtle meat (True 1887). Green turtles caught and processed in Texas were presumed to feed in the seagrass beds located between Matagorda Bay and Laguna Madre (Hildebrand 1982). Rapid depletion of green turtle populations in Texas and elsewhere occurred because the fishery targeted juvenile turtles, a highly vulnerable stage in the life cycle of this slow-growing, late-maturing species (Crouse et al. 1987; Witzell 1994). While the green turtle fishery in Texas developed very quickly, it declined abruptly after 1892 and ended shortly thereafter, presumably due to the scarcity of turtles and to a deep freeze along the Texas coast (Hildebrand 1982). A small turtle fishery in Louisiana remained open through the early 1970s (Figure 11.2).

Prior to 1860, green turtles caught on the Florida east coast, particularly in the Indian River Lagoon, were exchanged for goods with various merchant vessels; in later years, agents purchased the catch and then shipped it mainly to New York (True 1887). By 1887, green, loggerhead, and hawksbill turtles were hunted as far north as Beaufort and Morehead City, North Carolina, where green turtles in particular were a delicacy and consumed locally (True 1887). There was no mention of Kemp's ridley sea turtles being harvested in the early records, even though this species surely was present in Florida at the time (Carr 1957). The omission appears to have occurred because Kemp's ridleys were sold as loggerheads for years (Rebel 1974; Cato et al. 1978).

Two locations on Florida's west coast—Key West and Homosassa—appear to have supported the most abundant in-water populations of green turtles in the entire Gulf of Mexico in the late 1800s. Captured turtles weighing between 18 and 45 kg (40 and 100 lb) were kept alive in small *Kraals* (Dutch for corral) or seawater-filled holding pens until ready to be shipped to New York (True 1887). Estimates indicate that about 50 18-kg (40-pound) turtles per week were brought to Key West throughout the year (True 1887). The green turtle fishery in the Cedar Keys area presumably arose around 1878. Fishing was concentrated in an area 32–48 kilometers (km) (20–30 miles [mi]) north and south from the main Cedar Keys port, with the shallow foraging grounds being the most productive (True 1887). Large boats brought between 1,361 and 2,268 kg (3,000 and 5,000 lb) of green turtles to port; whereas, small boats brought in only 23–363 kg (50–800 lb). Interestingly, the largest green turtle recorded at the time weighed an impressive 544 kg (1,200 lb) (True 1887). The reported weight of this turtle was questioned by Carr and Caldwell (1956), who indicated that green turtles landed in the Cedar Keys fisheries in the 1950s were no larger than 52 kg (115 lb). Alternatively, it is possible that the discrepancy reflects the impact of the decades-long turtle fishery in the region, leaving no large adult turtles in the population.

Overall, statistics show that the Florida west coast produced about 81,647 kg (180,000 lb), Louisiana about 13,608 kg (30,000 lb), and Texas approximately 24,494 kg (54,000 lb) of green turtle meat in 1880, though apparently an unspecified amount of freshwater turtle meat was also included in these records (True 1887; Rebel 1974). No mention is made of significant sea turtle fisheries for any other Gulf coast state prior to 1880 besides Florida, Louisiana, and Texas. By 1887, the most important sea turtle fisheries in the Gulf of Mexico were those of the Cedar Keys area and around Key West (True 1887; Townsend 1899). Figure 11.3 shows statistics of turtle landings on the Florida Gulf coast between 1880 and 1897 for the years for which data are available (Townsend 1899), and provides a perspective of turtle demand in the northeastern Gulf of Mexico.

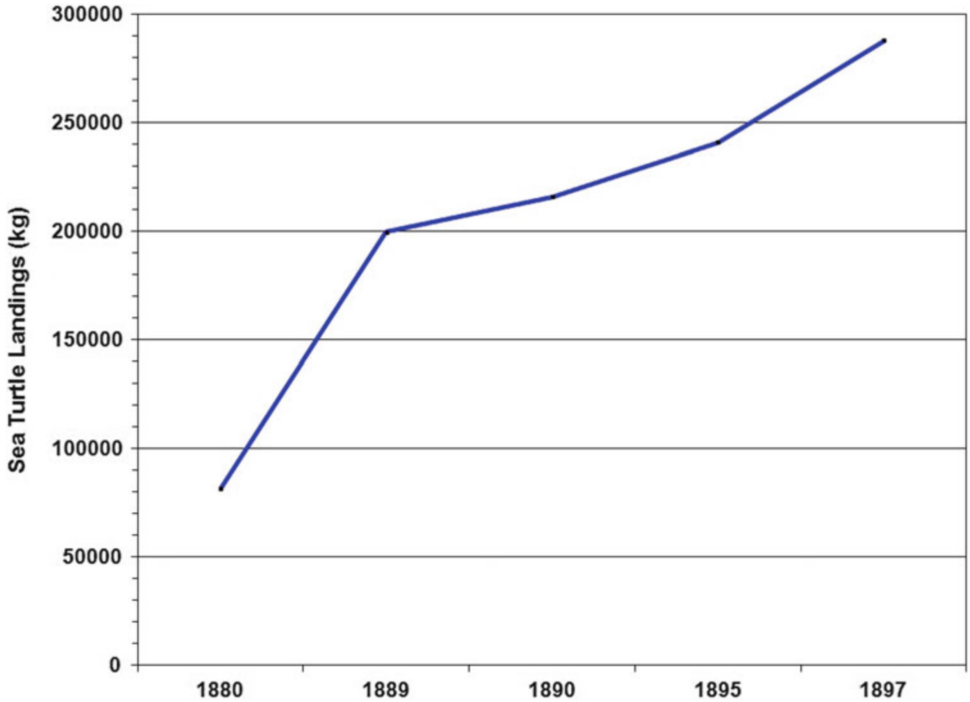

Figure 11.3. Turtle landings on the Florida Gulf coast for available years from 1880 through 1897. The data include mostly green sea turtles, along with loggerheads, hawksbills, and most likely Kemp's ridleys (from Townsend 1899).

Until 1890, the sea turtle fisheries in Florida occurred in eight counties, but by 1897 they were concentrated in four: Monroe, Levy, Franklin, and Escambia counties on the Gulf coast. The total production in 1897 alone was 287,857 kg (634,616 lb), with 86 % coming from Monroe County at the southern tip of Florida (Townsend 1899). However, an unspecified portion of the total production came from the Yucatán coast because turtles were already becoming scarce in Florida; by 1897, most of the turtles came from the Yucatán coast. Apparently, the decrease in the sea turtle populations along the Florida Gulf coast was due not only to the harvest of juveniles but also of eggs, as these were sought after eagerly by local people (Townsend 1899). Indeed, although few records of egg exploitation exist, it is believed that a large number of eggs were collected throughout the entire rim of the Gulf of Mexico (Hildebrand 1963; Witzell 1994). Turtles were so scarce by the late 1800s that Townsend (1899) called for the protection of the turtles and their eggs during their breeding season. The sea turtle fisheries continued through the early and mid-1900s on the Florida Gulf coast, though at a much lower rate (Figure 11.4), presumably due to decreases in the turtle populations (Rebel 1974). The landings records for Florida indicate a preference for green turtle meat (Figure 11.4).

In the 1900s, imports of live turtles into the United States were significant prior to the 1978 listing of sea turtles under the 1973 U.S. Endangered Species Act (ESA). The total live sea turtle imports into the United States from 1948 through 1976 amounted to 8,099,950 kg (17,857,334 lb), with a peak in 1951 and a decreasing trend to a minimum of 1,814 kg (4,000 lb) in 1975 (Figure 11.5) (Cato et al. 1978). The imported species of sea turtles included the green, olive ridley (*Lepidochelys olivacea*), loggerhead, and hawksbill, with the former two species being the most imported (Cato et al. 1978). These imports came from over 40 countries and demonstrate the demand for live sea turtles that existed in the United States until the

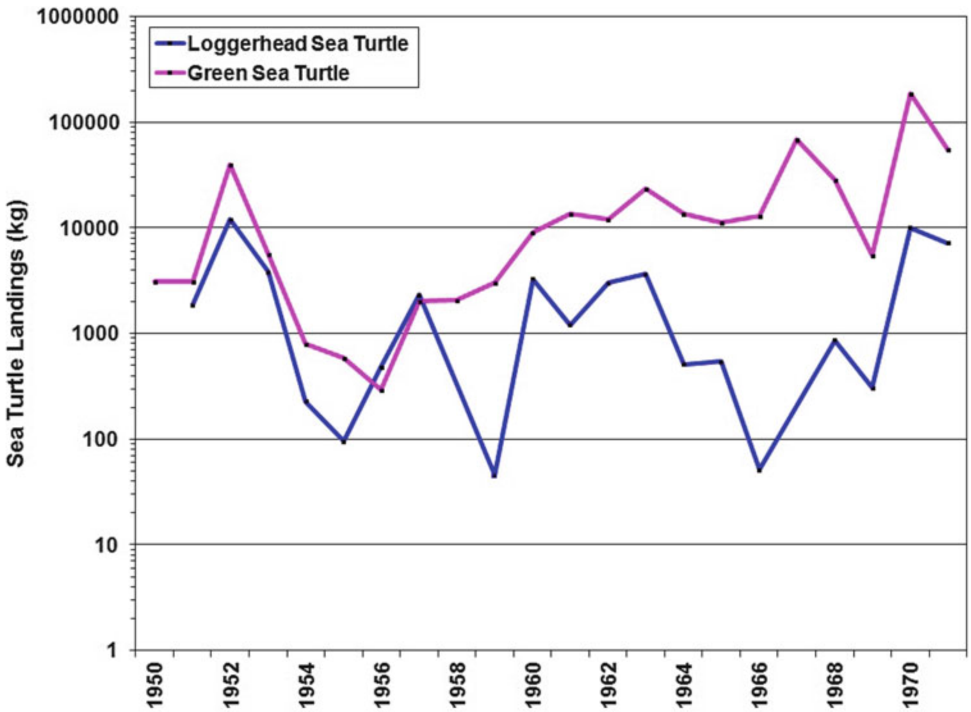

Figure 11.4. Turtle landings on the Florida Gulf coast for available years from 1950 through 1971. The data include green and loggerhead sea turtles, as well as Kemp's ridleys sold as loggerheads, and are plotted on a logarithmic scale to enhance contrast (from Rebel 1974).

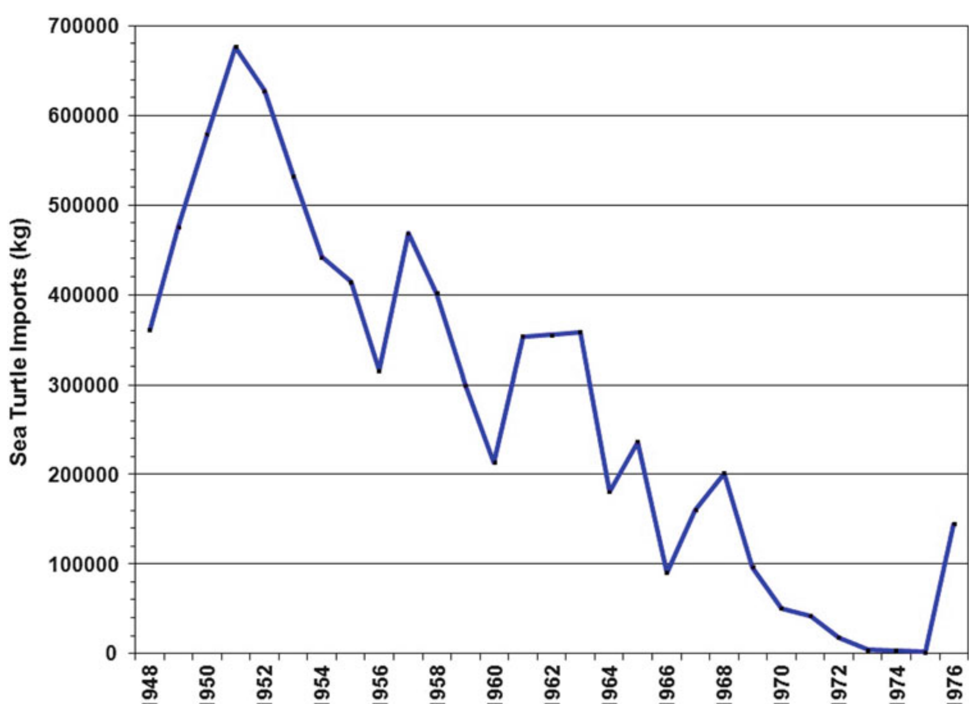

Figure 11.5. Live sea turtle imports into the United States from 1948 through 1976 (redrawn from Cato et al. 1978).

middle of the last century. The data show that demand was significant in the 1950s, but decreased steadily in the 1960s and 1970s, reaching its lowest volume from 1973 to 1975 (Figure 11.5). The spike observed in 1976 is believed to be flawed, since it does not match the overall trend up to that year (Figure 11.5) (Cato et al. 1978). Alternatively, this spike may reflect a last effort by the industry to import live sea turtles before the ban on these imports was fully implemented (Cato et al. 1978).

The green sea turtle fishery in Mexican waters of the Gulf of Mexico continued until 1990, when a total ban was imposed (DOF 1990). Some statistics are available quantifying the magnitude of this catch, which was mainly off the coasts of Quintana Roo and Campeche; for example, from 1964 through 1981, green turtles captured ranged from 14 to 74.7 % of the annual total of sea turtles captured (Márquez-M 2004). Although the commercial fishery also captured loggerheads and hawksbills, most of the take was green turtles, with an estimated average of 67.9 % from 1964 through 1981 (Márquez-M 2004).

In essence, the data provided above indicate that the sea turtle fishery that developed from the discovery of the New World through the mid-1900s was largely responsible for the decline of sea turtle populations in the Gulf of Mexico and the Caribbean. It is important to mention that the data do not include turtles and eggs consumed locally; therefore, the actual anthropogenic impact was likely of much greater magnitude on the populations of sea turtles in the Gulf. Sea turtle populations in the Gulf of Mexico have remained low to the present, mostly as a result of the impact of bycatch in various fisheries, mainly that of shrimp (McDaniel et al. 2000; Crowder and Heppell 2011; Finkbeiner et al. 2011).

11.1.3 General Nesting Abundance of Gulf of Mexico Sea Turtles

The beaches of east Texas, Louisiana, Mississippi, and Alabama in the north-central Gulf of Mexico are essentially devoid of any significant sea turtle nesting. Sea turtle nesting, in general, increases east, west, and southwest from this north-central location and reaches its zenith around the Florida and Yucatán peninsulas (Renaud 2001). Data available on nesting females show that Gulf of Mexico sea turtle populations generally exhibit a very low abundance relative to Atlantic regions outside the Gulf. This is particularly true for loggerhead sea turtles, whose nesting on Florida Gulf of Mexico beaches amounted to only 8.6 % of statewide nesting from 2001 through 2006 (Witherington et al. 2009). The only obvious exception to this general rule is the Kemp's ridley, whose main rookery is located along the beaches of Tamaulipas, Mexico, on the Mexican Gulf coast (NMFS et al. 2011). The very low nesting numbers indicate that all sea turtle populations in the Gulf of Mexico are particularly vulnerable to environmental and anthropogenic impacts, perhaps more so than populations outside the Gulf of Mexico.

An early assessment of the status of sea turtle populations in the western Gulf of Mexico was based largely on the presence/absence of turtles from Louisiana throughout the western rim of the Gulf and south to the state of Yucatán, Mexico (Hildebrand 1982). Unfortunately, no data on abundance were provided for the five species that nest in this large geographic area, which precludes establishing some sort of baseline to which current numbers can be compared. However, the review indicated that all sea turtle populations had undergone a significant decline by 1979 due to the exploitation of eggs, juveniles, and adult turtles (Hildebrand 1982).

A few years later, another attempt to assess the status of Gulf of Mexico sea turtle populations was published, which combined various sources of data, including nesting, in-water captures, aerial surveys, stranding, mortality, and bycatch data, among other datasets, providing hard numbers for the various U.S. stocks (Thompson 1988). Although the publication elicited some controversy (Dodd and Byles 1991; Thompson 1991), its message, that U.S. sea turtle stock assessments must be conducted regularly and frequently, was well taken.

In the last 30 years, many substantial efforts have been made to generate assessments of sea turtle populations on a regular basis (e.g., NMFS and USFWS 2007a, b, c, d, e, 2008; Conant et al. 2009; NMFS et al. 2011). However, most of these assessments focused on beach counts of nests and did not include in-water population assessments. This is not surprising, given the costs associated with studying highly migratory species with complex life cycles. To fully understand the health of sea turtle populations, it is imperative to generate reliable datasets of in-water turtle populations that include various demographic parameters suitable for analysis, such as age of hatchlings and juveniles and survival rates (Heppell et al. 2005), among many other parameters.

With regard to the southwestern Gulf of Mexico, all five species are present and nest on many beaches in the region (Hildebrand 1963; Sánchez-Pérez et al. 1989). On Mexican beaches of the Gulf, the Kemp's ridley and the green sea turtle are estimated to exhibit similar nesting abundances; whereas, the hawksbill and leatherback are less abundant (Márquez-M 2004). Most loggerheads nest on the Caribbean side of the Yucatán Peninsula, with low numbers of nesting occurring along the Gulf coast (Márquez-M et al. 2004). Only about ten leatherback nests were recorded on Mexican beaches of the Gulf in 2000 (Márquez-M 2004). At the level of the Gulf basin, these numbers, along with Florida numbers, confirm that the largest numbers of nesting sea turtles are located on the southwest and northeast rims of the Gulf of Mexico.

11.1.4 General In-Water Abundance of Gulf of Mexico Sea Turtles

The in-water abundance of the five species of sea turtles that inhabit the waters of the Gulf of Mexico is difficult to ascertain given the lack of long-term, systematic studies. Indeed, the Gulf may arguably be the most data-deficient basin in terms of its sea turtle populations. Efforts to determine the presence and abundance of all species in U.S. waters seem to have concentrated in Texas and Florida, likely due to the presence of nesting beaches in these states–Florida boasting by far the largest numbers (e.g., Meylan et al. 1995). Aerial surveys over Gulf of Mexico waters have been used frequently to address this deficiency of data. However, no reports exist regarding the southern Gulf of Mexico, and most of the reports available for the northern Gulf are point in time studies, over a season or a year, and lack the benefit of long-term, systematic records that could be used to establish population trends. This has led researchers to state that currently it is virtually impossible to assess and restore the sea turtle populations of the Gulf of Mexico in relation to their historical abundance (Bjorndal et al. 2011). While there are differences in the methodologies of aerial surveys that have been conducted in the Gulf and variables such as speed, altitude, visibility, and lack of consistency in the areas surveyed limit the accuracy of the observations, available aerial survey information is presented in the following paragraphs.

During aerial surveys conducted in the Gulf of Mexico from June 1980 to April 1981, loggerheads were observed nearly 50 times as often in waters off the Florida Gulf coast compared to those observed in the western Gulf (Fritts et al. 1983a). They were present throughout the year, mostly in waters less than 50 m (164 ft) deep, but the frequency of sightings was lowest during the winter. Green turtles were infrequently observed in the Gulf of Mexico. Kemp's ridleys were most frequently sighted off southwest Florida and rarely observed in the western Gulf. Leatherbacks were observed more often on the continental shelf than in deeper waters (Fritts et al. 1983a).

Differences in sea turtle distribution in the eastern and western U.S. Gulf of Mexico were found in an analysis of Sea Turtle Stranding and Salvage Network (STSSN) data (see Section 11.7.4 for an explanation of these data) collected from 1985 through 1991 (Teas 1993). Large numbers of juvenile and adult loggerheads occurred in the eastern Gulf of Mexico during the spring and summer, especially along the south Florida Gulf coast. During all seasons, juvenile and adult green and Kemp's ridley sea turtles used eastern Gulf waters extensively. Low numbers of hawksbills also used the eastern Gulf of Mexico, while leatherbacks were found in the eastern Gulf during the spring and fall as they migrated through to preferred feeding and nesting grounds (Teas 1993). The western Gulf of Mexico provided year-round habitat for juvenile loggerheads and for hatchlings to adult Kemp's ridleys. Throughout the year, juvenile green turtles used western Gulf of Mexico waters (Teas 1993). During the summer and fall when prevailing currents carried them into the western Gulf, hawksbills ranging from hatchling to juvenile were common. Leatherback sea turtles migrated through the western Gulf during the spring and fall (Teas 1993).

Surveys were conducted in 1991 and 1992 to establish relative sea turtle abundance and seasonality in the U.S. Gulf of Mexico (Braun-McNeill and Epperly 2002). The study was based on surveying fishermen along the Gulf coast from Louisiana to southern Florida year-round regarding the sighting of sea turtles. The surveys indicated that sea turtle abundance along the Gulf coast was seasonal, with turtles migrating northward in the warmer months and then migrating south in the colder months. This seasonality was in agreement with historical information obtained from turtle fisheries in the Gulf States (Stevenson 1893; Carr and Caldwell 1956). The study also demonstrated that the number of turtle sightings was significantly higher in the Florida Keys than in any other location and that, throughout the study area, most turtles tended to be located within 506 km (314 mi) from the coast (Braun-McNeill and Epperly 2002).

An analysis of National Marine Fisheries Service (NMFS) aerial survey data for September, October, and November of 1992, 1993, and 1994 was conducted to determine sea turtle spatial dynamics for the U.S. Gulf of Mexico (McDaniel 1998). The results of the study indicated that sea turtles were observed at much higher rates along the Florida Gulf coast than in the western Gulf, and the highest density of observed sea turtles occurred in the Florida Keys region (0.525 turtles per square kilometer [km^2] or 0.203 turtles per square mile [mi^2]) (McDaniel et al. 2000). These results are similar to those obtained by Fritts et al. (1983a) discussed above. Various hypotheses were proposed to explain the higher numbers of sea turtles observed in the eastern Gulf as compared to the western Gulf of Mexico; these included the following: more suitable sea turtle habitat in the eastern Gulf of Mexico, the reduction of turtles by the intense shrimp fishery in the western Gulf of Mexico, low oxygen levels off the Louisiana coast, sea turtles being attracted to shrimp vessel bycatch, and more turtles inhabiting nearshore areas compared to areas offshore (McDaniel 1998).

In the same study conducted by McDaniel et al. (2000), sea turtle abundance decreased significantly west from Florida toward the north-central Gulf of Mexico and increased 20-fold in south Texas as compared to other areas surveyed in the western Gulf, ranging from no turtles up to 0.10 turtles/km^2 (0.04 turtles/mi^2) (McDaniel et al. 2000). This is consistent with an earlier week-long aerial survey conducted in the fall of 1979 in the same south Texas area that reported a mean sea turtle density of 0.0196 turtle/km^2 (0.0076 turtles/mi^2) (Reeves and Leatherwood 1983). Interestingly, a small peak around the Chandeleur Islands in Louisiana was observed (McDaniel et al. 2000); this peak around the Chandeleur Islands is important because Louisiana waters are known foraging grounds for post-nesting Kemp's ridleys (Chávez 1968; Pritchard and Márquez-M. 1973; Ogren 1989).

In aerial surveys conducted along the U.S. Gulf Coast from September through November during 1992 through 1996, Kemp's ridleys were sighted primarily in inshore waters and most commonly occurred in the eastern Gulf of Mexico (Epperly et al. 2002). During the same surveys, loggerhead sea turtles were sighted throughout the Gulf but had a very low occurrence in offshore waters of the western Gulf. Green turtles occurred offshore and primarily were sighted in the southern portion of the Florida Gulf coast. Hawksbills occurred mainly in southwest Florida, and leatherback sea turtles were more broadly distributed and were observed predominantly in offshore waters (Epperly et al. 2002).

Loggerhead sea turtles are the most abundant sea turtle species in the Gulf of Mexico (Henwood 1987). The nearshore waters of the northwestern Gulf provide important foraging areas for loggerhead sea turtles (Plotkin et al. 1993; Plotkin 1996). Loggerhead densities of 0.04 turtles/km^2 (0.015 turtles/mi^2) were reported for the northeastern Gulf of Mexico during aerial and ship surveys conducted from 1996 through 1998 (Mullin and Hoggard 2000). In a survey of the eastern Gulf of Mexico continental shelf along a series of transects between Tampa Bay and Charlotte Harbor, Florida, conducted from November 1998 through November 2000 between the coast and the 180 m (591 ft) isobaths, the overall density of loggerhead sea turtles was estimated to be 0.013 turtles/km^2 (0.005 turtles/mi^2) (Griffin and Griffin 2003); since unidentified turtles were not included in the analyses, the abundance of loggerheads for the eastern continental shelf of the Gulf of Mexico was most likely underestimated.

11.1.5 Regulation and Protection of Gulf of Mexico Sea Turtles

Two pieces of legislation were crucial to the protection of sea turtle species around the world and in the United States: the U.S. ESA and the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), the latter being an international agreement regulating the international trade of endangered species. Both pieces were enacted or signed by the United States in 1973, a time when sea turtle populations, particularly those in the Gulf of Mexico and greater Caribbean, exhibited evident signs of overexploitation.

All sea turtles occurring in the Gulf of Mexico are listed under the U.S. ESA and are under the joint jurisdiction of the National Oceanic and Atmospheric Administration (NOAA), the NMFS, and the U.S. Fish and Wildlife Service (USFWS) (NOAA 2013). The USFWS has lead responsibility on the nesting beaches, while the NMFS is the lead agency in the marine environment (NMFS et al. 2011). Kemp's ridley, leatherback, and hawksbill sea turtles are listed as endangered under the ESA (NOAA 2013). The overall listing status for the loggerhead sea turtle is threatened; however, each of the nine distinct population segments (DPSs) of loggerheads has a separate listing (NOAA 2013). The Northwest Atlantic Ocean DPS of loggerheads, whose range includes the Gulf of Mexico, is listed as threatened (USFWS and NMFS 2011). Green sea turtles have two listed populations: the Florida and Mexican Pacific coast green turtle breeding colonies are listed as endangered, and green turtles in all other areas are listed as threatened (NOAA 2013).

The NMFS and USFWS established the DPS policy in 1996 (USFWS and NMFS 1996). A population is considered to be a DPS if it is both discrete and significant relative to its taxon (taxonomic group). A population may be considered discrete if it satisfies either of the following conditions: (1) it is markedly separated from other populations of the same taxon as a consequence of physical, physiological, ecological, and behavioral factors (often based on genetic evidence); or (2) it is delimited by international government boundaries within which significant differences in control of exploitation, management of habitat, conservation status, or regulatory mechanisms exist. If a population segment is considered to be discrete, the NMFS and/or the USFWS must then determine whether the DPS is significant relative to its taxon using established criteria (USFWS and NMFS 1996).

Sea turtles that occur in the Gulf of Mexico are also listed by many U.S. Gulf Coast states as threatened and endangered species. State agencies that protect, regulate, and study sea turtles along the U.S. Gulf Coast include the Florida Fish and Wildlife Conservation Commission; Alabama Department of Conservation and Natural Resources; Mississippi Department of

Wildlife, Fisheries, and Parks; Louisiana Department of Wildlife and Fisheries; and Texas Parks and Wildlife Department (ADCNR 2012; FFWCC 2012; LDWF 2012; MDWFP 2012; TPWD 2012). In Mexico, sea turtles are regulated by the Comisión Nacional de Areas Naturales Protegidas, Secretaría de Medio Ambiente y Recursos Naturales (SEMARNAT) (SEMARNAT 2012). The taking of all sea turtles in Mexico was prohibited by presidential decree in 1990 (DOF 1990), and the National Program for Protection, Conservation, Research, and Management of Marine Turtles was implemented in Mexico in 2000 (NMFS et al. 2011).

Kemp's ridley, leatherback, and hawksbill sea turtles are listed as critically endangered by the International Union for the Conservation of Nature, and loggerhead and green sea turtles are listed as endangered (IUCN 2012). All sea turtles that occur in the Gulf are listed in CITES's Appendix 1, which includes species identified as endangered and prohibits all commercial international trade (NMFS et al. 2011).

11.1.5.1 History of Kemp's Ridley Sea Turtle Protection in the United States and Mexico

Under the ESA, the Kemp's ridley sea turtle was listed as endangered throughout its range on December 2, 1970 (NMFS and USFWS 2007a). Five-year status reviews of the Kemp's ridley were conducted by the NMFS in 1985, by the USFWS in 1985 and 1991, and by the NMFS and USFWS in 1995 and 2007; no change in the Kemp's ridley endangered listing status was recommended as a result of these reviews (Mager 1985; Plotkin 1995; NMFS and USFWS 2007a). The initial recovery plan for the Kemp's ridley sea turtle, which was for all six sea turtle species occurring in the United States, was approved by the NMFS on September 19, 1984, and the first revision and separate recovery plan for the Kemp's ridley was approved by the USFWS and NMFS on August 21, 1992 (NMFS and USFWS 2007a). In 2002, the USFWS, NMFS, and Mexico's SEMARNAT initiated the process to revise the recovery plan for a second time, but this time as a binational recovery plan; the second revision of this plan was approved on September 22, 2011 (NMFS et al. 2011).

In Mexico, efforts to protect Kemp's ridleys and their nesting beaches have been ongoing since the 1960s (Márquez-M. 1994). The harvest of Kemp's ridleys in the Gulf of Mexico has been prohibited since the 1970s (Márquez-M. et al. 1989). In 1977, Rancho Nuevo, the only mass-nesting site for the Kemp's ridley, was declared a natural reserve and further protective measures were added in 1986 (DOF 1977, 1986; Márquez-M. et al. 1989). Rancho Nuevo was declared a sanctuary in 2002 and was included in the listing of Wetlands of International Importance under the Convention on Wetlands in 2004 (DOF 2002; NMFS et al. 2011).

11.1.5.2 History of Loggerhead Sea Turtle Protection in the United States

The loggerhead sea turtle was listed as threatened throughout its range under the ESA on July 28, 1978 (Conant et al. 2009). The initial recovery plan for the loggerhead sea turtle was approved by the NMFS on September 19, 1984 (NMFS et al. 2011), while the first revision of the loggerhead recovery plan, which focused on the U.S. population in the Atlantic Ocean, was approved by the USFWS and NMFS on December 26, 1991 (NMFS and USFWS 2008). No change in the loggerhead threatened listing status was recommended as a result of 5-year status reviews conducted by the NMFS in 1985 and the USFWS in 1991 (Mager 1985; USFWS 1991). While no change to the loggerhead's listing status as threatened was recommended as a result of the joint 5-year review conducted by the NMFS and USFWS in 1995, the review identified the need to conduct additional research regarding the existence of two separate nesting populations along the southeast U.S. coast: the Florida subpopulation and the subpopulation nesting from Georgia through southern Virginia (Plotkin 1995). The results of research conducted between the 1995 5-year status review and the 2007 joint review completed by the NMFS and USFWS indicated that loggerhead populations might be separated by ocean basins, and while no change to the threatened listing of the loggerhead was recommended as a result of the 2007 status review, a commitment was made to determine the applicability of the DPS policy (NMFS and USFWS 2007b).

Five recovery units/subpopulations of loggerheads within the northwest Atlantic Ocean were recognized in the second revision and most recent version of the *Recovery Plan for the Northwest Atlantic Population of the Loggerhead Sea Turtle* (NMFS and USFWS 2008). The recovery units/subpopulations include the northern subpopulation (southern Virginia through the Florida/Georgia border), Peninsular Florida subpopulation (Florida/Georgia border through Pinellas County, Florida), Dry Tortugas subpopulation (islands located west of Key West, Florida), northern Gulf of Mexico subpopulation (Franklin County, Florida through Texas), and Greater Caribbean subpopulation (Mexico through French Guiana, The Bahamas, Lesser Antilles, and Greater Antilles). A threats analysis was conducted for all loggerhead life stages in support of the recovery plan to prioritize conservation actions relative to their impact on population growth rate and to support development of management priorities (NMFS and USFWS 2008; Bolten et al. 2011). During the most recent loggerhead status review, nine loggerhead DPSs were identified (Conant et al. 2009). They include the DPSs for the North Pacific Ocean, South Pacific Ocean, North Indian Ocean, Southeast Indo-Pacific Ocean, Southwest Indian Ocean, Northwest Atlantic Ocean, Northeast Atlantic Ocean, Mediterranean Sea, and South Atlantic Ocean (Figure 11.6).

The NMFS and USFWS published a proposed rule in March 2010, in which a Northwest Atlantic Ocean DPS would be established and listed as endangered under the ESA (USFWS and NMFS 2011). However, prior to making a final determination, nesting data available after the proposed rule was published and information provided by reviewers was evaluated; it was ultimately determined that listing the Northwest Atlantic Ocean DPS as threatened was more appropriate because the nesting population was large, the overall nesting population remained

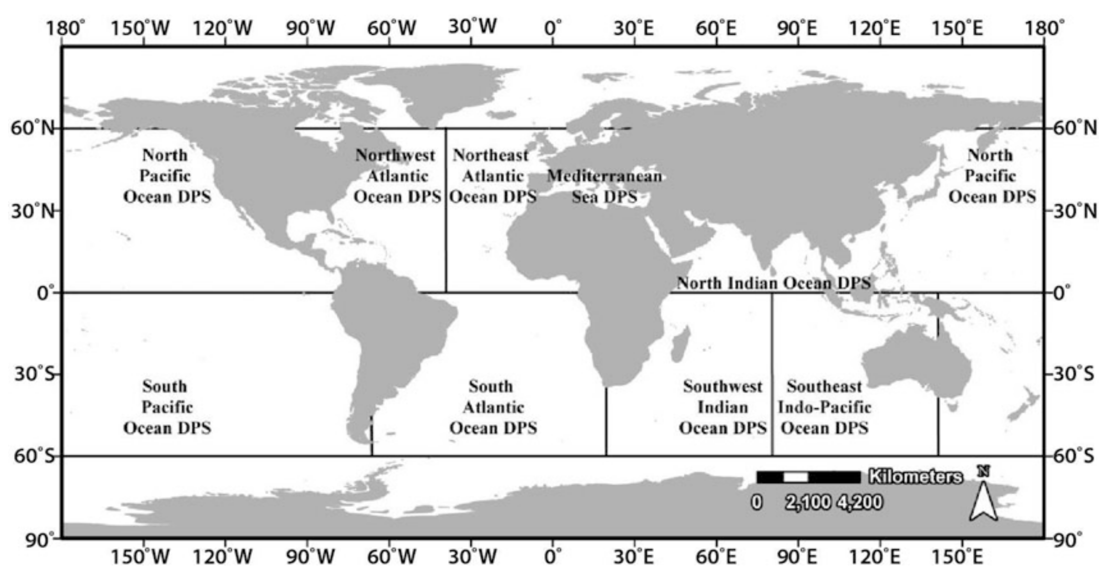

Figure 11.6. Location of the loggerhead sea turtle distinct population segment (DPS) boundaries (redrawn from USFWS and NMFS 2011).

widespread, the trend for the nesting population appeared to be stabilizing, and substantial conservation efforts were underway to address threats (USFWS and NMFS 2011).

In September 2011, the NMFS and USFWS listed nine DPSs of loggerhead sea turtles under the ESA (USFWS and NMFS 2011). Four DPSs (Southeast Indo-Pacific Ocean, Southwest Indian Ocean, Northwest Atlantic Ocean, and South Atlantic Ocean) were listed as threatened, and five DPSs (North Pacific Ocean, South Pacific Ocean, North Indian Ocean, Northeast Atlantic Ocean, and Mediterranean Sea) were listed as endangered. In addition, the rule stated that critical habitat for the two loggerhead DPSs occurring within the United States (Northwest Atlantic Ocean and Northeast Atlantic Ocean) will be proposed in future rulemaking, and information related to this effort was requested.

11.1.5.3 History of Green Sea Turtle Protection in the United States

Under the ESA on July 28, 1978, the green turtle breeding colony populations in Florida and on the Pacific coast of Mexico were listed as endangered, and green turtles in all other areas were listed as threatened (NMFS and USFWS 2007c). Five-year status reviews of the green sea turtle were conducted by the USFWS in 1983 and 1991, by the NMFS in 1985, and by the NMFS and USFWS in 1995 and 2007. In these reviews, no changes in the green turtle's listing status were recommended (Mager 1985; Plotkin 1995; NMFS and USFWS 2007c). The initial recovery plan for the green sea turtle was approved by the NMFS on September 19, 1984 (NMFS and USFWS 2007c). A recovery plan for the U.S. population of the Atlantic green turtle was approved on October 29, 1991, and recovery plans for the U.S. Pacific populations of the green turtle and the East Pacific green turtle were finalized in January 1998 (NMFS and USFWS 1991, 2007c). While the green turtle has no designated critical habitat in the Gulf of Mexico, marine critical habitat was designated in Puerto Rico on September 2, 1998 (NMFS and USFWS 2007c).

11.1.5.4 History of Leatherback Sea Turtle Protection in the United States

The leatherback sea turtle was listed as endangered throughout its range under the ESA on June 2, 1970, and the listing status has since remained unchanged (NMFS 2011b). Leatherbacks were included in the initial recovery plan for all sea turtle species in the United States approved by the NMFS on September 19, 1984 (NMFS and USFWS 2007d). The NMFS and USFWS completed recovery plans for leatherbacks in the U.S. Caribbean, Atlantic Ocean, and Gulf of Mexico in 1992 and for leatherbacks in the U.S. Pacific Ocean in 1998 (NMFS 1992; NMFS and USFWS 2007d). The leatherback has no designated critical habitat in the Gulf; however, terrestrial and marine critical habitat for the leatherback was designated on and around St. Croix, U.S. Virgin Islands in 1978 and 1979, respectively (NMFS and USFWS 2007d). The NMFS added waters adjacent to the U.S. west coast to the designated critical habitat in 2010. In 2011, waters surrounding a major nesting beach location in Puerto Rico were added (NMFS 2011b). Status reviews were conducted by the NMFS in 1985, by the USFWS in 1995 and 1991, and by the NMFS in 1995 (Plotkin 1995; NMFS 2011b). The most recent 5-year review was completed jointly by the NMFS and USFWS in 2007, and further review to determine the application of the DPS policy to leatherbacks was suggested (NMFS and USFWS 2007d).

11.1.5.5 History of Hawksbill Sea Turtle Protection in the United States

The hawksbill sea turtle was listed as endangered throughout its range under the ESA on June 2, 1970 (NMFS and USFWS 2007e). While the hawksbill has no designated critical habitat in the Gulf of Mexico, terrestrial and marine critical habitat was designated for the hawksbill in

Puerto Rico on June 24, 1982 and on September 2, 1998, respectively (NMFS and USFWS 2007e). Five-year status reviews of the hawksbill sea turtle were conducted by the NMFS in 1985, by the USFWS in 1985 and 1991, and by the NMFS and USFWS in 1995 and 2007. While no changes in the hawksbill's listing classification were recommended as a result of these reviews, a future analysis of the hawksbill was recommended in the most recent review to determine the application of the DPS policy to this species (Mager 1985; Plotkin 1995; NMFS and USFWS 2007e). The initial recovery plan for the hawksbill sea turtle was approved by the NMFS on September 19, 1984 (NMFS and USFWS 2007e). The first revision and separate recovery plan for the hawksbill in the U.S. Caribbean, Atlantic Ocean, and Gulf of Mexico was approved by the NMFS and USFWS on December 15, 1993, and a recovery plan for the U.S. Pacific populations of the hawksbill sea turtle was issued on January 12, 1998 (NMFS and USFWS 2007e).

11.2 KEMP'S RIDLEY SEA TURTLE (*LEPIDOCHELYS KEMPII*)

Along with the flatback sea turtle (*Natator depressus*), the Kemp's ridley sea turtle has the most geographically restricted distribution of all sea turtle species (Morreale et al. 2007). The smallest of the sea turtle species, the Kemp's ridley was first described in the late 1800s and named for Richard M. Kemp, a fisherman and naturalist from Key West, Florida, who submitted the type specimen (Figures 11.7 and 11.8) (NMFS et al. 2011). Unlike other species of sea turtles, which emerge individually on beaches to lay their eggs in the sand, the Kemp's ridley, as well as the closely related olive ridley, typically comes ashore in large, synchronous aggregations to lay their eggs; these events, or *arribadas*, occur at only a few beaches around the world (Plotkin 2007a). While olive ridleys typically nest at night like most sea turtles, Kemp's ridleys regularly nest during daylight hours (Safina and Wallace 2010). Another significant difference between the Kemp's ridley and the olive ridley is that the latter is the most abundant of all the sea turtle species (Valverde et al. 2012), while Kemp's ridleys are the least abundant species of sea turtles. In addition to the arribadas, the two species share the

Figure 11.7. Nesting Kemp's ridley sea turtle (from NOAA 2011).

Figure 11.8. Kemp's ridley sea turtle in the water (photograph by Kim Bassos-Hull, Mote Marine Laboratory) (NOAA 2011).

trademark *ridley dance*, in which the nesting females rock from side to side using their bodies to tamp sand on top of their nests (Safina and Wallace 2010).

The life history, particularly the nesting beach locations, of the Kemp's ridley sea turtle remained a mystery through the 1950s; some thought the Kemp's ridley was a hybrid of the loggerhead and green sea turtles (Carr 1979). However, the western Gulf of Mexico was determined to be important for this species when two Kemp's ridleys were found nesting during the day on Padre Island, Texas, in 1948 and 1950 (Werler 1951). The only mass-nesting site for the Kemp's ridley—the Rancho Nuevo area located in Tamaulipas, Mexico, already impacted from years of egg overexploitation—was not discovered by the scientific community until the early 1960s (Carr 1963; Hildebrand 1963).

Due to overexploitation and accidental mortality in fishing gear, the Kemp's ridley sea turtle came perilously close to extinction in the 1980s (Crowder and Heppell 2011). Due to the intensive, cooperative efforts by researchers and volunteers in Mexico and the United States (see Section 11.2.2), the Kemp's ridley rebounded from the brink of extinction (Heppell et al. 2007). The Kemp's ridley has recovered remarkably because conservation efforts have focused on stressors affecting all life stages, from eggs to juveniles and adults at sea. While the story of Kemp's ridley recovery is not finished, the trajectory is promising (Crowder and Heppell 2011). The combination of turtle excluder device (TED) use (see Sections 11.2.2 and 11.7.1.2), reductions in the shrimping effort, and nest protection on Mexican beaches has resulted in an unusually rapid recovery for a long-lived, slow-growing vertebrate. However, in spite of the recent gains over the lowest abundance of the 1980s and increased protection measures, Kemp's ridley populations remain significantly below historical levels.

11.2.1 Kemp's Ridley Life History, Distribution, and Abundance

The Kemp's ridley sea turtle occurs in the Gulf of Mexico and along the U.S. Atlantic Coast (Figure 11.9). The vast majority of Kemp's ridley nesting occurs on beaches in the western Gulf (Figure 11.10), and most juveniles spend time in the Gulf of Mexico oceanic zone after they

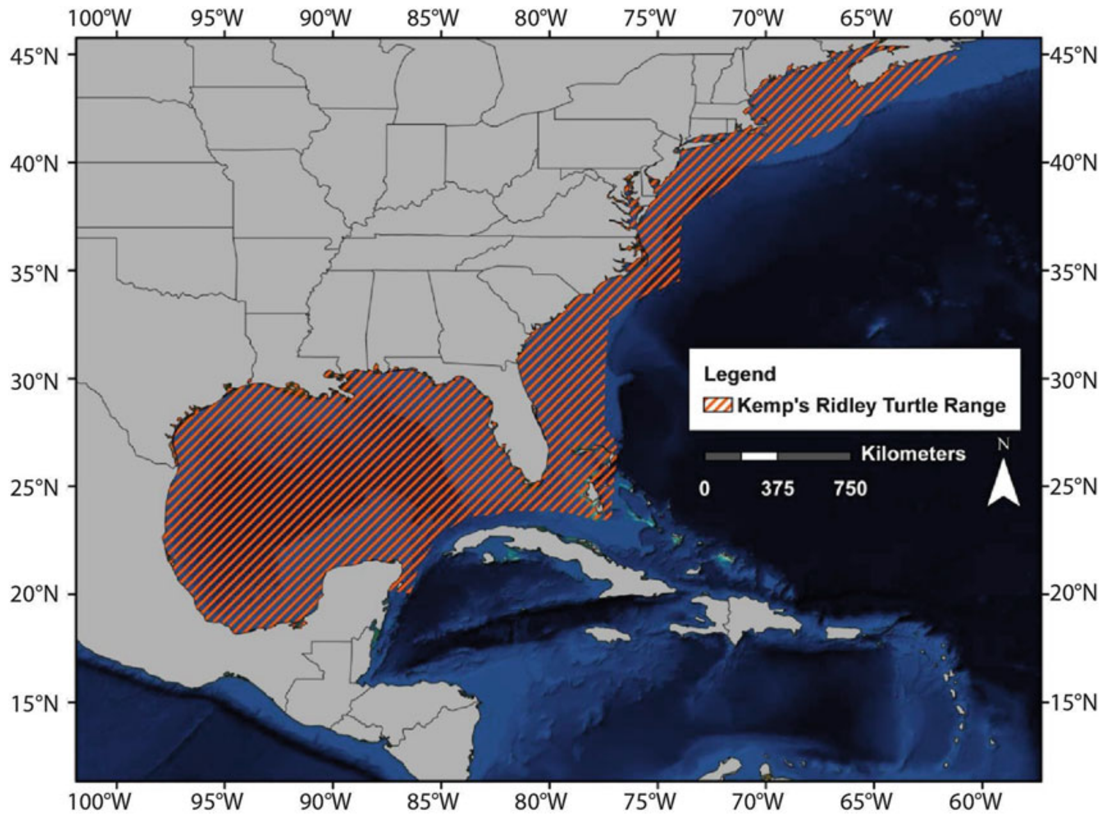

Figure 11.9. Range of the Kemp's ridley sea turtle (from NOAA 2007).

leave the beach as hatchlings (Collard and Ogren 1990; TEWG 2000; Putman et al. 2010). After the oceanic juvenile stage, juveniles recruit into the neritic zone (inshore marine environment), mostly along the Gulf continental shelf but also along the U.S. Atlantic Coast (Pritchard 1969; Ogren 1989; Schmid 1998; Witzell and Schmid 2004; Seney and Landry 2011), where they continue to feed and grow for a number of years until reaching sexual maturity. Oceanic juveniles that end up in the currents of the Atlantic Ocean move into coastal habitats along the east coast of the United States from Florida to New England, and as far north as the Grand Banks and Nova Scotia (Pritchard 1969; Ogren 1989; Morreale and Standora 1999; Watson et al. 2004; Morreale et al. 2007; Frazier et al. 2007; Landry and Seney 2008). Some Kemp's ridleys have been found in European Atlantic waters, the Mediterranean, and the Azores (Brongersma 1972; Brongersma and Carr 1983; Fontaine et al. 1989; Fretey 2001). Many of these juveniles return to the Gulf of Mexico to reproduce; for example, neritic juvenile Kemp's ridleys that were tagged along the U.S. Atlantic Coast have nested at Rancho Nuevo, Mexico (Witzell 1998; Schmid and Woodhead 2000). Adult Kemp's ridley sea turtles occur primarily in the Gulf of Mexico, typically in nearshore waters (Hildebrand 1982; Ogren 1989; USFWS and NMFS 1992; Pritchard 2007a). A summary of life history information for the Kemp's ridley specific to the Gulf of Mexico is included in Table 11.1; information available for specific Gulf beaches or regions is also included in the table.

Table 11.1. Summary of Life History Information for the Kemp's Ridley Sea Turtle

Parameter	Values	References
Nesting season: Gulf of Mexico	April through July	Hirth (1980)
Remigration interval: Rancho Nuevo, Mexico	Mean: 2 years	Márquez-M et al. (1982)
	Mean: 1.5 years	van Buskirk and Crowder (1994)
Nesting (arribada) interval: Rancho Nuevo, Mexico	Range: 20–28 days	Chávez (1969)
	Mean: 25 days	Rostal et al. (1997)
Number of nests/season: Rancho Nuevo, Mexico	Mean: 3.1 nests	Rostal (1991)
	Mean: 2.5 nests	Heppell et al. (2005)
	Mean: 3.1 nests	Rostal (2005)
Number of eggs/nest		
Rancho Nuevo, Mexico	Mean: 116 eggs, Range: 93–135 eggs	Pritchard and Márquez-M. (1973)
	Mean: 104 eggs, Range: 17–192 eggs	Márquez-M. (1994)
	Mean: 95 eggs	Coyne (2000)
Upper Texas Region	Mean: 99 eggs, Range: 71–119 eggs	Seney (2008)
Egg incubation time		
Rancho Nuevo, Mexico	Range: 50–70 days	Chávez et al. (1967)
	Range: 45–58 days	Márquez-M. (1990)
Padre Island National Seashore, Texas	Mean: 49.7 days	Shaver (2005)
Nest pivotal temperature	30.2 °C	Shaver et al. (1988)
Sex ratio of hatchlings from in situ nests (proportional female)		
Rancho Nuevo, Mexico	Mean: 0.80	Wibbels and Geis (2003)
	Mean: 0.64	T. Wibbels, UAB, unpublished data, cited in NMFS and USFWS (2007a)
Padre Island National Seashore, Texas	Mean: 0.60	Shaver (2005)
Emergence success of hatchlings from in situ nests		
Rancho Nuevo, Mexico	Mean: 0.66	USFWS (2006)
	Mean: 0.80	J. Pena, GPZ, personal communication, cited in NMFS et al. (2011)
Padre Island National Seashore, Texas	Mean: 0.62	Shaver (2005, 2006a, b, 2007, 2008), D. Shaver, PAIS, unpublished data, cited in NMFS et al. (2011)
Size of hatchlings	Mean: 4.4 cm SCL[a]	Márquez-M. (1972)
	Mean: 3.8 cm SCL	NOAA Fisheries OPR (2013)
Size of oceanic juveniles: Cedar Keys, Florida	Range: 5–19 cm SCL	Gregory and Schmid (2001)
Duration of oceanic juvenile stage: Cedar Keys and Cape Canaveral, Florida	Mean: 2 years	Schmid and Witzell (1997)
	Estimated maximum: 4 years	Putman et al. (2010)

(continued)

Table 11.1. (continued)

Parameter	Values	References
Diet of oceanic juveniles		
Lower Texas Region	Marine mollusks associated with the pelagic *Sargassum* community, including brown janthinas, *Cavolina longirostris*, *Sargassum* snails, and unidentifiable crabs, and *Sargassum*	Shaver (1991)
Texas and western Louisiana	Hardhead catfish, blue crabs, stone crabs, mottled purse crabs, and *Sargassum*	Zimmerman (1998)
Gulf Stream off Florida's Gulf coast	Marine animals associated with the pelagic *Sargassum* community, including hydroids, *Membranipora* sp., *Sargassum* anemones, serpulid polychaetes, gastropods, *Sargassum* snails, and *Sargassum* swimming crabs; *Sargassum*; and cladophora algae	Witherington et al. (2012)
Size of neritic juveniles: Sea Rim State Park, Texas to Cedar Keys, Florida	Range: 20–60 cm SCL	Ogren (1989)
Duration of neritic juvenile stage: Mississippi Sound, Mississippi to Ten Thousand Islands, Florida	Range: 8–9 years	Schmid and Barichivich (2005)
	Range: 7–8 years	Schmid and Woodhead (2000)
Diet of neritic juveniles		
Southern Texas	Speckled swimming crabs, blue crabs, mottled purse crabs, *Libinia* sp., calico crabs, surf hermits, Gulf stone crabs, bruised nassas, sharp nassas, moon snails, concentric nut clams, oysters, American stardrums, spot croakers, *Sargassum*, shoalgrass, *Gracilaria* sp., turtle grass, brown shrimp, and white shrimp	Shaver (1991)
Matagorda and Galveston Bays, Texas	Blue crabs, calico crabs, longnose spider crabs, *Ovalipes* sp., flat-clawed hermit crabs, mottled purse crabs, blood ark clams, transverse ark clams, *Anadara* sp., *Bittium* sp., angel wing clams, *Epitonium* sp., dwarf surf clams, bruised nassas, moon snails, *Terebra* sp., annelids, common sand dollars, mullet, and *Sargassum*	Seney (2008)

(continued)

Table 11.1. (continued)

Parameter	Values	References
Sabine Pass, Texas and Louisiana	Blue crabs, stone crabs, *Persephona aquilonaris*, thinstripe hermit crabs, dwarf surf clams, sharp nassas, oysters, catfish, *Sargassum*, shoalgrass, and bryozoans, including *Corallina cubensis*, common sheep's wool, and *Amathia distans*	Werner (1994)
Terrebonne Parish, Louisiana	Blue crabs, ornate blue crabs, *Nassarius* sp., and clams, including *Nuculana* sp., *Corbula* sp., and *Mulinia* sp.	Dobie et al. (1961)
Deadman Bay, Florida	Spider crabs, blue crabs, stone crabs, and mottled purse crabs	Barichivich et al. (1998)
Waccasassa Bay, Florida	Stone crabs, blue crabs, *Paguridae* sp., moon snails, bruised nassas, *Cantharus cancellarius*, eastern oysters, hooked mussels, shoalgrass, and star grass	Schmid (1998)
Charlotte Harbor Estuary, Florida	Spider crabs, mottled purse crabs, calico crabs, and blue crabs	Schmid (2011)
Gullivan Bay, Florida	Sea squirts, worm tubes, *Amathia* sp., hydroids, *Libinia* sp., mottled purse crabs, calico crabs, Atlantic horseshoe crabs, *Pitho* sp., *Hexapanopeus* sp., Florida stone crabs, giant marine hermit crabs, estuarine mud crabs, squatter pea crabs, *Marginella* sp., *Anadara* sp., *Lucina* sp., *Vermicularia* sp., turtle grass, shoalgrass, and manatee grass	Witzell and Schmid (2005)
Age at sexual maturity		
Rancho Nuevo, Mexico	Range: 5–7 years	Márquez-M (1972)
	Mean: 10 years	Coyne (2000)
Texas coast	Mean: 10 years	Caillouet et al. (1995)
	Range: 10–20 years	Shaver and Wibbels (2007)
Texas coast to southwest Florida	Range: 10–11 years	Schmid and Barichivich (2005)
Eastern Louisiana to southwest Florida	Range: 7–11 years	Schmid and Woodhead (2000)
Size of sexually mature adult females		
Rancho Nuevo, Mexico	Mean: 64 cm SCL, Range: 56–72.5 cm SCL	Burchfield et al. (1988)
	Minimum: 52.4 cm SCL	Márquez-M (1990)
Upper Texas Region to Louisiana coast	Mean: 60 cm SCL	Coyne and Landry (2000)

(continued)

Table 11.1. (continued)

Parameter	Values	References
Eastern Louisiana to southwest Florida	Mean: 60 cm SCL	Schmid and Barichivich (2005)
Diet of adults		
Lower Texas Region	Speckled swimming crabs, blue crabs, mottled purse crabs, *Libinia* sp., calico crabs, surf hermits, Gulf stone crabs, bruised nassas, sharp nassas, moon snails, concentric nut clams, oysters, star drums, spot croakers, *Sargassum*, shoalgrass, *Gracilaria* sp., turtle grass, brown shrimp, and white shrimp	Shaver (1991)
Gullivan Bay, Florida	Sea squirts, worm tubes, *Amathia* sp., hydroids, *Leptogoria* sp., *Libinia* sp., mottled purse crabs, calico crabs, Atlantic horseshoe crabs, *Pitho* sp., *Hexapanopeus* sp., Florida stone crabs, giant marine hermit crabs, estuarine mud crabs, blue crabs, squatter pea crabs, flatback mud crabs, *Nassarius* sp., *Marginella* sp., *Anadara* sp., eastern oysters, *Lucina* sp., *Vermicularia* sp., horse conches, turtle grass, shoalgrass, manatee grass, star grass, leafy caulerpa, and tonguefishes	Witzell and Schmid (2005)

[a] *SCL* straight carapace length, *cm* centimeters

11.2.1.1 Nesting Life History, Distribution, and Abundance for Gulf of Mexico Kemp's Ridleys

The single known aggregated nesting site and primary Kemp's ridley nesting beaches are located in Tamaulipas, Mexico, and include Rancho Nuevo, Tepehuajes, and Playa Dos (Figure 11.10) (Pritchard 2007b; NMFS et al. 2011). Nesting in Tamaulipas often occurs in arribadas, which may be triggered by strong onshore winds, especially north winds, as well as changes in barometric pressure (Jimenez et al. 2005). Individual nesting of Kemp's ridleys occurs from Texas to Veracruz, Mexico, and as far east as Campeche, Mexico (Figure 11.10) (Ross et al. 1989; Shaver 2005; Pritchard 2007a, b; Guzmán-Hernández et al. 2007). The majority of Kemp's ridley nesting in Texas occurs at PAIS, but low levels of nesting now regularly occur along the upper Texas coast, including in Matagorda, Brazoria, and Galveston Counties (Figure 11.10) (Shaver and Caillouet 1998; Shaver 2005; Seney 2008). The increase in nesting along the upper Texas coast represents either a northern expansion of Kemp's ridley nesting in the Gulf of Mexico or a reestablishment of its nesting range (Seney 2008).

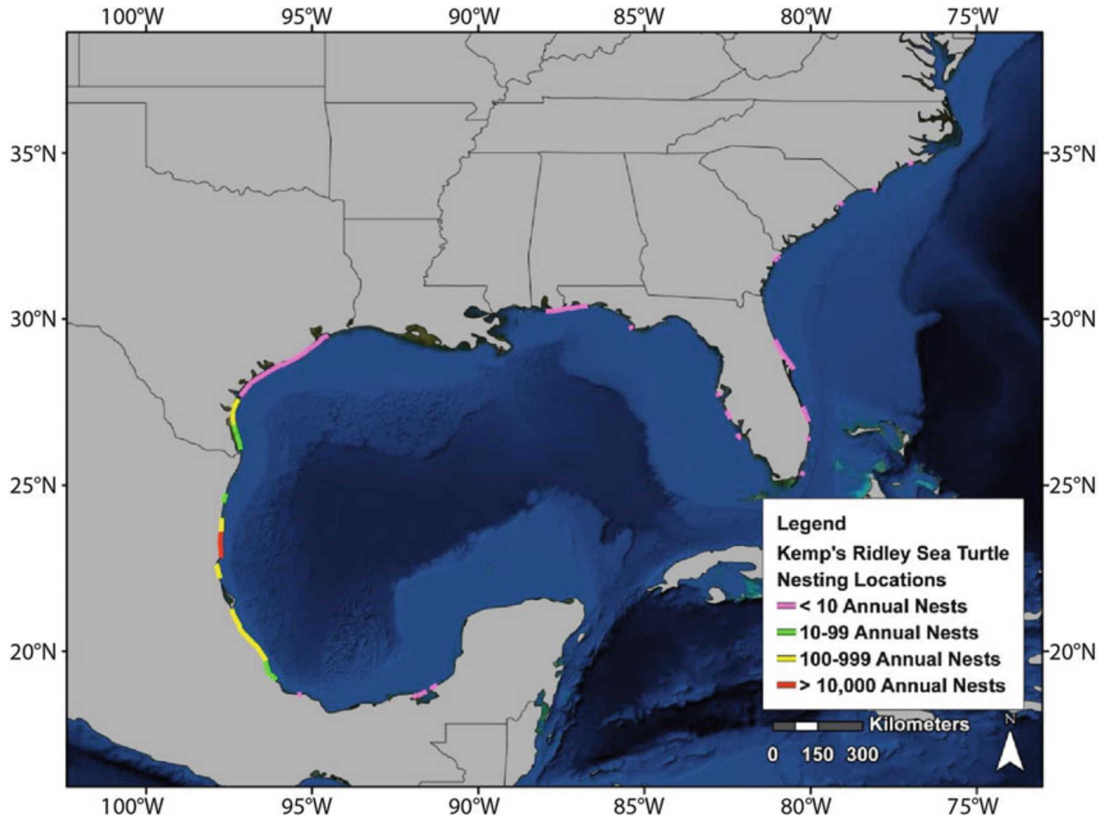

Figure 11.10. Generalized nesting beach locations of the Kemp's ridley sea turtle in the Gulf of Mexico and southeast U.S. Atlantic Coast (interpreted from Dow et al. 2007; SWOT 2010a; NMFS et al. 2011).

In addition, since the late 1980s, Kemp's ridley sea turtles have nested occasionally in Alabama, on Florida's Gulf and Atlantic coasts, and in Georgia, South Carolina, and North Carolina (Figure 11.10) (Meylan et al. 1991; Anonymous 1992; Márquez-M. et al. 1996; Johnson et al. 1999a; Williams et al. 2006).

Because of the limited nesting distribution, as well as the collaborative United States–Mexican recovery program (see Section 11.2.2), an entire time series of nesting information, beginning in 1966, is available for the Kemp's ridley for Rancho Nuevo and the adjacent beaches. This time series of information has little uncertainty after 1978, when nest protection methods became standardized and almost all nests were moved to a hatchery and recorded; however, since the mid-1990s, the level of uncertainty in estimating the population size has increased because of spatial expansion of the population and increased protection efforts (Márquez-M. et al. 1999; Heppell et al. 2007).

While more than 40,000 Kemp's ridleys were estimated to nest at Rancho Nuevo during an arribada in one day in 1947 (Carr 1963; Hildebrand 1963), only 924 nests were documented in 1978, and a low of 702 nests was recorded in 1985, representing about 300 and 228 nesting females, respectively (Figure 11.11). The number of nests observed at Rancho Nuevo and nearby beaches began to increase during the late 1980s and continued to increase at a rate of about 15 % per year (Figure 11.11) (Heppell et al. 2005; Crowder and Heppell 2011). In addition, the geographic range of nesting has expanded to the north and south of Rancho Nuevo (Heppell et al. 2007). Since 2005, the number of nests recorded in the Rancho Nuevo area each year

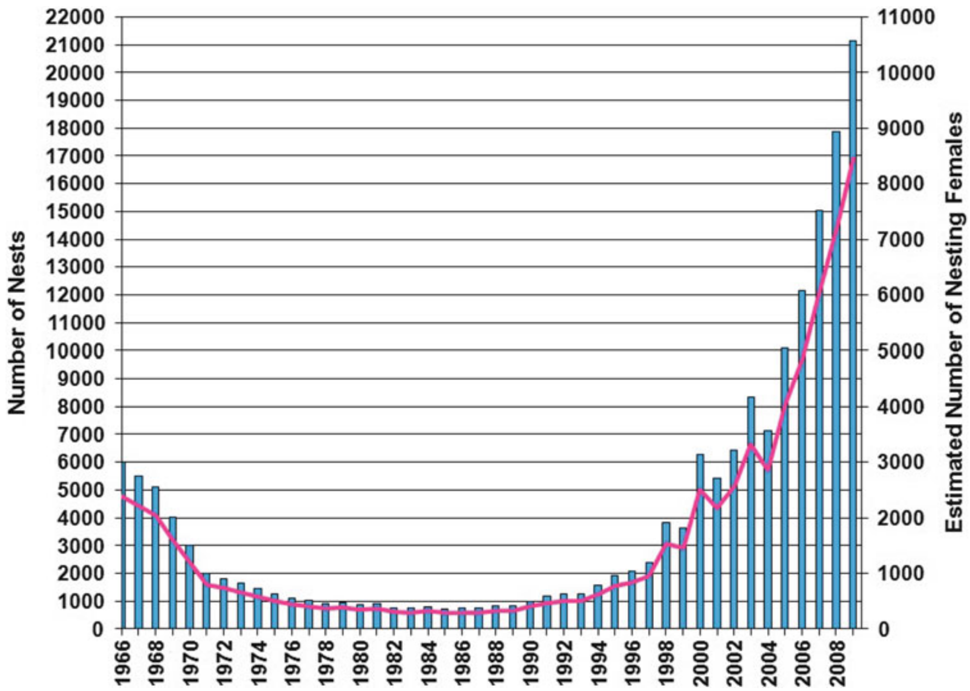

Figure 11.11. Annual number of nests (*bars*) and estimated number of nesting females (*line*), assuming 2.5 nests per female (Heppell et al. 2005), for Kemp's ridley sea turtles at Rancho Nuevo and adjacent beaches, Tamaulipas and Veracruz, Mexico from 1966 through 2009 (TEWG 2000; USFWS 2006; Alonso 2009).

consistently exceeded 10,000, indicating that at least 4,000 Kemp's ridleys were nesting each year (Figure 11.11). A record number of 21,144 nests (representing approximately 8,458 nesting Kemp's ridleys) were recorded at Rancho Nuevo and the adjacent beaches during 2009 (Figure 11.11). Approximately 13,000 nests were recorded from the Rancho Nuevo area during 2010, between 18,000 and 20,000 nests were recorded during 2011 (FuelFix 2011), and more than 21,000 nests were recorded in 2012 (NPS 2013a).

Besides its main nesting site at Tamaulipas, the beaches of Campeche, Mexico, are considered an important historic nesting site for the Kemp's ridley because regular nesting occurs, albeit at low levels (Guzmán-Hernández et al. 2007). The fact that nesting activity has been ongoing in this region, more than 1,200 km (746 mi) from Rancho Nuevo, corroborates the resilience of this highly vulnerable species and helps support the idea that Campeche was an important nesting region decades ago before the spread of overfishing and other human impacts (Márquez-M. 2004). From 1984 through 2003, 15 Kemp's ridley nests were recorded during the standardized patrolling and surveillance of sea turtle nesting beaches in Campeche. Nests were found on three beaches: ten nests on Isla Aguada, three on Isla del Carmen, and two in Sabancuy. While few in number compared to the hatchlings released at Rancho Nuevo, these nests contributed 1,109 hatchlings to the Kemp's ridley population in the Gulf of Mexico.

The number of Kemp's ridley nests along the Texas coast has increased dramatically since the late 1940s (Figure 11.12). The PAIS in Corpus Christi, Texas, is now considered a secondary nesting colony; more Kemp's ridley nests have been confirmed at PAIS than any other location in the United States during the last 50 years (Shaver 1999, 2005, 2006a). Kemp's ridleys that nest in Texas today are a mixture of head-started turtles—raised in captivity for a period of time and later released—and wild-stock turtles (Shaver 2005; Seney 2008). The larger size of the

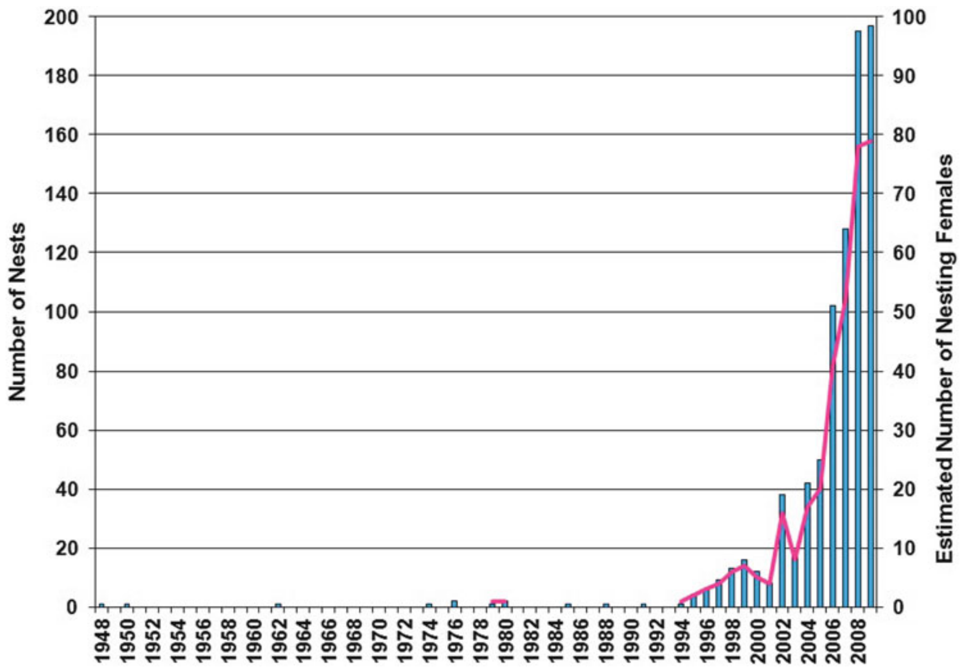

Figure 11.12. Annual number of nests (*bars*) and estimated number of nesting females (*line*), assuming 2.5 nests per female (Heppell et al. 2005), for Kemp's ridley sea turtles recorded on Texas beaches from 1948 through 2009 (Shaver and Caillouet 1998; NPS 2013b). Data were not available for 1949, 1951 through 1961, 1963 through 1973, 1975, 1977, 1978, 1981 through 1984, 1986, 1987, 1989, 1990, 1992, or 1993.

head-started turtles when released is assumed to decrease mortality (Section 11.2.2). Nesting surveys along the Texas coast, which have increased in recent years, have been extremely challenging due to the hundreds of kilometers of beaches that must be searched, limited resources, and logistical difficulties, and because of the nesting characteristics of Kemp's ridleys (e.g., nesting during the day on windy days) (Shaver and Rubio 2008). Eggs from nests found on many of the Texas beaches are moved to incubation facilities or protective corrals. While fewer than ten nests were recorded each year (for years for which data are available) prior to 1997, the number of nests on Texas beaches began to increase in 1998. Since 2006, more than 100 Kemp's ridley nests or at least 40 nesting Kemp's ridleys have been recorded for Texas beaches each year (Figure 11.12). In 2010, 141 nests were recorded along the Texas coast, and 199 nests were recorded in 2011 (NPS 2013a). A record number of 209 Kemp's ridley nests, representing about 84 nesting females, was recorded along the Texas coast in 2012 (NPS 2013b), and 153 nests were recorded during the 2013 nesting season (NPS 2013a). With increased productivity on the nesting beaches and improved survival at sea because of reductions in fishing mortality, conservation efforts boosted the current population of Kemp's ridleys from 7,000 to 8,000 breeding females nesting at multiple sites from Padre Island, Texas, to Veracruz in the southwestern Gulf of Mexico (Crowder and Heppell 2011).

11.2.1.2 Hatchling, Post-Hatchling, and Oceanic Juvenile Life History and Distribution for Gulf of Mexico Kemp's Ridleys

After the embryos have developed, the time depending on temperature and other incubation conditions (Table 11.1), Kemp's ridley hatchlings emerge from the nest *en masse* at night or

Figure 11.13. Kemp's ridley sea turtle hatchlings entering the sea after emerging from the nest (from NPS 2013c).

during the early morning and swim offshore into the oceanic zone to feed and grow (Figure 11.13) (NMFS et al. 2011). The estimated pivotal temperature for the Kemp's ridley is relatively high (30.2 °C) compared to those for other sea turtle species (29–29.6 °C) (Yntema and Mrosovsky 1982; Godfrey et al. 1999; Hulin et al. 2009); the high temperatures at the nesting beach at Rancho Nuevo appear to naturally produce a hatchling sex ratio that is female biased (Table 11.1) (Wibbels 2007).

Not much is known about the period after a Kemp's ridley hatchling leaves the beach, swims offshore, associates with boundary currents, and is transported by the open ocean currents, often known as the *lost years*. However, the Kemp's ridley lost years may be similar to what occurs for the loggerhead sea turtle (Collard and Ogren 1990; Bolten 2003; Witherington et al. 2012). The oceanic currents in the western Gulf of Mexico control Kemp's ridley hatchling transport; coastal, shelf, and offshore currents vary during the hatchling emergence period (Collard 1987). The migratory success of young turtles that quickly reach pelagic waters is highly variable and influenced by oceanic conditions across the Kemp's ridley nesting range (Putman et al. 2010).

A recent analysis of seven Kemp's ridley nesting regions indicated that Rancho Nuevo ranked highest for migratory success of hatchlings to pelagic habitat (Putman et al. 2010). The narrow continental shelf off Tamaulipas, as well as oceanic conditions offshore Tamaulipas and Veracruz, may facilitate hatchling transport to the pelagic environment within 1–4 days (Collard and Ogren 1990; Putman et al. 2010). Depending on the type, location, strength, and paths of surface currents, pelagic Kemp's ridleys may either complete the developmental phase of their life cycle in the western Gulf of Mexico or be transported to the east, entrained in the Loop Current, exit the Gulf through the Straits of Florida, and drift to the north on the western edge of the Florida Current/Gulf Stream (Collard 1987; Collard and Ogren 1990). Similar to loggerheads, post-hatchlings likely become passive migrants in oceanic currents and use the *Sargassum* community as developmental habitat (Shaver 1991; NMFS et al. 2011).

The oceanic juvenile stage can be divided into two groups: the majority that remain in the currents of the Gulf of Mexico and a smaller group that is entrained in the Florida Current and transported up the Atlantic coast by the Gulf Stream (Putman et al. 2010). Because of the variability in growth rates, there is a range in the time estimated for a hatchling to grow to a size of about 20 centimeters (cm) (7.9 in) straight carapace length (SCL), the size at which Kemp's ridleys typically transition to the next stage—the neritic juvenile stage (Table 11.1). Juvenile Kemp's ridleys in the oceanic zone feed mostly on pelagic invertebrate prey associated with the *Sargassum* community (Table 11.1).

No post-hatchling Kemp's ridleys were collected from the Gulf of Mexico off the Florida coast from 2005 through 2011 as part of a study to determine the importance of the pelagic *Sargassum*-dominated drift community to young sea turtles (Witherington et al. 2012). Hatchlings are typically around 4 cm (1.6 in) SCL in size (Table 11.1); the smallest Kemp's ridley collected by Witherington et al. (2012) was 17 cm (6.7 in) SCL. Witherington et al. (2012) sampled only in the eastern Gulf of Mexico from May through September, and post-hatchling Kemp's ridley would most likely occur in the western Gulf of Mexico off the principal nesting beaches.

Thirty-eight juvenile Kemp's ridleys, ranging in size from 17 to 28 cm (6.7 to 11 in) SCL, were captured from the *Sargassum*-dominated surface-pelagic drift community in the eastern Gulf of Mexico from 2005 through 2011 (Witherington et al. 2012). These turtles were estimated to be between 1 and 2 years old. Because they were similar in size to the lower size range observed in nearby neritic habitats (Schmid 1998; Witzell and Schmid 2004) and because most of these turtles were not found within currents that would transport them out of the Gulf, they were hypothesized to be on the cusp of recruiting into coastal habitats of the northern and eastern Gulf of Mexico. Similar to what was proposed by Collard and Ogren (1990), these data suggest that an important recruitment pulse occurs primarily in the northern and eastern Gulf of Mexico, marking the end of the oceanic juvenile stage and the beginning of the neritic juvenile stage and suggests that the open waters of the northern and eastern Gulf of Mexico are of unique importance to Kemp's ridley sea turtles (Witherington et al. 2012).

11.2.1.3 Neritic Juvenile Life History and Distribution for Gulf of Mexico Kemp's Ridleys

Juvenile Kemp's ridleys that remain in the Gulf of Mexico during the oceanic stage move into coastal waters and are known to concentrate in shallow coastal waters, bays, estuaries, and sounds of the Gulf from south Texas to southwest Florida (Ogren 1989; Rudloe et al. 1991; Schmid 1998; Witzell and Schmid 2004; Schmid and Barichivich 2005; Frazier et al. 2007; Seney 2008). Coastal developmental and foraging areas frequently used by neritic juvenile Kemp's ridleys include Bolivar Roads Channel, Sabine Pass, and Lavaca and Matagorda bays in Texas; Caillou Bay and Calcasieu Pass in Louisiana; Big Gulley in Alabama; and Charlotte Harbor, Apalachicola Bay, Apalachee Bay, Deadman Bay, Waccasassa Bay/Cedar Keys, and Gullivan Bay/Ten Thousand Islands in Florida (Landry et al. 1995, 2005; Schmid and Barichivich 2005, 2006; Renaud and Williams 1997, 2005; Eaton et al. 2008; Schmid 2011). Details regarding studies that have been conducted in these areas are presented in the following paragraphs. Juvenile neritic Kemp's ridley sea turtles have not been reported from the southern Gulf of Mexico (Carr 1984).

The initial transition, as well as subsequent movements, of juvenile Kemp's ridleys to and from these coastal habitats appears to be seasonal (NMFS et al. 2011). Data from capture-mark-recapture (CMR) and satellite telemetry studies in the Gulf of Mexico have documented that juvenile turtles leave the coastal foraging areas in the fall and move to more suitable overwintering habitat in deeper or more southern waters and return to the same coastal feeding areas the following spring (Ogren 1989; Schmid 1998; Witzell and Schmid 2004; Landry et al. 2005; Schmid and Barichivich 2005, 2006; Renaud and Williams 2005; Schmid and Witzell

2006; Seney and Landry 2011). For example, Renaud and Williams (2005) documented the fall and winter movements of Kemp's ridleys in the Gulf of Mexico in response to changing seawater temperature using satellite telemetry. Kemp's ridleys on the Florida Gulf coast moved in a southerly direction during the months of October through January as far south as the Florida Keys; once waters began to warm, they reversed their direction of movement. Southerly and southwesterly fall and winter migrations also were observed for turtles on the central and upper Texas coast (Renaud and Williams 2005).

Satellite telemetry was also used to monitor the winter migration of six neritic juveniles on the Florida Gulf coast (Schmid and Witzell 2006). All Kemp's ridleys departed from the Cedar Keys area in late November, when the average sea surface temperature dropped from 23.6 to 17.1 °C, migrated south in December, and overwintered in offshore waters from the Anclote Keys to Captiva Island during January. In February, when water temperatures increased to an average of 16.6 °C, the turtles started moving north and began returning to the Cedar Keys area in March.

Studies conducted in the early 1990s for the U.S. Army Corps of Engineers (USACE) to collect quantitative data to assess the risks of maintenance dredging to sea turtles demonstrated that the inshore and nearshore habitats of the upper Texas and Louisiana coasts were used by Kemp's ridleys on a seasonal basis and verified that jetties and channel entrances along the Texas and Louisiana coasts served as summer developmental habitat (Landry et al. 1992, 1993, 1994, 1995; Renaud et al. 1993a, 1995a). Small turtles arrived at Sabine Pass and Calcasieu Pass in April and May, and in June, July, and August, the mean size and overall abundance of turtles increased. Turtle abundance began to decrease in September, and by November, most, if not all, of the Kemp's ridleys had left the region. Kemp's ridley abundance was highest at Sabine Pass in Texas, followed by Calcasieu Pass and Bolivar Roads Channel. Small turtles (less than 18 kg or 39.7 lb) remained nearshore from May to October and moved less than large Kemp's ridleys (greater than 24 kg or 52.9 lb). Migration patterns of Kemp's ridleys varied by season and depended on turtle size; however, the majority of tracked turtles released near Sabine Pass and Calcasieu Pass remained within a few kilometers of shore and in relatively shallow waters. In addition, most maintained strong site fidelity to the westward side of both passes, most likely because current eddies and quiet water appeared to result in more favorable habitat and accumulation of prey (e.g., blue crabs), until cold fronts forced them to migrate south along the coast. Kemp's ridleys were found in dredged channels and moved back and forth across the passes and into inshore waters through shipping channels.

The nearshore Kemp's ridley assemblages in the western Gulf of Mexico were characterized from 1992 through 1998 by netting turtles at nine study areas from Grand Isle, Louisiana, to South Padre Island, Texas (Landry et al. 2005). The occurrence of Kemp's ridleys at Sabine and Calcasieu passes was typically limited to April through September, and no turtles were captured from December through February. The 429 Kemp's ridleys captured during the study ranged in size from 19.5 to 65.8 cm (7.7 to 25.9 in) SCL; 77 % of the turtles had an SCL of less than 40 cm (15.7 in), and about 2 % of the turtles captured were adults, with none being mature males. The results of the study indicated that nearshore Gulf of Mexico waters along the upper Texas and Louisiana coasts provide developmental habitat to neritic juveniles during late spring through summer, when blue crab abundance and discarded shrimp fishery bycatch were highest (Landry et al. 2005).

The long-term abundance and distribution of neritic juvenile Kemp's ridleys (20–40 cm [7.9–15.7 in] SCL) in the nearshore waters of the northwestern Gulf of Mexico was characterized using 10 years of entanglement netting data (Metz 2004). The nearshore waters included beachfront sites ranging in depth from 0.6 to 2 m (1.9 to 6.6 ft), while jetty sites ranged in depth from 1.5 to 3 m (4.9 to 9.8 ft). This 10-year survey, which was conducted by the Sea Turtle and

Fisheries Ecology Research Laboratory at Texas A&M University-Galveston, is the longest of its kind in the northwestern Gulf of Mexico and was conducted at locations (index habitats) that have a consistent occurrence of juvenile through adult Kemp's ridleys. The netting surveys were conducted primarily at Sabine Pass, Texas, and at Calcasieu Pass, Louisiana, as well as secondarily near the Mermentau River, Louisiana, from April through October 1993 through 2002. During the 10-year study, 600 Kemp's ridleys were captured, ranging in size from 19.5 to 66.3 cm (7.7 to 26.1 in) SCL; all annual mean size values were between 30 and 40 cm (11.8 and 15.7 in) SCL. Of all Kemp's ridleys captured during the study, 77 % were between 20 and 40 cm (7.9 and 15.7 in) SCL, about 20 % were between 40 and 60 cm (15.7 and 23.6 in) SCL, and 2 % were larger than 60 cm (23.6 in) SCL (Metz 2004). The size of Kemp's ridleys at Sabine Pass ranged from 19.5 to 64 cm (7.7 to 25.2 in) SCL, and no turtles larger than 55 cm (21.7 in) SCL were captured after 1998. Turtles captured at Calcasieu Pass were significantly larger and ranged in size between 22.4 and 66.3 cm (8.8 and 26.1 in) SCL.

Most likely in response to rising water temperatures and seasonal occurrence of blue crab prey, the overall monthly Kemp's ridley catch per unit effort (CPUE) peaked from April through June during the 10-year study in northwestern Gulf of Mexico nearshore waters (Metz 2004). The annual mean ridley CPUE across all study areas peaked in 1994, 1997, 1999, and 2002, which suggested a 2- to 3-year cycle in abundance that could be related to temporal patterns in clutch size or hatch success at the Rancho Nuevo nesting beach resulting from variability in nesting female fecundity and the remigration interval (Metz 2004). However, there was no significant relationship between Kemp's ridley CPUE in nearshore Texas and Louisiana waters and the number of hatchlings leaving the nesting beaches at Rancho Nuevo. In fact, juvenile ridley CPUE remained relatively constant or decreased slightly, even as the number of hatchlings released from Rancho Nuevo increased exponentially. Assuming that post-hatchling mortality rates did not increase during the study period, juvenile Kemp's ridleys may have been recruiting to coastal locations outside of the northwestern Gulf of Mexico study areas (Metz 2004). The annual declines in strandings in Texas since 1994, along with the subsequent increases in Florida strandings since 1995, suggested that a shift in Kemp's ridley distribution from the western to the eastern Gulf of Mexico may have occurred in the mid-1990s, which could have been related to fluctuations in circulation patterns. Significant declines in turtle CPUE at Sabine Pass since 1997 coincided with a concurrent reduction in blue crab size; however, a similar trend was not seen at Calcasieu Pass. When evaluating various biological and abiotic factors, nesting dynamics and prey availability appeared to have had the most influence on the nearshore occurrence of Kemp's ridleys (Metz 2004).

Hook-and-line captures data, stranding and nesting records, satellite telemetry, and diet analyses were used to characterize Kemp's ridley population dynamics and movements along the Texas coast from 2003 through 2007 (Seney 2008). The results of the analyses confirmed that Kemp's ridleys use the upper Texas coast and northwestern Gulf of Mexico throughout their life and that the region was used seasonally as developmental and nesting habitat, as well as a migration and foraging corridor. Recreational hook-and-line captures, which did not include oceanic juveniles or adults, made up about one-third of non-nesting encounters along the coasts of Galveston and Jefferson counties in Texas. Juveniles demonstrated a preference for habitat type or benthic prey concentrations, rather than specific locations, in the northwestern Gulf and were found in nearshore waters along the upper Texas coast primarily during the warmer months (March through October). They also entered inshore areas, such as bays and coastal lakes, along the Texas and Louisiana coasts (Seney 2008). Adult females that nested along the upper Texas coast occupied the region during the nesting season (April through July). Juvenile and internesting adults occurred in relatively shallow Texas state waters, and post-nesting females subsequently migrated through deeper, federal waters (Seney 2008).

In a related satellite telemetry study, the inshore and continental shelf waters of the northwestern Gulf of Mexico were shown to serve as developmental and migratory habitat for the Kemp's ridley sea turtle (Seney and Landry 2011). Fifteen juveniles were fitted with transmitters and released off the upper Texas coast from 2004 through 2007. Their movements were restricted to the continental shelf from Matagorda Bay, Texas, east to waters offshore Timbalier Bay, Louisiana, and during most or all of the tracking period, the juveniles remained primarily in waters less than 5 m (16.4 ft) deep (Seney and Landry 2011). While movement patterns varied among years, the juvenile Kemp's ridleys were tracked primarily during the warmer months and preferred tidal passes, bays, coastal lakes, and nearshore waters. In addition, this investigation suggested that the preferred habitat of juvenile Kemp's ridleys may differ among years and could be related to the locations and abundances of specific prey items (Seney and Landry 2011).

The movements of juvenile Kemp's ridleys in an understudied region of the Kemp's ridley range—the north-central Gulf of Mexico—were studied by satellite tracking 12 turtles that were captured incidentally by recreational fishermen on piers or stranded live in Mississippi and Alabama, and rehabilitated and released (Lyn et al. 2012). Six turtles were released in Mississippi waters, 3.2 km (2 mi) south of East Ship Island, in November 2010, and six were released near documented feeding grounds off the Cedar Keys in Florida in April 2011. The turtles released in Mississippi migrated to warmer waters offshore (when the water temperatures decreased) and stayed in the general area of Mississippi Sound and adjacent Louisiana waters. However, within days of being released, most of the turtles released in Florida quickly began swimming up the coastline toward Alabama and Mississippi. One of the turtles released in Florida, a newly mature male, was tracked all the way to Rancho Nuevo, Mexico; it remained in this area for 2 weeks in March before returning north to waters along the Texas/Louisiana border. The results of this study indicated that releasing turtles near their hooking/stranding location is preferred over releasing them in known feeding grounds (Lyn et al. 2012).

A number of in-water tagging studies have characterized sea turtle distribution, abundance, use, and ecology in nearshore waters along the Florida Gulf coast. Nearshore waters that have been studied include Apalachee Bay (Rudloe et al. 1991; Campbell 1996), Deadman Bay (Barichivich 2006), Cedar Keys/Waccasassa Bay (Schmid and Ogren 1990; Schmid 1998; Schmid et al. 2002, 2003), Tampa Bay (Nelson 2000), Charlotte Harbor Estuary (Schmid 2011), and Ten Thousand Islands/Gullivan Bay (Witzell and Schmid 2004, 2005). Details of these investigations are presented in the following paragraphs. However, information gaps still exist since the Florida Gulf coast is extensive, and long-term, in-water studies are needed to monitor the status of juvenile sea turtles at key foraging areas (Eaton et al. 2008).

Rudloe et al. (1991) conducted a tagging study of post-oceanic juvenile Kemp's ridleys that were incidentally captured during shrimp trawling, gill netting, or fish seining in the coastal waters of the northeastern Gulf of Mexico, including Apalachee, Levy, and Dickerson bays, from 1984 through 1988. A total of 106 turtles, ranging in size from 20.3 to 57.9 cm (8 to 22.8 in) SCL (mean 36.7 cm [14.4 in] SCL), were collected over a 97 km (60 mi) stretch of the Florida Gulf coast from Shell Point, Wakulla County to St. George Island, Franklin County. While turtles were collected every month, the highest numbers of turtles were collected during May and December. Turtles obtained during December, January, and February were significantly larger than those collected in June, July, August, and September (means of 40.4 cm [15.9 in] SCL and 30 cm [11.8 in] SCL, respectively). Kemp's ridleys were collected from seagrass, sand, and mud bottom substrates at depths ranging from 0.3 m (1 ft) inshore to 32 m (105 ft) 9.7 km (6 mi) offshore, with the smallest turtles collected from depths of less than 2 m (6.5 ft). Since turtles were only recaptured within a season, the results of the study indicated a transitory Kemp's ridley population along the northwest Florida Panhandle coast. In a netting study conducted from August 1995 through July 1997, Apalachee Bay was shown to be an important developmental habitat for juvenile Kemp's ridleys; the average size of captured turtles was 34 cm (13.4 in) SCL (Campbell 1996).

The importance of Waccasassa Bay, located along the Florida Gulf coast near the Cedar Keys, as developmental habitat for neritic juvenile Kemp's ridleys was demonstrated in an investigation conducted from June 1986 through October 1995 (Schmid 1998). Turtles were captured with large-mesh tangle nets from April to November when water temperatures were greater than 20 °C and near the oyster bars of Corrigan Reef. CMR data indicated that some Kemp's ridleys remained in the vicinity of Corrigan Reef during their seasonal occurrence and returned each year, and a mean annual population size of 159 turtles, with high rates of immigration and emigration, was estimated for this area. Captured Kemp's ridleys at Corrigan Reef (253 turtles) ranged in size from 26.8 to 58.6 cm (10.6 to 23.1 in) SCL and averaged 44.5 cm (17.5 in) SCL. Turtles captured during the summer were significantly larger than those captured during the fall (means of 45.5 cm [17.9 in] SCL and 43.1 cm [17 in] SCL, respectively). With the exception of 1991 when most turtles were in the 30–40 cm (11.8–15.7 in) SCL size class, the 40–50 cm (15.7–19.7 in) SCL size class dominated the catch. This investigation, as well as earlier work (Carr and Caldwell 1956; Schmid and Ogren 1990), confirmed the occurrence of a seasonal, resident population of juvenile Kemp's ridleys in Waccasassa Bay.

In a related investigation, juvenile Kemp's ridleys were studied in Waccasassa Bay from May through August 1994 and May through November 1995 (Schmid et al. 2002, 2003). Turtles occupied foraging areas ranging from 5 to 30 km^2 (1.9 to 11.6 mi^2) in size, and they used rock outcroppings more than expected (Schmid et al. 2003). In addition, live bottom and green macroalgae habitats were used more than seagrass habitats. Turtles increased their rate of movement with the increasing velocity of the tide (Schmid et al. 2002). The rates of Kemp's ridley movement were higher and surface and submergence durations were shorter during the day; the turtles' daily activities were attributed to food acquisition and bioenergetics.

Juvenile Kemp's ridleys inhabiting the nearshore waters of the northeastern Gulf of Mexico, specifically from Apalachee Bay to Suwannee Sound, including Deadman Bay, Florida, were studied from 1995 through 1999 (Barichivich 2006). The majority of the turtles captured were from 20 to 40 cm (7.9 to 15.7 in) SCL, and most were captured in Deadman Bay (121 of 126 turtles). While fewer captures were made during the cooler months, Kemp's ridleys were captured from March through December, and turtles were captured when water temperatures were between 19.7 and 34 °C. Annual growth rates ranged from 1.25 to 8.92 cm (0.49 to 3.51 in). The large number of short-term recaptures, as well as recaptures between seasons, indicated that Deadman Bay is an important developmental habitat for Kemp's ridleys (Barichivich 2006).

To assess sea turtle occurrence relative to the channel bottom, trawl surveys were conducted within the Tampa Bay Entrance Channel during the spring, summer, and fall of 1997, as well as in the spring of 1998, by the USACE (Nelson 2000). During the surveys, two juvenile Kemp's ridleys were captured and tracked. These turtles remained in the study area for days or months at a time and eventually moved in response to changing water temperature. They either moved offshore or southward as water temperatures decreased, and when water temperatures warmed, they returned to their original location. The results of the surveys indicated that dredging activities in the Tampa Bay Entrance Channel should be conducted during extremes in water temperature during either the winter or summer (Nelson 2000).

In a study conducted from 1997 through 2004, the nearshore waters of Gullivan Bay, in the Ten Thousand Islands area off the southwest Florida Gulf coast, were determined to be important developmental habitat for juvenile Kemp's ridleys (Witzell and Schmid 2004, 2005; Eaton et al. 2008). More than 190 Kemp's ridleys, ranging in size from 21.4 to 65.2 cm (8.43 to

25.7 in) SCL and averaging 40.4 cm (15.9 in) SCL, were captured. Kemp's ridley recaptures were documented within and between sampling seasons, indicating foraging-site fidelity. Some turtles set up home ranges in this area for as long as 3 years. Kemp's ridleys preferred areas of sand substrate with plumed worm tubes and live-bottom organisms (Witzell and Schmid 2004, 2005).

From August 2009 through April 2011, monthly in-water surveys of the southeastern portion of Pine Island Sound, part of the Charlotte Harbor estuary complex on the Gulf coast in Lee County, Florida, containing live bottom habitat were conducted. These in-water surveys were a continuation of earlier tagging studies conducted in the area (Schmid 2011). Almost 70 % of the sea turtles observed during the study were Kemp's ridleys. There were 50 sightings of Kemp's ridleys, and 45 Kemp's ridleys were captured and tagged. Similar to other nearshore areas of the Florida Gulf coast, most of the Kemp's ridleys were juveniles; captured turtles ranged in size from 24.2 to 62.7 cm (9.5 to 24.7 in) minimum SCL and averaged 40.9 cm (16.1 in) minimum SCL in size (Schmid 2011). CMR data indicated both within- and between-seasons fidelity to the study area. A satellite-tracked juvenile turtle demonstrated seasonal fidelity to the area by leaving Charlotte Harbor in late fall, heading south and wintering off the Florida and Marquesas keys, and returning to within a few kilometers of its capture site in early spring (Schmid 2011). An adult-sized Kemp's ridley was also tracked by satellite; it appeared to be a transient inhabitant in the area, immediately leaving Pine Island Sound after release and moving northward to a feeding area offshore from Homosassa Bay. The results of the study reinforced the importance of Charlotte Harbor Estuary, particularly Pine Island Sound, as Kemp's ridley developmental habitat (Schmid 2011).

Most of the diet of neritic juvenile Kemp's ridley sea turtles consists of crabs (Table 11.1). However, as indicated by the lists of food items for neritic juveniles from different locations included in Table 11.1, Kemp's ridleys appear to be opportunistic foragers and readily utilize prey in a particular area. Since neritic juvenile Kemp's ridley sea turtles continue the pattern of seasonal migrations and fidelity to foraging sites for many years until maturing and moving to adult foraging areas, both the nearshore foraging grounds discussed in the paragraphs above and the offshore overwintering areas in the Gulf of Mexico are important for this life stage of Kemp's ridleys (NMFS et al. 2011).

11.2.1.4 Adult Life History and Distribution for Gulf of Mexico Kemp's Ridleys

Adult Kemp's ridley sea turtles occur primarily in nearshore waters of the Gulf of Mexico. They are occasionally found in the coastal regions of the southeast U.S. Atlantic Coast and are rare off the northeast U.S. Atlantic Coast (Hildebrand 1982; Ogren 1989; USFWS and NMFS 1992; TEWG 2000; Pritchard 2007a). Important foraging areas where adult females reside seasonally consist of biologically productive locations in the waters off the western and northern Yucatán Peninsula, including the Laguna del Carmen area off Campeche, Mexico, and the northern Gulf of Mexico from southern Texas to western Florida, such as along the Louisiana coast near the mouth of the Mississippi River (Chávez 1968; Pritchard and Márquez-M. 1973; Guzmán-Hernández et al. 2007; Shaver and Rubio 2008). While there is some geographic variation, crabs (especially portunid crabs) are the primary prey of adult Kemp's ridleys (Table 11.1).

Early on Kemp's ridley adult migration was thought to occur within the continental shelf waters of the Gulf of Mexico, mainly between nesting sites in Tamaulipas, Mexico, and coastal feeding grounds (Morreale et al. 2007). In fact, adult female Kemp's ridleys have been tracked from foraging grounds in Louisiana and Texas to the Rancho Nuevo nesting beach (Renaud et al. 1996). This pattern of moderate-distance migration was consistent with the observed high frequency of annual return of nesting females to their main nesting beach in Rancho Nuevo, Mexico, as well as with their strategy of feeding primarily on benthic crustaceans in coastal waters (Morreale et al. 2007). Satellite tracking studies have indicated that post-nesting female Kemp's ridleys travel along coastal corridors (typically shallower than 50 m [164 ft]) along the rim of the Gulf of Mexico basin, extending from the Yucatán Peninsula to southern Florida (Byles 1989; Byles and Plotkin 1994; Renaud 1995; Renaud et al. 1996; Shaver 1999, 2001a, b; Morreale et al. 2007). Many of these turtles settled in resident feeding areas for up to several months after migrating, which demonstrated that Kemp's ridley post-nesting migrations could also be considered foraging migrations to fixed destinations (Byles and Plotkin 1994).

Between 1997 and 2006, the movements of 28 turtles (17 wild-stock and 11 head-started) that nested on North Padre Island or Mustang Island, Texas, were monitored using satellite telemetry to obtain habitat use and movement information for Kemp's ridleys that nested in south Texas (Shaver and Rubio 2008). Internesting residency was documented off south Texas, and post-nesting residency occurred in Gulf of Mexico waters from south Texas to the southern tip of Florida. After nesting for the season was complete, most of the Kemp's ridleys left south Texas and traveled, parallel to the coastline, to the northern or eastern Gulf of Mexico. This study demonstrated the importance of nearshore Gulf waters to post-nesting Kemp's ridleys, and the results were used to develop a regulation to close nearshore south Texas waters seasonally to shrimp trawling (Shaver and Rubio 2008).

The inshore and continental shelf waters of the northwestern Gulf of Mexico along the upper Texas coast were recently demonstrated to serve as migratory, internesting, and post-nesting habitat for adult Kemp's ridleys (Seney and Landry 2008, 2011). Six female Kemp's ridleys were fitted with satellite transmitters after nesting and tracked during 2005 and 2006 (Seney and Landry 2008). In a second investigation, seven adult females (six nesting and one trawl-caught) were fitted with transmitters and released off the upper Texas coast during 2004 through 2007 to characterize their movements, migration patterns, and foraging grounds in the northwestern Gulf of Mexico (Seney and Landry 2011). During both tracking studies, the females remained in the Galveston region and in the vicinity of the upper Texas coast during their internesting intervals and moved eastward along the continental shelf (20 m [66 ft] isobath) to offshore foraging areas of central Louisiana upon entering the post-nesting stage (Seney and Landry 2008, 2011).

The importance of nearshore Gulf of Mexico waters, specifically off the Louisiana coast, as critical foraging habitat for post-nesting Kemp's ridleys was recently demonstrated by Shaver et al. (2013). Satellite telemetry and switching state-space modeling was used to track 31 turtles after nesting at PAIS and Rancho Nuevo for 13 years from 1998 through 2011. Multiple turtles foraged along their migratory route before arriving at their final foraging sites. Nearshore Gulf of Mexico waters served as foraging habitat for all turtles tracked in the study, and final foraging sites were located in water less than 68 m (223 ft) deep and a mean distance of 33.2 km (20.6 mi) from the nearest mainland coast. The wide distribution of foraging sites indicates that a foraging corridor exists for Kemp's ridleys in the Gulf of Mexico (Shaver et al. 2013).

In the spring, mature female Kemp's ridleys undertake annual migrations to the western Gulf of Mexico and gather along the coast of Tamaulipas near the village of Rancho Nuevo to nest on the many kilometers of almost continuous sand beach (Pritchard 2007a, b). Reproductive females begin to arrive offshore Rancho Nuevo in March and April, with most arriving during May and June, and remain in the vicinity through the nesting season (Table 11.1) (Rostal 1991; Seney 2008). About a month before the nesting season begins, females and males aggregate to mate in nearshore waters near the beach at Rancho Nuevo (Pritchard 1969; Mendonca and Pritchard 1986).

In contrast to the pattern of female post-nesting migration, many adult male Kemp's ridleys remain in the vicinity of the nesting beach throughout the year (Shaver et al. 2005; Shaver 2006a). Shaver et al. (2005) monitored the movements of 11 adult male Kemp's ridleys captured near Rancho Nuevo, Mexico, using satellite telemetry, and while one traveled north and was last located offshore Galveston, Texas, ten remained in the vicinity of the nesting beach. As indicated by mating activities for the Kemp's ridley, which are more widespread than the nesting areas and occur in coastal and inshore waters from south Texas to Veracruz, Mexico, males that do not reside near the nesting beaches throughout the year mate with females in foraging areas or migration pathways (Shaver 1992; Morreale et al. 2007).

11.2.2 Kemp's Ridley Recovery Program

As mentioned previously, through a variety of intervention methods by Mexico and the United States that began almost 50 years ago, the Kemp's ridley has made a remarkable comeback from the brink of extinction (Márquez-M. et al., 1998, 1999; TEWG 1998; Heppell et al. 2007; Crowder and Heppell 2011). Egg protection efforts began at Rancho Nuevo in 1966 and were expanded in 1976, and a binational recovery plan was developed in 1977. Community involvement and education, which have changed attitudes regarding Kemp's ridley conservation, have been important elements of the recovery program since its implementation (Heppell et al. 2007). While no method exists to quantitatively assess the relative impact of all of the methods implemented, which is a critical issue, it is clear that conservation efforts have resulted in increased survival rates for all life stages of the Kemp's ridley (Heppell et al. 2007; Plotkin 2007b).

The Kemp's Ridley Recovery Program began in 1978, and 100 % of the nests in the Rancho Nuevo area began to be relocated to fenced corrals; these hatcheries eliminated land-based predation and human egg collection from the Kemp's ridley life cycle to ensure high egg survival rates (Cornelius et al. 2007). Over the years, nest protection efforts have expanded north and south of Rancho Nuevo, and additional corrals have been constructed. After the hatchlings emerge from the nests in these hatcheries, they are counted and released in large groups directly into the water at different locations along the beach (NMFS et al. 2011).

Currently, nests continue to be relocated to fenced corrals; however, a large number of nests are now left in situ. Many of these nests are covered to protect against predation, as was done in a 2007 arribada (NMFS et al. 2011). The slow and steady increase in the nesting population at Rancho Nuevo and the increase in numbers of females nesting in Texas have been aided by protective egg hatcheries in both Mexico and Texas (Márquez-M. et al. 1996; Heppell 1997; Shaver and Caillouet 1998).

At the same time, measures to reduce the at-sea, incidental mortality of juvenile and adult Kemp's ridleys resulting from shrimp trawling and other fishing operations began to be implemented (Heppell et al. 2007). In 1980, U.S. shrimp trawlers were excluded from Mexican waters; sailing and fishing within 6.44 km (4 mi) of the Rancho Nuevo beach was prohibited starting in the late 1980s; and by 1990, the sea turtle product trade was banned in Mexico, and TEDs were required in U.S. waters (NMFS et al. 2011). In the mid- and late-1990s, TEDs began to be required in Mexico, the fishing effort off the main nesting beaches was reduced, and a closure of the Mexican shrimping season during the primary ridley nesting period began. In 2000, an annual closure of shrimp trawling in Gulf of Mexico waters off North Padre Island, South Padre Island, and Boca Chica Beach began. Kemp's ridley feeding habitat off South Padre Island, Texas, was protected in 2002 (Márquez-M. et al., 1998, 1999; TEWG 1998; Shaver 2005, 2006b; Heppell et al. 2007).

In addition, the Kemp's Ridley Sea Turtle Head Start Experiment began in 1978, with a goal of establishing a second nesting population at PAIS in Texas where sporadic Kemp's ridley nesting had been documented in the past (Fontaine and Shaver 2005). To establish a nesting beach at PAIS, a small fraction (average of 2.8 %) of the total eggs from Rancho Nuevo was translocated for incubation, hatching, and experimental imprinting from 1978 through 1988 (Caillouet 1995). During the 23 years of the Head Start Experiment—1978 to 2000—more than 23,000 Kemp's ridley hatchlings were raised in captivity for approximately 1 year at the NMFS Laboratory in Galveston, Texas (see below; Fontaine and Shaver 2005; Shaver and Wibbels 2007). This prevented the high level of predation associated with the post-hatchling life stage in the wild, as these larger turtles would be less susceptible to predators upon their release (Shaver and Wibbels 2007). In case of extinction in the wild, a few hundred of the head-started ridleys were retained in captivity in various locations.

The majority of hatchlings from 1978 through 1988 were obtained from eggs that had been transferred from Rancho Nuevo to PAIS for experimental imprinting. Several hundred hatchlings from the 1978, 1979, 1980, and 1983 year classes were obtained directly from Rancho Nuevo (Fontaine and Shaver 2005; Shaver and Wibbels 2007). The eggs obtained from Rancho Nuevo from 1978 through 1988 (22,507 total eggs) were incubated in PAIS sand, and the hatchlings were released on the beach and allowed to swim briefly in the Gulf of Mexico (Shaver and Wibbels 2007). After the hatchlings swam approximately 5–10 m (16–33 ft), they were captured by dip net and shipped to the NMFS Laboratory in Galveston for head-starting. Approximately 77 % of the eggs incubated at PAIS produced hatchlings (15,875 total hatchlings) that were transferred to the Galveston Laboratory from 1978 through 1988 (Shaver 2005). All hatchlings reared in the Head Start Experiment were obtained directly from Rancho Nuevo from 1989 through 2000. Because of lower than pivotal incubation temperatures, males likely predominated in the 1978, 1979, 1981, and 1983 year classes. Conversely, about 78 % of the 1985 through 1988 year classes were estimated to be females given the above pivotal temperature used to incubate the eggs (overall, approximately 60 % of the 1978 through 1988 year classes were females), and over 90 % of the head-started turtles from the 1989 through 2000 year classes were likely females (Fontaine and Shaver 2005; Shaver and Wibbels 2007).

All released turtles were tagged using a variety of methods: external metal tags, *living tags* (a disk of light living tissue from the plastron transplanted to the dark carapace), internal magnetized wire tags, and passive integrated transponder tags (Fontaine and Shaver 2005; Shaver and Wibbels 2007). Except for 1983, and from 1993 through 2000, approximately 1,000–2,000 turtles were raised in captivity each year (Shaver and Wibbels 2007). Over 23 years, 27,137 Kemp's ridley hatchlings were transported to the NMFS Galveston Laboratory from either PAIS or directly from Rancho Nuevo; of those, about 88 % (23,473 hatchlings, which included 13,275 imprinted at PAIS and 10,198 imprinted at Rancho Nuevo) were successfully reared, tagged, and released into the Gulf of Mexico at sizes comparable to the juvenile oceanic or neritic stage of wild-stock Kemp's ridleys (Fontaine and Shaver 2005; Shaver and Wibbels 2007).

Head-started Kemp's ridleys, which typically weighed approximately 1 kg (2.2 lb), were released throughout the Gulf of Mexico in a variety of locations that represented habitats appropriate for late oceanic or neritic juvenile ridleys (Shaver and Wibbels 2007). In 1978 and 1979, most turtles were released off the Florida Gulf coast. However, there was concern that turtles were leaving the Gulf of Mexico and might not return to breed, since many of the released turtles were later recaptured along the Atlantic coast (Shaver and Wibbels 2007). Therefore, from 1980 through 2000, ridleys were released in the western Gulf of Mexico, primarily off the Texas coast in the waters off Padre Island.

While the site of release affected the movements and distributions of head-started turtles, overall, head-started Kemp's ridleys dispersed widely from release areas and were reported throughout the natural range and in habitats of wild-stock Kemp's ridleys (Shaver and Wibbels 2007). Growth and diet information obtained from recaptured and stranded turtles indicated that head-started ridleys adapted to feeding in the wild. Mortality rates were likely high for both wild-stock and head-started ridleys before the mandatory use of TEDs began in U.S. offshore waters in 1990, as indicated by the continued decline of Kemp's ridleys throughout the 1980s and early 1990s. In addition, because Kemp's ridley mortality was so high during the 1980s, survival to adulthood was unlikely for most turtles (Shaver and Wibbels 2007).

Although factors, such as insufficient monitoring of the nesting beaches and turtle tag loss, have affected the collection of head-started Kemp's ridley nesting data, some head-started Kemp's ridleys have nested on beaches of PAIS, North Padre Island, Mustang Island, Galveston Island, and Bolivar Peninsula in Texas, as well as in Tamaulipas, Mexico (Shaver and Wibbels 2007). The first head-started Kemp's ridley nests, laid by turtles from the 1983 and 1986 year classes, were documented at PAIS and Boca Chica Beach, Texas in 1996 (Shaver 1996; Fontaine and Shaver 2005; Shaver and Wibbels 2007). The Kemp's ridleys that have nested in Texas since 1996 have been a mixture of head-started and wild-stock turtles. However, since 2002, the majority of nesting in Texas has been by wild-stock Kemp's ridleys (NMFS et al. 2011). For example, of the 42 Kemp's ridley nests documented in Texas in 2004, 13 were from head-started turtles (Safina and Wallace 2010). Tracking studies have demonstrated that nesting head-started Kemp's ridleys have similar post-nesting movement behaviors as wild-stock ridleys (Landry and Seney 2006; Shaver and Wibbels 2007).

Because the Head Start Experiment was a large-scale experiment on the most endangered sea turtle in the world and was the subject of intense debate and controversy throughout the 1980s and early 1990s (Shaver and Wibbels 2007), it was officially terminated in 1993 but continued on a limited basis through 2000. While it is encouraging that some head-started Kemp's ridleys have survived to maturity and are nesting, it is not possible to accurately determine the effectiveness or ineffectiveness of the experiment (Shaver and Wibbels 2007). However, head-started females have nested and viable second-generation hatchlings have been produced both at PAIS and at Rancho Nuevo. In addition, a large amount of biological information was obtained as a result of the Head Start Experiment that might otherwise have never been discovered (Fontaine and Shaver 2005).

11.3 LOGGERHEAD SEA TURTLE (*CARETTA CARETTA*)

Loggerhead sea turtles are named for their large heads (Figures 11.14 and 11.15). The adults are slightly larger than hawksbills but slightly smaller than green sea turtles (Witherington et al. 2006a). Of all species of sea turtles, the life history of the loggerhead is probably the best understood (Bolten and Witherington 2003; Witherington et al. 2012). This is due to the concern regarding their decline because of the incidental capture in commercial fisheries (e.g., trawling, driftnet, longline) as well as the loss of nesting habitat caused by coastal development. In addition, loggerheads nest in areas with major conservation programs; for example, loggerheads nest on the beaches of the Archie Carr National Wildlife Refuge (ACNWR) in Florida (Figure 11.14), the most important nesting beach of loggerheads in the Western Hemisphere, and the location of a long-term research program. Compared to the other sea turtle species, the loggerhead sea turtle has the largest geographic nesting range, which includes both temperate and tropical latitudes. It is globally distributed in all temperate and tropical ocean basins, and its diet is the least specialized (Bolten and Witherington 2003).

Figure 11.14. Nesting loggerhead sea turtle on the beach at Archie Carr National Wildlife Refuge, Indian River and Brevard Counties, Florida, June 2012 (photograph by Steve A. Johnson, University of Florida, with permission).

Figure 11.15. Juvenile loggerhead sea turtle in the water (photograph by Marco Giuliano, Fondazione Cetecea) (from NOAA 2011).

11.3.1 Loggerhead Sea Turtle Life History, Distribution, and Abundance

The loggerhead sea turtle occurs throughout the temperate, subtropical, and tropical regions of the Atlantic, Pacific, and Indian oceans, and its range includes foraging areas, migration corridors, and nesting beaches (Figure 11.16) (Dodd 1988). Unlike most other sea turtle species, the loggerhead is less abundant in the tropics than it is in temperate waters, and most of its nesting beaches are located outside of the tropics (Witherington et al. 2006a). The majority of loggerhead nesting is located at the western rims of the Atlantic and Indian oceans, and only two loggerhead nesting aggregations have greater than 10,000 females nesting per year: Peninsular Florida in the United States and Masirah Island, Oman (Conant et al. 2009). Loggerhead nesting aggregations with 1,000–9,999 females nesting annually occur in Georgia through North Carolina in the United States, Quintana Roo and Yucatán in Mexico, Brazil, Cape Verde Islands, western Australia, and Japan (Márquez-M. 1990; Ehrhart et al. 2003; Conant et al. 2009). Aggregations with 100–999 females nesting annually occur in the northern Gulf of Mexico (USA), Dry Tortugas (USA), Cay Sal Bank (Bahamas), Tongaland (South Africa), Mozambique, Arabian Sea coast (Oman), Halaniyat Islands (Oman), Cyprus, Peloponnesus (Greece), Zakynthos (Greece), Crete (Greece), Turkey, and Queensland (Australia) (Conant et al. 2009).

Following are the five currently recognized life stages for the loggerhead sea turtle (TEWG 2009):

1. Year One: terrestrial zone to oceanic zone, size less than or equal to 15 cm (5.9 in) SCL.
2. Juvenile, Stage I: exclusively oceanic zone, size range of 15–63 cm (5.9–24.8 in) SCL.
3. Juvenile, Stage II: oceanic or neritic zones, size range of 41–82 cm (16.1–32.3 in) SCL.
4. Juvenile, Stage III: oceanic or neritic zones, size range of 63–100 cm (24.8–39.4 in) SCL.
5. Adult: neritic or oceanic zones, size greater than or equal to 82 cm (32.3 in) SCL.

Loggerhead sea turtles are represented by many distinct populations (Figure 11.6) (USFWS and NMFS 2011). Because the focus of this chapter is the Gulf of Mexico sea turtles, the

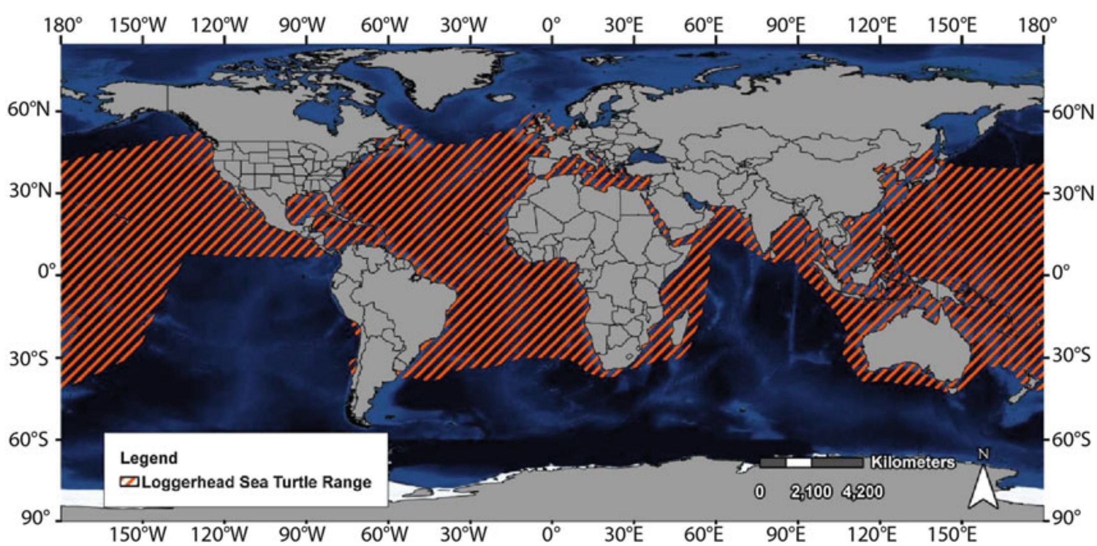

Figure 11.16. Range of the loggerhead sea turtle (from NOAA 2009a).

sections that follow focus on loggerheads that occur in the Gulf of Mexico during some portion of their life cycle. The Northwest Atlantic Ocean DPS of loggerhead sea turtles uses the Gulf of Mexico's beaches for nesting, oceanic currents for developmental habitat, and/or neritic and oceanic areas for foraging, resting, or migrating.

The Northwest Atlantic Ocean DPS of loggerhead sea turtles includes the following subpopulations (NMFS and USFWS 2008; USFWS and NMFS 2011):

- Northern Subpopulation: southern Virginia to Florida/Georgia border (rarely utilizes the Gulf of Mexico (Heppell et al. 2003)).
- Peninsular Florida Subpopulation: Florida/Georgia border south through Pinellas County, excluding the islands of Key West, Florida.
- Northern Gulf of Mexico Subpopulation: Franklin County, Florida, west through Texas.
- Dry Tortugas Subpopulation: islands west of Key West, Florida.
- Greater Caribbean Subpopulation: Mexico through French Guiana, Bahamas, Lesser and Greater Antilles.

Using nesting data from 2001 through 2010, Richards et al. (2011) estimated the Northwest Atlantic Ocean DPS adult female loggerhead population to range from 30,096 to 51,211 turtles. Individual subpopulations were estimated as follows: 258–496 adult females for Dry Tortugas, 323–634 adult females for northern Gulf of Mexico, 1,975–4,232 adult females for Greater Caribbean, and 23,655–45,058 adult females for Peninsular Florida. Richards et al. (2011) remarked that improved estimates of clutch frequency and breeding intervals, as well as better measures of temporal and spatial variation, are needed to improve population estimates based on nest counts.

11.3.1.1 Nesting Life History, Distribution, and Abundance for Gulf of Mexico Loggerheads

The generalized locations of loggerhead nesting beaches for all of the loggerhead subpopulations of the Northwest Atlantic Ocean DPS are included in Figure 11.17. Life history information, including nesting information, for loggerheads is included in Table 11.2; available information for specific Gulf of Mexico beaches or regions is also included.

The Peninsular Florida subpopulation of loggerheads is the largest nesting aggregation in the Atlantic Ocean, representing about 80 % of all nesting and about 90 % of all hatchlings in this DPS (Ehrhart et al. 2003; TEWG 2009; Witherington et al. 2009). The greatest proportion of nesting for the Peninsular Florida subpopulation occurs on the Atlantic coast in six Florida counties (Brevard, Indian River, St. Lucie, Martin, Palm Beach, and Broward); however, thousands of nests are laid each year on southwest Florida Gulf coast beaches (Figure 11.17) (TEWG 2000, 2009; Ehrhart et al. 2003; Witherington et al. 2009). In the Gulf of Mexico, loggerheads typically nest on barrier island beaches with moderate to high wave energy. They also nest on low-relief mangrove islands, such as those located in the Ten Thousand Islands in southwest Florida (Foley et al. 2000).

In 1979, approximately 10,000 loggerhead nests, or about 1,900 nesting females, were counted on beaches surveyed in Peninsular Florida (these data include beaches on both the Atlantic and Gulf coasts; separating out data for the Gulf of Mexico was not possible). In 1998, loggerhead nesting in Peninsular Florida reached a high of almost 84,600, which equals about 15,700 nesting females (Figure 11.18). From 1979 through 2000, a general increasing trend in annual loggerhead nesting occurred in Peninsular Florida; but, in 2001, annual nest counts began a decreasing trend through 2009, with a low of 44,512 nests (or about 8,243 nesting

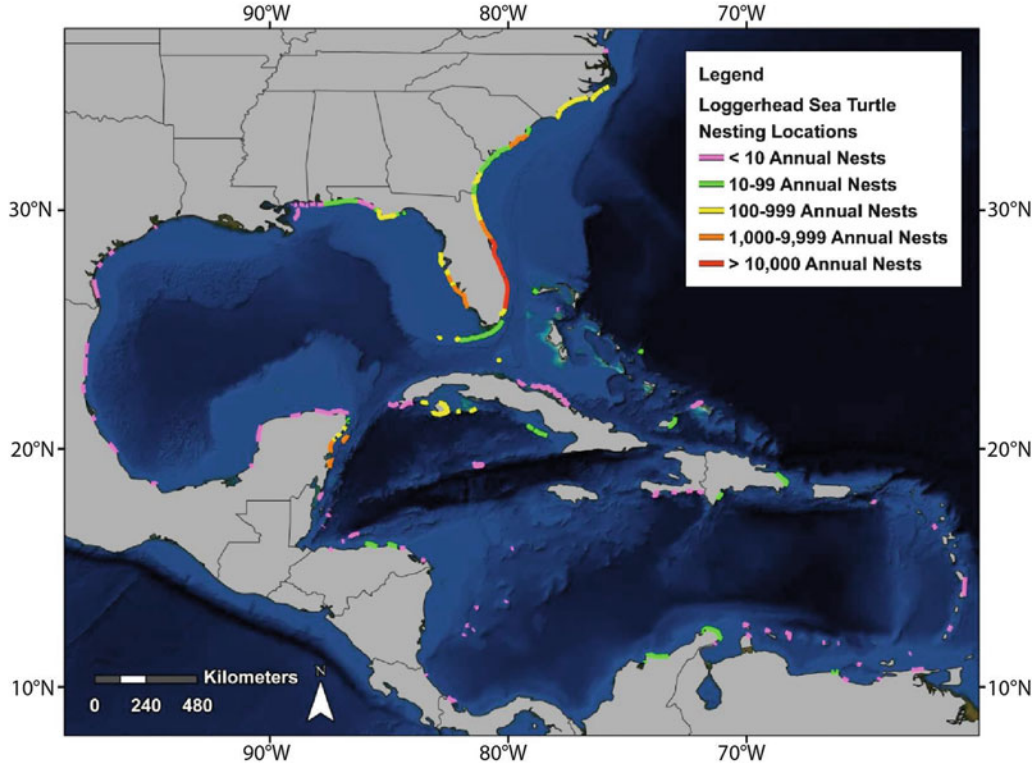

Figure 11.17. Generalized nesting beach locations of the Northwest Atlantic Ocean Distinct Population Segment of loggerhead sea turtles (interpreted from Dow et al. 2007; NMFS and USFWS 2008; SWOT 2007a).

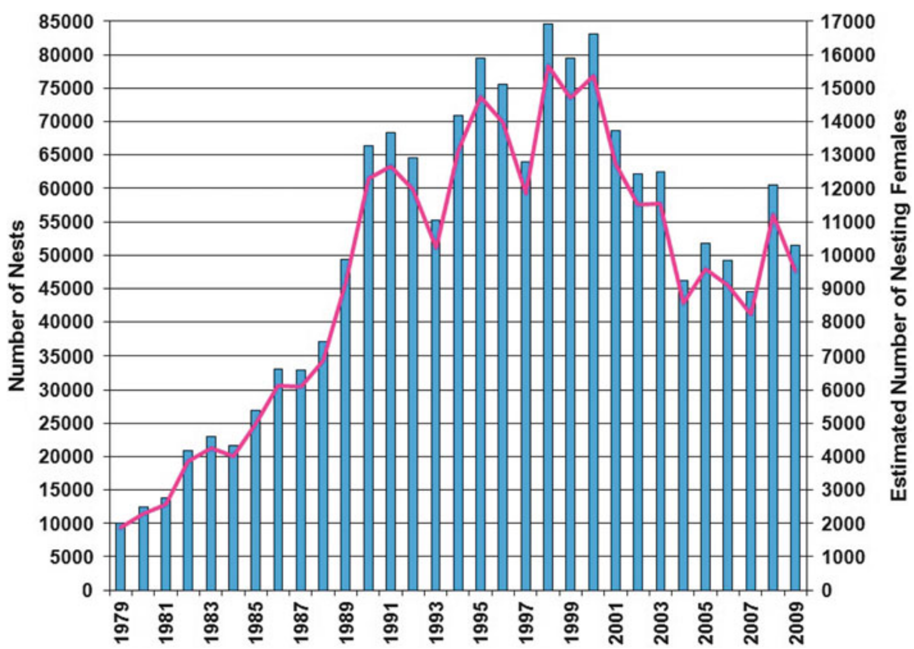

Figure 11.18. Annual number of nests (*bars*) and estimated number of nesting females (*line*), assuming 5.4 nests per female (Tucker 2010), for the Peninsular Florida subpopulation of loggerhead sea turtles from 1979 through 2009. Note that the survey effort was not consistent among years (from Meylan et al. 1995; NMFS and USFWS 2008; FFWCC FWRI 2012).

females) recorded for all surveyed Peninsular Florida beaches in 2007 (Figure 11.18). Results of a recent analysis of Florida Index Beach survey data, a subset of surveyed Florida beach data suitable for trend assessments because of consistent spatial and temporal nest counts (Witherington et al. 2009), for the Peninsular Florida subpopulation indicated a 26 % decrease in nesting from 1989 through 2008 and a 41 % decline since 1998 (NMFS and USFWS 2008). However, in 2010 and 2011, loggerhead nesting counts on all beaches surveyed in Peninsular Florida were back to numbers similar to those recorded in 2000 (73,066 nests and 67,701 nests, respectively), and a high of 97,000 nests, representing about 18,000 nesting females, was recorded in 2012 (FFWCC FWRI 2012).

The northern Gulf of Mexico subpopulation of loggerheads is one of the smallest nesting aggregations in the Atlantic and the second smallest in the western North Atlantic (TEWG 2009). The nesting beaches of this subpopulation are concentrated in the Florida Panhandle, with a consistent but small amount of nesting in other Gulf States, mostly Alabama and Texas (Figure 11.19). As part of this subpopulation, loggerhead sea turtles nest along Eglin Air Force Base on Cape San Blas and Santa Rosa Island. The number of nests laid at each location from 1994 through 1997 included the following: 53, 60, 25, and 54 nests on Cape San Blas and 32, 18, 28, and 22 nests on Santa Rosa Island (Lamont et al. 1998). Texas has almost 600 km (373 mi) of beach available to nesting sea turtles but, for unknown reasons, loggerheads do not nest regularly or in large numbers on Texas beaches (Plotkin 1989). Since 1994, annual nest counts for northern Gulf of Mexico loggerheads have consistently exceeded 600, or about 110 nesting females, with a high of 1,285 nests in 1999 and a low of 611 nests in 2007 (Figure 11.19). While it is difficult to evaluate long-term nesting trends for the northern Gulf of Mexico subpopulation

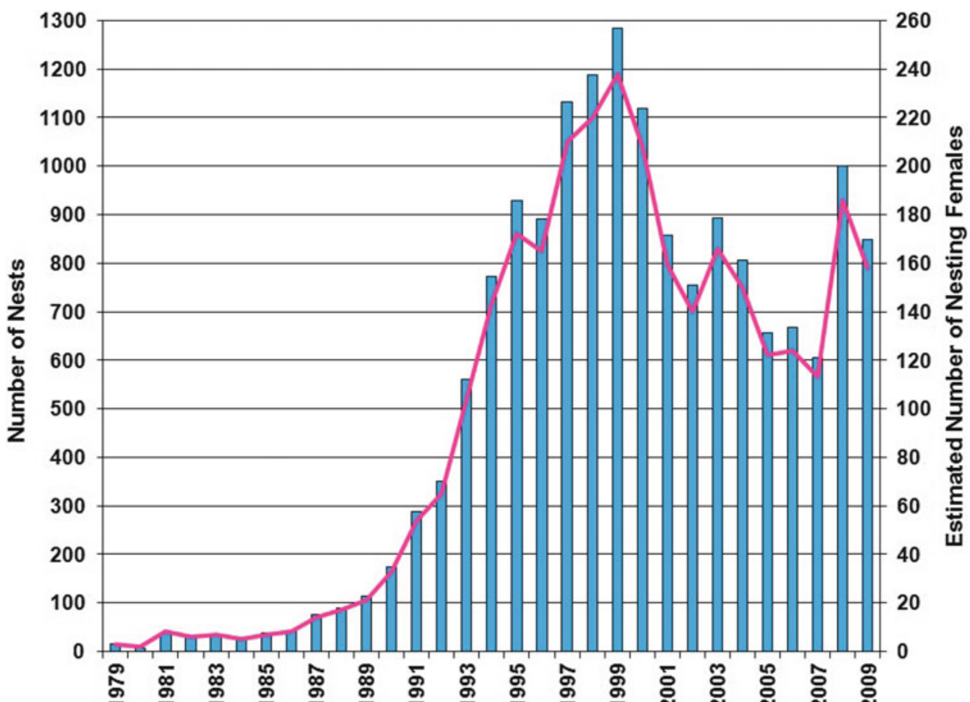

Figure 11.19. Annual number of nests (*bars*) and estimated number of nesting females (*line*), assuming 5.4 nests per female (Tucker 2010), for the northern Gulf of Mexico subpopulation of loggerhead sea turtles from 1979 through 2009. Note that the survey effort was not consistent among years (from Meylan et al. 1995; TEWG 2000; NMFS and USFWS 2008; Richards et al. 2011).

because of changed and expanded beach survey coverage, an analysis of 12 years (1995–2007) of Florida Index Beach survey data for this subpopulation indicated a significant declining trend of about 5 % per year (NMFS and USFWS 2008). In 2010 and 2011, 683 and 970 nests, respectively, were recorded for northern Gulf of Mexico loggerheads, and a high of 1,750 nests was recorded in 2012 (FFWCC FWRI 2012; Share the Beach 2013).

The Dry Tortugas subpopulation is the smallest loggerhead subpopulation of the Northwest Atlantic Ocean DPS (Figure 11.17) (TEWG 2009). This subpopulation is important not only for its genetic distinctiveness but also because it is isolated from many of the threats facing most sea turtle nesting areas (van Houtan and Pimm 2007). Loggerhead nesting activity is unevenly distributed among the seven islands within Dry Tortugas National Park (DTNP). About 90 % of loggerhead nests are laid on East Key and Loggerhead Key each year; the remaining 10 % of nesting activity occurs on Bush Key, Hospital Key, and Garden Key, and virtually no nesting occurs on Long Key and Middle Key (van Houtan and Pimm 2007). Annual nest counts for the Dry Tortugas subpopulation of loggerheads from 1995 through 2001 consistently exceeded 200 (about 40 nesting females) with a high of 340 nests recorded in 1995 (Figure 11.20). About 200 nests were also recorded for the Dry Tortugas in 2003 and 2009. A longer time series of nesting data for the Dry Tortugas subpopulation is necessary in order to detect a trend (NMFS and USFWS 2008).

The majority of nesting for the Greater Caribbean subpopulation of loggerhead sea turtles occurs in Quintana Roo, Mexico. The loggerhead nesting aggregation in Quintana Roo is the third largest in the western north Atlantic (Figure 11.17) (Ehrhart et al. 2003; TEWG 2009). Less frequent and scattered loggerhead nesting occurs along the Mexican Gulf coast from the

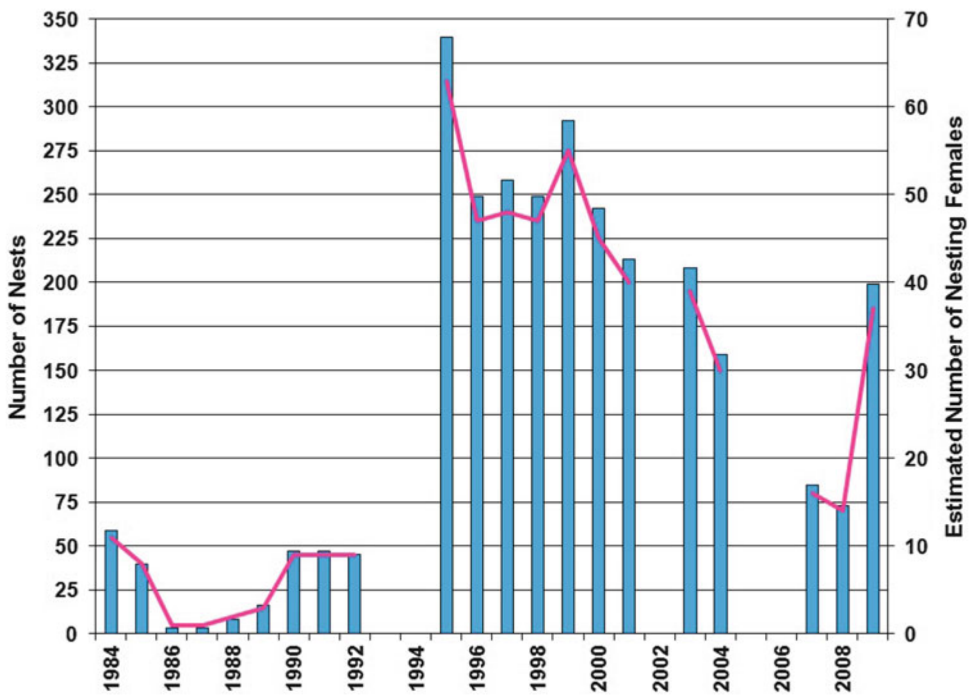

Figure 11.20. Annual number of nests (*bars*) and estimated number of nesting females (*line*), assuming 5.4 nests per female (Tucker 2010), for the Dry Tortugas subpopulation of loggerhead sea turtles from 1984 through 2009. Note that the survey effort was not consistent among years, and data were not available for 1993, 1994, 2002, 2005, or 2006 (from Meylan et al. 1995; NMFS and USFWS 2008; FFWCC FWRI 2012).

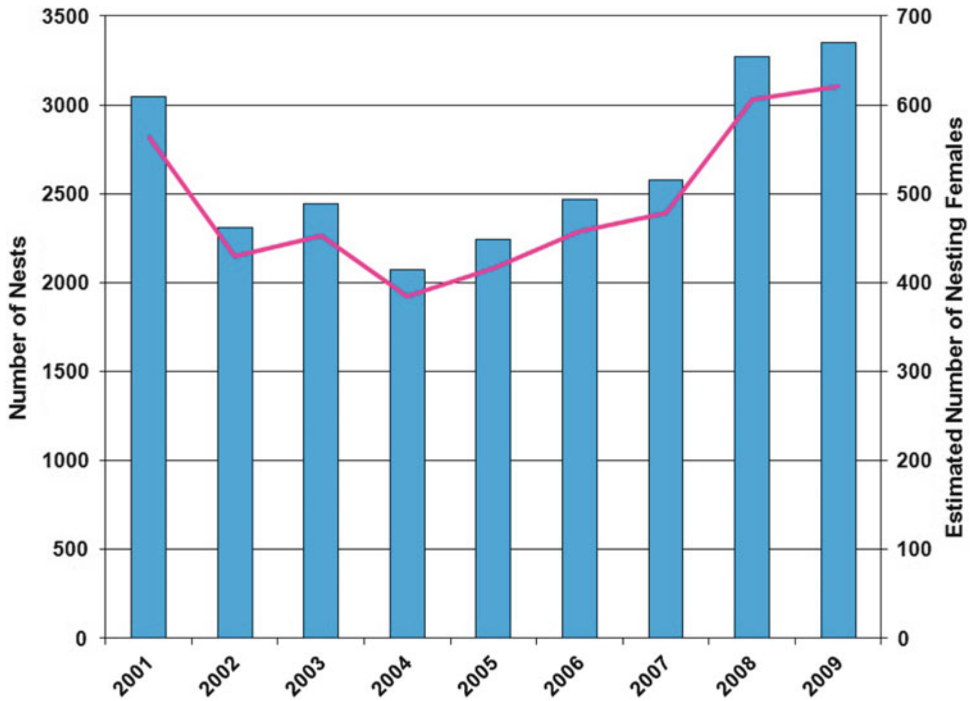

Figure 11.21. Annual number of nests (*bars*) and estimated number of nesting females (*line*), assuming 5.4 nests per female (Tucker 2010), for the Greater Caribbean subpopulation of loggerhead sea turtles from 2001 through 2009. Note that the survey effort was not consistent among years (from Richards et al. 2011).

Yucatán Peninsula to as far north as Tamaulipas (Figure 11.17) (Márquez-M. 1990; Ehrhart et al. 2003). Additional nesting locations for the Greater Caribbean subpopulation of loggerheads include Cay Sal Bank in the eastern Bahamas; along the coasts of Cuba, Central America, Colombia, and Venezuela; and the eastern Caribbean Islands (Figure 11.17) (Addison and Morford 1996; Addison 1997; Ehrhart et al. 2003; Conant et al. 2009). Annual nest counts for the Greater Caribbean subpopulation of loggerheads from 2001 through 2009 consistently exceeded 2,000 nests, or about 370 nesting females, with a high of 3,349 nests in 2009 (Figure 11.21). An analysis of trends in loggerhead nesting for the Greater Caribbean subpopulation is challenging because few long-term standardized nesting surveys are available for the region, the survey effort at monitored beaches has changed, and scattered, low-level loggerhead nesting can be found at many locations; however, nesting for this subpopulation appears to be stable (Figure 11.21).

11.3.1.2 Hatchling, Post-Hatchling, and Oceanic Juvenile Life History and Distribution for Gulf of Mexico Loggerheads

After loggerhead hatchlings emerge from the nest (Figure 11.22), they begin a period of frenzied activity, and during this period, they move from their nest to the surf, swim, and are swept through the surf zone (Witherington 1995; Conant et al. 2009). A magnetic compass and a progression of orientation cues guide the hatchlings as they swim offshore from the nesting beach (Lohmann et al. 2012). Once the swim frenzy stage ends, neonate loggerheads that have migrated offshore are mostly inactive and swim only occasionally and slowly. They begin to feed and are no longer relying on retained yolk (Witherington 2002). Pelagic post-hatchling

Figure 11.22. Loggerhead sea turtle hatchling in its frenzy stage as it approaches the sea (photograph by Burcin Tuncer) (Tuncer 2009).

northwest Atlantic Ocean loggerheads may inhabit the neritic waters just off the nesting beach for weeks to months, which may be a transition to the oceanic stage that loggerheads enter as they grow, or they may be transported by ocean currents into the Gulf of Mexico and the North Atlantic Ocean (Witherington 2002; Bolten 2003; Conant et al. 2009).

Since loggerhead hatchlings that emerge from nests on the Florida Gulf and Atlantic coasts must swim in opposite directions and search for different surface currents to migrate away from continental shelf waters, Wyneken et al. (2008) compared the pattern of swimming activity shown by the hatchlings from each coast over the first 6 days of migration. Hatchlings from both coasts were equally active during the first 24 h of swimming (the frenzy period), as well as during the daylight hours that followed (the post-frenzy period). However, the Gulf coast hatchlings were significantly more active than the Atlantic turtles during the nocturnal portion of the post-frenzy period (Wyneken et al. 2008). This difference could be related to the greater distance Gulf coast loggerhead hatchlings must negotiate to locate surface currents for transport out of the Gulf of Mexico and into the Atlantic Ocean. These behavioral differences could be determined genetically or may be due to phenotypic plasticity that occurs as the hatchlings respond to unique environmental cues on each coast (Wyneken et al. 2008).

As post-hatchlings, northwest Atlantic Ocean loggerheads inhabit areas where surface waters converge to form local downwellings, which are characterized by linear accumulations of floating material, especially *Sargassum*; these areas are common between the Gulf Stream and the southeast U.S. coast and between the Loop Current and the Florida coast in the Gulf of Mexico (Carr 1986; Witherington 2002; Witherington et al. 2012). During this time, the post-hatchlings feed on a wide variety of floating material, including organisms associated with the *Sargassum* community (Table 11.2), and are low-energy, float-and-wait foragers (Witherington 2002; Witherington et al. 2012).

In a study conducted from 2005 through 2011 to determine the importance of the pelagic *Sargassum*-dominated drift community to young sea turtles, 1,688 of 1,704 post-hatchlings that were observed were loggerheads (Witherington et al. 2012). While 30 of the post-hatchlings

Table 11.2. Summary of Life History Information for the Loggerhead Sea Turtle

Parameter	Values	References
Nesting season: Northwest Atlantic Ocean DPS	April through September	NMFS and USFWS (2008)
Remigration interval		
Quintana Roo, Mexico	Mean: 2.6 years	J. Zurita, ECOSUR, personal communication, cited in NMFS SEFSC (2009)
Casey Key, Florida	Mean: 3.7 years, Range: 1–8 years	Tucker (2010)
Cape Sable, Florida	Mean: 2 years	Davis and Whiting (1977)
Nesting interval		
Casey Key, Florida	Mean: 12 days, Range: 6–21 days	Tucker (2010)
Sanibel Island, Florida	Mean: 11 days	LeBuff (1990)
Key Island, Florida	Mean: 11 days	Addison (1996)
Cape Sable, Florida	Mean: 12 days, Range: 1–24 days	Davis and Whiting (1977)
Number of nests/season		
Casey Key, Florida	Mean: 5.4 nests, Range: 2–8 nests	Tucker (2010)
Sanibel Island, Florida	Mean: 3 nests	LeBuff (1990)
Key Island, Florida	Mean: 3.9 nests	Addison (1996)
Number of eggs/nest		
Isla Contoy, Quintana Roo, Mexico	Mean: 110 eggs, Range 71–177 eggs	Najera (1990)
Santa Rosa Island, Florida	Mean: 117 eggs, Range: 53–170 eggs	Atencio (1994)
	Mean: 116 eggs	Lamont et al. (1998)
Cape San Blas, Florida	Mean: 100 eggs	Lamont et al. (1998)
Casey Key, Florida	Mean: 102 eggs	Llew Ehrhart and Bill Redfoot, UCF, personal communication, cited in NMFS SEFSC (2009)
Cape Sable, Florida	Mean: 100 eggs, Range: 48–159 eggs	Davis and Whiting (1977)
Dry Tortugas, Florida	Mean: 102 eggs	van Houtan and Pimm (2007)
Egg incubation time		
Santa Rosa Island, Florida	Mean: 66.5 days, Range: 50–81 days	Atencio (1994)
	Mean: 54 days	Lamont et al. (1998)
Cape San Blas, Florida	Mean: 62 days	Lamont et al. (1998)
Cape Sable, Florida	Mean: 55 days	Davis and Whiting (1977)
Nest pivotal temperature	29 °C	Yntema and Mrosovsky (1982)
Sex ratio of hatchlings (proportional female)		
Sarasota, Florida	Mean: 0.71	Blair (2005)
Sanibel Island, Florida	Mean: 0.65	Blair (2005)

(continued)

Table 11.2. (continued)

Parameter	Values	References
Emergence success of hatchlings from nests		
Santa Rosa Island, Florida	Mean: 0.21	Lamont et al. (1998)
Cape San Blas, Florida	Mean: 0.27	Lamont et al. (1998)
Size of hatchlings		
Azores, Portugal	Estimated value: 15 cm SCL[a]	Bjorndal et al. (2000)
Southeastern Gulf Stream, Florida	Mean: 5.4 cm SCL, Range: 4.6–6.3 cm SCL	Eaton et al. (2008)
Size of post-hatchlings: East and west coast of Florida	Range: 3.9–7.8 cm SCL	Witherington et al. (2012)
Duration of hatchling stage: The Azores, Portugal	Estimated value: less than 1 year	Bjorndal et al. (2000)
Size of oceanic juveniles: East and west coast of Florida	Estimated range: 15–63 cm SCL[b]	Bjorndal et al. (2000), TEWG (2009)
Duration of oceanic juvenile stage: Cape Canaveral, Florida, and Madeira and the Azores, Portugal	Estimated range: 7–11.5 years	Bjorndal et al. (2003)
Diet of oceanic juveniles		
Lower Texas Region	*Sargassum*, pelagic crustaceans, and mollusks	Plotkin (1996)
East and west coast of Florida	Marine animals associated with the *Sargassum* community, including anemones, hydroids, *Aurelia* sp., and *Sargassum*	Witherington et al. (2012)
Size of oceanic juveniles at recruitment to neritic juvenile stage		
U.S. Gulf of Mexico	Range: 41.6–79.7 cm SCL[b]	Bjorndal et al. (2001)
East and west coast of Florida	Range: 31.7–98.7 cm SCL	Witherington et al. (2012)
Duration of neritic juvenile stage: U.S. Gulf of Mexico	Estimated value: 20 years	Bjorndal et al. (2001)
Diet of neritic juveniles: Lower Texas Region	Pipe cleaner sea pens, calico crabs, *Libinia* sp., blue crabs, *Persephona* sp., bivalves, gastropods, and carrion from fisheries bycatch	Plotkin et al. (1993), Plotkin (1996)
Age at sexual maturity: U.S. Gulf of Mexico	Estimated value: 27 years	Bjorndal et al. (2000, 2003)
Size of sexually mature adult females		
Quintana Roo, Mexico	Mean: 90.6 cm SCL[b], Range: 73.7–105.7 cm SCL	J. Zurita, ECOSUR, personal communication, cited in TEWG (2009)
Casey and Manasota Key, Florida	Mean: 89 cm SCL[b], Range: 74.1–105.7 cm SCL	T. Tucker, Mote Marine Laboratory, personal communication, cited in TEWG (2009)

(continued)

Table 11.2. (continued)

Parameter	Values	References
Cape Sable, Florida	Mean: 92.4 cm SCL, Range: 76.2–108 cm SCL	Davis and Whiting (1977)
U.S. Gulf of Mexico	Estimated value: 79.7 cm SCL[b]	Bjorndal et al. (2001)
Diet of adults: Lower Texas Region	Pipe cleaner sea pens, calico crabs, *Libinia* sp., blue crabs, *Persephona* sp., bivalves, gastropods, and carrion from fisheries bycatch	Plotkin et al. (1993)

[a]*SCL* straight carapace length, *cm* centimeters
[b]To convert from curved carapace length (*CCL*), the following equation was used: SCL = (0.948 × CCL) − 1.442 (Bjorndal et al. 2001)

were observed in the Gulf of Mexico, the majority of post-hatchling loggerheads were observed in the Atlantic Ocean, and only during the hatching season of adjacent Florida nesting beaches (July to October). All the loggerhead post-hatchlings were observed in both deep neritic and oceanic waters and were slightly larger than hatchlings measured on nearby nesting beaches, indicating that they had begun to feed and grow following their offshore recruitment (Witherington et al. 2012). It was not surprising that most of the observed post-hatchlings were loggerheads, given the large numbers of loggerheads that nest on Florida beaches (Figures 11.17, 11.18, and 11.19).

The oceanic juvenile life stage is better understood for loggerheads than for any of the other sea turtle species (Table 11.2) (Bolten 2003; Witherington et al. 2006a, 2012). Loggerhead sea turtle hatchlings that originate from nesting beaches in the northwest Atlantic Ocean appear to use oceanic developmental habitats and move with the North Atlantic gyre for several years before returning to their neritic foraging and nesting habitats (Bolten 2003). Using the North Atlantic gyre, these oceanic juveniles can be transported to the northeast Atlantic and Mediterranean Sea (Carr 1987; Carreras et al. 2006; Eckert et al. 2008).

Since the surface currents used by Florida Gulf coast hatchlings are unknown, hatchlings from Casey Key, Sarasota County, were evaluated to determine their likely migratory routes (Merrill and Salmon 2011). The Gulf of Mexico hatchlings were shown to possess a guidance system for responding to surface currents. A hypothesized migratory route for Casey Key hatchlings included turtles being initially carried northward by an along-shore countercurrent, then south by the eastern portion of the Loop Current. Some turtles could then exit the Gulf via the Florida Straits and become entrained within the Gulf Stream (Merrill and Salmon 2011).

The oceanic juvenile stage in the North Atlantic primarily has been studied in the waters around the Azores and Madeira. Juvenile loggerheads undergo a long period of residency around the Azores, but turtles in Madeiran waters appear to be passing through (Dellinger and Freitas 2000; Bolten 2003). While 10 % of oceanic juveniles in the Azores were determined to be from Mexico, approximately 70 % of juveniles were from the Peninsular Florida subpopulation. It could not be determined whether any of these turtles came from Gulf of Mexico nesting beaches (Bolten et al. 1998). However, none of the oceanic juveniles in the Azores were from the northern Gulf of Mexico subpopulation (Bolten et al. 1998). After many years as oceanic juveniles (Table 11.2), which could include time in the eastern Atlantic and Mediterranean Sea, northwest Atlantic Ocean loggerheads return to settle in coastal habitats as neritic juveniles (Heppell et al. 2003).

11.3.1.3 Neritic Juvenile Life History and Distribution for Gulf of Mexico Loggerheads

The neritic juvenile stage begins when loggerhead sea turtles leave the oceanic zone, and juvenile loggerheads continue to mature in the neritic zone until they reach adulthood (Bolten 2003). Juvenile loggerheads recruiting to neritic habitats in the Gulf are typically not seen until they are larger than about 30–40 cm (11.8–15.7 in) SCL (Table 11.2).

The coasts of the Yucatán Peninsula are regarded as major foraging areas for juvenile loggerheads (Ehrhart et al. 2003). Loggerheads are the most abundant sea turtle in the western Gulf of Mexico; the majority of loggerheads that occur there are neritic juveniles (Rabalais and Rabalais 1980; Plotkin 1989; Plotkin et al. 1993). In addition, large juveniles have been associated with hard substrates, such as reefs and oil production areas (Figure 11.23), and appear to use these areas for resting (Rosman et al. 1987). Renaud and Carpenter (1994) characterized the long-term movement and submergence patterns of loggerheads using satellite telemetry. Four loggerheads were captured under oil and gas platforms in the Gulf of Mexico and tracked for periods of 5–10.5 months from June 1989 through January 1991. Loggerheads spent an average of more than 90 % of their time underwater in any given season, and average submergence times ranged from 4.2 min in June to 171.7 min in January. The home ranges determined for the turtles extended from 954 to 28,833 km^2 (368 to 11,132 mi^2), while core areas ranged from 89.6 to 4,279 km^2 (35 to 1652 mi^2). The core areas included several oil and gas platforms that may have been visited on a daily, weekly, or monthly basis (Renaud and Carpenter 1994).

The Flower Garden Banks National Marine Sanctuary (FGBNMS), three offshore banks located in the northwestern Gulf of Mexico, has been determined to be the residence of a population of large juvenile loggerheads (Hickerson 2000). Underwater and above water surveys conducted by recreational scuba divers from August 1994 through April 2000 resulted in 152 sightings of sea turtles. Most of the sightings were loggerheads (87 %), but hawksbills, leatherbacks, and unidentified turtles were also observed. Six large juveniles (five females and

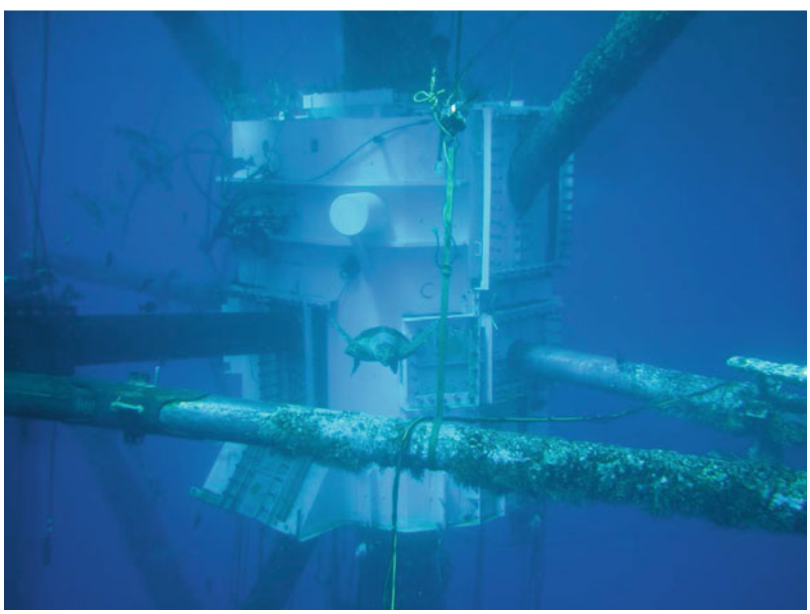

Figure 11.23. Loggerhead sea turtle swimming under an oil and gas platform (photograph courtesy of Ed Elfert, Chevron Corporation, photographer unknown).

one male) were captured and satellite tracked from June 1995 through September 1998. The male was recaptured three times over a 20-month period. More than 40 % of the satellite locations were within the boundaries of the FGBNMS (Hickerson 2000). An analysis using geographic information systems (GIS) indicated an average core range of the satellite-tracked loggerheads of 133.6 km^2 (52 mi^2), and an average home range of 1,074 km^2 (415 mi^2), which are similar to ranges determined for satellite-tracked loggerheads captured under oil and gas platforms in the Gulf of Mexico (Renaud and Carpenter 1994). The average core ranges of the juvenile loggerheads were within 1 km (0.6 mi) of FGBNMS boundaries, while the home range was within 30 km (18.6 mi) of the boundaries (Hickerson 2000).

During a Kemp's ridley investigation along the Texas coast from 2003 through 2007 (Seney 2008), four juvenile loggerheads, averaging 68.6 cm (23.6 in) SCL, were captured by recreational hook and line from Galveston County piers in April, August, and September. All four turtles were successfully rehabilitated and released, and none had been recaptured or stranded as of July 2008.

As part of an in-water study of juvenile Kemp's ridleys inhabiting Apalachee and Deadman bays in Florida that was conducted from 1995 through 1999, 11 loggerhead sea turtles were captured (Barichivich 2006). Loggerheads were captured in Deadman Bay from March through August; they ranged in size from 23.7 cm (9.3 in) SCL to more than 1 m (3.3 ft) SCL, with four being of adult size. Turtles were captured when water temperatures were between 20.7 and 32.7 °C. The large proportion of post-oceanic (less than 50 cm [19.7 in] SCL) loggerheads supports the hypothesis that this area may be an ejection point for turtles recruiting from the oceanic zone to the neritic zone. In addition, the observation of small loggerhead turtles suggests that they remained within the Gulf of Mexico during the oceanic juvenile stage (Barichivich 2006).

In a Kemp's ridley investigation conducted in Waccasassa Bay from June 1986 through October 1995, loggerhead sea turtles were captured from April through November (Schmid 1998). One loggerhead, measuring 86.4 cm (34 in) SCL, was netted on the seagrass shoals of Waccasassa Reefs, and 19 loggerheads, averaging 65 cm (25.6 in) SCL and ranging from 50 to 77.4 cm (19.7 to 30.5 in) SCL, were collected near the oyster bars of Corrigan Reef. Loggerhead turtles greater than 80 cm (31.5 in) SCL were caught at Corrigan Reef, but they could not be landed for data collection. Five loggerheads were recaptured, and recapture times ranged from 142 to 189 days. In an earlier survey, two loggerheads, ranging in size from 57 to 88 cm (22.4 to 34.6 in) SCL, were captured (Schmid and Ogren 1990). The results of the studies indicated the importance of oyster reefs and, to a lesser extent, seagrass beds in Waccasassa Bay as foraging habitat for juvenile and adult loggerheads (Schmid and Ogren 1990; Schmid 1998).

As a result of surveys conducted from July 1990 through December 1996, Florida Bay was shown to be an important developmental habitat for loggerhead sea turtles (Schroeder et al. 1998). The loggerheads captured averaged 80.1 cm (31.5 in) SCL and ranged in size from 48.9 to 98.7 cm (19.3 to 38.9 in) SCL; this size class represents juveniles that are just nearing maturation as well as adults.

Juvenile loggerhead sea turtles may periodically move between the neritic and oceanic zones, particularly during the winter (Bolten 2003; Morreale and Standora 2005; McClellan and Read 2007). Juvenile loggerhead sea turtles foraging in the Gulf of Mexico are primarily carnivorous but do consume some plant material in both the oceanic and neritic zones (Table 11.2). Loggerhead prey varies seasonally and geographically. For example, in the northwestern Gulf of Mexico, loggerheads feed primarily on sea pens during the spring, then primarily on crabs during the summer and fall, paralleling the annual increase in the abundance of crabs in the Gulf (Plotkin et al. 1993).

11.3.1.4 Adult Life History and Distribution for Gulf of Mexico Loggerheads

The age at which loggerhead sea turtles reach sexual maturity is variable (Bjorndal et al. 2000, 2001); however, the estimated age in the U.S. Gulf of Mexico is 27 years (Table 11.2). The duration of the adult female reproductive life stage is at least 25 years for the northwest Atlantic Ocean loggerhead nesting assemblages (Dahlen et al. 2000). While female loggerheads typically do not reproduce every year (Table 11.2), male loggerheads may breed every year (Wibbels et al. 1990). Limited studies of adult loggerheads indicate that their diet is similar to that of neritic juveniles (Table 11.2).

Essentially all shelf waters along the Gulf of Mexico shoreline are inhabited by loggerheads (Conant et al. 2009). Adult loggerheads have been associated with hard substrates, such as reefs and oil production areas, and appear to use these areas for resting (Rosman et al. 1987). Stranded adults have been observed in the Chandeleur Islands/Sound area in eastern Louisiana (Fuller 1988). During aerial surveys, loggerheads were associated with platforms off the Chandeleur Islands in the Gulf of Mexico (Lohoefener et al. 1989). Additionally, the FGBNMS coral reefs 150 km (93.2 mi) off the Louisiana/Texas coast typically have resting loggerheads present (Hickerson and Peccini 2000).

Continental shelf waters along the Florida west coast and the Yucatán Peninsula have been identified as important resident areas for adult female loggerheads from both the Peninsular Florida and northern Gulf of Mexico subpopulations (Meylan et al. 1983; Schroeder et al. 2003; Ehrhart et al. 2003; Foley et al. 2008; Conant et al. 2009; TEWG 2009). For example, approximately 98 % of the loggerheads tagged on Gulf of Mexico beaches as part of the Cooperative Marine Turtle Tagging Program, administered by the Archie Carr Center for Sea Turtle Research, from 1980 through 2007 were later recaptured in the Gulf of Mexico (TEWG 2009). Satellite telemetry and aerial/ship survey data consistently have shown that Gulf of Mexico adult female loggerheads likely remain within the Gulf or more southern geographic regions, such as Mexico and the Caribbean (TEWG 2009).

Post-nesting female loggerhead sea turtles leave the nesting beach area immediately (typically within 24 h) after the last clutch of eggs is deposited and often make directed migrations (Schroeder et al. 2003). The migratory route may be neritic or may involve crossing oceanic waters, and even if the foraging destinations are similar, turtles do not necessarily follow the same migratory routes. Ocean currents may affect migration routes; temporary course adjustments occur, and post-nesting females occasionally swim against the prevailing current. Post-nesting female loggerhead sea turtles have strong foraging area site fidelity, take up residence in discrete foraging areas on continental shelves, and may move among a number of preferred foraging sites within the larger foraging area (Schroeder et al. 2003).

During USACE trawl surveys conducted within the Tampa Bay Entrance Channel in 1997, five loggerheads that were captured—two adult males, one adult female, and two large juveniles; average size: 86.6 cm (34.1 in) SCL—were later tracked (Nelson 2000). The turtles, which were tracked from 13 to 376 days, remained in the study area for days or months at a time and eventually moved in response to changing water temperature. As water temperatures decreased, the turtles moved offshore or to the south and returned to their original location when water temperatures warmed. The results of the surveys indicated that dredging activities in the Tampa Bay Entrance Channel should be conducted during either the winter or summer when temperatures were at their extremes (Nelson 2000).

During 1998, 1999, and 2000, 38 nesting females were outfitted with transmitters and tracked by satellite after they had deposited their last clutch for the nesting season (Foley et al. 2008). Twenty-eight of the turtles were from the Peninsular Florida subpopulation (15 from the Atlantic coast and 13 from the Gulf of Mexico coast), and ten were from the northern Gulf of Mexico subpopulation. The females typically left the vicinity of the nesting beach within 24 h of nesting, and movements were usually highly directed. The post-nesting females took up residence in well-defined, relatively small (median size of 2,000 km^2 [772 mi^2]) areas on the continental shelf adjacent to Florida, Texas, Mexico, the Bahamas, and Cuba within a few weeks of departing from the nesting beaches (Foley et al. 2008). Sixty percent of the turtles (22 of 38) from both nesting assemblages took up residence off the Florida Gulf coast between the Dry Tortugas and Cape San Blas. The distribution of resident areas of female loggerheads from both nesting assemblages overlapped off the western coast of Florida, the western and northern coast of the Yucatán Peninsula, and the northern coast of Cuba (Foley et al. 2008).

Migrations of 28 post-nesting loggerheads from nesting beaches in Sarasota County—their most important Gulf of Mexico rookery—were tracked between May 2005 and December 2007 (Girard et al. 2009). Post-nesting migrations were completed in 3–68 days. Five different migration patterns were observed and included the following: six turtles remained in the vicinity of their nesting site; nine migrated to the southwestern part of the Florida Shelf; two migrated to the northeast Gulf of Mexico; five turtles migrated to the Yucatán Shelf in the southern Gulf of Mexico, Campeche Bay, or Cuba; and six loggerheads migrated to the Bahamas. Loggerheads moved along rather straight routes over the continental shelf but showed more indirect paths in oceanic waters. Smaller turtles remained on the Florida Shelf, and larger individuals showed various migration strategies, staying on the Florida Shelf or moving to long-distance foraging grounds (Girard et al. 2009).

Hart et al. (2012a) recently completed a study that identified shared regional at-sea foraging areas for female loggerheads, which was the first study to consolidate tracking data from three different nesting subpopulations in the Gulf. Ten females from nesting beaches of three subpopulations (St. Joseph Peninsula for northern Gulf of Mexico subpopulation, Casey Key for Peninsular Florida subpopulation, and DTNP for Dry Tortugas subpopulation) in the Gulf of Mexico were satellite tracked in 2008, 2009, and 2010. All turtles migrated to discrete foraging sites in two common areas located off southwest Florida and the northern Yucatán Peninsula, located 102–904 km (63.4–561.7 mi) from the nesting beaches. Within 3–35 days, turtles migrated to the foraging sites, where they all displayed high site fidelity over time. The results of the study indicated that different nesting aggregations of loggerheads in the Gulf of Mexico use common at-sea foraging areas, and these important areas should be protected (Hart et al. 2012a).

In a study of nearshore waters of Gullivan Bay, Ten Thousand Islands, that was conducted from 1997 through 2004, few loggerheads (15 turtles averaging 65.5 cm [25.8 in] SCL) were captured, possibly because they prefer deeper waters (Witzell and Schmid 2004, 2005). During surveys conducted at Key West National Wildlife Refuge (KWNWR) from 2002 through 2006, more than 470 loggerhead sea turtles were sighted, and 182 neritic juveniles and adults, ranging in size from 36.4 to 98.1 cm (14.3 to 38.6 in) SCL and averaging 75.5 cm (29.7 in) SCL, were captured, demonstrating the importance of this area as a foraging site for these size classes (Eaton et al. 2008).

A long-term study of Florida Bay demonstrated that this area provided year-round resident foraging areas for large numbers of juvenile and male and female adult loggerhead sea turtles (Schroeder et al. 2003; Eaton et al. 2008; Conant et al. 2009). From 1990 through 2006, 902 loggerheads were captured; they ranged in size from 33 to 98.7 cm (13 to 38.9 in) SCL, averaging 77.7 cm (30.6 in) SCL. Multiple recaptures of juvenile and adult loggerheads over periods of up to 10 years indicated some strong site fidelity within Florida Bay. Genetic studies have demonstrated that more than 80 % of the adult loggerheads that forage in Florida Bay are from the Peninsular Florida subpopulation, while about 10 % are from the Quintana Roo, Mexico nesting population (Heppell et al. 2003).

While the neritic zone provides important foraging, internesting, and migratory habitats for adult loggerhead sea turtles, some adults may also periodically move between the neritic and oceanic zones (Schroeder et al. 2003; Harrison and Bjorndal 2006). In addition, on a seasonal basis, loggerheads typically move from offshore to inshore and/or from south to north in the spring, reversing their direction in the fall. It is clear from many studies that water temperature is a critical environmental cue that loggerheads use to guide their movements in and out of shallow coastal waters (Hopkins-Murphy et al. 2003). Some loggerheads nesting in the Gulf may inhabit oceanic habitats and are significantly smaller than those in neritic habitats (Reich et al. 2007a).

11.4 GREEN SEA TURTLE (*CHELONIA MYDAS*)

The green sea turtle is the largest of the hard-shelled turtles (Figures 11.24, 11.25, and 11.26), weighing up to 395 kg (870 lb) (Pritchard 2010). The only other turtle that is larger than the green turtle is the leatherback sea turtle (Witherington et al. 2006b). The green turtle was named for the green cartilage found under its shell (Witzell 1994); other names given to this turtle include *edible turtle* and *soup turtle* (Hirth 1997).

The green sea turtle has been studied for centuries (Pritchard 2010). They were the first species for which a conservation and research program was established; this occurred in Tortuguero, Costa Rica, in 1955 and was the first of its kind in the world (Carr et al. 1978; Carr 1979). The Tortuguero program, which was started by the Caribbean Conservation Corporation (now known as the Sea Turtle Conservancy), continues today. Much of what is known about the biology of green sea turtles, as well as other sea turtle species, has been learned on the beaches of Tortuguero.

11.4.1 Green Sea Turtle Life History, Distribution, and Abundance

The distribution range of the green turtle includes nesting beaches, foraging areas, and migration corridors throughout the tropical and subtropical oceans of the world (Figure 11.27) (Hirth 1997). Although most green turtle populations are greatly depleted and many rookeries have been extirpated (Witherington et al. 2006b), green turtles nest in more than 80 countries and are thought to inhabit coastal areas of more than 140 countries (NMFS 2011a).

Figure 11.24. Green sea turtle nesting on the beach of Archie Carr National Wildlife Refuge, Brevard and Indian River counties, Florida, June 2012 (photograph by Steve A. Johnson, University of Florida, with permission).

Figure 11.25. Green sea turtle returning to the water after nesting at Archie Carr National Wildlife Refuge, Brevard and Indian River Counties, Florida, June 2012 (photograph by Steve A. Johnson, University of Florida, with permission).

Figure 11.26. Green sea turtle foraging on a seagrass bed (photograph by RP van Dam) (NOAA 2011).

11.4.1.1 Nesting Life History, Distribution, and Abundance for Gulf of Mexico Green Sea Turtles

Green sea turtles typically nest at night. In the tropics, nesting may span all seasons of the year, with a peak during the rainy season (Witherington et al. 2006b). The green turtle nesting life history varies throughout the Gulf of Mexico, Caribbean, and northwest Atlantic Ocean region (Table 11.3). Available green turtle life history information for specific Gulf of Mexico beaches or regions is included in Table 11.3.

Green turtles that nest on Gulf of Mexico beaches migrate to locations outside the Gulf. For example, turtles from Gulf of Mexico nesting beaches can be found foraging in the Bahamas,

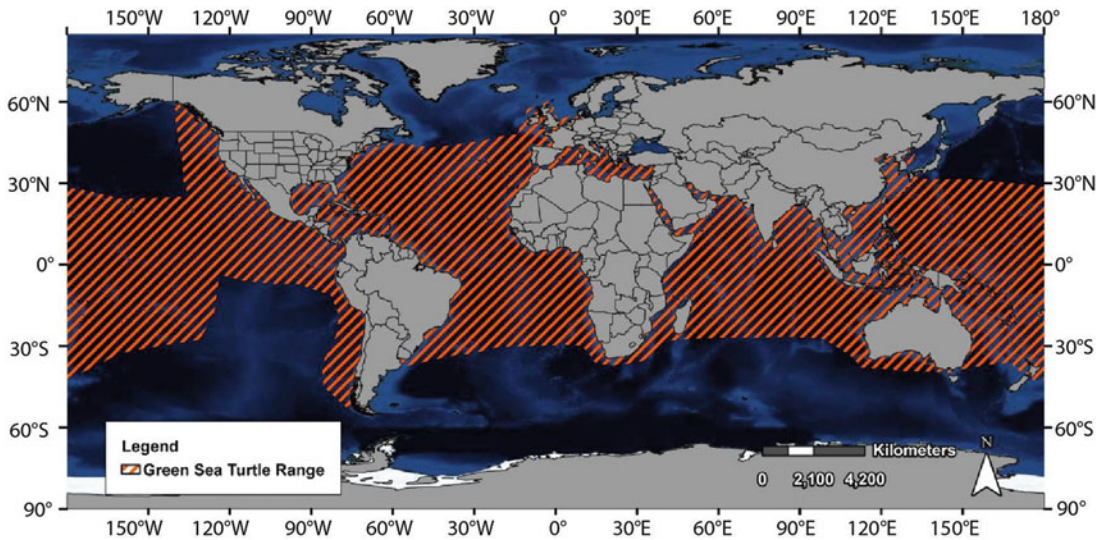

Figure 11.27. Range of the green sea turtle (from NOAA 2009b).

Table 11.3. Summary of Life History Information for the Green Sea Turtle

Parameter	Values	References
Nesting season		
Tortuguero, Costa Rica	May through September	Witherington et al. (2006b)
Santa Rosa Island, Florida	May through August	Atencio (1994)
Remigration interval		
Tortuguero, Costa Rica	Mean: 3 years	Carr et al. (1978)
Melbourne Beach, Florida	Mean: 2 years	Bjorndal et al. (1983)
Indian River Lagoon, Florida	Mean: 2 years	Witherington and Ehrhart (1989a)
Nesting interval		
Tortuguero, Costa Rica	Mean: 13 days, Range: 9–16 days	Carr and Hirth (1962)
El Cuyo, Yucatán, Yucatán Peninsula, Mexico	Mean: 12 days, Range: 10–20 days	Xavier et al. (2006)
Isla Aguada, Campeche, Yucatán Peninsula, Mexico	Mean: 11 days, Range: 8–13 days	Guzmán-Hernández et al. (2006)
Atlantic coast, Florida	Range: 9–15 days	Hirth (1997)
Number of nests/season		
Tortuguero, Costa Rica	Mean: 3 nests	Bjorndal (1982)
	Mean: 2.6 nests, Range: 2–7 nests	Bjorndal and Bolten (1992)
El Cuyo, Yucatán, Yucatán Peninsula, Mexico	Mean: 2.9 nests	Xavier et al. (2006)
Archie Carr National Wildlife Refuge, Brevard County, Florida	Mean: 3.6 nests	Johnson and Ehrhart (1994)

(continued)

Table 11.3. (continued)

Parameter	Values	References
Number of eggs/nest		
Tortuguero, Costa Rica	Mean: 104 eggs, Range: 7–178 eggs	Fowler (1979)
	Mean: 112 eggs, Range: 3–219 eggs	Bjorndal and Carr (1989)
Isla Contoy, Quintana Roo, Yucatán Peninsula, Mexico	Mean: 106 eggs, Range: 69–163 eggs	Najera (1990)
El Cuyo, Yucatán, Yucatán Peninsula, Mexico	Mean: 131 eggs	Xavier et al. (2006)
Rio Lagartos, Yucatán, Yucatán Peninsula, Mexico	Mean: 128 eggs, Range: 96–147 eggs	Najera (1990)
Santa Rosa Island, Florida	Mean: 131 eggs, Range: 76–172 eggs	Atencio (1994)
Dry Tortugas, Florida	Mean: 123 eggs	van Houtan and Pimm (2007)
Egg incubation time		
Tortuguero, Costa Rica	Mean: 56 days, Range: 48–70 days	Carr and Hirth (1962)
	Mean: 62 days, Range: 53–81 days	Fowler (1979)
Isla Aguada, Campeche, Yucatán Peninsula, Mexico	Mean: 52 days, Range: 41–66 days	Guzmán-Hernández et al. (2006)
Santa Rosa Island, Florida	Mean: 63 days, Range: 51–83 days	Atencio (1994)
Nest pivotal temperature: Tortuguero, Costa Rica	Range: 28.5–30.3 °C	Spotila et al. (1987)
Sex ratio of hatchlings from nests (proportional female): Tortuguero, Costa Rica	Range: 0.08–0.74	Spotila et al. (1987)
Emergence success of hatchlings from nests		
Tortuguero, Costa Rica	Mean: 0.51	Carr and Hirth (1962)
	Mean: 0.83	Fowler (1979)
El Cuyo, Yucatán, Yucatán Peninsula, Mexico	Mean: 0.86	Xavier et al. (2006)
Santa Rosa Island, Florida	Range: 0.13–0.48	Atencio (1994)
Size of hatchlings		
Tortuguero, Costa Rica	Mean: 5 cm SCL[a], Range: 4.6–5.6 cm SCL	Carr and Hirth (1962)
Merritt Island, Florida	Range: 4.4–5.8 cm SCL	Ehrhart (1980)
East and west coast of Florida	Range: 5.3–5.6 cm SCL	Witherington et al. (2012)
Size of oceanic juveniles		
St. Joseph Bay, Florida	Estimated mean: 20 cm SCL	Avens et al. (2012)
East and west coast of Florida	Range: 15–26.3 cm SCL	Witherington et al. (2012)

(continued)

Table 11.3. (continued)

Parameter	Values	References
Duration of oceanic juvenile stage: St. Joseph Bay, Florida	Estimated mean: 2 years	Avens et al. (2012)
Diet of oceanic juveniles: Gulf Stream off east and west coast of Florida	Marine animals related to pelagic *Sargassum*, including hydroids, *Membranipora* sp., portunid crabs, gastropods, serpulid polychaetes, *Porpita* sp., *Sargassum* nudibranchs, *Sargassum* snails, *Pyrosoma* sp.; planehead filefish; *Sargassum;* and coralline and cladophora algae	Witherington et al. (2012)
	Sargassum, *Sargassum*-affiliated invertebrates, including hydroids, bryozoans, *Porpita* sp., and *Vellela* sp.	Witherington, unpublished data, cited in Witherington et al. (2006b)
Size of oceanic juveniles at recruitment to neritic juvenile stage		
Mansfield Channel, Texas	Mean: 34.2 cm SCL, Range: 26.6–52 cm SCL	Shaver (1994)
St. Joseph Bay, Florida	Mean: 36.6 cm SCL, Range: 25–75.3 cm SCL	Foley et al. (2007)
	Mean: 36.3 cm SCL, Range: 18.1–78.5 cm SCL	Avens et al. (2012)
Cedar Key, Florida	Mean: 59.8 cm SCL	Eaton et al. (2008)
Corrigan Reef, Florida	Mean: 56.8 cm SCL, Range: 42.9–70.9 cm SCL	Schmid (1998)
Waccasassa Reef, Florida	Mean: 68 cm SCL, Range: 63–73.9 cm SCL	Schmid (1998)
Cape Sable, Florida	Mean: 40.1 cm SCL, Range: 32.8–51.9 cm SCL	Eaton et al. (2008)
Duration of neritic juvenile stage: St. Joseph Bay, Florida	Estimated range: 17–19 years	Avens et al. (2012)
Diet of neritic juveniles		
Great Inagua, Bahamas	Turtle grass, manatee grass, algae, jellyfish, sponges, and sea pens	Bjorndal (1980)
Caribbean coast of Nicaragua	Turtle grass, star grass, jellyfish, sponges, and sea pens	Mortimer (1981)
Caribbean	Turtle grass, manatee grass, and algae	Bjorndal (1985)
	Turtle grass, manatee grass, shoalgrass, star grass, eelgrass, and chicken liver sponge	Bjorndal (1997)

(continued)

Table 11.3. (continued)

Parameter	Values	References
St. Joseph Bay, Florida	Turtle grass, shoal grass, manatee grass, *Laurencia* sp., and *Entermorpha* sp.	Foley et al. (2007)
Mosquito Lagoon, Florida	Manatee grass and turtle grass	Mendonca (1981)
	Manatee grass, shoalgrass, star grass, and green and red algae	Mendonca (1983)
Age at sexual maturity		
St. Joseph Bay, Florida	Estimated range: 19–21 years	Avens et al. (2012)
Mosquito Lagoon, Florida	Estimated range: 18–27 years	Frazer and Ehrhart (1985)
Size of sexually mature adult females		
Tortuguero, Costa Rica	Mean: 100.3 cm SCL, Minimum: 69.2 cm SCL	Carr and Hirth (1962)
El Cuyo, Yucatán, Yucatán Peninsula, Mexico	Mean: 101.1 cm SCL[b]	Xavier et al. (2006)
Isla Aguada, Campeche, Yucatán Peninsula, Mexico	Mean: 101.8 cm SCL, Range: 92.3–114 cm SCL	Guzmán-Hernández et al. (2006)
Melbourne Beach, Florida	Range: 83–114 cm SCL	Witherington (1986)
Diet of adults		
Great Inagua, Bahamas	Turtle grass, manatee grass, algae, jellyfish, sponges, and sea pens	Bjorndal (1980)
Caribbean coast of Nicaragua	Turtle grass, star grass, jellyfish, sponges, and sea pens	Mortimer (1981)
Caribbean	Turtle grass, manatee grass, and algae	Bjorndal (1985)
	Turtle grass, manatee grass, shoalgrass, star grass, eelgrass, and chicken liver sponge	Bjorndal (1997)

[a]*SCL* straight carapace length, *cm* centimeters
[b]To convert from curved carapace length (*CCL*), the following equation was used: SCL = (0.9426 × CCL) − 0.0515 (Goshe 2009)

Barbados, Cuba, Puerto Rico, Venezuela, and the southeast United States (Bass et al. 2006; NMFS and USFWS 2007c). In addition, juveniles that forage in the Gulf of Mexico originate from Barbados, Costa Rica, Florida, Mexico, Venezuela, and Suriname (Bass and Witzell 2000). Tagging studies have also demonstrated that post-nesting Tortuguero green turtles migrate into the Gulf (Carr et al. 1978). Therefore, green turtle rookeries outside the Gulf of Mexico must be considered when assessing green turtles that occur in the Gulf of Mexico.

The largest green turtle rookery in the Gulf of Mexico, Caribbean, and northwest Atlantic Ocean, as well as in the entire Atlantic Ocean, is located in Tortuguero, Costa Rica (Figure 11.27) (Bjorndal et al. 1999; Witherington et al. 2006b; NMFS and USFWS 2007c). Additional major green turtle rookeries in the area include the Florida east coast, the Yucatán Peninsula, and areas along the Mexican Gulf coast (Figure 11.28). In the U.S. Gulf of Mexico, limited green turtle nesting has been documented along the Florida Gulf coast, as well as on the south Texas

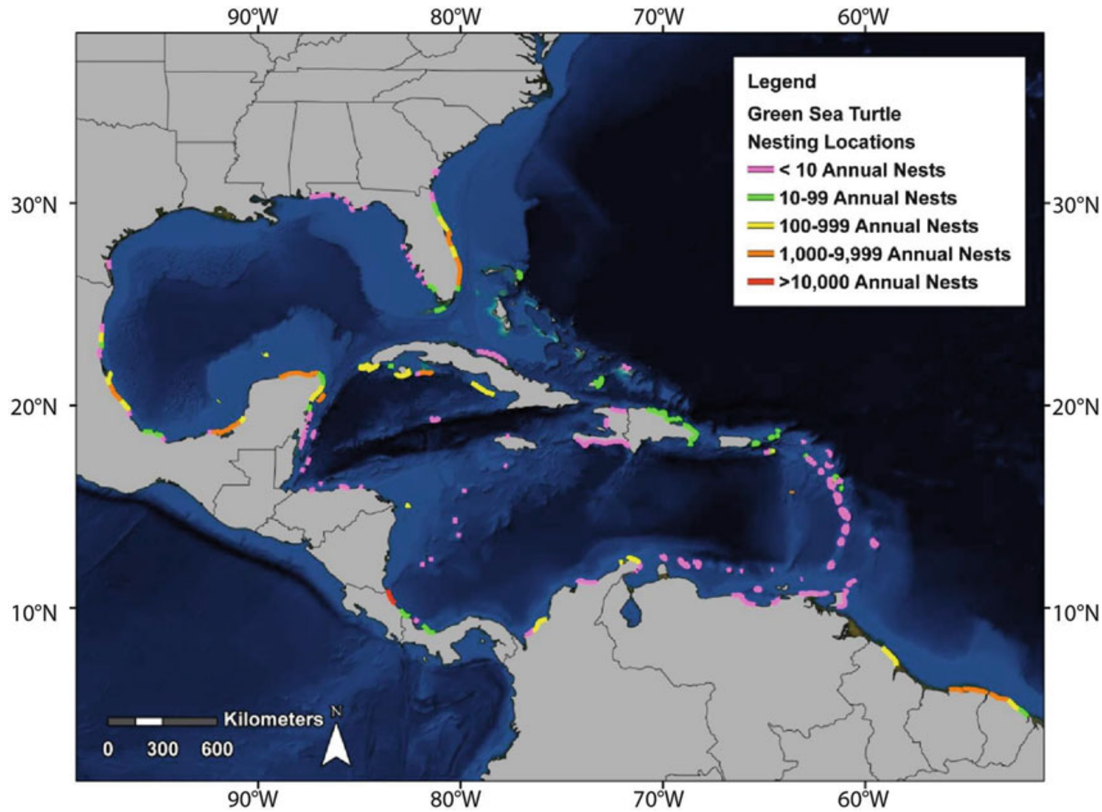

Figure 11.28. Generalized nesting locations of the green sea turtle in the Gulf of Mexico, Caribbean, and northwest Atlantic Ocean (from Dow et al. 2007; SWOT 2010b).

coast (Figure 11.28). Before 1956, no records documented green turtle nesting on Gulf of Mexico beaches or elsewhere in the continental United States However, green turtle populations are known to have been abundant historically as described in Section 11.1.2 (Meylan et al. 1995). Extensive seagrass beds in south Texas bays were once an important feeding ground for the green sea turtle (Owens et al. 1983).

A review of the green turtle nesting data through 2001 for the Tortuguero, Yucatán Peninsula, and Florida nesting beaches indicated that all three western Atlantic Ocean subpopulations were increasing (IUCN 2004). The population of nesting females for the western Atlantic Ocean and Caribbean Sea was estimated to range from 30,981 to 31,981 in 2001, a 13 to 66 % increase over past published estimates (IUCN 2004).

Green turtle nesting in Tortuguero has increased significantly since the 1970s (NMFS and USFWS 2007c). Evaluation of the trend in nesting activity on the Tortuguero beach indicated a relatively steady increase from 1971 to the mid-1980s, constant or possibly decreasing nesting during the late 1980s, and then resumption of an upward trend in the 1990s (Bjorndal et al. 1999). About 41,250 adult female green turtles emerged on beaches each year from 1971 through 1975, and from 1992 through 1996, approximately 72,200 females emerged per year (Bjorndal et al. 1999). Approximately 104,411 nests per year were laid on the beach at Tortuguero from 1999 through 2003, which corresponds to about 17,402–37,290 nesting female green turtles each year (Troëng and Rankin 2005). More than 80,000 green turtle nests were estimated for Tortuguero each year from 2003 through 2009, with a high of almost 178,000 estimated nests or about 68,000 estimated nesting females in 2007 (Figure 11.29).

Figure 11.29. Annual number of nests (*bars*) and estimated number of nesting females (*line*), assuming 2.6 nests per female (Bjorndal and Bolten 1992), for green sea turtles in Tortuguero, Costa Rica from 2003 through 2009 (Harrison and Troëng 2004a, 2005; Haro and Troëng 2006a; Haro and Harrison 2007a; Nolasco del Aguila et al. 2008a, 2009; Atkinson et al. 2010).

In Florida, approximately 99 % of green turtle nesting occurs on the Atlantic coast, with most of the activity occurring from Brevard through Broward counties (Figure 11.28) (Witherington et al. 2006b). While the Dry Tortugas historically supported an important green turtle nesting colony of approximately 2,800 nesting females each year that was thought to be extirpated (Thompson 1988), dozens of nests have been laid on Dry Tortugas beaches in recent years (Witherington et al. 2006b). Green turtle nesting was not recorded on Florida's Gulf coast before 1987; however, green turtle nesting now occurs regularly along most of the Gulf coast, with the exception of the Big Bend area, which is the area around Apalachee Bay from Franklin County on the west end through Jefferson, Taylor, and Dixie counties on the southeast end (Figure 11.28) (Witherington et al. 2006b). Very little green turtle nesting occurs on the Texas coast; for example, one nest was documented in 1987, five nests were documented during 1998, and 15 nests were recorded in 2013 (Shaver 2000; NPS 2013a).

From 1979 through 2009, green turtle nesting in Florida has increased significantly, with a high of 12,751 nests or 3,542 nesting females in 2007 (Figure 11.30). The data for all surveyed beaches in Florida include beaches on both the Atlantic and Gulf coasts; separating out data for the Gulf of Mexico was not possible. In 2010, 13,225 green turtle nests were recorded on all surveyed beaches in Florida (FFWCC FWRI 2011a). The increasing trend in green turtle nesting in Florida is in agreement with increases observed at Tortuguero, Costa Rica and Ascension Island since the mid-1970s (Bjorndal et al. 1999; Godley et al. 2001; Troëng and Rankin 2005). Interestingly, there are significant interannual fluctuations in green turtle nesting activity in Florida, which have been noticeable only since 1990 when nesting numbers began to increase significantly (Figure 11.30). These annual nesting fluctuations are characteristic of green turtle

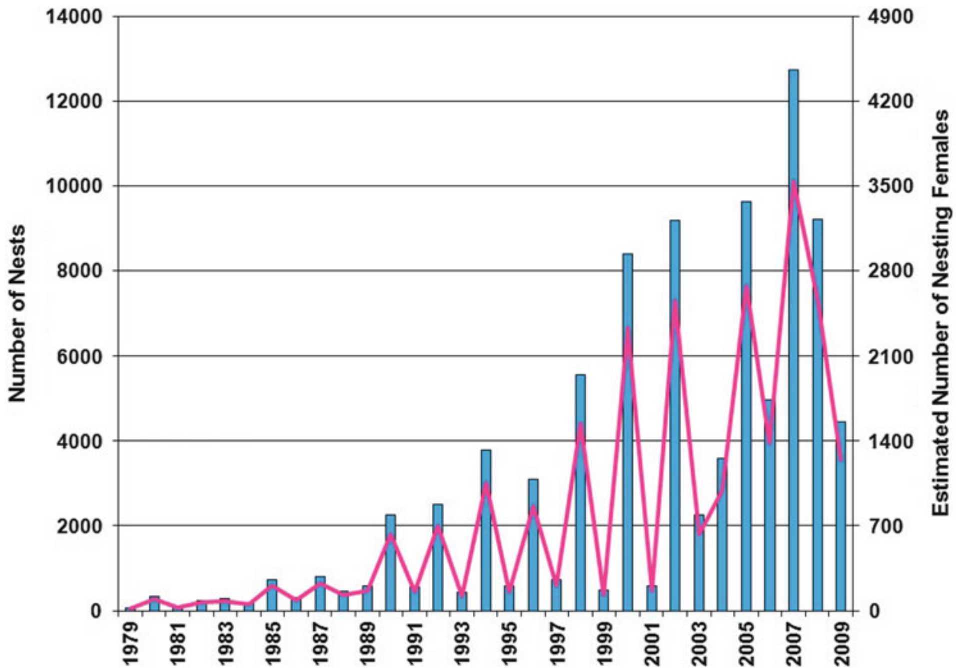

Figure 11.30. Annual number of nests (*bars*) and estimated number of nesting females (*line*), assuming 3.6 nests per female (Johnson and Ehrhart 1994), for green sea turtles for all surveyed beaches in Florida from 1979 through 2009. Note that the survey effort was not consistent among years (from FFWCC FWRI 2011a).

rookeries (Limpus and Nicholls 1988; Bjorndal et al. 1999). It has been proposed that such wide fluctuations are due to environmentally constrained primary productivity, which regulates energy budgets of this herbivorous turtle (Broderick et al. 2001).

In the Florida Panhandle, green turtles nest along Eglin Air Force Base on Santa Rosa Island; 16 and 14 nests were laid in 1994 and 1996, respectively, while no nests were recorded in 1995 or 1997 (Lamont et al. 1998). This small nesting population also displays the annual nesting fluctuations characteristic of green turtle rookeries mentioned above. Green turtles also nest on the low-relief mangrove islands located in the Ten Thousand Islands in southwest Florida (Foley et al. 2000).

From 1993 through 2002, the number of green turtle nests laid on Mexican beaches ranged from about 1,000 to over 7,000, with a high recorded in 2,000 of more than 7,200 nests, representing approximately 2,570 nesting females (Figure 11.31). Interannual fluctuations in green turtle nesting are also apparent for the Mexican Gulf coast (Figure 11.31). A summary of nesting data on Mexican beaches from 1993 through 2002 indicated that most green turtles nested in the state of Quintana Roo, followed by Veracruz, Yucatán, Campeche, and Tamaulipas, with an estimated mean annual total of 1,430, 730, 633, 535, and 141 nests, respectively (Márquez-M 2004). Daily nesting beach reconnaissance efforts along the Yucatán Peninsula indicates that green turtle nesting has increased. In the early 1980s, approximately 875 nests per year were laid; by 2000, nesting had increased to more than 1,500 nests per year (Instituto Nacional de Pesca unpublished data cited in NMFS and USFWS 2007c). By 2004, about 1,547 females were estimated to nest on Yucatán Peninsula beaches (IUCN 2004).

From 2002 through 2004, green turtle nesting activity at El Cuyo Beach, located within the Río Lagartos Biosphere Reserve on the Yucatán Peninsula in Mexico, was evaluated (Xavier

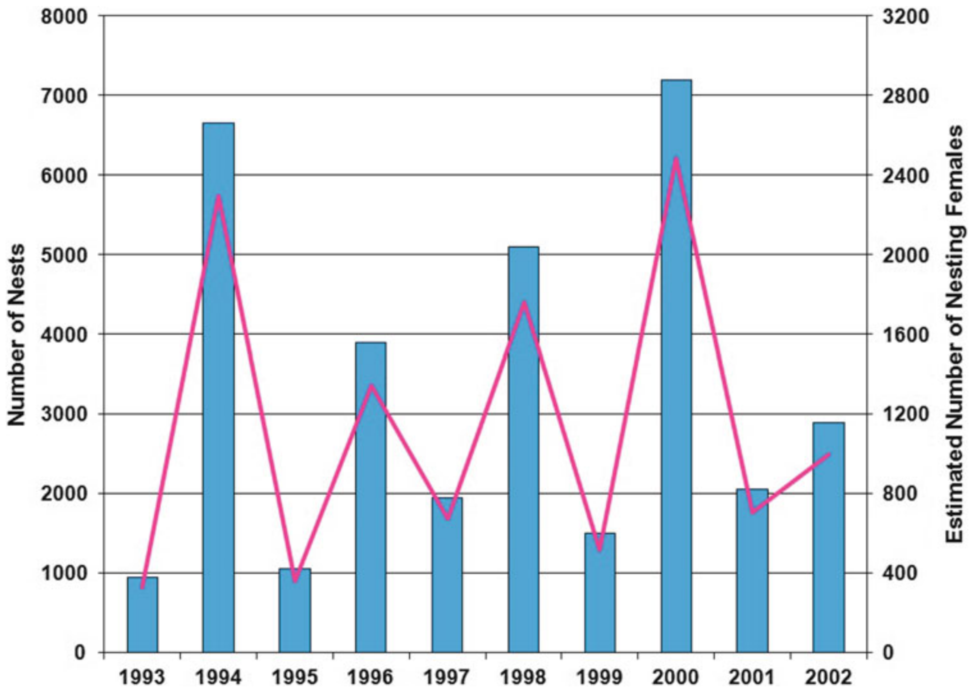

Figure 11.31. Annual number of nests (*bars*) and estimated number of nesting females (*line*), assuming 2.9 nests per female (Xavier et al. 2006), for green sea turtles for Mexican Gulf of Mexico beaches from 1993 through 2002 (Márquez-M 2004).

et al. 2006). Beach surveys were conducted from mid-April through September each year. Green turtle nesting activity fluctuated during the study period, from a high of 390 nests in 2002, followed by 157 nests in 2003 and 172 nests in 2004. There were no differences in the size of nesting females between nesting seasons. Green turtles exhibited high site fidelity, with an average distance of 1.8 km (1.1 mi) between nests. Hurricane Ivan destroyed the majority of green turtle nests in 2004 (Xavier et al. 2006). El Cuyo Beach is also one of the most important hawksbill nesting beaches on the Yucatán Peninsula. Compared to hawksbills, green turtles had a narrow range of preferences for beach morphological features and selected beaches with slightly steeper slopes, mainly in the berm zone, and were clearly restricted to nesting in the western portion of the beach (Cuevas et al. 2010).

11.4.1.2 Hatchling, Post-Hatchling, and Oceanic Juvenile Life History and Distribution for Gulf of Mexico Green Sea Turtles

Green sea turtle hatchlings emerge from the nest about four days after hatching, when sand surface temperatures are appropriate (Figure 11.32) (Godfrey and Mrosovsky 1997), and enter the sea, dispersing away from land and into the open ocean (Witherington et al. 2006b). While little is known about green turtle post-hatchling ecology, some hatchlings have been observed in convergence zones, drift lines, and *Sargassum* (Carr 1987; Witherington et al. 2006b, 2012). Post-hatchling and young juvenile green turtles are thought to be carnivorous or omnivorous (Table 11.3) (Hirth 1971).

The oceanic phase of juvenile green sea turtles remains one of the most poorly understood phases of green turtle life history (NMFS and USFWS 2007c). After leaving the nesting beach as 5 cm (2 in) SCL hatchlings, green turtles disappear until they recruit to neritic habitats as

Figure 11.32. Green sea turtle hatchling moving across the beach toward the sea (photograph by Kjersti Joergensen) (Joergensen 2012).

juveniles and feed primarily on seagrasses and algae (Table 11.3) (Reich et al. 2007b). While extensive surveys in the northwest Atlantic Ocean have resulted in the sightings of thousands of loggerheads, green turtles are rarely observed (Witherington 2002; Bolten 2003; Witherington et al. 2012). A recent analysis of stable isotopes in green turtle scute tissue suggested that, before recruiting to neritic habitats, green turtles occupy similar habitats and feed at the same trophic level as oceanic-stage loggerheads for 3–5 years (Reich et al. 2007b).

Fifteen post-hatchling green turtles, with an average size of 5.4 cm (2.1 in) SCL and 44 juveniles, with an average size of 20.6 cm (8.1 in) SCL, were observed and/or captured as part of a study to determine the importance of the pelagic *Sargassum*-dominated drift community to young sea turtles in the Atlantic and Gulf of Mexico from 2005 through 2011 (Witherington et al. 2012). In contrast to loggerheads, the post-hatchling green turtles were active within the *Sargassum* or swimming. There was a size gap of 9 cm (3.5 in) between the juvenile and post-hatchling green turtles. Juveniles were estimated to be 1–2 years old, indicating that two discrete life stages of green turtles were observed in the *Sargassum* community (Witherington et al. 2012).

11.4.1.3 Neritic Juvenile Life History and Distribution for Gulf of Mexico Green Sea Turtles

Juvenile green turtles typically shift from the oceanic phase and move into neritic waters, such as protected lagoons and open coastal areas. They may remain in these areas for many years and then shift to other sites as larger juveniles (Zug and Glor 1998; Witherington et al. 2006b). Some green turtles remain in the oceanic zone for extended periods (Pelletier et al. 2003).

When they move into neritic foraging grounds, they adopt an herbivorous diet, which is unique among sea turtles (Table 11.3) (Bjorndal 1985). Green turtles utilize seagrasses as a food source by frequently grazing in the same areas, thus promoting an abundance of young grass blades with high nutritional value (Bjorndal 1980, 1985). However, juvenile, as well as adult, green turtles do consume invertebrates as part of their diet (Table 11.3).

Tracking and CMR studies on green sea turtles in south Texas, as well as sightings at jetties and channel entrances along the central and south Texas coast during the summer, suggest that these areas serve as important developmental habitats for juvenile green turtles (Shaver 1990a, 1994, 2000; Manzella et al. 1990; Renaud et al. 1992, 1993b, 1995b; Renaud and Williams 1997). For example, juvenile green turtles were studied from 1989 through 1992 in the Laguna Madre and the Mansfield Channel in Texas. Turtles, ranging initially in size from 26.6 to 52 cm (10.5 to 20.5 in) SCL, were caught during all months except January, and 42 % of the turtles were recaptured at least once; the CPUE was positively correlated with water temperature, air temperature, and water salinity (Shaver 1994).

Ten juvenile green turtles, ranging in size from 26.6 to 47.9 cm (10.5 to 18.9 in) SCL, that were tracked from July through September 1992 and nine juveniles, ranging from 29.1 to 49.7 cm (11.5 to 19.6 in) SCL, that were tracked in August and September 1992 demonstrated a preference for the jetty habitat in Brazos Santiago Pass in south Texas (Renaud et al. 1993b, 1995b). The tracking data suggested that movement behaviors of juvenile green sea turtles in the Brazos Santiago Pass area did not threaten their lives with respect to the biannual hopper dredging of the Brownsville Ship Channel because they mainly stayed in the jetty habitat and rarely entered the channel (Renaud et al. 1993b).

In addition, neritic juvenile green turtles, ranging in size from 27.5 to 29.9 cm (10.8 to 11.8 in) SCL, were tracked in 1996 and 1997 to determine the spatial and temporal distribution of green turtles in Lavaca and Matagorda bays in Texas, as well as to determine the exposure risk associated with a point-source discharge (Renaud and Williams 1997). Coincident with the distribution of seagrasses, green turtles used the southwestern portion of Lavaca Bay and the western shores of Matagorda Bay. Their home range was greater than 19.5 km^2 (7.5 mi^2), and they moved into the Gulf of Mexico during the winter months seeking warmer temperatures.

From April 1991 through March 1993, the feeding ecology of juvenile green turtles at South Padre Island was characterized by capturing turtles from jetty habitat at Brazos Santiago Pass and seagrass beds at South Bay/Mexiquita Flats (Coyne 1994). There were differences in the sizes of the turtles and feeding selectivity between the sites. Green turtles using the jetty habitat averaged 31.3 cm (12.3 in) SCL in size (range of 22.2–47.9 cm [8.74–18.9 in] SCL) and fed strictly on algae. The turtles captured from the seagrass beds ranged in size from 29.6 to 81.5 cm (11.7 to 32.1 in) SCL (mean = 44.6 cm [17.6 in] SCL); they fed primarily on seagrasses and exhibited a preference for the least abundant taxon, shoalgrass (*Halodule wrightii*). The highest growth rates were observed in spring and summer (0.62 and 0.64 cm/month [0.24 and 0.25 in/month], respectively), while turtles grew the slowest during the winter (0.14 cm/month [0.06 in/month]) (Coyne 1994). There were also seasonal differences in activity patterns, with increased movement and strong site fidelity during the warmer months.

Using stable isotope analysis of scute tissues (Gorga 2010), an intermediate stage between the shift of green turtles from the oceanic juvenile stage (when they are omnivores) to the neritic juvenile stage (when they switch to foraging on seagrass and algae) was found for juvenile green turtles inhabiting south Texas bays, such as the Lower Laguna Madre and Aransas Bay. This intermediate stage consists of an initial recruitment of neritic juveniles to jetty habitat located on the channel passes Gulf-ward of adjacent bays to forage on algae before subsequently recruiting to seagrass beds in these bays. These results and those found earlier by Coyne (1994) indicated the use of a characteristic sequence of distinct habitats by multiple life-history stages of green turtles in Texas bays (Gorga 2010).

As a result of a large hypothermic-stunning event in St. Joseph Bay along the Florida Gulf coast in Gulf County in December 2000/January 2001 that stranded 388 green turtles, information on the assemblage of green turtles along the northeastern Gulf of Mexico, which had been observed in the past (Carr and Caldwell 1956), was obtained (Foley et al. 2007). All of the green turtles were neritic juveniles, with a mean size of 36.6 cm (14.4 in) SCL (range = 25–75.3 cm [9.8–29.6 in] SCL). Genetic analyses indicated that about 81 % of the turtles were from nesting populations in Florida and the Yucatán. This assemblage is interesting because it does not have substantial representation from the nesting population in Tortuguero, Costa Rica, the Atlantic's largest green turtle nesting population (Foley et al. 2007).

Green turtle CMR data from surveys conducted in St. Joseph Bay during 2002 and 39 green turtle strandings from a small hypothermic-stunning event in 2003 indicated site fidelity to St. Joseph Bay since more than 70 % of the recaptures were originally tagged in 2001 (McMichael et al. 2003, 2008). The recapture intervals ranged from 311 to 1,193 days (mean = 636 days). Turtles ranged in size from 27.4 to 56.9 cm (10.8 to 22.4 in) SCL (mean = 37.4 cm [14.7 in] SCL). Annual growth increments ranged from 1.2 to 8.4 cm/year (0.47 to 3.3 in/year) (McMichael et al. 2008). Size-specific growth rates were as follows:

- For 30–39.9 cm (11.8–15.7 in) SCL turtles, growth rates averaged 4.7 cm/year (1.9 in/year).
- For 40–49.9 cm (15.7–19.6 in) SCL turtles, growth rates averaged 4.3 cm/year (1.7 in/year).
- For 50–59.9 cm (19.7–23.2 in) SCL turtles, growth rates averaged 4.8 cm/year (1.9 in/year).
- For 60–69.9 cm (23.6–27.5 in) SCL turtles, growth rates averaged 1.2 cm/year (0.47 in/year).

In addition, as a result of comparing green turtle mean size data throughout the Gulf of Mexico, it appears that developmental migration occurs throughout the region, which was first discussed by Carr and Caldwell (1956), with the size of turtles increasing as they move from west to east (McMichael et al. 2003). Green turtles enter St. Joseph Bay at just under 30 cm (11.8 in) SCL, and the majority of turtles remain in this habitat until they reach a size of just over 60 cm (23.6 in) SCL. The estimated mean time of residency within the bay is 7 years (±1.5 years) (Eaton et al. 2008).

As a result of a massive hypothermic-stunning event in January 2010, the population of neritic juvenile green turtles inhabiting St. Joseph Bay was characterized using necropsy and skeletochronology by evaluating more than 400 dead turtles of the more than 4,600 turtles that stranded (Avens et al. 2012). The size range of the dead green turtles was not significantly different from those that survived the hypothermic-stunning event, indicating that the sample was representative. The age of the turtles ranged from 2 to 22 years, and SCLs ranged from 18.1 to 78.5 cm (7.1 to 30.9 in). The female age distribution was significantly greater than that of males, and the mean stage duration ranged from 17 to 20 years. Growth rates of the green turtles were significantly influenced by size, age, and calendar year; however, no effect of sex, fibropapilloma status, or body condition on growth rates was found (Avens et al. 2012).

Twenty-eight neritic juvenile green turtles were also captured during an in-water study conducted from 1995 through 1999 on juvenile Kemp's ridleys inhabiting Apalachee and Deadman bays, Florida (Barichivich 2006). One green turtle, which measured 37.3 cm (14.7 in) SCL, was captured in Apalachee Bay, and 27 green turtles, ranging in size from 27.9 to 70.7 cm (11 to 27.8 in) SCL (mean of 42.2 cm [16.6 in] SCL), were captured in Deadman Bay; one green turtle was recaptured in Deadman Bay during the study. The green turtles were captured when water temperatures were between 22.2 and 32.7 °C. The large proportion of post-oceanic (less than 40 cm [15.7 in] SCL) green turtles supported the hypothesis that the Big Bend region may be an ejection point for turtles recruiting from the oceanic zone to the neritic zone. In addition, Deadman Bay, located in the largest remaining seagrass bed in North America, is an important developmental habitat for green turtles (Barichivich 2006).

Green turtles were captured from April through November in a study conducted in Waccasassa Bay from June 1986 through October 1995 (Schmid 1998). Six green turtles were netted on the seagrass shoals of Waccasassa Reefs. They ranged in size from 63 to 73.9 cm (24.8 to 29 in) SCL and averaged 68 cm (26.8 in) SCL. The four green turtles collected near the oyster bars of Corrigan Reef averaged 56.8 cm (22.4 in) SCL, with a size range of 42.9–70.9 cm (16.9–27.9 in) SCL. The results of this study, as well as an earlier survey in which nine juvenile green turtles were captured (mean SCL: 66 cm [26 in], range: 49.5–74 cm [19.5–29.1 in] SCL) (Schmid and Ogren 1990), indicated the importance of the seagrass beds in Waccasassa Bay as foraging habitat for late-stage juvenile green turtles (Schmid 1998).

As a result of surveys conducted in Florida Bay from July 1990 through December 1996 and from 2000 through 2006, the bay was determined to be an important developmental habitat for juvenile green sea turtles (Schroeder et al. 1998; Eaton et al. 2008). The green turtles captured from 1990 through 1996 ranged in size from 25.5 to 52.9 cm (10 to 20.8 in) SCL and averaged 46.2 cm (18.2 in) SCL, while the 73 juveniles captured from 2000 through 2006 averaged 45.8 cm (18 in) SCL (range of 25.5–66.1 cm [10–26 in] SCL). This size class distribution was similar to those for other nearshore developmental habitats in the Gulf of Mexico. No adult green turtles were captured or sighted (Schroeder et al. 1998). Juvenile green turtles, averaging 51.6 cm (20.3 in) SCL, also have been captured in surveys of the Ten Thousand Islands area (Witzell and Schmid 2004).

The movements of six juvenile green sea turtles, ranging in size from 33.4 to 67.5 cm (13.1 to 26.6 in) SCL captured in southwest Florida within Everglades National Park were tracked using satellite telemetry during the spring for 27 days in 2007 and 62 days in 2008 (Hart and Fujisaki 2010). These turtles were observed to be resident for several months in coastal waters ranging up to 10 m (38.2 ft) in depth near their capture and release sites. The results of this study documented habitat use by juvenile green turtles in the mangroves of southwest Florida and highlighted the need to consider the impacts of Everglades restoration activities on juvenile green turtles and their habitat (Hart and Fujisaki 2010).

11.4.1.4 Adult Life History and Distribution for Gulf of Mexico Green Sea Turtles

While growth rates vary among populations, most green turtles grow very slowly because of their low energy, mostly herbivorous diet (Bjorndal 1982). The age to maturity for green turtles ranges from less than 20 years to more than 40 years (Table 11.3) (Hirth 1997; Zug et al. 2002). Reproductive longevity for green turtles ranges from 17 to 23 years (Carr et al. 1978). After leaving the nesting beach as hatchlings and living in a variety of marine habitats for up to 40 or more years, adult female green turtles return to the same beach from where they were hatched (Bowen et al. 1989, 1992).

A recent satellite tracking study provided the first available information on green turtle migratory corridors and post-nesting foraging locations in the Yucatán (Cuevas et al. 2012). In 2011, nine post-nesting females were tracked from eight different nesting beaches. Green turtles appeared to prefer a region known at Petenes-Celestun off the northwest corner of the Yucatán Peninsula for foraging (42 % of tracked turtles), while 22 % of the tracked turtles migrated to the Florida Keys. A well-known green turtle feeding and mating area for the region, the Catoche-Contoy area off the northeast corner of the Yucatán Peninsula, was confirmed. Yucatán waters within about 15 km (9.3 mi) from shore were shown to be important migratory corridors (Cuevas et al. 2012).

Green turtle juveniles and adults are found in inshore and nearshore waters of the U.S. Gulf of Mexico from Texas to Florida (NMFS 2011a). They are known to forage in Florida's coastal waters where there is sufficient seagrass or algae. Important green turtle foraging areas along the Florida Gulf coast include the Florida Keys, Marquesas, Florida Bay, Homosassa, Crystal River, the Cedar Keys, and St. Joseph Bay (Witherington et al. 2006b; NMFS 2011a). When not nesting, adult female green sea turtles reside in Gulf of Mexico foraging areas from throughout the Florida Keys to the Dry Tortugas and waters southwest of Cape Sable (NMFS and USFWS 2007c). From 2002 through 2006, more than 900 green turtles were sighted during surveys conducted at KWNWR. In addition, almost 90 juvenile and adult turtles were captured, ranging in size from 27 to 108.5 cm (10.6 to 42.7 in) SCL and averaging 61.3 cm (24.1 in) SCL (Eaton et al. 2008). Green turtles often return to the same foraging locations after subsequent nesting migrations, and after their arrival, they typically visit specific areas for foraging and resting (Broderick et al. 2006; Taquet et al. 2006).

The coastal foraging grounds, where green turtles spend the majority of their lives, are often highly dynamic, with annual fluctuations in seawater and air temperatures, which cause the distribution and abundance of green turtle food items to vary significantly between seasons and years (Carballo et al. 2002). This variability in food item abundance may explain, in part, the significant interannual fluctuations in green turtle nesting.

11.5 LEATHERBACK SEA TURTLE (*DERMOCHELYS CORIACEA*)

First described by Vandelli in 1761 (Fretey and Bour 1980), the highly specialized leatherback sea turtle is the only living member of the family Dermochelyidae (Eckert et al. 2012). Leatherbacks are more widely distributed than any other reptile species and are the largest species of sea turtles; these gigantic turtles can measure 2 m (6.5 ft) in length and weigh up to about 900 kg (2,000 lb) (Stewart and Johnson 2006; NMFS 2011b). The largest recorded leatherback is a male that weighed 916 kg (2,019 lb), found off the shores of Wales in 1988 (Morgan 1989). Leatherbacks are one of the deepest diving vertebrates, diving to depths greater than 1,000 m (3,280 ft), surpassed only by sperm whales and elephant seals (Eckert et al. 1989a).

The leatherback sea turtle is easily identified by its unique morphology (Figure 11.33). Leatherbacks do not have hard scutes made of keratin on their carapace or plastron like the other species of sea turtles; instead, they are completely covered by a thin layer of smooth, rubbery, oily skin, which is black with white mottling dorsally and lighter colored ventrally (NMFS 2011b). Beneath the skin of the carapace lies a nearly continuous layer of small dermal bones, and the carapace is made up of seven longitudinal bony ridges, which taper to a blunt point posteriorly (NMFS 2011b). There is no sharp angle between the carapace and plastron; therefore, the body of the leatherback is barrel shaped (NMFS 2011b). Leatherbacks have a distinctive upper jaw, with two tooth-like projections that are each flanked by deep cusps, rather than a hard beak-like structure like all other species of sea turtles (NMFS 2011b). Adult leatherbacks have no scales on their head or flippers and lack claws. Their proportionately long fore flippers and streamlined body shape make the leatherback highly adapted for long migrations and deep dives in a primarily pelagic habitat (NMFS 2011b).

Figure 11.33. Leatherback sea turtle covering her eggs after nesting (photograph by Paul Mannix) (Mannix 2012).

11.5.1 Leatherback Sea Turtle Life History, Distribution, and Abundance

Leatherback sea turtles are physiologically unique among all other species of sea turtles and are capable of maintaining a core body temperature several degrees higher than the surrounding water temperature (Frair et al. 1972; Standora et al. 1984). This trait is likely due to countercurrent mechanisms in the circulatory system, peripheral insulation, regional endothermy, and large body size; this suite of adaptations is sometimes referred to as *gigantothermy*, distinct from strict ectothermy and endothermy (Greer et al. 1973; Penick et al. 1998; Eckert et al. 2012). Leatherbacks are able to forage in cold water environments without becoming hypothermic stunned and, due to their ability to tolerate cold temperatures, have the most extensive global range of any reptile (Eckert et al. 2012).

Leatherback sea turtles occur in the Pacific, Atlantic, and Indian oceans (Figure 11.34). While they nest on tropical and subtropical beaches, wide-ranging foraging areas include temperate and subarctic waters (Stewart and Johnson 2006). Adult leatherbacks have the longest migration of any reptile, greater than 5,000 km (3,107 mi), traveling between high-latitude foraging grounds and low-latitude mating and nesting areas (Pritchard 1976). Satellite-tagged leatherbacks have frequently traversed entire ocean basins (Luschi et al. 2003; Hays et al. 2004, 2006; Fossette et al. 2010). In the western Atlantic Ocean, leatherbacks have been sighted as far north as Greenland and as far south as Argentina (Carriol and Vader 2002; González-C. et al. 2011).

Leatherback sea turtles nest on every continent except Europe and Antarctica (Eckert et al. 2012), and an estimated 652 nesting sites have been documented worldwide (Wallace et al. 2010). The largest nesting assemblages in the world are along the coasts of French Guiana and Suriname (4,500–7,500 females per year) and Gabon, West Africa (1,300–2,553 females per year) (NMFS 2011b). There is no evidence of substantial declines recently at the main western

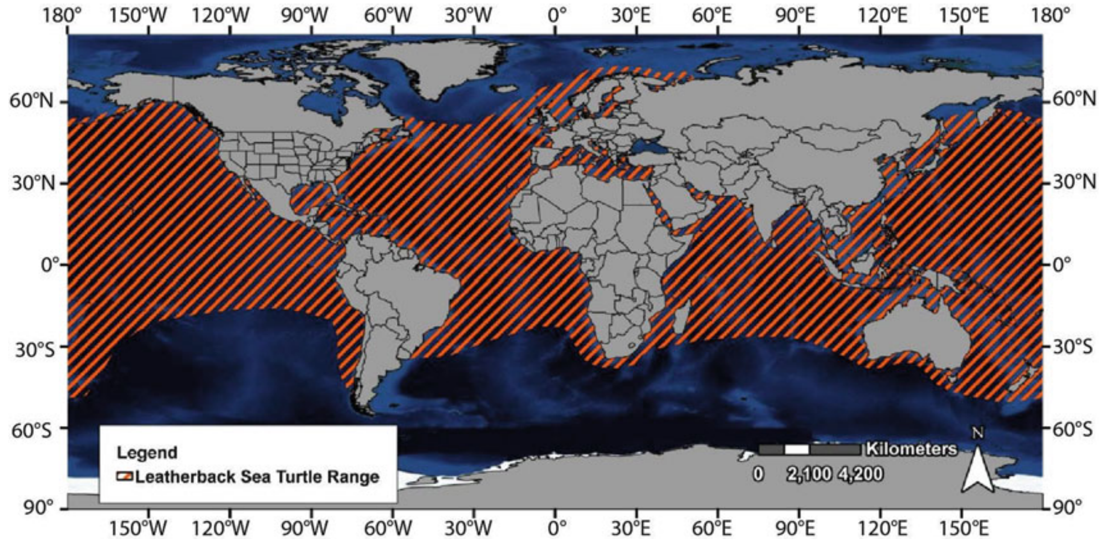

Figure 11.34. Range of the leatherback sea turtle (from NOAA 2009c).

Atlantic Ocean nesting grounds; however, significant threats are affecting populations in the eastern Atlantic, and populations in the Pacific Ocean have been decimated (Spotila et al. 2000; Eckert et al. 2012).

In 1982, 115,000 adult female leatherbacks were estimated to occur worldwide, of which 60 % nested along the Mexican Pacific coast (Pritchard 1982). Spotila et al. (1996) later estimated that only 34,500 nesting females remained worldwide in 1995. However, the most recent population estimate for the entire north Atlantic Ocean ranged from 34,000 to 94,000 adult leatherback sea turtles, and the global population was determined to be slightly female biased, ranging from 51.9 to 66.7 % female (TEWG 2007).

11.5.1.1 Nesting Life History, Distribution, and Abundance for Gulf of Mexico Leatherbacks

Compared to other species of sea turtles, genetic studies to date have revealed reduced global divergence and differentiation between leatherback nesting populations (Eckert et al. 2012). Some of the possible reasons for this include the extensive home ranges, large migrations between foraging and nesting areas, and weaker nesting beach fidelity than other sea turtle species.

The major leatherback nesting beaches in the northwest Atlantic Ocean are located along the coasts of French Guiana and Suriname, as well as along the coasts of Costa Rica and Panama (Figure 11.35). The large colony in French Guiana and Suriname appears to be stable or increasing; from 5,029 to 63,294 nests were laid each year from 1967 to 2002 (Girondot et al. 2007). The population of leatherbacks that nest on the beaches of Trinidad also appears to be stable or slightly increasing; an estimated 52,797 and 48,240 nests were laid in 2007 and 2008, respectively (Eckert et al. 2012).

In the USA, leatherback nesting occurs primarily in St. Croix in the U.S. Virgin Islands, Puerto Rico, and Florida (Figure 11.35). The first record of a leatherback nesting in Florida is from 1947 (Carr 1952), with additional reports coming after 1955 (Caldwell et al. 1955; Caldwell 1959). The highest density of leatherback nesting in Florida occurs along the Atlantic coast from Jensen Beach south to Palm Beach in Martin and Palm Beach counties (Stewart and Johnson

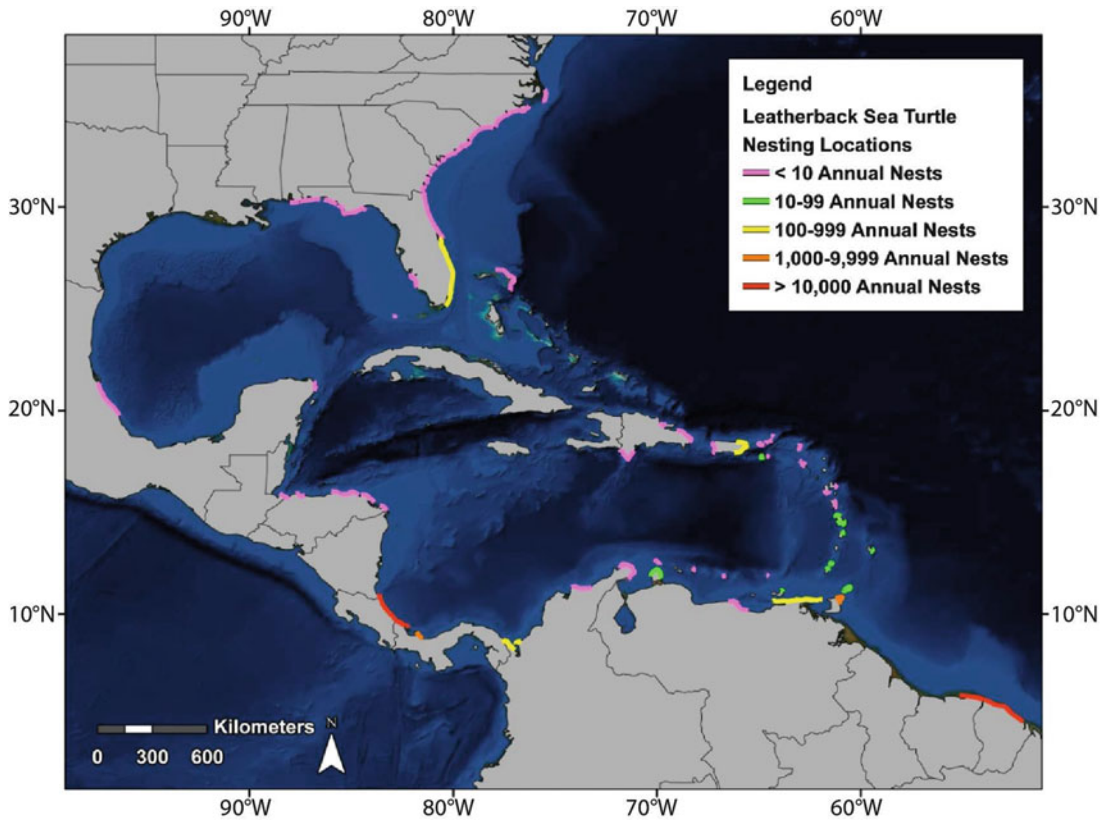

Figure 11.35. Generalized nesting beach locations of the leatherback sea turtle in the Gulf of Mexico, Caribbean, and northwest Atlantic Ocean (interpreted from Dow et al. 2007; SWOT 2007b).

2006). Low numbers of leatherback nesting also occur along the southeast Atlantic coast of the United States, from central Florida through North Carolina (Figure 11.35).

In the Gulf of Mexico, leatherback sea turtles nest at low levels along the Florida, Alabama, and Mexican coasts (Figure 11.35). Leatherback nesting was reported as early as 1962 on a beach near Destin along the Florida Gulf coast (Yerger 1965). As indicated by reports from local residents, leatherbacks may have nested on Texas beaches in the 1920s and 1930s (Hildebrand 1963). One leatherback nest was recorded at PAIS in 2008 (NPS 2013d).

Leatherback sea turtles that forage in and travel through the Gulf of Mexico originate from many different nesting beaches. For example, leatherbacks that were tagged while nesting on the Florida east coast and St. Croix were later recaptured along the Mexican Gulf coast (Eckert et al. 2012). In addition, leatherbacks that nest on Gulf of Mexico beaches migrate to and forage in waters outside the Gulf, and many satellite tracking studies have shown that leatherback sea turtles that nest in the western Atlantic migrate to Western Europe and West Africa (Eckert et al. 2012). Therefore, using data from nesting beaches of leatherbacks known to occur in the Gulf of Mexico to evaluate the population of leatherback sea turtles that use the Gulf is extremely challenging. However, to determine trends in leatherback nesting in the general western Atlantic Ocean area, data from two important nesting beaches in the area with long-term monitoring were evaluated: the Florida east coast and Tortuguero, Costa Rica. In addition, recent investigations have suggested that the Gulf of Mexico may be a significant year-round foraging ground for leatherbacks that nest along the Caribbean coast of Costa Rica, as well as Panama (Evans et al. 2007; Fossette et al. 2010).

The leatherback is a nocturnal nester and, on average, takes just over 100 min to complete the entire nesting procedure (Eckert 1987). While females generally return to their natal beaches to nest, some are known to also nest at beaches greater than 100 km (62.1 mi) apart (Eckert et al. 1989b). In comparison to the other sea turtle species, nesting leatherbacks have a warmer body temperature; therefore, they lay eggs that are at a higher temperature than eggs of other sea turtle species (Mrosovsky and Pritchard 1971). In addition, their eggs are the largest of any sea turtle species (Eckert et al. 2012). Leatherbacks are the only sea turtle species that normally lay small, irregular yolkless eggs along with viable eggs (Bell et al. 2003). The irregular eggs typically appear in a clutch during the latter half of egg laying (Eckert et al. 2012). Table 11.4 includes nesting life history information for leatherback sea turtles; since no life history

Table 11.4. Summary of Life History Information for the Leatherback Sea Turtle

Parameter	Values	References
Nesting season		
French Guiana	April through August	Hilterman and Goverse (2007)
Panama	February through August	Boulon et al. (1996)
Southeast Florida coast	March through June	Stewart and Johnson (2006)
Remigration interval		
St. Croix, U.S. Virgin Islands	Mode: 2 years, Mean: 2.2 years	Dutton et al. (2005)
Babunsanti, Samsambo, Kolukumbo, and Matapica, Suriname	Mode: 2 years	Hilterman and Goverse (2007)
Juno Beach, Florida	Mean: 2.9 years, Range: 1–6 years	Stewart (2007)
Nesting interval		
St. Croix, U.S. Virgin Islands	Mean: 9.6 days	Boulon et al. (1996)
Babunsanti, Samsambo, Kolukumbo, and Matapica, Suriname	Mean: 9.6 days	Hilterman and Goverse (2007)
Juno Beach, Florida	Mean: 10 days	Stewart and Johnson (2006)
Number of nests/season		
St. Croix, U.S. Virgin Islands	Mean: 5.3 nests	Boulon et al. (1996)
Babunsanti, Samsambo, Kolukumbo, and Matapica, Suriname	Mean: 4.6 nests	Hilterman and Goverse (2007)
Juno Beach, Florida	Estimated mean: 4.1 nests	Stewart (2007)
Number of eggs/nest		
St. Croix, U.S. Virgin Islands	Mean: 116.1 eggs	Boulon et al. (1996)
Babunsanti and Matapica, Suriname	Mean: 115.8 eggs	Hilterman and Goverse (2007)
Juno Beach, Florida	Mean: 98 eggs	Stewart and Johnson (2006)
Egg incubation time		
St. Croix, U.S. Virgin Islands	Mean: 63.2 days, Range: 57–76 days	Boulon et al. (1996)
Babunsanti, Samsambo, Kolukumbo, and Matapica, Suriname	Mean: 64 days	Hilterman and Goverse (2007)

(continued)

Table 11.4. (continued)

Parameter	Values	References
Southeast coast of Florida	Mean: 67 days	Stewart and Johnson (2006)
Nest pivotal temperature: French Guiana and Suriname	Mean: 29.5 °C	Hulin et al. (2009)
Sex ratio of hatchlings from nests (proportional female)		
St. Croix, U.S. Virgin Islands	Estimated mean: 0.65	Dutton et al. (1985)
Suriname	Mean: 0.53	Godfrey et al. (1996)
Tortuguero, Costa Rica	Estimated mean: 0.67	Leslie et al. (1996)
Emergence success of hatchlings from nests		
St. Croix, U.S. Virgin Islands	Mean: 0.64	Eckert and Eckert (1990)
Awala Yalimapo, French Guiana	Mean: 0.38	Caut et al. (2006)
Southeast coast of Florida	Mean: 0.47	Stewart and Johnson (2006), Stewart (2007)
Size of hatchling		
Culebra Island, Puerto Rico	Mean: 9.07 cm SCL[a], Range: 7.91–9.90 cm SCL	Tucker (1988)
Matura and Paria Bays, Trinidad	Mean: 6.50 cm SCL	Bacon (1970)
Suriname	Mean: 5.91 cm SCL (Babunsanti)	Hilterman and Goverse (2007)
	Mean: 5.95 cm SCL (Matapica)	
Tortuguero, Costa Rica	Mean: 6.28 cm SCL	Carr and Ogren (1959)
Duration of hatchling stage	Estimated value: 1 year	Spotila et al. (1996)
Size of oceanic juveniles: Juno Beach, Florida	Range: 10–134.7 cm SCL[b]	Tucker (1988), Stewart et al. (2007)
Duration of oceanic juvenile stage: St. Croix, U.S. Virgin Islands	Estimated range: 11–13 years	Dutton et al. (2005)
Diet of oceanic juveniles: Offshore from Boynton Beach, Florida	*Aurelia* sp., *Ocryopsis* sp., warty comb jellyfish, and tunicates	Salmon et al. (2004)
Age at sexual maturity		
St. Croix, U.S. Virgin Islands	Range: 12–14 years	Dutton et al. (2005)
Western North Atlantic	Range: 24.5–29 years	Avens et al. (2009)
Size of sexually mature adult females		
U.S. Virgin Islands	Range: 127.4–172.7 cm SCL[b]	Boulon et al. (1996)
Juno Beach, Florida	Mean: 147.7 cm SCL[b]; Range: 134.7–160.7 cm SCL	Stewart et al. (2007)
Diet of adults		
Offshore from Port Aransas, Texas	Cannonball jellyfish	Leary (1957)
North Sea	Hydrozoans, Siphonophorans, Scyphozoans, *Cyanea* sp., *Aurelia* sp., *Stomolophus* sp., comb jellies, tunicates, cephalopods, and gastropods	den Hartog and van Nierop (1984)

[a] *SCL* straight carapace length, *cm* centimeters
[b] To convert from curved carapace length (*CCL*), the following equation was used: SCL = (0.9781 × CCL) − 0.7714 (Avens et al. 2012)

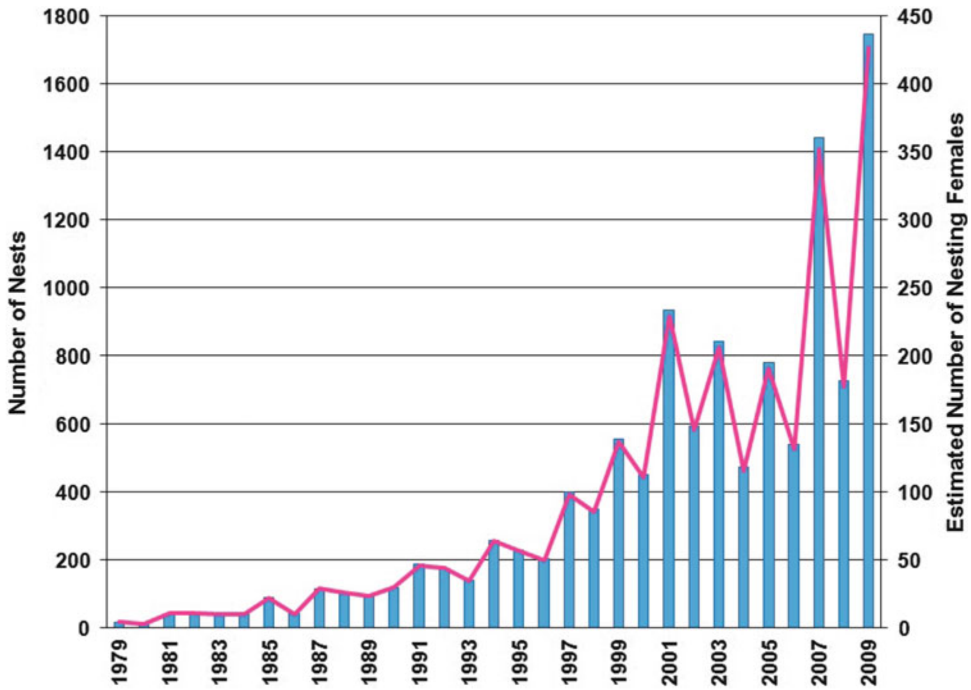

Figure 11.36. Annual number of nests (*bars*) and estimated number of nesting females (*line*), assuming 4.1 nests per female (Stewart 2007), for leatherback sea turtles for all surveyed beaches in Florida from 1979 through 2009. Note that the survey effort was not consistent among years (FFWCC FWRI 2011a).

information for leatherbacks was available specifically for the Gulf of Mexico, values are from nesting beaches in the Caribbean or northwest Atlantic Ocean or leatherback populations from other locations.

Leatherback nesting data for Florida presented below includes beaches on both the Gulf and Atlantic coasts, since separating Gulf of Mexico data was not possible. There is an increasing trend in the number of leatherback nests recorded on Florida beaches (Figure 11.36). From 1979 through 2009, the number of leatherback nests recorded each year in Florida has increased significantly, from 18 recorded nests in 1979 to 1,747 nests in 2009, representing about 5 to 420 nesting females, respectively. In 2010, 1,334 leatherback nests were recorded on Florida beaches (FFWCC FWRI 2011a).

Some of the increased leatherback nesting in Florida could be the result of increased survey and documentation efforts since the late 1970s. Leatherbacks begin their nesting season in Florida early in the year (Table 11.4), and because the leatherback nesting season starts before most nesting surveys begin, the number of nests reported in Florida is considered to be a minimum (Meylan et al. 1995).

No trend is indicated in the number of leatherback nests recorded at Tortuguero, Costa Rica from 1998 through 2009 (Figure 11.37). The lowest number of leatherback nests was recorded in 1998 (94 nests), and nest numbers have ranged from 481 to 1,107 nests, representing about 115 to 264 nesting females, from 1999 through 2009.

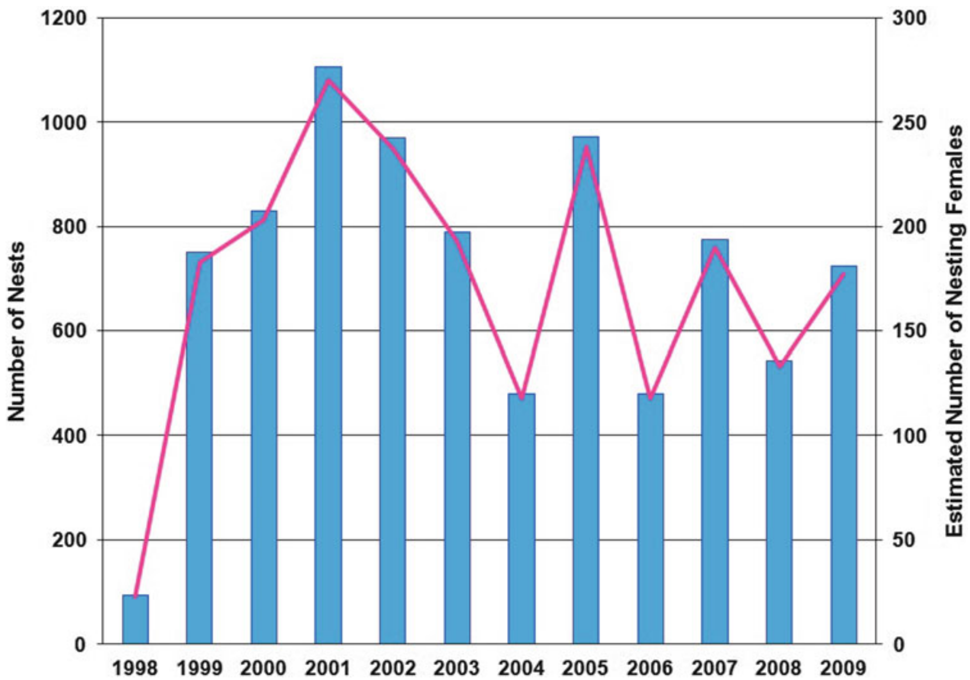

Figure 11.37. Annual number of nests (*bars*) and estimated number of nesting females (*line*), assuming 4.1 nests per female (Stewart 2007), for leatherback sea turtles in Tortuguero, Costa Rica from 1998 through 2009 (Troëng 1998, 2000; Troëng and Cook 2000; Reyes and Troëng 2001; Harrison and Troëng 2003a, b, 2004b; Haro and Troëng 2006b; Haro and Harrison 2007b; Nolasco del Aguila et al. 2008b; Debade et al. 2009; Sarmiento Devia and Harrison 2010).

11.5.1.2 Hatchling, Post-Hatchling, and Oceanic Juvenile Life History and Distribution for Gulf of Mexico Leatherbacks

In addition to being the largest species of sea turtle, leatherback hatchlings are also larger than the hatchlings of other species (Figure 11.38, Table 11.4). Immediately after emerging and crawling to the water, hatchling leatherbacks go through the swim frenzy stage, similar to hatchlings of other sea turtle species, and swim continuously for the first 24 h (Wyneken and Salmon 1992). In contrast to loggerhead and green sea turtle hatchlings, which eventually stop all swimming activities during the night, leatherback hatchlings begin a daily swimming pattern after the first 24 h and decrease swimming to 15–45 % of nighttime (Wyneken and Salmon 1992).

Hatchling leatherbacks are capable of diving soon after entering the ocean. Hatchlings between 2 and 8 weeks of age have been documented to dive deeper and longer with age (Eckert et al. 2012), while foraging exclusively on gelatinous prey throughout the water column (Table 11.4). The post-hatchling habitat remains obscure, and nothing is known about the dispersal or distribution of post-hatchling leatherbacks in the open ocean (Eckert et al. 2012). In contrast to other species of sea turtles, there is no evidence that young leatherbacks associate with *Sargassum* or epipelagic debris (Carr 1987).

Little is known about the life history or distribution of juvenile leatherbacks (Table 11.4) (Eckert et al. 2012). Leatherbacks have a unique diet that consists primarily of jellyfish, salps, and other soft-bodied coelenterates that inhabit the mid-water column in the open ocean (Table 11.4). The distribution of juvenile, as well as adult, leatherbacks is likely to be closely

Figure 11.38. Leatherback sea turtle hatchlings leaving the nesting beach (photograph by Scott R. Benson, NMFS Southwest Fisheries Science Center) (NOAA 2011).

linked to the distribution and abundance of their prey—jellyfish and other soft-bodied invertebrates—as well as their preferred temperature tolerances (Eckert et al. 2012).

A study by Eckert (2002) indicated that juvenile leatherbacks were found exclusively in waters warmer than 26 °C, but larger juveniles and subadults would venture into waters as cold as 8 °C. Leatherbacks, therefore, spend the first portion of their lives in tropical waters, venturing into cooler latitudes only after reaching a size of 97 cm (38.2 in) SCL (Eckert 2002; Avens et al. 2009). The restriction of smaller leatherbacks to warmer waters suggests that size may play a role in the ability of the species to exist in colder waters. The warm water restrictions also suggest that the onset of thermogenerating capability, which is not found in younger or smaller turtles, occurs after reaching a size of about 97 cm (38.2 in) SCL (Eckert 2002).

11.5.1.3 Adult Life History, Distribution, and Abundance for Gulf of Mexico Leatherbacks

Adult leatherbacks, which have the most extensive range of any living reptile, are primarily pelagic and are generally only seen in coastal waters when nesting (Eckert et al. 2012). While the life history information for adults is incomplete (Table 11.4), male and female leatherbacks generally return to their native nesting locales to mate and nest. They presumably mate in the waters adjacent to the nesting beaches (Reina et al. 2005; Eckert et al. 2006). Males may migrate to the area annually or may mate opportunistically at foraging grounds or nesting areas other than their own native area (Eckert and Eckert 1988).

While reproductively active females and males arrive seasonally at preferred subtropical and tropical nesting locations, nonbreeding adults range further north and south into temperate zones seeking areas containing oceanic jellyfish and other soft-bodied invertebrates (Eckert et al. 2012). Nearly 25 % of leatherback sightings away from nesting areas are associated with aggregations of jellyfish, suggesting that jellyfish distribution may drive the distribution of leatherback foraging areas (Houghton et al. 2006). Foraging occurs on both the continental shelf and in pelagic waters, and nonrandom, long-distance migrations between foraging and nesting grounds are typical (Eckert et al. 2012).

In the Gulf of Mexico, information regarding the distribution and abundance of leatherback sea turtles, which is summarized in the following paragraphs, is incomplete. However, adult leatherback distribution in the Gulf of Mexico—whether in deep-sea, pelagic waters, along the continental shelf, or in nearshore waters—has historically been associated with dense concentrations of jellyfish (Leary 1957; Fritts et al. 1983a, b; den Hartog and van Nierop 1984; Lohoefener et al. 1989; Eckert et al. 1989a, 2012; Houghton et al. 2006).

In the 1950s, Florida fishermen reported occasional sightings of leatherbacks off the coast of Sarasota and claimed they had been seen occasionally in the area since at least the 1930s (Yerger 1965). A leatherback carcass was found in Copano Bay, Texas in 1951 (Gunter 1951). In 1956, 100 leatherbacks were reported only 75 yards from the beach in Port Aransas, Texas. This group of leatherbacks appeared to be associated with a large abundance of jellyfish in the area (Leary 1957). Multiple sightings and captures of leatherbacks occurred along the Florida Gulf coast during the early 1960s (Yerger 1965). In fact, leatherbacks were once reported to be seasonally abundant off the coast of Panama City, Florida (Pritchard 1976). In 1975, leatherbacks were observed off the coast of Alabama (Mount 1975).

In 1979, aerial surveys of the Texas and Florida Gulf coasts reported more than 97 % of all turtle sightings to be in Florida, four of which were leatherbacks (Fritts and Reynolds 1981). Aerial surveys conducted in the Gulf of Mexico from May 1980 to April 1981 reported a total of 47 leatherbacks across all areas (except off the coast of south Texas), and found leatherbacks to be most common off the coast of Florida (Fritts et al. 1983a, b). During the survey, leatherbacks were more conspicuous on the continental shelf than in adjacent deeper waters (Fritts et al. 1983a, b).

Between 1988 and 1990, NMFS aerial surveys in the Gulf of Mexico reported infrequent sightings of leatherbacks, with most sightings occurring in July through November 1989 (Lohoefener et al. 1990). A survey of sea turtles in southeastern Louisiana in 1988 reported that the only evidence of leatherbacks in the area was one report from a diver in July 1988 (Fuller 1989). During the summer of 1989, six leatherback sightings were reported in south Louisiana, five of which were by divers most likely diving offshore at oil platforms (Fuller 1989). During a 1989 cruise in the Gulf of Mexico by the University of West Florida to the head of De Soto Canyon to collect neuston and *Sargassum*, eight leatherbacks were sighted near a coastal-subtropical water mass boundary region that contained high densities of jellyfish (Collard 1990).

Beginning in 1992, the U.S. Geological Survey performed a 3-year aerial and ship survey of the entire northern Gulf of Mexico. The study estimated an overall abundance of 168 leatherbacks in the continental slope area and also estimated leatherbacks to be 12 times more abundant in the winter than in the summer (Davis et al. 2000). Aerial surveys by the Southeast Fisheries Science Center during the fall of 1992, 1993, 1994, and 1996 sighted a total of three leatherbacks in the western Gulf of Mexico and eight in the eastern Gulf; only two of the turtles sighted were in water less than approximately 18 m (59 ft) deep (Epperly et al. 2002).

In CMR studies, the low recapture rate of leatherbacks, compared to other species of sea turtles, is likely due to their highly pelagic lifestyle and uncommon occurrence in coastal waters. However, captures and tag returns of leatherbacks have indicated that adult females use the Gulf of Mexico as a foraging ground. From 1970 to 1973, researchers in Suriname and French Guiana tagged more than 2,000 nesting female leatherbacks. Two of the turtles that were tagged on French Guiana beaches in 1970 and 1972, respectively, were recaptured in western Gulf of Mexico waters in 1973 (Pritchard 1976). Also, a nesting female tagged on the Costa Rican coast in 1985 was captured a year later by a shrimp fisherman off the Mississippi

Gulf coast (Hirth and Ogren 1987). In addition, tag returns from turtles tagged while nesting at index beaches in Costa Rica from 1976 to 2003 demonstrated that the northern Gulf of Mexico is a common location of dispersal for post-nesting females, as five of 21 tag returns were from this region (Troëng et al. 2004).

From 2003 through 2006, 12 adult female leatherbacks were satellite tracked from their nesting beaches at Tortuguero and Gandoca in Costa Rica and Chiriquí Beach in Panama (Evans et al. 2007). Of the four turtles that migrated to the Gulf of Mexico, three stayed within the eastern part of the Gulf off the Florida and Alabama coasts and the fourth leatherback stayed within the western Gulf. This research suggested that the Gulf of Mexico may represent a significant year-round foraging ground for leatherbacks from the Caribbean coast of Central America and not just a seasonal feeding area or pass-through region for migrating leatherbacks. For unknown reasons, jellyfish populations in the Gulf of Mexico have been increasing in recent years; it is possible that year-round foraging has increased as a response to increased jellyfish densities (Evans et al. 2007).

To assess the potential determinants of intra- and interpopulation variability in migratory patterns over the north and south Atlantic Ocean, the movements and diving behavior of 16 Atlantic leatherback turtles from different nesting sites (Chiriquí Beach in Panama, Samsambo Beach in Suriname, Awala-Yalimapo Beach in French Guiana, and Kinguere Beach in Gabon) and one foraging site (waters off Uruguay) were satellite tracked during their post-breeding migrations between 2005 and 2008 (Fossette et al. 2010). Two of the three turtles from Panama migrated to the Gulf of Mexico. After crossing the Caribbean Sea in 1 month, one turtle explored the eastern side of the Gulf, spending 2 months (September and October 2005) along the northeastern continental slope and 4 months (November 2005 through March 2006) south of the Loop Current (Fossette et al. 2010). The second turtle first moved toward the northern continental shelf of the Gulf of Mexico, and then traveled to the western and southwestern shelves from August through September 2006 toward an area between Veracruz and Yucatán, Mexico, where she remained for 6 months until March 2007. These turtles spent most of their time along the continental slope of the Gulf of Mexico, possibly foraging on gelatinous zooplankton aggregated along the front of the shelf break. By monitoring turtles from different nesting sites and one foraging area over the Atlantic Ocean, this study clearly illustrated that the general dispersal patterns and temporary residence areas used by the leatherback turtles may vary among individuals of the same nesting population and among populations (Fossette et al. 2010).

11.6 HAWKSBILL SEA TURTLE (*ERETMOCHELYS IMBRICATA*)

Linnaeus originally described the hawksbill in 1766 as *Testudo imbricata*; it was later transferred to its own genus, *Eretmochelys*, in 1843 by Fitzinger (Meylan and Redlow 2006). The specific name, *imbricata*, refers to the overlapping nature of the carapace scutes (Amorocho 2001). Unlike other sea turtles, the scutes of the hawksbill's beautiful shell or carapace are overlapping, and the rear edge of the carapace is almost always serrated (Figure 11.39) (NMFS and USFWS 1993; Meylan and Redlow 2006). The scutes are often richly patterned with irregularly radiating streaks of brown, black, orange, or red on an amber background. The small- to medium-sized hawksbill sea turtle is named for its strongly hooked beak.

Its beauty cursed the hawksbill sea turtle. As the sole source of commercial tortoiseshell, it has been exploited for centuries (Mortimer 2008). Tortoiseshell from the attractive carapace of the hawksbill can be used to make products, such as jewelry, combs, embellishments on furniture, and rims for eye glasses (Witzell 1983). Millions of hawksbills have been killed for

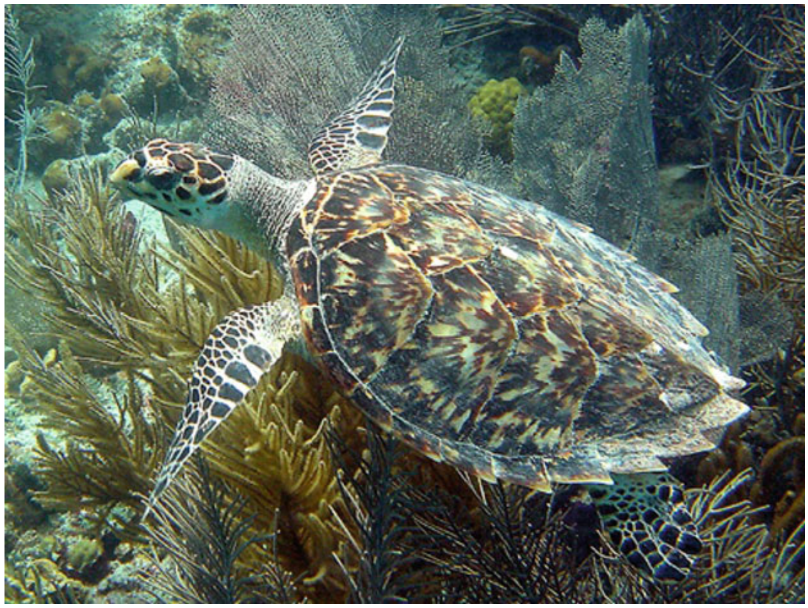

Figure 11.39. Hawksbill sea turtle using a coral reef (photograph by Caroline Rogers, USGS) (NOAA 2011).

the tortoiseshell markets of Asia, Europe, the Caribbean, and the USA over the past 100 years (NMFS and USFWS 2007e). Japan was historically the major importer of tortoiseshell or *bekko* from the Caribbean (NMFS and USFWS 1993). They agreed to stop importing bekko in 1993 (USFWS 2012). Although the volume of international trade has declined significantly in the past 20 years, it remains active, especially in Southeast Asia and the Americas (Mortimer 2008). In Southeast Asia, the extensive practice of selling whole, stuffed hawksbills is a relatively new threat (Mortimer 2008).

Hawksbills are the most tropical of the sea turtle species and typically nest at low densities throughout their range (NMFS and USFWS 1993, 2007e). In the past, hawksbill sea turtles were considered to be naturally rare and to have a more dispersed nesting pattern than the other sea turtle species (Groombridge and Luxmoore 1989). However, the dispersed pattern currently observed is now believed to be the result of overexploitation of previously large colonies (Meylan and Donnelly 1999). Hawksbill sea turtles are often associated with coral reefs (Meylan 1988; Meylan and Redlow 2006).

There are many gaps in the understanding of hawksbill sea turtle biology, and the oceanic phase of the post-hatchlings remains one of the most poorly understood aspects of hawksbill life history (NMFS and USFWS 2007e). Because of an almost total lack of long-term trend data at hawksbill foraging sites, nesting beach data are the primary information source used to evaluate trends in hawksbill populations. Few data are available on the at-sea mortality of hawksbills in fisheries (NMFS and USFWS 2007e).

11.6.1 Hawksbill Sea Turtle Life History, Distribution, and Abundance

Hawksbill sea turtles occur in the tropical and subtropical seas of the Atlantic, Pacific, and Indian oceans and are widely distributed in the Caribbean Sea and western Atlantic Ocean (Figure 11.40) (NMFS and USFWS 1993; USFWS 2012). They were once abundant in the tropical

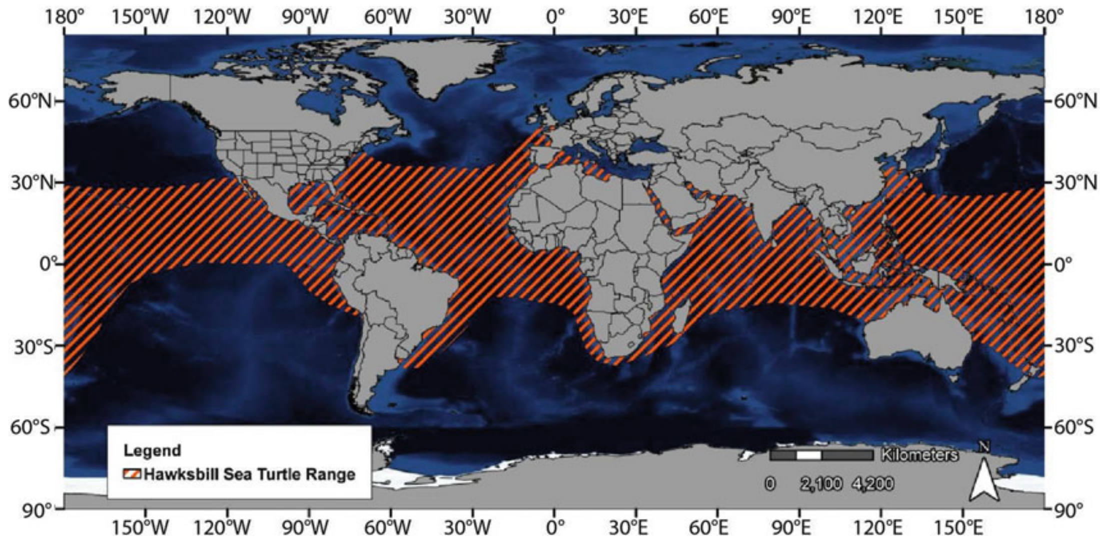

Figure 11.40. Range of the hawksbill sea turtle (from NOAA 2009d).

and subtropical regions of the world and now occur at only a fraction of their historical distribution (NMFS and USFWS 2007e). Unlike other sea turtle species, hawksbills nest in low densities on scattered, small beaches (USFWS 2012). Throughout their range, hawksbills nest in at least 70 countries on insular and mainland sandy beaches and inhabit coastal waters of more than 100 countries NMFS and USFWS (1993, 2007e). In 2007, at 83 nesting sites distributed among ten ocean regions around the world, approximately 21,212–28,138 hawksbills were estimated to nest each year (NMFS and USFWS 2007e; USFWS 2012).

Although greatly depleted compared to historical levels, nesting populations in the Atlantic Ocean are generally doing better than those in the Indo-Pacific (NMFS and USFWS 2007e). Hawksbill nesting in the Caribbean accounts for approximately 20–30 % of the world's hawksbill sea turtle population, and the number of hawksbill sea turtles living in the Caribbean was estimated at 27,000 in 2003 (Lutz et al. 2003; USFWS 2012). In the Atlantic, more population increases have been recorded in the Insular Caribbean, as compared to populations on the western Caribbean mainland or in the eastern Atlantic Ocean (NMFS and USFWS 2007e). Historically, Panama supported the single most important nesting population in the Caribbean, but hawksbill nesting at Chiriquí Beach in Panama has declined by more than 95 % during the past 50 years (Carr 1979; Meylan and Donnelly 1999). The Yucatán Peninsula in Mexico now supports the largest nesting hawksbill population in the area (Figure 11.41) (Cuevas et al. 2010).

The Insular Caribbean has eight nesting concentrations of hawksbills: Antigua/Barbuda (especially Jumby Bay), Bahamas, Barbados, Cuba (Doce Leguas Cays), Jamaica, Puerto Rico (especially Mona Island), Trinidad and Tobago, and the U.S. Virgin Islands (especially Buck Island Reef National Monument [BIRNM], an uninhabited island about 2.4 km [1.5 mi] north of the northeast coast of St. Croix) (Figure 11.41) (NMFS and USFWS 2007e). In 2002, 400–833 nesting females, or 2,000–2,500 nests, were estimated for Doce Leguas Cays, Cuba (NMFS and USFWS 2007e; Mortimer and Donnelly 2008; USFWS 2012). From 2001 through 2005, 199–332 nests were recorded each year on Mona Island, Puerto Rico, and 51–85 nests were recorded each season for Culebra Island, Caja de Muertos, and Humacao, Puerto Rico (NMFS and USFWS 2007e; USFWS 2012). From 2001 through 2006, an average of 56 nests were laid each year at BIRNM in the U.S. Virgin Islands, while 30–222 nests were laid in 2006 on U.S. Virgin Islands beaches outside BIRNM (NMFS and USFWS 2007e; USFWS 2012).

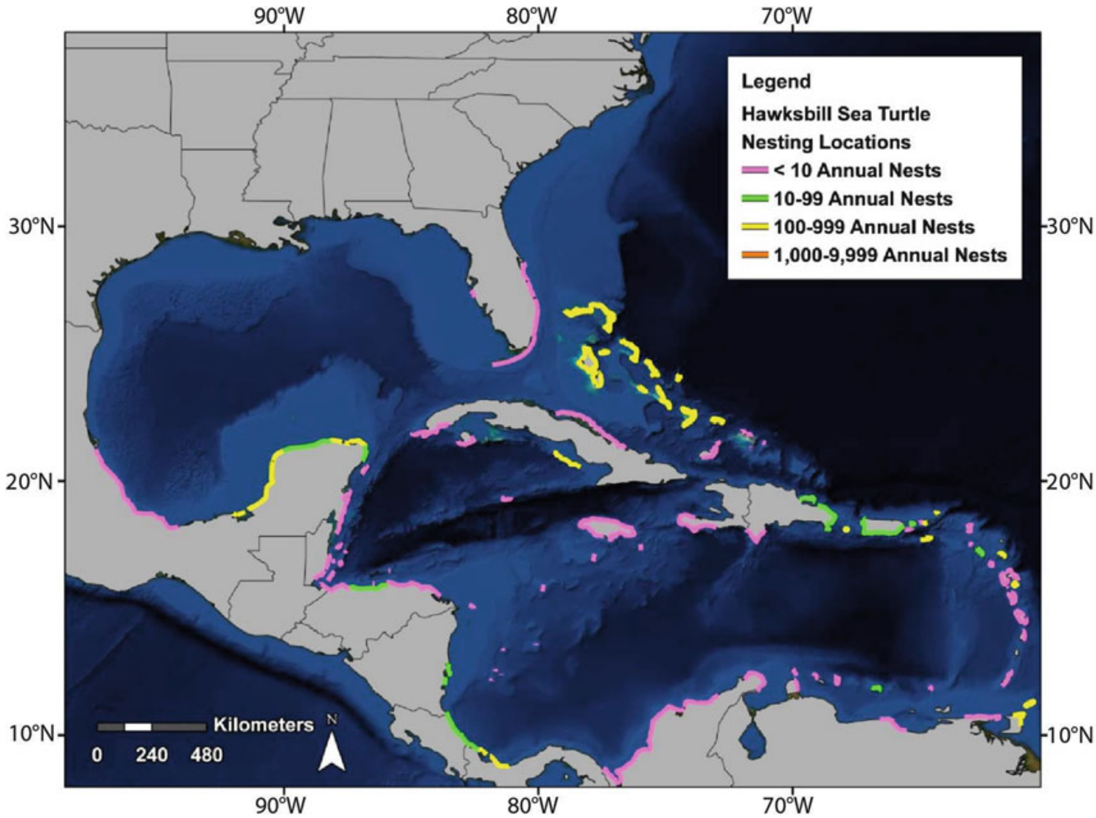

Figure 11.41. Generalized nesting beach locations of the hawksbill sea turtle in the Gulf of Mexico, Caribbean, and northwest Atlantic Ocean (interpreted from Dow et al. 2007; SWOT 2008).

11.6.1.1 Nesting Life History, Distribution, and Abundance for Gulf of Mexico Hawksbills

In the Gulf of Mexico, hawksbill nesting occurs mostly on Yucatán Peninsula beaches in the southern Gulf (Figure 11.41), and these beaches are also the location of the most important hawksbill rookery in the Atlantic (Cuevas et al. 2010). Hatchlings from this rookery are likely to be carried by the current through the Yucatán Channel north into the Gulf of Mexico (Blumenthal et al. 2009). For example, a genetic contribution of 93 % from Yucatán Peninsula nesting beaches was estimated for a group of 42 juvenile hawksbills, ranging in size from 5.2 to 36.8 cm (2 to 14.5 in) SCL, that were stranded on Texas beaches (Bowen et al. 2007). In addition, strandings of juvenile hawksbills along the Florida Gulf coast, which commonly occur, are most likely from this rookery (Meylan and Redlow 2006). Life history information for the hawksbill sea turtle is summarized in Table 11.5, including all available information for specific Gulf of Mexico beaches or regions.

The hawksbill nesting population of the Yucatán Peninsula was declining until 1978 when local and regional protection was implemented (Meylan and Donnelly 1999). As a result of an analysis of long-term trends of hawksbill nesting from 1977 through 1996 on Yucatán Peninsula beaches, the steady improvement in monitoring effort was determined to be the major explanation for the gradual increase in nesting from 1977 to 1992 (Garduño-Andrade et al. 1999). However, the average annual increase of 270 nests per year from 1992 through

Table 11.5. Summary of Life History Information for the Hawksbill Sea Turtle

Parameter	Values	References
Nesting season: Yucatán Peninsula, Mexico	April through September	Cuevas et al. (2010)
Remigration interval		
Yucatán Peninsula, Mexico	Range: 2–3 years	Garduño-Andrade (1999)
Isla Aguada, Campeche, Yucatán Peninsula, Mexico	Range: 2–4 years	Guzmán-Hernández et al. (2006)
Nesting interval		
El Cuyo, Yucatán, Yucatán Peninsula, Mexico	Mean: 18 days	Xavier et al. (2006)
Isla Aguada, Campeche, Yucatán Peninsula, Mexico	Mean: 14 days, Range: 11–15 days	Guzmán-Hernández et al. (2006)
Laguna de Términos, Campeche, Yucatán Peninsula, Mexico	Range: 14–16 days	Guzmán-Hernández and García-Alvarado (2010), Amorocho (2001)
Yucatán Peninsula, Mexico	Mean: 21 days (Isla Contoy, Quintana Roo)	Najera (1990)
	Mean: 21 days (Isla Holbox, Quintana Roo)	
	Mean: 23 days (Rio Lagartos, Yucatán)	
Number of nests/season		
Las Coloradas, Quintana Roo, Yucatán Peninsula, Mexico	Mean: 2.1 nests	Garduño (1998)
El Cuyo, Yucatán, Yucatán Peninsula, Mexico	Mean: 2.3 nests	Xavier et al. (2006)
Campeche, Yucatán Peninsula, Mexico	Mean: 3.1 nests	Guzmán et al. (1996)
Isla Aguada, Campeche, Yucatán Peninsula, Mexico	Mean: 2.8 nests, Range: 2.5–3.2 nests	Guzmán-Hernández et al. (2006)
Laguna de Términos, Campeche, Yucatán Peninsula, Mexico	Mean: 3 nests	Guzmán-Hernández and García-Alvarado (2010)
Number of eggs/nest		
El Cuyo, Yucatán, Yucatán Peninsula, Mexico	Mean: 149 eggs	Xavier et al. (2006)
Las Coloradas, Quintana Roo, Yucatán Peninsula, Mexico	Mean: 157 eggs	Garduño-Andrade (2000)
Isla Aguada, Campeche, Yucatán Peninsula, Mexico	Mean: 137 eggs	Frazier (1993)
Isla del Carmen, Chenkan, and Isla Aguada beaches, Campeche, Yucatán Peninsula, Mexico	Range: 96–183 eggs	Cuevas et al. (2008)
Yucatán Peninsula, Mexico	Mean: 148 eggs	Echeverría-García and Torres-Burgos (2007)
	Mean: 159 eggs (Telchac Puerto, Yucatán)	Echeverría-García et al. (2008)
	Mean: 161 eggs (Sisal, Yucatán)	
	Mean: 140 eggs, Range: 46–244 eggs	Frazier (1993)

(continued)

Table 11.5. (continued)

Parameter	Values	References
Yucatán Peninsula, Mexico	Mean: 149 eggs, Range: 47–194 eggs (Isla Contoy, Quintana Roo)	Najera (1990)
	Mean: 152 eggs, Range: 100–188 eggs (Isla Holbox, Quintana Roo)	
	Mean: 153 eggs, Range: 19–229 eggs (Rio Lagartos, Yucatán)	
	Mean: 140 eggs, Range: 60–247 eggs (Celestun, Yucatán)	Pérez-Castañeda et al. (2007)
	Mean: 142 eggs, Range: 60–257 eggs (Isla Holbox, Quintana Roo)	
	Mean: 145 eggs, Range: 62–241 eggs (El Cuyo, Quintana Roo)	
Egg incubation time		
Isla Aguada, Campeche, Yucatán Peninsula, Mexico	Mean: 57 days, Range: 51–64 days	Guzmán-Hernández et al. (2006)
Yucatán Peninsula, Mexico	Mean: 62 days, Range: 51–83 days (Celestun, Yucatán)	Pérez-Castañeda et al. (2007)
	Mean: 63 days, Range: 50–80 days (El Cuyo, Quintana Roo)	
	Mean: 65 days, Range: 50–80 days (Isla Holbox, Quintana Roo)	
Nest pivotal temperature		
Antigua, West Indies	29.3 °C	Hulin et al. (2009)
Bahia, Brazil	29.6 °C	Godfrey et al. (1999)
Sex ratio of hatchlings from nests (proportional female)		
Mona Island, Puerto Rico	Mean: 0.44	Diez and van Dam (2003)
Bahia, Brazil	Range: 0.91–1.0	Godfrey et al. (1999)
Emergence success of hatchlings from nests		
El Cuyo, Yucatán, Yucatán Peninsula, Mexico	Mean: 0.81	Xavier et al. (2006)
Yucatán, Yucatán Peninsula, Mexico	Mean: 0.76	Echeverría-García and Torres-Burgos (2007)
	Mean: 0.83 (Sisal, Yucatán)	Echeverría-García et al. (2008)
	Mean: 0.86 (Telchac Puerto, Yucatán)	
	Mean: 0.59 (Dzilam de Bravo, Yucatán)	Echeverría-García et al. (2009)
	Mean: 0.67 (Telchac Puerto, Yucatán)	
	Mean: 0.68 (Sisal, Yucatán)	
Yucatán Peninsula, Mexico	Mean: 0.82 (El Cuyo, Quintana Roo)	Pérez-Castañeda et al. (2007)
	Mean: 0.85 (Celestun, Yucatán)	
	Mean: 0.88 (Isla Holbox, Quintana Roo)	

(continued)

Table 11.5. (continued)

Parameter	Values	References
Size of hatchlings		
Tortuguero, Costa Rica	Mean: 4.24 cm SCL[a], Range: 3.91–4.60 cm SCL	Carr and Ogren (1966)
Wider Caribbean Region	Mean: 4.20 cm SCL, Range: 3.90–4.60 cm SCL	Amorocho (2001)
Mustang Island, Texas	Range: 5–21 cm SCL	Carr (1987)
Diet of hatchlings		
Caribbean	*Sargassum*, manatee grass, crab chela, eggs of flying fish, half-beaks, and needlefish	Meylan (1984)
Florida	Sargassum	Meylan and Redlow (2006)
Size of oceanic juveniles		
Rio Lagartos Sea Turtle Sanctuary, Yucatán Peninsula, Mexico	Range: 20–30 cm SCL	Cuevas et al. (2007)
Padre Island National Seashore, Mustang Island, and Port Aransas, Texas	Range: 20.1–29.1 cm SCL[b]	Amos (1989)
East and west coast of Florida	Mean: 20.6 cm SCL, Range: 13.4–24.8 cm SCL	Witherington et al. (2012)
Diet of oceanic juveniles		
Caribbean	*Sargassum,* manatee grass, crab chela, eggs of flying fish, half-beaks, and needlefish	Meylan (1984)
Florida	Sargassum	Meylan and Redlow (2006)
Size of oceanic juveniles at recruitment to neritic juvenile stage		
Rio Lagartos, Las Colorados, Quintana Roo, Yucatán Peninsula, Mexico	Range: 20–65 cm SCL	R. Pérez-Castañeda, Universidad Autonoma Tamaulipas, unpublished data, cited in Garduño-Andrade et al. (1999)
Size of oceanic juveniles at recruitment to neritic juvenile stage		
Florida Keys	Range: 21.4–69 cm SCL	M. Bressette, Inwater Research Group, unpublished data, cited in Witherington et al. (2012)
Broward County to St. Lucie Nuclear Plant, Florida	Range: 25.7–34 cm SCL	M. Bressette and R. Wershoven, Quantum Resources, Inc. and Broward County Audubon Society, personal communication, cited in Meylan and Redlow (2006)
Diet of neritic juveniles: Rio Lagartos Sea Turtle Sanctuary, Quintana Roo, Yucatán Peninsula, Mexico	Sponges, including *Chondrilla* sp., *Dictyopteris* sp., *Hypnea* sp., *Jania* sp., *Laurencia* sp., *Ceramium* sp., *Codium* sp., and *Gracilaria* sp.	Cuevas et al. (2007)

(continued)

Table 11.5. (continued)

Parameter	Values	References
Age at sexual maturity		
Las Coloradas, Quintana Roo, Yucatán Peninsula, Mexico	Mean: 24 years, Minimum: 14 years	Garduño (1998)
Yucatán Peninsula, Mexico	Mean: 31.2 years	IUCN (2012)
Size of sexually mature adult females		
Las Coloradas, Quintana Roo, Yucatán Peninsula, Mexico	Mean: 90 cm SCL, Minimum: 80 cm SCL	Garduño (1998)
El Cuyo, Yucatán, Yucatán Peninsula, Mexico	Mean: 94.4 cm SCL[b]	Xavier et al. (2006)
Isla Aguada, Campeche, Yucatán Peninsula, Mexico	Mean: 92.1 cm SCL[b], Range: 85.7–98.6 cm SCL	Guzmán-Hernández et al. (2006)
Isla del Carmen, Chenkan, and Isla Aguada beaches, Campeche, Yucatán Peninsula, Mexico	Range: 82.7–95.6 cm SCL[b]	Cuevas et al. (2008)
Yucatán, Yucatán Peninsula, Mexico	Mean: 93.1 cm SCL (Telchac Puerto, Yucatán), Mean: 96.5 cm SCL (Sisal, Yucatán)	Echeverría-García et al. (2008)
	Mean: 92.7 cm SCL (Sisal, Yucatán)	Echeverría-García et al. (2009)
	Mean: 99.6 cm SCL (Telchac Puerto, Yucatán)	
Yucatán Peninsula, Mexico	Mean: 94 cm SCL[b], Range of modes: 94.6–98.6 cm SCL (Celestun, Yucatán);	Pérez-Castañeda et al. (2007)
	Mean: 93.7 cm SCL[b], Range of modes: 94.6–98.6 cm SCL (El Cuyo, Quintana Roo)	
	Mean: 94.3 cm SCL[b], Range of modes: 89.6–93.6 cm SCL (Isla Holbox, Quintana Roo)	
Diet of adults		
Caribbean	Sponges, demosponges, and button polyp, *Ricordea florida*	Meylan (1984)
	Sponges, including chicken liver sponge, *Ancorina* sp., *Geodia* sp., *Placospongia* sp., *Suberites* sp., *Myriastra* sp., *Ecionemia* sp., *Chondrosia* sp., *Aaptos* sp., and *Tethya actinia*	Meylan (1988)

[a] *SCL* straight carapace length, *cm* centimeters
[b] To convert from curved carapace length (*CCL*): SCL = (0.9927 × CCL) + 0.2782 (Garduño 1998)

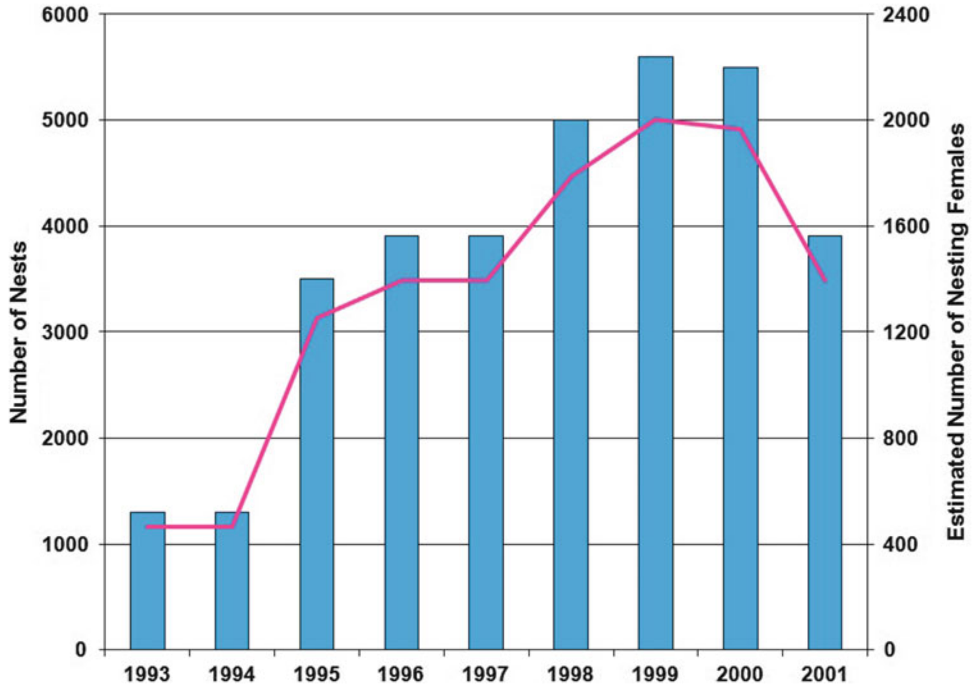

Figure 11.42. Annual number of nests (*bars*) and estimated number of nesting females (*line*), assuming 2.8 nests per female (Guzmán-Hernández et al. 2006), for the hawksbill sea turtle for selected Mexican Gulf of Mexico beaches along the coasts of Tamaulipas, Veracruz, Campeche, Yucatán, and Quintana Roo from 1993 through 2001 (Márquez-M. 2004).

1996 was considered to be indicative of real population increases, since beach coverage had peaked, with consistent monitoring efforts each year. In 1996, 4,522 nests were reported, which was equivalent of up to 2,200 nesting females; the Yucatán Peninsula hawksbill nesting population was estimated to range from 1,900 to 4,300 turtles during 1996, the largest in the western Atlantic Ocean. Most nesting in 1996 occurred on Campeche beaches; average nesting densities for the 16 monitored beaches ranged from 0.8 to 64 nests/km (0.5 to 40 nests/mi) (mean = 16.7 nests/km [10.4 nests/mi]). The increase in the nesting population after 1993 was thought to be a result of increased survival rates of juveniles and adults due to regional conservation measures and increased recruitment into the breeding stock from protected Yucatán Peninsula beaches (Garduño-Andrade et al. 1999). Similar trends in hawksbill nesting were observed along the entire Mexican Gulf coast (Figure 11.42).

In 1999, more than 6,000 nests were recorded on Yucatán Peninsula beaches (Garduño-Andrade et al. 1999; Cuevas et al. 2008). However, nesting numbers declined by 63 % from 1999 through 2004. The cause of this decline in nesting numbers was suspected to be the taking of turtles and/or impacts to the hawksbill's marine habitats (Abreu-Grobois et al. 2005). At three Yucatán Peninsula beaches—Celestun, El Cuyo, and Isla Holbox—hawksbill nesting decreased significantly from 2000 to 2001 (Pérez-Castañeda et al. 2007). Nesting numbers on Yucátan Peninsula beaches have increased since 2004 but are still below 1999 numbers (NMFS and USFWS 2007e; del Monte-Luna et al. 2012). From 2001 through 2006, 2,672 nests or 534–891 nesting females were recorded in the Yucatán Peninsula each year (NMFS and USFWS 2007e; Mortimer and Donnelly 2008; USFWS 2012).

A study was conducted on El Cuyo Beach to evaluate hawksbill nesting activity from 2002 through 2004 (Xavier et al. 2006). This beach is one of the most important hawksbill nesting beaches on the Yucatán Peninsula. During the 3-year study period, hawksbill nesting decreased by 40 %; there were 373 nests in 2002, 311 in 2003, and 217 nests in 2004. Hawksbill nesting on this beach averaged 659 nests/km (409 nests/mi) in 1996 (Garduño-Andrade et al. 1999) and decreased to about 300 nests between 1999 and 2001 (Salum-Fares 2003). Between nesting seasons, there was no difference in the size of nesting females. High site fidelity was demonstrated by the nesting hawksbills, with an average distance of 3 km (1.9 mi) between nests. Predation has risen on El Cuyo Beach, which has affected hawksbill nests (Xavier et al. 2006). El Cuyo Beach is also an important nesting beach for green turtles. Compared to green turtles, hawksbills had a wider nesting distribution on the beach and seemed to have a wider range of preferences for beach morphological features (Cuevas et al. 2010).

Hawksbill nesting data from the southern Gulf of Mexico from 1980 through 2010 were recently evaluated by del Monte-Luna et al. (2012) to determine the cause of the long-term decline. Since nesting hawksbills along the Campeche coast can reasonably be considered as representative of the entire nesting population of the Yucatán Peninsula, hawksbill nesting data from Isla Aguada, Savancuy, Chencan, and Punta Xen, Campeche, Mexico, were analyzed. A 7-year cycle in annual relative number of nesting turtles in the southern Gulf of Mexico was found, which was inversely correlated with cycles of similar periodicity in the North Atlantic sea surface temperature. Long-term population dynamics in the southern Gulf were related to a basin-wide, quasi-decadal temperature fluctuation in the North Atlantic. Other threats that also may have contributed to the long-term decline of nesting hawksbills in the southern Gulf of Mexico include increased hurricane activity in the Caribbean, regional sea-level rise, and the constant expansion of beach development (del Monte-Luna et al. 2012).

In addition to the rookery in the Yucatán Peninsula, hawksbill sea turtles lay a small number of nests each year along the Gulf coasts of Mexico and Florida (Figure 11.41). In Florida, hawksbill nesting also occurs along the east coast from Volusia through Miami-Dade counties, including Soldier Key in Biscayne Bay and the Florida Keys (Figure 11.41) (Dalrymple et al. 1985; Meylan 1992). From 1979 through 2003, 31 nests were documented in Florida and were distributed along the Atlantic coast from Volusia County to Monroe County, with a single record on the Gulf coast in Manatee County and a maximum of four nests recorded in any year (Meylan and Redlow 2006). Hawksbill nesting on both the Florida Atlantic and Gulf coasts is most likely underestimated for the following reasons: (1) beaches in areas known to be used by hawksbills are incompletely surveyed (e.g., Florida Keys), (2) beach surveys are typically not conducted during the fall months, (3) hawksbill and loggerhead sea turtle tracks are similar, (4) hawksbills nest in or under vegetation and sometimes on narrow beaches, and (5) hawksbill and loggerhead hatchlings are similar in appearance (Meylan and Redlow 2006). Low levels of hawksbill nesting are also suspected to occur in the Marquesas and Dry Tortugas (NMFS and USFWS 1993). In 1998, the first hawksbill nest was recorded on the Texas coast at PAIS (Mays and Shaver 1998).

The nesting season of hawksbills is longer than that of other sea turtle species (Table 11.5), and the small, agile females have the ability to climb over reefs and rocks to nest in beach vegetation (Figure 11.43) (NMFS and USFWS 1993; USFWS 2012). Tagging and genetic studies have demonstrated that female hawksbills have strong site fidelity and return to nest in the vicinity where they hatched (Witzell 1983; Bass 1999). Although rare daytime nesting is known, hawksbill sea turtles typically nest at night (Meylan and Redlow 2006).

Figure 11.43. Hawksbill sea turtle returning to the sea (from Scarygami 2012).

11.6.1.2 Hatchling, Post-Hatchling, and Oceanic Juvenile Life History and Distribution for Gulf of Mexico Hawksbills

Similar to other species of sea turtles, hatchling hawksbills enter an oceanic phase (Figure 11.44). This phase may involve long-distance travel carried by surface gyres, with eventual recruitment to neritic foraging habitat (NMFS and USFWS 2007e). Hatchlings entrained in the Loop Current could be expected to remain in the Gulf of Mexico for differing periods of time, depending on which branch of the Loop Current they enter (Meylan and Redlow 2006). Both newly hatched and early juvenile hawksbills have been found in association with *Sargassum* and floating weed in the Atlantic and Caribbean (Table 11.5) (Carr 1987; Mellgren and Mann 1996; Musick and Limpus 1997; Meylan and Redlow 2006). No post-hatchlings but six juvenile hawksbill sea turtles, with an average size of 20.6 cm (8.1 in) SCL, were captured from the *Sargassum*-dominated, surface-pelagic drift community in the eastern Gulf of Mexico and Atlantic Ocean from 2005 through 2011; some of these hawksbills were large enough to be on the cusp of recruitment into the neritic zone, most likely to foraging habitat in the Florida Keys (Witherington et al. 2012). Weedlines in the Gulf of Mexico likely serve as habitat for post-hatchling hawksbills from nesting beaches in Mexico and Central America (NMFS and USFWS 1993).

Between 1972 and 1984, 77 strandings of post-hatchling and juvenile hawksbills were recorded in Texas, with most occurring near Corpus Christi; these turtles most likely originated from nesting beaches on the Yucatán Peninsula in Mexico (Amos 1989; Bowen et al. 2007). Limited tagging data indicates that some post-hatchling hawksbills from the western Gulf of Mexico disperse into the Atlantic Ocean, most likely through the Florida Straits, and move northward along Florida’s east coast (Meylan and Redlow 2006).

Figure 11.44. Hawksbill sea turtle hatchling moving across the beach toward the sea (from Serge_Vero 2007).

11.6.1.3 Neritic Juvenile Life History and Distribution for Gulf of Mexico Hawksbills

Oceanic juvenile hawksbills recruit to foraging habitat in the neritic zone starting at around 20 cm (7.9 in) SCL (Table 11.5). The origin of juveniles found in neritic foraging areas is related to nesting population size, geographic distance from the nesting areas, and ocean currents. Juveniles typically occupy a series of habitats as they increase in size, with larger turtles often inhabiting deeper sites (Bowen et al. 2007). Large juveniles may be associated with the same feeding location for more than 10 years (Musick and Limpus 1997). Neritic juvenile hawksbills may occupy a range of habitats, including coral reefs, rocky areas, other hard bottom habitats, seagrass and algae beds, shallow coastal areas, lagoons and oceanic islands, narrow creeks, and mangrove bays and are rarely found in water deeper than about 20 m (66 ft) (Musick and Limpus 1997; USFWS 2012). Throughout their range, neritic juvenile hawksbill sea turtles typically feed on sponges (Table 11.5). However, hawksbills are not always mainly spongivorous; for example, in a recent long-term study in the northern Great Barrier Reef, Australia, hawksbills fed mainly on algae (*Laurencia* sp. and *Gelidiella* sp.; 72 % of ingesta), along with some sponges (10 %), soft corals, and other prey (12 %) (Bell 2012).

Cuevas et al. (2007) evaluated the benthic foraging habitat of juvenile hawksbills in the Rio Lagartos Sea Turtle Sanctuary in Yucatán, Mexico, an important feeding and development area for juvenile hawksbill sea turtles. Hawksbills were found to be distributed mainly on hard bottom sites covered by octocorals, such as *Pseudopterogorgia*, and sponges of the genera *Chondrilla* and *Spheciospongia*. Based on tracking data, the average home range of the turtles was larger during the day (0.123 km^2 [0.048 mi^2]) than that used at night (0.021 km^2 [0.008 mi^2]). In addition, there were differences in habitat preferences between day and night. During the day, hawksbills mainly occupied habitats with 20–40 % octocoral cover; at night they tended to occupy bare substratum areas (Cuevas et al. 2007).

Along the U.S. Gulf Coast, juvenile hawksbills are associated with stone jetties in Texas (Amos 1989). Hawksbills are rare along the north Florida Gulf coast; juveniles are more abundant in Gulf of Mexico waters off west-central Florida than anywhere else along the Florida Gulf coast (Meylan and Redlow 2006). Hard-bottom communities on the west Florida Shelf, the southern Pulley Ridge, and the Florida Middle Ground reef complex represent potential hawksbill foraging habitat in the Gulf of Mexico off the Florida west coast; the distribution of post-pelagic hawksbills corresponds closely to the Florida Reef Tract (Meylan and Redlow 2006). During surveys conducted at KWNWR from 2002 through 2006, almost 60 hawksbills were sighted, and 19 juveniles, ranging in size from 28.2 to 69 cm (11.1 to 27.2 in) SCL and averaging 46.4 cm (18.3 in) SCL, were captured (Eaton et al. 2008).

There is little to no evidence that hawksbills use Florida's major Gulf coast estuaries, such as Tampa Bay and Charlotte Harbor (Meylan and Redlow 2006). Neritic juvenile hawksbills were captured during a study conducted from 1997 through 2004 in the nearshore waters of Gullivan Bay, in the Ten Thousand Islands off the southwest Florida Gulf coast (Witzell and Schmid 2004; Eaton et al. 2008). Three juvenile hawksbills, averaging 49.8 cm (19.6 in) SCL in size (range of 38.2–58.1 cm [15–23 in] SCL) were captured during surveys conducted in Florida Bay from 2000 through 2006 (Eaton et al. 2008).

Three large juvenile hawksbills, ranging in size from 51.9 to 69.8 cm (20.4 to 27.5 in) SCL (mean = 61.5 cm or 24.2 in SCL), were captured within DTNP in the Gulf and tracked from August 2008 through January 2011 to determine patterns of habitat use (Hart et al. 2012b). Core use areas within the park ranged from 9.2 to 21.5 km^2 (3.6 to 8.3 mi^2) and were concentrated around the flats surrounded by Garden Key, Bush Key, and Long Key. The turtles were more active during the day than at night, which could indicate active foraging during the day and resting behavior at night. After between 263 and 699 days residing within the park, two turtles migrated to Cuba, while the third hawksbill migrated toward Key West, Florida. The turtles that migrated to Cuba ceased transmitting after 320 and 687 days, while the turtle that migrated toward Key West stopped transmitting after 884 tracking days. This study highlighted unknown regional connections for hawksbills, possible turtle harvest incidents, and fine-scale habitat use of juvenile turtles (Hart et al. 2012b).

11.6.1.4 Adult Life History and Distribution for Gulf of Mexico Hawksbills

Most of the adult life history information available for hawksbills in the Gulf of Mexico is for the Yucatán Peninsula in Mexico (Table 11.5). Most hawksbills have slow growth rates, which vary within and among populations (NMFS and USFWS 2007e); growth rates of 2–4 cm (0.8–1.6 in) per year are typical for the Caribbean (Boulon 1994). Adult hawksbills may reach up to 1 m (3.3 ft) in length and weigh up to 140 kg (308.6 lb); however, they typically average about 0.75 m (2.5 ft) in length and weigh around 80 kg (176.4 lb) or less (USFWS 2012).

Recent satellite tracking studies conducted to determine the migratory patterns and feeding ground locations for hawksbills nesting on Yucatán Peninsula beaches have indicated that turtles remain in Mexican waters (Cuevas et al. 2008, 2012). In 2006 and 2007, three post-nesting females were tracked for up to 510 days from three of the major nesting beaches in Campeche; two migrated to foraging grounds off the coast of Campeche, and one migrated to the Mexican Caribbean (Cuevas et al. 2008). Ten post-nesting hawksbill sea turtles were tracked from nine different nesting beaches on the Yucatán Peninsula in 2006 and 2007; turtles that nested on the western side of the peninsula migrated to the east, while those that nested on the eastern side migrated to the west (Cuevas et al. 2012). In a second, similar study conducted in 2011, tracked hawksbills also stayed in Mexican waters; however, one female migrated north to the border of the continental shelf, while a male stayed close to the nesting beaches (Cuevas et al. 2012).

While hawksbills are not encountered in the Gulf of Mexico as frequently as some of the other species of sea turtles (Thompson et al. 1990), they regularly occur in U.S. Gulf of Mexico waters off the southern Florida coast and in the northern Gulf, especially in Texas coastal waters (NMFS and USFWS 1993). Hawksbills have been recorded in waters of all U.S. Gulf Coast states and are regularly observed in the Florida Keys (Lund 1985; NMFS and USFWS 1993; Meylan and Redlow 2006). The distribution and abundance of hawksbill sea turtles in the Florida Keys is largely unknown, and few studies have been conducted to document their distribution and abundance; however, the Florida Keys National Marine Sanctuary, KWNWR, and DTNP contain important hawksbill habitat (Meylan and Redlow 2006).

Adult hawksbills are often associated with coral reefs, where they typically forage on a limited number of sponge species (Table 11.5); however, as already mentioned, hawksbills are not always mainly spongivorous (Bell 2012). The ledges and caves of coral reefs provide important shelter for resting hawksbills both during the day and night (NMFS and USFWS 1993). Similar to other species of sea turtles, hawksbills are integral components of marine and coastal food webs (Bouchard and Bjorndal 2000). In addition, because they eat sponges, they help keep coral reefs healthy (Bjorndal and Jackson 2003).

11.7 THREATS TO GULF OF MEXICO SEA TURTLE POPULATIONS

Many anthropogenic and natural threats affect all ecosystem zones used by sea turtle populations in the Gulf of Mexico (Table 11.6). These threats, which occur either in the Gulf of Mexico or within the distribution range of the sea turtle species that occur in the Gulf, are discussed below. In addition, if available, examples of quantified impacts associated with some of these threats for each of the sea turtle species that occurs in the Gulf of Mexico are presented.

11.7.1 Incidental Capture of Sea Turtles in Commercial and Recreational Fisheries

In 1990, the incidental capture of sea turtles in shrimp trawls was identified by the U.S. National Academy of Sciences as the major cause of turtle mortality associated with human activities; in fact, this incidental capture was determined to kill more sea turtles than all other human activities combined (Magnuson et al. 1990). In addition, most of the sea turtles that are killed in shrimp trawls are neritic juveniles—the life stage most critical to the stability and recovery of sea turtle populations (Crouse et al. 1987; Crowder et al. 1994). In the first cumulative estimates of sea turtle bycatch across fisheries of the United States between 1990 and 2007, the southeast U.S./Gulf of Mexico shrimp trawl fishery was estimated to be responsible for up to 98 % of all sea turtle interactions and for more than 80 % of all sea turtle mortality. However, due to the lack of observer coverage, estimates of bycatch for this fishery are highly uncertain (Epperly et al. 2002; Finkbeiner et al. 2011). The Gulf of Mexico portion of the fishery was estimated to comprise 73 % of total interactions and 96 % of mortality; the shrimp trawl fishery was estimated to account for about 69,300 lethal takes of sea turtles before the 2003 TED enlargement requirements, and approximately 3,700 mortalities following the TED enlargement requirements and the reduction of fishing effort in the Gulf of Mexico (Finkbeiner et al. 2011).

By the late 1970s, before TEDs were developed to prevent turtles from entering the back of shrimp trawl nets and provide for escape (Figure 11.45), the only major nesting population of Kemp's ridleys was close to extinction (Henwood et al. 1992; Frazier et al. 2007). The NMFS

Table 11.6. Summary of Anthropogenic and Natural Threats Affecting the Various Ecosystem Zones used by Sea Turtle Populations in the Gulf of Mexico (NMFS and USFWS 2008; Bolten et al. 2011; NMFS et al. 2011)

Threat	Terrestrial Zone[a]	Neritic Zone[a]	Oceanic Zone[a]
Incidental capture in commercial and recreational fisheries			
Trawls		X	X
Gill nets		X	X
Dredges		X	X
Pelagic and bottom long lines		X	X
Seines		X	
Pound nets and weirs		X	
Pots and traps		X	
Hook and line		X	X
Illegal harvest			
Eggs	X		
Juveniles		X	
Adults	X	X	
Nesting beach alterations			
Cleaning	X		
Human presence	X		
Driving on beach (cars and off-road vehicles)	X		
Artificial lighting	X	X	
Construction	X		
Nourishment and restoration	X	X	
Sand mining	X	X	
Armoring and shoreline stabilization (drift fences, groins, jetties)	X		
Other anthropogenic impacts			
Channel dredging and bridge building		X	
Boat strikes		X	X
Oil and gas exploration (including seismic activity), development, and production	X	X	X
Stormwater runoff		X	X
Oil and chemical pollution and toxins	X	X	X
Algal blooms, including red tides		X	
Hypoxia		X	

(continued)

Table 11.6. (continued)

Threat	Terrestrial Zone[a]	Neritic Zone[a]	Oceanic Zone[a]
Marine debris ingestion and entanglement	X	X	X
Military activities and noise pollution	X	X	X
Industrial and power plant intake, impingement, and entrainment		X	
Dams and water diversion		X	
Sea level rise due to climate change	X		
Temperature change due to climate change	X	X	X
Trophic changes due to fishing and benthic habitat alteration		X	X
Natural impacts			
Predation	X	X	X
Beach erosion and vegetation alteration	X		
Habitat modification by invasive species	X	X	
Pathogens and disease	X	X	X
Hurricanes and severe storms	X	X	
Droughts		X	
Hypothermic stunning		X	

[a]Terrestrial zone = Nesting beach where females excavate nests and lay eggs, where embryos develop; Neritic zone = inshore marine environment from the surface to the seafloor, including bays, sounds, and estuaries, as well as the continental shelf, where water depths do not exceed 200 m (656.2 ft); and Oceanic zone = open ocean environment from the surface to the seafloor where water depths are greater than 200 m (656.2 ft)

published final regulations requiring TEDs in shrimp trawlers in June 1987; however, implementation was delayed as a result of legal and congressional action (NMFS and USFWS 2008). By the early 1990s, TEDs were required at all times of the year in all U.S. waters where the southeast U.S. shrimp fishery operated (Epperly 2003). Turtle excluder devices were also required in Mexican waters beginning in 1995 (Crowder and Heppell 2011).

An evaluation of monthly sea turtle stranding data and shrimp fishing effort from 1986 through 1989 for the northwestern Gulf of Mexico demonstrated a significant relationship: turtle strandings increased as fishing effort increased in waters landward of 15 fathoms or 9.1 m (29.9 ft) (Caillouet et al. 1991). Despite the requirement of TEDs beginning in 1990, there was no change in the relationship of monthly sea turtle stranding rates and monthly shrimp fishing intensities in the northwestern Gulf of Mexico when data collected in 1986 through 1989 and 1990 through 1993 were compared (Caillouet et al. 1996). This lack of change in sea turtle stranding rates indicated that the problem of sea turtle mortality at sea had not been solved and

Figure 11.45. Loggerhead sea turtle escaping a net equipped with a turtle excluder device (TED) (from NOAA 2011).

that further efforts were necessary. An analysis by Epperly and Teas (2002) indicated that the minimum openings of TEDs were too small to exclude large leatherback, loggerhead, and green sea turtles, but they were effective at excluding Kemp's ridleys and juvenile loggerheads. Therefore, the NMFS enacted new regulations in 2003, which required that TED openings be large enough to allow all sea turtles to escape. While TED regulations already in place were likely effective for reducing fishery-induced mortality of smaller turtles (e.g., Kemp's ridley), 2003 was considered the beginning of effective reduction of sea turtle bycatch for the shrimp trawl fishery in the Gulf of Mexico, especially for loggerheads and leatherbacks (Finkbeiner et al. 2011). Among the Gulf States, the State of Louisiana stands out because in the late 1980s, it enacted legislation prohibiting state authorities from enforcing the federal law requiring the use of TEDs by the shrimp fisheries in State waters (Louisiana Revised Statutes 1987). It is unknown how many sea turtles may have drowned in shrimp nets due to this lack of enforcement. Fortunately, the Louisiana legislature approved in 2015 a bill to repeal the old prohibition of TED enforcement law. Thus, Louisiana authorities are now able to enforce the use of TED by the Louisiana shrimp industry (Hill 2015).

While compliance and enforcement has been spotty, the correct and consistent use of TEDs in the United States and Mexican Gulf of Mexico shrimp fisheries has been effective (Lewison et al. 2003). In addition, both shrimp fisheries have declined in recent years because of many factors, including the decline in shrimp abundance, increased fuel costs, reduced shrimp prices, competition with farmed and imported shrimp, and the recent, active hurricane seasons (NMFS 2007; Caillouet et al. 2008; NMFS et al. 2011; Ponwith 2011). Therefore, the decline in shrimp fishing effort in the Gulf of Mexico since the early 1990s, as well as the spatial and temporal closures of Gulf shrimp fisheries, has also reduced bycatch mortality from shrimp trawling (Lewison et al. 2003; Shaver 2005; Caillouet et al. 2008; Crowder and Heppell 2011).

In the Gulf of Mexico, TEDs are not required for many trawl fisheries that could kill sea turtles; however, tow times are often restricted to reduce the probability of sea turtle mortality (Epperly et al. 2002). For example, skimmer trawls are allowed to use restricted tow times

(55 min from April through October and 75 min from November through March) in lieu of TED requirements as a sea turtle bycatch mitigation measure. However, recent observations have indicated that the tow times are often exceeded (Price and Gearhart 2011). Because a mass sea turtle stranding event that occurred in late spring 2010 along the Mississippi coast was attributed to skimmer trawl activity, the feasibility of using TEDs in these fisheries was recently investigated (Price and Gearhart 2011). A rule was proposed by the NMFS requiring the use of TEDs on skimmer, pusher-head, and wing-net trawls in May 2012 (NMFS 2012); however, the rule was withdrawn in late November 2012 because data gathered from a recent investigation did not support the implementation of the rule (Pulver et al. 2012; NOAA 2012).

Both pelagic and bottom longline fisheries in the Gulf of Mexico are known to incidentally take sea turtles. These longline fisheries include the U.S. Gulf of Mexico yellowfin tuna fishery, the U.S. distant water (outside the U.S. Exclusive Economic Zone) swordfish fishery, the Mexican Gulf of Mexico tuna fishery, the U.S. Gulf of Mexico shark fishery, the U.S. Gulf of Mexico grouper/snapper/reef fish/tilefish fishery, and the Mexican Gulf of Mexico shark fishery (NMFS et al. 2011). Gill net fisheries operate off the U.S. Gulf Coast as well as in nearshore state waters and incidentally capture sea turtles; however, gill nets have been banned in Florida, Louisiana, and Texas (NMFS et al. 2011). Sea turtles have also become entangled in dredges, seines, pound nets, and weirs, as well as pots and traps that are used to capture crabs, lobster, eels, and fish (NMFS et al. 2011). In addition, sea turtles are known to bite a baited hook; they have been hooked in both commercial and recreational fishing (TEWG 2000).

In addition to TED requirements, the banning of gill nets, and the permanent and temporary spatial and temporal closures of fisheries, additional measures have been implemented since the 1980s to reduce the incidental bycatch of sea turtles in fisheries. These efforts have included observer programs; developing gear solutions, such as circle hooks and bait combinations; modifying gear, such as reduced pound net mesh sizes and chain mats to prevent turtles from entering the dredge bag; and implementing careful release protocols (Conant et al. 2009; Stokes et al. 2012).

11.7.1.1 Incidental Capture of Kemp's Ridley Sea Turtles in Commercial and Recreational Fisheries

The documentation of the incidental capture of Kemp's ridleys during commercial shrimping operations, particularly in the northern Gulf of Mexico, began in 1973 when the shrimp fishery in the U.S. Gulf of Mexico was becoming highly mechanized (Frazier et al. 2007). Between 500 and 5,000 Kemp's ridleys were estimated to be killed each year prior to the requirement that the offshore shrimping fleet in the southeast United States and Gulf of Mexico use TEDs (Magnuson et al. 1990). Largely because of shrimp trawling in the southeast United States and Gulf of Mexico, 2,700 juvenile and adult Kemp's ridleys were estimated to die annually from interactions with the fisheries even after TED enlargement requirements were implemented in 2003. However, these bycatch estimates are highly uncertain due to the lack of observer coverage (Finkbeiner et al. 2011). Nevertheless, because up to 5,000 Kemp's ridleys were estimated to be killed each year prior to the requirement of TEDs in the shrimp trawl fishery (Magnuson et al. 1990), the estimate by Finkbeiner et al. (2011) may represent a significant reduction to Kemp's ridley annual mortality from shrimp trawling. The decline in shrimp fishing effort in the Gulf of Mexico since the early 1990s, as well as the spatial and temporal closures of Gulf of Mexico shrimp fisheries, has reduced bycatch mortality from shrimp trawling and has contributed to the Kemp's ridley population increase (Lewison et al. 2003; Shaver 2005; Caillouet et al. 2008; Crowder and Heppell 2011).

In the Gulf of Mexico, Kemp's ridleys rarely interact with, or are incidentally captured in low numbers by fisheries, other than shrimp trawling. From 1994 through 2006, 11 Kemp's ridleys were incidentally captured during 4,096 trips of the Mexican pelagic longline tuna fishery (Ramirez and Ania 2000; J. Molina, Instituto Nacional de Pesca, personal communication, 2007, cited in NMFS et al. 2011). No Kemp's ridleys were observed as bycatch for the U.S. shark bottom longline fishery in the Gulf from 1994 through 2002 or from July 2005 through 2010 (Hale and Carlson 2007; NMFS et al. 2011). In the Gulf of Mexico bottom longline fishery for shark, grouper, snapper, tilefish, and reef fish, no Kemp's ridleys were observed incidentally captured from 2005 through 2010 (Hale and Carlson 2007; Hale et al. 2009; NMFS et al. 2011). In the gill net fisheries operating off the U.S. Gulf Coast, no Kemp's ridleys were observed taken from 2000 through 2008, and only one Kemp's ridley, which was released alive and uninjured, was incidentally captured in 2009 (Garrison 2007; Baremore et al. 2007; Passerotti and Carlson 2009; Passerotti et al. 2010).

Kemp's ridleys are caught in both commercial and recreational hook-and-line fisheries along the U.S. Gulf Coast (Cannon et al. 1994; Seney 2008; NMFS et al. 2011). From 1980 through 1992, 112 Kemp's ridleys interacted with recreational hook-and-line gear at piers along the Texas coast, with 39 turtles documented in 1992 alone; 62 of these live captures were between the Bolivar Peninsular in Galveston County and the Texas-Louisiana border, and 63 % of the Kemp's ridleys were turtles that had been head-started (Cannon et al. 1994). These turtles, which were primarily juveniles ranging in size from 25 to 45 cm (9.8 to 17.7 in) SCL, represented a cost-effective means of gathering important Kemp's ridley data, while providing an opportunity for rehabilitation, if necessary (Seney 2008). A total of 170 Kemp's ridley hook-and-line encounters, which included 154 live captures, was recorded along the upper Texas coast from 1980 through 1995, with almost 90 % (135 turtles) occurring from 1992 through 1995 (C.W. Caillouet, NOAA NMFS, unpublished memo, 1996, cited in Seney 2008). The increased reports of hook-and-line captures during the mid-1990s may have been due to public education efforts targeting anglers, better survival of juvenile Kemp's ridleys due to shrimping regulations, and/or the initial stages of recovery exhibited by the overall population (Cannon et al. 1994; Cannon 1995; Lewison et al. 2003).

From 2003 through 2007, 42 Kemp's ridleys, with an average size of 34.6 cm (13.6 in) SCL, were captured on hook and line along the upper Texas coast in Galveston (45 turtles) and Jefferson (two turtles) counties (Seney 2008). Most of the captures (74 %) were reported from a single pier on Galveston Island. No hatchling or oceanic juveniles (less than 25 cm [9.8 in] SCL) or subadults and adults (greater than 45 cm [17.7 in] SCL) were captured on hook and line, which was similar to what occurred for Kemp's ridleys caught on hook and line along the Florida Panhandle from 1991 through 2003 (Rudloe and Rudloe 2005). However, in contrast to the ridleys caught on hook and line at piers in the Florida Panhandle, the Kemp's ridleys caught by hook and line along the upper Texas coast demonstrated low site fidelity. The hook-and-line captures were retrieved between March and October, with 81 % occurring from April through June. Forty of the hook-and-line captures were successfully rehabilitated and released (Seney 2008).

11.7.1.2 Incidental Capture of Loggerhead Sea Turtles in Commercial and Recreational Fisheries

The incidental capture of loggerheads in fisheries was concluded to be the most important threat to the northwest Atlantic Ocean loggerhead population in the most recent recovery plan (Bolten et al. 2011) and was determined to be a significant threat to loggerheads in the northwest Atlantic in the most recent status review completed by the NMFS and USFWS (Conant et al. 2009). In observed U.S. fisheries, loggerheads that are taken range from moderately sized juveniles through adults. Loggerhead hatchlings and small size classes are rarely seen as bycatch (TEWG 2009).

An estimated 63,500 loggerheads died annually from fishery interactions before bycatch mitigation strategies were mandated; an estimated 1,400 loggerheads died each year after TED opening enlargements were implemented in 2003 (Finkbeiner et al. 2011). The southeast U.S./Gulf of Mexico shrimp trawl fishery (responsible for an estimated 23,300 annual interactions) was responsible for the most loggerhead interactions on an annual basis. In addition, due to their large nesting assemblages in the southeast United States, including the Gulf of Mexico, and their annual migrations to higher latitudes (Plotkin and Spotila 2002), loggerheads interact with more fisheries than any other sea turtle species in the United States (17 of 18 analyzed fisheries). New loggerhead bycatch estimates available for the southeast U.S./Gulf of Mexico shrimp trawl fishery, updated using 2009 data, suggest that 28,200 interactions occur each year in the Gulf, resulting in 785 loggerhead deaths (Ponwith 2011).

The estimated number of loggerhead sea turtles caught by the U.S. pelagic longline fishery in the Gulf of Mexico from 1993 and 2009 is summarized in Figure 11.46. The highest loggerhead bycatch was estimated to occur in 2002; the second highest occurred in 2003. Loggerhead bycatch estimates resulting from the Gulf of Mexico pelagic longline fishery have declined in recent years, and levels from 2005 through 2009 appear to be similar to those estimated for the 1990s (Figure 11.46).

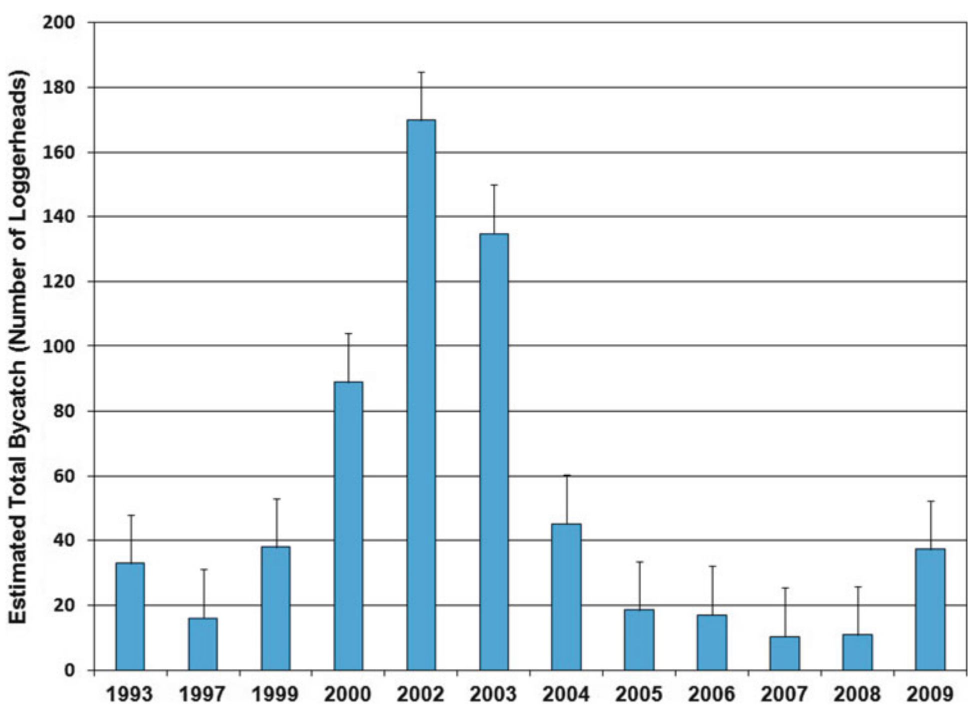

Figure 11.46. Estimated number of loggerhead sea turtle incidental captures by the U.S. pelagic longline fishery in the Gulf of Mexico from 1993 through 2009; no estimates available for 1994, 1995, 1996, 1998, or 2001; error bars = standard error (Johnson et al. 1999b; Yeung 1999, 2001; Garrison 2003, 2005; Garrison and Richards 2004; Fairfield-Walsh and Garrison 2006, 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010).

11.7.1.3 Incidental Capture of Green Sea Turtles in Commercial and Recreational Fisheries

Green turtles are killed as bycatch in coastal fisheries, including drift nets, longlines, set nets, pound nets, and trawl fisheries (Magnuson et al. 1990). Prior to the implementation of TED requirements, an estimated 57 % of green turtle mortalities occurred in the Gulf of Mexico as a result of shrimping activity; most of the turtles killed were juveniles (less than 60 cm [23.6 in] SCL), with the majority occurring in the central, northern Gulf (Henwood and Stuntz 1987; Thompson 1988). By the early 1980s, an estimated 229 green turtles drowned in shrimp nets annually, and most captures were in neritic Gulf of Mexico waters with a depth of less than 25 m (82 ft) (Henwood and Stuntz 1987). After TED opening enlargements were mandated in 2003, 300 green turtles were estimated to die each year (Finkbeiner et al. 2011); this study demonstrated that green turtles interacted primarily with the southeast U.S./Gulf of Mexico shrimp trawl fishery (11,300 bycatch events).

11.7.1.4 Incidental Capture of Leatherback Sea Turtles in Commercial and Recreational Fisheries

Since the mid-1900s, numerous instances of leatherbacks becoming hooked or tangled on longlines, buoy anchor lines, and other ropes and cables, leading to injury and/or death have been documented (NMFS 1992). Of 30 sea turtles caught in the Gulf of Mexico by the Japanese tuna longline fishery from 1978 to 1981, 12 (40 %) were leatherback sea turtles (Witzell 1984). Between 1992 and 1995, the U.S. pelagic longline fleet caught 73 leatherbacks in the Gulf of Mexico (Witzell 1999). Of the 621 turtles taken in the U.S. pelagic longline fishery in the Gulf of Mexico from 1992 through 2005, more than 85 % of them were leatherbacks (Kot et al. 2010).

A summary of the estimated number of leatherbacks captured by the U.S. pelagic longline fishery in the Gulf from 1993 and 2009 is presented in Figure 11.47. Although a careful statistical analysis is not available, the highest numbers of leatherback bycatch appeared to have occurred from 2002 through 2004. Leatherback bycatch resulting from the Gulf of Mexico pelagic longline fishery appears to have declined in recent years.

The first edition of the *U.S. National Bycatch Report* summarized sea turtle bycatch from 2001 through 2006 for the Atlantic and Gulf of Mexico fisheries (NMFS 2011c). The report estimated the following for bycatch of leatherback sea turtles: 63 killed annually in the Gulf due to capture in shrimp trawls; 83 caught annually by Atlantic and Gulf of Mexico shark bottom longline fisheries; and 351 caught annually by Atlantic and Gulf of Mexico pelagic longline fisheries.

While high uncertainty is associated with these estimates, approximately 2,300 leatherbacks were estimated to have died annually from fisheries interactions before bycatch mitigation strategies were mandated, and an estimated 40 leatherbacks died each year after TED opening enlargements were mandated in 2003 (Finkbeiner et al. 2011). For leatherbacks, the Atlantic/Gulf of Mexico pelagic longline fishing was estimated to be responsible for the most interactions, followed by the southeast U.S./Gulf of Mexico shrimp trawl fishery (Finkbeiner et al. 2011). For 2009, the NMFS estimated 623 interactions between leatherbacks and shrimp trawls in the Gulf, 18 of which were estimated to result in mortality (Ponwith 2011).

11.7.1.5 Incidental Capture of Hawksbill Sea Turtles in Commercial and Recreational Fisheries

Hawksbill sea turtles are caught much less frequently as bycatch than the other four species of sea turtles in the Gulf of Mexico. Nevertheless, hawksbills are susceptible, particularly in

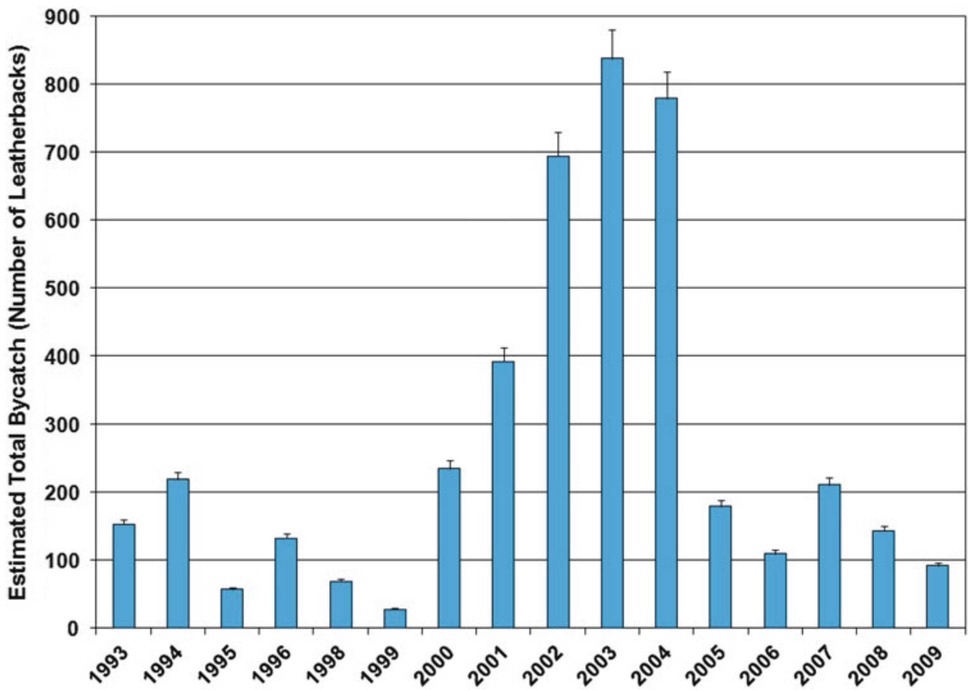

Figure 11.47. Estimated number of leatherback sea turtle incidental captures by the U.S. pelagic longline fishery in the Gulf of Mexico from 1993 through 2009; no estimates available for 1997; error bars = standard error (Johnson et al. 1999b; Yeung 1999, 2001; Garrison 2003, 2005; Garrison and Richards 2004; Fairfield-Walsh and Garrison 2006, 2007; Fairfield and Garrison 2008; Garrison et al. 2009; Garrison and Stokes 2010).

nearshore fisheries, such as drift netting, longlining, set netting, pound netting, gill netting, and trawl fisheries (Magnuson et al. 1990; NMFS and USFWS 2007e). For example, hawksbill turtle bycatch was not quantified in the 2011 national bycatch report for the southeast United States, but they may have been included in the 0.4 % of unidentified turtles (NMFS 2011c). In an effort to evaluate the effectiveness of the use of TEDs, NMFS observers documented only one hawksbill of the 13 turtles caught by U.S. shrimp fisherman in the Gulf of Mexico from March 1988 through July 1989 (Renaud et al. 1990); the same program recorded no hawksbills of two documented sea turtles caught in the Gulf of Mexico by the same fishery from September 1989 through August 1990 (Renaud et al. 1991). The sea turtle bycatch report from the 1998 pelagic longline fishery recorded no hawksbill captures in the Gulf (Yeung 1999). The limited data on hawksbill bycatch suggests that the risk of being killed or injured as bycatch in commercial and recreational fisheries in the Gulf of Mexico is low. However, given the low population abundance of hawksbills in the Gulf of Mexico, efforts must continue to ensure low bycatch impact on this species.

11.7.2 Terrestrial Zone Threats

Although uncommon, the poaching of eggs from nesting female Kemp's ridleys still occurs in Mexico and has occurred in south Texas (NMFS et al. 2011). In Florida, egg poaching does occur, and from 1980 through 2002, more than 60 arrests were made for the possession or sale of sea turtle eggs (NMFS and USFWS 2008).

Along the Gulf of Mexico coast, beaches are cleaned by mechanical raking, scraping with large machinery, hand raking, and picking up debris by hand. These activities can directly and indirectly affect sea turtles (NMFS et al. 2011). Driving is permitted on many Gulf of Mexico beaches. Nesting sea turtles have been run over and killed by vehicles, and vehicles have crushed emerging hatchlings (NMFS et al. 2011). Because the Kemp's ridley has only one primary nesting beach in Rancho Nuevo, Mexico and a secondary nesting colony at PAIS in Texas, it is particularly susceptible to beach disturbance, alteration, and destruction by natural and anthropogenic events. Beach cleaning has been documented to affect Kemp's ridleys. For example, 12 Kemp's ridley hatchlings became trapped by a sand ridge created by heavy equipment cleaning the beach on North Padre Island, Texas in 2002 and were later crushed and killed by passing vehicles (Shaver 2004).

The presence of artificial lighting on or near the beach adversely affects both nesting and hatchling sea turtles (Witherington and Martin 1996). Mortality from misdirection by artificial lighting on both the Gulf of Mexico and Atlantic coasts of Florida kills thousands of loggerhead hatchlings each year (Witherington 1997). The proportion of all emerging loggerhead hatchlings that died because of beach lighting was estimated in the early 1990s to be as high as 5–10 % (Witherington et al. 1996). Loggerheads abort nesting attempts at a greater frequency in lighted areas. Artificial lighting also deters females from emerging from the ocean to nest (Witherington 1986, 1992). Hatchling orientation of nests located at 23 representative beaches in six Florida counties was surveyed in 1993 and 1994, and approximately 10–30 % of nests showed evidence of hatchling disorientation by lighting (Witherington et al. 1996). Similar to other nocturnal nesting sea turtles, nesting leatherbacks and hatchlings can be disoriented by artificial lighting on the beach (NMFS 2011b).

A significant negative relationship was found between sea turtle nesting density and distance from the nearest of 17 ocean inlets (Witherington et al. 2005). Beach instability from both erosion and sand accretion may discourage sea turtle nesting, since the effect of inlets in lowering nesting density was found both updrift and downdrift of the inlets. When sea turtles emerged to nest in the presence of armoring structures, more returned to the water without nesting compared to turtles that emerged on nonarmored beaches (Mosier 1998; Mosier and Witherington 2002). Fewer sea turtles made nesting attempts on beaches fronted by seawalls than on adjacent beaches where armoring structures were absent (Mosier 1998). In addition, sea turtles on armored sections of beach had a tendency to wander great distances as compared to turtles that emerged to nest on adjacent natural beaches (Mosier 1998).

Since oil exploration and production occurs south of Tamaulipas and Veracruz in Mexico and at PAIS in Texas, the Kemp's ridley nesting beaches, as well as the nesting turtles, eggs, and hatchlings, could be impacted by oil spills and related activities (NMFS et al. 2011). For example, in 1979, the Ixtoc I oil well blew out and caused a fire in the Bay of Campeche in Mexico. The nesting beach at Rancho Nuevo was affected by the Ixtoc I well blowout, and large amounts of oil were released daily into the Gulf of Mexico for several months. However, the oil reached the beach after the nesting season, and nesting females were not present. Also, a loaded supertanker, the *Mega Borg*, exploded near Galveston, Texas in 1990, causing more than 121,000 barrels of crude oil to be released into the Gulf of Mexico; sea turtles covered in oil were found after this spill (Yender and Mearns 2003).

Oil spills have affected loggerhead nesting beaches in the Gulf of Mexico. In August 1993, approximately 350,000 gal of fuel oil spilled into Tampa Bay and washed onto nesting beaches in Pinellas County, Florida (Conant et al. 2009). Impacts to loggerheads resulting from the spill included 31 dead hatchlings, 176 oil-covered nests, and 2,177 eggs and hatchlings exposed to oil or disturbed by response activities (FDEP et al. 1997).

Tropical coastlines are rapidly being developed, often leading to the destruction of hawksbill nesting habitat (Mortimer and Donnelly 2008). For example, critical hawksbill habitats are quickly being impacted by the development along the Gulf coast of the Yucatán Peninsula (Garduño-Andrade et al. 1999). Beachfront development and the clearing of dune vegetation significantly affect hawksbill sea turtles because they prefer to nest under vegetation (Mortimer and Donnelly 2008).

Global climate change may affect sea turtle nesting beaches in several ways, including sea level rise, higher ambient temperatures, and changes in hurricane/cyclone activity (Hawkes et al. 2009; Witt et al. 2010). Higher water levels associated with sea level rise will gradually and directly decrease the availability of suitable nesting sites (Witt et al. 2010). Increasing temperatures may open up areas that were previously unavailable for nesting (Witt et al. 2010), but recent studies have suggested that up to half of the currently available sea turtle nesting areas could be lost with predicted sea level rise (Fish et al. 2008). Sea turtle nesting is significantly affected by temperature, and incubation temperatures can affect incubation success, duration, and the sex of hatchlings (Mrosovsky and Yntema 1980; Witt et al. 2010; Valverde et al. 2010). Therefore, increasing temperatures have the potential to change current nest incubation regimes, as well as to skew sex ratios (Hawkes et al. 2007a). Changes in the global climate are predicted to increase the frequency and intensity of hurricanes (Webster et al. 2005), which can significantly affect the reproductive success of sea turtles.

On Gulf of Mexico beaches, predation of sea turtle eggs and hatchlings can be significant. Known predators of sea turtle eggs and hatchlings include raccoons, ghost crabs, coyotes, foxes, armadillos, domestic dogs and cats, feral pigs, skunks, bobcats, badgers, gulls, fish crows, and larval insects (Witherington et al. 2006a; NMFS et al. 2011). Invasive fire ants are also significant predators of sea turtle eggs and hatchlings on Gulf coast nesting beaches (Witherington et al. 2006a; NMFS et al. 2011). Invasive plant species are known to invade and desiccate eggs, trap hatchlings, interfere with nest construction, and lower nest incubation temperatures because of shading (Conant et al. 2009). For example, the invasive Australian pine has caused shading of beaches, lowered sea turtle nest incubation temperatures, limited accessibility to suitable nest sites, entrapped nesting turtles, interfered with nest construction, and caused sea turtle nesting activity to decline on a remote nesting beach in Everglades National Park along the Florida Gulf coast (Davis and Whiting 1977; Schmeltz and Mezich 1988; Reardon and Mansfield 1997; Hanson et al. 1998).

The hurricane season for the Gulf overlaps closely with the sea turtle nesting season (Magnuson et al. 1990). While sea turtles have evolved to deal with erosion, flooding, storm surges, and other disturbances caused by hurricanes and other severe storms by laying large numbers of eggs and distributing their nests spatially and temporally, hurricanes can affect the reproductive success of sea turtles, since they rely on specific beaches for reproduction (Carr and Carr 1972). In addition, the effects of hurricanes vary by species (Pike and Stiner 2007). For example, in the southeastern United States, leatherback turtles nest the earliest, and most hatchlings emerge before the hurricane season starts, while loggerhead turtles nest intermediately, and only nests laid late in the season would be at risk. However, green turtles nest the latest, and their entire nesting season occurs during the hurricane season; therefore, their developing eggs and nests are extremely vulnerable to hurricanes (Pike and Stiner 2007).

In 1989, the effects of Hurricane Gilbert were documented on the Kemp's ridley nesting beach in Mexico. When debris was deposited, the beach was eroded, and coral rock was exposed along the central portion of Rancho Nuevo. About 20 % of the Kemp's ridley nesting activity was displaced to the north that year (Márquez-M. 1990).

Hurricane Andrew, a Category 4 hurricane that hit south Florida on August 24, 1992, provided an opportunity to quantify the impacts of a major hurricane on six beaches where loggerheads of the Peninsular Florida subpopulation nest (Milton et al. 1994). Sea turtle nests on more than 145 km (90 mi) of beaches on the Gulf and Atlantic coasts of Florida were affected by the hurricane. The associated storm surge produced the greatest mortality through nest flooding. Loggerhead egg mortality was 100 % on beaches closest to the eye; mortality decreased with distance from the eye (Milton et al. 1994). Hurricane Andrew affected about 68 km (42 mi) of beach on the Florida Gulf coast. Within this zone, there was about a 40–50 % mortality of loggerhead eggs and hatchlings, and about 22 % of the 2,762 loggerhead nests were partially or completely destroyed (Milton et al. 1994).

Although hurricanes periodically remove the sand from the typically small hawksbill nesting beaches, the sand is usually replaced by wind and wave action. However, hurricanes may cause trees to fall and debris to be deposited on beaches; this debris hinders or prevents hawksbills from reaching their nesting habitat.

11.7.3 Neritic and Oceanic Zone Threats

The illegal poaching of juvenile and adult sea turtles in the marine environment is uncommon in the Gulf of Mexico (NMFS et al. 2011). However, sea turtles that use the Gulf could be harvested legally in nearby countries (e.g., Turks and Caicos Islands) (NMFS and USFWS 2008).

In the past, Gulf of Mexico loggerhead sea turtles could have been taken in Cuba, since an active harvest continued through the mid-1990s (Moncada Gavilan 2000). The estimated harvest of loggerheads for meat in Cuba was as follows: (1) from 1968 through 1975, at least 4,300 turtles were harvested each year; (2) from 1976 through 1987, at least 2,600 turtles were harvested each year; and (3) from 1988 through 1994, an initial level of at least 1,750 turtles were harvested each year, declining to at least 660 turtles in later years (TEWG 2009). Interestingly, there was a concurrent increase in the number of loggerhead nests laid on Florida Index Beaches, including beaches on the Florida Gulf coast, as fewer loggerheads were harvested in Cuban waters from the late 1980s through the late 1990s. In addition, the annual number of nests in Florida was still increasing when the loggerhead fishery ended in Cuba in 1996, with little sign of the decline that has characterized loggerhead nesting in Florida in recent years (TEWG 2009).

While it has declined dramatically over the last 20 years, the most significant threat to hawksbill populations is the continued illegal trade of hawksbill products. While the legal hawksbill tortoiseshell trade ended in 1993 when Japan, historically the major importer of bekko from the Caribbean, agreed to cease the imports, a significant illegal trade continues (USFWS 2012). Because of the migratory nature of hawksbill sea turtles, this trade threatens hawksbills that occur in the Gulf of Mexico.

In addition, hawksbills may still be captured illegally in the general Gulf of Mexico and Caribbean region. While hawksbills were harvested in Cuba from at least the 1500s and thousands of nesting females were captured annually through the nineteenth and twentieth centuries, a seasonal closure was introduced in 1936, and the Cuban government prohibited egg collection and the disturbance of nesting females beginning in 1961 (Carrillo et al. 1999; McClenachan et al. 2006; Moncada et al. 2012). In Cuba, the annual legal foraging ground exploitation of 5,000 hawksbills was reduced to 3,000 in 1993, 1,000 in 1994, and 500 turtles in 1995 (Carrillo et al. 1999). Cuba closed their hawksbill harvest in 2008 (Moncada et al. 2012); however, illegal subsistence fishing may still be occurring (Hart et al. 2012b).

The Gulf of Mexico is an area of high-density offshore oil and gas exploration, development, and production, and related activities affect all life stages of sea turtles. In addition to oil and natural gas being released into the Gulf from natural seeps and low-level spills, large events do occur on occasion (NMFS et al. 2011). The impacts of seismic surveys associated with oil exploration activities on sea turtles have not been fully studied (Cuevas et al. 2008). The few available studies suggest responses of sea turtles include an alarm reaction, subsequent avoidance, and sometimes temporary or permanent hearing loss (MMS 2004).

Explosives are used to remove oil platforms in the Gulf of Mexico when they are no longer in operation. These explosions have affected sea turtles in the past (Klima et al. 1988). However, an intensive observer program has been in place since 1987 to minimize sea turtle impacts from these explosions (Gitschlag 1992). The program has been successful in mitigating impacts to sea turtles associated with the explosive removal of offshore structures (Viada et al. 2008).

Leatherbacks have infrequently been observed near offshore developments associated with oil and gas operations. For example, in a survey of turtles around energy structures off the coasts of Texas and Louisiana in 1992, only two leatherbacks of 47 individual turtles were observed (Gitschlag and Herczeg 1994). During a 2-year survey, 15 leatherbacks were sighted within 8,000 m (2,625 ft) of petroleum platforms (Lohoefener et al. 1990).

Eight oiled, dead sea turtles, including one juvenile Kemp's ridley, washed up on Texas beaches after the Ixtoc I spill in the Gulf in 1979 (Rabalais and Rabalais 1980). Chronic oil exposure may have led to their poor body condition and, ultimately, their death (Hall et al. 1983). Juvenile Kemp's ridleys that were released into *Sargassum* patches offshore Padre and Mustang Islands, Texas in 1983 were found stranded on Padre and Mustang Island beaches with oily residues in their mouth, esophagus, and stomach; the residues were later determined to most likely be a result of tanker cleaning operations (Overton et al. 1983).

Van Vleet and Pauly (1987) determined that crude oil tanker discharge was significantly affecting sea turtle populations in the eastern Gulf of Mexico based on an analysis of oil residues from Kemp's ridley, loggerhead, green, and hawksbill sea turtles stranded on U.S. Gulf of Mexico beaches. There is evidence that oil pollution has a greater impact on hawksbills than on other species of sea turtles (Meylan and Redlow 2006). For example, approximately 12 % of stranded hawksbills on Florida beaches had evidence of fouling by oil, compared to 1.1 % of the strandings of other species. In addition, 22.4 % of stranded hawksbills smaller than 22 cm (8.7 in) SCL had evidence of oil (Meylan and Redlow 2006). However, from 1980 through 2002, no oil-affected turtles were found stranded on Florida Gulf coast beaches (Meylan and Redlow 2006). Ingested tar or tar on their bodies has been documented for Kemp's ridleys found along the Texas coast (Shaver 1991).

In addition to oil, other contaminants could affect sea turtles. For example, polychlorinated biphenyls (PCBs), as well as dichlorodiphenyltrichloroethane (DDT) and its breakdown products, dichlorodiphenyldichloroethylene and dichlorodiphenyldichloroethane (DDE and DDD, respectively), have been found in tissues of stranded Kemp's ridley and loggerhead sea turtles (Rybitski et al. 1995). Polycyclic aromatic hydrocarbons (PAHs), DDD, PCBs, and metals were found in loggerhead eggs that failed to hatch that were collected from 20 nests on Florida Gulf coast beaches in 1992. The failure of the eggs to hatch may have been related to the additive or synergistic toxicity of the low levels of the contaminants found in the eggs (Alam and Brim 2000). Exposure to organochlorines, such as PCBs and pesticides, has been suggested to modulate immunity in loggerhead sea turtles (Keller et al. 2006).

A large number of persistent organic pollutants (POPs) were measured in egg yolks of unhatched loggerhead eggs collected from 11 nests along the Gulf coast in Sarasota in 2002 (Alava et al. 2011). Levels of POPs were lowest in eggs collected from the Gulf coast compared to eggs collected from loggerhead nests along the Atlantic coast. Foraging ground locations used by the nesting females may have caused the differences. Data from a satellite tracking study conducted by Girard et al. (2009) indicated that females nesting on Sarasota beaches forage in the Gulf of Mexico and Caribbean Sea, where the prey is less contaminated than from sites used by females nesting along the Atlantic coast (Alava et al. 2011).

Although the effects of exposure to POPs in sea turtles are unknown, POPs have been measured in tissue, blood, and eggs from leatherbacks on the U.S. east coast (Stewart et al. 2011). In addition, trace metals have been found in blood and eggs from females in French Guiana (Guirlet et al. 2008). Since nesting females from French Guiana are known to spend time in the Gulf of Mexico (Pritchard 1976), it is possible that contaminant exposure occurs there.

Sea turtles in the Gulf are differentially affected by the ingestion of and entanglement in marine debris (Bjorndal et al. 1994; Witzell and Schmid 2005). However, all five species of sea turtles that occur in the Gulf of Mexico are significantly affected by the ingestion of and, to a lesser extent, by entanglement in marine debris, typically plastic, in the northwestern Gulf (Plotkin and Amos 1990). Compared to other species of sea turtles, the ingestion of marine debris by Kemp's ridleys is thought to be minimal because they eat more active prey, and their foraging areas are in locations where wind and currents do not concentrate marine debris (Bjorndal et al. 1994; Witzell and Schmid 2005). However, Kemp's ridleys have been documented to ingest plastic, rubber, fishing line and hooks, tar, string, Styrofoam, and aluminum (Shaver 1991; Werner 1994; Witherington et al. 2012). They also have been killed as a result of entanglement in plastic, fishing line, discarded netting, and other debris (Plotkin and Amos 1988).

Because post-hatchling and small oceanic juvenile loggerheads occupy convergences, rips, and driftlines in the open ocean, the likelihood of becoming caught in and consuming marine debris is significant; marine debris accumulates in these same areas (Carr 1986; Witherington 2002; Witherington et al. 2012). In addition to leatherbacks, loggerhead sea turtles appear to ingest more debris in all of its life stages because of habitat choice and feeding behavior (Lutcavage et al. 1997); the ingestion of debris, including plastic, Styrofoam, balloons, and tar balls occurs when debris is mistaken for or associated with prey items (Conant et al. 2009).

Since at least 1970, there have been numerous reports of leatherbacks ingesting plastic debris (Mrosovsky 1981; Mrosovsky et al. 2009). Researchers have suggested that leatherbacks are unable to distinguish between floating plastic debris and gelatinous prey; therefore, they purposely ingest plastic as though it were a prey item (Balazs 1985). In a recent analysis of autopsy records of more than 400 leatherbacks spanning 23 years, plastic was reported in 34 % of these cases, and blockage of the gastrointestinal tract by plastic was documented in multiple cases (Mrosovsky et al. 2009).

The increasing frequency of red tides and other harmful algae blooms in the Gulf of Mexico and the increasing number, geographic extent, and duration of anoxic and hypoxic dead zones caused by agricultural runoff in Mississippi River outflow to the Gulf directly and indirectly affect sea turtles (NMFS et al. 2011). Since 1995, red tide blooms, caused by the marine alga, *Karenia brevis*, have been detected every year in the Gulf of Mexico near southwest Florida. This algal species produces brevetoxin, which kills sea turtles (Pierce and Henry 2008; FFWCC unpublished data, cited in TEWG 2009). Since the early 1990s, at least 100 Kemp's ridleys found dead on Florida Gulf coast beaches have been associated with red tide events (NMFS et al. 2011).

The absence of dissolved oxygen, which is essential to most animals and plants that inhabit the Gulf of Mexico, in the dead zones kills benthic invertebrates, including crabs, the main prey item for Kemp's ridleys. Due to the reduced abundance of food, Kemp's ridley sea turtles are not likely to forage in or inhabit hypoxic areas for any length of time (McDaniel et al. 2000). Aerial surveys have indicated an absence of sea turtles in these zones (Craig et al. 2001).

The frequency of fibropapilloma tumors, which are often linked to debilitation and death, is much higher in green turtles than for any of the other sea turtle species (Witherington et al. 2006b; NMFS and USFWS 2007c). In an analysis of STSSN data for 4,328 green turtles found dead or debilitated from Massachusetts to Texas from 1980 through 1998, fibropapillomatosis was reported in green turtles only in the southern half of Florida, and 22 % of turtles in this region had tumors. The disease was more prevalent in turtles found on the Florida Gulf coast (52 %) as compared to turtles found along the Florida Atlantic coast (12 %) (Foley et al. 2005). The disease was more common in coastal waters characterized by habitat degradation and pollution, large shallow water areas, and low wave energy. In addition, 22 % of the 6,027 green turtles stranded on both the Gulf and Atlantic coasts of Florida from 1980 through 2005 had external fibropapilloma tumors (FFWCC FWRI 2011b).

In the Gulf of Mexico, sea turtles have been documented to entrain in and impinge on cooling water intake structures associated with power plant operations, and while the majority of turtles are released unharmed, some turtles are injured and killed (NMFS 2002). For example, in 1998, approximately 40 Kemp's ridleys were trapped in intake structures associated with the Crystal River Energy Complex located near the foraging grounds in the Cedar Keys, Florida area (NMFS 2002), and 92 Kemp's ridleys were entrapped from 1999 through 2004 (Eaton et al. 2008). When a significant increase in the number of Kemp's ridleys stranded on the intake bar racks of the power plant was observed in 1998, sea turtle protection measures were implemented (Eaton et al. 2008). In addition, 38 juvenile green turtles, ranging in size from 24 to 50 cm (9.5 to 19.7 in) SCL, and some loggerhead turtles were trapped in the intake structures at the energy complex during the same time period (Eaton et al. 2008).

Crabs are the main prey of Kemp's ridleys, and blue crab populations in the Gulf of Mexico have declined significantly since the late 1960s/early 1970s (TPWD 2006; Murphy et al. 2007). Crab populations off the Texas and Florida Gulf coasts reached lows in 2000 and have not rebounded. Many causes for the decline have been suggested, including drought and the subsequent reduction of freshwater flow into bays and estuaries, overharvesting, and coastal wetland loss (TPWD 2006). Recovering the blue crab stocks is challenging because of significant gaps in knowledge not only about the commercial and recreational blue crab fishery, but also about blue crab life history (Murphy et al. 2007). This decline could have significant impacts on the recovering Kemp's ridley population, especially as the ridley population continues to increase.

In the marine environment, the effects of climate change on sea turtles are difficult to study because sea turtles range across entire ocean basins, are late maturing, and are long lived (Zug et al. 2002). Temperature is known to influence the distribution and behavior of sea turtles (Hawkes et al. 2007b). Ocean currents, which are important for dispersing hatchling sea turtles, could change in magnitude or direction (Bolten 2003); these changes could influence the duration of future juvenile development (Hamann et al. 2003). Sea turtle trophic dynamics and juvenile growth and development could be altered as a result of changes to the pelagic community resulting from climate change (Bjorndal 1997; Bjorndal et al. 2000; Witt et al. 2010). Adult foraging habitat and the location and sizes of home ranges and diet could be altered as a result of changes to thermal regimes and sea surface currents resulting from climate change (Bjorndal 1997; Polovina et al. 2004; Witt et al. 2010).

Of all coastal marine habitats, seagrass beds—the primary foraging areas for green turtles—are the most susceptible and have low resiliency to disturbance. Seagrass beds occur along sheltered coasts with good water quality, are often located in areas of port development, and are downstream from point and nonpoint source discharges (Waycott et al. 2005). The hawksbill's dependence on coral reefs for shelter and food links its well-being to the conditions of coral reefs, and coral reefs are one of the most endangered ecosystems in the world (NMFS and USFWS 1993, 2007e). Because they prefer hard bottom habitats, hawksbills are thought to frequent underwater structures and hard platforms, such as those provided by offshore continental shelf development in the Gulf of Mexico. In addition, habitat loss in the Caribbean could cause more hawksbills to move into the Gulf of Mexico (Hopkins and Richardson 1984; Thompson et al. 1990).

11.7.3.1 Hypothermic (Cold) Stunning of Sea Turtles

Deep freeze events, in which water temperatures drop below 10 °C, are uncommon and brief in the northern Gulf of Mexico and typically last for only one to a few days; however, when these events occur, they can have a significant impact on coastal sea turtles, especially the juveniles (Henry et al. 1994). Most hypothermic or cold-stunning events, in which sea turtles become incapacitated and lose their ability to swim and dive as a result of rapidly dropping water temperatures, occur in shallow coastal lagoons and bays (Ogren and McVea 1982; Turnbull et al. 2000). The decreases in water temperature occur so rapidly that the turtles have little time to move to warmer waters (McMichael et al. 2008). Low water temperatures become lethal between 5 and 6.5 °C (Schwartz 1978; Moon et al. 1997). Hypothermic-stunned sea turtles may exhibit metabolic and respiratory acidosis (Innis et al. 2007). If hypothermic-stunned turtles are left untreated, they may perish in the water or when stranded on the beach, since air temperatures are even lower than water temperatures.

Adult but mainly juvenile sea turtles that overwinter in inshore waters of the Gulf of Mexico, such as along PAIS in Texas and St. Joseph Bay in Florida (Shaver 1990b; Foley et al. 2007), are most susceptible to hypothermic stunning because the temperature change is most rapid in shallow waters (Witherington and Ehrhart 1989b). A recent study suggested that it is not the rate of water temperature decrease but the duration and magnitude of the drop that drives the severity of the phenomenon (Roberts et al. 2014). Sea turtles often die from hypothermia, unless rescued, rehabilitated, and later released.

Kemp's ridleys, loggerheads, green turtles, and hawksbills have been affected by hypothermic stunning in waters along the northern Gulf of Mexico in Texas and Florida (Hildebrand 1982; Shaver 1990b; Foley et al. 2007; Roberts et al. 2014). However, historically, green turtles, especially juveniles, have been most affected by hypothermic stunning (Figure 11.48). Leatherbacks are not susceptible to hypothermic-stunning events due to their unique ability to maintain a higher body temperature than the surrounding water temperature (Frair et al. 1972) and to the fact that they spend most of their lives in deep waters. Leatherbacks have never been reported stranded due to hypothermic stunning in the Gulf of Mexico.

Historically, severe freezes in the late 1800s are thought to have contributed to the decline in the green turtle fishery in Texas (Hildebrand 1982). More recently, juvenile green turtles, and to a lesser extent Kemp's ridleys, loggerheads, and hawksbills were hypothermic stunned and stranded in Texas inshore waters during severe winters in 1971, 1979, 1983, and 1989 (Hildebrand 1982; Shaver 1990b). During a severe cold front in February 1989, 46 sea turtles were found stranded as a result of hypothermic stunning in Laguna Madre near Port Mansfield, Texas (Shaver 1990b; Figure 11.48); the hypothermic-stunned turtles included 45 green turtles (31 were found dead) and one dead loggerhead. Hypothermic-stunning events occurred in Port Isabel and South Padre Island, Texas during the winters of 2007 and 2010, in which 150 and 200 green turtles, respectively, were hypothermic stunned (Figure 11.48).

During late December 2000 and early January 2001, an unprecedented hypothermic-stunning event occurred in St. Joseph Bay along the Florida Gulf coast (Foley et al. 2007). More than 400 turtles were hypothermic stunned, which included 388 green turtles (55 found dead) (Figure 11.48), ten Kemp's ridleys (four found dead), and three loggerhead sea turtles (one found dead) (Foley et al. 2007; McMichael et al. 2008). All of the hypothermic-stunned green turtles were neritic juveniles, and most were from the Florida and Yucatán populations

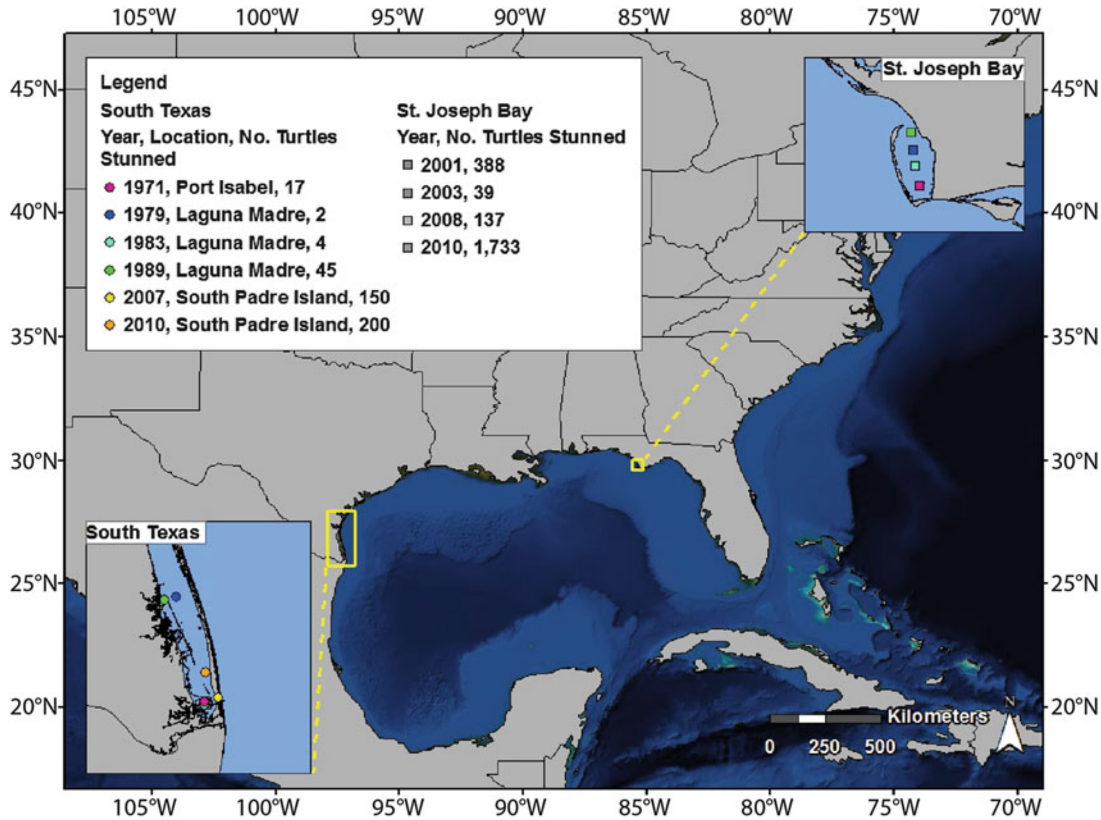

Figure 11.48. Hypothermic stunning locations of green sea turtles in the northern Gulf of Mexico from 1971 through Winter 2010 (interpreted from Shaver 1990b; H. Hildebrand, personal communication, cited in Shaver 1990b; McMichael et al. 2003, 2008; Foley et al. 2007; Texas A&M University 2011; Avens et al. 2012; Roberts et al. 2014).

(Foley et al. 2007). The 10 Kemp's ridleys ranged in size from 26.5 to 46 cm (10.4 to 18.1 in) SCL, with an average SCL of 33.4 cm (13.1 in), and one of the four Kemp's ridleys found dead was a head-started turtle from the 1998 year class. Most (337 of 401) of the sea turtles survived and were later released (Foley et al. 2007). The rehabilitated and released sea turtles included 329 green turtles, six Kemp's ridleys, and two loggerheads.

In 2003, a small hypothermic-stunning event occurred in St. Joseph Bay (McMichael et al. 2003, 2008). Forty-two turtles (39 green turtles, two Kemp's ridleys, and one loggerhead) were hypothermic stunned (Figure 11.48), and 30 of the sea turtles survived. In January 2008, another moderate hypothermic-stunning event occurred in St. Joseph Bay in which more than 100 sea turtles, mostly green turtles, were hypothermic stunned (Roberts et al. 2014).

During January 2010, Florida experienced below freezing temperatures for 12 consecutive days, resulting in a hypothermic-stunning event in St. Joseph Bay of unprecedented magnitude (Avens et al. 2012). A total of 1,733 sea turtles (mostly green sea turtles) were hypothermic stunned. While the majority of the 1,670 green turtles that were hypothermic stunned survived, 434 green turtles died from the hypothermic-stunning event (Avens et al. 2012). Air temperatures below 10 °C along with strong winds were responsible for the mass hypothermic stunning event, in which some of the turtles died when water temperatures remained between 5 and 6 °C for 3 days or so (Roberts et al. 2014; Figure 11.49).

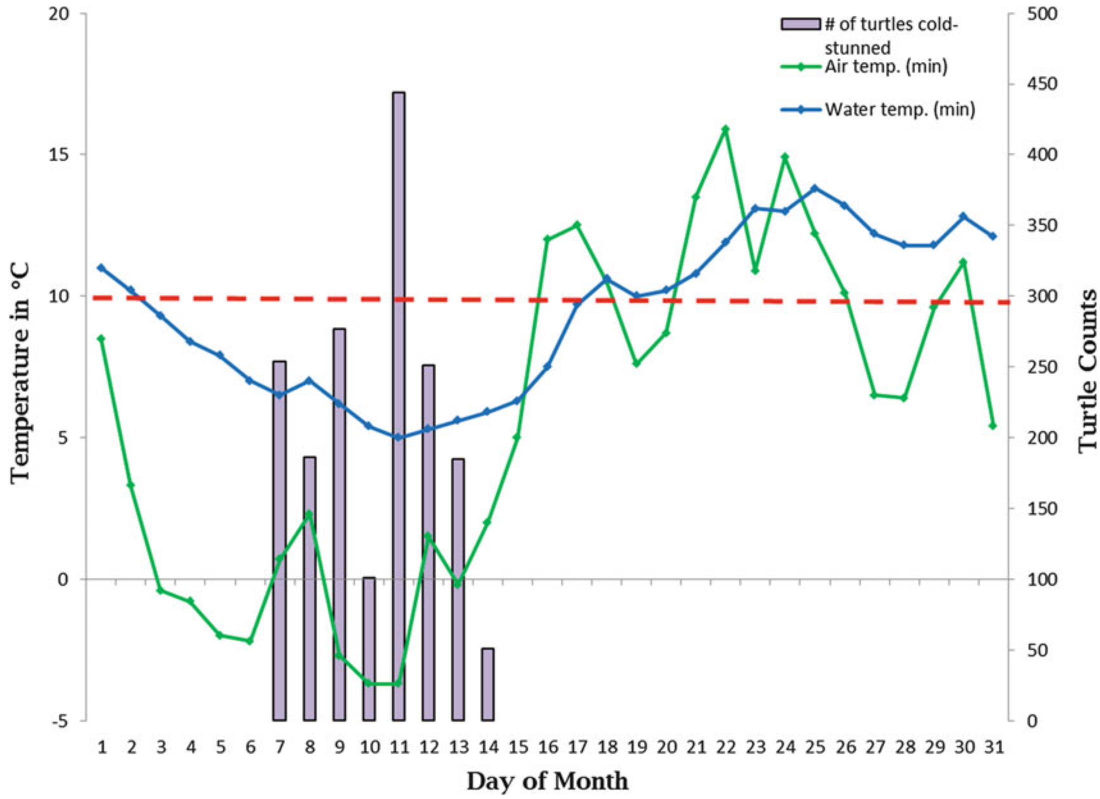

Figure 11.49. Air and water temperatures in relation to hypothermic stunned green sea turtles in Saint Joseph Bay, FL in January of 2010. The horizontal dashed red line indicates the hypothermic stunning temperature threshold for sea turtles (Roberts et al. 2014).

The frequency of occurrence, duration, and severity of hypothermic-stunning events in the Gulf of Mexico are unpredictable; however, as sea turtle populations continue to recover in the Gulf, the number of individuals impacted by hypothermic stunning may likely increase. However, because the majority of the hypothermic-stunned sea turtles are rescued, rehabilitated, and later released (as seen in hypothermic-stunning events in the northern Gulf of Mexico during the winters of 2011 and 2013/2014), the impacts of these events to the sea turtle populations are most likely minimal.

11.7.4 Sea Turtle Stranding Data

Occasionally, sea turtles wash up or strand (dead or alive) on the beaches of the Gulf of Mexico and of the U.S. Atlantic Ocean. The stranding of sea turtles is caused by many factors, including incidental capture in shrimping and fishing operations, entanglement, ingestion of marine debris, boat strikes, disease, storms, and hypothermic stunning. The STSSN was established in 1980 to collect information on and document strandings of live and dead sea turtles along the U.S. Gulf of Mexico and Atlantic coasts. The area includes the coasts from Maine through Texas, as well as portions of the U.S. Caribbean (STSSN 2012). As part of STSSN methodology, the United States coastline was divided into statistical stranding zones. Zones 1 through 21 include the Gulf coast from the Florida Keys through Texas (Figure 11.50). Important information regarding species composition, stock structure, life-history stage,

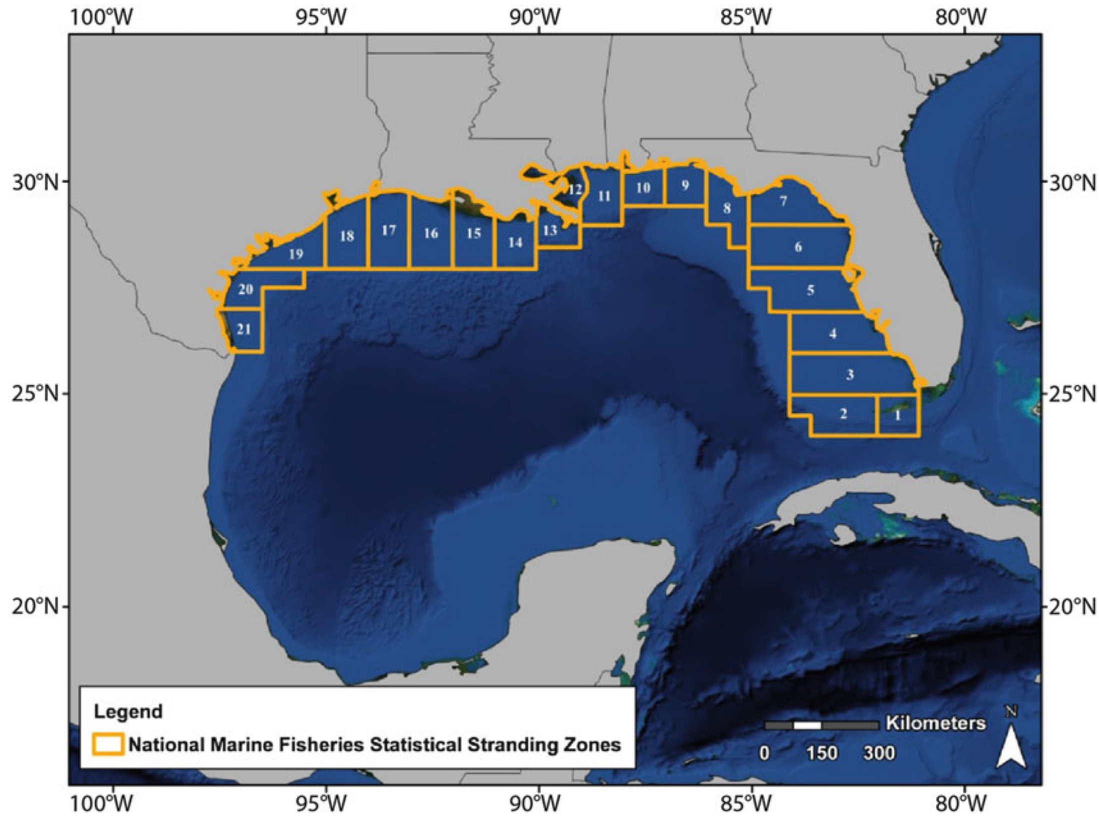

Figure 11.50. National Marine Fisheries Service statistical stranding zones for the U.S. Gulf of Mexico (from STSSN 2012).

distribution, migratory patterns, seasonality, habitat use, and causes of mortality may be inferred from the stranding data collected by the STSSN (NMFS et al. 2011). However, stranding data must be interpreted with caution, as the number of strandings recorded depends on many factors, including the surface currents, winds, time that has passed since the turtle died or was affected by the stressor that caused it to strand, and search effort (Epperly et al. 1996). In addition, while stranding data may represent a biased sample of the population, stranding distribution patterns can provide useful information on the distribution and abundance of juvenile and adult sea turtles, especially when large sample sizes and long time series are available (Meylan and Redlow 2006).

Available STSSN data prior to the Deepwater Horizon incident have been summarized for the Gulf of Mexico for each of the sea turtle species. The data available include preliminary data on the number of sea turtles stranded in each zone by species from 1986 through 2009. Strandings are defined as turtles that wash ashore, dead or alive, or are found floating dead or alive (generally in weak condition). In addition, per STSSN methodology, stranded hatchlings, as well as head-started turtles are excluded from the dataset, since their stranding may be an artifact of captive rearing and release (Teas 1993). Since the only data available from the STSSN were the number of strandings by species, an additional dataset was summarized; this dataset included sea turtle stranding data available for Florida from 1986 through 2006 and contained county, month, and size information for each stranded turtle, in addition to the number of strandings by species (FFWCC FWRI 2011b). When possible, the stranding data were summarized to demonstrate the data variability.

11.7.4.1 Kemp's Ridley Sea Turtle Strandings

From 1986 through 2009, 4,960 live and dead Kemp's ridleys were reported stranded on U.S. Gulf of Mexico beaches (STSSN 2012). More Kemp's ridley strandings (432 turtles) were reported on Gulf of Mexico beaches in 1994 than in any other year from 1986 through 2009 (Figure 11.51). The number of reported strandings along U.S. Gulf of Mexico beaches is not increasing in conjunction with the Kemp's ridley population increases based on the increased Gulf of Mexico nest counts (Figure 11.51), which illustrates the shortcomings of using strandings data to describe population abundance. Alternatively, this lack of increase in strandings, along with the increase in the nesting population in recent years, suggests that general conservation measures in the Gulf of Mexico are effective.

Of the 4,960 live and dead Kemp's ridleys reported stranded on Gulf of Mexico beaches from 1986 through 2009, approximately 45 % of them (2,242 turtles) were reported from Texas beaches (STSSN 2012). More Kemp's ridleys were typically reported stranded on Texas beaches each year from 1986 through the early 2000s than on Gulf coast beaches in Louisiana, Mississippi, Alabama, and Florida (Figure 11.52). While there is a lot of variability and a careful and detailed analysis is not available, the number of Kemp's ridley strandings reported for Texas and Louisiana beaches appears to be decreasing since a high in 1994, and Kemp's ridley strandings reported for Florida Gulf coast beaches appears to be significantly higher than in 1986 (Figure 11.52).

Along the Florida Gulf coast, few Kemp's ridleys were reported stranded along the Nature coast from Pasco through Jefferson counties. More Kemp's ridley strandings were reported on the southwest Gulf coast than on Panhandle beaches (Figure 11.53). Similar to what has occurred along the Florida Gulf coast beaches as a whole (Figure 11.52), Kemp's ridley strandings

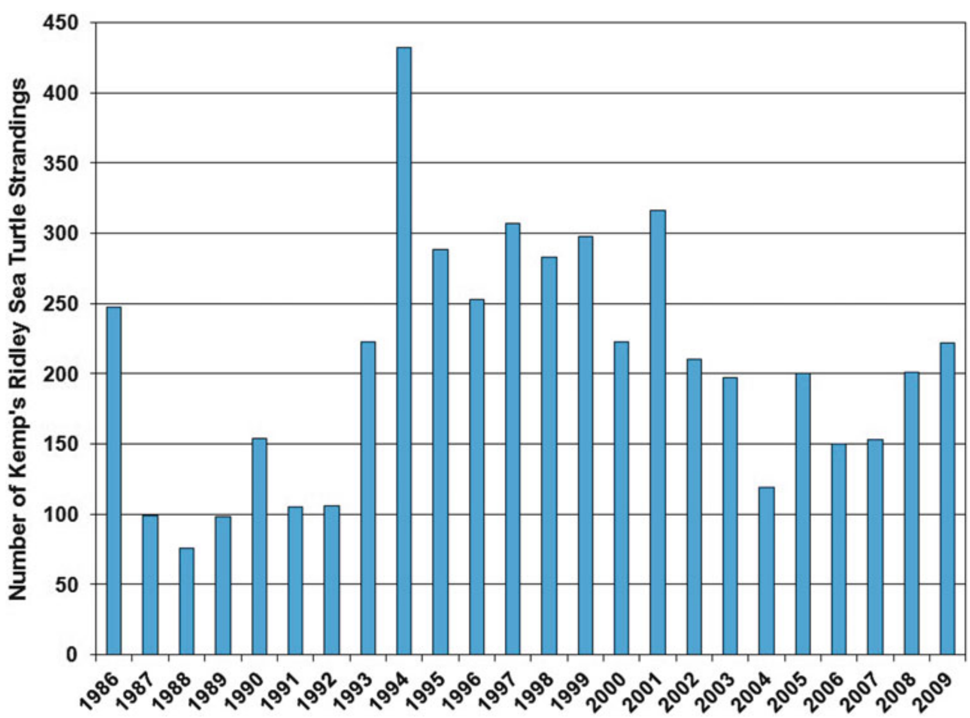

Figure 11.51. Number of reported Kemp's ridley sea turtle strandings (both live and dead turtles) on U.S. Gulf of Mexico beaches from 1986 through 2009 (from STSSN 2012).

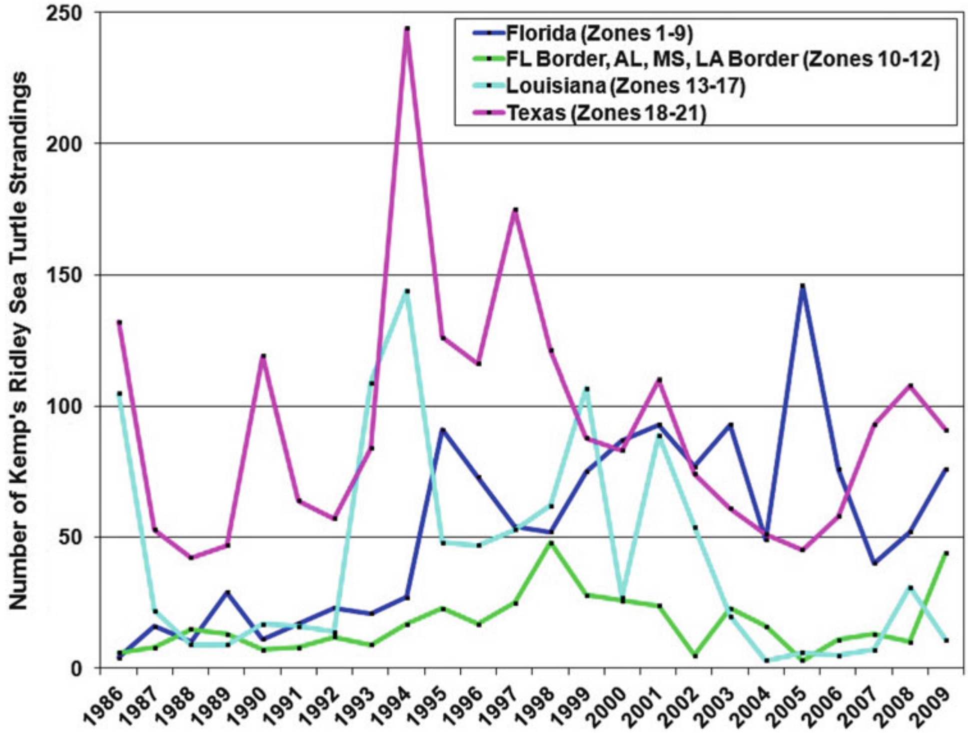

Figure 11.52. **Number of reported Kemp's ridley sea turtle strandings (both live and dead turtles) on Florida, Alabama, Mississippi, Louisiana, and Texas Gulf coast beaches from 1986 through 2009 (from STSSN 2012).**

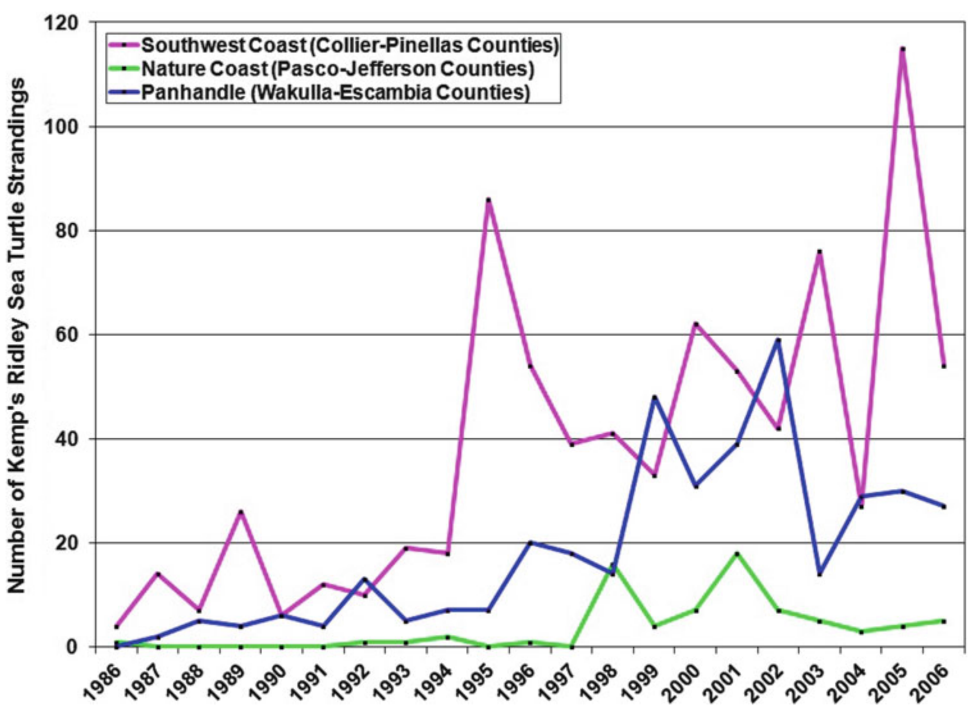

Figure 11.53. **Number of reported Kemp's ridley sea turtle strandings (both live and dead turtles) along the Florida Gulf coast by region from 1986 through 2006 (from FFWCC FWRI 2011b).**

reported for Florida Gulf coast regions appear to be increasing, with most of the strandings driven by southwest and panhandle regions of Florida (Figure 11.53).

Shaver (2005, 2006a) reported that more adult Kemp's ridleys were found stranded in Texas (mostly dead) than in any other state in the United States during each year from 1986 to 2003, and most of the turtles found on south Texas Gulf beaches occurred when Gulf of Mexico waters were open to shrimp trawling during the spring and summer. Despite the reported high compliance with TED regulations since they were implemented in 1990, a relationship continued on the Texas coast between shrimping and strandings on beaches through 2003 (Shaver 1997; Lewison et al. 2003). In late 2000, an annual shrimp-trawling closure of Gulf waters off south Texas beaches out to 8 km (5 mi) from shore from December 1 through mid-May was established; therefore, since 2001, South Texas nearshore waters have been closed to shrimp trawling during the entire Kemp's ridley mating and nesting seasons (Shaver 2006b). This closure, as well as the existing annual Texas closure, which extends out to 200 nautical miles from mid-May until mid-July each year, may be contributing to the lower numbers of strandings in Texas in recent years (Figure 11.52), as well as to the recent significant increase in nesting on Texas beaches (Figure 11.12).

A comparison of Kemp's ridley strandings reported for the Florida Gulf coast indicates that most turtles that stranded from 1986 through 2006 were neritic juveniles between 20 and 64 cm (7.9 and 25.2 in) SCL (Figure 11.54). The highest average numbers of strandings were reported for the 30–34 cm (11.8–13.4 in) SCL size class, followed by the 40–44 cm (15.7–17.3 in) SCL size class. From 1986 through 2006, Kemp's ridleys appeared to strand along the Florida Gulf coast most often during May and April, while the lowest average numbers of strandings

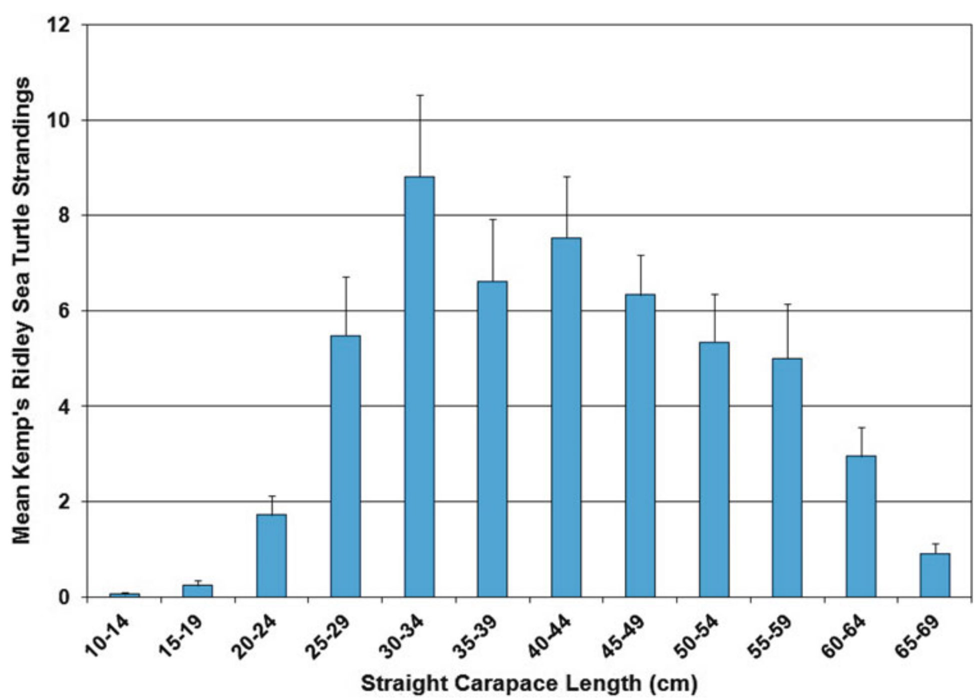

Figure 11.54. Number of reported Kemp's ridley sea turtle strandings (both live and dead turtles) along the Florida Gulf coast (Collier to Escambia counties) by size class from 1986 through 2006; error bars = standard error (from FFWCC FWRI 2011b).

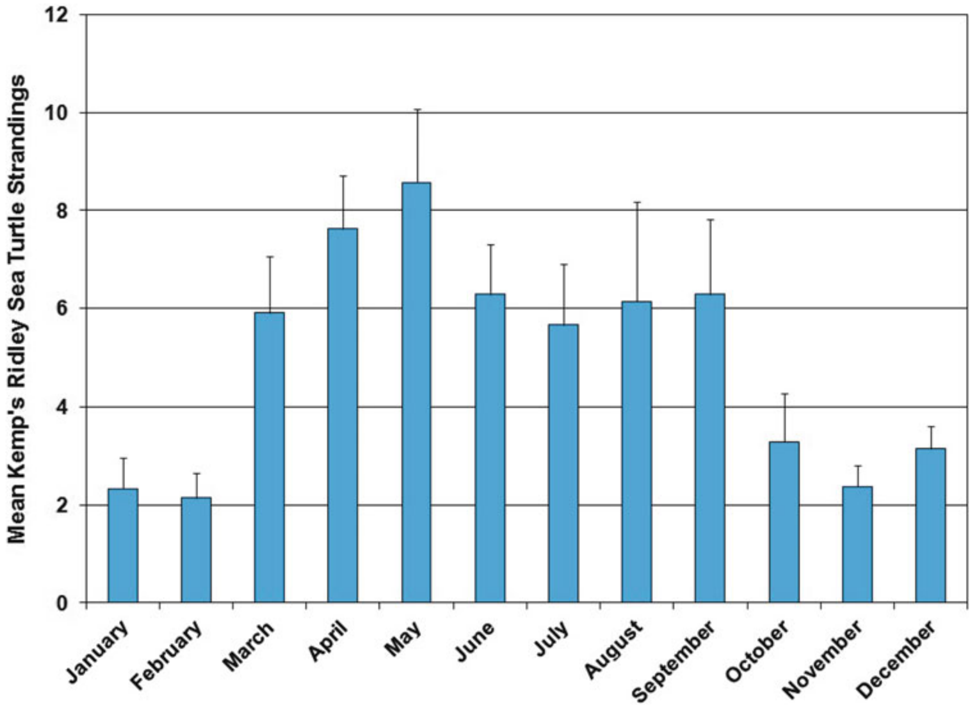

Figure 11.55. Number of reported Kemp's ridley sea turtle strandings (both live and dead turtles) along the Florida Gulf coast (Collier to Escambia counties) by month from 1986 through 2006; error bars = standard error (from FFWCC FWRI 2011b).

were reported for the winter months (Figure 11.55), which suggests that cold weather is not the main force driving strandings.

Prior to 2001, stranding data in Mexico were anecdotal or were reported only during the months of nesting activity at the main nesting beaches (NMFS et al. 2011). However, since 2001, year-round surveys of dead turtles stranded on the beaches of Tamaulipas have occurred (Figure 11.56). Strandings of Kemp's ridleys are most likely lower during the nesting season because Mexico implements a shrimp closure during the nesting season (NMFS et al. 2011). The increase in dead turtle strandings on Mexican beaches is thought to be due to an increase in the number of turtles available to be caught and killed. The data trends have been interpreted as an indication that TEDs and shrimp closures have decreased mortality of incidentally caught Kemp's ridleys (TEWG 2000; Frazier et al. 2007).

11.7.4.2 Loggerhead Sea Turtle Strandings

From 1986 through 2009, 9,289 live and dead loggerheads were reported stranded on U.S. Gulf of Mexico beaches (STSSN 2012). Approximately 15 % of all stranded loggerheads reported for the U.S. Gulf of Mexico from 1997 through 2005 were documented as having sustained some type of propeller or collision injury (NMFS and USFWS 2008). Loggerheads become caught in many materials, including fishing line, rope sacks, netting, and trap line, and from 1997 through 2005, almost 2 % of loggerheads reported stranded on Gulf of Mexico beaches were entangled in fishing gear, primarily monofilament line (NMFS and USFWS 2008). In the Gulf, high numbers of reported loggerhead strandings were associated with strong red tides in southwest Florida during some years (1995, 1996, 2000, 2001, 2003, and 2005) from

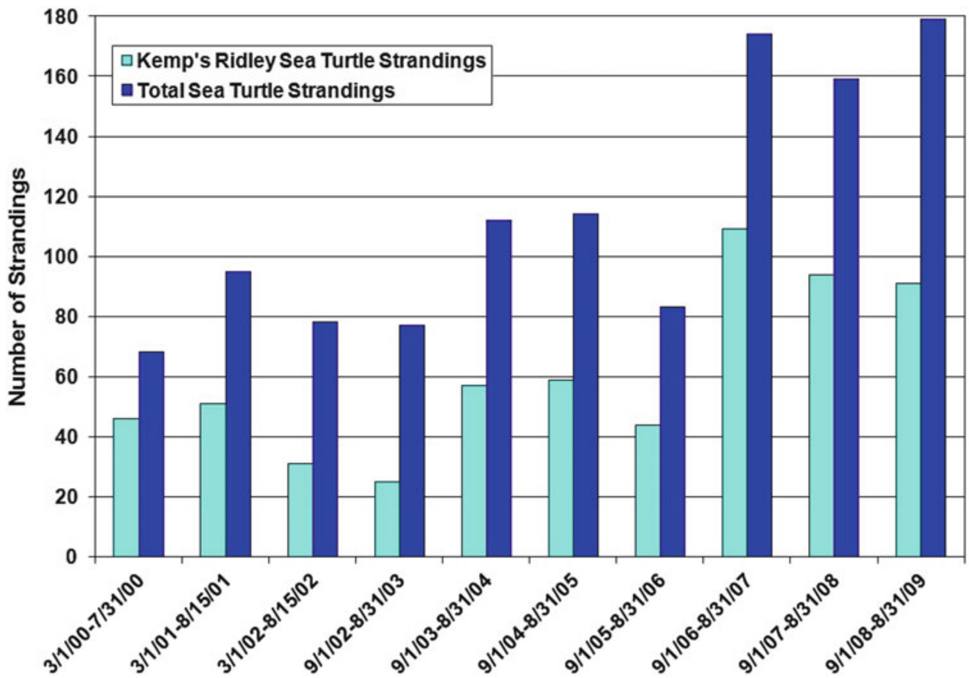

Figure 11.56. Number of dead turtles reported stranded along the coast of Tamaulipas, Mexico from March 2000 through August 2009 (from NMFS et al. 2011).

1995 through 2005 (FFWCC unpublished data, cited in TEWG 2009). From 1986 through 2009, more loggerhead sea turtles were reported stranded on Gulf of Mexico beaches in 2009 (790 turtles), than in prior years (Figure 11.57). Although no detailed and careful analysis is available, loggerhead strandings reported for the U.S. Gulf Coast appeared to be increasing, particularly since 1992 (Figure 11.57).

Of the 9,289 live and dead loggerhead strandings reported for U.S. Gulf of Mexico beaches from 1986 through 2009, very few were reported stranded on Alabama, Mississippi, or Louisiana beaches (Figure 11.58). From 1986 through 2009, loggerhead strandings reported for Texas beaches were within a similar range each year, approximately 100–200 turtles (Figure 11.58). The majority of loggerhead strandings for the U.S. Gulf of Mexico were reported for Florida beaches (Figure 11.58), with a high of 579 stranded loggerheads reported in 2009. Almost no loggerheads were reported stranded on beaches from Pasco to Jefferson counties, and fewer than 100 loggerheads were reported stranded each year on Florida Panhandle beaches from 1986 through 2006 (Figure 11.59). Fairly similar numbers of loggerheads were reported stranded along the southwest coast of Florida from 1986 through 2004; however, more than 300 loggerhead strandings were reported for the southwest coast in 2005, and almost 400 turtles were stranded on southwest coast beaches in 2006 (Figure 11.59).

From 1986 through 2006, most of the loggerheads reported stranded along the Florida Gulf coast ranged in size from 85 to 99 cm (33.5 to 38.9 in) (Figure 11.60); all of these size classes are considered adults (TEWG 2009). More loggerheads appeared to be stranded along the Florida Gulf coast during May and April, while the lowest average numbers of strandings occurred during January and December (Figure 11.61).

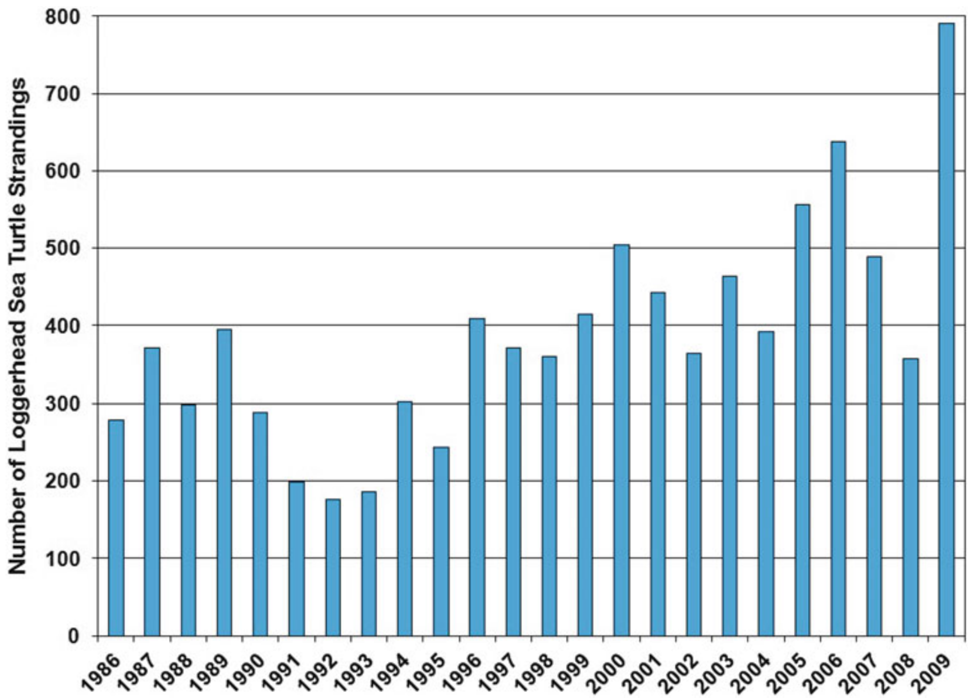

Figure 11.57. Number of reported loggerhead sea turtle strandings (both live and dead turtles) on U.S. Gulf of Mexico beaches from 1986 through 2009 (from STSSN 2012).

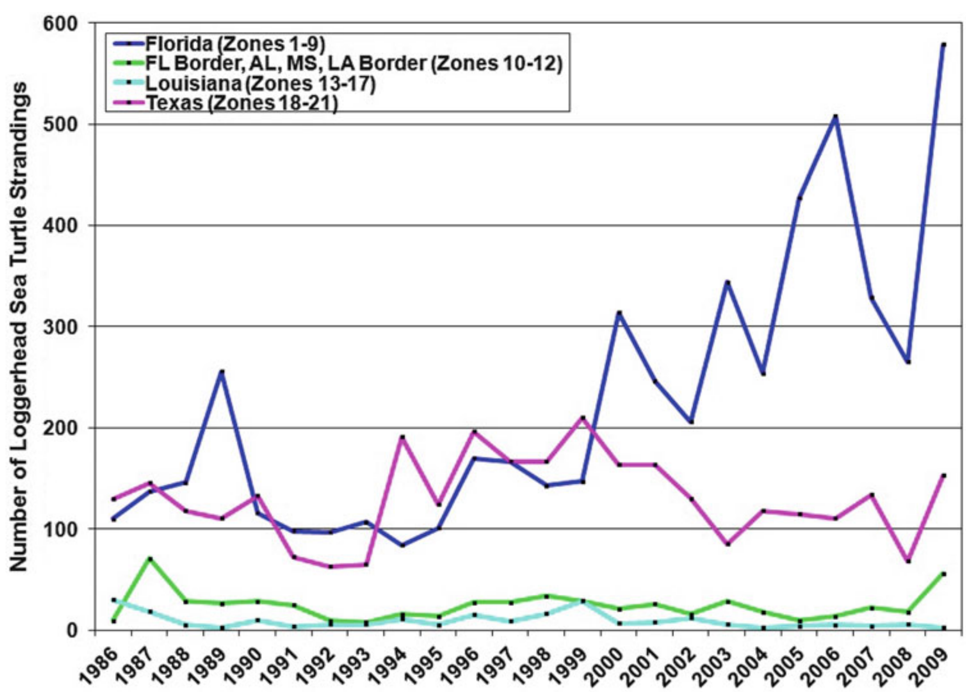

Figure 11.58. Number of reported loggerhead sea turtle strandings (both live and dead turtles) on Florida, Alabama, Mississippi, Louisiana, and Texas Gulf coast beaches from 1986 through 2009 (from STSSN 2012).

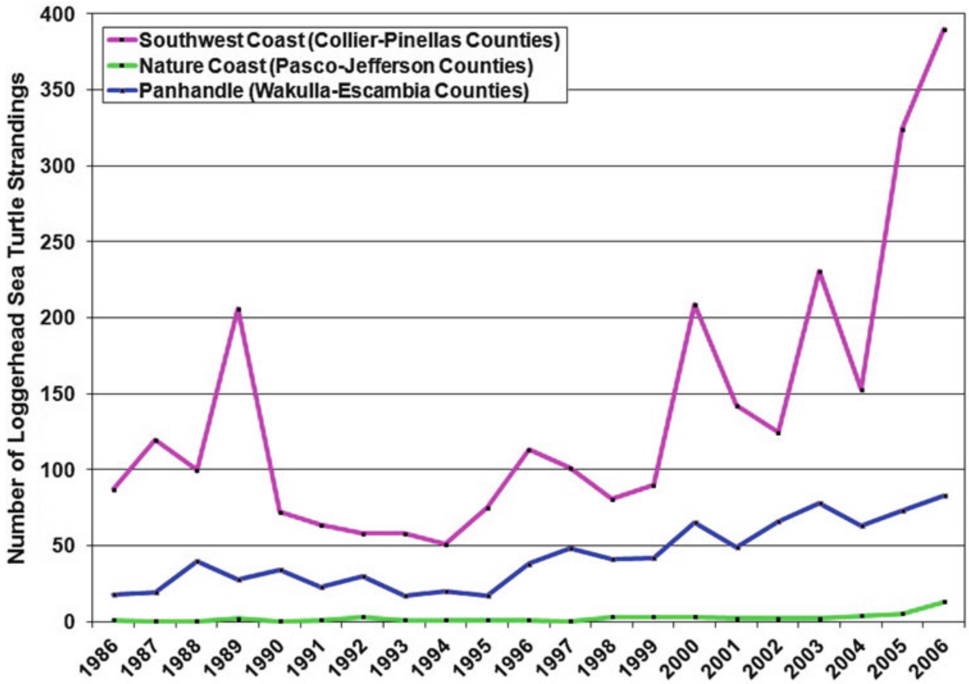

Figure 11.59. Number of reported loggerhead sea turtle strandings (both live and dead turtles) along the Floria Gulf coast by region from 1986 through 2006 (from FFWCC FWRI 2011b).

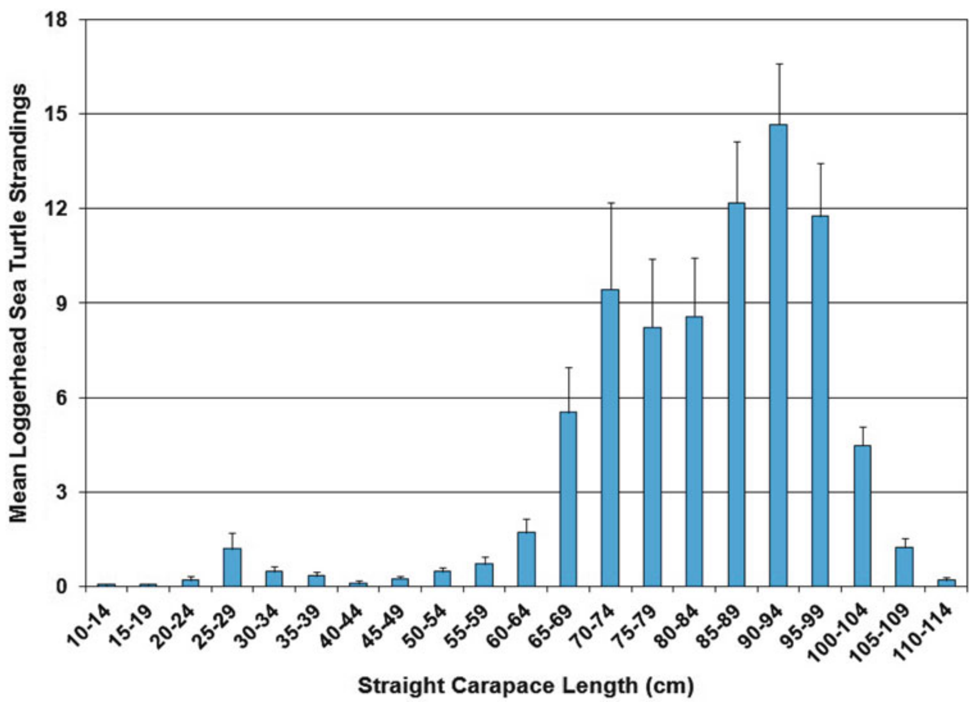

Figure 11.60. Number of reported loggerhead sea turtle strandings (both live and dead turtles) along the Florida Gulf coast (Collier to Escambia counties) by size class from 1986 through 2006; error bars = standard error (from FFWCC FWRI 2011b).

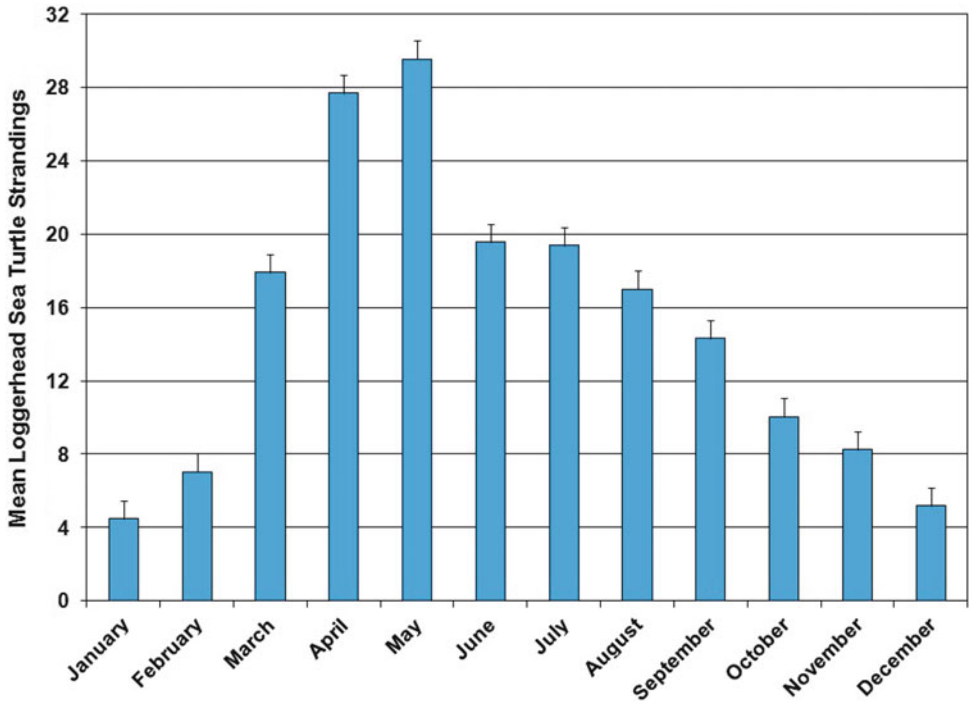

Figure 11.61. Number of reported loggerhead sea turtle strandings (both live and dead turtles) along the Florida Gulf coast (Collier to Escambia counties) by month from 1986 through 2006; error bars = standard error (from FFWCC FWRI 2011b).

11.7.4.3 Green Sea Turtle Strandings

From 1986 through 2009, 4,222 live and dead green sea turtles were reported stranded on U.S. Gulf of Mexico beaches (STSSN 2012). Green turtle strandings were most numerous in 2001 (almost 600 reports), followed by 2007 (more than 500 reports) (Figure 11.62).

Approximately equal proportions of the 4,222 green turtles stranded on U.S. Gulf of Mexico beaches from 1986 through 2009 were reported for Florida and Texas, and with few exceptions, similar numbers stranded on the beaches in both states were reported each year (Figure 11.63). Almost no green turtles were reported stranded on Alabama, Mississippi, or Louisiana beaches from 1986 through 2009 (Figure 11.63).

The majority of green turtles that were reported stranded on Florida Gulf coast beaches from 1986 through 2006 stranded on the southwest coast (Figure 11.64). Very few green turtles, typically fewer than ten turtles per year, were reported stranded on Florida Nature coast or Florida Panhandle beaches (Figure 11.64).

The highest numbers of reported green sea turtle strandings along the Florida Gulf coast from 1986 through 2006 were in the 30–44 cm (11.8–17.3 in) SCL size classes (Figure 11.65); all of these sizes classes are juvenile green turtles (Table 11.3). More green sea turtles were reported stranded along the Florida Gulf coast during January through March than any other period of the year from 1986 through 2006, while the lowest average numbers of reported strandings occurred during June (Figure 11.66).

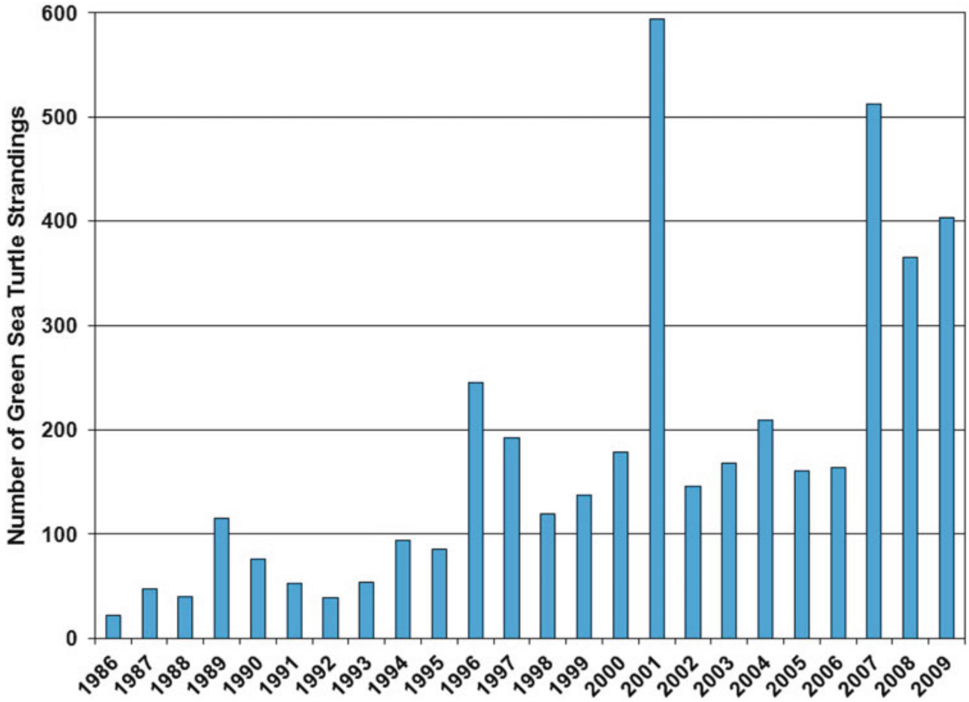

Figure 11.62. Number of reported green sea turtle strandings (both live and dead turtles) on U.S. Gulf of Mexico beaches from 1986 through 2009 (from STSSN 2012).

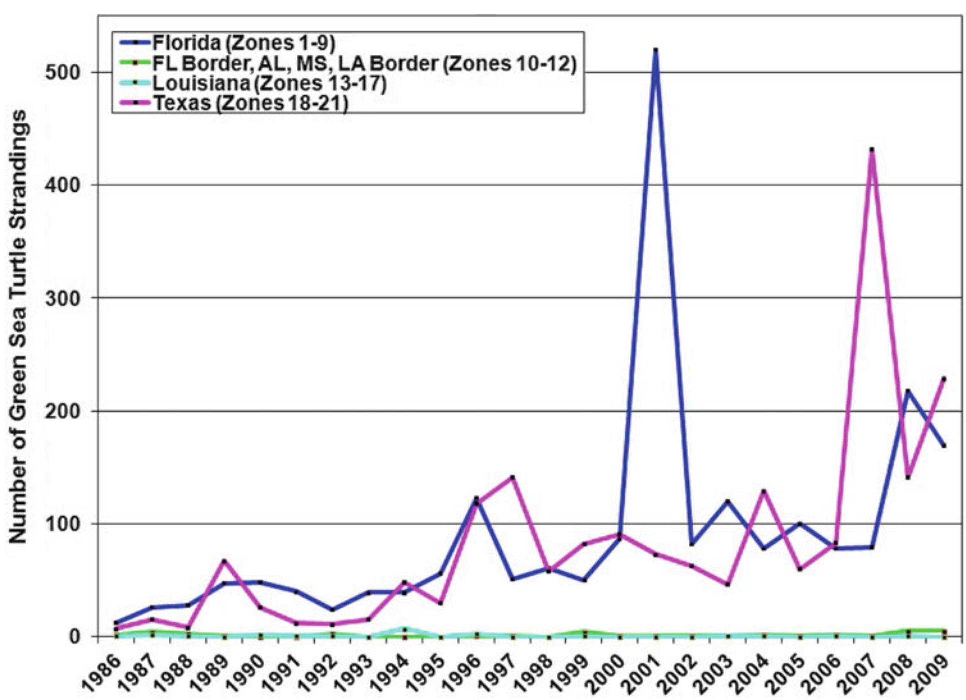

Figure 11.63. Number of green sea turtle strandings (both live and dead turtles) on Florida, Alabama, Mississippi, Louisiana, and Texas Gulf coast beaches from 1986 through 2009 (from STSSN 2012).

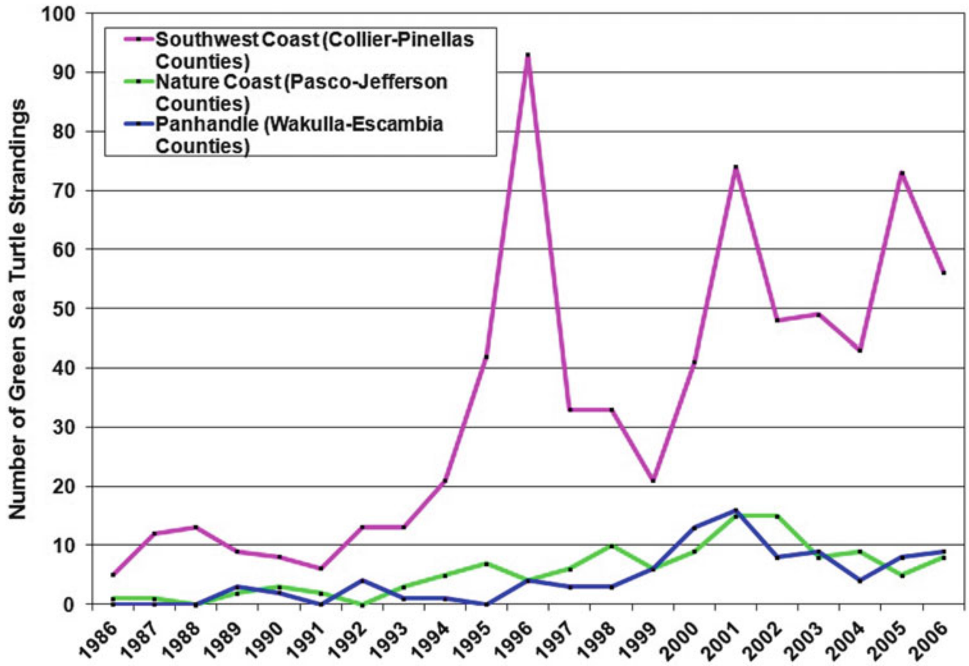

Figure 11.64. Number of reported green sea turtle strandings (both live and dead turtles) along the Florida Gulf coast by region from 1986 through 2006 (from FFWCC FWRI 2011b).

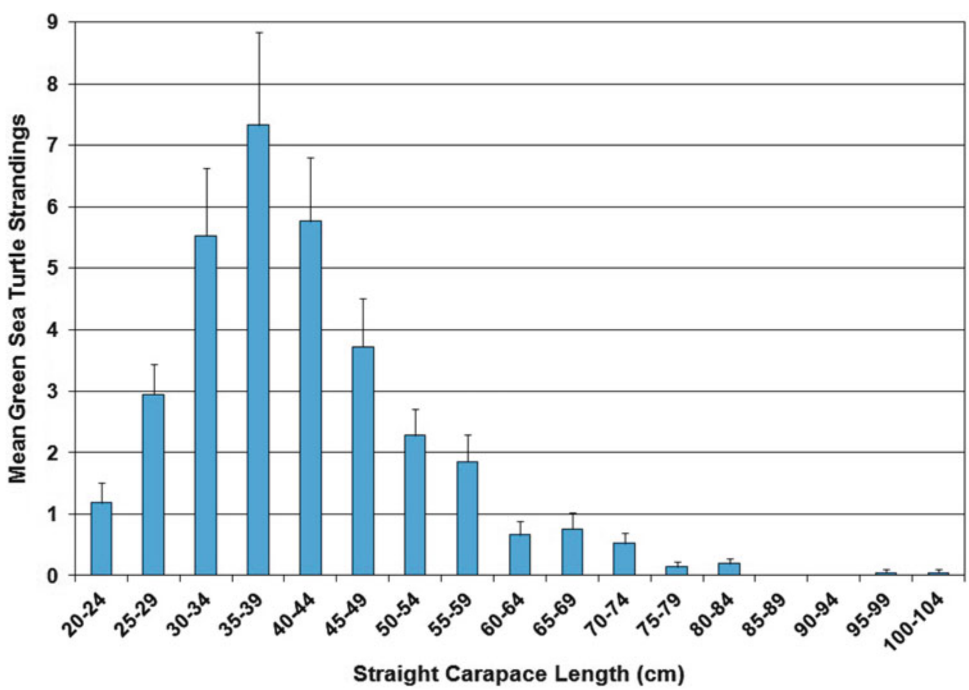

Figure 11.65. Number of reported green sea turtle strandings (both live and dead turtles) along the Florida Gulf coast (Collier to Escambia counties) by size class from 1986 through 2006; error bars = standard error (from FFWCC FWRI 2011b).

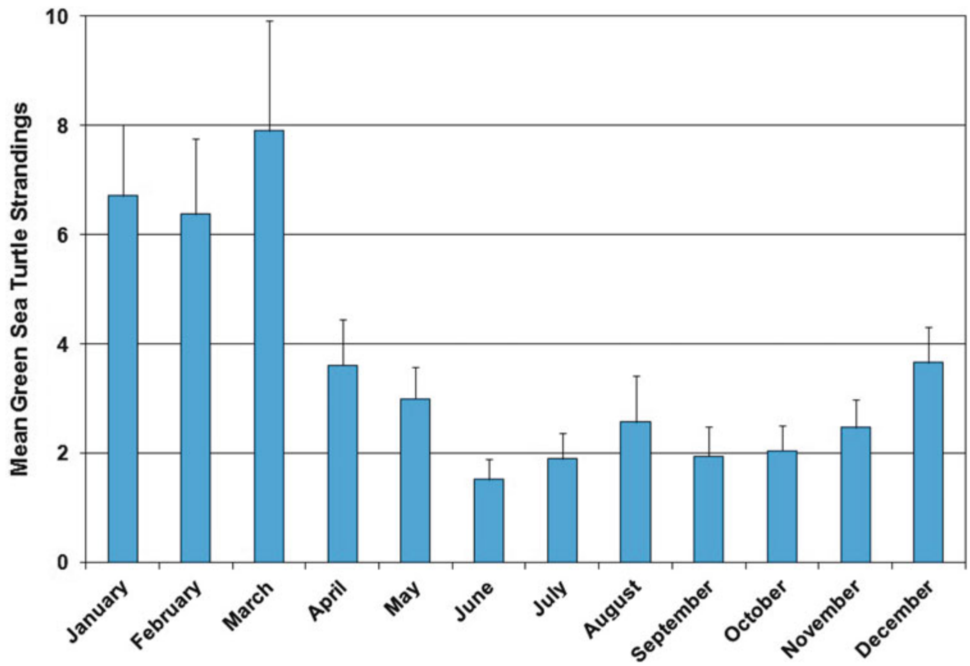

Figure 11.66. Number of reported green sea turtle strandings (both live and dead turtles) along the Florida Gulf coast (Collier to Escambia counties) by month from 1986 through 2006; error bars = standard error (from FFWCC FWRI 2011b).

11.7.4.4 Leatherback Sea Turtle Strandings

In the U.S. Gulf of Mexico, strandings of live and dead leatherbacks are far less frequent than strandings of other sea turtle species (STSSN 2012). For example, from 1986 to 2009, fewer than 30 leatherbacks typically stranded each year in the Gulf of Mexico (Figure 11.67). In addition, leatherback sea turtles represented about 2 % of the U.S. Gulf of Mexico strandings from 1986 through 2009. The yearly proportion of total strandings that consisted of leatherbacks ranged from 0.35 to 4.38 % (STSSN 2012).

The causes of leatherback strandings are most frequently attributed to boat strikes; interactions with fisheries, including entanglement in line, nets, and other gear; and ingestion of marine debris (NMFS SEFSC 2001). More frequent leatherback strandings in the Gulf tend to occur during the spring, coinciding with nearshore shrimp trawling activity (NMFS SEFSC 2001). Between 1980 and 1981, three leatherback carcasses washed ashore in Louisiana and their deaths appeared to be a result of interactions with shrimp trawls (Fritts et al. 1983a, b). In May and June 1993, 107 turtles were stranded around Grand Isle, Louisiana; the stranded turtles included two leatherbacks. These strandings were attributed to fatal interactions with the offshore longline ground fishery (Thompson 1993). In 1994, a total of 16 dead leatherback strandings in the Gulf of Mexico were attributed to interactions with shrimp trawlers (Steiner 1994).

While records of leatherback boat strikes were nonexistent prior to the late 1980s, 10 % of the 231 leatherback strandings involving boat strikes from 1980 through 1999 occurred in states on the Gulf of Mexico. Whether or not those strandings were caused directly by the boat strikes is unknown (NMFS SEFSC 2001).

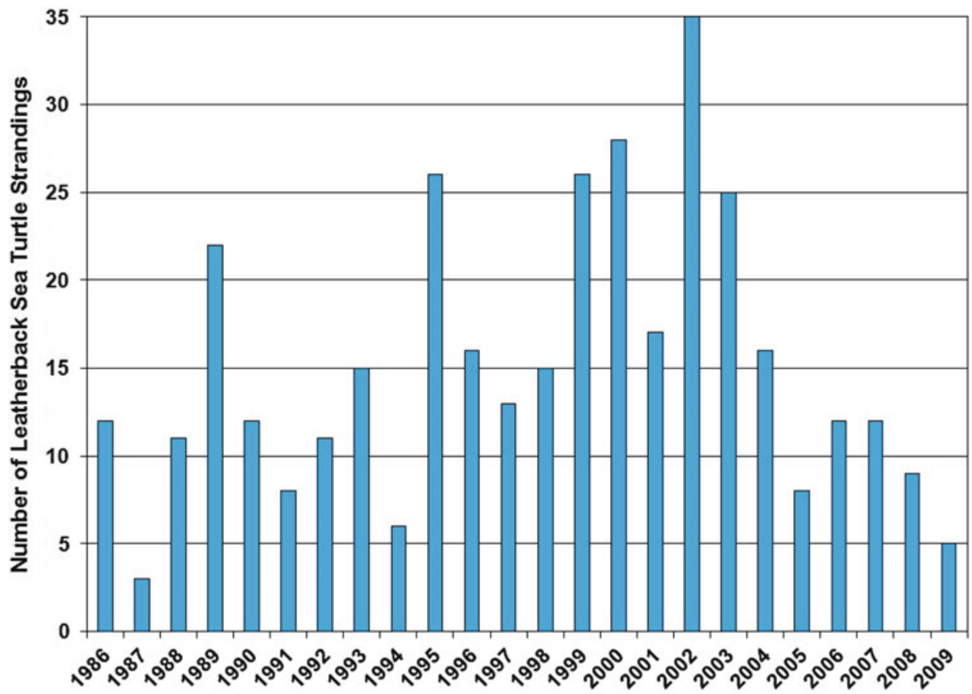

Figure 11.67. Number of reported leatherback sea turtle strandings (both live and dead turtles) on U.S. Gulf of Mexico beaches from 1986 through 2009 (from STSSN 2012).

From 1986 through 2009, 363 leatherbacks were reported to have stranded on U.S. Gulf of Mexico beaches (STSSN 2012). The highest number of leatherbacks (35) that were reported stranded on Gulf of Mexico beaches occurred in 2002 (Figure 11.67). Of the 363 leatherbacks reported stranded on U.S. Gulf of Mexico beaches, the majority was reported for Texas beaches, followed by beaches on the Florida Gulf coast (Figure 11.68). Fewer than five leatherbacks were typically reported stranded on Louisiana, Alabama, and Mississippi beaches each year (Figure 11.68).

From 1986 through 2006, more leatherbacks were reported stranded on Florida Panhandle beaches than on the remainder of the Florida Gulf coast (Figure 11.69). For each year, no more than seven leatherbacks were reported stranded in a given Florida Gulf coast region (Figure 11.69).

From 1986 through 2006, the highest average numbers of leatherback sea turtle strandings along the Florida Gulf coast were in the 140–149 cm (57.1–58.7 in) SCL size classes (Figure 11.70); this size range represents adult leatherbacks (Table 11.4). No leatherbacks were reported stranded along the Florida Gulf coast in September from 1986 through 2006, and the most reported average strandings occurred in March, followed by April (Figure 11.71).

11.7.4.5 Hawksbill Sea Turtle Strandings

Hawksbills are stranded on Gulf of Mexico beaches during all months of the year (NMFS and USFWS 1993). From 1986 through 2009, 474 live and dead hawksbill sea turtles were reported stranded on U.S. Gulf of Mexico beaches (STSSN 2012). Unlike other species of sea turtles, hawksbill strandings are often live turtles (Amos 1989). Hawksbills represented 1–5.6 % of the total turtles stranded each year on U.S. Gulf beaches. The 474 hawksbills that stranded on

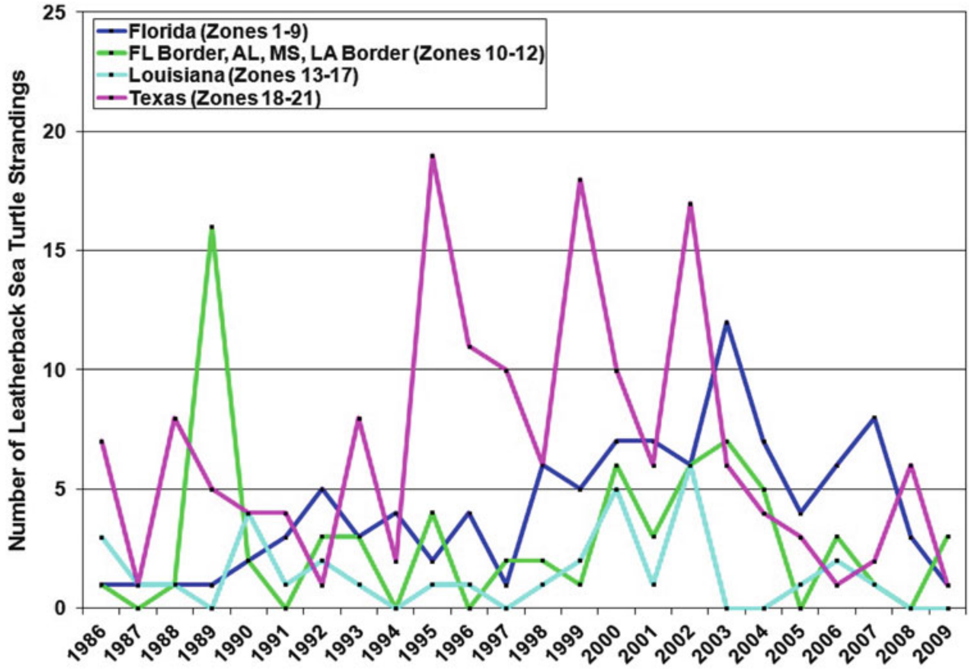

Figure 11.68. Number of reported leatherback sea turtle strandings (both live and dead turtles) on Florida, Alabama, Mississippi, Louisiana, and Texas Gulf coast beaches from 1986 through 2009 (from STSSN 2012).

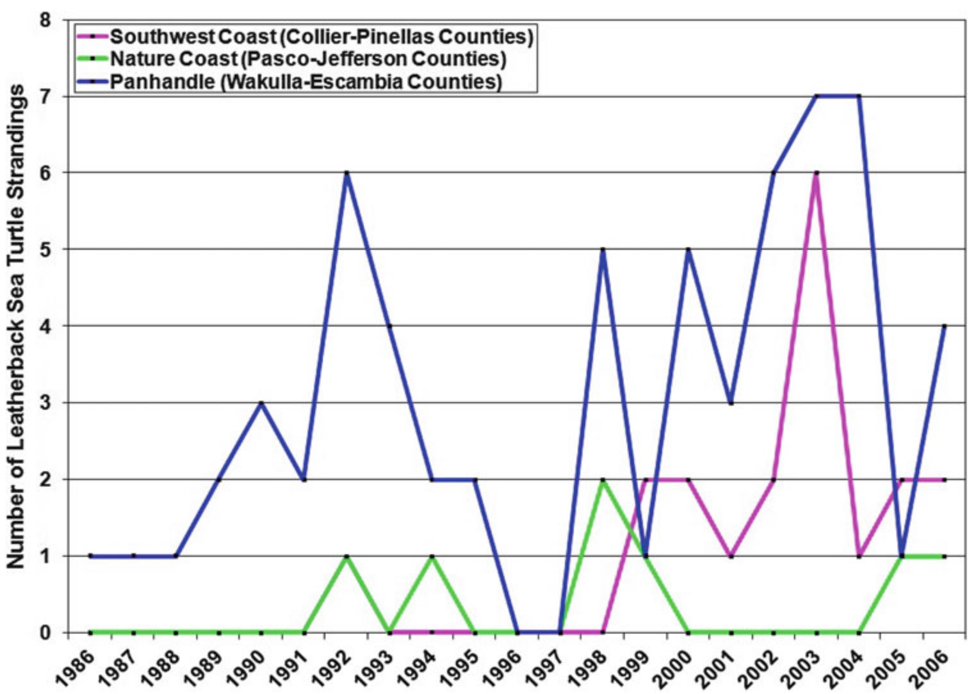

Figure 11.69. Number of reported leatherback sea turtle strandings (both live and dead turtles) along the Florida Gulf coast by region from 1986 through 2006 (from FFWCC FWRI 2011b).

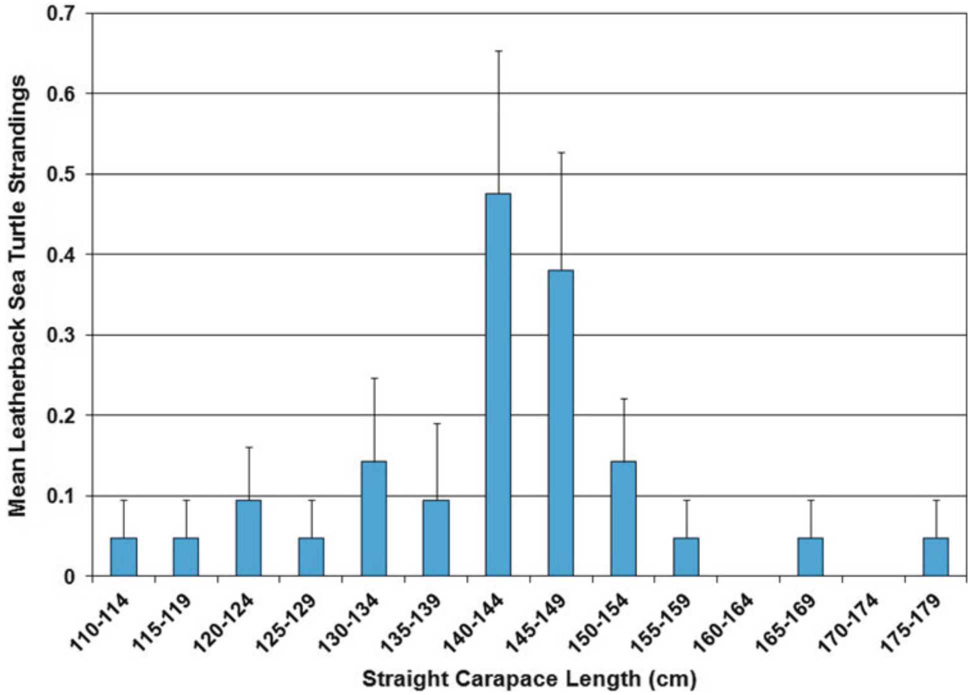

Figure 11.70. Number of reported leatherback sea turtle strandings (both live and dead turtles) along the Florida Gulf coast (Collier to Escambia counties) by size class from 1986 through 2006; error bars = standard error (from FFWCC FWRI 2011b).

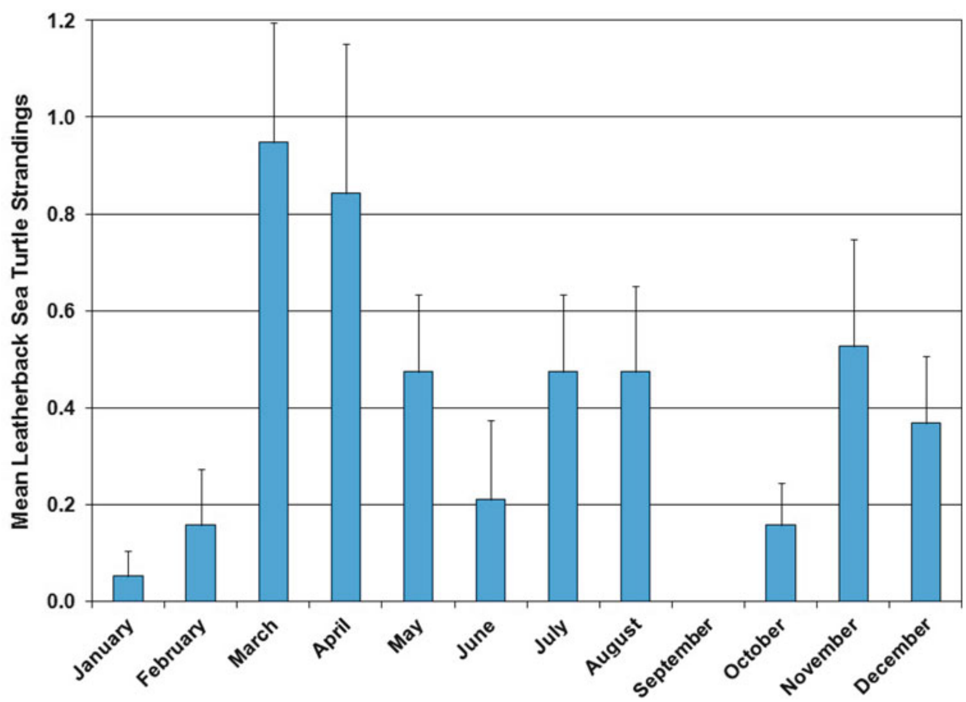

Figure 11.71. Number of reported leatherback sea turtle strandings (both live and dead turtles) along the Florida Gulf coast (Collier to Escambia counties) by month from 1986 through 2006; error bars = standard error (from FFWCC FWRI 2011b).

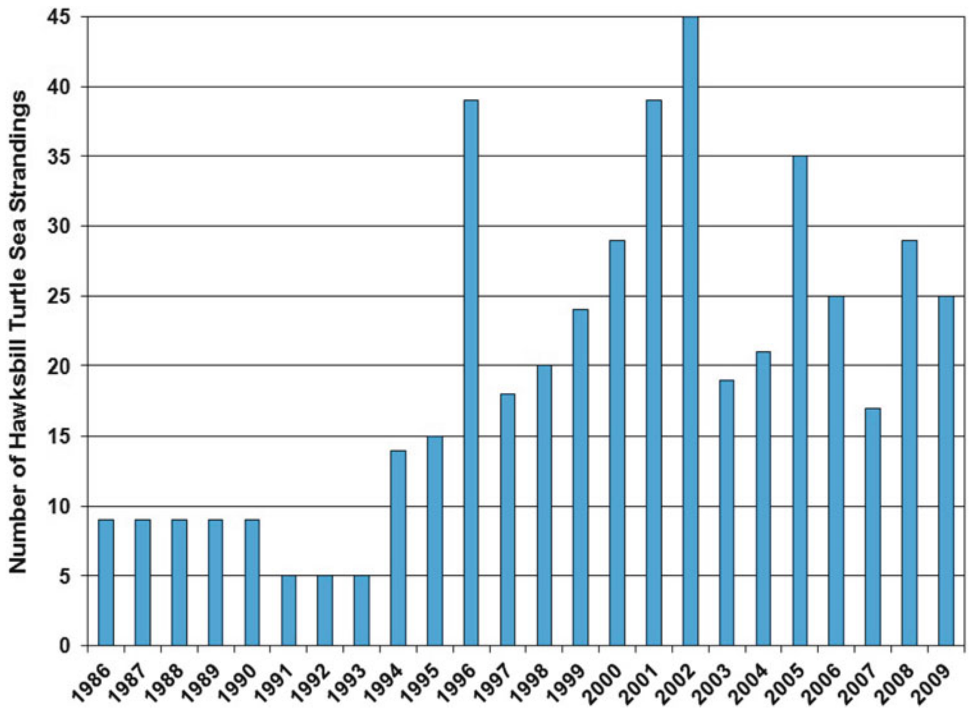

Figure 11.72. Number of reported hawksbill sea turtle strandings (both live and dead turtles) on U.S. Gulf of Mexico beaches from 1986 through 2009 (from STSSN 2012).

Gulf of Mexico beaches from 1986 through 2009 represented 2.5 % of the total strandings. Fewer than ten hawksbills were reported stranded each year on U.S. Gulf Coast beaches from 1986 through 1993. However, in 1994, the number of hawksbill strandings began to increase, with a high of 45 turtles in 2002 (Figure 11.72).

Most of the 474 hawksbill strandings reported on U.S. Gulf of Mexico beaches from 1986 through 2009 occurred in Texas (301 turtles) (Figure 11.73), with a high of 35 stranded hawksbills reported in 2002. No hawksbills were reported stranded in Alabama or Mississippi, and few to none were stranded on the Louisiana coast (Figure 11.73). On the Florida Gulf coast, 166 hawksbill strandings were reported from 1986 through 2009, and with the exception of 2001 when 27 hawksbill strandings were recorded, fewer than 15 hawksbills were reported stranded each year (Figure 11.73).

With the exception of 2001 when 23 hawksbills strandings were recorded, fewer than ten hawksbills were reported stranded each year on the southwest Florida coast from 1986 through 2006 (Figure 11.74). Few hawksbill sea turtles were reported stranded on the Florida coast from Pasco through Escambia counties from 1986 through 2006 (Figure 11.74).

The majority of hawksbill strandings reported along the Florida Gulf coast from 1986 through 2006 were juveniles (less than 60 cm [23.6 in] SCL); the highest average number of reported hawksbill strandings was in the 25–29 cm (9.8–11.4 in) SCL size class (Figure 11.75). From 1986 through 2006, more hawksbill sea turtles were reported stranded along the Florida Gulf coast during March than any other month of the year; the lowest average numbers of reported strandings occurred during November (Figure 11.76).

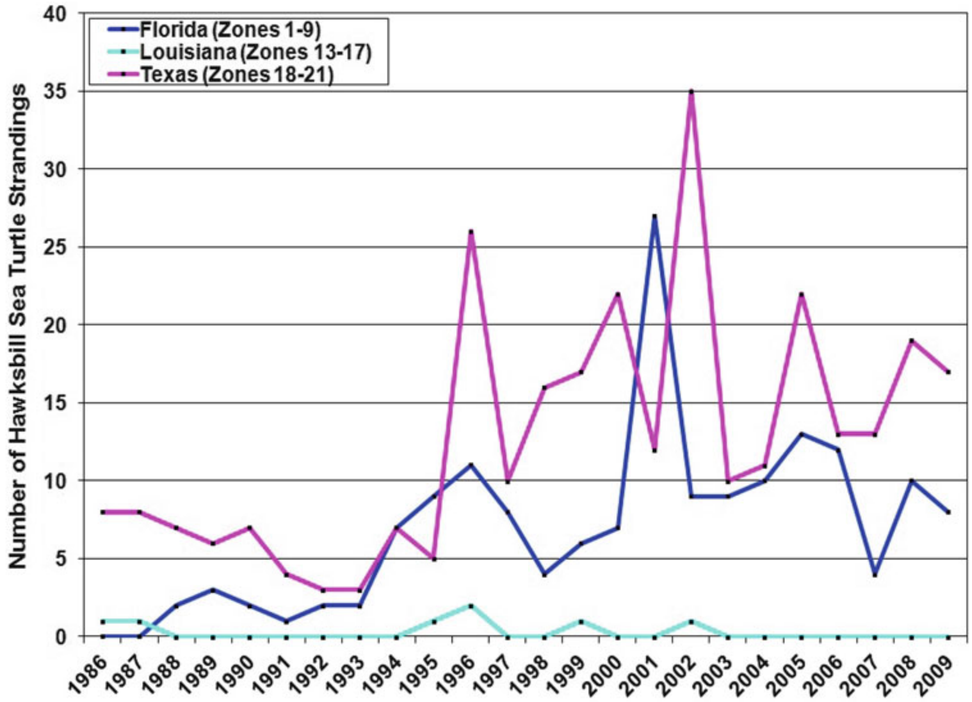

Figure 11.73. Number of reported hawksbill sea turtle strandings (both live and dead turtles) on Florida, Louisiana, and Texas Gulf coast beaches from 1986 through 2009. No hawksbill strandings were reported for Alabama or Mississippi (from STSSN 2012).

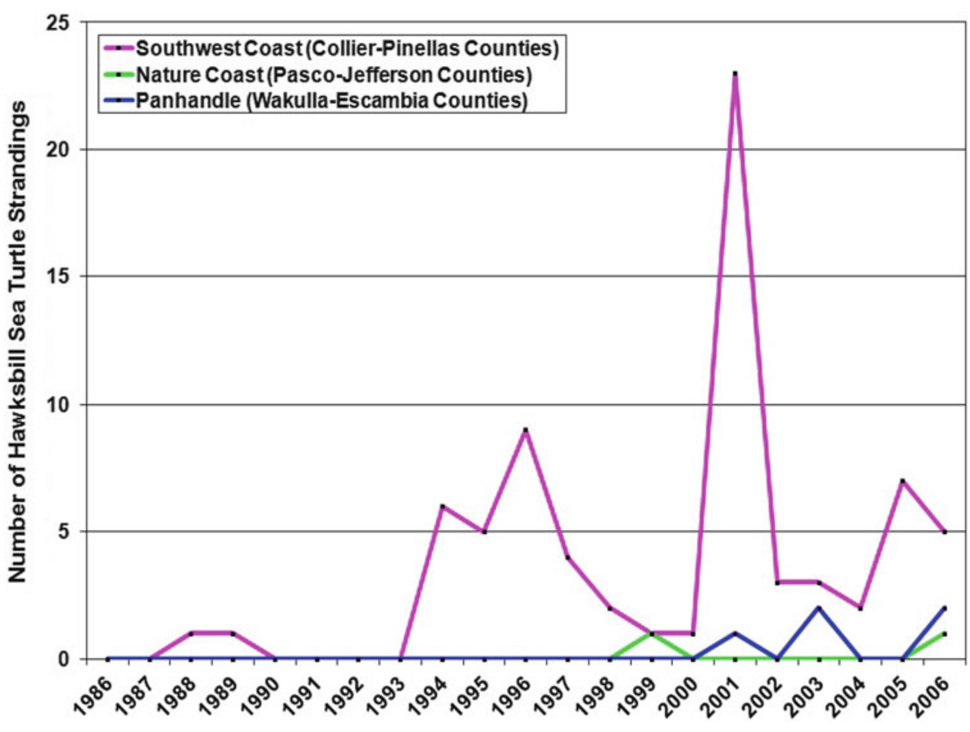

Figure 11.74. Number of reported hawksbill sea turtle strandings (both live and dead turtles) along the Florida Gulf coast by region from 1986 through 2006 (from FFWCC FWRI 2011b).

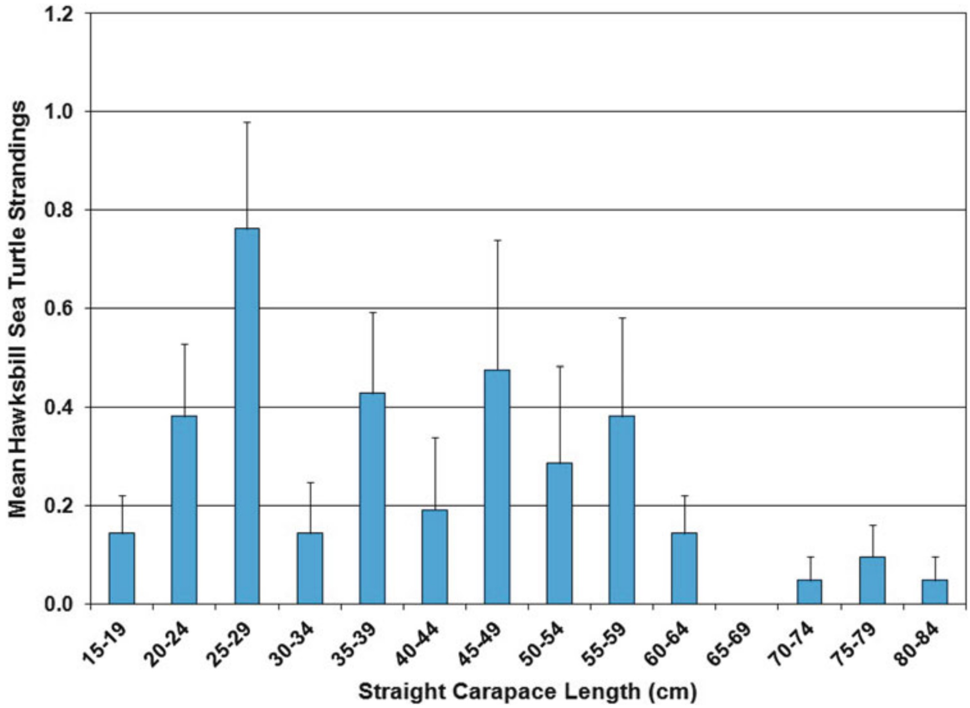

Figure 11.75. Number of reported hawksbill sea turtle strandings (both live and dead turtles) along the Florida Gulf coast (Collier to Escambia counties) by size class from 1986 through 2006; error bars = standard error (from FFWCC FWRI 2011b).

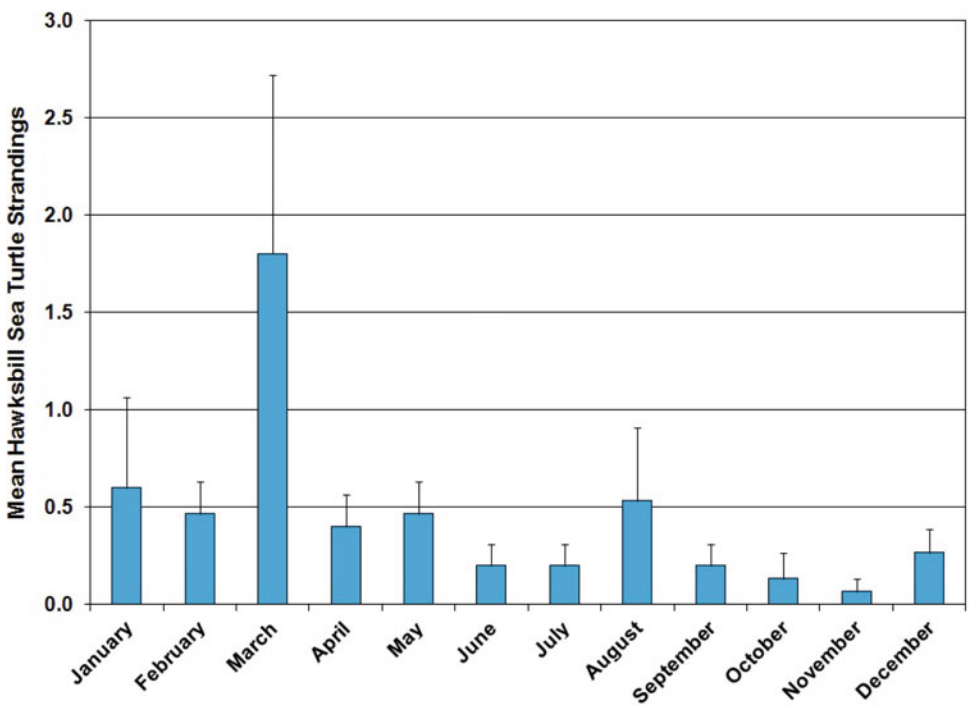

Figure 11.76. Number of reported hawksbill sea turtle strandings (both live and dead turtles) along the Florida Gulf coast (Collier to Escambia counties) by month from 1986 through 2006; error bars = standard error (from FFWCC FWRI 2011b).

11.8 SUMMARY AND DISCUSSION

In this chapter, available nesting beach data, tagging studies, satellite tracking studies, genetics studies, in-water observation and capture program data, stranding data, and other types of data have been summarized for Kemp's ridley, loggerhead, green, leatherback, and hawksbill sea turtles in order to characterize the distribution and abundance of sea turtles in the Gulf of Mexico prior to the Deepwater Horizon event. Life history information was also given for each of the five species of sea turtles that occur in the Gulf, and when available, Gulf of Mexico-specific data were presented. The threats associated with anthropogenic activities, as well as from natural events, that affect Gulf of Mexico sea turtle populations were described, and the impacts quantified, when possible, by synthesizing data on fisheries bycatch, turtle strandings, power plant entrapment and impingement, hypothermic-stunning events, and hurricane impacts. Because sea turtles are highly migratory and use terrestrial, neritic, and oceanic ecosystems throughout their long lifetimes, activities both within and outside the Gulf affect Gulf of Mexico sea turtle populations (NRC 2010).

The amount and quality of available distribution, abundance, and life history data, as well as information regarding threats, varied widely by sea turtle species, and the data are best described as a mosaic of information, with many significant spatial and longitudinal gaps throughout the Gulf of Mexico basin. For example, large amounts of high-quality information are available for Kemp's ridleys and loggerhead sea turtles because they have been the focus of significant research, recent recovery plans, status determinations, and comprehensive books (Bolten and Witherington 2003; Plotkin 2007c; NMFS and USFWS 2008; NMFS et al. 2011; USFWS and NMFS 2011). The amount of sea turtle data available for nesting beaches and nearshore waters along the U.S. Gulf Coast also varied; more information was available for Texas and Florida than for Louisiana, Mississippi, and Alabama. In addition, data available for sea turtle nesting beaches, as well as nearshore waters, along the Mexican Gulf coast varied by species and region, and long-term datasets are few and limited.

Sea turtles are difficult to study because they spend most of their lives in the ocean, are widely distributed, and have long life spans (Holder and Holder 2007; Witherington et al. 2009; NRC 2010). Consequently, significant gaps exist in the data available by species, as well as by life stage. In addition, the time frames over which data collection must occur to adequately assess population parameters for slow-to-mature, long-lived species, such as sea turtles, are daunting (Heppell et al. 2003). The most common datasets available were typically for long-term beach nesting. These data were heavily relied upon to describe Gulf of Mexico sea turtle nesting populations, aware of the fact that nesting females represent a small portion of the overall population and that extrapolating from individual nesting beaches to the entire population is inadequate at best (Heppell et al. 2007; NRC 2010). In-water studies, which have their own limitations, such as limited geographic scope and differential habitat use, continue to be lacking, although recent studies are encouraging (Cuevas et al. 2008, 2012; Fossette et al. 2010; Hart and Fujisaki 2010; Hart et al. 2012a, b; Avens et al. 2012; Witherington et al. 2012). Even though stranding data represents a biased population sample (Epperly et al. 1996; Meylan and Redlow 2006; Frazier et al. 2007), the information was used to provide a basic view of the threats sea turtles face in the Gulf of Mexico and the relative presence of a species in a given geographic area. Both the STSSN and Florida datasets that were summarized are long term, have large sample sizes, and provide rudimentary information regarding the distribution and relative abundance of juvenile and adult sea turtles, migratory patterns, seasonality, habitat use, and causes of mortality. Recent analyses of long-term fisheries bycatch data make it possible to estimate the impacts associated with the incidental bycatch of sea turtles in commercial fisheries, although the uncertainty associated with the estimates is high (Conant et al. 2009; Kot et al. 2010; Bolten et al. 2011; Finkbeiner et al. 2011; Ponwith 2011). Despite the data gaps and limitations associated with specific datasets, the current conditions of the five species of sea turtles that occur in the Gulf of Mexico have been described qualitatively, and sometimes quantitatively.

Kemp's Ridley Sea Turtles. In the Gulf of Mexico, the Kemp's ridley sea turtle has made a remarkable recovery from the brink of extinction (Heppell et al. 2007; Crowder and Heppell 2011), although the current nesting populations remain far below historical levels. The increased abundance and nesting of Kemp's ridleys in the Gulf of Mexico in recent years is most likely a result of many activities that affect all Kemp's ridley life stages—conservation and education efforts in both Mexico and the United States, the Kemp's Ridley Recovery Program, the Kemp's Ridley Head Start Experiment, the elimination of direct harvest of eggs and adult turtles, nest and hatchling protection, TED use, shrimp fishery closures, and the reduced Gulf of Mexico shrimp trawling effort in both the United States and Mexico (Shaver and Wibbels 2007; NMFS et al. 2011; Crowder and Heppell 2011). Despite the partial success story, there are threats to Kemp's ridleys in the Gulf that could cause significant impacts to the population and affect its continued recovery, particularly the incidental bycatch. The future of Kemp's ridleys is optimistically bright, and nesting numbers at Rancho Nuevo and PAIS continue to increase; however, because of their mass nesting at a single site in the western Gulf and their restricted Gulf of Mexico distribution, a significant event or the synergistic effects of multiple threats in the Gulf could have catastrophic effects on the recovering Kemp's ridley population.

Loggerhead Sea Turtles. The life history, distribution, and abundance of the loggerhead sea turtle is probably better understood than all other sea turtle species that inhabit the Gulf of Mexico (Bolten and Witherington 2003; USFWS and NMFS 2011). While the loggerhead is distributed globally, the goal of this chapter was to describe contemporary conditions of Gulf sea turtles; information for populations that occur in the Gulf of Mexico during some portion of their life cycle was the focus. These populations included the Peninsular Florida, northern Gulf of Mexico, Dry Tortugas, and Greater Caribbean subpopulations of the Northwest Atlantic Ocean DPS (USFWS and NMFS 2011). The annual nest counts from 2001 through 2009 on Florida Gulf coast and Atlantic beaches showed a decreasing trend, but in 2010 and 2011, loggerhead nest counts on surveyed Florida beaches were back to numbers similar to those recorded in 2000, indicating that the population of nesting females may have stabilized. Given the lack of a long historical record, it is not known whether these types of fluctuations are common or if the recent temporary decrease observed was caused by an acute factor. A recent analysis of annual loggerhead nest counts for the northern Gulf of Mexico subpopulation indicated a significant declining trend; however, a longer time series of nesting data is needed for an adequate evaluation (NMFS and USFWS 2008). More years of data are also needed in order to detect a trend in loggerhead nest counts for the Dry Tortugas subpopulation. Nesting for the Greater Caribbean subpopulation appears to be stable; however, a trend analysis is challenging because standardized surveys are few, survey efforts have changed, and low-level nesting occurs in many locations (NMFS and USFWS 2008).

High cumulative anthropogenic threat levels were estimated for oceanic and neritic juveniles and adults of the Northwest Atlantic Ocean DPS during the most recent loggerhead status review (Conant et al. 2009; Bolten et al. 2011). The loggerhead mortalities associated with these high threat levels resulted primarily from fisheries bycatch. Significant loggerhead mortality occurred in longline fisheries, bottom and mid-water trawl fisheries, dredge fisheries, gillnet fisheries, and pot/trap fisheries that occur not only in U.S. waters (Conant et al. 2009; Bolten et al. 2011; Ponwith 2011), but also in the Azores, the Mediterranean, on the Grand Banks, in Canadian waters, and in other locations throughout the range of the northwest Atlantic Ocean loggerhead population. Loggerhead sea turtles interacted with more U.S. fisheries (17 of 18 analyzed fisheries) than any other sea turtle species in a recent cumulative estimate of U.S. fisheries bycatch from 1990 through 2007 (Finkbeiner et al. 2011). This high level of interaction was thought to be due to their large nesting assemblages in Florida (as well as throughout the southeast and along the Gulf of Mexico) and their annual migrations to higher latitudes (Plotkin and Spotila 2002). The large range of northwest Atlantic Ocean neritic and oceanic juvenile and adult loggerheads overlaps significantly with coastal and oceanic areas where many fisheries occur, which unfortunately results in the death of thousands of loggerheads each year.

Green Sea Turtles. Despite being greatly depleted in the past, green turtle populations in the Gulf of Mexico are increasing. Green turtle nesting along the Mexican Gulf coast has increased in recent years and remains relatively stable. In addition, nesting at major rookeries in the region, such as Tortuguero, Costa Rica, and the Florida east coast, including ACNWR, has increased significantly since the 1970s. While fibropapilloma tumors have been reported for all sea turtle species, the frequency of these tumors is much higher in green turtles than in other species of sea turtles; this disease remains a threat to green sea turtles (Witherington et al. 2006b; NMFS and USFWS 2007c). Green turtles are also dependent on the continued maintenance of healthy seagrass meadows in their foraging areas. Although the impacts to green turtles resulting from the incidental bycatch in fisheries are not as significant as those for loggerheads, many green turtles die each year from fisheries interactions.

Leatherback Sea Turtles. Because leatherback sea turtles spend most of their lives in the oceanic zone, distribution and abundance data for leatherbacks in the Gulf of Mexico are incomplete. In addition, large life history data gaps still exist, especially for post-hatchlings and juveniles. However, the available data do verify that leatherbacks use the Gulf as a foraging area, and recent tracking studies have demonstrated that the Gulf of Mexico may be a significant year-round foraging ground for leatherbacks that nest along the Caribbean coast (Evans et al. 2007; Fossette et al. 2010). In the Gulf of Mexico, leatherbacks are often found in areas containing an abundance of jellyfish, their main prey; and they are less abundant than other sea turtle species, such as Kemp's ridleys, loggerheads, and hawksbills. Because of the low numbers of leatherback nesting in the Gulf, the significant gaps in available leatherback data and information, as well as their extensive migrations and large home ranges, determining the status of the Gulf of Mexico leatherback population is uncertain. However, leatherback nesting has increased significantly in Florida since the late 1970s, which may indicate that the leatherback population in the general Gulf of Mexico, Caribbean, and northwest Atlantic Ocean area is stable or increasing. A major threat to leatherbacks in the Gulf continues to be mortality as bycatch in pelagic longline fisheries (Kot et al. 2010; Finkbeiner et al. 2011).

Hawksbill Sea Turtles. Because millions of hawksbill sea turtles have been killed globally for the tortoiseshell markets of Asia, Europe, and the United States over the past 100 years, the current abundance of hawksbills in the general Gulf of Mexico, Caribbean, and northwest Atlantic Ocean area is only a fraction of what occurred historically (NMFS and USFWS 2007e). In the Gulf, the most important hawksbill rookery is located on Yucatán Peninsula beaches in Mexico (Abreu-Grobois et al. 2005; NMFS and USFWS 2007e; del Monte-Luna et al. 2012). Analyses of hawksbill nesting data from 1980 through 2010 for the Mexican Gulf of Mexico/Caribbean coasts attributed the apparent recent declines in nesting to the low-level taking of turtles, impacts to the hawksbill's marine habitats, constant expansion of beach development, and increases in sea surface temperature, hurricane activity, and regional sea level rise (Abreu-Grobois et al. 2005; del Monte-Luna et al. 2012).

The destruction of nesting and foraging habitat is affecting hawksbills that nest along the beaches and use the nearshore areas of the Gulf of Mexico (Garduño-Andrade et al. 1999). Hawksbills are also dependent on coral reefs—one of the world's most endangered ecosystems—for food and shelter (NMFS and USFWS 1993, 2007e). In addition, although the trade in hawksbill products has declined significantly compared to historical levels, both the illegal and legal trade is still active and significant (Mortimer 2008). The low hawksbill bycatch data levels suggest that the threat of death or injury from bycatch in Gulf of Mexico fisheries is not substantial. However, given the low abundance of these populations, which renders them significantly vulnerable, it is crucial to ensure the continued protection of this species from all threats in general, and from fisheries bycatch, specifically.

Overview. This chapter has presented a significant body of data and literature on sea turtle nesting populations in the Gulf of Mexico. Most of that information focused on the northern and eastern Gulf coast, particularly on Florida beaches (e.g., Meylan et al. 1995). This information showed that north-central Gulf of Mexico beaches are essentially devoid of any significant nesting, in particular the beaches of east Texas, Louisiana, Mississippi, and Alabama. Sea turtle nesting in general increases east, west, and southwest from this north-central location and reaches its zenith around the Florida and Yucatán Peninsulas (Renaud 2001). Data available on nesting females showed that sea turtle populations in the Gulf generally exhibit a very low abundance relative to Atlantic regions outside the Gulf. This is particularly true for the loggerhead, whose nesting on Florida west coast beaches amounted to only 8.6 % of statewide nesting from 2001 through 2006 (Witherington et al. 2009). The only obvious exception to this general rule is the Kemp's ridley, whose main rookery is located in Rancho Nuevo, in northeast Mexico (NMFS et al. 2011). The very low nesting numbers indicate that all sea turtle populations in the Gulf of Mexico are particularly vulnerable to natural and anthropogenic impacts, perhaps more so than populations outside the Gulf. Moreover, the current abundance of sea turtles is so low in the Gulf mainly due to the myriad of impacts associated with the anthropogenic activities summarized in Table 11.6.

It is difficult to ascertain the abundance and trends of Gulf of Mexico sea turtle populations given the lack of long-term, in-water, systematic studies. Indeed, the Gulf may arguably be the most data-deficient basin in terms of its sea turtle populations. Efforts to determine the presence and abundance of all species in U.S. waters seem to have concentrated in the states of Florida and Texas, likely due to the presence of nesting beaches in these states, with Florida boasting by far the largest numbers (e.g., Meylan et al. 1995). Aerial surveys over Gulf of Mexico waters frequently have been used to address this deficiency of data. However, no reports exist regarding the southern and southwest Gulf of Mexico, and most of the reports available for the northern Gulf are point in time studies (e.g., over a season or a year) and lack the benefit of long-term systematic records that could be used to establish population trends.

11.9 FUTURE CONSIDERATIONS AND RESEARCH NEEDS

It has been recognized that commercial sea turtle fisheries, habitat destruction, and pollution, among other causes, have played a critical role in the decimation of sea turtles in U.S. waters (Witzell 1994). Because sea turtle nesting currently is scattered and not abundant throughout most of the rim of the Gulf of Mexico basin, future work should focus on collecting information on in-water populations in order to characterize the status of sea turtle populations in the basin. For example, large amounts of essential data on population abundance, particularly with regard to juveniles and adult turtles, can be gathered at foraging grounds, such as in the Bay of Campeche, off the Louisiana coast, and in northwest Florida waters in the case of the Kemp's ridley (Márquez-M. 1999).

A series of index sites have been identified in Florida waters that have yielded valuable information about the biology, distribution, and abundance of sea turtle populations in that state (Eaton et al. 2008). The concept of in-water index sites should be expanded to the entire Gulf of Mexico by creating a basin-wide network. Many techniques and approaches have been discussed to conduct this type of in-water work (Bjorndal and Bolten 2000). Work on in-water index sites may be complemented by aerial surveys (e.g., Witzell and Azarovitz 1996). A few major technical hurdles will need to be overcome if this gigantic task is to be pursued. Two of these hurdles are the cost of undertaking such a project in a large body of water like the Gulf of Mexico and the logistical constraints associated with flight altitude, typically around 150 m (492 ft), that complicate the detection of turtles comparable in size to adult Kemp's ridleys (Witzell and Azarovitz 1996). To address these two significant problems, a possible approach would be to use high-resolution cameras mounted on surveyor airplanes. This would keep surveying crews to a minimum, while allowing for the possibility of zooming in on the images for positive species identification and the creation of an archive of important data. Future technological and legislative advances should make it possible for high-resolution imagery for the entire Gulf of Mexico from geostationary satellites, which would provide information in real time on relative abundance on every corner of the Gulf, not only of sea turtles but also of every major vertebrate that inhabits the Gulf, including birds, cetaceans, and schools of commercial fish. Until these advances occur, a more realistic approach might be to set up observers at a fraction of the more than 3,800 oil rigs that currently exist in the Gulf of Mexico, which are mostly concentrated in Louisiana and Texas waters. Such a program may provide a reasonably good idea of the seasonal relative abundance of sea turtles, at least in the northwestern Gulf. For other areas of the Gulf, index sites, such as the Chandeleur Islands in Louisiana and Waccasassa Bay in Florida, could be monitored.

Because of the many significant data gaps discussed above, the information needed for accurate abundance assessments, as well as for the calculation of important demographic parameters, for most sea turtle populations is typically not available (NRC 2010). Additional challenges to accurately estimate sea turtle populations include the limited spatial and temporal scopes of most sea turtle research projects and the lack of coordination of sea turtle data for a given species or region (e.g., the lack of comprehensive databases) (NRC 2010). Bjorndal et al. (2011) recently recommended that the following seven elements be included in strategic plans for the collection of essential sea turtle data: (1) integrate demography with abundance trends for multiple life stages and determine environmental effects on those parameters; (2) emphasize analyses of cumulative effects; (3) elucidate links among and within populations with new tools in genetics, statistical models, and tracking; (4) revise the permitting processes that now hinder peer-reviewed studies of critical processes and management alternatives for protected species; (5) encourage data sharing; (6) improve assessment tools for evaluation of anthropogenic impacts on populations by fostering interdisciplinary research among scientists, students, and managers; and (7) prioritize investments for research and monitoring.

In addition to revealing data gaps, this compilation of information for Gulf of Mexico sea turtle populations shows that sea turtle data are highly variable from year to year; an excellent example of this variability is the interannual fluctuations in green turtle nesting data. This variability highlights the importance of long-term datasets and explains why long-term trends, not year-to-year fluctuations, are critical in determining changes in sea turtle nesting populations. While determining if changes to sea turtle populations have occurred is extremely difficult, it is also not possible to determine the cause of population variability. Because multiple human-made and natural threats affect all life stages of sea turtles, each threat may affect a life stage or species differently. The effects of multiple threats may be synergistic, and impacts to a specific year class may not appear on the nesting beach for many years. These issues will continue to be a challenge in the future as new threats emerge and attempts are made to quantify impacts on sea turtle health and populations associated with environmental change due to urbanization and coastal development, global warming and sea level rise, fisheries exploitation and regulation, oil and gas exploration and production, and other anthropogenic effects, such as nutrient enrichment in the Gulf of Mexico. These factors and many others interact to challenge the development of effective strategies and measures to promote sea turtle conservation.

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REFERENCES

Abreu-Grobois FA, Guzmán V, Cuevas E, Alba Gamio M (eds) (2005) Proceedings, Memorias del Taller Rumbo a la COP 3: diagnóstico del estado de la tortuga carey (*Eretmochelys imbricata*) en la Peńinsula de Yucatán y determinación de acciones estratégicas. Telchac Puerto, Yucatán, Mexico, Marzo, pp xiv–75

Ackerman RA (1997) The nest environment and the embryonic development of sea turtles. In: Lutz PL, Musick JA (eds) The biology of sea turtles, vol 1. CRC Press, Boca Raton, FL, USA, pp 83–106

Addison DS (1996) Mean annual nest frequency for renesting loggerhead turtles (*Caretta caretta*) on the southwest coast of Florida. Mar Tur Newsl (MTN) 75:13–15

Addison DS (1997) Sea turtle nesting on Cay Sal, Bahamas, recorded June 2–4, 1996. J Bahamas Sci 5:34–35

Addison DS, Morford B (1996) Sea turtle nesting activity on the Cay Sal Bank, Bahamas. J Bahamas Sci 3:31–36

Alabama Department of Conservation and Natural Resources [ADCNR] (2012) Homepage. http://www.dcnr.state.al.us/. Accessed 20 November 2013

Alam SK, Brim MS (2000) Organochlorine, PCB, PAH, and metal concentrations in eggs of loggerhead sea turtles (*Caretta caretta*) from northwest Florida, USA. J Environ Sci Health B35:705–724

Alava JJ, Keller JM, Wyneken J, Crowder L, Scott G, Kucklick JR (2011) Geographical variation of persistent organic pollutants in eggs of threatened loggerhead sea turtles (*Caretta caretta*) from southeastern United States. Environ Toxicol Chem 30:1677–1688

Allard MW, Miyamoto MM, Bjorndal KA, Bolten AB, Bowen BW (1994) Support for natal homing in green turtles from mitochondrial DNA sequences. Copeia 1994:34–41

Alonso GT (2009) Situación Actual de la Tortuga Lora (*Lepidochelys kempii*) en el Golfo de Mexico. Proceedings, Memorias de la Reunión Nacional sobre Conservación de Tortugas Marinas. Veracruz, Mexico, Noviembre, pp 33–38

Amorocho DF (2001) Status and distribution of the hawksbill turtle, *Eretmochelys imbricata*, in the wider Caribbean region. In: Eckert KL, Abreu Grobois FA (eds) Marine turtle conservation in the wider Caribbean region—a dialogue for effective regional management. Santo Domingo, Dominican Republic, 16–18 November 1999, WIDECAST, IUCN-MTSG, WWF, and UNEP-CEP, pp 41–45

Amos AF (1989) The occurrence of hawksbills *(Eretmochelys imbricata*) along the Texas Coast. Proceedings, 9th Annual Workshop on Sea Turtle Conservation and Biology, Jekyll Island, GA, USA, August, Part 1, pp 9–12

Anonymous (1992) First Kemp's ridley nesting in South Carolina. MTN 59:23

Atencio DE (1994) Marine turtle nesting activity on Eglin AFB, Florida, 1987–1992. Proceedings, 13th Annual Symposium on Sea Turtle Biology and Conservation, NOAA Technical Memorandum NMFS-SEFSC-341. National Technical Information Service, Springfield, VA, USA, pp 201–204

Atkinson C, Berrondo L, Harrison E (2010) Report on the 2009 Green turtle program at Tortuguero, Costa Rica. Caribbean Conservation Corporation, Gainesville, FL, USA and Ministry of Environment and Energy of Costa Rica, San Pedro, Costa Rica. 66 p

Avens L, Taylor JC, Goshe LR, Jones TT, Hastings M (2009) Use of skeletochronological analysis to estimate the age of leatherback sea turtles *Dermochelys coriacea* in the western North Atlantic. Endang Species Res 8:165–177

Avens L, Goshe LR, Harms CA, Anderson ET, Hall AG, Cluse WM, Godfrey MH, Braun-McNeill J, Stacy B, Bailey R, Lamont MM (2012) Population characteristics, age structure, and growth dynamics of neritic juvenile green turtles in the northeastern Gulf of Mexico. Mar Ecol Prog Ser 458:213–229

Bacon PR (1970) Studies on the leatherback turtle, *Dermochelys coriacea* (L) in Trinidad, West Indies. Biol Conserv 2:213–217

Balazs GH (1985) Impact of ocean debris on marine turtles: Entanglement and ingestion. Proceedings, Workshop on the Fate and Impact of Marine Debris, Honolulu, HI, USA, November, Session II, pp 387–429

Baremore IE, Carlson JK, Hollensead LD, Bethea DM (2007) Catch and bycatch in U.S. Southeast Gillnet Fisheries, 2007. NOAA Technical Memorandum NMFS-SEFSC-565. National Marine Fisheries Service, Panama City, FL, USA, December. 20 p

Barichivich WJ (2006) Characterization of a marine turtle aggregation in the big bend of Florida. MS Thesis, University of Florida, Gainsville, FL, USA. 46 p

Barichivich WJ, Sulak KL, Carthy RR (1998) Feeding ecology and habitat affinities of Kemp's ridley sea turtles (*Lepidochelys kempi*) in the Big Bend, Florida. Southeast Fisheries Science Center, NMFS, Panama City, FL, USA. December. 18 p

Bass AL (1999) Genetic analysis to elucidate the natural history and behavior of hawksbill turtles (*Eretmochelys imbricata*) in the Wider Caribbean: A review and re-analysis. Chel Conserv Biol 3:195–199

Bass AL, Witzell WN (2000) Demographic composition of immature green turtles (*Chelonia mydas*) from the east central Florida coast: Evidence from mtDNA markers. Herpetologica 56:357–367

Bass AL, Epperly SP, Braun-McNeill J (2006) Green turtle (*Chelonia mydas*) foraging and nesting aggregations in the Caribbean and Atlantic: Impact of currents and behavior on dispersal. J Hered 97:346–354

Bell BA, Spotila JR, Paladino FV, Reina RD (2003) Low reproductive success of leatherback turtles, *Dermochelys coriacea*, is due to high embryonic mortality. Biol Conserv 115:131–138

Bell IP (2012) The hawksbill turtle, *Eretmochelys imbricata*, (Linnaeus 1766): Ecological insights of a resident population in the northern Great Barrier Reef, Queensland, Australia. PhD Thesis, James Cook University, Queensland, Australia. 143 p

Bjorndal KA (1980) Nutrition and grazing behavior of the green turtle *Chelonia mydas*. Mar Biol 56:147–154

Bjorndal KA (1982) The consequences of herbivory for the life history pattern of the Caribbean green turtle, *Chelonia mydas*. In: Bjorndal KA (ed) Biology and conservation of sea turtles. Smithsonian Institution Press, Washington, DC, USA, pp 111–116

Bjorndal KA (1985) Nutritional ecology of sea turtles. Copeia 1985:736–751

Bjorndal KA (1997) Foraging ecology and nutrition of sea turtles. In: Lutz PL, Musick JA (eds) The biology of sea turtles, vol 1. CRC Press, Boca Raton, FL, USA, pp 199–232

Bjorndal KA, Bolten AB (1992) Spatial distribution of green turtle (*Chelonia mydas*) nests at Tortuguero, Costa Rica. Copeia 1992:45–53

Bjorndal KA, Bolten AB (eds) (2000) Proceedings of a Workshop on Assessing Abundance and Trends for In-Water Sea Turtle Populations. NOAA Technical Memorandum NMFS-SEFSC-445. National Marine Fisheries Service, Miami, FL, USA. October. 83 p

Bjorndal KA, Carr A (1989) Variation in clutch size and egg size in the green turtle nesting population at Tortuguero, Costa Rica. Herpetologica 45:181–189

Bjorndal KA, Jackson JBC (2003) Roles of sea turtles in marine ecosystems: Reconstructing the past. In: Lutz PA, Musick JA, Wyneken J (eds) The biology of sea turtles, vol 2. CRC Press, Boca Raton, FL, USA, pp 259–273

Bjorndal KA, Meylan AB, Turner BJ (1983) Sea turtle nesting at Melbourne Beach, Florida, 1. Size, growth, reproductive biology. Biol Conserv 26:65–77

Bjorndal KA, Bolten AB, Lagueux CJ (1994) Ingestion of marine debris by juvenile sea turtles in coastal Florida habitats. Mar Pollut Bull 28:154–158

Bjorndal KA, Wetherall JA, Bolten AB, Mortimer JA (1999) Twenty-six years of green turtle nesting at Tortuguero, Costa Rica: An encouraging trend. Conserv Biol 13:126–134

Bjorndal KA, Bolten AB, Martins HR (2000) Somatic growth model of juvenile loggerhead sea turtles *Caretta caretta*: duration of pelagic stage. Mar Ecol Prog Ser 202:265–272

Bjorndal KA, Bolten AB, Koike B, Schroeder BA, Shaver DJ, Teas WG, Witzell WN (2001) Somatic growth function for immature loggerhead sea turtles, *Caretta caretta*, in southeastern U.S. waters. Fish Bull 99:240–246

Bjorndal KA, Bolten AB, Martins HR (2003) Estimates of survival probabilities for oceanic-stage loggerhead sea turtles (*Caretta caretta*) in the North Atlantic. Fish Bull 101:732–736

Bjorndal KA, Bowen BW, Chaloupka M, Crowder LB, Heppell SS, Jones CM, Lutcavage ME, Policansky D, Solow AR, Witherington BE (2011) Better science needed for restoration in the Gulf of Mexico. Science 331:537–538

Blair K (2005) Determination of sex ratios and their relationship to nest temperature of loggerhead sea turtle (*Caretta caretta*, L.) hatchlings produced along the southeastern Atlantic coast of the United States. Florida Atlantic University, Boca Raton, FL, USA. 89 p

Blumenthal JM, Abreu-Grobois FA, Austin TJ, Broderick AC, Bruford MW, Coyne MS, Ebanks-Petrie G, Formia A, Meylan PA, Meylan AB, Godley BJ (2009) Turtle groups or turtle soup: Dispersal patterns of hawksbill turtles in the Caribbean. Mol Ecol 18:4841–4853

Bolten AB (2003) Active swimmers-passive drifters: The oceanic juvenile stage of loggerheads in the atlantic system. In: Bolten AB, Witherington BE (eds) Loggerhead sea turtles. Smithsonian Books, Washington, DC, USA, pp 63–78

Bolten AB, Witherington BE (eds) (2003) Loggerhead sea turtles. Smithsonian Books, Washington, DC, USA. 352 p

Bolten AB, Bjorndal KA, Martins HR, Dellinger T, Biscoito MJ, Encalada SE, Bowen BW (1998) Transatlantic developmental migrations of loggerhead sea turtles demonstrated by mtDNA sequence analysis. Ecol Appl 8:1–7

Bolten AB, Crowder LB, Dodd MG, MacPherson SL, Musick JA, Schroeder BA, Witherington BE, Long KJ, Snover ML (2011) Quantifying multiple threats to endangered species: An example from loggerhead sea turtles. Front Ecol Environ 9:295–301

Bouchard SS, Bjorndal KA (2000) Sea turtles as biological transporters of nutrients and energy from marine to terrestrial ecosystems. Ecology 81:2305–2313

Boulon RH Jr (1994) Growth rates of wild juvenile hawksbill turtles, *Eretmochelys imbricata*, in St. Thomas, United States Virgin Islands. Copeia 1994:811–814

Boulon RH Jr, Dutton PH, McDonald DL (1996) Leatherback turtles (*Dermochelys coriacea*) on St. Croix, U.S. Virgin Islands: Fifteen years of conservation. Chel Conserv Biol 2:144–147

Bowen BW, Avise JC (1995) Conservation genetics of marine turtles. In: Avise JC, Hamrick JL (eds) Conservation genetics: Case histories from nature. Chapman and Hall, New York, NY, USA, pp 190–237

Bowen BW, Meylan AB, Avise JC (1989) An odyssey of the green sea turtle: Ascension Island revisited. Proc Natl Acad Sci Biol 86:573–576

Bowen BW, Meylan AB, Ross JP, Limpus CJ, Balazs GH, Avise JC (1992) Global population structure and natural history of the green turtle (*Chelonia mydas*) in terms of matriarchal phylogeny. Evolution 46:865–881

Bowen BW, Grant WS, Hillis-Starr Z, Shaver DJ, Bjorndal KA, Bolten AB, Bass AL (2007) Mixed-stock analysis reveals the migrations of juvenile hawksbill turtles (*Eretmochelys imbricata*) in the Caribbean Sea. Mol Ecol 16:49–60

Braun-McNeill J, Epperly SP (2002) Spatial and temporal distribution of sea turtles in the western North Atlantic and the U.S. Gulf of Mexico from Marine Recreational Fishery Statistics Survey (MRFSS). Mar Fish Rev 64:50–56

Broderick AC, Godley BJ, Hays GC (2001) Trophic status drives interannual variability in nesting numbers of marine turtles. Proc Roy Soc Lond B Bio 268:1481–1487

Broderick AC, Coyne MS, Glen F, Fuller WJ, Godley BJ (2006) Foraging site fidelity of adult green and loggerhead turtles. Proceedings, 26th Annual Workshop on Sea Turtle Conservation and Biology, Island of Crete, EL, Greece. 83 p

Brongersma LD (1972) European atlantic turtles. Zoological treatises. National Museum of Natural History, Leiden, ZH, The Netherlands. 318 p

Brongersma LD, Carr AF (1983) *Lepidochelys kempi*, (Garman) from Malta. Proc Roy Neth Acad Art Sci Series C 86:445–454

Burchfield PM, Byles R, Vicente Mongrell MVZJ, Bartlett M, Rostal D (1988) Report on Republic of Mexico/United States of America: Conservation effort on behalf of Kemp's ridley sea turtle at Playa de Rancho Nuevo, Tamaulipas, Mexico, 1988. Gladys Porter Zoo, Brownsville, TX, USA. 83 p

Byles RA (1989) Satellite telemetry of Kemp's ridley sea turtle, *Lepidochelys kempi*, in the Gulf of Mexico. Proceedings, 9th Annual Workshop on Sea Turtle Conservation and Biology, Jekyll Island, GA, USA, August, Part 1, pp 25–26

Byles RA, Plotkin PT (1994) Comparison of the migratory behavior of the congeneric sea turtles *Lepidochelys olivacea* and *L. kempii*. Proceedings, 13th Annual Symposium on Sea Turtle Biology and Conservation, Jekyll Island, GA, USA, January, Part 1. 39 p

Caillouet CW Jr (1995) Egg and hatchling take for the Kemp's ridley headstart experiment. MTN 68:13–15

Caillouet CW Jr, Duronslet MJ, Landry AM Jr, Revera DB, Shaver DJ, Stanley KM, Heinly RW, Stabenau EK (1991) Sea turtle strandings and shrimp fishing effort in the northwestern Gulf of Mexico, 1986–1989. Fish Bull 89:712–718

Caillouet CW Jr, Fontaine CT, Manzella-Tirpak SA, Shaver DJ (1995) Survival of head-started Kemp's ridley sea turtles (*Lepidochelys kempii*) released into the Gulf of Mexico or adjacent bays. Chel Conserv Biol 1:285–292

Caillouet CW Jr, Shaver DJ, Teas WG, Nance JM, Revera DB, Cannon AC (1996) Relationship between sea turtle stranding rates and shrimp fishing intensities in the northwestern Gulf of Mexico: 1986–1989 versus 1990–1993. Fish Bull 94:237–249

Caillouet CW Jr, Hart RA, Nance JM (2008) Growth overfishing in the brown shrimp fishery of Texas, Louisiana, and adjoining Gulf of Mexico EEZ. Fish Res 92:289–302

Caldwell DK (1959) On the status of the Atlantic leatherback sea turtle, *Dermochelys coriacea*, as a visitant to Florida nesting beaches, with natural history notes. Q J Florida Acad Sci 21:285–291

Caldwell DK, Carr A, Hellier TR Jr (1955) A nest of the Atlantic leatherback turtle, *Dermochelys coriacea* (Linnaeus), on the Atlantic coast of Florida, with a summary of American nesting records. Q J Florida Acad Sci 18:279–284

Campbell CL (1996) Capture of Juvenile Kemp's ridleys in the nearshore waters of Apalachee Bay, Florida. Proceedings, 16th Annual Symposium on Sea Turtle Biology and Conservation, Hilton Head, SC, USA, February, pp 28–30

Cannon AC (1995) Incidental catch of Kemp's ridley sea turtles (*Lepidochelys kempii*) by Hook-and-Line along the Upper Texas Coast. Task final report for fiscal year 1994. NMFS Marine Entanglement Research Program, Galveston, TX, USA

Cannon AC, Fontaine CT, Williams TD, Revera DB, Caillouet CW Jr (1994) Incidental catch of Kemp's ridley sea turtles (*Lepidochelys kempi*), by hook and line, along the Texas Coast, 1980–1992. Proceedings, 13th Annual Symposium on Sea Turtle Biology and Conservation, Jekyll Island, GA, USA, February, Part I, pp 40–42

Carballo JL, Olabarria C, Garza OT (2002) Analysis of four macroalgal assemblages along the Pacific Mexican coast during and after the 1997–1998 El Niño. Ecosystems 5:749–760

Carr A (1952) Handbook of turtles. Comstock Publishing Associates, Cornell University, Ithaca, NY, USA. 542 p

Carr A (1957) Notes on the zoogeography of the Atlantic sea turtles of the genus *Lepidochelys*. Rev Biol Trop 5:45–61

Carr AF (1963) Panspecific reproductive convergence in *Lepidochelys kempi*. Ergeb Biol 25:297–303

Carr AF (1979) The windward road: Adventures of a naturalist on remote Caribbean shores, Revised edn. University of Florida Press, Gainsville, FL, USA. 258 p

Carr AF Jr (1984) So excellent a fishe: A natural history of sea turtles, Revised edn. Scribner's, New York, NY, USA. 280 p

Carr A (1986) Rips, FADS, and little loggerheads. BioScience 36:92–100

Carr A (1987) New perspectives on the pelagic stage of sea turtle development. Conserv Biol 1:103–121

Carr AF, Caldwell DK (1956) The ecology and migrations of sea turtles, 1. Results of field work in Florida, 1955. Am Mus Novit 1793:1–23

Carr A, Carr MH (1972) Site fixity in the Caribbean green turtle. Ecology 53:425–429

Carr A, Hirth H (1962) The ecology and migrations of sea turtles, 5. Comparative features of isolated green turtle colonies. Am Mus Novit 2091:1–42

Carr A, Ogren L (1959) The ecology and migrations of sea turtles, 3. *Dermochelys* in Costa Rica. Am Mus Novit 1958:1–29

Carr A, Ogren L (1966) The ecology and migrations of sea turtles, 6. The hawksbill turtle in the Caribbean Sea. Am Mus Novit 2248:1–29

Carr A, Carr MH, Meylan AB (1978) The ecology and migrations of sea turtles, 7. The west Caribbean green turtle colony. B Am Mus Nat Hist 162:1–46

Carreras C, Pont S, Maffucci F, Pascual M, Barceló A, Bentivegna F, Cardona L, Alegre F, San Félix M, Fernández G, Aguilar A (2006) Genetic structuring of immature loggerhead sea turtle (*Caretta caretta*) in the Mediterranean Sea reflects water circulation patterns. Mar Biol 149:1269–1279

Carrillo E, Webb GJW, Manolis SC (1999) Hawksbill turtles (*Eretmochelys imbricata*) in Cuba: An assessment of the historical harvest and its impacts. Chel Conserv Biol 3:264–280

Carriol RP, Vader W (2002) Occurrence of *Stomatolepas elegans* (Cirripedia: Balanomorpha) on a leatherback turtle from Finmark, northern Norway. J Mar Biol Assoc UK 82:1033–1034

Cato JC, Prochaska FJ, Pritchard PCH (1978) An analysis of the capture, marketing, and utilization of marine turtles. Environmental Assessment Division, National Marine Fisheries Service, St. Petersburg, FL, USA. 238 p

Caut S, Guirlet E, Jouquet P, Girondot M (2006) Influence of nest location and yolkless eggs on the hatching success of leatherback turtle clutches in French Guiana. Can J Zool 84:908–915

Chávez H (1968) Marcado y recaptura de individuos de tortuga lora, *Lepidochelys kempi* (Garman). Instituto Nacional de Investigaciones Biológico Pesqueras 19:1–28

Chávez H (1969) Tagging and recapture of the lora turtle (*Lepidochelys kempii*). Int Turtle Tortoise Soc J 2:16–19, 32–36

Chávez H, Contreras GM, Hernández DE (1967) Aspectos biologicos y proteccion de la tortuga lora, *Lepidochelys kempii* (Garman), en la costa de Tamaulipas, Mexico. Publicación Número 17. Instituto Nacional de Investigaciones Biológico Pesqueras, Ciudad de Mexico, 40 p

Collard SB (1987) Review of oceanographic features relating to neonate sea turtle distribution and dispersal in the pelagic environment: Kemp's ridley (*Lepidochelys kempi*) in the Gulf of Mexico. Final Report. Contract Number 40-GFNF-5-00193. National Marine Fisheries Service, Panama City, FL, USA, Dec. 42 p

Collard SB (1990) Leatherback turtles feeding near a water mass boundary in the eastern Gulf of Mexico. MTN 50:12–14

Collard SB, Ogren LH (1990) Dispersal scenarios for pelagic post-hatchling sea turtles. Bull Mar Sci 47:233–243

Conant TA, Dutton PH, Eguchi T, Epperly SE, Fahy CC, Godfrey MH, MacPherson SL, Possardt EE, Schroeder BA, Seminoff JA, Snover ML, Upite CM, Witherington BE (2009) Loggerhead sea turtle (*Caretta caretta*) 2009 status review under the U.S. endangered species act. A report of the loggerhead biological review team to the National Marine Fisheries Service. NMFS, Silver Spring, MD, USA. 222 p

Cornelius SE, Arauz R, Fretey J, Godfrey MH, Márquez-M R, Shanker K (2007) Effect of land-based harvest of *Lepidochelys*. In: Plotkin PT (ed) Biology and conservation of ridley sea turtles. The Johns Hopkins University Press, Baltimore, MD, USA, pp 231–252

Coyne MS (1994) Feeding ecology of subadult green sea turtles in South Texas Waters. MS Thesis, Texas A&M University, College Station, TX, USA. 76 p

Coyne MS (2000) Population sex ratio of the Kemp's ridley sea turtle (*Lepidochelys kempii*): problems in population modeling. PhD Thesis, Texas A&M University, College Station, TX, USA. 124 p

Coyne MS, Landry AM Jr (2000) Plasma testosterone dynamics in the Kemp's ridley sea turtle. Proceedings, 18th International Sea Turtle Symposium, Mazatlán, Sinaloa, Mexico, March, Part VI. 286 p

Craig JK, Crowder LB, Gray CD, McDaniel CJ, Henwood TA, Hanifen JG (2001) Ecological effects of hypoxia on fish, sea turtles, and marine mammals in the northwestern Gulf of Mexico. Coast Estuar Stud 58:269–292

Crouse DT, Crowder LB, Caswell H (1987) A stage-based population model for loggerhead sea turtles and implications for conservation. Ecology 68:1412–1423

Crowder L, Heppell S (2011) The decline and rise of a sea turtle: How Kemp's ridleys are recovering in the Gulf of Mexico. Solutions 2:67–73

Crowder LB, Crouse DT, Heppell SS, Martin TH (1994) Predicting the impact of turtle excluder devices on loggerhead sea turtle populations. Ecol Appl 4:437–445

Cuevas E, de los Ángeles Liceaga-Correa M, Garduño-Andrade M (2007) Spatial characteristics of a foraging area for immature hawksbill turtles (*Eretmochelys imbricata*) in Yucatán, Mexico. Amphibia-Reptilia 28:337–346

Cuevas E, Abreu-Grobois FA, Guzmán-Hernández V, Liceaga-Correa MA, van Dam RP (2008) Post-nesting migratory movements of hawksbill turtles *Eretmochelys imbricata* in waters adjacent to the Yucatán Peninsula, Mexico. Endang Species Res 10:123–133

Cuevas E, de los Ángeles Liceaga-Correa A, Mariño-Tapia I (2010) Influence of beach slope and width on hawksbill (*Eretmochelys imbricata*) and green turtle (*Chelonia mydas*) nesting activity in El Cuyo, Yucatán, Mexico. Chel Conserv Biol 9:262–267

Cuevas E, González-Garza BI, Guzmán-Hernández V, van Dam RP, García-Alvarado P, Abreu-Grobois FA, Huerta-Rodríguez P (2012) Tracking turtles off Mexico's Yucatán Peninsula. In SWOT (State of the Worlds Sea Turtles) Report Volume VII. Arlington, VA, USA, pp 8–9

Dahlen MK, Bell R, Richardson JI, Richardson TH (2000) Beyond D-0004: Thirty-four years of loggerhead (*Caretta caretta*) research on Little Cumberland Island, Georgia, 1964–1997. Proceedings, 18th International Sea Turtle Symposium, Mazatlán, Sinaloa, Mexico, March, Part VI, pp 60–62

Dalrymple GH, Hampp JC, Wellins DJ (1985) Male-biased sex ratio in a cold nest of a hawksbill sea turtle *Eretmochelys imbricata*. J Herpetol 19:158–159

Davis GE, Whiting MC (1977) Loggerhead sea turtle nesting in Everglades National Park, Florida, USA. Herpetologica 33:18–28

Davis RW, Evans WE, Wursig B (2000) Cetaceans, sea turtles and seabirds in the Northern Gulf of Mexico: Distribution, Abundance and Habitat Associations. Texas A&M at Galveston, National Marine Fisheries Service OCS Study MMS 2000-002. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA, USA. 28 p

Debade X, Nolasco del Aguila D, Harrison E (2009) Report on the 2008 leatherback program at Tortuguero, Costa Rica. Caribbean Conservation Corporation, Gainesville and Ministry of Environment and Energy of Costa Rica, San Pedro, Costa Rica. 46 p

del Monte-Luna P, Guzmán-Hernández V, Cuevas EA, Arreguín-Sánchez F, Lluch-Belda D (2012) Effect of North Atlantic climate variability on hawksbill turtles in the Southern Gulf of Mexico. J Exp Mar Biol Ecol 412:103–109

Dellinger T, Freitas C (2000) Movement and diving behaviour of pelagic stage loggerhead sea turtles in the North Atlantic: Preliminary results obtained through satellite telemetry. Proceedings, 19th Annual Symposium on Sea Turtle Conservation and Biology, South Padre Island, TX, USA, March, pp 155–157

den Hartog JC, van Nierop MM (1984) A study on the gut contents of six leathery turtles *Dermochelys coriacea* (Linnaeus) (Reptilia: Testudines: Dermochelyidae) from British waters and from the Netherlands. Zoologische Verhandelingen 209:1–36

Diez CE, van Dam RP (2003) Sex ratio of an immature hawksbill sea turtle aggregation at Mona Island, Puerto Rico. J Herpetol 37:533–537

Dobie JL, Ogren LH, Fitzpatrick JF Jr (1961) Food notes and records of the Atlantic ridley turtle (*Lepidochelys kempi*) from Louisiana. Copeia 1961:109–110

Dodd CK Jr (1988) Synopsis of the biological data on the loggerhead sea turtle *Caretta caretta* (Linnaeus 1758). USFWS Biological Report 88-14. U.S. Fish and Wildlife Service, Washington, DC, USA. 110 p

Dodd CK Jr, Byles R (1991) The status of loggerhead, *Caretta caretta*; Kemp's ridley, *Lepidochelys kempi*; and green, *Chelonia mydas*, sea turtles in U.S. waters: A reconsideration. Mar Fish Rev 53:30–33

DOF (Diario Oficial de la Federación) (1977) Acuerdo que establece como Zona de Refugio y de Veda para la proteccion de la Tortuga lora. Department de Pesca, Ciudad de Mexico, pp 1–2

DOF (1986) Decreto por el que se determinan como Zonas de Reserva y Sitios de Refugio para la proteccion, conservacion, repoblacion, desarrollo y control de las diversas species de Tortuga marina, los lugares donde anida y desovan dichas species. Department de Pesca, Ciudad de Mexico, pp 1–5

DOF (1990) Acuerdo pro el que se establece veda total para todas las species y subspecies de Tortugas marinas en aguas de jurisdicción nacional de los litorales del Océano Pacífico, Golfo de Mexico y Mar Caribe. Department de Pesca, Ciudad de Mexico, pp 21–22

DOF (2002) Acuerdo por el que se determinan como áreas naturales protegidas, con la categoría de santuarios, a las zonas de reserva y sitios de refugio para la protección, conservación, repoblación, desarrollo y control de las diversas especies de tortuga marina, ubicadas en los estados de Chiapas, Guerrero, Jalisco, Michoacán, Oaxaca, Sinaloa, Tamaulipas y Yucatán, identificadas en el decreto publicado el 29 de Octubre de 1986. Department de Pesca, Ciudad de Mexico, pp 268–269

Doughty RW (1984) Sea turtles in Texas: A forgotten commerce. Southwestern Historical Quarterly 88:43–70

Dow W, Eckert K, Palmer M, Kramer P (2007) An atlas of sea turtle nesting habitat for the wider Caribbean region. WIDECAST Technical Report No. 6. Wider Caribbean Sea Turtle Conservation Network, Beaufort, NC, USA. 267 p

Dutton PH, Whitmore CP, Mrosovsky N (1985) Masculinisation of leatherback turtle, *Dermochelys coriacea*, hatchlings from eggs incubated in Styrofoam boxes. Biol Conserv 31:249–264

Dutton DL, Dutton PH, Chaloupka M, Boulon RH (2005) Increase of a Caribbean leatherback turtle *Dermochelys coriacea* nesting population linked to long-term nest protection. Biol Conserv 126:186–194

Eaton C, McMichael E, Witherington B, Foley A, Hardy R, Meylan A (2008) In-water sea turtle monitoring and research in Florida: Review and recommendations. NOAA Technical Memorandum NMFS-OPR-38, St. Petersburg, FL, USA, June. 233 p

Echeverría-García A, Torres-Burgos E (2007) 14 Años de Monitoreo de Tortugas Marinas en Yucatán. Memorias de la Reunión Nacional sobre Conservación de Tortugas Marinas, Veracruz, VE, Mexico, Noviembre. 21 p

Echeverría-García AW, Torres-Burgos E, Quezada-Domínguez C, López-Alonzo MA (2008) Informe Técnico de la Temporada 2007 en los Centros para la Protección y Conservación de Tortugas Marinas, operados por la Secretaría de Ecología del Gobierno del Estado de Yucatán, Mexico. Centros Para la Protección y Conservación de las Tortugas Marinas, Gobierno de Estado de Yucatán, Merida, Yucatán, Mexico. 17 p

Echeverría-García AW, Torres-Burgos E, Quezada-Domínguez C, López-Alonzo MA (2009) Informe Técnico de la Temporada 2008 de los Centros para la Protección y Conservación de Tortugas Marinas (Campamentos Tortugueros), Operados por la Secretaría de Desarrollo Urbano y Medio Ambiente (Seduma) del Gobierno del Estado de Yucatán, Mexico.

Centros Para la Protección y Conservación de las Tortugas Marinas, Gobierno de Estado de Yucatán, Merida, Yucatán, Mexico

Eckert KL (1987) Environmental unpredictability and leatherback sea turtle (*Dermochelys coriacea*) nest loss. Herpetologica 43:315–323

Eckert SA (2002) Distribution of juvenile leatherback sea turtle *Dermochelys coriacea* sightings. Mar Ecol Prog Ser 230:289–293

Eckert KL, Eckert SA (1988) Pre-reproductive movements of leatherback sea turtles (*Dermochelys coriacea*) nesting in the Caribbean. Copeia 1988:400–406

Eckert KL, Eckert SA (1990) Embryo mortality and hatch success in *in situ* and translocated leatherback sea turtle *Dermochelys coriacea* eggs. Biol Conserv 53:37–46

Eckert SA, Eckert KL, Ponganis P, Kooyma GL (1989a) Diving and foraging behavior of leatherback sea turtles (*Dermochelys coriacea*). Can J Zool 67:2834–2840

Eckert KL, Eckert SA, Adams TW, Tucker AD (1989b) Inter-nesting migrations by leatherback sea turtles (*Dermochelys coriacea*) in the West Indies. Herpetologica 45:190–194

Eckert SA, Bagley D, Kubis S, Ehrhart L, Johnson C, Stewart K, DeFreese D (2006) Internesting and postnesting movements and foraging habitats of leatherback sea turtles (*Dermochelys coriacea*) nesting in Florida. Chel Conserv Biol 5:239–248

Eckert SA, Moore JE, Dunn DC, van Buiten RS, Eckert KL, Halpin PN (2008) Modeling loggerhead turtle movement in the Mediterranean: Importance of body size and oceanography. Ecol Appl 18:290–308

Eckert KL, Wallace BP, Frazier JG, Eckert SA, Pritchard PCH (2012) Synopsis of the biological data on the leatherback sea turtle (*Dermochelys coriacea*). U.S. Department of Interior, Fish and Wildlife Service, Biological Technical Publication BTP-R4015-2012. Washington, DC, USA. 160 p

Ehrhart LM (1980) A survey of marine turtles nesting at the Kennedy Space Center Cape Canaveral Air Force Station, North Brevard County, Florida. Report to the Division of Marine Resources, Florida. 216 p

Ehrhart LM, Bagley DA, Redfoot WE (2003) Loggerhead turtles in the atlantic ocean: Geographic distribution, abundance, and population status. In: Bolten AB, Witherington BE (eds) Loggerhead sea turtles. Smithsonian Books, Washington, DC, USA, pp 157–174

Epperly SP (2003) Fisheries-related mortality and turtle excluder devices. In: Lutz PA, Musick JA, Wyneken J (eds) The biology of sea turtles, vol 2. CRC Press, Boca Raton, FL, USA, pp 339–353

Epperly SP, Teas WG (2002) Turtle excluder devices—are the escape openings large enough? Fish Bull 100:466–474

Epperly SP, Braun J, Chester AJ, Cross FA, Merriner JV, Tester PA, Churchill JH (1996) Beach strandings as an indicator of at-sea mortality of sea turtles. Bull Mar Sci 59:289–297

Epperly S, Avens L, Garrison L, Henwood T, Hoggard W, Mitchell J, Nance J, Poffenberger J, Sasso C, Scott-Denton E, Yeung C (2002) Analysis of sea turtle bycatch in the commercial shrimp fisheries of the southeast U.S. Waters and the Gulf of Mexico. NOAA Technical Memorandum NMFS-SEFSC-490. National Marine Fisheries Service, Miami, FL, USA, November. 88 p

Evans D, Ordonez C, Troëng S, Dreus C (2007) Satellite tracking of leatherback turtles from Caribbean Central America Reveals Unexpected Foraging Grounds. Proceedings, 27th Annual Symposium in Sea Turtle Conservation and Biology, Myrtle Beach, SC, USA, April, pp 40–41

Fairfield CP, Garrison LP (2008) Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic Pelagic Longline Fleet During 2007. NOAA Technical Memorandum. NMFS-SEFSC-572. National Marine Fisheries Service, Miami, FL, USA. August. 62 p

Fairfield-Walsh C, Garrison LP (2006) Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic Pelagic Longline Fleet During 2005. NOAA Technical Memorandum. NMFS-SEFSC- 539. National Marine Fisheries Service, Miami, FL, USA. May. 52 p

Fairfield-Walsh C, Garrison LP (2007) Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic Pelagic Longline Fleet During 2006. NOAA Technical Memorandum. NMFS-SEFSC-560. National Marine Fisheries Service, Miami, FL, USA. June. 54 p

FDEP (Florida Department of Environmental Protection), NOAA (National Oceanic and Atmospheric Administration), USDOI (U.S. Department of the Interior) (1997) Damage assessment and restoration plan/environmental assessment for the August 10, 1993 Tampa Bay Oil Spill: Volume I—Ecological injuries—Final. Florida Department of Environmental Protection, Tallahassee, FL, USA. June. 91 p

FFWCC (Florida Fish and Wildlife Conservation Commission) (2012) Homepage. http://myfwc.com/. Accessed 20 November 2013

FFWCC FWRI (Fish and Wildlife Research Institute) (2011a) Marine turtle program: Statewide nesting totals 1979–2009. http://myfwc.com/media/1313557/Statewide-1979-2010.pdf. Accessed 20 November 2013

FFWCC FWRI (2011b) Marine turtle program: Sea turtle stranding. http://ocean.floridamarine.org/TRGIS/Description_Layers_Terrestrial.htm. Accessed 20 November 2013

FFWCC FWRI (2012) Statewide nesting beach program: Loggerhead nesting data, 2007–2012. http://myfwc.com/media/2078432/LoggerheadNestingData.pdf. Accessed 20 November 2013

Finkbeiner EM, Wallace BP, Moore JE, Lewison RL, Crowder LB, Read AJ (2011) Cumulative estimates of sea turtle bycatch and mortality in USA fisheries between 1990 and 2007. Biol Conserv 144:2719–2727

Fish MR, Cote IM, Horrocks JA, Mulligan B, Watkinson AR, Jones AP (2008) Construction setback regulations and sea-level rise: Mitigating sea turtle nesting beach loss. Ocean Coast Manage 51:330–341

Foley AM, Peck SA, Harman GR, Richardson LW (2000) Loggerhead turtle (*Caretta caretta*) nesting habitat on low-relief mangrove islands in southwest Florida and consequences to hatchling sex ratios. Herpetologica 56:436–447

Foley AM, Schroeder BA, Redlow AE, Fick-Child KJ, Teas WG (2005) Fibropapillomatosis in stranded green turtles (*Chelonia mydas*) from the eastern United States (1980–1998): Trends and associations with environmental factors. J Wildlife Dis 41:29–41

Foley AM, Singel KE, Dutton PH, Summers TM, Redlow AE, Lessman J (2007) Characteristics of a green turtle (*Chelonia mydas*) assemblage in northwestern Florida determined during a hypothermic stunning event. Gulf Mex Sci 2007:131–143

Foley AM, Schroeder BA, MacPherson SL (2008) Post-nesting migrations and resident areas of Florida Loggerhead Turtles (*Caretta caretta*). Proceedings, 25th Annual Symposium on Sea Turtle Biology and Conservation, Savannah, GA, USA, January, pp 75–76

Fontaine C, Shaver D (2005) Head-starting the Kemp's ridley sea turtle, *Lepidochelys kempii*, at the NMFS Galveston Laboratory, 1978–1992: A review. Chel Conserv Biol 4:838–845

Fontaine CT, Manzella SA, Williams TD, Harris RM, Browning WJ (1989) Distribution, growth and survival of head started, tagged and released Kemp's ridley sea turtles (*Lepidochelys kempi*) from year-classes 1978–1983. Proceedings, 1st International Symposium on Kemp's Ridley Sea Turtle Biology, Conservation and Management, Galveston, TX, USA, conservation and management, Galveston, October, Session III, pp 124–144

Fossette S, Girard C, Lopez-Mendilaharsu M, Miller P, Domingo A, Evan D, Kelle L, Plot V, Prosdocimi L, Verhage S, Gaspar P, Georges JY (2010) Atlantic leatherback migratory pathways and temporary residence areas. PLoS One 5(e13908):1–12

Fowler LE (1979) Hatching success and nest predation in the green sea turtle, *Chelonia mydas*, at Tortuguero, Costa Rica. Ecology 60:946–955

Frair W, Ackman RG, Mrosovsky N (1972) Body temperature of *Dermochelys coriacea*: Warm turtle from cold water. Science 177:791–793

Frazer NB, Ehrhart LM (1985) Preliminary growth models for green, *Chelonia mydas*, and loggerhead, *Caretta caretta*, turtles in the wild. Copeia 1985:73–79

Frazier J (1993) Una evaluación del manejo de nidos de tortugas marinas en la Península de Yucatán. Memorias del IV Taller Regional Sobre Programas de Conservación de Tortugas Marinas en la Península de Yucatán, Mérida, Yucatán, Mexico, Marzo, pp 37–76

Frazier J, Arauz R, Chevalier J, Formia A, Fretey J, Godfrey MH, Márquez-M R, Pandav B, Shanker K (2007) Human-turtle interactions at sea. In: Plotkin PT (ed) Biology and conservation of ridley sea turtles. The Johns Hopkins University Press, Baltimore, MD, USA, pp 253–296

Fretey J (2001) Biogeography and conservation of marine turtles of the Atlantic Coast of Africa. CMS Technical Series 6. United Nations Environment Programme, Convention on Migratory Species, Bonn, Germany. 429 p

Fretey J, Bour J (1980) Redécouverte du type de *Dermochelys coriacea* (Vandelli) (Testudinata, Dermochelydae). Bolletin di Zoologia 47:193–205

Fritts TH, Reynolds RP (1981) Pilot study of the marine mammals, birds and turtles in OCS Areas of the Gulf of Mexico. FWS/OBS-81/36. U.S. Fish and Wildlife Service and Bureau of Land Management Gulf of Mexico OCS Office, New Orleans, LA, USA. 139 p

Fritts TH, Hoffman W, McGehee MA (1983a) The distribution and abundance of marine turtles in the Gulf of Mexico and nearby Atlantic waters. J Herpetol 17:327–344

Fritts TH, Irvine AB, Jennings RD, Collum LA, Hoffman W, McGehee M (1983b) Turtles, birds and mammals in the Northern Gulf of Mexico and Nearby Atlantic Waters. FWS/OBS-82/65. U.S. Fish and Wildlife Service, Washington, DC, USA. 455 p

FuelFix (2011) Despite BP oil spill, Kemp's ridley turtle numbers at record levels in Texas. http://fuelfix.com/blog/2011/07/19/despite-bp-oil-spill-kemps-ridley-turtle-numbers-at-record-levels-in-texas. Accessed 20 November 2013

Fuller DA (1988) Occurrences of sea turtles in Eastern Louisiana. LSU-CFI-88-12. National Marine Fisheries Service, Panama City, FL, USA. 10 p

Fuller DA (1989) Sea turtle strandings and sightings in southeastern Louisiana. LSU-CFI-89-05. National Marine Fisheries Service, Panama City, FL, USA. 11 p

Garduño M (1998) Ecología de la tortuga carey (*Eretmochelys imbricata*) en la zona de Las Coloradas, Yucatán, Mexico. Universidad de Colima, Colima, Mexico

Garduño-Andrade M (1999) Nesting of the hawksbill turtle, *Eretmochelys imbricata*, at Río Lagartos, Yucatán, Mexico, 1990–1997. Chel Conserv Biol 3:281–285

Garduño-Andrade M (2000) Fecundidad de la tortuga de carey *Eretmochelys imbricata* en las Colorados, Yucatán, Mexico. Ciencia Pesquera 14:67–70

Garduño-Andrade M, Guzmán V, Miranda E, Briseño-Deuñas R, Abreu-Grobois FA (1999) Increases in hawksbill turtle (*Eretmochelys imbricata*) nestings in the Yucatán Peninsula, Mexico, 1977–1996: Data in support of successful conservation? Chel Conserv Biol 3:286–295

Garrison LP (2003) Estimated bycatch of marine mammals and turtles in the U.S. Atlantic Pelagic Longline Fleet during 2001–2002. NOAA Technical Memorandum NMFS-SEFSC-515. National Marine Fisheries Service, Miami, FL, USA. December. 52 p

Garrison LP (2005) Estimated bycatch of marine mammals and turtles in the U.S. Atlantic Pelagic Longline Fleet during 2004. NOAA Technical Memorandum NMFS-SEFSC-531. National Marine Fisheries Service, Miami, FL, USA. June. 57 p

Garrison LP (2007) Estimated marine mammal and turtle bycatch in shark gillnet fisheries along the Southeast U.S. Atlantic Coast: 2000–2006. NMFS Southeast Fisheries Science Center Contribution PRD-07/08-02. National Marine Fisheries Service, Silver Spring, MD, USA. November. 22 p

Garrison LP, Richards PM (2004) Estimated bycatch of marine mammals and turtles in the U.S. Atlantic Pelagic Longline Fleet during 2003. NOAA Technical Memorandum NMFS-SEFSC-527. National Marine Fisheries Service, Miami, FL, USA. September. 57 p

Garrison LP, Stokes L (2010) Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic Pelagic Longline Fleet During 2009. NOAA Technical Memorandum NMFS-SEFSC-607. National Marine Fisheries Service, Miami, FL, USA. November. 63 p

Garrison LP, Stokes L, Fairfield C (2009) Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic Pelagic Longline Fleet during 2008. NOAA Technical Memorandum NMFS-SEFSC-591. National Marine Fisheries Service, Miami, June, 63 p

Girard C, Tucker AD, Calmettes B (2009) Post-nesting migrations of loggerhead sea turtles in the Gulf of Mexico: Dispersal in highly dynamic conditions. Mar Biol 156:1827–1839

Girondot M, Godfrey MH, Ponge L, Rivalan P (2007) Modeling approaches to quantify leatherback nesting trends in French Guiana and Suriname. Chel Conserv Biol 6:37–47

Gitschlag G (1992) Offshore oil and gas structures as sea turtle habitat. Proceedings, 11th Annual Workshop on Sea Turtle Biology and Conservation, Jekyll Island, GA, USA, Jekyll Island, March, Part 1, pp 49–50

Gitschlag G, Herczeg B (1994) Sea turtle observations at explosive removals of energy structures. Mar Fish Rev 56:1–8

Godfrey MH, Mrosovsky N (1997) Estimating the time between hatching of sea turtles and their emergence from the nest. Chel Conserv Biol 2:581–585

Godfrey MH, Barreto R, Mrosovsky N (1996) Estimating past and present sex ratios of sea turtles in Suriname. Can J of Zool 74:267–277

Godfrey MH, D'Amato AF, Marcovaldi MA, Mrosovsky N (1999) Pivotal temperature and predicted sex ratios for hatchling hawksbill turtles from Brazil. Can J Zool 77:1465–1473

Godley BJ, Broderick AC, Hays GC (2001) Nesting of green turtles (*Chelonia mydas*) at Ascension Island, South Atlantic. Biol Conserv 97:151–158

González-C V, Álvarez KC, Prosdocimi L, Inchaurraga MC, Dellacasa RF, Faiella A, Echenique C, González R, Andrejuk J, Mianzin HW, Campagna C, Albareda DA (2011) Argentinean coastal waters: A temperate habitat for three species of threatened sea turtles. Mar Biol Res 7:500–508

Gorga CCT (2010) Foraging ecology of green turtles (*Chelonia mydas*) on the Texas Coast, as Determined by Stable Isotope Analysis. MS Thesis, Texas A&M University, College Station, TX, USA. 53 p

Goshe LR (2009) Age at Maturation and Growth Rates of Green Sea Turtles (*Chelonia mydas*) along the Southeastern U.S. Atlantic Coast Estimated Using Skeletochronology. University of North Carolina Wilmington, Wilmington, NC, USA. 69 p

Greer AE, Lazelljun JD, Wright RM (1973) Anatomical evidence for a counter-current heat exchange in the leatherback turtle (*Dermochelys coriacea*). Nature 244:181

Gregory LF, Schmid JR (2001) Stress responses and sexing of wild Kemp's ridley sea turtles (*Lepidochelys kempii*) in the northeastern Gulf of Mexico. Gen Comp Endocr 124:66–74

Griffin RB, Griffin NJ (2003) Distribution, habitat partitioning, and abundance of Atlantic spotted dolphins, bottlenose dolphins, and loggerhead sea turtles on the eastern Gulf of Mexico continental shelf. Gulf Mex Sci 2003:23–34

Groombridge B, Luxmoore R (1989) The green turtle and hawksbill (Reptilia: Cheloniidae): World status, exploitation, and trade. Convention on International Trade in Endangered Species of Wild Fauna and Flora, Cambridge, England. 573 p

Guirlet E, Das K, Girondot M (2008) Maternal transfer of trace elements in leatherback turtles (*Dermochelys coriacea*) of French Guiana. Aquat Toxicol 88:267–276

Gunter G (1951) Destruction of fishes and other organisms on the South Texas coast by the cold wave of January 28-February 3, 1951. Ecology 32:731–736

Guzmán V, Rejón JC, Gómez R, Silva J (1996) Informe Final del Programa de Investigación y Protección de las Tortugas Marinas de Campeche, Mexico. Temporada 1996. Situación Actual. Bol Téc 1 INP-CRIP Carmen. 40 p

Guzmán-Hernández V, García-Alvarado PA (2010) Informe Técnico 2009 del Programa de Conservación de Tortugas Marinas en Laguna de Términos, Campeche, Mexico. Contiene Informacion de: 1. CPCTM Xicalango-Victoria, 2. CPCTM Chacahito, 3. CPCTM Isla Aguada y 4. Reseña estatal regional. APFFLT/RPCyGM/CONANP. 67 p

Guzmán-Hernández V, García Alvarado PA, Rejón JC, Gómez C, Gómez J (2006) Informe Técnico Final del Programa de Conservación de Tortugas Marinas de Campeche, Mexico en 2005. Programa Nacional de Tortugas Marinas. 36 p

Guzmán-Hernández V, Cuevas-Flores EA, Márquez-Millán R (2007) Occurrence of Kemp's ridley (*Lepidochelys kempii*) along the coast of the Yucatán Peninsula, Mexico. Chel Conserv Biol 6:274–277

Hale LF, Carlson JK (2007) Characterization of the shark bottom longline fishery: 2005–2006. NOAA Technical Memorandum NMFS-SEFSC-554. National Marine Fisheries Service, Silver Spring, MD, USA. 28 p

Hale LF, Gulak SJB, Carlson JK (2009) Characterization of the shark bottom longline fishery: 2008. Panama City, Florida. NOAA Technical Memorandum NMFS-SEFSC-586. National Marine Fisheries Service, Silver Spring, MD, USA. April. 23 p

Hall RJ, Belisle AA, Sileo L (1983) Residues of petroleum hydrocarbons in tissues of sea turtles exposed to the Ixtoc I oil spill. J Wildlife Dis 19:106–109

Hamann M, Limpus CJ, Owens DW (2003) Reproductive cycles of males and females. In: Lutz PA, Musick JA, Wyneken J (eds) The biology of sea turtles, vol 2. CRC Press, Boca Raton, FL, USA, pp 135–162

Hanson J, Wibbels T, Martin RE (1998) Predicted female bias in sex rations of hatchling loggerhead sea turtles from a Florida nesting beach. Can J Zool 76:1850–1861

Haro A, Harrison E (2007a) Report on the 2006 green turtle program at Tortuguero, Costa Rica. Caribbean Conservation Corporation, Gainesville and Ministry of Environment and Energy of Costa Rica, San Pedro, Costa Rica. 48 p

Haro A, Harrison E (2007b) Report on the 2006 leatherback program at Tortuguero, Costa Rica. Caribbean Conservation Corporation, Gainesville and Ministry of Environment and Energy of Costa Rica, San Pedro, Costa Rica. 33 p

Haro A, Troëng S (2006a) Report on the 2005 green turtle program at Tortuguero, Costa Rica. Caribbean Conservation Corporation, Gainesville and Ministry of Environment and Energy of Costa Rica, San Pedro, Costa Rica. 49 p

Haro A, Troëng S (2006b) Report on the 2005 Leatherback program at Tortuguero, Costa Rica. Caribbean Conservation Corporation, Gainesville and Ministry of Environment and Energy of Costa Rica, San Pedro, Costa Rica. 25 p

Harrison AL, Bjorndal KA (2006) Connectivity and wide-ranging species in the ocean. In: Crooks KR, Sanjayan M (eds) Connectivity conservation. Cambridge University Press, Cambridge, MA, USA, pp 213–232

Harrison E, Troëng S (2003a) Report on the 2002 leatherback program at Tortuguero, Costa Rica. Caribbean Conservation Corporation, Gainesville and Ministry of Environment and Energy of Costa Rica, San Pedro, Costa Rica. 32 p

Harrison E, Troëng S (2003b) Report on the 2003 leatherback program at Tortuguero, Costa Rica. Caribbean Conservation Corporation, Gainesville and Ministry of Environment and Energy of Costa Rica, San Pedro, Costa Rica. 29 p

Harrison E, Troëng S (2004a) Report on the 2003 green turtle program at Tortuguero, Costa Rica. Caribbean Conservation Corporation, Gainesville and Ministry of Environment and Energy of Costa Rica, San Pedro, Costa Rica. 52 p

Harrison E, Troëng S (2004b) Report on the 2004 leatherback program at Tortuguero, Costa Rica. Caribbean Conservation Corporation, Gainesville and Ministry of Environment and Energy of Costa Rica, San Pedro, Costa Rica. 30 p

Harrison E, Troëng S (2005) Report on the 2004 green turtle program at Tortuguero, Costa Rica. Caribbean Conservation Corporation, Gainesville and Ministry of Environment and Energy of Costa Rica, San Pedro, Costa Rica. 51 p

Hart KM, Fujisaki I (2010) Satellite tracking reveals habitat use by juvenile green sea turtles *Chelonia midas* in the Everglades, Florida, USA. Endang Species Res 11:221–232

Hart KM, Lamont MM, Fujisaki I, Tucker AD, Carthy RR (2012a) Common coastal foraging areas for loggerheads in the Gulf of Mexico: Opportunities for marine conservation. Biol Conserv 145:185–1948

Hart KM, Sartain AR, Fujisaki I, Pratt HL Jr, Morley D, Feeley MW (2012b) Home range, habitat use, and migrations of hawksbills turtles tracked from Dry Tortugas National Park, Florida, USA. Mar Ecol Prog Ser 457:193–207

Hawkes LA, Broderick AC, Godfrey MH, Godley BJ (2007a) Investigating the potential impacts of climate change on a marine turtle population. Glob Chang Biol 13:923–932

Hawkes LA, Broderick AC, Coyne MS, Godfrey MH, Godley BJ (2007b) Only some like it hot—quantifying the environmental niche of the loggerhead sea turtle. Diversity Distrib 13:447–457

Hawkes LA, Broderick AC, Godfrey MH, Godley BJ (2009) Climate change and marine turtles. Endang Spec Res 7:137–154

Hays GC, Houghton JDR, Myers AE (2004) Endangered species: Pan-Atlantic leatherback turtle movements. Nature 429:522

Hays GC, Hobson VJ, Metcalfe JD, Righton D, Sims DW (2006) Flexible foraging movements of leatherback turtles across the North Atlantic Ocean. Ecology 87:2647–2656

Henry JA, Portier KM, Coyne J (1994) The climate and weather of Florida. Pineapple Press, Inc., Coconut Grove, FL, USA. 216 p

Henwood TA (1987) Sea turtles of the Southeastern United States, with emphasis on the life history and population dynamics of the turtle, *Caretta caretta*. PhD Thesis, Auburn University, Auburn, AL, USA. 170 p

Henwood TA, Stuntz WE (1987) Analysis of sea turtle captures and mortalities during commercial shrimp trawling. Fish Bull 85:813–817

Henwood TA, Stuntz W, Thompson N (1992) Evaluation of U.S. turtle protective measures under existing TED regulations, including estimates of shrimp trawler related turtle mortality in the wider Caribbean. NOAA Technical Memorandum NMFS-SEFSC-303. National Marine Fisheries Service, Spring, MD, USA. 21 p

Heppell SS (1997) On the importance of eggs. MTN 76:6–8

Heppell SS, Crowder LB, Crouse DT, Epperly SP, Frazer NB (2003) Population models for atlantic loggerheads: Past, present, and future. In: Bolten AB, Witherington BE (eds) Loggerhead sea turtles. Smithsonian Books, Washington, DC, USA, pp 255–276

Heppell SS, Crouse DT, Crowder LB, Epperly SP, Gabriel W, Henwood T, Márquez-M R, Thompson NB (2005) A population model to estimate recovery time, population size, and management impacts on Kemp's ridley sea turtles. Chel Conserv Biol 4:767–773

Heppell SS, Burchfield PM, Peña LJ (2007) Kemp's ridley recovery: How far have we come, and where are we headed? In: Plotkin PT (ed) Biology and conservation of ridley sea turtles. The Johns Hopkins University Press, Baltimore, MD, USA, pp 325–336

Hickerson EL (2000) Assessing and tracking resident, immature loggerheads (*Caretta caretta*) in and around the flower garden banks, Northwest Gulf of Mexico. MS Thesis, Texas A&M University, College Station, MD, USA. 102 p

Hickerson EL, Peccini MB (2000) Using GIS to study habitat use by subadult loggerhead sea turtles (*Caretta caretta*) at the Flower Garden Banks National Marine Sanctuary in the Northwest Gulf of Mexico. Proceedings, 19th Annual Symposium on Sea Turtle Conservation and Biology, South Padre Island, TX, USA, March, pp 179–181

Hildebrand HH (1963) Hallazgo del area de anidacion de la Tortuga marina "lora", Lepidochelys kempii (Garman), en la costa occidental del Golfo de Mexico (Rept., Chel.). Science 22:105–112

Hildebrand H (1982) A historical review of the status of sea turtle populations in the Western Gulf of Mexico. In: Bjorndal K (ed) Biology and conservation of sea turtles. Smithsonian Institution Press, Washington, DC, USA, pp 447–453

Hilterman ML, Goverse E (2007) Nesting and nest success of the leatherback turtle (*Dermochelys coriacea*) in Suriname, 1999–2005. Chel Conserv Biol 68:87–100

Hill, D. 2015. FISHING/SHRIMP: Repeals the prohibition on enforcement of the federal TEDS in shrimp nets requirement. HLS 15RS-1313. 2 pp

Hirth HF (1971) Synopsis of the biological data on the green turtle *Chelonia mydas*. FAO Fisheries Synopsis 85. Food and Agriculture Organization of the United Nations, Rome, Italy. 84 p

Hirth HF (1980) Some aspects of the nesting behavior and reproductive biology of sea turtles. Am Zool 20:507–523

Hirth HF (1997) Synopsis of biological data on the green turtle *Chelonia mydas* (Linnaeus 1758). U.S. Fish and Wildlife Biological Report 97(1). U.S. Fish and Wildlife Service, Washington, DC, USA. 120 p

Hirth HF, Ogren LH (1987) Some aspects of the ecology of the leatherback turtle *Dermochelys coriacea* at Laguna Jalova, Costa Rica. NOAA Technical Report NMFS 56. National Marine Fisheries Service, Spring, MD, USA. 14 p

Holder KK, Holder MT (2007) Phylogeography and population genetics. In: Plotkin PT (ed) Biology and conservation of ridley sea turtles. The Johns Hopkins University Press, Baltimore, MD, USA, pp 107–117

Hopkins SR, Richardson JI (1984) Recovery plan for marine turtles: Agency review draft. National Marine Fisheries Service, Silver Spring, MD, USA. 355 p

Hopkins-Murphy SR, Owens DW, Murphy TM (2003) Ecology of immature loggerhead on foraging grounds and adults in internesting habitat in the Eastern United States. In: Bolten AB, Witherington BE (eds) Loggerhead sea turtles. Smithsonian Books, Washington, DC, USA, pp 79–92

Houghton JD, Doyle TK, Wilson MW, Davenport J, Hays GC (2006) Jellyfish aggregations and leatherback turtle foraging patterns in a temperate coastal environment. Ecology 87:1967–1972

Hulin V, Delmas V, Girondot M, Godfrey MH, Guillon JM (2009) Temperature-dependent sex determination and global change: Are some species at greater risk? Oecologia 160:493–506

Innis CJ, Tlusty M, Merigo C, Weber ES (2007) Metabolic and respiratory status of cold-stunned Kemp's ridley sea turtles (*Lepidochelys kempii*). J Comp Phys B 177:623–630

IUCN (International Union for the Conservation of Nature) (2004) Green turtle (*Chelonia mydas*), Version 2012.1. http://www.iucnredlist.org/apps/redlist/details/4615/0. Accessed 20 November 2013

IUCN (2012) IUCN Red list of threatened species, Version 2012.2. http://www.iucnredlist.org/. Accessed 20 November 2013

Jackson JBC (1997) Reefs since Columbus. Coral Reefs 16:S23–S33

Jimenez MC, Filonov A, Tereshchenko I, Márquez-M R (2005) Time-series analyses of the relationship between nesting frequency of the Kemp's ridley sea turtle and meteorological conditions. Chel Conserv Biol 4:774–780

Joergensen K (2012) Green sea turtle hatchling. iStockphoto, Calgary, Alberta, Canada. http://www.istockphoto.com/stock-photo-22302729-green-sea-turtle-hatchling.php. Accessed 20 November 2013

Johnson SA, Ehrhart LM (1994) Nest-site fidelity of the Florida green turtle. Proceedings, 13th Annual Symposium on Sea Turtle Biology and Conservation, Jekyll Island, GA, USA, February, Part I. 83 p

Johnson SA, Bass AL, Libert B, Marshall M, Fulk D (1999a) Kemp's ridley (*Lepidochelys kempi*) nesting in Florida. Florida Scientist 62:194–204

Johnson DR, Yeung C, Brown CA (1999b) Estimates of marine mammal and marine turtle bycatch by the U.S. Atlantic Pelagic Longline Fleet in 1992–1997. NOAA Technical Memorandum NMFS-SEFSC-418. National Marine Fisheries Service, Miami, FL, USA. April. 70 p

Karnauskas M, Schirripa MJ, Kelble CR, Cook GS, Craig JK (2013) Ecosystem status report for the Gulf of Mexico. NOAA Technical Memorandum NMFS-SEFSC-653. National Marine Fisheries Service, Southeast Fisheries Science Center, Miami, FL, USA. December. 52 p

Keller JM, McClellan-Green PD, Kucklick JR, Keil DE, Peden-Adams MM (2006) Effects of organochlorine contaminants on loggerhead sea turtle immunity: Comparison of a correlative field study and *in vitro* exposure experiments. Environ Health Perspect 114:70–76

Klima EF, Gitschlag GR, Renaud ML (1988) Impacts of the explosive removal of offshore petroleum platforms on sea turtles and dolphins. Mar Fish Rev 50:33–42

Kot CY, Boustany AM, Halpin PN (2010) Temporal patters of target catch and sea turtle bycatch in the U.S. Atlantic pelagic longline fishing fleet. Can J Fish Aquat Sci 67:42–57

Lamont MA, Percival HF, Pearlstine LG, Colwell SV, Carthy RR (1998) Sea turtle nesting activity along Eglin Air Force Base on Cape San Blas and Santa Rosa, Island, Florida from 1994 to 1997. Technical Report 59. U.S. Geological Survey, Biological Resources Division, Florida Cooperative Fish and Wildlife Research Unit, Gainsville, FL, USA. 52 p

Landry AM Jr, Seney EE (2006) Nest site selection and post-nesting movements of Kemp's ridley sea turtles along the Upper Texas Coast. Final Report to the Texas General Land Office Coastal Impact Assessment Program GLO Contract 05-233N. Texas A&M University, College Station, TX, USA

Landry AM Jr, Seney EE (2008) Movement and behavior of Kemp's ridley sea turtles in the Northwestern Gulf of Mexico during 2006 and 2007. Final Report to the Schlumberger Excellence in Educational Development Program, Sugar Land, Texas. Texas A&M University, College Station

Landry A Jr, Costa D, Williams B, Coyne M (1992) Turtle capture and habitat characterization study. Final Report. U.S. Army Corps of Engineers, Galveston District, Galveston, TX, USA. 118 p

Landry A Jr, Costa D, Williams B, Coyne M (1993) Sea turtle capture and habitat characterization: South Padre Island and Sabine Pass, TX Environs. Final Report. U.S. Army Corps of Engineers, Galveston and New Orleans Districts. Galveston, TX, USA. 119 p

Landry AM Jr, Costa DT, Coyne MS, Kenyon FL, Werner SA, Fitzgerald PS, St. John KE, Williams BB (1994) Sea turtle capture/population index and habitat characterization: Bolivar Roads and Sabine Pass, TX and Calcasieu Pass, Louisiana. Final Report. U.S. Army Corps of Engineers, Galveston District. Galveston, TX, USA. 332 p

Landry AM Jr, Costa DT, Coyne MS, Kenyon FL, Werner SA, Fitzgerald PS, St. John KE, Williams BB (1995) Sea turtle capture/ population index and habitat characterization: Bolivar Roads and Sabine Pass, Texas, and Calcasieu Pass, Louisiana. Final Report. U.S. Army Corps of Engineers, Galveston District. Galveston. Texas A&M University, College Station, TX, USA. 173 p

Landry AM Jr, Costa DT, Kenyon FL, Coyne MS (2005) Population characteristics of Kemp's ridley sea turtles in nearshore waters of the upper Texas and Louisiana coasts. Chel Conserv Biol 4:801–807

LDWF (Louisiana Department of Wildlife and Fisheries) (2012) Homepage. http://www.wlf.louisiana.gov/. Accessed 20 November 2013

Leary TR (1957) A schooling of leatherback turtles, *Dermochelys coriacea*, on the Texas coast. Copeia 1957:232

LeBuff CR Jr (1974) Unusual nesting relocation in the loggerhead turtle, *Caretta caretta*. Herpetologica 30:29–31

LeBuff CR Jr (1990) The loggerhead turtle in the Eastern Gulf of Mexico. Caretta Research, Sanibel, 216 p

Leslie AJ, Penick DN, Spotila JR, Paladino FV (1996) Leatherback turtle, *Dermochelys coriacea*, nesting and nest success at Tortuguero, Costa Rica, in 1990–1991. Chel Conserv Biol 2:159–168

Lewison RL, Crowder LB, Shaver DJ (2003) The impact of turtle excluder devices and fisheries closures on loggerhead and Kemp's ridley strandings in the western Gulf of Mexico. Conserv Biol 17:1089–1097

Limpus CJ, Nicholls N (1988) The Southern Oscillation regulates the annual numbers of green turtles (*Chelonia mydas*) breeding around northern Australia. Aust Wildlife Res 15:157–161

Lohmann KJ, Lohmann CMF (2003) Orientation mechanisms of hatchlings loggerheads. In: Bolten AB, Witherington BE (eds) Loggerhead sea turtles. Smithsonian Books, Washington, DC, USA, pp 44–62

Lohmann KJ, Putman NF, Lohmann CMF (2012) The magnetic map of hatchling loggerhead sea turtles. Curr Opin Neurobiol 22:336–342

Lohoefener R, Hoggard W, Mullin K, Roden C, Rogers C (1989) Are sea turtles attracted to petroleum platforms? Proceedings, 9th Annual Workshop on Sea Turtle Conservation and Biology, Jekyll Island, GA, USA, August, Part 1, pp 103–104

Lohoefener R, Hoggard W, Mullin K, Roden C, Rogers C (1990) Association of sea turtles with petroleum platforms in the North-Central Gulf of Mexico. OCS Study/MMS 90-0025. Minerals Management Service, New Orleans, LA, USA. 90 p

Louisiana Revised Statutes (1987) RS 56:57.2. http://www.legis.state.la.us/lss/lss.asp?doc=105394. Accessed 20 November 2013

Lund PF (1985) Hawksbill turtle *Ermetmochelys imbricata* nesting on the east coast of Florida. J Herpetol 19:164–166

Luschi P, Hays GC, Papi F (2003) A review of long-distance movements by marine turtles, and the possible role of ocean currents. Oikos 103:293–302

Lutcavage ME, Plotkin P, Witherington B, Lutz PL (1997) Human impacts on sea turtle survival. In: Lutz PL, Musick JA (eds) The biology of sea turtles, vol 1. CRC Press, Boca Raton, FL, USA, pp 387–409

Lutz PL, Musick JA, Wyneken J (eds) (2003) The biology of sea turtles, vol 2. CRC Press, Boca Raton, FL, USA. 472 p

Lyn H, Coleman A, Broadway M, Klaus J, Finerty S, Shannon D, Solangi M (2012) Displacement and site fidelity of rehabilitated immature Kemp's ridley sea turtles (*Lepidochelys kempii*). MTN 135:10–13

Mager AM Jr (1985) Five-year status reviews of sea turtles listed under the endangered species act of 1973. National Marine Fisheries Service, St. Petersburg, FL, USA. 90 p

Magnuson JJ, Bjorndal KA, DuPaul WD, Graham GL, Owens DW, Pritchard PCH, Richardson JI, Saul GE, West CW (1990) Decline of the sea turtles, causes and prevention. National Academy Press, Washington, DC, USA. 260 p

Mannix P (2012) A leatherback turtle covering her eggs, Turtle Beach, Tobago stock photo. Flickr.com. http://www.flickr.com/photos/paulmannix/2633233033/. Accessed 20 November 2013

Manzella SA, Williams JA, Caillouet CW Jr (1990) Radio and sonic tracking of juvenile sea turtles in inshore waters of Louisiana and Texas. Proceedings, 10th Annual Workshop on Sea Turtle Biology and Conservation, Hilton Head Island, SC, USA, February, pp 115–120

Márquez-M R (1972) Resultados prelimiares sobre edad y crecimiento de la tortuga lora, *Lepidochelys kempii* (Garman). Memorias IV Congreso Nacional de Oceanografia, Mexico, pp 419–427

Márquez-M R (1990) FAO species catalogue: An annotated and illustrated catalogue of sea turtles known to date. FAO Fisheries Synopsis No. 125, Volume 11. Food and Agriculture Organization of the United Nations, Rome, Italy. 81 p

Márquez-M R (2004) Las tortugas marinas del Golfo de Mexico: abundancia, distribución y protección. In: Caso M, Pisanty I, Ezcurra E (eds) Diagnóstico ambiental del Golfo de Mexico. Instituto Nacional de Ecología (INE-SEMARNAT), Mexico City, DF, Mexico, pp 173–198

Márquez-M R, Villanueva OA, Sánchez PM (1982) The population of the Kemp's ridley sea turtle in the Gulf of Mexico—*Lepidochelys kempii*. In: Bjorndal KA (ed) Biology and conservation of sea turtles. Smithsonian Institution Press, Washington, DC, USA, pp 159–164

Márquez-M R, Carrasco-A M, Jiménez-O C, Byles R, Burchfield P, Sanchez-P M, Diaz-F J, Leo-P A (1996) Good news! Rising numbers of Kemp's ridley nest at Rancho Nuevo, Tamaulipas, Mexico. MTN 73:2–5

Márquez-M R, Jiménez MC, Carrasco MA, Villanueva NA (1998) Comments concerning the population tendencies of the marine turtle genus *Lepidochelys* after the year 1990. Oceánides 13:41–62

Márquez-M R, Díaz J, Sánchez M, Burchfield P, Leo A, Carrasco M, Peña J, Jiménez C, Bravo R (1999) Results of the Kemp's ridley nesting beach conservation efforts in México. MTN 85:2–4

Márquez-M R, Díaz-F J, Guzmán-H V, Bravo-G R, Jimenez-Q M (2004) Marine turtles of the Gulf of Mexico. Abundance, distribution and protection. In: Caso M, Pisanty I, Ezcurra E (eds) Environmental analysis of the Gulf of Mexico. Harte Research Institute for Gulf of Mexico Studies, Corpus Christi, TX, USA, pp 89–107

Márquez-MR (1994) Synopsis of biological data on the Kemp's ridley sea turtle, *Lepidochelys kempi* (Garman, 1880). NOAA Technical Memorandum NMFS-SEFSC-343. National Marine Fisheries Service, Southeast Fisheries Science Center, Miami, FL, USA. March. 91 p

Márquez-MR (1999) Status and distribution of the Kemp's ridley turtle, *Lepidochelys kempii,* in the wider Caribbean region. Proceedings, The Regional Meeting: Marine Turtle Conservation in the Wider Caribbean Region: A Dialogue for Effective Regional Management, Santo Domingo, Dominican Republic, pp 16–18

Márquez-MR, Ríos OD, Sánchez PJM, Díaz J (1989) Mexico's contribution to Kemp's ridley sea turtle recovery. Proceedings, 1st International Symposium on Kemp's Ridley Sea Turtle Biology, Conservation and Management, Galveston, TX, USA, October, Session 1, pp 4–6

Mays JL, Shaver DJ (1998) Nesting trends of sea turtles in national seashores along Atlantic and Gulf Coast Waters of the United States. Final report for Natural Resources Preservation Program, Project Number 95-15. National Park Service and U.S. Geological Survey, Washington, DC, USA. 61 p

McClellan CM, Read AJ (2007) Complexity and variation in loggerhead sea turtle life history. Biol Lett 3:592–594

McClenachan L, Jackson JBC, Newman MJH (2006) Conservation implications of historic sea turtle nesting beach loss. Front Ecol Environ 4:290–296

McDaniel CJ (1998) A spatial analysis of sea turtle abundance and shrimping intensity in the Gulf of Mexico: Recommendations for conservation and management. Duke University, Durham, NC, USA. 141 p

McDaniel CJ, Crowder LB, Priddy JA (2000) Spatial dynamics of sea turtle abundance and shrimping in the U.S. Gulf of Mexico. Conserv Ecol 4:15

McMichael E, Carthy RR, Seminoff JA (2003) Ecology of Juvenile Sea Turtles in the Northeastern Gulf of Mexico. Proceedings, 23rd Annual Symposium on Sea Turtle Biology and Conservation, Kuala Lumpur, Malaysia, March, pp 20–21.

McMichael E, Seminoff J, Carthy R (2008) Growth rates of wild green turtles, *Chelonia mydas*, at a temperate foraging habitat in the northern Gulf of Mexico: Assessing short-term effects of cold-stunning on growth. J Nat Hist 42:2793–2807

MDWFP (Mississippi Department of Wildlife, Fisheries, & Parks) (2012) Homepage. http://www.mdwfp.com/. Accessed 20 November 2013

Mellgren RL, Mann MA (1996) comparative behavior of hatchling sea turtles. Proceedings, 15th Annual Symposium on Sea Turtle Biology and Conservation, Hilton Head, SC, USA, February, pp 52–53

Mendonca MT (1981) Comparative growth rates of wild immature *Chelonia mydas* and *Caretta caretta* in Florida. J Herpetol 154:444–447

Mendonca MT (1983) Movements and feeding ecology of immature green turtles (*Chelonia mydas*) in a Florida lagoon. Copeia 1983:1013–1023

Mendonca MT, Pritchard PCH (1986) Offshore movements of post-nesting Kemp's ridley sea turtles (*Lepidochelys kempi*). Herpetologica 42:373–381

Merrill MW, Salmon M (2011) Magnetic orientation by hatchling loggerhead sea turtles (*Caretta caretta*) from the Gulf of Mexico. Mar Biol 158:101–112

Metz TL (2004) Factors influencing Kemp's ridley sea turtle (*Lepidochelys kempii*) distribution in nearshore waters and implications for management. Texas A&M University, College Station, TX, USA. 163 p

Meylan A (1984) Feeding ecology of the hawksbill turtle *Eretmochelys imbricata*: spongivory as a feeding niche in the coral reef community. PhD Thesis, University of Florida, Gainsville, FL, USA. 117 p

Meylan A (1988) Spongivory in hawksbill turtles: A diet of glass. Science 22:393–395

Meylan A (1992) Hawksbill turtle *Eretmochelys imbricata*. In: Moler P (ed) Rare and endangered biota of Florida. University Press of Florida, Gainesville, FL, USA, pp 95–99

Meylan AB, Donnelly M (1999) Status justification for listing the hawksbill turtle (*Eretmochelys imbricata*) as critically endangered on the 1996 IUCN Red List of Threatened Animals. Chel Conserv Biol 3:200–224

Meylan A, Redlow A (2006) *Eretmochelys imbricata*—hawksbill turtle. In: Meylan PA (ed) Biology and conservation of florida turtles. Chelonian Research Foundation, Lunenburg, MA, USA, pp 105–127

Meylan AB, Bjorndal KA, Turner BJ (1983) Sea turtles nesting at Melbourne Beach, Florida. II. Post-nesting movements of *Caretta caretta*. Biol Conserv 26:79–90

Meylan A, Castaneda P, Coogan C, Lozon T, Fletemeyer J (1991) First recorded nesting by Kemp's ridley in Florida, USA. MTN 48:8–9

Meylan A, Schroeder B, Mosier A (1995) Sea turtle nesting activity in the State of Florida 1971–1992. Florida Marine Research Publications Number 52. Florida Marine Research Institute, St. Petersburg, FL, USA. 51 p

Milton SL, Leone-Kabler S, Schulman AA, Lutz PL (1994) Effects of Hurricane Andrew on the sea turtle nesting beaches of South Florida. Bull Mar Sci 54:974–981

MMS (Minerals Management Service) (2004) Geological and geophysical exploration for mineral resources on the Gulf of Mexico Outer Continental Shelf. OCS EIA/EA MMS 2004-054. U.S. Department of the Interior, Gulf of Mexico OCS Region, Washington, DC, USA. 487 p

Moncada Gavilan F (2000) Impact of regulatory measures on cuban marine turtle fisheries. Proceedings, 18th International Sea Turtle Symposium, Mazatlán, Sinaloa, Mexico, March, Part VI, pp 108–109

Moncada FG, Hawkes LA, Fish MR, Godley BJ, Manolis SC, Medina Y, Nodarse G, Webb GJW (2012) Patterns of dispersal of hawksbill turtles from the Cuban shelf inform scale of conservation and management. Biol Conserv 148:191–199

Moon DY, MacKenzie DS, Owens DW (1997) Simulated hibernation of sea turtles in the laboratory: I. Feeding, breathing frequency, blood pH, and blood gases. J Exp Zool 278:372–380

Morgan PJ (1989) Occurrence of leatherback turtles (*Dermochelys coriacea*) in the British Isles in 1988 with Reference to a Record Specimen. Proceedings, 9th Annual Workshop on Sea Turtle Conservation and Biology, Jekyll Island, FL, USA, February, Part I, pp 119–120

Morreale SJ, Standora EA (1999) Migration patterns of Northeastern U.S. sea turtles. New York Department of Environmental Conservation NOAA Award NA36FL0430. National Marine Fisheries Service, Silver Spring, MD, USA

Morreale SJ, Standora EA (2005) Western North Atlantic waters: Crucial developmental habitat for Kemp's ridley and loggerhead sea turtles. Chel Conserv Biol 4:872–882

Morreale SJ, Plotkin PT, Shaver DJ, Kalb HJ (2007) Adult migration and habitat utilization: Ridley turtles in their element. In: Plotkin PT (ed) Biology and conservation of ridley sea turtles. The Johns Hopkins University Press, Baltimore, MD, USA, pp 213–230

Mortimer JA (1981) The feeding ecology of the west Caribbean green turtle (*Chelonia mydas*) in Nicaragua. Biotropica 13:49–58

Mortimer JA (2008) The State of the World's Hawksbills. In SWOT (State of the Worlds Sea Turtles), Report, Volume III, Arlington, VA, USA, pp 10–13

Mortimer JA, Donnelly M (2008) Hawksbill turtle (*Eretmochelys imbricata*), Marine Turtle Specialist Group 2008 IUCN Red List Status Assessment. 108 p

Mosier A (1998) The impact of coastal armoring structures on sea turtle nesting behavior at three beaches on the East Coast of Florida. MS Thesis, University of South Florida, Tampa, FL, USA. 112 p

Mosier AE, Witherington BE (2002) Documented effects of coastal armoring structures on sea turtle nesting behavior. Proceedings, 20th Annual Symposium on Sea Turtle Biology and Conservation, Orlando, FL, USA, pp 304–306

Mount RH (ed) (1975) The reptiles and amphibians of alabama. Auburn Printing Company, Auburn, AL, USA. 347 p

Mrosovsky N (1981) Plastic jellyfish. MTN 17:5–7

Mrosovsky N, Pritchard PCH (1971) Body temperatures of *Dermochelys coriacea* and other sea turtles. Copeia 1971:624–631

Mrosovsky N, Yntema CL (1980) Temperature dependence of sexual differentiation in sea turtles: Implications for conservation practices. Biol Conserv 18:271–280

Mrosovsky N, Ryan GD, James MC (2009) Leatherback turtles: The menace of plastic. Mar Pollut Bull 58:287–289

Mullin KD, Hoggard W (2000) Visual surveys of cetaceans and sea turtles from aircraft and ships. In: Davis RW, Evans WE, Würsig B (eds) Cetaceans, sea turtles and seabirds in the Northern Gulf of Mexico: Distribution, abundance and habitat associations, Volume II. Technical Report. Texas A&M at Galveston, National Marine Fisheries Service OCS Study MMS 200-2003. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA, USA, pp 111–172

Murphy MD, McMillen-Jackson AL, Mahmoudi B (2007) A STOCK ASSESSMENT FOR BLUE CRab, *Callinectes sapidus*, in Florida Waters. In House Report 2007-006. Florida Fish and Wildlife Conservation Commission, Division of Marine Fisheries Management, St. Petersburg, FL, USA. June. 85 p

Musick JA, Limpus CJ (1997) Habitat utilization and migration in juvenile sea turtles. In: Lutz PL, Musick JA (eds) The biology of sea turtles, vol 1. CRC Press, Boca Raton, FL, USA, pp 137–163

Najera JJD (1990) Nesting of three species of sea turtles in the northeast coast of the Yucatán Peninsula, Mexico. Proceedings, 10th Annual Workshop on Sea Turtle Biology and Conservation, Hilton Head, SC, USA, pp 29–33

Nelson DA (2000) Winter movements of sea turtles. Proceedings, 19th Annual Symposium on Sea Turtle Biology and Conservation, South Padre Island, TX, USA. March. 26 p

NMFS (National Marine Fisheries Service) (1992) Recovery plan for leatherback turtles in the U.S. Caribbean, Atlantic and Gulf of Mexico. National Marine Fisheries Service, Silver Spring, MD, USA. 65 p

NMFS (2002) Endangered species act-section 7 consultation biological opinion on the cooling water intake system at the crystal river energy complex. National Marine Fisheries Service, Silver Spring, MD, USA. 26 p

NMFS (2007) Report to Congress on the impacts of Hurricanes Katrina, Rita, and Wilma on Alabama, Louisiana, Florida, Mississippi, and Texas Fisheries. National Marine Fisheries Service, Silver Spring, MD, USA. July. 133 p

NMFS (2011a) Green turtle (*Chelonia mydas*) species account. http://www.nmfs.noaa.gov/pr/species/turtles/green.htm. Accessed 20 November 2013

NMFS (2011b) Leatherback turtle (*Dermochelys coriacea*) species account. http://www.nmfs.noaa.gov/pr/species/turtles/leatherback.htm. Accessed 20 November 2013

NMFS (2011c) U.S. National Bycatch Report. NOAA Technical Memorandum NMFS-F/SPO-117C. National Marine Fisheries Service, Silver Spring, MD, USA. September. 508 p

NMFS (2012) Sea turtle conservation; Shrimp trawling requirements. Fed Regist 77:27411–27415

NMFS SEFSC (Southeast Fisheries Science Center) (2001) Stock assessments of loggerhead and leatherback sea turtles and an assessment of the impact of the pelagic longline fishery on the loggerhead and leatherback sea turtles of the western North Atlantic. NOAA Technical Memorandum NMFS-SEFSC-455. National Marine Fisheries Service, Miami, FL, USA. March. 343 p

NMFS SEFSC (2009) An assessment of loggerhead sea turtles to estimate impacts of mortality reductions on population dynamics. NMFS Southeast Fisheries Science Center Contribution PRD-08/09-14. National Marine Fisheries Service, Miami, FL, USA. July. 45 p

NMFS, USFWS (U.S. Fish and Wildlife Service) (1991) Recovery plan for U.S. population of atlantic green turtle *Chelonia mydas*. National Marine Fisheries Service, Washington, DC, USA and Southeast Region, U.S. Fish and Wildlife Service, Atlanta, GA, USA. October. 52 p

NMFS, USFWS (1993) Recovery plan for hawksbill turtles in the U.S. Caribbean Sea, Atlantic Ocean, and Gulf of Mexico. National Marine Fisheries Service, Washington, DC and Southeast Region, U.S. Fish and Wildlife Service, Atlanta, GA, USA. December. 52 p

NMFS, USFWS (2007a) Kemp's ridley sea turtle (*Lepidochelys kempii*): 5-year review Summary and evaluation. Office of Protected Resources, National Marine Fisheries Service, Silver Spring and Southwest Region, U.S. Fish and Wildlife Service, Albuquerque, NM, USA. August. 50 p

NMFS, USFWS (2007b) Loggerhead sea turtle (*Caretta caretta*): 5-year review: Summary and evaluation. Office of Protected Resources, National Marine Fisheries Service, Silver Spring and Jacksonville Ecological Services Field Office, U.S. Fish and Wildlife Service, Jacksonville, FL, USA. August. 55 p

NMFS, USFWS (2007c) Green sea turtle (*Chelonia mydas*): 5-year review: Summary and evaluation. Office of Protected Resources, National Marine Fisheries Service, Silver Spring, MD, USA and Jacksonville Ecological Services Field Office, U.S. Fish and Wildlife Service, Jacksonville, FL, USA. August. 102 p

NMFS, USFWS (2007d) Leatherback sea turtle (*Dermochelys coriacea*): 5-year review: summary and evaluation. Office of Protected Resources, National Marine Fisheries Service, Silver Spring and Jacksonville Ecological Services Field Office, U.S. Fish and Wildlife Service, Jacksonville, FL, USA. August. 79 p

NMFS, USFWS (2007e) Hawksbill sea turtle (*Eretmochelys imbricata*): 5-year review: Summary and evaluation. Office of Protected Resources, National Marine Fisheries Service, Silver Spring and Jacksonville Ecological Services Field Office, U.S. Fish and Wildlife Service, Jacksonville, FL, USA. August. 90 p

NMFS, USFWS (2008) Recovery plan for the Northwest Atlantic population of the loggerhead sea turtle (*Caretta caretta*): Second revision. Office of Protected Resources, National Marine Fisheries Service, Silver Spring, MD, USA. January. 325 p

NMFS, USFWS, SEMARNAT (Secretary of Environment and Natural Resources, Mexico) (2011) Bi-national recovery plan for the Kemp's ridley sea turtle (*Lepidochelys kempii*), 2nd Rev. National Marine Fisheries Service, Silver Spring, MD, USA. 174 p

NOAA (National Oceanic and Atmospheric Administration) (2007) Kemp's ridley sea turtle range. http://www.nmfs.noaa.gov/pr/species/turtles/. Accessed 20 November 2013

NOAA (2009a) Loggerhead sea turtle range. http://www.nmfs.noaa.gov/pr/species/turtles/. Accessed 20 November 2013

NOAA (2009b) Green sea turtle range. http://www.nmfs.noaa.gov/pr/species/turtles/. Accessed 20 November 2013

NOAA (2009c) Leatherback sea turtle range. http://www.nmfs.noaa.gov/pr/species/turtles/. Accessed 20 November 2013

NOAA (2009d) Hawksbill sea turtle range. http://www.nmfs.noaa.gov/pr/species/turtles/. Accessed 20 November 2013

NOAA (2011) Sea turtle photos. National Oceanic and Atmospheric Administration, Silver Spring, MD, USA. http://www.nmfs.noaa.gov/pr/species/turtle/photos.htm. Accessed 20 November 2013

NOAA (2012) New data prompts NOAA fisheries to withdraw proposed rule to require turtle excluder devices in certain shrimp trawls. http://sero.nmfs.noaa.gov/news_room/press_releases/2012/press_release_skimmer_trawl_proposed_rule.pdf. Accessed 20 November 2013

NOAA Fisheries OPR (Office of Protected Resources) (2013) Sea turtles. http://www.nmfs.noaa.gov/pr/species/turtles/. Accessed 20 November 2013

Nolasco del Aguila D, Debade X, Harrison E (2008a) Report on the 2007 green turtle program at Tortuguero, Costa Rica. Caribbean Conservation Corporation, Gainesville, FL, USA and Ministry of Environment and Energy of Costa Rica. San Pedro, Costa Rica, 63 p

Nolasco del Aguila D, Debade X, Harrison E (2008b) Report on the 2007 leatherback program at Tortuguero, Costa Rica. Caribbean Conservation Corporation, Gainesville and Ministry of Environment and Energy of Costa Rica, San Pedro, Costa Rica. 37 p

Nolasco del Aguila D, Debade X, Harrison E (2009) Report on the 2008 green turtle program at Tortuguero, Costa Rica. Caribbean Conservation Corporation, Gainesville and Ministry of Environment and Energy of Costa Rica, San Pedro, Costa Rica. 66 p

NPS (National Park Service) (2013a) Padre Island: Current sea turtle nesting season. http://www.nps.gov/pais/naturescience/current-season.htm. Accessed 20 November 2013

NPS (2013b) Padre Island: Sea turtle recovery project. http://www.nps.gov/pais/naturescience/strp.htm. Accessed 20 November 2013

NPS (2013c) Padre Island National Seashore: Young Kemp's ridley hatchlings entering water. http://www.nps.gov/pais/naturescience/kridley.htm. Accessed 20 November 2013

NPS (2013d) The leatherback sea turtle. http://www.nps.gov/pais/naturescience/leatherback.htm. Accessed 20 November 2013

NRC (National Research Council) (2010) Sea turtle status and trends: Integrating demography and abundance. Prepared for the National Ocean and Atmospheric Administration, National Academies Press, Washington, DC, USA. 162 p

Ogren LH (1989) Distribution of Juvenile and Subadult Kemp's ridley turtles: Preliminary results from the 1980–1987 surveys. Proceedings, 1st iInternational Symposium on Kemp's Ridley Sea Turtle Biology, Conservation and Management, Galveston, TX, USA. October, Session VI, pp 116–123

Ogren L, McVea CJ (1982) Apparent hibernation by sea turtles in North American Waters. In: Bjorndal KA (ed) Biology and conservation of sea turtles. Smithsonian Institution Press, Washington, DC, USA, pp 127–132

Overton EB, Byrne CJ, McFall JA, Antoine SR, Laseter JL (1983) Results from the chemical analyses of oily residue samples taken from stranded juvenile sea turtles collected from Padre and Mustang Islands, Texas. Prepared for the Minerals Management Service, Special Report 1983-32. Center for Bio-Organic Studies, New Orleans, LA, USA. December. 25 p

Owens D, Crowell D, Dienberg G, Grassman M, McCain S, Morris Y, Schwantes N, Wibbels T (eds) (1983) Western Gulf of Mexico Sea Turtle Proceedings, January 13–14, 1983. TAMU-SG-84-105. Sea Grant College Program, Texas A&M University, College Station, TX, USA. October. 74 p

Parsons JJ (1962) The green turtle and man. Rose Printing Company, Tallahassee, FL, USA. 126 p

Passerotti MS, Carlson JK (2009) Catch and bycatch in the U.S. Southeast Gillnet Fisheries, 2008. NOAA Technical Memorandum NMFS-SEFSC-583. National Marine Fisheries Service, Panama City, FL, USA. February. 19 p

Passerotti MS, Carlson JK, Gulak SJB (2010) Catch and bycatch in the U.S. Southeast Gillnet Fisheries, 2009. NOAA Technical Memorandum NMFS-SEFSC-600. National Marine Fisheries Service, Panama City, FL, USA. February. 20 p

Pelletier D, Roos D, Ciccione S (2003) Oceanic survival and movements of wild and captive-reared immature green turtles (*Chelonia mydas*) in the Indian Ocean. Aquat Living Resour 16:35–41

Penick DN, Spotila JR, O'Connor MP, Steyermark AC, George RH, Salice CJ, Paladino FV (1998) Thermal independence of muscle tissue metabolism in the leatherback turtle, *Dermochelys coriacea*. Comp Biochem Phys A 120:399–403

Pérez-Castañeda R, Salum-Fares A, Defeo O (2007) Reproductive patterns of the hawksbill turtle *Eretmochelys imbricata* in sandy beaches of the Yucatán Peninsula. J Mar Biol Assoc UK 87:815–824

Pierce RH, Henry MS (2008) Harmful algal toxins of the Florida red tide (*Karenia brevis*): natural chemical stressors in South Florida coastal ecosystems. Ecotoxicology 17:623–631

Pike DA, Stiner JC (2007) Sea turtles vary in their susceptibility to tropical cyclones. Oecologia 153:471–478

Plotkin PT (1989) The feeding ecology of the loggerhead sea turtle in the Northwestern Gulf of Mexico. Texas A&M University, Station, TX, USA. 124 p

Plotkin PT (1995) National Marine Fisheries Service and U.S. Fish and Wildlife Service status reviews for sea turtles listed under the endangered species act of 1973. National Marine Fisheries Service, Silver Spring, MD, USA. 139 p

Plotkin PT (1996) Occurrence and diet of juvenile loggerhead sea turtles, *Caretta caretta*, in the northwestern Gulf of Mexico. Chel Conserv Biol 2:78–80

Plotkin PT (2007a) Introduction. In: Plotkin PT (ed) Biology and conservation of ridley sea turtles. The Johns Hopkins University Press, Baltimore, MD, USA, pp 3–5

Plotkin PT (2007b) Near extinction and recovery. In: Plotkin PT (ed) Biology and conservation of ridley sea turtles. The Johns Hopkins University Press, Baltimore, MD, USA, pp 337–340

Plotkin PT (2007c) Biology and conservation of ridley sea turtles. The Johns Hopkins University Press, Baltimore, MD, USA. 368 p

Plotkin P, Amos AF (1988) Entanglement in and ingestion of marine debris by sea turtles stranded along the south Texas coast. Proceedings, 8th Annual Workshop on Sea Turtle Conservation and Biology, Fort Fisher, NC, USA, February, pp 79–82

Plotkin P, Amos AF (1990) Effects of anthropogenic debris on sea turtles in the Northwestern Gulf of Mexico. Proceedings, 2nd International Workshop on Marine Debris, Honolulu, HI, USA, April, pp 736–743

Plotkin PT, Spotila JR (2002) Post-nesting migrations of loggerhead turtles *Caretta caretta* from Georgia, USA: Conservation implications for a genetically distinct subpopulation. Oryx 36:396–399

Plotkin PT, Wicksten MK, Amos AF (1993) Feeding ecology of the loggerhead sea turtle, *Caretta caretta*, in the northwestern Gulf of Mexico. Mar Biol 115:1–15

Polovina JJ, Balazs GH, Howell EA, Parker DM, Seki MP, Dutton PH (2004) Forage and migration habitat of loggerhead (*Caretta caretta*) and olive ridley (*Lepidochelys olivacea*) sea turtles in the central North Pacific Ocean. Fish Oceanogr 13:36–51

Ponwith B (2011) Data analysis request: Update of turtle bycatch in the Gulf of Mexico and Southeastern Atlantic Shrimp Fisheries. NMFS Memorandum for Roy E. Crabtree, January 5, 2011. 10 p. http://seaturtles.org/downloads/UPR_SEFSC_shrimp_bycatch_2011.pdf. Accessed 20 November 2013

Price AB, Gearhart JL (2011) Evaluations of Turtle Excluder Device (TED) Performance in the U.S. Southeast Atlantic and Gulf of Mexico Skimmer Trawl Fisheries. NOAA Technical Memorandum NMFS-SEFSC-615. National Marine Fisheries Service, Beaufort, NC and Pascagoula, MS, USA. May. 15 p

Pritchard PCH (1969) Studies of the systematics and reproductive cycles of the genus *Lepidochelys*. University of Florida, Gainsville, FL, USA. 226 p

Pritchard PCH (1976) Post-nesting movements of marine turtles (Cheloniidae and Dermochelyidae) tagged in the Guianas. Copeia 1976:749–754

Pritchard PCH (1982) Nesting of the leatherback turtle, *Dermochelys coriacea* in Pacific Mexico, with a new estimate of the world population status. Copeia 1982:741–747

Pritchard PCH (2007a) Evolutionary relationships, osteology, morphology, and zoogeography of ridley sea turtles. In: Plotkin PT (ed) Biology and conservation of ridley sea turtles. The Johns Hopkins University Press, Baltimore, MD, USA, pp 45–58

Pritchard PCH (2007b) Arribadas I have known. In: Plotkin PT (ed) Biology and conservation of ridley sea turtles. The Johns Hopkins University Press, Baltimore, MD, USA, pp 7–22

Pritchard PCH (2010) The most valuable reptile in the world—the green turtle. In SWOT (State of the World's Sea Turtles), Report Volume VI, Arlington, VA, USA, pp 24–29

Pritchard PCH, Márquez-M. R (1973) Kemp's ridley or Atlantic ridley, *Lepidochelys kempii*. IUCN Monograph No. 2 (Marine Turtle Series), Morges, VD, Switzerland. 30 p

Pulver JR, Scott-Denton E, Williams JA (2012) Characterization of the U.S. Gulf of Mexico Skimmer Trawl Fishery Based on Observer Coverage. NOAA Technical Memorandum NMFS-SEFSC-636. National Marine Fisheries Service, Galveston, TX, USA. October. 27 p

Putman NF, Shay TJ, Lohmann KJ (2010) Is the geographic distribution of nesting in the Kemp's ridley turtle shaped by the Migratory Needs of Offspring? Proceedings, integrative and comparative biology, Seattle, WA, USA, January, pp 305–314

Rabalais SC, Rabalais NN (1980) The occurrence of sea turtles on the South Texas coast. Contrib Mar Sci 23:123–129

Ramirez PA, Ania LV (2000) Incidence of marine turtles in the Mexican Long-line Tuna Fishery in the Gulf of Mexico. Proceedings, 18th Annual Workshop on Sea Turtle Biology and Conservation, Mazatlan, Sinaloa, Mexico, March, Part XI. 110 p

Reardon R, Mansfield K (1997) Annual report-1997 season: Dry Tortugas National Park, Sea Turtle Monitoring Program, Monroe County, Florida. National Park Service, Key West, FL, USA. 37 p

Rebel TP (1974) Sea turtles and the turtle industry of the West Indies, Florida, and the Gulf of Mexico. University of Miami Press, Coral Gables, FL, USA. 250 p

Reeves RR, Leatherwood S (1983) Autumn sightings of marine turtles (Cheloniidae) off South Texas. Southwest Nat 28:281–288

Reich KJ, Bjorndal KA, Bolten AB, Witherington BE (2007a) Do some loggerheads nesting in Florida have an oceanic foraging strategy? An assessment based on stable isotopes. Proceedings, 24th Annual Symposium on Sea Turtle Biology and Conservation, San Jose, Costa Rica. February. 32 p

Reich KJ, Bjorndal KA, Bolten AB (2007b) The 'lost years' of green turtles: Using stable isotopes to study cryptic life stages. Biol Lett 3:712–714

Reina RD, Abernathy KJ, Marshall GJ, Spotila JR (2005) Respiratory frequency, dive behaviour and social interactions of leatherback turtles, *Dermochelys coriacea* during the internesting interval. J Exp Mar Biol Ecol 316:1–16

Renaud ML (1995) Movements and submergence patterns of Kemp's ridley turtles (*Lepidochelys kempii*). J Herpetol 29:370–374

Renaud ML (2001) Sea turtles of the Gulf of Mexico. Proceedings, Gulf of Mexico Marine Protected Species Workshop, June 1999. New Orleans, LA, USA, June, pp 41–48

Renaud ML, Carpenter JA (1994) Movements and submergence patterns of loggerhead turtles (*Caretta caretta*) in the Gulf of Mexico determined through satellite telemetry. Bull Mar Sci 55:1–15

Renaud ML, Williams JA (1997) Movement of Kemp's ridley sea turtles (*Lepidochelys kempii*) and green (*Chelonia mydas*) sea turtles using Lavaca Bay and Matagorda Bay, 1996–1997. Final Report Submitted to the U.S. Environmental Protection Agency, Office of Planning and Coordination, Region 6, Dallas, TX, USA. 54 p

Renaud ML, Williams JA (2005) Kemp's ridley sea turtle movements and migrations. Chel Conserv Biol 4:808–816

Renaud M, Gitschlag G, Klima E, Shah A, Nance J, Caillouet C, Zein-Eldin Z, Koi D, Patella K (1990) Evaluation of the impacts of turtle excluder devices (TEDs) on Shrimp Catch Rates in the Gulf of Mexico and South Atlantic, March 1988 through July 1989. NOAA Technical Memorandum NMFS-SEFSC-254. National Marine Fisheries Service, Galveston, TX, USA. May. 165 p

Renaud M, Gitschlag G, Klima E, Shah A, Koi D, Nance J (1991) Evaluation of the impacts of turtle excluder devices (TEDs) on Shrimp Catch Rates in Coastal Waters of the United States along the Gulf of Mexico and Atlantic, September 1989 through August 1990. NOAA Technical Memorandum NMFS-SEFSC-288. National Marine Fisheries Service, Galveston, TX, USA. September. 80 p

Renaud M, Gitschlag G, Manzella S, Williams J (1992) Tracking of Green (*Chelonia mydas*) and Loggerhead (*Caretta caretta*) sea turtles using radio and sonic telemetry at South Padre Island, Texas, June-September 1991. Final Report. National Marine Fisheries Service, Galveston, TX, USA. 52 p

Renaud ML, Carpenter JA, Williams JA, Manzella SA (1993a) Telemetric tracking of Kemp's ridley sea turtles (*Lepidochelys kempii*) in Relation to Dredged Channels at Bolivar Pass and Sabine Pass, TX and Calcasieu Pass, LA, May 1993 through February 1994. Final Report. National Marine Fisheries Service, Galveston, TX, USA. 76 p

Renaud ML, Carpenter JA, Manzella SA, Williams JA (1993b) Telemetric tracking of green sea turtles (*Chelonia mydas*) in Relation to Dredged Channels at South Padre Island, Texas July through September 1992. Final Report. National Marine Fisheries Service, Galveston, TX, USA. 56 p

Renaud M, Carpenter J, Williams J, Carter D, Williams B (1995a) Movement of Kemp's ridley sea turtles (*Lepidochelys kempii*) near Bolivar Roads Pass and Sabine Pass, Texas and Calcasieu Pass, Louisiana. Final Report. National Marine Fisheries Service, Galveston, TX, USA. 75 p

Renaud ML, Carpenter JA, Williams JA, Manzella-Tirpak SA (1995b) Activities of juvenile green turtles, *Chelonia mydas*, at a jettied pass in south Texas. Fish Bull 93:586–593

Renaud ML, Carpenter JA, Williams JA, Landry AM Jr (1996) Kemp's ridley sea turtle (*Lepidochelys kempii*) tracked by satellite telemetry from Louisiana to nesting beach at Rancho Nuevo, Tamaulipas, Mexico. Chel Conserv Biol 2:108–109

Reyes C, Troëng S (2001) Report on the 2000 leatherback program at Tortuguero, Costa Rica. Caribbean Conservation Corporation, Gainesville, and Ministry of Environment and Energy of Costa Rica, San Pedro, Costa Rica. 31 p

Richards PM, Epperly SP, Heppell SS, King RT, Sasso CR, Moncada F, Nodarse G, Shaver DJ, Medina Y, Zurita J (2011) Sea turtle population estimates incorporating uncertainty: a new approach applied to western north Atlantic loggerheads *Caretta caretta*. Endang Species Res 15:151–158

Roberts K, Collins J, Paxton CH, Hardy R, Downs J (2014) Weather patterns associated with green turtle hypothermic stunning events in St. Joseph Bay and Mosquito Lagoon, Florida. Physical Geography 35:134–150

Rosman I, Boland GS, Martin LR, Chandler CR (1987) Underwater sightings of sea turtles in the Northern Gulf of Mexico. OCS Study MMS 87/0107. Minerals Management Service, Washington, DC, USA. 37 p

Ross JP, Beavers S, Mundell D, Airth-Kindree M (1989) The status of Kemp's ridley. Center for Marine Conservation, Washington, DC, USA. 51 p

Rostal DC (1991) The reproductive behavior and physiology of the Kemp's ridley sea turtle, *Lepidochelys kempii* (Garman, 1880). PhD Thesis, Texas A&M University, College Station, TX, USA. 138 p

Rostal DC (2005) Seasonal reproductive biology of the Kemp's ridley sea turtle (*Lepidochelys kempii*): Comparison of captive and wild populations. Chel Conserv Biol 4:788–800

Rostal DC, Grumbles JS, Byles RA, Márquez-M R, Owens DW (1997) Nesting physiology of Kemp's ridley sea turtles, *Lepidochelys kempi*, at Rancho Nuevo, Tamaulipas, Mexico, with observations on population estimates. Chel Conserv Biol 2:538–547

Rudloe A, Rudloe J (2005) Site specificity and the impact of recreational fishing activity on subadults endangered Kemp's ridley sea turtles in estuarine foraging habitats in the northeastern Gulf of Mexico. Gulf Mex Sci 2005:186–191

Rudloe A, Rudloe J, Ogren L (1991) Occurrence of immature Kemp's ridley turtles, *Lepidochelys kempi*, in coastal waters of northwest Florida. Northeast Gulf Science 12:49–53

Rybitski MJ, Hale RC, Musick JA (1995) Distribution of organochlorine pollutants in Atlantic sea turtles. Copeia 1995:379–390

Safina C, Wallace B (2010) Solving the "Ridley Riddle." In SWOT (State of the Worlds Sea Turtles). Report Volume V. Arlington, VA, USA, pp 26–30

Salmon M, Jones TT, Horch KW (2004) Ontogeny of diving and feeding behavior in juvenile sea turtles: Leatherback sea turtles (*Dermochelys coriacea* L) and green sea turtles (*Chelonia mydas* L) in the Florida Current. J Herpetol 38:36–43

Salum-Fares AD (2003) Analisis demográfico-reproductivo de la tortuga *Eretmochelys imbricata* (Linnaeus 1766) en playas arenosas de la Península de Yucatán: un estudio de largo plazo. Centro de Investigación y de Estudios Avanzados del Instituto Politécnico Nacional, Merida, Yucatán, Mexico

SEMARNAT (Secretaria de Medio Ambiente y Recursos Naturales) (2012) Homepage. http://www.semarnat.gob.mx/Pages/Inicio.aspx. Accessed 20 November 2013

Sánchez-Pérez JM, Vasconcelos-Pérez J, Díaz-Flores J (1989) Campamentos para Investigación, Proteccion y Fomento de Tortugas Marinas. In Tortugas: Golfina, Laud, Prieta, Blanca, Cahuama y Carey, Compilacion 1999. Instituto Nacional de Ecología, DF, Mexico, pp 17–28

Sarmiento Devia R, Harrison E (2010) Report on the 2009 leatherback program at Tortuguero, Costa Rica. Caribbean Conservation Corporation, Gainesville and Ministry of Environment and Energy of Costa Rica, San Pedro, Costa Rica. 39 p

Scarygami (2012) Hawksbill turtle stock photo. Flickr.com. http://www.flickr.com/photos/scarygami/5254129116/. Accessed 20 November 2013

Schmeltz GW, Mezich RR (1988) A preliminary investigation of the potential impact of Australian pines on the nesting activities of the Loggerhead Turtle. Proceedings, 8th Annual Workshop on Sea Turtle Conservation and Biology, Fort Fisher, NC, USA, December, pp 63–66

Schmid JR (1998) Marine turtle populations on the west-central coast of Florida: Results of tagging studies at the Cedar Keys, Florida, 1986–1995. Fish Bull 96:589–602

Schmid JR (2011) Assessment of marine turtle aggregations in the Coastal Waters of Lee County, Florida. Final Report to Sea Turtle Grants Program for Contract No. #09-050R, Submitted by Conservancy of Southwest Florida to Sea Turtle Conservancy, Gainsville, FL, USA. May. 28 p

Schmid JR, Barichivich WJ (2005) Developmental biology and ecology of Kemp's ridley turtles in the eastern Gulf of Mexico. Chel Conserv Biol 4:828–834

Schmid JR, Barichivich WJ (2006) *Lepidochelys kempii*—Kemp's ridley turtle. In: Meylan PA (ed) Biology and conservation of Florida turtles. Chelonian Research Foundation, Lunenburg, MA, USA, pp 128–141

Schmid JR, Ogren LH (1990) Results of a tagging study at Cedar Key, Florida, with comments on Kemp's ridley distribution in the southeastern U.S. Proceedings, 10th Annual Workshop on Sea Turtle Biology and Conservation, Hilton Head, SC, USA, August, pp 129–130

Schmid JR, Witzell WN (1997) Age and growth of wild Kemp's ridley sea turtles, *Lepidochelys kempi*: Cumulative results of tagging studies in Florida. Chel Conserv Biol 2:532–537

Schmid JR, Witzell WN (2006) Seasonal migrations of immature Kemp's ridley turtles (*Lepidochelys kempii* Garman) along the west coast of Florida. Gulf Mex Sci 24:28–40

Schmid JR, Woodhead A (2000) Von Bertalanffy growth models for wild Kemp's ridley turtles: Analysis of the NMFS miami laboratory tagging database. In: Group TEW (ed) Assessment update for the Kemp's ridley and loggerhead sea turtle populations in the Western North Atlantic. National Marine Fisheries Service, Silver Spring, MD, USA, pp 94–102

Schmid JR, Bolten AB, Bjorndal KA, Lindberg WJ (2002) Activity patterns of Kemp's ridley turtles, *Lepidochelys kempii*, in the coastal waters of the Cedar Keys, Florida. Mar Biol 140:215–228

Schmid JR, Bolten AB, Bjorndal KA, Lindberg WJ, Percival HF, Zwick PD (2003) Home range and habitat use by Kemp's ridley turtles in west-central Florida. J Wildl Manage 67:196–206

Schroeder B, Murphy S (1999) Population surveys (Ground and Aerial) on nesting beaches. In: Eckert KL, Bjorndal KA, Abreu-Grobois FA, Donnelly M (eds) Research and Management Techniques for the Conservation of Sea Turtles. IUCN/SSC Marine Turtle Specialist Group Publication 4. Washington, DC, USA, pp 45–55

Schroeder BA, Foley AM, Witherington BE, Mosier AE (1998) Ecology of marine turtles in Florida Bay: Population structure, distribution, and occurrence of Fibropapilloma. Proceedings, 17th Annual Turtle Symposium, Orlando, FL, USA, pp 281–283

Schroeder BA, Foley AM, Bagley DA (2003) Nesting patterns, reproductive migrations, and adult foraging areas of loggerhead turtles. In: Bolten AB, Witherington BE (eds) Loggerhead sea turtles. Smithsonian Books, Washington, DC, USA, pp 114–124

Schwartz FJ (1978) Behavioral and tolerance responses to cold water temperatures by three species of sea turtles (Reptilia, Cheloniidae) in North Carolina. Fla Mar Res Publ 33:16–18

SeaTurtle.org (2011) News coverage: Scientist letter to Louisiana's Governor Makes Waves. http://seaturtles.org/article.php?id=2188. Accessed 20 November 2013

Seney EE (2008) Population dynamics and movements of the Kemp's ridley sea turtle, *Lepidochelys kempii*, in the Northwestern Gulf of Mexico. PhD Thesis, Texas A&M University, College Station, TX, USA. 168 p

Seney EE, Landry AM Jr (2008) Movements of Kemp's ridley sea turtles nesting on the upper Texas coast: Implications for management. Endang Species Res 4:73–84

Seney EE, Landry AM Jr (2011) Movement patterns of immature and adult female Kemp's ridley sea turtles in the northwestern Gulf of Mexico. Mar Ecol Prog Ser 440:241–254

Serge_Vero (2007) Sea turtle hatchling. iStockphoto, Calgary, Alberta, Canada. http://www.istockphoto.com/stock-photo-4557351-sea-turtle-hatchling.php?st=874d172. Accessed 20 November 2013

Share the Beach (2013) Nesting season statistics. http://www.alabamaseaturtles.com/nesting-season-statistics/. Accessed 20 November 2013

Shaver DJ (1990a) Sea turtles in South Texas Inshore Waters. Proceedings, 10th Annual Workshop on Sea Turtle Biology and Conservation, Hilton Head Island, SC, USA, August, pp 131–132

Shaver DJ (1990b) Hypothermic stunning of sea turtles in Texas. MTN 48:25–27

Shaver DJ (1991) Feeding ecology of Kemp's ridley in South Texas waters. J Herpetol 25:327–334

Shaver DJ (1992) Kemp's ridley sea turtle reproduction. Herpetol Rev 23:59

Shaver DJ (1994) Relative abundance, temporal patterns, and growth of sea turtles at the Mansfield Channel, Texas. J Herpetol 28:491–497

Shaver DJ (1996) Head-started Kemp's ridley turtles nest in Texas. MTN 74:5–7

Shaver DJ (1997) Kemp's ridley turtles from international project return to Texas to Nest. Proceedings, 16th Annual Gulf of Mexico Information Transfer Meeting, New Orleans, LA, USA, December, pp 38–40

Shaver DJ (1999) Kemp's ridley sea turtle project at Padre Island National Seashore, Texas. Proceedings, 17th Annual Gulf of Mexico Information Transfer Meeting, New Orleans, LA, USA, December, Session 1G, pp 342–347

Shaver DJ (2000) Distribution, residency, and seasonal movements of the green sea turtle, *Chelonia mydas* (Linnaeus, 1758) in Texas. PhD Thesis. Texas A&M University, College Station, TX, USA. 237 p

Shaver DJ (2001a) Padre Island National Seashore Kemp's ridley sea turtle project and Texas sea turtle nesting and stranding 2000 report. U.S. Geological Survey, Reston, VA, USA. 61 p

Shaver DJ (2001b) U.S. Geological Survey/National Park Service Kemp's ridley sea turtle research and monitoring programs in Texas. Proceedings, Gulf of Mexico marine protected species workshop, New Orleans, LA, USA, June, pp 121–124

Shaver DJ (2004) Kemp's ridley project at Padre Island National Seashore and Texas Sea Turtle Nesting and Stranding 2002 Report. Proceedings, 17th Annual Gulf of Mexico Information Transfer Meeting, New Orleans, LA, USA, December, Session 1G, pp 342–347

Shaver DJ (2005) Analysis of the Kemp's ridley imprinting and headstart project at Padre Island National Seashore, Texas, 1978–1988, with subsequent nesting and stranding records on the Texas coast. Chel Conserv Biol 4:846–859

Shaver DJ (2006a) Kemp's ridley sea turtle habitat use in Mexico. Final Programmatic Report to the National Fish and Wildlife Foundation (2003-0212-009). National Park Service, Washington, DC, USA. 59 p

Shaver DJ (2006b) Kemp's ridley sea turtle nesting increasing in Texas. In: Harmon D (ed) People, places, and parks: Proceedings, 2005 George Wright Society Conference on Parks, Protected Areas, and Cultural Sites, Philadelphia, PA, USA, April, pp 391–397

Shaver DJ (2007) Texas sea turtle nesting and stranding 2006 report. National Park Service, Washington, DC, USA. 33 p

Shaver DJ (2008) Texas sea turtle nesting and stranding 2007 report. National Park Service, Washington, DC, USA. 36 p

Shaver DJ, Caillouet CW (1998) More Kemp's ridley turtles return to South Padre Island to nest. MTN 82:1–5

Shaver DJ, Rubio C (2008) Post-nesting movement of wild and headstarted Kemp's ridley sea turtles *Lepidochelys kempii* in the Gulf of Mexico. Endang Spec Res 4:43–55

Shaver DJ, Wibbels T (2007) Head-starting the Kemp's ridley sea turtle. In: Plotkin PT (ed) Biology and conservation of ridley sea turtles. The Johns Hopkins University Press, Baltimore, MD, USA, pp 297–324

Shaver DJ, Owens DW, Chaney AH, Caillouet CW Jr, Burchfield PB, Márquez-M. R (1988) Styrofoam box and beach temperatures in relation to incubation and sex ratios of Kemp's ridley sea turtles. Proceedings, 8th Annual Workshop on Sea Turtle Conservation and Biology, Fort Fisher, NC, USA, December, pp 103–108

Shaver DJ, Schroeder BA, Byles RA, Burchfield PM, Peña J, Márquez-M R, Martinez HJ (2005) Movements of home ranges of adult male Kemp's ridley sea turtles (*Lepidochelys kempii*) in the Gulf of Mexico investigated by satellite telemetry. Chel Conserv Biol 4:817–827

Shaver DJ, Hart KM, Fujisaki I, Rubio C, Sartain AR, Peña J, Burchfield PM, Gomez Gamez D, Ortiz J (2013) Foraging area fidelity for Kemp's ridleys in the Gulf of Mexcio. Ecol Evol 3:2002–2012

Spotila JR, Standora EA, Morreale SJ, Ruiz GJ (1987) Temperature dependent sex determination in the green turtle (*Chelonia mydas*) effects on the sex ratio on a natural beach. Herpetologica 43:74–81

Spotila JR, Dunham AE, Leslie AJ, Steyermark AC, Plotkin PT, Paladino FV (1996) Worldwide population decline of *Dermochelys coriacea*: are leatherback turtles going extinct? Chel Conserv Biol 2:209–222

Spotila JR, Reina RD, Steyermark AC, Plotkin PT, Paladino FV (2000) Pacific leatherback turtles face extinction. Nature 405:529–530

Standora EA, Spotila JR, Keinath JA, Shoop CR (1984) Body temperatures, diving cycles, and movement of a subadult leatherback turtle, *Dermochelys coriacea*. Herpetologica 40:169–176

Steiner T (1994) Shrimpers implicated as strandings soar in the USA. MTN 67:2–5

Stevenson CH (1893) Report on the coast fisheries of Texas. Report of the Commissioner for 1889 to 1891. U.S. Commission of Fish and Fisheries, Washington, DC, USA, pp 373–420

Stewart KR (2007) Establishment and growth of a sea turtle rookery: The population biology of the leatherback in Florida. PhD Thesis. Duke University, Durham, NC, USA. 129 p

Stewart K, Johnson C (2006) *Dermochelys coriacea*—leatherback sea turtle. In: Meylan PA (ed) Biology and conservation of Florida Turtles. Chelonian Research Monographs No. 3, Lunenburg, MA, USA, pp 144–157

Stewart K, Johnson C, Godfrey MH (2007) The minimum size of leatherbacks at reproductive maturity, with a review of sizes for nesting females from the Indian, Atlantic and Pacific ocean basins. J Herpetol 17:123–128

Stewart KR, Keller JM, Templeton R, Kucklick JR, Johnson C (2011) Monitoring persistent organic pollutants in leatherback turtles (*Dermochelys coriacea*) confirms maternal transfer. Mar Pollut Bull 62:1396–1409

Stokes LW, Epperly SP, McCarthy KJ (2012) Relationship between hook type and hooking location in sea turtles incidentally captured in the United States Atlantic pelagic longline fishery. Bull Mar Sci 88:703–718

STSSN (Sea Turtle Stranding and Salvage Network) (2012) STSSN database. Southeast Fisheries Science Center, Miami. http://www.sefsc.noaa.gov/species/turtles/strandings.htm. Accessed 20 November 2013

SWOT (State of the World's Sea Turtles) (2007a) A global glimpse of loggerhead nesting. Report Volume II. Arlington, VA, USA. 50 p

SWOT (2007b) Burning issues in conservation. Leatherback sea turtles of the world. Report Volume I. Arlington, VA, USA. 38 p
SWOT (2008) Where the hawksbill are. Report Volume III. Arlington, VA, USA. 44 p
SWOT (2010a) Kemp's and olive ridleys: Small turtles, big secrets. Report Volume V. Arlington, VA, USA. 54 p
SWOT (2010b) The green turtle: The most valuable reptile in the world. Report Volume VI. Arlington, VA, USA. 62 p
Taquet C, Taquet M, Dempster T, Soria M, Ciccione S, Roos D, Dagorn L (2006) Foraging of the green sea turtle *Chelonia mydas* on seagrass beds at Mayotte Island (Indian Ocean), determined by acoustic transmitters. Mar Ecol Prog Ser 306:295–302
Teas WG (1993) Species composition and size class distribution of marine turtle strandings on the Gulf of Mexico and Southeast United States Coasts, 1985–1991. NOAA Technical Memorandum NMFS-SEFSC-315, National Marine Fisheries Service, Miami, FL, USA. January. 43 p
TEWG (Turtle Expert Working Group) (1998) An assessment of the Kemp's ridley (*Lepidochelys kempii*) and loggerhead (*Caretta caretta*) sea turtle population in the Western North Atlantic. NOAA Technical Memorandum NMFS-SEFSC-409. Southeast Fisheries Science Center, National Marine Fisheries Service, Miami, FL, USA. March. 96 p
TEWG (2000) Assessment update for the Kemp's ridley and loggerhead sea turtle populations in the Western North Atlantic. NOAA Technical Memorandum NMFS-SEFSC-444. Southeast Fisheries Science Center, National Marine Fisheries Service, Miami, FL, USA. November. 115 p
TEWG (2007) An assessment of the leatherback turtle population in the Atlantic Ocean. NOAA Technical Memorandum NMFS-SEFSC-555. Southeast Fisheries Science Center, National Marine Fisheries Service, Miami, FL, USA. April. 116 p
TEWG (2009) An assessment of the loggerhead turtle population in the Western North Atlantic Ocean. NOAA Technical Memorandum NMFS-SEFSC-575. Southeast Fisheries Science Center, National Marine Fisheries Service, Miami, FL, USA. 131 p
Texas A&M University (2011) GEOSCIENCES NEWS: To the rescue. Texas A&M, College Station, TX, USA. http://geonews.tamu.edu/latestnews/775-to-the-rescue.html. Accessed 20 January 2014
Thompson NB (1988) The status of loggerhead, *Caretta caretta*; Kemp's ridley, *Lepidochelys kempi*; and green, *Chelonia mydas*, sea turtles in U.S. waters. Mar Fish Rev 50:16–23
Thompson N (1991) A response to Dodd and Byles. Mar Fish Rev 53:31–33
Thompson NB (1993) Preliminary analysis of sea turtle stranding event: Northern Gulf of Mexico, May/June 1993. Miami Laboratory Contribution MIA-93/94-20. National Marine Fisheries Service, Miami, FL, USA. 19 p
Thompson N, Henwood T, Epperly S, Lohoefener R, Gitschlag G, Ogren L, Mysing J, Renaud M (1990) Marine turtle habitat plan. NOAA Technical Memorandum NMFS-SEFSC-255. National Marine Fisheries Service, Miami, FL, USA. March. 20 p
Townsend CH (1899) Statistics of the fisheries of the Gulf States. In Division of Statistics and Methods of the Fisheries. U.S. Government Printing Office, Washington, DC, USA, pp 105–169
TPWD (Texas Parks and Wildlife Department) (2006) Fewer crabs—fewer fish. http://www.tpwd.state.tx.us/fishboat/fish/didyouknow/bluecrabdecline.phtml. Accessed 20 November 2013
TPWD (2012) Homepage. http://www.tpwd.state.tx.us/. Accessed 20 November 2013
Troëng S (1998) Report on the 1998 leatherback program at Tortuguero, Costa Rica. Caribbean Conservation Corporation, Gainesville and Ministry of Environment and Energy of Costa Rica, San Pedro, Costa Rica. 30 p

Troëng S (2000) Report on the 1999 leatherback program at Tortuguero, Costa Rica. Caribbean Conservation Corporation, Gainesville and Ministry of Environment and Energy of Costa Rica, San Pedro, Costa Rica. 29 p

Troëng S, Cook G (2000) Report on the 2000 leatherback program at Tortuguero, Costa Rica. Caribbean Conservation Corporation, Gainesville and Ministry of Environment and Energy of Costa Rica, San Pedro, Costa Rica. 28 p

Troëng S, Rankin E (2005) Long-term conservation efforts contribute to positive green turtle *Chelonia mydas* nesting trend at Tortuguero, Costa Rica. Biol Conserv 121:111–116

Troëng S, Chacon D, Dick B (2004) Possible decline in leatherback turtle *Dermochelys coriacea* nesting along the coast of Caribbean Central America. Oryx 38:395–403

True FW (1887) The turtle and terrapin fisheries. In: Goode GB (ed) The fisheries and fishery industries of the United States. U.S. Government Printing Office, Washington, DC, USA, pp 493–504

Tucker AD (1988) A summary of leatherback turtle *Dermochelys coriacea* nesting at Culebra, Puerto Rico, from 1984–1987 with Management Recommendations. Report. U.S. Fish and Wildlife Service, Vero Beach, FL, USA. 34 p

Tucker AD (2010) Nest site fidelity and clutch frequency of loggerhead turtles are better elucidated by satellite telemetry than by nocturnal tagging efforts: implications for stock estimation. J Exp Mar Biol Ecol 383:48–55

Tuncer B (2009) New born *Caretta* (loggerhead) sea turtle crawling on golden sands. iStockphoto. Calgary, Alberta, Canada. http://www.istockphoto.com/stock-photo-8799774-new-born-caretta-loggerhead-sea-turtle-crawling-on-golden-sands.php. Accessed 20 November 2013

Turnbull BS, Smith CR, Stamper MA (2000) Medical implications of hypothermia in threatened loggerhead (*Caretta caretta*) and endangered Kemp's ridley (*Lepidochelys kempi*) and green (*Chelonia mydas*) sea turtles. Proceedings, Annual Conference-American Association of Zoo Veterinarians, 1998, Omaha, NE, USA, October, pp 31–35

USFWS (U.S. Fish and Wildlife Service) (1991) Endangered and threatened wildlife and plants; 5-year review of listed species. Fed Regist 56:56882–56900

USFWS (2006) Preliminary report on the Mexico/United States of America Population Restoration Project for the Kemp's Ridley Sea Turtle, *Lepidochelys kempii*, on the Coasts of Tamaulipas and Veracruz, Mexico. Report Prepared by Gladys Porter Zoo and CONANP/SEMARNAT. U.S. Fish and Wildlife Service, Washington, DC, USA. 8 p

USFWS (2012) Hawksbill sea turtle (*Eretmochelys imbricata*) fact sheet. http://www.fws.gov/northflorida/SeaTurtles/Turtle%20Factsheets/hawksbill-sea-turtle.htm. Accessed 20 November 2013

USFWS, NMFS (1992) Recovery plan for the Kemp's ridley sea turtle (*Lepidochelys kempii*). U.S. Fish and Wildlife Service, Washington, DC and National Marine Fisheries Service, Silver Spring, MD, USA. 40 p

USFWS, NMFS (1996) Policy regarding the recognition of distinct vertebrate population segments under the endangered species act. U.S. Fish and Wildlife Service, Washington, DC and National Marine Fisheries Service, Silver Spring, MD, USA. Federal Register 61:4722–4725.

USFWS, NMFS (2011) Endangered and threatened species: Determination of nine distinct population segments of loggerhead sea turtles as endangered or threatened. U.S. Fish and Wildlife Service, Washington, DC and National Marine Fisheries Service, Silver Spring, MD, USA. Federal Register 76:58868–58952.

Valverde RA, Wingard S, Gómez F, Tordoir MT, Orrego CM (2010) Field lethal incubation temperature of olive ridley sea turtle *Lepidochelys olivacea* embryos at a mass nesting rookery. Endang Spec Res 12:77–86

Valverde RA, Orrego CM, Tordoir MT, Gómez FM, Solís DS, Hernández RA, Gómez GB, Brenes LS, Baltodano JP, Fonseca LG, Spotila JR (2012) Olive ridley mass nesting ecology and egg harvest at Ostional Beach, Costa Rica. Chel Conserv Biol 11:1–11

van Buskirk J, Crowder LB (1994) Life-history variation in marine turtles. Copeia 1994:66–81

van Houtan KS, Pimm SL (2007) Assessment of the Dry Tortugas National Park Sea Turtle Monitoring Program 1982–2006: Ecological Trends & Conservation Recommendations. NPS Report #H5299-05-1010. National Park Service, Washington, DC, USA. 29 p

Van Vleet ES, Pauly GG (1987) Characterization of oil residues scraped from stranded sea turtles from the Gulf of Mexico. Carib J Sci 23:77–83

Viada ST, Hammer RM, Racca R, Hannay D, Thompson MJ, Balcom BJ, Phillips NW (2008) Review of potential impacts to sea turtles from underwater explosive removal of offshore structures. Environ Impact Assess 28:267–285

Wallace BP, DiMatteo AD, Hurley BJ, Finkbeiner EM, Bolten AB, Chaloupka MY, Hutchinson BJ, Abreu-Grobois FA, Amorocho D, Bjorndal KA, Bourjea J, Bowen BW, Briseño D, Paolo C, Choudhury BC, Costa A, Dutton PH, Fallabrino A, Girard A, Girondot M, Godfrey MH, Hamann M, López-Mendilaharsu M, Marcovaldi MA, Mortimer JA, Musick JA, Nel R, Pilcher NJ, Seminoff JA, Troëng S, Witherington B, Mast RB (2010) Regional management units for marine turtles: A novel framework for prioritizing conservation and research across multiple scales. PLoS One 5, e15465

Watson JW, Foster DG, Epperly S, Shah A (2004) Experiments in the Western Atlantic Northeast distant waters to evaluate sea turtle mitigation measures in the pelagic longline fishery: Report on experiments conducted in 2001–2003. NMFS Technical Report. National Marine Fisheries Service, Silver Spring, MD, USA. 123 p

Waycott M, Longstaff BJ, Mellors J (2005) Seagrass population dynamics and water quality in the Great Barrier Reef region: A review and future research directions. Mar Pollut Bull 51:343350

Webster PJ, Holland GJ, Curry JA, Chang H-R (2005) Changes in tropical cyclone number, duration, and intensity in a warming environment. Science 309:1844–1846

Werler J (1951) Miscellaneous notes on the eggs and young of Texan and Mexican reptiles. Zoologica 36:37–48

Werner SA (1994) Feeding ecology of wild and head started Kemp's ridley sea turtles. MS Thesis, Texas A&M University, College Station, TX, USA. 65 p

Wibbels T (2007) Sex determination and sex ratio in ridley sea turtles. In: Plotkin PT (ed) Biology and conservation of ridley sea turtles. The Johns Hopkins University Press, Baltimore, MD, USA, pp 167–189

Wibbels T, Geis A (2003) Evaluation of hatchling sex ratios of Kemp's ridley *in situ* nests and egg corral nests in the Kemp's ridley recovery program during the 2002 nesting season. Final Report. National Marine Fisheries Service. National Marine Fisheries Service, Silver Spring, MD, USA

Wibbels TR, Owens DW, Limpus CJ, Reed PC, Amoss MS Jr (1990) Seasonal changes in serum gonadal steroids associated with migration, mating, and nesting in the loggerhead sea turtle (*Caretta caretta*). Gen Comp Endocr 79:154–164

Williams KL, Frick MG, Pfaller JB (2006) First report of green, *Chelonia mydas*, and Kemp's ridley, *Lepidochelys kempii*, turtle nesting on Wassaw Island, Georgia, USA. MTN 113:8

Witherington BE (1986) Human and natural causes of marine turtle clutch and hatchling mortality and their relationship to hatchling production on an important Florida nesting beach. MS Thesis, University of Central Florida, Orlando, FL, USA. 141 p

Witherington BE (1992) Behavior responses of nesting sea turtles to artificial lighting. Herpetologica 48:31–39

Witherington BE (1995) Observations of hatchling loggerhead turtles during the first few days of the lost year(s). Proceedings, 12th Annual Workshop on Sea Turtle Biology and Conservation, Jekyll Island, FL, USA, February, pp 154–157

Witherington BE (1997) The problem of photopollution for sea turtles and other nocturnal animals. In: Clemmons JR, Buchholz R (eds) Behavioral approaches to conservation in the wild. Cambridge University Press, Cambridge, UK, pp 303–328

Witherington BE (2002) Ecology of neonate loggerhead turtles inhabiting lines of downwelling near a Gulf Stream front. Mar Biol 140:843–853

Witherington BE, Ehrhart LM (1989a) Status and reproductive characteristics of green turtles (*Chelonia mydas*) nesting in Florida. Proceedings, 2nd Western Atlantic Turtle Symposium. National Marine Fisheries Service, Mayaguez, Puerto Rico, October, pp 351–352

Witherington BE, Ehrhart LM (1989b) Hypothermic stunning and mortality of marine turtles in the Indian River Lagoon System, Florida. Copeia 1989:696–703

Witherington BE, Martin RE (1996) Understanding, assessing, and resolving light-pollution problems on sea turtle nesting beaches. Technical Report TR-2. Florida Marine Research Institute, St. Petersburg, FL, USA. 73 p

Witherington BE, Crady C, Bolen L (1996) A 'Hatchling Orientation Index' for assessing orientation disruption from artificial lighting. Proceedings, 15th Annual Symposium on Sea Turtle Biology and Conservation, Hilton Head, SC, USA, February, pp 344–347

Witherington B, Lucas L, Koeppel C (2005) Nesting sea turtles respond to the effects of ocean inlets. Proceedings, 21st Annual Symposium on Sea Turtle Biology and Conservation. Philadelphia, PA, USA, February, Part II, pp 355–356

Witherington B, Herren R, Bresette M (2006a) *Caretta caretta*—loggerhead sea turtle. In: Meylan PA (ed) Biology and conservation of Florida turtles, vol 3, Chelonian research monographs. Massachusetts Chelonian Research Foundation, Lunenburg, MA, USA, pp 74–89

Witherington B, Bresette M, Herren R (2006b) *Chelonia mydas*—green turtle. In: Meylan PA (ed) Biology and conservation of Florida turtles, vol 3, Chelonian research monographs. Massachusetts Chelonian Research Foundation, Lunenburg, MA, USA, pp 90–104

Witherington BE, Kubilis P, Brost B, Meylan A (2009) Decreasing annual nest counts in a globally important loggerhead sea turtle population. Ecol Appl 19:30–54

Witherington B, Hirama S, Hardy R (2012) Young sea turtles of the pelagic *Sargassum*-dominated drift community: Habitat use, population density, and threats. Mar Ecol Prog Ser 463:1–22

Witt MJ, Hawkes LA, Godfrey MH, Godley BJ, Broderick AC (2010) Predicting the impacts of climate change on a globally distributed species: The case of the loggerhead turtle. J Exp Biol 213:901–911

Witzell WN (1983) Synopsis of biological data on the hawksbill turtle, *Eretmochelys imbricata* (Linnaeus, 1766). FAO Fisheries Synopsis 137. Food and Agriculture Organization of the United Nations, Rome, Italy. 78 p

Witzell WN (1984) The incidental capture of sea turtles in the Atlantic U.S. Fishery Conservation Zone by the Japanese Tuna Longline Fleet, 1978–1981. MTN 46:56–58

Witzell WN (1994) The origin, evolution, and demise of the U.S. sea turtle fisheries. Mar Fish Rev 56:8–23

Witzell WN (1998) Long-term tag returns from juvenile Kemp's ridleys. MTN 79:20

Witzell WN (1999) Distribution and relative abundance of sea turtles caught incidentally by the U.S. pelagic longline fleet in the western North Atlantic Ocean, 1992–1995. Fish Bull 97:200–211

Witzell WN, Azarovitz T (1996) Relative abundance and thermal and geographic distribution of sea turtles off the U.S. Atlantic Coast based on aerial surveys (1963–1969). NOAA

Technical Memorandum NMFS-SEFSC-381. National Marine Fisheries Service, Miami, FL, USA. March. 10 p
Witzell WN, Schmid JR (2004) Immature sea turtles in Gullivan Bay, Ten Thousand Islands, Southwest Florida. Gulf Mex Sci 4:54–61
Witzell WN, Schmid JR (2005) Diet of immature Kemp's ridley turtles (*Lepidochelys kempi*) from Gullivan Bay, Ten Thousand Islands, Southwest Florida. Bull Mar Sci 77:191–199
Wyneken J, Salmon M (1992) Frenzy and postfrenzy swimming activity in loggerhead, green, and leatherback hatchling sea turtles. Copeia 1992:478–484
Wyneken J, Madrak SV, Salmon M, Foote J (2008) Migratory activity by hatchling loggerhead sea turtles (*Caretta caretta* L.): Evidence for divergence between nesting groups. Mar Biol 156:171–178
Xavier R, Barata A, Cortez LP, Queiroz N, Cuevas E (2006) Hawksbill turtle (*Eretmochelys imbricata* Linnaeus 1766) and green turtle (*Chelonia mydas* Linnaeus 1754) nesting activity (2002–2004) at El Cuyo beach, Mexico. Amphibia-Reptilia 27:539–547
Yender RA, Mearns AJ (2003) Case studies of spills that threaten sea turtles. Oil and sea turtles: Biology, planning, and response. Office of Response and Restoration, National Oceanic and Atmospheric Administration, Seattle, WA, USA, pp 69–86
Yerger RW (1965) The leatherback turtle on the Gulf coast of Florida. Copeia 1965:365–366
Yeung C (1999) Estimates of marine mammal and marine turtle bycatch by the U.S. Pelagic Longline Fleet in 1998. NOAA Technical Memorandum NMFS-SEFSC-430. National Marine Fisheries Service, Miami, FL, USA. November. 26 p
Yeung C (2001) Estimates of marine mammal and marine turtle bycatch by the U.S. Atlantic Pelagic Longline Fleet in 1999–2000. NOAA Technical Memorandum NMFS-SEFSC-467. National Marine Fisheries Service, Miami, FL, USA. August. 43 p
Yntema CL, Mrosovsky N (1982) Critical periods and pivotal temperatures for sexual differentiation in loggerhead sea turtles. Can J Zool 60:1012–1016
Zimmerman R (ed) (1998) Characteristics and causes of Texas marine strandings. NOAA Technical Report NMFS-143. National Marine Fisheries Service, Galveston, TX, USA. 85 p
Zug GR, Glor RE (1998) Estimates of age and grown in a population of green sea turtles (*Chelonia mydas*) from the Indian River Lagoon system, Florida: A skeletochronological analysis. Can J Zool 76:1497–1506
Zug GR, Balazs GH, Wetherall JA, Parker DM, Murakawa SKK (2002) Age and growth of Hawaiian green sea turtles (*Chelonia mydas*): An analysis based on skeletochronology. Fish Bull 100:117–127