

DIRECT AND INDIRECT EFFECTS OF URBANIZATION ON DIAMOND-BACKED TERRAPINS OF THE HUDSON RIVER BIGHT: DISTRIBUTION AND PREDATION IN A HUMAN-MODIFIED ESTUARY

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Abstract — Populations of Diamond-backed Terrapins (*Malaclemys terrapin*) are declining throughout their range as a result of habitat degradation, urbanization, incidental trapping, and predation. Large populations of Terrapins still live in the Hudson River Bight (HRB), a large estuary mostly within the political borders of New York City, the second largest city in the world. Two centuries of intensive urbanization, pollution, and dredging have resulted in dramatic changes to HRB shorelines and marshes. Proximity to the city has also increased Terrapin harvest rates in the past and urbanization currently subsidizes predators.

Gateway National Recreation Area (GNRA) is a relatively new U.S. National Park within HRB that protects essential Terrapin habitat. We studied distribution of and predation rates on Terrapin nests at the three management units of the GNRA: heavily urbanized Staten Island Unit (SIU), less urbanized Jamaica Bay Unit (JBU), and least urbanized Sandy Hook Unit (SHU). The Terrapin population at SIU appears to be extirpated. JBU Terrapins nest on the mainland coast and on three islands. Approximately 20 nests are deposited annually on two of these islands, predation is low, and about 300 hatchlings are produced. Approximately 2000 Terrapin nests are laid annually on the largest JBU island (Ruler's Bar Hassock), but 95% of these are predated by Raccoons (*Procyon lotor*). Over 92% of nests were predated at SHU despite relatively low levels of human disturbance. We conclude that human-induced habitat changes affect Terrapin nesting success even at low levels of urbanization because these changes result in high levels of subsidized predators such as Raccoons, Norway rats (*Rattus norvegicus*), and gulls (*Larus atricilla* and *L. argentatus*).

Key words — Diamond-backed Terrapin, GIS, Habitat, Malaclemys, Reproduction, Urbanization

Diamond-backed Terrapins (*Malaclemys terrapin*) (Terrapins hereafter) are medium-sized, brackish water North American turtles whose range is a narrow strip approximately 5764 km long along the Atlantic coast from Cape Cod to the Gulf Coast of Texas. Terrapins are the only American turtles that regularly inhabit tidal creeks, bays, estuaries, and salt marshes, where the salinity ranges from fresh water to almost full strength sea water. Their diet consists mainly of invertebrates such as crabs, snails, shrimps, and clams (Ernst et al. 1994). Terrapins may be important components of estuarine food webs (Hurd et al. 1979), but more research is needed to clarify their precise role in estuarine ecosystems.

Other than for occasional terrestrial basking, Terrapins only come on land to nest. During their late May to early August nesting season, female Terrapins cross the intertidal zone to oviposit in nearby upland areas (Ernst et al. 1994). In much of their range, Terrapins are dependent on tidal salt marshes, dominated by *Spartina spp.*, within which Terrapins feed and thermoregulate (Brennessel 2006).

It is difficult to quantify the effects that urbanization has had on Terrapins, mostly because few data are available on Terrapin population levels prior to urbanization. However, we speculate that Terrapin populations were probably enormous,

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as they commonly shared habitat with oysters, and only 300 yrs ago oysters formed vast reefs all along the shallow coasts of eastern United States (Jackson 2001; Jackson et al. 2001). Terrapins were so abundant in the 1700s that they were provided as a monotonously common food for slaves (Carr 1952) and regularly clogged the nets of fishermen (Coker 1920). Terrapins were still numerous into the mid-1800s, when, due to a change in culinary fashions, large-scale Terrapin harvests began (Hay 1904; McCauley 1945; Carr 1952). Heavy harvest of Terrapins continued into the early 1900s; by then many populations had been reduced to low levels (Babcock 1926; De Sola 1931; Carr 1952; Conant 1952) and imminent extinction was feared (Hay 1904). Although no firm data are available, harvest range-wide has dropped considerably since the early 1900s and many Terrapin populations have managed a partial comeback (Carr 1952; Klemens 1993; Burke et al. 2000).

In addition to harvest, Terrapins have been directly impacted by other human activities (Roosenburg 1991; Burger and Garber 1995). Currently, Terrapins face threats from habitat degradation, pollution, road and boat traffic (Wood and Herlands 1997), drowning in crab pots (Roosenburg et al. 1997; Wood 1997, see also Hoyle 2008 this volume), and predation (Feinberg and Burke 2003). Many sections of the Terrapin range are heavily urbanized and thus subject to massive amounts of pollution of many types, from multiple sources, both large and small (O'Connor and Huggett 1988; Daskalakis and O'Connor 1995). The effects of pollutants on Terrapins are poorly studied, but are likely to be severe (e.g., Bishop et al. 1991, 1998).

Two indirect ways that humans and urbanization affect Terrapins are through subsidized predators (Feinberg and Burke 2003) and destruction of salt marshes. Although many species are known to prey on Terrapins and their eggs, Raccoons (*Procyon lotor*) are by far the most important overall. Raccoon predation on nests is common in many North American turtle populations (Mitchell and Klemens 2000), but this may be a recent phenomenon. United States Raccoon populations generally grew 15–20 times larger in the 20th century, and spread into many new areas of North America (Obbard et al. 1987; Sanderson 1987). The reasons for these increases are not well understood, but urbanization is a contributing factor (Prange and Gehrt 2004). Raccoons adapt well to urbanization and urban areas have some of the highest Raccoon densities known (Riley et al. 1998; Larivière 2004). Raccoon predation is a potentially limiting factor for Terrapin populations not only because Raccoons eat turtle eggs, but because Raccoons also eat adults (Seigel 1980; Feinberg and Burke 2003). Predation on adults can have severe impacts on turtle populations (Congdon et al. 1993). Terrapin populations can be eliminated by Raccoon predation on reproductive females (e.g., Seigel 1980).

Besides subsidizing predators, urbanization has also impacted Terrapins indirectly through damage to salt marshes. Many of the world's largest cities are located near estuaries, because such sites provide commercially and militarily valuable link-

ages between ocean travel and inland waterways. As a result, over one-third of the population of the United States lives close to salt marshes (McLusky and Elliot 2004). Beginning in the 1700s, impacts of such urbanization were primarily through increased sediments and sewage loads and through intensive resource use, but with increased industrialization pollutants changed rapidly to more complex chemicals and increased movement of surface material (Odum et al. 1984; Hanson and Lindh 1993; Ehrenfeld 2001). Wide-scale diking, dredging, and filling of urban salt marshes became common around cities beginning in the mid-1800s with the advent of heavy machinery, often associated with the maintenance of shipping channels and development of industrial sites. Dredging has continued; development and fill deposition accounted for 73% of estuarine losses in the United States from 1986 to 1997 (Dahl 2000). Four of the 5 states with the highest levels of estuarine wetland losses are found within the range of the Terrapin: Florida, Louisiana, New Jersey, and Texas (Tiner 1984), and the coastlines of these four states together comprise 67% of Terrapin range. Three of the five largest cities in the United States, New York City, Houston, and Philadelphia, are located on estuaries within Terrapin range. Clearly Terrapins and their habitat have been strongly impacted by urbanization.

We investigated the distribution of Terrapin nesting activity and associated nest success along a gradient of human disturbance within the boundaries of Gateway National Recreation Area, a U.S. National Park spanning the Hudson River Bight in New York and New Jersey. Urbanization has had an important effect on Terrapins in this region. Prior to initial urbanization, shallow reefs spread across much of New York Harbor (Waldman 2000), and the *Spartina* marshes of New York Harbor and neighboring Long Island probably supported large Terrapin populations. Terrapins of the New York/New Jersey area were heavily affected by the trade in Terrapin meat because of their proximity to major food markets in New York City and Philadelphia, and their reputation as high quality meat (Murphy 1916; Carr 1952). Subsequent development of New York City led to the loss of most regional *Spartina* marshes (Waldman 2000).

Our objective was to investigate the past and current impacts that urbanization has had on Terrapin populations in the Hudson River Bight. We studied nest characteristics because they are relatively easily surveyed and may serve as good proxies for recruitment rates. Because we expected that human impact would be seen indirectly through presence and abundance of nest predators, we also censused nest predators at these sites.

MATERIALS AND METHODS

Study Sites — Gateway National Recreation Area (GNRA) consists of uplands, salt water marshes, and ocean, mostly within the political borders of New York City (Fig. 1). Established in 1972 by the National Park Service (NPS) as America's first urban national park, GNRA is responsible for providing rec-

reational opportunities for 8.5 million visitors annually (NPS, unpubl. data) and for supplying habitat for wildlife.

GNRA spans the mouth of the Hudson River, an area known as the Hudson River Bight (HRB). The HRB has a 250-yr history of development and pollution, including a complex mix of organic and inorganic toxins (Waldman 2000). The lower Hudson River currently absorbs treated sewage from most of New York City's 14 wastewater treatment plants, which emit 1–2 billion gallons of treated sewage per day. The discharge plume is mostly inorganic nitrogen, but also carries many heavy metals and chemicals, mostly in trace amounts. Other active sources of pollution include industrial discharge and leaks from landfills. While levels of many of these pollutants, such as mercury, PCBs, and DDTs, have dropped in recent decades (Steinberg et al. 2004), dioxin levels in Hudson River Bight sediments have risen steadily since measurements were begun in the 1960s. Thus, the waters of GNRA have a complex and changing history of pollution.

GNRA is divided into three management units (Fig. 1). The most urbanized (Staten Island Unit, SIU), is located in the New York City borough of Staten Island. The SIU consists of several properties, including Miller Field, Great Kills Park (GKP), and Hoffman and Swinburne Islands (Figs. 1, 2). The lack of *Spartina* marshes at Miller Field and Hoffman and Swinburne Islands probably precludes Terrapins there, but small *Spartina* marshes still exist within GKP. GKP was dramatically altered by dredging and land fill activity, especially in the 1940s (Barlow 1971; Tanacredi and Badger 1995; Wrenn 1975). As a result, nearly all of the *Spartina* marsh that once existed in SIU has been eliminated.

Jamaica Bay Unit (JBU) is located in the boroughs of Brooklyn and Queens and includes mainland coast, parts of a barrier island, and seven upland islands ("upland" here means habitat that is not flooded by high tides, and therefore utilizable by Terrapins for nesting): Ruler's Bar Hassock (RBH) the largest island, Canarsie Pol (CP), Ruffle Bar (RB), Little Egg Marsh (LEM), Subway Island (SI), Pumpkin Patch (PP), and Elder's Point (EP) (Figs. 1, 3). Jamaica Bay was relatively unaffected by urbanization until the mid-1800s (Black 1981) and was known for abundant fish, oysters, and clams that supported a substantial fishery throughout the 1800s. In 1904, household sewage and industrial pollution from rapidly expanding New York City resulted in serious human illness in people who ate shellfish from Jamaica Bay, closing the shellfish industry. The *Spartina* marshes, islands, and ocean floor of Jamaica Bay were greatly altered by dredging operations in the early 1900s to provide greater ocean access to urban areas of New York City. Most coastal marshes in Jamaica Bay were covered with dredge material, and the freshwater streams into the bay were largely eliminated (Black 1981). Canarsie Pol was created in the 1930s by the dumping of ocean dredge material onto a tiny marsh. Ocean gaps that separated most of the original upland islands were filled, so these islands became connected to the mainland, providing direct access for potential terrestrial nest predators. A roadway bridge was built in 1923 that connected

the mainland to RBH, similarly providing a potential corridor for dispersing wildlife. Two large freshwater ponds were excavated on RBH in the 1950s, providing the only permanent fresh water source for wildlife on any of the Jamaica Bay islands. Several smaller islands were eliminated and dredge material was dumped on a central group of marshes, creating one big upland island (RBH) where none had existed before. The only upland island existing today that existed prior to this period of dredging is RB (Black 1981). Loss of Terrapin habitat in JBU continues because the *Spartina* marshes are currently deteriorating, due to combined effects of the changes in sediment deposition patterns and recent ocean level increases (NPS 2001; Gornitz et al. 2002; Hartig et al. 2002).

Sandy Hook Unit (SHU) is a barrier beach peninsula located at the northern tip of the New Jersey shore (Figs. 1, 4). The west side of the peninsula faces the Hudson River Bight and includes significant *Spartina* marshes. SHU has undergone intense military use from the early 1800s until 1972, which resulted in significant upland alterations (Tanacredi and Badger 1995). The military has made dramatic efforts to protect the peninsula from erosion by using a series of sand groins and beach refurbishment. Although approximately 2.3 million people visit SHU annually, the SHU coastline is mostly undeveloped.

Measuring Wetlands Loss at the Three GNRA Management Units — As a measure of the effects of urbanization of Terrapin habitat, we measured the loss of estuarine wetlands in the three GNRA management units since the major urbanization of New York City. We digitized the earliest maps that we could find that contained relevant information for each of the GNRA management units to measure the area of estuarine habitat before large-scale modification occurred (before the mid-1800s, see above). We assumed that all coastal habitat labeled as wetlands or marsh was *Spartina* marsh. We compared the amount of *Spartina* in these maps to surveys done in the mid-1970s, which is after major modification and after the area was protected as a national park (NPS 1979).

Identifying Potential Nesting Areas — We located potential Terrapin nesting areas in two ways: using previous reports (Cook 1989; Feinberg 2000) and analyzing GIS data. The latter was accomplished for SIU and JBU using 1994 aerial imagery obtained from NPS, including 1976 vegetation maps. We used the ArcView GIS program to find possible Terrapin nesting grounds with suitable habitat. Terrapins nest in a variety of upland habitats (Roosenburg 1994); we defined suitable nesting habitat for Terrapins as "bare sand," "beach," "beachgrass dune," "mixed grassland," "open shrubland," "sand flat," "sand dunes," "sparsely vegetated dune areas," "low thicket," "lawn," and "flat sand areas with low vegetation cover," as defined on the vegetation maps.

We rectified a time series of historical, 2001 aerial photographs, and 1996 vegetation maps from Monmouth County, New Jersey provided by NPS to identify possible Terrapin

nesting areas in the SHU. As described above, we categorized sites into habitat types based on the same vegetation categories. We also made a preliminary visit to SHU and located nesting areas based on where we found evidence of predated nests in 2001.

Surveys of Potential Nesting Areas — We trained a NPS employee working at SIU to survey (see protocol below) potential Terrapin nesting areas daily in May, June, and July 2000. With the assistance of volunteers, we surveyed all suitable island nesting areas at JBU (CP, EP, LEM, PP, RB, and SI) for any sign of nesting Terrapins at least twice per week in June, July, and August 2000. We recorded observations of available habitat and compared it with the vegetation maps and aerial photoimagery.

In 2001, we focused on the areas where we found evidence of nesting in 2000. We spent little time on RBH because it was already well studied (Feinberg 2000); instead we concentrated our work on LEM and RB because nesting had been reported there previously (Feinberg and Burke 2003) and we found eggshells there in 2000. Volunteers were posted on LEM and RB each day of the nesting season (June–July 2001) to watch for nesting activity. When a nesting female was detected, we let her nest, recorded nest location and captured the Terrapin to obtain measurements and uniquely mark her. Upon first capture each female was marked by notching marginal scutes and injecting a passive integrated transponder (PIT). Nest locations were marked by orange polyvinyl flags and monitored until predation or hatching. Such marking probably does not affect predation rates (Burke et al. 2005), and we did not protect JBU nests from predators.

The remaining islands (CP, EP, and PP) were visited once each week June–July 2001 for turtle tracks or predated nests. SI was visited biweekly. In early August, after the nesting season, CP, EP, LEM, PP, and RB were visited twice weekly to observe either evidence of predation or signs of hatchling emergence. Hatchling signs include post-emergence holes, which are small openings in the ground from which hatchlings have emerged. From 9 June to 26 October 2001, we spent 521 total person hrs on LEM, RB, PP, EP and CP waiting for nesting females, looking for nests, and searching for evidence of predation and signs of hatchling emergence on possible Terrapin nesting beaches.

We visited SHU briefly in 2001 and performed an intensive study there in 2002. In 2001, we searched all feasible areas for evidence of nesting and performed counts of predated nests. In 2002, we visited nesting areas daily and monitored for nesting females. Each nesting female and nest was treated as described above for LEM and RB, except we also protected 19 nests to determine clutch sizes and measure egg viability. From 1 June to 1 August 2002, we spent 518 total person hrs looking for Terrapin nests and nesters at SHU. As described above for LEM and RB, we surveyed all feasible nesting areas for predation or signs of hatchling emergence during August and September. In some cases post-emergence holes allowed us to find nests that we had missed during the nesting period.

We excavated these to count eggs, eggshells, and remaining hatchlings. We spent 164 person hrs at SHU searching for post-emergence holes.

Measuring Mortality of Nests and Adults — We surveyed SIU in 2000 for predated nests, dead Terrapins, and nesting females, as described above. Similarly, we searched for predated nests and dead Terrapins on CP, EP, LEM, RB, PP and SI, from 1 June to 31 July and sporadically in August 2000. In 2001, volunteers who were stationed on LEM, RB and CP to observe nesting also monitored unprotected nests and searched for predated nests. We searched SHU for predated nests and dead Terrapins briefly in 2001 and more intensively 1 June to 30 October 2002.

During all of these surveys we only counted a nest as predated if the excavated hole had eggshells associated with it, so as not to inflate our count with other holes that were not nests. We identified nest predators by species, where possible, based on tracks, nest scars or scat. Predation rates were calculated only using nests for which we observed oviposition, and that later appeared to be predated, using the field signs described above.

Measuring Egg Viability — We defined “egg viability” as the number of eggs that resulted in emergent hatchlings, excluding eggs that were predated, flooded, died while hatching, or eaten by fly larvae. We determined this by excavating nests either upon hatchling emergence or in the late fall. We counted the number of unhatched eggs, eggs infested with fly larvae, egg shells (indicating successful hatching), and dead hatchlings. We assumed that eggs were killed by flooding when nests were below the tide line and no eggs hatched. To improve the probability that some nests at SHU would survive predation and could be used to measure egg viability, we protected 19 nests with hardware cloth predator excluders (Feinberg and Burke 2003).

Predator Surveys — In addition to observations of mammals made during all visits to Terrapin nesting areas, we surveyed LEM and RB more intensively for potential predators from 3 to 17 September 2001 using Tomahawk live-traps (80 X 30 X 30 cm), baited with marshmallows, donuts, and cat food to capture mammals such as rats, Raccoons, and Opossums. We placed 3 Tomahawk traps on LEM (18.3 ha) and 5 Tomahawk traps on RB (56.8 ha). We also used Sherman traps baited with peanut butter and a seed mixture to live-trap smaller mammals, such as mice, meadow voles, and shrews. We set 40 Sherman traps on LEM and 20 traps on RB for the same time periods as the Tomahawk traps. We checked and re-baited traps daily.

RESULTS

Measuring Wetlands Loss at the Three GNRA Management Units — The best pre-major-modification estuarine marsh coverage data showed that 84% of the marsh was within what is now JBU, 16% was within what is now SIU, and 2% was within

what is now SHU (Table 1). By the time of the creation of GNRA in the 1972, JBU contained 92% of GNRA's total estuarine marsh, SHU contained 6.5%, and SIU contained 1.5%. The SIU had experienced nearly complete loss of its original salt marsh, while JBU lost more than half of its original salt marsh, and SHU has gained salt marsh.

Identification of Potential Nesting Areas — All of the study areas appeared to have at least some suitable nesting habitat (Table 2, Figs. 2, 3, 4) based on GIS analysis of the vegetation maps and surveys. The SIU and SHU had about the same amount of potential nesting habitat as RBH. Most islands in JBU had little potential nesting habitat, but CP and RBH had large amounts of potential nesting habitat, together comprising 65% of the potential nesting habitat on JBU islands.

Surveys of Potential Nesting Area — Despite the availability of nesting areas, we found no any evidence of Terrapins in SIU, at either GK or MF. However, in summer 2002 one Terrapin hatchling was found in a parking lot in GK (R. Cook, pers. comm.).

Results were more varied in the JBU. We observed 43 Terrapin nests (29 in 2000, 14 in 2001) on LEM and RB; both are fairly small islands (Table 2). Despite having more potential nesting habitat than any other island in JBU, we found little evidence of nesting Terrapins on CP; a single Terrapin crawl trail led from the beach into a nesting area in 2000 and 2001, although no nests were found. We found no evidence of nesting on the tiny islands of PP or EP, though we observed nine turtles swimming near EP. Nesting density was highest at RBH, where nearly 6% of available nesting habitat was used for nesting (Table 2).

We found 203 nests at SHU in 2002; 49 of these were found during oviposition, 140 after predation, and 14 using post-emergence hatchling signs. Nesting density was nearly as high as at RBH: 4% of available nesting habitat was used for nesting (Table 2).

Mortality of Nests and Adults — Because we were unable to find nests at SIU, we report no further data from that site. Nest mortality rates on LEM and RB were low at JBU during 2000 and 2001. Predation accounted for loss of 33% of the nests, whereas flooding accounted for the loss of another 8% of nests (Table 3). All predation events were discovered post facto; we did not directly observe any predation on these islands on either Terrapins or their nests. All but one of the nests appeared to have been predated by rats and/or birds; eggshells were scattered 3–4 m away, and rat tracks were found around nests. The exception to this pattern was a RB nest in 2000, which appeared to have been Raccoon-predated (see Aresco 1996 for description of field sign). Predated and flooded nests were always completely destroyed. Other sources of mortality accounted for partial loss of nests (Table 3).

Almost all predation at SHU was by Raccoons. The 30 nests discovered during oviposition and left unprotected were all

predated. Fourteen emergence holes were discovered in SHU, leading to nests that had survived Raccoon predation and successfully produced hatchlings. Raccoons predated eight of the 19 SHU nests that had been protected with predator excluders; predation occurred after the protection had been removed, 63 days post-oviposition.

We found a Terrapin carcass, unidentified sex, on the RB beach in 2001; cause of death could not be determined. We found carcasses of 7 nesting Terrapins killed by Raccoons at SHU in 2002.

Egg Viability — Overall egg viability on LEM and RB in 2000 and 2001 combined was 95% (322/340 eggs). At SHU, egg viability from protected nests was 73.3% (85/116 eggs, 11 nests). This low rate was partially due to one Terrapin that laid two clutches ($n = 15, 16$ eggs) that had 0% survivorship. If nests from this female are excluded from the egg viability calculations, then viability at SHU was 97.6% (83/85).

Surveys for Potential Terrapin Predators — Three Meadow Voles (*Microtus pennsylvanicus*) and 8 Norway Rats (*Rattus norvegicus*) were captured on LEM and RB in 14 trap-nights. Voles were captured only on RB, at an overall rate of three captures per 980 trap-nights (0.003 captures/trap night) using Sherman traps. Rats were captured on both LEM and RB, at an overall rate of eight per 112 trap-nights (0.071 captures/trap night) using Tomahawk traps. No Raccoons were captured at either island, perhaps because human visitors to the islands tampered with the Tomahawk traps on LEM and RB.

We did not survey for predators at SIU because of the absence of Terrapin nesting. Large nesting colonies of Common Terns (*Sterna hirundo*) and Herring Gulls (*Larus argentatus*) were found on LEM and CP. RB had a large breeding colony of Herring Gulls on the side of the island opposite from where the Terrapins nested.

Although Raccoons and Muskrats (*Ondatra zibethicus*) were never captured in traps, their tracks were observed on both LEM and RB, and in 2001 we found a Raccoon skull on RB. We also found one Terrapin nest on RB that appeared to have been predated by a Raccoon (see above). Otherwise, a lack of tracks throughout the area and bite patterns in the eggshells suggested that Raccoons were not the primary predators on LEM and RB.

Raccoons at SHU were observed at dusk searching the Terrapin nesting areas. We did not directly observe predation, but evidence of Raccoon predation on nests and adults was commonly observed. We observed foxes and their tracks in the area, but none of the nests showed evidence of fox predation.

DISCUSSION

Terrapins are restricted to a narrow strip of habitat along the heavily urbanized east coast of the United States. We examined some impacts of urbanization at GNRA, a new national park near one of the world's largest cities. We are

unable to estimate past Terrapin populations in this area because no appropriate records are available. Terrapins were too unimportant as a commercial food source to be recorded along with the accounts of the vast oyster beds and fisheries of this area through the 1800s. However, we found that the salt marshes upon which Terrapins depend for food and shelter were once far more extensive than they are currently and that marsh loss can be primarily attributed to urbanization. Nearly all of the salt marshes of SIU were covered with landfill during the urbanization of Staten Island in the early 1900s. Many of the salt marshes of JBU were either dredged or covered with dredge material, partly to develop a major airport and a sea port. Only the salt marshes of SHU, those farthest from New York City, remain largely intact.

Substantial Terrapin nesting habitat remains and foraging habitat is greatly reduced in GNRA. Some parts of SIU may have more dune habitat now than in the past, as marshes were converted to uplands. Although much of the Jamaica Bay shoreline has been made inaccessible to nesting Terrapins through bulkheading, new nesting areas were created when dredged material was dumped high enough to make upland islands. For example, CP, a large human-made island in Jamaica Bay, now provides substantial potential nesting habitat. It is, however, rarely used by nesting Terrapins and it may take still more time for Terrapins to colonize new nesting beaches. The availability of nesting habitat in the upland areas of SHU appears to be much the same as it has been for the last few centuries.

In addition to habitat degradation, urbanization affects Terrapins by supporting populations of nest predators. In most of North America, Raccoons are the most important predators on turtle nests and generally benefit from all levels of urbanization (Riley et al. 1998; Mitchell and Klemens 2000; Larivière 2004). Congdon et al. (1993) attributed a decrease in Blanding's turtle (*Emydoidea blandingii*) nest survivorship over a 15 yr period as being the result of increased Raccoon and fox populations and blamed decreased fur trapping (also see Roosenburg 1992). However, urbanization may have been a contributing factor. Raccoons occur in higher densities and have smaller home ranges in urban and suburban than in rural areas, primarily because of the availability of stable and spatially fixed food sources (Prange et al. 2004). It is not surprising that we detected a dense Raccoon population even at SHU, the least urbanized of our study sites. SHU is heavily used throughout the year by beach visitors that supply ample refuse as a food resource.

The major source of egg mortality throughout GNRA was predation by Raccoons, and not surprisingly, we found a close relationship between the presence of nest predators, primarily Raccoons, and levels of nest predation. Egg predation levels were relatively low on the two Jamaica Bay islands where Raccoons occurred only sporadically and in low numbers. In the 1980s, before Raccoons occurred on the island of Ruler's Bar Hassock, egg predation was very low there as well (Cook 1989). Raccoon populations on RBH increased with urban-

ization of the island; a large Raccoon population occurs there now. RBH predation rates now match those of SHU—consistently above 90% annually over five yrs of study (Feinberg and Burke 2003; R. Burke unpubl. data). These findings are consistent with those of Christiansen and Gallaway (1984) and Garmestani and Percival (2005), who found high turtle nest predation rates when raccoons were common on their study sites, and much lower predation rates when raccoon numbers were reduced.

Other researchers have found generally lower predation rates on Terrapin nests in less urbanized areas. Burger (1977) reported terrapin nest predation rates of 51% to 71% in 1973 and 1974, respectively, on a barrier island in Brigantine National Wildlife Refuge, New Jersey. Roosenburg (1992, pers. comm.) and Roosenburg and Place (1994) found Terrapin nest predation rates of 30%–94% over 5 yrs on neighboring beaches (averages were 84% and 41%) in a rural/suburban site on the Patuxent River in Maryland. Butler et al. (2004, J. Butler pers. comm.) found nest predation rates of 34% in 1997 and 51% in 2000, primarily by Raccoons, in rural Talbot Island State Park in Florida. In all of these studies, multiple predator species were important, whereas at RBH and SHU, only Raccoons are important nest predators.

The possible effect of other egg and nest predators is unclear, although two other subsidized predators, Herring Gulls and Norway Rats, are also common in GNRA. We found Norway Rats on the Jamaica Bay islands LEM and RB, and they also occur on RBH (R. Burke pers. obs.). We found no unambiguous evidence that birds or rats predated Terrapin eggs or hatchlings, and we did not observe predation on hatchlings. Draud et al. (2004) found that Norway Rats can be important predators on hatchling Terrapins, and rats may be similarly important on Jamaica Bay islands.

We did not measure pollution or pollution effects directly, but we did measure egg viability. We assumed that if pollutants, such as the organochlorines and heavy metals known to occur in GNRA, were having a dramatic effect on the Terrapins, this effect would be evident in reproductive failure, as has been seen elsewhere (Guillette et al. 1994, 1999). Instead, we found relatively high levels of egg viability at JBU (as did Giambanco 2003) and SHU. However, pollutants could express their effects in more subtle ways. For example, we have noted high levels of scute abnormalities in many hatchlings from all three units (S.E. Ner and A.L. Widrig, unpubl. data).

Because of the near complete loss of salt marshes in SIU, we conclude that the Terrapin population there is effectively extirpated, despite the fact that not long ago Terrapins were common on Staten Island (Kieran 1959). Terrapins are not likely to be found in any significant numbers in SIU again until salt marshes are restored. Terrapin numbers at JBU appear to be very high and numerically stable currently (R.L. Burke, unpubl. data), but it is unclear whether this will continue under the impacts of new, high levels of egg predation on RBH (Feinberg and Burke 2003) and current high levels of marsh loss (Gornitz et al. 2002; Hartig et al. 2002).

Because of lower egg predation rates on the smaller islands of JBU, they may be an important recruitment source of Jamaica Bay's Terrapin population, whereas RBH now may be a sink population. Despite much lower levels of human disturbance at SHU, predation rates there were similar to those in RBH. It is unknown whether Raccoon densities at SHU have changed in recent years.

The relationships between urbanization and factors affecting Terrapin populations are complex. Factors that may have been important in the past, such as Terrapin harvest and dredging, may still be influencing the Terrapin habitat and Terrapin numbers today, both directly and indirectly. Relatively modern environmental impacts, such as air pollution, affect the amount of Terrapin habitat, because one effect of global climate change is increasing ocean levels, which are reducing the size of JBU salt marshes (Gornitz et al. 2002) and likely other marshes throughout Terrapin range (Kennish 2001). In addition, even relatively subtle human-induced upland habitat changes, such as light urbanization at SHU, can affect Terrapin nesting success because these changes result in high levels of subsidized predators (e.g., Raccoons and Norway Rats).

Because of their dependence on coastal wetlands, the primary impacts of urbanization on Terrapins throughout much of their range have been through conversion of their habitat to other uses. Current and future impacts of urbanization, such as subsidized predators, water and air pollution, are more indirect. As increasing numbers of people move to coastal cities (Crossett et al. 2004), the impacts of urbanization on Diamond-backed Terrapins are likely to become even more severe.

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