2011

Biology and Conservation of the Diamondback Terrapin, Malaclemys terrapin pileata, in Alabama

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Recommended Citation

Coleman, Andy, "Biology and Conservation of the Diamondback Terrapin, Malaclemys terrapin pileata, in Alabama" (2011). *All ETDs from UAB*. 1403.
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BIOLOGY AND CONSERVATION OF THE DIAMONDBACK TERRAPIN, *MALACLEMYS TERRAPIN PILEATA*, IN ALABAMA

by

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A DISSERTATION

Submitted to the graduate faculty of The University of Alabama at Birmingham,
in partial fulfillment of the requirements for the degree of
Doctor of Philosophy

BIRMINGHAM, ALABAMA

2011

BIOLOGY AND CONSERVATION OF THE DIAMONDBACK TERRAPIN, *MALACLEMYS TERRAPIN PILEATA*, IN ALABAMA

ANDREW THOMAS COLEMAN

BIOLOGY

ABSTRACT

The diamondback terrapin is the only obligate estuarine turtle in North America and is considered an integral member of the salt marsh ecosystem. Unfortunately, many populations throughout the terrapin's range have experienced declines due to past overexploitation and have been unable to rebound due to current threats, including crab trap mortality, habitat degradation, nest predation, and road mortality. The current study was the first comprehensive study examining various population and conservation parameters of Mississippi diamondback terrapins in Alabama. Through various field survey methods conducted in numerous salt marshes along the Gulf Coast of Alabama, it was concluded that Cedar Point Marsh supported the largest aggregation of terrapins in Alabama, and the beach bordering Cedar Point Marsh represented the most important nesting habitat. However, population estimates indicated a significant size reduction from historical levels, and crab trap mortality and nest predation were identified as major threats currently impacting this population. The major population decline in Alabama was also reflected in the terrapins' genetic diversity, whose low diversity was similar to other sampled terrapin populations. By-catch reduction devices were shown to be an effective management tool to prevent terrapin entry into crab traps, although decreases in crab capture were observed. Obtaining eggs from nesting females to help offset nest predation allowed investigations of female allocation strategies and post-emergence orientation behavior of hatchlings. Larger and older females produced larger eggs and hatchlings, but the advantage of larger hatchling sizes was not detected with the potential fitness indicators examined in this study. The consequences of high levels of road mortality, which would theoretically result in removing older females, were examined, and the Alabama population, which does not experience high road mortality, produced larger eggs than the Georgia population that does suffer from this threat. Terrapin hatchlings utilized the same orientation cues as sea turtle hatchlings but moved toward the higher marsh areas rather than open water. This underscored the necessity of healthy marsh habitat adjacent to nesting beaches. The initiation of this long-term dataset is crucial in developing optimal management strategies for ensuring the future survival of diamondback terrapins in Alabama.

conservation—bycatch—nest predation—habitat loss—microsatellites—turtles

DEDICATION

To my mother and father, Connie and Mark Coleman, whose love and support made this dissertation possible. They never wavered when their youngest son decided to study salamanders and turtles when other sons and daughters pursued normal 9-5 careers.

To the residents of the Gulf Coast who truly care for their surrounding natural beauty and whose resilience and determination are unsurpassed.

ACKNOWLEDGEMENTS

I first would like to thank the members of my committee: Thane Wibbels, Ken Marion, Bob Thacker, Willem Roosenburg, and David Nelson. The insight and advice they provided with the research and dissertation have been greatly appreciated. I would also like to acknowledge the major funding resources for this project: Alabama Center for Estuarine Studies and Alabama Department of Conservation and Natural Resources. I also received funding from University of Alabama at Birmingham Biology Department, Alabama Academy of Science, Birmingham Audubon Society, and University of Alabama Graduate Student Association. I would like to thank our collaborators: John Dindo, Kristen Hart, and Jordan Gray. The support I received from the Department Chair, Bud Fischer, and Graduate Program Director, Steve Watts, was extremely helpful. I have also been the recipient of great help from the Biology Department office staff and lab coordinators. Additionally, I would like to thank my friends who endured countless turtle anecdotes and enjoyed sharing their favorite turtle jokes. And, I would be remiss if I did not thank Mrs. Kathy Wibbels for all of the time and effort she provided to me and the rest of the Wibbels' lab. I consider myself lucky to have had the great opportunity to study under Thane Wibbels and Ken Marion whose guidance made me a better scientist and whose stories in the field made my time here just that much more enjoyable. Finally, this degree would have been impossible without the love, compassion, and care given freely by my parents, Mark and Connie Coleman, and the rest of my family including Jeremy, Julie, Julian, and Rebecca Coleman.

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CHAPTER 1

INTRODUCTION

DEVELOPMENT OF RECOVERY PROGRAMS FOR THE DIAMONDBACK TERRAPIN: EVALUATION OF THREATS AND CONSERVATION STRATEGIES

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Submitted to Chelonian Conservation and Biology

Format adapted for dissertation

Abstract

The survival status of the diamondback terrapin, the only North American estuarine turtle, is being impacted by a wide variety of threats. Terrapins were once abundant in the salt marshes lining the Atlantic and Gulf coasts of the U.S. They represented a valuable economic resource that was exploited as a culinary delicacy. Although, they are no longer subject to range wide commercial exploitation, terrapin populations have not rebounded to historic levels. To the contrary, many populations have declined to the point at which their survival requires protection and management through conservation programs. Current threats, including habitat loss, crab trap and road mortality, and nest predation, are significantly affecting terrapins throughout their range. The following review discusses the wide variety of threats that are impacting terrapins and explores conservation strategies that could mitigate these threats, and thus enhance the recovery of terrapin populations.

Key Words

diamondback terrapins—conservation—habitat loss—bycatch—head-starting—depredation—road mortality—crap trap

Introduction

The diamondback terrapin (*Malaclemys terrapin*) is of distinct ecological interest because it is the only turtle, and one of the few reptiles, to exclusively inhabit bays, estuaries, and salt marshes in North America (Brennessel, 2006). While it is an obligate inhabitant of a brackish water environment, it belongs to the freshwater turtle family Emydidae, and its closest relatives are the map turtles of the genus *Graptemys* (Lamb and Osentoski, 1997). Despite their freshwater ancestry, terrapins have evolved certain physiological and behavioral adaptations that allow them to thrive in these harsh brackish water environments (Dunson, 1970; Gilles-Baillen, 1970; Gilles-Baillen, 1973; Robinson and Dunson, 1976; Davenport and Macedo, 1990; Davenport and Magill, 1996; Brennessel, 2006). It is also of ecological interest because it is an integral part of the salt marsh ecosystem representing a top level predator and potential keystone species (Tucker *et al.*, 1995; Silliman and Zieman 2001; Silliman *et al.*, 2005; Gustafson *et al.*, 2006). It is known to feed on a variety of invertebrates including periwinkle snails of the genus *Littorina,* blue crabs, *Callinectes sapidus,* and a variety of other mollusks, crustaceans, and fishes (Tucker *et al.*, 1995).

Diamondback terrapins have experienced a rich and diverse cultural history over the past three centuries in North America (Carr, 1952; Brennessel, 2006; Hart and Lee, 2006; Schaffer *et al.*, 2008). This history indicates that they were once numerous throughout most of their range and were used as an abundant food source as well as an important economic resource. Interestingly, terrapins made a transition from a nuisance species to a top culinary delicacy (Carr, 1952; Brennessel, 2006; Hart and Lee, 2006; Schaffer *et al.*, 2008). During the 1700's terrapins were often considered nuisance by-catch and represented an inexpensive food source. For example terrapins were fed to servants and slaves in some coastal plantations as well as the Continental Army (Brennessel, 2006; Hart and Lee, 2006; Schaffer *et al.*, 2008). During the early 1800's, terrapins made a remarkable transition in popularity as stew and soup recipes proliferated and turned the diamondback terrapin into a culinary delicacy. This popularity remained high and by the mid 1800's there was commercial harvesting in areas ranging from Maryland to Texas (Brennessel, 2006). By the end of nineteenth and beginning of the twentieth centuries, terrapin stew became almost a mandatory dish at upscale restaurants and social gatherings. The demand for terrapin was so great during the mid 1800’s through the early 1900’s that natural stocks declined due to decades of overharvesting, and by the early 1900’s stocks in many areas were too low to sustain commercial harvest (Coker, 1920; Babcock, 1926; Finneran, 1948; McCauley, 1945; Schaffer *et al.*, 2008). In an effort to meet the demand for terrapin, commercial aquaculture was initiated in the late 1800’s and was enticing enough that a stocking program was initiated by the federal fisheries at Beaufort, North Carolina (Hildebrand and Hatsel, 1926).

A decline in the demand for terrapins began in early 1900’s for a variety of reasons. In addition to declining terrapin populations, the social and economic atmosphere associated with World War I and then the Great Depression decreased the demand for this high priced reptile, and Prohibition prevented the purchase of the sherry wine, popular in terrapin soups and stews (Brennessel, 2006; Hart and Lee, 2006; Schaffer *et al.*, 2008). Collectively, these factors resulted in a distinct decline in the demand for terrapins during the first few decades of the 20th century. Although some local populations may have been wiped out by the decades of overharvesting, some began to slowly recover due to the decreased harvesting (Coker, 1951; Hurd *et al.*, 1979; Brennessel, 2006; Hart and Lee, 2006; Schaffer *et al.*, 2008). However, during the 1940’s and 1950’s the crab fisheries along the Atlantic and Gulf Coasts of U.S. began large scale utilization of crab traps as an efficient method for harvesting crabs (Kennedy *et al.*, 2007). As the crab fishery flourished, the impact on terrapin populations significantly increased due to incidental capture and mortality of terrapin in crab traps (Roosenburg *et al.*, 1997; Wood 1997; Dorcas *et al.*, 2007).

As reviewed below, crab traps represent a major threat to terrapin populations and have caused significant declines in populations throughout the Atlantic and Gulf coastal waters of the U.S. Further, other threats including increased coastal development along with heavy predator load on nesting beaches have also impacted terrapins in recent years (Feinburg and Burke, 2003; Butler *et al.*, 2004; Seigel, 1980). Collectively, such threats have impacted terrapin populations throughout their range and have prevented their recovery. The purpose of the current manuscript is to provide a comprehensive overview of the major threats affecting the diamondback terrapin and review potential strategies for enhancing their recovery, including the review of strategies implemented for other reptile recovery programs when appropriate.

Habitat Loss

Although the range of terrapins extends along the majority of the Atlantic and Gulf coasts of the United States, it exists as a relatively thin strip of estuarine habitat, (Carr, 1952). Further, the total amount of this estuarine habitat has been decreasing yearly. According to a 2008 study, over 350,000 acres were lost in coastal watersheds in the eastern U.S. between 1998-2004, and losses in the Gulf of Mexico region were 25 times higher in that same time period (Stedman and Dahl, 2008). Of the designated wetland types, salt marshes sustained the heaviest losses. Many of these losses can be attributed to anthropogenic effects associated with coastal development (Stedman and Dahl, 2008; Hartig *et al.*, 2002). Dredging, filling, shoreline hardening, and a variety of other activities alter natural processes and contribute to the loss of salt marsh habitat (Stedman and Dahl, 2008) and diamondback terrapin nesting habitat (Roosenburg, 1991). As an example, it has been estimated that 38% to 78% of salt marsh habitat associated with islands in Jamaica Bay (New York City, NY) has been lost since 1974 (Hartig *et al.*, 2002). Compounding the loss of habitat from anthropogenic effects, loss of salt marsh habitat can also be accelerated episodically by natural events as exemplified by Hurricane Katrina in 2005 (Stedman and Dahl, 2008).

Impact of Global Climate Change on Terrapin Habitat

Another threat which could significantly impact the terrapin's habitat is global climate change (IPCC, 2007). Of particular importance, models project significant increases in sea level and temperature over the 21st Century (IPCC, 2007). The potential effects of global climate change on salt marshes have been discussed by a variety of authors (Donnelly and Bertness, 2001; Simas *et al.*, 2001; Hartig *et al.*, 2002; Scavia *et al.*, 2002; Hughes, 2004; Najjar, *et al.*, 2000). Sea level rise has the potential of drowning salt marshes, but it has been suggested that is some cases, salt marshes may be able to survive by accreting sedimentation vertically (Simas *et al.*, 2001). Salt marshes with large tidal ranges and high sediment transport would be predicted to more capable of compensating for sea level increases, whereas those with small tidal ranges and low sediment transport would be more susceptible to drowning (Simas *et al.*, 2001).

Terrapins inhabit a relatively thin strip of estuarine environment lining the Atlantic and Gulf coasts of the U.S., and in many cases, those habitats are now bordered by coastal development. Therefore, if sea level rise negatively impacts salt marsh, this would result in the narrowing or disappearance of terrapin habitats in some locations (Donnelly and Bertness, 2001; Hartig *et al.*, 2002; Hughes, 2004).

In addition to sea level rise, temperature change could also have a variety of impacts on the salt marshes, including primary productivity, eutrophication, and dissolved oxygen content (Najjar *et al.*, 2000; Scavia *et al.*, 2002), all of which could affect the ecology of the diamondback terrapin. Global temperature change could also affect the reproductive ecology of terrapins, including the timing of reproduction (Marion, 1982; Ganzhorn and Licht, 1983; Mendonça, 1987) as well as sex determination. In the case of the terrapin's temperature-dependent sex determination, a 1.0 ºC increase temperature could result in a shift from an approximate 1:1 sex ratio to the production of all female hatchlings (Jeyasuria and Place, 1997; Wibbels, 2003), and current models suggest a temperature increase ranging from 1.8 to 4.0 C during the 21st Century (IPCC, 2007). Thus, although there are many immediate threats that are typically addressed when generating a recovery strategy, global climate change could emerge as a major factor threatening the survival of terrapin over the next century.

Habitat Quality and Habitat Pollution

In addition to the loss of salt marsh habitat, factors such as pollution, invasive species, freshwater diversion, shoreline development, and loss of natural species can adversely affect the structure and quality of the salt marsh ecosystem (Bertness *et al.*, 2002; Stedman and Dahl, 2008). For instance, shoreline development has been shown to increase invasive species and decrease species diversity in adjacent salt marshes (Bertness *et al.*, 2002; Silliman and Bertness, 2004). The preservation of salt marsh quality is of paramount importance to maintaining the diversity and stability of the salt marsh (Pennings *et al.*, 2002). A variety of ecosystem functions have been attributed to salt marshes, some of which extend to adjacent ecosystems (Richardson, 1994). For example, the salt marsh represents a nursery ground for many fishes and invertebrates which may play vital roles in the food web of the salt marsh as well as adjacent ecosystems (e.g. adjacent marine and freshwater habitats). In the case of the diamondback terrapin, its survival is dependent upon a healthy salt marsh ecosystem. Unfortunately, certain pollutants such as heavy metals have been shown to accumulate in salt marshes (Giblin *et al.*, 1980; Williams *et al.*, 2003), and terrapins have been shown to accumulate heavy metals in salt marshes that are exposed to this form of pollution (Burger, 2002; Blanvillian *et al.*, 2007).

The Deepwater Horizon oil spill in the northern Gulf of Mexico during 2010 and the Chalk Point oil spill in the Patuxent River in Maryland during 2000 (Michel *et al.*, 2001) have highlighted the potential impact of oil pollution on estuaries inhabited by terrapin. In the case of the Chalk Point oil spill, it was estimated to cause the mortality of approximately 826 hatchlings and 122 adults and juveniles (Michel *et al.*, 2001), however, these predictions were based on a model. Holliday *et al.* (2008) measured polycyclic aromatic hydrocarbon (PAH) levels in terrapin eggs a year after the Chalk Point oil spill. The observed levels of PAH's were low and were not strongly correlated with shoreline oiling. A nest from a low impacted site displayed the highest PAH concentrations. The authors attributed the oil exposure to maternal transfer reflecting background levels instead of direct effects of the oil spill (Holliday *et al.*, 2008).
Toxic effects of PAH's have also been examined in freshwater turtle species. Van Meter *et al.* (2006) observed snapping turtle (*Chelydra serpentine*) embryo mortality as well as developmental abnormalities after exposure to oil. Bell *et al.* (2006) also detected deformities in adult snapping turtles from the same population in addition to painted turtles (*Chrysemys picta*). The authors credited these effects to PAH exposure because

high levels of PAH’s were observed in the fat reserves of the snapping turtles (Bell *et al.*, 2006).

The toxic effects of oil pollution have also been examined in sea turtles (Milton *et al.*, 2003). Sea turtles may have a higher risk oil exposure due to the wide range of habitats that sea turtles utilize during various stages in their life history (Milton *et al.*, 2003). That study noted direct effects of oil pollution on the physiology and behavior of sea turtles. The authors suggested that oil pollution could also have a variety of indirect effects such as masking odors that may be important as foraging or orientation cues. In addition, exposure during the smaller life history stages (i.e. hatchlings) could be more harmful due to smaller overall size, immature metabolic physiology that is not able to properly detoxify, and higher concentration of lipids to which contaminants can attach (Milton, *et al.*, 2003). The authors indicated that oiled nesting beaches could represent another negative impact for sea turtles. Both females and hatchlings have to travel across oil that is present on the beach and oil can also have damaging effects on the nests. It can significantly modify gas exchange, the surrounding hydric environment, and the incubation temperatures of the nests. Altered incubation temperatures can affect embryonic development and sex determination (Milton *et al.*, 2003). However, these impacts appear to dissipate over time because sand measured the year after an oil spill did not have measureable effects on hatchling survival or morphology (Milton *et al.*, 2003).

Therefore, loss of salt marsh habitat, salt marsh pollution, and decrease in salt marsh quality represent significant threats to terrapin conservation. While terrapins are dependent upon salt marsh for their survival, they also enhance the stability and health of the salt marsh. Terrapins are an important component in the salt marsh food web feeding on a wide variety of prey (Tucker *et al.*, 1995). Their predation on *Littorina* periwinkle snails can play a pivotal role in maintaining overall salt marsh health. One of the major vegetation types in southeastern U.S. salt marshes is *Spartina alterniflora*, and Silliman *et al.* (2005) and Gustafson *et al.* (2006) both observed the deleterious effects of unchecked grazing of *Littorina* snails on *Spartina* vegetation. Plant biomass decreased to such an extent that all that was left were exposed mudflats (Silliman *et al.*, 2005). Gustafson *et al.* (2006) suggested that predator control of *Littorina* snails (such as diamondback terrapins) is important in preserving ecosystem integrity and function of salt marshes. The connection between diamondback terrapin abundance and salt marsh health provides an ecosystem-wide rationale for terrapin conservation. If threats result in the loss of a terrapin population from a salt marsh, it may represent a long-term or permanent loss for the salt marsh. Though adult males disperse during mating season (Hauswaldt and Glenn, 2005), overall, terrapins display high site fidelity. So if a local aggregation is extirpated, it may take years if ever for it to become re-established (Gibbons *et al.*, 2001).

Road Mortality and Habitat Fragmentation

The construction of roadways in environmentally sensitive areas has led to habitat fragmentation and road mortality in a variety of organisms (Gloyne and Clevenger, 2000; Aresco, 2003; Szerlag and McRoberts, 2006). For terrapins, road mortality represents a significant threat to adults during reproductive migration and can significantly decrease a population's reproductive potential (Wood and Herlands, 1997). In particular, adult females are vulnerable as they migrate to suitable nesting sites through habitats that have been fragmented by roadways. For example, on the Cape May Peninsula of New Jersey, over four thousand adult females were killed between 1989 and 1995 on 11.5 km of roadways that dissect the terrapins' habitats (Wood and Herlands, 1997). Because natural sand dunes that were once used for nesting had been removed during coastal development, increased numbers of females nest along those roads' embankments (Wood and Herlands, 1997). Road mortality may have resulted in a change in this local terrapin population structure with a decrease in the number of adult females in comparison to the same population in the late 1980s (Avissar, 2006). Further, the adult females that were captured were smaller, and because size can correlate with age, the author suggested that the average age of adult females in the population had decreased (Avissar, 2006).

Mitigating Road Mortality

The threat of road mortality has been addressed in other turtle species. Aresco (2003) provided a detailed account of how high amounts of traffic on U.S. Highway 27 adjacent to Lake Jackson, FL affect the local herpetological fauna. Before fencing was inserted, over 350 turtles were found on the short stretch of highway in a 44-day monitoring effort. It was estimated that a turtle had a 98% chance of being struck if attempted a crossing. As observed in the Cape May population of terrapins, a male-biased sex ratio was detected in three species of turtles, *Pseydemys floridana, Trachemys scripta*, and *Sternotherus odoratus*, due to the road mortality (Aresco, 2003). Steen and Gibbs (2004) also observed male-biased sex ratios in populations of painted turtles (*Chrysemys picta*) and snapping turtles (*Chelydra serpentina*) that inhabited high road density freshwater wetlands. The authors noted that because turtles' are a longed live species, populations can endure for years before the consequences from a threat is noticeable (Steen and Gibbs, 2004). However, insertion of fencing around the busy roadway intersecting Lake

Jackson led to a decrease in the amounts of dead turtles, from 9.7 individuals/km/day to 0.08 individuals/km/day (Aresco, 2003).

Road fencing has been one strategy for lessening road mortality associated with habitat fragmentation. Fencing along U.S. Highway 90 north of Mobile Bay in AL was constructed to significantly decrease the high levels of both female and hatchling mortality of the federally endangered Alabama red-bellied turtle, *Pseudemys alabamensis* (Nelson and Turner, 2004; David Nelson, University of South Alabama, pers. comm.). Aresco (2003) noted the success of the inserted fences to preventing turtle mortality on U.S. Highway 27 in FL, but the fencing was not nearly as successful in preventing road mortality in other herpetofauna species. Less than one-half of all individuals (other than turtles) were prevented from being struck by cars. Even though the fencing prevented most turtles from accessing the road, many were killed due to predation behind the fences. Only one culvert, allowing animals to access the other side of the highway, exists in the area, so chances for predation for turtles as well as other herpetofauna increases as they travel farther distances to find the culvert (Aresco, 2003).

The findings by Aresco (2003) suggested that some form of tunnel structure might be an effective strategy for facilitating movement of turtles and other herpetofauna under roadways separating habitats. In fact, Jackson and Griffin (2000) argued that fencing without any crossing structure will only increase the fragmentation of habitats. They listed the various factors that should be considered when constructing movement tunnels. They suggest that the placement of such structures could be the most important factor because, if they are too widely spaced, the tunnels may not properly serve their purpose. Other factors include size, light, moisture, temperature, noise, substrate, approaches, and species interactions, although different species will have different requirements for these factors. The authors suggested that more cost efficient strategy would include a mixture of widely spaced large structures and more frequently spaced small structures (Jackson and Griffin, 2000). However, Simberloff *et al.* (1992) contended that despite possible rationales for the construction of migration corridors (to maintain overall number of species, to decrease demographic stochasticity, to allow for gene flow to prevent inbreeding depression, and to provide for inherent need of certain species to move) data is needed to evaluate the cost effectiveness of this strategy. Until such data exist, other management options must not be blindly discarded (Simberloff *et al.*, 1992).

The development or preservation of wildlife migration corridors has also been suggested as a method of alleviating problems associated with habitat fragmentation and road mortality. Simberloff *et al.* (1992) discussed the various rationales for the construction of migration corridors: to maintain overall number of species, to decrease demographic stochasticity, to allow for gene flow to prevent inbreeding depression, and to provide for inherent need of certain species to move. However, they contend that data is needed to evaluate the cost effectiveness of migration corridors, and until such data exits, other management options must not be blindly discarded (Simberloff *et al.*, 1992).

Crab Trap-Induced Mortality

Another major threat to terrapin populations is crab trap-induced mortality (reviewed below). The crab fishery along the Atlantic and Gulf coasts of the U.S. primarily targets the blue crab (*Callinectes sapidus*) (Kennedy *et al*. 2007). The perishability and marketing of the blue crab initially limited the proliferation of this fishery, but by the late 1800’s and early 1900’s commercial crab fisheries were established in various locations along the Atlantic and Gulf coasts of the U.S (Stagg and Whilden, 1997; Guillory *et al*., 2001b; Kennedy *et al*., 2007). The modern crab trap was introduced in 1927 in Chesapeake Bay (Kennedy *et al*., 2007). The design was improved in 1938 and became the "gear of choice" for crab capture by the 1950's as the crab fisheries proliferated throughout the Atlantic and Gulf coast regions (Stagg and Whilden, 1997; Guillory *et al*., 2001b; Kennedy *et al*., 2007). Yearly landings have fluctuated, but the total amount of blue crab landed along the Atlantic and Gulf coast of the U.S. gradually increased from the 1950's through 1970's (Stagg and Whilden, 1997; Guillory *et al*., 2001b; Kennedy *et al*., 2007). In general, this was concomitant with an increase in the number of crab traps and crab fisherman (Hill *et al*., 1989; Guillory *et al*., 2001b). Since the 1970's, fishing effort has remained relatively high in the blue crab fisheries, although significant declines in landings have occurred in recent years in many areas due to factors such as overharvesting, pollution, and loss of habitat (Murphy *et al.*, 2007; Sutton and Wagner, 2007; CBSAC Report, 2010).

The expansion of blue crab fishery along the Atlantic and Gulf coasts of the U.S. from the middle of the 20th century to present day has significantly impacted terrapin populations. The crab trap is the preferred capture method in most areas and a numerous studies over the past several decades have documented their impact on terrapin populations. An initial study by Bishop (1983) examined the incidence of terrapin by-catch in crab traps in South Carolina estuaries. Crab traps were observed to capture high numbers of terrapins, and the catch was male biased (2.3:1). Based on his sampling results and the size of the crab fishery in South Carolina, he estimated that 2,853 terrapin would be captured daily in that fishery from May through April (Bishop, 1983).

However, he indicated that the mortality rate was lower than the capture rate depending on factors such as the frequency at which traps were checked and water temperature (ranging to as low as 10% of the capture rate) (Bishop, 1983).

Subsequent to the study by Bishop (1983), the widespread impact of crab traps on terrapin populations has been documented in numerous studies. Crab trap-induced mortality of terrapins has been documented in New Jersey (Wood, 1997), Maryland (Roosenburg, 1991; Roosenburg *et al.*, 1997; Roosenburg and Green, 2000), South Carolina (Dorcas *et al.*, 2007), Georgia (Grosse *et al.*, 2009), Florida (Siegel, 1993), and Alabama (Coleman *et al.*, unpublished data). The study in Maryland estimated that more than 2000 terrapins were caught annually by crab traps (Roosenburg *et al.*, 1997). Roosenburg *et al.* (1997) indicate that if all those terrapins died, this would represent 15 to 78% of local population being removed annually. A male-biased capture ratio (3:2), was observed in that study, and it probably contributed to the female biased sex ratio (2:1) that was observed. Terrapins exhibit sexually dimorphic body size with females growing significantly larger than males, and Roosenburg *et al.* (1997) indicate that once females reach a certain size, they are too large to enter the crab trap funnels. However, males never reach that size, so they are vulnerable to crab trap mortality throughout their lives (Roosenburg *et al.*, 1997). Both commercial and recreational crab trapping disproportionally remove juveniles and adult males creating populations with female biased sex ratios. However, the elimination of juvenile females might have the greatest impact because these individuals have not yet contributed reproductively (Roosenburg *et al.*, 1997). In fact, Tucker *et al.* (2001) calculated the mean life span for females in the sampled population in South Carolina and found that the average female did not survive to reproductive maturity.

A long-term study by Dorcas *et al.* (2007) indicated that crab trap-induced mortality was a major factor causing declines and demographic changes in the terrapin population inhabiting the salt marshes of Kiawah Island, GA. That study was based on more than twenty years of mark-recapture data and found a significant decrease in population size based on long-term sampling from multiple locations and found a demographic shift to a female-biased population comprised of larger and older individuals.

Roosenburg *et al*., (1997) suggested that the recreational and commercial crab fisheries could have varying impacts on terrapin populations. Recreational crab traps are often in more near-shore salt marsh habitats (Roosenburg *et al.*, 1997) where the terrapins (juveniles and adult males) that are more susceptible to crab trap mortality are more apt to be found (Roosenburg *et al.*, 1999). In South Carolina, recreational crab traps could possibly outnumber commercial crab traps, suggesting a relatively high impact of the recreational crab fishery on terrapin mortality (Hoyle and Gibbons, 2000).

Data from these studies also suggested that abandoned or lost crab traps (often referred to as derelict or “ghost” crab traps) may pose a greater risk of mortality for terrapins than those that are checked regularly. Bishop (1983) initially indicated that derelict crab traps could pose an increased risk due to numerous records suggesting that terrapins captured in crap traps tend to attract other terrapins. There have been several published anecdotes of derelict traps containing large number of carcasses: 29 terrapins in one trap in South Carolina (Bishop, 1983), 49 in one trap in Maryland (Roosenburg, 1991), and 94 in one trap in Georgia (Grosse *et al.*, 2009). This problem is amplified by the relatively large number of derelict crab traps that are added yearly to estuaries along the Atlantic and Gulf coasts of the U.S. As examples, it has been estimated that 25% to 30% of crab traps are lost or abandoned yearly resulting in estimates of 250,000 traps per year added to the Gulf of Mexico (Guillory and Perret, 1998; Guillory et al., 2001a), and 100,000 traps per year added to the Chesapeake Bay in the State of Virginia (Havens *et al.*, 2008).

Mitigating Crab Trap-Induced Mortality

Two mitigation measures have been developed to modify crab traps in order to decrease terrapin mortality; (1) modified crab traps (Roosenburg *et al.*, 1997) and (2) by-catch reduction devices or BRD's (Wood, 1997) Modified crab traps are taller than standard crab traps and allow terrapins to enter the upper tier of the trap, to surface and breath. In a study examining the efficacy of modified crab traps, no significant differences in crab captures were observed between modified and standard traps, and the modified crab traps did not display the terrapin mortality. However, due to their increased costs in building and awkwardness in handling, modified crab traps are better suited as recreational crab traps (Roosenburg *et al.,* 1997).

By-catch reduction devices are metal wire or plastic rectangles initially developed by Wood (1997), which fit into the crab trap funnel openings and are intended to prevent terrapin entry into the traps while still allowing crabs to enter. Wood (1997) found that traps outfitted with 5 x 10 cm BRDs performed the best in preventing terrapin capture and permitting crab capture. In fact, capture of marketable sized crabs was significantly enhanced in these traps. This scenario was also observed by Guillory and Prejean (1998) and Roosenburg and Green (2000). Further, no significant differences in crab captures between traps fitted with BRD's and non-fitted traps were detected in other studies (Cole and Helser, 2001; Butler and Heinrich, 2007; Morris *et al.*, 2010; Rook *et al.*, 2010). Guillory and Prejean (1998) suggested that the presence of BRD's may prohibit the egress of crabs by preventing manipulation of the funnel openings by the crabs.

Both Roosenburg and Green (2000) and Dorcas *et al.* (2007) contended that the size of BRD's should not be uniform across the terrapin's range but instead be adjusted based on regional size differences. While Wood (1997) found that 5 x 10 cm BRD performed best in New Jersey, the 4.5 x 12 cm performed best in the Chesapeake Bay (Roosenburg and Green, 2000) and Delaware Bay (Cole and Helser, 2001). The orientation of the BRD in funnel opening could also affect its performance. Crabs would presumably still enter traps fitted with BRD's in the vertical position, but terrapins could not (Hoyle and Gibbons, 2000; Dorcas *et al.*, 2007).

Programs for mitigating the impact of derelict crab trap-induced mortality have been initiated in a variety of states. Trap removal programs have been initiated in both Gulf and Atlantic coast states in the past (Guillory *et al.*, 2001a), and now many have yearly derelict trap removal programs (Perry *et al.*, 2008).

Finally, restricting the use of crab traps in specific terrapin habitats and/or during specific times of the year could represent a powerful mitigation tool for initiating and facilitating the recovery of populations which are in danger of extirpation (Butler, 2002).

Nest and Hatchling Predation

The predation of nests and hatchlings also represents a major threat to terrapin populations. While some predation can be considered a natural threat which has long been an integral part of terrapin ecology, anthropogenic factors such as coastal development can increase the natural predator load and the access of predators to terrapin nesting areas. Further, the decline of terrapin populations associated with other threats such as habitat loss and crab trap-induced mortality could leave a population in a precarious state in which nest depredation could have a much greater effect on the survival status of a population.

High levels of egg and hatchling predation have been observed on numerous terrapin nesting beaches. Butler *et al.*, (2004) reviewed a variety of terrapin studies that reported nest predation rates from 41 to 88 %. Feinburg and Burke (2003) observed over 3,000 depredated nests over two nesting periods and attributed over 98% of the depredations were due to raccoons. In Maryland, 94% of nests were taken from a sampled nesting location with raccoons as the main predator (Roosenburg and Place, 1995). Butler *et al.* (2004) detected over 80% of nests were depredated over two nesting seasons in a Florida terrapin population, and raccoons were the primary nest predator. A study of a terrapin population in Rode Island found that 87% of the monitored nests were depredated and raccoons were the primary predator (Goodwin, 1994). Raccoons appear to use cues such as scent and habitat disturbance when locating nests. In a study by Burke *et al.* (2005) at a nesting habitat at Jamaica Bay Wildlife Refuge, cues that seemed to be most important for raccoons were disturbance of nesting soils and as well as disturbance of the natural beach scents. Further, the majority of nests were preyed upon within 24 hours of oviposition (Burke *et al*., 2005).

In addition to raccoons, other reported predators include foxes (Burger, 1977; Roosenburg and Place, 1995), otters (Roosenburg and Place, 1995), gulls (Burger, 1977), crows, boat-tailed grackles, armadillos, and ghost crabs (Butler *et al.*, 2004). Beach vegetation can also contribute to egg and hatchling mortality. The roots of various beach grasses have also been observed to invade terrapin nests (Lazell and Auger, 1981; Butler *et al.*, 2004). Beach grasses have been planted for erosion control as alternative to other measures such as bulkheading (Roosenburg, 1991). The roots of beach grasses can impact eggs and hatchlings, including the direct penetration and mortality of eggs. Nests can also become root bound, which can result in embryonic death, prevent successful pipping and hatching, and/or prevent the emergence of hatchlings (Lazell and Auger, 1981; Butler *et al.*, 2004).

A variety of human activities can increase the population density of predators for terrapin eggs and hatchlings. For example, nuisance raccoons were illegally released on an island in the Jamaica Bay Wildlife Refuge whose terrapin nesting beach subsequently experienced high levels of nest predation (Feinburg and Burke, 2003). Human-induced reduction of natural predators of raccoons and foxes (e.g. wolves, coyotes, etc.) combined with the reduction of the fur market has also contributed to increases in terrapin nest depredation (Congdon, *et al.*, 1993; Roosenburg and Place, 1994). Additionally, raccoons can thrive in suburban environments, which are increasing in many coastal areas (Roosenburg and Place, 1994). Finally, the increase in roads and bridges associated with coastal development can increase predator access to salt marshes inhabited by terrapin.

Mitigating Nest and Hatchling Predation

Different methods of raccoon control have been attempted in order to decrease nest predation in terrapin populations as well as in other turtle species. Munscher (2007) found that removal of raccoons significantly decreased nest predation in a terrapin population in northeastern Florida. However, after the control efforts were ended, raccoons quickly repopulated the area (Munscher, 2007). A study by Christiansen and Galloway (1984) documented a significant decrease in both nest and hatchling predation of freshwater turtles at Big Sand Mound, Iowa, when raccoons were removed from the area. A study by Ratnaswamy *et al.* (1997) found that lethal removal of raccoons did not significantly affect sea turtle nest predation, even though the authors noted that this method may need to occur over a longer time period than it did in this study to have a significant effect. At Ten Thousand Islands, FL, raccoons were preying on 76-100% of sea turtle nests on the four sampled islands over four years. Sixteen raccoons were removed over two years on one of these islands (Panther Key), and nest depredation fell to 0% (Garmestani and Percival, 2005). However, it was suggested that the repopulation of raccoons in that area might occur more quickly if a given site were better connected to surrounding habitats, thus overcoming the initial benefit of predator removal (Garmestani and Percival, 2005).

Ratnaswamy *et al.* (1997) compared the efficacy of three different control methods for raccoons on a sea turtle nesting beach: lethal removal, non-lethal conditioned taste aversion, and screening of nests. The most labor intensive and costly method was nest screening, but it was also the most effective at preventing raccoon predationIn the case of a terrapin population. Nest screening has also been used on terrapin nesting beaches as a method for decreasing predation (Goodwin, 1994; Giambanco, 2002; Butler *et al*., 2006). However, the strategy requires close monitoring of nesting beaches in order to identify and screen nests soon after they are laid.

Ratnaswamy and Warren (1998) contended that it is pivotal to truly understand the complete role a predator has in an ecosystem before it is removed. Raccoons may provide benefits to the ecology of a specific habitat. For example, raccoons could serve as seed dispersers as well as important predators on invertebrate and other vertebrate species in coastal systems, and altering these relationships could have unforeseen detrimental effects (Ratnaswamy and Warren, 1998). Barton (2005) observed that ghost crab abundance increased when raccoons were removed, and this shift resulted in overall increases in sea turtle egg mortality due to ghost crab predation. However, if predator removal is undertaken, efforts have to be consistent at least in the short term. Engeman *et al.* (2006) detected an almost immediate increase to high levels of sea turtle nest predation in the middle of the 2004 nesting season after several years because predator removal efforts were abandoned due to budgetary constraints.

As an alternative to predator removal, efforts have been undertaken to exclude predators from nesting beaches. Bennett *et al.* (2009) constructed electric fences on terrapin nesting beaches to keep out raccoons and foxes. Although no significant differences in predator exclusion were observed between control and treatment plots, predation was less in the treatment plots, so absence of statistical significance was attributed to low sample size. The authors noted that this method could have complications based on the conductivity of the nesting beach substrate, amount of vegetation on nesting beach, and relative size of nesting beach (Bennett *et al.*, 2009).

Head-Starting as a Strategy for Decreasing Egg and Hatchling Mortality

Another potential alternative for decreasing predation in terrapin populations is through head-starting programs. This method involves the artificial incubation of eggs followed by the rearing of hatchlings to a certain size before releasing them into the environment. This circumvents the high mortality associated with early life history stages of terrapins (i.e. egg incubation, hatchling, and post hatchling stages). Head-start projects have been incorporated into ongoing terrapin conservation programs in New Jersey (Herlands *et al.*, 2004), Maryland (Smeenk, 2010) and Alabama (Coleman *et al.*, 2010). A terrapin head-starting project was initiated in southern New Jersey in 1989 in response to high levels of road mortality. Annually, 200-300 hatchlings are released back into the salt marshes, although 400-600 adult females are killed each year due to road mortality (Herlands *et al.*, 2004), thus the possible benefits of head-starting may not be able to compensate for the high level of road mortality. Smeenk (2010) studied the effectiveness of head-starting as a conservation strategy in a terrapin population in Maryland. That project released 664 terrapins that were 2 -5 years old. The results suggest that potential benefits from head-starting project were not sufficient to compensate for the high mortality from crab traps in that area (Smeenk, 2010).

A factor that should be considered when head-starting terrapins is the threat of introducing pathogens into the wild populations (Cunningham, 1996). Werner *et al.* (2002) screened both captive-reared diamondback hatchlings as well as wild hatchlings for pathogens and found both groups relatively healthy. It was concluded that introducing head-started individuals did not pose any health dangers to the wild terrapins (Werner *et al.*, 2002).

The effectiveness of head-starting has also received attention in the Kemp's ridley sea turtle (Dodd and Siegel, 1991; Frazer, 1992; Heppell *et al.*, 1996). Heppell *et al.* (1996) created a model that examined the effects of head-starting on a population of

Kemp's ridley sea turtle. That model indicted that the beneficial effects of head-starting would be negated if there were high levels of threat-induced mortality of the sub-adult and adult portions of the population. This conclusion aligned with those of Frazer (1992). He described head-starting as a "half-way technology", because it did nothing to address the threats faced by juvenile and adult portions of the population. However, data from that project indicate that head start Kemp's ridleys can survive and reproduce in the wild (Shaver and Caillouet, Jr., 1998; Shaver and Wibbels, 2007). Head-started female western pond turtles (*Actinemys marmorata*) have also been documented nesting (Vander Haegen *et al.*, 2009). To properly assess the effectiveness of any head-start program, long-term monitoring on both the nesting beaches and foraging habitats will be required (Shaver and Wibbels, 2007), an idea also argued by Dodd and Siegel (1991).

Protection and Legal Take of Terrapins

Data indicate that terrapin populations throughout their range are being impacted by the various threats discussed above (Butler et al., 2006). However, the survival status varies between populations that have been studied, and the survival status of many populations has not been well documented (Butler et al., 2006). Because of the variability in the survival status and the paucity of data from most locations, the laws protecting terrapins and regulating their legal take vary greatly. The U.S. Fish and Wildlife Service (USFWS) has listed the diamondback terrapin as a "status review species" for several decades (Hart and Lee, 2006). However, range-wide population surveys data are lacking, so the USFWS has not been able to determine if federal protection is necessary. Therefore, the current protection and regulation of legal harvest of terrapins is under state control. The state laws regulating the protection and legal take of terrapins vary widely (Reviewed in Figure 1). Of the 16 states along the Atlantic and Gulf coasts of the U.S. with terrapin populations, seven have regulations that prevent the legal take of terrapins (AL, CT, GA, MA, RI, TX and VA). Six states (DE, FL, MD, MS, NC and SC) have laws that limit daily or yearly take of terrapin (Figure 1), and three states (LA, NJ and NY have open harvest during a specific season (Figure 1). The laws regulating the protection and harvesting of terrapin can obviously impact the survival status and recovery of a given population. However, to fully understand that impact, the overall magnitude of the legal take must be taken into account relative to the size and stability of the population. It seems intuitive that decreasing legal harvest would enhance the stability and recovery in a population, but this needs to be evaluated on a population by population basis.

Summary

Diamondback terrapins endured historical exploitation and their current survival is being impacted by a spectrum of new anthropogenic threats. The purpose of the current review was to highlight and evaluate the primary threats to terrapin populations and potential strategies for their recovery. Surveys of terrapin researchers from along the Atlantic and Gulf coasts indicated that crab trap-induced mortality, habitat loss, nesting beach alteration, predation, and road mortality represented major threat to populations in most areas (Butler *et al.*, 2006). Terrapins face additional threats such as pollution, legal harvest, and the potential for the renewal of a commercial market (Gibbons *et al.*, 2001). However, most of these are clearly defined threats, and a variety of strategies have been developed and evaluated for mitigating their negative impact on terrapin populations.

effective management strategies should target the threats that have the greatest impact on a particular population. Preservation of appropriate habitat is a prerequisite to the recovery and maintenance of a stable terrapin population. The use of BRD's on crab traps is an obvious and well-documented strategy for decreasing mortality of juveniles and adults. Decreasing the high road mortality experienced by some populations would directly enhance the survival of the reproductively active portion of the population. Attempting to decrease the predation of eggs and hatchlings could also enhance the recovery of a population if mortality rates in the juvenile and adult portions of the population are not excessively high. Thus, a variety of conservation measures have been shown to be effective means of enhancing the recovery of terrapin populations. Such methods can be selectively incorporated into a conservation strategy that is targeted for enhancing the recovery of a specific population. Ideally, it would be advantageous to include an ecological model that utilizes population survey data to estimate survival rates at various life history stages as well as mortality rates associated with major threats (Heppell *et al.*, 1996). This would provide a prioritized list of threats which need to be addressed in a recovery program.   It is imperative that the overall strategy ensures that survival rates at various life stages are sufficient to ensure a stable (and preferably increasing) reproductively active portion of the population. The current review provides an overview of the viable options which can be implemented into effective management strategies for enhancing the recovery of a terrapin population.

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Table 1. State regulations regarding terrapin status, bycatch and take.

State	Status	BRD Regulations	Legal Take Regulations
Alabama	"Species of Greatest Conservation Need"	None	Unlawful to take, possess or sell
Connecticut	State regulated species	None	No open season for taking
Delaware	"Species of Concern"	1.75"x4.75"—recreational traps	Unlawful to take or destroy eggs Open season between September 1 and November 15, with a limit of 4/person/day Buying/selling of lawfully taken terrapins is permitted
Florida	None	None	No more than 1 terrapin/day (no eggs) No person may possess more than 2 terrapins
Georgia	"Unusual Species"	None	Unlawful to harass, capture, kill, sell or purchase
Louisiana	"Species of Concern"	None	6" plastron minimum size limitation Unlawful to take eggs Unlawful to take through use of a trap May not be captured or shipped out of state between April 15 and June 15
Maryland	None	1.75"x4.75" on recreational traps	Unlawful to take or possess for commercial purposes May possess 3 or less for non-commercial purposes
Massachusetts	"Threatened"	None	Unlawful to take, disturb, or harass terrapins or eggs
Mississippi	"Species in need of management"	None	Unlawful to possess or sell more than a total of 20 non-game reptiles, and no more than 4 of a single species, for personal use
New Jersey	"Species of Special Concern"	Funnel openings have to reduced to 2"x 6" in small water bodies (less than 150 feet wide) and man-made lagoons	Unlawful to take or sell except between November 1 and March 31, and plastron may not be less than 5" long (no limit during open season)
New York	None	None presently, under review	Open season from August 1 to April 30 (upper shell length must be between 4 and 7 straight-line inches); license required Lawfully taken terrapins may be purchased and sold
North Carolina	"Special Concern Species"	None presently, under review	Commercial take is unlawful (i.e. 5 or more/year)
Rhode Island	"Endangered"	None	Unlawful to possess, buy or sell terrapins or eggs
South Carolina	None	None	Unlawful to take or possess for commercial purposes May possess up to 2 for non-commercial purpose
Texas	None	None	Unlawful to take or possess terrapins or eggs
Virginia	None	None	Unlawful to take for private use

CHAPTER 2

POPULATIN ECOLOGY OF THE DIAMONDBACK TERRAPIN (*MALACLEMYS TERRAPIN PILEATA*) IN ALABAMA

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In preparation for Journal of Herpetology

Format adapted for dissertation

Abstract

Long-term studies can provide critical demographic and ecological data on long-lived species such as turtles. A comprehensive survey of the diamondback terrapin population in Alabama was initiated in 2004 to investigate several population parameters. Various field methods sampled potential salt marsh habitats along the Gulf Coast of Alabama. Although diamondback terrapins were historically abundant in Alabama, terrapins are currently present in several isolated remnant aggregations with the largest aggregation existing in Cedar Point Marsh, which was once the site of a large terrapin farm in the late 1800's. This aggregation was estimated to be fewer than 400 individuals, and a female-biased sex ratio was detected. Crab trap mortality and nest predation represent the biggest threats impacting the Alabama population. This long-term data set will be crucial in developing optimal management strategies aimed at initiating the recovery of diamondback terrapins in Alabama.

Introduction

Long-term studies with associated demographic and life history data are required to gather a complete picture of an organism's ecological and conservation status (Tinkle, 1979). These studies are critical for long-lived species, such as turtles; however, they are time and effort intensive to gather the necessary data (Gibbons, 1987). Madsen and Shine (2001) contrasted the findings of their long-term data set on a long-lived species of snake, the Arafura filesnake (*Acrochordus arafurae*), to short-term studies that made conclusions about growth rates, age at maturity, and mean reproductive rates of this snake species. The authors argued that environmental stochasticity could have a greater influence on short-term studies examining life history traits of long-lived organisms, leading to inaccurate conclusions (Madsen and Shine, 2001). This comparison between conclusions of long and short-term studies indicates the need for long-term data sets to better understand life history characteristics and current status of long-lived species.

Several hypotheses have been proposed to explain the longevity of turtles: their low metabolism, a large energetic investment in developing the adult's protective shell, and an apparent lack of anatomical and physiological senility (Gibbons, 1987). Certain life history characteristics accompany turtle longevity, including delayed maturity, greater adult survival, high lifespan fecundity, and greater generational times.

A long-term study of snapping turtles (*Chelydra serpentina*) performed at the University of Michigan's E.S. George Reserve in Southeastern Michigan has generated reliable data on life history traits of this species. The age at maturity of females was 12 years, and mature females produced no more than one clutch per nesting season (Congdon *et al.*, 1987). The annual survivorship of juveniles (ages 2-12) was calculated to be between 0.65-0.82, and adult females had a survivorship of 0.88. The authors utilized these data to support actions that would improve the conservation of this exploited population (Congdon *et al.*, 1994).

Similar conclusions were made based on a long-term data set concerning Blanding's turtles (*Emydoidea blandigii*) at the same field site (Congdon *et al.*, 1993). The average age at maturity for females was 17.7 years of age and mean clutch size was 10.2 eggs. As seen in snapping turtle females, no females nested more than once a field season and some did not nest every year. However, older females did reproduce more often than younger mature females (Congdon and van Loben Sels, 1993; Congdon *et al.*, 2001) and produced larger clutch sizes (Congdon *et al.*, 2001), although egg and hatchling sizes produced by older females did not increase (Congdon *et al.*, 2001). Also similarly observed in snapping turtles, adult Blanding's turtles displayed high adult survivorship (Congdon *et al.*, 1993).

Frazer *et al.* (1991) calculated much lower survivorship for painted turtles (*Chrysemys picta*) in a population of southwestern Michigan. Annual survivorship for adult females ranged from .29-.50, whereas survivorship for adult males ranged from .64-.83. This discrepancy most likely led to the male-biased sex ratio that was present in the population. The oldest female that was observed was 34 years old even though very few turtles survived a decade (Frazer *et al.*, 1991). From a population of painted turtles inhabiting E.S. George Reserve, Congdon *et al.* (2003) found that these turtles displayed indeterminate growth, but growth rates of older turtles were slower than younger turtles. Older females did not have larger clutches or greater among year reproductive frequency, though egg size, hatchling size, and within year reproductive frequency did increase with age (Congdon *et al.*, 2003).

In addition to better understanding basic life history parameters, long-term studies can elucidate patterns of both spatial and temporal change due to environmental variation that short-term studies might miss or misinterpret. Utilizing almost a decade's worth of growth data for male painted turtles, Frazer *et al.* (1992) noted an increase in growth rates of juveniles and age at maturity for male turtles sampled in the late 1980's compared to earlier in the 1980's. The authors attributed these changes to the longer growing seasons that occurred in the latter half of the decade (Frazer *et al.*, 1992).

Spatial variation was detected when a population of common mud turtles (*Kinosternon subrubrum*) in South Carolina was compared to a conspecific population in

Virginia and a congener population of yellow mud turtles (*Kinosternon flavscens*) in Nebraska (Frazer *et al.*, 1991). Females in the mud turtle population in South Carolina matured at smaller sizes and younger ages than the yellow mud turtle females. The yellow mud turtles had a slightly lower average of clutches per year but a higher percentage of reproductively active females within a season. The yellow mud turtle population also displayed a longer generation time (Frazer *et al.*, 1991).

Finally, long-term studies have provided insight to ecological attributes and responses to environmental disturbances of turtle populations. Burke *et al.* (1995) examined 26 years of mark- recapture data from the Ellenton Bay population of red-eared sliders (*Trachemys scripta*) in South Carolina. The authors recognized that this population was a part of a larger metapopulation consisting of nine populations inhabiting various aquatic habitats such as permanent streams, seasonal ponds, and semi-permanent Carolina Bays (Burke *et al.*, 1995). Gibbons *et al.* (1983) was able to differentiate between natural variation observed in several populations of freshwater turtles inhabiting Ellenton Bay and their specific responses to the environmental disturbance in the form of a drought that occurred from 1980-1981 by using 15 years of mark-recapture data.

A long-term study was initiated in 2004 to assess the current population status of the Mississippi diamondback terrapin (*Malaclemys terrapin pileata*) along the Gulf Coast of Alabama. The diamondback terrapin is an obligate estuarine turtle inhabiting salt marshes, bays, and estuaries from Cape Cod, MA, along the Atlantic and Gulf Coasts to Corpus Christi, TX. Seven subspecies are currently recognized, and the range for the Mississippi diamondback terrapins extends from the Panhandle region of Florida to the Louisiana/Texas border (Brennesel, 2006). Terrapins potentially play a significant role in maintaining salt marsh ecosystem integrity through their predation on salt marsh periwinkle snails (*Littoraria irrorata*) (Silliman and Bertness, 2002); however, many populations are experiencing drastic population declines (Butler *et al.*, 2006), and the Alabama population is no exception. Jackson and Jackson (1970) noted the presence of terrapins on Dauphin Island, AL, but more recent cursory surveys concluded that terrapin aggregations in Alabama appear to be greatly reduced and isolated (Marion 1986; Nelson and Marion, 2004). The study presented here was the first comprehensive study that utilized a variety of sampling methods in marsh channels and on potential nesting beaches to determine present day viability of diamondback terrapins in Alabama. These data will be critical in developing optimal management strategies for this unique and vital turtle.

Methods

To identify habitats along the Gulf Coast of Alabama that support diamondback terrapins, numerous salt marshes in the Mississippi Sound were sampled (Figure 1). In the marsh channels, head counts (Harden *et al.*, 2009) and trapping with modified crab traps (Roosenburg *et al.*, 1997) were conducted. For head counts, the number of individuals at the water's surface was counted as the boat traveled down the marsh channel. These surveys usually lasted from 30 minutes to an hour and would be done on the way into the marsh. They provided very rough estimates of terrapin abundance in the marshes. Modified crab traps were set at various places within the sampled marsh and were usually baited with menhaden purchased from a local bait shop. They were checked every two to three days, but crab trap mortality was not a threat because of these traps. Because the top of the traps were never below the water, terrapins were not in danger of drowning. Each trap was attached to a PVC pipe inserted into the marsh channel bed to prevent the trap from falling over. In 2006, eight traps were placed in Cedar Point Marsh, while four were placed in Jemison Marsh. In 2007, the four traps were moved from Jemison Marsh to Airport Marsh along with four more traps in Little Dauphin Island. In addition to these, sixteen traps were placed in Cedar Point Marsh. Traps fitted with 2”x 6” by-catch reduction devices (BRD’s) and not fitted with BRD’s were set side by side at eight locations for a comparison of both terrapin and crab capture (will be further discussed in Chapter 4).

Survey methods were also conducted on the potential nesting beaches surrounding the sampled salt marshes. Depredated nest surveys indicated relative importance of each potential nesting beach. Due to the shell hash substrate of the beaches, only depredated nests could be counted as opposed to intact nest counts. Trapping of nesting females also was conducted on nesting beaches using drift fences with pitfall traps (Gibbons, 1970). Drift fences were standard silt fences purchased from local hardware stores. They were inserted into the beach substrate (20-30 centimeters deep) and roughly ran parallel to the marsh. Four pitfall traps were associated with each drift fence, one at each end, one in the middle facing the marsh, and one in the middle facing the open water. As females approached the fence, they would be forced to walk along it until reaching one of the traps. As they walked on top of the trap, the lid would flip, allowing the turtle to fall into the bucket. Nest surveys were done at various times during the summer, but the traps were checked every day. Effort was calculated for the head surveys and the trapping methods. For modified crab traps and pitfall traps, catch per unit effort was calculated as the number of terrapins per trap per day. For trawling, catch per unit effort was calculated as the number of terrapins caught per 45 minutes.

After initial surveys, survey and trapping efforts were concentrated on Cedar Point Marsh. In 2008 and 2009, the same side by side comparison of modified crab traps with and without BRD's was completed. From 2008-2010, three to four drift fences with accompanying pitfall traps were inserted in the surrounding nesting beach. In 2009 and 2010, otter trawling in the marsh channel was utilized as another trapping method. Trawling was done at various times of the spring and summer starting in March. Catch per unit effort was also calculated for this sampling method.

A suite of morphological measurements were obtained from each captured individual (Table 1). Age for each terrapin was estimated using growth annuli on carapacial and plastral scutes. In addition, a blood sample was taken from the caudal vein. The blood was separated so the red blood cells could be used for a genetic study and serum for a reproductive physiology study. In 2006 and beginning of 2007, the marginal carapace scutes were notched (Cagle, 1939) for identification purposes. Starting in 2007, a carapace tag that consisted of a yellow plastic tag attached to a black zip tie was used. Each of these tags contained a unique number and was attached to a marginal scute above the terrapin's right hind limb. Also, each year, PIT (Passive Integrated Transponder) tags were inserted beneath the skin of the terrapin's left hind limb.

Table 1. Morphological measurements taken from each captured terrapin.

curved carapace length (CCL)	straight-lined carapace length (SCL)
curved carapace width (CCW)	straight-lined carapace width (SCW)
plastron length (PL)	shell depth (SD)
head width (HW)	Weight
plastron notch to vent length (NTVL)	vent to tail tip length (VTTL)

Population estimates were calculated using the modified Schnabel method (Schnabel, 1938) to obtain estimates of total population (juveniles, adult males and females) and the nesting female population inhabiting Cedar Point Marsh. The following equation was used:

$$\hat{N} = \sum(C_t * M_t)/(\sum R_t)+1$$

C_t is the number of captures in the n^{th} sample
M_t is the total number of marked individuals in the population before the n^{th} sample
R_t is the number of recaptures in the n^{th} sample

The 95% upper and lower limits for each estimated was calculated using the number of total recaptures following the methodology of Chapman and Overton (1966).

Results

In 2004-2005, terrapin heads were observed in several Alabama marshes in the Mississippi Sound, although heads per unit effort were all low (Table 2). Heron Bay, Cedar Point Marsh, Little Dauphin Island, and Mon Louis/Cat Island were the only marshes for which the effort for 30 minutes yielded multiple heads. Depredated nests were also observed on numerous potential nesting beaches (Figure 2). The majority of nests were observed on the beach surrounding Cedar Point Marsh in both years. The difference between the two years at Cedar Point Marsh was not due to fewer surveys in 2005.

Terrapins were captured in several marshes using both modified crab traps and pitfall traps in 2004 and 2005. No terrapins were caught with modified crab traps in 2004, but in 2005 eight terrapins were captured in Jemison Marsh along with three at Mon Louis and two at Baron Point (Figure 3). With pitfall traps, nine females were captured on

Barton Island, with one at Bayou La Fourche in 2004 (Figure 4). In 2005, five adult females were caught at Cedar Point Marsh and one at Mon Louis Island.

Table 2. Heads per unit effort by location in 2004-2005.

Location	# of surveys	# of heads/min	# of heads/30 min
Cedar Point Marsh	5	0.12	3.6
Heron Bay	1	0.27	8
Airport Marsh	3	0.03	0.8
Little Dauphin Island	2	0.12	3.6
Barton Island	1	0	0
Mon Louis/Cat Island	2	0.7	2.2
Fowl River	1	0	0
Point Aux Pines	1	0	0

Based on results of surveys and trapping results from 2004-2005, trapping and monitoring efforts in 2006-2010 were mostly focused on terrapin habitats situated near Dauphin Island, AL: Jemison Marsh, Airport Marsh, Little Dauphin Island, and Cedar Point Marsh. Modified crab traps were only used in Jemison Marsh, Airport Marsh, and Little Dauphin Island, whereas both crab traps and pitfall traps were placed in Cedar Point Marsh.

Increased head surveying effort in marshes did not result in greatly increased heads per unit effort (Tables 3 & 4). Other than 2006, heads per unit effort for Cedar Point Marsh for each year was greater than the combined survey efforts for the other sampled marshes. The same pattern was observed in depredated nest survey results (Figures 5 & 6). Of the other marshes, most depredated nests were observed at Airport Marsh from 2006-2010. More than 100 nests were observed on the nesting beach surrounding Cedar Point Marsh each year except for 2007. A prolonged drought could have affected the nesting activity of females that year.

Table 3. Heads per unit effort for various salt marshes from 2006-2010.

Location	# of surveys	# of heads/min	# of heads/30 min
Airport Marsh	20	0.030	0.895
Little Dauphin Island	15	0.0132	0.396
Jemison Marsh	10	0.064	1.915
Mon Louis/Cat Island	3	0.0286	0.857
Barton Island	1	0	0

Table 4. Heads per unit effort for Cedar Point Marsh from 2006-2010.

Year	# of surveys	# of heads/min	# of heads/30 min
2006	16	0.053	1.58
2007	23	0.084	2.53
2008	42	0.093	2.79
2009	37	0.13	3.99
2010	28	0.14	4.07

Sixty-three terrapin were captured with modified crab traps from 2006-2009, with the majority of those caught in Cedar Point Marsh (Figure 7). There were numerous instances where multiple terrapins were captured in the same trap. In fact, seven terrapins were found in one trap in 2009. Sixty-two adult females were caught with pitfall traps on the nesting beach surrounding Cedar Point Marsh from 2006-2010 (Figure 8). The highest total was 18 captured in 2009 with the lowest of 8 in 2007. Trawling was initiated in 2009, and a total of 28 terrapins were caught in 2009-2010 (Figure 9). Catches per unit effort (C.P.U.E.s) were calculated for each trapping method per year and averaged (Table 5). The low captures with high amounts of trapping efforts resulted in very low calculated C.P.U.E.s for each trapping method. For crab traps, 0.016 terrapins/trap/day were captured, while .012 nesting females/trap/day were captured with the pitfall traps. For every 45 minutes of trawling, 0.39 terrapins were caught.

Table 5. Catch per unit effort for each trapping method by year.

Year	Modified crab traps (# terrapins/trap/day)	Pitfall traps (# terrapins/trap/day)	Trawling (# terrapins/45 min.)
2006	0.027	0.013	N/A
2007	0	0.012	N/A
2008	0.008	0.015	N/A
2009	0.027	0.013	0.056
2010	N/A	0.008	0.022
Average	0.016	0.012	0.039

Adult females comprised the majority of the catch each year in Cedar Point Marsh (Table 6). Seventy-four adult females were captured with all three capture methods, with only 19 adult males and 4 juveniles. This represented a 4:1 female:male sex ratio for the capture rate. However, based on the captures that both sexes could be caught (modified crab traps and trawling), 35 females and 19 males were captured, shifting the sex ratio to approximately 2:1. When divided into age classes, most captures (43) were in adult females 5-8 years of age (Figure 10). The next highest classes were the adult females 9+ years of age (34) and adult males 3-6 years of age (18). Only three adult males 7+ years of age were captured over the five field seasons.

Table 6. Number of juveniles, females, and males captured in Cedar Point Marsh by year.

Sex	2006	2007	2008	2009	2010
Juvenile	3	0	0	0	1
Female	20	6	13	26	9
Male	1	0	0	17	1

The average morphological measurements of each sex indicated the sexual dimorphism that has been observed in this species (Table 7). Adult males had noticeably swollen tail bases and longer notch to vent lengths. Adult females had significantly larger heads than males. The smallest nesting female captured on the nesting beach had a PL measurement of 14.6 cm.

Table 7. Average morphological measurements by sex.

Sex	**CCL**	**SCL**	**CCW**	**SCW**	**Height**
Female	17.75921	16.51171	15.08684	12.40645	6.839412
S.D.	2.785356	2.576626	2.365363	1.864884	0.97267
Range	12.2-22.7	11.43-21.7	11-19.5	9.1-15.8	4.87-8.4
Male	13.21429	12.30571	10.81905	9.271905	49.02105
S.D.	0.558825	0.477939	0.393216	0.270215	3.251936
Range	12.4-14.6	11.51-13.25	10.3-11.5	8.73-9.83	44.1-59.1
Sex	**PL**	**HW**	**NTVL**	**VTTL**	**Weight**
Female	14.96158	37.30147	22.58487	30.34342	781.8247
S.D.	2.409321	7.395702	5.042538	5.512383	372.485
Range	10.04-19.1	23.3-50.4	10.69-39.8	21.2-42.9	272.99-1537.14
Male	104.8024	23.545	33.48524	28.10476	284.1757
S.D.	3.611111	1.299585	4.044195	2.769201	26.06049
Range	98.2-110.65	20.7-25.1	25.2-40.9	25.2-34.6	247.85-315.3

From 2006-2010, 97 individuals in Cedar Point Marsh were captured with all three methods, with 38 of those terrapins being recaptured at some point. Using the Schnabel method with Chapman's modification, the total population estimate was approximately 336 (Figure 11). This estimate did not include hatchlings, because trapping methods did not target this age group. The total nesting female population for Cedar Point Marsh was also estimated. Approximately, 83 nesting females utilize the nesting beach, which correlated with the number of depredated nest surveys given that terrapins can lay 2-3 clutches per nesting season (Seigel, 1984).

Discussion

Diamondback terrapins were observed existing in a number of isolated remnant aggregations in the Mississippi Sound along the Gulf Coast of Alabama. Cedar Point Marsh supported the largest of these aggregations. The heads per unit effort was relatively high in Cedar Point Marsh in 2004-2005, along with Heron Bayou, Mon Louis Island, and Little Dauphin Island. The greatest number of depredated nests were observed on the nesting beach surrounding Cedar Point Marsh in both years. When the surveying efforts in 2006-2010 were focused on marshes located near Dauphin Island, the largest number of terrapins was consistently detected in Cedar Point Marsh. Other than Cedar Point Marsh, more than one head was observed every thirty minutes in only one location, Jemison Marsh, when all of the surveys from 2006-2010 were combined. However, the effort for 30 minutes was above one for each year in Cedar Point Marsh, with the last two years reaching approximately 4 heads per 30 minutes. Other than 2007, over 100 depredated nests were found each year from 2006-2010 on the nesting beach surrounding Cedar Point Marsh. This greatly outweighs the nesting beach with next highest total, Airport Marsh with 43 depredated nests. Even though the total for Airport Marsh was over 5 seasons, the majority of those nests were found in one survey in 2010, which indicates that Airport Marsh still represents an important nesting beach and that nest surveys could overlook potential nesting beaches if not performed intensively or at regular intervals.

A number of individuals from several locations were captured with modified crab traps and pitfall traps in 2004 and 2005, but the majority of the captures occurred in Cedar Point Marsh from 2006-2010. A total of 97 individuals were captured in Cedar Point Marsh during this time using all three trapping methods, and 38 of those individuals were captured more than once. Each trapping method displayed a low catch per unit effort. The average C.P.U.E. for modified crab traps in this study was calculated to be 0.016 terrapins/trap/day compared to Roosenburg et al. (1997) calculated C.P.U.E. of 0.17 terrapins/trap/day from a population in Maryland, which was an order of magnitude higher than the Alabama C.P.U.E. In capture/trap day, the modified crab trap average

C.P.U.E. was .11 terrapins/trap day, which was higher than the average C.P.U.E. (.02 terrapins/trap day) for modified crab traps over two seasons in a population in Florida (Butler, 2002). However, based on total capture data for this Florida population, the population size was estimated to be over 3,000 individuals (Butler, 2002), indicating that reliance on limited data sets can result in misguided conclusions.

A female-biased sex ratio (4:1) was detected based on the total capture data. Overall, both sexes should have been equally targeted. Pitfall traps only captured adult females coming up on the beach to nest, but modified crab traps should have selectively targeted adult males along with juveniles. However, if the pitfall trap capture data were removed, a female-biased sex ratio (2:1) was still detected. Diamondback terrapins display sexual dimorphism in which the adult females grow to a much larger size than adult males (Lovich and Gibbons, 1990). Because of size differences, adult females should grow to a size where entry into crab traps are inhibited (Roosenburg *et al.*, 1997). It should be noted that some adult females were captured in modified crab traps, so the traps used in this study may not have been as selective as in other studies. Trawling should non-randomly fish anything in the channel large enough not to pass through the mesh, so both sexes should have been targeted. Interestingly, most captured adult males were done so with trawling, 20 males compared to 7 with modified crab traps.

When the captures were divided into age classes, the number of older adult males and younger adult males caught was higher than that of adult females. This could be an indication that crab trap mortality is a major threat affecting the Cedar Point Marsh aggregation. If the apparent lack of older adult males is real, then sex-biased mortality exists for these terrapins and should be addressed. The Cedar Point Marsh aggregation has experienced a size reduction over a long period of time. According to a New York Times article, the world’s largest terrapin farm existed in this location in the late 1800’s. It reportedly housed over 20,000 terrapins, which was a combination of resident terrapins along with terrapins that were brought in by local residents, and each year, thousands of turtles were shipped to the markets of the Northeastern U.S. for human consumption. This occurred at the height of the culinary demand for the gourmet terrapin soup, which raised the market prices to historical levels. This led to declines in numerous populations, which were exacerbated by the creation of new anthropogenic threats that have intensified in recent decades, such as crab trapping. Based on population estimates using mark-recapture data, the Cedar Point Marsh aggregation was estimated to be 336 individuals (not including 0-2 year olds), down from 20,000 individuals, a 98% decrease.

Other long-term studies have been completed with diamondback terrapin populations throughout its range. Analyzing a 15-year dataset of Carolina diamondback terrapins (*Malaclemys terrapin centrata*) inhabiting marshes around Kiawah Island, SC, Gibbons *et al.* (2001) detected that males matured at a PL of 90 mm, which occurred between their third and fourth year, and females matured at a PL of 138 mm, which occurred between their sixth and seventh year. The population displayed a male-biased sex ratio that had been attributed to males maturing at an earlier age (Lovich and Gibbons, 1990). Terrapins also displayed high site fidelity because only a few individuals were recaptured in a marsh different from their original capture site (Gibbons *et al.*, 2005).

Roosenburg (1991) began a mark-recapture study with a population of Northern diamondback terrapins (*Malaclemys terrapin terrapin*) inhabiting the Patuxent River, MD, in 1987. More than 2,800 individuals were marked, and using the Jolly-Seber model, the population was estimated to be 2,293. Both males and females matured at an older age than the SC population (4-7 years and 8-13 years, respectively), and females were larger (175 mm) at first reproduction (Roosenburg, 1991). A very low survivorship (1-3%) of eggs to hatchlings was observed, and it was suggested that an adult female must be at the highest reproductive output for three years to replace her (Roosenburg, 1991).

Seigel (1984) and Butler (2002) both reported population parameters of diamondback terrapin populations in Florida. In his two populations of Florida East Coast terrapins (*Malaclemys terrapin tequesta*), Seigel (1984) observed heavily female-biased sex ratio (5:1), whereas Butler (2002) detected a slight male-biased sex ratio (1.39:1) in the studied population of Carolina terrapins. A female-biased sex ratio was also observed in the previously mentioned Maryland population (Roosenburg *et al.*, 1997). The smallest *M. t. tequesta* mature female had a PL of 13.5, and females reached this length between their third and fourth year (Seigel, 1984). The smallest gravid female in the Carolina terrapin population had a PL of 144 mm, and females spent most of the year in the same marsh. High nest depredation was also observed in this population, with over 700 depredated nests over the two years surveyed, and of the nests with known deposition dates, almost 80% were depredated within 48 hours of oviposition (Butler, 2002).

A stage-specific population model was developed for a population of Northern diamondback terrapin in Rhode Island utilizing an eleven-year dataset. Females averaged 1.2 clutches per season although some nested three times (Mitro, 2003), similar to the Maryland population (Roosenburg, 1991). There was less than a 10% chance of producing a successful clutch, and the survivorship of nesting females declined slightly over the study period. The PL's of females increased over the years, suggesting females were larger and older (Mitro, 2003), a trend also observed by Roosenburg (1991). It was concluded that survival of females was more important to population stability than recruitment or reproductive success (Mitro, 2003).

Before the current study, Mann (1995) completed the only terrapin survey in recent years in the Northern Gulf of Mexico populations. The study focused on a Mississippi diamondback terrapin (*Malaclemys terrapin pileata*) population in Mississippi. Assuming females laid three clutches per season, it was estimated that a total of 535 females nested on 37 kilometers of surveyed beaches based on the number of observed depredated nests. It was then hypothesized that the proportion of adult females captured with crab traps to the total number of captured terrapins should be similar to the proportion of nesting females to total population size. Based on this methodology, the total population size for Mississippi was calculated to be 3,375 individuals (Mann, 1995). Also based on depredated nest surveys, the average clutch size for terrapins in Mississippi was 8.25 eggs, and it ranged from 4-15 eggs (Mann, 1995).

The Alabama population of diamondback terrapins appears to have experienced a severe population decline. It was estimated that fewer than 400 individuals (excluding 0-2 age class) inhabit Cedar Point Marsh, and this marsh supports the largest aggregation in the state. Although basic assumptions of the Schnabel method, which was used to calculate the estimate, could have been violated (e.g. the population should be closed), this method has been used in other terrapin studies (Seigel, 1984; Butler, 2002; Morris, 2010; Rook, 2010). The only populations comparable in size to Alabama's were those studied by Seigel (1984). The Banana River population contained an estimated 405 individuals, and the Indian River population contained 213 (Seigel, 1984). But populations have since been depleted (Seigel, 1993).

Based on anecdotal evidence from local fishermen and past evaluations, crab trap mortality represents the major cause of the terrapin decline in Alabama (Marion, 1986; Nelson and Marion, 2004). Because of the initiation of the long term study in Maryland, Roosenburg *et al.* (1997) documented the drastic consequences of crab trap mortality. Based on his previous population estimate (Roosenburg, 1991), crab trap mortality could result in a 19-78% annual loss, which would be unsustainable (Roosenburg *et al.*, 1997). Differential survivorship was observed, with higher mortality in adult males, and this contributed to the overall female-biased sex ratio in the population. However, crab trap mortality also posed a threat to juvenile females in the Maryland population. Because they do not have the opportunity to fulfill their reproductive potential, the loss of these females could prove more detrimental than the loss of adult males (Roosenburg *et al.*, 1997). Crab traps that are no longer fished (derelict or ghost traps) can remove large numbers of individuals from a population. Roosenburg (1991) found one ghost trap with 49 terrapins, and one trap in Georgia contained 94 carcasses, which was more than the estimated population for the creek in which that particular trap was found (Grosse *et al.*, 2009). Mann (1995) noted finding seven dead terrapin in three ghost traps in Mississippi. With the number of ghost traps added annually to the Gulf and Atlantic coasts estimated to be 250,000 (Guillory *et al.*, 2001), terrapin bycatch in ghost traps represent a major threat. Terrapin capture in crab traps was also episodic in Alabama. Seven terrapins were caught in the same trap at once in 2009, representing 33% of the terrapin capture for modified crab traps that year. Butler (2002) observed multiple terrapins in 25% of the crab capture events.

The placement of crab traps also plays a major role in the degree of impact they have on terrapin populations. The smaller-sized adult male and juvenile terrapins stay closer to shore (Roosenburg *et al.*, 1999), so crab traps set in these areas have a greater chance of capturing these turtles. Thus, recreational crab traps that are more numerous inshore and potentially checked less regularly pose a greater threat than commercial crab traps (Roosenburg, 1991; Mann, 1995; Roosenburg *et al.*, 1997). However, for the Cedar Point Marsh aggregation, commercial crab traps pose a greater threat. Terrapins were captured with modified crab traps, and numerous commercial traps mainly fish waters surrounding the marsh channels. Additionally, commercial traps have also been observed in the main channel at Cedar Point (Coleman, personal observation).

Nest predation also contributes to the inability of the Alabama population to rebound. Although Mann assumed that females laid 3 clutches per season, no females were captured more than twice on the nesting beach in Cedar Point Marsh. So assuming female terrapins in Alabama nest twice a year, the majority of nests at Cedar Point Marsh (166 hypothetical nests based on the estimate of 83 nesting females) are depredated each year (over 150 nests were found in 2009 and 2010). This is consistent with the nest survivorships observed in other populations (Roosenburg, 1991; Butler, 2002; Mitro, 2003). Road mortality (Wood and Herlands, 1997), boat propeller injuries (Roosenburg, 1991; Cecala *et al.*, 2008) and habitat loss (Roosenburg, 1991; Butler *et al.*, 2006) also represent major threats to terrapin survival. Mann (1995) suggested that sufficient nesting beaches may be a limiting factor to terrapin distribution. Suitable salt marsh habitat appears to be abundant in Alabama, but there may be a lack of elevated nesting beach surrounding the salt marshes.

Complete long-term data sets are critical in evaluating population trends and life history parameters in long-lived species such as turtles. Gibbons *et al.* (2001) documented the extirpation of a local population of diamondback terrapins that was attributed to recreational crab trap mortality. Local populations could be removed in a short time period leading to a wrong conclusion that terrapin were never locally abundant. Also, an assessment of population stability could be inaccurate if the number of nesting females is only estimated (Roosenburg *et al.*, 1997). Low survivorship of eggs and hatchlings is present in terrapin populations which have led to low population recruitment. Other populations in Maryland and South Carolina contained higher numbers of older individuals (Roosenburg, 1991; Gibbons *et al.*, 2001), but this pattern was only observed in females in the Alabama population. The majority of females captured in Alabama were older than age 5, although all may not have been reproductively active yet. With adult males, only one adult male above the age of six was captured. This underscores the impact that crab traps have represented on terrapins in Alabama. Congdon *et al.* (1993, 1994) stated that long-lived species have a suite of co-evolved life history traits that will preclude the ability to properly adapt to chronic disturbances and will ultimately result in the species' demise if the disturbances are not properly reversed or mitigated. Numerous anthropogenic threats are negatively affecting terrapins throughout their range. Long-term datasets such as the one that was initiated during the current study will play a prominent role in developing optimal management strategies aimed at ensuring the survival of the terrapin.

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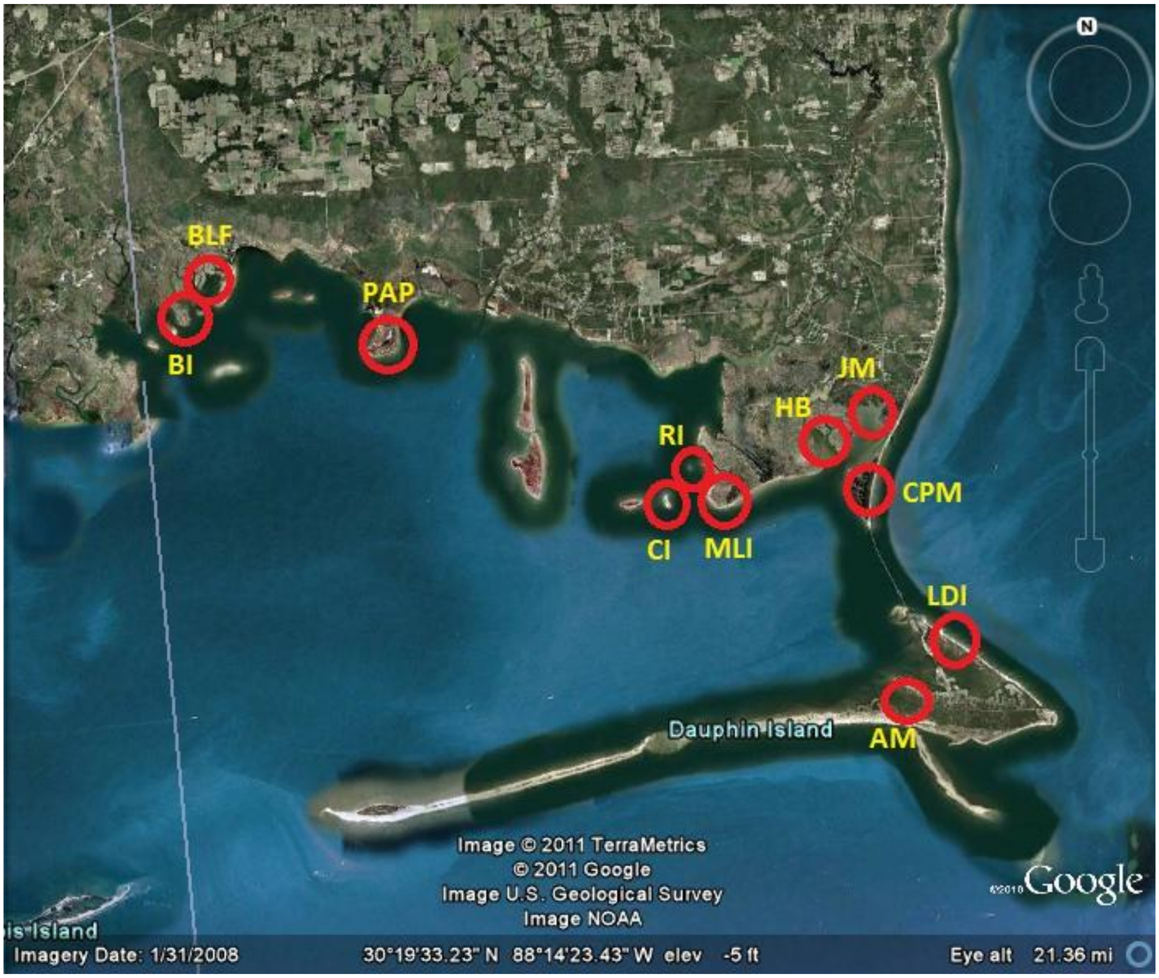

Figure 1. Salt marshes along the Gulf Coast of Alabama that were sampled from 2004-2010. (BI=Barton Island, BLF=Bayou La Fourche, PAP=Point Aux Pines, CI=Cat Island, RI=Raccoon Island, ML=Mon Louis, HB=Heron Bayou, JM=Jemison Marsh, CPM=Cedar Point Marsh, LDI=Little Dauphin Island, AM=Airport Marsh)

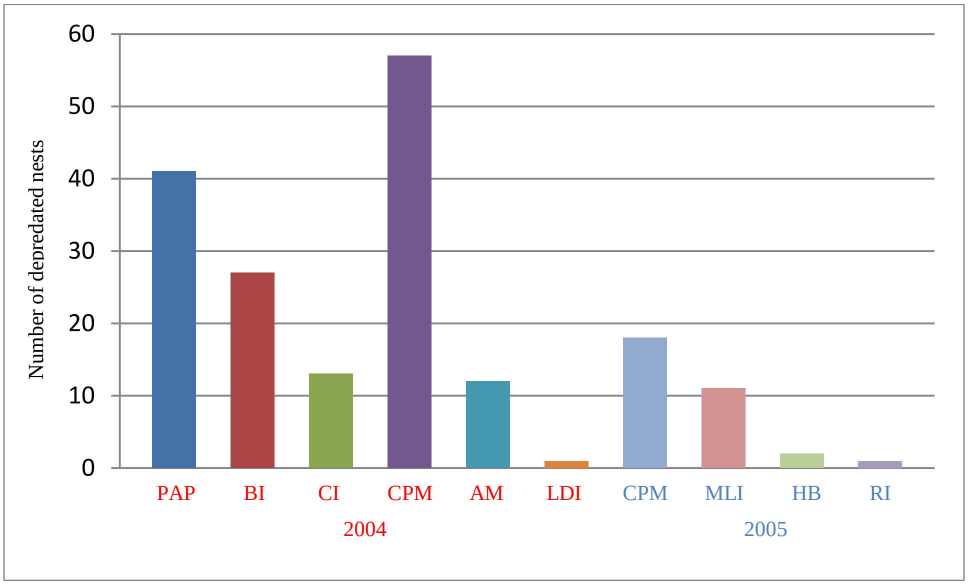

Figure 2. Number of depredated nests by location per year.

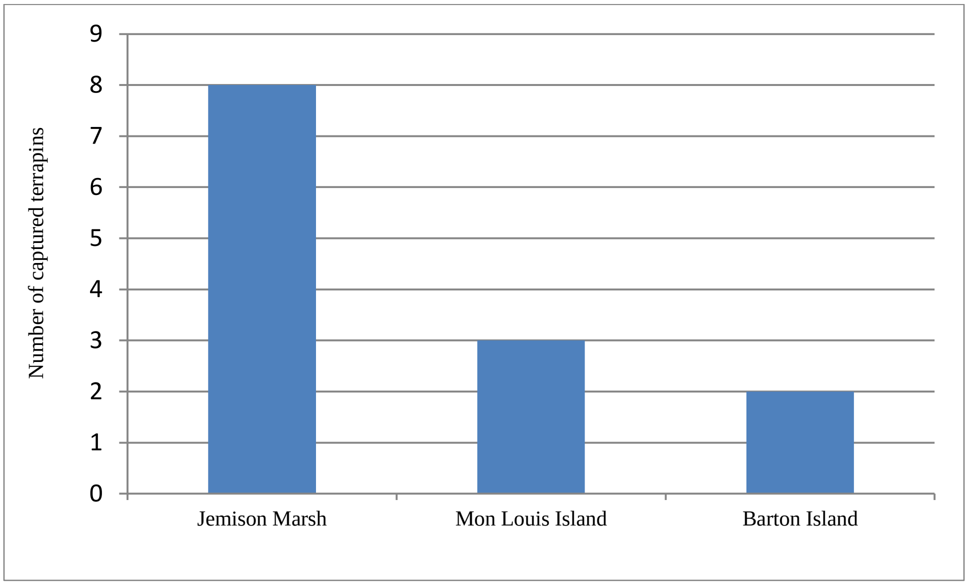

Figure 3. Number of captured terrapins in 2005 with modified crab traps by location.

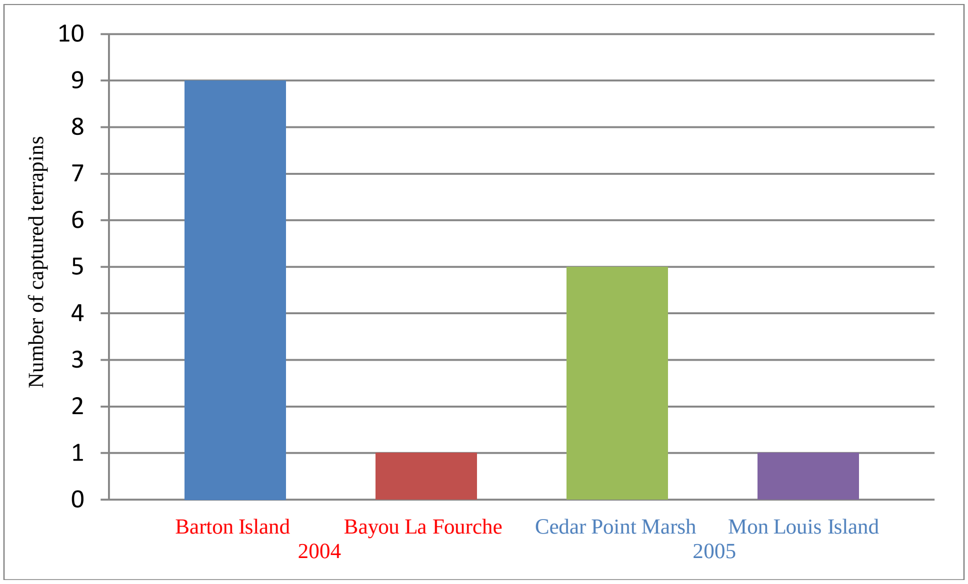

Figure 4. Number of captured terrapins with pitfall traps by location per year.

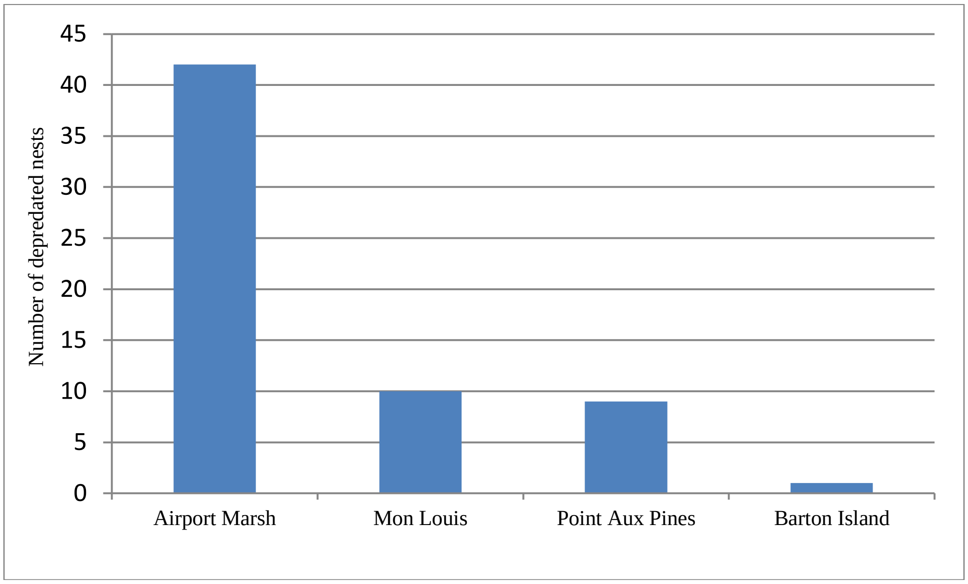

Figure 5. Number of depredated nests found at various locations from 2006-2010.

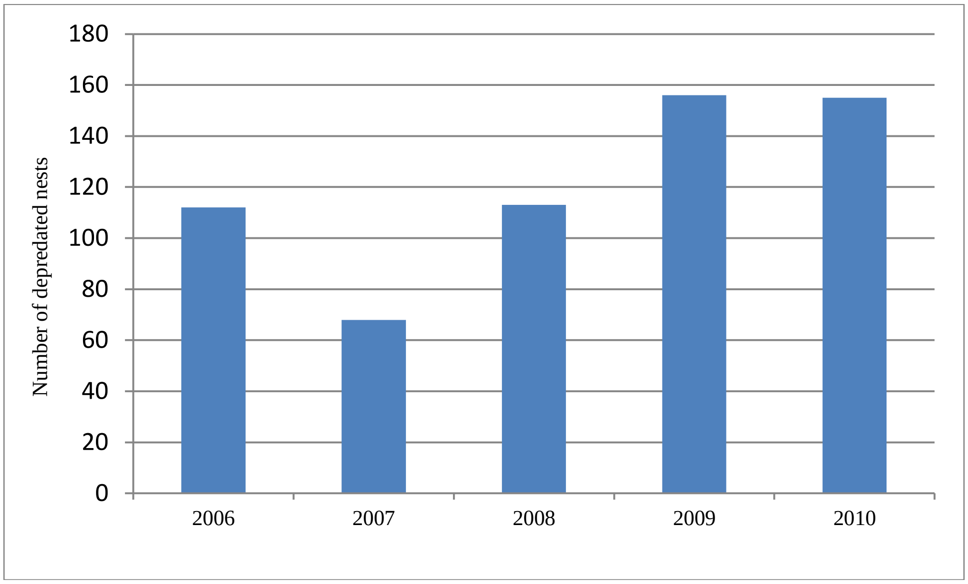

Figure 6. Summary of depredated nest surveys completed at Cedar Point Marsh from 2006-2010.

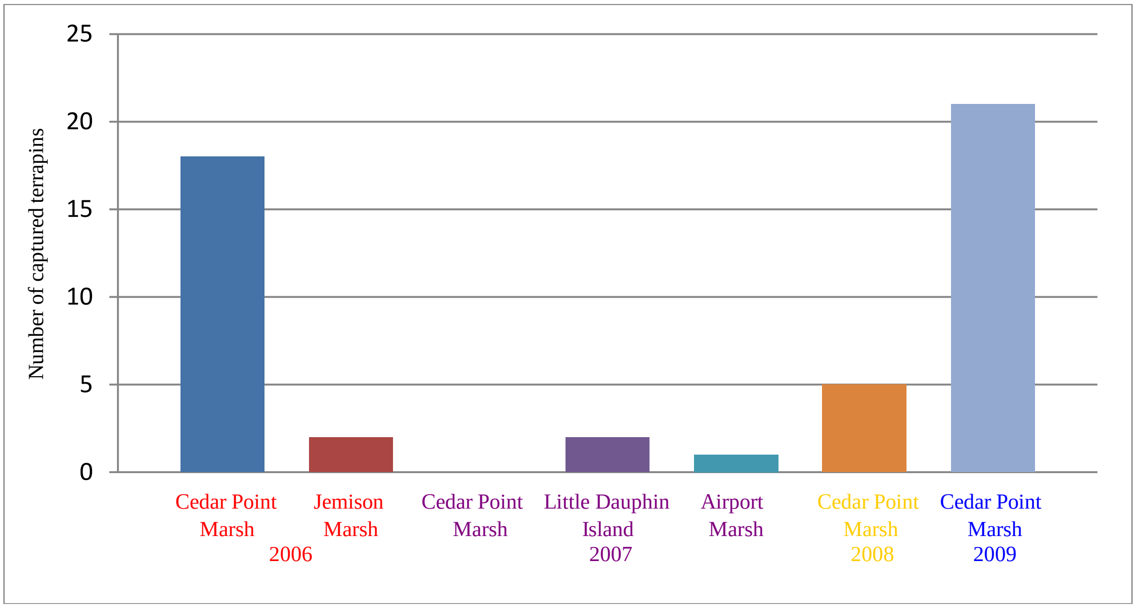

Figure 7. Number of captures with modified crab traps by location per year.

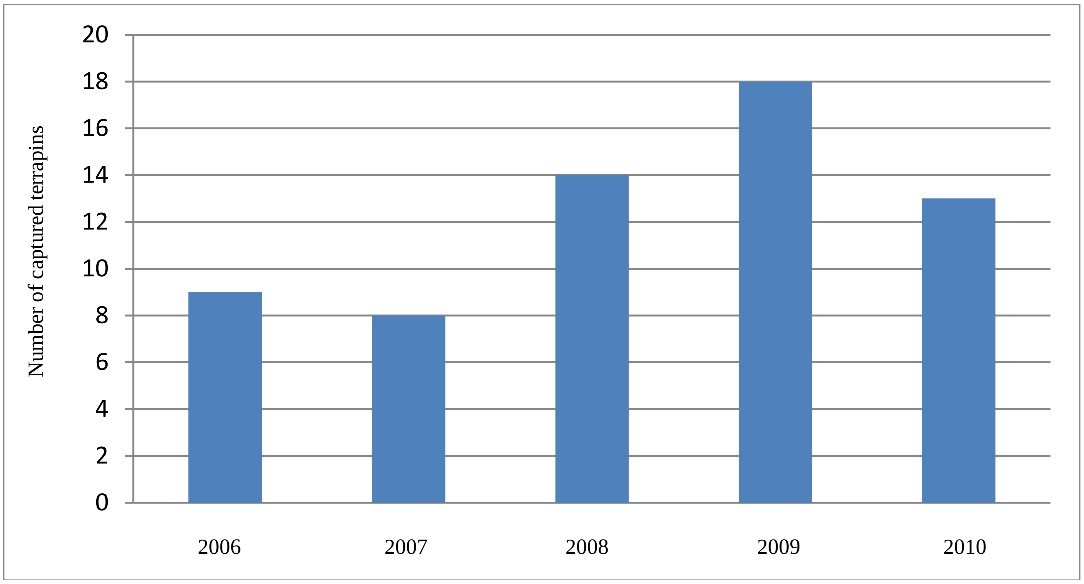

Figure 8. Number of captures with pitfall traps in Cedar Point Marsh per year.

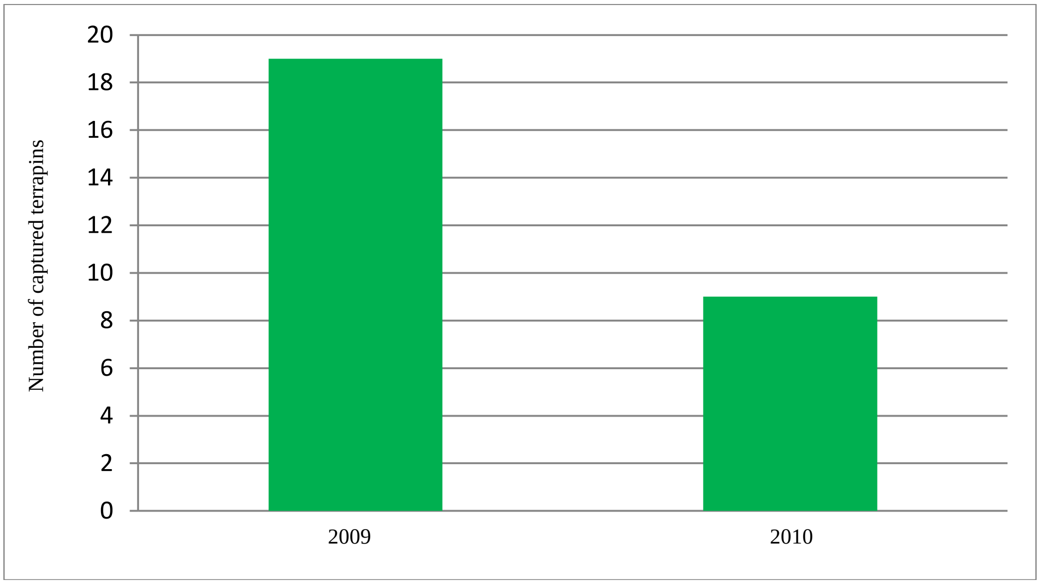

Figure 9. Number of captures in Cedar Point Marsh with trawling by year.

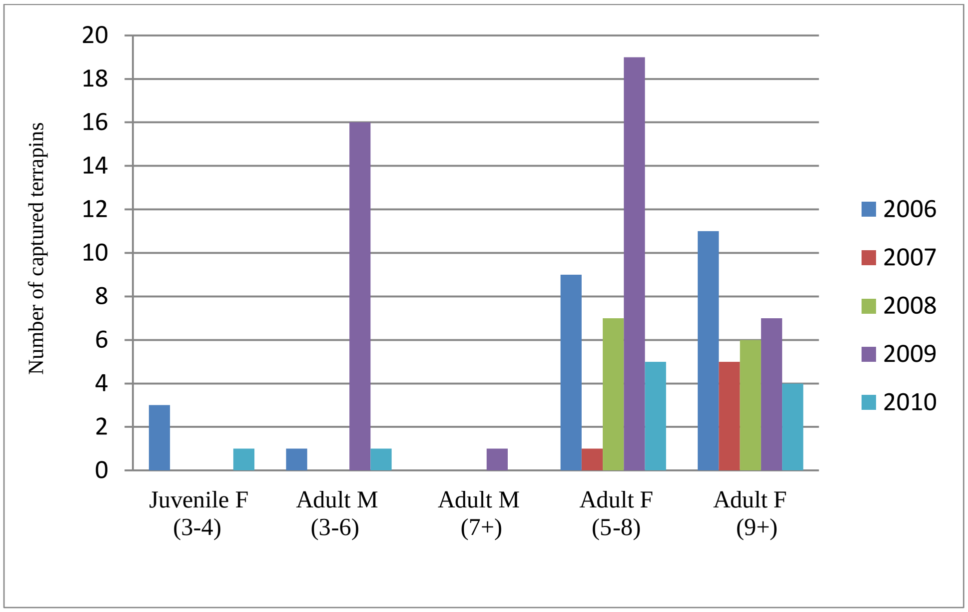

Figure 10. Captures of age classes by sex from 2006-2010 in Cedar Point Marsh.

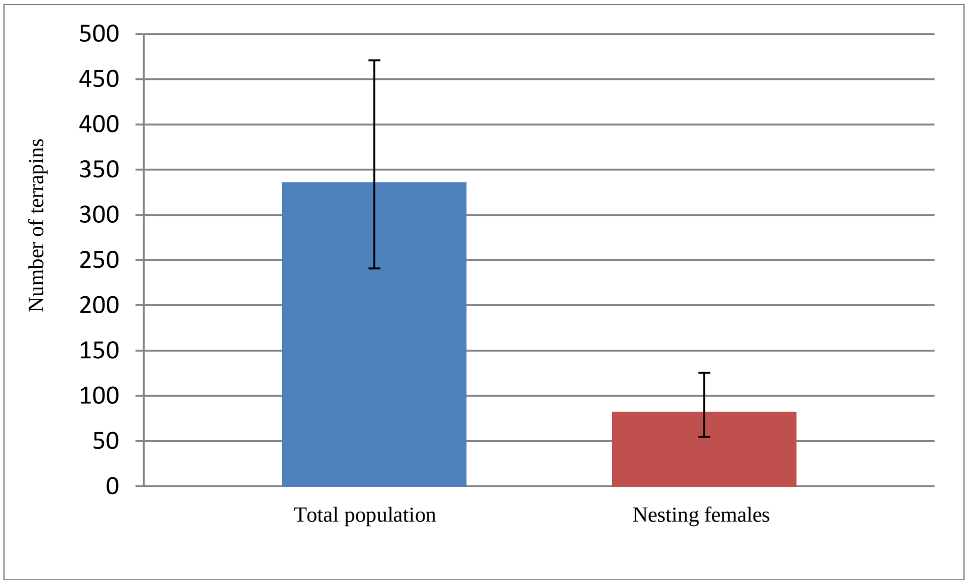

Figure 11. Population estimates for the population not including hatchlings as well as nesting females. Error bars represent 95% upper and lower confidence limits.

CHAPTER 3

POPULATION GENETIC EVALUATION OF THE DIAMONDBACK TERRAPIN (*MALACLEMYS TERRAPIN PILEATA*) IN ALABAMA

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In preparation for Journal of Herpetology

Format adapted for dissertation

Abstract

Molecular data can complement ecological data to provide a more clear understanding of the evolutionary processes that a population could be undergoing. The genetic diversity of twelve microsatellite loci was sampled from a severely depleted population of diamondback terrapins in Alabama, and the genetic uniqueness of this population was compared to other terrapin populations present along the Gulf Coast and South Atlantic Coast. Overall, reduced genetic diversity was detected in all sampled populations, but the Alabama population represented the greatest source of genetic diversity along the northern Gulf of Mexico. Four out the twelve loci significantly deviated from Hardy-Weinberg equilibrium in the Alabama population, and its *M* ratio was 0.34. These data indicated the population has experienced genetic drift most likely caused by a severe population size reduction. Based on results of Analysis of Molecular Variance and STRUCTURE analyses, the seven sampled populations fell under three groupings: the South Carolina group, the Florida group, and the Northern Gulf of Mexico group.

Introduction

Molecular data have provided insight into ecological and evolutionary processes at the species and population levels (Moritz, 1995) as well as indicated the importance of genetic diversity in conservation (Frankham, 1996). Haig (1998) discussed the wide range of information that can be elucidated with molecular tools, including taxonomic definition, hybridization, and individual identification. Within and among population genetic diversity along with rates of gene flow can also be examined, which has great implications for population viability. The combination of reliable molecular information with sound ecological and demographic data results in the best understanding of current status of the concerned population (Haig, 1998).

Bowen *et al.* (2005) presented a clear example of how population structure is revealed when field work is properly complemented with molecular data. The loggerhead sea turtle (*Caretta caretta*) is a highly migratory marine reptile that utilizes different habitats during its life stage. The amount of population admixture, detected by mitochondrial and microsatellite DNA variation, decreased as juveniles in pelagic habitats transitioned to adults in natal coastal habitats. Thus, the authors argued that management strategies at these various habitats would have different spatial influences (Bowen *et al.*, 2005).

Pearse *et al.* (2006) also incorporated both mitochondrial and microsatellite DNA in their investigation of population genetic structure of the giant Amazon River turtle (*Podocnemis expansa*). This exploited species is indigenous to the Amazon, Orinoco, and Essequibo River basins, and individuals can migrate long distances between feeding and nesting grounds, although not as extensive as marine turtles. In contrast to *C. caretta* females, the genetic data indicated that *P. expansa* females did not adhere to strict natal homing at the sub-basin level. In addition, populations were panmictic at this level due to extensive gene flow, although, populations between river basins were genetically isolated (Pearse *et al.*, 2006).

Previous Genetic Analyses

Several studies have examined population genetic structure in the diamondback terrapin. Avise *et al.* (1992) investigated the mutation rate of mitochondrial DNA (mtDNA) in six species of turtles, including diamondback terrapin. Based on restriction site assays, very low mtDNA variation was observed in each of the species suggesting remarkably lower mtDNA evolution in turtles. For the diamondback terrapin, the observed rate was 14-fold slower than the widely accepted 2%/Myr calibrated clock for mtDNA in derived vertebrate lineages (Avise *et al.*, 1992). Nucleotide diversity was 0.0002, and two haplotypes were present in more than one of the 53 individuals sampled. One haplotype was fixed in populations north of Cape Canaveral, and the other one fixed in those south of Cape Canaveral. This phylogeographic pattern was also observed in other sampled vertebrate and invertebrate species indicating a similar vicariant history in these species (Avise *et al.*, 1992).

To better elucidate finer scale genetic variation within and among populations in diamondback terrapins, variation in microsatellite DNA was examined (Hauswaldt and Glenn, 2005; Hart 2005). Microsatellites are relatively short pieces of nuclear DNA that consist of tandem repeats of 1-6 nucleotides, usually 5-40 repeats. Repeat motifs of dinucleotides, trinucleotides, and tetranucleotides are most commonly employed. Microsatellites are non-coding regions of DNA and thus have a high mutation rate (Frankham, 1996; Selkoe and Toonen, 2006). They are useful in answering questions of population structure or demographic changes in recent past (Haig, 1998; Selkoe and Toonen, 2006). Avise *et al.* (1992) argued that the low mtDNA diversity seen in turtles could be due to long generation times and low metabolic rates, so microsatellites should be more appropriate when investigating within species variation.

Hauswaldt and Glenn (2003) developed primers for six microsatellite markers that they employed to examine population genetic structure in several terrapin populations. Like Avise *et al.* (1992), they observed lower genetic variation than expected with the use of microsatellites. No genetic structure existed with the sampled Charleston, SC, estuary, despite some sites being 30km apart. This finding was surprising given the high site fidelity of terrapins when examining mark recapture data (Gibbons *et al.*, 2001). The authors postulated that juvenile dispersal and mating aggregations with males venturing to neighboring marshes may promote the gene flow that prevents the development of genetic structure. Low genetic variation was also observed between estuaries along the East Coast; and interestingly, the East Coast populations were more similar to the sampled Texas population than they were to the sampled Florida population. This was most likely an artifact of humans mixing the populations during the heyday of the terrapin market demand (Hauswaldt and Glenn, 2005).

Hart (2005) performed a range wide genetic analysis with diamondback terrapins, but she utilized markers developed for bog turtles by King and Julian (2004). Seven subspecies of terrapins are currently recognized based on differences in coloration and morphology (Brennessel, 2006). Based on her results, Hart (2005) argued for the establishment of six management units that did not coincide with the accepted subspecies boundaries. A management unit consists of one or more genetically homogenous populations that contain a significantly divergent set of allele frequencies of nuclear or mtDNA from other management units (Moritz, 1995). As observed by Hauswaldt and Glenn (2005), low population genetic structure existed between local and regional populations, with the highest levels between the various management units. Also, male-biased dispersal was detected thus facilitating local gene flow (Hart, 2005).

Objectives and Hypotheses

The current study utilized the twelve microsatellites used by Hart (2005) to investigate the genetic health of a diamondback terrapin population along the Gulf Coast of Alabama. Based on mark-recapture data, this population has suffered a significant population decline from historic levels of over 20,000 individuals to now approximately 400 (Chapter 2). Because of this decline, the potential for reduced genetic health is very high. The null hypothesis tested was that high levels of genetic variation exist in the Alabama population. A subset of Hart's (2005) data was incorporated to examine the genetic relatedness of the Alabama population to other Gulf Coast populations. The null hypothesis tested was that the Alabama population was genetically homogenous with the other sampled Gulf Coast populations.

Methods

Blood samples were obtained from terrapins that were trapped for a mark-recapture study performed at Cedar Point Marsh along the Gulf Coast of Alabama (Chapter 2). Populations from Hart (2005) included in the analysis as well as sample sizes are listed in Table 1. The blood samples were spun down to separate the whole blood from the serum. DNA was extracted with Fast Prep Bio 101 Kits (MP Biomedicals, Solon, OH) using the default protocol for animal tissues.

Table 1. Sampled populations included in genetic analysis and sample sizes.

Population	n	Population	n
South Carolina	48	Alabama	53
Nest Key, Florida	13	Louisiana	31
Everglades, Florida	50	Texas	14
Tampa, Florida	6		

The same twelve microsatellite loci examined by (Hart, 2005) were investigated in this study (Table 2). Primers for these loci were developed by King and Julian (2004).

Primers were constructed by Integrated DNA Technologies (Coralville, IA), and a fluorescently dye tag (blue colored FAM) was attached to each forward primer. PCR reactions were completed in a GeneAmp 2400 PCR System (Perkin Elmer, Waltham, MA). The following parameters were set for each reaction: 94°C for 2 minutes for the initial denaturing stage, 35 cycles of 94°C for 45 seconds, 56°C for 45 seconds, and 72°C for 2 minutes, and a final extension stage of 72°C for 10 minutes.

Successful amplified reactions were taken to the University of Alabama at Birmingham Center for AIDS Research sequencing facility for analysis. Samples were analyzed with an Applied Biosystems 3130xl Genetic Analyzer (Carlsbad, CA). Results were scored and binned with ABI Peak Scanner software v1.0. A binning procedure of adding or subtracting one or two base pairs was used depending on the consensus of the readouts. Because most repeat motifs for the loci were tetranucleotides, no bias was introduced into the interpretations.

Table 2. Microsatellite primers developed by King and Julian (2004) and used in Hart (2005) as well as the current study.

Locus	Accession Number	Size Range (# of bp)
GmuA18	AF337648	101-139
GmuB08	AF517229	193-264
GmuB67	AF517233	140-162
GmuB91	AF517234	115-150
GmuD21	AF517236	145-159
GmuD55	AF517240	153-220
GmuD62	AF517241	125-185
GmuD87	AF517244	212-292
GmuD90	AF517247	106-165
GmuD93	AF517248	113-196
GmuD114	AF517251	85-122
GmuD121	AF517252	120-190

Basic population genetic parameters were analyzed in Arlequin v3.5 (Excoffer and Lischer, 2010). A subset of Hart's (2005) data was included in the analysis to compare the Alabama population to other Gulf Coast and Atlantic Coast populations: one from Louisiana, Texas, and South Carolina. Three populations from Florida were also included: Tampa, the Everglades, and Nest Key. The observed and expected heterozygosity for each locus in each population were determined. Also, deviations from the Hardy Weinberg equilibrium for each population were calculated. The number of steps set in the Markov Chain was set at 1,000,000, and the number of dememorisation steps was set at 1,000. The test type was set at locus by locus. The mean ratio of the number of alleles to the range of allele size (M) was calculated for each population in order to detect possible population bottlenecks (Garza and Williamson, 2001). Three different Analyses of Molecular Variance (AMOVAs) were run with 1,000 permutations to examine population differentiation under three different groupings. The first analysis compared the four Gulf Coast populations (Texas, Louisiana, Alabama, and Tampa, FL) to the Southern FL/Atlantic Coast populations (Everglades, FL, Nest Key, FL, and SC). The second analysis switched the Tampa samples to the FL/Atlantic Coast group, and the third analysis switched the Louisiana samples with the Tampa samples. These different analyses were completed to examine the genetic uniqueness of the Gulf Coast populations. Population pairwise F_{ST} values were calculated to compare genetic variation between populations. One hundred permutations were run at an α level of 0.05, and a distance matrix was computed.

STRUCTURE (Pritchard *et al.*, 2000), which employs a Bayesian clustering method, was used to infer population genetic structure. The samples were assigned to their sampled populations, and admixture was allowed. The program evaluated the number of assumed populations, K, from 2-7 with a burnin of 50,000 iterations followed by 500,000 iterations for each value of K. K values were run 2-3 times to examine the precision of the results.

Results

A total of 53 terrapins from Alabama were sampled, and 68 alleles were observed for the 12 microsatellites. The mean allelic diversity (Figure 1) and allelic range (Figure 2) for the Alabama population was among the highest for the Gulf Coast populations. The observed heterozygosities for 4 of the 12 loci significantly deviated from Hardy-Weinberg equilibrium (Table 3), and the mean observed heterozygosity across the 12 loci were similar to the other Gulf populations (Figure 3). Each population had low *M* ratios (Figure 4), much lower than the .68 *M* ratio threshold proposed by Garza and Williamson (2001). The Nest Key, FL, population had the highest *M* ratio, 0.40, and the Alabama population's ratio was 0.34. However, four loci in the Nest Key, FL, population were monomorphic and did not contribute to the *M* ratio.

The highest F_{ST} value, 0.196, was in the analysis with Louisiana in the Southern Florida/Atlantic group (Table 4). The lowest F_{ST} value, 0.153, was observed when the Tampa, Florida, population was included in the Southern Florida/Atlantic group. In the population specific comparison, the lowest F_{ST} values for Alabama was between Louisiana, 0.026, and Texas, 0.058. (Table 5). F_{ST} values for the Tampa, Florida, population were lowest for the Nest Key and Everglades, Florida, populations.

Table 3. Observed and expected heterzygosities for the 12 sampled loci in the Alabama population. Significant deviations are denoted with asterisks.

Loci	**obs. H**	**Exp. H**	**p**
B91	0.01887	0.01887	1
B08	0.83019	0.78095	0.90771
D93	0.69811*	0.62695	0.01477
A18	0.50943*	0.70872	0.00038
D87	0.92453	0.85517	0.24146
B67	0.01887	0.01887	1
D90	0.66038*	0.75813	0.0346
D55	0.11321	0.10979	1
D114	0.03774*	0.07403	0.01941
D21	0.62264	0.63827	0.85654
D121	0.84906	0.72399	0.0755
D62	0.71698	0.75238	0.76213

Table 4. Analyses of Molecular Variance results for the three different groupings. The first analysis evaluated all Gulf populations together. The second analysis grouped the Tampa, FL, population with the other Florida and South Carolina populations. The third analysis switched the Louisiana samples with the Tampa samples in the groupings. All F_{ST} values were statistically significant.

AMOVA	**F_{IS}**	**F_{ST}**	**F_{IT}**
1	0.065	0.162	0.055
2	0.065	0.153	0.071
3	0.065	0.196	-0.011

Table 5. Pairwise population F_{ST} comparisons. All comparisons were significant except for the Tampa and Nest Key, Florida, comparison. (AL—Alabama, LA—Louisiana, TX—Texas, TFL—Tampa, Florida, EFL—Everglades, Florida, NKFL—Nest Key, Florida, SC—South Carolina)

	AL	SC	NKFL	Everglades	Tampa	Louisiana
AL						
SC	0.196					
NKFL	0.263	0.225				
Everglades	0.248	0.269	0.036			
Tampa	0.199	0.178	0.03*	0.043		
Louisiana	0.026	0.175	0.228	0.21	0.155	
Texas	0.058	0.158	0.228	0.22	0.158	0.040

Utilizing the Bayesian clustering method under STRUCTURE, three groups were detected, the South Carolina population, the Florida populations, and the Northern Gulf of Mexico populations (Figure 5). When the number of populations (K) were set at three, the Northern Gulf of Mexico group included 98% of the Alabama, Louisiana, and Texas samples. The same was observed for the other two groups, except for the Tampa population, which 76% were included. The next largest percentage, 16%, was observed in the South Carolina group. The net nucleotide distance between the Northern Gulf of Mexico group and the Florida and South Carolina groups were 0.136 and 0.127, respectively. However, variation did exist within the groups, especially the Florida and Northern Gulf of Mexico groups. The F_{ST} value for the Florida group was .32 whereas for the Northern Gulf of Mexico group, it was 0.193.

Discussion

The Alabama population of diamondback terrapins along with the other sampled populations included in this study displayed decreased genetic variation, so the null hypothesis for high levels of genetic variation in Alabama terrapins was rejected. The mean allelic diversity was below 6 alleles per loci for each population except for South Carolina, which had an average over 8. The observed heterozygosity for four out of the twelve loci in the Alabama population deviated significantly from Hardy Weinberg equilibrium. Five loci in the South Carolina population significantly deviated from Hardy Weinberg equilibrium. The main cause for the depletion of heterozygosity was genetic drift resulting from drastic population size reduction. Overall size for the Cedar Point Marsh aggregation in Alabama was diminished from an historical level of reportedly 20,000 individuals to a current size of 336 (excluding 0-2 year olds). The other terrapin populations included have experienced similar historical exploitations and are currently facing the same threats (Chapter 1).

This conclusion was also supported by the populations' *M* ratios, which are derived from dividing the total number of alleles by the allelic range. Garza and Williamson (2001) theorized populations that have undergone recent bottlenecks would have lower *M* ratios. Through the loss of random alleles, genetic drift would cause larger gaps in the allelic range without drastically decreasing the overall range size. The decreased *M* ratios will take longer to recover if the population size remains depleted, and this measure could be utilized to distinguish between populations that have been recently reduced and those that are small yet stable populations (Garza and Williamson, 2001). Various studies have employed this methodology in detecting population bottlenecks in numerous species: elephants (Whitehouse and Harley, 2001), monk seals (Pastor *et al.*, 2004), salmon (Garza *et al.*, 2004), salamanders (Spear *et al.*, 2006), and pine trees (Al-Rabab'ah and Williams, 2004). It has also been utilized in genetic studies of *P. expansa* (Pearse *et al.*, 2006) and ornate box turtles (*Terrapene ornata*) (Kuo and Janzen, 2004). Each of the terrapin populations displayed *M* ratios less than 0.40, which is well below the threshold value for bottlenecked populations of 0.68 (Garza and Williamson, 2001). Genetic variation is directly related to population size, and this relationship should be most evident for neutral genetic markers, such as non-coding microsatellites (Frankham, 1996). Kuo and Janzen (2004) did not observe reduced *M* ratios in a depleted population of ornate box turtles in which a population bottleneck was known to have occurred. The authors attributed this to the bottleneck being more gradual in nature than a drastic occurrence (Kuo and Janzen, 2004). This conclusion supports the argument that the population bottleneck experienced by the examined terrapin populations was quite severe.

In contrast to these results, terrapin populations included in the Hauswaldt and Glenn (2005) study did not have *M* ratios below the threshold value. Also, values for allelic diversity and observed heterozygosity observed in the Hauswaldt and Glenn (2005) study differed from the current study, even though the same populations in South Carolina, Florida, and Texas were included in both studies. Allelic richness for the Texas population in the Hauswaldt and Glenn (2005) study was almost twice that observed in the present study. This disparity is most likely due to different loci being examined in the two studies indicating how varying conclusions could be drawn depending on which loci are utilized.

Currently, 7 subspecies of diamondback terrapins are recognized, and five were included in this study: *Malaclemys terrapin centrata* (South Carolina), *M. t. rhizophrarum* (Nest Key, Florida), *M. t. macrospilata* (Everglades and Tampa, Florida), *M. t. pilieata* (Alabama and Louisisana), and *M. t. littoralis* (Texas). Based on her range-wide study, Hart (2005) concluded that terrapins exist in 6 genetic management units (MUs). MUs are delineated based on differences in allele frequency and are more prudent for short term management as opposed to evolutionary significant units (ESUs), which should be viewed on a long term time scale (Moritz, 1994, 1995). The SC population would fall under the Coastal Carolina MU including the range of *M. t. centrata*. The Nest Key, FL, population would fall under the Southern Florida MU, and the three Gulf of Mexico subspecies would be included into one Gulf of Mexico MU. Low amounts of variation were observed among the populations within their respective regions (Hart, 2005). Hauswaldt and Glenn (2005) also observed little genetic structure on a regional spatial scale.

Based on the results in this study, there would be hesitancy to group the three Gulf of Mexico subspecies into one MU, thus the null hypothesis stating that the Gulf populations were genetically homogenous was rejected. The low level of genetic differentiation between the Alabama and Louisiana population was expected because both are *M. t. pileata* subspecies populations. Also, the F_{ST} values between Texas and Alabama and Louisiana (0.058 and 0.040, respectively) indicated that *M. t. pileata* and *M. t. littoralis* can be grouped within one MU. F_{ST} values for the comparisons between these populations and the Tampa, FL, population were all above 0.15, which Hartl (1988) proposed as indicating high levels of genetic differentiation. It should be noted similar pairwise values were observed between these populations in Hart (2005). The overall AMOVA results also supported this conclusion. The highest F_{ST} value was observed when the Louisiana samples were placed in the Atlantic/Florida group, while the lowest F_{ST} value was observed when the Tampa samples were included. The STRUCTURE analysis designated three basic groupings: the SC group, the FL group, and Northern Gulf of Mexico group. The greatest amount of variation was detected in the Northern Gulf group when K was set from 5-7.

However, the sample collection of Hart (2005) was more widespread than the current study, and the more narrow scope could have led to an overestimation of genetic divergence in this study. In addition, more samples from the Tampa population would have provided greater ability to correctly identify to which MU this population belongs. Further, sampling *M. t. macrospilata* populations closer to the boundary with *M. t.*

pileata, such as from the Big Bend area or the panhandle of Florida, would better elucidate the genetic differentiation between these two subspecies.

Possible juvenile dispersal or mating aggregations were suggested to promote the lack of genetic structure observed between local populations (Hauswaldt and Glenn, 2005). Hart (2005) found that males are the dispersing sex migrating to neighboring populations during the mating season and maintaining the genetic connections. But, Hauswaldt and Glenn (2005) postulated that longer term evolutionary processes, e.g., reestablishment of populations after the last glacial period, are more important in explaining the lack of genetic differentiation between estuaries, although sex biased dispersal and gene flow do have an influence.

Because terrapin populations appear to be isolated in some areas thus making gene flow improbable, relocating individuals within estuaries or to neighboring estuaries represents a viable management option (Hauswaldt and Glenn, 2005). Local extirpation has been observed (Gibbons *et al.*, 2001), and although males migrate to other marshes for mating, this will not necessarily result in recolonization of extirpated populations (Hart, 2005). Translocation should only be done within ESUs, so that long term evolutionary processes are not disrupted or modified (Moritz, 1995). This was echoed by Pearse *et al.* (2006) when the authors argued that *P. expansa* hatchlings could be transplanted to other populations within the same river sub-basin but not between rivers. Introducing genetic diversity through introducing individuals into a depleted population has been successful in other reptile species (Madsen *et al.*, 1999). However, there are concerns that translocation has the potential to introduce foreign genes that could negatively impact local adaptation (Moritz, 1995). Also, unless the threats that the population currently face are properly addressed, introduction of new individuals will not have the desired effect (Frazer, 1992). On the other hand, these concerns should be balanced against the notion that depleted population size could continue to promote genetic drift such that variation will be lost more quickly along with the ability of the population to adapt to environmental changes (Frankham, 1996; Hart, 2005).

Methodology limitations

The use of microsatellites to examine genetic diversity can present inherent problems. First, the mutation processes that the specific microsatellites follow may not meet the assumptions of the traditionally used infinite allele model (IAM) and stepwise mutational model (SMM). However, new statistical packages are addressing this problem (Selkoe and Toonen, 2006). Second, with only using differences in base pair sizes of alleles to assess diversity, some genetic diversity may be lost without examining the specific sequences (Selkoe and Toonen, 2006). Third, using non-coding DNA could overestimate genetic divergence between populations because they are not under the degree of selection that coding regions of DNA are (Frankham, 1996). Given this, terrapins may not be as homogenous at loci that determine fitness (Hauswaldt and Glenn, 2005). A limitation of using the *M* ratio (Garza and Williamson, 2001) is the inability to include fixed alleles. Because they have an allele size range of zero, this value cannot be calculated for fixed alleles. As seen in the Nest Key, FL, *M* ratio values can misjudge genetic loss because monomorphic alleles cannot be included in the calculations.

Conclusions

The Alabama population of diamondback terrapins displayed lower levels of genetic variation, and the genetic signature reflects a massive reduction in historical population size. Low levels of genetic variation were also detected in the other sampled populations of diamondback terrapins. Based on the amount of genetic differentiation between the populations, the Alabama, Louisiana, and Texas populations could be managed as one MU, but more samples are needed to discriminate the differentiation between this MU and the populations of *M. t. macrospilata*. Although heavily depleted, the Alabama population represents a relatively important source of genetic diversity for the Northern Gulf of Mexico MU. The historical gene flow that influenced genetic connectivity may not be occurring to the same degree today, so future monitoring is necessary to assess the genetic viability of terrapin populations throughout their range.

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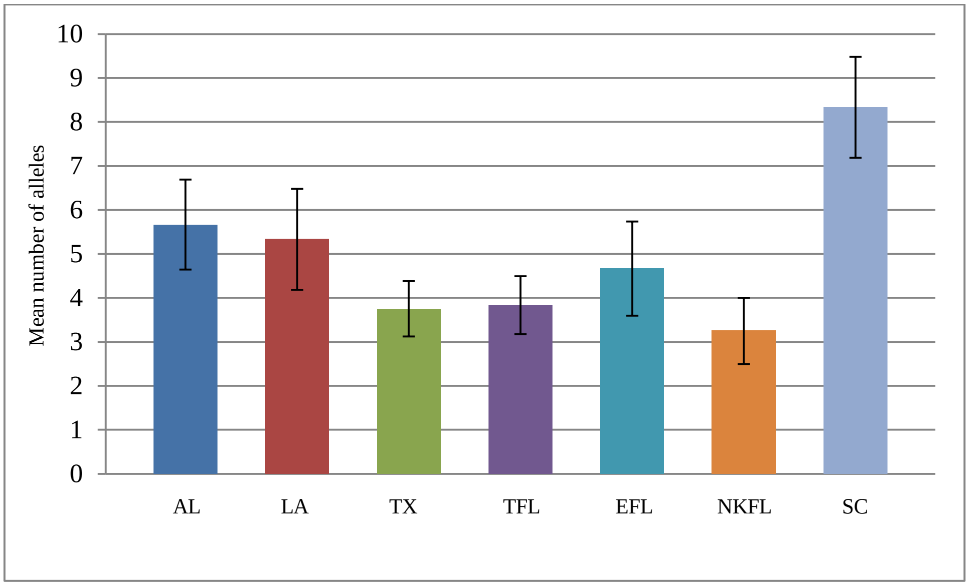

Figure 1. Mean allelic diversity of the sampled populations across 12 loci. Standard error bars are included. (AL—Alabama, LA—Louisiana, TX—Texas, TFL—Tampa, Florida, EFL—Everglades, Florida, NKFL—Nest Key, Florida, SC—South Carolina)

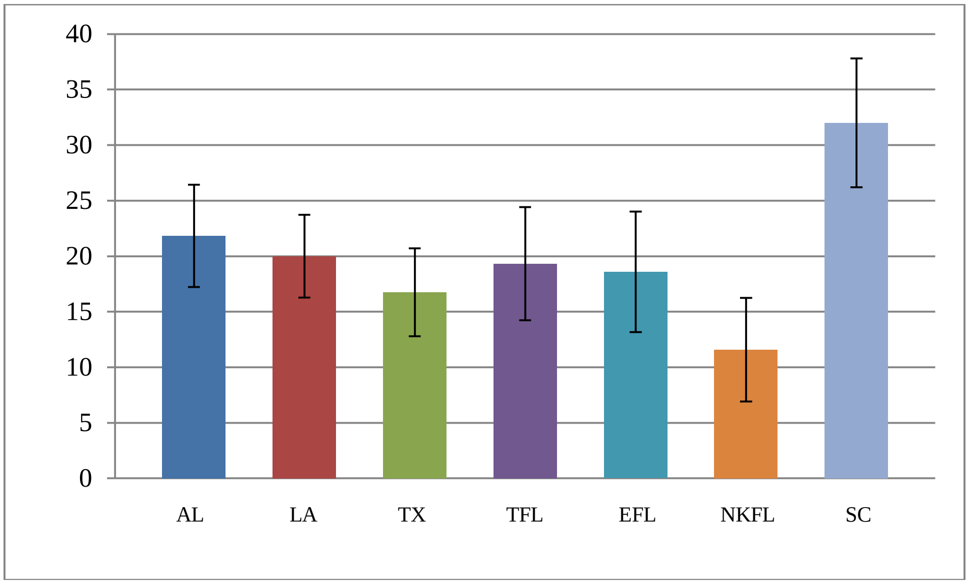

Figure 2. Mean allelic range of the sampled populations across 12 loci. Standard error bars are included. (AL—Alabama, LA—Louisiana, TX—Texas, TFL—Tampa, Florida, EFL—Everglades, Florida, NKFL—Nest Key, Florida, SC—South Carolina)

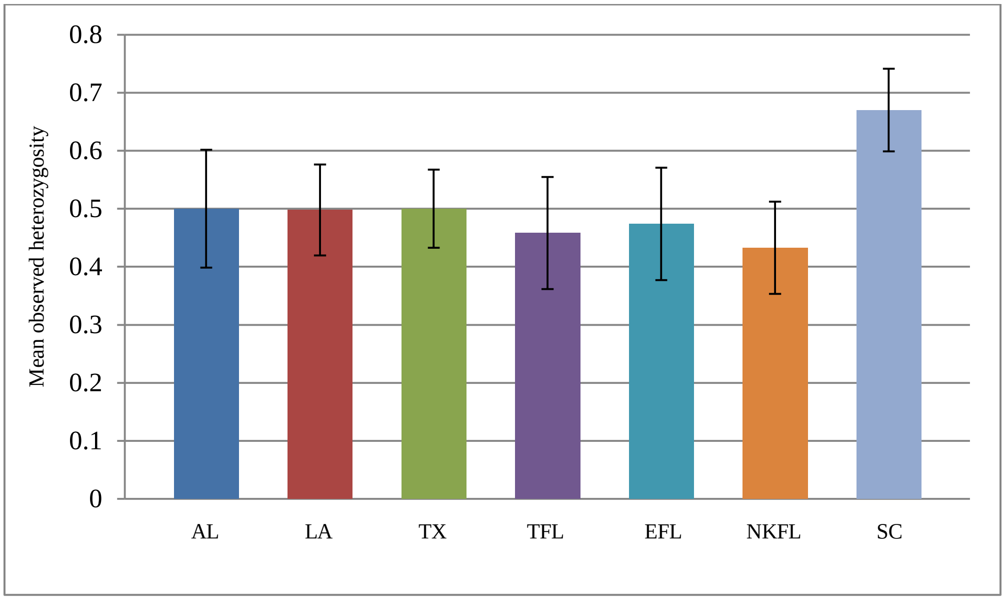

Figure 3. Observed heterozyogsities of the sampled locations across 12 loci. Standard error bars are included. (AL—Alabama, LA—Louisiana, TX—Texas, TFL—Tampa, Florida, EFL—Everglades, Florida, NKFL—Nest Key, Florida, SC—South Carolina)

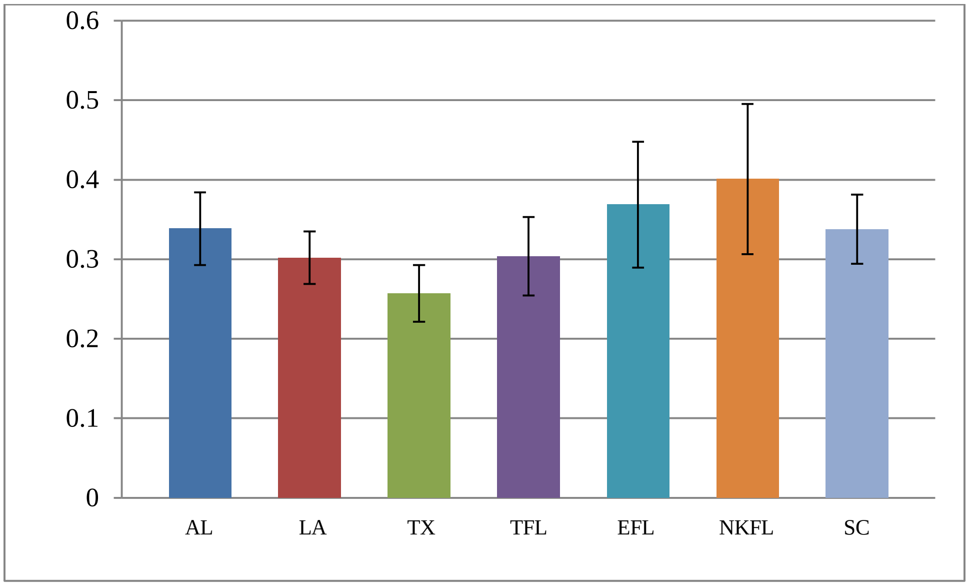

Figure 4. Mean *M* ratio for the sampled populations across 12 loci. Standard error bars are included. (AL—Alabama, LA—Louisiana, TX—Texas, TFL—Tampa, Florida, EFL—Everglades, Florida, NKFL—Nest Key, Florida, SC—South Carolina)

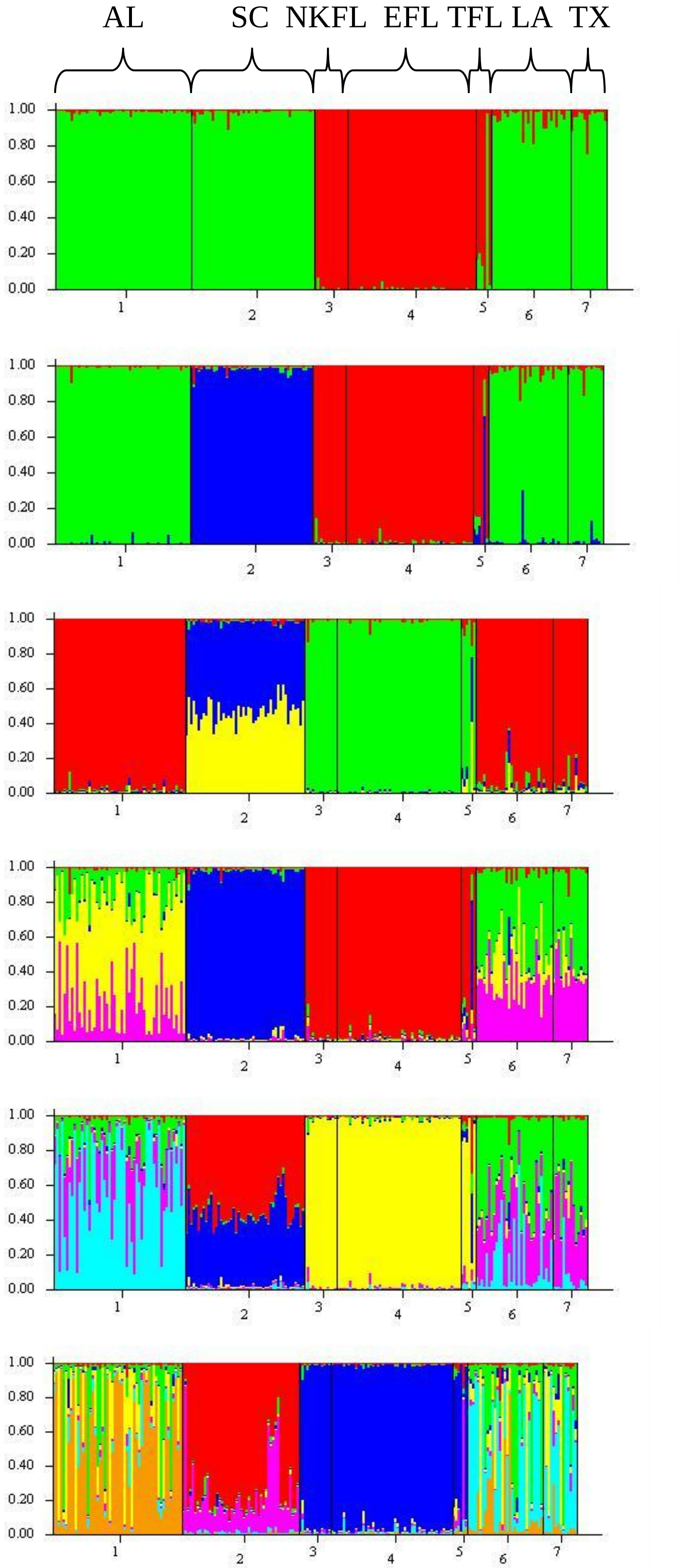

Figure 5. Evaluation of K (2-7) populations using Bayesian clustering method. In each K scenario, different colors denote different populations.

CHAPTER 4

EFFECT OF BYCATCH REDUCTION DEVICES (BRDS) ON THE CAPTURE OF DIAMONDBACK TERRAPINS (*MALACLEMYS TERRAPIN*) IN CRAB TRAPS IN AN ALABAMA SALT MARSH

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In preparation for Journal of Herpetology

Format adapted for dissertation

Abstract

Bycatch is a serious conservation threat to populations of numerous marine species, including diamondback terrapins, and requires immediate attention. Terrapins are estuarine turtles that can have an important role in maintaining ecosystem integrity of their salt marsh habitats. Unfortunately, numerous populations are being depleted due to drowning in submerged commercial and recreational crab traps. Bycatch reduction devices (BRDs) were developed to prevent terrapin bycatch without affecting crab capture, and they have been shown in previous studies to be successful. The current study examined the efficacy of BRDs in a depleted population of terrapins along the Gulf Coast of Alabama. The BRDs significantly prevented terrapin bycatch; however, crab capture was significantly lower in traps modified with BRDs. This result was in stark contrast to past studies that had much larger sample sizes and found no significant effects in crab capture with the use of BRDs. Therefore, BRDs represent an effective and practical management tool in preventing terrapin bycatch in commercial and recreational crab traps. Further, management strategies should include derelict crab trap removal programs.

Introduction

Bycatch from marine fisheries has been documented to significantly impact a variety of vertebrates. As examples, the impact has been well documented and has become a conservation concern in a variety of species such as sea turtles (Hall *et al.*, 2000; Lewison *et al.*, 2003; Pinedo and Polancheck, 2004; Gilman *et al.*, 2006; Moore *et al.*, 2009), sharks (McKinnell and Seki, 1998; Beerkircher *et al.*, 2002; Campana *et al.*, 2009), seabirds (Melvin *et al.*, 1999; Lewison and Crowder, 2003; Gilman *et al.*, 2005; Moore *et* *al.*, 2009) and marine mammals (Cox *et al.*, 1998; Read *et al.*, 2006; Moore *et al.*, 2009). These are examples of taxa displaying certain K-selected traits such as long life spans and delayed sexual maturity; therefore, the survival status of these species is particularly susceptible to bycatch-induced mortality of adults and/or subadults.

The diamondback terrapin (*Malaclemys terrapin*), is an exclusively estuarine turtle whose range extends from Cape Cod, MA, along the Atlantic and Gulf Coasts to Corpus Christi, TX, and displays similar life history traits. Unfortunately, many populations have declined significantly due to a variety of natural and anthropogenic threats. Major threats include crab-trap induced mortality (Roosenburg *et al.*, 1997; Wood 1997; Roosenburg, 2004), as well as habitat loss (Roosenburg, 1991), nest predation (Feinberg and Burke 2003, Lazell Jr and Auger 1981, Draud, Bossert and Zimnavoda 2004), and road mortality (Wood and Herlands , Szerlag and McRobert 2006). In the case of crab-trap-induced mortality, juvenile and adult terrapin represent bycatch that is attracted to bait, captured blue crabs (*Callinectes sapidus*), or other bycatch. Once in the submerged crab traps, terrapins will drown unless the trap is checked soon after they are captured, especially in oxygen-depleted waters during the summer months (Roosenburg *et al.*, 1997; Wood 1997; Roosenburg, 2004).

The proliferation of the blue crab fishery along the Atlantic and Gulf coasts of the U.S. from the middle of the 20th century to present day (Kennedy et al., 2007) has significantly impacted terrapin populations because the range of terrapins greatly overlaps with the habitat that is heavily fished for that resource. Roosenburg *et al.* (1997) described just how devastating crab trap mortality can be to a population. Based on both trapping and mark recapture data, the authors estimated that crabbing could potentially remove 58-78% of the population annually. Even if the estimates were overstated by a factor of four, it still could result in annual mortality of 20% of the population (Roosenburg *et al.*, 1997).

A long-term mark recapture study by Dorcas *et al.* (2007) indicated that crab trap-induced mortality was a major factor causing the decline of a terrapin population inhabiting the salt marshes of Kiawah Island, GA. The decline was accompanied by a shift in age structure to older individuals and a shift in sex ratio to more females, which would be the predicted impact of crab traps (Dorcas *et al.*, 2007). Crab traps were indicated as the primary factor affecting the decline and demographic changes in the Kiawah Island, GA, population, including both the commercial as well as the recreational crab fishery (Hoyle and Gibbons, 2000; Dorcas *et al.*, 2007). Roosenburg *et al.* (1997) also suggested that crab trap mortality was the cause of the female-biased sex ratio observed in their Maryland population. Terrapins exhibit sexual dimorphism, with adult females growing to a much larger size than adult males (Lovich and Gibbons, 1990), to account for divergent reproductive priorities (Gibbons *et al.*, 2001). As a consequence, adult males never reach the size that most adult females do, which would exclude them from entering crab traps (Roosenburg *et al.*, 1997).

Not only do regularly fished traps pose a danger to terrapins, but so do abandoned traps that are termed “derelict” or “ghost traps”. Derelict traps have been found to contain numerous terrapin carcasses: 49 in one trap in Maryland (Roosenburg, 1991) and 133 in two traps in Georgia (Grosse *et al.*, 2009). Based on mark-recapture estimates for the tidal creek from which the two traps were recovered in Georgia, these dead individuals represented approximately double the number of remaining live terrapins in that population (Grosse *et al.*, 2009). In the Gulf of Mexico, yearly crab trap loss can be quite high (20-100%) (Guillory *et al.*, 2001) and has been conservatively estimated to be approximately 250,000 derelict traps per year (Perry *et al.*, 2003), thus, reinforcing the magnitude of the threat these abandoned traps pose.

In addition of the direct mortality of bycatch, authors have suggested ecological effects of the discarded bycatch need to be examined. Both noncommercial and commercial species rely on a functioning ecosystem, and the removal or severe reduction of certain species can greatly alter this (Hall *et al.*, 2000; Kennelly and Broadhurst, 2002). Recent research has indicated that diamondback terrapins could possibly represent a keystone predator in their salt marsh habitats (Silliman and Zieman, 2001; Silliman *et al.*, 2005; Gustafson, 2006). A major prey item is the periwinkle snail (*Littoraria irrorata*), and these snails have been shown to significantly decimate salt marsh vegetation through their fungal farming activities (Silliman and Zieman, 2001; Silliman *et al.*, 2005; Gustafson, 2006).

Modifications to crab traps have been developed in an effort to decrease terrapin mortality. Roosenburg *et al.* (1997) changed the overall shape and size of crab traps by removing the top of the trap and adding PVC and netting so that the top of the trap remains above the water's surface. Wood (1997) developed a bycatch reduction device (BRD), which is a metal or plastic rectangle and fits into the funnel openings of the trap. It reduces the size of the openings to theoretically prevent terrapin entry without affecting crab capture.

The purpose of the current study was to assess the effectiveness of BRDs in reducing terrapin captures in crab traps in the salt marshes of Alabama. Diamondback terrapins were once an abundant and economically important species in Alabama, but now the state lists the terrapin as a species of "highest conservation concern" (Mirarchi *et al.*, 2004). In addition to the capture of terrapins, the study also evaluated the effect of BRDs on the capture of marketable-sized blue crabs. The results provide insight on whether or not BRDs represent an effective and practical method for decreasing crab trap-induced mortality of the diamondback terrapin in the saltmarshes of Alabama.

Methods

A side-by-side comparison of crab traps fitted with BRDs and not fitted was completed from May to August in 2007-2009. Eight pairs of traps were placed at various locations in Cedar Point Marsh (N 30° 19' 33.70" W 88° 08' 36.36"). The Alabama population of diamondback terrapins exists in isolated remnant aggregations, and the largest one inhabits Cedar Point Marsh. This terrapin aggregation has been extensively studied starting in 2004 (Chapter 2). Cedar Point Marsh is dominated by black needle rush (*Juncus roemerianus*), salt marsh cord grass (*Spartina alterniflora*), and coastal salt grass (*Distichlis spicata*). Invertebrates and vertebrates inhabiting the marsh include fiddler crabs (*Uca panacea*), periwinkle snails, striped mullet (*Mugil cephalus*), Atlantic croaker (*Micropognias undulates*), and Southern flounder (*Paralichthys lethostigma*). Modified crab traps following the design of Roosenburg *et al.* (1997) were utilized. The BRDs were metal rectangles measuring 5.08cm x 15.24 cm (2" x 6") and were placed midway into the funnel openings. Traps were checked and baited with menhaden at least twice a week during the sampling periods.

Carapace width (point to point) of crabs was measured to the nearest inch and released, unless kept for bait, when the traps were checked. For the terrapins that were captured, a suite of morphological measurements were collected (Table 1). The sex of each terrapin was determined based on tail length. A shell tag with a unique identification number was attached to the terrapin's carapace along with a PIT (passive integrated transponder) tag inserted into the musculature of the left hind limb.

Table 1. Morphological measurements obtained from each captured terrapin.

Straight-line carapace length (SLCL)	Straight-line carapace width (SLCW)
Plastron length (PL)	Shell depth (SD)
Weight	

Two sampled paired t-tests at an α level of 0.05 were used to observe any significant differences between marketable sized crab (> 5") capture, total crab capture, and terrapin capture per week as well as catch per unit effort (C.P.U.E.) for both crab categories for the two types of traps. C.P.U.E. was calculated for each set of traps per week to examine crab capture by trap day.

Results

Terrapin Bycatch

A total of 24 terrapins were captured in the modified crab traps in Cedar Point Marsh during the study period. A significant difference (t = -2.198, p = 0.032) was observed between the weekly terrapin capture for the crab traps, with 2 terrapin captured in crab traps fitted with BRDs and 22 terrapins in traps not fitted with BRDs (Figure 1). Nineteen females and three males with two recaptures were caught. The C.P.U.E. was also significantly different (t = -2.178, p = 0.033) (Figure 2). The average measurements of the captured terrapins are shown in Table 2.

Table 2. Average morphological measurements of terrapin bycatch.

	SLCL (cm)	SLCW (cm)	PL (cm)	SD (cm)	Weight (g)
BRD Traps	13.36	9.65	12.19	5.6	412.25
Non BRD Traps	14.62	11.11	13.18	5.98	542.54

Crab Catch

Overall, non-BRD crab traps caught more marketable sized crabs and total crabs. A significant difference in the average weekly crab capture was observed in both categories between the trap types (marketable-sized crabs: $t = -3.430$, $p = 0.0001$; total crabs: $t = -2.938$, $p = 0.005$). A significant difference was also observed in the C.P.U.E. for marketable-sized crab capture ($t = -2.788$, $p = 0.007$) (Figure 3), but the C.P.U.E. for total crab capture was not significantly different ($t = -1.147$, $p = 0.256$) (Figure 4).

Discussion

Kennelly and Broadhurst (2002) provided a framework to effectively address bycatch issues. These steps include quantifying the amount of bycatch and identifying the species comprising the bycatch, developing or improving gear modifications to reduce bycatch amounts and completing reliable field studies, and finally earning the approval of affected fisheries and concerned interest groups. The issue of crab trap mortality of diamondback terrapins is easily viewed within this framework, and the current study adds to the wealth of information indicating that BRDs can effectively prevent terrapin mortality. The results of the current study indicate that BRDs significantly reduced the capture of terrapins in crab traps. Of the 24 terrapins caught in crab traps in Cedar Point Marsh, only two were captured in traps fitted with BRDs, suggesting an approximate 90% reduction in the capture of terrapins. The two terrapins that entered traps fitted with BRDs must have distorted the BRDs because their shell height (5.5 cm and 5.7 cm) was more than the height of the BRDs (5.08 cm). Thus the use of a more rigid BRD or the smaller version of the BRD could potentially have prevented the capture of these two terrapins. A smaller version of BRD has been tested in previous studies (described below). Regardless, the results of the current study indicate that BRDs represent an effective means of significantly reducing the capture of terrapins in crab traps in an Alabama salt marsh (90% reduction in the current study).

The results of the current study are consistent with those from a variety of previous studies throughout the range of terrapins. BRDs have been shown to significantly inhibit terrapin trap entry in every study that has examined their efficacy. Wood (1997) and Roosenburg and Green (2000) both compared a variety of BRD sizes in crab traps. In New Jersey, no terrapins were caught in traps fitted with 4x8 cm and 4.5x10 cm BRDs, as opposed to 46 in non-BRD traps. Eleven terrapins were captured when testing 5x10 cm BRDs versus 84 in traps without BRDs (Wood, 1997). In Maryland, 56 terrapins were caught in traps fitted with 5x10 cm BRDs, 19 with 4.5x12 cm, and none with 4x10 cm BRDs. In non-BRD traps, 105 terrapins were captured (Roosenburg and Green, 2000). Butler and Heinrich (2007), Morris *et al.* (2010), and Rook *et al.* (2010) utilized the 4.5x12 cm BRD in their studies and observed the high efficiency of BRDs at excluding terrapins from entering the crab traps. Morris *et al.* (2010) noted that 92% of all captured females and 70% of all captured males caught in non-BRD traps in their study would have been prevented from entering traps with the 4.5x12 cm BRDs.

While BRDs represent an effective means of decreasing the capture of terrapins, their benefit must be weighed relative to the potential negative impacts on crab capture rates. In the current study, the weekly capture of marketable-sized crabs and total crabs, as well as the C.P.U.E. of marketable sized crabs, was significantly lower in traps fitted with BRDs than in non-BRD traps. Crab traps fitted with BRDs showed an approximate 27% reduction in the capture of blue crabs. Interestingly, this finding contradicted the findings of a variety of larger scale studies. The sample size of captured crabs in the current study (543) paled in comparison to other studies that have examined this topic. Cole and Helser (2001) captured 3,522 crabs of marketable size in their BRD comparison study. Wood (1997) found that traps fitted with 5x10 cm caught 12,912 marketable-sized crabs, compared to 10,873 crabs in non-BRD traps. Roosenburg and Green (2000), in a combination of standard and taller crab traps used in this study, caught 762 in non-BRD traps, 782 in traps fitted with 4.5x12 cm BRDs, and 710 in traps fitted with 5x10 cm BRDs. Butler and Heinrich (2007) observed 1,398 marketable-sized crabs in traps fitted with 4.5x12 cm BRD's and 1,355 in non-BRD traps. More recently an ongoing long-term study by the Gulf Coast Research Laboratory (GCRL, Ocean Springs, MS) and the Mississippi Department of Marine Resources have utilized several commercial crab fishermen for evaluating the effect of BRDs on crab capture. With over two years of data collected and a total sample size of approximately 140,000 crabs, their results imply the crab capture is not affected in traps fitted with BRDs (D. Graham, GCRL).

Thus, in contrast to the findings from the current study, a variety of larger scale studies indicated that BRDs do not hamper crab capture; in fact it has been suggested that in some situations, the use of BRDs may actually increase crab yield. Roosenburg and Green (2000), Cole and Helser (2001), Butler and Heinrich (2007), Morris *et al.* (2010), and Rook *et al.* (2010) observed no significant differences in crab C.P.U.E.s between trap types using certain sized BRDs. In Maryland, the C.P.U.E. of standard traps with 4.5x12 cm BRDs was 2.69 crabs/trap/day versus 2.55 crabs/trap/day in non-BRD traps (Roosenburg and Green, 2000). Guillory and Prejean (1998) and Wood (1997) found higher crab captures in traps fitted with BRDs. Rook (2010) also detected larger crab biomass and size in modified traps, and Roosenburg and Green (2000) noted the largest crabs were in the traps with 4.5x12cm BRDs. It has been speculated that the presence of the BRD may provide increased rigidity to the funnel opening, thus preventing manipulation by the crabs and eventual egress (Guillory and Prejean, 1998; Roosenburg and Green, 2000). Moreover, Morris (2010) observed that non-BRD traps that contained live terrapins had lower crab capture than other non-BRD traps that did not have captured terrapins.

The lower crab capture rate in BRD-fitted traps in the current study could be due to a variety of factors. The side-by-side arrangement of BRD versus non-BRD crab traps could have influenced the results. Roosenburg (2004) noted that a side-by-side comparison could result in affecting crab behavior by permitting them to enter the more easily accessed non-BRD trap as opposed to a trap fitted with BRDs. The study location could have also affected the results. As exemplified by the low number of crabs captured in the current study, this is not an optimal crabbing area. Crab fishermen rarely utilize this channel, even though traps are typically very numerous in the bay surrounding the marsh. Thus, factors such as the low abundance of crabs together with side-by-side arrangement of BRD versus non-BRD traps could help explain why the current results were in direct contrast to a variety of previous and ongoing studies from other areas.

If BRDs are adopted as a management strategy, the size of the BRD can affect the capture rate of terrapins as well as crabs. Roosenburg and Green (2000) concluded that in Maryland the 4.5x12 cm BRD performed the best at preventing terrapin while still allowing sufficient crab capture. This size BRD was also used in the studies completed by Butler and Heinrich (2007), Morris (2010), and Rook (2010). However, Wood (1997) found that the 5x10 cm BRD performed best in New Jersey, and this BRD was also utilized in the Guillory and Prejean (1998) study. For recreational crabbing, tall crab traps developed by Roosenburg *et al.* (1997) with an even smaller BRD (even though its crab C.P.U.E. is lower) might be most appropriate. Roosenburg and Green (2000) noted that comparison studies in specific regions would need to be performed to see which BRD would be most successful there. Based on the shell height data collected for all captured terrapins in the larger population study (Chapter 2), 79% of adult male terrapins and 6% of adult female terrapins were small enough to enter traps fitted with the size of BRDs (2”x6”) utilized in this study, so a BRD with a smaller height would be more successful in preventing terrapin bycatch.

In the current study, a female-biased sex ratio was observed in terrapins captured in the crab traps (3 males:19 females). The finding of a female bias is consistent with results of our ongoing mark recapture study in Cedar Point Marsh (Chapter 2). This could be due to a variety of factors, including the effect of crab traps on terrapin population demographics. Crab trap-induced mortality has been shown to significantly alter population structure, resulting in a female biased sex ratio (Roosenburg *et al.*, 1997; Dorcas *et al.,* 2007). Additionally, Roosenburg *et al.* (1997) suggested the loss of juvenile females could be more detrimental to the survival status of the population because of the loss of their reproductive potential. The low terrapin C.P.U.E. in non BRD traps in the current study (0.012 terrapins/trap/day) compared to other areas (0.17 terrapins/trap/day in Maryland (Roosenburg *et al.*, 1997) and 0.20 terrapins/trap/day in Virginia (Rook *et al.*, 2010)) suggested a small population size, which was likely the result of the historic impact of the crab fishery (Wood, 1997) on the terrapin population in Alabama.

In addition to the implementation of BRDs, another effective management strategy would be a consistent derelict crab trap removal program. In recent years, several Gulf Coast states have initiated such efforts with great success. The Derelict Trap Task Force under the Gulf States Marine Fisheries Commission authored a set of guidelines for the development of derelict trap removal program (Perry *et al.*, 2003). Also, closing Cedar Point Marsh to crab traps would be a worthwhile measure, especially if this habitat is not conducive to successful crabbing as our results indicated. Lewison *et al.* (2003) observed a decline in Kemp’s ridley sea turtle strandings in areas that experienced seasonal closures to shrimp trawling.

The issue of terrapin bycatch has met the first four steps of the bycatch framework described by Kennelly and Broadhurst (2002). Terrapin bycatch has been clearly defined and quantified as a major problem associated with the crab fishery. BRDs were developed and tested by reliable field studies to be successful at preventing terrapin bycatch. The last step in the framework is to involve the affected fishing industry (crab fishery) in the implementation process to gain their approval, a notion that has been proposed by several studies (Melvin *et al.*, 1999; Hall *et al.*, 2000; Gilman *et al.*, 2005; Hall and Mainprize, 2005). As mentioned earlier, the researchers performing the on-going study in Mississippi examining the efficacy of BRDs are partnering with commercial crab fishermen to collect data from their traps (D. Graham, GCRL). This type of synergistic relationship between science and industry will benefit the ultimate goal of terrapin conservation.

Conclusions and Recommendations

Bycatch is a serious conservation concern with biodiversity and ecological implications. Diamondback terrapin mortality as bycatch in crab traps is affecting most populations throughout their range and is threatening their future survival status (Seigel and Gibbons, 1995; Butler *et al.*, 2006). The current study, along with a variety of past studies, has shown that the use of bycatch reduction devices (BRDs) can significantly decrease terrapin capture in crab traps (a 90% reduction in terrapin capture was recorded in the current study). A 27% reduction in crab capture was also recorded in the current study in the BRD-equipped crab traps. However, this finding is not consistent with several larger scale studies which have not detected significant changes in crab capture in BRD-equipped crab traps. The results of the current study also indicated a relatively low C.P.U.E for terrapins, suggesting relatively low numbers of terrapins in the Cedar Point Marsh study area. Although the number of terrapins appears to be relatively low, Cedar Point Marsh has been identified as the location with the largest aggregation of terrapins identified to-date in Alabama (Chapter 2). This suggested that there is a distinct need for a management program for terrapins. The results of the current study indicated that the implementation of BRDs on crab traps could represent an effective component of a management plan for initiating the recovery of the diamondback terrapin in Alabama.

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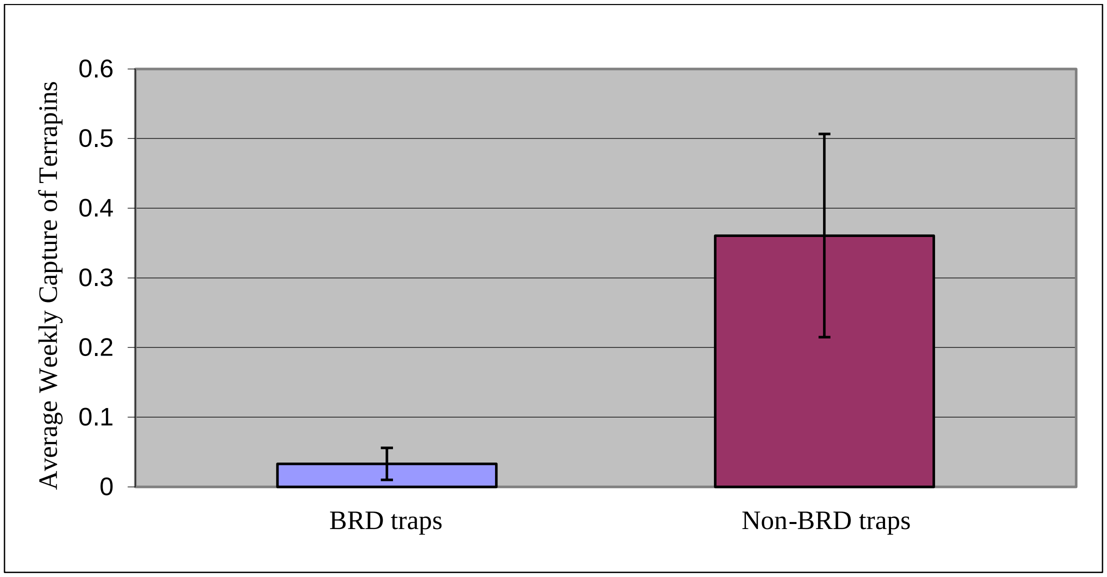

Figure 1. Average weekly capture of terrapin bycatch by traps fitted with BRDs and non-BRD traps. A significant difference between the traps' means was observed.

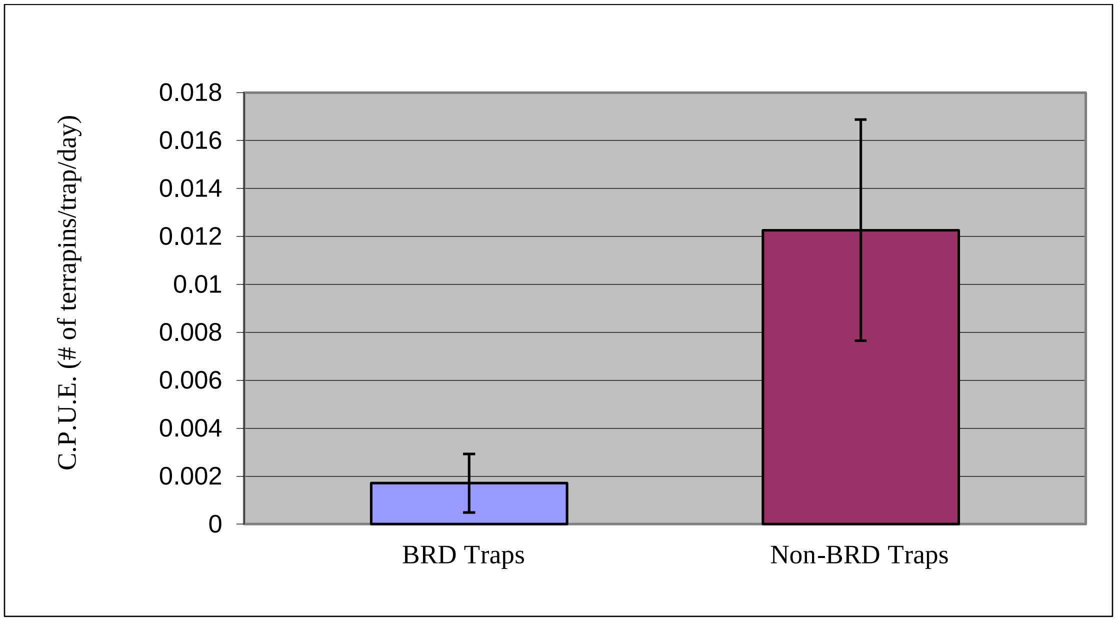

Figure 2. Catch per unit effort (C.P.U.E.) of terrapins caught in traps fitted with BRDs and non-BRD traps. A significant difference was observed.

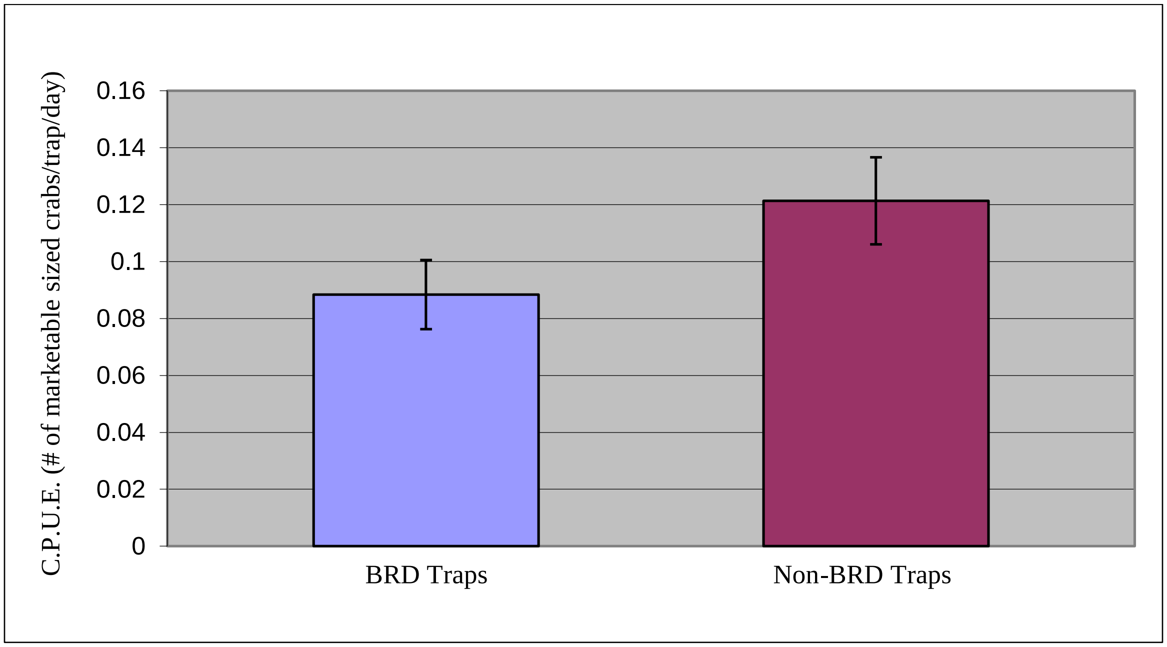

Figure 3. Catch per unit effort (C.P.U.E.) of marketable-sized crabs caught with traps fitted with BRDs and non-BRD traps. Traps without BRDs displayed significantly higher C.P.U.E.

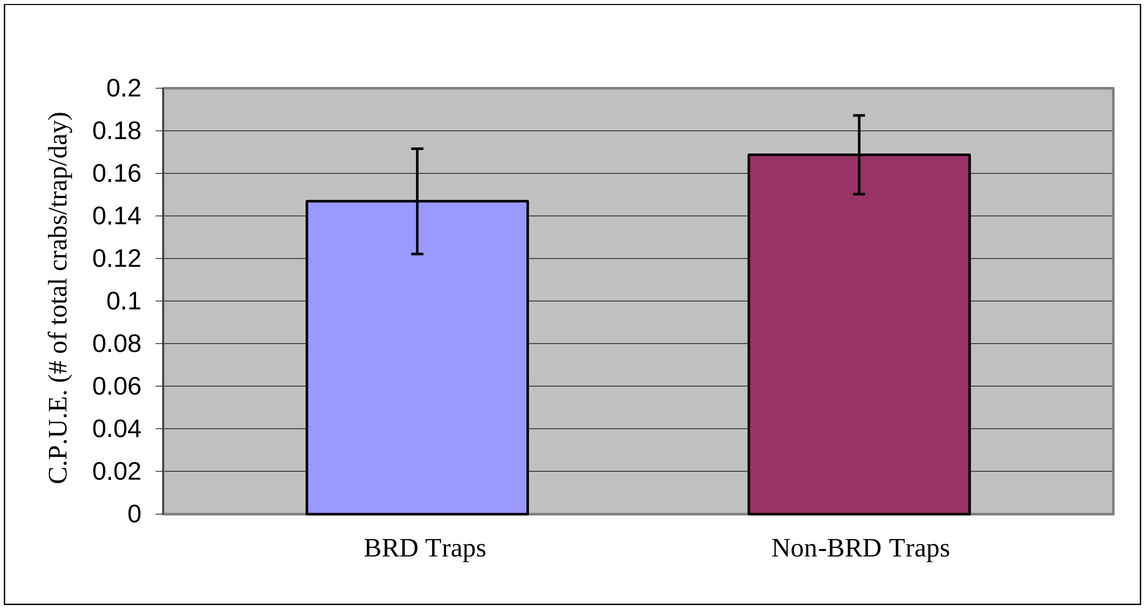

Figure 4. Catch per unit effort (C.P.U.E.) for total crabs caught with traps fitted with BRDs and non-BRD traps. No significant difference was observed for total crab capture C.P.U.E. between trap types.

CHAPTER 5

EVALUATION OF MATERNAL INVESTIMENT, EGG SIZE AND HATCHLING FITNESS IN THE DIAMONDBACK TERRAPIN (*MALACLEMYS TERRAPIN*)

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In preparation for Journal of Herpetology

Format adapted for dissertation

Abstract

The optimal egg size theory (Smith and Fretwell, 1974) attempted to explain how limited resources are allocated by females to reproduction. It states that, in a population, more variation would be observed in clutch sizes rather than egg and hatchling sizes. Larger females would produce larger clutches than smaller females. The current study investigated the effects of female size and age on egg size and hatchling fitness in a depleted population of Mississippi diamondback terrapins, *Malaclemys terrapin pileata*, in Alabama. In contrast to the predictions of the optimal egg size theory, larger females produced larger eggs and hatchlings but not larger clutches. This suggested that larger hatchlings could experience a survival advantage even though none of the potential fitness indicators that were tested significantly correlated with larger hatchling size. This topic also has potential conservation implications if the larger females that are producing the more successful offspring are being removed from the population. Results from the comparison of the egg and hatchling sizes from the Alabama population, which is not experiencing road mortality, to a population from Georgia that is experiencing high road mortality indicated larger eggs and heavier hatchlings were produced by Alabama terrapins.

Introduction

Smith and Fretwell (1974) proposed the optimal egg size (o.e.s.) theory to explain how females allocate energy to reproduction. Resources are limited, and natural selection theory will favor females that best apportion limited energy reserves between growth, maintenance, and propagule production. The optimal egg size theory predicts that females in a population will produce offspring at the minimum size necessary for hatching; thus, while egg and hatchling size should be consistent, clutch sizes should vary due to differing total female resources (Figure 1). The authors noted that their theory would fit best to animals that produced large clutches and offered no classic parental care (Smith and Fretwell, 1974), such as is the case in turtles.

Numerous studies have examined the topic of female allocation strategies in turtles, but the results and conclusions of these studies do not necessarily fit an orderly pattern. In a population of chicken turtles (*Deiroochelys reticularia*) from South Carolina, Congdon *et al.* (1983) observed a positive relationship between adult female size and egg size and only weakly with clutch size. The negative relationship between clutch and egg size as predicted by the o.e.s. theory was not detected (Congdon *et al.*, 1983). A positive correlation between clutch size and female body size was generally observed by Congdon and Gibbons (1985) in a study examining these parameters in 12 species of turtles, although it was not detected in box turtles (*Terrapene carolina*) and chicken turtles. Egg size increased with female body size in 3 of the 12 examined species, chicken turtles, mud turtles (*Kinosternon subrubrum*), and red-eared sliders (*Trachemys scripta*). As in the previous study, no overall negative relationship between egg size and clutch size was observed in any of the turtles (Congdon and Gibbons, 1985).

Some supporting evidence for the o.e.s. theory was observed in a population of green sea turtles (*Chelonia mydas*) from Tortuguero, Costa Rica (Bjorndal and Carr, 1989). Clutch sizes within the population varied more than egg size. Female carapace length accounted for more variation in clutch sizes than egg sizes, and a non-significant relationship existed between egg size and female body mass. The average egg size was

44.4 mm, which agreed with a previously proposed optimal egg size of 45 mm for green turtles (Bjorndal and Carr, 1989).

To help explain the presence of the strong correlation between adult female size and egg size in some turtle species, Congdon and Gibbons (1987) argued that pelvic girdle size constrains egg size in smaller-sized turtles so that the optimal egg size in smaller species cannot be obtained. Egg size increased with pelvic girdle width in the two smaller examined species, chicken turtles and painted turtles (*Chrysemys picta*), but it did not in the larger slider turtle (Congdon and Gibbons, 1987). The authors suggested that the size of the pelvic girdle is being influenced by two contrasting selection pressures, locomotor performance and larger egg size (Congdon and Gibbons, 1987). However, the selection for a larger egg size has been strong enough to cause pelvic girdles of females in some turtle species to be significantly larger than in conspecific males (Long and Rose, 1989). The pelvic aperture of mud turtles has also been observed to constrain egg size (Wilkinson and Gibbons, 2005). Although pelvic girdle size was implicated in these studies, the distance between the carapace and the plastron, the caudal gap, was also suggested as representing another morphological constraint on egg size (Clark *et al.*, 2001).

An analysis of a long- term data set from a population of painted turtles somewhat supported the argument for size -specific constraints on optimal egg size (Rollinson and Brooks, 2008). Within the population, egg width was limited in smaller individuals, and mean egg mass increased with female body size. Clutch size varied more over the years than egg size, although considerable variation within egg size was observed, and the tradeoff between egg size and clutch size was similar between small and large females

(Rollison and Brooks, 2008). The authors noted that environmental variation could impact optimal egg size, thus leading to the observed egg size variation. Also, other factors, such as nest site selection and hatchling migration, could play a role in maternal investment and resulting egg size, so focusing only on morphological constraints could be too narrow of scope (Rollison and Brooks, 2008).

The influence of maternal size on egg size suggests that the optimal egg size that maximizes female reproductive fitness may be different than the size that maximizes offspring fitness. Janzen and Warner (2009) examined the optimal egg size dichotomy in 3 species of turtles: smooth softshell turtles (*Apalone mutica*), common snapping turtles (*Chelydra serpentine*) and painted turtles. For smooth softshell turtles and painted turtles, the optimal egg size that maximized female fitness was most similar to the actual population mean, indicating that the conflict between maximizing maternal and offspring fitness was resolved in favor of females. For common snapping turtles, no optimal egg size for female reproductive fitness could be determined because of strong directional selection on egg size (Janzen and Warner, 2009).

Utilizing common snapping turtles, Congdon *et al.* (1999) tested the “bigger is better” hypothesis that states that larger hatchlings experience greater survival success. Long- term and short- term survival rates from several experiments were compared with hatchling body parameters, and selection for a specific body size was not particularly evident. When selection was observed, it was concluded to be stabilizing in nature because the intermediate forms were favored over both extremes (Congdon *et al.*, 1999).

In the present study, effect of female size and age on egg and hatchling size, as well as hatchling fitness, was examined in a population of Mississippi diamondback terrapins

(*Malaclemys terrapin pileata*). The data were analyzed in the context of optimal egg size theory, because this is the classical theory to explain female allocation strategies, but other theories will also be explored. The following questions were tested: 1) Are female size and female age correlated? 2) Do larger females produce larger eggs and larger hatchlings? 3) Are larger hatchlings more fit? Righting response times (Steyermark and Spotila, 2001; Delmas *et al.*, 2007) and hatchling growth were examined as potential fitness indicators.

This topic also has potential conservation implications. The Alabama population of terrapins has experienced a historic population decline and has not recovered. The largest aggregation in the state, located in Cedar Point Marsh, has been reduced from approximately 25,000 individuals to approximately 435 individuals (Chapter 2). The main threats to the survival of this population are crab trap mortality and nest predation. Road mortality is also a major threat to many terrapin populations (Wood and Herlands, 1997; Butler *et al.*, 2006), including the population at Tybee Island, GA (J. Gray, personal communication), but it is rarely observed in Alabama. Adult females are struck by automobiles as they leave the marsh to migrate to suitable nesting habitats. If older females have a greater chance of being killed, simply by making more migrations over their life time, than younger females and older females produce more successful hatchlings, then road mortality could not only remove reproductively active individuals from the population, but also remove individuals that are contributing most to the future survival of the population. Accordingly, egg and hatchling sizes from Cedar Point Marsh, AL, and Tybee Island, GA, populations were compared to examine if road mortality is potentially affecting the size of the eggs and hatchlings produced by nesting females.

Methods

To address the population decline in Alabama, a head-start program was initiated at University of Alabama at Birmingham. Eggs were obtained starting in 2008 from adult females who were captured in pitfall traps on the nesting beach surrounding Cedar Point Marsh. Captured females were checked to determine whether they were gravid, and if so, they were injected with a safe level of oxytocin (Ewert and Legler, 1978). Doses of 5 I.U.s were injected every few hours, but no more than 25 I.U.s were given. A suite of morphological measurements was taken from each female (Chapter 2). Weight (Fsize), age (Fage), and plastron length (FPL) of females were utilized for this study. Egg and hatchling data were collected in 2009 and 2010. Length (EL), width (EW), and weight (Eweight) were measured for each egg, and they were placed in incubators in the U.A.B. animal facility. Because terrapins display temperature dependent sexual determination (Roosenburg and Kelley, 1996), eggs were incubated at either 26°C (male-producing temperature) or 31°C (female-producing temperature). However, because the overall purpose of the head-start program was to provide a population boost, the majority of eggs were placed at 31°C to produce more females. In 2010, three clutches were separated between temperatures to investigate temperature effects within clutches.

After hatching, initial carapace length (HCL), carapace width (HCW), plastron length (HPL), and weight (Hweight) were measured on each hatchling. Initial plastron length was only measured for hatchlings in 2010. Hatchlings were separated by clutch, and these measurements were taken each week. Because clutches were kept separate, but not the hatchling individuals, clutch averages of the various measurements were included in the analyses. Hatchlings were fed to satiation daily. To measure righting response time, hatchlings were placed into a plastic container on their carapace, and the time it took for the hatchlings to right themselves was measured. Two trials were completed for each hatchling. In 2009, righting response times were measured within two months of the hatching of the last clutch. In 2010, righting response times were measured within two weeks of each clutch hatching. Growth of the hatchlings was analyzed via the slopes of the CL, CW, PL, and Weight growth curves after week 19 because variation was basically undetectable before this time.

To test for data normality, Shapiro-Wilk tests were run on each dataset. If the data were normally distributed, Pearson's correlation coefficient was utilized. If the data were not normally distributed, Spearman's correlation coefficient for non-parametric data was utilized. Correlations of the weight, age, and plastron length of females with each hatchling measurement were run in SAS$^{©}$ (Cary, NC) v.8 at an α level of 0.05.

For the Alabama-Georgia comparison, clutch averages for egg and initial hatchling size parameters were compared. The average female weight for the *Malaclemys terrapin centrata* subspecies, 667 g, as reported by Gibbons *et al.* (2001), was used to standardize the Alabama measurements to account for subspecies differences. The percentage between the average adult female weight in the Cedar Point Marsh population (Chapter 2), 781.82 g, and the average adult female weight for the females in this study, 1040.09 g, was used to calculate the hypothetical average adult female weight for the Tybee Island population, 886.97 g. The values for the Cedar Point Marsh clutch averages were adjusted to fit the slope of the line produced by adjusting the adult female weights of the Cedar Point Marsh population to the Tybee Island average. The adjusted clutch values were then compared to Tybee Island clutch averages to test for significant differences utilizing 2-tailed t-tests at an α 0.05. These analyses were run in Microsoft© Excel 2007.

Results

For this study, 22 clutches were obtained from 21 females for a total of 165 eggs (average clutch size of 7.5 eggs) (Table 1). Based on the results of the Shapiro-Wilk tests, adult female weight and plastron length were normally distributed. For the eggs and hatchlings, egg length, egg width, egg weight, clutch size, initial carapace length, initial carapace width, and initial hatchling mass were normally distributed. Slopes of growth curves for each measurement and righting response times were not normally distributed.

Table 1. Averages of various egg and hatchling parameters obtained from 22 clutches.

	Clutch Size	EL (mm)	EW (mm)	EWeight (g)	CL (mm)	CW (mm)	PL (mm)	HWeight (g)
Average	7.5	36.871	23.699	11.944	31.414	25.038	26.796	8.672
Std. Error	0.445	0.399	0.329	0.407	0.269	0.262	0.337	0.220

Both adult female age and plastron length significantly correlated with adult female weight (Table 2). As females age, the annuli on both carapace and plastron scutes become worn making it more difficult to estimate age. So, age estimates for the older females were most likely underestimates, furthering supporting the significant correlation.

Table 2. Results from Pearson's correlations of adult female weight with both adult female age and plastron length.

Correlation	R	p
Fweight_Fage	0.675	0.0006
Fweight_FPL	0.847	<0.001

None of the correlations between clutch size and any of the adult female parameters were significant (Figure 2), so no appreciable clutch size differences were observed between larger and smaller adult females (Table 3). Egg length was also not significantly correlated with any of the adult female parameters, but the other two egg measurements, egg width and egg weight (Figure 3), were significantly correlated. Larger females produced larger eggs.

Table 3. Results from Pearson's correlations between the adult female parameters and egg parameters. R values are shown and p values are in parentheses.

Correlations	Fweight	Fage	FPL
Clutch Size	-0.004 (0.984)	0.037 (0.869)	0.113 (0.618)
EL	0.209 (0.350)	0.390 (.0723)	0.111 (0.622)
EW	0.733 (0.001)	0.583 (0.004)	0.576 (0.005)
Eweight	0.590 (0.004)	0.633 (0.002)	0.415 (0.055)

Larger females also produced larger hatchlings (Table 4). Adult female size, age, and plastron length were significantly correlated with initial hatchling carapace length, width, and weight (Figure 4). They were not significantly correlated with initial hatchling plastron length. However, the adult female parameters were not positively correlated with any of the hypothesized fitness indicators (Table 5). Adult female weight, age, and plastron length were not significantly correlated with the growth curve slopes of the hatchling parameters. The righting response times were also not significantly correlated with any of the adult female parameters, so larger hatchlings did not have faster righting response times.

Table 4. Results from Pearson's correlations between adult female parameters and initial hatchling parameters. R values are shown and p values are in parentheses.

Correlations	Fweight	Fage	FPL
HCL	0.611 (0.001)	0.638 (0.006)	0.419 (0.037)
HCW	0.567 (0.003)	0.590 (0.002)	0.401 (0.047)
HPL	0.458 (0.012)	0.651 (0.016)	0.208 (0.496)
Hweight	0.574 (0.003)	0.636 (0.001)	0.421 (0.036)

Table 5. Results from Spearman's correlations between adult female parameters and potential hatchling fitness indicators. R values are shown and p values are in parentheses.

Correlations	Fweight	Fage	FPL
CL	0.102 (0.628)	0.077 (0.713)	0.053 (0.800)
CW	-0.080 (0.702)	-0.020 (0.923)	-0.146 (0.485)
PL	0.076 (0.717)	0.088 (0.676)	-0.003 (0.988)
Weight	0.052 (0.804)	0.063 (0.765)	-0.020 (0.926)
Righting	-0.171 (0.413)	0.002 (0.992)	-0.099 (.638)

Incubation temperatures did not have an effect on either initial hatchling size or hatchling growth (Table 6). None of the correlations between incubation temperature and initial hatchling size and hatchling growth curve slopes were significant.

Significant differences were observed between the Cedar Point Marsh and Tybee island terrapins (Table 7). Cedar Point Marsh females produced larger eggs, significant differences were observed for egg length, width, and weight. Cedar Point Marsh hatchlings were also significantly larger in initial weight; however, Tybee Island hatchlings displayed significantly higher carapace length and width and plastron length.

Table 6. Results from Pearson's and Spearman's correlations between incubation temperature and initial hatchling size and hatchling growth parameters. Pearson's correlations were run on initial hatchling sizes, and Spearman's correlations were run on hatchling growth parameters. R and p values are shown.

Correlations	R	p
HCL	-0.291	0.578
HCW	-0.802	0.055
HPL	-0.141	0.79
Hweight	0.433	0.392
CL	-0.098	0.854
CW	0.488	0.326
PL	0.293	0.573
Weight	0.293	0.573

Table 7. Results of two-tailed t tests of comparisons of egg and initial hatchling parameters between Cedar Point Marsh, AL, and Tybee Island, GA. Number of clutches included in each analyses are shown.

Parameters	n	C.P.M. (AL)	T.I. (GA)	t	p
EL	49	36.55	33.962	4.62	<0.0001
EW	49	22.772	20.354	7.163	<0.0001
Eweight	49	11.023	9.78	2.665	0.011
HCL	36	30.834	31.873	-2.139	0.046
HCW	36	24.504	26.759	-5.439	<0.0001
HPL	24	26.529	28.458	-3.468	0.0022
Hweight	36	8.601	7.909	2.105	0.043

Discussion

Overall, egg and hatchling size increased with female size and age in the diamondback terrapin population in Alabama (Figure 5), suggesting support for the "bigger is better" hypothesis. The results did not support predictions of the optimal egg size theory because egg size varied but not clutch size. However, none of the potential fitness indicators significantly correlated with female size or age. Larger hatchlings did not show faster growth or quicker righting response times when compared to smaller hatchlings. This does not indicate that larger hatchlings did not possess a fitness advantage; other fitness indicators need to be examined to better understand if any advantage exists. A significant correlation was also not observed between incubation temperature and initial hatchling size or hatchling growth.

Reproductive population parameters and effects of female allocation have been reported in various diamondback terrapin populations (Montevecchi and Burger, 1975; Seigel, 1980; Roosenburg, 1996; Roosenburg and Kelley, 1996; Roosenburg and Dunham, 1997; Roosenburg and Dennis, 2005; Allman, 2006). Egg and hatchling averages from the Alabama population were similar to those observed by Seigel (1980) at a *Malaclemys terrapin tequesta* population from Merritt Island National Wildlife Refuge and larger than those observed by Montevecchi and Burger (1975) at a *Malaclemys terrapin terrapin* population from southern New Jersey. The average clutch size from southern New Jersey (9.7 eggs/clutch) were greater than both the Florida population (6.7 eggs/clutch) and the Alabama population (7.5 eggs/clutch). This latitudinal trend was also observed by Allman (2006). In contrast to the present study, neither the Seigel (1980) nor the Montevecchi and Burger (1975) studies detected a significant correlation between egg size and female size.

In agreement with the present study's data, larger terrapin eggs have been shown to result in larger terrapin hatchlings (Roosenburg and Kelley, 1996). But, whereas differences in hatchling growth were not detected in this study, egg size and incubation temperatures affected growth in their study. Larger eggs incubated at the female-producing temperature, (32°C), grew a higher rate than smaller eggs incubated at the same temperature, but this was not observed at the male-producing temperature (26°C). The effect of egg size at 32°C could aid larger females in reaching sexual maturity three years quicker than smaller females (Roosenburg and Kelley, 1996). Additional studies have investigated potential temperature effects on hatchlings of other turtle species. Páez *et al.* (2009) observed incubation temperature influences on growth rates and hatchling size but not hatchling weight of Magdalena River turtles (*Podocnemis lewyana*). Interestingly, hatchlings from cooler incubation temperatures grew faster than hatchlings from warmer temperatures (Páez *et al.*, 2009). Booth *et al.* (2004) detected differential effects of incubation temperature on two species of turtles, green turtles and Brisbane River turtles (*Emydura signata*). Brisbane River turtle hatchlings incubated at warmer temperatures grew faster than hatchlings from cooler temperatures. In green turtles, hatchlings incubated at 26°C were larger in size than hatchlings incubated at warmer temperatures; however, these larger hatchlings displayed poorer swimming (Booth *et al.*, 2004). Although incubation temperatures affected a hatchling fitness indicator in the Booth *et al.* (2004) study, a significant correlation between incubation temperatures and righting response was not observed in this study.

Maternal allocation of resources to eggs has been delineated into two basic categories: parental investment into embryogenesis (PIE) and parental investment into care (PIC) (Congdon 1989). Roosenburg and Dennis (2005) observed larger terrapin eggs contained more PIC components, such as non-polar lipids, protein, structural lipids, and water, than smaller eggs. Clutch effects were also observed in relative amounts of non-polar lipids and water content indicating females have the ability to vary relative allocations of these egg components. The increased PIC in eggs could enhance the ability of the resulting hatchlings to survive (Roosenburg and Dennis, 2005). Roosenburg and Dennis (2005) postulated that the latitudinal variation in terrapin eggs could be explained by females in southern populations allocating more components to their eggs. Allman (2006) found this to be true; eggs from South Carolina contained higher levels of non-polar lipids than eggs laid in Maryland and Rhode Island. These increased egg components were needed because hatchlings from South Carolina displayed a higher maintenance energetic demand (Allman, 2006). Nagle *et al.* (2003) also observed high levels of egg nutrients donated to turtle hatchlings (smooth softshell turtles, *Apalone mutica*) that experience variable environments that require high energy reserves.

Allocation of energy to eggs is not the only avenue of female care that has been studied in terrapins. Roosenburg (1996) noted how nest site selection by females can affect offspring phenotype. Larger eggs were found to be laid in nests in the open, which would experience greater incubation temperatures. This would result in more females being produced, and because it had been shown that larger egg sizes would promote quicker growth rates in females but not males (Roosenburg and Kelley, 1996), nest site selection represents another source of female care (Roosenburg, 1996).

Roosenburg and Dunham (1997) examined female allocation strategies in diamondback terrapins within the context of the optimal egg size theory as well as another allocation theory, developmental plasticity, which states that females can adjust allocation between clutches within a nesting season to account for environmental variation. In contrast to the current study's results, female size did not contribute to egg mass variation, and various aspects of their data seemingly supported both theories. The coefficients of variation for clutch sizes were greater than egg size coefficients of variation, which would be predicted by optimal egg size theory, but within-female variation for both egg and clutch size was observed, supporting the theory for developmental plasticity (Roosenburg and Dunham, 1997). The authors stated that the process to determine clutch and egg sizes may be decoupled. Clutch size may be determined when the maturing follicles reach the energetic threshold to ensure successful embryogenesis, and then egg size could be based on the availability of environmental resources (Roosenburg and Dunham, 1997).

Wilkinson and Gibbons (2005) also examined these two theories with three freshwater turtles: mud turtles, common musk turtles (*Sternotherus odoratus*), and Florida cooter turtles (*Pseudemys floridana*). Like the Roosenburg and Dunham (1997) study, different data lent credence to both theories. Low within-clutch egg variation in all three species was observed, consistent with the optimal egg size theory. However, consistent with the developmental plasticity theory, interannual variation in egg and/or clutch size was observed in all three species. Additionally, female body size and age both contributed to egg size variability (Wilkinson and Gibbons, 2005).

In this study, female size and age were positively correlated, so their relative effects on egg and initial hatchling size could not be differentiated. Age has been implicated in other studies as having a significant effect on resource allocation. Harms *et al.* (2005) tested the relative reproductive rate hypothesis, which contends that allocation to reproduction will increase with age in long-lived organisms, by examining egg size and components in painted turtles. High nesting experienced (HNE) females produced larger eggs with greater yolk mass, lipids, and proteins than low nesting experienced (LNE) females, but the lipid and protein differences disappeared once egg size and yolk mass was taken into account. The results were consistent with predictions of the relative reproductive rate hypothesis (Harms *et al.*, 2005), which was also supported by Congdon *et al.* (2003). Larger offspring produced by older painted turtles have been shown to experience greater survival on migrations to water habitats than hatchlings produced by younger females (Paitz *et al.*, 2007).

Bowden *et al.* (2004) explored the potential effects of testosterone levels in younger painted turtles on constraining egg sizes. HNE females produced significantly larger eggs than similarly-sized LNE females, but yolks from LNE female eggs contained significantly more testosterone. The presence of testosterone in the yolk was probably an indirect result of high amounts of the hormone circulating in the LNE females, because more resources were being devoted to growth rather than reproduction, as in the HNE females. Testosterone constrained egg size by inhibiting vitellogenic activity. So, physiological constraints may have more bearing on egg sizes in some turtle species, rather than physical constraints such as pelvic girdle size (Bowden *et al.*, 2004).

In this study, no significant correlation was observed between any of the female parameters and hatchling righting response times as well as between incubation temperature and hatchling righting response times. Freedberg *et al.* (2004) detected an influence of incubation temperature on righting response times in red-eared sliders and Ouachita map turtles (*Graptemys ouachitensis*), and in the case of Ouachita map turtles, those effects lasted for at least one year, with hatchlings incubated at higher temperatures displaying faster times. Steyermark and Spotila (2001), however, did not observe incubation temperature effects, or egg mass effects on righting response times in common snapping turtle hatchlings. Delmas *et al.* (2007) concluded that incubation regimes of constant versus fluctuating temperatures can influence hatchling righting response times.

Righting response times appear to be an appropriate fitness measure to examine, given its potentially crucial role in determining survival at such a vulnerable life stage. However, the methods and/or timeframe incorporated in the current study may not have been rigorous enough to detect any female or egg influences.

Another potential fitness indicator that has been studied in turtle hatchlings is predation avoidance. Tucker (2000) examined recapture rates of red-eared slider and painted turtle hatchlings that were released near aquatic habitats. More of the smaller painted turtles were recaptured than the larger sliders, but larger conspecifics of both species were more likely to be recaptured. Even though the painted turtle hatchlings reached the destination faster than the slider hatchlings, the author argued that this was due to the different migration strategies employed by the two turtles (Tucker, 2000). Smaller red-eared slider hatchlings also experienced greater differential mortality in the study performed by Janzen *et al.* (2000a). This differential mortality was due to high amounts of avian predation (Janzen *et al.*, 2000b), which was found to indirectly promote greater hatchling size, because larger hatchlings undergo less exposure to predation because of their enhanced locomotor performance (Janzen *et al.*, 2007). Investigating this potential fitness indicator with this study's terrapin hatchlings was not a viable option, given that the ultimate goal of the head-start program was to provide a boost to the population size.

Conservation Implications

Based on the Alabama-Georgia comparison, diamondback terrapin females from Cedar Point Marsh, AL, produced larger eggs than the Tybee Island, GA, females after accounting for potential subspecies size differences. However, Alabama hatchlings only displayed larger initial weights; the Georgia hatchlings had larger initial carapace lengths and widths and plastron lengths. So, clear conclusions on whether females from a population not experiencing road mortality produce more successful offspring than those from a population threatened by road mortality cannot be drawn. This aspect of the current study does have possible methodology limitations. The Alabama data were modified based on adult female weight estimate for a population of the same *M. t. centrata* subspecies from Kiawah Island, SC. Then the percentage that the mean female weight for the Cedar Point Marsh population differed from the mean weight of Cedar Point Marsh females incorporated in this study was used to calculate the average female weight of the Tybee Island population that Alabama data were adjusted. The data adjustment methodology could have introduced unwarranted bias or employed incorrect female weight means for the Tybee Island population. Also, the analysis was suggested ad hoc of the populations' measurements, and the author was not involved in the Tybee Island measurements. Despite these potential limitations, the purpose of this comparison was to investigate how road mortality indirectly affected egg and hatchling sizes, and the results indicated that possible effects were present and should be further examined.

Avissar (2006) compared the demographic structure of a terrapin population in southern New Jersey to the results of a study done in the same population completed over a decade earlier. A smaller percentage of adult females comprised the total catch in the more recent survey. Also, the average adult CL significantly decreased from approximately 154 mm to approximately 118 mm, and the largest CL observed decreased from 250 mm to 190 mm. The earlier survey caught more individuals belonging to larger size classes than the more recent survey. The author attributed this drastic decline in female numbers and overall size to a high degree of road mortality that is threatening the future viability of the population (Avissar, 2006).

The effect of body condition of females on their reproductive output has been examined in the spotted turtle (Litzgus *et al.*, 2007; Rasmussen and Litzgus, 2010). For these studies, body condition was denoted as the adjusted mass based on the female's CL. Females with better body condition increased their reproductive output through higher clutch sizes over multiple years (Litzgus *et al.*, 2007; Rasmussen and Litzgus, 2010). Also, Litzgus *et al.* (2007) observed that females with better body condition produced larger eggs as well. This should also be examined in diamondback terrapins. Rate of injuries to terrapin adults increased over time in the Kiawah Island, SC, population, and survivorship of injured terrapins was less than uninjured terrapins (Cecala *et al.*, 2008). Although, decreases in female body condition due to injuries were not detected (Cecala *et al.*, 2008), effects of injuries on female reproductive success could exist.

Conclusions

Egg sizes and initial hatchling sizes increased with increased female size and age in the Alabama population of diamondback terrapins. This pattern has been observed in other turtle species (Valenzuela, 2001), but it is in contrast to other terrapin populations previously discussed. The differences observed between this study and other terrapin studies could result from the different environments the populations inhabit. The results from Alabama did not meet the predictions of the optimal egg size theory, and no study discussed did so explicitly. Physical and physiological constraints could be inhibiting smaller females of certain species from producing optimally- sized eggs. Optimality could also shift with environment and age. It is also unclear if the larger hatchlings from

Alabama experienced a fitness advantage over smaller hatchlings, although studies completed with other species concluded that larger hatchlings could enjoy greater predator avoidance. This possibility could not be investigated in the present study. With anthropogenic threats increasing in many turtle populations, the conservation implications of variation in female reproductive output requires immediate attention. The comparison between the Alabama population, which is not experiencing road mortality, and Georgia population, which is, did not elucidate any reproductive output differences. The levels of road mortality in the Tybee Island population may not represent a strong enough selective pressure to elicit a shift yet, but if not properly mitigated, could have a much greater influence on the future viability of the population than previously thought.

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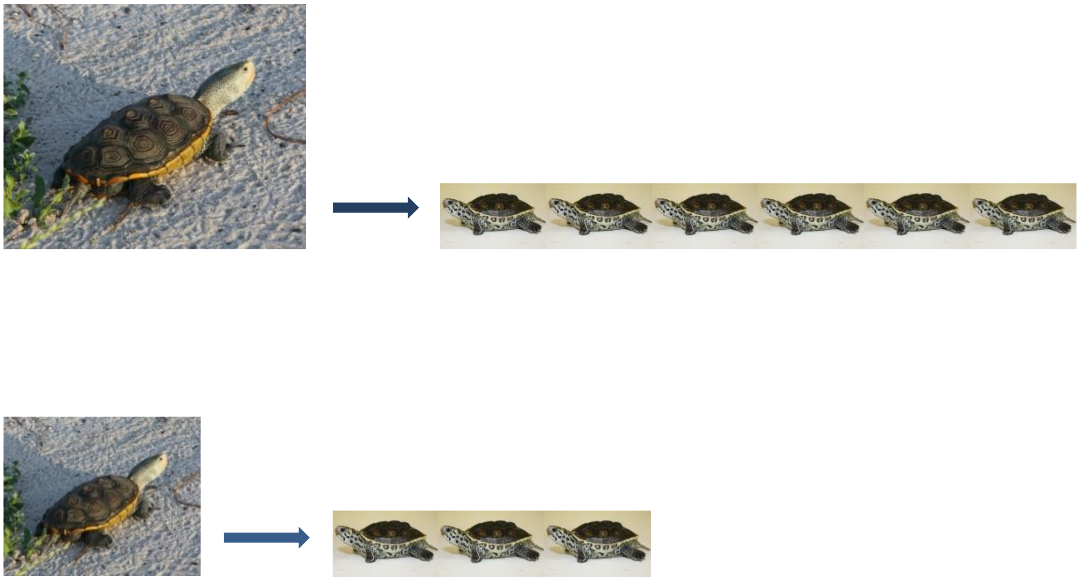

Figure 1. Pictorial representation of the predictions of the optimal egg size theory (Smith and Fretwell, 1974). Larger females would produce larger clutches than smaller females, but the egg sizes would be similar.

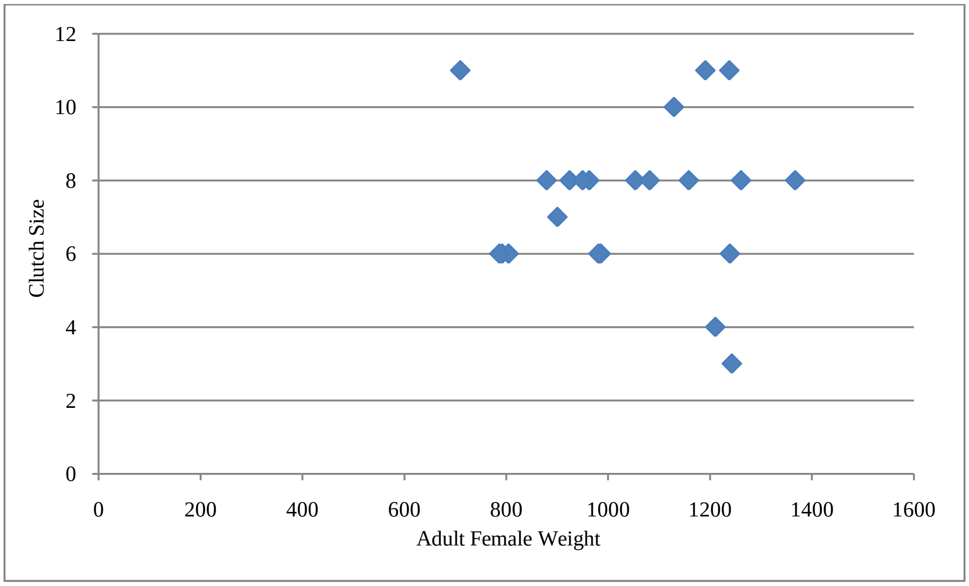

Figure 2. Scatter plot of clutch sizes by adult female weight. No signficant correlation was detected.

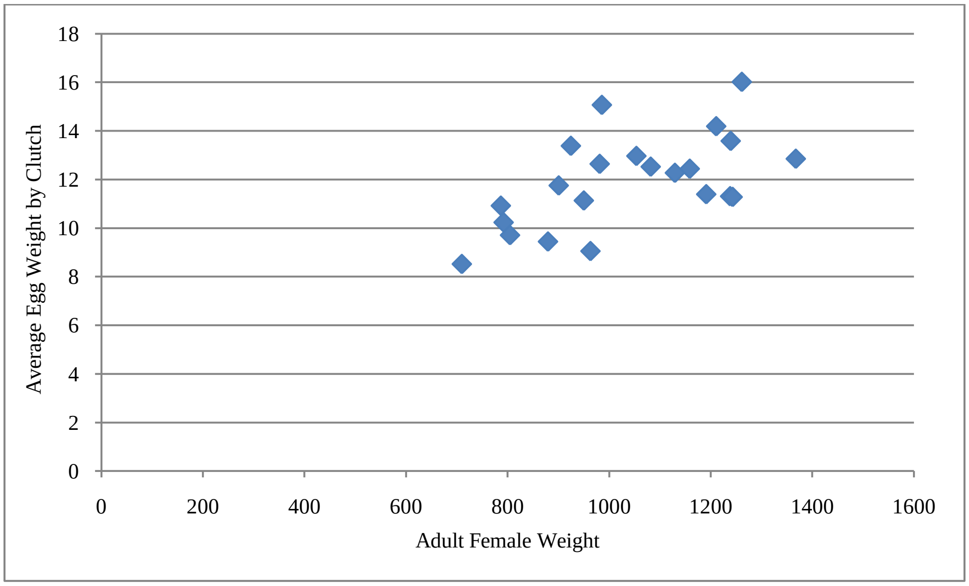

Figure 3. Scatter plot of average egg weight by clutch by adult female weight. A signficant correlation was detected.

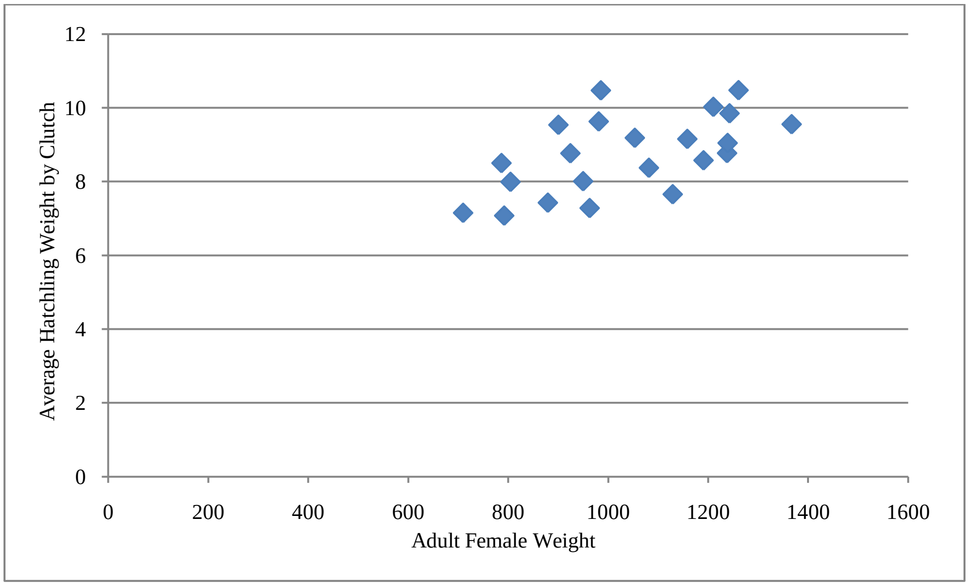

Figure 4. Scatter plot of average hatchling weight by clutch by adult female weight. A signficant correlation was detected.

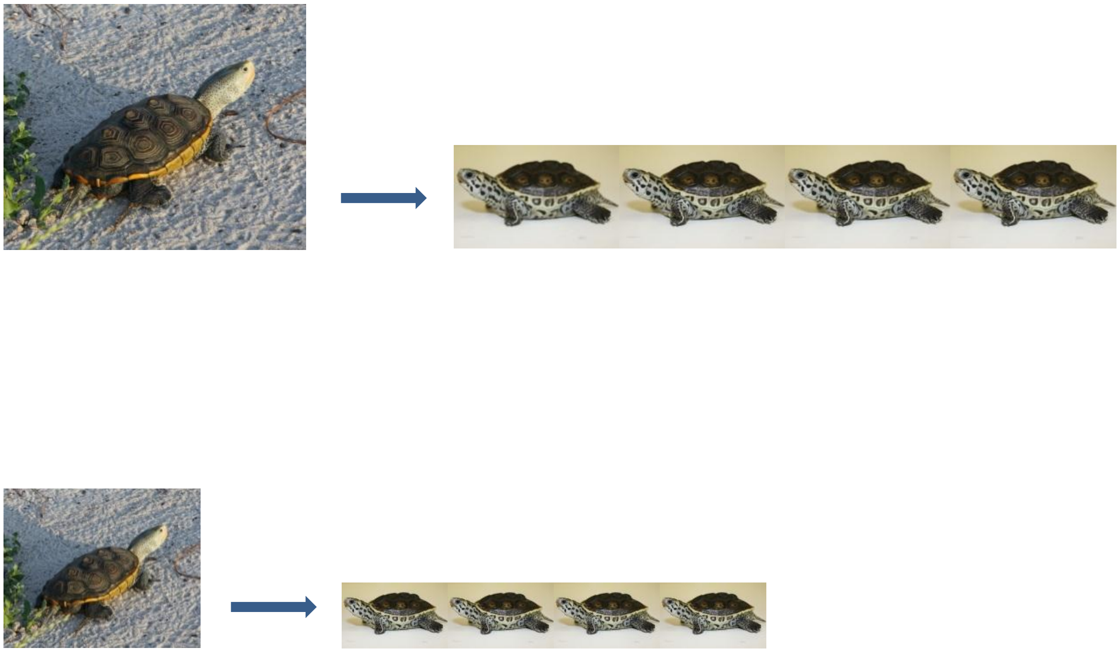

Figure 5. Pictorial representation of the overall results from the current study. Larger females produced larger eggs and hatchlings, but there was no size effects on clutch sizes.

ORIENTATION OF DIAMONDBACK TERRAPIN (MALACLEMSY TERRAPIN) HATCHLINGS ON A NATURAL NESTING BEACH

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In preparation for Journal of Herpetology

Format adapted for dissertation

Abstract

Post emergence orientation behavior of hatchling turtles has been investigated in several turtle species, and freshwater species do not follow the strong behavioral pattern as has been observed in sea turtles. Sea turtle hatchlings rely on visual cues to orient towards the brightest horizon in the seaward direction as well as orient away from tall dark silhouettes such as sand dunes. The orientation behavior of diamondback terrapin hatchlings was examined utilizing an orientation arena on a natural nesting beach, which was a similar habitat experienced by sea turtle hatchlings. As opposed to sea turtles, terrapin hatchlings displayed strong orientation towards high marsh vegetation instead of open water. The results stressed the importance of having healthy marsh habitat adjacent to nesting areas to provide critical habitat to these vulnerable life history stages of diamondback terrapins.

Introduction

Orientation behavior of turtle hatchlings has been thoroughly studied, most notably in sea turtles. Lohmann et al. (1997) reviewed previous research on this topic and summarized that sea turtle hatchlings primarily utilize visual cues to guide them from the nest to the open water to begin their migration. Hatchlings collect these cues within their “cone of acceptance”, which is a visual field with a wide horizontal angle and a narrow vertical angle. They display brightest direction orientation, and because of the narrow vertical angle within their “cone of acceptance”, the light closest to the horizon, which is usually moonlight reflecting on the water, has the greatest influence (Lohmann et al., 1997). Unfortunately, artificial lighting that is located near the nesting beach can be brighter than natural light, thus resulting in hatchlings orienting away from open water and travelling inland (Salmon *et al.*, 1995). Hatchlings also show negative orientation away from dark silhouettes indicating the positive influence natural sand dunes can have on hatchlings reaching the ocean (Lohmann et al., 1997).

Orientation behavior of freshwater turtle hatchlings has been examined in numerous studies; yet, their conclusions do not convey a succinct pattern as with sea turtles. Anderson (1958) observed hatchlings of *Trionyx muticus*, *Graptemys oculifera*, and *Graptemys pulchra* display negative orientation to tall dark forms similar to sea turtles. But although it was noted that hatchlings of these species ventured from their nests to the water after sunset, they did not utilize light reflected off the water in their migration (Anderson, 1958).

Standing *et al.* (1997) and McNeil *et al.* (2000) found that Blanding's turtle (*Emydoidea blandingii*) hatchlings from a population in Nova Scotia do not seek water after emergence despite the seemingly strong selective pressures favoring entering the water. Visual cues appeared to be utilized by the hatchlings even though silhouettes of nearby vegetation along with slope and open horizon were not important cues (Standing *et al.*, 1997). Hatchlings' movements showed some evidence of cover seeking behavior (McNeil *et al.*, 2000), and their migration was more direct under vegetative cover (Standing *et al.*, 1997). Their seemingly random movements after emergence could be an adaptive "bet-hedging" strategy in which they respond differently to various environmental stimuli to better ensure higher overall survival rates (Standing *et al.*, 1997; McNeil *et al.*, 2000). Tuttle and Carroll (2005) observed similar scattered dispersal in wood turtle hatchlings (*Glyptemys insculpta*); however, in addition to visual cues, olfactory and auditory cues and positive geotaxis appeared to influence hatchling migration.

Several studies have investigated factors that could affect survival probabilities of red-eared slider hatchlings (*Trachemys scripta*) on their journey from terrestrial nests where they overwinter to the water. These factors include body size (Janzen *et al.*, 2000), nest-site characteristics (Kolbe and Janzen, 2001), water loss (Kolbe and Janzen, 2002), and exposure to predation (Janzen *et al.*, 2007).

The current study examined the orientation behavior of Mississippi diamondback terrapin hatchlings (*Malaclemys terrapin pileata*) on a natural nesting beach. Terrapins exclusively inhabit brackish water environments of salt marshes, bays, and estuaries (Carr, 1952). Terrapin hatchlings are believed to spend their first few years in the upper salt marsh habitat (Brennessel, 2006). Lovich *et al.* (1991) observed hatchlings released in water swim directly to land and venture to wrack located at the high tide line. Thus, having ample upper salt marsh habitat appears to be critical in the life cycle of diamondback terrapins. However, their journey after emergence is perilous. Numerous animals prey on terrapin nests and hatchlings, including raccoons, fish and American crows, ghost crabs (Butler *et al.*, 2004), and Norway rats (Draud *et al.*, 2004). In this study, hatchlings were able to choose between migrating to the open water or towards salt marsh vegetation. The results will help elucidate the relative importance of the surrounding environments to migrating diamondback terrapin hatchlings.

Methods

Diamondback terrapin hatchlings were obtained from nesting females captured using drift fences with pitfall traps on the nesting beach surrounding Cedar Point Marsh located along the Gulf Coast of Alabama. Captured females were palpated to determine if they were gravid. If so, they were given a safe dose of oxytocin, a naturally occurring hormone that stimulates egg laying. Terrapins display temperature-dependent sex determination, and eggs were incubated at either 26°C (male producing temperature) or 31°C (female producing temperature). Eggs and hatchlings were separated by clutch, and hatchlings were fed daily to satiation.

Orientation arenas were constructed on the nesting beach surrounding Cedar Point Marsh and another nesting beach at Airport Marsh located behind Dauphin Island, AL. The nesting beach surrounding Cedar Point Marsh has a North-South orientation whereas the Airport Marsh nesting location runs East-West. Arenas had a diameter of 6 meters and consisted of 12 gates bordered by short PVC pipes inserted into the ground (Figure 1). Each PVC pipe was at a 30° angle from the center of the arena. Hatchlings were tested in 2008 only at Cedar Point Marsh whereas, in 2009, both Cedar Point Marsh and Airport Marsh were utilized. In 2010, only Airport Marsh was tested.

Hatchlings were tested after their yolk sac had been reabsorbed within the plastron and at most within 3 months of hatching. On site, hatchlings were tested individually and placed under a container in the center of the arena for a few minutes before starting. The container was lifted via string by a tester that was outside of the arena. The tester's position around the arena was changed periodically to prevent any bias in the hatchlings' movements. The gate the hatchling passed through along with its time to leave arena and any possible orientation behavior was noted. A Chi-square goodness of fit analysis at α level of 0.05 was run on the results to examine if hatchlings displayed nonrandom movement in their emergence migration.

Results

When examining results from individual gates, diamondback terrapin hatchlings displayed a strong orientation preference for the marsh vegetation over the open water at Cedar Point Marsh (Table 1; Figures 2&3). The results at Airport Marsh indicated random movements of hatchlings through the orientation arena (Figures 4&5).

Table 1. Results of Chi-square goodness of fit analyses examining orientation preferences of diamondback terrapin hatchlings. Each analysis had 11 degrees of freedom. Asterisk denotes levels of significance.

Year	Location	n	X^2	$X^2_{0.05,11}$	p
2008	Cedar Point Marsh	60	142.8*	19.675	<0.0001
2009	Cedar Point Marsh	78	239.2*	19.675	<0.0001
2009	Airport Marsh	27	17.4	19.675	0.097
2010	Airport Marsh	41	16.02	19.675	0.140

Although the results from Airport Marsh suggested random hatchling movement on the nesting beach, a preference for marsh vegetation was indicated in 2009 when the gates were grouped into marsh facing gates and seaward gates (Table 2). In 2009, eighteen hatchlings ventured through gates facing the marsh (Gates 4-9) whereas only 3 passed through seaward gates (Gates 1-3, 10-12). The significant value for 2010 was influenced by the high number of hatchlings that did not make it out of the orientation arena.

Table 2. Results of Chi-square goodness of fit analysis of orientation behavior on the natural nesting beach surrounding Airport Marsh. Gates were grouped into marsh facing gates (Gates 4-9) and seaward gates (Gates1-3, 10-12). Each analysis had one degree of freedom. Asterisks denote levels of significance at α level of 0.05.

Year	n	X^2	$X^2_{0.05,1}$	p
2009	27	9.67	3.84	0.0002
2010	41	4.22	3.84	0.040

Consistent behavior patterns were observed throughout the study. Hatchlings would extend their head and turn their body usually in a complete circle before any movement. Once movement was initiated, it was not continuous to the destination. Hatchlings would stop intermittently to perform their “orientation circle” with heads extended before continuing their migration. Sometimes, they would change course or persist on their original track. Also, a number of hatchlings buried themselves into the beach substrate instead of migrating.

A wide range of times for the hatchlings’ to exit the arena was measured. Numerous instances occurred where hatchlings left the arena in 60 seconds or less. Interestingly, the weather for the day where most of these quick times were recorded was overcast. However, the second highest percentage of hatchlings that buried themselves was also observed on this day. The average time for those hatchlings that made it out of the arena in the allotted ten minutes was 285.08 seconds.

Discussion

Burger (1976) performed field and laboratory experiments examining orientation behavior of terrapin hatchlings. Hatchlings emerged between 700 and 1900, with most between 1200 and 1700. When the hatchlings emerged from nests on flat areas, tracks were observed in random directions; however, they ventured down the gradient from nests laid on slopes (Burger, 1976). Results indicated that hatchlings would migrate down inclines in different compass orientations. Although, individuals chose moving to vegetation regardless of incline (Burger, 1976).

In the present study, diamondback terrapin hatchlings displayed a significant preference for migrating to marsh vegetation over the open water. Visual cues seemed most important during the hatchlings' movements. Terrapins would perform "orientation circles" with extended heads before initial movements and then stop at various times to perform additional ones. Burger (1976) observed similar behavior with hatchlings raising their heads and looking around. Butler *et al.* (2004) also observed 160 of 172 hatchlings' discernible tracks leading directly to marsh vegetation. Tuttle and Carroll (2005) suggested that vision also was the major cue for wood turtle hatchlings. Wood turtle hatchlings exhibited saltatory searches with stop and go movements. The same saltatory searches seem to be the most important cue for terrapin hatchling migrations as well.

Other cues could have influence on terrapin hatchlings. The arenas were constructed in relatively flat areas, so the influence of positive geotaxis was not examined in this study. As in Burger's (1976) study, compass direction did not affect hatchlings' migration. The arena at Cedar Point Marsh had a N-S orientation whereas the Airport Marsh had a E-W orientation. The results obtained from Airport Marsh were not as clear as those from Cedar Point Marsh. The higher amount of vegetation surrounding the Airport Marsh arena probably skewed the ultimate destination given the strong preference for vegetation observed by Burger (1976). Tuttle and Carroll (2005) argued that olfactory and auditory cues could also have influence on wood turtle hatchlings. Neither of these possibilities was explored in the present study.

It should not be surprising that hatchlings burying themselves into the beach substrate were observed. Terrapins emerge during daylight hours, with the highest emergence occurring during the hottest portion (Burger, 1976). Desiccation has been shown to be a powerful influence on turtle hatchlings' migrations (Kolbe and Janzen, 2002). Dead hatchlings that apparently succumbed to overheating have been found on

Cedar Point Marsh (Andrew Coleman, pers. obs.). So, burying themselves into the substrate appears to be an alternative strategy to direct movement to the marsh vegetation. We did not wait to investigate whether hatchlings that buried themselves later completed their migration when ambient temperatures decreased. However, this strategy could increase their chances of falling prey to nocturnal raccoons.

This study is the first to utilize an orientation arena on a natural nesting beach so the migration preferences of diamondback terrapin hatchlings could be quantitatively assessed. Hatchlings significantly preferred venturing to marsh vegetation rather than open water. This is in stark contrast to the migration preferences of sea turtles, which display negative orientation from dark silhouettes and positive orientation towards the open horizon of the ocean (Lohmann *et al.*, 1997). The results also stress the importance of having healthy upland marsh habitat adjacent to terrapin nesting beaches. It is critical for the success of hatchling survival and eventual recruitment into the adult population. Unfortunately, loss of marsh habitat is rampant throughout coastal ecosystems (Stedman and Dahl, 2008), and it represents a major threat to the future viability of terrapin populations (Roosenburg, 2001; Butler and Seigel, 2006).

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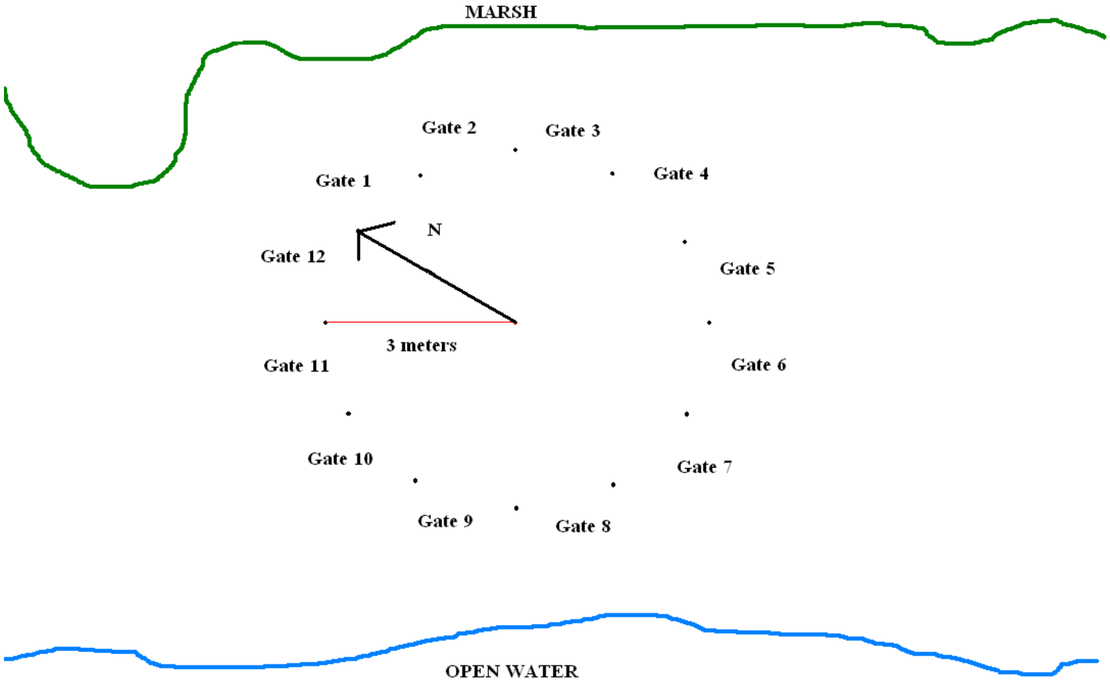

Figure 1. Diagram of orientation arena on natural nesting beach. The arena was composed of twelve gates that were delimited by PVC pipes positioned at a 30° angle from the center. Gate 1 always faced due North.

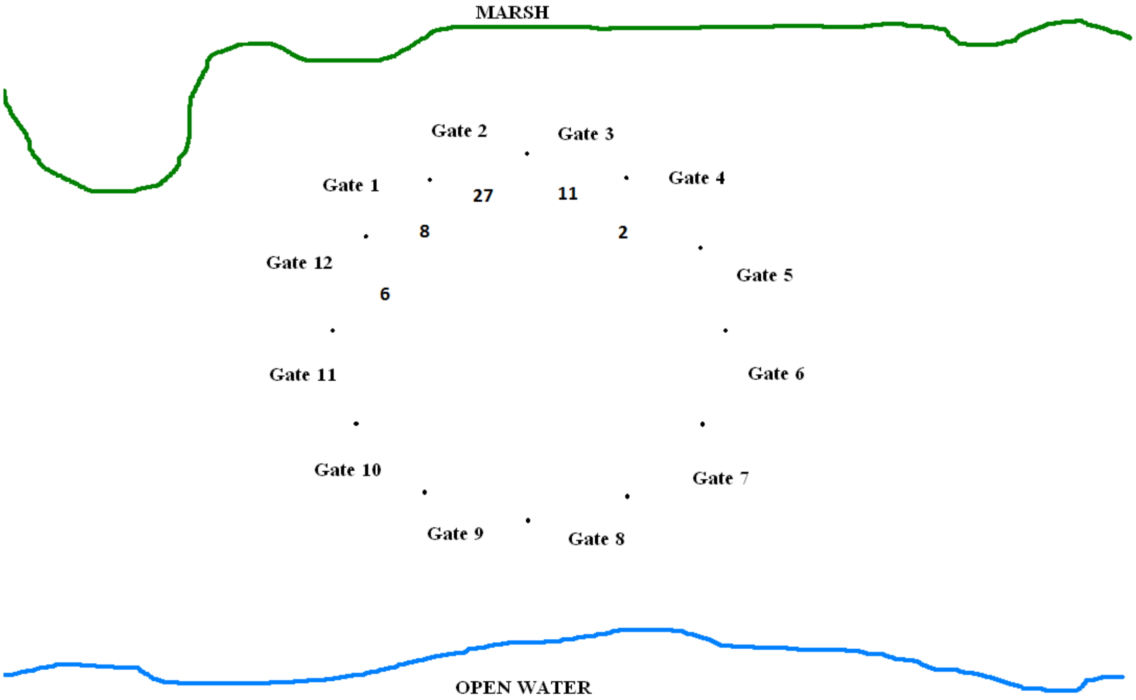

Figure 2. 2008 results of diamondback terrapin hatchling orientation behavior on a natural nesting beach surrounding Cedar Point Marsh. Six hatchlings did not make it out of the arena in the allotted ten minutes.

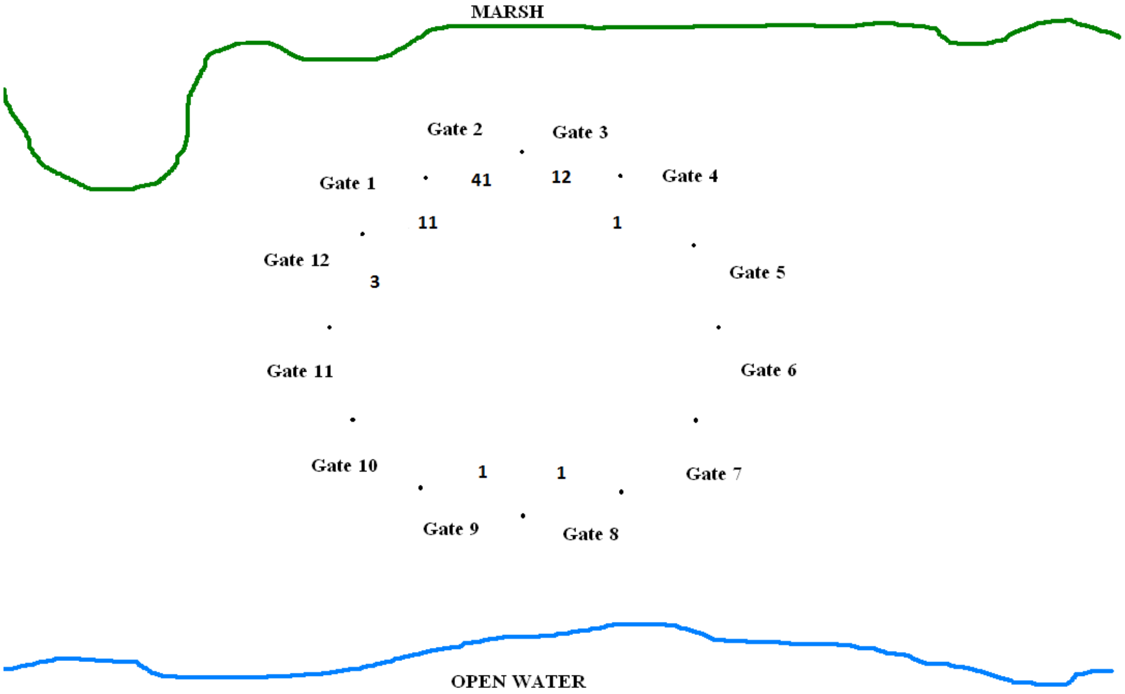

Figure 3. 2009 results of diamondback terrapin hatchling orientation behavior on a natural nesting beach surrounding Cedar Point Marsh. Eight hatchlings did not make it out of the arena in the allotted ten minutes.

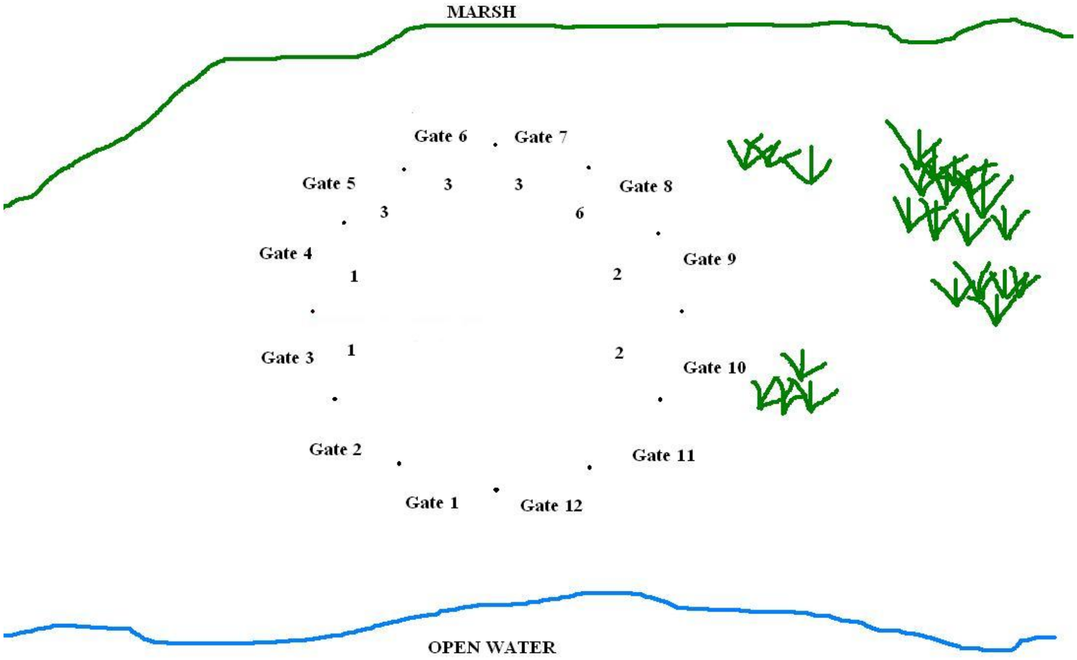

Figure 4. 2009 results of diamondback terrapin hatchling orientation behavior on a natural nesting beach surrounding Airport Marsh. Six hatchlings did not leave the orientation arena in the ten minutes allotted.

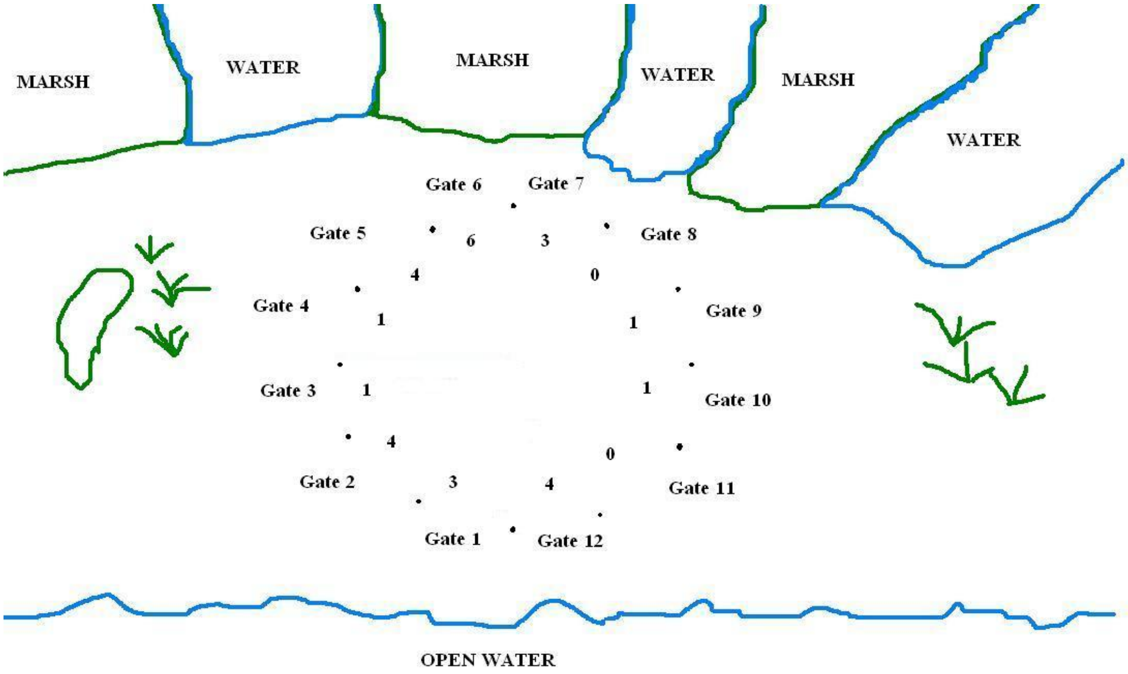

Figure 5. 2010 results of diamondback terrapin hatchling orientation behavior on a natural nesting beach surrounding Airport Marsh. Thirteen hatchlings did not leave the orientation arena in the ten minutes allotted.

FINAL DISCUSSION

The diamondback terrapin is an obligate estuarine turtle inhabiting salt marshes, bays, and estuaries from Cape Cod, MA, along the Atlantic and Gulf Coasts to Corpus Christi, TX. Seven subspecies are currently recognized, and the range for the Mississippi diamondback terrapins extends from the Panhandle region of Florida to the Louisiana/Texas border (Brennesel, 2006). Terrapins potentially play a significant role in maintaining salt marsh ecosystem integrity through their predation on salt marsh periwinkle snails (*Littoraria irrorata*) (Silliman and Bertness, 2002). Terrapins were once abundant in the salt marshes lining the Atlantic and Gulf coasts of the U.S. They represented a valuable economic resource that was exploited as a culinary delicacy (Carr, 1952; Hart and Lee, 2006). Although, they are no longer subject to range-wide commercial exploitation, terrapin populations have not rebounded to historic levels. Surveys of terrapin researchers from along the Atlantic and Gulf coasts indicated that crab trap-induced mortality, habitat loss, nesting beach alteration, predation, and road mortality represented major threats to populations in most areas (Butler *et al.*, 2006). Terrapins face additional threats such as pollution, legal harvest, and the potential for the renewal of a commercial market (Gibbons *et al.*, 2001).

A long-term study was initiated in 2004 to assess the current population status of the Mississippi diamondback terrapin (*Malaclemys terrapin pileata*) along the Gulf Coast of Alabama. Diamondback terrapins were observed existing in a number of isolated remnant aggregations in the Mississippi Sound along the Gulf Coast of Alabama. Cedar Point Marsh supported the largest of these aggregations. Other than 2007, over 100 depredated nests were found each year from 2006-2010 on the nesting beach surrounding Cedar Point Marsh. A number of individuals from several locations were captured with modified crab traps and pitfall traps in 2004 and 2005, but the majority of the captures occurred in Cedar Point Marsh from 2006-2010. A total of 97 individuals were captured in Cedar Point Marsh during this time using all three trapping methods (modified crab traps, pitfall traps, and trawling), and 38 of those individuals were recaptured at least once. The Cedar Point Marsh aggregation has experienced a size reduction over a long period of time. According to a New York Times article, the world's largest terrapin farm existed in this location in the late 1800's. It housed over 20,000 terrapins, which was a combination of resident terrapins along with terrapins that were brought in by local residents, and each year, thousands of turtles were shipped to the markets of the Northeastern U.S. for human consumption. Based on population estimates using mark-recapture data, the Cedar Point Marsh aggregation was estimated to be 336 individuals (not including 0-2 year olds), down from reportedly 20,000 individuals, a 98% decrease.

In addition to credible ecological and demographic data, the attainment of reliable molecular information results in the best understanding of the current status of the concerned population (Haig, 1998). The current study utilized the twelve microsatellites used by Hart (2005) to investigate the genetic health of a diamondback terrapin population along the Gulf Coast of Alabama. The Alabama population of diamondback terrapins along with the other sampled populations included in this study displayed decreased genetic variation. The observed heterozygosity for four out of the twelve loci in the Alabama population deviated significantly from Hardy Weinberg equilibrium. The main cause for the depletion of heterozygosity was genetic drift resulting from drastic population size reduction. This conclusion was supported by the population's *M* ratios, which are derived from dividing the total number of alleles by the allelic range. Each of the sampled terrapin populations displayed *M* ratios less than 0.40, which is well below the threshold value for bottlenecked populations of 0.68 (Garza and Williamson, 2001). According to her range-wide study, Hart (2005) concluded that terrapins exist in six genetic management units (MUs). However, there would be hesitancy to group the three Gulf of Mexico subspecies into one MU based on this study's results. F_{st} values for the comparisons between Alabama, Louisiana, and Texas populations with the Tampa, FL, population were all above 0.15, which Hartl (1988) proposed as indicating high levels of genetic differentiation. The STRUCTURE analysis designated three basic groupings: the SC population, the FL populations, and Northern Gulf of Mexico populations. Sampling *M. t. macrospilata* populations closer to the boundary with *M. t. pileata*, such as from the Big Bend area or the Florida panhandle, would better elucidate the genetic differentiation between these two subspecies. Although heavily depleted, the Alabama population represents a relatively important source of genetic diversity for the Northern Gulf of Mexico MU. The historical gene flow that influenced genetic connectivity between populations may not be occurring to the same degree today, so future monitoring is necessary to assess the genetic viability of terrapin populations throughout their range.

The crab fishery along the Atlantic and Gulf coasts of the U.S. primarily targets the blue crab (*Callinectes sapidus*) (Kennedy *et al*. 2007). Yearly landings have fluctuated, but the total amount of blue crabs landed along the Atlantic and Gulf coast of the U.S. gradually increased from the 1950's through 1970's (Stagg and Whilden, 1997; Guillory *et al*., 2001; Kennedy *et al*., 2007). In general, this was concomitant with an increase in the number of crab traps and crab fisherman (Hill *et al*., 1989; Guillory *et al*., 2001). Since the 1970's, fishing effort has remained relatively high in the blue crab fisheries, although significant declines in landings have occurred in recent years in many areas due to factors such as overharvesting, pollution, and loss of habitat (Murphy *et al.*, 2007; Sutton and Wagner, 2007; CBSAC Report, 2010). The crab trap is the preferred capture method in most areas and a numerous studies over the past several decades have documented their impact on terrapin populations. Crab trap-induced mortality of terrapins has been documented in New Jersey (Wood, 1997), Maryland (Roosenburg, 1991; Roosenburg *et al.*, 1997; Roosenburg and Green, 2000), South Carolina (Dorcas *et al.*, 2007), Georgia (Grosse *et al.*, 2009), Florida (Siegel, 1993), and Alabama (Coleman *et al.*, unpublished data; Marion, 1986; Nelson and Marion, 2004). The study in Maryland estimated that more than 2000 terrapins were caught annually by crab traps (Roosenburg *et al.*, 1997). Data from these studies also suggested that abandoned or lost crab traps (often referred to as derelict or "ghost" crab traps) may pose a greater risk of mortality for terrapins than those that are checked regularly.

By-catch reduction devices are metal wire or plastic rectangles initially developed by Wood (1997), which fit into the crab trap funnel openings and are intended to prevent terrapin entry into the traps while still allowing crabs to enter. The results of the current study indicate that BRDs significantly reduced the capture of terrapins in crab traps. Of the 24 terrapins caught in crab traps in Cedar Point Marsh, only two were captured in traps with BRDs, suggesting an approximate 90% reduction in the capture of terrapins. BRDs have been shown to significantly inhibit terrapin entry in every study that examined their efficacy. Wood (1997) found that traps outfitted with 5 x 10 cm BRDs performed the best in preventing terrapin capture and permitting crab capture. In fact, capture of marketable-sized crabs was significantly enhanced in these traps. This scenario was also observed by Guillory and Prejean (1998) and Roosenburg and Green (2000). Further, no significant differences in crab captures between traps fitted with BRD's and non-fitted traps were detected in other studies (Cole and Helser, 2001; Butler and Heinrich, 2007; Morris *et al.*, 2010; Rook *et al.*, 2010). However, in the current study, the weekly capture of marketable sized crabs and total crabs, as well as the C.P.U.E. of marketable sized crabs, were significantly lower in traps fitted with BRDs than traps not fitted with BRDs. Crab traps fitted with BRDs showed an approximate 27% reduction in the capture of blue crabs. But, the sample size of captured crabs of the current study (543) paled in comparison to other studies that have examined this topic. The results of the current study indicate that the implementation of BRDs on crab traps could represent an effective component of a management plan for initiating the recovery of the diamondback terrapin in Alabama. The low terrapin C.P.U.E. in non-BRD traps in the current study (0.012 terrapins/trap/day) compared to other areas (0.17 terrapins/trap/day in Maryland (Roosenburg *et al.*, 1997) and 0.20 terrapins/trap/day in Virginia (Rook *et al.*, 2010)) suggests a small population size which could be an indication of the historic impact of the crab fishery (Wood, 1997) on the terrapin population in Alabama.

Habitat loss is another threat impacting terrapin survival. According to a 2008 study, over 350,000 acres of marsh were lost in coastal watersheds in the eastern U.S. between 1998-2004, and losses in the Gulf of Mexico region were 25 times higher in that same time period (Stedman and Dahl, 2008). Of the designated wetland types, salt marshes sustained the heaviest losses. Many of these losses can be attributed to anthropogenic effects associated with coastal development (Stedman and Dahl, 2008; Hartig *et al.*, 2002). Dredging, filling, shoreline hardening, and a variety of other activities alter natural processes and contribute to the loss of salt marsh habitat (Stedman and Dahl, 2008) and diamondback terrapin nesting habitat (Roosenburg, 1991). The Deepwater Horizon oil spill in the northern Gulf of Mexico during 2010 and the Chalk Point oil spill in the Patuxent River in Maryland during 2000 (Michel *et al.*, 2001) have highlighted the potential impact of oil pollution on estuaries inhabited by terrapin. Therefore, loss of salt marsh habitat, salt marsh pollution, and decrease in salt marsh quality represent significant threats to terrapin conservation. While terrapins are dependent upon salt marsh for their survival, they also enhance the stability and health of the salt marsh (Silliman *et al.*, 2005; Gustafson *et al.*, 2006). Mann (1995) suggested that sufficient nesting beaches may be a limiting factor to terrapin distribution. Suitable salt marsh habitat appears to be abundant in Alabama, but there may be a lack of elevated nesting beach surrounding the salt marshes. The presence of healthy marsh habitat was also shown to be critical to supporting terrapin hatchlings. Post-emergence diamondback terrapin hatchlings displayed a significant preference to orient toward marsh vegetation over the open water. Visual cues seemed most important during the hatchlings' movements. Terrapins would perform "orientation circles" with extended heads before initial movements and then stop at various times to perform additional ones. This is in stark contrast to the migration preferences of sea turtles, which display negative orientation toward dark silhouettes and positive orientation towards the open horizon of the ocean (Lohmann *et al.*, 1997). The results stressed the importance of having healthy upland marsh habitat adjacent to terrapin nesting beaches. It is critical for the success of hatchling survival and eventual recruitment into the adult population.

Concomitant with the increase in coastal development, populations of major terrapin nest predators, primarily raccoons, have increased (Roosenburg, 1991; Feinburg and Burke, 2003). Feinburg and Burke (2003) observed over 3,000 depredated nests over two nesting periods and attributed over 98% of the depredations to raccoons. In Maryland, 94% of nests were taken from a sampled nesting location, with raccoons as the main predator (Roosenburg and Place, 1995). Butler *et al.* (2004) detected over 80% of nests were depredated over two nesting seasons in a Florida terrapin population, and raccoons were the primary nest predator. A study of a terrapin population in Rhode Island found that 87% of the monitored nests were depredated and raccoons were the primary predator (Goodwin, 1994). Other than 2007, over 100 depredated nests were found each year from 2006-2010 on the nesting beach surrounding Cedar Point Marsh. This greatly outnumbered the nesting beach with next highest total, Airport Marsh with 43 depredated nests. Even though the total for Airport Marsh was over five seasons, the majority of those nests were found in one survey in 2010, which indicates that Airport Marsh still represents an important nesting beach and that nest surveys could overlook potential nesting beaches if not performed intensively or at regular intervals. Assuming female terrapins in Alabama nest twice a year, the majority of nests at Cedar Point Marsh (166 hypothetical nests based on the estimate of 83 nesting females) are depredated each year (over 150 nests were found in 2009 and 2010). This is consistent with the nest survivorships observed in other populations (Roosenburg, 1991; Butler, 2002; Mitro, 2003).

For terrapins, road mortality represents a significant threat to adults during reproductive migration and can significantly decrease a population's reproductive potential (Wood and Herlands, 1997). In particular, adult females are vulnerable as they migrate to suitable nesting sites through habitats that have been fragmented by roadways. For example, on the Cape May Peninsula of New Jersey, over four thousand adult females were killed between 1989 and 1995 on 11.5 km of roadways that dissect the terrapins' habitats (Wood and Herlands, 1997). Road mortality may have resulted in a change in this local terrapin population structure, with a decrease in the number of adult females in comparison to the same population in the late 1980s (Avissar, 2006). Further, the adult females that were captured were smaller, and since size can correlate with age, the author suggested that the average age of adult females in the population had decreased (Avissar, 2006). The loss of older and larger females could have consequences beyond losing these individuals. Egg sizes and initial hatchling sizes increased with increased female size and age in the Alabama population of diamondback terrapins. It was unclear if the larger hatchlings from Alabama experienced a fitness advantage over smaller hatchlings, although studies completed with other species concluded that larger hatchlings could enjoy greater predator avoidance (Janzen *et al.* 2000, Janzen *et al.*, 2007). A comparison between the Alabama population, which is not experiencing high road mortality, with one that is, Tybee Island, Georgia, indicated that Alabama terrapin females produced larger eggs, resulting in heavier hatchlings. These differences could be a result from the loss of larger and older adult females from the Tybee Island population due to road mortality. If larger hatchlings are more successful, then road mortality could be preventing the introduction of the more successful hatchlings into the population.

Complete long-term data sets are critical in evaluating population trends and life history parameters in long-lived species such as turtles. Congdon *et al.* (1993, 1994) stated that long-lived species have a suite of co-evolved life history traits that will preclude the ability to properly adapt to chronic disturbances and will ultimately result in the species' demise if the disturbances are not properly reversed or mitigated. Numerous anthropogenic threats are negatively affecting terrapins throughout their range. In Alabama, crab trap mortality and nest predation have been identified as the top threats inhibiting the diamondback terrapin population from recovering from historical exploitation. Long-term datasets, such as the one that was initiated by the current study, that include population size and genetic diversity estimates will play a prominent role in developing optimal management strategies aimed at ensuring the survival of the terrapin. Also, examining various aspects of terrapin biology and ecology, such as hatchling orientation behavior and female allocation strategies, provided crucial information that can be utilized for its conservation. Management strategies should include implementing BRD's into commercial crab traps and the banning of crab trapping in Cedar Point marsh, which supports the largest aggregation of terrapins in Alabama.

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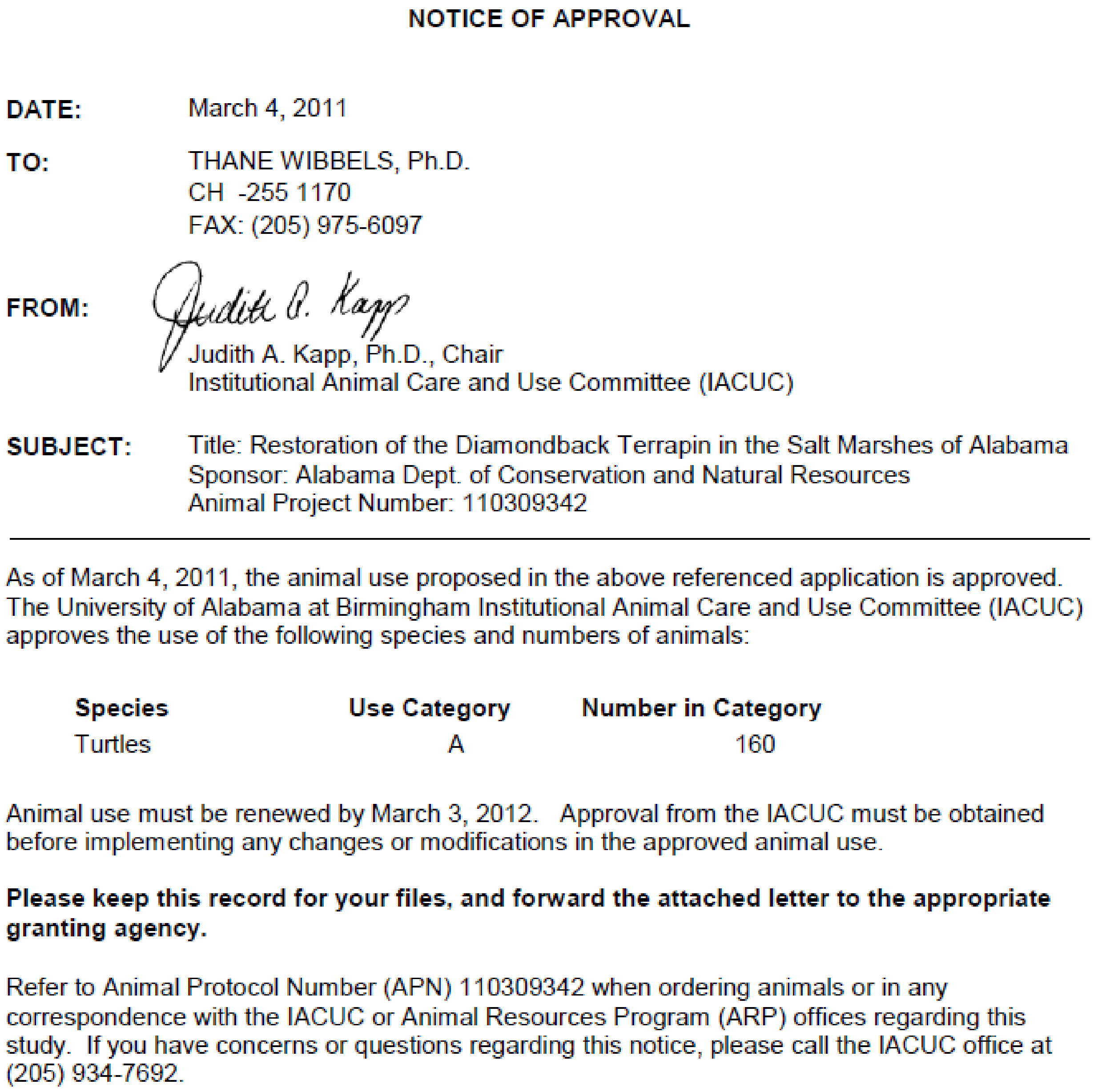
APPENDIX A

Institutional Animal Care and Use Committee (IACUC)



NOTICE OF APPROVAL

DATE: March 4, 2011

TO: THANE WIBBELS, Ph.D.
CH -255 1170
FAX: (205) 975-6097

FROM: Judith A. Kapp
Judith A. Kapp, Ph.D., Chair
Institutional Animal Care and Use Committee (IACUC)

SUBJECT: Title: Restoration of the Diamondback Terrapin in the Salt Marshes of Alabama
Sponsor: Alabama Dept. of Conservation and Natural Resources
Animal Project Number: 110309342

As of March 4, 2011, the animal use proposed in the above referenced application is approved. The University of Alabama at Birmingham Institutional Animal Care and Use Committee (IACUC) approves the use of the following species and numbers of animals:

Species	Use Category	Number in Category
Turtles	A	160

Animal use must be renewed by March 3, 2012. Approval from the IACUC must be obtained before implementing any changes or modifications in the approved animal use.

Please keep this record for your files, and forward the attached letter to the appropriate granting agency.

Refer to Animal Protocol Number (APN) 110309342 when ordering animals or in any correspondence with the IACUC or Animal Resources Program (ARP) offices regarding this study. If you have concerns or questions regarding this notice, please call the IACUC office at (205) 934-7692.

APPENDIX B

HEAD SURVEY RESULTS 2004-2010

CEDAR POINT MARSH

Date	Heads in	Heads out	Elapsed time in	Tide
7/9/2004	4	2	30	
7/22/2004	1	1	30	
4/22/2005	6		30	
6/16/2005	6		30	
8/5/2005	1		30	
5/15/2006	5		30	falling tide
5/17/2006	1		30	falling tide
5/18/2006	2		30	
5/19/2006	1		30	
5/25/2006	2		30	
6/6/2006	5		30	
6/29/2006	1		30	
6/30/2006	4		30	
7/20/2006	1		30	
7/21/2006	0		30	high tide
8/1/2006	0	1	30	falling tide
8/3/2006	2		30	high tide
8/4/2006	0	2	30	
9/13/2006	0		45	
9/15/2006	1		30	
10/20/2006	1		30	
3/19/2007	2		30	
3/23/2007	2		30	
4/9/2007	2		30	Very High
4/13/2007	8		30	
5/1/2007	17		30	
5/4/2007	1		30	Over mid rising
5/14/2007	5		30	
5/17/2007	0		30	over mid rising
5/21/2007	2		30	near high tide
5/26/2007	1		30	
6/1/2007	0		30	
6/7/2009	1		30	
6/14/2007	1		39	high tide
6/19/2007	0		30	

6/26/2007	1		30	
7/3/2007	0		30	
7/10/2007	0		30	
7/17/2007	0		30	
7/24/2007	0		30	below mid tide
8/2/2007	5		30	high tide
9/14/2007	2		30	near high tide
10/15/2007	0		30	mid rising tide
10/19/2007	9		30	mid tide
3/27/2008	6		30	
3/28/2008	1		30	mid rising tide
3/31/2008	4		30	
4/10/2008	5		60	low tide
4/11/2008	3		39	low but rising tide
4/12/2008	10	7	30	low but rising tide
4/14/2008	2		30	mid tide
4/24/2008	8		30	high tide
4/25/2008	3		30	
5/2/2008	8		30	mid
5/8/2008	0	4	30	
5/19/2008	4		30	
5/20/2008	3		30	
5/21/2008	0		30	
5/23/2008	5		30	
5/26/2008	0		30	high tide
5/28/2008	3		30	
5/30/2008	1		30	
6/3/2008	0		30	
6/6/2008	0		30	
6/10/2008	6		30	
6/12/2008	6		30	
6/17/2008	0		30	falling tide low
6/19/2008	1		30	
6/24/2008	4		30	
6/27/2008	2		30	rising tide
7/1/2008	0		30	falling tide
7/3/2008	0		30	
7/8/2008	5	3	30	
7/10/2008	3	2	30	
7/16/2008	2		30	
7/17/2008	9		30	
7/22/2008	3		30	
7/24/2008	2		30	
7/29/2008	0		30	
7/31/2008	1		30	
8/8/2008	2	1	30	
8/12/2008	2		30	

9/25/2008	4	3	30	high tide
9/26/2008	0		30	neap tide
10/9/2008	3		30	neap tide
10/10/2008	0	3	30	rising tide
3/11/2009	12	5	30	high tide
3/12/2009	6		30	high tide
3/19/2009	17		60	rising tide
3/25/2009	3		20	rising tide
4/10/2009	7		30	falling tide
4/17/2009	9		30	
4/30/2009	6		30	falling tide
5/15/2009	14		90	high tide
5/18/2009	12		30	high tide
5/19/2009	16		30	falling tide
5/26/2010	1		30	normal tide
5/27/2009	0		25	low tide
5/28/2009	1		30	rising tide
5/29/2009	2		30	high tide
6/3/2009	1		30	rising tide
6/9/2009	2		30	over mid rising
6/11/2009	4		30	high neap tide
6/12/2009	1		35	
6/16/2009	5		30	low tide at 6:50
6/18/2009	4		30	high at 7:50
6/24/2009	1		30	high tide
6/30/2009	13		25	mid tide
7/13/2009	0		35	rising mid tide
7/7/2009	0		30	mid tide
7/9/2009	3	2	30	low tide at 6:30
7/14/2009	2	1	30	high tide
7/17/2009	3		30	high at 12:00
7/17/2009	1		30	high at 5am
7/21/2009	0		30	low at 5:30
7/23/2009	2		30	high tide
7/27/2009	4	3	20	mid to high tide
7/30/2009	0	2	30	high tide
7/31/2009	0	2	30	high tide
8/4/2009	1		30	mid tide
8/13/2009	0		30	low tide
8/28/2009	0		30	high tide
10/2/2009	0		30	high tide
3/16/2010	1	1	30	falling tide
4/2/2010	0		30	high tide
4/19/2010	2		30	
4/26/2010	10	4	30	rising tide
4/27/2010	1		30	
5/3/2010	4		30	high tide

5/6/2010	23		25	high tide
5/12/2010	0		28	
5/17/2010	0		30	high tide
5/18/2010	5		30	falling tide
5/19/2010	1		30	high tide
6/1/2010	0	3	30	high neap tide
6/8/2010	12		30	mid tide
6/15/2010	2		20	falling tide
6/17/2010	3		30	rising tide
6/18/2010	3		30	mid tide
6/24/2010	1		25	high to mid
6/29/2010	1		22	high tide
7/15/2010	11	3	30	mid tide
7/16/2010	6		30	high tide
7/23/2010	0		30	high tide
7/27/2010	3		30	high to mid tide
8/5/2010	1	1	30	rising tide
8/6/2010	0		30	low tide
8/20/2010	0		30	mid to low tide
8/27/2010	1		30	
9/1/2010	15		30	
9/24/2010	4	3	30	

AIRPORT MARSH

Date	Heads in	Heads out	Elapsed time in	Tide
6/11/2004	0		30	
7/23/2004	1		15	
6/17/2005	1		30	
5/22/2007	1		30	
5/26/2007	0		30	
6/1/2007	0		30	
6/14/2007	0		30	
6/19/2007	0		30	
6/26/2007	0		30	high tide
7/3/2007	0		30	
7/10/2007	0		30	
7/17/2007	0		30	
7/24/2007	0		30	
8/2/2007	0		30	
4/10/2008	5		30	
4/24/2008	2		30	
6/20/2008	0		30	
7/16/2008	9		30	
6/7/2009	0		30	
6/11/2009	0		15	

8/7/2009	0		15	high tide
8/12/2010	0		30	
8/13/2010	0		30	

LITTLE DAUPHIN ISLAND

Date	Heads in	Heads out	Elapsed time in	Tide
6/11/2004	0		20	
6/17/2005	6		30	
5/22/2007	0		30	
6/1/2007	0		30	
6/7/2009	0		30	
6/14/2007	0		39	
6/19/2007	1		30	
6/26/2007	0		30	high tide
7/3/2007	0		30	
7/10/2007	0		30	
7/17/2007	0		30	
7/24/2007	0		30	
8/2/2007	0		30	
6/20/2008	4		30	
8/7/2008	1		30	
6/11/2009	0		25	high tide
8/20/2010	0		30	

JEMISON MARSH

Date	Heads in	Heads out	Elapsed time in	Tide
5/17/2006	4		30	
6/6/2006	4		10	
7/20/2006	1		15	
7/21/2006	0		20	
8/1/2006	0		30	
8/3/2006	1		40	
9/13/2006	1		20	falling tide
9/15/2006	0		20	high tide
10/20/2006	0		20	
4/12/2008	4		30	

BARTON ISLAND

Date	Heads in	Heads out	Elapsed time in	Tide
5/5/2005	0		30	
8/6/2008	0		30	

MON LOUIS ISLAND/CAT ISLAND

Date	Heads in	Heads out	Elapsed time in	Tide
5/6/2004	4		48	
5/5/2005	5	1	76	
4/25/2008	2		30	
8/6/2009	0		10	high tide
8/6/2009	0		30	high tide

MISCELLANEOUS

Date	Location	Heads in	Heads out	Elapsed time in
5/6/2004	Fowl River	0		25
5/7/2004	Point Aux Pines	0		30
6/10/2004	Heron Bay	4	2	15

APPENDIX C

DEPREDATED NEST SURVEY RESULTS

2004

Date	Lattitude	Longitude	Location	Type of cover
5/20/2004	N 30° 19.285	W 88° 08.759	Cedar Point Marsh	
5/20/2004	N 30° 19.378	W 88° 08.787	Cedar Point Marsh	
5/20/2004	N 30° 19.127	W 88° 08.696	Cedar Point Marsh	
5/20/2004	N 30° 18.819	W 88° 08.496	Cedar Point Marsh	
6/10/2004	N 30° 19.182	W 88° 08.727	Cedar Point Marsh	
6/10/2004	N 30° 19.194	W 88° 08.736	Cedar Point Marsh	
6/10/2004	N 30° 19.197	W 88° 08.738	Cedar Point Marsh	
6/10/2004	N 30° 19.202	W 88° 08.740	Cedar Point Marsh	
6/10/2004	N 30° 19.197	W 88° 08.741	Cedar Point Marsh	
6/10/2004	N 30° 19.140	W 88° 08.703	Cedar Point Marsh	veg
6/10/2004	N 30° 19.140	W 88° 08.703	Cedar Point Marsh	veg
6/10/2004	N 30° 19.138	W 88° 08.700	Cedar Point Marsh	
6/10/2004	N 30° 19.121	W 88° 08.692	Cedar Point Marsh	
6/10/2004	N 30° 19.085	W 88° 08.669	Cedar Point Marsh	
6/10/2004	N 30° 18.871	W 88° 08.341	Cedar Point Marsh	
6/10/2004	N 30° 18.833	W 88° 08.503	Cedar Point Marsh	
6/10/2004	N 30° 18.758	W 88° 08.463	Cedar Point Marsh	
6/10/2004	N 30° 18.838	W 88° 08.520	Cedar Point Marsh	
7/8/2004	N 30° 19.175	W 88° 08.458	Cedar Point Marsh	
7/8/2004	N 30° 19.176	W 88° 08.460	Cedar Point Marsh	
7/8/2004	N 30° 20.238	W 88° 11.541	Cat Island	
7/8/2004	N 30° 20.228	W 88° 11.543	Cedar Point Marsh	
7/8/2004	N 30° 19.351	W 88° 08.425	Cedar Point Marsh	
7/8/2004	N 30° 19.302	W 88° 08.456	Cedar Point Marsh	
7/8/2004	N 30° 19.226	W 88° 08.472	Cedar Point Marsh	
7/8/2004	N 30° 19.085	W 88° 08.424	Cedar Point Marsh	
7/8/2004	N 30° 19.084	W 88° 08.421	Cedar Point Marsh	
7/8/2004	N 30° 19.082	W 88° 08.420	Cedar Point Marsh	
7/8/2004	N 30° 19.082	W 88° 08.420	Cedar Point Marsh	
7/8/2004	N 30° 19.080	W 88° 08.420	Cedar Point Marsh	

7/8/2004	N 30° 19.081	W 88° 08.420	Cedar Point Marsh	
7/8/2004	N 30° 19.081	W 88° 08.418	Cedar Point Marsh	
7/8/2004	N 30° 19.059	W 88° 08.409	Cedar Point Marsh	
7/8/2004	N 30° 19.046	W 88° 08.398	Cedar Point Marsh	
7/8/2004	N 30° 19.017	W 88° 08.387	Cedar Point Marsh	
7/8/2004	N 30° 19.107	W 88° 08.436	Cedar Point Marsh	
7/8/2004	N 30° 19.106	W 88° 08.436	Cedar Point Marsh	
7/8/2004	N 30° 19.110	W 88° 08.436	Cedar Point Marsh	
7/8/2004	N 30° 19.110	W 88° 08.439	Cedar Point Marsh	
7/8/2004	N 30° 19.115	W 88° 08.440	Cedar Point Marsh	
7/8/2004	N 30° 19.513	W 88° 08.317	Cedar Point Marsh	
7/8/2004	N 30° 19.513	W 88° 08.316	Cedar Point Marsh	
7/8/2004	N 30° 19.473	W 88° 08.285	Cedar Point Marsh	
7/22/2004	N 30° 19.156	W 88° 08.448	Cedar Point Marsh	
7/22/2004	N 30° 19.170	W 88° 08.456	Cedar Point Marsh	
7/22/2004	N 30° 19.154	W 88° 08.450	Cedar Point Marsh	
7/22/2004	N 30° 19.160	W 88° 08.423	Cedar Point Marsh	
7/22/2004	N 30° 19.105	W 88° 08.434	Cedar Point Marsh	
7/22/2004	N 30° 19.112	W 88° 08.439	Cedar Point Marsh	
7/22/2004	N 30° 19.032	W 88° 08.393	Cedar Point Marsh	
7/22/2004	N 30° 20.227	W 88° 11.542	North Cat Island	
7/22/2004	N 30° 20.226	W 88° 11.541	North Cat Island	
7/22/2004	N 30° 20.229	W 88° 11.544	North Cat Island	
7/22/2004	N 30° 20.238	W 88° 11.542	North Cat Island	
7/22/2004	N 30° 20.229	W 88° 11.543	North Cat Island	
7/22/2004	N 30° 20.229	W 88° 11.542	North Cat Island	
7/22/2004	N 30° 20.207	W 88° 11.550	North Cat Island	
7/22/2004	N 30° 20.176	W 88° 11.366	North Cat Island	
7/22/2004	N 30° 19.210	W 88° 08.468	Cat Island	
7/22/2004	N 30° 19.204	W 88° 08.469	Cat Island	
7/22/2004	N 30° 19.198	W 88° 08.467	Cat Island	
7/22/2004	N 30° 19.197	W 88° 08.466	Cat Island	
7/23/2004	N 30° 15.067	W 88° 07.466	Airport Marsh	
7/23/2004	N 30° 15.066	W 88° 07.466	Airport Marsh	
7/23/2004	N 30° 15.063	W 88° 07.467	Airport Marsh	
7/23/2004	N 30° 15.062	W 88° 07.469	Airport Marsh	
7/23/2004	N 30° 15.063	W 88° 07.468	Airport Marsh	
7/23/2004	N 30° 15.063	W 88° 07.468	Airport Marsh	
7/23/2004	N 30° 15.108	W 88° 08.063	Airport Marsh	
7/23/2004	N 30° 15.104	W 88° 08.065	Airport Marsh	
7/23/2004	N 30° 15.110	W 88° 08.064	Airport Marsh	

7/23/2004	N 30° 15.110	W 88° 08.064	Airport Marsh	
7/23/2004	N 30° 15.109	W 88° 08.067	Airport Marsh	
7/23/2004	N 30° 15.110	W 88° 08.64	Airport Marsh	
7/23/2004	N 30° 16.183	W 88° 07.062	Little Dauphin Island	

2005

Date	Latitude	Longitude	Location	Type of cover
5/25/2005	N 30° 18.837	W 88° 08.506	Cedar Point Marsh	
6/16/2005	N 30° 19.249	W 88° 08.746	Cedar Point Marsh	
6/16/2005	N 30° 19.161	W 88° 08.712	Cedar Point Marsh	
6/16/2005	N 30° 19.047	W 88° 08.645	Cedar Point Marsh	
6/16/2005	N 30° 19.315	W 88° 11.782	Cedar Point Marsh	
6/16/2005	N 30° 19.240	W 88° 11.053	Mon Louis Island	
6/21/2005	N 30° 19.200	W 88° 08.737	Cedar Point Marsh	
6/21/2005	N 30° 19.199	W 88° 08.737	Cedar Point Marsh	
6/21/2005	N 30° 19.133	W 88° 08.698	Cedar Point Marsh	
6/21/2005	N 30° 19.054	W 88° 08.654	Cedar Point Marsh	
6/21/2005	N 30° 19.190	W 88° 08.732	Cedar Point Marsh	
6/21/2005	N 30° 19.199	W 88° 08.739	Cedar Point Marsh	
6/21/2005	N 30° 19.207	W 88° 08.743	Cedar Point Marsh	
8/5/2005	N 30° 19.273	W 88° 08.755	Cedar Point Marsh	
8/5/2005	N 30° 19.163	W 88° 08.713	Cedar Point Marsh	
8/5/2005	N 30° 19.155	W 88° 08.707	Cedar Point Marsh	
8/5/2005	N 30° 19.192	W 88° 08.733	Cedar Point Marsh	
8/5/2005	N 30° 19.222	W 88° 08.746	Cedar Point Marsh	
8/5/2005	N 30° 19.229	W 88° 08.748	Cedar Point Marsh	

2006

Date	Latitude	Longitude	Location	Type of cover
5/15/2006	N 30° 19.203	W 88° 08.732	Cedar Point Marsh	
5/15/2006	N 30° 19.282	W 88° 08.753	Cedar Point Marsh	
5/15/2006	N 30° 19.428	W 88° 08.774	Cedar Point Marsh	
5/15/2006	N 30° 19.413	W 88° 08.785	Cedar Point Marsh	
5/15/2006	N 30° 19.188	W 88° 08.723	Cedar Point Marsh	
5/15/2006	N 30° 19.184	W 88° 08.721	Cedar Point Marsh	
5/15/2006	N 30° 19.174	W 88° 08.716	Cedar Point Marsh	
5/15/2006	N 30° 19.170	W 88° 08.714	Cedar Point Marsh	
5/15/2006	N 30° 18.899	W 88° 08.462	Cedar Point Marsh	

5/15/2006	N 30° 18.884	W 88° 08.497	Cedar Point Marsh	
5/15/2006	N 30° 19.099	W 88° 08.673	Cedar Point Marsh	
5/15/2006	N 30° 18.759	W 88° 08.458	Cedar Point Marsh	
5/17/2006	N 30° 19.208	W 88° 08.659	Cedar Point Marsh	
5/17/2006	N 30° 19.112	W 88° 08.607	Cedar Point Marsh	
5/17/2006	N 30° 19.337	W 88° 08.702	Cedar Point Marsh	
5/17/2006	N 30° 19.471	W 88° 08.726	Cedar Point Marsh	
5/22/2006	N 30° 19.207	W 88° 08.652	Cedar Point Marsh	
5/24/2006	N 30° 19.351	W 88° 08.715	Cedar Point Marsh	
5/24/2006	N 30° 19.216	W 88° 08.661	Cedar Point Marsh	
5/26/2006	N 30° 19.207	W 88° 08.654	Cedar Point Marsh	veg
5/28/2006	N 30° 18.987	W 88° 08.730	Cedar Point Marsh	
5/30/2006	N 30° 19.216	W 88° 08.666	Cedar Point Marsh	
6/4/2006	N 30° 19.192	W 88° 08.654	Cedar Point Marsh	
6/6/2006	N 30° 19.307	W 88° 08.690	Cedar Point Marsh	
6/6/2006	N 30° 19.200	W 88° 08.661	Cedar Point Marsh	
6/6/2006	N 30° 19.179	W 88° 08.643	Cedar Point Marsh	
6/8/2006	N 30° 18.814	W 88° 08.407	Cedar Point Marsh	
6/8/2006	N 30° 18.845	W 88° 08.421	Cedar Point Marsh	
6/8/2006	N 30° 19.178	W 88° 08.638	Cedar Point Marsh	
6/8/2006	N 30° 19.217	W 88° 08.661	Cedar Point Marsh	
6/8/2006	N 30° 19.217	W 88° 08.661	Cedar Point Marsh	
6/8/2006	N 30° 19.246	W 88° 08.681	Cedar Point Marsh	
6/8/2006	N 30° 19.388	W 88° 08.728	Cedar Point Marsh	
6/8/2006	N 30° 18.926	W 88° 08.486	Cedar Point Marsh	
6/11/2006	N 30° 18.786	W 88° 08.394	Cedar Point Marsh	
6/14/2006	N 30° 18.913	W 88° 08.473	Cedar Point Marsh	
6/14/2006	N 30° 19.217	W 88° 08.660	Cedar Point Marsh	
6/14/2006	N 30° 19.205	W 88° 08.655	Cedar Point Marsh	
6/16/2006	N 30° 19.107	W 88° 08.661	Cedar Point Marsh	
6/20/2006	N 30° 18.873	W 88° 08.440	Cedar Point Marsh	
6/20/2006	N 30° 18.886	W 88° 08.450	Cedar Point Marsh	
6/20/2006	N 30° 18.892	W 88° 08.457	Cedar Point Marsh	
6/20/2006	N 30° 18.903	W 88° 08.464	Cedar Point Marsh	
6/20/2006	N 30° 18.931	W 88° 08.487	Cedar Point Marsh	
6/20/2006	N 30° 19.110	W 88° 08.602	Cedar Point Marsh	
6/20/2006	N 30° 19.204	W 88° 08.654	Cedar Point Marsh	
6/20/2006	N 30° 19.230	W 88° 08.668	Cedar Point Marsh	
6/20/2006	N 30° 19.282	W 88° 08.693	Cedar Point Marsh	
6/20/2006	N 30° 19.302	W 88° 08.692	Cedar Point Marsh	
6/20/2006	N 30° 19.382	W 88° 08.719	Cedar Point Marsh	

6/20/2006	N 30° 19.448	W 88° 08.733	Cedar Point Marsh	
6/20/2006	N 30° 19.219	W 88° 08.665	Cedar Point Marsh	
6/20/2006	N 30° 19.176	W 88° 08.635	Cedar Point Marsh	
6/23/2006	N 30° 19.116	W 88° 08.608	Cedar Point Marsh	
6/23/2006	N 30° 19.244	W 88° 08.679	Cedar Point Marsh	
6/23/2006	N 30° 19.267	W 88° 08.690	Cedar Point Marsh	
6/23/2006	N 30° 19.348	W 88° 08.712	Cedar Point Marsh	
7/2/2006	N 30° 18.864	W 88° 08.432	Cedar Point Marsh	
7/2/2006	N 30° 18.893	W 88° 08.459	Cedar Point Marsh	
7/2/2006	N 30° 19.177	W 88° 08.635	Cedar Point Marsh	
7/2/2006	N 30° 19.204	W 88° 08.653	Cedar Point Marsh	
7/2/2006	N 30° 19.209	W 88° 08.654	Cedar Point Marsh	
7/2/2006	N 30° 19.210	W 88° 08.656	Cedar Point Marsh	
7/2/2006	N 30° 19.209	W 88° 08.660	Cedar Point Marsh	
7/2/2006	N 30° 19.206	W 88° 08.656	Cedar Point Marsh	
7/2/2006	N 30° 19.219	W 88° 08.662	Cedar Point Marsh	
7/2/2006	N 30° 19.220	W 88° 08.661	Cedar Point Marsh	
7/2/2006	N 30° 19.285	W 88° 08.692	Cedar Point Marsh	
7/2/2006	N 30° 19.285	W 88° 08.694	Cedar Point Marsh	
7/2/2006	N 30° 19.293	W 88° 08.692	Cedar Point Marsh	
7/2/2006	N 30° 19.353	W 88° 08.703	Cedar Point Marsh	
7/2/2006	N 30° 19.339	W 88° 08.708	Cedar Point Marsh	
7/6/2006	N 30° 18.346	W 88° 08.714	Cedar Point Marsh	
7/6/2006	N 30° 18.359	W 88° 08.716	Cedar Point Marsh	
7/6/2006	N 30° 18.874	W 88° 08.445	Cedar Point Marsh	
7/6/2006	N 30° 18.989	W 88° 08.529	Cedar Point Marsh	
7/6/2006	N 30° 19.210	W 88° 08.658	Cedar Point Marsh	
7/6/2006	N 30° 19.317	W 88° 08.693	Cedar Point Marsh	
7/9/2006	N 30° 18.701	W 88° 08.314	Cedar Point Marsh	
7/9/2006	N 30° 18.949	W 88° 08.496	Cedar Point Marsh	
7/6/2006	N 30° 19.220	W 88° 08.663	Cedar Point Marsh	
7/11/2006	N 30° 18.790	W 88° 08.398	Cedar Point Marsh	
7/11/2006	N 30° 19.237	W 88° 08.674	Cedar Point Marsh	
7/11/2006	N 30° 19.314	W 88° 08.694	Cedar Point Marsh	
7/14/2006	N 30° 19.203	W 88° 08.651	Cedar Point Marsh	
7/14/2006	N 30° 19.223	W 88° 08.665	Cedar Point Marsh	
7/14/2006	N 30° 19.319	W 88° 08.699	Cedar Point Marsh	
7/14/2006	N 30° 19.335	W 88° 08.704	Cedar Point Marsh	
7/17/2006	N 30° 19.335	W 88° 08.704	Cedar Point Marsh	
7/17/2006	N 30° 19.256	W 88° 08.683	Cedar Point Marsh	
7/17/2006	N 30° 19.293	W 88° 08.693	Cedar Point Marsh	

7/17/2006	N 30° 19.318	W 88° 08.698	Cedar Point Marsh	
7/21/2006	N 30° 19.299	W 88° 08.697	Cedar Point Marsh	
7/21/2006	N 30° 19.259	W 88° 08.693	Cedar Point Marsh	
7/21/2006	N 30° 19.219	W 88° 08.669	Cedar Point Marsh	
7/21/2006	N 30° 19.549	W 88° 08.707	Cedar Point Marsh	
7/21/2006	N 30° 19.552	W 88° 08.708	Cedar Point Marsh	
7/21/2006	N 30° 19.564	W 88° 08.706	Cedar Point Marsh	
7/21/2006	N 30° 19.644	W 88° 08.652	Cedar Point Marsh	
7/21/2006	N 30° 19.637	W 88° 08.654	Cedar Point Marsh	
7/21/2006	N 30° 19.638	W 88° 08.654	Cedar Point Marsh	
7/21/2006	N 30° 19.630	W 88° 08.578	Cedar Point Marsh	
7/21/2006	N 30° 19.631	W 88° 08.582	Cedar Point Marsh	
7/21/2006	N 30° 19.636	W 88° 08.580	Cedar Point Marsh	
7/21/2006	N 30° 19.638	W 88° 08.582	Cedar Point Marsh	
7/21/2006	N 30° 19.640	W 88° 08.584	Cedar Point Marsh	
7/26/2006	N 30° 19.218	W 88° 08.660	Cedar Point Marsh	
7/31/2006	N 30° 19.201	W 88° 08.659	Cedar Point Marsh	
8/3/2006	N 30° 19.220	W 88° 08.667	Cedar Point Marsh	
8/3/2006	N 30° 19.222	W 88° 08.669	Cedar Point Marsh	
8/3/2006	N 30° 19.301	W 88° 08.696	Cedar Point Marsh	
7/26/2006	N 30° 19.	W 88° 08.	Cedar Point Marsh	
7/26/2006	N 30° 19.	W 88° 08.	Cedar Point Marsh	
7/26/2006	N 30° 19.	W 88° 08.	Cedar Point Marsh	
7/26/2006	N 30° 19.	W 88° 08.	Cedar Point Marsh	
7/26/2006	N 30° 19.	W 88° 08.	Cedar Point Marsh	
7/26/2006	N 30° 19.	W 88° 08.	Cedar Point Marsh	
7/26/2006	N 30° 19.	W 88° 08.	Cedar Point Marsh	
7/26/2006	N 30° 19.	W 88° 08.	Cedar Point Marsh	
7/26/2006	N 30° 19.	W 88° 08.	Cedar Point Marsh	
7/26/2006	N 30° 19.	W 88° 08.	Cedar Point Marsh	
7/26/2006	N 30° 19.	W 88° 08.	Cedar Point Marsh	
7/26/2006	N 30° 19.	W 88° 08.	Cedar Point Marsh	
7/26/2006	N 30° 19.	W 88° 08.	Cedar Point Marsh	
7/26/2006	N 30° 19.	W 88° 08.	Cedar Point Marsh	
7/26/2006	N 30° 19.	W 88° 08.	Cedar Point Marsh	
7/26/2006	N 30° 19.	W 88° 08.	Cedar Point Marsh	
7/26/2006	N 30° 19.	W 88° 08.	Cedar Point Marsh	
7/26/2006	N 30° 19.	W 88° 08.	Cedar Point Marsh	

2007

Date	Lattitude	Longitude	Location	Type of cover
5/21/2007	N 30° 18.823	W 88° 08.408	Cedar Point Marsh	
6/14/2007	N 30° 19.274	W 88° 08.696	Cedar Point Marsh	
6/14/2007	N 30° 19.274	W 88° 08.694	Cedar Point Marsh	
6/14/2007	N 30° 19.303	W 88° 08.692	Cedar Point Marsh	
6/14/2007	N 30° 19.386	W 88° 08.722	Cedar Point Marsh	
6/19/2007	N 30° 19.205	W 88° 08.650	Cedar Point Marsh	
6/19/2007	N 30° 19.210	W 88° 08.659	Cedar Point Marsh	
6/19/2007	N 30° 19.298	W 88° 08.690	Cedar Point Marsh	
6/26/2007	N 30° 19.274	W 88° 08.701	Cedar Point Marsh	
6/26/2007	N 30° 19.204	W 88° 08.657	Cedar Point Marsh	
6/26/2007	N 30° 19.174	W 88° 08.640	Cedar Point Marsh	
6/26/2007	N 30° 19.180	W 88° 08.643	Cedar Point Marsh	
6/26/2007	N 30° 19.408	W 88° 08.731	Cedar Point Marsh	
7/10/2007	N 30° 19.241	W 88° 08.682	Cedar Point Marsh	
7/10/2007	N 30° 19.075	W 88° 08.581	Cedar Point Marsh	
7/10/2007	N 30° 19.225	W 88° 08.671	Cedar Point Marsh	
7/10/2007	N 30° 19.261	W 88° 08.690	Cedar Point Marsh	
7/10/2007	N 30° 19.285	W 88° 08.626	Cedar Point Marsh	
7/10/2007	N 30° 19.290	W 88° 08.695	Cedar Point Marsh	
7/10/2007	N 30° 19.407	W 88° 08.727	Cedar Point Marsh	
7/17/2007	N 30° 19.203	W 88° 08.655	Cedar Point Marsh	
7/17/2007	N 30° 19.237	W 88° 08.678	Cedar Point Marsh	
7/17/2007	N 30° 19.239	W 88° 08.681	Cedar Point Marsh	
7/17/2007	N 30° 19.317	W 88° 08.700	Cedar Point Marsh	
7/17/2007	N 30° 19.852	W 88° 08.430	Cedar Point Marsh	
7/17/2007	N 30° 19.973	W 88° 08.515	Cedar Point Marsh	
7/17/2007	N 30° 19.010	W 88° 08.542	Cedar Point Marsh	
7/24/2007	N 30° 19.171	W 88° 08.638	Cedar Point Marsh	
7/24/2007	N 30° 19.167	W 88° 08.640	Cedar Point Marsh	
7/24/2007	N 30° 19.155	W 88° 08.629	Cedar Point Marsh	
9/14/2007	N 30° 19.422	W 88° 08.733	Cedar Point Marsh	
9/14/2007	N 30° 19.333	W 88° 08.709	Cedar Point Marsh	
9/14/2007	N 30° 19.327	W 88° 08.709	Cedar Point Marsh	
9/14/2007	N 30° 19.290	W 88° 08.644	Cedar Point Marsh	
9/14/2007	N 30° 19.253	W 88° 08.692	Cedar Point Marsh	
9/14/2007	N 30° 19.196	W 88° 08.654	Cedar Point Marsh	

2008

Date	Lattitude	Longitude	Location	Type of cover
5/7/2008	N 30° 19.277	W 88° 08.696	Cedar Point Marsh	
5/7/2008	N 30° 19.202	W 88° 08.655	Cedar Point Marsh	
5/7/2008	N 30° 19.196	W 88° 08.650	Cedar Point Marsh	
5/7/2008	N 30° 19.190	W 88° 08.645	Cedar Point Marsh	
5/12/2008	N 30° 19.196	W 88° 08.684	Cedar Point Marsh	
5/19/2008	N 30° 19.279	W 88° 08.695	Cedar Point Marsh	
5/19/2008	N 30° 19.261	W 88° 08.692	Cedar Point Marsh	
5/19/2008	N 30° 19.251	W 88° 08.680	Cedar Point Marsh	
5/19/2008	N 30° 19.350	W 88° 08.714	Cedar Point Marsh	
5/21/2008	N 30° 19.348	W 88° 08.715	Cedar Point Marsh	
6/3/2008	N 30° 19.200	W 88° 08.652	Cedar Point Marsh	
6/6/2008	N 30° 19.284	W 88° 08.703	Cedar Point Marsh	
6/6/2008	N 30° 19.287	W 88° 08.704	Cedar Point Marsh	
6/6/2008	N 30° 19.627	W 88° 08.580	Cedar Point Marsh	
6/6/2008	N 30° 19.658	W 88° 08.639	Cedar Point Marsh	veg
6/6/2008	N 30° 19.635	W 88° 08.654	Cedar Point Marsh	
6/6/2008	N 30° 19.642	W 88° 08.649	Cedar Point Marsh	
6/10/2008	N 30° 19.240	W 88° 08.678	Cedar Point Marsh	
6/13/2008	N 30° 19.220	W 88° 08.664	Cedar Point Marsh	
6/13/2008	N 30° 19.227	W 88° 08.668	Cedar Point Marsh	
6/17/2008	N 30° 19.174	W 88° 08.634	Cedar Point Marsh	
6/17/2008	N 30° 19.170	W 88° 08.631	Cedar Point Marsh	
6/17/2008	N 30° 19.148	W 88° 08.622	Cedar Point Marsh	
6/19/2008	N 30° 19.298	W 88° 08.692	Cedar Point Marsh	
6/19/2008	N 30° 19.341	W 88° 08.707	Cedar Point Marsh	veg
6/19/2008	N 30° 19.442	W 88° 08.731	Cedar Point Marsh	veg
6/19/2008	N 30° 19.455	W 88° 08.732	Cedar Point Marsh	veg
6/20/2008	N 30° 15.554	W 88° 07.641	Airport Marsh	
6/20/2008	N 30° 15.552	W 88° 07.639	Airport Marsh	
6/23/2008	N 30° 19.191	W 88° 08.643	Cedar Point Marsh	
6/24/2008	N 30° 18.837	W 88° 08.414	Cedar Point Marsh	none
6/24/2008	N 30° 19.109	W 88° 08.594	Cedar Point Marsh	none
6/24/2008	N 30° 19.117	W 88° 08.610	Cedar Point Marsh	veg
6/24/2008	N 30° 19.173	W 88° 08.635	Cedar Point Marsh	veg
6/24/2008	N 30° 19.173	W 88° 08.635	Cedar Point Marsh	veg
6/24/2008	N 30° 19.272	W 88° 08.694	Cedar Point Marsh	veg
6/24/2008	N 30° 19.568	W 88° 08.611	Cedar Point Marsh	
6/24/2008	N 30° 19.453	W 88° 08.594	Cedar Point Marsh	

6/24/2008	N 30° 19.446	W 88° 08.638	Cedar Point Marsh	
6/24/2008	N 30° 19.440	W 88° 08.572	Cedar Point Marsh	
7/1/2008	N 30° 19.420	W 88° 08.730	Cedar Point Marsh	
7/1/2008	N 30° 19.375	W 88° 08.721	Cedar Point Marsh	
7/3/2008	N 30° 18.891	W 88° 08.458	Cedar Point Marsh	none
7/3/2008	N 30° 19.420	W 88° 08.732	Cedar Point Marsh	veg
7/3/2008	N 30° 19.391	W 88° 08.746	Cedar Point Marsh	veg
7/3/2008	N 30° 19.349	W 88° 08.720	Cedar Point Marsh	veg
7/3/2008	N 30° 18.815	W 88° 08.407	Cedar Point Marsh	veg
7/3/2008	N 30° 18.806	W 88° 08.407	Cedar Point Marsh	veg
7/3/2008	N 30° 18.779	W 88° 08.399	Cedar Point Marsh	veg
7/6/2008	N 30° 18.865	W 88° 08.434	Cedar Point Marsh	
7/6/2008	N 30° 19.119	W 88° 08.610	Cedar Point Marsh	
7/8/2008	N 30° 19.282	W 88° 08.675	Cedar Point Marsh	veg
7/8/2008	N 30° 19.205	W 88° 08.655	Cedar Point Marsh	veg
7/8/2008	N 30° 19.452	W 88° 08.735	Cedar Point Marsh	veg
7/8/2008	N 30° 19.228	W 88° 08.667	Cedar Point Marsh	veg
7/9/2008	N 30° 18.950	W 88° 08.502	Cedar Point Marsh	
7/9/2008	N 30° 19.101	W 88° 08.594	Cedar Point Marsh	
7/9/2008	N 30° 19.107	W 88° 08.601	Cedar Point Marsh	
7/10/2008	N 30° 19.139	W 88° 08.398	Cedar Point Marsh	
7/10/2008	N 30° 19.156	W 88° 08.410	Cedar Point Marsh	
7/10/2008	N 30° 19.164	W 88° 08.415	Cedar Point Marsh	
7/10/2008	N 30° 19.174	W 88° 08.418	Cedar Point Marsh	
7/10/2008	N 30° 19.203	W 88° 08.652	Cedar Point Marsh	
7/10/2008	N 30° 19.235	W 88° 08.676	Cedar Point Marsh	
7/10/2008	N 30° 19.261	W 88° 08.690	Cedar Point Marsh	
7/12/2008	N 30° 19.333	W 88° 08.702	Cedar Point Marsh	
7/12/2008	N 30° 19.333	W 88° 08.702	Cedar Point Marsh	
7/16/2008	N 30° 19.232	W 88° 08.668	Cedar Point Marsh	
7/16/2008	N 30° 19.257	W 88° 08.682	Cedar Point Marsh	
7/16/2008	N 30° 19.262	W 88° 08.687	Cedar Point Marsh	
7/16/2008	N 30° 19.445	W 88° 08.731	Cedar Point Marsh	
7/16/2008	N 30° 19.369	W 88° 08.722	Cedar Point Marsh	
7/16/2008	N 30° 19.280	W 88° 08.694	Cedar Point Marsh	
7/16/2008	N 30° 15.566	W 88° 07.641	Airport Marsh	
7/16/2008	N 30° 15.554	W 88° 07.642	Airport Marsh	
7/17/2008	N 30° 19.332	W 88° 08.702	Cedar Point Marsh	
7/17/2008	N 30° 20.437	W 88° 11.850	Mon Louis	
7/17/2008	N 30° 20.425	W 88° 11.849	Mon Louis	
7/17/2008	N 30° 20.425	W 88° 11.850	Mon Louis	

7/17/2008	N 30° 20.407	W 88° 11.855	Mon Louis	
7/17/2008	N 30° 20.395	W 88° 11.857	Mon Louis	
7/17/2008	N 30° 20.369	W 88° 11.863	Mon Louis	
7/17/2008	N 30° 20.357	W 88° 11.862	Mon Louis	
7/17/2008	N 30° 20.410	W 88° 11.854	Mon Louis	
7/17/2008	N 30° 20.424	W 88° 11.852	Mon Louis	
7/17/2008	N 30° 20.445	W 88° 11.847	Mon Louis	
7/17/2008	N 30° 22.283	W 88° 18.300	Point Aux Pines	
7/17/2008	N 30° 22.287	W 88° 18.310	Point Aux Pines	
7/17/2008	N 30° 22.291	W 88° 18.315	Point Aux Pines	
7/17/2008	N 30° 22.286	W 88° 18.314	Point Aux Pines	
7/17/2008	N 30° 22.293	W 88° 18.317	Point Aux Pines	
7/17/2008	N 30° 22.293	W 88° 18.322	Point Aux Pines	
7/17/2008	N 30° 22.293	W 88° 18.322	Point Aux Pines	
7/17/2008	N 30° 22.249	W 88° 18.323	Point Aux Pines	
7/17/2008	N 30° 22.395	W 88° 18.324	Point Aux Pines	
7/21/2008	N 30° 19.332	W 88° 08.702	Cedar Point Marsh	
7/22/2008	N 30° 19.270	W 88° 08.700	Cedar Point Marsh	
7/22/2008	N 30° 19.284	W 88° 08.695	Cedar Point Marsh	
7/22/2008	N 30° 19.437	W 88° 08.731	Cedar Point Marsh	
7/22/2008	N 30° 19.254	W 88° 08.685	Cedar Point Marsh	
7/22/2008	N 30° 19.171	W 88° 08.633	Cedar Point Marsh	
7/22/2008	N 30° 19.285	W 88° 08.694	Cedar Point Marsh	
7/23/2008	N 30° 18.827	W 88° 08.409	Cedar Point Marsh	
7/23/2008	N 30° 19.228	W 88° 08.669	Cedar Point Marsh	
7/23/2008	N 30° 19.105	W 88° 08.600	Cedar Point Marsh	
7/23/2008	N 30° 19.103	W 88° 08.595	Cedar Point Marsh	
7/24/2008	N 30° 18.825	W 88° 08.411	Cedar Point Marsh	veg
7/24/2008	N 30° 18.826	W 88° 08.413	Cedar Point Marsh	veg
7/24/2008	N 30° 18.830	W 88° 08.414	Cedar Point Marsh	veg
7/24/2008	N 30° 19.115	W 88° 08.600	Cedar Point Marsh	veg
7/24/2008	N 30° 19.172	W 88° 08.636	Cedar Point Marsh	veg
7/24/2008	N 30° 19.211	W 88° 08.638	Cedar Point Marsh	veg
7/24/2008	N 30° 19.300	W 88° 08.691	Cedar Point Marsh	none
7/24/2008	N 30° 19.016	W 88° 08.545	Cedar Point Marsh	veg
7/29/2008	N 30° 19.233	W 88° 08.669	Cedar Point Marsh	none
7/29/2008	N 30° 19.363	W 88° 08.714	Cedar Point Marsh	veg
7/29/2008	N 30° 19.402	W 88° 08.730	Cedar Point Marsh	veg
7/31/2008	N 30° 19.170	W 88° 08.636	Cedar Point Marsh	veg
7/31/2008	N 30° 19.417	W 88° 08.724	Cedar Point Marsh	veg
7/31/2008	N 30° 19.145	W 88° 08.623	Cedar Point Marsh	veg

7/31/2008	N 30° 19.116	W 88° 08.609	Cedar Point Marsh	veg
7/31/2008	N 30° 19.114	W 88° 08.612	Cedar Point Marsh	none
7/31/2008	N 30° 19.157	W 88° 08.411	Cedar Point Marsh	veg
7/31/2008	N 30° 19.157	W 88° 08.411	Cedar Point Marsh	none
7/31/2008	N 30° 19.157	W 88° 08.411	Cedar Point Marsh	veg
7/31/2008	N 30° 19.157	W 88° 08.411	Cedar Point Marsh	veg
7/31/2008	N 30° 19.157	W 88° 08.411	Cedar Point Marsh	veg
7/31/2008	N 30° 19.157	W 88° 08.411	Cedar Point Marsh	veg
7/31/2008	N 30° 19.157	W 88° 08.411	Cedar Point Marsh	none
7/31/2008	N 30° 19.157	W 88° 08.411	Cedar Point Marsh	veg
7/31/2008	N 30° 19.157	W 88° 08.411	Cedar Point Marsh	none
7/31/2008	N 30° 19.157	W 88° 08.411	Cedar Point Marsh	veg
7/31/2008	N 30° 19.157	W 88° 08.411	Cedar Point Marsh	none
8/6/2008	N 30° 23.004	W 88° 22.829	Barton Island	
8/6/2008	N 30° 23.016	W 88° 22.843	Cedar Point Marsh	
8/6/2008	N 30° 23.021	W 88° 88.843	Cedar Point Marsh	

2009

Date	Lattitude	Longitude	Location	Type of cover
5/15/2009	N 30° 19.274	W 88° 08.704	Cedar Point Marsh	
5/15/2009	N 30° 19.275	W 88° 08.701	Cedar Point Marsh	
5/15/2009	N 30° 19.279	W 88° 08.702	Cedar Point Marsh	
5/15/2009	N 30° 19.284	W 88° 08.699	Cedar Point Marsh	
5/15/2009	N 30° 19.300	W 88° 08.698	Cedar Point Marsh	
5/15/2009	N 30° 19.319	W 88° 08.700	Cedar Point Marsh	
5/18/2009	N 30° 18.828	W 88° 08.409	Cedar Point Marsh	
5/18/2009	N 30° 18.850	W 88° 08.430	Cedar Point Marsh	
5/18/2009	N 30° 18.852	W 88° 08.427	Cedar Point Marsh	
5/18/2009	N 30° 18.853	W 88° 08.428	Cedar Point Marsh	
5/18/2009	N 30° 18.926	W 88° 08.484	Cedar Point Marsh	
5/18/2009	N 30° 18.969	W 88° 08.517	Cedar Point Marsh	
5/18/2009	N 30° 18.987	W 88° 08.530	Cedar Point Marsh	
5/18/2009	N 30° 18.997	W 88° 08.533	Cedar Point Marsh	
5/18/2009	N 30° 19.016	W 88° 08.540	Cedar Point Marsh	
5/18/2009	N 30° 19.017	W 88° 08.540	Cedar Point Marsh	
5/18/2009	N 30° 19.075	W 88° 08.550	Cedar Point Marsh	
5/18/2009	N 30° 19.147	W 88° 08.621	Cedar Point Marsh	
5/20/2009	N 30° 19.319	W 88° 08.699	Cedar Point Marsh	
5/20/2009	N 30° 19.325	W 88° 08.697	Cedar Point Marsh	

5/20/2009	N 30° 19.340	W 88° 08.702	Cedar Point Marsh	
5/20/2009	N 30° 19.341	W 88° 08.707	Cedar Point Marsh	
5/20/2009	N 30° 19.384	W 88° 08.720	Cedar Point Marsh	
5/29/2009	N 30° 19.449	W 88° 08.739	Cedar Point Marsh	
5/29/2009	N 30° 19.430	W 88° 08.736	Cedar Point Marsh	
5/31/2009	N 30° 18.909	W 88° 08.469	Cedar Point Marsh	
5/31/2009	N 30° 18.920	W 88° 08.479	Cedar Point Marsh	
6/3/2009	N 30° 19.340	W 88° 08.703	Cedar Point Marsh	veg
6/4/2009	N 30° 18.835	W 88° 08.422	Cedar Point Marsh	veg
6/4/2009	N 30° 18.892	W 88° 08.462	Cedar Point Marsh	veg
6/4/2009	N 30° 18.950	W 88° 08.504	Cedar Point Marsh	veg
6/4/2009	N 30° 18.969	W 88° 08.518	Cedar Point Marsh	veg
6/4/2009	N 30° 19.013	W 88° 08.540	Cedar Point Marsh	veg
6/4/2009	N 30° 19.258	W 88° 08.684	Cedar Point Marsh	veg
6/4/2009	N 30° 19.291	W 88° 08.695	Cedar Point Marsh	veg
6/4/2009	N 30° 19.336	W 88° 08.703	Cedar Point Marsh	veg
6/4/2009	N 30° 19.383	W 88° 08.723	Cedar Point Marsh	veg
6/4/2009	N 30° 19.444	W 88° 08.730	Cedar Point Marsh	
6/4/2009	N 30° 19.452	W 88° 08.739	Cedar Point Marsh	veg
6/4/2009	N 30° 19.451	W 88° 08.734	Cedar Point Marsh	veg
6/4/2009	N 30° 18.991	W 88° 08.530	Cedar Point Marsh	veg
6/4/2009	N 30° 18.867	W 88° 08.437	Cedar Point Marsh	veg
6/5/2009	N 30° 18.829	W 88° 08.410	Cedar Point Marsh	veg
6/11/2009	N 30° 19.106	W 88° 08.600	Cedar Point Marsh	veg
6/11/2009	N 30° 19.075	W 88° 08.576	Cedar Point Marsh	veg
6/11/2009	N 30° 19.257	W 88° 08.682	Cedar Point Marsh	veg
6/11/2009	N 30° 19.280	W 88° 08.692	Cedar Point Marsh	veg
6/11/2009	N 30° 19.618	W 88° 08.569	Cedar Point Marsh	veg
6/11/2009	N 30° 19.631	W 88° 08.565	Cedar Point Marsh	veg
6/11/2009	N 30° 19.630	W 88° 08.573	Cedar Point Marsh	veg
6/11/2009	N 30° 19.632	W 88° 08.573	Cedar Point Marsh	riprap
6/11/2009	N 30° 19.655	W 88° 08.618	Cedar Point Marsh	veg
6/11/2009	N 30° 19.655	W 88° 08.612	Cedar Point Marsh	veg
6/11/2009	N 30° 19.659	W 88° 08.622	Cedar Point Marsh	veg
6/11/2009	N 30° 19.646	W 88° 08.637	Cedar Point Marsh	none
6/11/2009	N 30° 19.637	W 88° 08.647	Cedar Point Marsh	none
6/11/2009	N 30° 19.635	W 88° 08.647	Cedar Point Marsh	none
6/11/2009	N 30° 19.635	W 88° 08.648	Cedar Point Marsh	none
6/11/2009	N 30° 19.363	W 88° 08.646	Cedar Point Marsh	none
6/11/2009	N 30° 19.643	W 88° 08.645	Cedar Point Marsh	veg
6/11/2009	N 30° 19.296	W 88° 08.690	Cedar Point Marsh	veg

6/11/2009	N 30° 19.328	W 88° 08.699	Cedar Point Marsh	veg
6/11/2009	N 30° 19.345	W 88° 08.707	Cedar Point Marsh	veg
6/11/2009	N 30° 19.351	W 88° 08.718	Cedar Point Marsh	veg
6/11/2009	N 30° 19.366	W 88° 08.716	Cedar Point Marsh	veg
6/15/2009	N 30° 19.234	W 88° 08.668	Cedar Point Marsh	none
6/16/2009	N 30° 19.327	W 88° 08.707	Cedar Point Marsh	veg
6/16/2009	N 30° 19.334	W 88° 08.701	Cedar Point Marsh	veg
6/18/2009	N 30° 19.174	W 88° 08.637	Cedar Point Marsh	veg
6/18/2009	N 30° 19.147	W 88° 08.627	Cedar Point Marsh	none
6/18/2009	N 30° 18.894	W 88° 08.465	Cedar Point Marsh	veg
6/18/2009	N 30° 18.889	W 88° 08.461	Cedar Point Marsh	veg
6/18/2009	N 30° 18.885	W 88° 08.451	Cedar Point Marsh	veg
6/18/2009	N 30° 19.273	W 88° 08.690	Cedar Point Marsh	veg
6/18/2009	N 30° 19.268	W 88° 08.698	Cedar Point Marsh	veg
6/18/2009	N 30° 19.361	W 88° 08.717	Cedar Point Marsh	veg
6/18/2009	N 30° 19.375	W 88° 08.722	Cedar Point Marsh	veg
6/18/2009	N 30° 19.373	W 88° 08.720	Cedar Point Marsh	veg
6/24/2009	N 30° 19.275	W 88° 08.692	Cedar Point Marsh	veg
6/24/2009	N 30° 18.292	W 88° 08.485	Cedar Point Marsh	veg
6/24/2009	N 30° 18.943	W 88° 08.492	Cedar Point Marsh	
6/24/2009	N 30° 18.968	W 88° 08.514	Cedar Point Marsh	veg
6/25/2009	N 30° 18.819	W 88° 08.407	Cedar Point Marsh	veg
6/25/2009	N 30° 18.823	W 88° 08.411	Cedar Point Marsh	veg
6/25/2009	N 30° 18.828	W 88° 08.414	Cedar Point Marsh	veg
6/25/2009	N 30° 18.929	W 88° 08.486	Cedar Point Marsh	
6/25/2009	N 30° 18.924	W 88° 08.492	Cedar Point Marsh	
6/25/2009	N 30° 18.969	W 88° 08.515	Cedar Point Marsh	
6/25/2009	N 30° 18.968	W 88° 08.516	Cedar Point Marsh	veg
6/25/2009	N 30° 19.012	W 88° 08.538	Cedar Point Marsh	veg
6/25/2009	N 30° 19.013	W 88° 08.536	Cedar Point Marsh	none
6/25/2009	N 30° 19.018	W 88° 08.539	Cedar Point Marsh	veg
6/25/2009	N 30° 19.287	W 88° 08.692	Cedar Point Marsh	veg
6/25/2009	N 30° 19.313	W 88° 08.694	Cedar Point Marsh	veg
6/25/2009	N 30° 19.343	W 88° 08.708	Cedar Point Marsh	veg
6/25/2009	N 30° 19.352	W 88° 08.715	Cedar Point Marsh	veg
6/25/2009	N 30° 19.371	W 88° 08.716	Cedar Point Marsh	veg
6/25/2009	N 30° 19.376	W 88° 08.718	Cedar Point Marsh	veg
6/25/2009	N 30° 18.856	W 88° 08.428	Cedar Point Marsh	veg
7/3/2009	N 30° 19.109	W 88° 08.610	Cedar Point Marsh	none
7/3/2009	N 30° 19.081	W 88° 08.590	Cedar Point Marsh	veg
7/3/2009	N 30° 19.077	W 88° 08.586	Cedar Point Marsh	veg

7/3/2009	N 30° 19.075	W 88° 08.585	Cedar Point Marsh	veg
7/3/2009	N 30° 18.988	W 88° 08.535	Cedar Point Marsh	veg
7/3/2009	N 30° 18.896	W 88° 08.467	Cedar Point Marsh	veg
7/3/2009	N 30° 18.968	W 88° 08.519	Cedar Point Marsh	veg
7/3/2009	N 30° 19.241	W 88° 08.677	Cedar Point Marsh	none
7/3/2009	N 30° 19.357	W 88° 08.720	Cedar Point Marsh	veg
7/3/2009	N 30° 19.381	W 88° 08.724	Cedar Point Marsh	veg
7/3/2009	N 30° 19.399	W 88° 08.733	Cedar Point Marsh	veg
7/3/2009	N 30° 19.440	W 88° 08.727	Cedar Point Marsh	veg
7/3/2009	N 30° 19.452	W 88° 08.740	Cedar Point Marsh	veg
7/3/2009	N 30° 19.286	W 88° 08.698	Cedar Point Marsh	veg
7/10/2009	N 30° 18.752	W 88° 08.364	Cedar Point Marsh	veg
7/10/2009	N 30° 19.078	W 88° 08.578	Cedar Point Marsh	veg
7/10/2009	N 30° 19.214	W 88° 08.656	Cedar Point Marsh	none
7/10/2009	N 30° 19.298	W 88° 08.691	Cedar Point Marsh	veg
7/17/2009	N 30° 19.343	W 88° 08.715	Cedar Point Marsh	
7/17/2009	N 30° 19.360	W 88° 08.720	Cedar Point Marsh	
7/17/2009	N 30° 19.377	W 88° 08.728	Cedar Point Marsh	
7/17/2009	N 30° 19.401	W 88° 08.734	Cedar Point Marsh	
7/17/2009	N 30° 19.146	W 88° 08.629	Cedar Point Marsh	
7/17/2009	N 30° 19.226	W 88° 08.671	Cedar Point Marsh	
7/21/2009	N 30° 19.280	W 88° 08.642	Cedar Point Marsh	
7/23/2009	N 30° 19.104	W 88° 08.399	Cedar Point Marsh	none
7/23/2009	N 30° 18.832	W 88° 08.414	Cedar Point Marsh	veg
7/23/2009	N 30° 18.818	W 88° 08.408	Cedar Point Marsh	veg
7/23/2009	N 30° 19.147	W 88° 08.626	Cedar Point Marsh	
7/30/2009	N 30° 19.115	W 88° 08.616	Cedar Point Marsh	veg
7/30/2009	N 30° 18.801	W 88° 08.411	Cedar Point Marsh	veg
7/30/2009	N 30° 19.659	W 88° 08.596	Cedar Point Marsh	veg
7/30/2009	N 30° 19.659	W 88° 08.611	Cedar Point Marsh	veg
7/30/2009	N 30° 19.658	W 88° 08.626	Cedar Point Marsh	veg
7/30/2009	N 30° 19.658	W 88° 08.627	Cedar Point Marsh	veg
7/30/2009	N 30° 19.643	W 88° 08.645	Cedar Point Marsh	veg
7/30/2009	N 30° 19.643	W 88° 08.645	Cedar Point Marsh	veg
7/30/2009	N 30° 19.639	W 88° 08.648	Cedar Point Marsh	veg
7/30/2009	N 30° 19.638	W 88° 08.649	Cedar Point Marsh	veg
7/30/2009	N 30° 19.638	W 88° 08.649	Cedar Point Marsh	veg
7/30/2009	N 30° 19.635	W 88° 08.650	Cedar Point Marsh	veg
7/30/2009	N 30° 19.640	W 88° 08.581	Cedar Point Marsh	veg
7/30/2009	N 30° 19.635	W 88° 08.578	Cedar Point Marsh	veg
7/30/2009	N 30° 19.626	W 88° 08.573	Cedar Point Marsh	veg

7/30/2009	N 30° 19.622	W 88° 08.574	Cedar Point Marsh	veg
7/30/2009	N 30° 19.629	W 88° 08.575	Cedar Point Marsh	veg
7/30/2009	N 30° 19.619	W 88° 08.573	Cedar Point Marsh	veg

2010

Date	Lattitude	Longitude	Location	Type of cover
5/7/2010	N 30° 18.819	W 88° 08.410	Cedar Point Marsh	veg
5/7/2010	N 30° 18.856	W 88° 08.427	Cedar Point Marsh	none
5/7/2010	N 30° 18.933	W 88° 08.486	Cedar Point Marsh	veg
5/7/2010	N 30° 19.072	W 88° 08.574	Cedar Point Marsh	veg
5/7/2010	N 30° 19.106	W 88° 08.600	Cedar Point Marsh	none
5/7/2010	N 30° 19.193	W 88° 08.639	Cedar Point Marsh	none
5/7/2010	N 30° 19.249	W 88° 08.671	Cedar Point Marsh	veg
5/7/2010	N 30° 19.277	W 88° 08.692	Cedar Point Marsh	none
5/7/2010	N 30° 19.332	W 88° 08.699	Cedar Point Marsh	veg
5/7/2010	N 30° 19.335	W 88° 08.701	Cedar Point Marsh	veg
5/7/2010	N 30° 19.412	W 88° 08.724	Cedar Point Marsh	veg
5/7/2010	N 30° 19.332	W 88° 08.698	Cedar Point Marsh	veg
5/12/2010	N 30° 19.336	W 88° 08.701	Cedar Point Marsh	
5/12/2010	N 30° 19.332	W 88° 08.703	Cedar Point Marsh	
5/17/2010	N 30° 18.817	W 88° 08.407	Cedar Point Marsh	veg
5/17/2010	N 30° 18.830	W 88° 08.410	Cedar Point Marsh	none
5/17/2010	N 30° 18.838	W 88° 08.417	Cedar Point Marsh	
5/17/2010	N 30° 18.866	W 88° 08.435	Cedar Point Marsh	none
5/17/2010	N 30° 18.888	W 88° 08.449	Cedar Point Marsh	veg
5/17/2010	N 30° 19.090	W 88° 08.593	Cedar Point Marsh	veg
5/17/2010	N 30° 19.090	W 88° 08.593	Cedar Point Marsh	veg
5/17/2010	N 30° 19.269	W 88° 08.693	Cedar Point Marsh	
5/17/2010	N 30° 19.333	W 88° 08.704	Cedar Point Marsh	
5/17/2010	N 30° 19.326	W 88° 08.695	Cedar Point Marsh	
5/17/2010	N 30° 19.364	W 88° 08.712	Cedar Point Marsh	
5/17/2010	N 30° 19.366	W 88° 08.714	Cedar Point Marsh	
5/17/2010	N 30° 19.310	W 88° 08.692	Cedar Point Marsh	
5/19/2010	N 30° 18.967	W 88° 08.513	Cedar Point Marsh	veg
5/19/2010	N 30° 19.023	W 88° 08.545	Cedar Point Marsh	veg
5/19/2010	N 30° 19.090	W 88° 08.593	Cedar Point Marsh	veg
5/19/2010	N 30° 19.091	W 88° 08.593	Cedar Point Marsh	veg
6/2/2010	N 30° 19.376	W 88° 08.718	Cedar Point Marsh	veg
6/2/2010	N 30° 19.382	W 88° 08.718	Cedar Point Marsh	none

6/2/2010	N 30° 19.472	W 88° 08.721	Cedar Point Marsh	veg
6/2/2010	N 30° 19.264	W 88° 08.690	Cedar Point Marsh	veg
6/2/2010	N 30° 19.249	W 88° 08.684	Cedar Point Marsh	veg
6/2/2010	N 30° 19.200	W 88° 08.644	Cedar Point Marsh	none
6/2/2010	N 30° 18.960	W 88° 08.511	Cedar Point Marsh	veg
6/2/2010	N 30° 18.871	W 88° 08.439	Cedar Point Marsh	veg
6/2/2010	N 30° 18.860	W 88° 08.432	Cedar Point Marsh	
6/8/2010	N 30° 19.254	W 88° 08.684	Cedar Point Marsh	
6/8/2010	N 30° 19.263	W 88° 08.694	Cedar Point Marsh	
6/10/2010	N 30° 18.806	W 88° 08.401	Cedar Point Marsh	veg
6/10/2010	N 30° 18.846	W 88° 08.427	Cedar Point Marsh	veg
6/10/2010	N 30° 18.852	W 88° 08.457	Cedar Point Marsh	veg
6/10/2010	N 30° 18.893	W 88° 08.464	Cedar Point Marsh	marsh edge
6/10/2010	N 30° 18.940	W 88° 08.497	Cedar Point Marsh	none
6/10/2010	N 30° 18.948	W 88° 08.506	Cedar Point Marsh	veg
6/10/2010	N 30° 19.064	W 88° 08.572	Cedar Point Marsh	none
6/10/2010	N 30° 19.089	W 88° 08.597	Cedar Point Marsh	veg
6/10/2010	N 30° 19.110	W 88° 08.604	Cedar Point Marsh	none
6/10/2010	N 30° 19.348	W 88° 08.722	Cedar Point Marsh	veg
6/10/2010	N 30° 19.363	W 88° 08.712	Cedar Point Marsh	none
6/10/2010	N 30° 19.363	W 88° 08.722	Cedar Point Marsh	veg
6/10/2010	N 30° 19.369	W 88° 08.717	Cedar Point Marsh	none
6/10/2010	N 30° 19.475	W 88° 08.732	Cedar Point Marsh	none
6/10/2010	N 30° 19.430	W 88° 08.732	Cedar Point Marsh	veg
6/10/2010	N 30° 18.762	W 88° 08.387	Cedar Point Marsh	
6/15/2010	N 30° 19.275	W 88° 08.692	Cedar Point Marsh	veg
6/15/2010	N 30° 19.271	W 88° 08.698	Cedar Point Marsh	veg
6/15/2010	N 30° 19.275	W 88° 08.692	Cedar Point Marsh	veg
6/15/2010	N 30° 19.280	W 88° 08.693	Cedar Point Marsh	veg
6/15/2010	N 30° 19.283	W 88° 08.692	Cedar Point Marsh	veg
6/15/2010	N 30° 19.285	W 88° 08.690	Cedar Point Marsh	veg
6/15/2010	N 30° 19.299	W 88° 08.695	Cedar Point Marsh	veg
6/15/2010	N 30° 19.313	W 88° 08.697	Cedar Point Marsh	veg
6/16/2010	N 30° 18.832	W 88° 08.415	Cedar Point Marsh	veg
6/16/2010	N 30° 18.843	W 88° 08.421	Cedar Point Marsh	veg
6/16/2010	N 30° 18.861	W 88° 08.442	Cedar Point Marsh	
6/16/2010	N 30° 18.880	W 88° 08.446	Cedar Point Marsh	
6/16/2010	N 30° 18.879	W 88° 08.447	Cedar Point Marsh	veg
6/16/2010	N 30° 18.884	W 88° 08.449	Cedar Point Marsh	veg
6/16/2010	N 30° 18.886	W 88° 08.450	Cedar Point Marsh	veg
6/16/2010	N 30° 18.914	W 88° 08.474	Cedar Point Marsh	

6/16/2010	N 30° 19.068	W 88° 08.571	Cedar Point Marsh	
6/16/2010	N 30° 19.358	W 88° 08.721	Cedar Point Marsh	veg
6/16/2010	N 30° 19.372	W 88° 08.719	Cedar Point Marsh	veg
6/16/2010	N 30° 19.385	W 88° 08.723	Cedar Point Marsh	none
6/16/2010	N 30° 19.387	W 88° 08.725	Cedar Point Marsh	none
6/16/2010	N 30° 19.399	W 88° 08.728	Cedar Point Marsh	veg
6/17/2010	N 30° 19.368	W 88° 08.715	Cedar Point Marsh	none
6/17/2010	N 30° 19.366	W 88° 08.715	Cedar Point Marsh	veg
6/17/2010	N 30° 19.384	W 88° 08.724	Cedar Point Marsh	none
6/17/2010	N 30° 19.406	W 88° 08.730	Cedar Point Marsh	veg
6/18/2010	N 30° 19.165	W 88° 08.640	Cedar Point Marsh	veg
6/22/2010	N 30° 19.075	W 88° 08.576	Cedar Point Marsh	veg
6/22/2010	N 30° 19.090	W 88° 08.594	Cedar Point Marsh	veg
6/22/2010	N 30° 19.105	W 88° 08.597	Cedar Point Marsh	none
6/22/2010	N 30° 19.361	W 88° 08.713	Cedar Point Marsh	veg
6/22/2010	N 30° 19.362	W 88° 08.714	Cedar Point Marsh	veg
6/22/2010	N 30° 19.423	W 88° 08.725	Cedar Point Marsh	none
6/22/2010	N 30° 19.434	W 88° 08.722	Cedar Point Marsh	veg
6/24/2010	N 30° 19.297	W 88° 08.692	Cedar Point Marsh	veg
6/24/2010	N 30° 19.316	W 88° 08.702	Cedar Point Marsh	veg
6/24/2010	N 30° 19.387	W 88° 08.723	Cedar Point Marsh	veg
6/24/2010	N 30° 19.384	W 88° 08.723	Cedar Point Marsh	veg
6/29/2010	N 30° 19.282	W 88° 08.692	Cedar Point Marsh	none
6/29/2010	N 30° 19.295	W 88° 08.695	Cedar Point Marsh	veg
6/29/2010	N 30° 19.096	W 88° 08.577	Cedar Point Marsh	veg
6/29/2010	N 30° 18.973	W 88° 08.519	Cedar Point Marsh	veg
6/29/2010	N 30° 18.928	W 88° 08.487	Cedar Point Marsh	none
6/29/2010	N 30° 18.891	W 88° 08.461	Cedar Point Marsh	veg
6/29/2010	N 30° 18.883	W 88° 08.454	Cedar Point Marsh	veg
6/29/2010	N 30° 18.818	W 88° 08.412	Cedar Point Marsh	veg
7/6/2010	N 30° 18.890	W 88° 08.455	Cedar Point Marsh	
7/6/2010	N 30° 18.894	W 88° 08.462	Cedar Point Marsh	
7/6/2010	N 30° 19.066	W 88° 08.573	Cedar Point Marsh	
7/6/2010	N 30° 19.092	W 88° 08.596	Cedar Point Marsh	
7/6/2010	N 30° 19.112	W 88° 08.612	Cedar Point Marsh	
7/6/2010	N 30° 18.875	W 88° 08.448	Cedar Point Marsh	
7/15/2010	N 30° 19.357	W 88° 08.719	Cedar Point Marsh	
7/15/2010	N 30° 19.360	W 88° 08.719	Cedar Point Marsh	
7/15/2010	N 30° 19.358	W 88° 08.719	Cedar Point Marsh	
7/15/2010	N 30° 19.367	W 88° 08.721	Cedar Point Marsh	
7/15/2010	N 30° 19.376	W 88° 08.724	Cedar Point Marsh	

7/15/2010	N 30° 19.381	W 88° 08.725	Cedar Point Marsh	
7/15/2010	N 30° 19.415	W 88° 08.725	Cedar Point Marsh	
7/15/2010	N 30° 19.270	W 88° 08.634	Cedar Point Marsh	
7/15/2010	N 30° 19.257	W 88° 08.689	Cedar Point Marsh	
7/15/2010	N 30° 20.301	W 88° 09.139	Oyster Island Near Heron Bay	
7/15/2010	N 30° 19.652	W 88° 08.599	North Cedar Point Marsh	
7/15/2010	N 30° 19.652	W 88° 08.599	North Cedar Point Marsh	
7/15/2010	N 30° 19.658	W 88° 08.609	North Cedar Point Marsh	
7/15/2010	N 30° 19.658	W 88° 08.611	North Cedar Point Marsh	
7/15/2010	N 30° 19.659	W 88° 08.612	North Cedar Point Marsh	
7/15/2010	N 30° 19.659	W 88° 623	North Cedar Point Marsh	
7/15/2010	N 30° 19.659	W 88° 08.623	North Cedar Point Marsh	
7/15/2010	N 30° 19.659	W 88° 08.623	North Cedar Point Marsh	
7/15/2010	N 30° 19.658	W 88° 08.630	North Cedar Point Marsh	
7/15/2010	N 30° 19.654	W 88° 08.627	North Cedar Point Marsh	
7/15/2010	N 30° 19.654	W 88° 08.627	North Cedar Point Marsh	
7/15/2010	N 30° 19.654	W 88° 08.627	North Cedar Point Marsh	
7/15/2010	N 30° 19.654	W 88° 08.627	North Cedar Point Marsh	
7/15/2010	N 30° 19.658	W 88° 08.631	North Cedar Point Marsh	
7/15/2010	N 30° 19.654	W 88° 08.635	North Cedar Point Marsh	
7/15/2010	N 30° 19.637	W 88° 08.650	North Cedar Point Marsh	
7/15/2010	N 30° 19.635	W 88° 08.652	North Cedar Point Marsh	
7/15/2010	N 30° 18.483	W 88° 08.238	Cedar Point Marsh	
7/15/2010	N 30° 18.537	W 88° 08.274	Cedar Point Marsh	
7/15/2010	N 30° 18.556	W 88° 08.288	Cedar Point Marsh	
7/15/2010	N 30° 18.581	W 88° 08.307	Cedar Point Marsh	
7/15/2010	N 30° 19.043	W 88° 08.347	Cedar Point Marsh	
7/15/2010	N 30° 19.043	W 88° 08.356	Cedar Point Marsh	
7/15/2010	N 30° 19.642	W 88° 08.580	North Cedar Point Marsh	
7/15/2010	N 30° 19.623	W 88° 08.573	North Cedar Point Marsh	
7/15/2010	N 30° 19.659	W 88° 08.496	North Cedar Point Marsh	
7/15/2010	N 30° 19.664	W 88° 08.496	North Cedar Point Marsh	
7/23/2010	N 30° 19.247	W 88° 08.683	Cedar Point Marsh	
7/23/2010	N 30° 19.273	W 88° 08.700	Cedar Point Marsh	
7/23/2010	N 30° 19.282	W 88° 08.702	Cedar Point Marsh	
7/23/2010	N 30° 19.374	W 88° 08.727	Cedar Point Marsh	
7/23/2010	N 30° 19.164	W 88° 08.637	Cedar Point Marsh	
8/13/2010	N 30° 15.560	W 88° 07.647	Airport Marsh	
8/13/2010	N 30° 15.494	W 88° 07.565	Airport Marsh	
8/13/2010	N 30° 15.487	W 88° 07.560	Airport Marsh	
8/13/2010	N 30° 15.560	W 88° 07.647	Airport Marsh	

8/13/2010	N 30° 15.560	W 88° 07.647	Airport Marsh	
8/13/2010	N 30° 15.560	W 88° 07.647	Airport Marsh	
8/13/2010	N 30° 15.560	W 88° 07.650	Airport Marsh	
8/13/2010	N 30° 15.568	W 88° 07.657	Airport Marsh	
8/13/2010	N 30° 15.568	W 88° 07.657	Airport Marsh	
8/13/2010	N 30° 15.568	W 88° 07.657	Airport Marsh	
8/13/2010	N 30° 15.568	W 88° 07.657	Airport Marsh	
8/13/2010	N 30° 15.572	W 88° 07.657	Airport Marsh	
8/13/2010	N 30° 15.574	W 88° 07.657	Airport Marsh	
8/13/2010	N 30° 15.575	W 88° 07.659	Airport Marsh	
8/13/2010	N 30° 15.577	W 88° 07.660	Airport Marsh	
8/13/2010	N 30° 15.578	W 88° 07.654	Airport Marsh	
8/13/2010	N 30° 15.580	W 88° 07.657	Airport Marsh	
8/13/2010	N 30° 15.580	W 88° 07.658	Airport Marsh	
8/13/2010	N 30° 15.580	W 88° 07.660	Airport Marsh	
8/13/2010	N 30° 15.581	W 88° 07.650	Airport Marsh	
8/13/2010	N 30° 15.582	W 88° 07.658	Airport Marsh	
8/13/2010	N 30° 15.586	W 88° 07.659	Airport Marsh	
8/13/2010	N 30° 15.587	W 88° 07.659	Airport Marsh	
8/13/2010	N 30° 15.589	W 88° 07.659	Airport Marsh	
8/13/2010	N 30° 15.584	W 88° 07.659	Airport Marsh	
8/13/2010	N 30° 15.594	W 88° 07.659	Airport Marsh	
8/13/2010	N 30° 15.598	W 88° 07.658	Airport Marsh	
8/13/2010	N 30° 15.161	W 88° 07.735	Four Tree Island (AM)	
8/13/2010	N 30° 15.154	W 88° 07.730	Four Tree Island (AM)	
8/13/2010	N 30° 15.152	W 88° 07.772	Four Tree Island (AM)	
8/13/2010	N 30° 15.147	W 88° 07.738	Four Tree Island (AM)	
8/13/2010	N 30° 15.135	W 88° 07.755	Four Tree Island (AM)	
8/13/2010	N 30° 15.129	W 88° 07.765	Four Tree Island (AM)	
8/13/2010	N 30° 15.127	W 88° 07.771	Four Tree Island (AM)	
8/13/2010	N 30° 15.156	W 88° 07.773	Four Tree Island (AM)	
8/13/2010	N 30° 15.192	W 88° 07.923	Spoil Island (AM)	
8/13/2010	N 30° 15.219	W 88° 07.954	Spoil Island (AM)	
8/13/2010	N 30° 15.221	W 88° 08.028	Spoil Island (AM)	

APPENDIX D

CAPTURE DATA RESULTS

2006

Tag #	PIT Tag #	Location	Date	Sex	Trap
30	#4444377F5D	Cedar Point Marsh	5/17/2006	F	crab trap
31	#4440434867	Cedar Point Marsh	5/17/2006	F	crab trap
32	#4441252947	Cedar Point Marsh	5/18/2006	F	crab trap
33	#4441405E20	Cedar Point Marsh	5/19/2006	J	crab trap
34	#444144103D	Cedar Point Marsh	5/19/2006	F	crab trap
35	#4441532170	Cedar Point Marsh	5/18/2006	J	crab trap
36	#44413D6F2B	Cedar Point Marsh	5/17/2006	F	crab trap
37	#4441500023	Cedar Point Marsh	5/23/2006	F	crab trap
38	#44441F7262	Jemison Marsh	5/23/2006	M	crab trap
39	#444146317A	Cedar Point Marsh	5/23/2006	M	crab trap
40	#4441264F60	Cedar Point Marsh	5/23/2006	F	crab trap
41	#4441400576	Jemison Marsh	5/23/2006	F	crab trap
42	#4441492D70	Cedar Point Marsh	5/23/2006	F	crab trap
43	#44405E3B57	Cedar Point Marsh	5/25/2006	J	crab trap
44	#44456F2E47	Cedar Point Marsh	5/26/2006	F	pitfall trap
45	#44442B572F	Cedar Point Marsh	6/4/2006	F	pitfall trap
46	#44440230B7F	Cedar Point Marsh	6/4/2006	F	by hand
47	#4444377E7E	Cedar Point Marsh	6/1/2006	F	pitfall trap
48	#4444321B25	Cedar Point Marsh	6/4/2006	F	pitfall trap
49	#44440553101	Cedar Point Marsh	6/8/2006	F	pitfall trap
50	#4440454A38	Cedar Point Marsh	6/11/2006	F	pitfall trap
51	#4440472174	Cedar Point Marsh	6/29/2006	F	crab trap
31	#4440434867	Cedar Point Marsh	6/29/2006	F	crab trap
33	#4441405E20	Cedar Point Marsh	6/29/2006	J	crab trap
52	#44415B2313	Cedar Point Marsh	7/14/2006	F	pitfall trap
53	#444427736D	Cedar Point Marsh	7/21/2006	F	crab trap
54	#44413E6130	Cedar Point Marsh	7/21/2006	F	crab trap
55	#4440444493B	Cedar Point Marsh	7/21/2006	F	pitfall trap
54	#44413E6130	Cedar Point Marsh	8/4/2006	F	pitfall trap

2007

Tag#	PIT Tag #	Location	Date	Sex	Trap
#0161	44413B1E67	Cedar Point Marsh	4/9/2007	F	by hand
48	4444321B25	Cedar Point Marsh	5/30/2007	F	pitfall trap
#0168	44401E1E0C	Cedar Point Marsh	6/7/2007	F	pitfall trap
46	4440230B7F	Cedar Point Marsh	6/12/2007	F	pitfall trap
#0159	4440580975	Cedar Point Marsh	6/12/2007	F	pitfall trap
#0157	44444D5F74	Little Dauphin Island	6/14/2007	M	crab trap
56	4445712839	Airport Marsh	7/10/2007	F	crab trap
#0172	44402B2543	Cedar Point Marsh	7/6/2007	F	pitfall trap
58	4444506A13	Little Dauphin Island	7/10/2007	F	crab trap
#0171	444052543D	Cedar Point Marsh	7/20/2007	F	pitfall trap
#0172	44402B2543	Cedar Point Marsh	7/20/2007	F	pitfall trap
2 zip tags	444139D17	Cedar Point Marsh		F	pitfall trap

2008

Tag #	PIT Tag #	Location	Date	Sex	Trap
#0173	44441F6511	Cedar Point Marsh	4/14/2008	F	crab trap
#0178	44415B2842	Cedar Point Marsh	5/19/2008	F	crab trap
#0175	44441F186A	Cedar Point Marsh	5/21/2008	F	pitfall trap
#0174	444044493B	Cedar Point Marsh	5/24/2008	F	pitfall trap
#0176	4441261602	Cedar Point Marsh	5/26/2008	F	crab trap
#0177	44414E1D72	Cedar Point Marsh	6/3/2008	F	pitfall trap
#0179	44443D4E33	Cedar Point Marsh	5/28/2008	F	crab trap
#0180	4440454A38	Cedar Point Marsh	6/7/2008	F	pitfall trap
#0181	44403E0534	Cedar Point Marsh	6/8/2008	F	pitfall trap
#0182	44402F5834	Cedar Point Marsh	6/7/2008	F	pitfall trap
#0183	4440505B11	Cedar Point Marsh	6/11/2008	F	pitfall trap
#0175	44441F186A	Cedar Point Marsh	6/11/2008	F	pitfall trap
#0177	44414E1D72	Cedar Point Marsh	6/11/2008	F	pitfall trap
#0184	444452153B	Cedar Point Marsh	6/18/2008	F	crab trap
#0171	444052543D	Cedar Point Marsh		F	pitfall trap
#0185	44457E256E	Cedar Point Marsh		F	pitfall trap
#0186	444125317F	Cedar Point Marsh	7/6/2008	F	pitfall trap
#0180	4440454A38	Cedar Point Marsh	7/14/2008	F	pitfall trap
#0187	4441260C0E	Cedar Point Marsh	7/14/2008	F	pitfall trap

2009

Tag #	PIT Tag #	Location	Date	Sex	Trap
#0188	44414B225C	Cedar Point Marsh	3/11/2009	M	otter trawl
#0189	4441421262	Cedar Point Marsh	3/11/2009	M	otter trawl
#0190	4440353177	Cedar Point Marsh	3/12/2009	M	otter trawl
#0191	4441301431	Cedar Point Marsh	3/12/2009	M	otter trawl
#0192	44443A1865	Cedar Point Marsh	3/12/2009	M	otter trawl
#0193	4440471109	Cedar Point Marsh	3/19/2009	F	otter trawl
#0194	4444477416	Cedar Point Marsh	3/19/2009	F	otter trawl
#0195	444138183A	Cedar Point Marsh	3/19/2009	M	otter trawl
#0196	44415B7120	Cedar Point Marsh	3/19/2009	M	otter trawl
#0197	444129470E	Cedar Point Marsh	3/19/2009	M	otter trawl
#0198	4441247A43	Cedar Point Marsh	3/19/2009	M	otter trawl
#0199	44412D412C	Cedar Point Marsh	3/19/2009	M	otter trawl
#0200	444432284F	Cedar Point Marsh	3/25/2009	M	otter trawl
#0201	444041003F	Cedar Point Marsh	4/10/2009	F	otter trawl
#0202	4444502E7	Cedar Point Marsh	4/17/2009	M	otter trawl
#0203		Cedar Point Marsh	4/17/2009	M	otter trawl
#0204	444121794C	Cedar Point Marsh	4/30/2009	M	otter trawl
#0188	44414B225C	Cedar Point Marsh	5/19/2009	M	otter trawl
#0205	44405C7636	Cedar Point Marsh	5/19/2009	M	otter trawl
#0206	444051394B	Cedar Point Marsh	5/27/2009	F	pitfall trap
#0207	44443D1565	Cedar Point Marsh	6/1/2009	F	pitfall trap
#0208	4444283D54	Cedar Point Marsh	6/3/2009	F	crab trap
#0209	44456D7A3D	Cedar Point Marsh	6/3/2009	F	crab trap
#0210	4441285764	Cedar Point Marsh	6/3/2009	F	crab trap
#0211	4441510037	Cedar Point Marsh	6/3/2009	F	crab trap
#0212	4445796172	Cedar Point Marsh	6/3/2009	M	crab trap
#0213	44442A7A47	Cedar Point Marsh	6/3/2009	M	crab trap
#0214	4440427745	Cedar Point Marsh	6/3/2009	F	crab trap
#0178	44415B2842	Cedar Point Marsh	6/3/2009	F	crab trap
#0187	4441260C0E	Cedar Point Marsh	6/7/2009	F	pitfall trap
#0215	44442B0728	Cedar Point Marsh	6/11/2009	F	crab trap
#0216	44414C552A	Cedar Point Marsh	6/11/2009	F	pitfall trap
#0217	44415C6236	Cedar Point Marsh	6/11/2009	F	pitfall trap
#0218	4441531025	Cedar Point Marsh	6/12/2009	F	crab trap
#0219	4441317F0E	Cedar Point Marsh	6/12/2009	F	crab trap
#0172	44402B2543	Cedar Point Marsh	6/11/2009	F	pitfall trap
#0213	44442A7A47	Cedar Point Marsh	6/12/2009	M	crab trap
#0196	44415B7120	Cedar Point Marsh	6/16/2009	M	crab trap

#0220	4440302428	Cedar Point Marsh	6/17/2009	F	pitfall trap
#0201	444041003F	Cedar Point Marsh	6/17/2009	F	pitfall trap
#0221	4441584700	road	6/18/2009	F	by hand
#0222	44412C5A17	Cedar Point Marsh	6/18/2009	F	by hand
#0223	44404A1C39	Cedar Point Marsh	6/18/2009	F	pitfall trap
#0208	4444283D54	Cedar Point Marsh	6/18/2009	F	crab trap
#0224	44403E0534	Cedar Point Marsh	6/22/2009	F	pitfall trap
#0225	44402F3A7A	Cedar Point Marsh	6/22/2009	F	pitfall trap
#0159	4440580975	Cedar Point Marsh	6/19/2009	F	pitfall trap
#0175	44441F186A	Cedar Point Marsh	6/22/2009	F	pitfall trap
#0226	44415B2313	Cedar Point Marsh	6/24/2009	F	pitfall trap
#0183	4440505B11	Cedar Point Marsh	6/28/2009	F	pitfall trap
#0227	4440260A44	Cedar Point Marsh	7/3/2009	F	crab trap
#0171	444052543D	Cedar Point Marsh	7/14/2009	F	by hand
#0228	4441491A00	Cedar Point Marsh	7/14/2009	F	crab trap
#0229	4440366020	Cedar Point Marsh	7/17/2009	F	crab trap
#0230	44413A5B33	Cedar Point Marsh	7/21/2009	F	crab trap
#0171	444052543D	Cedar Point Marsh	7/27/2009	F	crab trap
#0230	44413A5B33	Cedar Point Marsh	7/31/2009	F	crab trap
#0238	44412B5F51	Cedar Point Marsh	7/31/2009	F	crab trap

2010

Tag #	PIT Tag #	Location	Date	Sex	Trap
#0195	444138183A	Cedar Point Marsh	4/2/2010	M	otter trawl
#0192	44443A1865	Cedar Point Marsh	4/19/2010	M	otter trawl
#0189	4441421262	Cedar Point Marsh	4/26/2010	M	otter trawl
#0239	44412E0A77	Cedar Point Marsh	4/26/2010	F	otter trawl
#0240	4441441427	Cedar Point Marsh	4/26/2010	F	otter trawl
#0241	44411F3134	Cedar Point Marsh	4/27/2010	F	otter trawl
#0242	4441226B75	Cedar Point Marsh	5/6/2010	F	otter trawl
#0243	44411E7D05	Cedar Point Marsh	5/6/2010	M	otter trawl
#0171	444052543D	Cedar Point Marsh	5/7/2010	F	by hand
#0214	4440427745	Cedar Point Marsh	5/15/2010	F	pitfall trap
#0252	44442B572F	Cedar Point Marsh	5/18/2010	F	pitfall trap
#0175	44441F186A	Cedar Point Marsh	5/19/2010	F	pitfall trap
#0242	4441226B75	Cedar Point Marsh	5/25/2010	F	pitfall trap
#0253	44403B2850	Cedar Point Marsh	6/8/2010	F	pitfall trap
#0254	44441E2271	Cedar Point Marsh	6/17/2010	F	pitfall trap
#0255	44414F547A	Cedar Point Marsh	6/18/2010	F	pitfall trap

#0256	44456F2E47	Cedar Point Marsh	6/18/2010	F	by hand
#0257	4440434867	Cedar Point Marsh	7/1/2010	F	pitfall trap
#0258		Cedar Point Marsh	7/1/2010	F	pitfall trap
#0271	4441374B02	Cedar Point Marsh	7/25/2010	F	pitfall trap
#0175	44441F186A	Cedar Point Marsh	7/30/2010	F	pitfall trap
#0294	44404F3C22	Cedar Point Marsh	9/24/2010	F	otter trawl