



**Life history of a native emydid turtle
(*Malaclemys terrapin centrata*) on the remote
oceanic islands of Bermuda**

Submitted by

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This thesis is presented to the University College Cork, in
candidature for the degree of Doctor of Philosophy.

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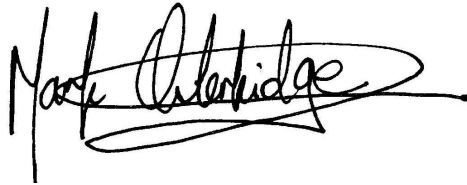
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December 2014

Declaration

I declare that this thesis represents work carried out by myself, except where note is made to the contrary, and has not been submitted for another degree, either at University College Cork or elsewhere.

A handwritten signature in black ink, reading "Mark Outerbridge". The signature is stylized with a large, sweeping loop at the end of the last name.

Mark E. Outerbridge

December 17, 2014

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Acknowledgements

I would first like to thank my supervisors Professor John Davenport, Dr Ruth Ramsay and Dr Robbie Smith for their valuable advice and their critical reviews of this dissertation. Particular thanks are due to numerous individuals for their invaluable assistance with field work, which was often at times very demanding. These people include Simieon Massey for her help with the mark-recapture and nesting surveys over the course of many seasons, Ronald Burchall who checked and re-baited the terrapin traps in 2010, Patrick Harris and Micah Hoder who performed the benthic sediment surveys in Mangrove Lake and South Pond and Erin Limerick who assisted with the benthic mangrove surveys during the summer of 2011. Don Lewis gave practical advice on record keeping as well as handling and marking wild caught terrapins, Dr Matthew Draud provided technical assistance with the hatchling radio-telemetry, Dr Douglas Fort helped with the interpretation of the eco-toxicological data and Dr Tom Quirke provided much valued assistance on data analysis. The Bermuda Aquarium, Museum and Zoo and Dolphin Quest generously donated large quantities of herring over the years for use in the terrapin traps.

I am grateful to the Mid Ocean club board of directors as well as the residents who live adjacent to Mangrove Lake, for generously allowing me to access the study site. Many of the Mid Ocean golf course maintenance staff were helpful in reporting terrapin movements and nesting activity, including Justin Lindo, Kemo Bean, Joe Cabral, Bobbie McDonald and Norman Furtado. I also wish to express my gratitude to all of the anonymous golfers who alerted me to terrapin nests by marking their locations in the sand bunkers.

I am deeply indebted to all of the individuals and organisations who generously donated money. Funding for this project was largely provided by the Bermuda Government's Department of Conservation Services and the Atlantic Conservation Partnership. Additional support was provided by the Robert Steinhoff Scholarship for Environmental Sciences, the Bermuda Zoological Society, and many members of the Mid Ocean Club.

Furthermore, I would like to thank Evan Outerbridge and Simieon Massey for their encouragement, understanding and unwavering support throughout the many years of these investigations. And finally I wish to express my gratitude to the diamondback terrapins who endured my frequent presence and intrusion as they went about their seasonal activities. It is to them that this work is dedicated.

Abstract

Diamondback terrapins (*Malaclemys terrapin*) are native to the remote oceanic islands of Bermuda and presently inhabit only four small brackish water ponds on a private golf course. The life history of this species is poorly understood on Bermuda and so the aim of this study was to fill these knowledge gaps, to compare the results with what is known from other areas in the North American range, and to inform the development of a local management plan.

The results of a mark-recapture census revealed that *ca.* 100 individuals ≥ 81 mm straight carapace length live on Bermuda, of which nearly half (48.5%) were considered sexually mature. The population is dominated by females (sex ratio 2.9:1) and annual recruitment over the three year period was found to be extremely low (approximately two terrapins).

Female diamondback terrapins in Bermuda nest almost exclusively within a limited number of sand bunkers on the golf course. Nesting commenced in late March or early April and ended in late August. Peak oviposition was observed in May and June. Clutch size averaged 5.1 eggs (range 0-10; SD 2.4) and the incubation period averaged 61.8 days (range 49-83; SD 10.5). Delayed emergence was documented, with 43.8% of the hatchlings remaining in their natal nests over the winter months. The mean annual hatching success rate was determined to be 19% (range 17.6-21; SD 1.9).

Radio-telemetry was used to investigate the movements and survivorship of post-emergent hatchling diamondback terrapins. The results indicated that mangrove swamps and grass-dominated marshes adjacent to the ponds are important developmental habitats for hatchlings. Yellow-crowned night herons (*Nyctanassa violacea*) were found to be significant predators of small terrapins during spring emergence.

Small aquatic gastropods comprised 66.7% of the faecal samples analysed from the Bermudian population. Scavenged fish and vertebrate animal remains, terrestrial arthropods, polychaete worms and bivalves were consumed in lesser amounts. Sediment from the pond environment was found in 74% of the faecal samples analysed and is believed to have been incidentally ingested while foraging for the small benthic gastropods.

Eco-toxicological analyses of the pond sediment, prey and terrapin eggs showed that the Bermudian diamondback terrapins live and feed in wetland habitats characterised by chronic, multifactorial contamination; principally total petroleum hydrocarbons, polycyclic aromatic hydrocarbons and a variety of heavy metals. This study found that some of those contaminants are accumulating in the gastropod prey as well as being transferred to terrapin eggs. This may be reducing the incidence of successful embryonic development for this species in Bermuda and may likely contribute to the observed low hatching rates.

These collective findings indicate that the Bermudian population is very vulnerable to local extirpation.

Chapter 1: General Introduction to Bermuda and Diamondback Terrapins

Abstract

Bermuda is an archipelago of islands totaling 54 km² in area and situated in the Atlantic Ocean, north of the Caribbean. The climate and natural history of Bermuda is heavily influenced by the Gulf Stream which delivers warm oceanic water and biota from the Caribbean and the southeastern coastal region of the U.S.A. Despite the isolation and 110 million year age of Bermuda, the overall endemism rate is low (*ca.* 3%), having been greatly affected by the habitat loss and species extinction events associated with multiple Pleistocene sea level fluctuations. Human colonization in 1609 resulted in further dramatic changes to Bermuda's biodiversity, particularly with regards to exotic species introductions and significant habitat modification as a result of development. Currently over 70% of Bermuda's land area is considered developed. With a population of nearly 65,000 Bermuda is one of the most densely populated countries in the world.

Diamondback terrapins (*Malaclemys terrapin*) are a small to medium sized turtle that inhabit a variety of brackish water habitats along the Gulf and Atlantic coastlines of the U.S.A. They are thought to have arrived on Bermuda via oceanic currents before human colonization and presently reside within the land-locked, brackish water pond environment. These ponds (n=4) are situated upon a single square kilometre of land on a private golf course at the eastern end of the islands and have been used as water hazards since the 1920s. Diamondback terrapins represent one of only two non-marine species of indigenous reptiles currently living on Bermuda (the other is an endemic skink). Much is known about the life history of diamondback terrapins in the North American range; however there is a paucity of information regarding their status, biology and ecology on the islands of Bermuda, despite having had a presence there for over 400 years.

An introduction to Bermuda

Geography and geology

The Bermuda Rise comprises three steep-sided seamounts, two of which presently only rise to within about 50 m of the sea surface. However the third, and most northeasterly seamount, supports the Bermuda islands, which are located at 32° 19'N and 64° 46'W - some 965 km ESE of Cape Hatteras, North Carolina. The total land area of the Bermuda islands is 54 km² and consists of a crescent-shaped chain of approximately 150 low-lying islands of various sizes. All islands are concentrated along the southern edge of the seamount, the larger ones forming a narrow chain linked by causeways and bridges. This archipelago sits on top of a volcanic seamount that rises 4000 m from the seafloor. The seamount forms a shallow 100,000 hectares (ha) shelf around the islands. This shelf is made up of coral reefs, sediments, shallow lagoons and seagrass meadows and is termed the Bermuda Platform (Anderson et al., 2001). The seamount is believed to be part of a group of submarine volcanoes that formed approximately 110 million years ago along the mid-Atlantic ridge (Aumento and Sullivan, 1974). Subsequently, this complex migrated 1200 km northwestwards for 60-80 million years, at which point it passed over a 'hot spot' in the earth's crust and commenced a second period of volcanic activity that produced the Bermuda Rise. Over a period of 30-40 million years, the Bermuda Rise continued its northwest motion for about 600 km during which time there was no volcanic activity (Vacher, 1986). The Bermuda islands were formed from a combination of volcanic activity, extensive dune building and coral reef formation. The limestone that forms the surface rocks originates from various types of calcareous algae, foraminiferans, corals, and mollusc shells, the most important components being crustose coralline algae and corals. During the Pleistocene period, remains of these organisms began to accumulate as coastal beds of sediment. Lowering of sea levels exposed reefs and these calcareous sands to the influence of wind and rain. Wind action resulted in the formation of sand dunes, and, at the same time, freshwater from rain dissolved limestone, percolated down through the sand and, through evaporation, cemented grains together to form huge amounts of

soft rock called aeolianite. At the same time, where water flow was especially concentrated, it dissolved away channels creating fissures, cracks and caves. Over the ages, episodes of rising and falling sea levels, as well as further erosion, resulted in re-working of the deposits on a cyclical basis (Thomas, 2004). Maximum elevation is 76 m above sea level at Town Hill in Smith's Parish; however, the average height of land above sea level is less than 30 m (Thomas and Logan, 1992). Bermuda is divided administratively into nine parishes which are, from west to east; Sandys, Southampton, Warwick, Paget, Pembroke, Devonshire, Smith's, Hamilton, and St. George's (Fig. 1.1).

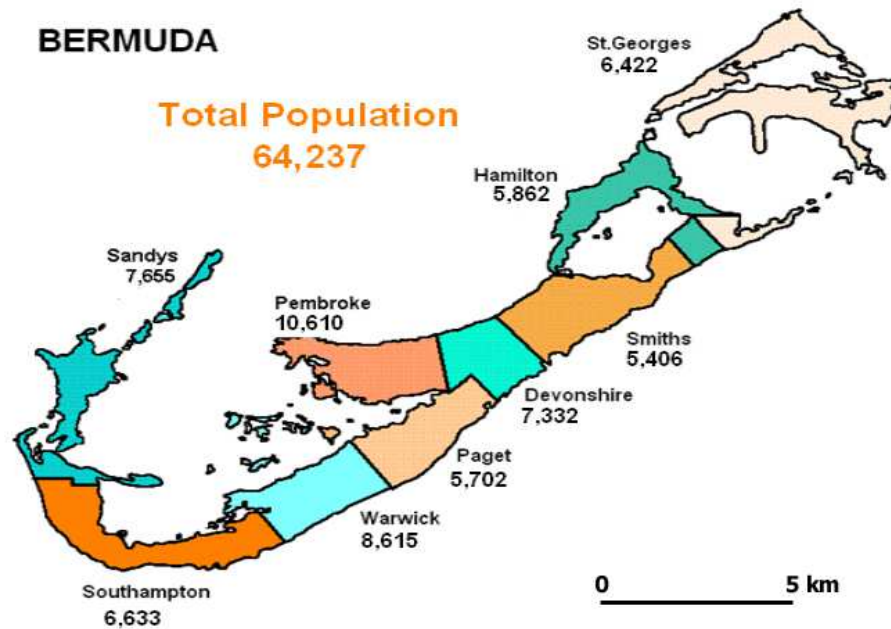


Figure 1.1. 2010 human population census map of Bermuda
Source: Bermuda Department of Statistics

Climate

Bermuda is bordered by the Sargasso Sea to the south east and by the Gulf Stream to the west, and is situated in an area of the North Atlantic Ocean which regularly receives spin-off eddies from the Gulf Stream (Fig. 1.2). These eddies deliver warm water and biota from the Caribbean and the southeastern coastal region of the U.S.A. (Glasspool, 1994; Meylan and Sterrer, 2000; Grady et al., 2001; Parham et al., 2008) and account for the mild climate of the islands. The climate and waters in Bermuda are unusually warm for this latitude, since heated water transported north in the Gulf Stream has created a northerly extension of subtropical systems. Consequently, Bermuda possesses the most northerly extensions of mangrove habitats and coral reef systems in the North Atlantic (Thomas and Logan, 1992). As there are only two distinct weather patterns, the year in Bermuda is generally divided into two climatological seasons; winter (November-April) and summer (May-October).

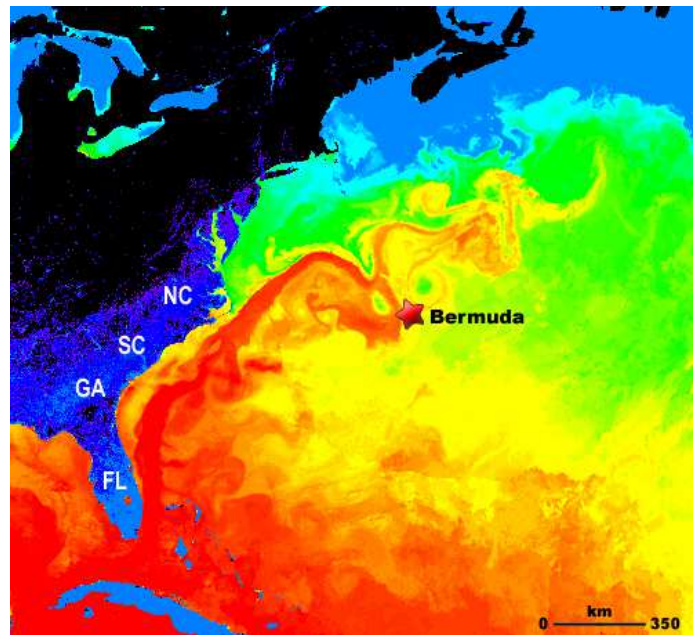


Figure 1.2. Infrared NOAA satellite imagery showing the path of the Gulf Stream into the North Atlantic (warmest water is red). NC=North Carolina, SC=South Carolina, GA=Georgia, FL=Florida. (Adapted from a National Oceanic Atmospheric Administration satellite image of the Western Atlantic).

The weather of Bermuda is greatly influenced by two features. Firstly, the waters of the Gulf Stream surround the area with a warm water mass, which in turn results in elevated air temperatures. Secondly, during the summer months, the islands are under the influence of the 'Azores High', an area of high pressure in the Atlantic that normally lies between the Azores and Bermuda. Summer frontal systems associated with westerly blowing winds are deflected to the north by the presence of the high. Consequently, summer winds are typically light and southeasterly in direction. The winter months, by contrast, are characterized by frequent northwesterly gales particularly in January, February and March since the more southerly positioning of the high pressure area gives little protection from weather associated with the westerlies (Thomas and Logan, 1992). Data from the Bermuda Weather Service shows that between 1949 and 1999 the mean monthly air temperatures varied from 17.8°C in January-March and 26.4°C in July-September (annual average 21.8°C). The annual inshore ocean surface temperatures followed a similar pattern, ranging from 18.5°C in January-March and 27.4°C in July-September (annual average 22.6°C). Rainfall does not show a marked seasonal pattern, and is the principal source of fresh water for Bermuda. During the same 50 year period, the mean annual rainfall was approximately 1410 mm. Summer rains tend to be heavy, but of short duration, though droughts lasting up to three months are not considered unusual. Winter rains are lighter, but more protracted. Humidity was uniformly high, at 73-83%, year-round (www.weather.bm; accessed in October 2012). Generally, in this part of the Atlantic Ocean, evaporation exceeds precipitation, but Bermuda's land mass is large enough to ensure that a fairly persistent cloud bank occurs at 1000 m over the islands, greatly increasing the frequency of showers, so that evaporation and precipitation are almost in balance (Thomas and Logan, 1992).

Human population and demography

Currently, approximately 50% of Bermuda's land area is used for housing and over 70% of Bermuda is considered developed (Anderson et al., 2001; Thomas, 2004). With a population of 64,237 and a population density of 1168 people per km² (Anonymous, 2011), Bermuda is among the most

densely populated oceanic islands in the world. The population density is unevenly distributed across Bermuda with 50.2% of the residents concentrated in the four central parishes; Pembroke with 10,610 residents (16.5% of total population), Warwick with 8615 (13.4%), Devonshire with 7332 (11.4%), and Paget with 5702 (8.9%) (see Fig 1.1.) The western parishes are Sandy's with 7655 residents (11.9%) and Southampton with 6633 (10.3%), while the eastern parishes comprise St. George's with 6422 residents (10%), Hamilton with 5862 (9.1%), and Smith's with 5406 (8.4%) (Anonymous, 2011).

Bermuda's terrestrial habitats

Glacial and inter-glacial Pleistocene sea level fluctuations around Bermuda were believed to have had an amplitude of 150 m which would have caused the land area to fluctuate between approximately 1000 km² to less than 50 km² over a time period spanning many hundreds of thousands of years, thereby greatly affecting Bermuda's terrestrial biogeography (Sterrerr, 1998; Sterrer et al., 2004; Olsen et al., 2006). Falling sea levels would have eliminated many of the shallow tropical marine habitats (e.g. coral reefs, seagrass meadows and mangrove swamp communities), replacing them with terrestrial and marsh habitats. Rising sea levels, conversely, would have flooded and drowned most of the terrestrial and in-land marsh habitats, extirpating a variety of terrestrial species, but creating coral reef and seagrass habitats instead.

The main natural terrestrial communities of Bermuda currently include upland hillsides, upland valleys, caves, limestone sinks (which are depressions caused by the collapse of caves), coastal uplands, rocky coastal (the area of shoreline that extends 15 m inland from the high water mark), and beaches and sand dunes (Anderson et al., 2001). Bermuda, at the time of permanent settlement in 1612, was estimated to comprise approximately 2303 ha of upland hillside, 921 ha of upland valley, 125 ha of limestone sink, 1382 ha of coastal upland, 162 ha of rocky coastal and 76 ha of beaches and sand dunes (Sterrerr et al., 2004). Four centuries later, the total area comprising upland hillside was reduced by 39%, 100% of the upland valleys had been lost (mostly to farmlands, housing and gardens), the area of

limestone sink had been reduced by 54%, the rocky coastal habitat was reduced by 56%, and upland coastal was reduced by 25%. Only the area representing the beach and dune habitat remained unchanged, mainly due to its unsuitability for development (Sterrers et al., 2004). Human modification to the environment has also created a number of habitats that would otherwise not normally exist in nature; gardens, golf courses, agricultural fields, hedgerows and wayside (un-mown grassy areas and un-tended sites where building demolition or land-fill has occurred) (Anderson et al., 2001). By 2000 more than half of Bermuda was considered developed, with an additional 20% of the land mass taken up by gardens, golf courses, agricultural fields, hedgerows and wayside (Sterrers et al., 2004).

Despite the isolation and age of Bermuda (110 million years old) the overall endemism rate is rather low (*ca.* 3%), having been greatly affected by the habitat loss and species extinction events associated with multiple Pleistocene sea level fluctuations (Sterrers, 1998). Bermuda's native vertebrate biota primarily comes from the Caribbean and south-eastern North America regions, having arrived either via the Gulf Stream or wind-borne means. Examples include post-larval reef fishes (Glasspool, 1994), an extinct land tortoise (*Hesperotestudo bermudae*) (Meylan and Sterrer, 2000) and a skink (*Plestiodon* (formerly *Eumeces*) *longirostris*) (Brandley et al., 2010) from Florida, killifishes (*Fundulus* spp.) (Smith-Vaniz et al., 1999; Grady et al., 2001) and the diamondback terrapin (*Malaclemys terrapin*) (Parham et al., 2008) from the Carolinas, and green sea turtles (*Chelonia mydas*) originating from a variety of countries throughout the Caribbean and Central America (Engstrom et al., 1998).

Bermuda's pond habitats

Modern day Bermuda has no permanent surface freshwater streams or lakes and hundreds of ponds scattered across the islands, some of which are anchialine (isolated, saline, land-locked bodies of water with permanent - often subterranean - connections to the ocean) (Thomas and Logan, 1992). Most of these anchialine ponds are very small in size and only five are over 0.5 hectare in area (Thomas et al., 1991). Man-made ponds outnumber those formed by natural processes (M. Outerbridge, unpublished data). Ponds

created on land that do not drop to the water table need liners of plastic or concrete to prevent their draining. Water hazards on golf courses are examples of these artificial ponds. Some of the anchialine ponds have acted as refuges for various rare native and endemic fauna, including Bermuda's killifishes *Fundulus bermudae* and *F. relictus* (Outerbridge et al., 2007), the flat mangrove oyster (*Isognomon alatus*) (Dangeubun, 1994), and the diamondback terrapin *Malaclemys terrapin* (Davenport et al., 2005).

All of the ponds presently found in Bermuda can be divided into two groups; natural or man-made ponds, and are either freshwater, saltwater (marine), or brackish. By definition, marine ponds have salinities of 30-35 practical salinity units (psu) when measured with a refractometer, whereas brackish ponds range from 0.5–29 psu, and freshwater ponds are less than 0.5 psu (Mantyla, 1987). Following these definitions, 60% of Bermuda's ponds are currently considered to be brackish and 40% are marine (M. Outerbridge, unpublished data). Strictly speaking, none of Bermuda's ponds can be considered freshwater since their average annual salinities exceed 0.5 psu; however, after heavy rainfall some of the slightly brackish ponds do have salinities close to zero for short periods of time (J. Bacon, personal communication).

The present day anchialine ponds in Bermuda vary both in size and in structure. Nearly all date back in formation to the late Holocene era (Neumann, 1971; Rueger, 2001). The most important factor influencing physical stability in the saline ponds is the amount of tidal exchange (Thomas et al., 1992). Temperature and salinity are dependent upon the amount of sea water that enters from the ocean, thus ponds close to the sea with relatively large connections have a higher flushing rate, narrower ranges of salinity and temperature and therefore provide a more stable environment than those of ponds further from the sea. The mean ocean tidal range in Bermuda is only 75 cm, but is greatly reduced in the anchialine ponds where there are more restrictions to tidal flow. While proximity to the ocean and the nature of the connections influence salinity level, the locations and sizes of these saltwater inlets in relation to the tide level also affect the flushing rate. Salinity stratification can occur in poorly mixed ponds, or where the connection to the sea is in the deepest part, due to the

different densities of fresh and saltwater, although this phenomenon is unlikely to occur in very shallow ponds. Thomas et al. (1991) described the physical characteristics of the six largest ponds; surface salinities ranged from 6.5 to 42.5 psu, temperatures varied from 15.0°C to 37.5°C and most possess a single connection to the sea (Thomas et al., 1991). More limited data exist for Bermuda's remaining ponds; however, it appears that salinity and temperature also follow predictable seasonal patterns. The small and shallow nature of some of these ponds means that temperatures can vary greatly from 10.6°C to 38.9°C (M. Outerbridge, unpublished data).

Bermuda's anchialine ponds generally have a rich biota. Species richness increases with increasing physical stability and diversity of habitat. Thus ponds that have submerged rock substrata, an abundant submerged mangrove root community along the periphery of the pond, and bottom sediment show greater diversity than ponds that feature sedimentary substrata only (Thomas et al., 1992). Bermuda's anchialine ponds all have deep benthic deposits of highly organic sediments and are subject to large changes in oxygen, temperature, salinity and nutrient levels (Thomas et al., 1992). Surface run-off from surrounding land transports particulate matter and plant nutrients into the ponds. Fringing mangrove trees (both red mangrove *Rhizophora mangle* and black mangrove *Avicennia germinans*) are a common feature of these saline ponds. These trees constantly drop leaves that slowly decompose, forming a highly organic layer on the pond bottom that enhances the base of the food web. Due to their small physical size and accumulated sediments, the anchialine ponds are usually quite shallow, averaging depths of only 180 cm. Because of this, ambient light levels at the bottom can be high, despite the fact that these ponds are typically very turbid due to the high levels of suspended organic material (Thomas et al., 1991). Plants, however, do not usually grow on the deeper bottoms of the ponds due to the unstable and anoxic environment created by the large quantities of extremely fine sediment and decay of organic matter. The levels of dissolved oxygen also vary considerably between ponds as well as diurnally and seasonally. Daytime photosynthesis can supersaturate pond water with oxygen, while the consumption of oxygen at night from fishes and microbial life on the sediment can reduce oxygen levels to zero,

at least in patches, resulting in transitory night-time anoxia. Anoxic events are routine in some of the poorly flushed anchialine ponds in summer and are thought partly responsible for their low species diversity, which is typically much reduced below that of open water marine habitats (Thomas and Logan, 1992). The biotic characteristics of Bermuda's ponds are highly variable. Pond size, volume, and physical stability, as well as the stochastic nature of species' colonization and the ability of these species to adapt and survive in the ponds are all factors responsible for this biological variability. One of the curious features of the ponds is that there is great variability of biota amongst the ponds. Quite often a species is found in only one or a few ponds and few species occur in all ponds. For example, red mangroves were found in four out of five ponds surveyed during the 1980s, the coffee bean snail (*Melampus coffeus*) was found in two ponds, and widgeon grass (*Ruppia maritima*) was only found in one of the ponds (Thomas et al., 1992). Surveys undertaken in 2004 and 2005 to determine the distribution of the endemic killifishes across Bermuda revealed that nine out of 27 ponds contained populations of them (Outerbridge, 2006), and Davenport et al., (2005) reported that diamondback terrapins were only found in two brackish ponds on Bermuda.

Native and endemic reptiles

The Bermuda skink (*Plestiodon* (formerly *Eumeces*) *longirostris*) is considered to be Bermuda's only endemic terrestrial vertebrate, and has the longest herpetological and paleontological history on the islands which suggests that it may have been residing on Bermuda for more than two million years (Olsen et al., 2006). It is a diurnal, ground-dwelling lizard that is thought to have evolved after an ancestral species colonised the islands after having rafted from the east coast of North America (Brandley et al., 2010). Population surveys undertaken between 1997 and 2004 suggest that Bermuda's skinks are declining in abundance and distribution (Bacon et al., 2006) when compared to their status and range in the nineteenth century (Jones, 1859). This decline has been attributed primarily to habitat loss (Bacon et al., 2006) and predation from introduced species (Davenport et al., 2001).

Five species of sea turtles are native to Bermuda; the green turtle (*Chelonia mydas*), the hawksbill turtle (*Eretmochelys imbricata*), the loggerhead turtle (*Caretta caretta*), the leatherback (*Dermochelys coriacea*), and Kemp's Ridley turtle (*Lepidochelys kempi*) (Bacon et al., 2006). All have been protected in Bermuda under national law since 1972. The green turtle nesting population that once used the beaches of Bermuda is extinct due to overharvest (Babcock, 1938), and only two isolated loggerhead nesting events have been recorded in the recent past (1990 and 2005) (Bacon et al., 2006). Juvenile green and hawksbill turtles are the two most commonly encountered species on the Bermuda platform, which serves as their developmental habitat (Meylan et al., 2011). Loggerhead juveniles are not known to inhabit the Bermuda platform, but pelagic-phase juveniles sometimes strand on beaches, often in association with winter storms and rafts of *Sargassum* weed. Leatherback turtles are occasionally sighted off the edge of the Bermuda platform, and three specimens have stranded on the islands' shores since the early 1980s. Kemp's ridleys are the least encountered species, with only three confirmed records since the 1940s (Outerbridge et al., In prep.).

Historical accounts of Bermuda's diamondback terrapins first appear in writings that date back to the 1950s (D. Wingate, unpublished notes), however it was not until 2007 that their origin on these remote oceanic islands was tested using a combination of palaeontologic (fossil, radiometric and palaeoenvironmental) and genetic data (Parham et al., 2008). These lines of evidence supported the hypothesis that these terrapins are natural colonizers of Bermuda, having arrived between 3000 and 400 years ago. Bermuda is situated in a part of the North Atlantic Ocean which regularly receives spin-off eddies from the Gulf Stream, which is thought to have served as vector for transporting diamondbacks to the islands (Davenport et al., 2005; Parham et al., 2008).

In addition to the reptilian herpetological fauna that is presently alive and living on Bermuda, one species of extinct land tortoise known from the fossil record is included in the native-endemic herpetological lists (Meylan and Sterrer, 2000; Olsen and Meylan, 2009). The land tortoise (*Hesperotestudo bermudae*) was described from the mid-Pleistocene of

Bermuda and is considered an island endemic. The holotype specimen was discovered in 1991 during the excavation of a fossilized sand dune and is thought to be 310,000 years old $\pm 10,000$ years (Meylan and Sterrer, 2000). The genus *Hesperotestudo* has a long record in North America and the authors hypothesized that it rafted from Florida to Bermuda using the Gulf Stream in as short a time as two weeks. Olsen et al. (2006) postulated that this species had only the Marine Isotope Stage (MIS) 10 glacial interval to evolve into an endemic before being extirpated by the following rise in sea level during MIS 9, thus confining the entire existence of *H. bermudae* on Bermuda to a maximum span of approximately 100,000 yr.

Introduced reptiles

Four species of *Anolis* lizards have been introduced and become naturalised on Bermuda since the early 19th century; Graham's anole (*Anolis grahami*), the Panther anole (*A. leachii*), the Barbados anole (*A. extremus*) (Bacon et al., 2006) and the brown anole (*Anolis sagrei*) (M. Outerbridge, unpublished data). Additionally, the tropical house gecko (*Hemidactylus mabouia*) and the Mediterranean or Turkish gecko (*Hemidactylus turcicus*) have also been introduced and become naturalised (R. Mariera and A. Copeland, personal communication).

Anolis grahami is the most common of the introduced lizards. It was introduced intentionally in 1905 to control a species fruit fly that was a horticultural pest at the time (Harris, 1905). The precise origin of *A. leachii* on Bermuda is unknown but it is believed to have been introduced ca. 1940 (Wingate, 1965). Both *A. grahami* and *A. leachii* currently have an island-wide distribution (Bacon et al., 2006). *Anolis extremus* is believed to have been accidentally introduced via ship(s) some time prior to 1945 to the Royal Naval Dockyard in Sandy's Parish (Wingate, 1965), and its present day distribution is still largely confined to the western parishes (Bacon et al., 2006). *Anolis sagrei* is considered to be the least common anole on the Bermuda islands. This species presence was only brought to the attention of the Bermuda Government's Department of Conservation Services in 2011 when a local farmer found one individual in a shipment of sphagnum moss that had been shipped from Florida. More individuals have been found in

the wild since then, however its present distribution is limited to a few small localities (M. Outerbridge, personal observation). *Anolis sagrei* is an exotic species of small lizard that was introduced from the Caribbean to south Florida in the 1940s and subsequently became established in several states along the Gulf and Atlantic coasts (Vigil, 2006). *Hemidactylus mabouia* and *H. turcicus* are nocturnal lizards native to continental Africa and the Mediterranean basin respectively; however they have been introduced to many other parts of the globe, including several New World countries (Lever, 2003). Human-mediated dispersal is believed to be the cause of their relatively recent and dramatic range expansion. They are associated with human development and are highly adaptable, thereby making them successful invaders throughout their non-native range. Both species were first discovered in 2005 in a cargo hangar at the Bermuda International Airport, but are now naturalized and spreading (M. Outerbridge, personal observation). The point of origin on Bermuda suggests that both species of gecko may have accidentally arrived via air freight and then subsequently been transported to other locations across the islands.

The red-eared slider (*Trachemys scripta elegans*) is an introduced and invasive species on Bermuda. Anecdotal evidence suggests that it was first sold as a pet in Bermuda during the 1950s, and it continues to be sold as a pet over 60 years later. Throughout this time individuals have escaped from captivity, or unwanted individuals have been deliberately released, which led to this species becoming firmly established in the wild. Surveys undertaken in 2005 and 2006 revealed that virtually all freshwater and slightly brackish ponds and canals on Bermuda (salinities $\leq 12\text{‰}$) were found to contain populations of red-eared sliders existing in densities as high as 981 turtles ha^{-1} (Pitman's Pond) (Outerbridge, 2008).

Other non-indigenous reptiles have also been deliberately imported to Bermuda and kept as pets in private captivity by members of the general public. The Care and Protection of Animals Act (1975) requires that permits be obtained from the Department of Environmental Protection for the importation of exotic species that are to be kept locally as pets. This act also provides for the prohibition of animals likely to become a hazard to the ecology of Bermuda. The North American box turtle (*Terrapene* spp.) has

traditionally been among the more popular reptile pets to be kept. These turtles are known to be breeding in captivity and on at least one occasion a hatchling was found in the wild (M. Outerbridge, personal observation). Snakes, banned under the Care and Protection of Animals Act, have been illegally smuggled into Bermuda (Anderson et al., 2001) and escape into the wild from time to time where they are found by members of the public. Every effort is made by the Department of Conservation Services (DCS) to capture these individuals. DCS occasionally receives non-indigenous reptiles from members of the general public who have encountered individuals in the wild, often in their gardens (B. Outerbridge, personal communication). Examples include the common snapping turtle (*Chelydra serpentina*) and green iguana (*Iguana iguana*).

Diamondback terrapins

Taxonomic classification

Diamondback terrapins belong to the Family Emydidae, a large and diverse group of reptiles collectively known as ‘pond turtles’ that are naturally found throughout North America, much of Europe, and eastward into Russia, the Near East, and North Africa (Meylan, 2006), and are the only member of the genus *Malaclemys*. Seven subspecies of diamondback terrapin are currently recognized; Northern diamondback (*Malaclemys terrapin terrapin*), Carolina diamondback (*Malaclemys terrapin centrata*), Florida east coast diamondback (*Malaclemys terrapin tequesta*), Mangrove diamondback (*Malaclemys terrapin rhizophororum*), Ornate diamondback (*Malaclemys terrapin macrospilota*), Mississippi diamondback (*Malaclemys terrapin pileata*), and Texas diamondback (*Malaclemys terrapin littoralis*). These seven subspecies have been divided into northern populations (*M. t. terrapin*, *M. t. centrata*) and southern populations (*M. t. tequesta*, *M. t. rhizophororum*, *M. t. macrospilota*, *M. t. pileata*, *M. t. littoralis*) with Merritt Island, Florida, providing a break between the two. Genetic studies, however, do not fully agree with the existence of these seven subspecies (Lamb and Avise, 1992; Hart, 2005; Hauswaldt and Glen, 2005).

Species description

Diamondback terrapins are small to medium sized turtles that show distinctive shell and soft tissue markings; however, these markings vary greatly throughout their range. The carapace is typically oblong in shape and possesses a mid-dorsal keel which is more visibly raised, or knobbled, in the southern subspecies. Carapace colour is highly variable, but usually of earth tones ranging from light olive and brown to dark brown and black. The carapace is also marked with concentric growth rings that are most pronounced on younger individuals (Fig. 1.3), from which this species gets its common name; however, these disappear with age. The circular depressions that these rings make extend below the veneer of each scute and are imprinted upon the dorsal surface of the underlying bones of the carapace. The plastron, in contrast to the carapace, is more brightly coloured with yellowish or orange hues and can be either plain in appearance or smudged with varying amounts of dark blotches. Sometimes, however, the plastral scutes can have a dark base colour with lighter colourful edges. The plastral scutes may also show growth rings. These rings, or annuli, have been used by some researchers to estimate the age of individuals (Seigel, 1984; Tucker et al., 1995; Gibbons et al., 2001); however, this technique remains a contentious method of aging terrapins and many agree that it is not possible to use it on older individuals whose rings have disappeared with the passage of time (Morreale, 1992; Gibbons et al., 2001). Skin colour also varies throughout the range, but is generally shades of grey with dark spots, flecks or lines (the latter have not been observed in the Bermuda population – M. Outerbridge, personal observation) (Fig. 1.4).

Diamondback terrapins show sexual dimorphism; with males being considerably smaller than females (Fig. 1.5) and having proportionally smaller heads, but wider and longer tails, with a cloaca situated posterior to the edge of the carapace when the tail is fully extended (Fig. 1.6).

The diamondback terrapin carapace normally features 38 named scutes:- one nuchal, five vertebrals, four pairs of costals (also known as pleurals), eleven pairs of marginals, and two supracaudals (Fig. 1.7). The plastron is normally composed of twelve named scutes; one pair of gulars, one pair of humerals, one pair of pectorals, one pair of abdominals, one pair

of femorals, and one pair of anals (Fig. 1.8). Both carapace and plastron are joined by a bridge. Variations in the number of vertebral, costal or marginal scutes are not uncommon, and may involve an extra, split, or distorted scute. These anomalies are believed to be caused by high incubation temperatures (Wood and Herlands, 1997; Herlands et al., 2004) and possibly embryological exposure to petroleum crude oil and polycyclic aromatic hydrocarbons (Van Meter et al., 2006).



Source: Mark Outerbridge

Figure 1.3. Photograph of a Bermuda specimen of *Malaclemys terrapin* illustrating the concentric rings on the scutes of the carapace, giving the appearance of the facets on a diamond, from which this species derives its common name.



Source: Mark Outerbridge

Figure 1.4. Photograph of a Bermuda specimen of *Malaclemys terrapin* illustrating the dark speckled pattern on the skin.



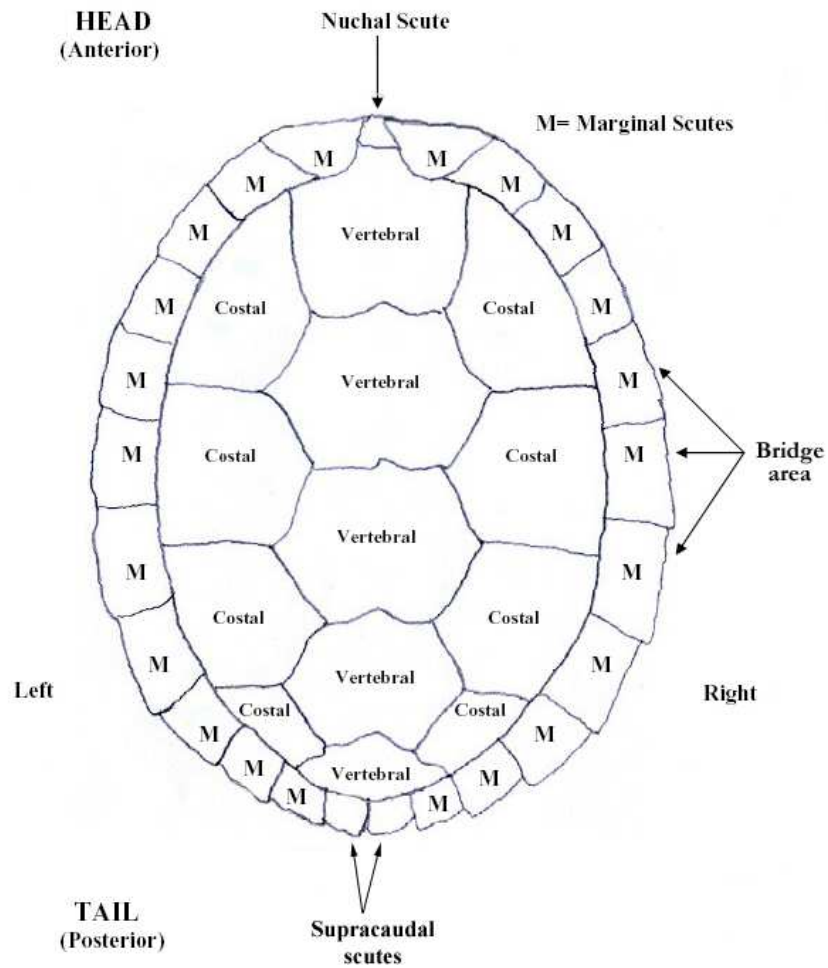
Source: Mark Outerbridge

Figure 1.5. Photograph showing the difference in body size between a mature male (left) and a mature female (right) diamondback terrapin.



Source: Mark Outerbridge

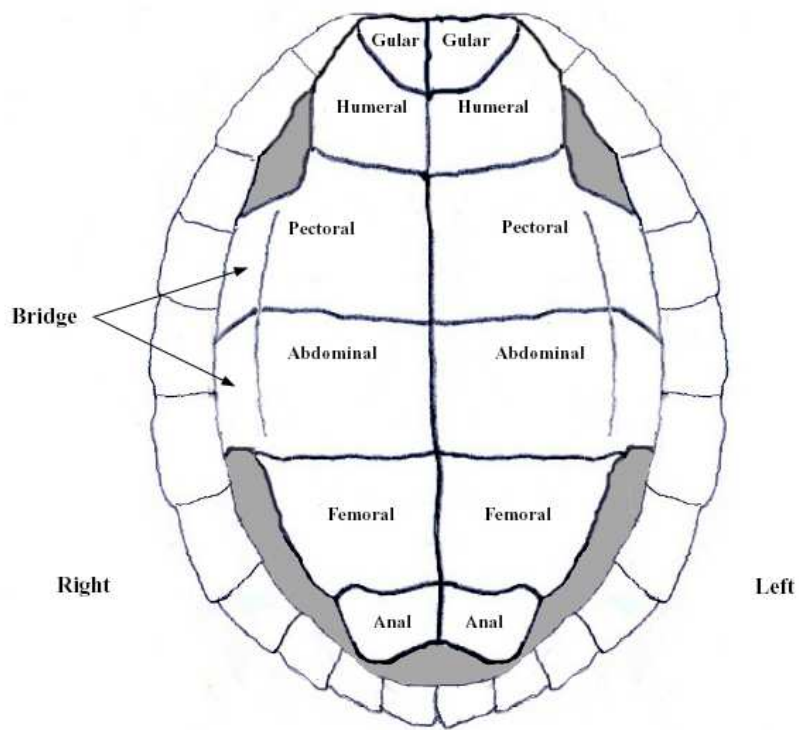
Figure 1.6. Photograph showing the tail size relative to body size of a mature male terrapin. White arrow shows the location of the cloaca.



Source: Mark Outerbridge

Figure 1.7. Illustration of the carapace of *Malaclemys terrapin* with named scutes.

HEAD
(Anterior)



TAIL
(Posterior)

Source: Mark Outerbridge

Figure 1.8. Illustration of the plastron of *Malaclemys terrapin* with named scutes.

Global geographic distribution and range-wide status

Diamondback terrapins are endemic to the Atlantic and Gulf Coasts of the United States of America. They have a discontinuous range which extends across 16 states from Cape Cod, Massachusetts, in the north to Corpus Christi, Texas, in the south (Fig. 1.9).

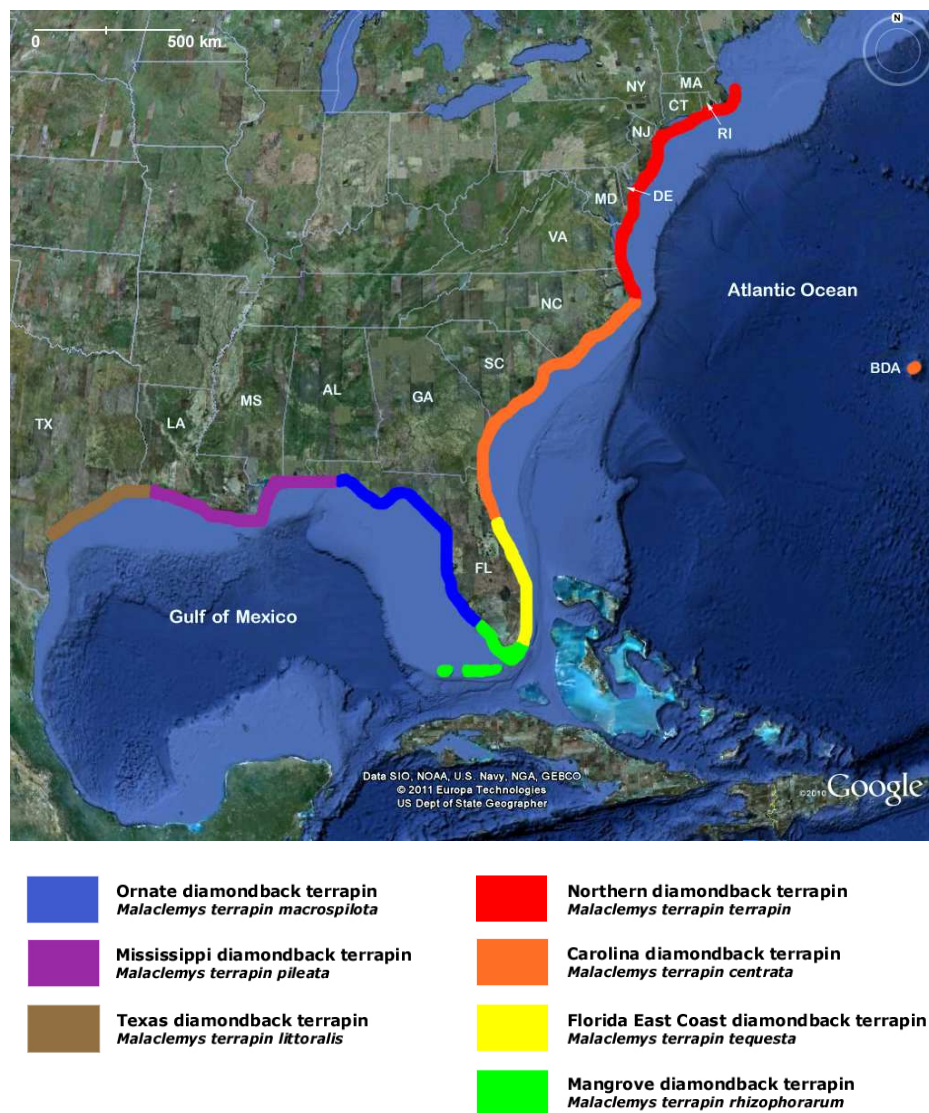
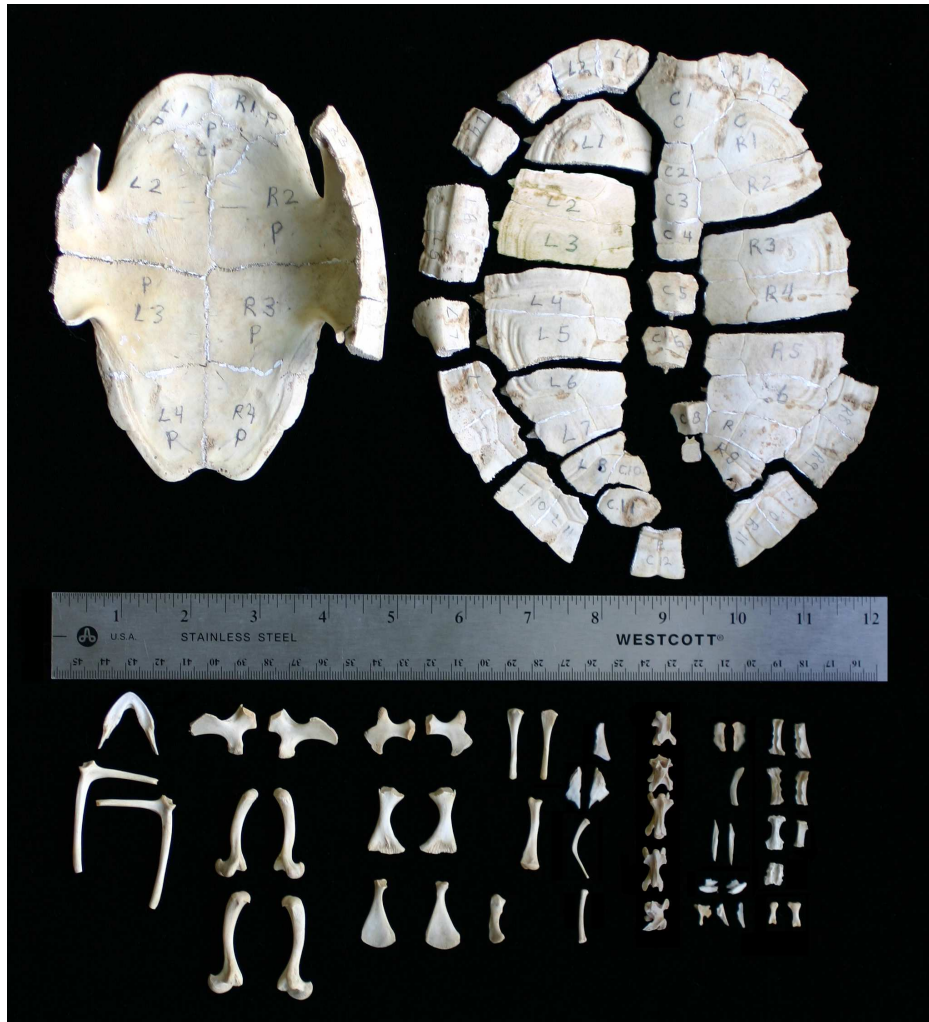


Figure 1.9. Map illustrating the range-wide distribution of the seven recognized diamondback terrapin subspecies. (Adapted from Butler et al., 2006; Lee and Chew, 2008).

Diamondback terrapins are restricted to coastal saline and brackish waters and are found in salt marsh, river estuary, tidal creek, lagoon, and mangrove habitats (Butler et al., 2006; Ernst and Lovich, 2009). Five of the seven subspecies occur within Florida, of which three are considered to reside exclusively in that state. The northern diamondback terrapin (*Malaclemys terrapin terrapin*) ranges from Cape Cod in Massachusetts to Cape Hatteras in North Carolina. The Carolina diamondback (*M. t. centrata*) ranges from Cape Hatteras southwards to Volusia County in Florida. The Florida East Coast diamondback (*M. t. tequesta*) ranges from Volusia County to Miami-Dade County, as well as possibly into the upper Keys in Monroe County. The mangrove diamondback (*M. t. rhizophorarum*) occurs in Monroe County from Fort Myers to Florida Bay and throughout the Florida Keys and the Marquesas. The ornate diamondback (*M. t. macrospilota*) occurs from Florida Bay to the western part of the Florida Panhandle in Walton County. The Mississippi diamondback (*M. t. pileata*) ranges from western Choctawhatchee Bay in Okaloosa County, Florida, westwards through the state of Louisiana. The Texas diamondback (*M. t. littoralis*) is found from western Louisiana to Corpus Christi in Texas (Butler et al., 2006; Lee and Chew, 2008; Ernst and Lovich, 2009). The only geographic region where diamondback terrapins appear to reside naturally outside their U.S.A. range is in Bermuda.

The origin of Bermuda's diamondback terrapins was tested by Parham et al. (2008) using a combination of fossil, radiometric, geological and genetic data, which lines of evidence supported the hypothesis proposed by Davenport et al. (2005) that diamondback terrapins could have naturally colonized the remote oceanic islands of Bermuda using the Gulf Stream as the transport mechanism. A comparative genetic survey conducted by Parham et al. (2008) of 27 terrapins collected throughout their natural global range found that the Bermudian samples most closely resembled samples from the Carolina region of the U.S.A. Thus, the diamondback terrapins currently inhabiting Bermuda have been designated as *M. t. centrata*. Furthermore, the Parham et al. (2008) study subjected a fragment of scute taken from a sub-fossil (Fig. 1.10) to radiocarbon dating. The Bermudian

sub-fossil had been discovered in 1974 inside a cave approximately one kilometre from the current location of the extant diamondback population.



Source: Mark Outerbridge

Figure 1.10. Sub-fossil remains of a Bermudian diamondback terrapin discovered in a cave in 1974.

The oldest calibrated dates that resulted from the radiocarbon dating were AD 1222-1276, whereas the most recent dates were AD 1427-1620 (with a most likely age range of AD 1452-1554). The majority of these periods predate human settlement of Bermuda (1609), and thus the authors suggested that the 400-600 year antiquity of the recovered bones was also consistent with the natural origin hypothesis. It is not at all unreasonable to imagine founding populations of diamondback terrapins arriving on oceanic

currents, particularly in light of the fact that a now extinct species of land tortoise (*Hesperotestudo bermudae*) arrived in Bermuda the same way over 300,000 years ago (Meylan and Sterrer, 2000). It is also quite feasible that diamondbacks have reached Bermuda naturally on several occasions, before and after 17th century human colonization.

Diamondback terrapins were listed as a globally near-threatened species by the International Union for the Conservation of Natural Resources (IUCN) in 1996. Their status, which varies from state to state in the U.S.A., ranges from ‘endangered’ to ‘a species of special concern’ (Lee and Chew, 2008). More recently (2011), the Centre for Biological Diversity advocated that diamondback terrapins should be considered as a candidate for a U.S.A. proposal to amend Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) at the 16th meeting of the Conference of the Parties. Their justification was that the species is “documented to be vulnerable to over-exploitation” and has an “intrinsically slow capacity to recover.” In 2013, diamondback terrapins were included in Appendix II in an attempt to regulate international trade so that exports from the native range are not detrimental to the species’ survival in the wild.

As a direct result of this doctoral investigation, Bermuda’s diamondback terrapins were classified in 2012 as a level II protected species and declared to be ‘Vulnerable’ under the Bermuda Protected Species Act (2003). Furthermore, the Protected Species Amendment Act (2011) now considers it an offence for an unauthorized person to willfully damage, destroy, injure, disturb, uproot, fell, kill, take, import, export, sell or purchase a level II protected species or any part of a level II protected species. Offenders are liable, on summary of conviction, to a fine of \$15,000 or one year of imprisonment (www.laws.bm; accessed October 2012). Despite this new legislative protection, there are no conservation measures currently in place. Diamondback terrapins are not harvested for food in Bermuda or caught as bycatch in commercial or recreational shellfish pots, and none of the ponds have boating traffic (M. Outerbridge, personal observation), however, the area in which the terrapins reside is

currently, and has also historically been, heavily impacted upon by other anthropogenic activities.

Biology and ecology of diamondback terrapins in the North American range

Habitat requirements

Diamondback terrapins have a life cycle comprised of distinct phases that have different habitat requirements. Adult and sub-adult terrapins have need of brackish bodies of water in which they feed, mate and, for populations residing in cooler regions, brumate (the reptilian equivalent of hibernation); mature female terrapins require sandy substrate for egg laying; hatchlings and small juveniles require dense vegetation that grows adjacent to the adult wetland environment in which they forage, grow and hide from predators (Pilter, 1985; Lovich et al., 1991; Roosenburg, 1991). Diamondback terrapins are the only species of turtle that have specialized to inhabit the tidal salt marsh and estuarine environment, and show unique physiological and behavioural adaptations that enable them to live there (Cowan, 1971; Gilles-Baillien, 1973; Cowan, 1990; Davenport and Macedo, 1990; Hart and Lee, 2006).

General biology

The annual activity cycle of adult diamondback terrapins is one that generally begins with emergence from winter-induced brumation during the spring which is then quickly followed by a period of courtship and mating. Nesting soon follows and often lasts for many months during which females can deposit multiple clutches of eggs (Seigel, 1980; Goodwin, 1994; Roosenburg and Dunham, 1997). Diamondback terrapins are believed to have a very small home range (Lovich and Gibbons, 1990; Gibbons et al., 2001; Baldwin et al., 2005) and some mature females are known to return to the same nesting beaches annually (Jeyasuria et al., 1994). The incubation period and the sex of the developing embryos are determined by the incubation temperatures; cooler temperatures produce male offspring while warmer temperatures produce female (Roosenburg and Kelley, 1996).

Hatchlings will, upon emergence, typically seek refuge within the closest vegetation and show avoidance of open water (Burger, 1977; Draud et al., 2004). Very little information exists in the literature about the life history of hatchlings and juveniles from the time they depart the nest to the time that they recruit to the sub-adult population, but it is believed that this period of development occurs in a separate habitat from that where the adults reside. Growth is most rapid during the first few years after hatching, but then slows down considerably after sexual maturity has been attained (Tucker et al., 1995; Roosenburg and Kelley, 1996). Diamondback terrapins usually enter brumation in November and December and remain in that state either buried in sediment or beneath undercut banks through February or March the following year (Yearicks et al., 1981; Seigel, 1984); however, some populations in Florida were observed to be active on warm days during the winter (Hart, 2005). The lifespan of diamondback terrapins in the wild has been estimated to be approximately 20 years (Seigel, 1984), but may last as long as 40 years in captivity (Hildebrand, 1932).

Population biology

The Diamondback Terrapin Working Group (DTWG) has participated in at least two questionnaire based surveys concerning the status and research needs of terrapins in the U.S.A. involving a variety of scientists, state agency biologists, and educators from all 16 states where diamondback terrapins naturally occur. These questionnaires asked respondents to assess the status of terrapins in their state as declining, stable, increasing, or unknown. The majority (*ca.* 50-55%) of the respondents said that the status of their terrapin populations was unknown, a further 30-33% said that terrapin populations were declining, and only 15-17% said that their terrapin populations were stable. Not one considered their terrapin populations to be increasing (Seigel and Gibbons, 1995; Butler et al., 2006). Roosenburg et al. (1997) reported a population estimate of 2778-3730 individuals in the Patuxent River Estuary of Chesapeake Bay; Seigel (1984) estimated populations of 213 and 404 at two sites in east central Florida; Hurd et al. (1979) suggested that as many as 1655 terrapins inhabited the Canary Creek salt marsh in Delaware; Butler (2002) reported a population of 3147

terrapins were found to be using a northeastern Florida nesting beach; and Hart (2005) estimated the Big Sable Creek population within the Everglades National Park in southwest Florida to be 1545 individuals. Density estimates of terrapins are less available in the literature, but were reported to range from 53-72 terrapins ha⁻¹ in central Florida (Seigel, 1984). Sex ratios in terrapin populations vary from being strongly female biased (Seigel, 1984; Roosenburg et al., 1997) to being male biased (Lovich and Gibbons, 1990). Hart (2005) reported that the sex ratio in the Big Sable Creek population was 1:1. Female terrapins can reach maximum carapace lengths of 238 mm range wide in North America; males 140 mm (Ernst et al., 1994). These size differences may reflect resource partitioning between the sexes (Tucker et al., 1995); however it has also been suggested that it may be influenced by sexual selection. Gibbons and Lovich (1990) suggested that sexual size differences in turtles were determined by sex-specific maturity patterns. Early maturity at a small body size may allow males to begin breeding earlier instead of investing energy into somatic growth, whereas females may benefit from continued growth beyond the size that males mature because of the advantage that body size confers on increased reproductive output (Gibbons and Lovich, 1990). It has also been suggested, for species that do not exhibit male to male combat for mates, that smaller male turtles may have an advantage of increased mobility over larger males when vying for females (Berry and Shine, 1980).

Reproduction

Diamondback terrapins form breeding aggregations, which has been hypothesised to increase the probability of successful mating (Seigel, 1980), during which females are approached and courted by males. Copulation appears to be brief and male territorial defence or male to male combat has not been reported for this species (see review in Ernst and Lovich, 2009).

Nesting ecology shows variability throughout the terrapin range. Females mature at ages between four and 13 years while males mature at much younger ages between two and seven years (Cagle, 1952; Seigel, 1984; Lovich and Gibbons, 1990; Lovich et al., 1991; Roosenburg, 1991; Gibbons et al., 2001). Terrapins in the northern parts of the range take

longer to reach sexual maturity than those in the southern regions. The nesting season typically begins in late April and ends in late July for terrapins in Florida (Seigel, 1980; Butler et al., 2004), while the nesting seasons in the extreme northern range are restricted to only June and July (Burger and Montevecchi, 1975; Lazell and Auger, 1981; Goodwin, 1994; Jeyasuria et al., 1994; Feinberg and Burke, 2003). In Louisiana, nesting may occur as late as September (Burns and Williams, 1972). Terrapins are reported to nest on sand dunes, beaches and along the sandy margins of marshes and islands (Burger and Montevecchi, 1975; Burger, 1977; Seigel, 1980; Roosenburg, 1994). Nest sites are generally flat (which facilitates the postures that females assume during digging and egg deposition) with low vegetative cover (which minimizes the destruction of the nests via mammalian and plant root predation). Clutch size ranges from 4-22 eggs; northern subspecies have the greatest mean clutch sizes of approximately 16 eggs in Rhode Island (Goodwin, 1994) and 13 eggs in Maryland (Roosenburg and Dunham, 1997), while those in Florida have mean clutch sizes of approximately seven eggs (Seigel, 1980; Butler, 2000). Estimated nesting densities range from 0.52 ha⁻¹ in Massachusetts (Auger and Giovannone, 1979) to 157.1 ha⁻¹ in New Jersey (Burger and Montevecchi, 1975), to 1125 ha⁻¹ in Maryland (Roosenburg, 1994). Diamondback terrapins exhibit temperature-dependent sex determination (TSD), whereby the ambient temperature of the nest medium affects the sex of the developing embryos (see Ewert and Nelson, 1991; Jeyasuria et al., 1994; Roosenburg and Place, 1994; Roosenburg and Kelley, 1996). TSD has been suggested as being a factor in biased sex ratios observed in some terrapin populations (Lovich and Gibbons, 1990; Ewert and Nelson, 1991). Incubation periods (the time it takes for eggs to develop and hatch) vary from 50-120 days. In New Jersey the mean incubation period was reported to be 76.2 days (Burger, 1977), while terrapins on the east Florida coast had a mean period of 65.6 days (Seigel, 1980). Hatching occurs from early August through to mid-October in northern terrapin populations (Burger, 1977; Roosenburg, 1991), and from early July to early October in some Florida populations (Butler et al., 2004). Emergence periods (the time hatchlings spend in the nest prior to leaving it) show tremendous variability

throughout the range; hatchlings may depart within hours after hatching (Roosenburg and Kelley, 1996) or they may spend months over-wintering in the nest chamber and emerge the following spring (Lazell and Auger, 1981; Roosenburg and Kelley, 1996; Baker et al., 2006). The timing of nest emergence is influenced by biological factors (e.g. evolutionary response and internal timing) as well as physical factors (e.g. rainfall and temperature) (see review in Costanzo et al., 2008). Gibbons and Nelson (1978) postulated that delayed emergence is a strategy employed by species in which high environmental uncertainty exists for hatchlings that emerge immediately after hatching. The suggested benefits of delayed emergence (over-wintering) include avoidance of predators and avoidance of exposure to adverse environmental conditions. Conversely, the benefits of early emergence (summer/fall) include the potential to begin feeding and growth immediately (Gibbons and Nelson, 1978).

Diet and feeding

Diamondback terrapins have been identified as an important component of the trophic dynamics of the salt marsh ecosystem (Silliman and Bertness, 2002; Davenport, 2011). Diamondbacks are carnivorous and feed mostly upon a variety of marine molluscs and crustaceans (namely periwinkles, crabs, mussels and clams) within the salt marsh, estuarine and mangrove ecosystems throughout their North American range (see reviews in Butler et al., 2006; Ernst and Lovich, 2009). They also show resource partitioning, whereby individuals with wider heads (the largest females) consume larger snails and crabs than terrapins possessing smaller heads (Tucker et al., 1995). Diamondbacks appear to be predators that use visual cues while foraging and they also show selectivity in the prey that they eat (Davenport et al., 1992; Tucker et al., 1995; Tucker et al., 1997). The food consumption of diamondbacks is considered to be ten times higher than that of other closely related aquatic emydid turtles of the same size (Davenport and Ward, 1993); however, studies have shown that diamondback appetite reduces (by up to 50%) when held in full sea water (34 psu) without access to freshwater (Davenport and Ward, 1993). Davenport and Macedo (1990) have shown that diamondback terrapins have fine salinity discrimination

and employ behavioural and postural responses designed to maximize exploitation of rainfall in the freshwater-limited environments throughout their North American coastal range.

Threats

Diamondback terrapins have had a long history of exploitation as a food source in the U.S.A. (Hart and Lee, 2006), and were harvested from the wild for centuries before their over-exploitation led to the raising of captive stock in the 1920s (Coker, 1906; Hildebrand, 1929). The demand for terrapin meat peaked between the late 19th and the early 20th centuries when the species became regarded as a gourmet food item which made them “one of the most economically important reptiles in the world” during that period (Ernst and Lovich, 2009). As a consequence, natural populations across the North American range were decimated (Carr, 1952).

The incidental capture and drowning of terrapins in blue crab traps along the Atlantic and Gulf coasts has been identified as the primary current threat to terrapin survival (Butler et al., 2006) and has prompted some states to require the use of by-catch reduction devices (BRDs) (also known as terrapin excluder devices or TEDs) on crab traps in order to minimize terrapin by-catch (Wood, 1997; Hart and Lee, 2006).

Predation has also been identified as a significant threat to terrapin populations. Terrapin nests and hatchlings are preyed upon by a wide variety of predators throughout the North American range that include small mammals (raccoons, skunks, foxes, rats) and birds (gulls, crows, herons, bald eagles), as well as ghost crabs, ants, and plant roots (most notably dune grass) (see review in Ernst and Lovich, 2009). Raccoons (*Procyon lotor*) are recognized as a major terrapin nest predator (Burger, 1977; Roosenburg, 1992; Goodwin, 1994; Butler et al., 2004), and were responsible for destroying 87-99% of terrapin nests at various study sites along the Atlantic coast of the U.S.A. (Roosenburg, 1992; Feinberg and Burke, 2003; Butler et al., 2004). Adult terrapins (particularly nesting females) are also occasionally preyed upon by raccoons (Seigel, 1980; Feinberg and Burke, 2003). Draud et al. (2004) reported that the Norway rat (*Rattus norvegicus*)

was a major predator on terrapin hatchlings and juveniles (25-41 mm SCL) in a New York population.

Road-associated mortality of nesting females has also been identified as a threat in some terrapin populations (Wood and Herlands, 1997). Adult females are killed every nesting season as they attempt to cross roads in search of nesting sites. During a seven year period, over 4000 terrapins were discovered as road kill during routine patrols at one study site in New Jersey (Wood and Herlands, 1997).

Pollution is considered to be a relatively new threat to diamondback terrapins. This species has been shown to be a bio-indicator of environmental contaminants in salt marsh ecosystems (Kannan et al., 1998; Burger, 2002), however the degree to which these contaminants affect terrapin health is largely unknown.

Historical ecology of diamondback terrapins and their wetland habitats in Bermuda

Historical ecology of diamondback terrapins in Bermuda

The first written historical accounts of Bermuda's biodiversity date back to the early 17th century when a violent storm caused the *Sea Venture*, the flagship of a fleet of ships bearing English settlers and supplies towards the Virginian Jamestown colony in the New World, to become wrecked upon the reefs of Bermuda whereupon the 150 passengers and crew were forced to land and take up residence in 1609. The ship-wrecked survivors described finding sea turtles among Bermuda's fauna (Lefroy, 1876), which they soon discovered were highly edible, and yet never mentioned finding an endemic skink or diamondback terrapins – two species that are still present on the islands of Bermuda over 400 years later. This absence in the early records may imply that both species were never particularly noticeable elements of Bermuda's wildlife, or perhaps they were not written about because neither were considered important food items for the inhabitants and thus not worthy of mention.

An examination of the literature published in the mid 19th and early 20th centuries on Bermuda's natural history reveals a conspicuous absence

of diamondback terrapins from the various lists of herpetological fauna. Jones (1859) and Hurdis (1897) list green turtles, hawksbill turtles, blue-tailed skinks and an isolated record of a small snake. Heilprin (1889) mentions sea turtles and the Bermuda skink as the only reptiles found on Bermuda, while Cope (1861) only lists a skink and omits the sea turtles entirely from the Bermudian herpetological fauna. Similarly, the works published as a result of the H.M.S. *Challenger* expedition only list a single species of reptile described as ‘a lizard common in Carolina’ (believed to be the endemic Bermuda skink), and completely fail to mention the sea turtles (Thompson, 1877). Garman (1884) lists four species of sea turtles (green turtle, hawksbill, loggerhead and leatherback) and one species of skink, while Jones and Goode (1884) fail to catalogue any reptiles at all in their *Contributions to the Natural History of the Bermudas*, and instead only mention that migratory water birds frequent ‘the land which lies between Paynter Vale and the south shore’ including the ‘ponds of Tucker’s Town (where)...along the shores of these ponds the mangrove grows luxuriantly wherever congenial mud affords its roots a resting place.’ Verrill (1902) and Verrill et al. (1903) describe four species of sea turtles (green turtle, hawksbill, loggerhead and leatherback) and an American blue-tailed lizard as being the only species of terrestrial reptile existing on Bermuda at the time of their surveys. However Verrill, in his 1902 publication, does concede the opinion that “the early writers (of Bermuda’s natural history)...were not close observers of small creatures”. The following year, Verrill continued to attempt to explain how it was possible to inadvertently omit animals from the earlier taxonomic lists by stating “so many species of comparatively large and conspicuous marine animals could be added in a few weeks to the fauna of a locality, where so many previous collections have been made, may seem strange. This is due, however, partly to a careful scrutiny of the hiding places of those forms that depend upon concealment for their safety, partly upon the fact that localities were visited...in which certain species seem to be localized, and perhaps, in some cases, upon the earlier season of the year (March), when some of the new forms come into shallow water to spawn” (Verrill et al., 1903).

Most of these early natural historians were visitors to Bermuda and did not have the time to make long-term observations, which take into account the effects of seasonal variation with respect to the habits and activities of some of the islands' biodiversity. An exception was John Hurdis, who was a resident of Bermuda from 1840-1855 and spent many hours hunting for birds in the various ponds and marshes. Hurdis began taking detailed notes on the islands' natural history in 1846, publishing over 650 diary entries from 1847-1854 and paid particular attention to collecting and describing the avifauna. In one specific diary entry dated November 4th – 7th 1847, Hurdis mentioned that he visited 'the ponds between a marsh and Harrington Sound', and did not report observing any small turtles or terrapins at that locality. He visited that area again three years later on July 23rd 1850, specifically naming Trott's Pond this time, and once more did not mention any terrapin sightings. It is most unlikely that Hurdis would have seen terrapins in November of 1847 as they would probably have been brumating at that time of year; however, it is curious that he did not observe any on his 1850 visit. It is possible that the population of diamondback terrapins residing in those ponds at that time may have been very low and, coupled with their naturally cryptic habits and the turbid water of both Mangrove Lake and Trott's Pond, may have contributed to Hurdis having overlooked the terrapins, particularly if he was there to collect ornithological specimens. It is also possible that he was simply not there long enough to observe one, or that the sighting of a small pond turtle might not have been a significant enough event to record in his diary on that day. If some of the other 19th century natural historians were able to overlook some of the more obvious elements of the Bermudian herpetological fauna, such as the sea turtles which at that time were a fisheries resource, then it is not surprising that the diamondback terrapins (with their much smaller size and more limited distribution) were also omitted from the written records.

The first written account describing the presence of diamondback terrapins in Bermuda is credited to David Wingate circa 1952. Wingate discovered a recently dead terrapin which had been partially consumed by rats within the saw-grass marsh at South Pond. The specimen was later positively identified as a diamondback terrapin (Parham et al., 2008);

however, it is unknown whether the terrapin was killed by a rat prior to partial consumption or whether it had died from another cause and subsequently been scavenged.

The historical distribution of diamondback terrapins on Bermuda remains unclear. It can be inferred from Hurdis (1897) that this species may already have had a very limited distribution by the mid 19th century; however a recent discussion with an elderly resident, Mr Teddy Tucker, suggests that diamondback terrapins may have been present in an area of land between two marsh complexes formerly known as the Collector's Hill Marsh in Smith's Parish and Camden Marsh in Paget Parish. Mr Tucker remembers exploring these marshlands, which he described as being interconnected by a series of ponds and canals, as a young boy in the 1940s. It was during this period that he observed 'many terrapins' inhabiting the area, and was even able to capture two small specimens which he kept as pets for nearly four decades. These two terrapins were described as being oval in shape and having a darkly coloured carapace and a plastron with an orange ring around the periphery and a dark centre. Mr Tucker claims that these terrapins are still a common species in Virginia, Georgia and the Carolinas where he used to routinely observe them while on vacation in those states. When presented with photographs of *T. s. elegans* and *M. terrapin*, Mr Tucker identified the latter as looking most alike the terrapins that he observed in the marshes as a boy, stating that 'they definitely did not possess a red stripe along the sides of their heads' – a characteristic feature that a layman can use to quickly differentiate between the two emydid turtles in Bermuda. Unfortunately Mr Tucker cannot identify the place where he buried his pet terrapins after they died in captivity, so it is not possible to exhume their remains for additional analysis. Until further diamondback terrapin remains are discovered in caves, buried in peat marshes, or excavated from the limestone rock, the historical distribution of this species across Bermuda will remain uncertain.

Historical ecology of mangrove swamps, ponds, and inland marshes in Bermuda

Human activities have caused nearly all of Bermuda's wetlands to decline, although some natural processes have also had an impact. Since Bermuda's permanent settlement from 1612 onwards, humans have filled, drained, denuded, and polluted the mangrove swamps, ponds, and inland marshes in an effort to create more arable land, residential and commercial building sites, and to dispose of unwanted waste material. Records indicate that at the beginning of the 20th century approximately 169.2 ha, or roughly 3% of Bermuda's total land area, comprised wetlands which included 20.4 ha of mangrove swamp, 29.6 ha of ponds and 119.2 ha of inland marsh. By 1980, these wetland areas had been reduced to an estimated 94.3 ha (16.7 ha of mangrove swamp, 29.2 ha of ponds, and 48.4 ha of inland marsh), representing a decrease of 44.5% (Sterrers and Wingate, 1981) and contributing to major losses of biodiversity in those areas (Sterrers, 1998; Sterrer et al., 2004). Widespread drainage of the inland marshes was employed as part of the mosquito control methods in the first half of the 20th century, as health officials attempted to prevent the spread of mosquito borne diseases. Furthermore, wetlands across the islands of Bermuda were historically used to dispose of domestic trash, thereby filling many of them and creating toxic conditions as chemicals slowly leached out over time (Sterrers and Wingate, 1981; Fort et al., 2006; Fort et al., 2006). During the period when the most intensive efforts were being made towards marsh reclamation (1920-1970) by the Bermuda Government, that had assumed the responsibility for garbage collection and a policy of disposing it in the wetlands, nearly 60% of the inland marsh habitat was lost and at least five ponds (three of which were in close proximity to the present day diamondback terrapin ponds - see Figs. A1.1 and A1.2 in Appendix 1) were completely filled in (Sterrers and Wingate, 1981). However, it has been suggested that the most concentrated destruction of Bermuda's wetland communities occurred between 1941 and 1943, when an estimated 33% of the islands' total mangrove acreage was destroyed on Longbird and St. David's Islands by the construction of the American-operated Kindley Air Force Base (Sterrers and Wingate, 1981).

However, since the 1960s, local organizations including the Bermuda Audubon Society and the Bermuda National Trust have raised funds to purchase wetland habitats, holding them in trust as nature reserves, thus ensuring some protection against further development. At the government level, protective planning legislation in 1983 designated all of the remaining wetland areas in Bermuda as nature reserves. Additional efforts have been made by a number of conservation agencies to raise public awareness regarding the ecological and aesthetic value of Bermuda's limited wetland habitats. Deliberate restoration projects have focused on the fresh-brackish marsh and pond habitats, with the end result that a variety of ponds and marshes have been physically and biologically re-established island wide.

Yet not all of the modifications to Bermuda's wetlands can be attributed to anthropogenic causes. Storm damage and other natural processes including ecological succession, sedimentation, and erosion have altered – and will continue to alter – Bermuda's wetland habitats. Close examination of maps produced over a 300 year period (see Figs. A1.3 – A1.7 in Appendix 1), beginning in the 17th century, illustrate the changing character of Mangrove Lake. John Speed published a map of the Somer (Bermuda) Islands in 1627 that shows a number of discrete bodies of water south of Harrington Sound including the present day diamondback terrapin ponds (i.e. Mangrove Lake, Trott's Pond, South Pond and North Pond) as distinct land-locked ponds (Fig. 1.11). Those bodies of water were still evident as ponds in a map produced by Henry Durnford in 1793 (Fig. 1.12); however, a map created by Thomas Hurd and published in 1797 clearly shows Mangrove Lake with a breach to the sea in the south west corner of the pond (Fig. 1.13). The Hurd map also shows this entrance as having a sandy substrate, giving the impression that oceanic water could inundate this area and access the interior of the pond perhaps during periods of high tide or stormy weather. Richard Nelson's map published in 1840, and a map engraved by Edward Weller and published in 1870, both still show Mangrove Lake as an embayment along the southern shoreline of Bermuda rather than as a distinct pond (Figs. 1.14 and 1.15). However, in the comprehensive ordnance survey performed by Arthur Savage in the years

1897-1899, which resulted in the publication of a highly detailed and accurate map of the Bermuda Islands in 1901, Mangrove Lake was once again depicted as a land-locked pond, cut-off from the southern shore by a neck of land and the Military Road (now known as the Southshore Road) (Fig. 1.16). Aerial photographs of Bermuda taken in 1940 and in 2003 (see Figs. A1.1 and A1.2 in Appendix 1) clearly show the ponds as discrete land-locked bodies of water (including Mangrove Lake).

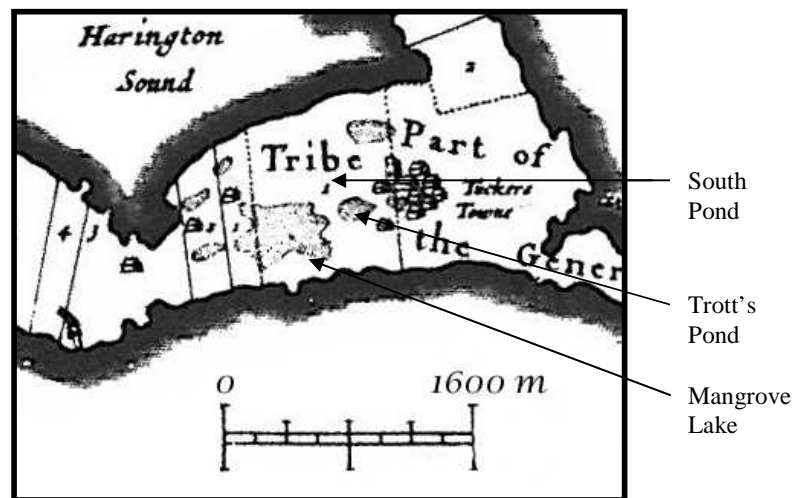


Figure 1.11. John Speed's 1627 map of the Somer (Bermuda) Islands showing Mangrove Lake, Trott's Pond and South Pond.

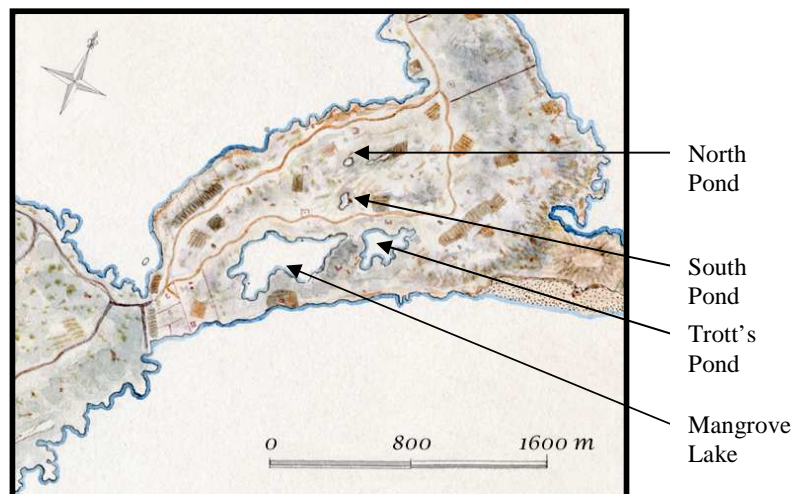
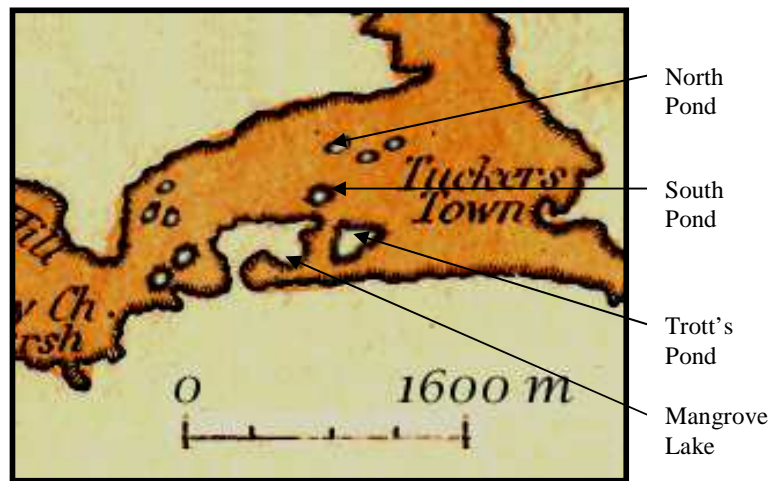


Figure 1.12. Captain Henry Durnford's (Royal Engineer) 1793 map of the Bermuda Islands showing Mangrove Lake, Trott's Pond, South Pond and North Pond.



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Figure 1.13. Lieutenant Thomas Hurd's (Royal Navy) 1797 map of the Bermuda Islands showing Mangrove Lake, Trott's Pond, South Pond and North Pond.



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Figure 1.14. Lieutenant Richard Nelson's (Royal Engineer) 1840 map of the Bermuda Islands showing Mangrove Lake, Trott's Pond, South Pond and North Pond.

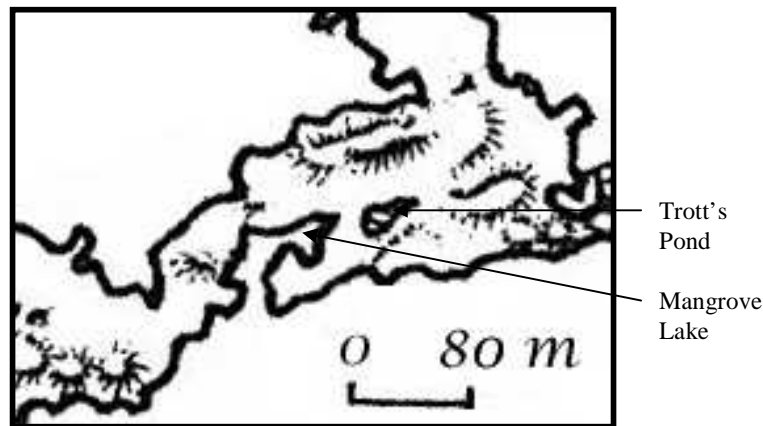
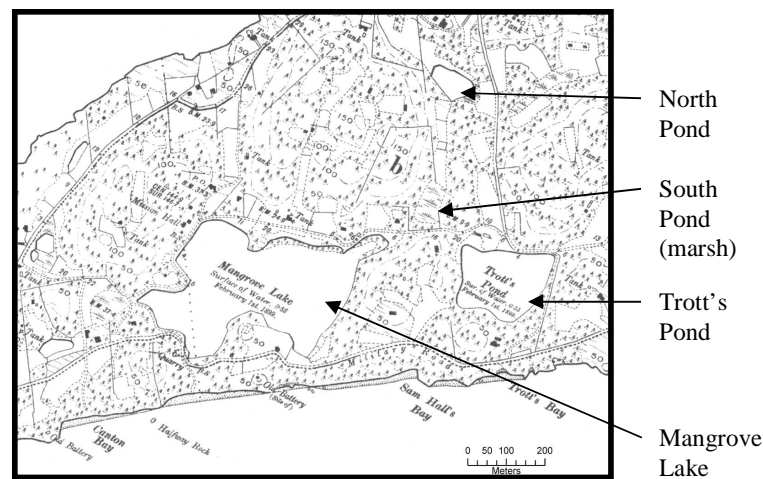


Figure 1.15. Edward Weller's 1870 map of the Bermuda Islands showing Mangrove Lake and Trott's Pond.



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Figure 1.16. Lieutenant Arthur Savage's (Royal Engineer) 1901 map of the Bermuda Islands showing Mangrove Lake, Trott's Pond, South Pond (marsh) and North Pond.

Further examination of the literature published in the mid 19th and early 20th centuries on Bermuda's natural history reveals a possible explanation for the changing nature of Mangrove Lake. Agassiz (1895) wrote, "The shore of the island is gradually being eaten away at all the low points leading either into sinks like those of Sinky or Hungry Bay, or into more elongated sinks like those which will be formed when the ponds lying close to the shore to the westward of Tucker's Town are invaded by the sea." He continued by saying, "The lagoons of the south shore between Tucker's Town and

Newton (John Smith's) Bay are brackish pools...the shores of which are protected by mangroves...(and) separated by low hills from the sea. In many places it would require comparatively slight inroads of the sea, or but a little subsistence, to change them into diminutive harbors or sounds." Verrill (1902) described the following about Peniston's (Spittal) Pond, "In severe storms the sea pours in large quantities over the low divide into Peniston's Pond...so that ultimately, and at no distant time, it will doubtless form a breach and thus convert the pond into a bay or harbour, like Hungry Bay and many others." In his 1903 publication *Zoology of the Bermudas* Verrill compared the early survey maps of Bermuda with the most recent Admiralty charts of the time, stating "in some cases small bays or coves have been converted into lagoons by the formation of sandbars across the mouth. In other cases such bars have been washed away, converting a small lagoon into an open cove. These phenomena are common on all sandy shores, and may take place during a single severe storm." Such a storm may well have converted Mangrove Lake from a pond into a bay during the last decade of the 18th century.

In September 2003, Bermuda was hit by a category 3 hurricane (named Fabian) producing a one-minute average wind speed of 120 mph (195 km h⁻¹), while a peak wind gust of 164 mph (264 km h⁻¹) was recorded. Extremely large waves were also associated with the hurricane, which battered the southern portion of the island for several days, reaching heights of seven to ten metres at the worst of the storm. While passing the island, the hurricane produced a storm surge exceeding three metres in height. Powerful storms, such as Hurricane Fabian, would be more than capable of creating the conditions described by Agassiz (1895) and Verrill (1902, 1903).

Natural forces may also have been responsible for subsequently occluding the breach in Mangrove Lake. Jones (1859) wrote, "on the south shore (of Bermuda) the sand has made several encroachments supported by constant supplies from the sea", and the works published by the *Challenger* expedition in 1877 mention a "sand glacier" at Elbow Beach. The 'glacier' was described having entirely filled up a valley and "is steadily progressing inland in a mass about five-and-twenty feet thick...(having) partially overwhelmed a garden and is moving slowly on." The account continued to

describe how a cottage was almost entirely covered by the shifting beach sands; “all that now remains of the cottage is the top of one of the chimneys projecting above the white sand like a tombstone.” If the natural deposition of sand blocked the mouth of the embayment to Mangrove Lake then it is not unlikely that the persons responsible for the construction of the roadways across Bermuda might have capitalized on the event and built the Military Road illustrated in Arthur Savage’s 1901 map (Fig. 1.16).

A final event took place in the vicinity of the diamondback ponds thereby significantly changing the ecology of the area. In 1921 the Furness Withy Steamship Company constructed a hotel and an 18-hole golf course, named the Mid Ocean, incorporating the existing ponds, mangrove swamps and peat marshes that naturally existed in the vicinity into the play area (Figs. 1.17 and 1.18). Prior to the construction of the golf course, the valleys surrounding the diamondback terrapin ponds had been used for agricultural purposes. A downturn in the global economy in the aftermath of the Second World War forced the Steamship Company to divest itself of the hotel and golf course, which were purchased by a group of Bermudians who then established The Mid Ocean Club, as a private members’ club, in 1951.



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Figure 1.17. Fifth hole on the Mid Ocean golf course showing the pedestrian bridge and a portion of Mangrove Lake (circa 1930).



Source: Mark Outerbridge

Figure 1.18. Fifth hole on the Mid Ocean golf course showing the pedestrian bridge and a portion of Mangrove Lake (2011).

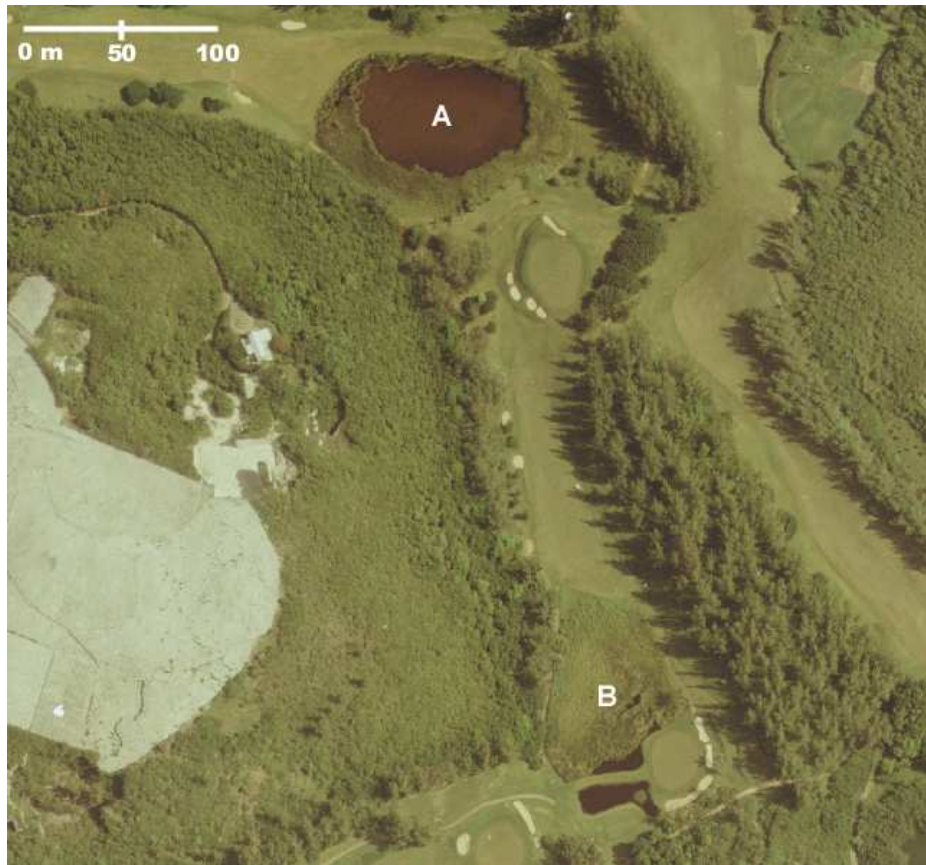
The golf course has undergone some slight re-modelling since the 1950s, leaving Mangrove Lake and Trott's Pond less affected than North Pond and South Pond. A 1941 aerial photograph shows North Pond as a distinct body of water and South Pond as a marsh with no open water visible (Fig. 1.19); however, a 1981 aerial photograph of the same area shows North Pond as a slightly reduced body of water (due to expansion of the marsh grass community) and the southern portion of the South Pond marsh bisected from west to east by a land bridge with two small bodies of water to the north and south (Fig. 1.20). In 1980, the North Pond marshland was comprised of 100% cattail (*Typha angustifolia*) and the South Pond marshland comprised 57% cattail and 43% saw-grass (*Cladium jamaicense*) (Sterrer and Wingate, 1981). South Pond was dredged for a second time circa 1993 during which the present day moat that constitutes South Pond major was constructed, while South Pond minor was deepened. The excavation material was observed to be partially composed of old bottles and other forms of refuse (N. Furtado, personal communication). A 2003 aerial photograph clearly shows the changes made to South Pond in the early 1990s (Fig. 1.21). North Pond also shows some change, with a greatly reduced marshland community. In an effort to maintain a constant high water level throughout the year, the Mid Ocean Club drilled a well beside North Pond in the late 1990s and began pumping water from the well into

the pond. While this had the desired effect on the water level, it had the unforeseen effect of causing the cattail marsh to die-off completely, as it was later discovered that the well-water was considerably more saline than cattails are capable of surviving. The marshland in North Pond is presently comprised 100% of sheathed paspalum grass (*Paspalum vaginatum*) and the South Pond major marshland is presently composed of approximately 97% saw-grass, 2% cattail, while the remaining 1% of the vegetation is a mixture of sheathed paspalum, giant fern (*Achrosticum excelsum*), wax myrtle (*Myrica cerifera*), Australian pine (*Casuarina equisetifolia*) and Brazil pepper (*Schinus terebinthifolius*) (M. Outerbridge, personal observation). In 2008, a small saline pond complete with three islets was created via excavation immediately to the east of North Pond in an effort to increase the aesthetic appeal of the ninth hole.



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Figure 1.19. Aerial photograph from 1941 showing North Pond (A) and South Pond (B).



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Figure 1.20. Aerial photograph from 1981 showing North Pond (A) and South Pond (B).



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Figure 1.21. Aerial photograph from 2003 showing North Pond (A) and South Pond (B).

Objectives of This Study

Much has been learned about the life history of diamondback terrapins in the North American range during the past four decades; however there is a dearth of information regarding the status, biology and ecology of diamondback terrapins on the islands of Bermuda, despite having had a presence there for over 400 years (Parham et al., 2008). This has constrained conservation efforts and has limited the ability to make informed management decisions for the species. While it could be assumed that the ecology of diamondback terrapins in Bermuda would be similar to that of its counterparts in the southern regions of the U.S.A., Bermuda's population differs in two fundamental ways: (1) it resides exclusively within a limited number of land-locked, brackish water ponds (rather than the brackish coastal environments that are typical along the Atlantic Ocean and the Gulf

of Mexico), (2) it is situated upon a private golf course that is heavily impacted by anthropogenic activities. This study therefore aimed to gain a greater understanding of the ecology of this species in Bermuda, to compare this with what is known from other areas of its range, and hence to inform the development of a management plan detailing the short-term and long-term survival goals for the species on Bermuda.

Overview of Thesis

The thesis contains five data chapters focusing on different aspects of the biology and ecology of Bermuda's isolated population of diamondback terrapins.

Chapter 2 describes the physical and biological characteristics of the four land-locked, brackish water ponds inhabited by Bermuda's population of diamondback terrapins.

Chapter 3 is a comprehensive demographic assessment of the population. This chapter provides, for the first time, an estimate of population size and describes various aspects of its structure (sex ratio, size classes, annual recruitment rates, density, and terrapin biometrics). In particular, its goals were:

- (1) to collect baseline data for future population monitoring,
- (2) to provide evidence-based data to the Government of Bermuda to advocate the legislative protection of diamondback terrapins,
- (3) to determine if Bermuda's diamondback terrapin population is vulnerable to local extirpation,
- (4) to determine if other methods of terrapin capture could be devised to increase the published catch-per-unit-effort.

Chapter 4 examines the feeding ecology of diamondback terrapins in Bermuda using direct observation, necropsies, faecal analyses and gastric lavage. The diet of diamondback terrapins has been studied in various regions throughout their North American range; however, nothing is known about their diet on Bermuda. It was envisaged that detailed knowledge of terrapin diet in Bermuda would allow appropriate conservation and management efforts to be directed towards protecting the areas in which

they forage. The primary goal was to examine the diet and foraging ecology of Bermuda's terrapin population, with specific aims to:

- (1) determine particular food preferences within the land-locked, brackish water pond environment,
- (2) to provide insight into any behavioural foraging adaptations that diamondback terrapins might display within this environment,
- (3) to assess the abundance and distribution of gastropods within the ponds and adjacent wetland communities.

Chapter 5 describes the nesting ecology of diamondback terrapins in Bermuda. This population appear to be the only wild breeding population outside the North American range, and anecdotal evidence suggests that this species has been nesting for many years in an artificial habitat (sand bunkers on the golf course). Quantitative assessments of nesting activity in Bermuda are lacking, and knowledge of reproductive output is needed in order to perform population modelling. The main goals of this chapter were:

- (1) to determine the frequency of sand bunker nesting in Bermuda,
- (2) to determine the duration of the nesting season,
- (3) to describe clutch size, egg morphology and hatchling biometrics,
- (4) to establish the incubation and emergence periods,
- (5) to report hatching success rates for Bermuda's diamondback terrapin population.

Following consideration of some findings of Chapters 3 and 4, during which yellow-crowned night herons (*Nyctanassa violacea*) were identified as predators upon small diamondback terrapins in Bermuda and the annual rate of recruitment to the adult terrapin population was observed to be very low, Chapter 6 investigates the post-emergent movements and survival of diamondback terrapin hatchlings in Bermuda for one month following departure from their natal nests (using radio telemetry). Specifically, the aims were:

- (1) to quantify the level of mortality during the period when hatchling terrapins may be most vulnerable to predation,
- (2) to identify areas of residency for terrapin hatchlings and small juveniles,

(3) to compare hatchling activity levels and movement patterns between those emerging in summer and those emerging in spring.

Chapter 7 examines whether petroleum hydrocarbons, polycyclic aromatic hydrocarbons and heavy metals are being bioaccumulated by Bermuda's diamondback terrapins. Recent investigations of the health status of the pond environment in Bermuda suggest that there is a suite of contaminants of concern that are having detrimental effects (e.g. inducing developmental malformations, endocrine disruption and immunological stress) on a range of taxa which constitute the resident fauna (Fort et al., 2006; Fort et al., 2006; Bacon, 2010; Bacon et al., 2012). Given this earlier research, it seemed possible that terrapins in Bermuda might be negatively affected by such contaminants which could put the population at risk. The specific objectives of this chapter were to examine the levels of toxic contaminants in benthic sediments in water bodies where diamondback terrapins have been recorded, as well as in aquatic gastropods on which they feed, and also in terrapin eggs. Examining the extent to which Bermuda's diamondback terrapins are impacted by contaminants, and how this influences survival, is critical to the design of appropriate management initiatives and wetland remediation activities.

Chapter 8 considers the main discoveries of this research, identifies their implications for the continued survival of diamondback terrapins on Bermuda and suggests ways in which research can be expanded in the future.

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Chapter 2: Description of the Study Site

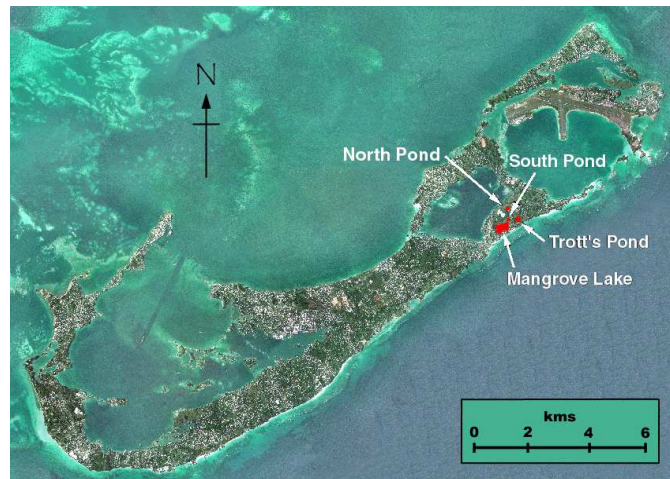
Abstract

Bermuda's native population of diamondback terrapins inhabit four brackish water ponds located on one square kilometre of private land. Three of the ponds are naturally occurring and one is man-made. All are relatively shallow and have bottoms comprised of deep organic sediment. Water temperature and salinity show great variability depending upon the season. The wetlands associated with the larger of the two ponds are dominated by red mangrove trees while the smaller two ponds have marshes comprised exclusively of grasses. All four ponds have been incorporated into a golf course since the 1920s and are therefore heavily impacted by anthropogenic activities.

General overview of the diamondback terrapin ponds

Despite the presence of hundreds of pools and ponds on Bermuda, the entire population of diamondback terrapins (*Malaclemys terrapin*) is found in only four brackish water ponds on a private golf course, the Mid Ocean Club, located in Smith's Parish at the eastern end of the islands; Mangrove Lake, South Pond, North Pond, and Trott's Pond (Fig. 2.1) (M. Outerbridge, personal observation). Mangrove Lake and Trott's Pond are the largest of these ponds and both are simple basins fringed by red mangrove trees (*Rhizophora mangle*) and characterized by shallow depths with bottoms comprised of deep deposits of highly organic sediment (Thomas et al., 1991). North Pond and South Pond are considerably smaller in area, shallower in depth, and lack mangrove vegetation; however both have small marshes in their centres dominated by grasses. All four bodies of water are situated upon a single square kilometre of Bermuda and are only separated from each other by, at most, 380 m of land (straight-line distance between North Pond and Trott's Pond). All four ponds have been incorporated into the golf course as water hazards found between the fifth and eleventh holes. No other diamondback terrapins have been discovered in any other bodies of water on Bermuda despite a series of extensive surveys of its wetland communities conducted between 2004 and 2007 (Outerbridge et al., 2007;

Outerbridge, 2008). Mangrove Lake, Trott's Pond and North Pond have been designated as 'nature reserves' under the 2008 Bermuda Development Plan; however, South Pond is currently zoned as a 'recreational area' (see Fig. A2.5 in Appendix 2).



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Figure 2.1. Aerial photograph of Bermuda showing the location of the four diamondback terrapin ponds. (Modified from an aerial map of the Bermuda Islands).



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Figure 2.2. Aerial photograph from 2003 showing the four diamondback terrapin ponds situated on the Mid Ocean golf course (A=Mangrove Lake, B=Trott's Pond, C=South Pond, D=North Pond).

Mangrove Lake

Mangrove Lake (Fig. 2.3 below and Fig. A2.1 in Appendix 2), is presently a simple basin that is 9.89 hectares (ha) in area (calculated in ArcGIS 9.0 using a 2003 digitized aerial orthophotograph of the Bermuda islands) and characterized by shallow depths, averaging only 134 cm (maximum depth 223 cm; minimum depth, and standard deviation not reported) (Thomas et al., 1991), fairly even contours and a gently sloping shoreline (Fig. 2.4).

Mangrove Lake is currently the largest of Bermuda's anchialine ponds and is believed to have formed during the last 11,000 years (Watts and Hansen, 1986). The pond bottom comprises deep deposits of gelatinous, sapropelic sediment from which patches of widgeon grass (*Ruppia maritima*) grow in dense clumps. Stratigraphic evidence indicates that Mangrove Lake's sedimentary environment has undergone three major depositional changes over time as a result of sea level changes; peat, freshwater gel, and brackish water gel (Hatcher et al., 1982). Thomas et al. (1991) reported that the present-day sediment comprises a matrix of mostly silt-clay, organic mud and detritus. Mangrove Lake is often subject to considerable changes in oxygen, redox potential, temperature, salinity and nutrient levels (Thomas et al., 1991). A few small subterranean fissures ensure that ocean water still enters this pond from the south shore; however, there is a very low flushing rate (calculated from the mean tidal exchange as a percentage of low tide volume) of 1% as well as a small tidal range of 1.4 cm (Thomas et al., 1992). Water temperatures were reported to range from 20-29.1°C (mean 24.6°C; SD 3.4°C); surface salinities (measured using an optical refractometer) from 27-33 psu (mean 29.7 psu; SD 2.2 psu) from 1980-1989 (Thomas et al., 1991). More recent data are reported below.

The pond is surrounded by a mangrove swamp totalling 2.3 ha in area and dominated almost exclusively by red mangrove trees that reach heights of 8 m (Thomas, 1993) (Fig. 2.5). The mean width of this fringing swamp is approximately 12 m; however it does attain a width of over 60 m in the NE and SW corners of the pond (M. Outerbridge, personal observation). The sediment within the swamp is high in organic content resulting primarily from leaf-fall and decay and is inhabited by a number of different invertebrate species (Thomas et al., 1992).

Mangrove Lake and the surrounding land are owned by a variety of private individuals and organizations. The pond is mostly owned by the Tucker's Point Club, although this club has recently offered to donate its entire holding to the Bermuda Government to be held as a nature reserve. The owners of the surrounding land include the Mid Ocean Club (which owns the land adjacent to the northern shoreline of the pond), the Bermuda National Trust (which owns the land bordering the western end of the pond), and a number of private individuals who live along the southern and eastern shorelines of the pond.



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Figure 2.3. Aerial photograph from 2005 showing Mangrove Lake.

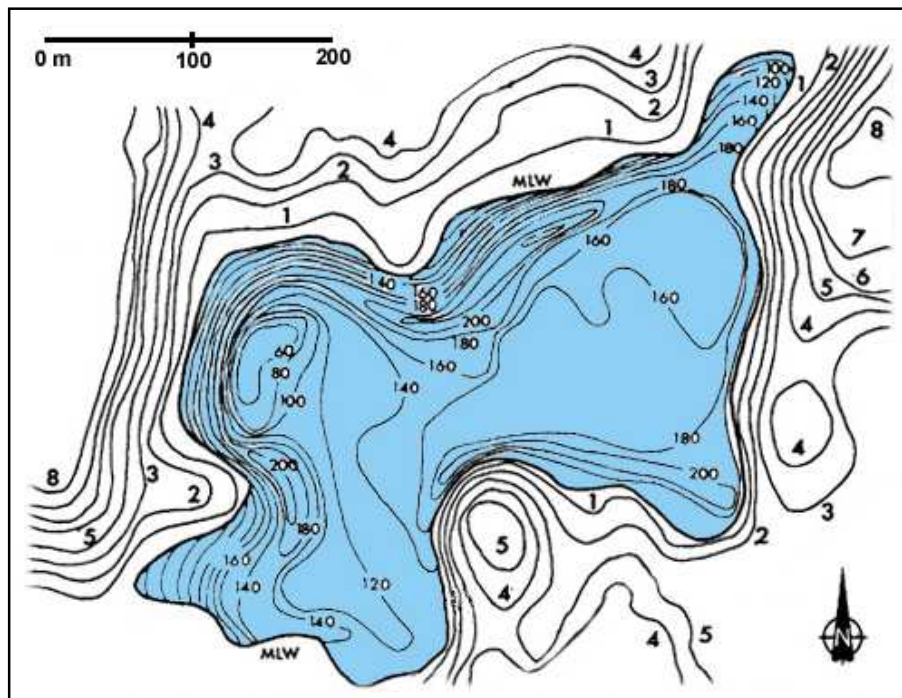


Figure 2.4. Bathymetry and surrounding topography of Mangrove Lake. Depth contours are shown in centimetres below mean low tide level and height contours are shown in metres. (Adapted from Thomas et al., 1991).

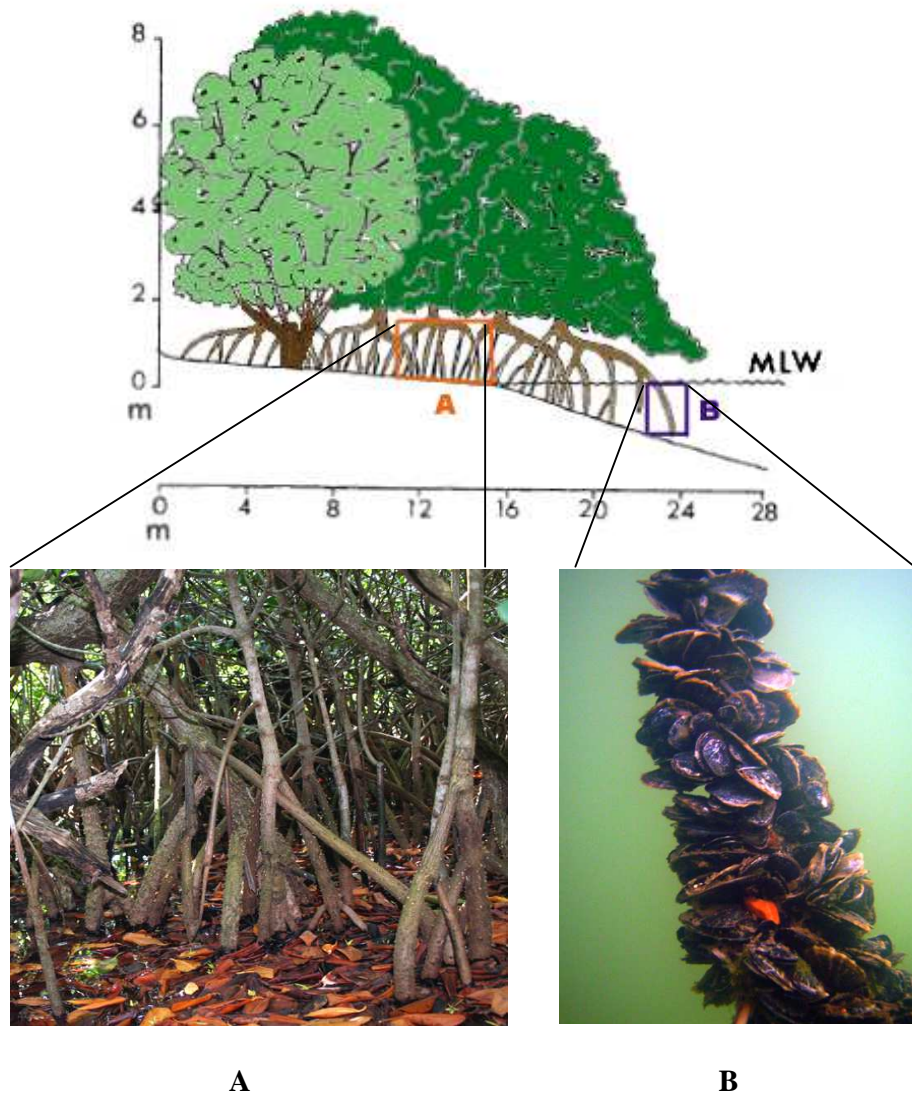


Figure 2.5. Cross section diagram of the mangrove swamp profile in Mangrove Lake. MLW=mean low water. The light green tree represents the transition zone from terrestrial to swamp community and comprises species such as the introduced and invasive Brazil pepper tree (*Schinus terebinthifolia*); the dark green tree represents the mangrove community (e.g. native red mangrove *Rhizophora mangle*); box A shows the adventitious mangrove prop root and benthic leaf-litter communities of the intertidal zone; box B shows a cluster of mangrove oysters (*Isognomon alatus*) growing on a mangrove prop root below water level. (Adapted from Thomas, 1993).

Trott's Pond

Trott's Pond (Fig. 2.6 below and Fig. A2.2 in Appendix 2) is situated less than 300 m to the east of Mangrove Lake. It is 2.88 ha in area (calculated in ArcGIS 9.0 using a 2003 digitized aerial orthophotograph of the Bermuda islands) and formed between low Pleistocene sand dunes that were inundated by postglacial seas. During interglacial periods of low sea level, freshwater slowly eroded away the depression creating fissures through which saltwater entered from the south shore as the sea level rose around Bermuda. Trott's Pond is currently a simple basin characterized by fairly shallow depths, with the deepest part at its centre. It has fairly even contours and a gently sloping shoreline (Thomas et al., 1992). The connection to the ocean is small and located near the surface. This connection gives Trott's Pond a very low flushing rate (calculated from the mean tidal exchange as a percentage of low tide volume) of 0.5% and a small tidal range of 1.5 cm per tidal cycle. Rainfall and surface runoff from the surrounding area usually do not mix with the saltwater below, but instead float as a distinct layer on top, eventually draining off through the surface connection (Thomas, 2002). The pond has a bottom that comprises deep deposits of highly organic sediments consisting of a matrix of mostly sand, silt-clay, organic mud and detritus (Thomas et al., 1991), and is surrounded by a mangrove swamp totalling 0.8 ha in area dominated almost exclusively by red mangrove trees (Thomas, 1993). The mean width of this fringing swamp is less than 10 m; however it does attain a width of approximately 30 m in the SW corner of the pond (M. Outerbridge, personal observation). The mean depth in Trott's Pond was reported to be 269 cm; the maximum was 320 cm (minimum depth, and standard deviation not reported) (Fig. 2.7). Water temperatures were reported to range from 16-31°C (mean 24.5°C; SD 4.8°C) and surface salinities varied from 23.5-33.5 psu (mean 27.4 psu; SD 2.6 psu) from 1980-1989 (Thomas et al., 1991).

The sediment within the swamp surrounding Trott's Pond is high in organic content, resulting primarily from leaf-fall and decay and is inhabited by a number of different invertebrate species. In fact, Trott's Pond shares at least twelve species in common with the neighbouring Mangrove Lake, including the mangrove oyster, the coffee bean marsh snail (*Melampus*

coffeus), the Bermuda killifish (*Fundulus bermudae*), and the diamondback terrapin (Thomas et al., 1992).



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Figure 2.6. Aerial photograph from 2005 showing Trott's Pond.

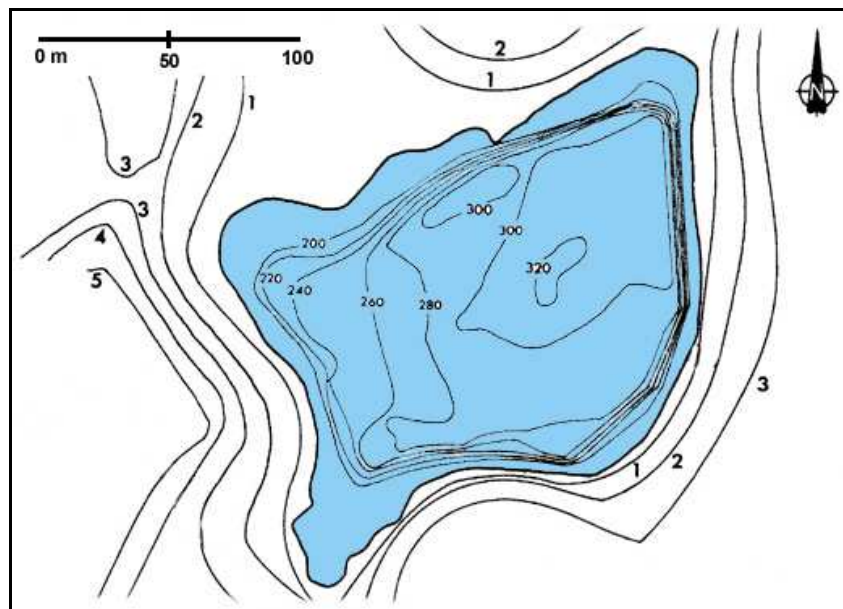


Figure 2.7. Bathymetry and surrounding topography of Trott's Pond. Depth contours are shown in centimetres below mean low tide level and height contours are shown in metres. (Adapted from Thomas et al., 1991).

South Pond

South Pond (Fig. 2.8 below and Fig. A2.3 in Appendix 2) lies to the north of Mangrove Lake and Trott's Pond, virtually equidistant from both bodies of water. It is much smaller than both ponds, having been deliberately dredged to create a golf course water hazard during the 1990s (see Chapter 1). A land bridge separates this pond into two distinct bodies of water; the moat-like pond to the north (South Pond major) and a small pond to the south (South Pond minor). However, these two bodies of water are collectively known as 'South Pond', unless otherwise stated. South Pond totals approximately 0.45 ha in area (major and minor combined), of which 0.27 ha comprises the central saw-grass (*Cladium jamaicense*) marsh. (Note: all areas were calculated in ArcGIS 9.0 using a 2003 digitized aerial orthophotograph of the Bermuda islands). The salinity of South Pond is much lower than in neighbouring Mangrove Lake and Trott's Pond, and varies seasonally between the major and minor ponds (see Table 2.3 below). Mangrove trees are not present at this site, but the small marsh located in the centre of South Pond major comprises mostly saw-grass, and to a lesser extent cattail (*Typha angustifolia*). The emergent vegetation that grows around the perimeter of South Pond minor is exclusively sheathed paspalum (*Paspalum vaginatum*) which is periodically trimmed by the course maintenance staff of the Mid Ocean Club. Widgeon grass grows seasonally within South Pond, and the pond bottom is comprised of highly organic sediment. The mean depth in South Pond major is 36 cm (range 20-52 cm; SD 8.7 cm) while South Pond minor averages 89 cm (range 30-122 cm; SD 23 cm) (M. Outerbridge, unpublished data). The water levels in South Pond vary considerably according to the amounts received through rainfall. In periods of drought it is not uncommon for areas of South Pond major to be reduced to depths of < 5 cm, or even to dry up completely (M. Outerbridge, personal observation) (Fig. 2.9). Conversely, during periods of very heavy rainfall the water level rises, floods the saw-grass marsh and spills over on to the surrounding land (Fig. 2.10).

South Pond is used by a variety of waterbirds that include both resident species (e.g. yellow-crowned night heron (*Nyctanassa violacea*)), moor hen (*Gallinula chloropus*) and migratory species (e.g. American coot

(*Fulica americana*), American bittern (*Botaurus lentiginosus*)). It is also inhabited by another emydid turtle; the red-eared slider (*Trachemys scripta elegans*), which was introduced to Bermuda via the pet trade by the mid 20th century. This species has established feral populations in at least 20 fresh and slightly brackish water ponds throughout Bermuda and was found to be living in densities estimated to be as high as 981 turtles ha⁻¹ (Outerbridge, 2008). Between 2005 and 2009, 86 red-eared sliders (ten juveniles, 20 males, and 56 females) ranging in size from 84-237 mm straight carapace length (SCL) were captured and permanently removed from South Pond (M. Outerbridge, unpublished data). The ecological impact that feral red-eared sliders have on native chelonians in other regions has been reported by others (Hays et al., 1999; Cadi and Joly, 2003; Spinks et al., 2003; Cadi and Joly, 2004); however, the impact on the Bermudian population of diamondback terrapins is currently unknown. Further studies are needed to determine whether diamondback terrapins are being negatively affected by this introduced species.



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Figure 2.8. Aerial photograph from 2005 showing South Pond. South Pond major is the top body of water and South Pond minor is the bottom body of water.



Source: Mark Outerbridge

Figure 2.9. South Pond major during a drought. Note the diamondback terrapin tracks in the mud on the right. These tracks show that an individual entered from South Pond minor but turned around shortly thereafter and departed.



Source: Mark Outerbridge

Figure 2.10. South Pond major during a flood.

North Pond

North Pond (Fig. 2.11 below and Fig. A2.4 in Appendix 2) is located some 220 m north of South Pond and is approximately 0.4 ha in area (calculated in ArcGIS 9.0 using a 2003 digitized aerial orthophotograph of the Bermuda islands). Although it is a naturally occurring pond, it has also been periodically dredged over the years since the construction of the golf course. The salinity in this pond is slightly lower than in the neighbouring Mangrove Lake and Trott's Pond, but higher than that of South Pond (Table 2.3). Mangrove trees are also not present at this site; however, there are five small islets located in the pond which are dominated by sheathed paspalum. As with the three other diamondback terrapin ponds, North Pond's bottom is comprised of highly organic sediment. The mean depth of water is 22 cm (range 10–48 cm; SD 8.4 cm) (M. Outerbridge, unpublished data). The water levels in North Pond vary considerably, according to the amounts received through rainfall. The pond is greatly reduced in area during periods of drought (Fig. 2.12), when a great deal of this wetland is converted to a mud flat which is frequented by foraging birds. During periods of heavy rainfall the water level rises, floods the grass marshes and may even spill over on to the surrounding golf course (Fig. 2.13) (M. Outerbridge, personal observation).



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Figure 2.11. Aerial photograph from 2005 showing North Pond.



Source: Mark Outerbridge

Figure 2.12. North Pond during a drought.



Source: Mark Outerbridge

Figure 2.13. North Pond during a flood.

Water temperatures and salinities of the diamondback terrapin ponds

The surface salinities in all four ponds were recorded between 2009 and 2011, and temperatures were recorded in South Pond major (2009-2011) and Mangrove Lake (2010 and 2011 only). Temperature was recorded every six hours using permanently deployed digital HOBO pendant data loggers (model # UA-002-08 from Onset Computer Corporation) suspended in the middle of each water body and salinity was measured at a depth of 10 cm in three separate locations within each pond on a bi-monthly basis, whenever possible, using an optical refractometer.

The mean monthly mid-water temperatures for South Pond major between 2009 and 2011 are summarised in Table 2.1. The 2009 mean temperature was 23.7°C and the mean range was 17°C (February)-30.6°C (August); however the coldest temperature occurred in January (12.2°C) while the warmest occurred in July (38.9°C). The 2010 mean was 24.7°C (based upon an eight month period) with a mean monthly range of 18.1°C (January)-31.4°C (July); the coldest temperature occurred in March (11.6°C) while the warmest occurred in June (39°C). The 2011 mean was 23.3°C with a mean monthly range of 15.7°C (February)-29.9°C (August); the coldest temperature occurred in February (12°C) while the warmest occurred in July (35.1°C) (Fig. 2.14).

The mean monthly mid-water temperatures for Mangrove Lake in 2010 and 2011 are summarised in Table 2.2. The 2010 mean was 23.5°C (based upon an eight month period) with a mean monthly range of 15.6°C (February)-30.6°C (July); the coldest temperature occurred in February (13.7°C) while the warmest occurred in July (33.6°C). The 2011 mean was 23.5°C (also based upon an eight month period) with a mean monthly range of 22.1°C (December)-33.5°C (July); the coldest temperature occurred in December (16.3°C) while the warmest occurred in July (33.5°C) (Fig. 2.14).

Table 2.1. (left) Minimum, maximum and mean monthly mid-water temperatures for South Pond major from January 2009 to December 2011 (db=dead battery).

Table 2.2. (right) Minimum, maximum and mean monthly mid-water temperatures for Mangrove Lake from January 2010 to December 2011 (ns=not sampled, db=dead battery).

South Pond major	min (°C)	max (°C)	mean (°C)
Jan-09	12.2	23.8	17.2
Feb-09	12.4	22.9	17.0
Mar-09	13.2	28.5	19.6
Apr-09	16.0	31.6	22.0
May-09	18.8	33.9	24.7
Jun-09	23.2	36.6	28.2
Jul-09	24.2	38.9	30.5
Aug-09	26.7	36.3	30.6
Sep-09	25.1	33.9	28.9
Oct-09	20.6	29.1	25.2
Nov-09	15.8	25.1	20.9
Dec-09	15.8	24.7	19.7
Jan-10	13.1	23.8	18.1
Feb-10	15.6	26.5	20.6
Mar-10	11.6	28.9	19.8
Apr-10	14.6	31.7	23.4
May-10	18.2	31.3	24.2
Jun-10	21.9	39.0	29.6
Jul-10	25.4	38.5	31.4
Aug-10	24.7	37.0	30.3
Sep-10	db	db	db
Oct-10	db	db	db
Nov-10	db	db	db
Dec-10	db	db	db
Jan-11	12.4	23.3	18.4
Feb-11	12.0	20.7	15.7
Mar-11	13.1	26.1	19.0
Apr-11	14.2	27.9	21.5
May-11	17.5	30.3	24.6
Jun-11	19.3	34.4	27.7
Jul-11	24.3	35.1	29.6
Aug-11	25.9	34.8	29.9
Sep-11	25.6	32.7	28.7
Oct-11	18.7	29.4	24.9
Nov-11	16.9	25.0	21.3
Dec-11	13.4	23.3	18.7

Mangrove Lake	min (°C)	max (°C)	mean (°C)
Jan-09	ns	ns	ns
Feb-09	ns	ns	ns
Mar-09	ns	ns	ns
Apr-09	ns	ns	ns
May-09	ns	ns	ns
Jun-09	ns	ns	ns
Jul-09	ns	ns	ns
Aug-09	ns	ns	ns
Sep-09	ns	ns	ns
Oct-09	ns	ns	ns
Nov-09	ns	ns	ns
Dec-09	ns	ns	ns
Jan-10	16.3	18.3	17.2
Feb-10	13.7	18.0	15.6
Mar-10	14.8	22.4	18.6
Apr-10	18.3	24.7	21.6
May-10	20.7	29.2	25.3
Jun-10	24.6	32.7	29.1
Jul-10	28.1	33.6	30.6
Aug-10	27.8	32.4	29.7
Sep-10	db	db	db
Oct-10	db	db	db
Nov-10	db	db	db
Dec-10	db	db	db
Jan-11	db	db	db
Feb-11	db	db	db
Mar-11	db	db	db
Apr-11	db	db	db
May-11	25.1	29.5	27.6
Jun-11	25.4	32.5	28.0
Jul-11	26.1	33.5	30.4
Aug-11	27.8	32.3	30.0
Sep-11	27.0	31.5	29.1
Oct-11	21.7	29.1	25.6
Nov-11	19.4	25.0	21.9
Dec-11	16.3	22.1	19.2

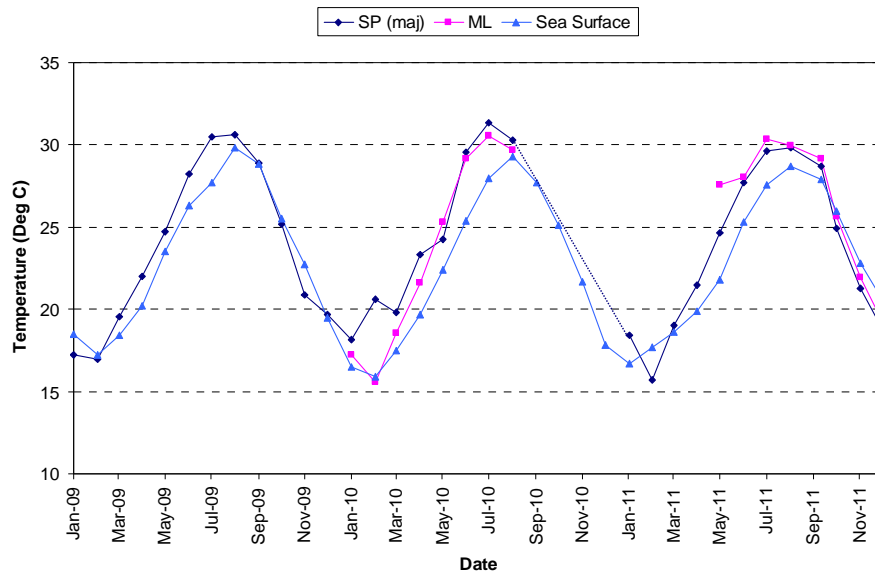


Figure 2.14. Monthly mean mid-water temperatures for South Pond (major) from 2009-2011 and Mangrove Lake from 2010-2011. Sea surface temperatures have been added for comparative purposes and were obtained from the Bermuda Weather Service. (Note: water temperature data does not exist for South Pond during the last third of 2010 or for Mangrove Lake during the last third of 2010 and the first third of 2011 because the batteries in the data loggers failed during those periods).

The mean monthly surface salinity for South Pond major, South Pond minor, North Pond, Trott's Pond, and Mangrove Lake from January 2009 to August 2011 are summarised in Table 2.3 and illustrated in Fig 2.15. Salinity varied between South Pond major and South Pond minor; with the latter being more saline throughout the study period. This increased salinity is probably linked to the fact that the dredging activities during the 1990s exposed South Pond minor to the saline water lens immediately below the pond. South Pond major is shallower and therefore less influenced by this lens. The mean salinity for 2009 in South Pond major was 3.7 practical salinity units (psu) [range 0.3 (September)-6.7 (January)], while in South Pond (minor) it was 6.3 psu [range 1.3 (April)-10.7 (January)]. In 2010 the annual mean in South Pond major increased to 10.4 psu [range 4.8 (March)-16.8 (July)], and in South Pond minor it was 13.5 psu [range 7.8 (April)-

18.7 (June)]. In 2011 the mean in South Pond major increased to 14.7 psu [range 6.3 (January)-21.3 (July)], and in South Pond minor it was 17 psu [range 12 (January)-21.3 (July)].

North Pond showed the greatest variation in monthly salinity during this period. The mean surface salinity in 2009 was 23.5 psu; range 18.7 (August)-27.7 (March). The mean salinity in 2010 was 19.1 psu, range 10 (March)-31 (July); and in 2011 the mean salinity (between January and August only) was 22.3 psu, range 13 (January)-29.3 (August).

The mean surface salinity for Trott's Pond in 2009 was 27.4 psu; range 22.5 (August)-29.7 (January). The mean salinity in 2010 was 29.3 psu, range 25.5 (March)-31.7 (July); and in 2011 the mean salinity (between January and August only) was 30.1 psu, range 27.3 (May)-31.3 (July and August).

Mangrove Lake was the most saline of the four ponds studied during this period. The mean surface salinity in 2009 was 28.3 psu; range 23.3 (September)-30.7 (April). The mean salinity in 2010 was 30.6 psu, range 26 (January)-34.3 (July); and in 2011 the mean salinity (between January and August only) was 32.8 psu, range 30.3 (April)-35.7 (August) (Fig. 2.15).

Monthly rainfall values between June 2008 and December 2011 for Bermuda were obtained from the Bermuda Weather Service (www.weather.bm; accessed in March 2014) and graphed (Fig. 5.16).

Table 2.3. Minimum, maximum, and mean monthly salinities (practical salinity units or psu) at a depth of 10 cm for South Pond major, South Pond minor, North Pond, Trott's Pond, and Mangrove Lake from January 2009 to August 2011 (ns=not sampled).

	South Pond major (psu)			South Pond minor (psu)			North Pond (psu)			Trott's Pond (psu)			Mangrove Lake (psu)		
	min	max	mean	min	max	mean	min	max	mean	min	max	mean	min	max	mean
Jan-09	6	7	6.7	10	11	10.7	26	26	26.0	29	30	29.7	29	31	30.0
Feb-09	6	7	6.3	9	10	9.7	23	23	23.0	28	29	28.7	29	29	29.0
Mar-09	4	5	4.7	8	9	8.7	27	28	27.7	29	30	29.5	30	30	30.0
Apr-09	6	7	6.3	1	2	1.3	24	24	24.0	28	28	28.0	30	32	30.7
May-09	4	5	4.7	3	3	3.0	23	23	23.0	28	28	28.0	29	29	29.0
Jun-09	1	3	2.0	5	6	5.7	22	23	22.5	26	28	27.0	29	30	29.3
Jul-09	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
Aug-09	1	2	1.7	5	6	5.7	18	19	18.7	22	23	22.5	25	28	26.7
Sep-09	0	1	0.3	5	6	5.7	ns	ns	ns	ns	ns	ns	22	24	23.3
Oct-09	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
Nov-09	1	1	1.0	6	7	6.3	22	22	22.0	25	25	25.0	26	26	26.0
Dec-09	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
Jan-10	5	7	6.0	8	8	8.0	10	13	11.7	27	27	27.0	25	27	26.0
Feb-10	6	7	10	10	10	10.0	13	13	13.0	27	28	27.3	27	29	28.3
Mar-10	3	7	4.8	9	10	9.4	9	11	10.0	25	26	25.5	27	28	27.3
Apr-10	4	7	6.0	4	11	7.8	13	16	14.5	25	28	27.0	27	30	29.0
May-10	10	17	13.4	10	21	15.8	22	30	26.8	29	31	30.0	30	32	31.0
Jun-10	12	19	16.7	18	19	18.7	27	31	28.8	28	31	30.3	32	35	34.0
Jul-10	14	20	16.8	16	20	17.3	30	33	31.0	30	33	31.7	33	36	34.3
Aug-10	8	18	12.8	15	19	17.0	17	20	18.6	29	30	29.4	30	35	32.3

Table 2.3. (continued) Minimum, maximum, and mean monthly salinities (practical salinity units or psu) at a depth of 10 cm for South Pond major, South Pond minor, North Pond, Trott's Pond, and Mangrove Lake from January 2009 to August 2011 (ns=not sampled).

	South Pond major (psu)			South Pond minor (psu)			North Pond (psu)			Trott's Pond (psu)			Mangrove Lake (psu)		
	min	max	mean	min	max	mean	min	max	mean	min	max	mean	min	max	mean
Sep-10	6	10	8.5	11	12	11.3	15	18	16.5	29	30	29.2	28	32	30.0
Oct-10	9	11	10.0	15	16	15.3	20	21	20.7	30	30	30.0	28	29	28.3
Nov-10	10	11	10.7	15	15	15.0	20	20	20.0	30	31	30.7	31	32	31.7
Dec-10	9	12	10.8	14	15	14.7	14	20	17.5	28	31	29.5	32	33	31.3
Jan-11	5	7	6.3	12	12	12.0	12	14	13.0	29	32	30.7	31	32	31.7
Feb-11	13	13	13.0	16	17	16.3	14	16	15.0	29	29	29.0	30	31	30.7
Mar-11	13	14	13.3	13	14	13.7	19	20	19.7	29	30	29.7	32	33	32.7
Apr-11	11	12	11.3	14	15	14.3	ns	ns	ns	ns	ns	ns	30	31	30.3
May-11	17	17	17.0	18	19	18.7	27	27	27.0	27	28	27.3	31	32	31.7
Jun-11	17	18	17.3	20	20	20.0	27	28	27.3	30	32	30.7	34	35	34.7
Jul-11	20	22	21.3	20	22	21.3	32	34	33.0	30	34	31.3	35	35	35.0
Aug-11	18	18	18.0	18	20	19.3	28	30	29.3	30	32	31.3	35	36	35.7

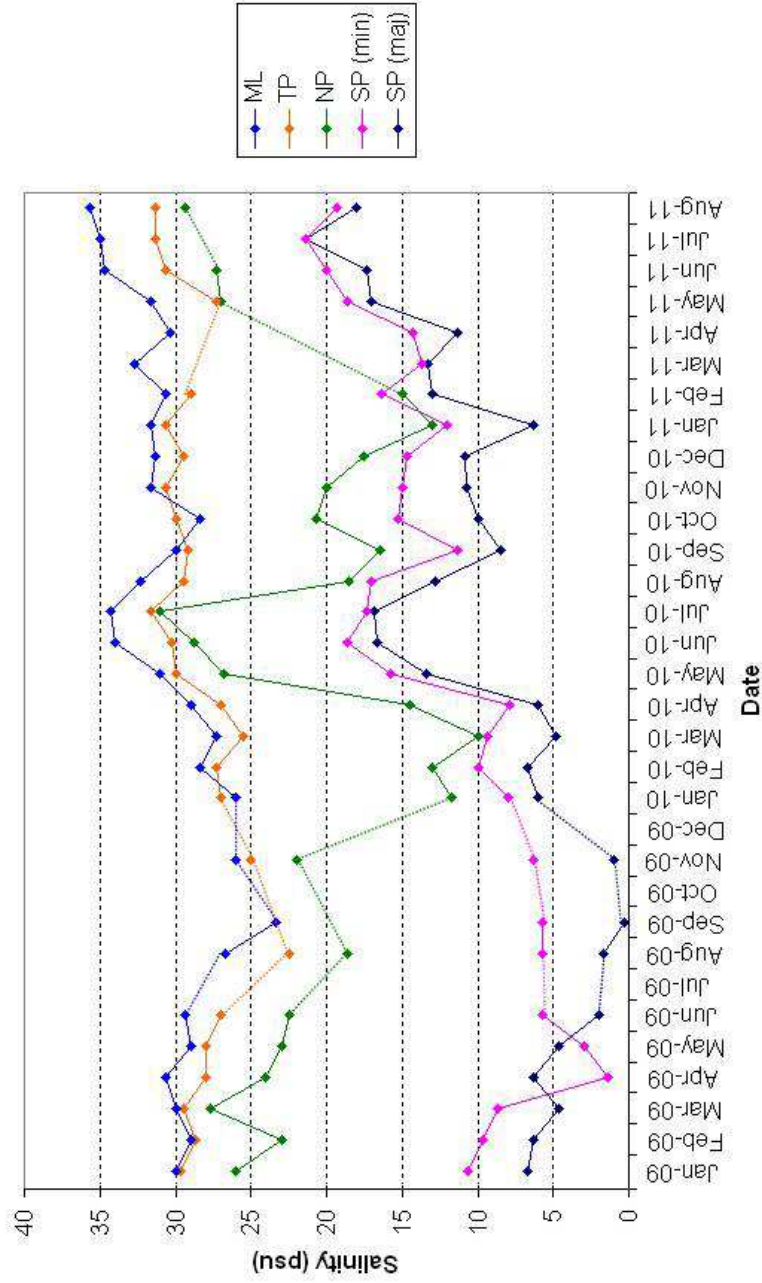


Figure 2.15. Monthly mean salinities (practical salinity units or psu) at depth of 10 cm for Mangrove Lake (ML), Trott's Pond (TP), North Pond (NP), and South Pond (SP) from January 2009 to August 2011.

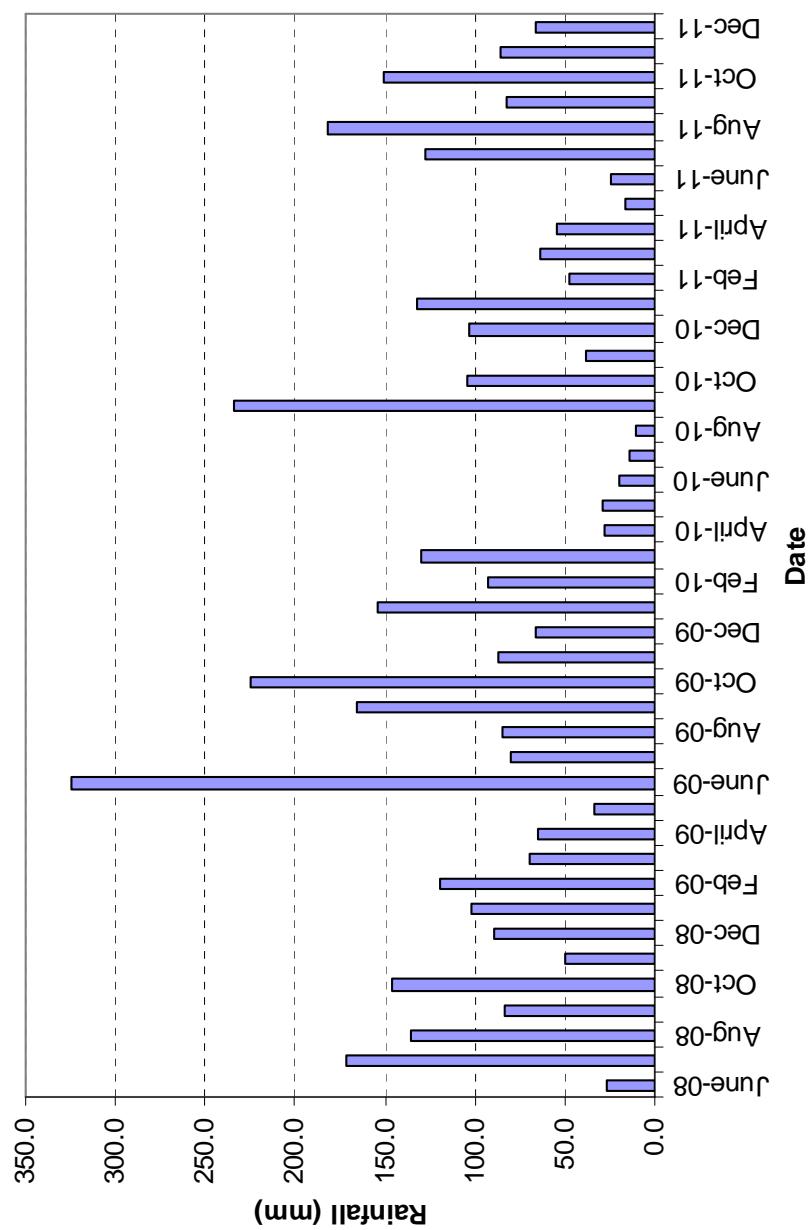


Figure 2.16. Monthly rainfall recorded on Bermuda between June 2008 and December 2011.

Mid Ocean golf course management activities around the terrapin ponds

The Mid Ocean golf course has a full-time course maintenance department, whose staff members are employed to maintain the golfing greens and surrounding environment for all 18 holes. These activities vary throughout the year and are performed by a variety of specialised machinery. The degree to which these activities affected the diamondback terrapins residing in the ponds and their environs was largely unknown prior to the onset of this doctoral study. Daily golf course maintenance activities include trimming, weeding, and mowing the greens. Less frequent activities include the edging and excavation of pre-existing sand bunkers, trimming the emergent vegetation (e.g. sheathed paspalum grass) that grows around the periphery of the ponds, trimming mangrove trees that obscure the views of the greens, and aerating and fertilizing the fairways.

The club currently tries to use environmentally friendly products (e.g. poultry manure) and practices (e.g. manual weeding) as often as possible rather than apply synthetic chemicals (e.g. Roundup). This ethos, however, has not always been practiced. The latter half of the 20th century saw a massive increase in the production and sale of synthetic pesticides, herbicides, fungicides, and fertilizers used for horticulture and agriculture across the globe (Robinson and Sutherland, 2002; Robbins and Sharp, 2003; Mikkelsen and Bruulsena, 2005). Many chemicals (e.g. lead arsenate) were imported into Bermuda and routinely applied to the fairways and to the ponds on the Mid Ocean golf course (as well as others) in order to promote the growth of desired grasses and deter the growth of unwanted fungi, weeds, and algae (N. Furtado, personal communication). Decades of applying these chemicals, some of which have lengthy half-lives, have created toxic conditions on Bermuda (Fort et al., 2006). Many of these chemicals have leached into the wetlands and harmful contaminants, particularly heavy metals, petroleum hydrocarbons (both gasoline-range and diesel-range), and polycyclic aromatic hydrocarbons are now found within the benthic sediment of a number of ponds across Bermuda, including South Pond, Mangrove Lake, and Trott's Pond. (J. Bacon, personal communication).

Another practice of the 1950s, now long-abandoned, was the burning of the saw-grass marsh at South Pond. The use of fire to eliminate undesirable vegetation in this marsh was frequently employed for decades before being abandoned in favour of more environmentally sensitive practices (N. Furtado, personal communication). Presently, the golf course maintenance staff allow the saw-grass in this marsh to naturally decompose in-situ.

The control of feral chickens (*Gallus domesticus*) and feral cats (*Felis catus*) occurs sporadically on the Mid Ocean property. The chickens are periodically culled using traps baited with grain while the resident cats are fed at established feeding stations. The Mid-Ocean Club tries to keep these cats from reproducing by having them captured and spayed or neutered, before releasing them back into the environment.

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Chapter 3: Demographic Assessment of the Diamondback Terrapin Population in Bermuda

Abstract

Diamondback terrapins (*Malaclemys terrapin*) are native to the remote oceanic islands of Bermuda, and presently inhabit only four small brackish water ponds. A three year mark-recapture study was performed to collect baseline abundance and demographic data for future monitoring. It is estimated that just 100 individuals with a straight carapace length of ≥ 81 mm live on Bermuda, of which 48.5% were considered sexually mature. The population is dominated by females (sex ratio 2.9:1), with a mean straight carapace length of 158 mm (range 116-196 mm; SD 22.6 mm; n=64) and a mean mass of 720 g (range 270-1340 g; SD 286 g; n=64). Males had a mean straight carapace length of 123 mm (range 109-134 mm; SD 8 mm; n=22) and a mean mass of 281 g (range 200-350 g; SD 47 g; n=22). Annual growth rates varied by sex and stage; juvenile terrapins displayed the greatest change in straight carapace length (SCL) growing 22.4 mm yr^{-1} (range $17\text{-}30.9 \text{ mm yr}^{-1}$; SD 7.5 mm yr^{-1}), female SCL growth was found to be 7.9 mm yr^{-1} (range $1\text{-}20.7 \text{ mm yr}^{-1}$; SD 6.6 mm yr^{-1}) while male SCL growth was 0.8 mm yr^{-1} (range $0\text{-}2.1 \text{ mm yr}^{-1}$; SD 0.8 mm yr^{-1}). Over the three year period annual recruitment was estimated to be two terrapins. The findings indicate that the Bermudian population is very vulnerable to local extirpation. The data informed advocates for the legislative protection of Bermuda's diamondback terrapins. Because of this study, diamondback terrapins received legislative protection from the Government of Bermuda in 2012.

Introduction

The diamondback terrapin is one of only two emydid turtles living in the inland pond environments of the isolated oceanic islands of Bermuda. The other, *Trachemys scripta elegans*, was introduced to Bermuda via the pet trade (Bacon et al., 2006), and numerous feral populations are present throughout the islands (Outerbridge, 2008). Diamondback terrapins are less abundant and have a greatly restricted local distribution (Davenport et al.,

2005). Parham et al. (2008), using a combination of fossil, radiometric, geological and genetic data, confirmed the earlier hypothesis that diamondback terrapins could have naturally colonized the remote oceanic islands of Bermuda from mainland North America, using the Gulf Stream as the transport mechanism (Davenport et al., 2005). Limited data on the population status of diamondback terrapins in Bermuda have constrained conservation efforts. Knowledge of basic population size and demographics was deemed necessary to make informed management decisions and support construction of a species recovery plan for Bermuda.

Diamondbacks terrapins have been overexploited and affected by habitat loss and other anthropogenic influences in the U.S.A. (see reviews by Butler et al., 2006; Ernst and Lovich, 2009). They were listed as a globally near-threatened species by the International Union for the Conservation of Natural Resources (IUCN) in 1996. Their status within the U.S.A., ranges from 'endangered' to 'a species of special concern' (Lee and Chew, 2008). In 2013, this species was included in Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) in an attempt to regulate international trade so that exports from the native range are not detrimental to the species' survival in the wild.

Population estimates can be accomplished using several capture-mark-recapture approaches. Methods for determining abundance in diamondback terrapin populations in North America have included the Petersen (or Lincoln- Petersen) method (Hurd et al., 1979; Avissar, 2006), the Jolly-Seber (also known as the Cormack-Jolly-Seber) stochastic method (Roosenburg et al., 1997; Hart, 2005), and the Schnabel method (Seigel, 1984; Butler, 2002).

Effective sampling of any species requires specialized techniques that suit the habitat and reflect the ecology of the target species (Akre et al., 2012). The heterogeneity of diamondback terrapin habitats throughout the North American range has required the use of different methods, some of which are inherently biased towards a particular sex (e.g. mature females hand-captured at nesting sites) or size class (e.g. adults and large juveniles are present within the aquatic environment whereas neonates and small

juveniles are typically not). Most of the published literature for terrapins has involved studies within the salt marsh environment, and the capture gear has included otter trawls (Hurd et al., 1979; Lovich and Gibbons, 1990; Butler, 2000), seine and trammel nets (Lovich and Gibbons, 1990; Tucker et al., 1995; Simoes and Chambers, 1999; Gibbons et al., 2001; Harden et al., 2007), gill nets (Seigel, 1984; Butler, 2000), peeler bank traps and fyke nets (Roosenburg et al., 1999), hoop and cast nets (Butler, 2000), and crab traps (both commercial and modified varieties) (Bishop, 1983; Roosenburg et al., 1997; Wood, 1997; Roosenburg et al., 1999; Butler, 2000, 2002; Avissar, 2006). Fewer studies have been published within the mangrove environment, but Hart and McIvor (2008) used commercial crab traps and dip nets (favouring the latter) to capture terrapins in SW Florida. The hand-capture of adult females while at nesting sites has also occurred throughout the range (Burger and Montevecchi, 1975; Burger, 1977; Roosenburg and Dunham, 1997; Butler, 2002; Feinberg and Burke, 2003).

The present investigation had the following objectives: (1) to estimate the size of the Bermudian terrapin population, (2) to analyze the population structure to determine demographic characteristics (sex ratio, size classes, annual recruitment rates, and density), (3) to test the effectiveness of two different trapping methods, (4) to determine somatic growth rates in Bermuda's terrapin population and compare them with known rates in North American terrapins, and (5) to provide scientific data on their abundance to the Government of Bermuda so that this native species could be included in the Bermuda Protected Species Act (2003).

Methods

Study site

The entire known Bermudian terrapin population of is found in four neighbouring brackish water ponds (Mangrove Lake, South Pond, North Pond, and Trott's Pond (Figs. 3.1 and 3.2)) on a private golf course located at the eastern end of the islands (32.32858°N, 64.70547°W; WGS 84). All were incorporated into the golf course as water hazards during the 1920s and are situated upon a single square kilometre of land. The total surface

area of each of the ponds was calculated in ArcGIS 9.0 using a 2003 digitized aerial orthophotograph of the Bermuda islands. Mangrove Lake is the largest pond on Bermuda, approximately 10 ha in area, and is characterized by shallow depths (averaging 134 cm), fairly even contours, and a gently sloping shoreline (Thomas et al., 1991). The pond bottom comprises deep deposits of gelatinous, sapropelic sediment (Hatcher et al., 1982) from which patches of widgeon grass (*Ruppia maritima*) grow in dense clumps. This entire body of water is surrounded by a mangrove swamp totalling 2.3 hectares in area and dominated by red mangrove trees (*Rhizophora mangle*) (Thomas, 1993). The mean monthly mid-water temperature during 2010 was 23.5°C (range 15.6-30.6; SD 6.0). Surface salinity averaged 29.4 psu (range 23.3-34.0; SD 2.7) between 2009 and 2010.

Trott's Pond is approximately 3 ha in area. The pond has a mean depth of 269 cm, and a bottom that comprises deep deposits of highly organic sediments consisting of a matrix of mostly sand, silt-clay, organic mud and detritus (Thomas et al., 1991). Trott's Pond is also surrounded by a mangrove swamp, totalling 0.8 hectares in area and dominated by red mangrove trees (Thomas, 1993). Mangrove Lake and Trott's Pond are both anchialine ponds (isolated, saline, land-locked bodies of water with permanent - often subterranean - connections to the ocean), and are refugia for various rare native and endemic fauna, including the Bermuda killifish (*Fundulus bermudae*) (Outerbridge et al., 2007), the flat mangrove oyster (*Isognomon alatus*) (Thomas and Dangeubun, 1994), as well as the diamondback terrapin (Davenport et al., 2005). The mean monthly mid-water temperature between 2009 and 2010 was not available. The surface salinity averaged 28.3 psu (range 22.5-31.7; SD 2.2).

North Pond and South Pond are considerably smaller in area (both approximately 0.4 ha), much shallower in depth (averaging 30 cm), and lack mangrove vegetation; however both ponds have small marshes in their centres dominated by grasses (*Cladium jamaicense* and *Paspalum vaginatum*). Neither pond is directly connected to the ocean, but water levels in both vary considerably with rainfall throughout the year, which causes great variability in temperature and salinity (see Chapter 2). The

mean monthly mid-water temperature between 2009 and 2010 at South Pond was 24.1°C (range 17.0-31.4; SD 5.0), and the surface salinity averaged 7.5 psu (range 0.3-16.8; SD 4.8) during the same period. The mean monthly mid-water temperature between 2009 and 2010 at North Pond was not available. The surface salinity averaged 20.8 psu (range 10.0-31.0; SD 5.8).

For a comprehensive description of these study sites please refer to Chapter 2.



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Figure 3.1. Aerial photograph of Bermuda showing the location of the four diamondback terrapin ponds.



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Figure 3.2. Aerial photograph from 2003 showing the four diamondback terrapin ponds situated on the Mid Ocean golf course (A=Mangrove Lake, B=Trott's Pond, C=South Pond, D=North Pond).

Mark and Recapture Sampling (Schnabel method)

Based on the conditions in the Schnabel formula and what was previously known about the restricted occurrence of diamondback terrapins in Bermuda, the Schnabel method was determined to be the most suitable method to obtain estimates for Bermuda's terrapin population, and was achieved by performing a three year survey based on mark and multiple recapture sampling. Bermuda's terrapins are considered to be a geographically closed population (e.g. no immigration or emigration). However, since the surveys spanned a three year period the population was demographically open (e.g. growth through recruitment and loss through death). Animals are captured on several occasions over time; all unmarked terrapins in each capture session are marked in a unique and readily identifiable way at the time of their first capture and then released back into

the pond from which they had been trapped. By recording the total number of terrapins captured in each trapping event, the number of marked terrapins among them, and the number of unmarked terrapins among them, the total population can be continuously estimated by using the following formula:

$$\text{Total Population (N)} = \frac{\sum n_i M_i^2}{\sum m_i M_i}$$

Where:

$i = i^{\text{th}}$ sample

n_i = number of animals in the i^{th} sample

m_i = number of animals in the i^{th} sample that are carrying marks

u_i = number of unmarked animals in the i^{th} sample ($n_i - m_i$)

M_i = number of animals marked prior to the i^{th} sample

Confidence in the Schnabel formula is maintained provided the following conditions are met:

1. The initial sample taken is representative of the entire population,
2. The probability of recapture and the survival of the marked terrapins are not compromised by the marking technique,
3. That marked terrapins are as vulnerable to the trapping technique as are unmarked terrapins,
4. That marked terrapins become randomly mixed with the unmarked terrapins in the pond,
5. That marked terrapins do not lose their mark over time,
6. That all marks are recognized and reported upon recovery.

Violating any of these assumptions can bias estimates to varying degrees (see Pollock et al., 1990).

Trapping

Between the months of June and September in each year of 2008, 2009, and 2010, mark-recapture trapping surveys were undertaken in Mangrove Lake, South Pond, North Pond, and Trott's Pond. This was the only instance in which Trott's Pond was used in the research. The differences in the physical characteristics between these ponds (Trott's Pond and Mangrove Lake are both deeper and more tidal than South Pond and North Pond) necessitated the use of two different traps which had been modified to capture terrapins safely.

The first type of trap was constructed of standard crab trap wire (16 gauge, 3.75 cm mesh), had four funnels (20 cm in width) at the base to allow terrapins points of entry, and had dimensions of 60 cm in length, 60 cm in width, and 200 cm in height (Fig. A3.2 in Appendix 3) (see Roosenburg et al., 1997). The height ensured that the trap's top remained above the surface of the water at all times, thus allowing captured terrapins a breathing space; buoys were attached to opposing corners at the top of each trap to prevent them from falling over. Three traps of this type were built for use in the current investigation, but were only used in Trott's Pond and Mangrove Lake.

The second type of trap was a simple modification of a collapsible fish trap (Memphis Net and Twine Co., Inc, U.S.A.). This funnel-style trap was made from 1.3 cm vinyl mesh, measured 80 cm in length, 60 cm in width, 28 cm in height, and had a 55 cm wide funnel opening at each end through which terrapins could enter (Fig. A3.3 in Appendix 3). Six traps of this type were used and were fitted internally with short segments of cylindrical Styrofoam held in place with locking nylon ties. The Styrofoam prevented the traps from sinking and also provided a breathing space for terrapins. These traps were used in all four ponds.

All traps were baited with frozen herring (*Clupea harengus*) which was secured within perforated bait-boxes to prevent ingestion by the terrapins (Fig. A3.4 in Appendix 3). Since Wood (1997) found that floating traps were ineffective in catching terrapins, all traps used in the present study were deployed in areas where the bottom of the trap was either upon the pond bottom or floating approximately 2.5 cm above it so that terrapins

foraging upon the bottom could gain easy access. Hence modified fish traps set in Mangrove Lake and Trott's Pond were always placed in shallow water (15-30 cm deep), typically between the mangrove prop roots under the tree canopy (Fig. A3.5 in Appendix 3), whereas the modified crab traps were set in the open water immediately seaward of the fringing mangrove trees in depths 1-2 m (Fig. A3.6 in Appendix 3). The modified fish traps set in South Pond and North Pond were also placed in shallow water (15-30 cm deep) in haphazard locations to maximise trapping effectiveness (Fig. A3.7 in Appendix 3).

Each trapping period consisted of daily trapping for 14 consecutive days every month (June-September) between 2008 and 2010, during which each trap was checked daily for the presence of terrapins and re-baited as necessary. Traps were removed from the four ponds for two weeks at the end of each trapping period, after which they were re-deployed for another 14 day trapping period.

Catch per unit effort (CPUE) for each type of trap was calculated as the daily number of terrapins captured (including recaptures), divided by the number of trap-days (total number of traps multiplied by number of days deployed).

Measurements

All terrapins caught for the first time were characterized as female, male or juvenile, digitally photographed, measured, weighed, and examined for general health and physical anomalies (e.g. missing or extra scutes and shell damage). The terrapins were then released at their original capture locations. Recaptured terrapins were simply re-weighed and re-measured.

Sex was determined by examining tail thickness as well as the position of the cloaca in relation to the margin of the supracaudal scutes; males have longer thicker tails with cloacal openings situated posterior to these scutes (Lovich and Gibbons, 1990). Maturity status was determined via plastron length following Lovich and Gibbons (1990) whereby individuals were classified as juvenile if straight plastron length measured less than 91 mm, males were classified as sexually mature if straight plastron length measured 91-137 mm and females were classified as

sexually mature if straight plastron length was equal to or greater than 138 mm. Estimates of size at maturity for the classification of mature individuals were based upon Lovich and Gibbons (1990) because Bermuda's population of terrapins are apparently descendants of terrapins from the Carolinas (Parham et al., 2008).

For each terrapin captured, four straight-line shell measurements were taken using vernier calipers following Bolten (1999); minimum straight carapace length (SCL), straight carapace width (SCW), straight plastron length (SPL), and shell height (SH) (Fig. A3.8 in Appendix 3). All measurements were recorded to the nearest 1.0 mm. The minimum straight carapace length was measured from the anterior edge of the nuchal scute to the posterior edge of the shell between the supracaudal scutes along the midline. Straight carapace width was measured at the widest point across the carapace. Plastron length was measured along the midline from the anterior edge of the gular scutes to the posterior edge of the anal scutes. Shell height was defined as the maximum distance between the lowest point of the plastron and the highest point of the carapace. While that the anatomical locations on the shell where SCW and SH were measured for each terrapin varied between individuals, due to slight variations in the shape of the terrapin, those used to determine SCL and SPL were consistent between individuals. All terrapins were measured by the same researcher (M.O.) throughout.

Body mass (g) was recorded for every capture and recapture event using three spring balances. Terrapins <100 g were measured to the nearest 1 g, terrapins 100-500 g were measured to the nearest 5 g and larger terrapins (>500 g) to the nearest 10 g. Each terrapin was placed in a nylon mesh bag to allow for accurate weighing. Total mass was calculated by subtracting the weight of the bag from the total weight of the bag and terrapin.

Marking

Each terrapin caught for the first time was given a unique mark using a marginal scute notching technique adapted from Cagle (1939). A triangular file was used to incise a V-shaped notch into either the left or right marginal

scutes, or a combination of both. Smaller individuals (i.e. sub-adults and mature males) were notched using a 5 mm file, larger individuals (i.e. mature females) with a 10 mm file. Notch locations were treated with antiseptic (betadine) immediately following the procedure. The twelve marginal scutes on the right side of the carapace were assigned single digit values (i.e. 1-9), the twelve marginal scutes on the left side given ten digit values (i.e. 10, 20, 30 – 90), and notches were cut into the centre of each corresponding scute. Hundred digit values (i.e. 100, 200, 300 – 900), were assigned to the sutures between the left marginal scutes, and thousand digit values (i.e. 1000, 2000, 3000 – 9000) to the sutures between the right marginal scutes (Figs. A3.9 and A3.10 in Appendix 3). Marginal scutes lying immediately over the bridge (i.e. the fifth, sixth, and seventh) on both sides of the carapace were not notched. Extra marginal scutes do not affect this numbering system and, when found on an individual, were not incised with a V-shaped notch.

Somatic growth

Somatic growth was examined by studying (1) the change in SCL over time, and (2) the allometric growth relationships (the relative change in shape) between four linear dimensions (SCL, SPL, SCW, SH) and mass. Annual growth rates were calculated for recaptured individuals by subtracting SCL at the time of original capture from the SCL measurement at final recapture for terrapins recaptured at time intervals of approximately 365 days (+/- 30 days), and multiples thereof. Intervals of approximately one year were chosen to minimise the possible distortion of growth data by seasonal effects. Recapture intervals that yielded no measurable growth were retained within the dataset. The relative change in shape was evaluated by analysing the relationships between SCL and SCW, SCL and SPL, SCL and SH, SCL and masses of all captured female and male terrapins. Allometric and isometric growth calculations were performed using the free palaeontological software program PAST. Datasets were normalised using log-transformations before performing linear regressions with 95% confidence intervals, and strength of regression and statistical significance were recorded as coefficients of determination (R^2) and P-values.

Results

Capture summaries and trapping effort

Nine traps (six modified fish traps and three modified crab traps) were used for a total of 140 trap days (51 in 2008, 45 in 2009, and 44 in 2010) between 2008 and 2010. Tables 3.1 and 3.2 summarise the number of terrapins captured (by observation and location respectively) during the three year period. A total of 317 terrapins were captured (including recaptures), which includes 171 captured in 2008 (80 first captures and 91 recaptures), 120 captured in 2009 (14 first captures and 106 recaptures), and 26 captured in 2010 (five first captures and 21 recaptures). There were no incidences of terrapin mortality associated with either trap type during the three year survey period.

Table 3.1. Summary of diamondback terrapin captures by observation (2008-2010).

Year	First captures	Recaptures	Total captures
2008	80	91	171
2009	14	106	120
2010	5	21	26
Total	99	218	317

Thirty-nine marked terrapins (39.4% of the total) were only captured once; however individual terrapins were often recaptured multiple times. The overall recapture rate was 60.6% over the three year survey period, and ranged from once only (15 individuals) to 21 times (one individual). Movement between the various brackish water ponds was detected using the traps throughout the three year study period. The recapture histories of twelve marked terrapins showed that movement occurred between South Pond and Mangrove Lake (n=11) and between Mangrove Lake and North Pond (n=1) (Table A3.1 in Appendix 3). Additionally, diamondback terrapins were frequently observed travelling overland from one pond to another throughout the study period. Consequently, the terrapins are considered here to be one rather than four discrete populations.

Of the 317 terrapins captured, 302 (95.3%) were caught using the modified fish traps and 15 (4.7%) were caught using the modified crab traps. 269 terrapins (84.9% of all captures) came from South Pond, 47 terrapins (14.8% of all captures) came from Mangrove Lake, and one terrapin (0.3% of all captures) came from Trott's Pond.

Table 3.2. Summary of diamondback terrapin captures by location (2008-2010).

Year	South Pond	Mangrove Lake	Trott's Pond	Total captures
2008	142	28	1	171
2009	108	12	0	120
2010	19	7	0	26
Total	269	47	1	317

Catch per unit effort (CPUE) for the modified fish traps was 0.36 terrapins trap-day⁻¹, the CPUE for the modified crab traps was far lower (0.04 terrapins trap-day⁻¹). Overall CPUE for the combined fish and crab traps was 0.25 terrapins trap-day⁻¹.

Population estimate

Table A3.2 in Appendix 3 summarises the Schnabel calculations for each sampling session between 2008 and 2010. The estimated population was 94 individuals (≥ 81 mm SCL, the minimum size captured in either trap type) at the end of the sampling session in 2008. This estimate had increased to 98.1 individuals by the end of the 2009 sampling session, and by 2010 it was 100.3 individuals. The 95% confidence intervals for the overall (2010) estimate were 97.8 and 102.8.

Population structure and sex ratio

Ninety-nine individual terrapins were marked over the three year period using the marginal scute notching technique; 64 female, 22 male and 13 juveniles (Fig. 3.3). Mature adults comprised almost half (48.5%) of the terrapins in the sampled population. Most male terrapins (72.7%) were considered to be sexually mature following the criteria established for the

Carolina diamondback terrapin by Lovich and Gibbons (1990) (i.e. SPL \geq 91 mm), whereas 50% of the females were considered mature (i.e. SPL \geq 138 mm). The remaining male and female terrapins were classified as immature. Juvenile terrapins only comprised 13.1% of the sampled population.

The sex ratio of adult females to adult males in the Bermuda population was 2.9(F):1.0(M); however, the functional sex ratio (defined as the relative proportion of sexually mature females to sexually mature males) was 2.2(F):1.0(M). Pearson's chi-square test (using Yates' correction for continuity) on the dataset gave a P-value <0.001 , indicating that there was a highly significant difference in the observed sex frequency from a 1:1 expected Fisherian frequency; female diamondback terrapins were significantly more numerous than males in the Bermuda population.

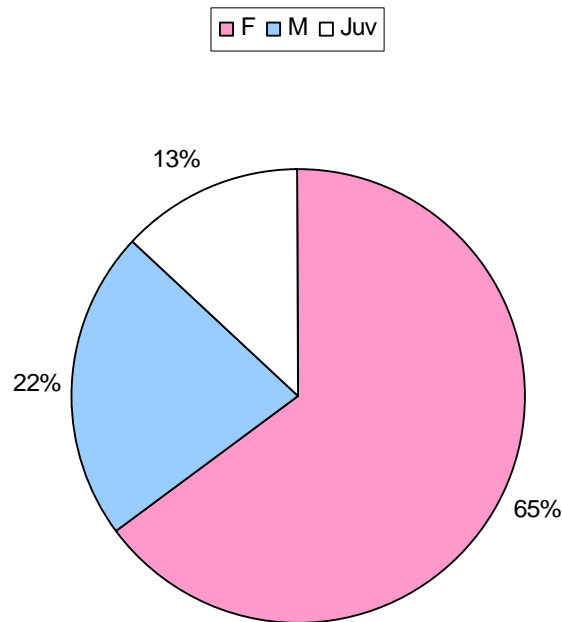


Figure 3.3. Demographic composition (2008-2010) of the sampled population of Bermuda's diamondback terrapins.

Terrapin biometrics

Table 3.3 and Fig. 3.4 summarise the biometric data for all female (n=64), male (n=22), and juvenile (n=13) diamondback terrapins at the time of first capture between 2008 and 2010. Full biometric datasets for female, male and juvenile terrapins are given in Tables A3.3 – A3.5 in Appendix 3. All

data were tested for normality. All data for females were non-normal (Anderson-Darling tests; SCL $p=0.016$, SCW $p=0.002$, SPL $p=0.003$, SH $p=0.034$, Mass $p=0.007$), some male data were non-normal (Anderson-Darling tests; SCL $p=0.030$, SPL $p=0.019$), others were normal (Anderson-Darling tests; SCW $p=0.400$, SH $p=0.118$, Mass $p=0.220$). All data for juveniles were normally-distributed (Anderson-Darling tests; SCL $p=0.188$, SCW $p=0.837$, SPL $p=0.193$, SH $p=0.502$, Mass $p=0.108$). Because some datasets were non-normal, non-parametric statistical tests were required in comparisons. A Mood's Median test of SCL showed that medians of all three categories differed significantly ($p<0.05$) (juvenile median SCL 101 mm, male median SCL 126 mm, female median SCL 160 mm). A similar test of body mass also showed significant differences ($p<0.05$) (juvenile median mass 185 g, male median mass 295 g, female median mass 710 g). Hence median female body mass was $2.4 \times$ male body mass.

Table 3.3. Biometric data summary for all female, male and juvenile terrapins encountered at first capture during the 2008, 2009 and 2010 population surveys (SCL=straight carapace length; SCW=straight carapace width; SPL=straight plastron length; SH=shell height).

	SCL (mm)	SCW (mm)	SPL (mm)	SH (mm)	Mass (g)
Females (n=64)					
Median	160.0	125.0	138.0	66.0	710.0
Q1	137.5	107.0	120.0	56.2	466.0
Q3	179.8	137.0	154.8	69.0	992.0
Range:	116-196	94-150	99-196	48-80	270-1340
Males (n=22)					
Median	126.0	94.0	101.0	45.5	295.0
Q1	114.0	87.8	114.0	43.8	233.7
Q3	128.5	97.2	128.5	47.0	320.0
Range:	109-134	85-102	87-111	40-48	200-350
Juveniles (n=13)					
Mean:	98	78	81.8	41.8	168.1
SD:	9.5	7.7	9.0	4.0	42.6
Range:	81-108	65-89	67-92	34-47	95-215

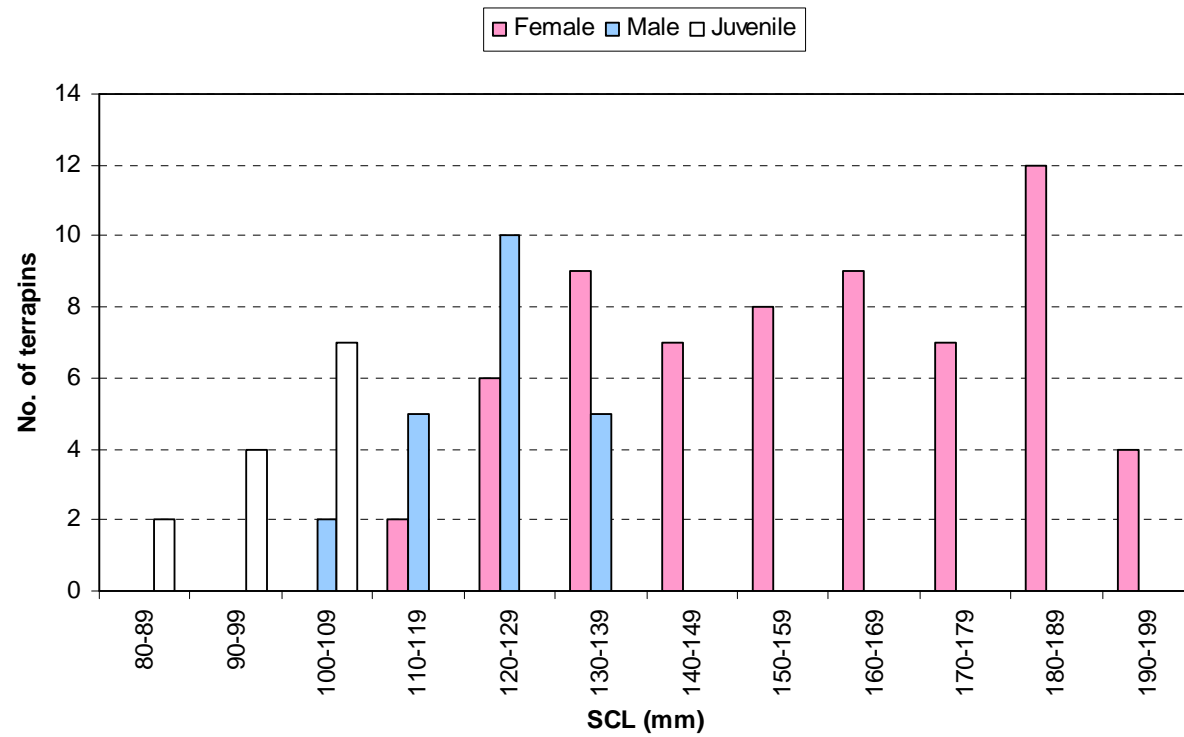


Figure 3.4. Size-frequency histogram for all female, male, and juvenile diamondback terrapins encountered at first capture in Bermuda (n=99) during the 2008, 2009 and 2010 population surveys.

Digital photographs of the carapace and plastron of each terrapin were obtained from all but two individuals. These photographs (see Figs. A3.11 – A3.20 in Appendix 3) show that Bermuda's terrapins exhibit variation in both carapace colour (which usually ranged from light olive and brown to dark brown and black) as well as plastron colour (which was typically shades of orange, sometimes flecked with dark blotches or smudges). In a few instances the plastral scutes had a dark base colour with bright orange highlights. Individuals found with darkly coloured plastrons also had a carapace colouration that was nearly black. Skin colour showed much less variation, always being shades of grey upon which many dark spots or flecks were visible. Dark lines or bars were never observed in the Bermuda population.

Thirty four out of 99 individuals (34.3%) in the sampled population showed carapace scute anomalies. The most common anomalies observed in the Bermuda population were extra vertebral scutes (15.2% frequency of occurrence), extra costal scutes (15.2% frequency of occurrence), and extra marginal scutes (18.2% frequency of occurrence). Missing vertebral, costal and marginal scutes were found less frequently (7.2% of the sampled population). Two individuals had the correct number of carapace scutes, but these scutes varied in size which had the effect of making the shells asymmetrical in shape. Only two individuals in the surveyed population had plastral scute anomalies; a large female that possessed a double set of abdominal, femoral and anal scutes, and a juvenile that possessed an extra right pectoral scute. Interestingly, neither of these individuals had abnormal carapace scute patterns.

No epibionts (e.g. barnacles) or major injuries (e.g. missing limbs or catastrophic shell damage) were observed in any of the sampled terrapins; however, minor damage to the carapace was occasionally observed. Nine terrapins were observed throughout the three year trapping period to have abrasions and scarring upon the carapace scutes, including two terrapins that had scarring upon the third and fourth vertebral scutes (one appeared to be an old, healed injury; the other appeared to be a more recent injury that was in the process of healing).

Population density

Population density (number of terrapins ha^{-1}) was calculated using the Schnabel population estimate and therefore represents a conservative estimate rather than an absolute estimate. The total area of open water inhabited by Bermuda's diamondback terrapins is 13.4 ha (Mangrove Lake is 9.9 ha, Trott's Pond is 2.9 ha, North Pond is 0.4 ha, and South Pond is 0.2 ha). Furthermore, the total area of wetlands associated with these ponds is 3.4 ha (2.3 ha and 0.8 ha of mangrove swamp surrounding Mangrove Lake and Trott's Pond respectively (Thomas, 1993), and 0.3 ha of saw-grass marsh at South Pond (M. Outerbridge, unpublished data)). Assuming the diamondback terrapins in Bermuda are considered to be one discrete population (given that movement between the ponds was observed), density estimates were calculated to range from 6.0 terrapins ha^{-1} (open water and adjacent wetlands) to 7.5 terrapins ha^{-1} (open water only).

Growth rates

Elapsed time between first capture and last recapture ranged from 1-824 days (mean 368 days). Forty one individual diamondback terrapins from the study population in Bermuda were recaptured after spending at least 365 days (± 30 days) at liberty, of which 16 terrapins were recaptured at the one year interval and six terrapins were recaptured at the two year interval. No terrapins were recaptured at the three year interval (see Table A3.6 in Appendix 3). Annual changes in SCL growth were detected in all but two of the diamondback terrapins recaptured at the one year and two year time intervals ($n=3$ juveniles; $n=6$ males; $n=13$ females), and no negative growth increments were recorded. Tables A3.7 – A3.9 in Appendix 3 summarise the annual growth rates measured in juvenile ($n=3$), male ($n=6$) and female ($n=13$) diamondback terrapins from the Bermuda population. Growth rate varied by sex/stage; the mean annual growth for the subset of juvenile terrapins that displayed a change in SCL was 22.4 mm yr^{-1} (range $17\text{-}30.9 \text{ mm yr}^{-1}$; SD 7.5 mm yr^{-1}). Mean annual growth for the subset of male terrapins that displayed a change in SCL was 0.8 mm yr^{-1} (range $0\text{-}2.1 \text{ mm}$

yr⁻¹; SD 0.8 mm yr⁻¹); for the subset of female terrapins it was 7.9 mm yr⁻¹ (range 1-20.7 mm yr⁻¹; SD 6.6 mm yr⁻¹).

Some further statistical analyses were performed, despite the small sample size (though juveniles (n=3) had to be excluded). A general additive model (GAM) was conducted upon the data for males and females, though interaction was not modelled because of low male sample size. Two factors had statistically significant influence upon growth rate, initial SCL (p=0.000321) and sex (0.0364). Additionally, there was a weak ($R^2 = 0.32$) but highly significant (p<0.0005) negative linear correlation between initial SCL and growth rate in adult female diamondbacks (n=46), indicating that growth slows with increasing size (and presumably age) (Fig. 3.5). The dataset contained several examples of zero growth, so further investigation of correlation (e.g. by logging data) or curve fitting was not feasible. Data for males (n=9) showed no significant correlation ($R^2 = 0.00$, p= 0.831); data for juveniles (n=4) also showed no significant correlation ($R^2 = 0.00$, p= 0.879).

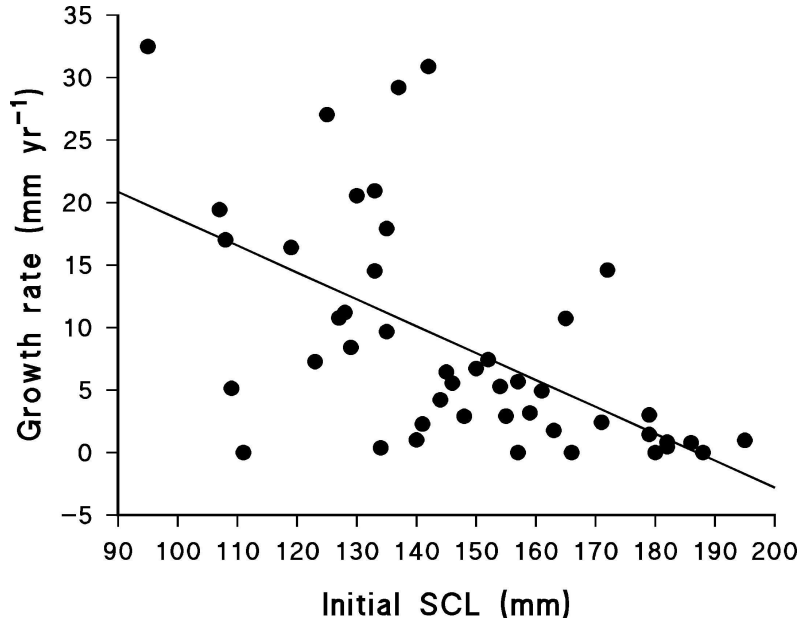


Figure 3.5. Relationship between initial SCL and growth rate of adult female diamondback terrapins. Solid circles represent data (n=46), solid line indicates linear regression of data. Growth rate (mm yr⁻¹) = 40.2-0.215 initial SCL (mm) ($R^2 = 0.32$, p<0.0005).

Morphometrics

Female diamondback terrapins in Bermuda (n=64) showed strong correlations in the relationships between the four linear shell dimensions and mass. Table 3.4 summarises the various log-transformed allometric relationships between straight carapace length (SCL) and straight carapace width (SCW), straight plastron length (SPL), shell height (SH) and mass. A significant negative allometric relationship existed between SCL and SCW. Carapace width increased proportionately less than carapace length (i.e. they got narrower as they grew longer). However, the relationship of SCL to SPL was not significantly different from the isometric condition. A significant negative allometric relationship existed between SCL and SH. Shell height increased proportionately less than carapace length (i.e. they became relatively flatter as they grew longer). Finally, the relationship of SCL to mass was not significantly different from the isometric condition. Mass increased with the cube of carapace length. Graphs A-D, Table A3.10 in Appendix 3 show the log-transformed regressions between SCL and the other three linear shell dimensions and mass.

Partially similar growth relationships were found for 22 male diamondback terrapins in Bermuda (see Table 3.5). A significant negative allometric relationship existed between SCL and SCW, but the relationship between SCL and SPL was isometric. Unlike in females, the relationship between SCL and SH was isometric, so male shells did not get flatter as they became larger. Finally, the relationship of SCL to mass had a slightly negative allometric relationship with SCL, thus it increased proportionately less than with the cube of carapace length in male terrapins. Graphs A-D, Table A3.11 in Appendix 3 show the log-transformed regressions between SCL and the other three linear shell dimensions plus mass.

Table 3.4. Log-transformed allometric relationships between straight carapace length (SCL) and straight carapace width (SCW), straight plastron length (SPL), shell height (SH) and mass for 64 female diamondback terrapins in Bermuda.

	R ² value	p-value	Growth
$\log \text{SCW} = 0.9187 + 0.0707 \log \text{SCL}$	0.965	<0.0001	negative allometric
$\log \text{SPL} = 0.9991 - 0.0629 \log \text{SCL}$	0.967	<0.0001	isometric
$\log \text{SH} = 0.8927 - 0.1598 \log \text{SCL}$	0.874	<0.0001	negative allometric
$\log \text{mass} = 2.952 - 3.659 \log \text{SCL}$	0.942	<0.0001	isometric

Table 3.5. Log-transformed allometric relationships between straight carapace length (SCL) and straight carapace width (SCW), straight plastron length (SPL), shell height (SH) and mass for 22 male diamondback terrapins in Bermuda.

	R ² value	p-value	Growth
$\log \text{SCW} = 0.8260 + 0.2454 \log \text{SCL}$	0.876	<0.0001	negative allometric
$\log \text{SPL} = 1.0702 - 0.2383 \log \text{SCL}$	0.899	<0.0001	isometric
$\log \text{SH} = 0.7112 - 0.1679 \log \text{SCL}$	0.214	<0.0291	isometric
$\log \text{mass} = 2.5607 - 2.9031 \log \text{SCL}$	0.788	<0.0001	negative allometric

Discussion

Traps and trapping effort

The results of the current study suggest that the modified fish traps were ten fold more effective in catching diamondback terrapins within Bermuda's pond environment than the modified crab traps. The modified fish traps were collapsible and readily stacked making them easy to transport and handle under field conditions. Modifications made to the buoyancy of these traps were inexpensive and quickly accomplished, while the trap size

allowed easy deployment in the shallow and structurally complex mangrove swamp environment. Similar traps have previously been used to capture the painted turtle (*Chrysemys picta*) in shallow water (<15 cm) (Zweifel, 1989) and traps modified to float have been used for surface trapping of red-eared sliders (*Trachemys scripta elegans*) in water >30 cm (Outerbridge, 2008; Ng, 2009).

The modified crab traps were more difficult to transport and handle under field conditions; they could only be deployed in areas of open water. Modified and un-modified crab traps have been used to capture terrapins in different regions throughout the North American range (Bishop, 1983; Mann, 1995; Roosenburg et al., 1997; Wood, 1997; Butler, 2000, 2002; Avissar, 2006; Hart and McIvor, 2008), but high terrapin mortality (8% to 28%) has been reported in some cases (Mann, 1995; Wood, 1997; Butler, 2000). Overall it appears that the modified fish traps are more effective and safer to use.

The marked drop in captures over the 2010 trapping session may be the result of a summer drought which lasted from April-August (see Figs 2.9 and 2.16 in Chapter 2) that led to significant losses of water (through evaporation) in South Pond and thus precluded the effective use of traps. The drop in capture rate over the same period at Mangrove Lake may also be explained by conditions brought on by the drought. Experimental studies have shown that diamondback terrapins are capable of drinking from a thin layer of fresh water lying on top of a saline water column (Davenport and Macedo, 1990) but they reduce food consumption when exposed to full seawater (34 psu) with no access to freshwater (Davenport and Ward, 1993). It is therefore likely that the prolonged lack of rainfall and a near-absence of potable water in South Pond may have suppressed appetite to an extent that many terrapins in the population were no longer attracted to the baited traps in Mangrove Lake.

Population estimate

Based upon the mark-recapture data, the entire population calculated using the Schnabel method was about 100 adults and large juveniles. This estimate is conservative as it does not include the smallest size classes of the

population (i.e. neonate and young juveniles), since they do not occur in open water pond environments (Lovich et al., 1991; Draud et al., 2004). A total of 99 individual terrapins had been uniquely marked at the conclusion of the three year trapping period (2008-2010). Confidence in this Schnabel estimate was maintained because: (1) The initial sample taken during the first trapping period (June 2008) was representative of the entire population; (2) No recaptured terrapin ever showed signs of shell necrosis or disability as a result of having been notched, so the notching technique did not compromise the survival and overall health of the marked individuals; (3) The relatively high recapture rate (61%) suggested that the trapping technique did not bias against marked terrapins, and the relatively low mean number of times an individual terrapin was recaptured ($n=3.4$) suggested that the trapping technique did not bias in favour of the marked terrapins either; (4) The prolonged time frame of the trapping investigation, combined with the fact that terrapins were recaptured in different ponds from their original capture, ensured that marked terrapins were randomly mixed with unmarked terrapins in the population; (5) The notches from terrapins originally marked in 2008 and subsequently recaptured in 2009 and 2010 were easily recognisable, indicating that the notched terrapins did not lose their marks over the trapping period.

Diamondback terrapins occupy a large coastal range in the U.S.A. and van Dijk (2011) estimated that the total population of diamondback terrapins in North America exceeds 100,000. However, the status of the species differs in various regions throughout its range, and local population estimates vary greatly. While Roosenburg et al. (1997) reported an estimated 2778-3730 individuals in the Patuxent River Estuary of Chesapeake Bay and Forstner et al. (2000) estimated that 1300 terrapins resided in the Everglades National Park. The latter also estimated that only 200 terrapins inhabited the Lower Florida Keys, while Seigel (1984) estimated a combined population of approximately 618 terrapins at two study sites within the Merritt Island National Wildlife Refuge in east central Florida.

Past Bermudian terrapin population sizes and trends are unknown. Diamondback terrapins were not recorded on 19th and 20th century

herpetological fauna lists (Jones, 1859; Garman, 1884; Heilprin, 1889; Hurdis, 1897; Verrill et al., 1903), despite being present for at least 400 years (Parham et al., 2008). Perhaps terrapins have always had a small population size and a limited distribution across Bermuda. Long-term monitoring is strongly recommended for the Bermuda population given its small size and limited distribution, which makes it vulnerable to local extirpation.

Population size is a major factor in the survival or extinction of populations: larger sizes provide insurance against unpredictable environmental events as well as stochastic changes in age structure, genetic drift and inbreeding depression (Thompson, 1991; Reed et al., 2003 for discussion). The concept of a minimum viable population (MVP) was introduced by Shaffer (1981) in an effort to provide criteria for the successful preservation of a species at the population level. He proposed that the MVP for any given species in any given habitat was 'the smallest isolated population having a 99% chance of remaining extant for 1000 years despite the foreseeable effects of demographic, environmental, and genetic stochasticity, and natural catastrophes.' Others have based the minimum population concept on genetic considerations. When considering the viability of vertebrate populations, there is a generally accepted 50/500 'rule of thumb' that a minimum effective population of 50 adults is required to prevent the deleterious effects of inbreeding, and a population of 500 adults is required to maintain genetic variability in order for a population to be able to adapt to environmental stochasticity (Franklin, 1980; Soule, 1980). This suggests that the Bermudian diamondback terrapin population is at best close to the MVP level and may be below it. Moreover, recent modelling exercises have taken stochastic events of demography and environment into consideration and suggest that long-term survival (>40 generations) requires minimum population sizes of the order of 5000 (Reed et al., 2003); if applied to diamondback terrapins this would suggest that most local populations in the U.S.A. (as well as Bermuda) would be doomed without human intervention. However, some studies indicate that long-lived species such as turtles may represent an exception to general MVP guidelines. Demographic modelling for a small and fragmented population of bog turtle

(*Glyptemys muhlenbergii*) in the U.S.A. suggests that colonies with as few as 10-15 breeding females have >90% probability of persisting for >100 years (Shoemaker et al., 2013). Those authors also suggested that some populations of long-lived species with <50 individuals may be able to persist provided vital rates and environmental variance remain favourable. Furthermore, they hypothesized that bog turtles and similar wetland turtles may be naturally adapted for persistence in small population units (Shoemaker et al., 2013). Such may have been the scenario for Bermuda's founding population of diamondback terrapins. In a critical review of MVP usefulness, Flather et al. (2011) argued that there is no particular population size that is likely to protect a species against extinction if conservation efforts fail to diagnose and treat the mechanisms responsible for a population decline. However, they suggested that MVPs can serve as a useful tool to persuade policy-makers that extinction for a particular species is a possibility.

High adult survivorship, delayed sexual maturity, longevity, and repeated reproductive cycles (iteroparity) once adulthood has been reached are key characteristics of the population biology of turtles (Gibbs and Amato, 2000); however, these demographic traits constrain a turtle population's capacity to absorb increases in mortality rates caused by anthropogenic factors. Furthermore, high post-hatching survival rates are necessary to ensure that enough individuals survive to reproduce and maintain populations, while small increases in mortality rates of adults can lead to declines in populations (Congdon et al., 1993, 1994).

Bermudian diamondback terrapins are not affected by targeted fishing or by-catch mortality. However, the population is threatened by low reproductive success (see Chapter 5), habitat fragmentation and avian predation (see Chapter 6), pollution (see Chapter 7), and to a limited extent by hatchlings killed by motorized golf course vehicles and historic golf course maintenance practices. This contrasts with North America, where terrapins are exploited or die as a result of interaction with fishing gear, and road mortality can be great; during a seven year period, over 4000 road kill terrapins were discovered at one study site (Wood and Herlands, 1997). Anecdotal evidence suggests that some terrapins have been removed from

Bermudian ponds as pets. This activity is of concern as it removes valuable individuals from the breeding population.

Population structure and sex ratio

Mature adults comprised almost half (48.5%) of the terrapins in the sampled population; juvenile terrapins only comprised 13.1% of the sampled population. Small juveniles and hatchlings were not captured. Hatchling and young juvenile diamondback terrapins are known to utilize different habitats from larger juvenile and adults in the U.S.A. (Lovich et al., 1991; Draud et al., 2004); this also appears to be true for Bermuda's terrapins (see Chapter 6). The low incidence of larger juveniles indicates limited recruitment. Poor recruitment may be caused by reproductive failure (i.e. low hatching success and/or infertility) or high levels of predation at the neonate and young juvenile life history stage. Investigations of the nesting ecology of terrapins in Bermuda (see Chapter 5) showed that hatching success was low (17.6-21%). Hatchlings are also vulnerable to avian predation (see Chapter 6), so both of these factors likely contribute to the observed recruitment of only two juveniles per annum. Continued trapping within South Pond, Mangrove Lake, and Trott's Pond is recommended in order to monitor future rates of recruitment to determine whether the low levels observed between 2008 and 2010 are typical for this population.

The female to male sex ratio in the Bermuda population was estimated to be 2.9(F):1(M) at the conclusion of the 2010 survey; however the functional (sometimes also known as operational) sex ratio was 2.2 sexually mature females to every sexually mature male. Examining the actual sex ratio in a population can help determine factors contributing to the differences observed, whereas determining the functional sex ratio gives more precision to demographic studies with regards to potential genetic exchange. Some authors have stated that only sexually mature individuals in a population should be included in the calculation of sex ratios, and that the functional sex ratio is important from a demographic perspective because of the potential influence that the relative proportion of the sexes can have on time spent searching for mates, intra-sexual competition, and annual propagule production (Lovich and Gibbons, 1990).

Sex ratios in diamondbacks in the North American range show marked variation. These can be strongly female-biased (as in Bermuda), male-biased or equal (Seigel, 1984; Lovich and Gibbons, 1990; Roosenburg et al., 1997; Butler, 2000; Baldwin et al., 2005; Hart and McIvor, 2008). Variation has been variously attributed to biases caused by size-selective capture methods, the influence of temperature-dependent sex determination (TSD), differential migration rates, differential rates of maturity, and differential rates of mortality from predation and anthropogenic causes (Gibbons, 1990; Lovich and Gibbons, 1990; Roosenburg et al., 1997; Baldwin et al., 2005).

The female biased sex ratio in the Bermuda population is not believed to be caused by biased sampling techniques because both of the trap designs used in the current investigation had entrances of sufficient width to allow entry of the largest females in the population. Neither is the skewed sex ratio believed to be caused by differential rates of mortality from predation or anthropogenic causes because there are no known predators of adult terrapins present on Bermuda and there are no direct sex-specific anthropogenic activities affecting adult mortality. It is, however, possible that the sex ratio reflects TSD. Diamondback terrapin eggs that are subjected to cooler temperatures within the tolerated thermal range for this species produce male hatchlings whereas warmer temperatures produce females (see Jeyasuria et al., 1994; Roosenburg and Place, 1994; Roosenburg and Kelley, 1996). Terrapin nests in Bermuda are subjected to relatively high temperatures as they are laid in un-shaded sand bunkers on the Mid Ocean golf course (Davenport et al., 2005); this is likely to favour female hatchlings.

Terrapin biometrics

Diamondback terrapins in North America exhibit sexual dimorphism, with mature females being considerably larger than mature males. The data from the present investigation show that terrapins in Bermuda have the same characteristics; females had greater mean straight-line carapace lengths, straight-line plastron lengths, straight-line carapace widths, and shell heights than male terrapins. Furthermore, the mean mass of female terrapins in

Bermuda was significantly greater than the mean mass for males. Maximum sizes for both sexes (female SCL 196 mm, male SCL 134 mm SCL) were rather lower than those reported for U.S.A. terrapins (238 mm and 140 mm respectively) by Ernst et al. (1994).

Scute anomalies were observed in 34.3% of the terrapins in the Bermudian population. Variations in the number of vertebral, costal or marginal scutes have been reported from the U.S.A. (Wood and Herlands, 1997; Herlands et al., 2004). These typically involve extra, split, or distorted scutes, caused by high (30-32°C) incubation temperatures (i.e. thermal shock) (Wood and Herlands, 1997; Herlands et al., 2004), or possibly changes in available oxygen supply during incubation (Hildebrand, 1932) or embryological exposure to polycyclic aromatic hydrocarbons (Van Meter et al., 2006). The most common anomalies observed in the Bermuda population were extra vertebral, costal, and marginal scutes. It is possible that high incubation temperatures may be partly responsible for the moderately high number of individuals observed to have scute irregularities; however the degree to which this small and isolated population is affected by inbreeding is currently poorly understood.

No epibionts or major injuries were observed in any of the sampled terrapins in Bermuda; however, minor damage to the carapace was occasionally observed. Abrasions were superficial and appeared to have been caused by repeated rubbing against limestone rocks during periods of brumation. Terrapins on both the Gulf and Atlantic coasts of Florida are known to host barnacles (Ross and Jackson, 1972; Seigel, 1983). Various physical injuries related to boat strikes (Roosenburg, 1991; Cecala et al., 2008) or encounters with terrestrial and aquatic predators (Lovich and Gibbons, 1990; Hart and McIvor, 2008) have been documented in terrapin populations throughout the U.S.A. Barnacles and motorized watercraft are absent from all ponds inhabited by terrapins in Bermuda, and the majority of terrestrial and aquatic terrapin predators in North America (e.g. raccoons (*Procyon lotor*) and alligators (*Alligator mississippiensis*)) are not elements of Bermuda's fauna.

Population density

Bermuda's diamondback terrapins have an estimated population density of only 6.0 terrapins ha⁻¹. This estimate is based upon the total pond area where trapping occurred as well as the surrounding wetlands (i.e. the mangrove swamp and saw-grass marsh environments), but it does not include the peripheral activity range of the terrapins (i.e. the golf course fairways which are used by the terrapins to move between ponds and the sand bunkers used as nesting habitat).

Seigel (1984) estimated that 53-72 terrapins ha⁻¹ inhabited a salt marsh in central Florida, and Avissar (2006) indicated that 24-27 terrapins ha⁻¹ inhabited a single sub-tidal salt marsh creek in southern New Jersey. The average population densities of other similar sized emydid turtles have been reported to be 40 turtles ha⁻¹ (Congdon et al., 1986), 137 turtles ha⁻¹ (Zweifel, 1989), and 576 turtles ha⁻¹ (Gibbons, 1968) for *C. picta* from various freshwater ponds and marshes in the U.S.A., and 43-981 turtles ha⁻¹ for *T. s. elegans* in the freshwater pond environment in Bermuda (Outerbridge, 2008). Clearly the Bermudian diamondback population density is extremely low.

The low density of 6.0 terrapins ha⁻¹ in Bermuda might suggest that their brackish pond environment is a sub-optimal habitat. However, food appears plentiful and predators on adults and large juveniles are absent, so the low density could be the result of years of systemic pressures and stochastic events (natural catastrophes or stochasticity associated with demographics, environment, genetics) causing poor recruitment and low survivorship.

Growth rates

The limited dataset (22 individuals) indicated that annual growth in Bermuda's diamondback terrapins was most rapid for individuals classified as juvenile, an observation consistent with other studies. Chelonian somatic growth rates are considered to be much more rapid during the juvenile stage of development than the adult stage, particularly during the first few years after hatching (Wilbur, 1975; Dunham and Gibbons, 1990; Bjorndal et al., 2000). Female terrapins in Bermuda displayed significantly greater annual

growth in SCL than did males, despite females being significantly larger in initial SCL. However, far more of the males were mature than females. Seigel (1984) found that the growth of male and female Florida east coast diamondback terrapins was relatively constant during the first two years of life but then began to diverge at age three, at which point growth rates declined in males but female growth continued at a steady rate. The Bermudian data are consistent with this scenario.

Habitat Suitability

Of the four brackish water ponds currently inhabited by diamondback terrapins, the trapping data suggests that Mangrove Lake and South Pond are most utilised of the ponds. Seasonal temperatures appear to be relatively consistent between Mangrove Lake, South Pond and Trott's Pond; however the surface salinities vary greatly (Thomas et al., 1991; this study). Diamondback terrapins are known to move between water of different salinities in order to feed, mate and brumate as well as to maintain proper osmotic balance (Hart and Lee, 2006). Davenport and Macedo (1990) showed that when salt-loaded, diamondback terrapins avoid drinking when salinities range from 27-34 psu (the salinity range of Mangrove Lake; see Chapter 2) but drink large amounts when they range from 0-10 psu (the typical salinity range of South Pond; see Chapter 2). After a drinking bout, individuals often exhibit prominent edematous swellings in the skin of the pelvic and pectoral regions (Robinson and Dunson, 1976). This has been attributed to subcutaneous storage of freshwater which is subsequently utilized during periods of dehydration. The comparatively higher capture rate of terrapins in South Pond versus Mangrove Lake and Trott's Pond throughout the 2008-2010 study period, and the fact that many individuals captured from South Pond had obvious edematous swellings (M. Outerbridge, personal observation), suggests that South Pond may act as an important hydrating area for terrapins in the Bermuda population, particularly during periods of reduced rainfall.

Analyses of terrapin faecal samples (see Chapter 4) have shown that small aquatic gastropods are frequently consumed by terrapins of all sizes. Benthic biotic surveys (also see Chapter 4) showed that small gastropods

were more abundant and diverse within the mangrove swamp community surrounding Mangrove Lake than in the grass-dominated marsh at South Pond. Additionally, the mangrove wetlands at Mangrove Lake and Trott's Pond are approximately ten times greater in area than the grass-dominated marshes at South Pond and North Pond. These combined findings suggest that the mangrove wetlands have greater foraging potential for Bermuda's diamondback terrapins. Furthermore, these swamps and marshes also serve as important developmental habitats for hatchling terrapins (see Chapter 6).

Concluding comments

The current investigation has shown that Bermuda's diamondback terrapins have a small population, a very limited distribution, and low annual rates of recruitment – factors which suggest that this population is at high risk of local extirpation. It is presently not known how stable the population is as there are no other population estimates with which to compare the current results with. However, the data collected over the three year study period (2008-2010) will serve as an effective base-line for future investigations. Long-term research and monitoring of this vulnerable population is highly recommended and the continued use of the modified fish traps is suggested for future studies. As a direct result of this investigation, Bermuda's diamondback terrapins were classified in 2012 as a level II protected species and declared to be 'Vulnerable' under the Bermuda Protected Species Act (2003). It is considered an offense for an unauthorised person to be in possession of, export, or otherwise harm diamondback terrapins and offenders are liable, on summary of conviction, to a fine of \$15,000 or one year of imprisonment (www.laws.bm; accessed October 2012). A management and recovery plan (see Appendix 8) detailing the short-term and long-term survival goals for this species has been drafted and is currently under review for implementation by the Bermuda Government's Department of Conservation Services.

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Chapter 4: Feeding Ecology of Diamondback Terrapins in Bermuda

Abstract

The foraging ecology of the diamondback terrapin (*Malaclemys terrapin*) was investigated in Bermuda using a combination of direct observation, faecal analyses, and to a limited extent, necropsy. Bermuda's diamondback terrapins do not have as varied a diet as that reported for the species from its North American range. The most frequently consumed dietary items in the Bermuda population constituted small gastropods <3 mm shell height (66.7% of the faecal samples analysed). Scavenged fish and vertebrate animal remains each occurred in 19% of the faecal samples, terrestrial arthropods occurred in 14.3% of the samples while polychaete worms and bivalves each occurred in less than 3%. Sediment from the pond environment was found in 74% of the faecal samples, presumably incidentally ingested while foraging for the small benthic gastropods inhabiting the gelatinous pond sediment. Terrapins were commonly observed moving slowly along the bottom of the pond taking successive mouthfuls of sediment, often with the head completely buried. This behaviour is a novel observation for this species, but may be exposing the terrapins to harmful contaminants present within this medium. The distribution and abundance of arthropods and molluscs resident within the brackish water terrapin wetland environment were also assessed in three different habitats; benthic pond, mangrove swamp and grass dominated marsh. These surveys suggested that Bermuda's terrapins do not fully exploit the food resources present within those environments.

Introduction

The diamondback terrapin is the only species of turtle that has become specialized to inhabit the tidal salt marsh and estuarine environment, and exhibits unique physiological and behavioural adaptations that enable them to live within coastal salt marshes, mudflats, river estuaries, tidal creeks, brackish lagoons, and mangrove habitats along the Atlantic and Gulf coasts of North America (Cowan, 1971; Gilles-Baillien, 1973; Cowan, 1990; Davenport and Macedo, 1990; Hart and Lee, 2006).

Diamondback terrapins have been identified as an important component of the trophic dynamics of the salt marsh ecosystem (Silliman and Bertness, 2002; Davenport, 2011) and are carnivorous, feeding mostly upon a variety of marine molluscs and crustaceans (namely periwinkles, crabs, mussels and clams) throughout the North American range (see reviews in Butler et al., 2006; Ernst and Lovich, 2009). There is, however, a growing body of evidence to support the hypothesis that this terrapin may be a dietary generalist that is opportunistic in its foraging habits (Spivey, 1998; Petrochic, 2009; Butler et al., 2012; Erasmus, 2012). Diamondback terrapins show resource partitioning, whereby individuals with wider heads (the largest females) consume larger snails and crabs than terrapins possessing smaller heads (Tucker et al., 1995). Diamondbacks appear to be predators that use visual cues while foraging, showing selectivity in the prey that they eat (Davenport et al., 1992; Tucker et al., 1995; Tucker et al., 1997; Butler et al., 2012), and their food consumption is considered to be ten times higher than that of other closely related aquatic emydid turtles of the same size (Davenport and Ward, 1993).

The analysis of faecal material is a non-destructive and non-invasive way of examining dietary preference and has previously been used on several species of small turtles (Demuth and Buhlmann, 1997; Lima et al., 1997), including diamondback terrapins (Tucker et al., 1995; Spivey, 1998; Roosenburg et al., 1999; King, 2007; Petrochic, 2009; Butler et al., 2012; Erasmus, 2012). This method of dietary determination also has the added benefit of allowing multiple samples to be taken from a single individual over time; however, it is limited by the differential digestibility of the various hard and soft-bodied dietary components which in turn affects their representation within the faeces. Gastric lavage is a technique that has been employed to examine dietary preference in a wide variety of chelonians (Legler, 1977; Parmenter and Avery, 1990; Fields et al., 2000; Seminoff et al., 2002; Witherington, 2002; Caputo and Vogt, 2008) and can provide a more comprehensive picture of diet when used in combination with faecal analysis.

The diet of diamondback terrapins has been studied in various regions throughout their North American range; however, nothing is known

about their diet on Bermuda. The primary objective of the current investigation was to examine the diet and foraging ecology of Bermuda's terrapin population, with specific aims to (1) determine particular food preferences within the land-locked, brackish water pond environment, (2) to provide insight into any foraging behavioural adaptations that diamondback terrapins might display within this environment, and (3) to assess the abundance and distribution of gastropods within the ponds and adjacent wetland communities. Furthermore, it was envisaged that detailed knowledge of terrapin diet in Bermuda would allow appropriate conservation and management efforts to be directed towards protecting the areas in which they forage.

Methods

Study site

The entire known Bermuda population of diamondback terrapins is found only in four brackish water ponds named Mangrove Lake, South Pond, North Pond, and Trott's Pond (Fig. 4.1). All are located on a private golf course and all are separated by, at most, 380 m of land (straight-line distance between North Pond and Trott's Pond. Refer to Chapter 2 for a more complete description of the study site.



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Figure 4.1. Aerial photograph from 2003 showing the four diamondback terrapin ponds situated on the Mid Ocean golf course (A=Mangrove Lake, B=Trott's Pond, C=South Pond, D=North Pond).

Direct observation

Opportunities to observe terrapin behaviour in the ponds were afforded during periods of good water clarity. Of the four terrapin ponds, North Pond and Trott's Pond typically had very poor water clarity, making direct observation of the terrapins within them unfeasible. Only South Pond and Mangrove Lake went through periods in which clarity improved to an extent that allowed the terrapins foraging along the bottom to be viewed; and of the two, South Pond's small size, shallow nature, and ease of accessibility to all parts of the pond made it the best pond to observe foraging behaviour. The terrapins that reside in this pond have become habituated to the frequent presence of people (golfers) and readily forage in the presence of observers. Many of the observations were opportunistic in nature and occurred when the study site was visited while investigating other aspects of diamondback terrapin ecology (e.g. to monitor nesting effort, check traps, or listen for

radio-telemetry signals). As such, these periods of observation were random in timing and varied in duration. They occurred throughout the calendar year, and took place during night and day. Nocturnal surveys were accomplished with a low intensity LED flashlight that emitted a diffuse white light to minimise disturbance. Only diurnal foraging episodes were timed (using a digital stop-watch). In an effort to describe aspects of foraging behaviour for this species, the time that each individual terrapin spent actively foraging upon the bottom of the pond (bottom time) was recorded, as was the period that each individual spent at the surface of the pond between successive feeding sessions (surface interval).

Faecal analyses

Juvenile, immature and adult diamondback terrapins were opportunistically captured using a long-handled dip net from Mangrove Lake, South Pond, North Pond and Trott's Pond from March-September 2010 and January-October 2011. Biometrics and sex/stage classification for each terrapin were recorded following the methods described in Chapter 3. Each individual was kept outside in the shade for 48 hrs in covered, plastic storage bins that measured 55 cm long, 45 cm wide and 30 cm deep. Freshwater was added to a depth of 2.5 cm for ingestion to facilitate the passing of faeces. Any faecal material obtained was strained through a 1 mm sieve, oven dried at 80°C for 48 hours, and stored in a sealed glass vial for subsequent identification. Furthermore, faecal samples were collected from neonate terrapins (i.e. individuals that were less than one year old) that were followed as part of a radio-telemetry study (see Chapter 6). At the end of the tracking period, each individual was placed in a 500 ml plastic bowl containing enough freshwater to cover the carapace and held in a room with an ambient temperature of 30°C for 48 hours. Any faecal material obtained was strained through Whatman Grade No.1 46 mm filter paper, allowed to air dry for 48 hours and stored in a sealed glass vial. All terrapins captured during the faecal analysis investigation were released at their original capture location.

Each faecal sample was examined at magnifications between 10× and 25× using a stereoscopic microscope with an ocular scale. Food items

were identified to the lowest possible taxonomic level, and weighed to the nearest 0.0001g. The shells of gastropods, when encountered whole, were counted and total height was measured to the nearest 1.0 mm. For samples containing more than 200 shells, abundance was determined via estimation through use of a 1 cm² grid pattern on a Petri dish. The sub-sample was spread evenly over the grid and the number of shells within one square was counted. The final estimate was obtained by multiplying the number of shells in the square by the total number of squares containing shells.

Quantification of dietary items was accomplished by determining the percentage dry mass of each item relative to the total dry mass of each sample. The relative frequency of occurrence of each dietary item was determined by calculating the percentage of turtles containing a given food type in relation to the total number of turtles examined.

Gastric lavage

Gastric lavage was trialled, following Fields et al., (2000), on eight Bermuda red-eared sliders (*Trachemys scripta elegans*) hand netted from the wild. Feral populations of this introduced species exist across Bermuda (Outerbridge, 2008). Red-eared sliders are the subject of an eradication programme, and were chosen as a surrogate to practice the flushing technique upon because of availability and similarity in size. The gastric lavage results were partially successful (see Appendix 4); however, the decision was ultimately made not to use lavage on diamondback terrapins due to its invasive nature.

Necropsies

Herons have been identified as predators of small terrapins (Burger, 1976; Draud et al., 2004). The dissection of dead terrapins (e.g. those obtained from herons prior to consumption and those found freshly-dead in the wild) provided an opportunity to study feeding ecology through direct examination of the entire digestive tract. This method was highly opportunistic and limited, but provided a comprehensive picture of dietary items. Yellow-crowned night herons (*Nyctanassa violacea*) were observed foraging in South Pond between mid-April and mid-May 2010. Three days

were randomly chosen each week over a four week period during which three one hour surveys occurred at random times throughout each day between 06:00 and 19:00 hours. When a heron was observed attempting to ingest a small terrapin, hazing (the usage of loud noise) was used in an effort to scare the heron away from its prey (guns are illegal in Bermuda and could not be used to shoot the herons). Structured surveys looking for additional dead terrapins at the study site were not employed; instead the discovery of freshly dead specimens occurred stochastically while in the field. Those that were encountered were weighed, measured and necropsied.

Necropsies involved the removal of the oesophagus, stomach and intestines from the body cavity; their contents were scraped into a glass vial containing 75% ethanol alcohol for subsequent identification. Following identification, each sample was oven dried at 80°C for 48 hours, weighed to the nearest 0.0001g, and then stored in a sealed glass vial. The shells of gastropods, when encountered whole, were counted and total height (maximum measurement along the central axis) was measured to the nearest 1.0 mm.

Benthic biotic surveys within the terrapin wetland environments

Assessments of mollusc and crustacean abundance and distribution within the ponds and adjacent wetland environments were conducted to determine prey availability for Bermuda's diamondback terrapins. These assessments were accomplished by performing a series of benthic transects within three different habitats utilized by all size and age classes of Bermuda's diamondback terrapins; the sediment at the bottom of Mangrove Lake and South Pond, the red mangrove swamp community that surrounds Mangrove Lake, and the saw-grass marsh in the centre of South Pond.

Due to the different sizes of the water bodies, two belt transect surveys of benthic biota were performed in Mangrove Lake and one belt transect survey was performed in South Pond in July 2011. The Mangrove Lake transects were straight-line and followed an east-west direction (Transect 1) and a south-north direction (Transect 2) (Fig. 4.3), whereas the survey in South Pond was circular (Transect 3) (Fig. 4.4). Ten locations were haphazardly sampled along the path of each transect. The GPS

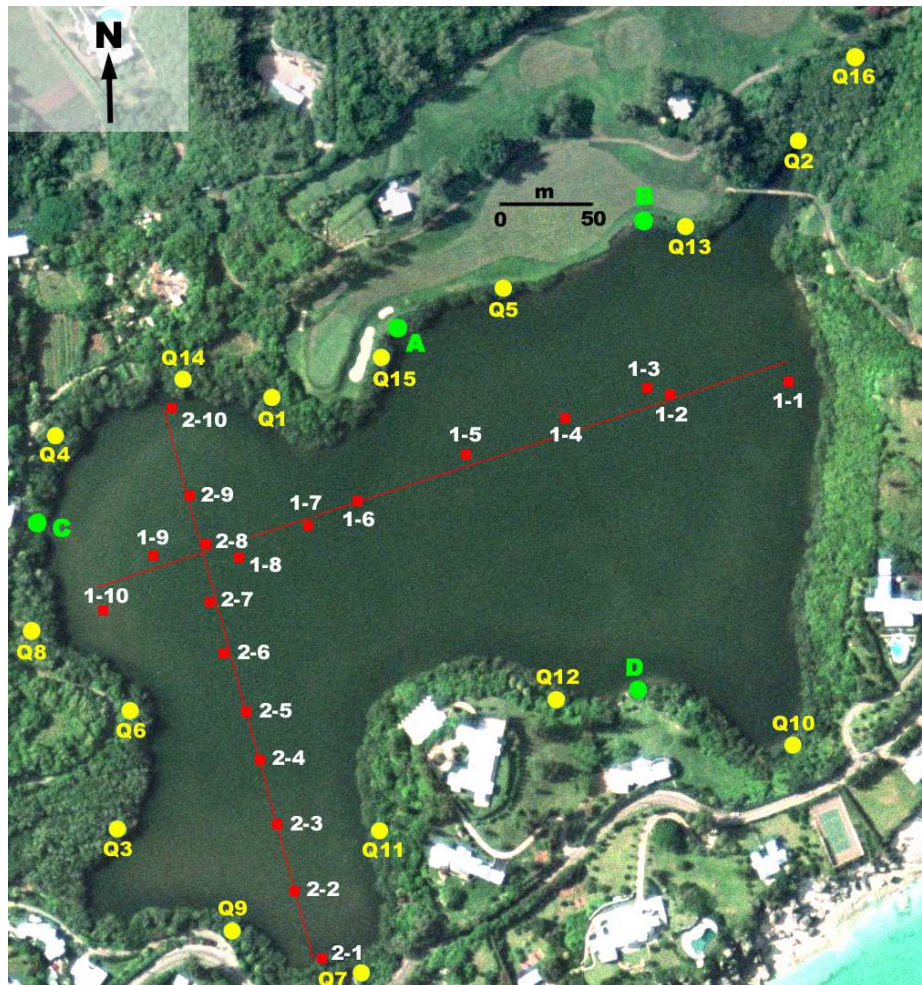
coordinates were recorded at each location along with a brief description of the benthic characteristics. Gastropod collection consisted of sweeping a dip net with 1 mm mesh and a square opening of 25 cm x 25 cm for a distance of 1 m and a depth of approximately 2.5 cm at the surface of the sediment (thereby sampling a linear area of 0.25m^2 at each location). The collected sediment was passed through a 1 mm mesh sieve at the surface of the pond and the material that remained was transferred into a one litre container. In addition to the belt transects, four replicate 25 cm x 25 cm quadrat surveys were performed at random in sand, rock, and gravel areas of Mangrove Lake (blue-coloured locations A, B, C, D in Fig 4.3.) The area defined by each quadrat was dredged to a depth of 2.5 cm and the contents transferred into a bucket and sorted by hand.

Sixteen replicate quadrat surveys were performed within the mangrove swamp that borders Mangrove Lake (Q1–Q16 in Fig. 4.3). The sites were haphazardly chosen, using an aerial map, at various locations around the periphery of the pond. Upon arrival in the field, a 25×25 cm quadrat was randomly placed upon the leaf litter immediately land-ward of the water-line. The area defined by each quadrat was dug to a depth of 2.5 cm and the contents transferred into a 3.8 litre sealable plastic bag. The contents of each bag were gently sifted in the laboratory using running water and a 5 mm sieve stacked on top of a 1 mm sieve.

Four replicate quadrat surveys were performed within the saw-grass marsh at the centre of South Pond (Q1–Q4 in Fig. 4.4). These sites were also haphazardly chosen using an aerial map. Upon arrival in the field, a 25×25 cm sample of saw-grass and turf was cut, to a depth of 2.5 cm, from the marsh at each of the four sites. The saw-grass blocks were transferred to separate 25 gallon buckets and taken to the laboratory for examination. Each sample was placed in a plastic bin measuring 60 cm long, 40 cm wide and 14 cm high, carefully broken apart and gently sifted in the laboratory using running water and a 5 mm sieve stacked on top of a 1 mm sieve. Shoot bundles were counted to determine saw-grass density.

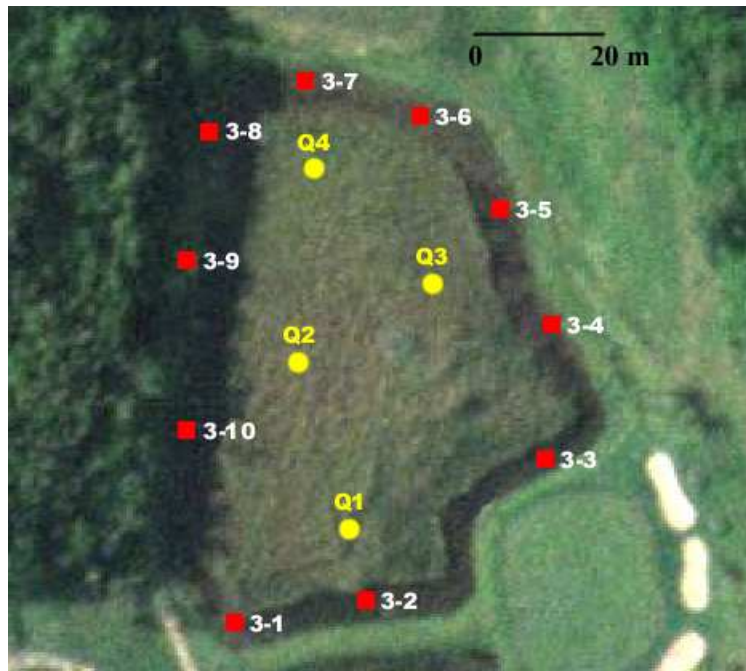
All biological specimens from the belt transect and quadrat surveys were kept for subsequent identification in the laboratory, but only living specimens were counted and measured (i.e. empty gastropod shells were

discarded). Live gastropods were counted, measured (total height mm), and frozen for eco-toxicological analyses (see Chapter 7). All other living biological specimens were returned to their original locations and released after identification. All transect and quadrat survey results were standardized on values m^{-2} .



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Figure 4.2. Benthic survey locations in Mangrove Lake, (red lines represent the belt transects while the red squares and associated white numerals represent the detritus sampling locations; green circles and associated letter notations represent the rocky substrate sample locations), and in the surrounding mangrove areas (yellow circles and alphanumeric notations represent the quadrat sample locations).



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Figure 4.3. Benthic survey locations in South Pond. (Red squares and white numerals show detritus sampling locations in this annular pond; yellow circles and alphanumeric notations represent the quadrat sample locations in the central marsh).

Results

Direct observation

Between 2008 and 2013, diamondback terrapins were commonly observed moving slowly along the bottom of South Pond taking successive mouthfuls of sediment during both diurnal and nocturnal surveys (Fig. 4.4). The head was often either fully or partially buried within the sediment during this behaviour, and sometimes the front limbs were used to rake the sediment laterally in front of the foraging individuals. These behaviours were observed being performed by female, male and juvenile terrapins. Individuals were seen foraging in this dredging-style manner upon open areas of sediment as well as immediately adjacent to the edges of the pond, often disappearing entirely from view under the banks of the pond or under the overhanging leaves of the saw-grass growing along the edges of the marsh at the centre of South Pond. Infrequent periods of water clarity in

Mangrove Lake permitted a limited amount of behavioural observations. Terrapins were observed at this location throughout the six year period, however no foraging was witnessed. Individuals were typically seen in transit, occasionally disappearing from view beneath mangrove branches overhanging the surface of the pond. Terrapins were frequently seen at the surface of both Trott's Pond and North Pond; however no foraging was witnessed within either due to poor water clarity.

The timed foraging observations in South Pond are summarized in Table 4.1. A total of twelve observations were made of foraging terrapins during the months April-August from 2010-2012 and included eleven adults (all female) and one juvenile. The submerged times that individual terrapins spent foraging on the bottom (bottom time) ranged from one minute and 15 seconds to 22 minutes and eleven seconds. Mean individual bottom times ranged from two minutes and 36 seconds to 16 minutes and ten seconds. The time spent at the surface between successive periods of foraging (surface interval) ranged from five to 54 seconds. Mean individual surface intervals ranged from 15 to 22 seconds (Table 4.1).



Source: Mark Outerbridge

Figure 4.4. Mature female diamondback terrapin showing dredging-style feeding behaviour while foraging in the sediment of South Pond. Note that head is completely buried in lower image.

Table 4.1. Timed foraging observations of twelve diamondback terrapins in South Pond.

Sex/stage	Bottom Time (min:sec)	Mean Bottom Time (min:sec)	Surface Interval (min:sec)	Mean Surface Interval (min:sec)
Juvenile	8:40	-	0:15	-
Adult female	2:02	2:36	0:18	0:16
	4:14		0:23	
	1:38		0:15	
	2:30		0:09	
Adult female	4:30	-	0:04	-
Adult female	5:35	-	0:10	-
Adult female	5:37	6:15	0:11	0:15
	5:18		0:19	
	7:05		0:07	
	7:03		0:22	
Adult female	4:13	-	0:11	-
Adult female	-	4:24	0:20	0:16
	6:07		0:16	
	2:41		0:12	
Adult female	1:15	3:21	0:54	0:22
	5:18		0:05	
	3:31		0:06	
Adult female	10:15	16:10	0:22	0:18
	15:20		0:17	
	22:11		0:14	
	16:56		0:20	
Adult female	3:54	-	0:27	-
Adult female	3:40	3:03	0:18	0:14
	2:27		0:10	
Adult female	5:58	-	0:15	-

Faecal analyses

A total of 54 diamondback terrapins were netted between March and September 2010 (n=21) and January and October 2011 (n=33) for the faecal analysis study, of which 42 (77.8%) produced faecal samples during the 48 hour confinement period (30 adults, four immature females, three juveniles of undetermined sex, and five neonates). Of the 54 terrapins, 30 were captured from South Pond (of which 23 or 76.7% produced faecal samples), 20 from Mangrove Lake (of which 15 or 75% produced faecal samples), three from North Pond (all of which produced faecal samples), and one was captured from Trott's Pond (which also produced a faecal sample).

Of the 42 terrapins that produced faecal matter, 28 (66.7%) were classified as female (24 mature, four immature) ranging from 126–196 mm

SCL (mean 172, SD 17.9) and six (14.3%) were classified as male (all mature) ranging from 114-134 mm SCL (mean 122, SD 8). Three (7.1%) were classified as juveniles (97-107 mm SCL, mean 102, SD 5), and five (11.9%) were classified as neonates (31-35 mm SCL, mean 33.7, SD 1.6).

Sediment occurred in 73.8% of the faecal samples, gastropods in 66.7%, plant material in 33.3%, fish and vertebrate animal bones in 19%, terrestrial arthropods in 14.3%, polychaete worms, bivalves, terrestrial crustaceans and trash (each 2.4% respectively) (Table 4.2).

Table 4.2. Dietary items obtained from 42 faecal samples of *Malaclemys terrapin* collected from four ponds in Bermuda. Symbols: n = number of samples containing a given food type; % = percentage of samples containing a given food type in relation to the total number of samples.

Dietary Item	n (%)
Sediment	31 (73.8%)
Plants (grass, seeds, algae)	14 (33.3%)
Gastropoda	28 (66.7%)
<i>Heleobops bermudensis</i>	24 (57.1%)
<i>Melanoides tuberculata</i>	15 (35.7%)
<i>Melampus coffeus</i>	2 (4.8%)
Insecta	6 (14.3%)
<i>Apis mellifera</i>	2 (4.8%)
<i>Berosus infuscatus</i>	1 (2.4%)
<i>Pheidole megacephala</i>	1 (2.4%)
<i>Julus</i> sp.	1 (2.4%)
Unidentified Lepidoptera larva	1 (2.4%)
Osteichthyes	
<i>Fundulus bermudae</i>	5 (11.9%)
Amphibia/Reptilia	3 (7.1%)
<i>Rhinella</i> (syn <i>Bufo</i>) <i>marinus</i>	2 (4.8%)
<i>Malaclemys terrapin</i>	1 (2.4%)
Polychaeta	
<i>Arenicola cristata</i>	1 (2.4%)
Bivalvia	
<i>Isognomon alatus</i>	1 (2.4%)
Crustacea	
<i>Armadillidium vulgare</i>	1 (2.4%)
Trash (cigarette filter)	1 (2.4%)

The gastropods comprised three species: *Heleobops bermudensis*, *Melanoides tuberculata*, (Fig. A4.5 in Appendix 4) and *Melampus coffeus*. *Heleobops bermudensis* occurred in 57.1% of all faecal samples and was

obtained from terrapins captured in South Pond, Mangrove Lake, and North Pond. *Melanoides tuberculata* occurred in 35.7% of the faecal samples but was only obtained from terrapins captured in South Pond, while *M. coffeus* only occurred in 4.8% of the faecal samples and was obtained from terrapins captured in Mangrove Lake.

The plant materials consisted mostly of mown grass fragments, saw-grass seeds and green algae. The terrestrial arthropods consisted of honey bees (*Apis mellifera*) (4.8% of the samples), small beetles (*Berosus infuscatus*), an isopod (*Armadillidium vulgare*), a millipede (*Julus* sp.), a big-headed ant (*Pheidole megacephala*), and an unidentified caterpillar (each represented in 2.4% of the samples). The fish and other vertebrate animal bones came from aquatic species and included fish from the family Cyprinodontidae - which occurred in 11.9% of the samples; an amphibian (*Rhinella marinus*) - which occurred in 4.8% of the samples; and a reptile (*Malaclemys terrapin*) - which occurred in 2.4% of the samples. The faecal samples containing arthropods and fish and vertebrate animal bones were acquired from terrapins captured in a variety of ponds. The samples that contained the polychaete worm (*Arenicola cristata*) and the shell fragments from the flat mangrove oyster (*Isognomon alatus*) came from terrapins captured in Mangrove Lake. The single sample that contained a wad of cotton (which was identified as having come from a cigarette filter) was obtained from a terrapin captured in South Pond. It is worth noting that most of the samples (n=33 or 78.6%) that contained sediment also contained other dietary items, whereas nine samples (21.4%) comprised only sediment. Female, male, and juvenile terrapins were all found to have ingested sediment, but none of the neonate terrapins produced faeces that contained sediment.

Table A4.5 in Appendix 4 summarises the dry mass of all dietary food items obtained from 33 terrapin faecal samples. The majority of the samples (n=20 or 47.6%) had a total dry mass of less than 1 g, and of the remaining 13 samples, only one had a total dry mass of over 10 g (note that sediment and vegetation were excluded from all calculations). The relative proportions of different dietary items in each faecal sample varied amongst terrapins (Table A4.6 in Appendix 4). Approximately half of the samples

(n=20) included more than one dietary item while nearly one third of the samples (n=13) consisted of only a single food item.

Tables A4.7 – A4.9 in Appendix 4 summarise the total number, size range, and mean size (with SD) for the three gastropod species obtained from the faecal samples of adult female, male and juvenile diamondback terrapins respectively. The mean size of *M. tuberculata* ingested by females (n=17) was 3.3 mm TH (SD 2.1 mm), the mean size of *H. bermudensis* was 1.8 mm TH (SD 0.8 mm), and the mean size of *M. coffeus* was 9.4 mm TH (SD 1.1 mm) (Table 4.7). The mean size of *M. tuberculata* ingested by males (n=3) was 2.1 mm TH (SD 1.0) and the mean size of *H. bermudensis* was 1.5 mm TH (SD 0.6 mm) (Table 4.8). The mean size of *M. tuberculata* ingested by juveniles (n=3) was 2.0 mm TH (SD 0.6 mm) and the mean size of *H. bermudensis* was 1.2 mm TH (SD 0.4 mm) (Table 4.9).

Table A4.10 in Appendix 4 summarises the total number, size range, and mean size (with SD) for the single species of gastropod obtained from the faecal samples of neonate diamondback terrapins (n=5). The mean size of *H. bermudensis* was 1.2 mm TH (SD 0.4 mm).

Table A4.11 in Appendix 4 shows the combined summaries of the total numbers, size ranges, and mean sizes (with SD) for the various species of gastropod obtained from the faecal samples of the adult female, adult male, juvenile, and neonate terrapins studied in this investigation. The total number of whole *M. tuberculata* from all faecal samples was estimated to be 2224 (size range 1-18 mm TH; mean 3.2 mm TH; SD 2.1 mm) of which 95% had been ingested by female terrapins, 4.5% by males and 0.6% by juveniles. The total number of whole *H. bermudensis* from all faecal samples was estimated to be 1910 (size range 1-5 mm TH; mean 1.8 mm TH; SD 1.1 mm) of which 86% had been ingested by females, 7.9% by males, 4% by juveniles and 2.1% by neonates. The total number of *M. coffeus* from all faecal samples was 13 (size range 7-11 mm TH; mean 9.4 mm TH SD 1.1 mm). These results show that *H. bermudensis* had been consumed by all age classes (i.e. adults, juveniles, and neonates), whereas *M. tuberculata* had been consumed by adults and juveniles and *M. coffeus* had only been consumed by adults.

The gastropod and bivalve shells were either found whole or in fragments in the faecal samples, often accompanied by their opercula. The majority of the insects were in various stages of disarticulation in the faecal samples (possibly because they were bitten at the time of ingestion or because of digestive processes) with the exception of the Lepidoptera larva, which was found whole and still vividly coloured. Small fish scales were occasionally found with the fish bones and one claw nail was discovered with the terrapin foot bones, which aided in the identification to species level.

Tables A4.12 – A4.15 in Appendix 4 show the number of the various gastropod sizes and Figs. 4.5 - 4.8 show the percentage frequency occurrence for the three different species of gastropods found within the pooled faecal samples of female, male, juvenile, and neonate diamondback terrapins. They clearly show that the vast majority of the snails ingested by the terrapins measured <3 mm TH, regardless of terrapin maturity status or sex. Most of the *H. bermudensis* snails measured <2 mm TH (85.5% within the female faecal samples, 92.2% within the male samples, 98.7% within the juvenile samples, and 100% within the neonate samples), and the majority of the *M. tuberculata* snails measured <3 mm TH (63.5% within the female faecal samples, 93% within the male samples, 100% within the juvenile samples). Nearly 70% of the *M. coffeus* snails ingested by the females measured between 9 mm and 10 mm TH.

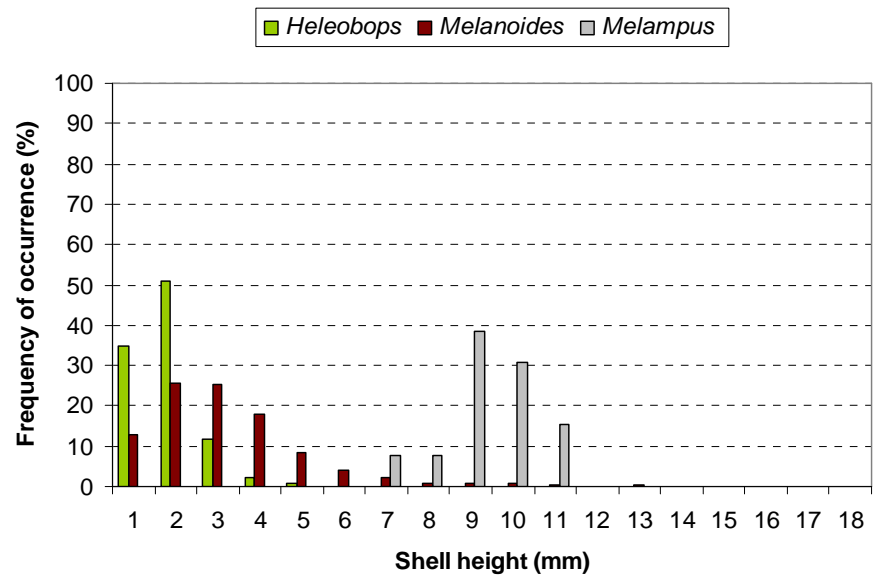


Figure 4.5. Frequency of occurrence of different sized *Heleobops bermudensis*, *Melanoides tuberculata* and *Melampus coffeus* in the faecal samples of female diamondback terrapins in Bermuda (n=17).

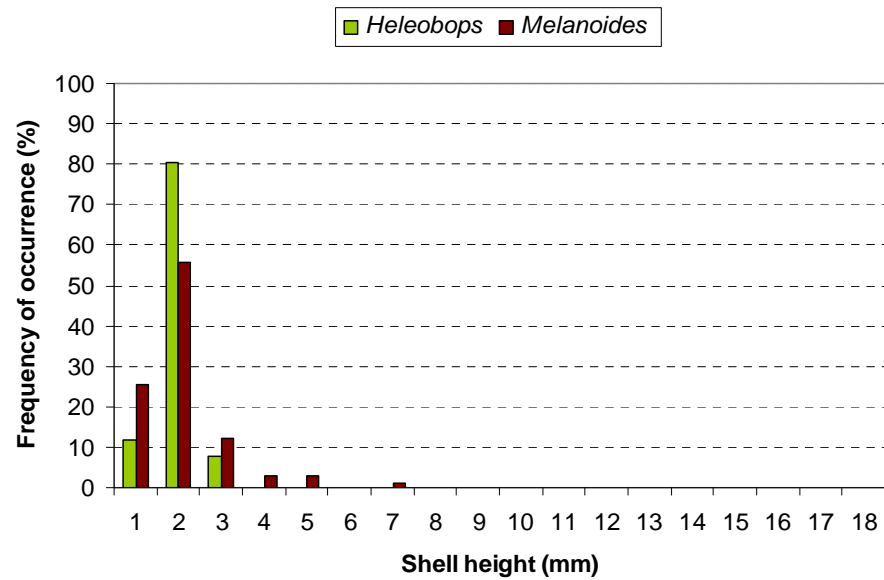


Figure 4.6. Frequency of occurrence of different sized *Heleobops bermudensis* and *Melanoides tuberculata* in the faecal samples of male diamondback terrapins in Bermuda (n=3).

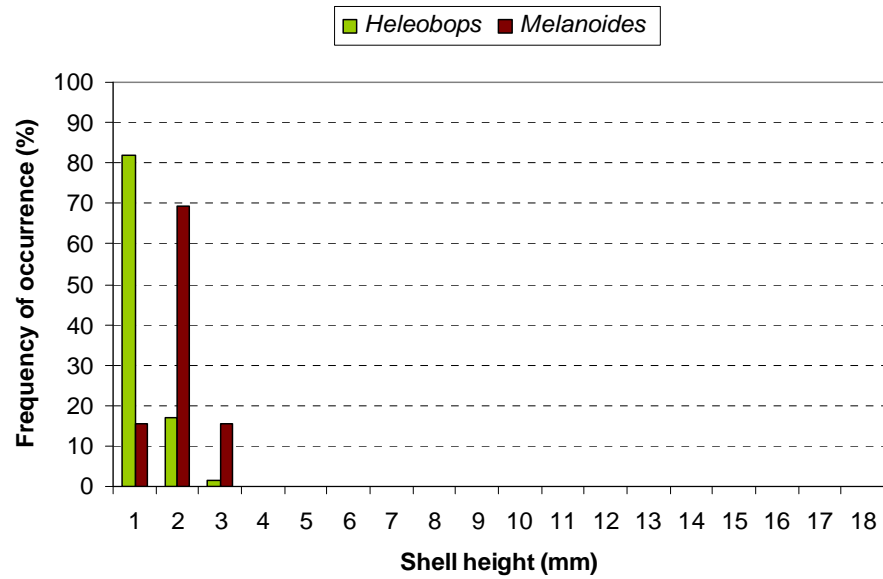


Figure 4.7. Frequency of occurrence of different sized *Heleobops bermudensis* and *Melanoides tuberculata* in the faecal samples of juvenile diamondback terrapins in Bermuda (n=3).

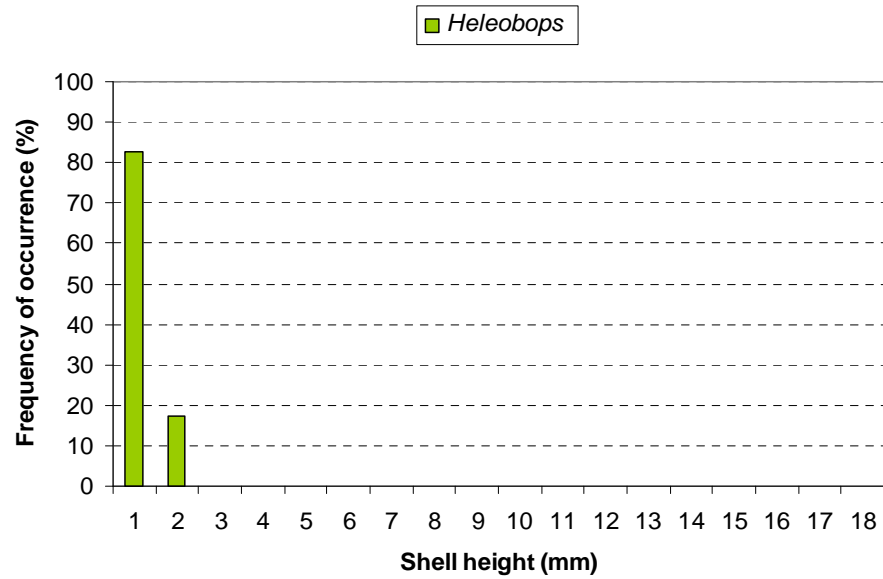


Figure 4.8. Frequency of occurrence of different sized *Heleobops bermudensis* in the faecal samples of neonate diamondback terrapins in Bermuda (n=5).

Necropsies

Ten small diamondback terrapins were recorded as being preyed upon by yellow-crowned night herons between April 19th and May 16th 2010; five during the scheduled heron observation surveys (see Tables A4.1 - A4.4 in Appendix 4) and an additional five were reported by members of the public over the same period. All occurred between 08:00 and 18:00 hours. Nine of these predation events were fatal to the terrapins (one neonate, with no obvious injuries, was rescued by Mid Ocean golf course maintenance staff and released back into South Pond). Of the nine fatally injured terrapins, only one (a 51 mm SCL juvenile) was obtained for necropsy (seven of the remaining eight were consumed by the herons and one was dropped into the pond and subsequently lost).

Additionally, six dead terrapins (two neonates, one juvenile, three adults) were discovered stochastically in South Pond and Mangrove Lake between 2010 and 2012. The juvenile measured 110 mm SCL, the three adults measured 142 mm, 151 mm, and 152 mm SCL respectively, and the neonates measured 29 mm and 30 mm SCL respectively. Only one neonate was fresh enough to necropsy; the remaining terrapins were either found as skeletons or in advanced stages of decomposition. Based on SCL, the three adults were determined to be females (see Chapter 3).

The stomach and intestinal contents of the predated juvenile (51 mm SCL) contained whole gastropod shells, crushed shell fragments, and flesh with the opercula still attached, all from the hydrobiid gastropod (*Heleobops* (syn *Paludetrina*) *bermudensis*) (Fig. A4.4 in Appendix 4). Shell heights ranged from 1-2 mm (mean 1.3 mm; SD 0.5; n=6), and the total dried mass of the stomach contents sample weighed 0.0345g. The stomach of the neonate encountered dead in the wild was found to be empty upon dissection; however the intestines contained whole shells and crushed shell fragments of *H. bermudensis*, as well as the chitinous body parts of a small water beetle (*Berosus infuscatus*). Shell heights ranged from 1-2 mm (mean 1.1 mm; SD 0.4; n=8), and the total dried mass of the intestine contents sample weighed 0.03494g. Thus, gastropods comprised 100% and 99.75% respectively of the total dried mass of these two samples. There was no significant difference between the shell heights of the *H. bermudensis* found

in the guts of the neonate and predated juvenile (Kruskal-Wallis; $H=0.82$, $p=0.365$), suggesting that both life history stages consume the same food resource.

Benthic biotic surveys within the terrapin wetland environments

Pond surveys

Only two species of aquatic gastropods were encountered during the Mangrove Lake surveys; the false horn shell (*Batillaria minima*) and the hydrobiid snail *Heleobops bermudensis*. Two species of aquatic gastropods were also encountered during the South Pond surveys; the hydrobiid snail *H. bermudensis* and the red-rimmed melania (*Melanooides tuberculata*).

Mangrove Lake:

Table A4.16 in Appendix 4 summarises the site location and benthic characteristics, as well as the relative abundance of *B. minima* and *H. bermudensis* encountered at each of the ten sample locations along Transect 1 in Mangrove Lake. The abundance of these gastropods varied along this transect; *B. minima* ranged from 0-56 snails m^{-2} (mean 15.2; SD 23.2; $n=152$), whereas *H. bermudensis* was more commonly encountered and ranged from 0-492 snails m^{-2} (mean 106.8; SD 160.7; $n=1068$). Both species were encountered in relatively low numbers at locations that comprised sediment only (*B. minima* 0-28 snails m^{-2} and *H. bermudensis* 0-192 snails m^{-2}). The abundance of both species increased significantly at locations where widgeon grass (*Ruppia maritima*) grew (*B. minima* 123 snails m^{-2} and *H. bermudensis* 252-492 snails m^{-2}) (see Fig. 4.9). Shell height of *H. bermudensis* along Transect 1 ranged from 1-3 mm (mean 1.6 mm; SD 0.5; $n=267$); *B. minima* ranged from 6.5-11 mm (mean 9.0; SD 1; $n=38$).

Table A4.17 in Appendix 4 summarises the site location and benthic characteristics, as well as the relative abundance of *B. minima* and *H. bermudensis* encountered at each of the ten sample locations along Transect 2 in Mangrove Lake. The abundance of these gastropods also varied along this transect; *B. minima* ranged from 0-20 snails m^{-2} (mean 3.2; SD 6.5; $n=32$), whereas *H. bermudensis* was more commonly encountered and

ranged from 0-772 snails m⁻² (mean 125.2; SD 255.5; n=1252). Both species were encountered in relatively low numbers at locations that comprised sediment only (*B. minima* 0-8 snails m⁻² and *H. bermudensis* 0-60 snails m⁻²). Again, the abundance of *H. bermudensis* increased significantly at locations where widgeon grass grew (range 380-772 snails m⁻²) (see Fig. 4.10). Shell heights of *H. bermudensis* along Transect 2 ranged from 1-4 mm (mean 1.8; SD 0.5; n=313); *B. minima* ranged from 7-10 mm (mean 8.3; SD 1.2; n=8). Pooling the data for each of the two separate transects in Mangrove Lake shows that *H. bermudensis* was more abundant than *B. minima* along the central axes of the pond.

South Pond:

Table A4.18 in Appendix 4 summarises the site location and benthic characteristics, as well as the relative abundance of *H. bermudensis* and *M. tuberculata* encountered at each of the ten sample locations along Transect 3 in South Pond. All of the sample locations comprised sediment and both snail species were encountered in very low numbers (*H. bermudensis* 0-4 snails m⁻², mean 0.4, SD 1.3; n=4; and *M. tuberculata* 4-20 snails m⁻², mean 13.2, SD 5.7; n=132) (see Fig. 4.11). Shell heights of *H. bermudensis* encountered along Transect 3 measured 1 mm TH and the shell heights of *M. tuberculata* ranged from 1-11 mm (mean 3.1, SD 2.0). The pooled data for Transect 3 shows that *M. tuberculata* was more abundant than *H. bermudensis* within the sediment of South Pond. Furthermore, *H. bermudensis* appeared to be more abundant within Mangrove Lake than in South Pond.

Further detailed analyses of gastropod abundances along all three study transects were attempted. The data were non-normal and variance was heterogenous whether the data were raw or square root transformed. The requirements of parametric statistics were therefore violated. Accordingly, a non-parametric approach was adopted. First, the abundances of *B. minima* were investigated. A Kruskal-Wallis test across the three transects showed that there were significant differences amongst the numbers of this species (Chi-Square = 7.885, df =2, p =0.019). Post-hoc tests using Mann Whitney

U tests were then conducted to compare Transect 1 with Transect 2, Transect 1 with Transect 3 and finally Transect 2 with Transect 3. This is not an ideal approach as there is an attendant risk of Type 1 error (i.e. incorrect rejection of a null hypothesis), but no better alternative is available. These post-hoc tests indicated that there were no significant differences in numbers of *B. minima* between Transects 1 and 2 (both from Mangrove Lake) (Mann-Whitney U = 36.50, Wilcoxon W = 91.50, Z= -1.153, p= 0.315). There were no significant differences in numbers of *B. minima* between Transects 1 and 3 (Mann-Whitney U = 33.00, Wilcoxon W = 88.00, Z= -1.302, p= 0.218), but there were significant differences between Transects 2 and 3 (Mann-Whitney U = 12.00, Wilcoxon W = 67.00, Z= -2.954, p= 0.003).

Second the same approach was adopted for the abundances of *H. bermudensis*. A Kruskal-Wallis test across the three transects showed that there were significant differences amongst the numbers of this species (Chi-Square = 12.76, df = 2, p = 0.002). Post-hoc Mann Whitney tests showed that abundances of *H. bermudensis* did not differ between Transects 1 and 2 (Mann-Whitney U = 39.00, Wilcoxon W = 94.00, Z= -2.954, p= 0.436), but did differ significantly between Transects 1 and 3 (Mann-Whitney U = 11.00, Wilcoxon W = 66.00, Z= -3.229, p= 0.002) and between Transects 2 and 3 (Mann-Whitney U = 12.50, Wilcoxon W = 67.50, Z= -3.117, p= 0.003). Overall these tests indicate that there is strong (but not conclusive) support for the abundance trends identified above.

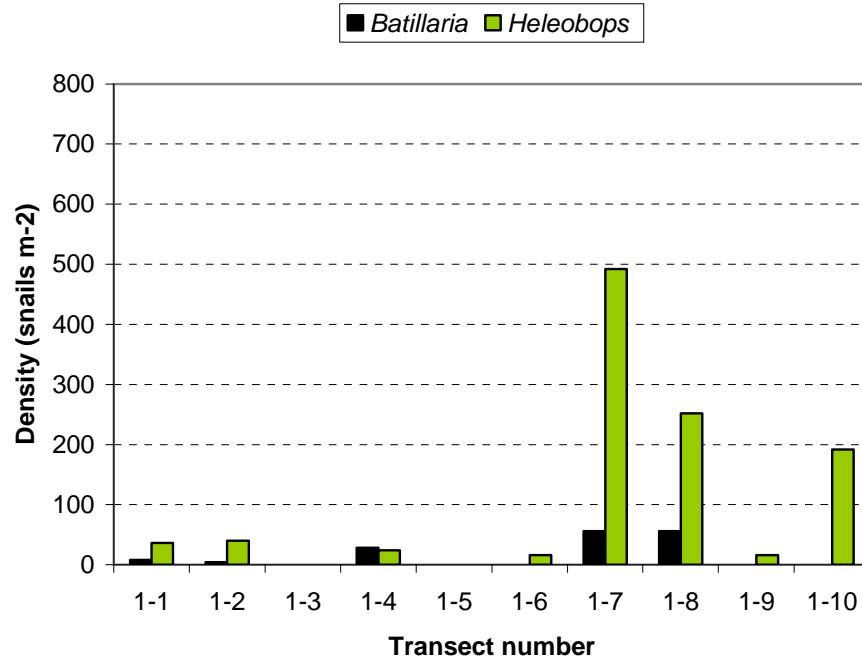


Figure 4.9. Densities of gastropods *Batillaria minima* and *Heleobops bermudensis* along Transect 1 (see Fig. 4.2) in Mangrove Lake.

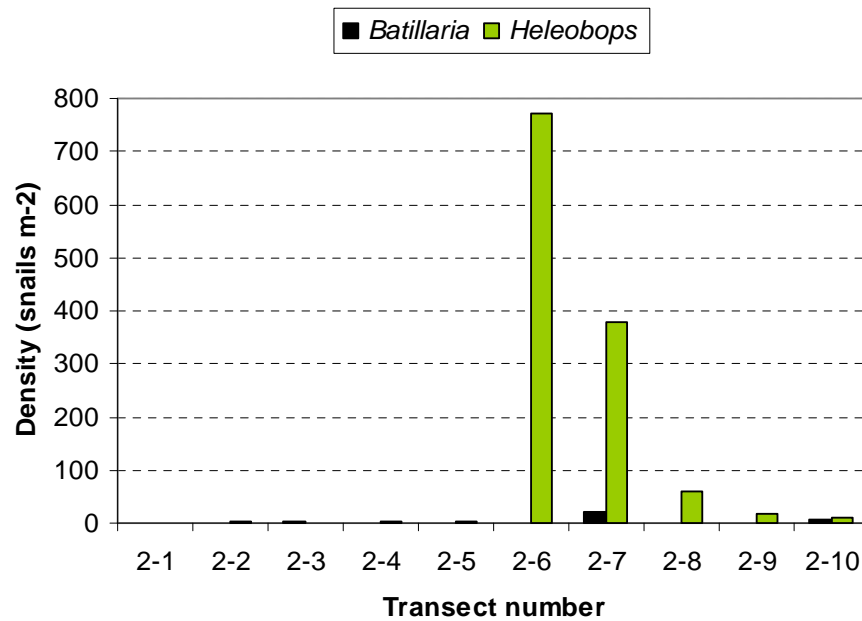


Figure 4.10. Densities of gastropods *Batillaria minima* and *Heleobops bermudensis* along Transect 2 (see Fig. 4.2) in Mangrove Lake.

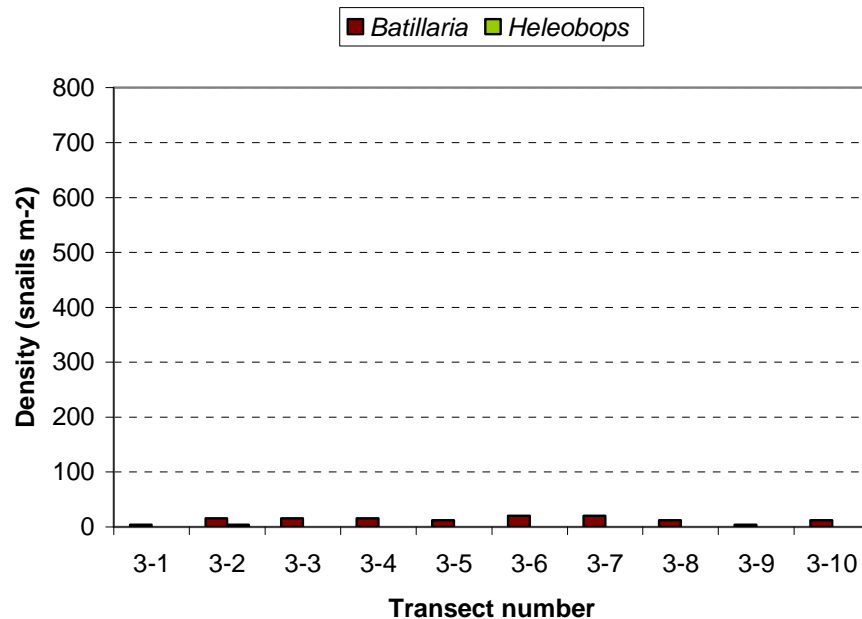


Figure 4.11. Densities of gastropods *Melanoides tuberculata* and *Heleobops bermudensis* along Transect 3 (see Fig. 4.3) in South Pond.

Table A4.19 in Appendix 4 shows the results of the four replicate quadrat surveys that were performed in the sandy, rocky, and gravelly areas of Mangrove Lake (locations A, B, C, D in Fig 4.3.) Only one species of gastropod (*Batillaria minima*) and one species of crustacean (the snapping shrimp *Alpheus armillatus*) were encountered. The snails were found most often attached to the rocky substrate, whereas the shrimp were found either buried within the gravel or hidden beneath rocks. The density of *B. minima* ranged from 2000-6752 snails m⁻² (mean 3596; SD 2211.4) and their sizes ranged from 3.5-10 mm TH (mean 6.4 mm); the density of *A. armillatus* ranged from 0-48 shrimp m⁻² (mean 20; SD 24) and their lengths ranged from 10-19 mm TL (mean 15.6 mm). These data suggest that the density of *B. minima* surveyed upon the rocky shoreline habitat (mean 3596 snails m⁻²) was nearly 400 times more than the mean density of live *B. minima* found upon the sediment along the central axes of Mangrove Lake (mean 9.2 snails m⁻²).

Mangrove swamp surveys

Figure A4.7 in Appendix 4 shows an example of the mangrove swamp habitat surrounding Mangrove Lake, and Table A4.20 in Appendix 4 summarises the various aquatic and terrestrial species discovered during the quadrat surveys (n=16) performed within this environment.

A total of five gastropod species were encountered; the coffee bean marsh snail (*Melampus coffeus*), the mouse-eared marsh snail (*Myosetella* (formerly *Ovatella*) *myosotis*), the Cuban marsh snail (*Laemodonta cubensis*), the small western marsh snail (*Microtralia occidentalis*), and the admirable stepping snail (*Pedipes mirabilis*). All were found within the detritus of the intertidal zone and some individuals of *M. coffeus* were also encountered attached to the red mangrove prop roots, usually in clusters, immediately above the water line of the pond. *Melampus coffeus* were most frequently encountered. Density for this species ranged from 0-1168 snails m⁻² (mean 282; SD 399.3; n=4512), and shell height ranged from 2-15 mm TH (mean 8.8, SD 3.2). *Myosetella myosotis* was the second most frequently encountered gastropod; however all were found in just one location (Fig. 4.3, Q12). Sizes ranged from 1-6 mm TH (mean 2.8; SD 1.2; n=848). *Laemodonta cubensis* was encountered in densities of 80 snails m⁻² and all occurred in one location (Fig. 4.3, Q12). Sizes ranged from 1-3 mm TH (mean 1.8; SD 0.8). *Microtralia occidentalis* and *Pedipes mirabilis* were infrequently encountered. Sizes of the former ranged from 6-7 mm TH (mean 6.3; SD 0.6; n=48), and the latter ranged from 2-3 mm TH (mean 2.3; SD 0.6; n=48).

In addition to the gastropods mentioned above, four species of crustaceans were encountered among the detritus; three were aquatic (unidentified amphipod species, the wharf louse (isopod) (*Ligia baudiniana*), and the isopod *Armadilloniscus ellipticus*) while the fourth was the terrestrial common sow bug (isopod) (*Armadillidium vulgare*). The amphipods were the most abundant crustaceans encountered during the mangrove swamp surveys, and were found in 81.3% of the quadrat locations. Densities ranged from 0-2272 m⁻² (mean 371; SD 656.8; n=5936). *Armadilloniscus ellipticus* was the second most frequently encountered crustacean, with densities of 0-1008 m⁻² (mean 197; SD 311.5; n=3152).

Ligia baudiniana and *A. vulgare* were not commonly encountered. Figure A4.8 in Appendix 4 shows the densities (expressed as the frequency of occurrence m^{-2}) of gastropods and crustaceans for all 16 mangrove swamp quadrat surveys performed in the mangrove swamp areas around Mangrove Lake.

Eggs from the endemic Bermuda killifish (*Fundulus bermudae*) were encountered in 25% of the quadrat surveys. Abundance varied from 0-3824 eggs m^{-2} (mean 313; SD 958.5; $n=5008$). The eggs were usually found hidden within the leaf detritus, but also attached to the red mangrove prop roots at the high water mark.

A variety of primarily terrestrial organisms were occasionally encountered in low densities within the 16 quadrat locations; these included millipedes, earwigs, small spiders, earthworms, small beetles, and a lepidopteran larva (Table A4.20 in Appendix 4).

Saw-grass marsh surveys

Figure A4.9 in Appendix 4 shows an example of the saw-grass marsh habitat in the centre of South Pond, and Table A4.21 in Appendix 4 summarises the various aquatic and terrestrial species discovered during the quadrat surveys performed within this environment.

Only one species of gastropod was found during the quadrat surveys (*Heleobops bermudensis*). Densities ranged from 176-272 snails m^{-2} (mean 208; SD 43.3; $n=832$), and shell heights ranged from 1-4 mm TH (mean 2.3 mm, SD 0.7 mm). Terrestrial organisms were infrequently encountered within the quadrats and included millipedes and small spiders. The number of saw-grass shoot bundles ranged from 16-48 m^{-2} .

Discussion

Molluscan and crustacean abundance and distribution within the terrapin wetland environments

Only two species of aquatic gastropods were encountered during the benthic surveys of Mangrove Lake; *B. minima* and *H. bermudensis*. Two species of aquatic gastropods were also encountered during the benthic South Pond

surveys; *H. bermudensis* and *M. tuberculata*. *Batillaria minima* is a small operculate snail (attaining 15 mm TH) belonging to the family Potamididae which inhabits intertidal habitats in Bermuda (e.g. mudflats and anchialine ponds), often occurring in enormous numbers (Sterrer, 1986). *Melanoides tuberculata* is a burrowing, deposit-feeding thiarid gastropod that is known to reach densities of 10,000-23,000 snails m⁻² (Roessler et al., 1977; Thompson, 2004). This gastropod is considered to be primarily a freshwater species that is native to tropical and sub-tropical regions of southern Asia and northern Africa (Clench, 1969). It was introduced to the U.S.A. during the 1930s (Benson and Neilson, 2012) by the aquarium pet trade (Murray, 1971) and is known to replace native snails in regions when introduced (Thompson, 2004). Roessler et al. (1977) reported that *M. tuberculata* had adapted to saline conditions within the estuarine environment in South Florida. *Melanoides tuberculata* appears also to have reached Bermuda through the aquarium trade (unpublished data). *Heleobops bermudensis* is a very small operculate snail, belonging to the family Hydrobiidae, which primarily inhabits brackish water ponds in Bermuda (see Pilsbry in Vanatta, 1911). Hydrobiid snails are a diverse group of gastropods, globally consisting of over 200 genera and approximately 1000 species (Thompson, 2004).

The results of the quadrat and transect surveys revealed that the sediment surface in Mangrove Lake and South Pond generally showed a paucity of gastropods; however *B. minima* and *H. bermudensis* were both found to exist in higher densities in localized patches throughout Mangrove Lake. *Batillaria minima* was most often associated with sand, rock and gravel substrate reaching densities *ca.* 6750 snails m⁻², whereas *H. bermudensis* was more commonly found within beds of widgeon grass in densities up to 772 snails m⁻². Benthic mapping of Mangrove Lake was not performed, but visual assessments of the pond in 2011 suggested that both the gravel/rock and widgeon grass environments comprised a very small proportion of the total pond area.

Gastropods were more abundant and diverse within the wetlands surrounding the ponds (Mangrove Lake swamp with 282 snails m⁻² and South Pond saw-grass marsh with 208 snails m⁻²) than in the benthic pond

sediments (125.2 snails m⁻² and 0.4 snails m⁻² at Mangrove Lake and South Pond respectively). Five species of gastropods (all pulmonates of the Family Melampidae) were encountered during the quadrat surveys within the detritus of the mangrove swamp intertidal zone surrounding Mangrove Lake. Pulmonate snails do not possess an operculum and are commonly found in moist, muddy areas at and slightly above the high tide line around marshes and mangrove swamps in Bermuda. *Melampus coffeus* grow to 20 mm TH, but the other remaining species rarely exceed 8 mm TH (Sterr, 1986). Thomas et al. (1992) and Herjanto (1994) reported that *M. coffeus* was frequently encountered upon the detritus and prop roots of mangrove trees in Mangrove Lake and Trott's Pond, often within 24 cm of mean low tide level. The survey results of the present investigation indicate that the gastropods within Bermuda's saw-grass marsh and mangrove swamp environments can reach densities of up to 1168 snails m⁻² (*M. coffeus*).

The flat mangrove oyster (*Isognomon alatus*) is a bivalve species that grows in clumps on the submerged prop roots of red mangrove trees in Mangrove Lake and Trott's Pond and has been reported to reach densities of 250 oysters/root or approximately 2700 oysters m⁻² of pond (Thomas and Dangeubun, 1994). The species was not surveyed during the present investigation as it was not recorded as a frequent dietary item in terrapins.

Crustaceans were rarely encountered within the aquatic environment of Mangrove Lake (none during the two transect surveys across the pond and only one species (*Alpheus armillatus*) was found in the rocky habitat quadrat surveys), and no crustaceans were encountered within South Pond. However, crustaceans (mostly small amphipods and isopods) were frequently encountered (87.5%) in the quadrat surveys performed in the mangrove swamp surrounding Mangrove Lake. The mangrove crab (*Goniopsis cruentata*) was not encountered during the present study though it was reported to inhabit the intertidal zone of Mangrove Lake and Trott's Pond two decades ago (Thomas et al., 1992).

Overall, the results of the current investigation indicate that the swamp wetlands adjacent to Mangrove Lake and South Pond do not appear to be food limited for the small population of terrapins that forage within them.

Foraging behaviour

Food consumption in diamondback terrapins has been shown to be affected by water temperature; appetite is stable from 20-35°C (Davenport and Ward, 1993). The mid-water temperatures for South Pond and Mangrove Lake were well within those values during the June-September trapping sessions (see Fig. 2.14, Table 2.1 and Table 2.2 in Chapter 2). Visual observations made at South Pond showed that Bermuda's diamondback terrapins spend relatively short periods of time submerged while foraging. Moll and Legler (1971) reported that juvenile red-eared sliders spend from 20 seconds to five minutes submerged while foraging, and adult sliders had submergence times of five to six minutes. Bermuda's adult terrapins spent comparable amounts of time submerged and foraging (mean six minutes 30 seconds), and the surface intervals between successive benthic foraging sessions were all brief in duration (mean 16 seconds) indicating that foraging bouts involve aerobic dives.

The terrapins observed in South Pond do not appear to use visual cues to identify gastropods for consumption. The benthic sediment in all of the terrapin ponds is gelatinous and extremely flocculent which allows the terrapins to both easily move through it and ingest it, apparently allowing them to consume *M. tuberculata*, the most frequently encountered gastropod within the pond's sediment (Outerbridge and Davenport, 2013). In support of this hypothesis, faecal analyses (from this study) confirm that Bermuda's terrapins consume large numbers of small *M. tuberculata* and *H. bermudensis* together with large quantities of sediment. The sediment is believed to have been incidentally rather than deliberately ingested.

Diamondback terrapins in the North American range live in environments affected by high tidal variability, and are able to forage in the upper reaches of salt marshes and mangrove swamps during periods of high tide. Tucker et al. (1995) surmised that feeding coincided with swimming activity at high tides when more snails were available, and marsh-dwelling crabs were more active. These authors reported observing large female terrapins feeding in areas of high snail density within the flooded *Spartina* marsh-land in South Carolina. Individual terrapins were seen consuming snails within reach and pushing over blades of *Spartina* to feed upon the

out-of-reach snails attached to the stems. Davenport et al. (1992) conducted a series of experiments to show how terrapins manage dangerous prey (crabs). Their results showed that terrapins were selective in the sizes of prey consumed and exhibited behavioural adaptations which allow them to minimize the risk of being injured by their prey; small crabs were eaten whole, medium-sized crabs were not killed, but rather had their rear legs cropped (i.e. the walking legs furthest from the chelipeds), and large crabs were generally avoided. Visual detection of moving prey has been identified as being of paramount importance to the foraging success of the slider turtle (Parmenter and Avery, 1990), and it generally appears to be the case for diamondback terrapins as well. The non-selective, deposit-feeding strategy observed in the Bermuda population of terrapins is evidently an adaptation that has allowed them to take advantage of the small benthic gastropods inhabiting the gelatinous pond sediment. This behaviour has not been reported previously, possibly because of the inconspicuous nature of diamondback terrapins and the turbid waters of the brackish coastal environments in North America where they feed.

Faecal analyses

Faecal analysis is the most common method for determining the dietary composition of wild diamondback terrapins (see reviews in Butler et al., 2006; Ernst and Lovich, 2009), and was the principal methodology used in the present study. Diamondback terrapins have been described as dietary generalists by some researchers (King, 2007; Petrochic, 2009), and this appears to be the case with Bermuda's population inhabiting the brackish water ponds. The vegetation found in the faeces of the terrapins studied in the current investigation is believed to have been inadvertently ingested. All of the insects are thought to have been consumed after falling into the ponds, rather than having been ingested in the terrestrial environment (with the exception of those consumed by the neonate terrapins which are residents of the intertidal mangrove and grass-dominated marsh environments adjacent to the ponds) (see Chapter 6). The polychaete worm and the fish, toad, and terrapin bones discovered in the faecal samples indicate that Bermuda's terrapins also scavenge on animal remains.

Carcasses of these species are periodically observed floating at the surface of the study ponds and it is likely that they are opportunistically ingested when encountered. Scavenging has been reported for other diamondback terrapin populations in the U.S.A. (Ehret and Werner, 2004; Petrochic, 2009; Butler et al., 2012).

A literature review revealed that at least 21 studies have reported on diamondback terrapin diet (17 involving wild terrapins and four involving captive terrapins), including eleven that used faecal analyses to determine diet composition (Table A4.22 in Appendix 4). These studies suggest that diamondback terrapins are opportunistic and carnivorous feeders that consume a wide variety of prey items within wetland habitats. Gastropods (e.g. *Littorina irrorata* and *Ilyanassa obsoleta*) predominate in the diet of some terrapin populations in South Carolina and New York (Tucker et al., 1995; Petrochic, 2009), while bivalves (e.g. *Mya arenaria* and *Mulinia lateralis*) predominate in the diet of other terrapin populations in Maryland, NE Florida and New York (Roosenburg et al., 1999; Butler et al., 2012; Erasmus, 2012), and crustaceans (e.g. *Callinectes sapidus*) were reported to dominate the diets of some terrapins in North Carolina (Spivey, 1998). Such investigations indicate that the diet of terrapins reflects geographic variations in prey availability and spatial distribution as well as food accessibility, especially in regions of high tidal amplitude. Butler et al. (2012) hypothesised that female terrapins, when forced to leave their resident tidal creeks where they normally forage, take advantage of alternative food sources in the areas where they nest. There is also evidence that indicates diamondbacks show dietary partitioning which is related to the ontogenic niche of terrapins. Tucker et al. (1995) demonstrated that the salt marsh periwinkle (*Littorina irrorata*) comprised up to 79% of the total dietary volume of the terrapins studied in their investigation, and that small terrapins specialised in consuming small snails, medium-sized terrapins consumed both small and large snails as well as crabs, and large terrapins spread their dietary preferences more evenly among the various sized snails and crabs. Davenport et al. (1992) and Petrochic (2009) studied jaw gape in relation to prey selection and both found that gape and bite force appeared to be the main constraints when feeding upon molluscs. Only those snails

and bivalves which could be either swallow whole or crushed between the terrapin's jaws were consumed; thus females (who possess the largest heads and therefore can deliver the strongest bite force) are more capable of ingesting larger, more heavily armoured prey items than smaller-sized terrapins. Tucker et al. (1997) found that diamondback terrapins in South Carolina did not eat a common and abundant gastropod (*Ilyanassa obsoleta*) despite the fact that it was readily accessible to terrapins foraging upon the salt marsh lower intertidal zone. They postulated that the greater shell strength of *Ilyanassa* led to higher processing costs which may deter predation by terrapins, despite the lower search costs and equivalent energetic returns relative to other gastropods (e.g. *Littorina*). This may partially explain why diamondback terrapins in Bermuda were not found to consume *B. minima*, which occurred in localized high densities (up to 6750 snails m⁻²) within Mangrove Lake. It is feasible that the robust shell architecture of *B. minima* may be providing protection from terrapin predation; however a crushing force investigation is needed to confirm this.

Analyses of the terrapin faecal samples of the Bermuda population suggest that *H. bermudensis* and *M. tuberculata* are the most frequently consumed gastropods. *Melampus coffeus* does not appear to be an important dietary food item for Bermuda's terrapins, and *M. myosotis*, *L. cubensis*, *M. occidentalis* and *P. mirabilis* do not appear to be consumed at all. Gastropods belonging to the genus *Melampus* were identified as a dietary component of wild diamondback terrapins (Coker, 1906; Spivey, 1998; King, 2007; Petrochic, 2009), and diamondback terrapins are known to readily consume *Melampus* in captivity (Allen and Littleford, 1955; Davenport et al., 1992). *Melampus coffeus* was found to be the most abundant gastropod within the intertidal zone of the mangrove swamp surrounding Mangrove Lake, however it was only found in 4.8% of the faecal samples examined in the present investigation. Petrochic (2009) reported that the bite force required to crush *M. bidentatus* was within the capabilities of male and female terrapins, thus it is likely that Bermuda's male and female terrapins are equally capable of consuming *M. coffeus*. It is currently unclear why Bermuda's diamondback terrapins are not exploiting this comparatively abundant food source. Future studies should focus on

examining additional faecal material, especially in the neonate and small juvenile size class (i.e. 30-90 mm SCL range) to determine the extent to which the small gastropods within the marsh wetlands are being consumed. It is equally unclear why Bermuda's terrapins do not appear to be exploiting the flat mangrove oyster as a food resource since it is among the most abundant, visible and sedentary of all the molluscs inhabiting Mangrove Lake and Trott's Pond.

Diamondback terrapins throughout the northern part of the North American range were found to frequently ingest the salt marsh snails *Littorina irrorata* and *L. saxatilis* (Coker, 1906; Tucker et al., 1995; Spivey, 1998; Petrochic, 2009; Butler et al., 2012; Erasmus, 2012). The modal number of *Littorina* excreted by terrapins captured from a South Carolina salt marsh was 16, but some passed as many as 90 snails (Lovich et al., unpublished data in Ernst and Lovich, 2009). Shell heights for *Littorina* excreted by male and female terrapins in the U.S.A. ranged from 2-15 mm and 4-21 mm respectively (Lovich et al., unpublished data in Ernst and Lovich, 2009). Nine species of periwinkles belonging to the family Littorinidae are described from Bermuda, but only one (the Atlantic mangrove periwinkle (*Littorina angulifera*)) is resident in the mangrove swamp community (Sterrer, 1986); however it is not present in Mangrove Lake nor Trott's Pond (Herjanto, 1994) and therefore is inaccessible to Bermuda's terrapins.

Hydrobiid snails have been identified as dietary items from terrapins in New York (Petrochic, 2009; Erasmus, 2012); however these snails did not comprise a large percentage of the terrapin diet, whereas they appear to be the most commonly consumed gastropod in the Bermuda population. The mean number of *H. bermudensis* defecated by male and female terrapins in Bermuda was 112 snails (range 1-926).

The thiarid snail *M. tuberculata*, the second most commonly consumed snail in the Bermuda terrapin population, has never been reported from previous studies and thus represents a novel food item for this species of reptile. *Melanoides tuberculata* is a relatively recent arrival to Bermuda (M. Outerbridge, unpublished data) and has also been identified as a host for several species of parasitic trematode worms (Pinto and de Melo, 2011)

which are known to affect the health of waterfowl, fishes and mammals (including humans) (Penner and Bernard, 1963; Mitchell et al., 2007). It is currently unknown whether this species of thiarid snail is host to any parasites in Bermuda and whether they would compromise the health of the terrapins that consume them. Bermuda's terrapins exploit this food resource in South Pond and may be regulating the population dynamics of this exotic prey item; such top-down control could explain the lower snail densities found in the present investigation in comparison with the significantly higher densities of *M. tuberculata* reported for areas in the U.S.A. (Roessler et al., 1977; Thompson, 2004). Exotic species that become established in non-native regions of the world can become extremely abundant and may eventually comprise a large part of the prey base available to native predators (see discussion in Carlsson et al., 2009). Further studies that examine the extent to which diamondback terrapins are providing biological resistance against the invasion of *M. tuberculata* (as well as the effect that consuming them has upon terrapin health and fitness) are warranted.

King (2007) examined the feeding ecology of neonate terrapins in the salt marshes of the Long Island Sound in New York state, and found that they were dietary generalists that selected food items based on abundance and availability. This smallest size class consumed a wide variety of prey organisms, with crustaceans (green crabs (*Carcinus maenas*) and amphipods belonging to the genus *Orchestia*) being the most commonly found in the faecal samples examined; however, insects and marsh snails (*Melampus bidentatus*) also occurred in the samples. *Heleobops bermudensis* was the only gastropod species found within the faecal samples of the neonate terrapins in the Bermuda study, despite the fact that small gastropods from numerous snail species inhabit the wetland communities. The quadrat surveys performed within the mangrove swamp surrounding Mangrove Lake suggested that the smallest cohorts of the pulmonate snails encountered are small enough to be ingested by neonate and small juvenile terrapins. It is possible that the small sample size in the faecal (five neonates) and necropsy (one neonate, one juvenile) investigations in the present study may be a reason why none of those gastropods were found in the diet of this smallest size class of terrapin in Bermuda. Other potential

food items for neonate and small juvenile terrapins encountered during the quadrat surveys included amphipods (which occurred in 81% of the survey sites and in densities of up to 2272 m⁻²) and killifish eggs (which occurred in 25% of the survey sites and in densities of up to 3824 m⁻².) In Jamaica Bay, New York, diamondback terrapins are known to ingest the eggs of Atlantic horseshoe crabs (*Limulus polyphemus*) (Erazmus, 2012), whose eggs are close in size to those deposited by killifish (*ca.* 2 mm) (Leschen et al., 2006; Outerbridge et al., 2007). More faecal samples are required from neonate and small juvenile terrapins inhabiting the Mangrove Lake swamp to determine whether pulmonate snails, amphipods, and *Fundulus* eggs are dietary items. Finally, the absence of sediment in the neonate faecal samples suggests that this size class in Bermuda shows more selectivity when foraging than do larger juveniles and adults within the benthic pond environment.

Necropsies

Care should be taken in the interpretation of results when analyzing samples from dead terrapins as the diets of these animals may not reflect the diets of healthy individuals. This caveat, however, is less applicable to terrapins that have been preyed upon by herons since the death of these terrapins may be independent of health status (i.e. illness). The necropsy results for the two terrapins examined in the current study confirm the results obtained via faecal analysis – chiefly that Bermuda’s terrapins predominantly consume small gastropods and insects found within their wetland environments. The small sizes of the necropsied terrapins (29 and 51 mm SCL) and the location where they were encountered suggest that these individuals were residents of the marsh at the centre of South Pond.

Concluding comments

The faecal analyses and, to a more limited extent, the necropsies have shown that diamondback terrapins in Bermuda are dietary generalists that appear to favour the consumption of small gastropods. The range of food items is less than those reported from North America, however this may be due to the fact that there is less diversity among prey species present within

the pond habitat in Bermuda in comparison with those found within the salt marshes of the U.S.A. For example crabs, which are a relatively abundant element of the salt marsh environment and an important food item for terrapins in North America, are cryptic and rare in Mangrove Lake and Trott's Pond (Thomas et al., 1992) and absent from South Pond and North Pond (M. Outerbridge, personal observation).

The quadrat survey results in the mangrove and saw-grass marshes of the present study indicate that these environments do not appear to be food limited, especially for neonate and small juvenile terrapins. In contrast, the benthic surveys within the ponds show that gastropod abundance is unevenly distributed and generally low within the sediment, but is higher in localized areas where rocky substrate or widgeon grass dominate.

The nutritional quality of diet and the quantity of food availability influences the physical health of turtles, affecting both growth rate and maturation time (Parmenter and Avery, 1990). No emaciation or other visible maladies resulting from nutritional deficiencies (i.e. goiters) was observed in individuals within the Bermuda population. The caloric content of the flat mangrove oyster was determined to be 5.23 cal mg⁻¹ (Thomas and Dangeubun, 1994), however this abundant bivalve does not appear to be heavily exploited by terrapins in Mangrove Lake. The energetic values for the most commonly ingested food items (e.g. *H. bermudensis* and *M. tuberculata*) should be calculated to ascertain if enough high quality prey are being consumed.

Evidence exists that suggests that Bermuda's terrapins are being exposed to a wide range of toxic compounds (e.g. heavy metals, gasoline-range and diesel-range petroleum hydrocarbons and polycyclic aromatic hydrocarbons) via food-chain contamination, specifically through the ingestion of gastropods. Furthermore, the high incidence of sediment consumption observed in the present study within juveniles and adult terrapins is of concern as this may be providing additional contaminant exposure (see Chapter 7). Exposure to polycyclic aromatic hydrocarbons and crude oil is known to cause embryological deformities and reduce embryo survival rates in the common snapping turtle (*Chelydra serpentina*) (Van Meter et al., 2006). Tissue residue analyses for Bermudian specimens

of adult cane toads (*Rhinella marina*) and red-eared sliders (*Trachemys scripta elegans*) have revealed significant levels of diesel range organics, polycyclic aromatic hydrocarbons and metals. Additionally, liver and gonad abnormalities have been documented within both species from a number of different locations throughout Bermuda which are associated with high levels of contaminants (Fort et al., 2006; Fort et al., 2006; J. Bacon, personal communication). The short-term and long-term effects that exposure to these contaminants may have upon the Bermuda terrapin population are unknown.

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Chapter 5: Nesting Ecology of Diamondback Terrapins in Bermuda

Abstract

Bermuda's small population of diamondback terrapins (*Malaclemys terrapin*) nests almost exclusively within a limited number of sand bunkers on a private golf course. Similar to elsewhere in their range, nesting was primarily diurnal, typically commenced in late March or early April and ended in late August. Peak oviposition was observed in May and June. The average clutch size was 5.1 (range 0-10 eggs; SD 2.4; n=163) and the average incubation period was 61.8 days (range 49-83 days; SD 10.5; n=26). Delayed emergence was documented, with as many as 43.8% of the hatchlings remaining in their natal nests over the winter months. The mean annual hatching success rate was determined to be 19% (range 17.6-21%; SD 1.9) from 2009-2011. The majority of nests monitored in 2010 (54.4%) and 2011 (69.6%) did not produce any hatchlings (i.e. experienced total hatching failure). No nest predation was documented. Statistical analyses confirmed that there were highly significant differences between hatching success in the different sand bunkers indicating that particular bunkers are important to nesting diamondback females and that some potential nesting sites are more valuable than others and therefore have greater conservation significance.

Introduction

Diamondback terrapins are small to medium sized emydid turtles whose native range in the U.S.A. is limited to brackish coastal waters of the Atlantic Ocean and the Gulf of Mexico between the states of Massachusetts and Texas. Seven sub-species of diamondback terrapin are currently recognized, and have been divided into northern and southern populations with Merritt Island on the east coast of Florida providing a break between the two (Butler et al., 2006; Ernst and Lovich, 2009). A breeding population is also found on Bermuda; (Davenport et al., 2005); this is considered native (Parham et al., 2008).

Nesting ecology in North America shows variability throughout the terrapin range. Diamondback terrapins in Florida form breeding aggregations in March and April which are soon followed by a period of courtship and mating (Seigel, 1980). The nesting season in the southern populations typically takes place between April and July in Florida (Seigel, 1980; Butler et al., 2004), but can occur as late as September in Louisiana (Burns and Williams, 1972). However, in the extreme northern range the season is short in duration and restricted to June and July (Burger and Montevecchi, 1975; Lazell and Auger, 1981; Goodwin, 1994; Jeyasuria et al., 1994; Feinberg and Burke, 2003). Sand is the preferred nesting medium, as it allows for sufficient gas exchange to occur between the developing embryo and the environment (Roosenburg, 1994). Terrapins throughout the U.S.A. are reported to nest on sand dunes, beaches and along the sandy margins of marshes and islands (Burger and Montevecchi, 1975; Burger, 1977; Seigel, 1980; Roosenburg, 1994). Diurnal nesting appears to be the standard for most terrapin populations (Burger and Montevecchi, 1975; Seigel, 1980; Goodwin, 1994), although nocturnal nesting has been documented in some populations (Auger and Giovannone, 1979; Roosenburg, 1992). Clutch size ranges from 4-22 eggs (Butler et al., 2006), with females in the northern part of the range having greater mean clutch sizes and comparatively smaller eggs than females in the southern part (Allman et al., 2012).

Diamondback terrapins, like many other reptiles, exhibit temperature-dependent sex determination (TSD) whereby the temperature of the sand affects the sex of the developing embryos. Cooler incubation temperatures are known to produce male hatchlings and warmer incubation temperatures produce female hatchlings (Jeyasuria et al., 1994; Roosenburg and Kelley, 1996; Wood and Herlands, 1997).

The incubation and emergence period also varies between regions. Incubation can last from 50-120 days (Burger, 1977; Jeyasuria et al., 1994; Butler et al., 2004), and hatching occurs from July - October (Burger, 1977; Roosenburg, 1991; Butler et al., 2004). Hatchlings may depart the nest within hours of hatching (Roosenburg and Kelley, 1996), or they may spend months over-wintering in the nest chamber and emerge in the following

spring (Lazell and Auger, 1981; Roosenburg and Kelley, 1996; Baker et al., 2006). Hatching success of wild nests has been reported to range from 1-84% (Burger, 1977; Roosenburg, 1992; Goodwin, 1994). Nest depredation, especially from small mammals, has been identified as a major determinant of hatching success in North America (see reviews in Butler et al., 2006; Ernst and Lovich, 2009).

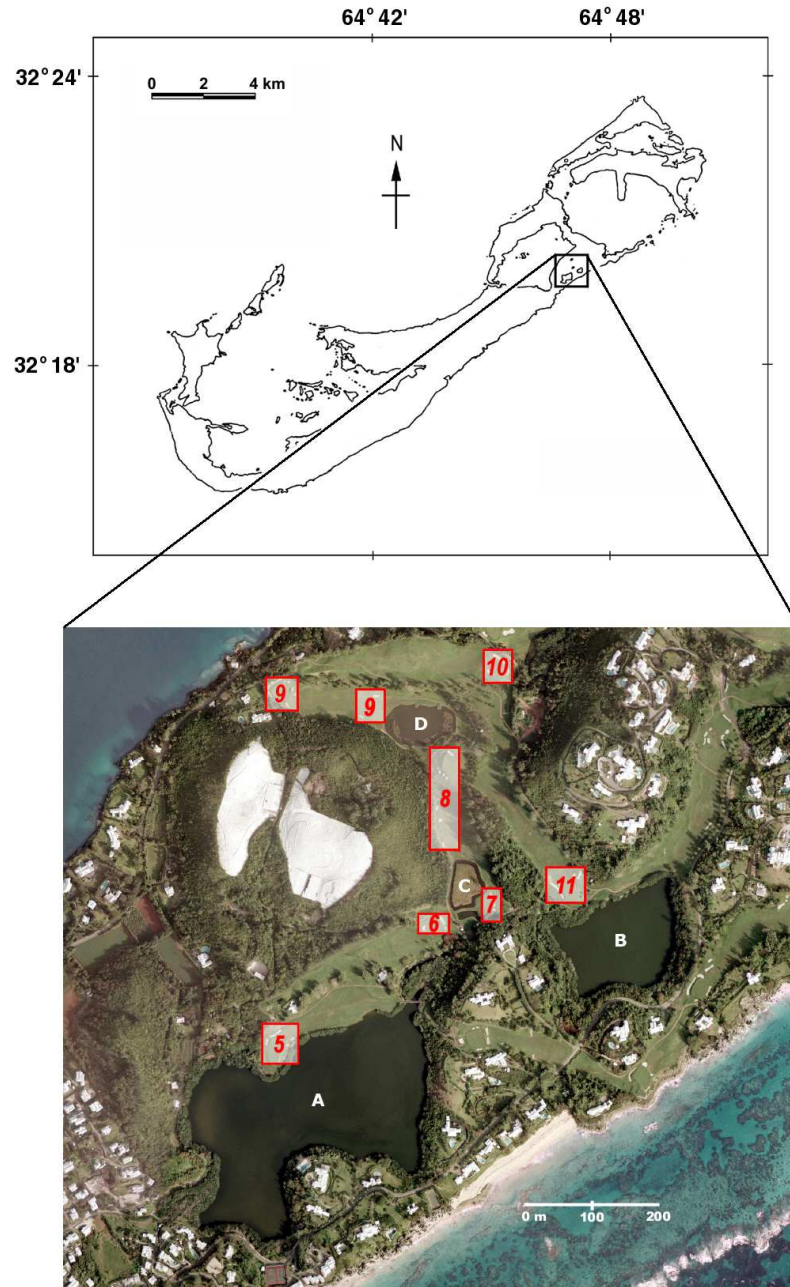
Bermuda's diamondback terrapins appear to be the only wild breeding population outside the North American range. Anecdotal evidence suggests that this species has been nesting for many years in a number of sand bunkers on a private golf course (Davenport et al., 2005), however quantitative assessments of nesting activity in Bermuda have been lacking for this species. Knowledge of the reproductive output is needed in order to perform population modelling and was deemed essential given the low number of females in the Bermuda population (see Chapter 3). The present investigation had the following objectives: (1) to determine the frequency of sand bunker nesting in Bermuda, (2) to determine the duration of the nesting season, (3) to describe clutch size, egg morphology and hatchling biometrics, (4) to establish the incubation and emergence periods and (5) to report hatching success rates for the Bermuda terrapin population.

Methods

Study site

The entire known Bermuda population of diamondback terrapins is found only in four brackish water ponds named Mangrove Lake, South Pond, North Pond, and Trott's Pond (Fig. 5.1). All are located on a private golf course and all are separated by, at most, 380 m of land (straight-line distance between North Pond and Trott's Pond. Refer to Chapter 2 for a more comprehensive description of the ponds and associated wetlands. A number of sand bunkers are also located at the site. They vary in size and all are un-shaded as they lack fringing vegetation. The sand bunkers found closest to the ponds inhabited by diamondback terrapins are located between the fifth and eleventh holes. The fifth hole has four sand bunkers situated around the putting green, two of which are in very close proximity to

Mangrove Lake. The sixth and seventh holes have three sand bunkers on each, between which lies South Pond. The eighth hole is located between South Pond and North Pond and has nine sand bunkers spread across the fairway and surrounding the putting green. The ninth hole has a total of seven sand bunkers upon it, of which two are in close proximity to North Pond and the tenth hole has three bunkers adjacent to the putting green. Finally, the eleventh hole has four sand bunkers surrounding the putting green, all of which are adjacent to Trott's Pond. Each of the sand bunkers mentioned above were assigned alphanumeric notations (Figs. A5.1 - A5.8, Appendix 5).



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Figure 5.1. (Top) aerial photograph of Bermuda showing the location of the four diamondback terrapin ponds. (Bottom) aerial photograph from 2003 showing the four diamondback terrapin ponds situated on the Mid Ocean golf course (A=Mangrove Lake, B=Trott's Pond, C=South Pond, D=North Pond) and the generalized locations of the sand bunkers associated with the fifth through eleventh holes (red boxes numbered 5-11).

Surveys

A pilot study was performed in 2009, during which the sand bunkers on the fifth, sixth, and seventh holes were haphazardly visited during daylight hours in May and June and examined for evidence of terrapin use that indicated recent oviposition (e.g. tracks in the sand and/or areas of disturbed sand patterns). The survey effort was increased in 2010 and 2011 so that daily nesting surveys between the hours of 06:00 and 24:00 hours were performed in all sand bunkers between the fifth and eleventh holes beginning in February and continuing until October during both years. The nesting survey effort was reduced to a once weekly schedule between the months of November and January. Nesting events were also reported by residents in the study area, golfers, and staff members of the Mid Ocean Club course maintenance department. The number of terrapin tracks (or crawls), number of excavated but un-used nests abandoned by female terrapins prior to oviposition, and the number of covered (e.g. used) nests (see Figs. A5.10 - A5.12, Appendix 5) were recorded during the 2010 and 2011 surveys. All signs of terrapin usage in the bunkers were raked away to prevent inclusion in subsequent surveys. Nesting was confirmed via discovery of terrapin eggs. For nests discovered only in 2010, the depth of sand covering the top egg in each clutch was recorded, after which the eggs were gently removed (taking care not to rotate the eggs during excavation), measured, weighed and counted. Maximum length and width was measured in millimetres using vernier calipers. Values were rounded to the nearest 1 mm. Mass was recorded in grams and obtained by placing each egg inside a plastic bag which was then hung from a 10 g spring scale with 1 g increments. The total mass of each egg was calculated by subtracting the mass of the empty bag from the total mass of the bag and egg. The maximum depth and maximum width of each nest chamber was also recorded prior to returning the eggs to their respective chambers. Nests discovered in 2009 and 2011 were not excavated upon discovery; however, 25 eggs were removed from 15 terrapin nests randomly chosen in June and July 2011 and preserved for various laboratory analyses. Nest locations within each bunker were recorded and marked with galvanised metal stakes and blue surveyors tape (Fig. A5.13, Appendix 5). The temperature in ten

haphazardly chosen nests (five in 2010 and five in 2011) was recorded every hour throughout the incubation period (IP) and the thermo-sensitive period (TSP) using digital HOBO Pendant data loggers (model # UA-002-08 from Onset Computer Corporation). These loggers measured temperature to the nearest 0.001 but the reported values were rounded to the nearest 0.1. The incubation period was defined as the elapsed time between the date of oviposition and first observed hatching. The thermo-sensitive period was defined as the period of embryonic development during which gonadal differentiation is most affected by temperature. The TSP for diamondback terrapins has been identified as occurring in the middle third of the incubation period (Roosenburg, 1994). Data loggers were buried in the sand immediately adjacent to each clutch of eggs and were retrieved at the time of nest excavation. The temperatures recorded by these devices were assumed to be similar to those that occurred within the nests. The effects of metabolic heating were assumed to be minimal, given the relatively small clutch size for this species in Bermuda.

Hatching and emergence were studied in 2010 to quantify the incubation and post-hatching nest residency periods. This was determined by carefully scraping away the sand 50 days after oviposition, exposing the top egg and checking for signs of hatching (e.g. tears or breaches in the egg shell). This was repeated daily until hatching was observed, at which point a 0.5 m diameter circular cage was placed on top of the nest site (Fig A5.14 in Appendix 5). The cages were checked daily for the presence of hatchlings. The emergence period was defined as the number of days between the first observed hatching event and hatchling emergence.

All marked nests were excavated in March 2011. The numbers of live and dead terrapin hatchlings encountered in the chamber of each nest were counted, as were the number of egg shells that comprise $\geq 50\%$ of the egg size and the number of whole (un-hatched) eggs. The latter were dissected and examined for terrapin remains (e.g. bones and scute material) in order to determine if embryos were present. Hatching success was defined as the percentage of eggs that hatched in each clutch. Annual hatching success (e.g. sum total for the year monitored) as well as individual nest hatching success was calculated.

In an effort to minimise disturbance to developing eggs, the incubation surveys were not performed on the nests created in 2011. Hatchling emergence was examined by using the 0.5 m cages, which were placed upon nest sites 50 days after oviposition had occurred and were monitored daily between June and November 2011. On November 30th all of the nests were excavated and the contents recorded as described earlier. Live hatchlings encountered in the nest chamber were assumed to be overwintering.

All hatchling terrapins encountered during the 2010 and 2011 nesting surveys were measured, weighed, and examined for general health and physical anomalies (e.g. missing or extra scutes). Three straight-line body measurements were taken using vernier calipers following Bolten (1999); minimum straight carapace length (SCL), straight carapace width (SCW), and straight plastron length (SPL). SCL was measured from the anterior edge of the nuchal scute to the posterior edge of the shell between the supracaudal scutes along the mid-line. SCW was measured at the widest point across the carapace and SPL was measured along the midline from the anterior edge of the gular scutes and the posterior edge of the anal scutes. Care was taken not to distort the shape of the shell, since the shells of hatchlings were flexible. All measurements were recorded to the nearest 0.1 mm. Discrepancies caused by differences in measurement technique between observers were eliminated by having each terrapin measured by the same researcher (M.O.). Body mass was measured to the nearest gram using a 10 g spring scale. Each hatchling was securely placed within a small plastic bag in order to reduce trauma and permit accurate weighing. Total mass was calculated by subtracting the mass of the empty bag from the total mass of the bag and hatchling.

Nesting area was determined by measuring the maximum straight-line length and width for each sand bunker situated between the fifth and eleventh holes. Nest density was reported as the number of nests m⁻².

Results

Nesting season and frequency

In 2010, the nesting season was observed to occur between March 22nd and August 11th (n=142 days). The nesting season in 2011 was observed to occur between April 16th and August 26th (n=133 days). Thus, the duration of the nesting season for diamondback terrapins in Bermuda is four to five months. A total of 57 terrapin nests were discovered in 2010 and 72 nests were discovered in 2011. May was identified as the peak nesting month in 2010 during which a total of 21 nests were discovered, whereas nesting activity peaked in June in 2011 during which 25 nests were discovered (Fig. 5.2). No evidence of nesting activity was observed between September and February during either year.

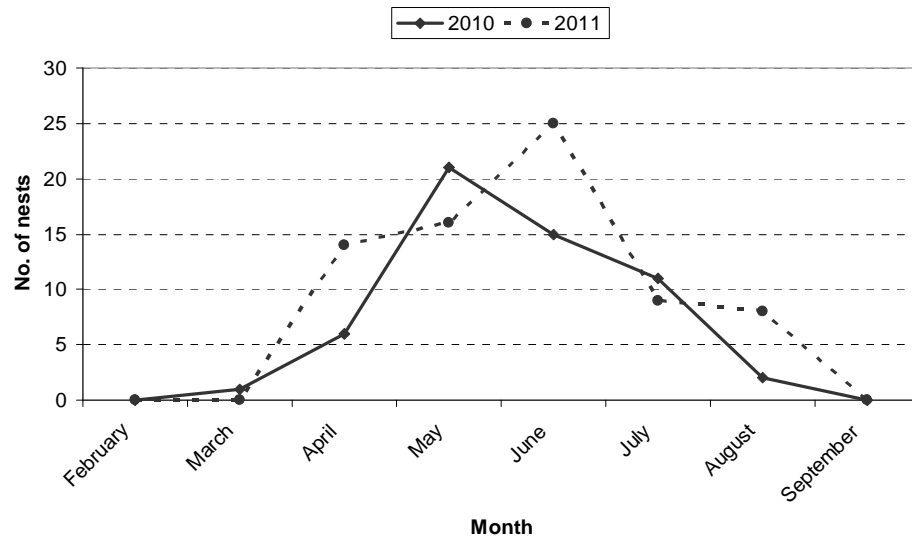


Figure 5.2. Observed nesting frequency for Bermuda's diamondback terrapins in 2010 (solid line) and 2011 (dashed line).

Nesting locations, densities and observations

The pilot nesting surveys during the months of May and June in 2009 resulted in a total of ten terrapin nests being discovered in the bunkers on the fifth, sixth, and seventh holes of the Mid Ocean golf course.

Of the 57 terrapin nests discovered in 2010, 56 (98.2%) were in sand bunkers on the fifth, sixth, seventh, eighth, and eleventh holes of the Mid Ocean golf course and one nest was discovered on a soil embankment in the NE corner of Mangrove Lake. The majority of these nests ($n=27$ or 47.4% of the total) were discovered in the sand bunkers on the seventh hole (Fig. A5.16 in Appendix 5). Seventeen nests (29.8%) were discovered in the sand bunkers on the fifth hole (Fig. A5.17 in Appendix 5) and ten nests (17.5%) were discovered in the sand bunkers on the sixth hole (Fig. A5.16 in Appendix 5). The eighth and eleventh holes had only one nest each (Figs. A5.18 and A5.19 in Appendix 5). The majority of the nests (53.6%) were found at various locations within the sand bunkers; however 46.4% were located around the margins. Furthermore, an additional seven nests were encountered that had been fully excavated by female terrapins but then abandoned before oviposition had occurred.

Of the 72 terrapin nests discovered in 2011, 70 (97.2%) were in the sand bunkers on the fifth, sixth, and seventh holes and two were discovered in the vicinity of the seventh hole; one upon a soil embankment and one beside a paved cart path. The majority of the nests ($n=34$ nests or 47.2% of the total) were discovered on the fifth hole bunker (Fig. A5.20 in Appendix 5). Twenty seven nests (37.5%) were discovered in the sand bunkers on the seventh hole and nine nests (12.5%) were discovered in the sand bunkers on the sixth hole (Fig. A5.21 in Appendix 5). The majority of the nests (63.9%) were found at various locations within the sand bunkers; but 36.1% were located around the margins. An additional five nests were encountered that had been fully excavated by female terrapins and then subsequently abandoned, including three which contained no eggs and two which contained one to three eggs.

A number of residents, golf course maintenance staff members and golfers reported observing female diamondback terrapins nesting or attempting to nest in 2010 and 2011. These sightings occurred primarily between the fifth and eleventh holes of the Mid Ocean golf course but also on private lands surrounding Mangrove Lake (Fig. A5.22 in Appendix 5).

Thirty three sand bunkers are situated between the fifth and eleventh holes on the Mid Ocean golf course and each varied in size and proximity to

the nearest pond. Table A5.1 in Appendix 5 summarises the dimensions, the number of nests discovered and the nesting density in each sand bunker. Individual bunker areas ranged from 12.3-181.7 m² (mean 56.8 m²) and the minimum straight-line distance to the nearest pond ranged from 5-207 m (mean 71.7 m). The total sand bunker area between the fifth and eleventh holes available to nesting female diamondback terrapins was calculated to be 1873.6 m² (0.19 ha). Mean annual nest density was estimated to be 0.035 nests m⁻² (347 nests ha⁻²) in 2010 and 0.044 nests m⁻² (443 nests ha⁻²) in 2011. The highest nesting density over the two year study period was recorded in bunker 7A (2011) which contained 0.278 nests m⁻² (approximately 2784 nests ha⁻²).

Seventeen nesting events were observed over the two year period, and all but one occurred during day-time hours. The single observed nocturnal nesting episode occurred at 21:10 (August 1st 2011). Mid Ocean golf course maintenance staff reported eight nesting events; golfers reported four and the author witnessed five - of which two were observed in their entirety. The first occurred on June 22nd 2010 at 09:45, during which the female terrapin was observed spending 40 minutes nesting: eleven minutes were spent excavating with her hind legs, ten minutes were spent lying motionless (presumably depositing eggs), and 19 minutes were spent re-covering the nest with her hind legs. Excavation of the nest revealed the presence of four eggs. The second nesting event occurred on May 28th 2011 at 16:00 on the north facing slope of the seventh hole at South Pond. The female was observed emerging from the pond and upon reaching the nesting location spent 37 minutes nesting, which included 28 minutes excavating, five minutes depositing eggs, and four minutes re-covering the nest. Excavation of the nest revealed the presence of eight eggs.

Nest chamber dimensions, clutch size and egg morphometrics

Nest chamber dimensions were recorded for 44 nests excavated in 2010. Maximum depth ranged from 11-16 cm (mean 13.7 cm; SD 1.4) and width ranged from 5-9 cm (mean 6.6 cm; SD 0.9). The depth of sand to the top eggs ranged from 7-13 cm (mean 9.6 cm; SD 1.6).

A total of 50 eggs were found within the ten nests discovered in 2009. Mean clutch size for these nests was 5.0 (range 1-9 eggs nest⁻¹; SD 2.5). In 2010, 268 eggs were recorded from the 57 nests. Mean clutch size was calculated to be 4.7 (range 0-10 eggs nest⁻¹; SD 2.6). In 2011, 397 eggs were recorded from 72 nests, giving a mean clutch size of 5.5 eggs (range 0-10 eggs nest⁻¹; SD 2.3). It is noteworthy to mention that two nests encountered in 2010 and one nest encountered in 2011 did not contain any eggs, despite the fact that the females responsible for their construction had gone through the entire process of excavating the sand and then re-covering the chambers.

A total of 174 eggs were measured and weighed from 37 nests excavated in 2010. All egg morphometric data were tested for normality. Data for egg length and egg mass were non-normal (Anderson-Darling tests; length $p=0.001$ and mass $p<0.0005$). Data for egg width were normally-distributed (Anderson-Darling test; width $p=0.095$). Maximum length ranged from 30.4-46.5 mm (median 35.7 mm; SD 2.0 mm); maximum width ranged from 18.7-25.5 mm (mean 21.8 mm; SD 1.3 mm); mass ranged from 7-16 g (median 11.0 g; SD 1.5 g). Because some datasets were non-normal, non-parametric statistical tests were required to make comparisons of egg length between different nests. A Kruskal-Wallis test showed highly significant differences amongst the median egg length found for individual nests ($H=141.06$; $DF=42$; $p<0.0005$).

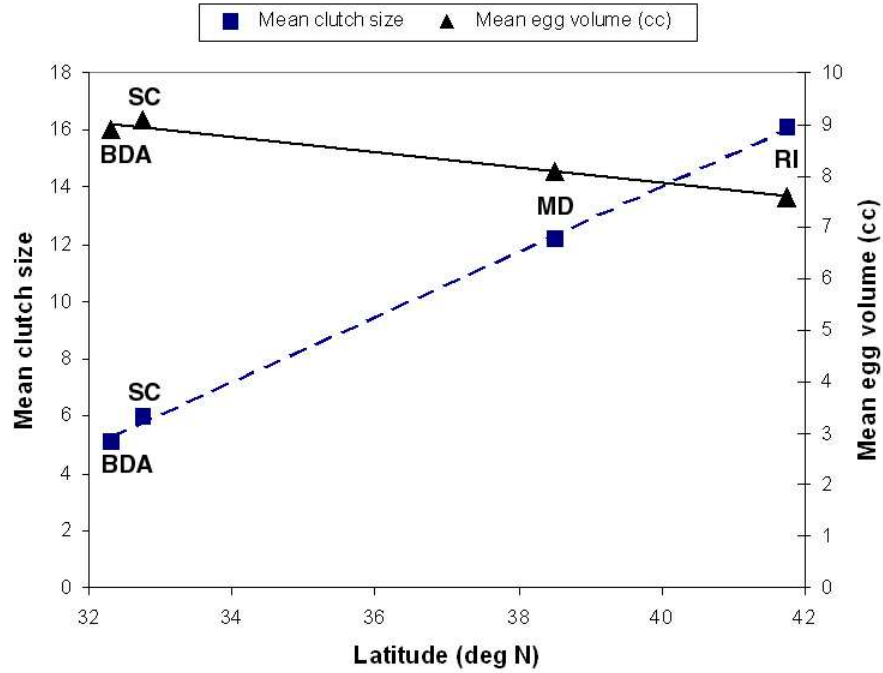


Figure 5.3. Geographic comparison of clutch size and egg volume for diamondback terrapins from Bermuda (this study) and the U.S.A (Allman et al., 2012). BDA=Bermuda, SC=South Carolina, MD=Maryland, RI=Rhode Island. Note: egg volume was calculated using the ellipsoid formula [volume = $(\pi/6)(\text{length})(\text{width}^2)$] following Allman et al., 2012).

Incubation and emergence periods

The incubation period for Bermuda's diamondback terrapins ranged from 49-83 days (mean 61.8 days; SD 10.5; n=26). Hatching was first observed on July 9th 2010 and last observed on September 14th 2010. The first terrapin hatchling to emerge from its nest was observed on July 20th 2010 and the last to emerge was observed on March 19th 2011, giving a range of 1-219 days (mean 83.6 days; SD 79.1; n=33). The frequency of emergence varied during the study period. Two distinct emergence patterns were documented; July-October 2010 and January-March 2011. The emergence period of the former ranged from 1-73 days (mean 31.4 days; SD 22.8; n=22) and the latter ranged from 140-219 days (mean 188.1 days; SD 30.6; n=11). No hatchling emergence was observed in November-December 2010 or April-June 2011.

Hatchling emergence was also observed to occur between July and October 2011, and live terrapin hatchlings were discovered in nest chambers during the excavations on November 30th 2011.

Hatching success

Table 5.1 summarises the clutch data for all nests monitored from 2009-2011. The overall mean hatching success rate for the 136 nests observed during this three year period was 19%. The 2009 pilot study revealed a hatching success rate of only 18%. Of the eggs that did not hatch, 35 (70%) appeared to show no evidence of embryonic development and six (12%) contained dead embryonic material (Table A5.2 in Appendix 5).

Table A5.3 in Appendix 5 summarises the clutch data for the 57 nests monitored in 2010. Excavation of these nests revealed that 165 eggs (61.6%) appeared to show no visible evidence of embryonic development, 33 (12.3%) contained dead embryonic material, and nine (3.4%) contained fully formed dead hatchlings - many of which had managed to break through the shell, but all failed to successfully emerge from their nest chambers. Individual hatching success for each clutch varied from 0–100% but the mean rate was 21%. Twenty six nests (45.6%) produced at least one hatchling, however 31 (54.4%) did not produce any hatchlings (i.e. experienced total failure). A total of 61 terrapin hatchlings successfully emerged from the monitored nests during this period. The number that was observed to emerge between July and October 2010 was 32 (52.5% of the total), while 14 hatchlings (23%) were observed emerging between January and March 2011. Fifteen hatchlings (24.6%) departed their nests during unobserved periods.

Table A5.4 in Appendix 5 summarises the clutch data for the 72 nests monitored in 2011. The locations of three nests containing 16 eggs were lost when the metal stakes marking their positions within the sand bunkers were removed by unknown people, and an additional 25 eggs were deliberately removed from 15 nests for toxicological analyses (see Chapter 7) and viability analyses (separate investigation). Thus, the total number of eggs monitored in 2011 was 356 from 69 nests. Upon excavation, 277 eggs (77.8%) appeared to show no visible evidence of embryonic development,

six (1.7%) contained dead embryonic material, and nine (2.5%) contained fully formed dead hatchlings. Individual clutch hatching successes varied from 0–100% but the mean rate for the 2011 nesting season was 17.6%; 21 nests (30.4%) produced at least one hatchling, however 48 (69.6%) experienced total failure. A total of 64 hatchlings emerged from the monitored nests; 36 between July and October, and 28 were excavated in November.

It is noteworthy to mention that no nest depredation was observed between 2009 and 2011.

Table 5.1. Summary of clutch data for nests monitored from 2009-2011.

Year	No. eggs monitored	No. of clutches	No. emerged hatchlings	No. dead hatchlings in chamber	No. un-hatched embryos	No. eggs with no embryo	Mean hatching success
2009	50	10	9	0	6	35	18.0 %
2010	268	57	61	9	33	165	21.0 %
2011	356	69	64	9	6	277	17.6 %

Relationship between sand bunkers, nesting frequency, incubation and hatching success.

To identify whether particular nesting bunkers were more important than others in terms of nesting and success of nests, further statistical investigations were carried out. From Table 5.2 it is evident that some of the eleven bunkers featured more nests than others. By inspection, it appears that bunkers 5D, 6B, 7A, 7B and 7C were especially important in terms of nest numbers. A replicated G test for goodness of fit (McDonald, 2009) was employed against the extrinsic hypothesis that numbers of nests would be distributed evenly (1:1:1:1 etc.) amongst bunkers in each of the two years and in the pooled data. For 2010, $G = 53.2$ (d.f.=10, $p < 0.0001$), for 2011 $G = 83.5$ (d.f.=10, $p < 0.0001$), for the pooled data $G = 125.8$ (d.f.=10, $p < 0.0001$). A heterogeneity G values for the pooled data indicated that there were no significant differences in ratio of nesting frequencies between years ($G = 10.96$, d.f.=10, $p = 0.360$). These analyses confirmed that there were highly significant differences between hatching success in different bunkers, but that, the ratios recorded in 2010 were similar to those found in 2011. This strongly indicates that particular bunkers are important to nesting diamondback females.

From Table 5.3 it can be seen that the number of emerged hatchlings also differed amongst bunkers, with bunkers 5A, 5D, 6B, 7A and 7B being especially important. Replicated G tests were again performed, yielding the following results: 2010: $G = 97.60$ (d.f.=10, $p < 0.0001$); 2011: $G = 85.14$ (d.f.=10, $p < 0.0001$); Pooled $G = 160.61$ (df=10, $p < 0.0001$); Heterogeneity $G = 22.14$ (d.f. =10, $p = 0.014$). Therefore, years 2010 and 2011 both do not follow a 1:1:1:1 etc. ratio for number of emerged hatchlings from the eleven bunkers (some bunkers showing more emerged hatchlings than others), but the two years individually have significantly different ratios from each other. By inspection there appeared to be a close match between the bunkers that yielded the majority of emerged hatchlings and those that featured the most nests.

The numbers of un-emerged hatchlings showed a similar pattern: 2010: $G = 43.73$ (d.f.=10, $p < 0.0001$); 2011: $G = 21.14$ (d.f.=10, $p < 0.0001$); Pooled $G = 51.1$ (df=10, $p < 0.0001$); Heterogeneity $G = 13.76$ (d.f. =10,

$p=0.183$). However, the ratios showed no significant differences between 2010 and 2011. Bunkers that produced high numbers of emerged hatchlings also resulted in high numbers of un-emerged hatchlings.

Table 5.4 shows the number of eggs that showed no sign of embryonic development recorded from each bunker. For 2010: $G=171.1$ (d.f.=10, $p<0.0001$); 2011: $G=402.6$ (d.f.=10, $p<0.0001$); Pooled $G= 532.8$ (df=10, $p<0.0001$); Heterogeneity $G= 40.85$ (d.f. =10, $p=0.183$). Hence, neither of years 2010 and 2011 followed a 1:1:1 etc. ratio for numbers of eggs lacking an embryo from the eleven bunkers (i.e. some bunkers showed more of such eggs than others) and the two years individually did not differ significantly in their ratios.

Finally, Spearman's Rank analysis (using PAST software) was conducted to further investigate the interrelationships between bunker, number of nests, number of emerged hatchlings, number of non-emerged hatchlings and numbers of eggs with no sign of embryonic development. It can be seen that, in all comparisons, the similarity in ranking between pairs of data is high and statistically significant (Table 5.5).

Table 5.2. Nesting frequencies in the different golf course bunkers (2010-2011).

Bunker ID	2010	2011	Pooled
5A	2	3	5
5C	1	0	1
5D	13	21	34
6A	1	0	1
6B	9	8	17
6C	0	1	1
7A	13	12	25
7B	8	4	12
7C	5	3	8
8H	1	0	1
11A	1	0	1
Total	54	52	106

Table 5.3. Frequencies of emerged hatchlings in the different golf course bunkers (2010-2011).

Bunker ID	2010	2011	Pooled
5A	3	7	10
5C	0	0	0
5D	8	12	20
6A	0	0	0
6B	12	16	28
6C	0	1	1
7A	25	10	35
7B	9	1	10
7C	1	0	1
8H	0	0	0
11A	3	0	3
Total	61	47	108

Table 5.4. Frequencies of eggs without embryos in nests laid on different golf course bunkers (2010-11).

Bunker ID	2010	2011	Pooled
5A	3	3	6
5C	1	0	1
5D	34	89	123
6A	9	0	9
6B	34	29	63
6C	0	0	0
7A	30	58	88
7B	22	15	37
7C	17	20	37
8H	0	0	0
11A	1	0	1
Total	151	214	365

Table 5.5. Results of Spearman's Rank analysis of bunker data (2010 and 2011 pooled). d.f. = 10 in each case.

Comparisons	Spearman's Rank	p
No. nests vs. No. emerged hatchlings	0.860	0.003
No. nests vs. No. non-emerged hatchlings	0.860	0.001
No. nests vs. No. egg with no embryonic development	0.953	<0.001
No. emerged hatchlings vs. No. non-emerged hatchlings	0.654	0.044

Nest temperatures

Table 5.6 summarises the range and mean temperatures of the ten terrapin nests monitored throughout their respective incubation and thermo-sensitive periods in 2010 and 2011. Figs. A5.23 - A5.32 in Appendix 5 show the temperatures at a depth of 10 cm for each of the monitored nests. Temperature showed great variability. Nests created earlier in the nesting season (e.g. in April and May) had lower mean daily temperatures than those created later in the nesting season (e.g. July and August). Daily temperatures during the incubation period ranged from 16.1-40.5°C, but averaged 25.4-31.6°C, while the daily temperatures during the TSP ranged from 17.2-40.5°C and averaged 24.6-30.7°C. All experienced lethal temperatures (e.g. $\geq 35^{\circ}\text{C}$) during the incubation period; however eight of the monitored nests experienced these temperatures for at least five consecutive days and sometimes for as long as eleven consecutive days. The individual hatching success rates for the ten monitored nests ranged from 0-100%.

Table 5.6. Temperatures (with ranges and means) and hatching success of ten diamondback terrapin nests monitored in 2010 and 2011.
IP=Incubation Period; TSP=Thermo-sensitive Period.

Nest ID	Incubation date	IP temperature (°C) range (mean)	TSP temperature (°C) range (mean)	Hatching success
7A1	April 18 – July 9, 2010	16.2 – 39.8 (27.4)	17.9 – 36.8 (27.0)	75%
5D1	April 19 – June 20, 2010	16.1 – 35.2 (25.4)	17.2 – 32.3 (24.6)	0%
11A1	June 22 – Aug 19, 2010	23.9 – 38.2 (29.9)	25.8 – 37.9 (30.7)	75%
7C4	July 9 – Aug 30, 2010	26.0 – 40.5 (31.6)	26.0 – 40.5 (30.5)	0%
7B8	July 13 – Sept 5, 2010	25.8 – 37.3 (30.5)	25.8 – 37.3 (30.0)	25%
5D5	April 29 – June 30, 2011	17.4 – 38.8 (26.7)	19.6 – 34.8 (26.7)	57%
6B4	May 5 – July 6, 2011	17.4 – 36.7 (26.4)	19.5 – 33.3 (26.1)	100%
7B4	May 16 – July 17, 2011	22.4 – 37.7 (28.0)	22.5 – 34.9 (27.1)	0%
5D9	May 31 – Aug 1, 2011	20.4 – 38.9 (28.8)	22.6 – 38.9 (30.7)	0%
5D14	June 18 – Aug 19, 2011	23.4 – 38.0 (29.6)	24.2 – 34.7 (28.8)	0%

Hatchling biometrics

Table 5.7 summarises the biometric data for 106 diamondback terrapin hatchlings measured during the 2010 and 2011 nesting surveys. All data (lengths and masses) were normally distributed (Anderson Darling test; $p > 0.05$) Straight carapace length (SCL) from 25.0 – 37.6 mm (mean 31.9 mm; SD 2.4 mm), straight carapace width (SCW) ranged from 22.9 – 32.4 mm (mean 27.8 mm; SD 2.0 mm), straight plastron length (SPL) ranged from 20.7 – 30.9 mm (mean 26.3 mm; SD 1.9 mm), and mass ranged from 4 – 10 g (mean 7.5 g; SD 1.4 g). One-way ANOVA showed that there was no significant difference between SCL values measured in 2010 and 2011 ($p = 0.849$). Reduced major axis regression of hatchling mass upon SCL revealed the following statistically significant relationship:

$$\text{Log mass} = 2.72 \log \text{SCL} - 3.21 \text{ (n=106, } r^2=0.70, p<0.0001)$$

The 95% confidence limits for the slope of the regression (2.44, 3.01) include three, so the relationship between body mass and SCL is isometric. Relationships between SCL, SCW and SPL were all isometric too. SCL did not differ between hatchlings collected in 2010 and 2011 (One-way ANOVA; $p = 0.849$). In addition, 54 hatchlings (50.9%) showed scute anomalies. The most common anomalies were extra vertebral scutes (33% frequency of occurrence), extra costal scutes (31.1% frequency of occurrence), and extra marginal scutes (25.5% frequency of occurrence). Only two individuals were encountered that had missing marginal scutes.

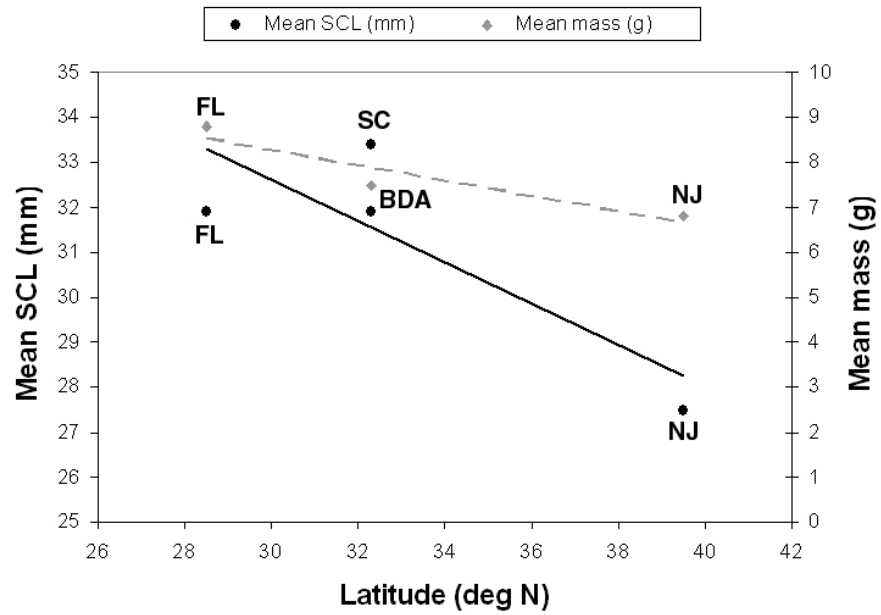


Figure 5.4. Geographic comparison of straight carapace length (SCL) and mass for diamondback terrapin hatchlings from Bermuda (this study) and the U.S.A (Burger, 1977; Seigel, 1980; Lovich et al., 1991). FL=Florida, SC=South Carolina, BDA=Bermuda, NJ=New Jersey.

Table 5.7. Biometric data summary for 106 diamondback terrapin hatchlings encountered during the 2010 and 2011 nesting surveys. Abbreviations are as follows: SCL = straight carapace length; SCW = straight carapace width; SPL = straight plastron length.

2010 (n=48)					2011 (n=58)			
	SCL (mm)	SCW (mm)	SPL (mm)	Mass (g)	SCL (mm)	SCW (mm)	SPL (mm)	Mass (g)
Range:	26.8- 37.6	22.9- 32.4	20.7- 30.9	5-10	25.0- 37.0	23.0- 32.0	22.0- 30.6	4-10
Mean:	31.9	27.8	26.1	7.7	31.8	27.8	26.4	7.4
SD:	2.5	2.0	2.1	1.4	2.3	2.0	1.8	1.4

Discussion

Nesting season, frequency, location and density

Diamondback terrapins in Bermuda have a nesting season (133-142 days) that is significantly longer in duration than those reported from different populations across the North American range, which vary from 34-44 days in New Jersey (Burger, 1977) to 56-71 days in Maryland (Roosenburg, 1991) and *ca.* 78 days in NE Florida (Butler, 2000). The climate in Bermuda is unusually warm for its latitude (32°N), since heated water transported north in the Gulf Stream has created a northerly extension of sub-tropical systems (Thomas and Logan, 1992). Presumably it is the sub-tropical climate in Bermuda that allows for the observed protracted terrapin nesting season.

The current investigation revealed a low monthly mean nesting frequency during the five month nesting period. This, however, is to be expected given that only 32 female diamondback terrapins are believed to be sexually mature within the Bermuda population (i.e. individuals with plastron lengths ≥ 138 mm total length) (see Chapter 3). It is likely that nests were missed during the daily surveys, thus the number of nests reported

represent conservative figures. It appears that individual female diamondback terrapins in Bermuda can lay at least 1.8-2.3 clutches annually; an observation consistent with others made from various regions throughout the U.S.A. (Auger and Giovannone, 1979; Feinberg and Burke, 2003). Three clutches of oviposited eggs have been reported from individual female terrapins on the Atlantic coast of Florida (Seigel, 1980) and in Chesapeake Bay (Roosenburg and Dunham, 1997).

The single observation of a female terrapin nesting during the night represents the first confirmed instance of nocturnal nesting in Bermuda; however, the local population appears to be comprised primarily of diurnal nesters.

Diamondback terrapins in Bermuda appear to primarily use the sand bunkers on the Mid Ocean golf course for nesting, however a limited amount occurs in other locations as well. The first confirmed report of diamondback terrapins nesting in sand bunkers occurred in the mid 1990s (Davenport et al., 2005), however terrapin nesting prior to this period is unknown. Diamondback terrapins were not recorded on 19th and 20th century herpetological fauna lists (Jones, 1859; Garman, 1884; Heilprin, 1889; Hurdis, 1897; Verrill et al., 1903), despite being present for at least 400 years (Parham et al., 2008). Detailed maps published between the late 18th and early 20th centuries show that the area currently inhabited by terrapins comprised at least eight brackish water ponds (including Mangrove Lake, Trott's Pond, South Pond and North Pond) interspersed with mangrove swamps and peat marshes. Furthermore, these maps show a one kilometre long beach on the coastline approximately 100 m to the south of Mangrove Lake and Trott's Pond. Assuming that diamondback terrapins have inhabited these wetlands since their arrival to Bermuda, it is probable that this beach served as a natural nesting ground. However, extensive habitat modification - particularly during the 20th century (see Sterrer and Wingate, 1981 for review) - has greatly affected Bermuda's natural landscape, including the wetlands inhabited by diamondback terrapins. Diamondback terrapins have been described as opportunists that find appropriate nesting areas throughout their North American range (Roosenburg, 1994), thus it is likely that the Bermudian terrapins started

using the sand bunkers (created in the 1920s) in addition to nesting on the coastal beach. As development in this locality (circa 1940s) increased over time, various obstacles (e.g. stone walls and fences, roads and buildings) would have prevented female terrapins from easily accessing the beach, and may explain why they now heavily use the sand bunkers as a surrogate nesting environment.

Estimated nesting densities in the U.S.A. range from 0.52 nests ha⁻¹ - 1125 nests ha⁻¹ (see review in Roosenburg, 1994). The mean annual nest density calculated in 2010 and 2011 for Bermuda's population of diamondback terrapins fits within the range reported from North America; however nesting was found to be highly localized. Only eleven of the 33 sand bunkers (33%) situated between the fifth and eleventh holes on the Mid Ocean golf course were used by female terrapins for nesting during the two year survey period. Nest density in the most heavily used bunkers (i.e. those closest to Mangrove Lake and South Pond) generally exceeded the North American upper range limit, and in one instance more than doubled it.

One of the consequences of nesting in the sand bunkers is that it greatly increases the chances of disturbance caused by anthropogenic activities. Golfers frequently enter the sand bunkers during the course of play and staff from the golf course maintenance department routinely use tools to manually trim the verges of the bunkers in order to prevent the incursion of grass. Some of these tools are capable of penetrating 15 cm into the sand. Furthermore, sand bunkers are periodically excavated using heavy machinery. These activities can negatively impact nesting females as well as the resident eggs and hatchlings, especially since the nest depth was fairly shallow (mean 13.7 cm). A relatively high proportion of the nests monitored between 2010 and 2011 (36-46%) were located along the margins of the sand bunkers and a high number of terrapin hatchlings (up to *ca.* 44%) remained in their natal nests for long periods of time.

Nest chamber dimensions, clutch size and egg morphometrics

The results of the present investigation show that diamondback terrapin nests in Bermuda are slightly shallower and have less sand covering the top-

most eggs than do nests and clutches reported from the U.S.A. (Burger, 1977; Roosenburg, 1992; Butler, 2000).

The mean clutch size from the Bermuda population (n=5) occurs at the lower end of the range reported from the U.S.A., where clutch size varies from 4-22 eggs, and the egg morphometrics of Bermuda's diamondback terrapins also falls within the mean egg dimensions reported from the U.S.A. (see reviews in Butler et al., 2006). A latitudinal gradient exists for egg and clutch size in North America, whereby female diamondback terrapins from the northern part of the range have comparatively smaller eggs but greater mean clutch sizes than do females from the southern part of the range (Seigel, 1980; Allman et al., 2012). The results of the current investigation show that the mean egg size measurements and clutch size of Bermuda's terrapins more closely matches those reported for South Carolina's diamondback terrapins than they do those reported from the northern range (e.g. Rhode Island and Maryland) (Allman et al., 2012) (see Fig.5.4). This lends additional support for the Carolina diamondback terrapin (*Malaclemys terrapin centrata*) designation of Bermuda's population (see Parham et al., 2008).

The entire nesting process has been reported to occur in approximately 20 minutes (Burger, 1977; Roosenburg, 1991; Goodwin, 1994), but can last as long as two hours (Roosenburg, 1994), and this also appears to be true for female diamondback terrapins in Bermuda.

The three fully constructed nests which contained no eggs were interpreted to be false nesting events. False nesting has been documented in some diamondback terrapin populations in Massachusetts and Maryland, U.S.A. (Auger and Giovannone, 1979; Roosenburg, 1994).

Annual reproductive output depends upon the number of clutches produced each season. If every sexually mature female terrapin in Bermuda (n=32) participates in annual nesting (mean clutch size of five eggs, nesting frequency $\times 3$ each year), then the average annual production of hatchlings in Bermuda is estimated to be not more than 91 individuals (assuming an annual hatching success rate of 19%).

Incubation and emergence periods

The incubation period for Bermuda's diamondback terrapins is typical for those reported in the U.S.A., which varies from 50-120 days (Burger, 1977; Jeyasuria et al., 1994; Butler et al., 2004). In New Jersey, the mean incubation period was 76.2 days (Burger, 1977), while terrapins on the east Florida coast had a mean incubation period of 65.6 days (Seigel, 1980). Hatching in Bermuda also occurred within the months (July-October) reported by Burger (1977), Roosenburg (1991) and Butler et al. (2004) for terrapin populations in North America.

The timing of nest emergence varies between chelonian taxa, populations and even between siblings within the same nest, and is influenced by biological factors (e.g. evolutionary response and internal timing) as well as physical factors (e.g. rainfall and temperature) (see review in Costanzo et al., 2008). The suggested benefits of delayed emergence (over-wintering) include avoidance of predators and avoidance of exposure to adverse environmental conditions. Conversely, the benefits of early emergence (summer/fall) include the potential to begin feeding and growth immediately (Gibbons and Nelson, 1978). Gibbons and Nelson (1978) postulated that delayed emergence is a strategy employed by species in which high environmental uncertainty exists for hatchlings that emerge immediately after hatching.

In the U.S.A, hatchling diamondback terrapins may depart the nest within hours after hatching (Roosenburg and Kelley, 1996), or they may spend months over-wintering in the nest chamber and emerge during the following spring (Lazell and Auger, 1981; Roosenburg and Kelley, 1996; Baker et al., 2006). This appears to be the case for Bermuda's terrapins as well. Approximately half of the hatchlings monitored in 2010 and 2011 emerged from their nests that season (e.g. between July and October). Nearly one quarter of the 2010 study group over-wintered in their natal nests within the sand bunkers on the Mid Ocean golf course (the remaining terrapins departed unobserved), and nearly half (43.8%) of the hatchlings from the 2011 study group were deemed to be over-wintering. It is believed that many of the hatchlings which departed unobserved in 2010 were deliberately released by well-meaning members of the public.

It is unclear why such a high percentage of Bermuda's terrapin hatchlings over-winter in their natal nests, but there are a number of potential hypotheses which may explain the benefits of delayed emergence in Bermuda. It may be an adaptation in response to avoiding desiccation during the summer months, but this has not been tested yet. Summer rains in Bermuda are typically of short duration and summer droughts lasting many months are not considered unusual. It is also possible that delayed emergence in Bermuda is a response to historical heavy avian predation (particularly from members of the Family Ardeidae) during the summer months in Bermuda's pre-colonial past (e.g. prior to 1609 AD), (but see Chapter 6). Historical writings indicate that herons (species unclear) inhabited Bermuda at the time of human settlement; however these breeding colonies were exterminated shortly thereafter (Wingate, 1982). By the early 21st century, the twelve species of heron that visited the islands of Bermuda primarily occurred either as vagrants or seasonal migrants (Raine, 2003).

Hatching success

The overall mean annual hatching success rate for Bermuda's diamondback terrapins during the three year study period was very low (19%) considering that no nest predation was observed. Nest predation from a variety of small mammals (most notably raccoons (*Procyon lotor*)) has been identified as a major source of egg mortality to diamondback terrapins in North America, accounting for the destruction of 82-99% of nests in some regions (Roosenburg, 1992; Feinberg and Burke, 2003; Butler et al., 2004). Nest survival rates in areas that do not have efficient nest predators are high, with mean hatching success rates reported to be 93% (Cook, 1989; Roosenburg et al., 2003; Roosenburg et al., 2009).

Butler et al. (2004) and Burger (1977) reported low percentages of un-hatched or undeveloped nests, but this was not true of the present investigation where the majority of the eggs (62-78%) did not appear to have any discernible embryos. The population contains adequate numbers of males (Chapter 3), so it appears unlikely that eggs were unfertilized. It is unclear why 54.4% of the nests monitored in 2010 and 69.6% of the nests monitored in 2011 failed to produce any hatchlings. Lethal incubation

temperatures may be partially responsible (see below) and recent toxicological investigations have shown that high levels of petroleum-hydrocarbons, heavy metals and polycyclic aromatic hydrocarbons exist within the pond environment (primarily the benthic sediment) and in terrapin eggs (see Chapter 7). Further research is warranted to determine whether terrapin eggs in Bermuda are being compromised by environmental factors.

Relationship between sand bunkers, nesting frequency, incubation and hatching success

The demonstration that particular bunkers were favoured by nesting female diamondbacks in both of the study years (2010 and 2011) indicates that some potential nesting sites are more valuable than others and therefore have greater conservation significance. The associated finding that frequencies of nesting choice, numbers of emerged hatchlings and numbers of unsuccessful incubations (eggs without embryos, un-emerged hatchlings) are all similarly-ranked suggests that bunker choice influences overall nest success. There was no clear relationship with the frequency of hatchling emergence and lay date, nor between nest ID and calendar date.

Nest temperatures

The temperature of the sand during the incubation period influences embryonic survival, determines the sex of the developing embryos and influences the duration of the incubation period (Ewert and Nelson, 1991; Miller, 1999). The tolerated constant thermal regime for artificially incubated diamondback terrapin eggs is 23-34°C. Eggs that are incubated at constant temperatures $<23^{\circ}\text{C}$ or $\geq 35^{\circ}\text{C}$ fail to hatch (Cunningham, 1939; Wood and Herlands, 1997), but eggs that have been exposed to 24-27°C are reported to have produced all male hatchlings, while those incubated at 30-32°C produced all females (Jeyasuria et al., 1994; Roosenburg and Kelley, 1996). The temperatures that produce mixed sexes for this species are reported to be 28.5-29.5°C (Jeyasuria et al., 1994; Roosenburg and Place, 1994). It is worthwhile to note that the diurnal temperatures in natural nests are seldom constant and the sex ratios produced from natural nests have

been found to be either males or females but rarely both (Roosenburg, 1992).

While natural nests are not subjected to constant incubation temperatures, TSD has been suggested as being a factor in biased sex ratios observed in some U.S.A. terrapin populations (Lovich and Gibbons, 1990; Morreale, 1992). Daily temperatures of the ten monitored nests in Bermuda reached, and exceeded, the lethal threshold for developing embryos ($35^{\circ}\text{C}+$) during the IP – such values sometimes lasting for ten consecutive hours each day. Furthermore, unpublished data show evidence of heat shock in terrapin eggs collected in 2011 (D. Fort, personal communication). The mean temperatures, however, were well within the tolerated thermal regime for diamondback terrapins. Furthermore, mean temperatures during the TSP showed variation which may have favoured the development of different sexes. Five nests (5D1, 5D5, 6B4, 7A1, 7B4) were recorded to have mean daily temperatures that are known to produce male hatchlings (e.g. $24.6\text{--}27.1^{\circ}\text{C}$), four nests (5D9, 7B8, 7C4, 11A1) were recorded to have mean daily temperatures that are known to produce female hatchlings (e.g. $30.0\text{--}30.7^{\circ}\text{C}$), and one nest (5D14) was recorded to have mean daily temperatures that are known to produce mixed sexes (e.g. 28.8°C). The small sample size of the current data set prevents a detailed examination of how temperature has affected sex determination and hatching success; however, it appears that there is no clear relationship between the mean daily incubation temperatures and hatching success in the ten monitored nests (see Fig. A5.33 in Appendix 5). Future studies examining the role that temperature plays in regulating sex determination and hatching success within the Bermuda population are recommended.

Hatchling biometrics

A longitudinal gradient occurs in the sizes of diamondback terrapins throughout the U.S.A. Typically, hatchlings in the northern part of the range have a smaller mean SCL and mass than hatchlings from the south. Burger (1977) reported a mean SCL of 27.5 mm and a mass of 6.8 g for hatchlings in New Jersey. Lovich et al. (1991) reported a mean SCL of 33.4 mm for hatchlings in South Carolina (mass not reported) and Seigel (1980) reported

a mean SCL of 31.9 mm and a mass of 8.8 g for hatchlings in Florida. The mean SCL and mass for Bermuda's hatchling terrapins (31.9 mm and 7.5 g) is more similar to those populations reported from the southern U.S.A. range than those from the northern (see Fig.5.3). Egg mass is considered to be the greatest single determinate of hatchling body mass (Roosenburg and Kelley, 1996).

Scute anomalies were observed in half (50.9%) of the Bermudian hatchlings studied (n=106). The most common anomalies observed involved extra vertebral, costal, and marginal scutes. Variations in the number of these scutes have been reported from terrapin hatchlings in the U.S.A. (Wood and Herlands, 1997; Herlands et al., 2004; Roosenburg et al., 2009). It is possible that high incubation temperatures may be partly responsible for the observed scute irregularities (see Wood and Herlands, 1997; Herlands et al., 2004) as well as exposure to polycyclic aromatic hydrocarbons (Van Meter et al., 2006); however the degree to which Bermuda's small and isolated population is affected by inbreeding is currently poorly understood.

Concluding comments

A recent investigation into the demographic characteristics of the isolated diamondback terrapin population in Bermuda has shown that it is small with a very limited distribution and suffers from low annual rates of recruitment (see Chapter 3). The results of the current study show that the annual production of hatchlings is limited, largely due to low rates of hatching success. All of these factors suggest that this population is at high risk of local extirpation. It is therefore strongly recommended that the in-situ monitoring of hatching success for Bermuda's terrapins be continued. Additional research is needed to determine the specific causes of the low hatching rates and ways to mitigate them. Future studies should examine how temperature control (via artificial egg incubation) affects hatching. Wood and Herlands (1997) and Herlands et al. (2004) reported hatching success rates between 32% and 50% of incubated terrapin eggs recovered from road-killed females. Egg viability should also be investigated within the Bermuda population. Examining oviposited terrapin eggs for the

presence of an embryonic disc will help to establish whether fertility (in either sex) is limiting the hatching rate, and a comprehensive genetic assessment of the population may help to determine if inbreeding is an issue.

Terrapin populations have been shown to decrease when females are forced to nest in marginal habitats where nest survivorship is low (Roosenburg, 1992). Furthermore, females need a wide range of nesting micro-habitats (e.g. variation in elevation, orientation to the sun and amount of shading provided by surrounding vegetation) to maintain balanced sex ratios within a population (Roosenburg and Place, 1994). Given that over 97% of the nests located in 2010 and 2011 occurred within the sand bunkers on the Mid Ocean golf course, and that these areas are greatly affected by anthropogenic activities, it would be prudent to increase and diversify the nesting habitat in the Mangrove Lake, Trott's Pond, South Pond and North Pond area. The creation of new nesting habitat should ideally occur in locations that will minimise the disturbance to nesting females, incubating eggs and over-wintering hatchlings. Any such area should also be relatively large, well-drained, and be easily accessible to female terrapins. Moreover, the finding that some of the existing sand bunkers on the Mid Ocean golf course are more important to nesting diamondback females than other sand bunkers and the associated finding that bunker choice influences overall nest success (reflected in the number of hatchlings that successfully emerged) should direct future conservation management efforts. This study has shown that bunkers 5D, 6B, 7A and 7B had comparatively higher nest numbers and greater hatchling emergence and should therefore be designated as critical nesting habitat for this species on Bermuda.

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Chapter 6: Post-emergent Movements and Survivorship of Diamondback Terrapin Hatchlings in Bermuda

Abstract

A small, native population of diamondback terrapins (*Malaclemys terrapin*) exists on the remote oceanic islands of Bermuda. Radio-telemetry was used to investigate the short-term movements and survivorship of post-emergent hatchling terrapins. Twenty hatchlings ranging from 30.3-34.5 mm straight carapace length (mean 32.6 mm; SD 1.3 mm) and weighing between 7-10g (mean 8.4g; SD 1g) were monitored over two tracking sessions; ten hatchlings were tracked over a 40 day period during the summer of 2010 and ten hatchlings were tracked over a 31 day period during the spring of 2011. The results indicated that mangrove swamps and grass-dominated marshes adjacent to the brackish water ponds inhabited by adult terrapins are important developmental habitats for hatchlings. Proportionally more movement was detected, and greater distances were recorded, during the spring tracking sessions than in the summer sessions. Yellow-crowned night herons (*Nyctanassa violacea*) were found to be predators of small terrapins during the spring (April) when the hatchlings were more active. At least 40% of the hatchlings monitored over that period were believed to have been consumed by herons within one week of deployment (range 2-6 days, mean 4.75 days). Young terrapins in Bermuda may remain susceptible to heron predation for three years following hatching and appear to be most vulnerable in areas affected by continued human disturbance, particularly the grass-dominated marshes that have been incorporated into a golf course.

Introduction

Diamondback terrapins are residents of coastal environments along the Atlantic and Gulf coasts of the U.S.A. Nesting typically occurs in sandy soil environments located above high tide (e.g. beaches and dunes) within these brackish habitats (see reviews in Butler et al., 2006; Ernst and Lovich, 2009). Nest predation from a variety of small mammals (most notably raccoons (*Procyon lotor*)) has been identified as a major source of mortality to diamondback terrapins, accounting for the destruction of 80-99% of nests

in some regions (Roosenburg, 1992; Goodwin, 1994; Feinberg and Burke, 2003; Butler et al., 2004). After emerging from the nest, hatchling terrapins typically seek refuge within the nearest vegetation, and generally show avoidance of open water (Burger, 1977; Lovich et al., 1991; Butler et al., 2004). Growth is most rapid during the first few years after hatching, but slows down considerably once sexual maturity has been attained (Tucker et al., 1995; Roosenburg and Kelley, 1996). Terrapin hatchlings in the U.S.A. have a variety of predators that include small mammals, birds and crabs (see Ernst and Lovich, 2009 for review), however studies quantifying the level of predation on hatchlings are limited.

A small, native population of diamondback terrapins exists on Bermuda (Davenport et al., 2005; Parham et al., 2008) which uses the sand bunkers on a private golf course as artificial nesting habitat. Surveys conducted in 2010 and 2011 revealed that 97% of the observed nesting occurred in only eleven sand bunkers, with those located immediately adjacent to two ponds (Mangrove Lake and South Pond) having the highest nest densities (up to 0.278 nests m⁻²). Diamondback terrapins in Bermuda nest between March and August. Hatchling emergence occurs during two distinct periods in the calendar year; summer/fall (July-October) and winter/spring (January-March) (see Chapter 5). Yellow-crowned night herons prey on small diamondback terrapins in Bermuda (see Chapter 4) and the results of a three year mark-recapture population assessment revealed a very low annual rate of recruitment (see Chapter 5).

Radio-telemetry has been used to study movement patterns, habitat use, and survivorship of different chelonian hatchlings and neonates, including box turtles (*Terrapene carolina*), gopher tortoises (*Gopherus polyphemus*), Blanding's turtles (*Emydoidea blandingii*) and diamondback terrapins (Butler and Graham, 1995; Butler and Sowell, 1996; Draud et al., 2004; Forsythe et al., 2004). Radio-telemetry was chosen as the primary means of investigating the survivorship of Bermuda's terrapin hatchlings that had newly-emerged from natal nests. This was required to inform effective conservation and management planning for this species in Bermuda. Secondary and tertiary goals were to identify areas of residency for hatchlings and small juveniles as well as comparing hatchling activity levels

and movement patterns between those emerging in summer and those emerging in spring.

Methods

Study Site

The entire known Bermuda population of diamondback terrapins lives in only four brackish water ponds on the Mid Ocean golf course, located upon one square kilometre of land at the eastern end of the islands (Fig. 6.1) All four bodies of water are separated from each other by, at most, 380 m of land (straight-line distance between North Pond and Trott's Pond) and all have been incorporated into the golf course as water hazards found between the fifth and eleventh holes. Two of the ponds (Mangrove Lake and Trott's Pond) are also important refugia for a species of endemic killifish (*Fundulus bermudae*) since they contain *ca.* 70% of Bermuda's total population (Outerbridge et al., 2007). Refer to Chapter 2 for more detailed descriptions of the physical and biological characteristics of the ponds and their surrounding wetlands.

Thirty three sand bunkers are situated between the fifth and eleventh holes (Fig. 6.1). Individual bunker areas ranged from 12.3-181.7 m² (mean 56.8 m²) and the minimum straight-line distance to the nearest pond ranged from 5-207 m (mean 71.7 m). Surveys performed in 2010 revealed that 77% of all discovered terrapin nests were located in the sand bunkers on fifth and seventh holes of the Mid Ocean golf course (see Chapter 5).



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Figure 6.1. Aerial photograph from 2003 showing the four diamondback terrapin ponds situated on the Mid Ocean golf course (A=Mangrove Lake, B=Trott's Pond, C=South Pond, D=North Pond) and the generalized locations of the sand bunkers associated with the fifth through eleventh holes (red boxes numbered 5-11).

Radio-telemetry surveys

Ten diamondback terrapin hatchlings were captured in 2010 after newly emerging from ten nests and small radio-transmitters (model BD-2, Holohil Systems Ltd.) with an expected 28-day battery life (range 21-35 days; J. Edwards, personal communication) were attached to the carapaces of these hatchlings following Draud et al. (2004) (see Fig. A6.1 in Appendix 6). Hatchling mass ranged from 7-10 g (mean 8.4 g; SD 0.8 g) and straight carapace length ranged from 31.1-34.4 mm (mean 32.9 mm; SD 1.0 mm). The transmitters weighed 0.7 g and thus were within the 10% limit for telemetry packages (see Beaupre et al., 2004). The hatchlings were subsequently released in sand bunkers on the fifth hole (n=5) and seventh

hole (n=5) on the Mid Ocean golf course and tracked one to two times daily from July 31st – September 8th 2010 using a telemetry receiver (R-1000, Communications Specialists, Inc.) fitted with a two element hand-held radio antenna (RA-2AK, Telonics Inc.). In 2011, an additional ten hatchlings (mass range 7-10 g; mean 8.3 g; SD 1.2 g and straight carapace length range 30.3-34.5 mm; mean 32.3 mm; SD 1.6 mm) which had recently emerged from over-wintering in ten nests were captured, fitted with new transmitters and released in sand bunkers on the fifth hole (n=5), sixth hole (n=2) and seventh hole (n=3) on the Mid Ocean golf course. They were tracked one to two times daily from March 27th – April 28th 2011. All tracking sessions were conducted haphazardly between the hours of 08:00 and 24:00.

Hatchlings were visually observed during each tracking session and a description of the habitat was recorded. Hatchling locations were discretely marked using stakes and the straight-line distances between the stakes and subsequently marked locations were measured. Searches continued for a three day period following the loss of a radio-transmitter signal, at which point the hatchling was no longer considered part of the investigation. The presence of herons (species and number counted) as well as the date and time of the observations were also documented from the area when applicable.

Results

Post-emergence movements

August 2010

Surveys were conducted over a 40-day period. Mean battery life for the BD-2 radio-transmitters was 33.5 days (range 23-40 days). Table A6.1 in Appendix 6 summarises the individual movement histories of the ten diamondback terrapin hatchlings tracked in August 2010. Overall, movement was detected in only 11% of the individual tracking sessions. The mean distance travelled over the survey period was 0.8 m (range 0-60 m).

Upon release, all ten hatchlings crawled immediately to the edge of the bunkers and either buried into the sand or burrowed into the grass

growing at the edge of the bunkers. Only three were observed to subsequently depart the bunkers. One hatchling (#H3) departed the bunker on the fifth hole by the second day following its release, travelled a straight-line distance of *ca.* 15 m and entered the mangrove swamp where it remained until the transmitter was lost 35 days later (Fig. 6.2). The second hatchling (#H6) departed the bunker on the seventh hole at the beginning of the fourth week of study, travelled a straight-line distance of *ca.* 60 m and entered the saw-grass marsh in the centre of South Pond where it remained until the transmitter was removed twelve days later. The third hatchling (#H8) departed the same bunker on the seventh hole at the end of the fourth week, travelled a straight-line distance of *ca.* 16 m and crawled into the grass bordering the seventh hole where it remained until the transmitter was removed ten days later (Fig 6.3).

The remaining seven hatchlings stayed concealed at the margins of their respective bunkers throughout the survey period; most were buried in the sand to depths ≤ 10 cm.

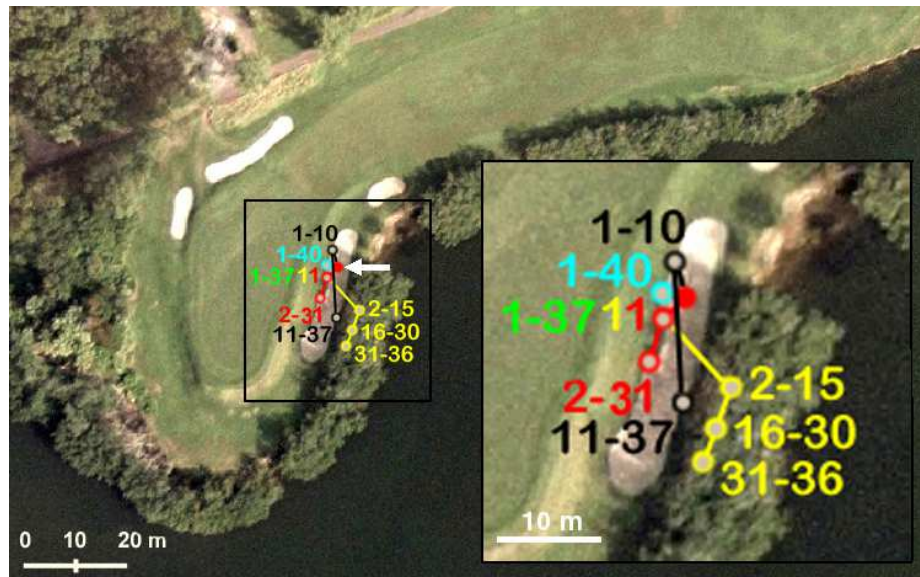


Figure 6.2. Diamondback terrapin hatchling movements from the sand bunkers adjacent to Mangrove Lake in August 2010 (white arrow indicates point of release, open circles represent observed locations of hatchlings, coloured numbers represent days after release). Hatchling identification is as follows: #H1 (red), #H2 (blue), #H3 (yellow), #H4 (green), #H5 (black).

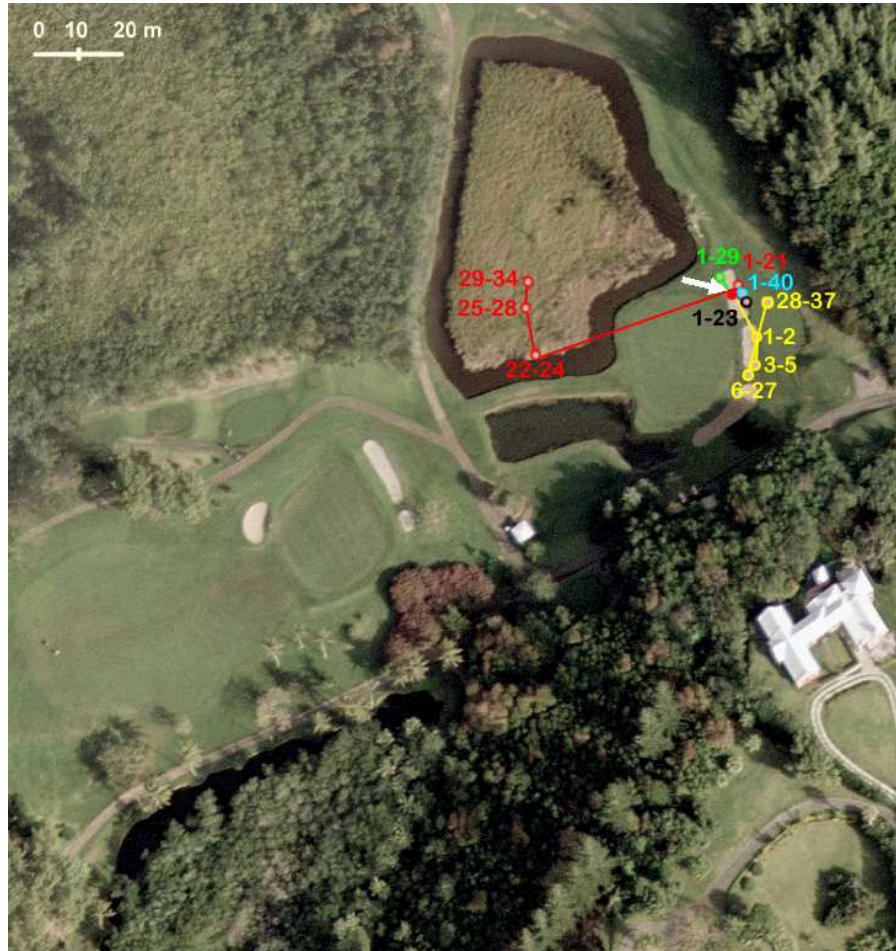


Figure 6.3. Diamondback terrapin hatchling movements from the sand bunkers adjacent South Pond in August 2010 (white arrow indicates point of release, open circles represent observed locations of hatchlings, coloured numbers represent days after release). Hatchling identification is as follows: #H6 (red), #H7 (black), #H8 (yellow), #H9 (green), #H10 (blue).

Thirty three heron sightings were recorded in total on 15 separate days over the 40-day survey period in 2010. All records were of yellow-crowned night herons and most constituted single heron observations, though the greatest number recorded in one tracking session was nine (seven on the fifth hole and two on the seventh hole). These observations occurred between the hours of 13:00-23:00 with the majority after 18:00.

March-April 2011

Surveys were conducted over a 31-day period. Mean battery life for the radio-transmitters that were recovered was 29.3 days (range 28-31 days). Table A6.2 in Appendix 6 summarises the individual movement histories of the ten diamondback terrapin hatchlings tracked between late March and late April 2011. Overall, movement was detected in 59.3% of the individual tracking sessions. The mean distance travelled over the survey period was 6.2 m (range 0-122.5 m).

As with the hatchlings studied in August 2010, all ten hatchlings released in March-April 2011 were observed to crawl immediately to the edge of the bunkers and either bury into the sand or burrow into the grass growing at the edge of the bunkers. By the end of the first week following their release, all of the terrapins had departed from their respective bunkers. The five hatchlings (#H11-H15) monitored at the fifth hole entered the mangroves and showed signs of dispersal along the swamp throughout the remainder of their respective tracking sessions (Fig. 6.4). These individuals were frequently observed seeking refuge within the mangrove leaf litter in close proximity to the water line or in shallow water among the mangrove prop roots. Occasionally individuals were discovered hiding under the pond embankment in areas lacking fringing vegetation (e.g. along the fifth fairway of the golf course).

Two of the hatchlings released in the seventh bunker (#H16 and #H20) moved to the dense mats of *Paspalum* that borders South Pond on the first day following release, travelling straight-line distances of 22.6 m and 35 m respectively, across the open lawn of a putting green (Fig. 6.5). Hatchling #H20 remained hidden within the *Paspalum* for the following 26 days; however, #H16 remained within the *Paspalum* for a two week period before taking up residency within the saw-grass marsh at the centre of South Pond. This individual was tracked for an additional 13 days during which it was repeatedly observed sheltering under dense saw-grass foliage, often partially buried in the marsh substrate. The third hatchling (#H17) released in the sand bunker on the seventh hole could not be located on the second day despite a thorough search of the area, however the transmitter (without terrapin) was found on the third day at a distance of 160 m (see below). The

two hatchlings released in the bunker on the sixth hole (#H18 and #H19) departed within two to three days and travelled straight-line distance of 92.4 m and 122.5 m respectively across the open lawn of the sixth fairway. One hatchling (#H18) crawled to the base of a tree where it remained until it disappeared on the sixth day following its release, and the other (#H19) entered the mangrove swamp adjacent to Mangrove Lake where it continued to move along the fringe of the swamp; it also disappeared on the sixth day following its release (Fig. 6.5).



Figure 6.4. Diamondback terrapin hatchling movements from the sand bunkers adjacent to Mangrove Lake in March-April 2011 (white arrow indicates point of release, open circles represent observed locations of hatchlings, coloured numbers represent days after release). Hatchling identification is as follows: #H11 (blue), #H12 (green), #H13 (black), #H14 (yellow), #H15 (red).

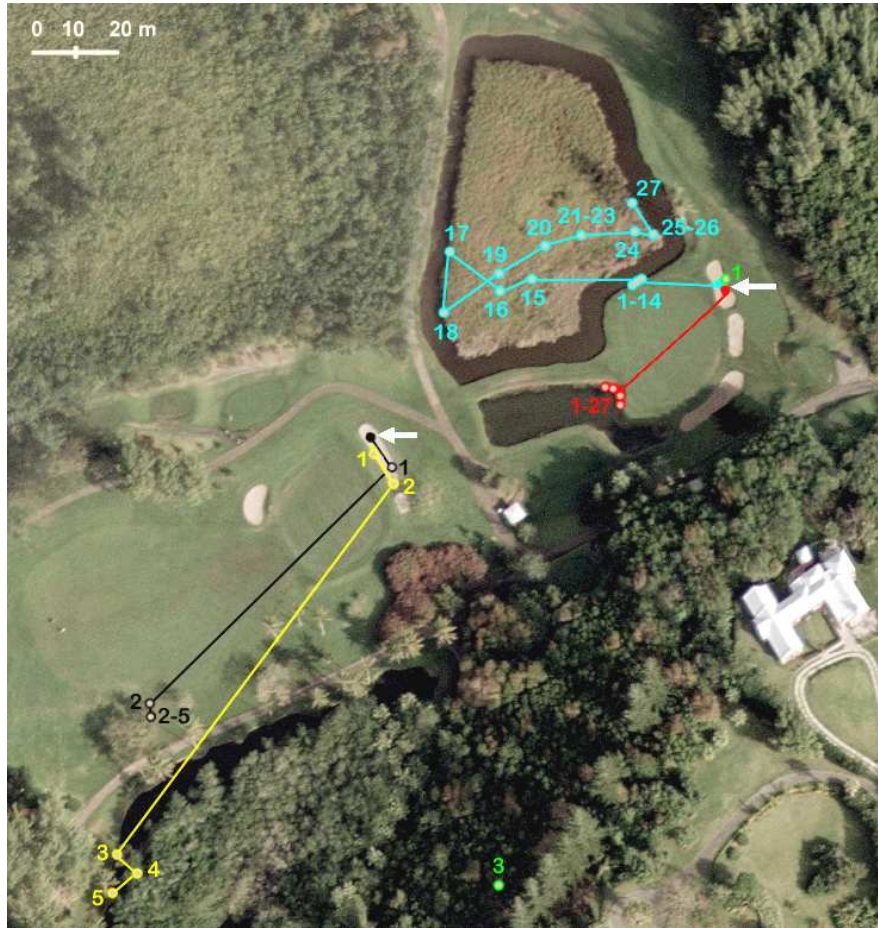


Figure 6.5. Diamondback terrapin hatchling movements from the sand bunkers adjacent to South Pond in March-April 2011 (white arrows indicate points of release, open circles represent observed locations of hatchlings, coloured numbers represent days after release). Hatchling identification is as follows: #H16 (blue), #H17 (green), #H18 (black), #H19 (yellow), #H20 (red).

A total of 49 heron sightings were recorded on 22 separate days over the 31-day survey period in 2011. These records comprised five different heron species; the yellow-crowned night heron, the little blue heron (*Egretta caerulea*), the great egret (*Ardea alba*), the snowy egret (*Egretta thula*), and the tricoloured heron (*Egretta tricolor*). Most constituted single heron observations and occurred between the hours of 08:30-23:00, including one observed predation event on April 20th of a small diamondback terrapin at South Pond (Fig. 6.6).

Survivorship

August 2010

Nine of the ten radio-transmitters were recovered from their respective hatchlings; four from Mangrove Lake and five from South Pond. Fading and/or erratic signals were associated with seven transmitters 23-40 days after deployment (mean 35.4 days) and two transmitters expired without any observed changes in the pulse rate 29 and 32 days following deployment. Only one hatchling vanished during this study due to loss of transmission signal 37 days after release. In total, nine transmitters functioned beyond the expected 28-day life span of their batteries.

March-April 2011

Three of the ten radio-transmitters were recovered from their respective hatchlings; two from Mangrove Lake and one from South Pond. Fading and/or erratic signals were associated with all three transmitters 28-31 days after deployment (mean 29.3 days). The remaining transmitters (and hatchlings) were not recovered (but see below) due to loss of radio signal. Four of these disappeared within one week of deployment (range 2-6 days, mean 4.75 days); the majority of which (n=3) occurred at South Pond. The final three hatchlings disappeared 17, 20 and 28 days following release. No changes in the pulse rates were observed in any of these transmitters prior to their disappearance. One transmitter was located 160 m from its last known location after vanishing for a 24 hr period (see #H17 in Fig. 6.5). This unit was recovered from a pellet that was believed to have been regurgitated from a yellow-crown night heron and which comprised chitinous fragments of terrestrial arthropods and carapace scutes from a diamondback terrapin hatchling (Fig. 6.7).



Figure 6.6. Yellow-crowned night heron with a dead juvenile diamondback terrapin in its bill (arrowed).



Figure 6.7. Regurgitated pellet believed to have come from a yellow-crowned night heron showing a BD-2 radio transmitter unit (arrowed).

Discussion

The use of radio-telemetry on terrapin hatchlings in Bermuda allowed for precise location of the tracked individuals, in spite of their cryptic nature, and the results of the present investigation have shown that the mangrove swamp and the grass-dominated marshes adjacent to the saline ponds on the Mid Ocean golf course are important habitats for the development of young diamondback terrapins in Bermuda. Similar to Bermuda, young diamondback terrapins (i.e. individuals ≤ 75 mm SCL) in the North American range have been found to be cryptic, having been observed hiding under accumulated surface debris, low growing vegetation, rocks and matted *Spartina* grass on tidal mud flats (Pilter, 1985), burrowing into the tidal wrack at the high tide line in salt marshes (Lovich et al., 1991) and within the intertidal vegetation of the high marsh zone (Draud et al., 2004; King, 2007). Furthermore, hatchlings display avoidance of open water and instead crawl to the nearest vegetation upon emerging from nests (Burger, 1976; Lovich et al., 1991; Butler et al., 2004). Muldoon and Burke (2012) performed a detailed study of seasonal movements of hatchling terrapins and found that post-emergent movements in the fall were typically upland, away from water but this trend reversed in the spring.

The mangrove swamps and grass-dominated marshes adjacent to Mangrove Lake and South Pond offer ample food resources (see Chapter 4) and the plant cover provides concealment from predators. These habitats are, however, limited in area. The mangrove swamps are confined to a relatively narrow band surrounding Trott's Pond and Mangrove Lake that quickly grades to open water on the seaward side and into golf course fairways, private gardens and forested regions on the landward side. The grass-dominated marshes in South Pond and North Pond are significantly smaller in area than the mangrove swamps, while the *Paspalum* mats that fringe these ponds are frequently cut back or removed entirely as part of the maintenance program of the golf course. The limited battery life of the BD-2 transmitters did not permit long-term monitoring of hatchling movement; however it is reasonable to assume that these areas are not temporary microhabitat choices. This is supported by the fact that no hatchling-sized or small juvenile terrapins (e.g. < 81 mm straight carapace length) were

encountered in the open water habitat of the ponds during a three year mark-recapture study of the Bermuda population (see Chapter 3).

Proportionally more movement was detected, and a greater mean travel distance was recorded, in the spring tracking sessions than in summer sessions. Terrapin hatchlings in Bermuda are clearly more active following emergence from brumation during the spring than after emergence from their nests in the summer. However, this increased spring activity occurs at a time of increased heron activity around the wetlands (M. Outerbridge, personal observation) which may make the terrapins more susceptible to avian predation.

None of the hatchlings monitored during the August 2010 tracking session were believed to have been lost via predation, but the results of the April 2011 study indicate that at least 40% ($n=4$) of the hatchlings monitored were probably consumed by herons in the vicinity of South Pond and Mangrove Lake. Similarly, Butler and Sowell (1996) reported higher rates of predation on hatchling and yearling gopher tortoises during the spring (April-May) than during other times of the year. Moreover, Bermuda's terrapin hatchlings from South Pond appear to be more at risk from avian predation than those from the Mangrove Lake area. This is believed to reflect the lack of adequate plant cover surrounding the sand bunkers at South Pond and in the area outside the saw-grass marsh.

One of the limitations of using radio-telemetry to examine survivorship is that without physical evidence (e.g. a carcass or a transmitter) it is not possible to differentiate mortality from tag failure or animal dispersal. It is unlikely, in the present study, that a hatchling moved beyond the range of the receiver since diamondback terrapins in Bermuda are known to only reside in a limited area (i.e. less than 1 km^2) and the daily tracking sessions often involved extensive searches of the wetlands. However, it is possible that the reason most of the transmitters were not recovered was because of heron-mediated dispersal away from the study area. All of the lost transmitter signals during the April 2011 tracking session were preceded by observations of herons foraging in the areas where the hatchlings were being monitored. Given that one hatchling was confirmed to be consumed by a heron during the first week of study in 2011,

it is reasonable to assume that the other three hatchlings which vanished during the same period were also due to avian predation. This is consistent with observations made during a four week period in the spring of 2010 when ten small diamondback terrapins were witnessed being preyed upon by yellow-crowned night herons foraging in the saw-grass marsh and *Paspalum* at South Pond (see Chapter 4). Draud et al. (2004) speculated that rats may become efficient hunters of terrapin hatchlings once they have acquired the appropriate search image and might then exploit these hatchlings so long as prey densities were high enough in the marsh habitat (e.g. at peak emergence times). Once the hatchling density dropped (e.g. via dispersal) the rats would be expected to switch to alternative prey sources. This may also be true of yellow-crowned night herons in Bermuda that prey upon terrapin hatchlings in the areas where the greatest nesting densities (e.g. the sand bunkers of the fifth, sixth and seventh holes on the Mid Ocean golf course) and the developmental areas for young terrapins (e.g. the wetlands adjacent to South Pond and Mangrove Lake) coincide, especially if this occurs at the time of greatest synchronous emergence (e.g. spring). It is unclear what caused the remaining three terrapins to vanish during the 2011 radio-telemetry investigation. It is possible that their transmitters expired since signal loss occurred close to or within the expected range of battery life (21-35 days), however avian predation cannot be ruled out.

Historical writings indicate that herons (species unclear) inhabited Bermuda at the time of human settlement (1609 AD). However, by the 19th century, records indicate that herons were no longer breeding in Bermuda but occurred as regular migrants to the islands (Wingate, 1982). Skeletal remains found in Pleistocene and Holocene cave and pond deposits indicate that an endemic, crab-eating heron (*Nyctanassa carcinocatactes*) was present on Bermuda, but went extinct subsequent to human colonization in the early 17th century (Olsen and Wingate, 2006). This is unsurprising as herons were regarded as delicacies for centuries in Europe before modern conservation measures (Holloway, 1996). During the late 1970s an attempt was made to establish a breeding population of an extant con-generic heron (the yellow-crowned night heron) as a potential agent of biological control of a terrestrial species of land crab (*Gecarcinus lateralis*) that was deemed a

pest at the time. Between 1976 and 1978, 46 chicks were translocated from Florida to Bermuda, hand-raised and released into the wild. The first confirmed breeding within this newly established population occurred in 1980 (Wingate, 1982). This species is now the most common heron on Bermuda, breeding has been documented island-wide and the present-day population is considered to be self-sustaining (J. Madeiros, personal communication). Herons of the genus *Nyctanassa* have been described as crustacean specialists (del Hoyo et al., 1992) and examination of regurgitated pellets on Bermuda during the early 1980s revealed that land crabs comprised approximately 97% of their diet; however the remains of terrestrial arthropods were also occasionally found, indicating that these herons were capable of preying upon non-crustacean species (Wingate, 1982). More recent foraging observations and examination of regurgitated pellets suggest that yellow-crowned night herons in Bermuda exhibit a much greater diversity of prey items than previously reported, that includes a wide variety of terrestrial arthropods, small fishes, marine crustaceans, small amphibians, and reptiles (M. Outerbridge, unpublished data). Diamondback terrapins ranging in size from 96-137 mm straight carapace length (SCL) were encountered during an assessment of the Bermuda population (see Chapter 3) and showed signs of carapace damage of a near-identical nature to those observed on a young specimen (51 mm SCL) that had been killed by a yellow-crowned night heron (Figs. A6.3 and A6.4 in Appendix 6). This suggests that yellow-crowned night herons prey on young diamondback terrapins until the latter attain a size of at least 96 mm SCL, when they are approximately three years old (Gibbons et al., 2001).

The other members of the Family Ardeidea that were observed frequenting the study area during the present telemetry investigation are reported as being primarily piscivorous (del Hoyo et al., 1992) and there are no published records of them preying upon diamondback terrapins in Bermuda.

Predators of small diamondback terrapins in the North American range include raccoons, the Norway rat (*Rattus norvegicus*), ghost crabs (*Ocypode quadrata*), and a variety of birds (including night herons) (Burger, 1976; Arndt, 1991, 1994; Draud et al., 2004; Rulison, 2009). Raccoons are

not present on Bermuda and ghost crabs have not been encountered within the area inhabited by Bermuda's diamondback terrapins. However, rats (*Rattus rattus* and *R. norvegicus*) and feral cats (*Felis catus*) have been seen. Draud et al. (2004) reported that the Norway rat was responsible for preying upon 67% of terrapin hatchlings and small juveniles (25-41 mm straight carapace length) studied in a New York population. Furthermore, the hatchlings appeared to be most vulnerable during the first few days following emergence from nests and hibernacula, when hatchling densities were highest due to synchronous emergence. Rats have been observed within the saw-grass and mangrove swamps adjacent to the ponds on the Mid Ocean golf course (M. O.), although predation by rats could not be verified by the Bermuda radio-telemetry investigation. Rats are not presently subject to rodent control measures in the area. Feral cats, widespread on Bermuda, are actually attracted to the Mid Ocean golf course property which has established feeding/watering stations, including one at South Pond. Such stations are associated with a feral cat sterilization program. It is not known whether the cats prey upon diamondback terrapin hatchlings, but Seabrook (1989) reported that approximately 90% of the cat scats collected from a study site on a single island in the Indian Ocean contained green turtle (*Chelonia mydas*) hatchling remains; cats are also known predators of Galapagos tortoise (*Geochelone nigra*) hatchlings and young juveniles (Swingland, 1989).

Concluding comments

Delayed sexual maturity, longevity and iteroparity are key characteristics of the population biology of chelonians (Gibbs and Amato, 2000), but these very characteristics also make their populations less capable of responding to elevated rates of juvenile mortality (Congdon et al., 1993). The observed level of avian predation on diamondback terrapin hatchlings during the first month following spring emergence in Bermuda may be partially responsible for the low levels of recruitment that have been documented in the adult population (approximately two terrapins per annum) (see Chapter 3). Furthermore, yellow-crowned night herons appear to remain predators of small juvenile terrapins for three years following hatching. This is of

particular concern given that the average annual production of hatchlings in Bermuda is estimated to be no more than 91 individuals (see Chapter 5). High post-hatching survival rates are necessary to ensure that adequate recruitment occurs to maintain a stable population.

Continued monitoring of this vulnerable population to determine temporal population trends is vital, as are further radio-telemetry studies to examine avian predation rates. It would also be prudent to control rats and to relocate the feral cat feeding station from South Pond to an area where terrapin hatchlings are absent. A limited cull of yellow-crowned night herons, particularly at South Pond, should be considered in the spring when terrapin hatchlings appear to be more conspicuous.

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Chapter 7: Eco-Toxicological Assessments of Diamondback Terrapin Habitat, Prey and Eggs in Bermuda

Abstract

Total petroleum hydrocarbons, polycyclic aromatic hydrocarbons and a variety of heavy metal residues (arsenic, cadmium, chromium, copper, iron, lead, nickel, zinc and mercury) were extracted and analyzed from fresh whole diamondback terrapin (*Malaclemys terrapin*) eggs, whole aquatic gastropods (*Heleobops bermudensis*, *Melanoides tuberculata*, *Melampus coffeus*) and benthic sediment from the pond environments in Bermuda inhabited by the terrapins. Biomagnification was detected, with the gastropods and the terrapin eggs showing elevated levels of heavy metals and organic pollutants by comparison with sediments. Conversely, polycyclic aromatic hydrocarbons were mostly found within the sediment and lesser amounts were detected in the gastropods and eggs. It is evident that contaminants are transferred to eggs, and that the concentration of several contaminants exceeds those known to cause damage in a range of aquatic vertebrates. Some of the contaminants are known to have mutagenic and teratogenic effects at the observed concentrations and may be reducing the incidence of successful embryonic development for this species in Bermuda. Bermudian diamondback terrapins evidently live and feed in wetland habitats characterized by chronic, multifactorial contamination that renders their main food source potentially dangerous to consume. This study suggests that environmental contamination may be contributing to the low hatching success shown by diamondback terrapin eggs on Bermuda.

Introduction

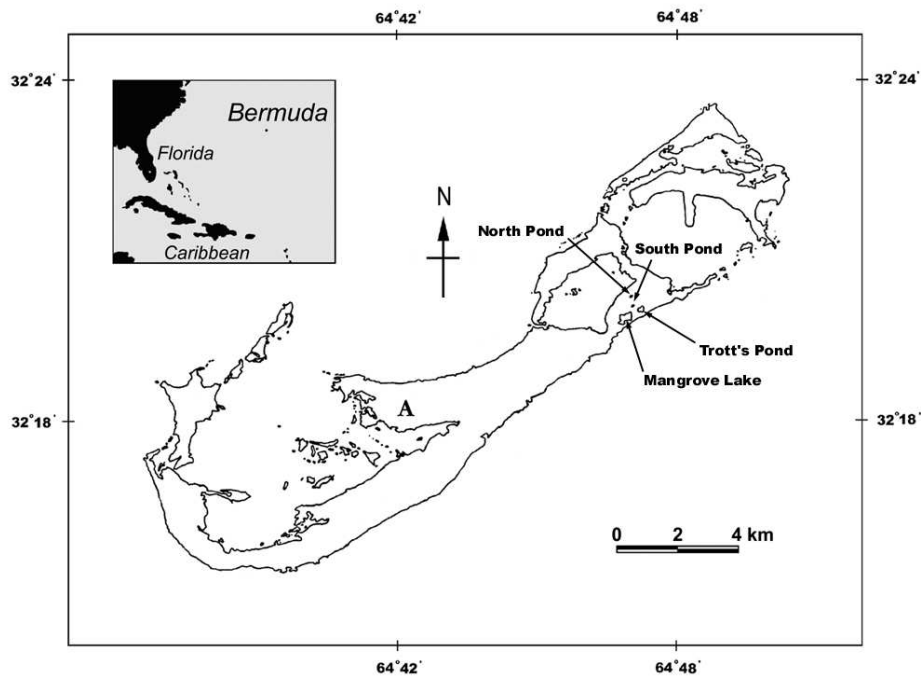
Bermuda has a long history of environmental contamination by chemicals of various types. Insecticides have been used for decades to control mosquitoes (D. Kendall, personal communication), while the great popularity of golf courses on the islands (there are nine in total) has promoted herbicide use (J. Bacon, personal communication). In addition, despite its small size (54 km²), Bermuda has a high human population (approximately 65,000) and is home to around 44,000 licensed road vehicles of various types. Bermuda

also has over 7,600 powered recreational watercraft (motorboats and jet skis) that heavily use the surrounding waters (Adwick et al., 2005). Many of such craft have two-stroke engines that are known to emit far greater quantities of hydrocarbons to the atmosphere than road vehicles (e.g. Davenport and Switalski, 2006). Throughout much of the 20th century, garbage (including metallic objects) has been disposed of in landfills, coastal waters and ponds across Bermuda (Sterrer and Wingate, 1981). Today, however, high temperature incineration is the main method of disposal of municipal solid waste, from residential and commercial sources. Although not industrialised, Bermuda is consequently characterised by high levels of localised anthropogenic pollution (e.g. Jones, 2011).

Recent investigations of the health status of the pond environment in Bermuda suggest that there is a suite of contaminants of concern that are having detrimental effects on the resident fauna (Fort et al., 2006; Fort et al., 2006; Bacon, 2010; Bacon et al., 2012). These contaminants include petroleum hydrocarbons (gasoline-range organics (TPH-GRO), diesel-range organics (TPH-DRO)), polycyclic aromatic hydrocarbons (PAH) and heavy metals. Entry of contaminants into the wetlands comes through storm-water run-off from adjacent roadways, car parks and house drives, aerial deposition and leachate from nearby landfills and ground-water sources (Fort et al., 2006). Ponds located within and adjacent to golf courses are among the most toxic wetlands in Bermuda (J. Bacon, personal communication). Tissue residue analyses from a range of taxa, including cane toads (*Rhinella marinus*), mosquitofish (*Gambusia holbrooki*), killifish (*Fundulus* spp.), and red-eared sliders (*Trachemys scripta elegans*) collected from a variety of contaminated wetlands across Bermuda have shown that petroleum hydrocarbons, polycyclic aromatic hydrocarbons and heavy metals are being accumulated and induce developmental malformations, endocrine disruption, liver and gonad abnormalities plus immunological stress (Bacon, 2010; Bacon et al., 2012).

Diamondback terrapins (*Malaclemys terrapin*) are considered to be native species to Bermuda (Parham et al., 2008), where they are residents of the land-locked, brackish water pond environment (Davenport et al., 2005) (Fig. 7.1). Bermuda's diamondback terrapins are presently considered to be

very vulnerable to local extirpation given the small population size (approximately 100 individuals ≥ 81 mm straight carapace length) and highly localized distribution (four brackish water ponds situated on one square km of land within a private golf course) (see Chapter 3). Furthermore, recent studies have shown that the annual hatching success for this population is low (*ca.* 19%) despite the total absence of nest predators (see Chapter 5).



Source: Mark Outerbridge

Figure 7.1. Map of Bermuda showing the location of the diamondback terrapin ponds; Mangrove Lake, South Pond, North Pond and Trott's Pond. (A=city of Hamilton).

Diamondback terrapins are known molluscivores throughout their North American range (Tucker et al., 1995), and investigations into the feeding ecology of Bermuda's diamondback terrapins have shown that they ingest substantial quantities of small gastropods, which are known bio-accumulators of both inorganic and organic toxic compounds (e.g. Walsh et al., 1995), together with appreciable quantities of surface benthic sediments (see Chapter 4). Terrapins in the U.S.A. accumulate heavy metals in liver

and muscle tissue (Burger, 2002), accumulate PAHs in eggs (Holliday et al., 2008), and have been used as bio-indicators of environmental contaminants in salt marsh ecosystems (Blanvillain et al., 2007; Basile et al., 2011), however the long-term effects of such exposure are unknown.

Cognisant of the aforementioned research, it seemed possible that terrapins in Bermuda, like other aquatic fauna, might be negatively affected by TPHs, PAHs and heavy metals which could put the population at risk. The objectives of the present study were to examine the levels of toxic contaminants in benthic sediments in water bodies where diamondback terrapins have been recorded, as well as in aquatic gastropods on which they feed, and also in terrapin eggs. Given the small population size of terrapins in Bermuda, destructive sampling of adults, juveniles or hatchlings was deemed unethical on conservation grounds. Thus, in order to investigate exposure to, and absorption of, toxic contaminants, it was decided to analyze samples of benthic sediment, aquatic gastropods as well as whole diamondback terrapin eggs for TPH, PAH and heavy metal residues. Funding was not available to permit analysis of levels of persistent organic pollutants (POPs) such as organochlorine pesticides (OCPs), polychlorobiphenyls (PCBs) or polybrominated diethyl ethers (PBDEs).

Examining the extent to which Bermuda's diamondback terrapins are impacted by contaminants, and how this influences survival, is critical to the design of appropriate management initiatives and wetland remediation activities.

Methods

Study sites

Mangrove Lake and Trott's Pond are among the largest ponds on Bermuda, consisting of approximately ten hectares (ha) and three hectares open water respectively, and both are surrounded by a narrow fringe of red mangrove trees (*Rhizophora mangle*) that have given rise to relatively small mangrove swamps totalling 2.3 ha and 0.8 ha respectively (Thomas et al., 1991; Thomas, 1993). Both bodies of water are saline (annual average salinity of 28.6, measured using a refractometer), situated <200 m from the coast and

are connected to the ocean via small subterranean fissures (Thomas et al., 1991). South Pond is one of the smallest ponds on Bermuda and consists of 0.18 ha of open water and 0.27 ha of saw-grass (*Cladium jamaicense*) marsh. This brackish water pond has an annual average surface salinity of 10.8 (see Chapter 2), and, as it is not connected to the ocean, the salinity and water levels of South Pond are greatly affected by rainfall. All three ponds are relatively shallow (mean depth range 35-269 cm) and have been incorporated into the Mid Ocean golf course as water hazards since the 1920s (Fig. 7.2). Mangrove Lake has deep deposits of gelatinous, sapropelic benthic sediment (Hatcher et al., 1982) and, based on preliminary examination, this appears to be true also of the other ponds as well (M. Outerbridge, personal observation). See Chapter 2 for more detailed descriptions of the physical and biological characteristics of these ponds and their associated wetlands.



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Figure 7.2. Aerial photograph from 2003 showing the four diamondback terrapin ponds situated on the Mid Ocean golf course (A=Mangrove Lake, B=Trott's Pond, C=South Pond, D=North Pond).

Sediment and tissue collection and analyses

Benthic sediment was collected haphazardly from a number of locations within Mangrove Lake (n=5 samples), Trott's Pond (n=3 samples) and South Pond (n=4 samples) in 2009 as part of an island-wide assessment of Bermuda's wetland health. The sediment was collected from a boat using a long-handled dip net with a mesh size of 1 mm. Each sample constituted sediment skimmed from the surface of the pond bottom, poured into sterile 4.5 litre glass bottles (giving a total of twelve 4.5 litres sediment samples) and refrigerated at 6°C prior to shipment.

Whole body samples of aquatic gastropods were haphazardly collected in July 2011 from Mangrove Lake and South Pond during a series of benthic biotic transect surveys, that formed part of a feeding ecology investigation of Bermuda's diamondback terrapins (see Chapter 4). Approximately 6 g of live hydrobiid snails (*Heleobops bermudensis*) was collected from these combined ponds, 6 g of live red-rimmed melania snails (*Melanoides tuberculata*) was collected from South Pond and 18 g of live coffee bean marsh snails (*Melampus coffeus*) was collected from Mangrove Lake, giving a total sample of 30 g of gastropods.

Eleven whole diamondback terrapin eggs were collected from eleven different nests, discovered during nesting surveys in June and July 2011, within sand bunkers on the Mid Ocean golf course (see Chapter 5). This represents about 4.5% of annual laying by the population. All gastropods and eggs were frozen following collection.

All collected samples were shipped to Fort Environmental Laboratories Inc. in Oklahoma, U.S.A. for analyses. Total Petroleum Hydrocarbons (TPH), both diesel range (DRO) and gasoline range (GRO), were extracted and analyzed in accordance with SW-846 under EPA method 3510 (DRO extraction) and OK8000/81 and OK 8020/80, respectively using GC-MS and 1 g of each sample. Polycyclic Aromatic Hydrocarbons (PAH) analyses were also performed in accordance with SW-846. One g of sample was extracted in 100 mL of hexane using supercritical fluid extraction (SFE) in accordance with EPA method 3560 and analyzed by GC-MS in accordance with EPA method 8270 corrected for small volumes. Metal analyses were performed in accordance with SW-846 under EPA method

200.7 and corrected for small volumes. One g of sample was digested in HNO₃/HCl and analyzed by inductively coupled plasma-atomic emission spectrometry (ICP-AES) (USEPA, 2008).

Data analysis for PAHs and heavy metals follows marine sediment guidelines adopted by the New Jersey Department of Environmental Protection (N.J. DEP). The low effects range (ERL) represents a concentration at which adverse benthic effects were found to have impacted 10% of cases studied, whereas the median effects range (ERM) represents a concentration at which adverse benthic effects were found to have impacted more than 50% of cases studied. As the state of New Jersey does not have guideline values for TPH-GRO and TPH-DRO, the low effects level guidelines were values established by the state of Oklahoma for soil remediation at contaminated sites and the severe effects level guidelines were values established by the state of California for soil screening. There are no comparable guidelines for safe amounts of TPHs, PAHs and metals in biological samples.

Results

Tables 7.1 and 7.2 summarise the amounts of total petroleum hydrocarbon, heavy metal and polycyclic aromatic hydrocarbon residues found in composite samples of pond sediment from Mangrove Lake, South Pond and Trott's Pond, aquatic gastropods collected from Mangrove Lake and South Pond and diamondback terrapin eggs collected from the sand bunkers on the Mid Ocean golf course. For the full dataset see Table A7.1 and A7.2 in Appendix 7. Table 7.3 summarises the regulatory values for heavy metals and total petroleum hydrocarbons established by the states of New Jersey, Oklahoma and California. Table 7.4 summarises the regulatory values for polycyclic aromatic hydrocarbons established by the state of New Jersey.

In general terms, the TPHs and heavy metals were found in greater concentrations in the aquatic gastropods and terrapin eggs than in the pond sediment, whereas the greatest amounts of the PAHs were mostly found within the sediment and lesser amounts were detected in the gastropods and eggs.

Sediment

Results showed that the sediment from Mangrove Lake, South Pond and Trott's Pond was highly contaminated with a variety of toxic compounds. Elevated amounts of diesel-range organic petroleum hydrocarbons and heavy metals were detected in all three ponds (see Table 7.1). The mean composited value for TPH-DRO was close to, and the maximum value exceeded, the severe effects level and the mean composited values for arsenic, cadmium, copper and mercury exceeded the low effects range for marine sediment screening guidelines established by the N.J. DEP; however the maximum amounts detected did not exceed the median effects range guidelines (see Table 7.3). PAHs were also detected in all three ponds at levels that exceed the low effects and median effects ranges. Eight (50%) of the PAHs examined had mean values that exceeded the low effects range and 6 PAHs exceeded the median effects range for sediment quality guidelines (compare Table 7.2 vs 7.4). Examination of the maximum PAH values shows that eight compounds, including acenaphthylene, acenaphthene, phenanthrene, anthracene, pyrene, benzo(a)anthracene, benzo(g,h,i)perylene and dibenz(a,h)anthracene, were found to greatly exceed (in some cases up to eight times) the median effects range (see Table 7.4).

Gastropods

The data show that the aquatic gastropods accumulate significant amounts of diesel-range petroleum hydrocarbons and all metals except iron. In some cases the mean values of metal residues found in the composite gastropod samples were $\times 10$ to $\times 20$ greater (e.g. lead, cadmium and zinc), and mercury residues were $\times 64$ greater, than the mean values found in the composite sediment samples (Table 7.1 and 7.2). Of the three different gastropod species examined, the coffee bean marsh snail (*Melampus coffeus*) was found to have the greatest TPH and metal residues (Table A7.1 in Appendix 7). A number of PAHs that were found in high amounts in the benthic sediment were not detected in the gastropods; however, elevated mean PAH values for fluorene, pyrene, chrysene and benzo(a)anthracene

were detected in greater amounts in the composite aquatic gastropod samples, indicating biomagnification of these compounds.

Diamondback terrapin eggs

The data show that mean values of cadmium, copper, lead, zinc, mercury and the TPH-DROs were significantly higher in the terrapin eggs than in the benthic sediment, but lower than those detected in the aquatic gastropods (see Table 7.1). Four PAHs (fluorene, fluoranthene, chrysene and benzo(a)anthracene) were also detected in greater amounts in the composite egg samples than in the sediment (see Table 7.2), including two (fluoranthene and chrysene) that showed clear biomagnification from sediment to gastropods to terrapin eggs.

Table 7.1. Summary of the heavy metal and total petroleum hydrocarbon residues (dry weight values) found in composite samples of pond sediment, aquatic gastropods and diamondback terrapin eggs from Bermuda. BDL=Below Detection Limit.

Sample (sample size)		Arsenic (mg/kg)	Cadmium (mg/kg)	Chromium (mg/kg)	Copper (mg/kg)	Iron (mg/kg)	Lead (mg/kg)	Nickel (mg/kg)	Zinc (mg/kg)	Mercury (mg/kg)	Gasoline-range petroleum hydrocarbons (mg/kg)	Diesel-range petroleum hydrocarbons (mg/kg)
Pond sediment (n=12 x 4.5 L)	max.	56.27	4.04	125.00	112.28	12884.62	42.20	8.79	64.95	0.27	0.77	148.33
	min.	19.04	BDL	27.02	21.63	2543.86	14.55	3.64	25.22	BDL	BDL	BDL
	mean	35.18	2.94	58.29	67.73	5990.40	27.96	5.61	46.05	0.21	0.77	104.58
	SD	13.31	0.86	32.97	27.75	3465.33	8.05	1.57	12.03	0.04	-	35.27
Pond gastropods (30g)	max.	69.03	80.96	428.71	545.10	353.49	673.13	23.43	1585.84	29.93	BDL	467.51
	min.	33.46	47.33	125.26	129.49	138.71	269.53	6.82	597.96	4.36	-	152.04
	mean	51.24	65.79	272.09	321.94	230.44	388.79	16.85	1043.19	13.48	-	278.49
	SD	18.95	13.86	125.39	183.07	89.85	190.57	7.60	488.66	11.52	-	134.85
Terrapin eggs (n=11)	max.	58.49	86.32	65.41	131.60	147.17	469.86	3.64	227.70	10.52	BDL	417.88
	min.	5.40	12.59	6.13	66.29	64.78	69.08	0.84	98.62	1.65	-	80.74
	mean	28.16	37.74	26.17	90.21	100.66	167.66	1.97	165.11	4.37	-	225.57
	SD	15.09	23.42	16.95	22.20	25.97	117.53	0.78	36.53	2.79	-	117.01

Table 7.2. Summary of the polycyclic aromatic hydrocarbon residues (dry weight values) found in composite samples of pond sediment, aquatic gastropods and diamondback terrapin eggs from Bermuda. BDL=Below Detection Limit.

Sample (sample size)		Naphthalene (µg/kg)	Acenaphthylene (µg/kg)	Acenaphthene (µg/kg)	Fluorene (µg/kg)	Phenanthrene (µg/kg)	Anthracene (µg/kg)	Fluoranthene (µg/kg)	Pyrene (µg/kg)	Benzo(a)anthracene (µg/kg)	Chrysene (µg/kg)	Benzo(b)fluoranthene (µg/kg)	Benzo(k)fluoranthene (µg/kg)	Benzo(a)pyrene (µg/kg)	Indeno(1,2,3-Cd)pyrene (µg/kg)	Dibenz(a,h)anthracene (µg/kg)	Benzo(g,h,i)perylene (µg/kg)
Pond sediment (n=12 x 4.5 L)	max.	BDL	1567.16	3789.47	BDL	2684.21	4333.33	BDL	2982.46	2754.39	BDL	BDL	BDL	BDL	BDL	2105.26	2456.14
	min.	BDL	BDL	BDL	BDL	BDL	1090.91	BDL	789.47	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
	mean	-	1567.16	2462.05	-	1856.35	2628.00	-	1827.77	1760.54	-	-	-	-	-	1396.92	2300.65
	SD	-	-	1087.96	-	620.78	1165.01	-	755.96	734.62	-	-	-	-	-	571.53	219.90
Pond gastropods (30g)	max.	BDL	BDL	2754.95	261.79	2218.98	1986.42	136.36	2667.36	474.30	172.65	BDL	BDL	690.30	BDL	1069.86	244.19
	min.	BDL	BDL	225.90	BDL	445.14	698.80	BDL	619.99	BDL	BDL	BDL	BDL	107.67	BDL	347.64	BDL
	mean	-	-	1417.03	158.34	1610.89	1502.94	136.36	1868.80	247.36	160.92	-	-	313.92	-	619.02	158.12
	SD	-	-	1073.78	91.37	792.72	557.98	-	901.98	197.11	16.59	-	-	259.92	-	343.92	74.55
Terrapin eggs (n=11)	max.	BDL	BDL	369.67	192.92	361.07	390.36	354.62	666.99	350.00	387.08	BDL	BDL	121.94	BDL	218.80	BDL
	min.	BDL	BDL	28.95	BDL	57.79	46.02	77.01	149.27	74.22	89.24	BDL	BDL	BDL	BDL	BDL	BDL
	mean	-	-	136.42	88.06	170.61	155.48	206.56	403.00	192.87	220.01	-	-	83.69	-	117.92	-
	SD	-	-	121.95	50.11	100.85	112.43	93.99	182.37	92.14	101.99	-	-	22.23	-	54.77	-

Table 7.3. Regulatory values for metals established for marine and estuarine environments by the New Jersey Department of Environmental Protection and total petroleum hydrocarbons established for soil screening and remediation by the states of Oklahoma and California.

	Arsenic (mg/kg)	Cadmium (mg/kg)	Chromium (mg/kg)	Copper (mg/kg)	Lead (mg/kg)	Nickel (mg/kg)	Zinc (mg/kg)	Mercury (mg/kg)	Gasoline-range petroleum hydrocarbons (mg/kg)	Diesel-range petroleum hydrocarbons (mg/kg)
Low effects range (ERL)	8.2	1.2	81	34	47	21	150	0.15	-	-
Median effects range (ERM)	70	9.6	370	270	218	52	410	0.71	-	-
Oklahoma (low effects)	-	-	-	-	-	-	-	-	50	50
California (severe effects)	-	-	-	-	-	-	-	-	110	110

Table 7.4. Regulatory values for polycyclic aromatic hydrocarbons established for marine and estuarine environments by the New Jersey Department of Environmental Protection.

	Naphthalene ($\mu\text{g/kg}$)	Acenaphthylene ($\mu\text{g/kg}$)	Acenaphthene ($\mu\text{g/kg}$)	Fluorene ($\mu\text{g/kg}$)	Phenanthrene ($\mu\text{g/kg}$)	Anthracene ($\mu\text{g/kg}$)	Fluoranthene ($\mu\text{g/kg}$)	Pyrene ($\mu\text{g/kg}$)	Benzo(a)anthracene ($\mu\text{g/kg}$)	Chrysene ($\mu\text{g/kg}$)	Benzo(k)fluoranthene ($\mu\text{g/kg}$)	Benzo(a)pyrene ($\mu\text{g/kg}$)	Indeno(1,2,3-Cd)pyrene ($\mu\text{g/kg}$)	Dibenz(a,h)anthracene ($\mu\text{g/kg}$)	Benzo(g,h,i)perylene ($\mu\text{g/kg}$)
Low effects range	160	44	16	19	240	85	600	665	261	384	240	430	200	63	170
Median effects range	2,100	640	500	540	1,500	1,100	5,100	2,600	1,600	2,800	1,340	1,600	320	260	320

Table 7.5. Comparison of mean dry weight specific metal levels recorded in diamondback terrapin eggs from Bermuda (this study) and Tuckerton, New Jersey (Burger, 2002). Only five metals were analysed in both studies; those of Burger (2002) have been converted assuming 70% of egg mass is made up of water.

Site	Arsenic (mg/kg)	Cadmium (mg/kg)	Chromium (mg/kg)	Lead (mg/kg)	Mercury (mg/kg)
New Jersey	0.04	0.0009	1.30	0.13	0.12
Bermuda	28.16	37.74	26.17	167.66	4.37

Table 7.6. Comparison of PAH concentrations recorded in diamondback terrapin eggs from Bermuda (this study) and eggs collected from a Maryland creek subject to an oil spill one year earlier (Holliday et al., 2008). A=‘Clean site’ (Golden Beach), B=‘Contaminated site’ (Sheridan Point), BDL= Below Detection Limit, NM= Not Measured.

PAH	Bermuda (µg/kg)	Maryland	
		A	B
Naphthalene	BDL	BDL	106.6
Acenaphthylene	BDL	BDL	46.9
Acenaphthene	136.4	BDL	52.4
Fluorene	88.1	BDL	BDL
Phenanthrene	170.6	BDL	BDL
Anthracene	155.5	BDL	67.9
Fluoranthene	206.6	BDL	433.3
Pyrene	403.0	BDL	88.1
Benzo(a)anthracene	192.9	BDL	60.6
Chrysene	220.0	BDL	BDL
Benzo(b)fluoranthene	BDL	BDL	BDL
Benzo(k)fluoranthene	BDL	BDL	82.1
Benzo(a)pyrene	83.7	116.6	95.5
Perlyene	NM	26.5	113.2
Indeno(1,2,3-Cd)pyrene	BDL	BDL	402.0
Dibenz(a,h)anthracene	117.9	BDL	BDL
Benzo(g,h,i)perylene	BDL	BDL	140.3
TOTAL	1774.7	143.1	1688.9

Discussion

This study was inevitably limited because of the ethical and conservation constraints that prohibited the sacrifice of hatchling, juvenile or adult diamondback terrapins. However, the results obtained confirmed that the sediments of the pond environments inhabited by the terrapins of Bermuda were heavily contaminated by heavy metals and organic pollutants; this is consistent with the general picture for Bermudian wetlands (Fort et al., 2006; Fort et al., 2006; Bacon, 2010; Bacon et al., 2012). It is also evident that the small benthic gastropods that inhabit the golf course ponds, and which are the main food item of the terrapins (Chapter 4), are also contaminated, showing biomagnification of all heavy metals except iron, as well as TPH-DRO. Such biomagnification has been repeatedly reported for freshwater and marine gastropods (e.g. Walsh et al., 1995). Broadly speaking there was little evidence of general biomagnification of PAHs by the molluscs, but a wide range of PAHs were present in their tissues (indicating bioaccumulation) and, while many were at lower concentrations than in sediments, in some cases there was evidence of great biomagnification (e.g. for benzo(a)pyrene, a powerful dietary carcinogenic mutagen (Lee and Shim, 2007)).

Terrapin eggs showed lower levels of all metals than did the gastropods. However, concentrations of mercury, zinc, lead and cadmium were all above those of the sediments. In contrast, levels of TPH-DRO in gastropods and terrapin eggs were similar, while PAH levels tended to be lower in eggs than either gastropods or sediments. Overall, these results might suggest that terrapin eggs were relatively uncontaminated. However, comparisons with other studies indicate that this is far from true. Burger (2002) measured metal levels in terrapin eggs, liver and muscle in material collected from Tuckerton, New Jersey, a rural and coastal area distant from pollution sources. Her published data were all wet-weight specific. Ricklefs (1977) reported that water made up 68.9% of terrapin egg mass; Roosenburg and Dennis (2005) found values ranging from 66.5% to 73.5% (mean 70.9%). In Table 7.5 the results of the Bermudian and New Jersey studies are compared, assuming that 70% of wet egg mass is made up of water. It is evident from this table that the eggs of Bermudian diamondbacks are

heavily contaminated by heavy metals, with concentrations $\times 20$ (chromium) to $\times 42,000$ (cadmium) greater than at the relatively pristine site of Tuckerton. Burger (2002) reported that egg metal levels were generally equal to or lower than maternal tissue levels, so the egg data provide strong indications that adult diamondbacks on Bermuda have high levels of metal contamination too. Female slider turtles (*Trachemys scripta*) are reported to sequester some heavy metals in their eggs as an excretion method to rid their bodies of toxic chemicals (Burger and Gibbons, 1998) and the same may be true of diamondback terrapins. It is also worth noting that eggs of green sea turtles (*Chelonia mydas*), collected for public health analysis, contained far lower levels of arsenic (*ca.* 0.32 mg/kg dry mass), cadmium (0.03 mg/kg dry mass) and lead (0.10 mg/kg dry mass) [no other heavy metals were analysed] (van de Merwe et al., 2009) than Bermudian diamondback eggs.

The cadmium levels recorded in sediments, gastropods and terrapin eggs are particularly disturbing, as this extremely toxic, non-essential metal has been subject to world-wide emission control for decades and global environmental levels have declined for many years (e.g. van Assche and Ciarletta, 1992). Cadmium, which is an endocrine disrupter in fish (Vettillard and Bailhache, 2005), is carcinogenic and possibly mutagenic (Burger, 2008). The observed value of 37.74 mg/kg dry weight for terrapin eggs in Bermuda is about $\times 38$ the maximum permitted concentration in human foodstuffs (CODEX [Codex Alimentarius Commission]: Eisler, 1985).

Chromium (in trivalent and especially hexavalent form) is known to be mutagenic, teratogenic, carcinogenic and an embryotoxin. Tissue levels in excess of 4 mg/kg dry weight indicate significant chromium contamination in a wide range of vertebrates (Eisler, 1986). The observed mean value of 28.16 mg/kg dry weight for terrapin eggs in Bermuda indicates that this metal is also present at deleterious levels.

Anthropogenic arsenic mainly enters the environment in the form of pesticides. Background arsenic concentrations in living organisms are usually < 3 mg/kg dry weight (calculated from Eisler, 1988), so the observed mean concentration of 28.16 mg/kg in terrapin eggs in Bermudian is nearly $\times 10$ background. However these levels are lower than those often found in

seafood (Eisler, 1988) and the effects of such levels are unknown. Arsenic, which occurs in many forms, is generally rather high in marine organisms and often forms non-toxic complexes (Eisler, 1988).

Lead levels in pond sediments were below ERL and ERM concentrations and therefore not dissimilar to those of coastal estuarine sediments, but mercury levels were intermediate between ERL and ERM; both were bioaccumulated and biomagnified by the gastropods and diamondbacks. Mercury is teratogenic, mutagenic and carcinogenic, and is known to cause embryocidal, cytochemical and histopathological effects in wildlife (Eisler, 1987); however there are few toxicological studies of mercury in reptiles. The mean mercury level in the diamondback terrapin eggs from Bermuda (4.37 mg/kg) was significantly higher than that reported by Burger and Gibbons (1998) for the slider turtle (*Trachemys scripta*) from the Savannah River Site, South Carolina (0.04 mg/kg) as well as that reported by Burger (2002) for diamondback terrapins from New Jersey (0.12 mg/kg). The biological transformation of mercury to the highly toxic methylmercury form and its subsequent accumulation in food chains is a threat to many species, especially those inhabiting aquatic environments (see reviews in USEPA, 1997).

Copper compounds (e.g. copper sulfate) are widely used as biocides to control nuisance algae and macrophytes (Bartley, 1967; Havens, 1994) and can concentrate in soil, water, and sediments after prolonged periods of application. While some studies suggest little or no biomagnification of copper in freshwater food chains (Stokes, 1979), others have found that among marine organisms, the highest accumulations of copper are generally found in molluscan tissues rather than vertebrate tissues (Eisler, 1979, 1981). Diet appears to be the most important route of copper accumulation in aquatic animals; however data are scarce on copper concentrations in field populations of amphibians and reptiles. Crocodile eggs may contain 60 mg/kg dry weight and livers of some toads may contain as much as 2100 mg/kg dry weight without apparent adverse effects (see Eisler, 1998), therefore the amount of copper detected in Bermuda's diamondback terrapin eggs (90.2 mg/kg) may be within tolerable limits.

No previous measurements of TPH-DRO appear to have been conducted upon diamondback terrapin eggs and interpretation of the data is difficult because of the high lipid content (26-30% of dry weight; Ricklefs (1977), Roosenburg and Dennis (2005)) of the eggs (D. Fort, personal communication). Despite this limitation, it is clear that terrapins accumulate TPH-DRO and transfer them to their eggs. In addition, the sedimentary TPH levels far exceed those already known to cause high levels of malformations in amphibians (Fort and McLaughlin, 2003).

There are however directly comparable data for PAHs. Holliday et al. (2008) made measurements of PAH content of terrapin eggs collected from various shores around Swanson's Creek, Maryland one year after a serious spill of crude and fuel oil. Data for the cleanest and most contaminated sites are shown in comparison with Bermudian data in Table 7.6. From this table it is clear that eggs from the Bermudian diamondback terrapins feature similar PAH levels to those collected from the most contaminated Maryland sites and are around twelve times the levels of (relatively) uncontaminated eggs. This indicates that Bermudian diamondbacks live in a chronically PAH-polluted habitat.

Van Meter et al. (2006) reported that exposure to crude oil and PAHs (particularly benzo[a]pyrene and 7,12-dimethylbenz-[a]anthracene) had a detrimental effect on the survival and development of common snapping turtle (*Chelydra serpentina*) embryos. Low hatching success and high deformity rates were reported from eggs collected from the John Heinz National Wildlife Refuge in Pennsylvania (a contaminated wetland). Bermuda's diamondback terrapin population has been characterized as having a very low annual hatching success rate (19%) despite an absence of nest depredation (see Chapter 5). Furthermore, this population is composed of individuals affected by a moderate level of minor deformities (e.g. misshaped carapace or plastron, extra scutes, misshapen scutes and deformed digits) (see Chapter 3). Such deformities have been attributed to embryological exposure to high incubation temperatures (Wood and Herlands, 1997; Herlands et al., 2004) as well as petroleum crude oil and PAHs (Van Meter et al., 2006). One potential route of egg PAH exposure is via incubation in contaminated beach sands; another route of exposure is

maternal transfer of lipophilic hydrocarbons (Nagle et al., 2001). The sources of the PAHs found within Bermuda's wetlands are currently being investigated.

Overall it is evident that the Bermudian diamondback terrapins live and feed in wetland habitats characterised by chronic, multifactorial contamination that renders their main food source dangerous to consume. While their own tissue contaminant concentrations are unknown, it is evident that contaminants are transferred to eggs, and that the concentration of several of these exceed those known to cause damage. Some of the contaminants are known to have mutagenic and teratogenic effects at the observed concentrations and may reduce the incidence of successful embryonic development.

Concluding comments

This study suggests that environmental contamination may be contributing to the low hatching success shown by diamondback terrapin eggs on Bermuda. Ideally, attempts should be made to store any hatchling, juvenile and adult material that results from mortalities, with a view to determining tissue contaminant concentrations. Similarly, it would be desirable to extend studies to measurements of persistent organic pollutants (POPS) in both benthic gastropods and terrapin eggs. However, from a conservation perspective these data already indicate that a programme of wetland remediation is urgently needed. Much of the observed contamination may be historical, but all efforts to reduce present and future contamination of the study ponds should be made.

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Chapter 8: Conclusions and Recommendations for Future Research

This investigation has shown that Bermuda's diamondback terrapins have a small population, a very limited distribution, and low annual rates of recruitment. Additionally, the annual production of hatchlings is limited, largely due to low rates of hatching success. All of these factors suggest that this population is at high risk of local extirpation. Delayed sexual maturity, longevity and iteroparity are key characteristics of the population biology of chelonians (Gibbs and Amato, 2000), but these very characteristics also likely make their populations less capable of responding to elevated rates of mortality (Congdon et al., 1993).

It is presently not known how stable the population in Bermuda is, since there are no other population estimates with which to compare the current findings. However, the data collected during the four years of field work (2008-2011) in this doctoral study will serve as an effective base-line for future investigations. Long-term monitoring of this vulnerable population, to determine temporal trends, is highly recommended and continued use of modified fish traps, rather than modified crab traps, is suggested for future studies – given that they were ten times more effective in catching diamondback terrapins within Bermuda's pond environment.

It is also strongly recommended that the monitoring of hatching success be continued. Additional research is needed to determine the specific causes of the low hatching rates observed in the Bermuda terrapin population, as well as ways to mitigate them. Egg viability should be investigated within the Bermuda population. Examining oviposited terrapin eggs for the presence of an embryonic disc will help to establish whether fertility (in either sex) is limiting the hatching rate, and a comprehensive genetic assessment of the population may help to determine if inbreeding is an issue. Future studies should also examine how temperature control affects hatching. This can be achieved via artificial egg incubation and use of sand temperature loggers.

The artificial incubation of terrapin eggs collected from sand bunkers on the Mid Ocean golf course, combined with a head-starting programme, might also prove to be a short-term means of numerically

enhancing Bermuda's diamondback terrapin population as well as increasing the relative proportion of males within the population. Head-starting has been used on a number of marine turtle species (see Meylan and Ehrenfeld, 2000) as well as diamondback terrapins (Wood and Herlands, 1997; Herlands et al., 2004) in an effort to replenish dwindling populations. However, this activity (and the subsequent reintroduction, repatriation or translocation of captive raised individuals) as a conservation tool is contentious (see Burke, 1991; Dodd and Seigel, 1991; McDougal, 2000; Meylan and Ehrenfeld, 2000) and should only be used in tandem with other conservation strategies, which target both species and habitat. If considered as an appropriate conservation tool, head-starting diamondback terrapins in Bermuda should be done to emulate, as much as possible, natural conditions (e.g. microhabitats, diet, seasonal temperatures and UV light levels) to minimize the risk of raising animals with anomalous behaviours (e.g. seeking and using inappropriate microhabitats, foraging poorly and lacking predator-avoidance behaviour), which would limit their ability to survive after release. Furthermore, adequate space, sanitation and veterinary care must be made available to minimize both the mortality rates of captive animals and the release of compromised individuals (i.e. diseased) into the wild. Juvenile terrapins raised in this manner would also be ideal candidates for translocation to new wetlands in Bermuda that are deemed suitable for terrapin survival (see below).

The artificial incubation of Bermudian diamondback terrapin eggs, and the subsequent release of hatchlings, has additional value as a public relations activity, by creating an educational opportunity to involve the public and raise awareness regarding the conservation and preservation of diamondback terrapins on Bermuda. This emydid turtle possesses the appearance and appeal to captivate the attention of individuals of all ages, and has proven in the U.S.A. to be an excellent motivator to teach environmental ethos and stewardship (D. Lewis personal communication).

Habitat loss and degradation are regarded as primary causes of population declines for many turtle species globally (see review in Mitchell and Klemens, 2000) and golf courses represent highly modified, human-dominated landscapes. The creation of the Mid Ocean golf course during the

1920s undoubtedly altered the terrapins' wetland habitats, however the effects that this had upon the terrapin population at that time are largely speculative. Activities associated with maintaining the golf course have also impacted upon the terrapin population. Decades of applying synthetic chemicals have created toxic conditions on Bermuda (Fort et al., 2006). Fire was frequently employed to eliminate undesirable vegetation in the marsh at South Pond for decades, before being abandoned in favour of more environmentally sensitive practices (N. Furtado, personal communication). This would have inevitably caused substantial mortality to resident young terrapins and would also have temporarily diminished the ability of the marsh to support new cohorts of terrapins. The current paucity of vegetation surrounding South Pond and the sand bunkers where nesting occurs most frequently increases exposure of hatchling terrapins to avian predators. Allowing native vegetation to grow around the edges of the pond and planting suitable vegetation around the sand bunkers that link them to neighbouring wetlands might help to make hatchlings less vulnerable to avian predators, particularly when they move towards the marshes after nest emergence. Any such modification to the vegetation on the golf course would require a balance between the survival needs of the terrapins with the needs of the Mid Ocean club, in order to be compatible with the aesthetic appearance of the golf course.

The protection of critical, or core, habitats is highly recommended. Core habitats can be defined as essential environments that are required to carry out critical life-history functions for a species (Semlitsch and Bodie, 2003). For hatchling and small juvenile diamondback terrapins in Bermuda, core habitats include the mangrove swamp and saw-grass marsh communities adjacent to Mangrove Lake, Trott's Pond, South Pond and North Pond. The swamps surrounding Mangrove Lake and Trott's Pond are presently afforded high levels of protection via their designation as nature reserves. However, the grass-dominated marshes at South Pond and North Pond are currently unprotected habitats. The results of this investigation support the legislative protection of these wetlands for Bermuda's diamondback terrapins. Furthermore, an island-wide assessment of all potential habitats suitable for diamondback terrapin growth, reproduction

and survival needs to be made and candidate habitats must be protected under Bermudian legislation before any potential terrapin translocations are carried out. Any area identified should have appropriate habitats available for all life stages of diamondback terrapins and have enough space to allow for a population to become sufficiently large, so that demographic stochasticity and environmental fluctuations do not lead to extirpation.

Diamondback terrapin populations have been shown to decrease when females are forced to nest in marginal habitats where nest survivorship is low (Roosenburg, 1992). Furthermore, females need a wide range of nesting micro-habitats (e.g. variation in elevation, orientation to the sun and amount of shading provided by surrounding vegetation) to maintain balanced sex ratios within a population (Roosenburg and Place, 1994). Given that over 97% of the nests located in 2010 and 2011 occurred within the sand bunkers on the Mid Ocean golf course, and that these areas are greatly affected by anthropogenic activities, it would be prudent to increase and diversify the nesting habitat in the Mangrove Lake, Trott's Pond, South Pond and North Pond area. The creation of new nesting habitat should ideally occur at locations that will minimise the disturbance to nesting females, incubating eggs and over-wintering hatchlings. Any such area should also be relatively large, well-drained, and be easily accessible to female terrapins. It should also incorporate varying levels of vegetational shading, to promote thermal variability. Examples of such habitat could include the creation of beaches along a 60 m length of un-vegetated shoreline on the fifth fairway adjacent to Mangrove Lake (region between Q5 and Q13 in Figure 4.2, Chapter 4) as well as along a 30 m length of un-vegetated shoreline on a private property adjacent to Mangrove Lake (region between Q10 and Q12 in Figure 4.2, Chapter 4). Substrate for the creation of these beaches can be taken from a neighbouring natural beach (Sam Hall's beach).

The observed level of avian predation on diamondback terrapin hatchlings during the first month following spring emergence in Bermuda may be partially responsible for the low levels of recruitment that have been documented in the adult population (approximately two terrapins per annum). Furthermore, yellow-crowned night herons appear to remain

predators of small juvenile terrapins for three years following hatching. This is of particular concern given that the annual production of hatchlings in Bermuda was estimated to be no more than 91 individuals. High post-hatching survival rates are necessary to ensure that adequate recruitment occurs to maintain a stable population. The initiation of a limited cull of yellow-crowned night herons, particularly South Pond, should be considered in the spring, when hatchlings appear to be more conspicuous. While it is presently unknown whether feral cats also pose a predatory threat to young diamondback terrapins in Bermuda, it would be prudent to relocate the cat feeding shelter from South Pond to another area where terrapin hatchlings are absent. Sustained control of rats from the terrapin wetlands area is also highly recommended, given that they have been identified as a significant threat to hatchlings in the U.S.A. (Draud et al., 2004).

The faecal analyses and, to a more limited extent, the necropsies have shown that diamondback terrapins in Bermuda are dietary generalists that appear to favour the consumption of small gastropods (primarily *Heleobops bermudensis* and *Melanoides tuberculata*). The range of food items is narrower than those reported from North America, however this may be due to the fact that there is less diversity among prey species present within the pond habitat in Bermuda in comparison with those found within the salt marshes of the U.S.A. For example crabs, which are a relatively abundant element of the salt marsh environment and an important food item for terrapins in North America, are cryptic and rare in Mangrove Lake and Trott's Pond (Thomas and Logan, 1992), and absent from South Pond and North Pond (M. Outerbridge, personal observation). The quadrat survey results in the mangrove and saw-grass marshes derived from the present study indicate that these environments do not appear to be food limited, especially for neonate and small juvenile terrapins. In contrast, the benthic surveys within the ponds show that gastropod abundance is unevenly distributed and generally low within the sediment, but is higher in localized areas where rocky substrate or widgeon grass dominate. It is currently unclear why Bermuda's diamondback terrapins are not exploiting the aquatic gastropods *Melampus coffeus* and *Batillaria minima*, since the benthic habitat surveys showed that both species were more abundant than

H. bermudensis and *M. tuberculata*. It is feasible that robust shell architecture may be providing protection from terrapin predation; however a crushing force investigation is needed to confirm this. It is equally unclear why Bermuda's terrapins do not appear to be exploiting the flat mangrove oyster as a food resource since it is among the most abundant, visible and sedentary of all the molluscs inhabiting Mangrove Lake and Trott's Pond. The caloric content of the flat mangrove oyster was determined to be 5.23 cal mg⁻¹ (Thomas and Dangeubun, 1994). The energetic values for the most commonly ingested food items (e.g. *H. bermudensis* and *M. tuberculata*) should be calculated to ascertain if enough high quality prey are being consumed. Future feeding ecology studies should also focus on examining additional faecal material, especially in the neonate and small juvenile size class (i.e. 30-90 mm SCL range) to determine the extent to which the small gastropods within the marsh wetlands are being consumed.

The thiarid snail *M. tuberculata*, the second most commonly consumed snail in the Bermuda terrapin population, has never been reported as prey in previous studies and thus represents a novel food item for this emydid turtle. *Melanoides tuberculata* has been identified as a host for several species of parasitic trematode worms (Pinto and de Melo, 2011) which are known to affect the health of waterbirds, fishes and mammals (including humans) (Penner and Bernard, 1963; Mitchell et al., 2007). It is currently unknown whether *M. tuberculata* is host to any parasites in Bermuda, but it would be prudent to determine if they could compromise the health of the terrapins that consume them.

The non-selective, deposit-feeding strategy observed in the Bermuda population of terrapins is evidently an adaptation that has allowed them to take advantage of the small benthic gastropods inhabiting the gelatinous pond sediment. This behaviour has not been previously reported; however, it is exposing them to the heavy metals, gasoline-range and diesel-range petroleum hydrocarbons and polycyclic aromatic hydrocarbons that exist within this medium. Furthermore, the high incidence of aquatic gastropod consumption observed within Bermuda's juvenile and adult terrapins is of concern as this is providing additional contaminant exposure.

Tissue residue analyses for Bermudian specimens of adult red-eared sliders (*Trachemys scripta elegans*) have revealed significant levels of diesel range organics, polycyclic aromatic hydrocarbons and metals. Additionally, liver and gonad abnormalities have been documented from a number of different locations throughout Bermuda which are associated with high levels of contaminants (Fort et al., 2006; Fort et al., 2006; J. Bacon, personal communication). Red-eared sliders are the subject of an eradication programme in Bermuda and make an ideal study proxy for diamondbacks because they inhabit nearby wetland environments and are readily available. The short-term and long-term effects that exposure to these contaminants may have upon the Bermuda diamondback terrapin population are unknown, but the results presented in Chapter 7 strongly suggest that environmental contamination is a probable contributor to the low hatching success shown by diamondback terrapin eggs on Bermuda. Ideally, any dead diamondback terrapins should be stored frozen, with a view to determining tissue contaminant concentrations, but future studies should focus on the necropsy and histological examination of tissue material obtained from red-eared sliders. Similarly, it would be desirable to extend studies to measurements of persistent organic pollutants (POPS) in both benthic gastropods and terrapin eggs. However, from a conservation perspective the data already available indicate that a programme of wetland remediation is urgently needed. Much of the observed contamination may be historical, but all efforts to reduce present and future contamination of the study ponds should be made.

Examples of wetland remediation include phytoremediation, in which plants are used to extract persistent contaminants from surrounding substrate, as well as employing various chemical and biological remediation techniques. Chemical remediation methods include reducing or eliminating inputs of contaminants from point sources, natural sediment remediation by biodegradation and chemical degradation, and active sediment remediation by removal or by in situ treatment; biological remediation methods include enhancing populations of target organisms (see reviews in Wilcox and Whillans, 1999). Some wetland plants have been shown to sequester petroleum hydrocarbons (Lin and Mendelssohn, 1998), PAHs (Lin and

Mendelssohn, 2009) and metals (Weis and Weis, 2004) from wetland sediment and store them below ground in roots or concentrate them in aerial tissues (e.g. leaves and stems). Introduction of oxygenated air into contaminated areas promotes natural biological degradation of contaminants by increasing the activity of indigenous bacteria that are capable of metabolizing pollutants (D. Fort, personal communication). Depositing clean sediment (e.g. diatomaceous earth) over contaminated sediment is yet another technique of wetland remediation that can diminish the risk of biological contact, however it should not be considered without first assessing its impact on the water column and aquatic biota of the ponds. Additionally, the creation of buffer zones between road drains and some of the ponds inhabited by the diamondback terrapins (e.g. Mangrove Lake and Trott's Pond) would help to reduce direct in-put of pollutants by serving as a filter for contaminants entering as road runoff. Presently, all road drains adjacent to Mangrove Lake and Trott's Pond channel storm water runoff directly into the ponds.

Finally, as a direct result of this doctoral investigation, Bermuda's diamondback terrapins were classified in 2012 as a level II protected species under the Bermuda Protected Species Act (2003) and declared to be 'Vulnerable'. A management and recovery plan (see Appendix 8) detailing the short-term and long-term survival goals for this species has been drafted and is currently under review for implementation by the Bermuda Government's Department of Conservation Services.

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Figure A1.1. Aerial photograph from 1940 showing Mangrove Lake (A), Trott's Pond (B), South Pond and marshland (C), North Pond and marshland (D), Compston Pond and marshland (E), Angel's Grotto Ponds (F and G), Somersall Marsh (H), and Somersall Pond (I). © Bermuda Ministry of Environment, Planning and Infrastructure.



Figure A1.2. Aerial photograph from 2005 showing Mangrove Lake (A), Trott's Pond (B), South Pond and marshland (C), North Pond (D), Compston Pond (E), fields where the Angel's Grotto Ponds were formerly located (F and G), Somersall Marsh (H), and the field where Somersall Pond was formerly located (I). © Bermuda Ministry of Environment, Planning and Infrastructure.

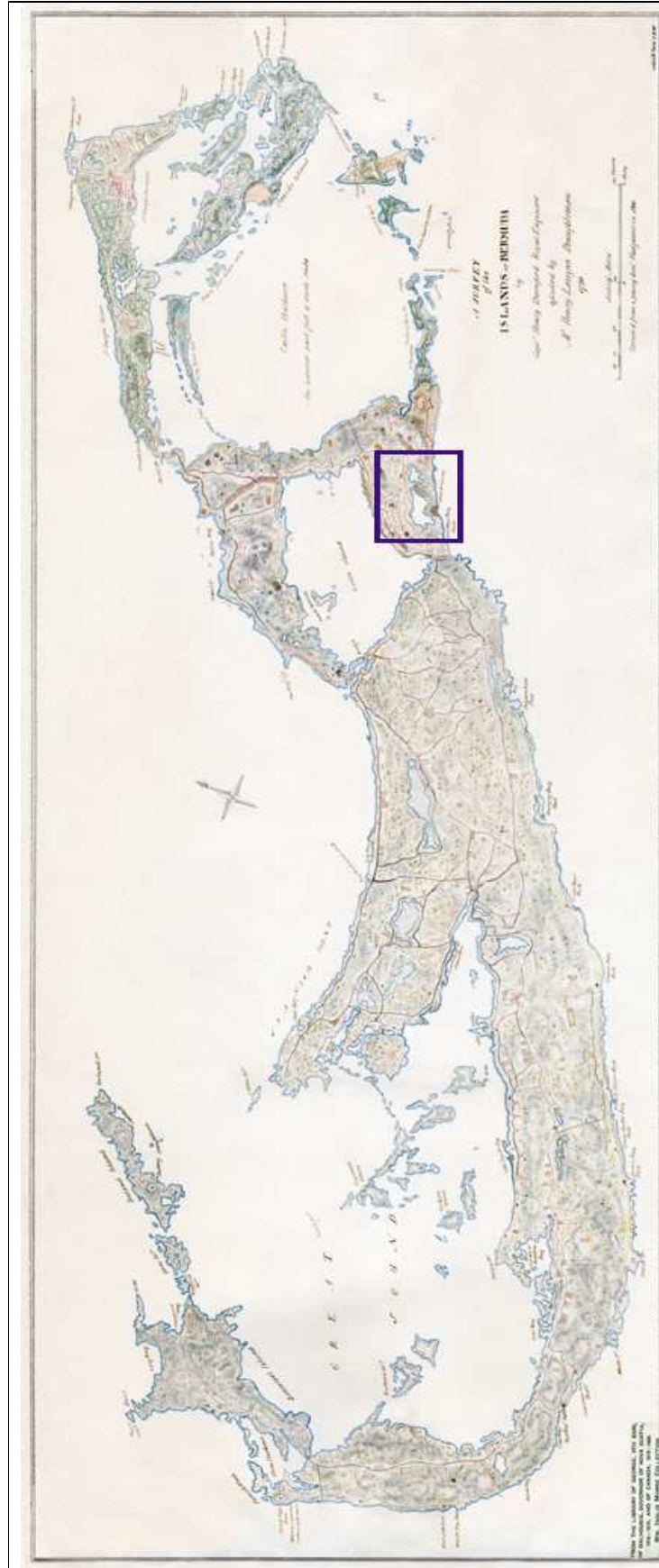


Figure A1.4. Captain Henry Durnford 1793 map of the Bermuda Islands showing Mangrove Lake, Trott's Pond, South Pond and North Pond (area bounded by purple box) © Bermuda Maritime Museum.

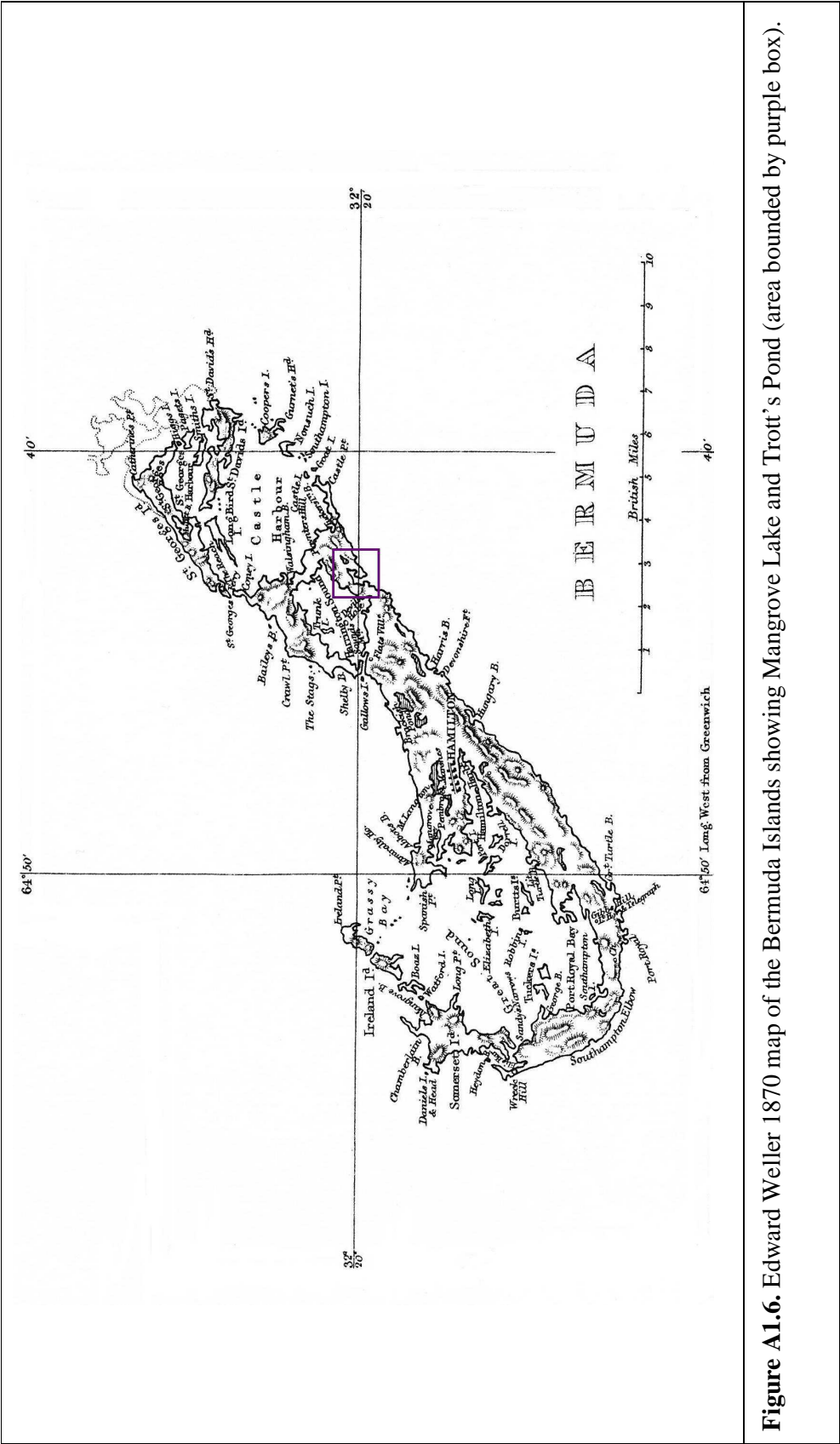
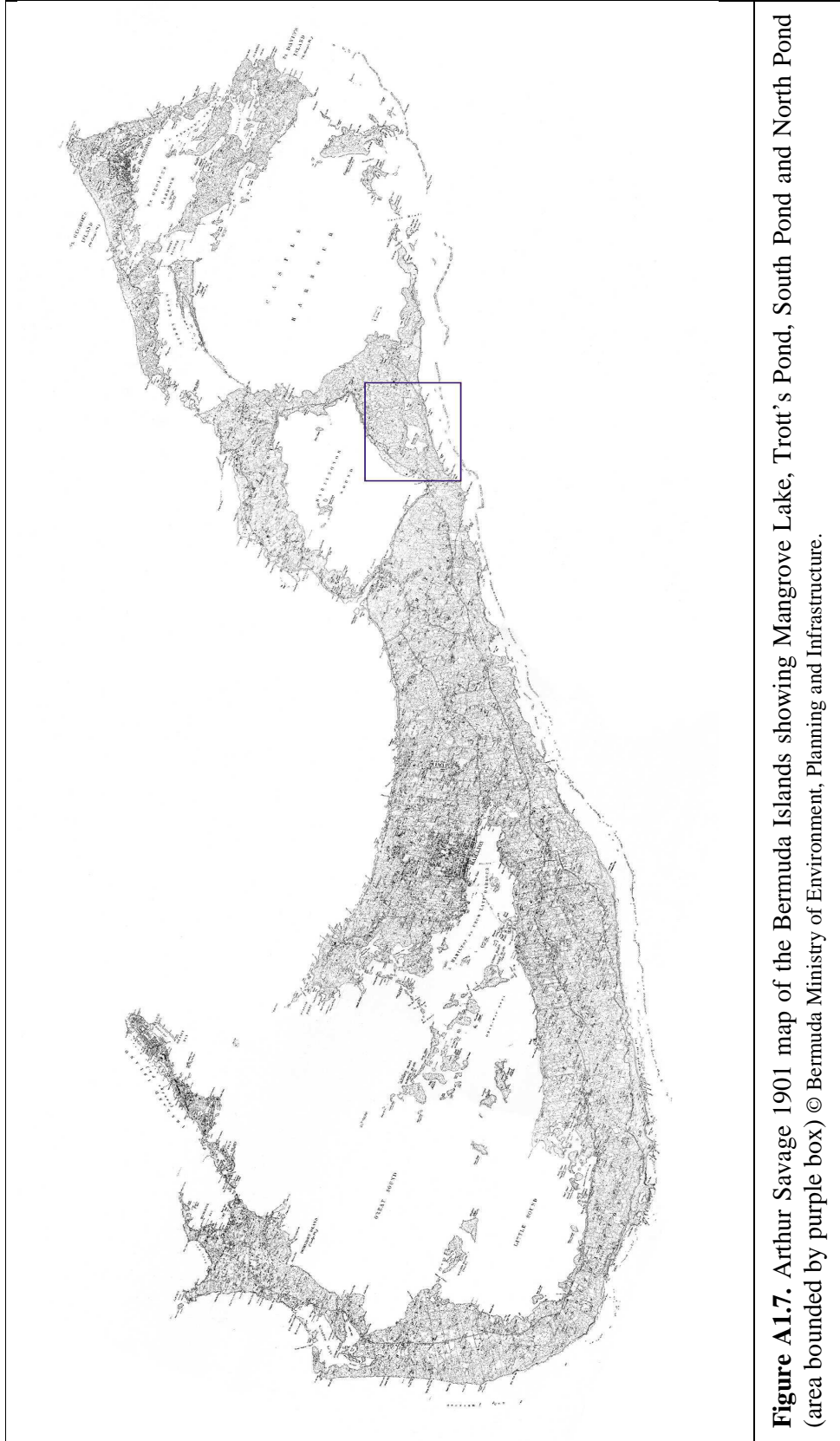


Figure A1.6. Edward Weller 1870 map of the Bermuda Islands showing Mangrove Lake and Trott's Pond (area bounded by purple box).



Appendix 2: Supplementary Material to Chapter 2



Figure A2.1. Panoramic vista of Mangrove Lake



Figure A2.2. Panoramic vista of Trott's Pond

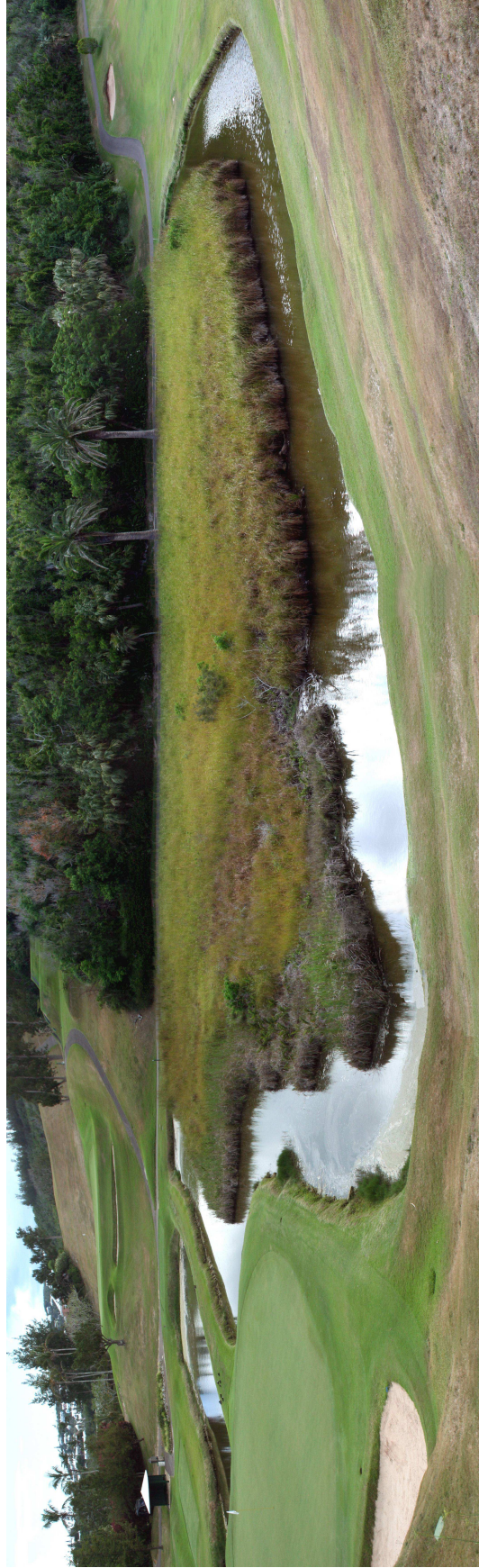
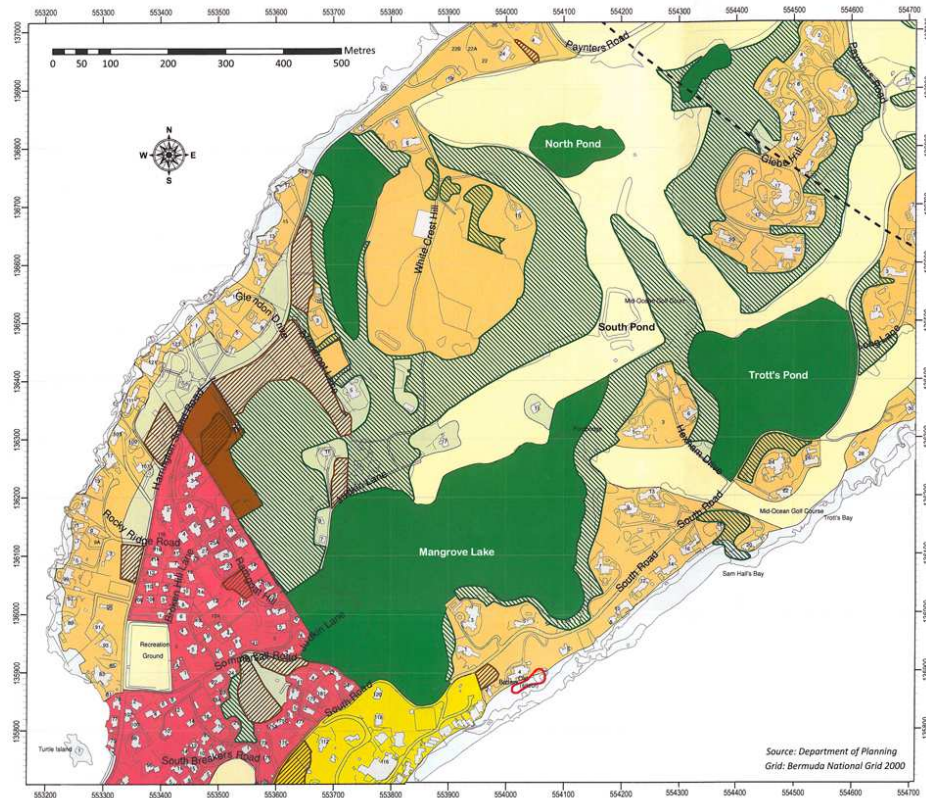


Figure A2.3. Panoramic vista of South Pond



Figure A2.4. Panoramic vista of North Pond



LEGEND

Base Zones

Residential 1	Airport
Residential 2	Open Space Reserve
Rural	Coastal Reserve
Commercial	Recreation
Institutional	Park
Industrial	Nature Reserve
Tourism	Special Study Area
Mixed Use	City Plan 2001

Protection Areas

Agricultural Reserve	Woodland Reserve
Historic Protection Area	Cave Protection Area
Known Cave Locations	

Figure A2.5. Development base zones for the area inhabited by Bermuda's population of diamondback terrapins. (Adapted from the 2008 Bermuda Plan).

Appendix 3: Supplementary Material to Chapter 3

Diamondback Terrapin Field Survey Form

Date: _____ Time: _____ Location: _____
Recorders: _____
Observation type: First / Recapture
Notch # _____
Female Male Juvenile Hatchling
S-Carapace length (mm): _____
S-Carapace width (mm): _____
S-Plastron length (mm): _____
Shell height (mm): _____
Mass (g): _____
Catch method: Modified crab trap
Modified fish trap
Dip net
Genetic Sample: Blood ☐
Tissue ☐
None ☐

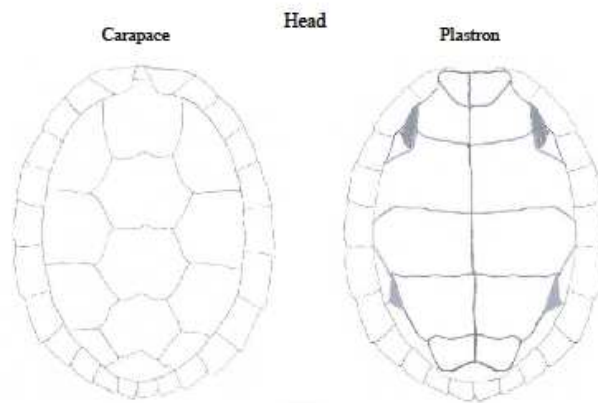


Photo # _____

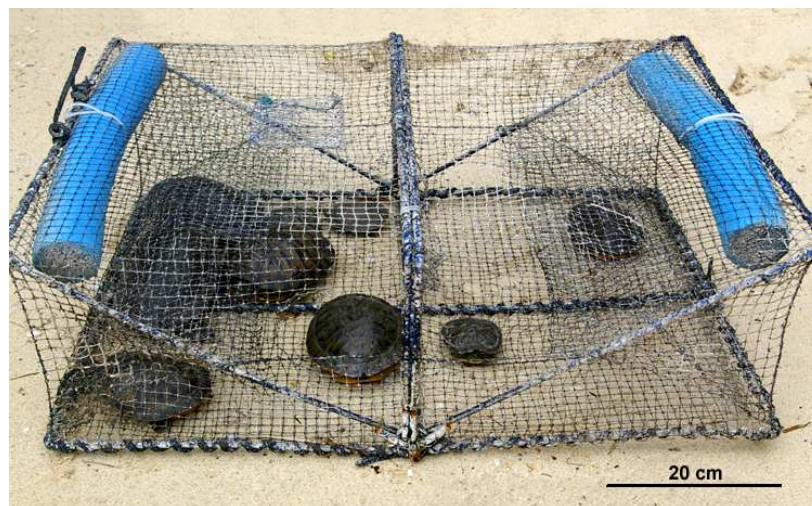
Notes:

Figure A3.1. Data sheet used during the 2008-2010 population surveys.



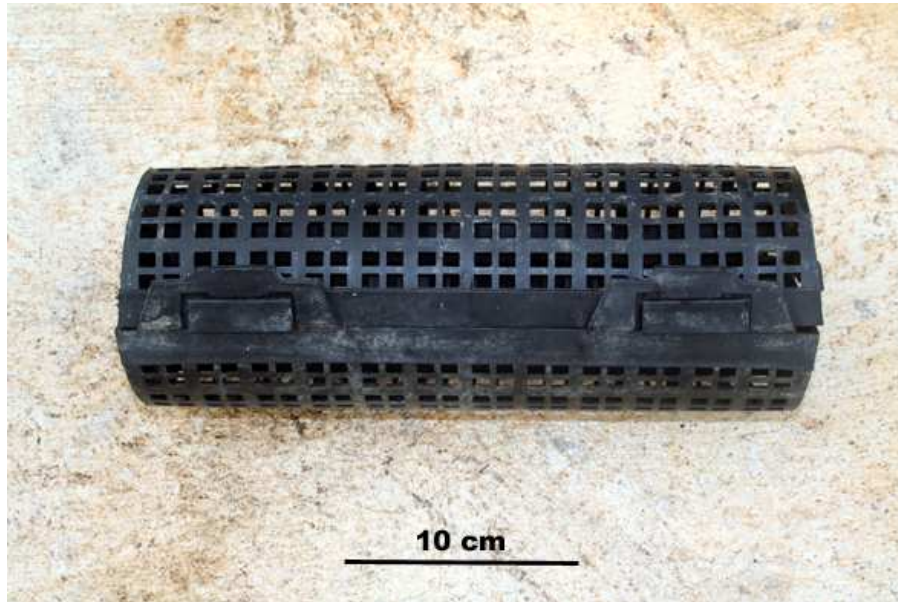
Source: Mark Outerbridge

Figure A3.2. Modified crab trap used to capture diamondback terrapins in Bermuda.



Source: Mark Outerbridge

Figure A3.3. Modified fish trap used to capture diamondback terrapins in Bermuda. (Note that the trap actually contains specimens of *Trachemys scripta elegans*).



Source: Mark Outerbridge

Figure A3.4. Bait box used to secure fish bait within the terrapin traps during the population surveys.



Source: Mark Outerbridge

Figure A3.5. Modified fish trap deployed among the prop roots of *Rhizophora mangle* in Mangrove Lake.



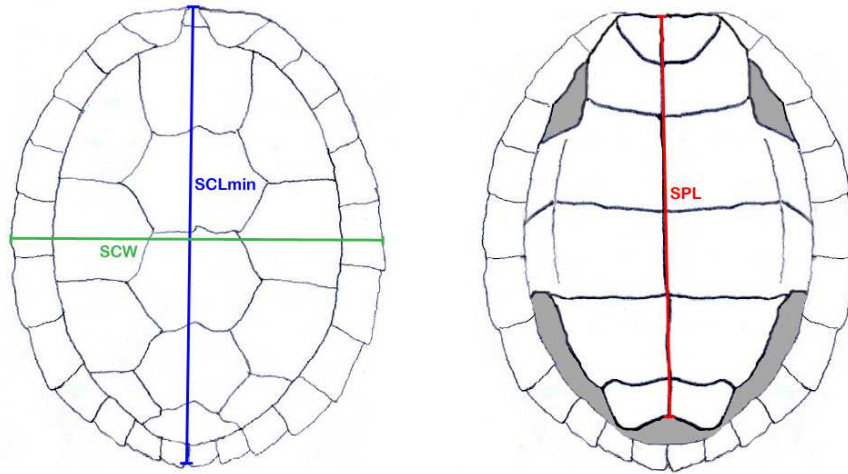
Source: Mark Outerbridge

Figure A3.6. Modified crab trap deployed in Trott's Pond.



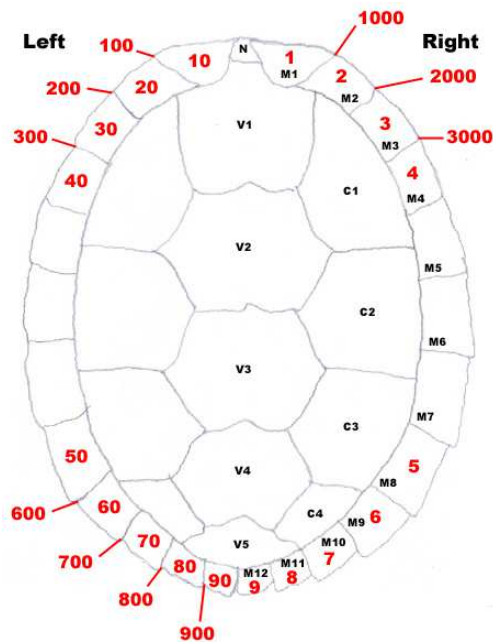
Source: Mark Outerbridge

Figure A3.7. Modified fish trap deployed in South Pond. (Note the diamondback terrapin to the left and the black bait-box to the right of the blue frame at centre).



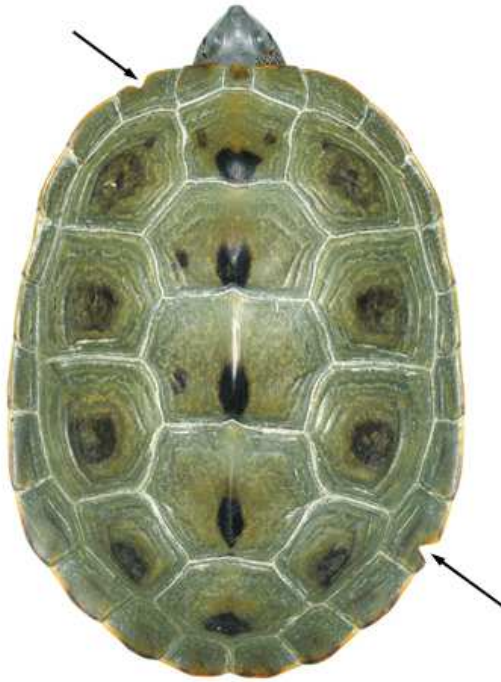
Source: Mark Outerbridge

Figure A3.8. The straight-line shell measurements for the carapace (left) and plastron (right). SCL = minimum straight carapace length, SCW = straight carapace width, SPL = straight plastron length. Shell height is not shown.



Source: Mark Outerbridge

Figure A3.9. Illustration showing the numerical values assigned to the marginal scutes used during the mark-recapture population surveys (N = nuchal scute, M = marginal scute, V = vertebral scute, C = costal scute).



Source: Mark Outerbridge

Figure A3.10. Photograph showing the notches incised into the marginal scutes of a mature female terrapin. In this example the second scute on the left and the ninth scute on the right of the carapace have been notched, indicating that this individual was assigned the number 26.

Table A3.1. Summary of diamondback terrapin movement between the brackish water ponds using recapture histories.

Notch #	Location	Date	Observation
7	South Pond	June 11 2008	first
7	Mangrove Lake	Sept 14 2009	recapture
8	South Pond	June 11 2008	first
8	Mangrove Lake	July 9 2008	recapture
8	South Pond	July 9 2009	recapture
9	South Pond	June 11 2008	first
9	Mangrove Lake	July 9 2009	recapture
19	South Pond	June 12 2008	first
19	Mangrove Lake	July 9 2008	recapture
33	South Pond	June 19 2008	first
33	Mangrove Lake	August 27 2008	recapture
38	Mangrove Lake	June 20 2008	first
38	South Pond	July 14 2008	recapture
42	South Pond	June 20 2008	first
42	Mangrove Lake	June 11 2009	recapture
45	South Pond	June 20 2008	first
45	Mangrove Lake	July 11 2008	recapture
45	South Pond	July 16 2009	recapture
50	South Pond	July 8 2008	first
50	Mangrove Lake	June 17 2009	recapture
69	Mangrove Lake	August 21 2008	first
69	North Pond	May 25 2010	recapture
76	South Pond	August 29 2008	first
76	Mangrove Lake	Sept 16 2009	recapture
82	South Pond	June 10 2009	first
82	Mangrove Lake	Sept 26 2010	recapture

Table A3.2. Data used to calculate the Schnabel population estimates for 2008, 2009, and 2010.

Samples 1-32 = 2008 (year 1); Samples 33-56 = 2009 (year 2);

Samples 57-74 = 2010 (year 3)

Where:

$i = i^{\text{th}}$ sample

n_i = number of animals in the i^{th} sample

m_i = number of animals in the i^{th} sample that are carrying marks

u_i = number of unmarked animals in the i^{th} sample ($n_i - m_i$)

M_i = number of animals marked prior to the i^{th} sample

Table A3.2. (continued). Data used to calculate the Schnabel population estimates for 2008, 2009 and 2010.

$$\text{Total Population } (N) = \frac{\sum n_i M_i^2}{\sum m_i M_i}$$

<i>i</i>	<i>n_i</i>	<i>m_i</i>	<i>u_i</i>	<i>M_i</i>	<i>n_iM_i²</i>	<i>m_iM_i</i>	<i>m_i²/n_i</i>	<i>Nth</i>
1	13	0	13	0	0	0	0	0
2	15	4	11	13	2535	52	1.1	48.8
3	11	6	5	24	6336	144	3.3	45.3
4	1	0	1	29	841	0	0	49.6
5	5	3	2	30	4500	90	1.8	49.7
6	11	7	4	32	11264	224	4.5	50.0
7	13	1	12	36	16848	36	0.1	77.5
8	6	2	4	48	13824	96	0.7	87.5
9	4	3	1	52	10816	156	2.3	83.9
10	3	2	1	53	8427	106	1.3	83.4
11	2	1	1	54	5832	54	0.5	84.8
12	5	2	3	55	15125	110	0.8	90.2
13	12	10	2	58	40368	580	8.3	83.0
14	6	5	1	60	21600	300	4.2	81.3
15	10	9	1	61	37210	549	8.1	78.3
16	1	1	0	62	3844	62	1	77.9
17	7	5	2	62	26908	310	3.6	78.9
18	6	3	3	64	24576	192	1.5	82.0
19	7	4	3	67	31423	268	2.3	84.8
20	2	2	0	70	9800	140	2	84.2
21	3	1	2	70	14700	70	0.3	86.7
22	4	1	3	72	20736	72	0.3	90.7
23	2	2	0	75	11250	150	2	90.1
24	4	3	1	75	22500	225	2.3	90.6
25	1	0	1	76	5776	0	0	92.1
26	2	2	0	77	11858	154	2	91.5

Table A3.2. (continued). Data used to calculate the Schnabel population estimates for 2008, 2009 and 2010.

i	n_i	m_i	u_i	M_i	$n_i M_i^2$	$m_i M_i$	m_i^2/n_i	N^{ith}
27	1	1	0	77	5929	77	1	91.3
28	3	3	0	77	17787	231	3	90.5
29	3	3	0	77	17787	231	3	89.8
30	5	5	0	77	29645	385	5	88.9
31	3	0	3	77	17787	0	0	92.4
32	6	4	2	80	38400	320	2.7	94.0
33	9	7	2	82	60516	574	5.4	95.1
34	6	6	0	84	42336	504	6	94.3
35	2	2	0	84	14112	168	2	94.0
36	4	4	0	84	28224	336	4	93.5
37	3	2	1	84	21168	168	1.3	94.3
38	9	8	1	85	65025	680	7.1	94.4
39	8	7	1	86	59168	602	6.1	94.7
40	3	2	1	87	22707	174	1.3	95.4
14	6	6	0	88	46464	528	6	95.0
42	6	5	1	88	46464	440	4.2	95.5
43	5	4	1	89	39605	356	3.2	96.0
44	5	4	1	90	40500	360	3.2	96.6
45	3	2	1	91	24843	182	1.3	97.3
46	7	7	0	92	59248	644	7	97.0
47	2	2	0	92	16928	184	2	96.9
48	7	6	1	92	59248	552	5.1	97.4
49	2	2	0	93	17298	186	2	97.3
50	2	2	0	93	17298	186	2	97.3
51	3	3	0	93	25947	279	3	97.2
52	1	1	0	93	8649	93	1	97.1
53	3	3	0	93	25947	279	3	97.0
54	6	5	1	93	51894	465	4.2	97.6
55	5	5	0	94	44180	470	5	97.4

Table A3.2. (continued). Data used to calculate the Schnabel population estimates for 2008, 2009 and 2010.

i	n_i	m_i	u_i	M_i	$n_i M_i^2$	$m_i M_i$	m_i^2/n_i	N^{ith}
56	1	0	1	94	8836	0	0	98.1
57	2	0	2	95	18050	0	0	99.4
58	1	1	0	97	9409	97	1	99.4
59	1	1	0	97	9409	97	1	99.3
60	2	2	0	97	18818	194	2	99.3
61	1	1	0	97	9409	97	1	99.3
62	1	1	0	97	9409	97	1	99.3
63	3	3	0	97	28227	291	3	99.2
64	3	3	0	97	28227	291	3	99.2
65	1	0	1	97	9409	0	0	99.8
66	2	2	0	98	19208	196	2	99.8
67	1	1	0	98	9604	98	1	99.8
68	1	1	0	98	9604	98	1	99.8
69	1	1	0	98	9604	98	1	99.8
70	2	2	0	98	19208	196	2	99.7
71	1	1	0	98	9604	98	1	99.7
72	3	3	0	98	28812	294	3	99.7
73	1	1	0	98	9604	98	1	99.7
74	1	0	1	98	9604	0	0	100.3

Table A3.2. (continued). Data used to calculate the Schnabel population estimates for 2008, 2009 and 2010.

		$n_i M_i^2$	$m_i M_i$	m_i^2/n_i
2008:	Σ	506232	5384	68.7
2009:	Σ	1352837	13794	154.2
2010:	Σ	1618056	16134	178.2

$$(N^{2008}) = \frac{\Sigma n_i M_i^2}{\Sigma m_i M_i} = \frac{506232}{5384} = \mathbf{94 \text{ individuals}}$$

$$(N^{2009}) = \frac{\Sigma n_i M_i^2}{\Sigma m_i M_i} = \frac{1352837}{13794} = \mathbf{98.1 \text{ individuals}}$$

$$(N^{2010}) = \frac{\Sigma n_i M_i^2}{\Sigma m_i M_i} = \frac{1618056}{16134} = \mathbf{100.3 \text{ individuals}}$$

Table A3.3. Biometric data summary for female diamondback terrapins (n=64) captured in South Pond, Mangrove Lake, and Trott's Pond between 2008 and 2010. All data distributions were non-normal (Anderson-Darling tests; SCL $p=0.016$, SCW $p=0.002$, SPL $p=0.003$, SH $p=0.034$, Mass $p=0.007$).

Observation	Sex/ stage	No.	SCL (mm)	SCW (mm)	SPL (mm)	SH (mm)	Mass (g)
first	female	1	134	108	113	54	400
first	female	3	152	114	135	63	565
first	female	4	163	129	147	66	560
first	female	5	146	117	124	58	580
first	female	6	182	144	159	80	1020
first	female	7	133	101	111	52	375
first	female	8	179	142	152	72	900
first	female	9	155	123	133	63	525
first	female	11	165	129	147	66	700
first	female	12	140	107	120	59	495
first	female	13	154	120	135	61	590
first	female	15	169	135	150	66	875
first	female	17	182	139	155	75	1060
first	female	18	166	128	146	71	880
first	female	19	157	130	134	63	750
first	female	20	180	136	155	73	960
first	female	21	128	101	113	55	400
first	female	22	150	123	139	66	760
first	female	23	150	120	135	66	625
first	female	24	125	98	108	54	365
first	female	25	183	137	160	74	1060
first	female	26	130	99	115	53	370
first	female	27	171	135	154	68	900
first	female	28	145	111	123	60	540
first	female	30	179	137	151	67	800
'first'	female	33	163	131	144	68	800
first	female	36	159	125	130	63	700
first	female	37	172	136	155	75	1020
first	female	38	166	133	146	68	880
first	female	40	144	111	128	60	460
first	female	41	133	105	114	54	380
first	female	42	123	98	105	52	325
first	female	43	188	141	157	73	1060
first	female	44	165	130	141	67	790
first	female	45	157	120	134	66	680
first	female	46	135	107	123	55	395
first	female	47	129	104	109	54	350
first	female	48	137	111	120	49	405
first	female	50	127	101	110	52	355
first	female	51	141	121	117	61	565
first	female	53	185	142	161	71	1000
first	female	54	139	107	120	52	435
first	female	55	119	94	103	51	310
first	female	57	135	107	113	56	485

Table A3.3. (continued). Biometric data summary for 64 female diamondback terrapins captured in South Pond, Mangrove Lake, and Trott's Pond between 2008 and 2010.

Observation	Sex/ stage	No.	SCL (mm)	SCW (mm)	SPL (mm)	SH (mm)	Mass (g)
first	female	58	134	107	110	59	540
first	female	59	191	144	163	73	1220
first	female	60	174	134	146	69	960
first	female	62	142	107	121	57	510
first	female	63	165	126	144	66	720
first	female	64	116	97	99	48	270
first	female	65	161	123	137	66	820
first	female	67	125	97	109	51	300
first	female	68	195	147	166	72	1300
first	female	69	194	148	169	79	1340
first	female	70	183	142	159	68	1080
first	female	71	187	141	159	71	1020
first	female	72	182	139	155	68	970
first	female	73	183	139	151	69	1080
first	female	76	148	118	126	57	560
first	female	80	179	136	152	65	1000
first	female	82	186	142	155	75	1050
first	female	84	173	132	145	64	730
first	female	95	187	141	155	69	1120
first	female	96	196	150	165	73	1050
		<i>Median</i>	<i>160.0</i>	<i>125.0</i>	<i>138.0</i>	<i>66.0</i>	<i>710</i>
		<i>Q1</i>	<i>137.5</i>	<i>107.0</i>	<i>120.0</i>	<i>56.2</i>	<i>466</i>
		<i>Q3</i>	<i>179.8</i>	<i>137.0</i>	<i>154.8</i>	<i>69.0</i>	<i>992</i>
		<i>Range</i>	<i>116.0-</i>	<i>94.0-</i>	<i>99.0-</i>	<i>48.0-</i>	<i>270-</i>
			<i>196.0</i>	<i>150.0</i>	<i>169.0</i>	<i>80.0</i>	<i>1340</i>

Table A3.4. Biometric data summary for male diamondback terrapins (n=22) captured in South Pond and Mangrove Lake between 2008 and 2010. Some data distributions were non-normal (Anderson-Darling tests; SCL p=0.030, SPL p=0.019), others were normal (Anderson-Darling tests; SCW p=0.400, SH p=0.118, Mass p=0.220).

Observation	Sex/ stage	No.	SCL (mm)	SCW (mm)	SPL (mm)	SH (mm)	Mass (g)
first	male	2	114	87	90	44	230
first	male	10	119	93	101	47	280
first	male	14	126	98	104	46	295
first	male	16	121	90	97	44	280
first	male	31	109	86	89	43	200
first	male	32	128	97	103	47	310
first	male	34	126	92	104	48	305
first	male	35	111	87	87	42	225
first	male	39	109	86	90	44	200
first	male	49	128	99	101	47	345
first	male	56	126	97	101	45	300
first	male	61	110	85	89	42	220
first	male	66	125	96	102	47	350
first	male	74	127	94	104	48	325
first	male	75	114	88	90	44	235
first	male	78	126	94	99	46	300
first	male	79	122	93	100	43	255
first	male	81	130	95	105	46	295
first	male	83	130	96	102	40	255
first	male	87	132	102	111	46	345
first	male	97	133	102	110	47	320
first	male	99	134	99	107	44	320
		<i>Median</i>	<i>126.0</i>	<i>94.0</i>	<i>101.0</i>	<i>45.5</i>	<i>295.0</i>
		<i>Q1</i>	<i>114.0</i>	<i>87.8</i>	<i>114.0</i>	<i>43.8</i>	<i>233.7</i>
		<i>Q3</i>	<i>128.5</i>	<i>97.2</i>	<i>128.5</i>	<i>47.0</i>	<i>320.0</i>
		<i>Range</i>	<i>108.0-</i>	<i>85.0-</i>	<i>87.0-</i>	<i>40.0-</i>	<i>200.0-</i>
			<i>134.0</i>	<i>102.0</i>	<i>111.0</i>	<i>48.0</i>	<i>350.0</i>

Table A3.5. Biometric data summary for juvenile diamondback terrapins (n=13) captured in South Pond and Mangrove Lake between 2008 and 2010. All data distributions were normal (Anderson-Darling tests; SCL p=0.188, SCW p=0.837, SPL p=0.193, SH p=0.502, Mass p=0.108).

Observation	Sex/ stage	No.	SCL (mm)	SCW (mm)	SPL (mm)	SH (mm)	Mass (g)
first	juvenile	29	108	85	92	46	215
first	juvenile	52	81	67	68	34	95
first	juvenile	77	102	80	86	47	190
first	juvenile	85	95	77	78	41	150
first	juvenile	86	95	73	76	39	150
first	juvenile	88	102	77	87	43	170
first	juvenile	89	107	88	92	44	210
first	juvenile	90	107	86	92	43	205
first	juvenile	91	108	89	92	46	210
first	juvenile	92	90	72	75	38	115
first	juvenile	93	81	65	67	36	100
first	juvenile	94	96	75	81	44	185
first	juvenile	98	101	80	78	42	190
		<i>mean</i>	98.0	78.0	81.8	41.8	168.1
		<i>SD</i>	9.5	7.7	9.0	4.0	42.6
		<i>Range</i>	81.0- 109.0	65.0- 89.0	67.0- 92.0	34.0- 47.0	92.0- 215.0

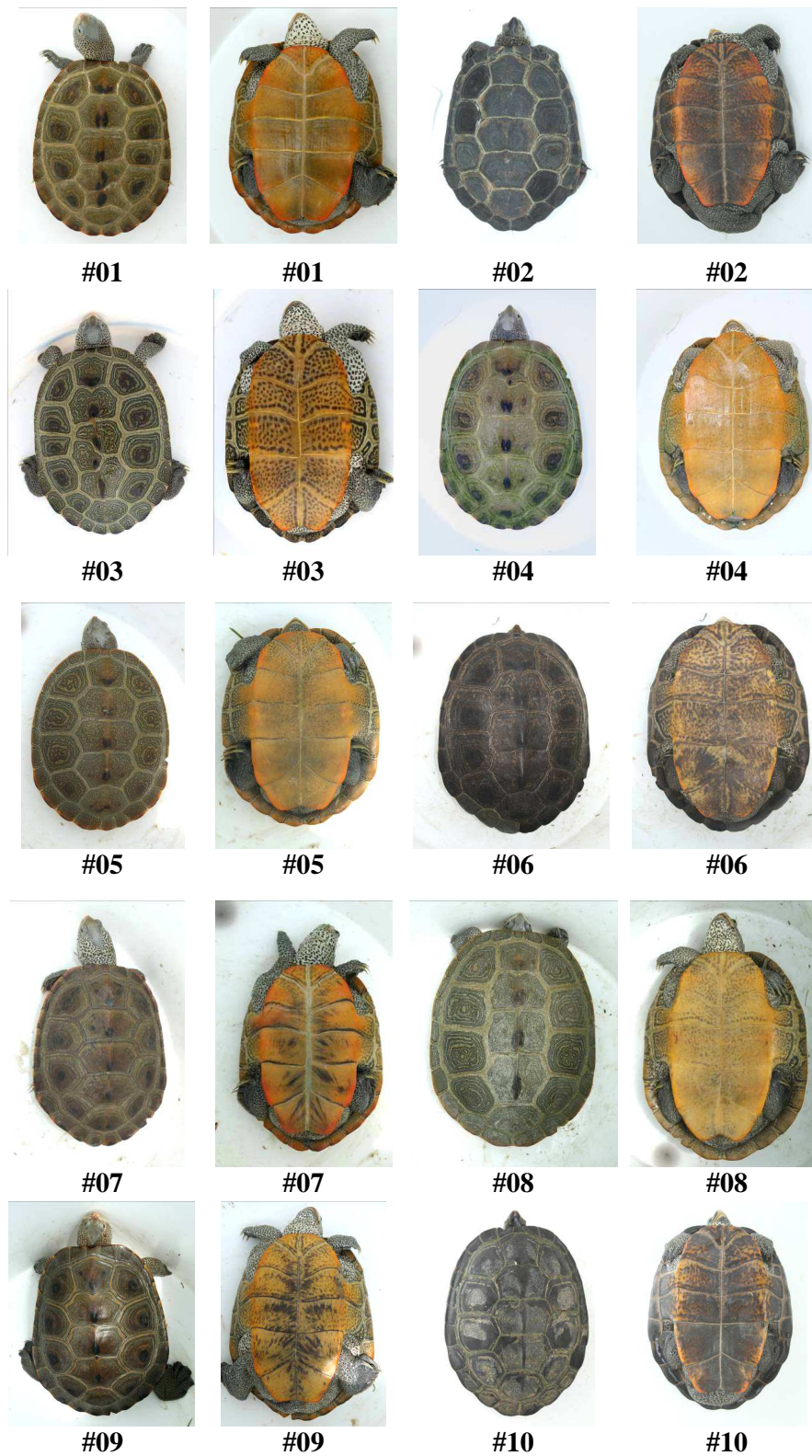


Figure A3.11. Carapace and plastron photographs of diamondback terrapins #1-10 (Note: not to scale).

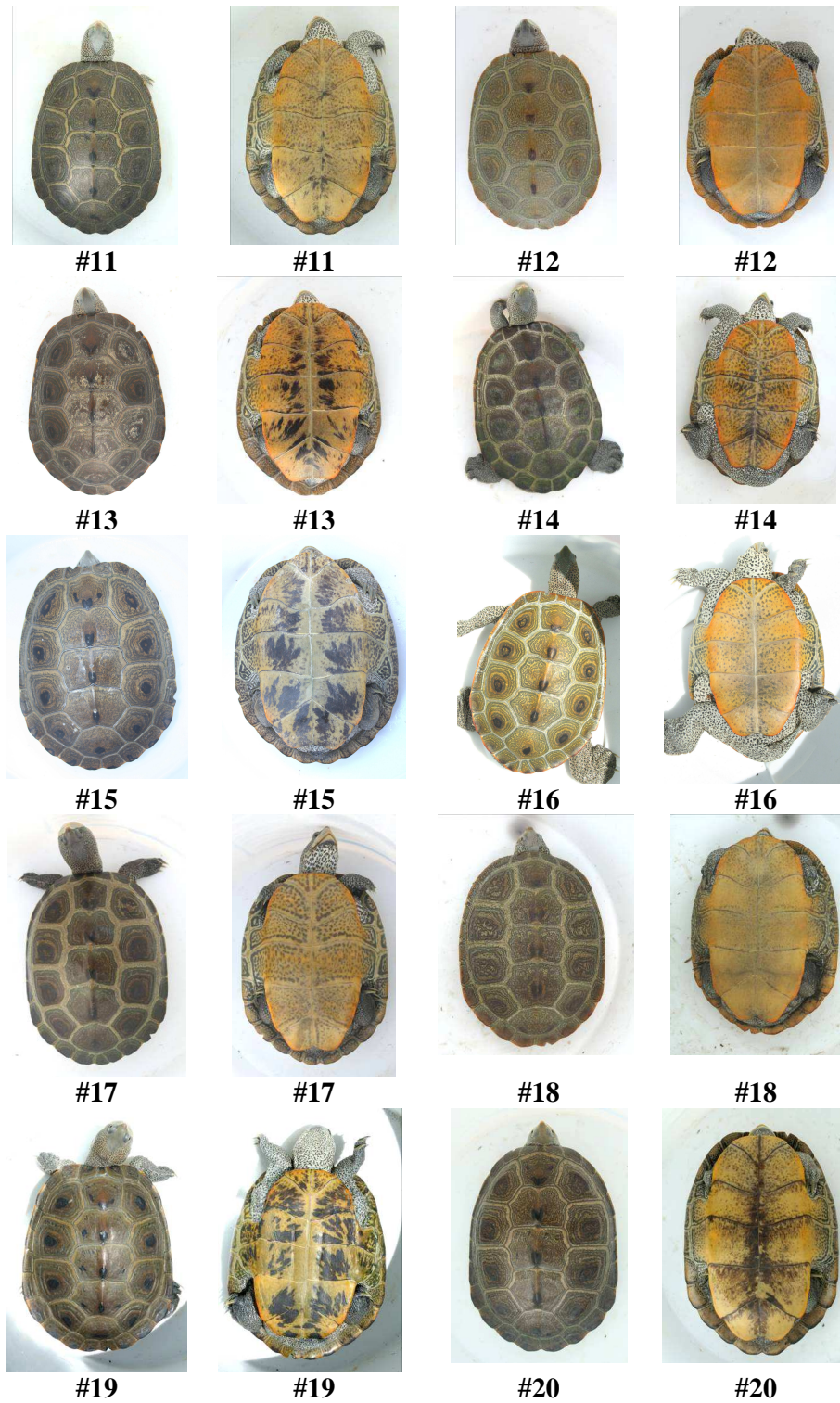


Figure A3.12. Carapace and plastron photographs of diamondback terrapins #11-20. (Note: not to scale).

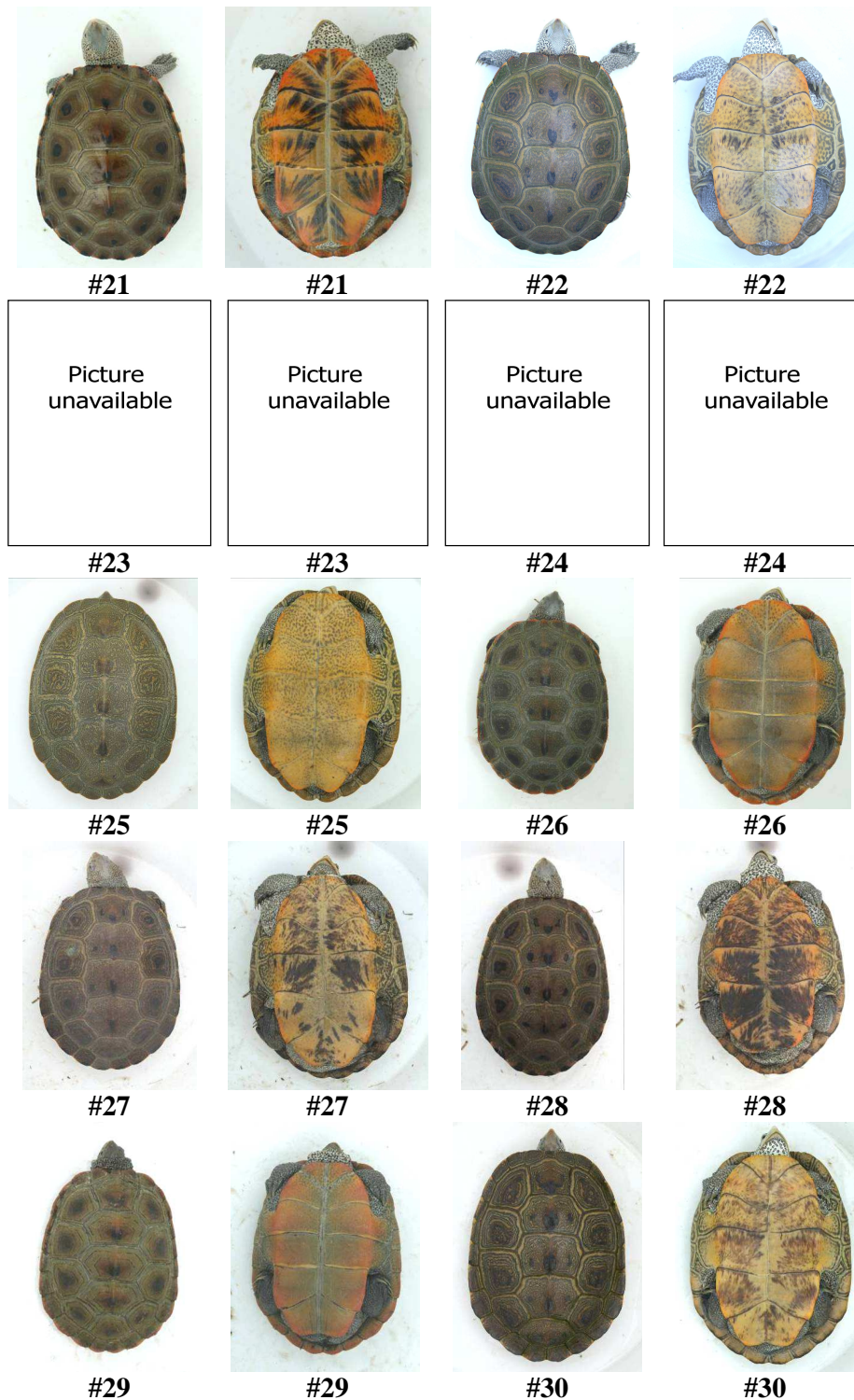


Figure A3.13. Carapace and plastron photographs of diamondback terrapins #21-30. (Note: not to scale).

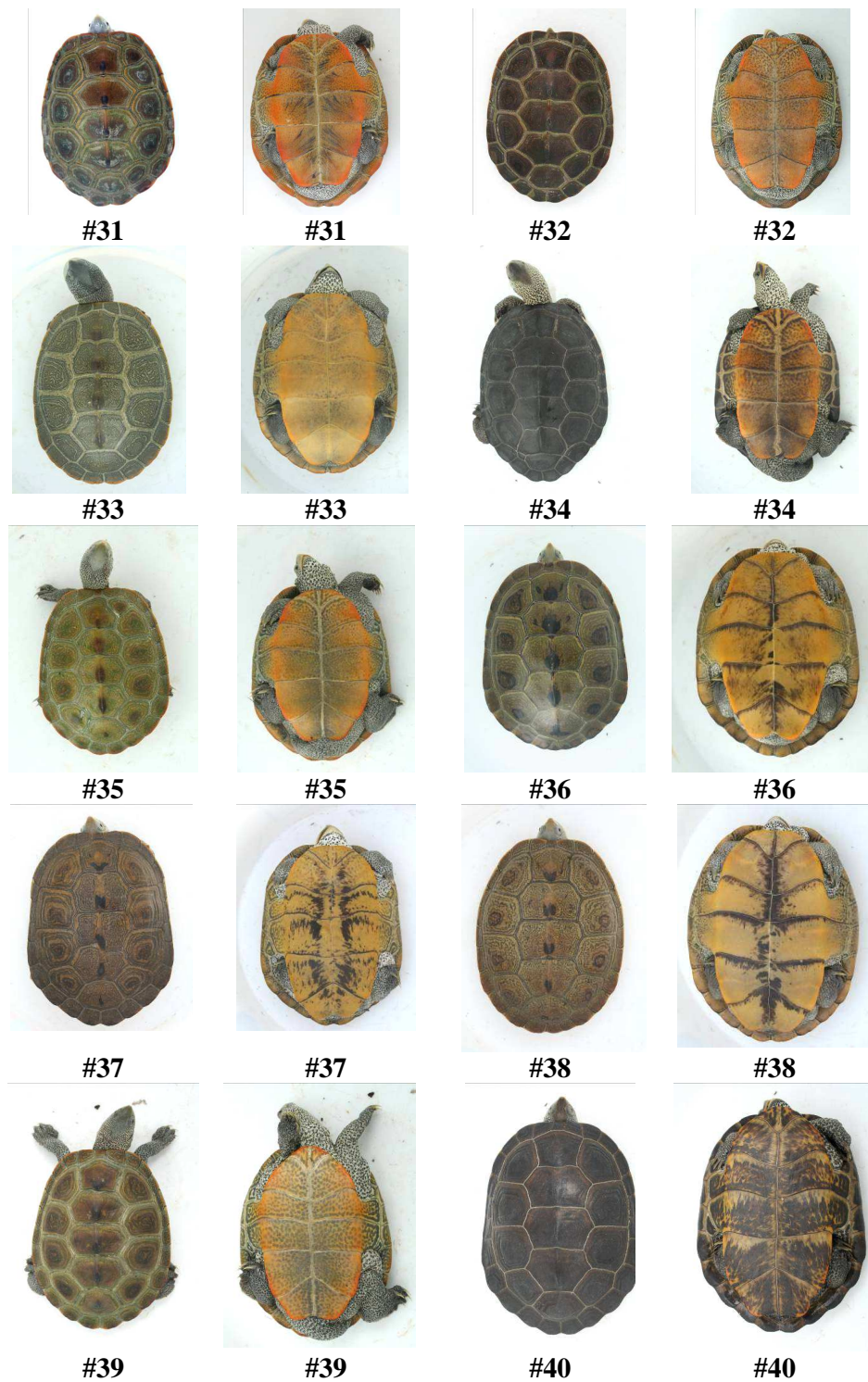


Figure A3.14. Carapace and plastron photographs of diamondback terrapins #31-40. (Note: not to scale).

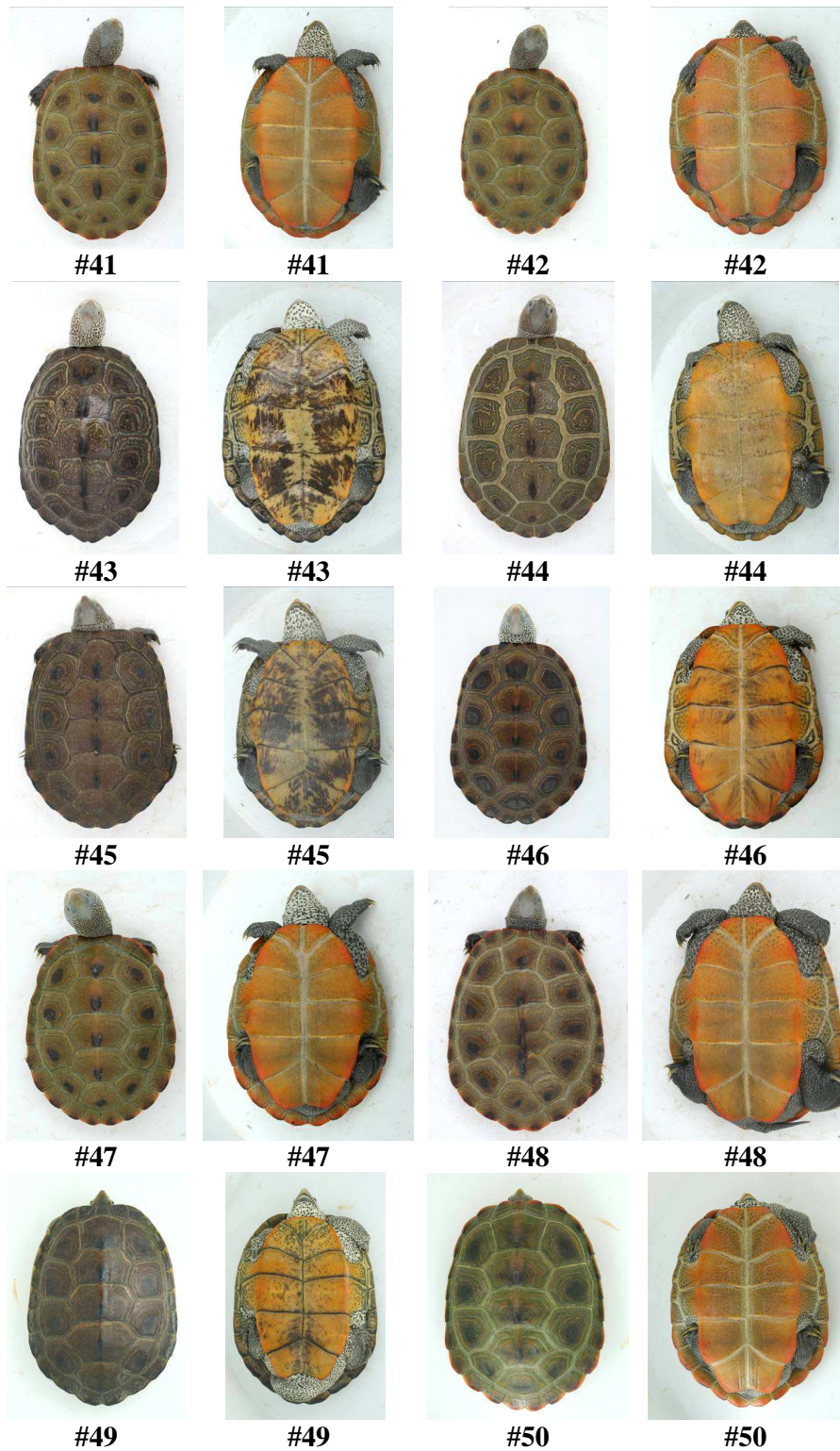


Figure A3.15. Carapace and plastron photographs of diamondback terrapins #41-50. (Note: not to scale).

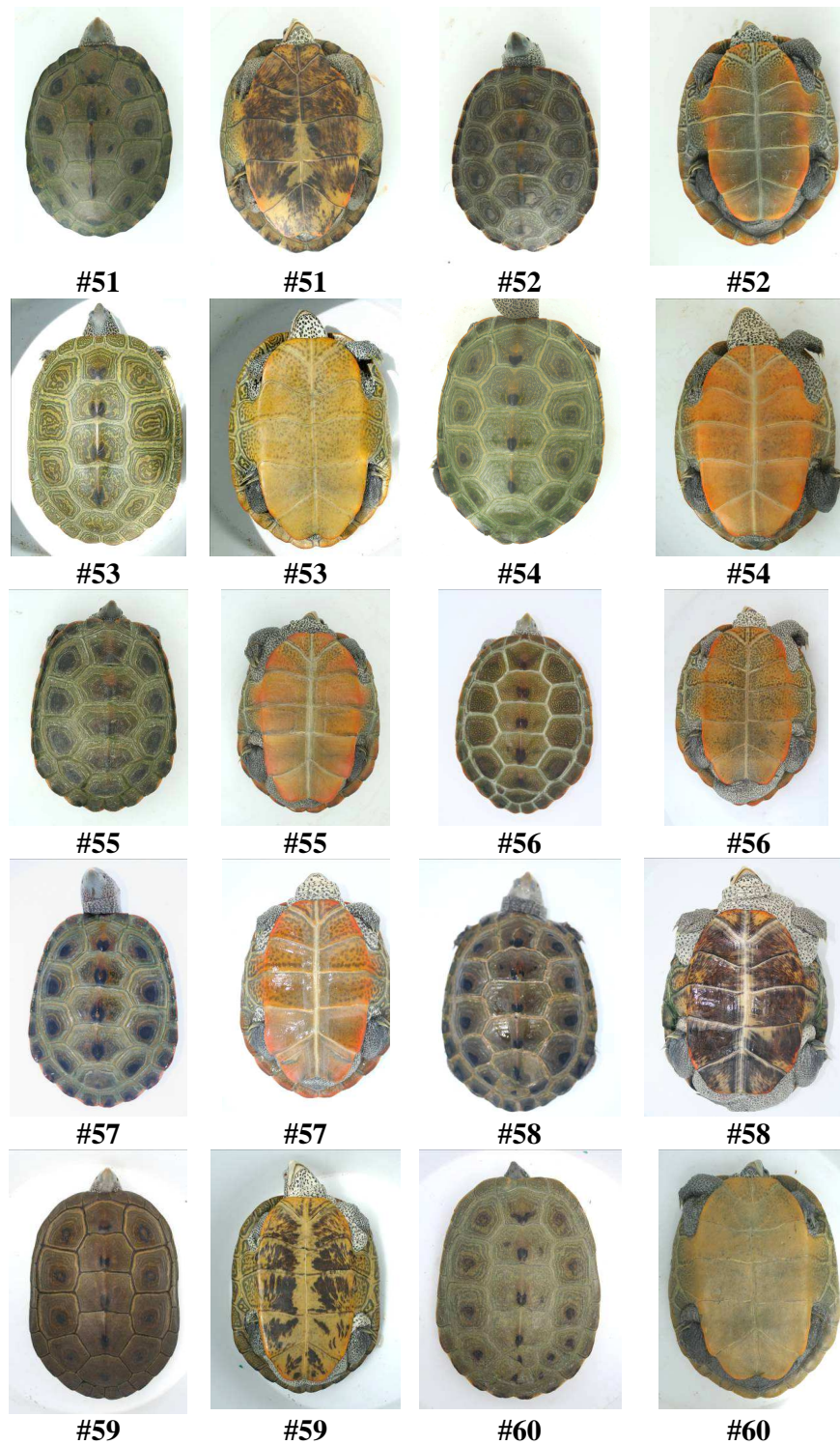


Figure A3.16. Carapace and plastron photographs of diamondback terrapins #51-60. (Note: not to scale).

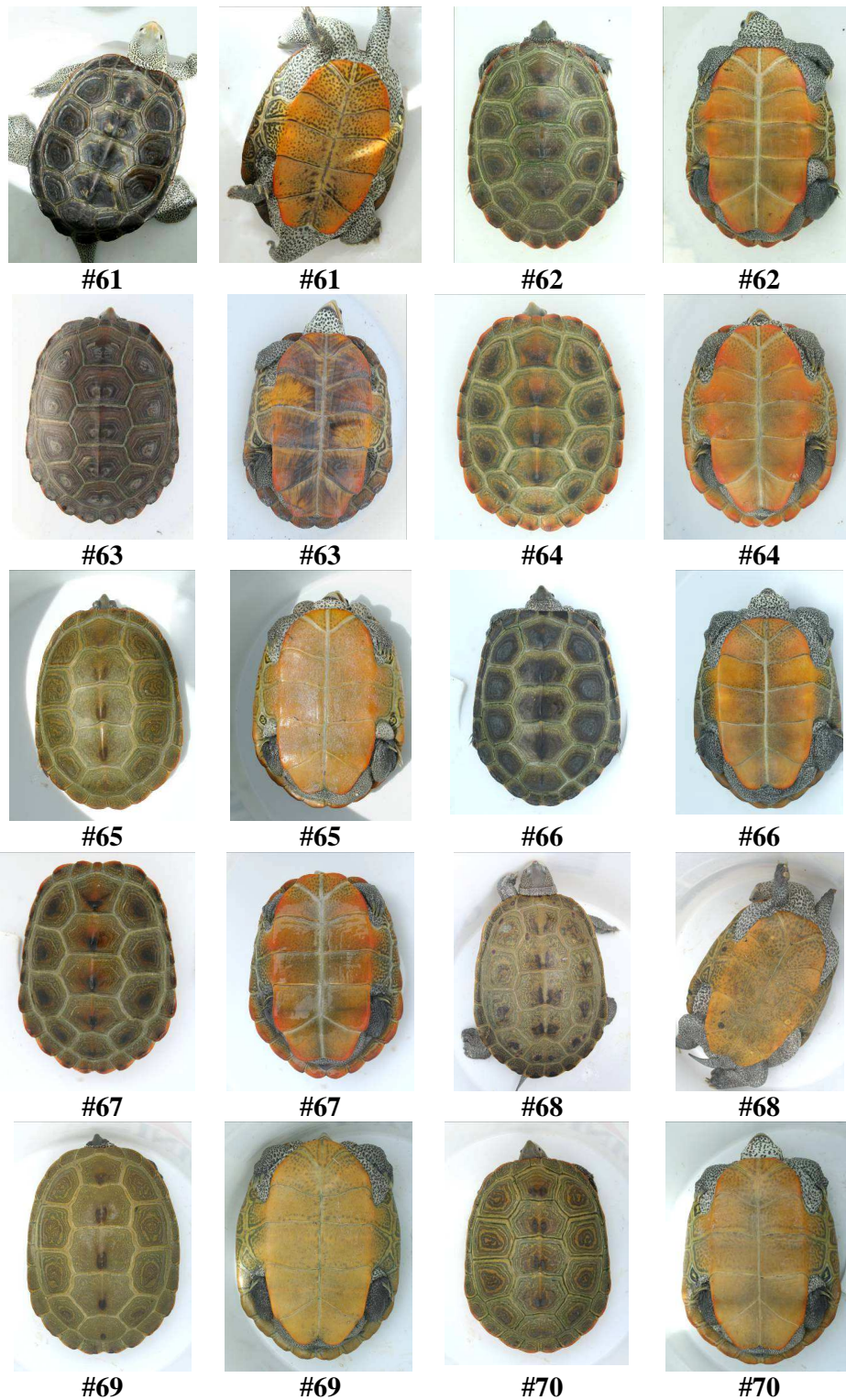


Figure A3.17. Carapace and plastron photographs of diamondback terrapins #61-70. (Note: not to scale).

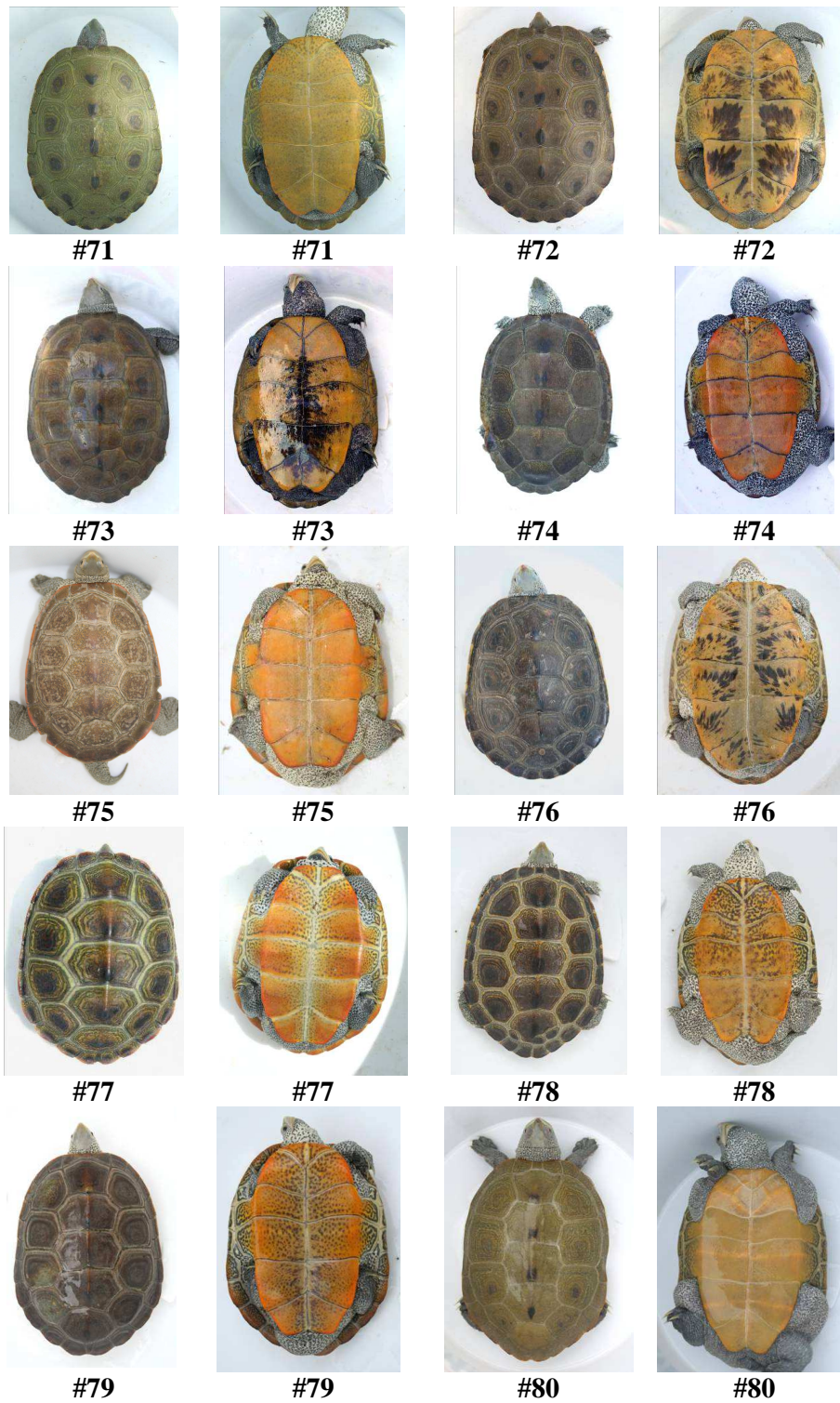


Figure A3.18. Carapace and plastron photographs of diamondback terrapins #71-80. (Note: not to scale).

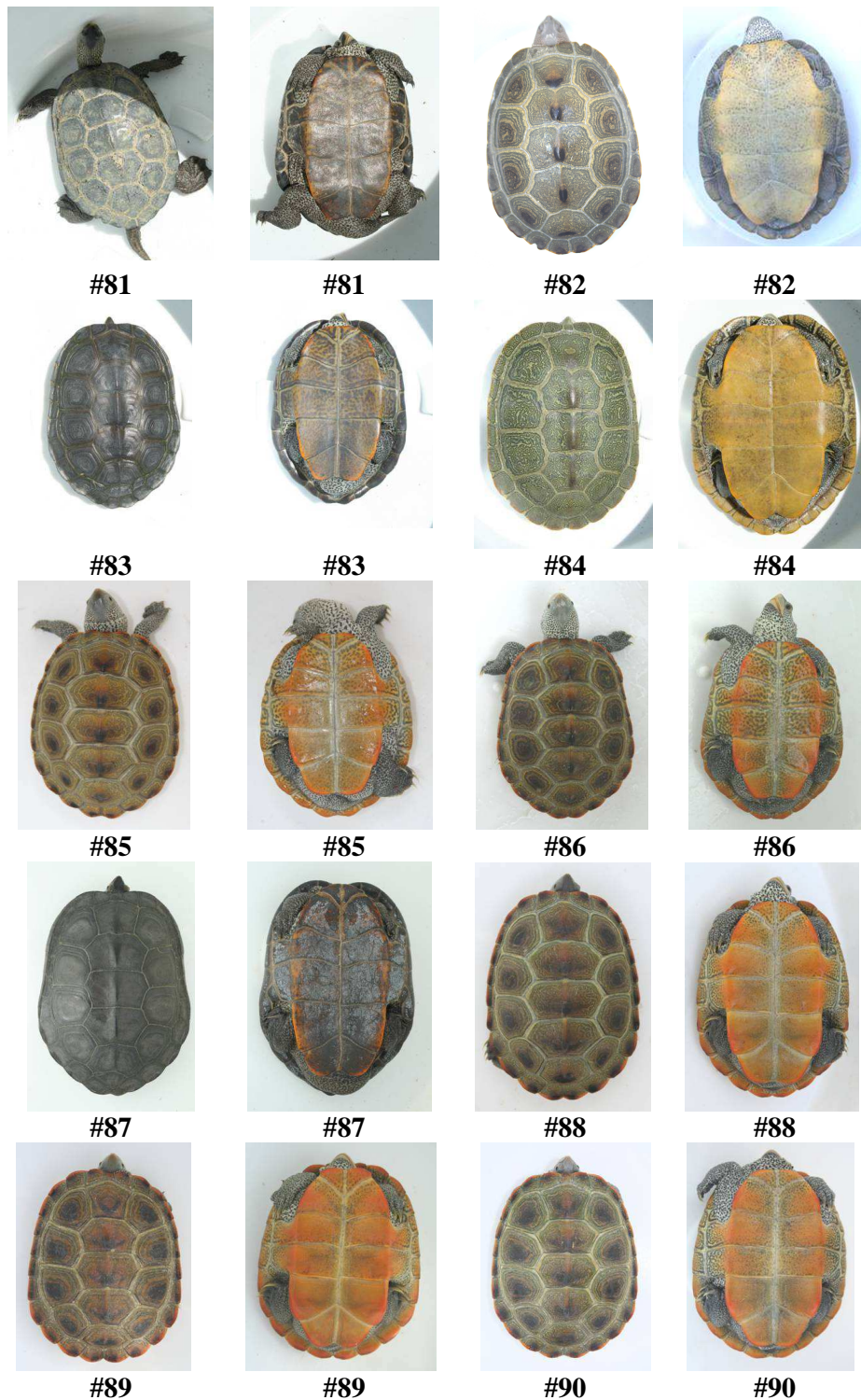


Figure A3.19. Carapace and plastron photographs of diamondback terrapins #81-90. (Note: not to scale).

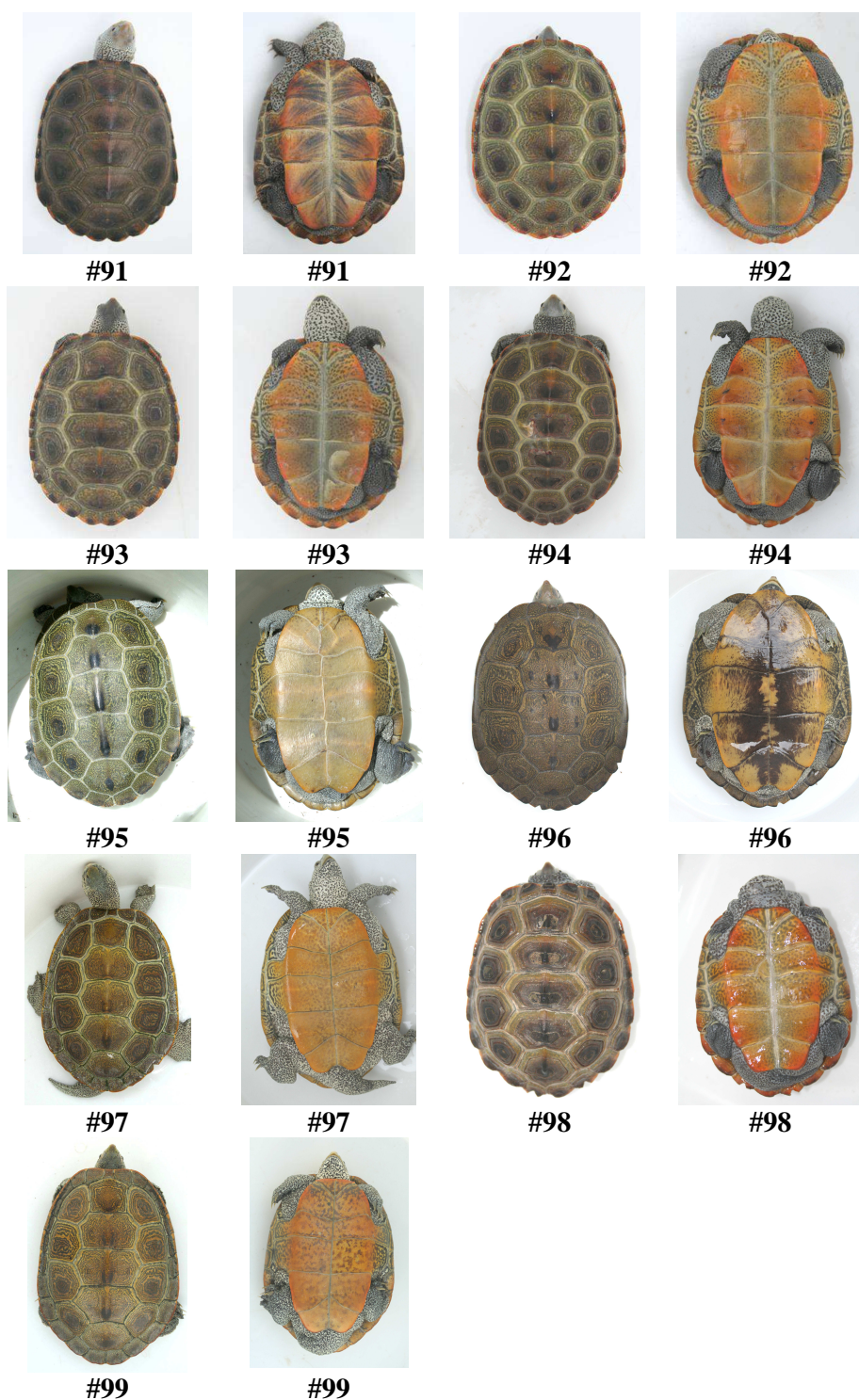


Figure A3.20. Carapace and plastron photographs of diamondback terrapins #91-99. (Note: not to scale).

Table A3.6. Growth data for Bermuda's diamondback terrapins (n=59) studied from 2008-2010.

Sex/Stage	No.	Initial SCL (mm)	Time Interval (days)	SCL growth increment (mm)	Growth rate (mm yr ⁻¹)
Juv	86	95	42	6	52.143
Juv	89	107	54	3	20.278
Juv	91	109	34	3	32.206
Juv	92	90	362	19	19.157
M	2	114	455	1	0.802
M	10	119	386	1	0.946
M	14	126	387	1	0.943
M	16	121	359	1	1.017
M	27	171	452	3	2.423
M	28	145	453	8	6.446
M	50	127	339	10	10.767
M	57	135	415	11	9.675
M	68	195	745	2	0.980
F	1	134	69	2	0.378
F	3	152	393	8	7.430
F	4	163	824	4	1.772
F	5	146	394	6	5.558
F	6	182	790	1	0.462
F	7	133	453	26	20.94
F	8	179	727	6	3.012
F	9	155	751	6	2.916
F	11	165	68	2	10.73
F	12	140	361	1	1.011
F	13	154	69	1	5.290
F	17	182	436	1	0.837
F	19	157	27	0	0
F	20	180	35	0	0
F	21	128	456	14	11.206
F	22	150	814	15	6.726
F	26	130	746	42	20.55
F	27	171	452	3	2.423
F	28	145	453	8	6.446
F	29	108	386	18	17.021
F	31	109	71	1	5.141
F	35	111	352	0	0
F	36	159	805	7	3.174
F	37	172	25	1	14.600
F	38	166	24	0	0
F	40	144	779	9	4.217
F	41	133	427	17	14.532
F	42	123	351	7	7.279
F	43	188	59	0	0
F	45	157	386	6	5.674
F	46	135	387	19	17.920
F	47	129	781	18	8.412
F	48	137	25	2	29.200
F	50	127	339	10	10.767
F	51	141	798	5	2.287
F	52	142	721	61	30.880
F	55	119	356	16	16.404
F	57	135	415	11	9.675
F	65	161	296	4	4.932

Table A3.6. (continued). Growth data for Bermuda's diamondback terrapins studied from 2008-2010.

Sex/Stage	No.	Initial SCL (mm)	Time Interval (days)	SCL growth increment (mm)	Growth rate (mm yr ⁻¹)
F	67	125	27	2	27.037
F	68	195	745	2	0.980
F	76	148	377	3	2.905
F	80	179	252	1	1.448
F	82	186	466	1	0.783
F	85	95	427	38	32.482
F	90	107	432	23	19.433

Table A3.7. Annual growth rates for juvenile diamondback terrapins (n=3) in Bermuda.

<i>SCL (mm) at first capture</i>	<i>mm yr⁻¹</i>
81	30.9
90	19.2
108	17.0
<i>mean</i>	<i>22.4</i>
<i>SD</i>	<i>7.5</i>

Table A3.8. Annual growth rates for male diamondback terrapins (n=6) in Bermuda. (* indicates sexually mature individual).

<i>SCL (mm) at first capture</i>	<i>mm yr⁻¹</i>
111	0.0
114	0.0
119*	0.9
121*	1.0
126*	0.9
126*	2.1
<i>mean</i>	<i>0.8</i>
<i>SD</i>	<i>0.8</i>

Table A3.9. Annual growth rates for female diamondback terrapins (n=13) in Bermuda. (* indicates sexually mature individual).

<i>SCL (mm) at first capture</i>	<i>mm yr⁻¹</i>
119	16.4
123	7.3
127	10.8
130	20.7
135	17.9
140	1.0
146	5.6
148	2.9
152	7.4
155	2.9
157	5.7
179*	3.0
195*	1.0
<i>mean</i>	<i>7.9</i>
<i>SD</i>	<i>6.6</i>

Table A3.10. Log-transformed regression lines of growth for female diamondback terrapins (n=64) in Bermuda.

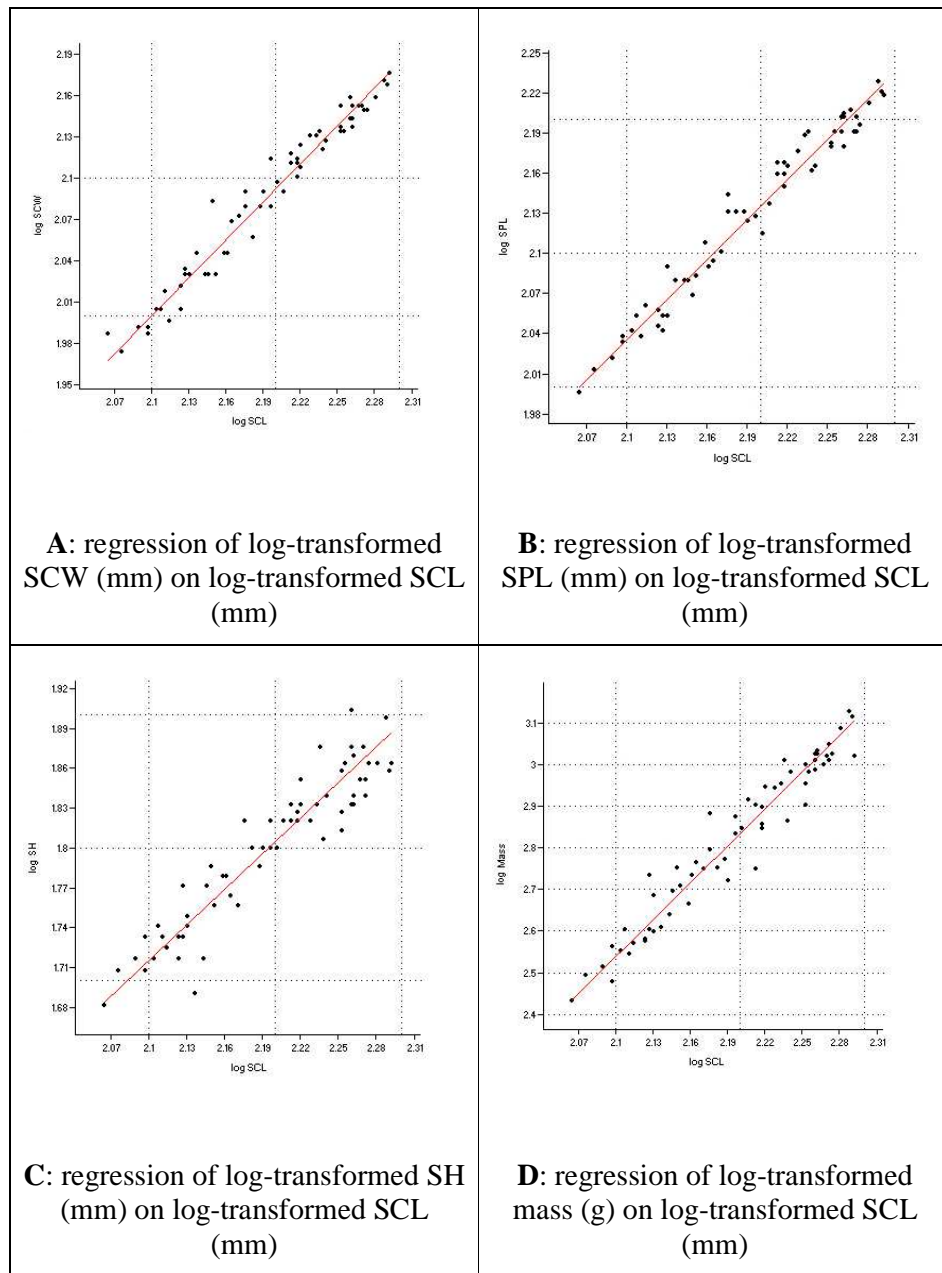
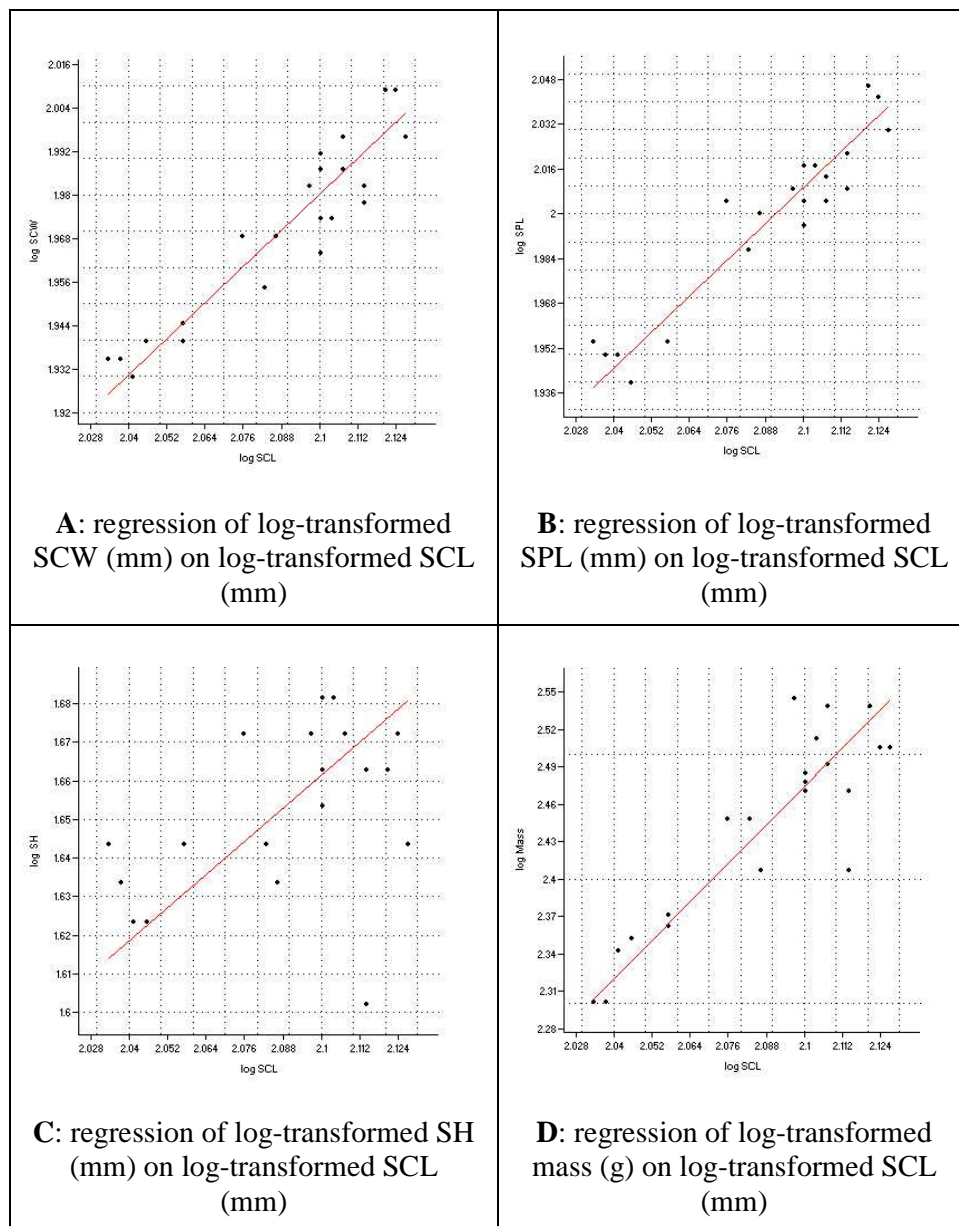


Table A3.11. Log-transformed regression lines of growth for male diamondback terrapins (n=22) in Bermuda.



Appendix 4: Supplementary Material to Chapter 4

Gastric Lavage

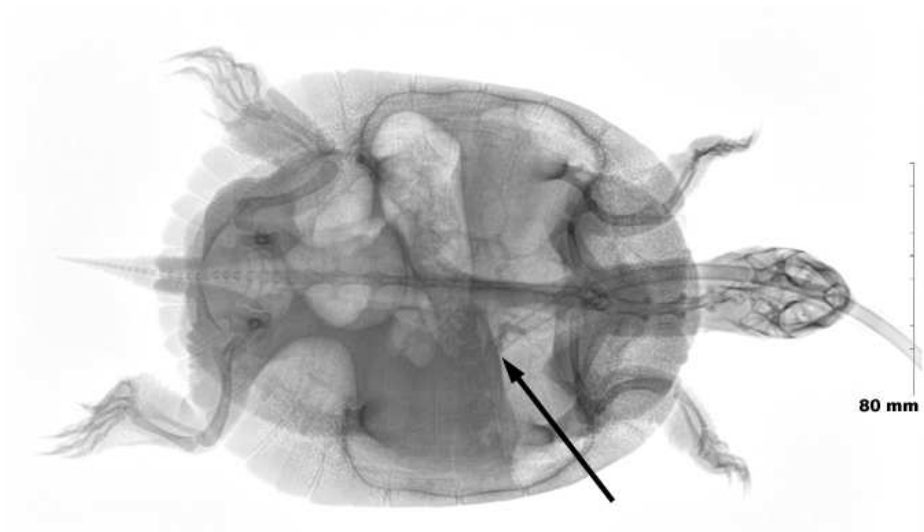
Three male and five female red-eared sliders, ranging in size from 125-190 mm straight carapace length, were sampled using gastric lavage. A restraining table was constructed using polyvinylchloride (PVC) board that measured 60 cm by 30 cm with a 45 degree downward tilt, onto which each slider was secured using a velcro belt (Fig. 4.1). A non-skid cloth was glued to the table surface and two foam blocks were placed either side of the sliders to prevent rolling and slipping during the procedure. Prior to being placed upon the restraining table, each slider was given an intramuscular injection of ketamine HCl, a dissociative anaesthetic, to promote general sedation and relaxation of the digestive tract. An initial dose of 25 mg/kg ketamine HCl was administered, followed by a 30 minute period of observation. If the individual was not totally sedated after this period, an additional 8 mg/kg was administered followed by a second 30 minute period of observation. The sedated slider was then secured to the table in a head-down and plastron-up position, and an adjustable test-tube clamp on a threaded rod was fastened to the extended neck, immediately behind the skull. This ensured that the head stayed extended and the neck in-line with the mid-line of the plastron. The jaw of each slider was held open by using a short length of inflexible vinyl tubing (10 mm outside diameter (OD) cut to a length of 30 mm), through which a flexible vinyl 5 mm OD delivery tube (used on five individuals collectively referred to as Group 1) and 3 mm OD (used on three individuals collectively referred to as Group 2) was inserted. Each delivery tube was coated with lubricating jelly and slowly inserted into the esophagus. The distance to the stomach was determined prior to tube insertion by laying the tube along the mid-line of the plastron and measuring from the junction of the pectoral and abdominal scutes to the tip of the mouth. Gentle twisting of the tube during insertion, and relaxation of the cardiac sphincter due to the ketamine HCl, facilitated the successful insertion of the tube into the stomach (Fig. 4.2). 250 ml of fresh water was then very slowly injected into the stomach using a plastic syringe. The

flushed contents passed out of the mouth, down the flushing trough and collected in the receiving bowl.



Source: John Davenport

Figure A4.1. Author using the stomach flushing table and delivery system on a sedated red-eared slider.



Source: Ian Walker

Figure A4.2. Reverse polarity x-ray of a sedated red-eared slider showing path of vinyl flushing tube through esophagus into stomach (insertion point into stomach is indicated by arrow).

All of the red-eared sliders were successfully sedated using the ketamine HCl, and subsequently recovered following the flushing procedure. However, despite flushing 250 mls of fresh water into the stomach of each slider, very little material was collected. Some material was collected from three of the five sliders using the 5 mm OD delivery tube (Group 1) and from one of the three sliders using the 3 mm OD delivery tube (Group 2). The recovered stomach content material consisted mostly of plant matter along with a few insect larvae, but none of the samples were large in volume.

To test the effectiveness of this flushing technique, all of the red-eared sliders were euthanised and necropsied immediately after the procedure. The post-procedural necropsies were performed in order to determine the degree of physical damage to each slider, as well as to evaluate the effectiveness of the flushing technique. The necropsies of the Group 1 sliders revealed that three still had full stomachs (the other two appeared to have had empty stomachs prior to the flushing procedure); while the necropsies of the Group 2 sliders revealed that all three still had stomachs containing food items. Furthermore, the intestines of these sliders were extremely distended with water (Fig. 4.3). Damage to the mouth, stomach, or esophagus was not observed in any of the sliders during the necropsies.



Source: John Davenport

Figure A4.3. Necropsied red-eared slider showing intestines inflated with water from the stomach flushing procedure.

The gastric lavage results were only partially successful. Some stomach content material was collected in the receiving bowl from half of the turtles; however the discovery of the water-filled intestines in the dissected red-eared sliders gave cause for alarm. It was hypothesized that the relatively large size of the 5 mm OD delivery tube used on the turtles in Group 1, combined with the highly fibrous material found in the stomachs, created a blockage that prevented successful flushing. This blockage forced the injected water deeper into the gastrointestinal tract of the sliders, thereby creating the observed intestinal distension. To increase the flushing success, the size of the delivery tube was reduced to 3 mm OD; however this neither resulted in procuring the stomach contents of the sliders in Group 2, nor did it result in reducing the incidence of intestinal distension. Because of the invasive nature of the procedure and the unwanted, and potentially damaging, water-filling effect it had upon the intestines of the red-eared sliders, it was decided that gastric lavage would not be performed upon Bermuda's diamondback terrapins.

Gastric lavage for use on wild populations of turtles was pioneered by Legler (1977) and has been successfully used on a wide variety of small to medium-sized chelonians, including yellow-bellied sliders (*Chrysemys scripta*) (Parmenter, 1980), post-hatchling (neonate) loggerhead turtles (*Caretta caretta*) (Witherington, 2002), green sea turtles (*Chelonia mydas*) (Seminoff et al., 2002), red side-necked turtles (*Phrynops rufipes*) (Caputo and Vogt, 2008), Texas river cooters (*Pseudemys texana*) and red-eared sliders (*Trachemys scripta elegans*) (Fields et al., 2000). The latter observed esophageal damage in some of the turtles during early trial experiments and attributed this damage to blockages caused by the delivery tube and trapped food masses. There are no published studies reporting the use of gastric lavage on diamondback terrapins; however, given the success reported in the above investigations, it is not unrealistic to assume that this technique would work on this species. The harm or destruction of any individual by performing gastric lavage on the Bermuda population was deemed unacceptable on conservation grounds, especially in pursuit of a meaningful sample size. Investigators in the U.S.A. who have access to much larger populations of terrapins may find success with this technique – particularly

if a small delivery tube is used (i.e. less than 3 mm OD); however the fact that some soft-bodied food items are able to pass through the gastrointestinal tract of a diamondback terrapin (see faecal analyses results in Chapter 4) suggests that the use of gastric lavage on this species may be unnecessary.

Table A4.1. Week 1 (April 19-25, 2010) survey schedule for yellow-crowned night heron predation observations at South Pond. (Note: The red “X” represents an observed predation event).

	Mon.	Tues.	Wed.	Thur.	Fri.	Sat.	Sun.
06:00			X				
07:00	X						
08:00							
09:00			X			X	
10:00							
11:00							
12:00							
13:00			X			X	
14:00							
15:00							
16:00	X						
17:00							
18:00	X					X	
19:00							

Table A4.2. Week 2 (April 2-May 2, 2010) survey schedule for yellow-crowned night heron predation observations at South Pond. (Note: The red “X” represents an observed predation event).

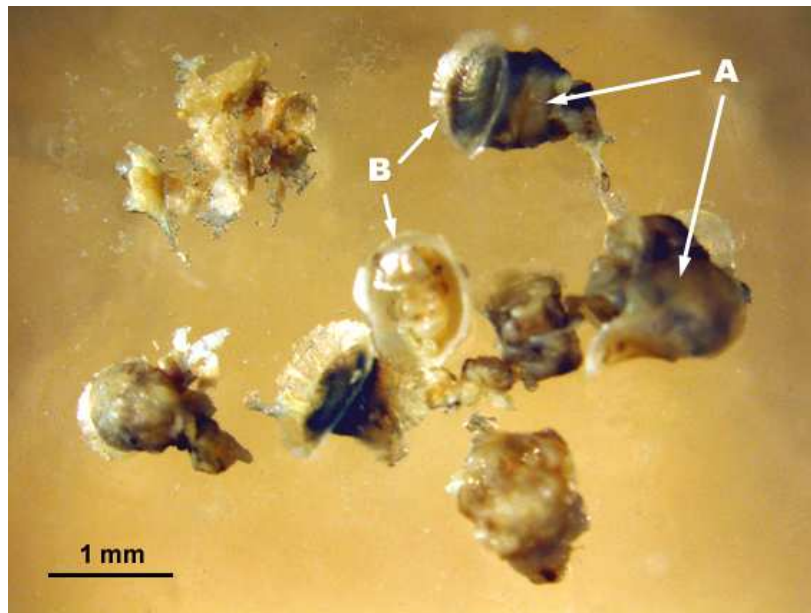
	Mon.	Tues.	Wed.	Thur.	Fri.	Sat.	Sun.
06:00							
07:00							
08:00	X	X			X		
09:00							
10:00	X						
11:00							
12:00					X		
13:00					X		
14:00							
15:00							
16:00							
17:00		X					
18:00	X						
19:00		X					

Table A4.3. Week 3 (May 3-9, 2010) survey schedule for yellow-crowned night heron predation observations at South Pond. (Note: The red “X” represents an observed predation event).

	Mon.	Tues.	Wed.	Thur.	Fri.	Sat.	Sun.
06:00	X						
07:00				X			
08:00			X				
09:00	X						
10:00			X				
11:00							
12:00							
13:00				X			
14:00				X			
15:00	X						
16:00			X				
17:00							
18:00							
19:00							

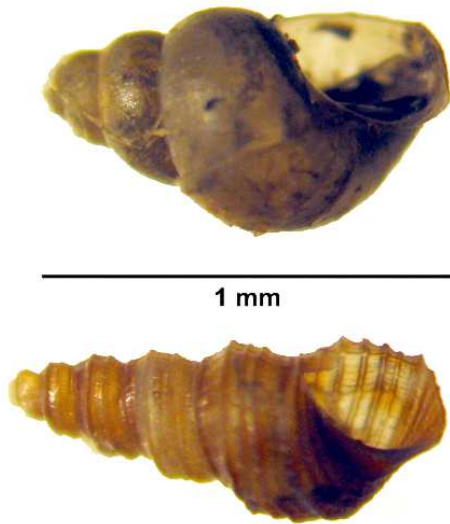
Table A4.4. Week 4 (May 10-16, 2010) survey schedule for yellow-crowned night heron predation observations at South Pond. (Note: The red “X” represents an observed predation event).

	Mon.	Tues.	Wed.	Thur.	Fri.	Sat.	Sun.
06:00							
07:00						X	
08:00		X					
09:00							X
10:00							
11:00						X	
12:00							X
13:00		X					
14:00							
15:00							X
16:00							
17:00		X					
18:00						X	
19:00							



Source: Mark Outerbridge

Figure A4.4. Flesh (A) and opercula (B) of the gastropod *Heleobops bermudensis* found within the stomach of a predated juvenile diamondback terrapin (51 mm SCL) from South Pond.



Source: Mark Outerbridge

Figure A4.5. Gastropod snails obtained from a faecal sample of a diamondback terrapin; *Heleobops bermudensis* (above) and *Melanoides tuberculata* (below).

Table A4.5. Dry mass summary of food items obtained from 33 faecal samples of diamondback terrapins collected from four sites (South Pond, Mangrove Lake, Trott's Pond, and North Pond).

ID	Sex	SCL (mm)	Site	Date	<i>Melanoides</i> dry mass (g)	<i>Heleobops</i> dry mass (g)	<i>Melampus</i> dry mass (g)	<i>Isognomon</i> dry mass (g)	Vegetation dry mass (g)	Insect dry mass (g)	Fish bone dry mass (g)	Toad bone dry mass (g)	Terrapin bone dry mass (g)	Polychaete dry mass (g)	Sediment dry mass (g)	TOTAL dry mass (g)
12	F	145	SP	02-04-11	-	0.1945	-	-	<i>not included</i>	-	-	0.3786	-	-	<i>not included</i>	0.5731
17	F	184	SP	15-10-11	0.8072	-	-	-	<i>not included</i>	-	-	0.7960	0.0003	-	<i>not included</i>	1.6035
27	F	175	SP	18-05-11	6.6892	-	-	-	<i>not included</i>	-	-	-	-	-	<i>not included</i>	6.6892
27	F	175	SP	02-04-11	13.4850	0.0026	-	-	-	-	-	-	-	-	<i>not included</i>	13.4876
33	F	172	ML	07-15-11	-	-	-	-	<i>not included</i>	-	0.0180	-	-	-	<i>not included</i>	0.0180
36	F	166	SP	14-09-10	4.7793	0.0645	-	-	<i>not included</i>	-	-	-	-	-	<i>not included</i>	4.8438
36	F	166	SP	11-04-11	-	-	-	-	-	0.0341	-	-	-	-	<i>not included</i>	0.0341
36	F	166	SP	02-02-11	3.2958	0.0080	-	-	<i>not included</i>	-	-	-	-	-	<i>not included</i>	3.3038
40 ³³	F	153	SP	19-08-10	0.6746	1.1129	-	-	<i>not included</i>	-	-	-	-	-	<i>not included</i>	1.7875
43 ³³	F	188	SP	19-08-10	1.4813	0.1140	-	-	-	-	0.0001	-	-	-	<i>not included</i>	1.5954
50	F	170	ML	28-07-11	-	0.0010	2.1850	-	-	-	-	-	-	-	-	2.1860
51	F	146	SP	11-04-11	2.4658	-	-	-	-	-	-	-	-	-	<i>not included</i>	2.4658
68	F	196	SP	09-04-10	0.5733	7.3795	-	-	<i>not included</i>	0.0060	-	-	-	-	<i>not included</i>	7.9588
68	F	196	SP	09-03-10	0.1573	0.5150	-	-	<i>not included</i>	0.0001	-	-	-	-	<i>not included</i>	0.6724
69	F	194	NP	25-05-10	-	-	-	-	-	0.0385	-	-	-	-	<i>not included</i>	0.0385
82	F	187	ML	26-09-10	-	0.8896	-	-	-	-	0.0625	-	-	-	-	0.9521
85	F	133	SP	26-08-10	1.5952	0.0737	-	-	<i>not included</i>	-	-	-	-	-	<i>not included</i>	1.6689
90	F	126	SP	19-08-10	0.2198	0.1871	-	-	<i>not included</i>	-	-	-	-	-	<i>not included</i>	0.4069
107	F	177	ML	31-05-11	-	-	0.0333	0.0595	-	-	-	-	-	-	-	0.0928
108	F	175	NP	01-07-11	-	1.3370	-	-	<i>not included</i>	-	-	-	-	-	-	1.3370

Table A4.5 (continued) Dry mass summary of food items obtained from 33 faecal samples of diamondback terrapins collected from four sites (South Pond, Mangrove Lake, Trott's Pond, and North Pond).

[illegible]

Table A4.6. Percentage frequency of occurrence of food items obtained from 33 diamondback terrapin faecal samples collected from four sites (South Pond, Mangrove Lake, Trott's Pond, and North Pond).

ID	Sex	SCL	Site	Date	<i>Melanoides</i> % total	<i>Heleobops</i> % total	<i>Melampus</i> % total	<i>Isognomon</i> % total	Insects % total	Fish bone % total	Toad bone % total	Terrapin bone % total	Polychaete % total
12	F	145	SP	02-04-11	-	33.94%	-	-	-	-	66.06%	-	-
17	F	184	SP	15-10-11	50.34%	-	-	-	-	-	49.64%	0.02%	-
27	F	175	SP	18-05-11	100.00%	-	-	-	-	-	-	-	-
27	F	175	SP	02-04-11	99.98%	0.02%	-	-	-	-	-	-	-
33	F	172	ML	07-15-11	-	-	-	-	-	100.00%	-	-	-
36	F	166	SP	14-09-10	98.67%	1.33%	-	-	-	-	-	-	-
36	F	166	SP	11-04-11	-	-	-	-	100.00%	-	-	-	-
36	F	166	SP	02-02-11	99.76%	0.24%	-	-	-	-	-	-	-
40	F	153	SP	19-08-10	37.74%	62.26%	-	-	-	-	-	-	-
43	F	188	SP	19-08-10	92.85%	7.15%	-	-	-	0.01%	-	-	-
50	F	170	ML	28-07-11	-	0.05%	99.95%	-	-	-	-	-	-
51	F	146	SP	11-04-11	100.00%	-	-	-	-	-	-	-	-
68	F	196	SP	09-04-10	7.20%	92.72%	-	-	0.08%	-	-	-	-
68	F	196	SP	09-03-10	23.39%	76.59%	-	-	0.01%	-	-	-	-
69	F	194	NP	25-05-10	-	-	-	-	100.00%	-	-	-	-
82	F	187	ML	26-09-10	-	93.44%	-	-	-	6.56%	-	-	-
85	F	133	SP	26-08-10	95.58%	4.42%	-	-	-	-	-	-	-
90	F	126	SP	19-08-10	54.02%	45.98%	-	-	-	-	-	-	-
107	F	177	ML	31-05-11	-	-	35.88%	64.12%	-	-	-	-	-
108	F	175	NP	01-07-11	-	100.00%	-	-	-	-	-	-	-

Table A4.6 (continued) Percentage frequency of occurrence of food items obtained from 33 diamondback terrapin faecal samples collected from four sites (South Pond, Mangrove Lake, Trott's Pond, and North Pond).

ID	Sex	SCL	Site	Date	<i>Melanoides</i> % total	<i>Heleobops</i> % total	<i>Melampus</i> % total	<i>Isognomon</i> % total	Insects % total	Fish bone % total	Toad bone % total	Terrapin bone % total	Polychaete % total
16	M	122	SP	01-05-10	-	100.00%	-	-	-	-	-	-	-
75	M	114	SP	11-03-10	26.36%	73.64%	-	-	-	-	-	-	-
75	M	114	SP	29-08-10	11.38%	88.62%	-	-	-	-	-	-	-
99	M	134	ML	30-09-10	-	-	-	-	-	0.65%	-	-	99.35%
109	M	121	ML	15-07-11	-	-	-	-	-	100.00%	-	-	-
88	Juv	107	SP	05-04-10	50.91%	48.54%	-	-	0.55%	-	-	-	-
98	Juv	101	ML	25-08-10	-	95.77%	-	-	4.23%	-	-	-	-
102	Juv	97	SP	07-04-11	-	100.00%	-	-	-	-	-	-	-
H5	neonate	31	ML	26-04-11	-	100.00%	-	-	-	-	-	-	-
H6	neonate	34.9	SP	02-09-10	-	100.00%	-	-	-	-	-	-	-
H6	neonate	34	SP	25-04-11	-	100.00%	-	-	-	-	-	-	-
H3	neonate	34.1	ML	26-04-11	-	93.43%	-	-	6.57%	-	-	-	-
H5	neonate	34.6	ML	28-04-11	-	100.00%	-	-	-	-	-	-	-

Table A4.7. Summary of total number, size range, mean size (with SD) for whole *Melanoides tuberculata*, *Heleobops bermudensis* and *Melampus coffeus* obtained from the faecal samples of 17 female diamondback terrapins. (Note: the blue text indicates estimated numbers derived from sub-sampling).

ID	Sex	SCL	Site	Date	<i>Melanoides</i> N	<i>Melanoides</i> size range TH (mm)	<i>Melanoides</i> mean TH (mm)	<i>Melanoides</i> SD	<i>Heleobops</i> N	<i>Heleobops</i> size range TH (mm)	<i>Heleobops</i> mean TH (mm)	<i>Heleobops</i> SD	<i>Melampus</i> N	<i>Melampus</i> size range TH (mm)	<i>Melampus</i> mean TH (mm)	<i>Melampus</i> SD
12	F	145	SP	02-04-11	-	-	-	-	41	1 - 4	1.4	0.6	-	-	-	-
17	F	184	SP	15-10-11	90	1 - 7	3.4	1.2	-	-	-	-	-	-	-	-
27	F	175	SP	18-05-11	159	1 - 18	5.0	3.8	-	-	-	-	-	-	-	-
27	F	175	SP	02-04-11	438	1 - 18	7.1	4.4	2	3 - 5	4.0	1.4	-	-	-	-
36	F	166	SP	14-09-10	463	1 - 11	3.5	1.3	31	1 - 4	1.8	0.8	-	-	-	-
36 ³³⁷	F	166	SP	02-02-11	175	1 - 10	3.3	1.4	15	1 - 3	1.7	0.6	-	-	-	-
40	F	153	SP	19-08-10	137	1 - 10	3.1	1.4	61	1 - 3	1.8	0.7	-	-	-	-
43	F	188	SP	19-08-10	99	1 - 16	3.3	2.5	74	1 - 4	1.8	0.8	-	-	-	-
50	F	170	ML	28-07-11	-	-	-	-	1	2	-	-	12	8 - 11	9.6	0.9
51	F	146	SP	11-04-11	90	1 - 11	4.0	2.2	-	-	-	-	-	-	-	-
68	F	196	SP	09-04-10	230	1 - 5	2.7	0.9	926	1 - 5	1.8	1.0	-	-	-	-
68	F	196	SP	09-03-10	68	1 - 4	1.7	1.0	26	1 - 3	2.0	0.7	-	-	-	-
82	F	187	ML	26-09-10	-	-	-	-	321	1 - 4	2.0	0.6	-	-	-	-
85	F	133	SP	26-08-10	82	1 - 9	3.4	1.8	33	1 - 3	1.5	0.6	-	-	-	-
90	F	126	SP	19-08-10	81	1 - 7	2.2	1.2	51	1 - 3	1.6	0.8	-	-	-	-
107	F	177	ML	31-05-11	-	-	-	-	-	-	-	-	1	7	-	-
108	F	175	NP	01-07-11	-	-	-	-	61	1 - 3	1.7	0.7	-	-	-	-

Table A4.8. Summary of total number, size range, mean size (with SD) for whole *Melanoides tuberculata* and *Heleobops bermudensis* obtained from the faecal samples of three male diamondback terrapins.

ID	Sex	SCL	Site	Date	<i>Melanoides</i> N	<i>Melanoides</i> size range TH (mm)	<i>Melanoides</i> mean TH (mm)	<i>Melanoides</i> SD	<i>Heleobops</i> N	<i>Heleobops</i> size range TH (mm)	<i>Heleobops</i> mean TH (mm)	<i>Heleobops</i> SD	<i>Melampus</i> N	<i>Melampus</i> size range TH (mm)	<i>Melampus</i> mean TH (mm)	<i>Melampus</i> SD
16	M	122	SP	01-05-10	-	-	-	-	21	1 - 2	1.6	0.5	-	-	-	-
75	M	114	SP	11-03-10	58	1 - 5	1.9	0.8	50	1 - 3	1.3	0.5	-	-	-	-
75	M	114	SP	29-08-10	41	1 - 7	2.2	1.2	79	1 - 3	1.6	0.6	-	-	-	-

Table A4.9. Summary of total number, size range, mean size (with SD) for whole *Melanoides tuberculata* and *Heleobops bermudensis* obtained from the faecal samples of three juvenile diamondback terrapins.

ID	Sex	SCL	Site	Date	<i>Melanoides</i> N	<i>Melanoides</i> size range TH (mm)	<i>Melanoides</i> mean TH (mm)	<i>Melanoides</i> SD	<i>Heleobops</i> N	<i>Heleobops</i> size range TH (mm)	<i>Heleobops</i> mean TH (mm)	<i>Heleobops</i> SD	<i>Melampus</i> N	<i>Melampus</i> size range TH (mm)	<i>Melampus</i> mean TH (mm)	<i>Melampus</i> SD
88	Juv	107	SP	05-04-10	13	1 - 3	2.0	0.6	34	1 - 2	1.1	0.4	-	-	-	-
98	Juv	101	ML	25-08-10	-	-	-	-	15	1 - 3	1.3	0.6	-	-	-	-
102	Juv	97	SP	07-04-11	-	-	-	-	28	1 - 2	1.2	0.4	-	-	-	-

Table A4.10. Summary of total number, size range, mean size (with SD) for whole *Melanoides tuberculata* and *Heleobops bermudensis* obtained from the faecal samples of five neonate diamondback terrapins.

ID	Sex	SCL	Site	Date	<i>Melanoides</i> N	<i>Melanoides</i> size range TH (mm)	<i>Melanoides</i> mean TH (mm)	<i>Melanoides</i> SD	<i>Heleobops</i> N	<i>Heleobops</i> size range TH (mm)	<i>Heleobops</i> mean TH (mm)	<i>Heleobops</i> SD	<i>Melampus</i> N	<i>Melampus</i> size range TH (mm)	<i>Melampus</i> mean TH (mm)	<i>Melampus</i> SD
H5	neonate	31	ML	26-04-11	-	-	-	-	4	1 - 2	1.5	0.6	-	-	-	-
H6	neonate	34.9	SP	02-09-10	-	-	-	-	4	1	1.0	0.0	-	-	-	-
H6	neonate	34	SP	25-04-11	-	-	-	-	6	1 - 2	1.2	0.4	-	-	-	-
H3	neonate	34.1	ML	26-04-11	-	-	-	-	14	1 - 2	1.1	0.4	-	-	-	-
H5	neonate	34.6	ML	28-04-11	-	-	-	-	12	1 - 2	1.2	0.4	-	-	-	-

Table A4.11. Pooled summaries of the total numbers, size ranges, mean sizes and standard deviations from the mean for whole *Melanoides tuberculata*, *Heleobops bermudensis* and *Melampus coffeus* obtained from 28 diamondback terrapin faecal samples. Note: the blue text indicates estimated numbers.

	<i>Melanoides</i> N	<i>Melanoides</i> size range TH (mm)	<i>Melanoides</i> mean TH (mm)	<i>Melanoides</i> SD	<i>Heleobops</i> N	<i>Heleobops</i> size range TH (mm)	<i>Heleobops</i> mean TH (mm)	<i>Heleobops</i> SD	<i>Melampus</i> N	<i>Melampus</i> size range TH (mm)	<i>Melampus</i> mean TH (mm)	<i>Melampus</i> SD
All pooled	2224	1 - 18	3.2	2.1	1910	1 - 5	1.7	0.7	13	7 - 11	9.4	1.1
Female pooled	2112	1 - 18	3.3	2.1	1643	1 - 5	1.8	0.8	13	7 - 11	9.4	1.1
Male pooled	99	1 - 7	2.1	1.0	150	1 - 3	1.5	0.6	-	-	-	-
Juvenile pooled	13	1 - 3	2	0.6	77	1 - 3	1.2	0.4	-	-	-	-
Neonate pooled	0	-	-	-	40	1 - 2	1.2	0.4	-	-	-	-

Table A4.12. Total number of the various sizes of *Heleobops bermudensis*, *Melanoides tuberculata* and *Melampus coffeus* found within the pooled faecal samples of female diamondback terrapins.

Shell height (mm)	<i>Heleobops</i> (n)	<i>Melanoides</i> (n)	<i>Melampus</i> (n)
1	289	117	-
2	423	236	-
3	97	231	-
4	19	166	-
5	5	77	-
6	-	38	-
7	-	20	1
8	-	8	1
9	-	7	5
10	-	8	4
11	-	3	2
12	-	1	-
13	-	3	-
14	-	0	-
15	-	1	-
16	-	1	-
17	-	1	-
18	-	1	-

Table A4.13. Total number of the various sizes of *Heleobops bermudensis* and *Melanoides tuberculata* found within the pooled faecal samples of male diamondback terrapins.

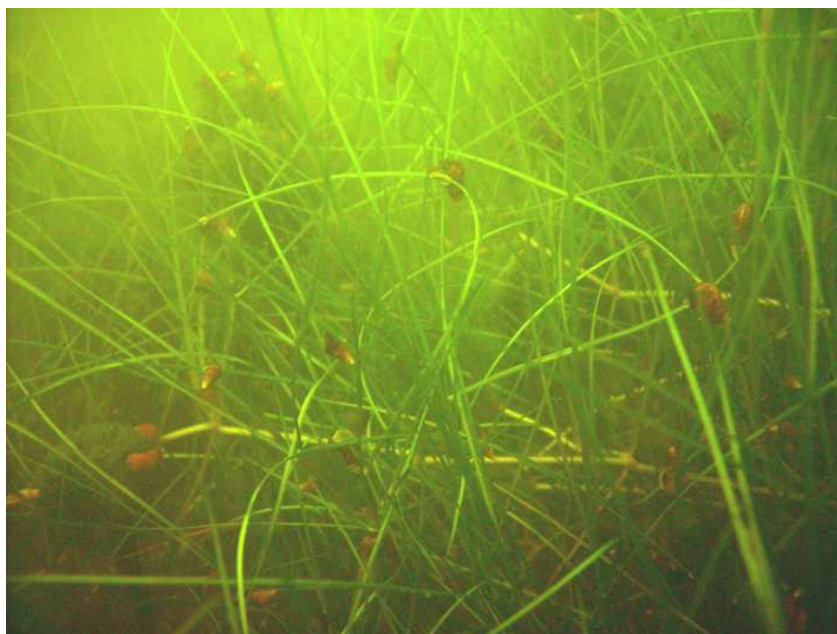
Shell height (mm)	<i>Heleobops</i> (n)	<i>Melanoides</i> (n)
1	9	25
2	62	55
3	6	12
4	-	3
5	-	3
6	-	0
7	-	1

Table A4.14. Total number of the various sizes of *Heleobops bermudensis* and *Melanoides tuberculata* found within the pooled faecal samples of juvenile diamondback terrapins.

Shell height (mm)	<i>Heleobops</i> (n)	<i>Melanoides</i> (n)
1	63	2
2	13	9
3	1	2

Table A4.15. Total number of the various sizes of *Heleobops bermudensis* found within the pooled faecal samples of neonate diamondback terrapins.

Shell height (mm)	<i>Heleobops</i> (n)
1	33
2	7



Source: Mark Outerbridge

Figure A4.6. Photograph showing *Batillaria minima* snails within a bed of widgeon grass (*Ruppia maritima*) at Mangrove Lake.

Table A4.16. Summary of gastropod abundance (number of snails m⁻²) at each site (n=10) along Transect 1 in Mangrove Lake.

Site No.	Lat.	Long.	Description	<i>Batillaria</i>	<i>Heleobops</i>
1-1	32.32646	64.70634	sediment	8	36
1-2	32.32640	64.70703	sediment	4	40
1-3	32.32643	64.70716	sediment	0	0
1-4	32.32628	64.70764	sediment	28	24
1-5	32.32610	64.70822	sediment	0	0
1-6	32.32587	64.70885	sediment	0	16
1-7	32.32575	64.70914	widgeon grass	56	492
1-8	32.32559	64.70954	widgeon grass	56	252
1-9	32.32560	64.71004	sediment	0	16
1-10	32.32533	64.71033	sediment	0	192

Table A4.17. Summary of gastropod abundance (number of snails m⁻²) at each site (n=10) along Transect 2 in Mangrove Lake.

Site No.	Lat.	Long.	Description	<i>Batillaria</i>	<i>Heleobops</i>
2-1	32.32361	64.70906	sediment	0	0
2-2	32.32394	64.70922	sediment	0	4
2-3	32.32427	64.70932	sediment	4	0
2-4	32.32459	64.70942	sediment	0	4
2-5	32.32483	64.70950	sediment	0	4
2-6	32.32512	64.70963	widgeon grass	0	772
2-7	32.32537	64.70971	widgeon grass	20	380
2-8	32.32566	64.70973	sediment	0	60
2-9	32.32590	64.70983	sediment	0	16
2-10	32.32633	64.70993	leaf litter	8	12

Table A4.18. Summary of gastropod abundance (number of snails m⁻²) at each site (n=10) along Transect 3 in South Pond.

Site No.	Lat.	Long.	Description	<i>Melanoides</i>	<i>Heleobops</i>
3-1	32.32890	64.70567	sediment	4	0
3-2	32.32893	64.70545	sediment	16	4
3-3	32.32913	64.70515	sediment	16	0
3-4	32.32932	64.70514	sediment	16	0
3-5	32.32948	64.70523	sediment	12	0
3-6	32.32961	64.70536	sediment	20	0
3-7	32.32966	64.70555	sediment	20	0
3-8	32.32959	64.70571	sediment	12	0
3-9	32.32941	64.70575	sediment	4	0
3-10	32.32917	64.70575	sediment	12	0

Table A4.19. Summary of gastropod and crustacean abundance (m^{-2}) at each sand, rock, and gravel site ($n=4$) within Mangrove Lake.

Site No.	Lat.	Long.	Description	<i>Batillaria minima</i>	<i>Alpheus armillatus</i>
A	32.326719	64.708638	Sand and gravel	2128	0
B	32.327237	64.707204	Rocks	2000	48
C	32.325764	64.710715	Rocks	3504	32
D	32.324942	64.717072	Rocks	6752	0



Source: Mark Outerbridge

Figure A4.7. Photograph showing a typical view within the mangrove swamp habitat surrounding Mangrove Lake; the swamp floor is dominated by leaf litter.

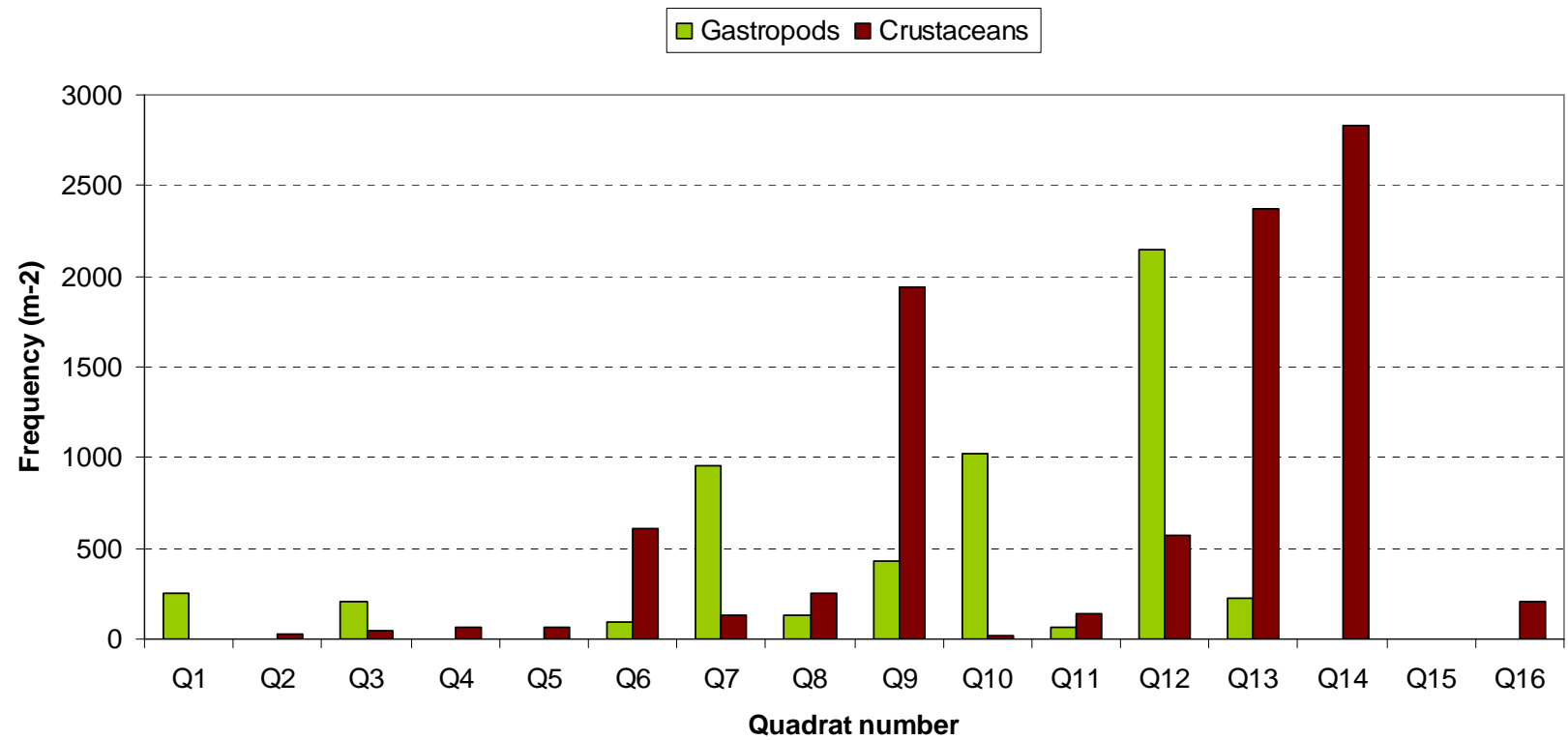


Figure A4.8. Frequency of occurrence of gastropods and crustaceans found within the mangrove swamp quadrat surveys (n=16) performed around Mangrove Lake.



Source: Mark Outerbridge

Figure A4.9. Photograph showing a typical view of the saw-grass marsh habitat at the centre of South Pond.

Table A4.21. Biotic summary of the quadrat surveys (n=4) performed within the saw-grass marsh habitat at the centre of South Pond. Note: results standardized on values m^{-2}

Site No.	Lat.	Long.	No. of shoot bundles	<i>Heleobops bermudensis</i>	Millipede sp.	UnID spiders
Q1	32.3290	64.7054	16	176	48	64
Q2	32.3292	64.7055	48	272	32	80
Q3	32.2937	64.7053	32	192	64	48
Q4	32.3295	64.7055	32	192	16	32

Table A4.22. Summary of the published studies describing diamondback terrapin diet throughout the North American range. (Note: the asterix indicates studies that used faecal analyses).

Sample size	Region	Diet composition	Literature source
14 mixed gender and age	North Carolina (wild)	Gastropods Crustaceans Annelid worms Algae, grass	Coker, 1906
unknown	North Carolina (captive)	Gastropods Crustaceans Bivalves Fish	Hildebrand, 1929
875 hatchlings	North Carolina (captive)	Gastropods Crustaceans Bivalves Fish, liver, beef	Allen & Littleford, 1955
70 mixed gender and age	Louisiana (wild)	Gastropods Bivalves	Cagle, 1952*
unknown	“East Coast” (wild)	Crustaceans Mollusks Arthropods	Carr, 1952*
unknown	Delaware (wild)	Bivalves	Hurd et al., 1979*
At least 65 hatchlings	Virginia and Florida (captive)	Bivalves Fish Squid	Dunson, 1985
11 adults (male)	“East Coast” (captive)	Gastropods Crustaceans Bivalves Fish	Davenport et al., 1992 Bels et al., 1998
unknown	Virginia (wild)	Gastropods	Mitchell, 1994
294 adults	South Carolina (wild)	Gastropods Crustaceans Bivalves	Tucker et al., 1992*;1995*
68 mixed gender and age	North Carolina (wild)	Gastropods Crustaceans Bivalves Fish	Spivey, 1998*
unknown	Maryland (wild)	Bivalves	Roosenburg et al., 1999
1 adult (female)	New Jersey (wild)	Arthropod larvae	Ehret & Werner, 2004

Table A4.22. (continued) Summary of the published studies describing diamondback terrapin diet throughout the North American range. (Note: the asterix indicates studies that used faecal analyses).

Sample size	Region	Diet composition	Literature source
114 hatchlings	New York (wild)	Gastropods Crustaceans Arthropods Arachnids Foraminifera	King, 2007*
4 hatchlings	New York (captive)	Annelid worms Mollusks Crustaceans Arthropods Fish	Kinneary, 2008
99 mixed gender and age	New York (wild)	Gastropods Crustaceans Bivalves Fish Plants	Petrochic, 2009*
81 mixed gender and age	Florida (wild)	Gastropods Crustaceans Bivalves Fish Unid animal tissue	Butler et al., 2000*; 2012*
129 adults (female)	New York (wild)	Gastropods Crustaceans Bivalves Annelid worms Bryozoans Arthropods Algae, plants Unid animal tissue	Erazmus, 2012*

Appendix 5: Supplementary Material to Chapter 5



Figure A5.1. Aerial photograph of the sand bunkers on the fifth hole of the Mid Ocean golf course showing the alphanumeric notations assigned to each bunker.



Figure A5.2. Aerial photograph of the sand bunkers on the sixth hole of the Mid Ocean golf course showing the alphanumeric notations assigned to each bunker.

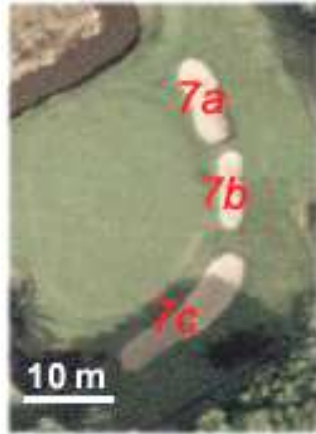


Figure A5.3. Aerial photograph of the sand bunkers on the seventh hole of the Mid Ocean golf course showing the alphanumeric notations assigned to each bunker.

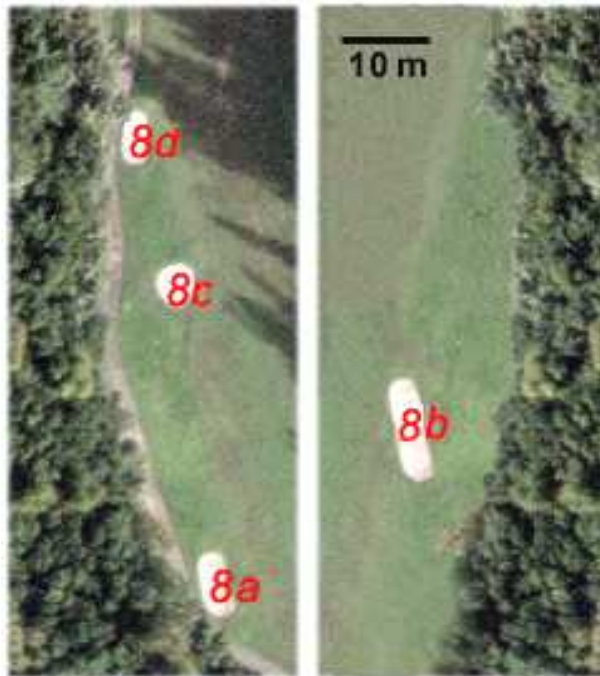


Figure A5.4. Aerial photograph of the sand bunkers on the fairway of the eighth hole of the Mid Ocean golf course showing the alphanumeric notations assigned to each bunker.

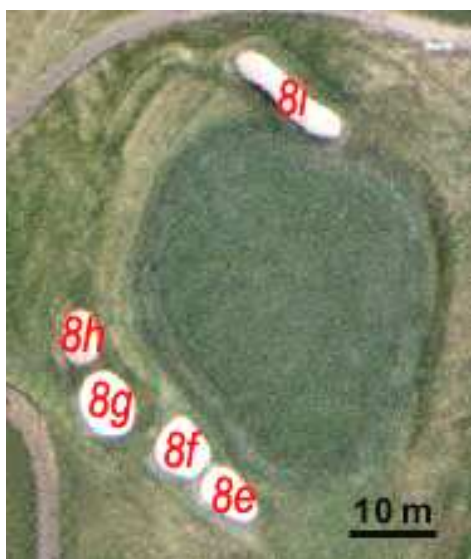


Figure A5.5. Aerial photograph of the sand bunkers surrounding the putting green on the eighth hole of the Mid Ocean golf course showing the alphanumeric notations assigned to each bunker.



Figure A5.6. Aerial photograph of the sand bunkers on the fairway (right) and surrounding the putting green (left) on the ninth hole of the Mid Ocean golf course showing the alphanumeric notations assigned to each bunker.

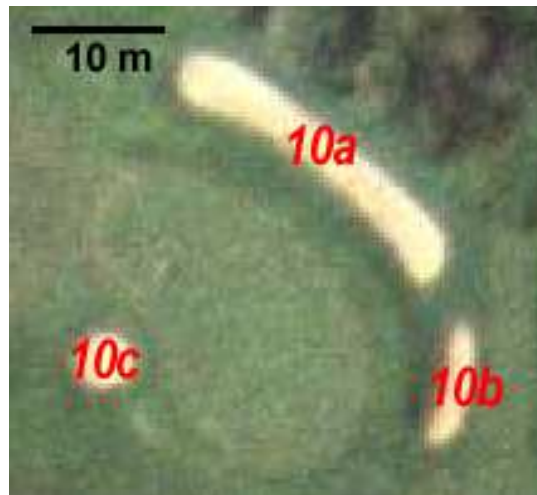


Figure A5.7. Aerial photograph of the sand bunkers on the tenth hole of the Mid Ocean golf course showing the alphanumeric notations assigned to each bunker.



Figure A5.8. Aerial photograph of the sand bunkers on the eleventh hole of the Mid Ocean golf course showing the alphanumeric notations assigned to each bunker.



Figure A5.9. Poster used to elicit help in reporting diamondback terrapin nesting activity on the Mid Ocean golf course.



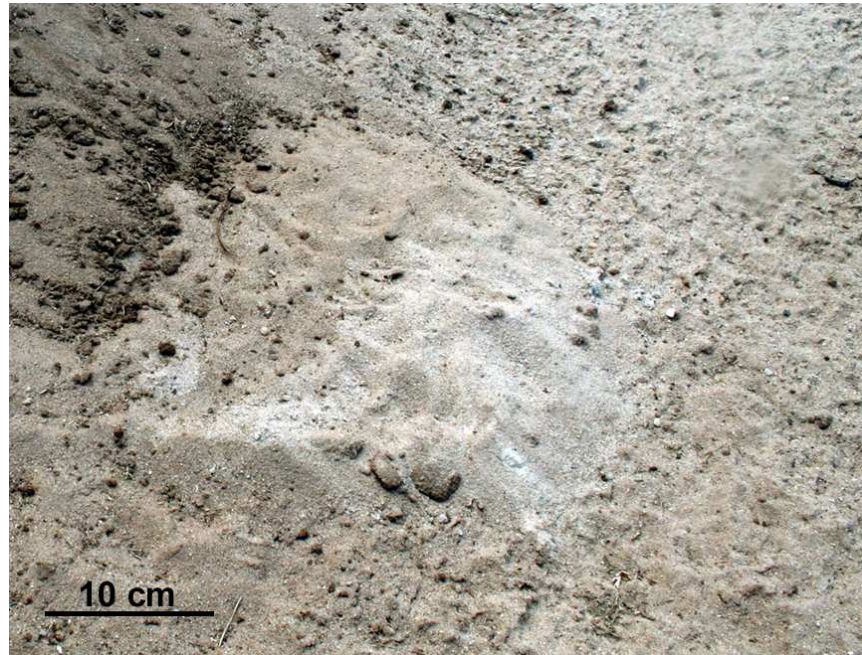
Source: Mark Outerbridge

Figure A5.10. Fresh diamondback terrapin tracks in a sand bunker on the Mid Ocean golf course.



Source: Mark Outerbridge

Figure A5.11. Abandoned diamondback terrapin nest.



Source: Mark Outerbridge

Figure A5.12. Mound of sand showing characteristic evidence of a diamondback terrapin nesting event.



Source: Mark Outerbridge

Figure A5.13. Sand bunker on the Mid Ocean golf course showing the metal stakes and blue surveyors tape that marked the locations of diamondback terrapin nests.



Source: Mark Outerbridge

Figure A5.14. Sand bunker on the Mid Ocean golf course showing the cages that were employed to determine the post-hatching residency periods for Bermuda's diamondback terrapins.


Diamondback Terrapin Nest Survey Form

Date: _____ Time: _____ Location: _____

Recorders: _____

Nest Data

Location: _____



Coordinates of nest:

Lat: _____

Long: _____

Relocated eggs? _____

Predator exclusion cage? _____

Temp logger deployed? _____ Date: _____

Date retrieved: _____

Nest Dimensions

Max width: _____

Max temp: _____

Max depth: _____

Min temp: _____

Depth of top egg: _____

Avg temp: _____

of samples _____

Egg Data

No. eggs laid: _____

Egg length (mm):

(1) (2) (3) (4) (5) (6) (7) (8) (9) (10)

Egg width (mm):

(1) (2) (3) (4) (5) (6) (7) (8) (9) (10)

Egg weight (g):

(1) (2) (3) (4) (5) (6) (7) (8) (9) (10)

Nest Excavation Data

emerged young _____

live hatchlings still in pipped shell _____

empty shells _____

dead hatchlings still in pipped shell _____

live young in nest _____

eggs with no embryo _____

dead young in nest _____

unhatched partial embryos _____

predated eggs _____

Notes:

Figure A5.15. Diamondback terrapin survey form used during the 2009-2012 nesting surveys.



Figure A5.16. Nesting locations for Bermuda's diamondback terrapins in 2010 on the sixth and seventh holes at the Mid Ocean golf course (red dots represent nests with confirmed eggs; red dot with yellow centre represents a nest containing no eggs; yellow dots represent unsuccessful nesting attempts).



Figure A5.17. Nesting locations for Bermuda's diamondback terrapins in 2010 on the fifth hole at the Mid Ocean golf course (red dots represent nests with confirmed eggs; red dot with yellow centre represents a nest containing no eggs; yellow dots represent unsuccessful nesting attempts).

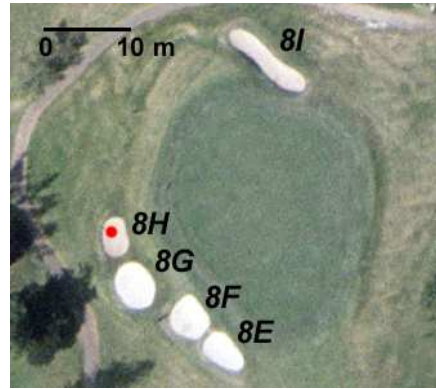


Figure A5.18. Nesting location for Bermuda's diamondback terrapins in 2010 on the eighth hole at the Mid Ocean golf course.



Figure A5.19. Nesting location for Bermuda's diamondback terrapins in 2010 on the eleventh hole at the Mid Ocean golf course.



Figure A5.20. Nesting locations for Bermuda's diamondback terrapins in 2011 on the fifth hole at the Mid Ocean golf course (red dots represent nests with confirmed eggs; red dot with yellow centre represents a nest containing no eggs; yellow dots represent unsuccessful nesting attempts).



Figure A5.21. Nesting locations for Bermuda's diamondback terrapins in 2011 on the sixth and seventh holes at the Mid Ocean golf course (red dots represent nests with confirmed eggs; yellow dots represent unsuccessful nesting attempts).



Figure A5.22. Ancillary nesting locations reported by members of the public for Bermuda's diamondback terrapins in 2010 and 2011 (red dots represent nests with confirmed eggs; yellow dots represent unsuccessful nesting attempts; A=Mangrove Lake, B=Trott's Pond, C=South Pond, D=North Pond, E=Compston's Pond).

Table A5.1. Sand bunker dimensions, number of nests containing terrapin eggs, and nesting density for the bunkers between the fifth and eleventh holes on the Mid Ocean golf course during the 2010 and 2011 nesting surveys.

Nest ID	Length (m)	Width (m)	Area (m ²)	2010 nest count	2011 nest count	2010 density (nests m ⁻²)	2011 density (nests m ⁻²)	2010 density (nests ha ⁻²)	2011 density (nests ha ⁻²)
5A	22.8	3.3	74.1	2	5	0.027	0.067	270.0	675.0
5B	15.8	3.7	58.4	0	0	0.000	0.000	0.0	0.0
5C	8.9	4.3	38.0	2	0	0.053	0.000	525.7	0.0
5D	32.8	5.1	165.6	18	31	0.109	0.187	1086.8	1871.8
6A	10.3	5.8	60.2	2	1	0.033	0.017	332.0	166.0
6B	17.1	3.7	62.9	9	10	0.143	0.159	1431.7	1590.7
6C	5.8	3.9	22.5	0	1	0.000	0.045	0.0	445.3
7A	14.0	4.9	68.3	15	19	0.220	0.278	2197.7	2783.8
7B	10.6	3.8	40.6	8	7	0.197	0.172	1969.3	1723.1
7C	19.6	4.6	89.7	7	9	0.078	0.100	780.8	1003.9
8A	7.6	4.6	35.0	0	0	0.000	0.000	0.0	0.0
8B	6.7	6.2	41.6	0	0	0.000	0.000	0.0	0.0
8C	9.1	4.0	36.6	0	0	0.000	0.000	0.0	0.0
8D	13.3	4.3	57.6	0	0	0.000	0.000	0.0	0.0
8E	6.1	4.0	24.4	0	0	0.000	0.000	0.0	0.0
8F	5.3	5.1	27.1	0	0	0.000	0.000	0.0	0.0
8G	6.5	5.3	34.2	0	0	0.000	0.000	0.0	0.0

Table A5.1. (continued) Sand bunker dimensions, number of nests containing terrapin eggs, and nesting density for the bunkers between the fifth and eleventh holes on the Mid Ocean golf course during the 2010 and 2011 nesting surveys.

Nest ID	Length (m)	Width (m)	Area (m ²)	2010 nest count	2011 nest count	2010 density (nests m ⁻²)	2011 density (nests m ⁻²)	2010 density (nests ha ⁻²)	2011 density (nests ha ⁻²)
8H	5.9	4.0	23.6	1	0	0.042	0.000	423.7	0.0
8I	9.4	3.3	30.7	0	0	0.000	0.000	0.0	0.0
9A	10.8	3.6	39.2	0	0	0.000	0.000	0.0	0.0
9B	11.6	3.3	38.4	0	0	0.000	0.000	0.0	0.0
9C	13.9	3.8	52.8	0	0	0.000	0.000	0.0	0.0
9D	4.8	4.0	18.9	0	0	0.000	0.000	0.0	0.0
9E	7.3	3.4	24.4	0	0	0.000	0.000	0.0	0.0
9F	3.7	3.4	12.3	0	0	0.000	0.000	0.0	0.0
9G	49.7	3.7	181.7	0	0	0.000	0.000	0.0	0.0
10A	32.6	4.4	144.1	0	0	0.000	0.000	0.0	0.0
10B	11.6	3.1	35.6	0	0	0.000	0.000	0.0	0.0
10C	5.9	4.3	25.2	0	0	0.000	0.000	0.0	0.0
11A	34.5	3.4	117.6	1	0	0.009	0.000	85.1	0.0
11B	11.6	5.0	58.0	0	0	0.000	0.000	0.0	0.0
11C	13.4	5.3	71.7	0	0	0.000	0.000	0.0	0.0
11D	16.7	3.8	62.8	0	0	0.000	0.000	0.0	0.0

Table A5.2. Summary of clutch data for nests monitored in 2009 (n=10).

Nest ID	Lay Date	No. eggs	Emerged hatchlings	Eggs with no embryo	Dead hatchlings found in nest chamber	Un-hatched embryos	Hatching success
7C1	9/5/2009	5	0	5	0	0	0 %
7A1	13/5/2009	9	3	4	0	2	33.3 %
slope behind 7C	14/5/2009	7	0	6	0	1	0 %
slope behind 7C	18/5/2009	8	0	8	0	0	0 %
slope b/w 6 & 7	21/5/2009	4	2	1	0	1	50 %
7C2	25/5/2009	6	0	6	0	0	0 %
slope behind 7C	1/6/2009	4	0	2	0	2	0 %
7A2	6/6/2009	4	4	0	0	0	100 %
7A3	4/6/ 2009	1	0	1	0	0	0 %
5D1	16/6/2009	2	0	2	0	0	0 %
TOTALS:		50	9	35	0	6	

Table A5.3. Summary of clutch data for nests monitored in 2010 (n=57).

Nest ID	Lay Date	No. eggs	Emerged hatchlings	Eggs with no embryo	Dead hatchlings found in nest chamber	Un-hatched embryos	Hatching success
5C1	22/3/2010	5	0	1	0	4	0 %
7B1	9/4/2010	3	0	3	0	0	0 %
6B1	17/4/2010	10	2	5	1	2	20 %
7A1	18/4/2010	4	3	0	0	1	75 %
5D1	19/4/2010	3	0	3	0	0	0 %
7B2	23/4/2010	10	0	8	0	2	0 %
5D2	27/4/2010	8	3	3	0	2	37.5 %
7A2	1/5/2010	6	6	0	0	0	100 %
7A3	1/5/2010	1	1	0	0	0	100 %
7B3	2/5/2010	6	2	3	1	0	33.3 %
5D	3/5/2010	0	0	0	0	0	0 %
7C	4/5/2010	8	0	8	0	0	0 %
5D3	4/5/2010	0	0	0	0	0	0 %
7C1	8/5/2010	7	1	2	1	3	14.3 %
7C2	10/5/2010	6	0	6	0	0	0 %
7A4	10/5/2010	7	2	4	0	1	28.6 %
6B2	20/5/2010	7	0	7	0	0	0 %
5A1	22/5/2010	6	3	1	0	1	50 %
5D4	22/5/2010	1	0	1	0	0	0 %
7A5	22/5/2010	8	3	4	0	1	37.5 %

Table A5.3. (continued) Summary of clutch data for nests monitored in 2010 (n=57).

Nest ID	Lay Date	No. eggs	Emerged hatchlings	Eggs with no embryo	Dead hatchlings found in nest chamber	Un-hatched embryos	Hatching success
7A6	26/5/2010	5	3	1	1	0	60 %
6B3	26/5/2010	2	1	0	0	1	50 %
5D5	26/5/2010	8	0	8	0	0	0 %
5D6	27/5/2010	6	2	1	0	3	33.3 %
6B4	27/5/2010	9	3	6	0	0	33.3 %
6B5	7/6/2010	4	1	1	1	1	25 %
5A2	8/6/2010	1	0	1	0	0	0 %
5D7	9/6/2010	5	0	5	0	0	0 %
7B5	11/6/2010	6	1	4	1	0	16.7 %
6B6	18/6/2010	5	0	4	0	1	0 %
7A7	18/6/2010	5	2	1	0	2	40 %
6B7	20/6/2010	9	4	5	0	0	44.4 %
5D8	22/6/2010	6	2	1	2	1	33.3 %
11A1	22/6/2010	4	3	1	0	0	75 %
7B6	22/6/2010	6	5	0	1	0	83.3 %
7A8	23/6/2010	6	0	6	0	0	0 %
5D9	24/6/2010	5	0	5	0	0	0 %
5D10	25/6/2010	4	0	4	0	0	0 %
7A9	28/6/2010	6	4	1	0	1	66.7 %
7B7	29/6/2010	1	0	1	0	0	0 %

Table A5.3. (continued) Summary of clutch data for nests monitored in 2010 (n=57).

Nest ID	Lay Date	No. eggs	Emerged hatchlings	Eggs with no embryo	Dead hatchlings found in nest chamber	Un-hatched embryos	Hatching success
7C3	23/5/2010	5	0	5	0	0	0 %
7B4	23/5/2010	1	0	1	0	0	0 %
8H1	24/5/2010	1	0	0	0	1	0 %
7A10	3/7/2010	3	0	3	0	0	0 %
soil berm	unknown	6	0	6	0	0	0 %
6B8	9/7/2010	2	0	2	0	0	0 %
7C4	9/7/2010	2	0	2	0	0	0 %
7B8	13/7/2010	4	1	2	0	1	25 %
7A11	14/7/2010	6	0	5	0	1	0 %
7A12	17/7/2010	2	1	1	0	0	50 %
5D11	19/7/2010	1	0	1	0	0	0 %
6A1	22/7/2010	9	0	9	0	0	0 %
6B9	22/7/2010	6	1	4	0	1	16.7 %
5D12	27/7/2010	3	1	1	0	1	33.3 %
5D13	28/7/2010	2	0	1	0	1	0 %
7C5	9/8/2010	2	0	2	0	0	0 %
7A13	11/8/2010	4	0	4	0	0	0 %
TOTALS:		268	61	165	9	33	

Table A5.4. Summary of clutch data for nests monitored in 2011 (n=72).

(Asterix indicates that eggs were removed from the nest chamber for viability and eco-toxicological analyses).

Nest ID	Lay Date	No. eggs	Emerged hatchings	Eggs with no embryo	Dead hatchlings found in nest chamber	Un-hatched embryos	Hatching success
7A1	16/4/2011	8	0	8	0	0	0 %
7A2	19/4/2011	7	1	6	0	0	14.3 %
5D1	19/4/2011	7	0	7	0	0	0 %
7A3	20/4/2011	3	0	3	0	0	0 %
7B1	21/4/2011	9	0	9	0	0	0 %
7B2	21/4/2011	7	0	5	0	2	0 %
5D2	22/4/2011	8*	0	6	0	0	0 %
5D3	25/4/2011	5	0	5	0	0	0 %
5D4	25/4/2011	5		<i>Stakes disappeared; nest location lost</i>			
6B1	25/4/2011	8	0	8	0	0	0 %
6B2	26/4/2011	3	0	3	0	0	0 %
6B3	27/4/2011	5	4	0	1	0	80 %
5D5	29/4/2011	7	4	3	0	0	57.1 %
5A1	30/4/2011	3	0	3	0	0	0 %
6B4	5/5/2011	4	4	0	0	0	100 %
7A4	8/5/2011	8*	0	6	0	0	0 %
7B3	8/5/2011	1	0	1	0	0	0 %
7A5	11/5/2011	8*	0	6	0	0	0 %
5A2	11/5/2011	8*	0	6	0	0	0 %

Table A5.4. (continued) Summary of clutch data for nests monitored in 2011 (n=72).

(Asterix indicates that eggs were removed from the nest chamber for viability and eco-toxicological analyses).

Nest ID	Lay Date	No. eggs	Emerged hatchings	Eggs with no embryo	Dead hatchlings found in nest chamber	Un-hatched embryos	Hatching success
7B5	18/5/2011	7*	0	5	0	0	0 %
5A3	22/5/2011	0	0	0	0	0	0 %
5D7	22/5/2011	2	1	1	0	0	50 %
5D8	22/5/2011	4*	0	3	0	0	0 %
7 th slope	28/5/2011	8	8	0	0	0	100 %
7C1	28/5/2011	6	0	6	0	0	0 %
5A4	31/5/2011	7	7	0	0	0	100 %
5D9	31/5/2011	9*	0	7	0	1	0 %
7A6	31/5/2011	6	4	2	0	0	66.7 %
7 th road	1/6/2011	3	3	0	0	0	100 %
7C2	2/6/2011	5	0	5	0	0	0 %
7C3	2/6/2011	9	0	9	0	0	0 %
5D10	2/6/2011	8*	0	7	0	0	0 %
5D11	9/6/2011	6	0	6	0	0	0 %
7A7	12/6/2011	7	0	7	0	0	0 %
7A8	12/6/2011	3*	0	2	0	0	0 %
7A9	12/6/2011	5	0	4	1	0	20 %
5D12	12/6/2011	6	4	2	0	0	66.7 %
5D13	14/6/2011	7	0	6	0	1	0 %

Table A5.4. (continued) Summary of clutch data for nests monitored in 2011 (n=72).

(Asterix indicates that eggs were removed from the nest chamber for viability and eco-toxicological analyses).

Nest ID	Lay Date	No. eggs	Emerged hatchings	Eggs with no embryo	Dead hatchlings found in nest chamber	Un-hatched embryos	Hatching success
5D16	20/6/2011	6*	2	1	0	2	33.3 %
7B6	21/6/2011	3	1	0	2	0	33.3 %
5D17	21/6/2011	4			<i>Stakes disappeared; nest location lost</i>		
7A10	21/6/2011	7*	0	5	0	0	0 %
7C4	27/6/2011	5*	3	0	0	0	60 %
5A5	28/6/2011	5*	0	3	0	0	0 %
5D18	28/6/2011	4*	1	1	0	0	25 %
5D19	28/6/2011	4	0	4	0	0	0 %
6B5	28/6/2011	6	0	6	0	0	0 %
7A11	28/6/2011	6	1	5	0	0	16.7 %
7A12	28/6/2011	5	0	5	0	0	0 %
5D20	29/6/2011	5	0	5	0	0	0 %
5D21	30/6/2011	10	0	10	0	0	0 %
7A13	4/7/2011	6	0	6	0	0	0 %
5D22	5/7/2011	6	0	6	0	0	0 %
5D23	6/7/2011	6	0	5	1	0	0 %
7A14	6/7/2011	5	2	3	0	0	40 %
6B6	7/7/2011	9	6	3	0	0	66.7 %
5D24	7/7/2011	6	3	3	0	0	50 %

Table A5.4. (continued) Summary of clutch data for nests monitored in 2011 (n=72).

(Asterix indicates that eggs were removed from the nest chamber for viability and eco-toxicological analyses).

Nest ID	Lay Date	No. eggs	Emerged hatchings	Eggs with no embryo	Dead hatchlings found in nest chamber	Un-hatched embryos	Hatching success
5D6	16/5/2011	2	0	2	0	0	0 %
7B4	16/5/2011	7*	0	5	0	0	0 %
5D14	18/6/2011	2	0	2	0	0	0 %
5D15	18/6/2011	2	0	2	0	0	0 %
6B7	11/7/2011	8	0	8	0	0	0 %
5D25	17/7/2011	10	0	10	0	0	0 %
5D26	22/7/2011	1	0	1	0	0	0 %
5D27	1/8/2011	1	0	1	0	0	0 %
7A15	1/8/2011	6	2	2	2	0	33.3 %
7C5	1/8/2011	7	<i>Stakes disappeared; nest location lost</i>				
7A16	3/8/2011	7	0	7	0	0	0 %
5D28	4/8/2011	5	0	5	0	0	0 %
6B8	4/8/2011	3	2	1	0	0	66.7 %
6C1	15/8/2011	3	1	0	2	0	33.3 %
5D29	26/8/2011	3	0	3	0	0	0 %
TOTALS:		397[§]	64	277	9	6	

[§] note that 41 eggs must be deducted from total (16 due to marker loss and 25 due to removal for viability assessment and eco-toxicological analyses)

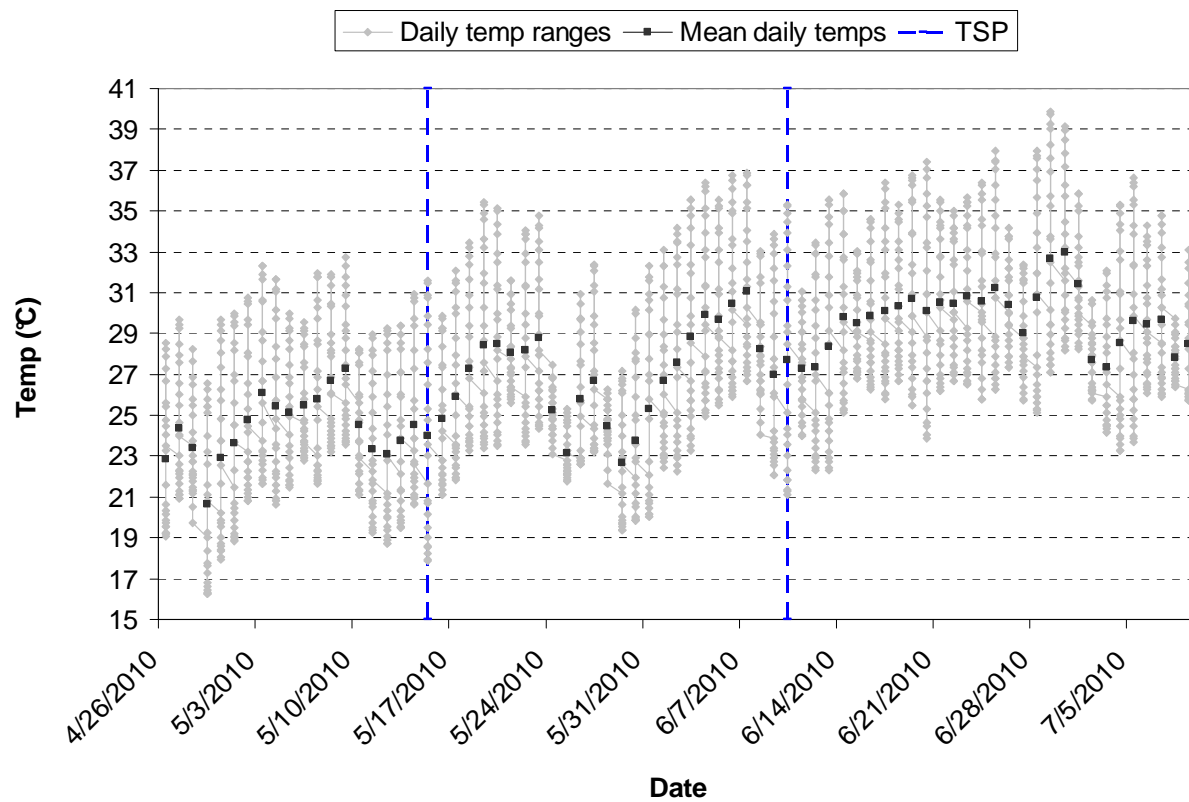


Figure A5.23. Temperature (daily ranges and means) at a depth of 10 cm for nest 7A1 on the Mid Ocean golf course between April 26th and July 9th 2010. TSP=Thermo-sensitive Period.

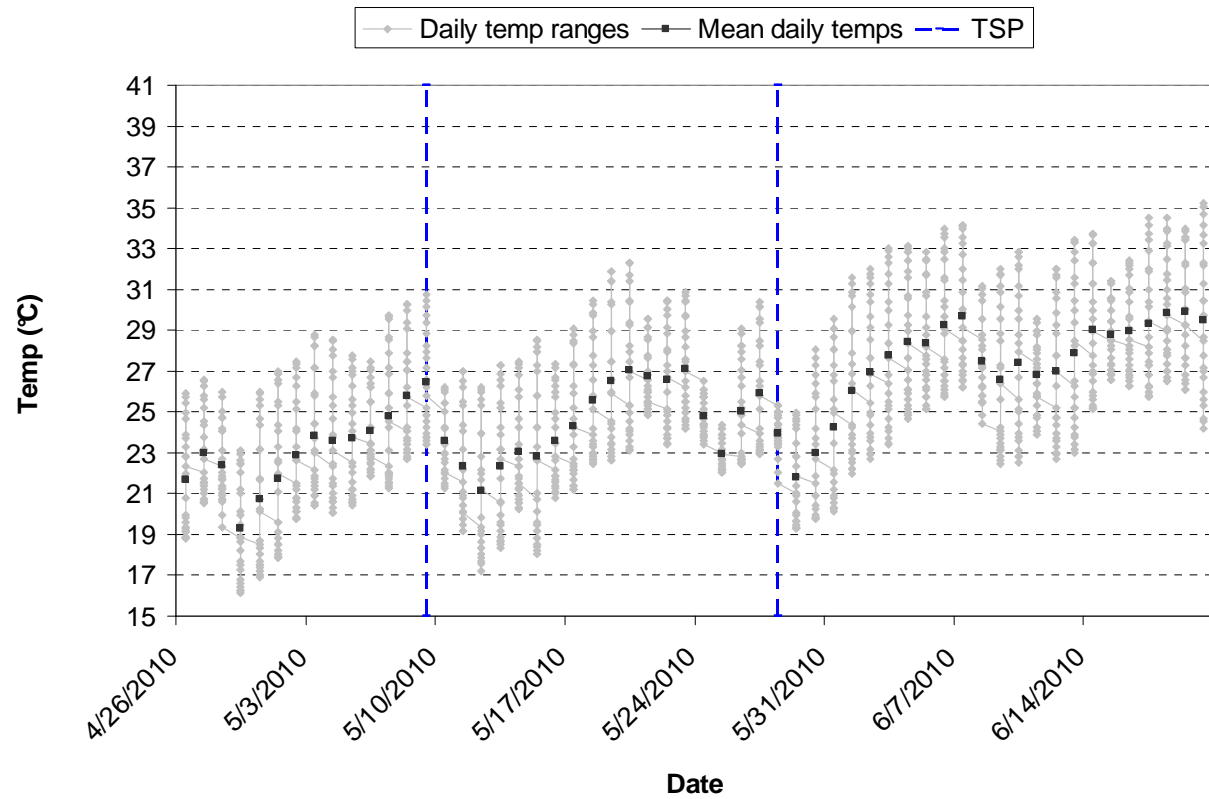


Figure A5.24. Temperature (daily ranges and means) at a depth of 10 cm for nest 5D1 on the Mid Ocean golf course between April 26th and June 20th 2010. TSP=Thermo-sensitive Period.

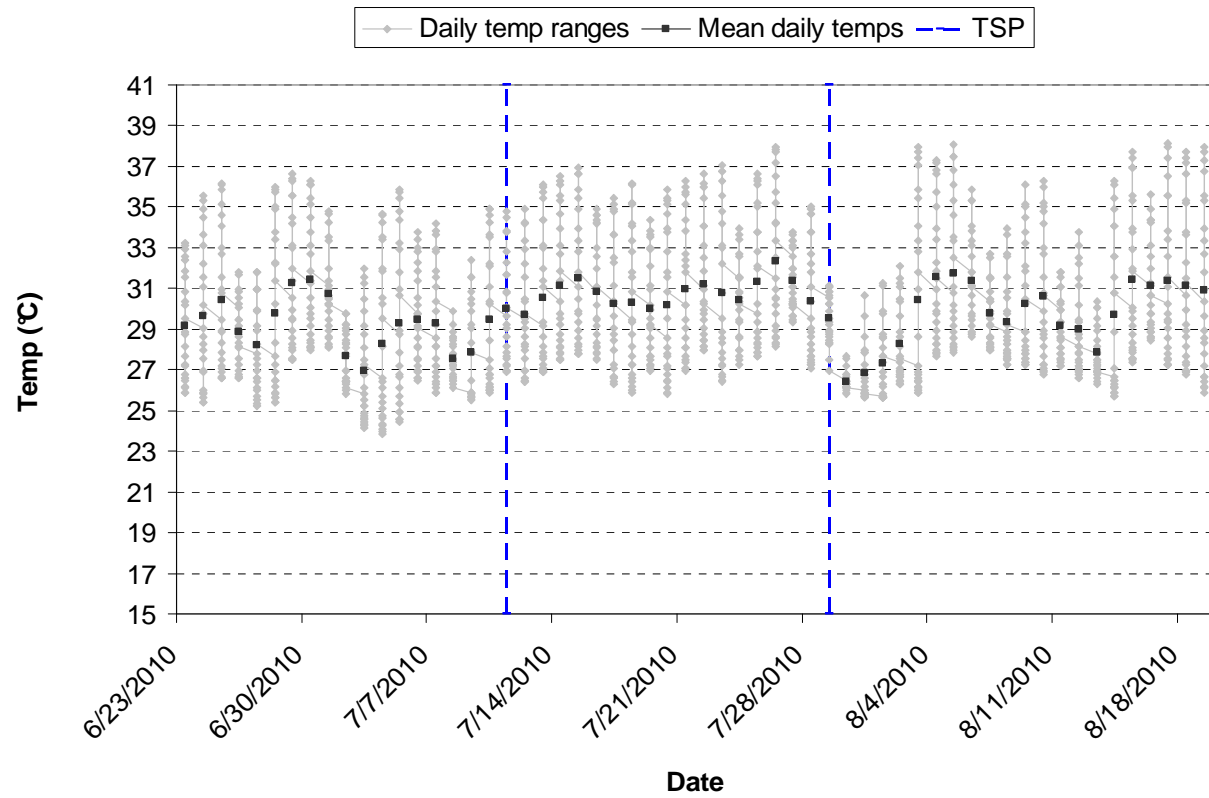


Figure A5.25. Temperature (daily ranges and means) at a depth of 10 cm for nest 11A1 on the Mid Ocean golf course between June 23rd and August 19th 2010. TSP=Thermo-sensitive Period.

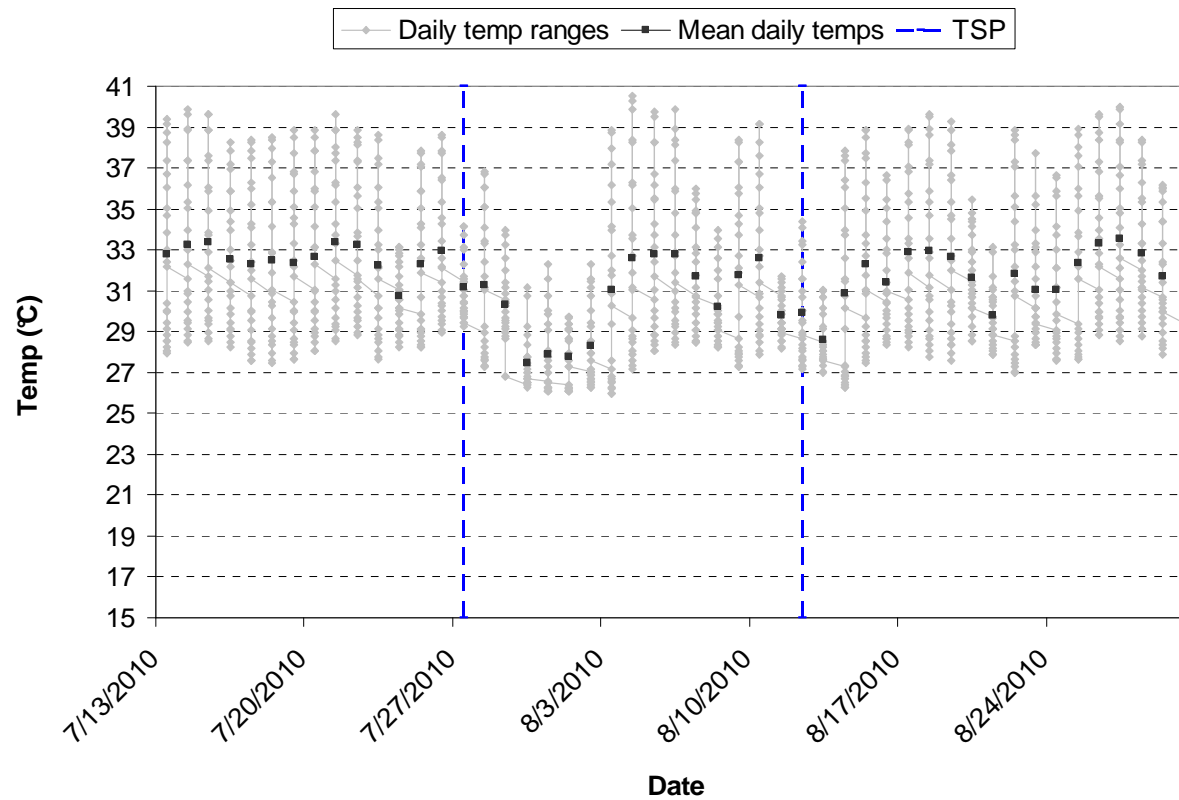


Figure A5.26. Temperature (daily ranges and means) at a depth of 10 cm for nest 7C4 on the Mid Ocean golf course between July 13th and August 30th 2010. TSP=Thermo-sensitive Period.

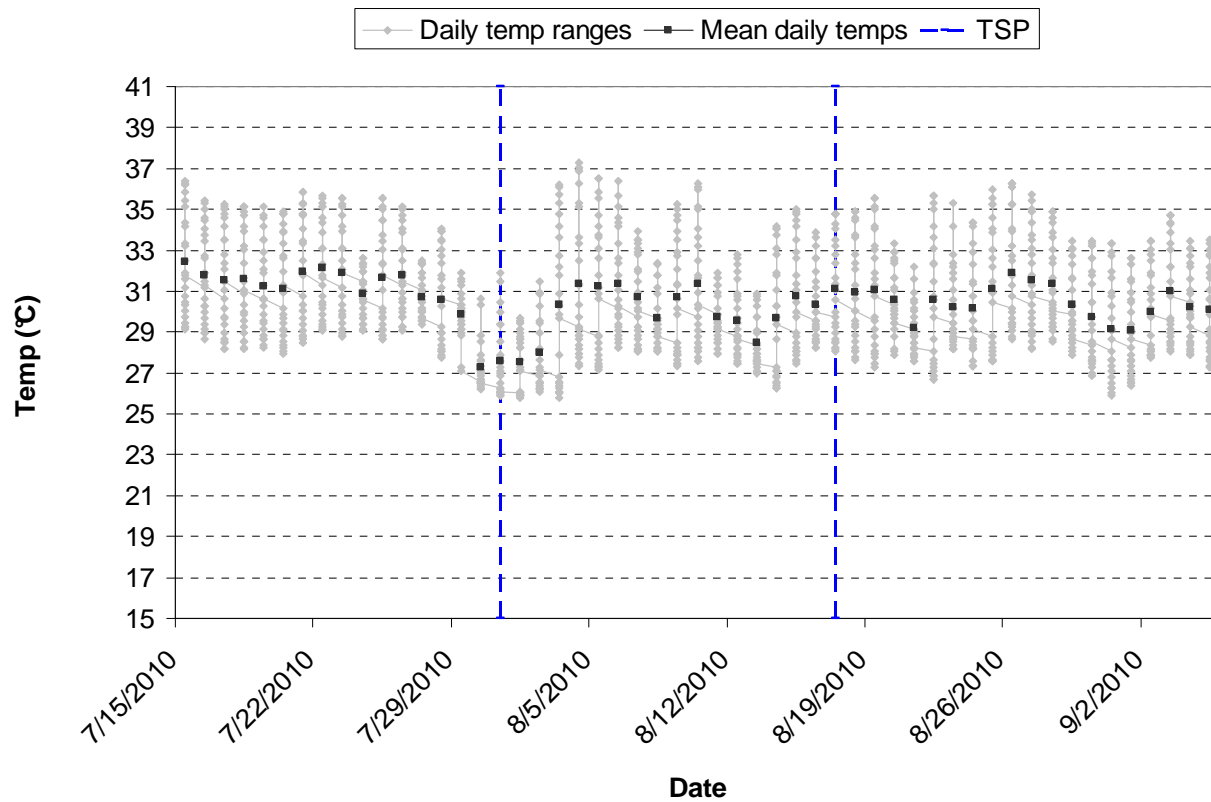


Figure A5.27. Temperature (daily ranges and means) at a depth of 10 cm for nest 7B8 on the Mid Ocean golf course between July 15th and September 5th 2010. TSP=Thermo-sensitive Period.

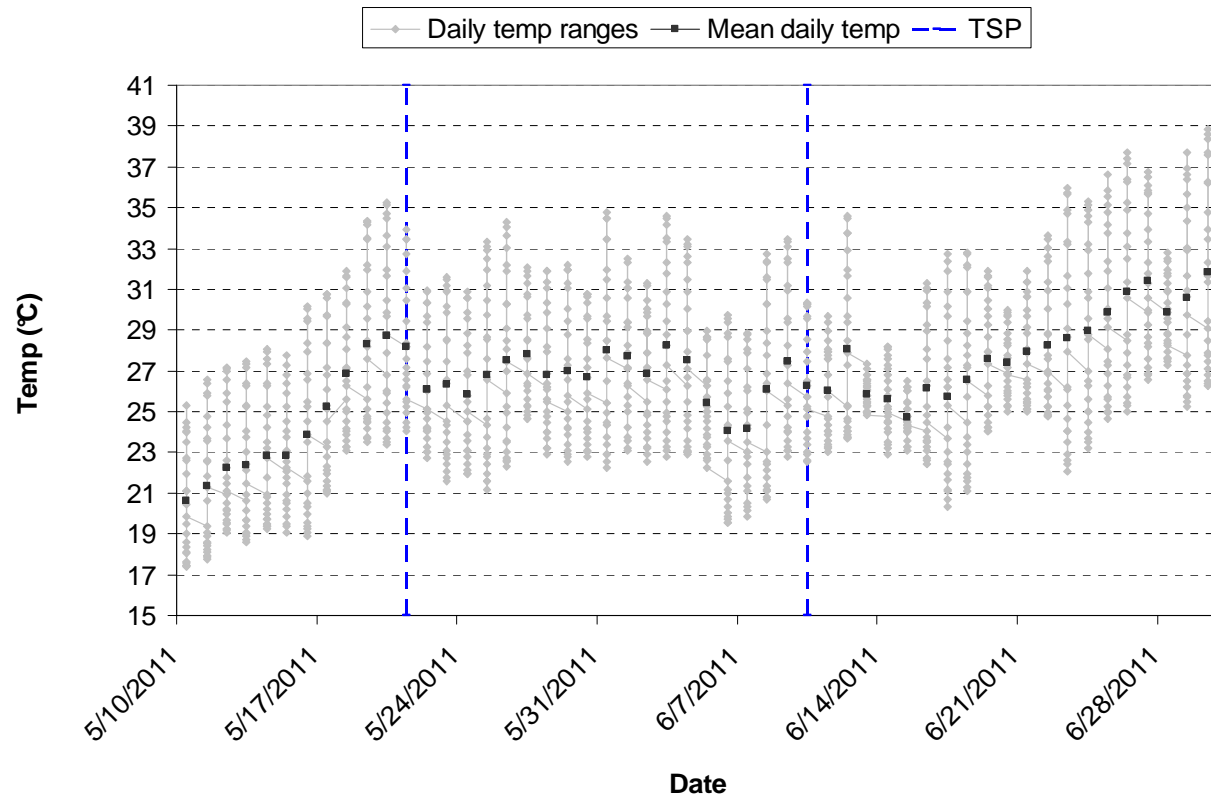


Figure A5.28. Temperature (daily ranges and means) at a depth of 10 cm for nest 5D5 on the Mid Ocean golf course between May 10th and June 30th 2011. TSP=Thermo-sensitive Period.

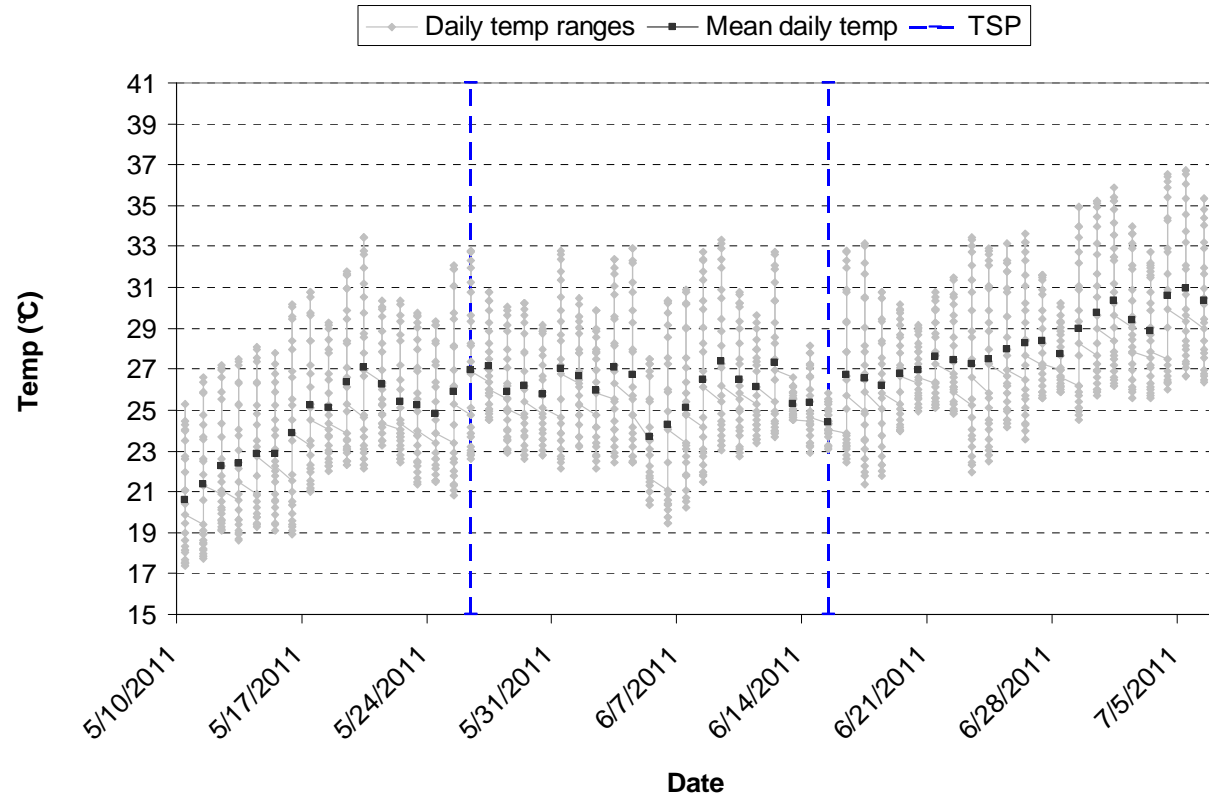


Figure A5.29. Temperature (daily ranges and means) at a depth of 10 cm for nest 6B4 on the Mid Ocean golf course between May 10th and July 6th 2011. TSP=Thermo-sensitive Period.

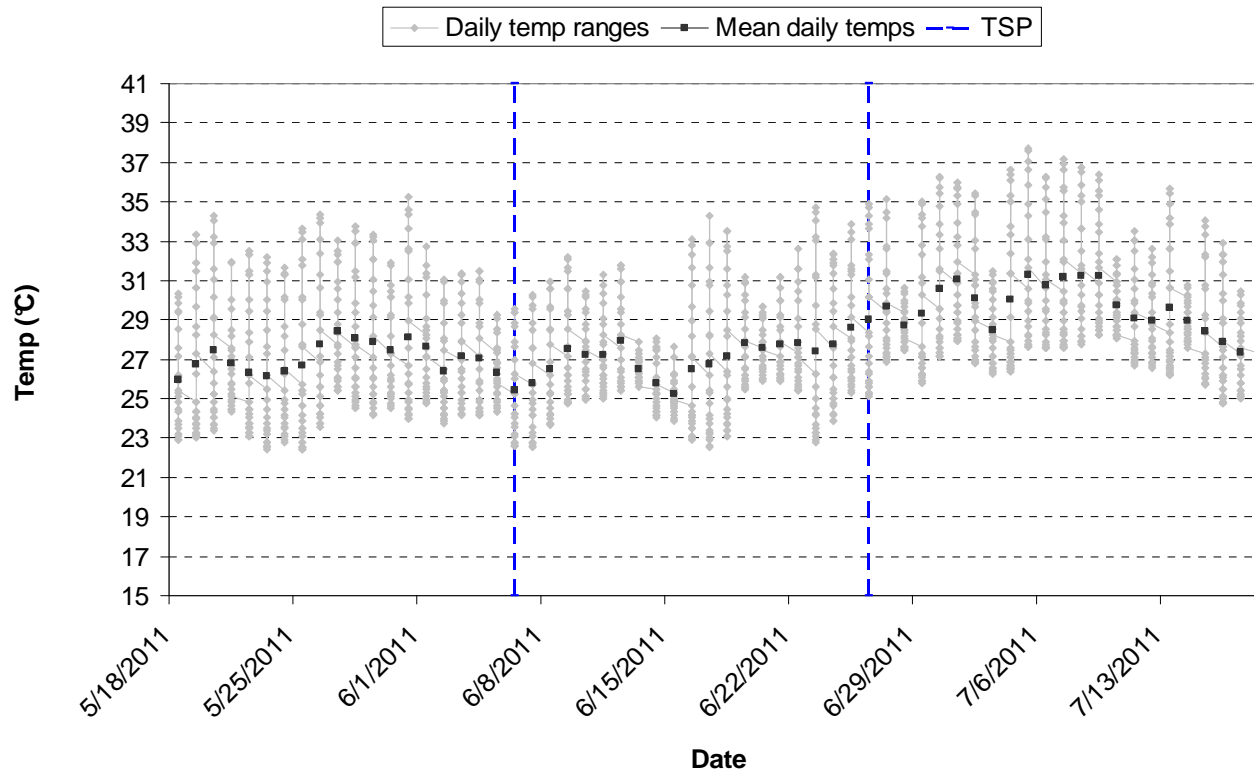


Figure A5.30. Temperature (daily ranges and means) at a depth of 10 cm for nest 7B4 on the Mid Ocean golf course between May 18th and July 26th 2011. TSP=Thermo-sensitive Period.

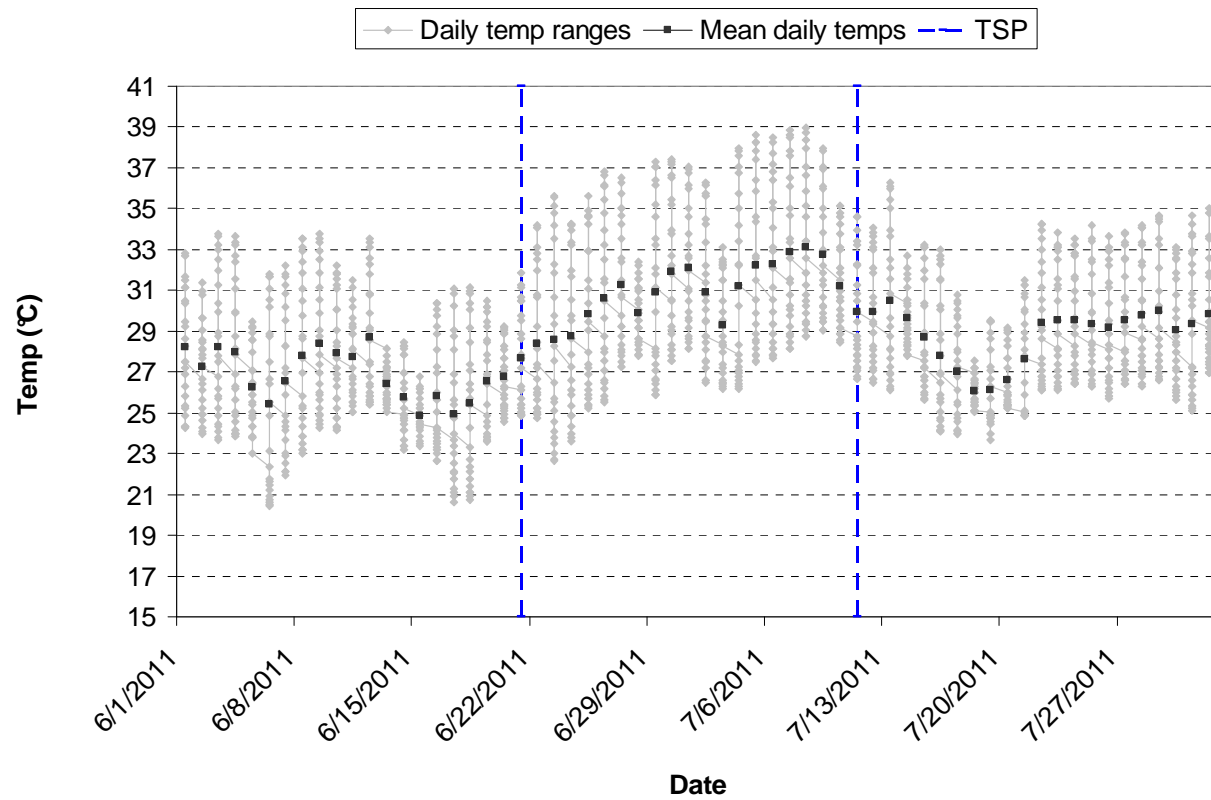


Figure A5.31. Temperature (daily ranges and means) at a depth of 10 cm for nest 5D9 on the Mid Ocean golf course between June 1st and August 1st 2011. TSP=Thermo-sensitive Period.

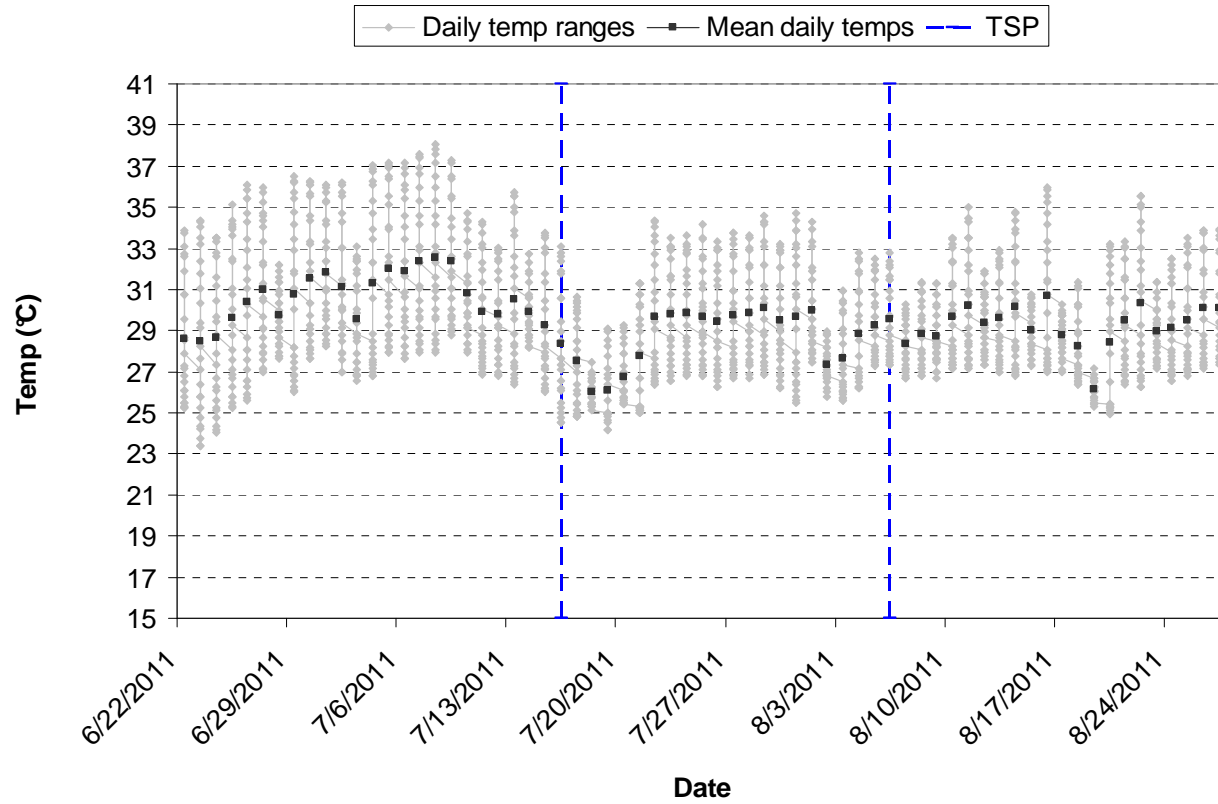


Figure A5.32. Temperature (daily ranges and means) at a depth of 10 cm for nest 5D14 on the Mid Ocean golf course between June 22nd and August 27th 2011. TSP=Thermo-sensitive Period.

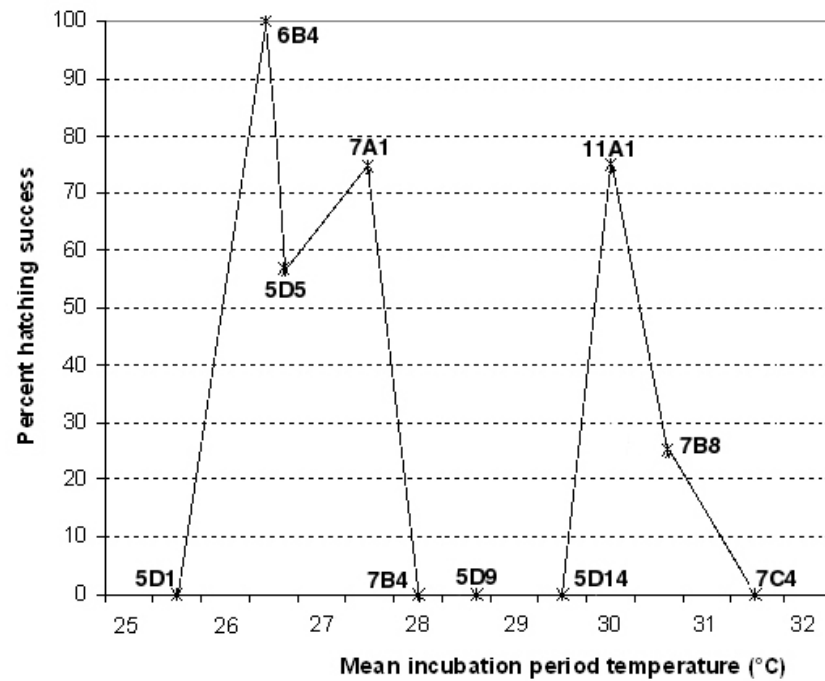


Figure A5.33. Graph showing percent hatching success plotted against mean temperature during the incubation period for ten nests monitored in 2010.

Appendix 6: Supplementary Material to Chapter 6

Table A6.1. Summary of the post-emergence movements for ten diamondback terrapin hatchlings in August 2010.

No. days after release	Distance moved on subsequent days (m)									
	# H1	# H2	# H3	# H4	# H5	# H6	# H7	# H8	# H9	# H10
1	2.1	3	3	3.5	2.5	6.6	13.2	12	5	2
2	4	0	15.2	0	0	0	0	0	0	0
3	0	0	0	0	0	0	0	8	0	0
4	0	0	0	0	0	0	0	0	0	0
5	0	0	0	0	0	0	0	0	0	0
6	0	0	1	0	0	0	0	2	0	0
7	0	0	0	0	0	0	0	0	0	0
8	0	0	0	0	0	0	0	0	0	0
9	0.6	0	0	0	0	0	0	0	0	0
10	0	0	0	0	0	0	0	0	0	0
11	0	0	0	0	13.7	0	0	0.15	0	0
12	0	0	0	0	0	0	0	0	0	0
13	0	0	0	0	0	0	0	0	0	0
14	0	0	0	0	0	0	0	0	0	0
15	0	0	0	0	0	0	0	0	0	0
16	0	0	4.3	0	0	0	0	0	0	0
17	0	0	0	0	0	0.3	0	0	0	0
18	0	0	0	0	0	0.75	0	0	0	0
19	0	0	0	0	0	0	0	0	0	0
20	0	0	0	0	0	0.15	0	0	0	0
21	0	0	0	0	0	0	0	0	0	0

Table A6.1. (continued) Summary of the post-emergence movements for ten diamondback terrapin hatchlings in August 2010.

No. days after release	Distance moved on subsequent days (m)									
	# H1	# H2	# H3	# H4	# H5	# H6	# H7	# H8	# H9	# H10
22	0	0	0	0	0	60	0	0	0	0
23	0	0	0	0	0	2.7	WS, RT	0	0	0
24	0	0	0	0	0	4		0	0	0
25	0	0	0	0	0	11		0	0	0
26	0	0	0	0	0	2.4		0	0	0
27	0	0	0	0	0	3.6		1.5	0	0
28	0	0	0	0	0	0		16.2	0	0
29	0	0	0	0	0	7		0	NS, RT	0
30	0	0	0	0	0	1.2		0		0
31	0	0	1.2	0	0	1.9		0		0
32	NS, RT	0	2.1	0	0	1.2		0		0
33		0	0	0	0	0		0		0
34		0	0	0	0	WS, RT		0		0
35		0	0.5	0	0			0		0
36		0	0	0	0			0		0
37		0	NS, LT	WS, RT	WS, RT			WS, RT		0
38		0	NS							0
39		0	NS							0
40		WS, RT	NS							WS, RT

Note:

WS = Weak transmitter signal

NS = No transmitter signal

RT = Removed terrapin hatchling

LT = Lost terrapin hatchling

Light grey = Range of battery life according to manufacturer

Dark grey = Estimated end of battery life according to manufacturer

Table A6.2. Summary of the post-emergence movements for ten diamondback terrapin hatchlings in March-April 2011.

No. days after release	Distance moved on subsequent days (m)									
	# H11	# H12	# H13	# H14	# H15	# H16	# H17	# H18	# H19	# H20
1	5.8	8.2	5.8	18.6	5.5	22.6	2	15.9	3	35
2	0	0	0	0	0.5	1.2	NS, LT	92.4	6.1	3.7
3	0	0	0	0	0	0	160§	0	122.5	1.8
4	45.7	0	0	0	0.3	0		0	7	2.1
5	NS, LT	21.3	0	0	62.8	0		0	8	0.6
6	NS	3.3	0	0	0	1		NS, LT	NS, LT	2.1
7	NS	3.5	0	20.1	10.1	1.5		NS	NS	2.4
8	NS	20.7	3.5	12.8	0	0.6		NS	NS	1.2
9	NS	0	0	0	0	0.6		NS	NS	1.2
10		1	16.8	0	2.1	0		NS	NS	0
11		1.5	0	0	1.8	0				1
12		0.5	14.3	6.7	0	2.4				0
13		0.5	1.5	0	0	1				0
14		0	0	0	0	0				0
15		6.1	7	10.7	1.8	30.5				0
16		6.1	11	0	0	17				0.3
17		12.8	1.8	NS, LT	5.2	24				0
18		30.5	4.6	NS	11	10				0.3
19		25.6	1.8	NS	22	13.7				0
20		NS, LT	18.6	NS	10	12				0
21		NS	1.5	NS	0	10				1.5
22		NS	5.5		0	1.5				2.7

Table A6.2. (continued) Summary of the post-emergence movements for ten diamondback terrapin hatchlings in March-April 2011.

No. days after release	Distance moved on subsequent days (m)									
	# H11	# H12	# H13	# H14	# H15	# H16	# H17	# H18	# H19	# H20
23		NS	0		0	0				0
24		NS	1.8		0	14				1.8
25			5.5		1	6				4.5
26			1.5		0	0				0
27			1		0	18.3				0.3
28			0		1	WS, RT				NS, LT
29			WS, RT		0					NS
30					0					NS
31					WS, RT					NS
32										
33										
34										
35										

Note:

WS = Weak transmitter signal

NS = No transmitter signal

RT = Removed terrapin hatchling
manufacturer

LT = Lost terrapin hatchling

§ = Transmitter located in a regurgitated food pellet

Light grey = Range of battery life according to manufacturer

Dark grey = Estimated end of battery life according to



Figure A6.1. Hatchling diamondback terrapin with BD-2 radio-transmitter attached to carapace.



Figure A6.2. Yellow-crowned night heron with a dead juvenile diamondback terrapin in its bill.



Figure A6.3. Dead juvenile diamondback terrapin that had been preyed upon by a yellow-crowned night heron.

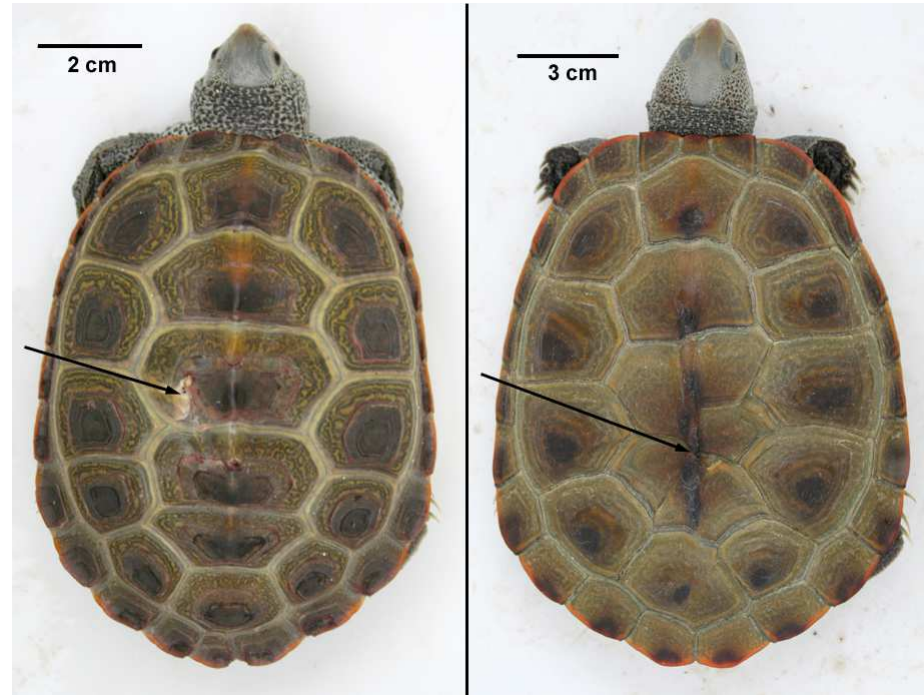


Figure A6.4. Juvenile (left) and adult female (right) diamondback terrapins from the Bermuda population showing carapace wounds believed to have been inflicted by herons during earlier attempted predation events.

Appendix 7: Supplementary Material to Chapter 7

Table A7.1. Total petroleum hydrocarbons and heavy metals (dry weight values) found in pond sediment, pond gastropods and diamondback terrapin eggs collected from Bermuda.

Sample	Gasoline-range petroleum hydrocarbons (mg/kg)	Diesel-range petroleum hydrocarbons (mg/kg)	Arsenic (mg/kg)	Cadmium (mg/kg)	Chromium (mg/kg)	Copper (mg/kg)	Iron (mg/kg)	Lead (mg/kg)	Nickel (mg/kg)	Zinc (mg/kg)	Mercury (mg/kg)
sediment - SP(A)	0.77	110.99	31.87	3.96	119.78	66.26	12637.36	42.20	8.79	64.95	BDL ⁴
sediment - SP(B)	BDL ¹	99.21	28.19	2.13	54.65	34.09	5535.43	38.35	3.78	32.91	BDL ⁴
sediment - SP(C)	BDL ¹	116.35	24.81	4.04	125.00	62.69	12884.62	38.85	8.27	53.17	BDL ⁴
sediment - SP(D)	BDL ¹	108.03	23.80	2.19	59.71	48.10	5956.20	23.50	5.04	31.53	BDL ⁴
sediment - TP(A)	BDL ¹	37.80	19.04	2.01	37.94	21.63	4095.69	14.55	3.64	25.22	BDL ⁴
sediment - TP(B)	BDL ¹	67.69	25.15	3.08	71.00	49.15	6876.92	26.08	5.38	41.77	BDL ⁴
sediment - TP(C)	BDL ¹	78.45	24.57	3.19	61.55	54.57	6758.62	30.43	5.52	49.57	0.19
sediment - ML(A)	BDL ¹	BDL ²	48.06	BDL ³	36.13	94.19	3596.77	25.48	6.13	56.94	0.27
sediment - ML(B)	BDL ¹	131.58	38.25	BDL ³	27.02	112.28	2701.75	24.91	4.91	60.35	0.25
sediment - ML(C)	BDL ¹	148.33	56.17	BDL ³	38.33	82.00	4133.33	24.17	5.67	45.33	0.18
sediment - ML(D)	BDL ¹	BDL ²	56.27	BDL ³	41.19	97.31	4164.18	25.07	5.82	42.99	0.16

Table A7.1. (continued) Total petroleum hydrocarbons and heavy metals (dry weight values) found in pond sediment, pond gastropods and diamondback terrapin eggs collected from Bermuda.

Sample	Gasoline-range petroleum hydrocarbons (mg/kg)	Diesel-range petroleum hydrocarbons (mg/kg)	Arsenic (mg/kg)	Cadmium (mg/kg)	Chromium (mg/kg)	Copper (mg/kg)	Iron (mg/kg)	Lead (mg/kg)	Nickel (mg/kg)	Zinc (mg/kg)	Mercury (mg/kg)
sediment - ML(E)	BDL ¹	147.37	45.96	BDL ³	27.19	90.53	2543.86	21.93	4.39	47.89	0.21
<i>M. coffeus</i> - ML	BDL ⁵	467.51	33.46	47.33	428.71	545.10	224.05	673.13	15.13	1585.84	12.80
<i>H. bermudensis</i> - ML	BDL ⁵	226.28	69.03	67.67	289.89	225.23	353.49	295.10	22.00	1325.34	29.93
<i>M. tuberculata</i> - SP	BDL ⁵	152.04	66.14	80.96	244.51	387.93	138.71	317.40	6.82	597.96	6.82
<i>H. bermudensis</i> - SP	BDL ⁵	268.12	36.31	67.21	125.26	129.49	205.49	269.53	23.43	663.62	4.36
Egg 01	BDL ⁵	160.62	25.21	15.86	15.86	66.29	128.61	103.40	1.42	194.62	6.26
Egg 02	BDL ⁵	305.19	27.27	32.53	28.72	120.07	81.31	98.62	2.08	155.71	6.47
Egg 03	BDL ⁵	242.19	13.95	45.18	39.53	72.76	64.78	253.16	1.86	192.69	3.26
Egg 04	BDL ⁵	87.97	35.71	40.98	65.41	95.86	100.75	125.56	2.74	163.16	6.62
Egg 05	BDL ⁵	80.74	27.46	72.95	15.78	87.30	125.00	75.41	1.84	227.70	10.52
Egg 06	BDL ⁵	225.54	5.40	37.05	16.37	68.35	98.56	162.59	2.09	171.94	2.05
Egg 07	BDL ⁵	154.97	18.78	17.68	6.13	68.23	64.92	152.21	1.30	98.62	2.15
Egg 08	BDL ⁵	293.98	39.64	22.13	34.74	105.62	102.41	69.08	1.37	181.53	4.10

Table A7.1. (continued) Total petroleum hydrocarbons and heavy metals (dry weight values) found in pond sediment, pond gastropods and diamondback terrapin eggs collected from Bermuda.

Sample	Gasoline-range petroleum hydrocarbons (mg/kg)	Diesel-range petroleum hydrocarbons (mg/kg)	Arsenic (mg/kg)	Cadmium (mg/kg)	Chromium (mg/kg)	Copper (mg/kg)	Iron (mg/kg)	Lead (mg/kg)	Nickel (mg/kg)	Zinc (mg/kg)	Mercury (mg/kg)
Egg 09	BDL ⁵	118.87	58.49	86.32	34.58	131.60	147.17	95.28	2.45	151.42	1.65
Egg 10	BDL ⁵	417.88	15.69	12.59	20.66	78.10	86.13	239.05	0.84	166.42	1.79
Egg 11	BDL ⁵	393.30	42.11	31.82	10.05	98.09	107.66	469.86	3.64	112.44	3.21

Notes for Table 7.1.

ML = Mangrove Lake; SP = South Pond; TP = Trott's Pond

BDL = Below Detection Limit reported as < method detection limits set at:

¹ <0.05 mg/kg

² <3.0 mg/kg

³ <0.012 mg/kg

⁴ <0.0002 mg/kg

⁵ <0.1 mg/kg

Table A7.2. Total polycyclic aromatic hydrocarbons (dry weight values) found in pond sediment, pond gastropods and diamondback terrapin eggs collected from Bermuda.

Sample	Naphthalene (µg/kg)	Acenaphthylene (µg/kg)	Acenaphthene (µg/kg)	Fluorene (µg/kg)	Phenanthrene (µg/kg)	Anthracene (µg/kg)	Fluoranthene (µg/kg)	Pyrene (µg/kg)	Benzo(a)anthracene (µg/kg)	Chrysene (µg/kg)	Benzo(b)fluoranthene (µg/kg)	Benzo(k)fluoranthene (µg/kg)	Benzo(a)pyrene (µg/kg)	Indeno(1,2,3-Cd)pyrene (µg/kg)	Dibenz(a,h)anthracene (µg/kg)	Benzo(g,h,i)perylene (µg/kg)
sediment - SP(A)	BDL ¹	BDL ²	BDL ³	BDL ⁴	1659.34	2527.47	BDL ⁶	1626.37	1681.32	BDL ⁸	BDL ⁹	BDL ¹⁰	BDL ¹¹	BDL ¹²	1296.70	BDL ¹⁴
sediment - SP(B)	BDL ¹	BDL ²	1574.80	BDL ⁴	1188.98	1645.67	BDL ⁶	1173.23	1259.84	BDL ⁸	BDL ⁹	BDL ¹⁰	BDL ¹¹	BDL ¹²	866.14	BDL ¹⁴
sediment - SP(C)	BDL ¹	BDL ²	BDL ³	BDL ⁴	1586.54	2144.23	BDL ⁶	1596.15	1528.85	BDL ⁸	BDL ⁹	BDL ¹⁰	BDL ¹¹	BDL ¹²	980.77	BDL ¹⁴
sediment - SP(D)	BDL ¹	BDL ²	1233.58	BDL ⁴	1080.29	1452.55	BDL ⁶	1094.89	1036.50	BDL ⁸	BDL ⁹	BDL ¹⁰	BDL ¹¹	BDL ¹²	875.91	BDL ¹⁴
sediment - TP(A)	BDL ¹	BDL ²	1033.49	BDL ⁴	BDL ⁵	1090.91	BDL ⁶	789.47	727.27	BDL ⁸	BDL ⁹	BDL ¹⁰	BDL ¹¹	BDL ¹²	569.38	BDL ¹⁴
sediment - TP(B)	BDL ¹	BDL ²	1630.77	BDL ⁴	1200.00	1607.69	BDL ⁶	1184.62	1292.31	BDL ⁸	BDL ⁹	BDL ¹⁰	BDL ¹¹	BDL ¹²	953.85	BDL ¹⁴
sediment - TP(C)	BDL ¹	BDL ²	1853.45	BDL ⁴	1362.07	1887.93	BDL ⁶	1284.48	1318.97	BDL ⁸	BDL ⁹	BDL ¹⁰	BDL ¹¹	BDL ¹²	BDL ¹³	BDL ¹⁴
sediment - ML(A)	BDL ¹	BDL ²	3483.87	BDL ⁴	2306.45	3822.58	BDL ⁶	2467.74	2661.29	BDL ⁸	BDL ⁹	BDL ¹⁰	BDL ¹¹	BDL ¹²	1951.61	2145.16
sediment - ML(B)	BDL ¹	BDL ²	3789.47	BDL ⁴	2631.58	4333.33	BDL ⁶	2982.46	2438.60	BDL ⁸	BDL ⁹	BDL ¹⁰	BDL ¹¹	BDL ¹²	2105.26	2456.14

Table A7.2. (continued) Total polycyclic aromatic hydrocarbons (dry weight values) found in pond sediment, pond gastropods and diamondback terrapin eggs collected from Bermuda.

Sample	Naphthalene (µg/kg)	Acenaphthylene (µg/kg)	Acenaphthene (µg/kg)	Fluorene (µg/kg)	Phenanthrene (µg/kg)	Anthracene (µg/kg)	Fluoranthene (µg/kg)	Pyrene (µg/kg)	Benzo(a)anthracene (µg/kg)	Chrysene (µg/kg)	Benzo(b)fluoranthene (µg/kg)	Benzo(k)fluoranthene (µg/kg)	Benzo(a)pyrene (µg/kg)	Indeno(1,2,3-Cd)pyrene (µg/kg)	Dibenz(a,h)anthracene (µg/kg)	Benzo(g,h,i)perylene (µg/kg)
sediment - ML(C)	BDL ¹	BDL ²	3450.00	BDL ⁴	2466.67	4216.67	BDL ⁶	2650.00	2666.67	BDL ⁸	BDL ⁹	BDL ¹⁰	BDL ¹¹	BDL ¹²	1983.33	BDL ¹⁴
sediment - ML(D)	BDL ¹	1567.16	3044.78	BDL ⁴	2253.73	3000.00	BDL ⁶	2522.39	BDL ⁷	BDL ⁸	BDL ⁹	BDL ¹⁰	BDL ¹¹	BDL ¹²	1835.82	BDL ¹⁴
sediment - ML(E)	BDL ¹	BDL ²	3526.32	BDL ⁴	2684.21	3807.02	BDL ⁶	2561.40	2754.39	BDL ⁸	BDL ⁹	BDL ¹⁰	BDL ¹¹	BDL ¹²	1947.37	BDL ¹⁴
<i>M. coffeus</i> -ML	BDL ¹⁵	BDL ¹⁶	1689.62	BDL ¹⁶	1911.74	1986.42	BDL ¹⁷	1827.35	474.30	172.65	BDL ¹⁷	BDL ¹⁷	273.52	BDL ¹⁷	705.14	BDL ¹⁷
<i>H. bermudensis</i> -ML	BDL ¹⁵	BDL ¹⁶	2754.95	88.63	2218.98	1629.82	BDL ¹⁷	2667.36	118.87	BDL ¹⁷	BDL ¹⁷	BDL ¹⁷	690.30	BDL ¹⁷	1069.86	115.75
<i>M. tuberculata</i> -SP	BDL ¹⁵	BDL ¹⁶	997.65	124.61	445.14	1696.71	136.36	2360.50	148.90	BDL ¹⁷	BDL ¹⁷	BDL ¹⁷	184.17	BDL ¹⁷	353.45	114.42
<i>H. bermudensis</i> -SP	BDL ¹⁵	BDL ¹⁶	225.90	261.79	1867.70	698.80	BDL ¹⁷	619.99	BDL ¹⁷	149.19	BDL ¹⁷	BDL ¹⁷	107.67	BDL ¹⁷	347.64	244.19
Egg 01	BDL ¹⁵	BDL ¹⁶	161.19	54.67	265.44	73.94	160.91	292.92	74.22	89.24	BDL ¹⁷	BDL ¹⁷	105.38	BDL ¹⁷	124.93	BDL ¹⁷
Egg 02	BDL ¹⁵	BDL ¹⁶	117.30	41.87	57.79	46.02	114.88	273.01	158.13	173.70	BDL ¹⁷	BDL ¹⁷	59.52	BDL ¹⁷	78.89	BDL ¹⁷

Table A7.2. (continued) Total polycyclic aromatic hydrocarbons (dry weight values) found in pond sediment, pond gastropods and diamondback terrapin eggs collected from Bermuda.

Sample		Naphthalene (µg/kg)	Acenaphthylene (µg/kg)	Acenaphthene (µg/kg)	Fluorene (µg/kg)	Phenanthrene (µg/kg)	Anthracene (µg/kg)	Fluoranthene (µg/kg)	Pyrene (µg/kg)	Benzo(a)anthracene (µg/kg)	Chrysene (µg/kg)	Benzo(b)fluoranthene (µg/kg)	Benzo(k)fluoranthene (µg/kg)	Benzo(a)pyrene (µg/kg)	Indeno(1,2,3-Cd)pyrene (µg/kg)	Dibenz(a,h)anthracene (µg/kg)	Benzo(g,h,i)perylene (µg/kg)
868	Egg 03	BDL ¹⁵	BDL ¹⁶	51.83	74.75	257.14	183.39	255.48	492.36	221.59	233.22	BDL ¹⁷	BDL ¹⁷	77.74	BDL ¹⁷	BDL ¹⁷	BDL ¹⁷
	Egg 04	BDL ¹⁵	BDL ¹⁶	28.95	102.26	210.15	78.95	161.28	301.88	139.85	166.17	BDL ¹⁷	BDL ¹⁷	BDL ¹⁷	BDL ¹⁷	218.80	BDL ¹⁷
	Egg 05	BDL ¹⁵	BDL ¹⁶	369.67	BDL ¹⁶	361.07	125.00	271.31	444.26	290.57	309.84	BDL ¹⁷	BDL ¹⁷	84.84	BDL ¹⁷	126.64	BDL ¹⁷
	Egg 06	BDL ¹⁵	BDL ¹⁶	101.44	67.99	219.42	256.47	252.52	519.06	350.00	374.82	BDL ¹⁷	BDL ¹⁷	121.94	BDL ¹⁷	51.44	BDL ¹⁷
	Egg 07	BDL ¹⁵	BDL ¹⁶	45.03	BDL ¹⁶	144.20	50.28	78.73	152.49	135.91	143.92	BDL ¹⁷	BDL ¹⁷	BDL ¹⁷	BDL ¹⁷	BDL ¹⁷	BDL ¹⁷
	Egg 08	BDL ¹⁵	BDL ¹⁶	84.74	81.93	67.47	390.36	354.62	663.45	140.16	163.45	BDL ¹⁷	BDL ¹⁷	96.79	BDL ¹⁷	149.40	BDL ¹⁷
	Egg 09	BDL ¹⁵	BDL ¹⁶	366.98	192.92	63.68	291.98	232.08	477.36	91.51	115.57	BDL ¹⁷	BDL ¹⁷	82.08	BDL ¹⁷	134.43	BDL ¹⁷
	Egg 10	BDL ¹⁵	BDL ¹⁶	40.51	BDL ¹⁶	156.20	121.53	77.01	149.27	202.92	263.14	BDL ¹⁷	BDL ¹⁷	75.18	BDL ¹⁷	BDL ¹⁷	BDL ¹⁷
	Egg 11	BDL ¹⁵	BDL ¹⁶	133.01	BDL ¹⁶	74.16	92.34	313.40	666.99	316.75	387.08	BDL ¹⁷	BDL ¹⁷	49.76	BDL ¹⁷	58.85	BDL ¹⁷

Notes for Table A7.2.

ML = Mangrove Lake; SP = South Pond; TP = Trott's Pond

BDL = Below Detection Limit reported as < method detection limits set at:

¹ <16.1 µg/kg

² <11.7µg/kg

³ <22.9 µg/kg

⁴ <20.4 µg/kg

⁵ <574 µg/kg

⁶ <20.0 µg/kg

⁷ <17.0 µg/kg

⁸ <12.3 µg/kg

⁹ <39.2 µg/kg

¹⁰ <60.5 µg/kg

¹¹ <10.8 µg/kg

¹² <43.6 µg/kg

¹³ <12.9 µg/kg

¹⁴ <15.1 µg/kg

¹⁵ <15.0 µg/kg

¹⁶ <7.5 µg/kg

¹⁷ <10.0 µg/kg

Recovery Plan for the Diamondback Terrapin, *Malaclemys terrapin*, in Bermuda



Government of Bermuda
Ministry of Environment and Planning
Department of Conservation Services

Recovery Plan for the Diamondback Terrapin, *Malaclemys terrapin*, in Bermuda

Prepared in Accordance with the Bermuda Protected Species Act 2003

Funded in part by:

Department of Conservation Services
Atlantic Conservation Partnership
Bermuda Zoological Society
Mid Ocean Club
OTEP

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Cover photo: Wild mature male diamondback terrapin, *Malaclemys terrapin*.

Photo by Mark Outerbridge

All photos throughout this document were taken by Mark Outerbridge

Maps were prepared by Mark Outerbridge

Published by



Government of Bermuda
Ministry of Environment and Planning
Department of Conservation Services

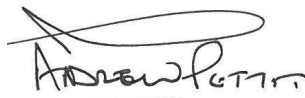
“To conserve and restore Bermuda’s natural heritage”

Disclaimer

Recovery plans delineate reasonable actions that are believed to be required to recover and/or protect listed species. We, the Department of Conservation Services, publish recovery plans, sometimes preparing them with the assistance of field scientists, other government departments, as well as other affected and interested parties, acting as independent advisors to us. Plans are submitted to additional peer review before they are adopted by us, and formulated with the approval of interested parties mentioned in Parts II and III of the plan. Objectives of the recovery plan will be attained and necessary funds made available subject to budgetary and other constraints affecting the parties involved. Recovery plans may not represent the views nor the official positions or approval of any individuals or agencies involved in the recovery plan formulation, other than our own. They represent our official position only after they have been signed by the Director of Conservation Services as approved. Approved recovery plans are subject to modifications as dictated by new findings, changes in species status, and the completion of recovery actions.

Literature citation of this document should read as follows: M. Outerbridge, 2013. Recovery Plan for Diamondback Terrapins (*Malaclemys terrapin*) in Bermuda. Department of Conservation Services, Government of Bermuda. 48 pages.

An electronic version of this recovery plan will also be made available at www.conservation.bm



Director
Department of Conservation Services
Government of Bermuda

16th OCTOBER 2013

Date

Acknowledgements

This investigation was initiated by grant money provided by Susan Ford. Additional funding was given by the Bermuda Government Department of Conservation Services, the Atlantic Conservation Partnership, the Bermuda Zoological Society and the Mid Ocean Club. The support of these sponsors is gratefully appreciated. Many thanks are due to numerous individuals who assisted with the various surveys in the field, most notably Simieon Massey and Ronald Burchall. Dr. Matthew Draud provided practical and technical assistance with the radio-telemetry investigations, and the Bermuda Aquarium, Museum and Zoo and Dolphin Quest generously donated large quantities of herring over the years for use in the terrapin traps.

Executive Summary

Current Species Status:

This recovery plan addresses the need for actions to conserve a native species of terrapin, *Malaclemys terrapin*, in Bermuda. This species is listed as Vulnerable (EN, B1a, biii) as per IUCN criteria, under the Protected Species Act 2003. Diamondback terrapins represent Bermuda's second naturally occurring non-marine reptile that still survives on the island (the other being the endemic skink *Plestiodon* (formerly *Eumeces*) *longirostris*.) The current terrapin population is estimated to comprise approximately 100 individuals ≥ 81 mm straight carapace length (SCL).

Habitat Requirements and Threats:

Diamondback terrapins are an inhabitant of the land-locked, brackish water pond environment in Bermuda. The entire population of diamondback terrapins can only be found in four ponds on the Mid Ocean golf course located in the eastern parishes of the islands. These ponds are known as Mangrove Lake, South Pond, North Pond, and Trott's Pond and all four bodies of water have been incorporated into the golf course as water hazards. Neonate and small juvenile terrapins use adjacent mangrove swamps and grass-dominated marshes as developmental habitat; larger juveniles (≥ 81 mm SCL), sub-adult and adult terrapins are found within the aquatic pond environment. It is thought that the principle factor which has led to the limited distribution of diamondback terrapins is loss of habitat through fragmentation of the wetlands in Bermuda. This restriction in habitat is due to both human development and natural processes. Pollution of ponds has also contributed to the decline and degradation of available habitat, as ponds and marshes were historically used as garbage disposal sites, and continue to receive run-off from roads and the surrounding golf course. In addition to having a limited distribution and a small population, Bermuda's terrapin population also suffers from low recruitment and poor annual hatching success which lends further support to the belief that it is vulnerable to local extirpation.

Recovery Objective:

The main goal of this plan is to increase both the population level and the areas of residency for diamondback terrapins in Bermuda.

Recovery Criteria:

Down listing of diamondback terrapins in Bermuda will be considered when:

- The genetic diversity of Bermuda's extant population is fully understood.
- All current and potential habitats suitable for diamondback terrapin growth, reproduction and survival are identified, assessed, restored and protected under legislation.
- Diamondback terrapins are viable residents in at least two separate geographic locations on Bermuda.

- Population levels in Bermuda indicate that terrapins are successfully maintaining themselves on a long-term basis and showing adequate levels of recruitment.

Actions Needed:

1. Protect wetland habitats of extant terrapin population through legislation,
2. Restore protected wetland habitats of current extant population,
3. Identify, assess, protect and restore wetland habitats deemed suitable for diamondback terrapin introduction,
4. Increase population size through increased hatching success and recruitment to the adult population,
5. Expand area of residency through translocation of individuals raised in captivity,
6. Identify the full genetic composition of existing population,
7. Develop research programmes on understanding the effects that environmental contaminants have upon the reproductive biology and overall health of terrapins in Bermuda,
8. Promote conservation education programmes concerning Bermuda's terrapin population,
9. Continued population monitoring.

Recovery Costs: The total cost of recovery actions cannot be defined at this point. Funding needs to be secured through Non-Governmental Organizations (NGO's), overseas agencies, and other interested parties for implementing the necessary research and monitoring studies on the biology of the diamondback terrapin. Developing budgets for each action are the responsibility of the leading party as outlined in the work plan.

Date of Recovery: Meeting the recovery objectives in Bermuda will depend on the restoration and protection of available habitats. Down listing will be considered following ten years of implementation (2023), once evaluation of conservation efforts is complete.

Part I: Introduction

A. Brief overview

Diamondback terrapins *Malaclemys terrapin* have been listed as a globally near threatened species by the International Union for Conservation of Natural Resources (IUCN). In 2013, diamondback terrapins were included in Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) in an attempt to regulate international trade so that exports from the native range are not detrimental to the species' survival in the wild. Diamondback terrapins are endemic to the coastal wetland environments along the east coast of the United States from Cape Cod in Massachusetts to Corpus Christi in Texas. Their status, which varies from state to state, ranges from endangered to a species of special concern. Massive over-harvesting for food consumption in the late 19th and early 20th centuries lead to huge declines in the North American populations, which continue to be affected by habitat loss, predation, crab trapping activities and commercial harvest for pet-trade and human consumption (Roosenburg et al., 1997; Hart and Lee, 2006).

Historical accounts of Bermuda's diamondback terrapins first appear in writings that date back to the 1950s (D. Wingate, unpublished notes), however it was not until 2007 when their origin on these remote oceanic islands was tested using a combination of palaeontologic (fossil, radiometric and palaeoenvironmental) and genetic data. These lines of evidence supported the hypothesis that these terrapins are natural colonizers of Bermuda, having arrived between 3000 and 400 years ago (Parham et al., 2008), and represents the second naturally occurring non-marine reptile that still survives on one of the most densely populated and heavily developed oceanic islands in the world (the other is an endemic skink). Bermuda is situated in a part of the North Atlantic Ocean which regularly receives spin-off eddies from the Gulf Stream. These eddies have been implicated in the transport of a great diversity of plants and animals from the Caribbean and eastern seaboard of North America to Bermuda (Glasspool, 1994; Meylan and Sterrer, 2000; Grady et al., 2001; Sterrer et al., 2004), and are most likely responsible for transporting diamondbacks as well (Davenport et al., 2005).

It appears that this Bermudian population is the only wild breeding population outside of the North American range. There has been a dearth of information regarding the health and status of this isolated oceanic population. Knowledge of their life history is necessary to make informed management decisions and was deemed critical for a species recovery plan. Consequently, work on this species was initiated in 2008 by Mark Outerbridge of the Bermuda Zoological Society as part of a Ph.D. programme. All of the Bermuda data presented in this recovery plan is the result of this doctoral investigation.

This recovery plan discusses threats and conservation efforts for Bermuda's diamondback terrapins, summarizing new and previously unavailable information about their local habitat and dietary requirements, reproductive biology, and threats to survival. In order to ensure sustainability of the terrapin population within Bermuda, an increase in the

area of occupancy as well as in population size is recommended and deemed possible through head-starting and translocations initiatives. The recovery of the population is heavily dependent on the availability of suitable habitats, hence the restoration of selected ponds is a priority in this plan. Should all of this be realized, it may be possible to down list diamondback terrapins to a lesser threatened status and/or remove it from the Protected Species list.

B. Current protection status

Bermuda's diamondback terrapins are classified as a level II protected species and declared to be Vulnerable under the Protected Species Act (2003). Diamondback terrapins are not harvested for food in Bermuda or caught as by-catch in commercial or recreational shellfish pots, and none of the ponds have boating traffic, however, the area in which the terrapins reside is currently, and has also historically been, heavily impacted upon by other anthropogenic activities. There have been few opportunities for range increase, due in great part to the restriction in habitat availability. The fragmentation of the wetland habitat in Bermuda, and the very limited distribution of the terrapins, makes this species very vulnerable to human impact.

Legal Protection

The Protected Species Amendment Act (2011) considers it an offence for an unauthorized person to willfully damage, destroy, injure, disturb, uproot, fell, kill, take, import, export, sell or purchase a level II protected species or any part of a level II protected species. Offenders are liable, on summary of conviction, to a fine of \$15,000 or one year of imprisonment.

Habitat Protection

Mangrove Lake, Trott's Pond and North Pond have been designated as 'nature reserves' under the 2008 Bermuda Development Plan; however, South Pond is currently zoned as a 'recreational area' (Fig. I in Appendix).

C. Taxonomy and description of species

Class: Reptilia (reptiles)
Order: Testudines (turtles, terrapins & tortoises)
Family: Emydidae (pond turtles)
Genus: *Malaclemys*
Species: *terrapin*
Common name: Diamondback terrapin

Diamondback terrapins belong to the Family Emydidae, a large and diverse group of reptiles collectively known as 'pond turtles' that are naturally found throughout North America, much of Europe, and eastward into Russia, the Near East, and North Africa (Meylan, 2006). They are the only

member of the genus *Malaclemys*. Seven subspecies of diamondback terrapin are currently recognized, which have been divided into northern (*M. t. terrapin*, *M. t. centrata*) and southern (*M. t. tequesta*, *M. t. rhizophorarum*, *M. t. macrospilota*, *M. t. pileata*, *M. t. littoralis*) populations with Merritt Island, Florida, providing a break between the two; however, genetic studies do not fully agree with the existence of these subspecies (Lamb and Avise, 1992; Hart, 2005; Hauswaldt and Glen, 2005).

Diamondback terrapins are small to medium sized turtles that show distinctive shell and soft tissue markings; however, these markings vary greatly throughout their range. The carapace is typically oblong in shape and possesses a mid-dorsal keel which is more visibly raised, or knobbed, in the southern subspecies. Carapace colour is highly variable but usually of earth tones ranging from light olive and brown to dark brown and black. The carapace is also marked with concentric growth rings that are most pronounced on younger individuals (Fig. 1 top), from which this species gets its common name, but disappear with age. The circular depressions that these rings make extend below the veneer of each scute and are imprinted upon the dorsal surface of the underlying bones of the carapace. The plastron, in contrast to the carapace, is more brightly coloured with yellowish or orange hues and can be either plain in appearance or smudged with varying amounts of dark blotches. Sometimes, however, the plastral scutes can have a dark base colour with lighter colourful edges. The plastral scutes may also show growth rings. These rings, or annuli, have been used by some researchers to estimate the age of individuals (Seigel, 1984; Tucker et al., 1995; Gibbons et al., 2001); however this technique remains a contentious method of aging terrapins and many agree that it is not possible to use it on older individuals whose rings have disappeared with the passage of time (Morreale, 1992; Gibbons et al., 2001). Skin colour also varies throughout the range, but is generally shades of gray with dark spots, flecks or lines (the latter having not been observed in the Bermuda population) (Fig. 1 bottom).

Diamondback terrapins show sexual dimorphism; with males being considerably smaller than females and having proportionally smaller heads, but wider and longer tails with a cloaca situated posterior to the edge of the carapace when the tail is fully extended.

The diamondback terrapin carapace normally features 38 named scutes:- one nuchal, five vertebrals, four pairs of costals (also known as pleurals), eleven pairs of marginals, and two supracaudals. The plastron is normally composed of twelve named scutes; one pair each of gular, humeral, pectoral, abdominal, femoral, and anal scutes. Both carapace and plastron are joined by a bridge. Variations in the number of vertebral, costal or marginal scutes are not uncommon, and may involve an extra, split, or distorted scute. These variations are believed to be caused by high incubation temperatures (Wood and Herlands, 1997; Herlands et al., 2004) and possibly embryological exposure to petroleum crude oil and polycyclic aromatic hydrocarbons (Van Meter et al., 2006).



Figure 1. Photographs of a typical diamondback terrapin from Bermuda

D. Current status

Global distribution

Diamondback terrapins are endemic to the Atlantic and Gulf Coasts of the United States of America, whose range extends across 16 states from Cape Cod, Massachusetts, in the north to Corpus Christi, Texas, in the south (Fig. 3). Their distribution across this range is not continuous, but rather consists of fragmented populations concentrated in a linear fashion along the coast. Five of the seven subspecies occur within Florida, of which three are considered to reside exclusively in that state. The northern diamondback terrapin, *Malaclemys terrapin terrapin*, ranges from Cape Cod in Massachusetts to Cape Hatteras in North Carolina. The Carolina diamondback, *M. t. centrata*, ranges from Cape Hatteras southwards to Volusia County in Florida. The Florida East Coast diamondback, *M. t. tequesta*, ranges from Volusia County to Miami-Dade County, as well as possibly into the upper Keys in Monroe County. The mangrove diamondback, *M. t. rhizophorarum*, occurs in Monroe County from Fort Myers to Florida Bay and throughout the Florida Keys and the Marquesas. The ornate diamondback, *M. t. macrospilota*, occurs from Florida Bay to the western part of the Florida Panhandle in Walton County. The Mississippi diamondback, *M. t. pileata*, ranges from western Choctawhatchee Bay in Okaloosa County, Florida, westwards through the state of Louisiana. The Texas diamondback, *M. t. littoralis*, is found from western Louisiana to Corpus Christi in Texas (Ernst et al., 1994; Butler et al., 2006). The only geographic region where diamondback terrapins appear to naturally reside outside of their North American range is in Bermuda.

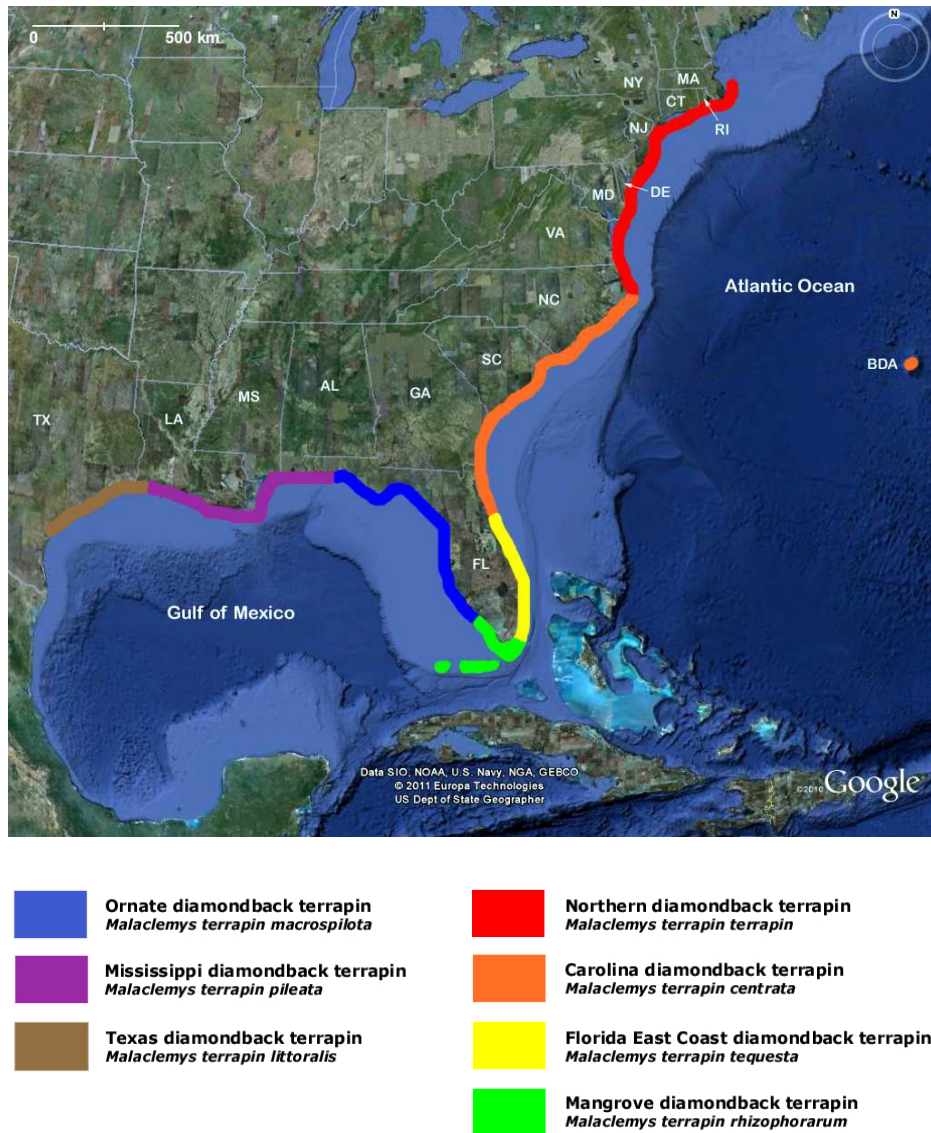


Figure 2. Map illustrating the range-wide distribution of the seven recognized diamondback terrapin subspecies (adapted from Butler et al., 2006; Lee and Chew, 2008).

Local distribution

The entire Bermuda population of diamondback terrapins is found only in four brackish water ponds named Mangrove Lake, South Pond, North Pond, and Trott's Pond. All four bodies of water are situated upon a single square kilometer of Bermuda and are only separated from each other by, at most, 380 meters of land. These ponds are located on a private golf course, the Mid Ocean Club, located in Smith's Parish at the eastern end of the islands (Figs. 3 and 4). Mangrove Lake and Trott's Pond are the largest of these ponds (approximately 10 ha and 3 ha respectively in area) and both are

simple basins fringed by red mangrove trees *Rhizophora mangle* and characterized by shallow depths (averaging 1.4 m and 2.7 m respectively) with bottoms comprised of deep deposits of highly organic sediment (Thomas et al., 1991). North Pond and South Pond are considerably smaller in area (both approximately 0.4 ha) and lack mangrove vegetation; however both have small marshes in their centers dominated by grasses. Mangrove Lake, South Pond, North Pond, and Trott's Pond have been incorporated into the golf course as water hazards found between the fifth and twelfth holes. No diamondback terrapins have been discovered in any other bodies of water on Bermuda despite a series of extensive wetland community surveys conducted between 2004 and 2007 (Outerbridge et al., 2007a; Outerbridge, 2008).

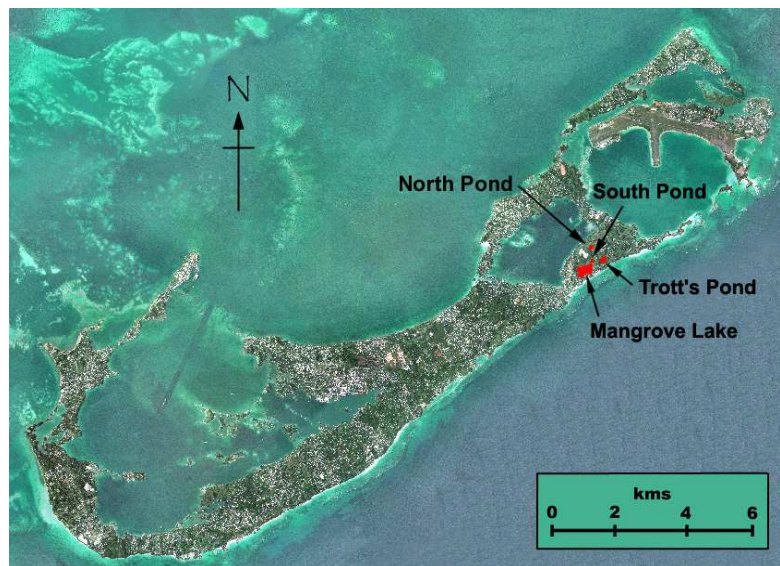


Figure 3. Aerial photograph of Bermuda showing the location of the diamondback terrapin ponds.



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Figure 4. Aerial photograph from 2003 showing the diamondback terrapin ponds situated on the Mid Ocean golf course (A=Mangrove Lake, B=Trott's Pond, C=South Pond, D=North Pond)

Mangrove Lake

Mangrove Lake is currently the largest of Bermuda's brackish water ponds and is believed to have formed during the last 11,000 years through the action of dissolution of calcium carbonate from either rock or sand, thereby creating a depression that gradually filled with saltwater as the seas rose (Watts and Hansen, 1986; Thomas, 2002). It is a simple basin approximately 10 ha, in area fringed almost exclusively by red mangrove trees *Rhizophora mangle* and characterized by shallow depths, averaging only 140 cm, fairly even contours and a gently sloping shoreline. The pond bottom comprises deep deposits of highly organic sediments, from which widgeon grass *Ruppia maritima* grows in dense clumps. Mangrove Lake is often subject to considerable changes in dissolved oxygen, temperature, salinity and nutrient levels (Thomas et al., 1991). A few small subterranean fissures ensure that ocean water still enters this pond from the south shore; however, it has a small tidal range of 1.4 cm (Thomas et al., 1992). Average mid-water temperatures ranged between 15.6°C (February) and 30.3°C (July); surface salinities between 26 practical salinity units (psu) (January) and 35.7 psu (August) (Outerbridge, unpublished data). Mangrove Lake and the surrounding land are owned by a variety of private individuals and organizations. The pond is mostly owned by the Tucker's Point Club, but the surrounding land is owned by the Mid Ocean Club, the Bermuda

National Trust, and a number of private individuals who live adjacent to the pond.

Trott's Pond

Trott's Pond is also partially situated on the Mid Ocean golf course. It is approximately 3 ha in area and formed between low Pleistocene sand dunes that were inundated by postglacial seas. Over time freshwater slowly eroded away the depression creating fissures through which saltwater enters from the south shore as the sea level rose around Bermuda. Trott's Pond is currently a simple basin characterized by fairly shallow depths, with the deepest part at its centre. It has fairly even contours and a gently sloping shoreline (Thomas et al., 1992). The connection to the ocean is small and located at the surface, giving Trott's Pond a very small tidal range of 1.5 cm. Rainfall and surface run-off from the surrounding area usually doesn't mix with the saltwater below, but instead floats as a distinct layer on top, eventually draining off through the surface connection (Thomas, 2002). The pond bottom comprises deep deposits of highly organic sediments. The mean depth in Trott's Pond is 269 cm; the maximum was 320 cm. Average annual surface water temperatures range from 16-31° C (+/- 4° 8 C) and salinities vary from 24-34 psu (+/- 2.6 psu) (Thomas *et al.*, 1991). Trott's Pond shares many species in common with neighbouring Mangrove Lake including the mangrove oyster *Isognomon alatus*, the Bermuda killifish *Fundulus bermudae* and the coffee bean snail *Melampus coffeus*. The shoreline of Trott's Pond is fringed almost entirely by red mangrove trees.

South Pond

South Pond is much smaller than Mangrove Lake and Trott's Pond and was deliberately dredged to create a golf course water hazard in the 1990s. A land bridge separates this pond into two distinct bodies of water; the moat-like pond to the north and a smaller pond to the south. These two bodies of water are collectively known as South Pond and comprise a combined area of approximately 0.4 ha. Mangrove trees are not present at this site, but there is a small 0.3 ha marsh located in the centre of the larger pond, made up predominantly of saw grass *Cladium jamaicense*, and to a lesser extent cattail *Typha angustifolia*. The emergent vegetation that grows around the perimeter of South Pond is exclusively sheathed paspalum *Paspalum vaginatum* which is periodically trimmed by the agronomy staff of the Mid Ocean Club. Widgeon grass *Ruppia maritima* grows seasonally within South Pond, and the pond bottom is comprised of highly organic sediment. The mean depth in the larger pond is 35 cm, while the smaller pond averages 81 cm. Average annual mid-water temperatures in 2011 ranged between 15.7 °C (Feb) and 29.8°C (August). The salinity of South Pond is much lower than in neighboring Mangrove Lake and Trott's Pond. Salinities in the larger pond ranged between 4.8 psu (March) and 16.8 psu (July) while in the smaller pond they ranged between 7.8 psu (April) and 18.7 psu (June) (Outerbridge, unpublished data). The water levels vary considerably according to the amounts received through rainfall, and in periods of drought it is not uncommon for some areas to dry up completely.

North Pond

North Pond is approximately the same size as South Pond (*ca.* 0.4 ha), is a naturally occurring pond, and has also been incorporated into the golf course. Mangrove trees are also not present at this site; however, there is a narrow band of marshland located in the pond which is dominated by sheathed paspalum *P. vaginatum*. As with all of the other diamondback terrapin ponds, North Pond's bottom is comprised of highly organic sediment. Mean depth of water is 30 cm. Mean mid-water temperatures are only available for a 6 month period in 2010, and ranged between 15.6°C (February) and 31.1°C (July). The salinity in this pond is slightly lower than in neighboring Mangrove Lake and Trott's Pond, but higher than South Pond. The mean mid-water salinities in 2010 ranged between 7.8 psu (April) and 18.7 psu (June) (Outerbridge, unpublished data). The water levels in North Pond vary considerably according to the amounts received through rainfall, and in periods of drought it is not uncommon for large areas of the pond to dry up completely.

E. Ecology

Habitat requirements

Diamondback terrapins have a life cycle comprised of distinct phases that have different habitat requirements. Adult and sub-adult terrapins have need of brackish bodies of water in which they feed, mate and, for populations residing in cooler regions, brumate (the reptilian equivalent of hibernation); mature female terrapins require sandy substrate for egg laying; hatchlings and small juveniles require dense vegetation which grows adjacent to the adult aquatic environment to forage, grow and hide from predators. Examples of this vegetation include salt marsh grasses (*Spartina spp.* in North America and *Paspalum vaginatum* and *Cladium jamaicense* in Bermuda) and red mangroves (*Rhizophora mangle*).

Diamondback terrapins are the only species of turtle that have specialized to inhabit the tidal salt marsh and estuarine environment along the Atlantic and Gulf coasts of North America (e.g. coastal marshes, mudflats, river estuaries, tidal creeks, brackish lagoons, and mangrove swamps). They exhibit unique physiological and behavioral adaptations that enable them to live within these habitats (Cowan, 1971; Gilles-Baillien, 1973; Cowan, 1990; Davenport and Macedo, 1990; Hart and Lee, 2006). Bermuda's extant terrapin population, however, is restricted to the brackish water pond environment. The present day saline pools and ponds in Bermuda vary both in size and in structure. Nearly all date back in formation to the Holocene era (approximately 10,000 years ago.) The sporadic addition of freshwater into these ponds, either directly in the form of rainfall or indirectly as surface run-off, means that salinities vary throughout the year. They are generally slightly lower than that of pure seawater, but do show predictable seasonal patterns. The primary factor influencing salinity is the size and location of the underground connections each pond has with the ocean. Pond size, depth and volume, the size and

nature of the connections to the ocean, the rate of freshwater inflow, and the tidal exchange of seawater all influence the hydrographic characteristics of each pond. Bermuda's marine ponds generally have a rich biota. Species richness increases with increasing physical stability and diversity of habitat. Thus ponds having submerged rock substrata, an abundant submerged mangrove root community along the periphery of the pond, and bottom sediment show greater diversity than ponds that feature sedimentary substrata only (Thomas et al., 1992).

Physical factors

The most important factor influencing physical stability in the saline ponds is the amount of tidal exchange (Thomas et al., 1992). Temperature and salinity are dependent upon the amount of sea water that enters from the ocean, thus ponds close to the sea with relatively large connections have a higher flushing rate, narrower ranges of salinity and temperature and therefore provide a more stable environment than those of ponds further from the sea. The mean ocean tidal range in Bermuda is only 75 cm, but is greatly reduced in the ponds where there are more restrictions to tidal flow. While proximity to the ocean and the nature of the connections influence salinity level, the locations and sizes of these saltwater inlets in relation to the tide level also affect the flushing rate. Salinity stratification can occur in poorly mixed ponds, or where the connection to the sea is in the deepest part, due to the different densities of fresh and saltwater, although this phenomenon is unlikely to occur in very shallow ponds. Thomas et al. (1991) described the physical characteristics of the six largest saline ponds, including Mangrove Lake and Trott's Pond. Surface salinities ranged from 6.5 to 42.5 psu and the temperatures varied from 15.0° to 37.5°C. More limited data exists for Bermuda's freshwater ponds; however, it appears that salinity and temperature also follow predictable seasonal patterns. Evaporation, coupled with the sporadic addition of freshwater either directly as rainfall or indirectly as surface run-off, typically via storm drains from neighboring roads, means that surface salinities can range from 0 (totally freshwater) to 12 psu (brackish water.) The small and shallow nature of most of these ponds means that temperatures can also vary greatly from 10.6°C to 34.6°C (Outerbridge, unpublished data). A shallow pond will show greater temperature range because it can exchange heat more rapidly with the atmosphere (e.g. North Pond).

Biological factors

Bermuda's brackish and marine ponds all have deep benthic deposits of highly organic sediments and are subject to large changes in dissolved oxygen, temperature, salinity and nutrient levels. Surface run-off from surrounding land transports particulate matter and plant nutrients into the ponds. Fringing mangrove trees are a common feature of these saline ponds. These trees constantly drop leaves that slowly decompose, forming a highly organic layer on the pond bottom that enhances the base of the food web. Due to their small physical size and accumulated sediments, the saline ponds are usually quite shallow. Because of this, ambient light levels at the

bottom can be high, despite the fact that these ponds are typically very turbid due to the high levels of suspended organic material. Plants, however, do not usually grow on the deeper bottoms of the ponds due to the unstable, anoxic environment created by the decomposition of the organic matter. The levels of dissolved oxygen also vary considerably between ponds as well as diurnally and seasonally. Daytime photosynthesis can supersaturate pond water with oxygen while the consumption of oxygen at night from fishes and microbial life on the sediment can reduce oxygen levels to zero, at least in patches, resulting in transitory night-time anoxia. Anoxic events are routine in some of the poorly flushed anchialine ponds in summer and are partly responsible for their low species diversity, which is typically much reduced below that of open water marine habitats (Thomas and Logan, 1992). The biotic characteristics of Bermuda's ponds are highly variable. Pond size, volume, and physical stability, as well as the stochastic nature of species' colonization and the ability of these species to adapt and survive in the ponds are all factors responsible for this biological variability. One of the curious features of the ponds is that there is great variability of biota amongst the ponds. Quite often a species is found in only one or a few ponds and few species occur in all ponds.

General biology

The annual activity cycle of adult diamondback terrapins from northern populations is one that generally begins with emergence from winter-induced brumation during the spring. Emergence is quickly followed by a period of courtship and mating. Nesting soon follows and often lasts for many months during which females can deposit multiple clutches of eggs (Seigel, 1980b; Goodwin, 1994; Roosenburg and Dunham, 1997). Diamondback terrapins are believed to have a very small home range (Lovich and Gibbons, 1990; Gibbons et al., 2001; Baldwin et al., 2005) and some mature females are known to return to the same nesting beaches annually (Jeyasuria et al., 1994). The incubation period and the sex of the developing embryos are determined by the incubation temperatures; cooler temperatures produce male offspring while warmer temperatures produce female. Hatchlings will, upon emergence, typically seek refuge within the closest vegetation and show avoidance of open water (Burger, 1977; Lovich et al., 1991). Very little exists in the literature about the life history of hatchlings and juveniles from the time they depart the nest to the time that they recruit to the sub-adult population. Growth is most rapid during the first few years after hatching, but then slows down considerably after sexual maturity has been attained (Tucker et al., 1995; Roosenburg and Kelley, 1996). Diamondback terrapins usually enter brumation in November and December and remain in that state either buried in sediment or beneath undercut banks through February or March the following year (Yearicks et al., 1981; Seigel, 1984); however, some populations in Florida were observed to be active on warm days during the winter (Hart, 2005). The lifespan of diamondback terrapins in the wild has been estimated to be approximately 20 years (Seigel, 1984), but may last as long as 40 years in captivity (Hildebrand, 1932).

Population biology

The results of a three year mark and recapture survey (2008-2010) suggest that the adult and sub-adult population of diamondback terrapins presently living on Bermuda comprises approximately 100 individuals. The recapture rate in this population was relatively high over the census period (60.6%), and coupled with the fact that 99 individuals were captured and marked (64 mature females, 22 mature males, 13 juveniles) suggests that the estimate may be very accurate. The Bermuda population is dominated by females (3:1), which ranged in size 116-196 mm straight carapace length (SCL notch-to-notch) (mean 158 mm; SD 22.6 mm) and 270-1340 grams (mean 720 g; SD 285.8 g). Males ranged in size from 109-134 mm SCL (mean 122.7 mm; SD 8.2 mm) and 200-350 grams (mean 281.4 g; SD 47.1 g); and juveniles ranged in size from 81-108 mm SCL (mean 98 mm; SD 9.5 mm) and 95-215 grams (mean 168 g; SD 42.6 g). Thirty four out of 99 individuals (34.3%) showed carapace scute anomalies. The most common anomalies were extra vertebral scutes (15.2% frequency of occurrence), extra costal scutes (15.2% frequency of occurrence), and extra marginal scutes (18.2% frequency of occurrence). The mean annual recruitment rate to the adult population throughout the three year census period was two terrapins; one new recruit was encountered in 2008, five in 2009 and none in 2010. The density of diamondback terrapins in Bermuda is estimated to be 6.0 terrapins/ha (Outerbridge, unpublished data.)

Information on the population biology of diamondback terrapins in their North American range shows variation in relative body sizes, sex ratios, estimates of population size and density. Roosenburg et al. (1997) reported a population estimate of 2778-3730 individuals in the Patuxent River Estuary of Chesapeake Bay; Seigel (1984) estimated populations of 213 and 404 at two sites in east central Florida; Hurd et al. (1979) suggested that as many as 1655 terrapins inhabited the Canary Creek salt marsh in Delaware; Butler (2002) reported a population of 3147 terrapins were found to be using a northeastern Florida nesting beach; and Hart (2005) estimated the Big Sable Creek population within the Everglades National Park in southwest Florida to be 1545 individuals. It is believed that the total number of diamondback terrapins in North America may exceed 100,000 individuals (van Dijk, 2011). Density estimates of terrapins in North America are less available in the literature, but were reported to range from 53-72 terrapins/ha in central Florida (Seigel, 1984). Sex ratios in terrapin populations vary from being strongly female biased (Seigel, 1984; Roosenburg et al., 1997) to being male biased (Lovich and Gibbons, 1990). Hart (2005) reported that the sex ratio in the Big Sable Creek population was 1:1. Female terrapins can reach carapace lengths of 238 mm range wide in North America; males 140 mm (Ernst et al., 1994).

Reproduction

Bermuda's population of diamondback terrapins typically commences mating in February-March and begins egg laying in late March or early April, with peak egg laying observed in May and June. Nesting is known to

occur through the summer until late August. The average clutch size is five eggs (range 0-10) and incubation (length of time between egg deposition and first hatching) takes 49-83 days (mean 61.8 days). Bermuda's terrapins exhibit delayed emergence, with as many as 44% of the hatchlings remaining buried in their natal nests during the winter months. The majority of nesting appears to occur within the sand bunkers on the fifth, sixth and seventh holes of the Mid Ocean golf course (most notably the fifth and seventh), although some nesting has been observed in the bunkers on the eighth, ninth and eleventh holes as well. Additionally, residents along the shoreline of Mangrove Lake have reported terrapins nesting occasionally in the soil of flower beds and vegetable gardens on their properties (Fig. 6). Nest densities in Bermuda are higher than those reported in the literature, reaching as many as 2,784 nests/ha (bunker on the seventh hole, 2011). The overall nest density in the bunkers on the Mid Ocean Golf course for 2010 and 2011 was calculated to be 347/ha and 443/ha respectively (Outerbridge, unpublished data.) The reason for these high densities is believed to be primarily due to limitations in suitable nesting habitat. The mean depth of nest chambers was 13.7 cm (range 11-16 cm), the mean width was 6.7 cm (range 5-9 cm), and the mean depth of sand over the top most eggs was 9.6 cm (range 7-13 cm). Terrapin eggs in Bermuda range in length from 29.6-46.5 mm (mean 35.6 mm, SD 2.1 mm); width from 18.0-25.5 mm (mean 21.8 mm, SD 1.4 mm); and mass from 7-16 mm (mean 10.7 mm; SD 1.5 mm). These biometrics fall well within the published mean egg dimensions throughout the North American range (Butler et al., 2006). As a general rule of thumb, the northern subspecies of terrapins exhibit smaller eggs sizes but larger clutches than those subspecies found in the south.

The overall annual hatching success of Bermuda's terrapin eggs from 2009-2011 was 19%, despite the complete absence of nest and egg predators. A pilot study initiated in 2009 revealed a very low hatching success rate (18%). Of the eggs that did not hatch, 35 (70%) appeared to show no evidence of embryonic development and six (12%) contained dead embryos in various stages of development. The overall hatching success for 57 monitored nests (collectively containing 268 eggs) during the 2010 nesting season was 21%. A total of 61 hatchlings emerged, 165 eggs (61.6%) appeared to show no evidence of embryonic development, 33 (12.3%) contained dead embryos in various stages of development, and nine (3.4%) contained fully formed dead hatchlings - many of which had managed to break through the shell, but all failed to successfully emerge from their nest chambers. Twenty six of the 57 nests (45.6%) produced at least one hatchling; however 31 nests (54.4%) did not produce any hatchlings (i.e. experienced total failure). In 2011, the overall hatching success for 69 monitored nests (collectively containing 356 eggs) was 17.6%. A total of 64 hatchlings emerged, 277 eggs (77.8%) appeared to show no evidence of embryonic development, six (1.7%) contained dead embryos, and nine (2.5%) contained fully formed dead hatchlings. 30.4% of the monitored nests produced at least one hatchling; however 69.6% nests did not produce any hatchlings (i.e. experienced total failure) (Outerbridge, unpublished data).

Hatchling emergence was also studied to quantify the post-hatching nest residency periods. Emergence periods (defined as the time between

hatching and full emergence from the nest) ranged from 1-219 days. Two distinct emergence patterns were documented; July-October (during which the mean emergence time was 31.4 days), and January-March (during which the mean emergence time was 188.1 days). No emergence was observed in November and December. A similar pattern was observed in 2011 and 2012.

The observed annual hatching rates in Bermuda are low in comparison to regions which experienced no mammalian depredation within the North American range; Feinburg and Burke (2003) reported 93% hatching success during the 1980s when raccoons were absent within the Jamaica Bay Wildlife Refuge and Roosenburg et al. (2003) reported a mean hatching success rate of 92.7% at a study site devoid of mammalian predators in Maryland. Nest depredation by small mammals has been identified as a significant source of egg mortality in North America (Burger, 1977; Feinberg and Burke, 2003) however, none of the Bermuda nests monitored in 2009, 2010 or 2011 experienced any nest depredation.

Nesting ecology in North America shows variability throughout the terrapin range. Females mature at ages of 4-13 years (Seigel, 1984; Lovich and Gibbons, 1990; Roosenburg, 1991a), with those in the northern parts of the range taking longer to reach sexual maturity than those in the southern range. Males mature at much younger ages of 2-7 years (Cagle, 1952; Seigel, 1984; Lovich and Gibbons, 1990; Lovich et al., 1991; Roosenburg, 1991a; Gibbons et al., 2001). The nesting season typically begins in late April and ends in late July for terrapins in Florida (Seigel, 1980b; Butler et al., 2004), while the nesting seasons in the extreme northern range are restricted to only June and July (Burger and Montevecchi, 1975; Lazell and Auger, 1981; Goodwin, 1994; Jeyasuria et al., 1994; Feinberg and Burke, 2003). In Louisiana, egg laying may occur as late as September (Burns and Williams, 1972).

Terrapins are reported to nest on sand dunes, beaches and along the sandy margins of marshes and islands (Burger and Montevecchi, 1975; Burger, 1977; Seigel, 1980b; Roosenburg, 1994). Sand is the preferred nesting medium as it allows for sufficient gas exchange to occur between the developing embryo and the environment (Roosenburg, 1994). Nest sites are generally flat (which facilitates the postures that females assume during digging and egg deposition) with low vegetative cover (which minimizes the destruction of the nests via mammalian and plant root predation.) Diurnal nesting appears to be the standard for most terrapin populations (Burger and Montevecchi, 1975; Seigel, 1980b; Goodwin, 1994), however nocturnal nesting has been documented in some populations (Auger and Giovannone, 1979; Roosenburg, 1992). Clutch size ranges from 4-22 eggs; northern subspecies have the greatest mean clutch sizes of approximately 16 in Rhode Island (Goodwin, 1994) and 13 in Maryland (Roosenburg and Dunham, 1997), while those in Florida have mean clutch sizes of approximately seven (Seigel, 1980b; Butler, 2000). Estimated nesting densities range from 0.52/ha in Massachusetts (Auger and Giovannone, 1979) to 157.1/ha in New Jersey (Burger and Montevecchi, 1975), to 1125/ha in Maryland (Roosenburg, 1994).

Terrapins exhibit temperature-dependent sex determination (TSD) whereby the ambient temperature of the nest medium affects the sex of the developing embryos. The thermo-sensitive period (the most critical period

for sexual development) has been identified as the middle third of the incubation period, and eggs that have been artificially incubated at constant temperatures between 24-27°C produced male hatchlings while those incubated at 30-32°C produced all females (Ewert and Nelson, 1991; Jeyasuria et al., 1994; Roosenburg and Kelley, 1996). The temperatures that produce mixed sex ratios in a nest are believed to be 28.5-29.5°C (Jeyasuria et al., 1994; Roosenburg and Place, 1994), however eggs that are incubated at constant temperatures of 35°C or higher fail to hatch entirely (Cunningham, 1939). TSD has been suggested as being a factor in biased sex ratios observed in some terrapin populations (Lovich and Gibbons, 1990; Ewert and Nelson, 1991). Incubation periods (the time it takes for eggs to develop and hatch) vary from 50-120 days; in New Jersey the mean incubation period was reported to be 76.2 days (Burger, 1977), while terrapins on the east Florida coast had a mean period of 65.6 days (Seigel, 1980c). Hatching occurs from early August through to mid-October in northern terrapin populations (Burger, 1977; Roosenburg, 1991b), and from early July to early October in some Florida populations (Butler et al., 2004). Emergence periods (the time hatchlings spend in the nest prior to leaving it) show tremendous variability throughout the range; hatchlings may depart hours after hatching (Roosenburg and Kelley, 1996) or they may spend months over-wintering in the nest chamber and emerge the following spring (Lazell and Auger, 1981; Roosenburg and Kelley, 1996; Baker et al., 2006).



Figure 5. Map illustrating diamondback terrapin nesting activity encountered during the 2010 and 2011 surveys. Red dots represent nests with egg clutches; yellow dots represent nesting attempts. (A=Mangrove Lake, B=Trott's Pond, C=South Pond, D=North Pond).

Diet and feeding

Diamondback terrapins are carnivorous and selectively feed upon a variety of marine molluscs and crustaceans (namely periwinkles, crabs, mussels and clams) within the salt marsh and mangrove ecosystems throughout their North American range (see reviews in (Butler et al., 2006; Ernst and Lovich, 2009). They also show resource partitioning whereby individuals with wider heads (the largest females) consume larger snails and crabs than those terrapins that possess smaller heads (Tucker et al., 1995). Terrapins have been identified as an important component of the trophic dynamics of the salt marsh ecosystem (Silliman and Bertness, 2002; Davenport, 2011).

The foraging ecology of Bermuda's terrapins was examined using a variety of methods (direct observation, necropsy and faecal analyses). Faecal analyses, and to a limited extent necropsies, revealed that Bermuda's terrapins are consuming a wide variety of marine and terrestrial food items, but show preference towards pond gastropods. The frequency of occurrence of each food item is as follows; aquatic gastropods (*Heleobops bermudensis*,

Melanoides tuberculata, *Melampus coffeus*) occurred in 66.7% of the faecal samples, while plant material (primarily mowed grass but also saw-grass seeds *Cladium jamaicense*) occurred in 33.3% of the samples. Terrestrial arthropods (e.g. bees, beetles, isopods, millipedes, caterpillars, ants) occurred in 14.3%, fish bones and fish scales occurred in 11.9%, and cane toad bones (*Rhinella* (formerly *Bufo*) *marinus*) occurred in 4.8%. Reptile bones (*Malaclemys terrapin*), bivalves (*Isognomon alatus*) and polychaete worms (*Arenicola cristata*) occurred in 2.4% of the faecal samples respectively. Additionally, 73.8% of the terrapins in this study excreted sediment, supporting the observation that many terrapins are ingesting the sediment found on the bottom of the ponds (Outerbridge, unpublished data.) Some of the plant material (especially the mowed grass) may have been ingested inadvertently while grazing upon invertebrates and the animal prey is believed to have been consumed as carrion. Carrion eating has been reported in a New Jersey population of terrapins (Ehret and Werner, 2004). The sediment consumption is also believed to be inadvertent since the targeted food items, *M. tuberculata* and *H. bermudensis*, are benthic gastropods that inhabit areas rich in detritus and silt (Dundee and Paine, 1977; Roessler et al., 1977). The occurrence of terrestrial arthropods is believed to be from terrapins encountering and ingesting arthropods that have fallen into the ponds rather than as a result of terrapins actively foraging within the terrestrial environment.

Habitat usage

Adult and sub-adult terrapins in Bermuda appear to spend most of the time within the aquatic environment; however their abundance varies seasonally. Monthly head count surveys were conducted at South Pond (following the methods described in Butler, 2002) for a five minute period each visit. The results show that the number of observed terrapins dropped during the winter months (Fig. II in Appendix). Brumation occurs within the benthic sediment of Mangrove Lake and under the embankment of South Pond (Outerbridge unpublished data). Direct observation and the results from the mark-recapture surveys indicate that Bermuda's terrapins move freely between the various ponds, traversing overland.

Radio-telemetry was used in August 2010 and April 2011 to investigate the survival rate, post hatching movement and habitat usage of hatchling diamondback terrapins in Bermuda. Ten transmitters (BD-2 model from Holohil Systems Ltd.) were attached to the carapaces of ten newly emerged hatchlings in both years following the method described by Draud et al (2004). The hatchlings were released in sand bunkers on the fifth and seventh holes and tracked on a daily basis for a four to five week period. The results from the August 2010 session revealed that upon release all of the hatchlings moved immediately to the edge of the bunkers and either buried into the sand or crawled under the grass growing at the edge of the bunkers. Eight of the ten hatchlings remained concealed in these locations throughout the survey period; however two made major moves over the open fairways into the mangrove and saw-grass marshes bordering the ponds. The results from the April 2011 tracking session, in contrast, revealed that virtually all of the hatchlings quickly moved away from the

sand bunkers and headed towards the mangrove trees and marsh grasses. These areas appear to be critical for the development of Bermuda's hatchling and juvenile terrapins. The high level of spring-time activity, however, also makes hatchlings vulnerable to avian predation, especially by yellow crowned night herons, *Nyctanassa violacea*.

Young terrapins in the U.S.A. have been reported to seek refuge within dense mats of vegetation and debris above mean high water levels in salt marshes and tidal mudflats (Pilter, 1985; Lovich et al., 1991; Roosenburg, 1991a).

F. Current threats

Diamondback terrapins have been listed as a globally near threatened species by the International Union for the Conservation of Natural Resources (IUCN). Their status, which varies from US state to state, ranges from 'endangered' to 'a species of special concern' (Hart and Lee, 2006; Lee and Chew, 2008). Massive over-harvesting for food consumption in the late 19th and early 20th centuries led to huge declines in the North American populations, which continue to be affected by habitat loss, predation, crab trapping activities and commercial harvest for pet trade and human consumption (Roosenburg et al., 1997; Hart and Lee, 2006; Ernst and Lovich, 2009). The incidental capture and drowning of terrapins in commercial and recreational traps designed to catch blue crabs along the Atlantic and Gulf coasts continues to threaten some terrapin populations (Roosenburg, 1992; Hoyle and Gibbons, 2000), and has prompted some states to require the use of by-catch reduction devices (BRDs) on crab traps in order to minimize terrapin by-catch (Wood, 1997; Hart and Lee, 2006). Road associated mortality of nesting females has also been identified as a significant threat in some terrapin populations (Wood and Herlands, 1997). Diamondback terrapins are presently not harvested for food in Bermuda, nor are they caught as by-catch in commercial or recreational crab traps; however they are threatened with habitat fragmentation, pollution, predation, and to a limited extent, motorized vehicles and human collection.

Lack of suitable habitat

Perhaps the greatest constraint to increasing the range of Bermuda's terrapin population is a lack of suitable wetland habitat. Human activities have caused nearly all of Bermuda's wetlands to fragment and decline through deleterious habitat modification. Since the island's colonization humans have filled, dredged, drained, denuded, and polluted the ponds, marshes, and mangrove swamps in an effort to create more arable land, residential and commercial building sites, as well as waste disposal sites. During the period of marsh reclamation by garbage disposal (1920-1970), five ponds totaling 1.6 hectares were completely filled in. Widespread drainage of marshes was employed as part of the mosquito control methods in the first half of the 20th century as health officials attempted to prevent the spread of malaria. Records indicate that in the 17th century approximately 127.5 hectares of freshwater ponds, marshes and swamps existed, representing 2.4% of the

total land area of Bermuda. It has been estimated that during the 1970's 100 tons of garbage was dumped daily into the Pembroke parish marsh complex (Sterrer and Wingate, 1981). By 1980 Bermuda's total freshwater wetland area had been reduced by 65% to only 58.9 hectares (Thomas, 2004). It has been suggested that the most concentrated destruction of Bermuda's wetland communities occurred between 1941 and 1943 when 32% of the island's total mangrove acreage was destroyed on Longbird and St. David's Islands by the construction of the American-operated Kindley Air Force Base (Sterrer and Wingate, 1981). Historical writings about Bermuda's natural history fail to mention diamondback terrapins as part of the herpetological fauna (Jones, 1859; Jones and Goode, 1884; Agassiz, 1895; Hurdis, 1897; Verrill, 1902, 1903; Verrill et al., 1903), thus preventing an estimate of the former population size and also making the former distribution of the Bermuda population unclear.

A lack of suitable nesting habitat has also been identified as a current constraint to the long term growth of the population. Presently there are a few high-density nesting areas on the fifth, sixth and seventh holes of the Mid Ocean golf course that are frequented by avian predators (most notably the yellow crowned night heron *Nyctanassa violacea*) and offer little in the way of shading to the incubating eggs. Roosenburg and Place (1994) suggested that preserving only high-density nesting areas which favour the production of one sex over the other may not adequately maintain a viable terrapin population. Instead the authors recommended that a wide variety of nesting micro-habitats is necessary to maintain balanced sex ratios.

Pollution

Pollution is considered to be a relatively new threat to diamondback terrapins. Recent investigations into the health status of the pond environment in Bermuda suggest that there is a suite of contaminants of concern that are having detrimental effects on the resident fauna (Fort et al., 2006; Fort et al., 2006; Bacon, 2010; Bacon et al., 2012). These contaminants include petroleum hydrocarbons - namely gasoline-range organics (TPH-GRO) and diesel-range organics (TPH-DRO), polycyclic aromatic hydrocarbons (PAH) and heavy metals. Entry into the wetlands comes through storm-water run-off from adjacent roadways, aerial deposition and leachate from nearby landfills and ground-water sources. Ponds located within and adjacent to golf courses are among the most toxic wetlands in Bermuda (J. Bacon personal communication). Water and sediment from three of the four diamondback ponds (Trott's Pond, Mangrove Lake, and South Pond) were collected and analyzed in 2009 by Fort Environmental Laboratories. Results showed that all three ponds had highly contaminated sediment (Bacon and Fort, 2010). Tissue residue analyses from cane toads (*Rhinella marinus*), mosquitofish *Gambusia holbrooki*, killifish *Fundulus* spp., and red-eared sliders *Trachemys scripta elegans* collected from a variety of contaminated wetlands across Bermuda have showed that petroleum hydrocarbons, polycyclic aromatic hydrocarbons and heavy metals are being accumulated and inducing developmental malformations, endocrine disruption, liver and gonad

abnormalities, and immunological stress (Bacon, 2010; Bacon et al., 2012). Diamondback terrapins are known molluscivores throughout their North American range (Tucker et al., 1995), and investigations into the feeding ecology of Bermuda's diamondback terrapins have shown that they are also consuming small gastropods, which are known bio-accumulators of toxic compounds along with large quantities of benthic sediment (Outerbridge, unpublished data). Terrapins in the U.S.A. accumulate heavy metals in liver and muscle tissue (Burger, 2002), accumulate PAHs in eggs (Holliday et al., 2008), and have been used as bio-indicators of environmental contaminants in salt marsh ecosystems (Blanvillain et al., 2007; Basile et al., 2011), however the long-term effects of such exposure are unknown. Evidence indicates that total petroleum hydrocarbons (particularly the diesel-range organics) as well as polycyclic aromatic hydrocarbons (most notably fluorene, pyrene, chrysene and benzo(a)anthracene) and heavy metals (including lead, cadmium, zinc and mercury) are being accumulated by aquatic gastropods and diamondback terrapins in Bermuda (Outerbridge unpublished data).

Predation

Terrapin nests and hatchlings are preyed upon by a wide variety of predators throughout the North American range. Predators include small mammals (raccoons, skunks, foxes, rats) and birds (gulls, crows, herons), as well as ghost crabs, ants, and plant roots (most notably dune grass) (see review in Ernst and Lovich, 2009). Adult terrapins (particularly nesting females) are also occasionally preyed upon by raccoons (Seigel, 1980a; Feinberg and Burke, 2003). Draud et al (2004) reported that the Norway rat *Rattus norvegicus* was a major predator on hatchlings and juveniles (25-41 mm SCL) in a New York population, but perhaps the greatest terrapin predator is the raccoon which has been responsible for destroying 87-99% of nests in various regions in North America (Roosenburg, 1992; Feinberg and Burke, 2003; Butler et al., 2004).

Yellow crowned night herons have been identified as a significant predator to hatchling and juvenile terrapins in Bermuda. This species was observed preying upon ten neonate terrapins among the emergent pond vegetation in South Pond over a four week period between 8:00 and 18:00 hrs in the spring of 2010. Subsequent radio-telemetry investigations suggested that yellow crowned night herons may be responsible for at least 40%, and possibly up to 70%, of the mortality of hatchlings within one month of emerging from hibernacula. Furthermore, this species of heron may remain a predator to neonate terrapins for three years following hatching (Outerbridge, unpublished data).

Motorized vehicles

Observations made between 2009 and 2012 indicated that hatchling terrapins are occasionally run over by motorized vehicles (golf carts, law mowers, trucks, etc.) operating on the Mid Ocean golf course. This source of mortality is thought to be low, but each year during the survey period at least one hatchling was discovered crushed upon the cart paths between

Mangrove Lake and South Pond. It is believed that they are accidentally killed by motorists unaware of their presence on the road as they wander in search of the wetland vegetation that borders the ponds. Road mortality has been identified as a major source of death among adult female terrapins in parts of their North American range. Adult females are killed every nesting season as they search for alternative nesting sites on highway embankments along the Atlantic coast of New Jersey. During a seven year period, over 4,000 terrapins were discovered as road kill during routine patrols at one study site (Wood and Herlands, 1997).

Human collection

Anecdotal evidence suggests that some diamondback terrapins in Bermuda have been removed from ponds as pets. The total number of terrapins currently kept in captivity by members of the general public is unknown. This activity is of concern as it removes valuable individuals from the local breeding population.

Commercial interest in diamondback terrapins remains high in the U.S.A. This interest is largely driven by the pet trade industry, and most specimens are exported to Asian markets where hatchlings can sell for US\$ 50-100 (Anonymous, 2013).

G. Current conservation action

Artificial incubation of terrapin eggs collected from the wild was first attempted at the Bermuda Aquarium Museum and Zoo in 1994. The hatching success was very limited (only four eggs produced hatchlings out of 18 eggs collected from three different clutches) and three of the individuals (one died) were subsequently kept on display at BAMZ for a number of years (*R. Marirea pers. comm.*) Egg incubation was re-attempted in 2012 during which 74 eggs were collected from ten nests located in the sand bunkers between the fifth and seventh holes on the Mid Ocean golf course. Thirty three eggs (44.6%) developed into hatchlings, of which 29 were subsequently released into the wild (four hatchlings died in captivity shortly after hatching).

Raising awareness about the vulnerable status of this fragile oceanic population is on-going, with organized public and private lectures occurring throughout the calendar year. Bermuda's terrapins have featured in several local newspaper articles, in local and international magazines as well as on a local television documentary. A representative from Bermuda has been actively participating in the triennial Diamondback Terrapin Working Group symposia since 2007 and maintains open dialogue with the south-eastern regional group (to which Bermuda is a member).

Part II: RECOVERY

A. Recovery goal

The principal aim of this Recovery Plan is to increase both the population level and the areas of residency for diamondback terrapins in Bermuda. The short term goal (five years) is to continue to research the biology and ecology of Bermuda's diamondback terrapins, as well as assess the suitability of appropriate habitats and ensure their protection, in order to promote effective management. The long term goal (30 years) is to increase the population levels and range of Bermuda's terrapins, enhancing natural recruitment and restoring wetland habitats.

B. Recovery objectives and criteria

Favorable conservation status will be achieved when:

- The genetic diversity of Bermuda's extant population is fully understood.
- All current and potential habitats suitable for diamondback terrapin growth, reproduction and survival are identified, assessed, restored and protected under legislation.
- Diamondback terrapins are viable residents in at least three separate geographic locations throughout Bermuda.
- Population levels in Bermuda indicate that terrapins are successfully maintaining themselves on a long-term basis and showing adequate levels of recruitment.

These overall objectives translate into specific targets outlined below:

Short-term target (five years): To ensure that by 2018 all studies necessary for development of effective management will be complete, and that both species and habitat will be protected under legislation. Habitats will be identified as "Critical Habitat" and designated as such under law, should they be considered crucial to the recovery of the species. This short-term goal includes examining the impact that environmental pollution has upon terrapin health and additional investigations to determine sources of threats to their survival. During this time, the identification and assessment of "health" status of current and potential habitats will be conducted.

Long-term target (30 years): Following the habitat assessments, restoration of habitats deemed suitable for diamondback terrapins will lead to the potential to increase both the area of occupancy and population within each pond. Artificial egg incubation and head-starting of hatchlings may be needed to achieve this long-term goal. Monitoring of efforts will be necessary to evaluate survival and growth of newly established populations, and determine their self-sustainability.

C. Recovery strategy

The species addressed in this recovery plan are currently restricted in both population size (approx. 100 individuals ≥ 81 mm SCL) and range (total area of residency is less than 1 km²). Bermuda's wetlands are easily impacted upon by physical disturbances (e.g. development), chemical processes (e.g. fertilizer, pesticide, herbicide and road run-off from surrounding lands) and ecological processes (e.g. encroachment of invasive species). In the case of the ponds on the Mid Ocean golf course, these activities are primarily via course maintenance which results in disturbance and fragmentation of the various habitats required during each stage of the life cycle (e.g. nesting and juvenile developmental habitats.) The strategy for recovery revolves around the protection of wetland habitats, the assessment of their "health" status, namely sediment and water quality, their remediation in some cases, and in the active intervention required for increasing the species distribution to a greater range. The selection of ponds for translocation is critical as habitat quality appears poor in several areas, based on previous sediment analyses and toxicological examination of red-eared sliders (J. Bacon, *pers. comm.*). This further drives the need for habitat protection of "healthier" ponds, controlling as much as possible input from external sources. It is believed that contaminants appear to be entering some of the ponds through groundwater, atmospheric deposition and/or road run-off (Bacon et al., 2013). Predator control should be seasonally employed in order to reduce hatchling mortality and increase recruitment to the existing population, stock enhancement via artificial egg incubation and captive rearing of hatchlings should be considered as a tool for the establishment of populations in sites considered adequate, and success for growth and survival of the species further ensured via legislated habitat protection.

D. Tools available for strategy

One tool is to seek collaboration with partner institutions that already have experience in successful research and conservation activities. In 2011 Bermuda, through a regional representative (MO), became a life-time member of the Diamondback Terrapin Working Group, which is a body of people and organizations committed to research, conservation management and education efforts that benefit terrapin populations and their associated ecosystems. Terrapins are an ideal species for captive rearing as demonstrated by the Wetlands Institute in New Jersey which has a 20 year history of successfully incubating eggs and head-starting young diamondback terrapins (Wood and Herlands, 1997; Herlands et al., 2004). Additionally, there is information available on the levels of contaminants, such as heavy metals, pesticides, pharmaceuticals, total petroleum hydrocarbons (TPHs) and polycyclic aromatic hydrocarbons (PAHs) for some of Bermuda's ponds, including South Pond, Trott's Pond and Mangrove Lake. Sediment analyses and red-eared slider tissue analyses have been conducted, providing data on suitability of selected ponds and the health of their resident sliders. Necropsies on sliders from a number of ponds have also indicated abnormalities in reproductive tissue and should be

taken into consideration when planning future translocation programmes. All of this data is documented by Drs Jamie Bacon (Bermuda Zoological Society) and Douglas Fort (Fort Environmental Laboratories Inc.)

E. Step-down narrative of work plan

Abbreviations:

DCS – Department of Conservation Services
DPW – Department of Public Works
Parks – Department of Parks
Planning – Department of Planning
DEH - Department of Environmental Health
AG - Attorney General's Chambers
MOC – Mid Ocean Club
BZS – Bermuda Zoological Society
BNT – Bermuda National Trust
BAMZ – Bermuda Aquarium Museum and Zoo
USGS – United States Geological Survey
FEL – Fort Environmental Laboratories

The actions needed to achieve recovery are as follows:

1. Protect wetland habitats of extant terrapin population through legislation,
2. Restore protected wetland habitats of current extant population,
3. Identify, assess, protect and restore wetland habitats deemed suitable for diamondback terrapin introduction,
4. Increase population size through increased hatching success and recruitment to the adult population,
5. Expand area of residency through translocation of individuals raised in captivity,
6. Identify the full genetic composition of existing population,
7. Develop research programmes on understanding the effects that environmental contaminants have upon the reproductive biology and overall health of terrapins in Bermuda,
8. Promote conservation education programmes concerning Bermuda's terrapin population,
9. Continued population monitoring.

1. Protect wetland habitats of extant terrapin population through legislation.

Actions proposed:

- Designation of Mangrove Lake, South Pond, North Pond, and Trott's Pond as "critical habitat" for Bermuda's diamondback terrapins.

Work team: DCS

Team leader: DCS

Assistance: AG

Outputs: Legislation for habitat protection

List of equipment required: GPS for boundary delineation, GIS mapping applications.

2. Restore protected wetland habitats of current extant population.

Actions proposed:

- Diversify and increase the area of nesting habitat,
- Increase the area required for neonate and juvenile development (including the establishment of terrapin corridors between nest sites and wetlands),
- Produce habitat management and landscaping guidelines for land owners bordering the ponds,
- Create buffer zones between road drains and ponds,
- Initiate remediation of select ponds where appropriate (e.g. use of diatomaceous earth to bind pollutants in sediment, cyclically plant and remove vegetation known to absorb pollutants and increase the activity of indigenous bacteria that are capable of metabolizing pollutants),
- Monitor sediment and water quality in South Pond, North Pond, Mangrove Lake and Trott's Pond.

Work team: DCS, MOC, DPW and collaborative institution for sample analyses

Team leader: DCS

Assistance: BZS, BNT and private land owners

Outputs: Creation of a more terrapin-friendly environment that promotes long-term population stability.

List of equipment required: Beach sand for creation of nesting habitat, funding required for laboratory analyses of sediment and water samples.

3. Identify, assess, protect and restore wetland habitats deemed suitable for diamondback terrapin introduction.

Actions proposed:

- Survey all of Bermuda's wetlands for suitable expansion habitats,
- Designate identified wetlands as "critical habitat" for diamondback terrapins,
- Produce habitat management guidelines for terrapins,
- Remove red-eared sliders from wetlands identified as suitable for terrapin introduction,
- Initiate remediation of select ponds where appropriate (e.g. use of diatomaceous earth to bind pollutants in sediment, cyclically plant and remove vegetation known to absorb pollutants and increase the activity of indigenous bacteria that are capable of metabolizing pollutants).

Work team: DCS, Parks, DPW, Planning and AG

Team leader: DCS

Assistance: BNT, DEH and private land owners

Outputs: Creation of a greater diversity of terrapin-friendly wetlands that promotes long-term population stability.

List of equipment required: Boat, traps and bait for the capture of feral red-eared sliders.

4. Increase population size through increased hatching success and recruitment to the adult population.

Actions proposed:

- Reduce and control predators (e.g. yellow-crowned night herons and rats), especially during periods of hatchling emerge, in areas where hatchlings and small juvenile terrapins reside,
- Increase ground cover between nest sites and wetlands by establishing terrapin corridors using natural vegetation,
- Relocate terrapin nests from areas subjected to frequent disturbance (i.e. sand bunkers on golf course) to areas subjected to less disturbance,
- Initiate an artificial egg incubation and head-starting programme.

Work team: DCS, MOC

Team leader: DCS

Assistance: BZS, members of the public

Outputs: Enhancing population size of natural stocks and engaging community in preservation of threatened native species.

List of equipment required: Egg incubator, head-starting tanks, rat poison.

5. Expand area of residency through translocation of individuals raised in captivity.

Actions proposed:

- Assess requirements for most favorable transfers of captive raised individuals to suitable wetlands,
- Introduce juvenile terrapins into suitable wetlands in equal sex ratio,
- Monitor populations via a mark-recapture programme.

Work team: DCS

Team leader: DCS

Assistance: Members of the public

Outputs: Assessment of terrapin populations following translocation, increasing range of occupancy and optimizing survival of the species, data on terrapin requirements for optimal growth and survival.

List of equipment required: Boat, traps and bait for capture of diamondback terrapins.

6. Identify the full genetic composition of existing population.

Actions proposed:

- Continued collection of tissue samples,
- Analysis of collected samples

Work team: DCS and USGS

Team leader: DCS

Assistance: Dr. Kristen Hart (USGS)

Outputs: Determination of genetic diversity of extant population in Bermuda and a population level genetic scientific publication.

List of equipment required: Boat, traps and bait for capture of diamondback terrapins and funding required for laboratory fees.

7. Develop research programmes on understanding the effects that environmental contaminants have upon the reproductive biology and overall health of terrapins in Bermuda.

Actions proposed:

- Collect terrapin blood samples for hormone and heavy metal analyses,

- Monitor red-eared sliders at select locations via necropsy and tissue analyses for metals, total petroleum hydrocarbons (TPHs) and polycyclic aromatic hydrocarbons (PAHs).

Work team: DCS and collaborative institution for necropsies, tissue and blood sample analyses (FEL)

Team leader: DCS

Assistance: Graduate student for research studies

Outputs: Determination of eco-toxicological effects on terrapins in Bermuda and a scientific publication.

List of equipment required: Boat, traps and bait for capture of sliders and terrapins, funding required for laboratory fees.

8. Promote conservation education programmes concerning Bermuda's terrapin population.

Actions proposed:

- Create and post cautionary and interpretive signage at relevant locations on the Mid Ocean golf course that explains the natural history of terrapins as well as the threats facing the species (e.g. turtle crossing signs at locations on the cart paths adjacent to Mangrove Lake and South Pond),
- Perform periodic presentations to public on the ecology and conservation of Bermuda's terrapin population,
- Publish scientific papers based upon research findings in addition to annual management plan progress reports.

Work team: DCS, MOC

Team leader: DCS

Assistance: BZS

Outputs: Engaging community in preservation of native terrapins.

List of equipment required: Text and image materials for signage

9. Continued population monitoring.

Actions proposed:

- Monitor all terrapin populations via a mark-recapture programme.

Work team: DCS

Team leader: DCS

Assistance: Volunteer interns

Outputs: Comprehensive assessment of existing and re-established populations.

List of equipment required: Materials for population surveys (boat, traps, bait, calipers, spring scales.)

F. Estimated date of down listing

It is anticipated that it will take at least five years to identify and restore key habitats for Bermuda's terrapins, and one year to complete the first head-starting and translocation initiative. Diamondback terrapins are a slow growing, long-lived species therefore programmes developed to aide in their recovery need to recognize that there may be long delays before favorable responses can be detected. It is only once implemented actions are evaluated that down listing (or removal) of this species will be considered, following assessments of population distribution and habitat quality monitoring. Re-assessment of this species should be done every ten years.

Part III: IMPLEMENTATION

Priority 1: An action that must be taken to prevent extinction or to prevent the species from declining irreversibly.

Priority 2: An action that must be taken to prevent a significant decline in the species population/habitat quality, or some other significant negative impact short of extinction.

Priority 3: All other action necessary to provide for full recovery of the species.

Priority #	Task #	Task description	Responsible Party
1		Protection of wetland habitats of extant population	
	1	Designation of current sites as 'critical habitat'	DCS, AG
1		Restoration of wetland habitats of extant population	
	2	Diversify and increase the area of nesting habitat	DCS, MOC
	3	Increase the area required for hatchling and juvenile development	DCS, MOC
	4	Produce habitat management guidelines	DCS
	5	Create buffer zones between road drains and ponds	DPW
	6	Initiate remediation of select ponds where appropriate	DCS
	7	Monitor sediment and water quality	DCS
2		Identification and assessment of additional wetland habitats for translocation	
	1	Survey for suitable expansion habitats	DCS
	2	Designate identified wetlands as 'critical habitat'	DCS, AG
	3	Produce habitat management guidelines	DCS
	4	Remove red-eared sliders	DCS
	5	Initiate remediation of select ponds where appropriate	DCS, DPW
2		Enhance population numbers	
	6	Control predators	DCS, MOC
	7	Increase ground cover between nest sites and wetlands	MOC
	8	Relocate terrapin nests	DCS
	9	Initiate an artificial egg incubation and head-starting programme	DCS, BAMZ

2		Expand area of occupancy through translocations	
	10	Assess requirements for successful transfers	DCS
	11	Introduce captive raised juvenile terrapins	DCS, BAMZ
	12	Monitor populations	DCS
3		Research genetic composition	
	1	Collection of tissue samples	DCS
	2	Analyses of collected samples	DCS, USGS
3		Research into effects of contaminants	
	3	Collect terrapin blood samples	DCS
	4	Monitor red-eared sliders at select locations	DCS, FEL
3		Promote conservation education	
	5	Create and post cautionary and interpretive signage	DCS, MOC
	6	Continue public presentations	DCS
	7	Publish scientific papers and annual reports	DCS
3		Continued population monitoring	
	8	Monitor all terrapin populations	DCS

APPENDIX

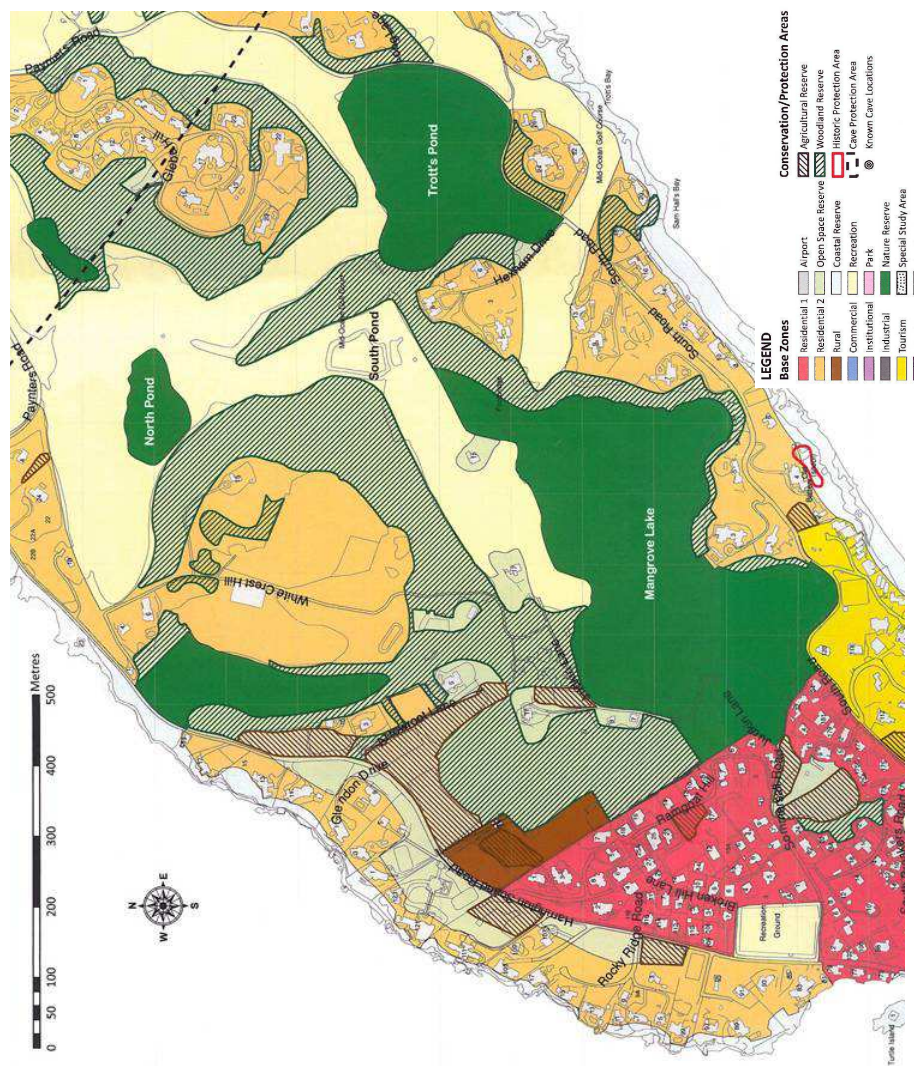


Figure I. Development base zones for the area inhabited by Bermuda's population of diamondback terrapins (adapted from the 2008 Bermuda Plan)

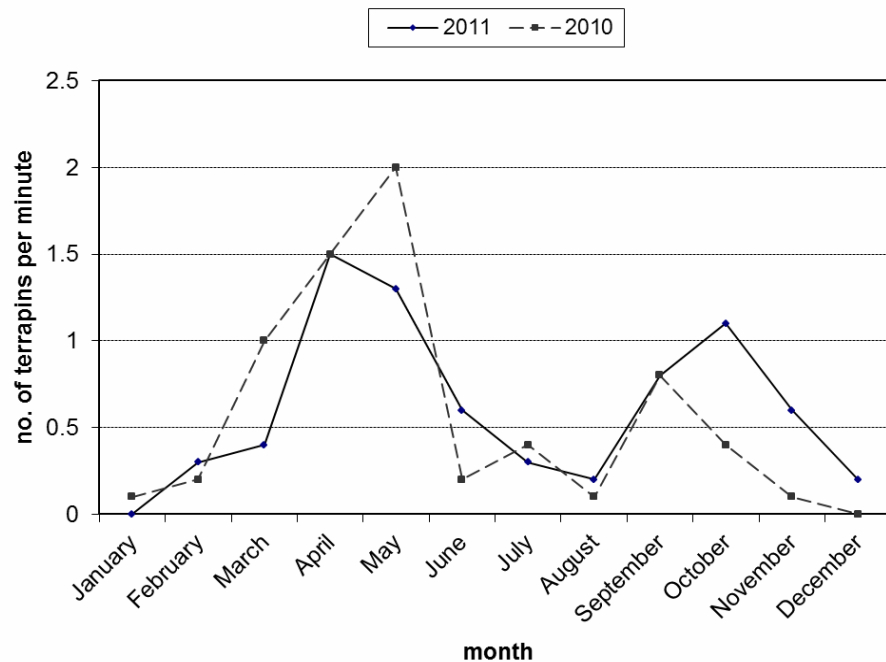


Figure II. Diamondback terrapin head count surveys in 2010 and 2011 at South Pond.

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