FINAL REPORT

South Carolina State Wildlife Grant T-60-R-1   SC-F-12AF01382
South Carolina Department of Natural Resources
January 1, 2013 – December 31, 2015

Project Title: Validation of Trammel Netting for Monitoring Population Trends and Assessment of Mortality Sources on Diamondback Terrapins (*Malaclemys terrapin*) in the Charleston Harbor Estuary.

Objective 1: Characterize demographic structure of terrapins captured in the Ashley River.

Accomplishments: 699 individual diamondback terrapins were captured, predominantly using trammel net and crab trap fishing gears, in a stretch of the Ashley River (to include several tributaries) coinciding with the SCDNR Inshore Fisheries trammel net survey (i.e. between US-17 and I-526 Bridges). Forty recapture events were recorded, which enabled examination of movement patterns. No change in size distribution or ratio of male to female terrapins was detected during the present study; however, fewer females were captured during the present study compared to a graduate thesis conducted in 2008–2009. Distance moved by recaptured terrapins was also greater during the present study than in 2008–009. For more detailed information on data collection and analysis, please refer to Chapter 1 of the Final Report.

Significant deviations: Original emphasis for this objective was to compare demographic structure of diamondback terrapins captured in creek vs. river habitats. In year one, sufficient data was collected to determine that diamondback terrapins captured in creek and river habitats were not distinct groups; thus, our focus shifted to better understanding demographic structure and movement patterns throughout the entire trammel net survey zone in the Ashley River. This focal shift also enabled temporal comparison between 2008 and 2015, a novel approach to terrapin data analysis for the trammel net survey.

Objective 2: Characterize habitat (river, creek) use by acoustically-tagged terrapins.

Accomplishments: Forty-four diamondback terrapins (26 female, 18 male) were tagged with acoustic transmitters and their occurrence at specific locations was monitored remotely between 11 April 2013 and 9 December 2015 by 13 to 24 acoustic receivers annually. All diamondback terrapins tagged with acoustic transmitters were detected by acoustic receivers and data for individual diamondback terrapins was collected for eight to 232 (outlier = 347) days, with a median of 81 days of data per terrapin. A total of 146,782 detections were recorded for these 44 diamondback terrapins, which were standardized to 21,610 hourly detection bins to assess detection frequency across habitats and several temporal scales. Acoustically-tagged diamondback terrapins were detected most frequently between April and June. Individual terrapins were typically (median) detected nine times as frequently in river habitats than in creek habitats, with significantly more river detections during nocturnal periods as well as increased frequency of creek detection during diurnal periods. In contrast to initial assessment in 2013,

tidal influence on detection in river vs. creek habitats was not significant. For more detailed information on data collection and analysis, please refer to Chapter 2 of the Final Report.

Significant deviations: Original emphasis was on comparing detection frequency in river vs. creek habitats surrounding Duck Island, a large hummock island centrally located within the Ashley River trammel net survey area. Similarly, in year two, we originally planned to move the telemetry array to the Wando River in order to provide a comparison of habitat use between the two river systems. However, more extensive movement in the Ashley River in year one and considerable variability among individuals suggested that a better approach would be to conduct a multi-year study in the Ashley River in order to (a) better characterize habitat use in a single trammel net survey area and (b) refine data collection and analysis protocols to facilitate more effective studies in other survey areas in the future. As such, the configuration of the receiver network varied among years concurrent with expanded spatial capture area for acoustically-tagged diamondback terrapins.

Objective 3: Design a superior by-catch excluder device (BRD) to reduce mortality of diamondback terrapins in crab traps that does not negatively impact capture of large, legal-sized blue crabs.

Accomplishments: A total of 3,062 crab trap soak hours were recorded during 689 trap sets that were evenly distributed between traps fished without and with five different BRD designs in 2014 and 2015. Trap sets captured 3,184 blue crabs [997 (31%) of which were legal-sized, ≥5 in. carapace width; CW] and 68 diamondback terrapins, with greatest capture of legal-sized crabs after August, but capture of diamondback terrapins occurred almost exclusively during April and May. BRD designs tested in 2014 captured significantly fewer legal-sized blue crabs than traps fished without BRDs; however, catch rates for legal-sized crabs were not significantly different between traps fished without or with the smallest BRD design when tested in both horizontal and vertical orientations. Traps fished with the small BRD were associated with a slightly greater frequency of not capturing any legal-sized blue crabs; however, visual observations recorded for a graduate thesis revealed no difference in the physical ability of large (up to 7.5 in. CW) legal-sized blue crabs to traverse through this BRD. For more detailed information on data collection and analysis, please refer to Chapter 3 of the Final Report.

Significant deviations: The original concept for this grant did not include BRD research; however, after discovering in year one how much time diamondback terrapins spent in the Ashley River, their potential vulnerability to entrainment in commercial crab traps could not be ignored. As such, we initiated ardent efforts to engineer a BRD configuration that would provide a superior (for target and non-target species) alternative to BRD designs tested in South Carolina during 2006–2008 that were not favorably received by commercial crabbers.

Objective 4: Monitor terrapin nesting at Duck Island using remotely operated cameras.

Accomplishments: Two site visits to Duck Island were conducted in 2014; however, very little habitat perceived to be suitable nesting was identified (see 2014 Annual Report for more details).

<u>Significant deviations</u>: Given extreme logistical challenges that would have been associated with this objective with a low probability of quality data collection, we abandoned this objective and re-allocated resources to improve the probability of successfully completing Objective 3. Seven of 38 (18%) un-solicited citizen-scientist reports during the present study involved terrapin nest sightings; thus, we are hopeful that active solicitation of citizen-scientist reporting across SC beginning in 2016 will lead to a better ability to characterize annual metrics associated with this important life history stage in the future.

Literature Cited: See Final Report

Estimated Federal Cost: $107,361 (award amount/spent, 01 January 2013 to 31 December

2015) Recommendations: Close the grant

Validation of Trammel Netting for Monitoring Population Trends and Assessment of Mortality Sources on Diamondback Terrapins (*Malaclemys terrapin*) in the Charleston Harbor Estuary

Final Grant Report
To
U.S. Fish and Wildlife Service

Prepared By:
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FINAL REPORT TO THE U.S. FISH AND WILDLIFE SERVICE

For

Validation of Trammel Netting for Monitoring Population Trends and Assessment of Mortality Sources on Diamondback Terrapins (*Malaclemys terrapin*) in the Charleston Harbor Estuary.

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EXECUTIVE SUMMARY

During this three-year grant, field and laboratory data collection and analysis as well as logistical support occurred over a span of nearly 200 work days by a host of SCDNR personnel, a cadre of volunteers, partners at the South Carolina Aquarium (SCA) and the Grice Marine Biology Laboratory, and a graduate student at the College of Charleston.

Nearly 30% of grant effort was associated with the processing of 699 terrapins captured at numerous locations throughout the Ashley River and tributaries between March and November, as well as one terrapin captured in the Stono River and five terrapins raised from hatchlings at the SCA.
No change in sex ratio or movement patterns (40 recapture events) was noted during 2013–2015, but comparison with demographic and recapture (55 events) data from 2008–2009 revealed relatively fewer female terrapins and more extensive movement during 2013–2015 (Chapter 1).

Twenty-six female and 18 male diamondback terrapins were tagged with acoustic transmitters to investigate spatio-temporal distribution patterns. Initial emphasis was placed on comparing resident and movement patterns between creek and river habitats, but was later expanded to better characterize along-river movements. Approximately 30% of grant effort was associated with maintaining between 13 and 24 acoustic receivers annually, which recorded 146,782 acoustic transmissions for terrapins between 11 April 2013 and 9 December 2015 (Chapter 2). Acoustically-tagged diamondback terrapins were detected most frequently between April and June. Individual terrapins were typically (median) detected nine times as frequently in river habitats than in creek habitats, with significantly more river detections during nocturnal periods as well as increased frequency of creek detection during diurnal periods. In contrast to initial assessment in 2013, tidal influence on detection in river vs. creek habitats was not significant.

The remainder of the grant effort was associated with evaluating five novel by-catch excluder device (BRD) designs that were systematically developed in 2014 and 2015 in the hopes of identifying a superior alternative to BRD designs tested in South Carolina during 2006–2008 that were not favorably received by commercial crabbers. A total of 3,062 crab trap soak hours were recorded during 689 trap sets that were evenly distributed between traps fished without and with BRDs in each year (Chapter 3). Trap sets captured 3,184 blue crabs (997 of which were legal-sized, 31%) and 68 diamondback terrapins, with greatest capture of legal-sized crabs after August but nearly exclusive capture of diamondback terrapins during April and May. BRD designs tested in 2014 captured significantly fewer legal-sized blue crabs than traps fished without BRDs; however, a non-significant difference in catch rates for legal-sized crabs across trap designs was associated with the BRD design tested in both horizontal and vertical orientations in 2015. Traps fished
with BRDs were associated with a slightly greater frequency of not capturing any legal-sized blue crabs; however, visual observations recorded for a graduate thesis revealed no difference in the physical ability of large (up to 19.1 cm) legal-sized blue crabs to traverse through BRDs. Given this observation, and nearly identical size distributions and sex ratios between control traps and all but one BRD design tested in 2014, reduced catch in BRD traps may stem from increased rates of crabs escaping through rigid funnel openings when BRDs are present, and future BRD research should systematically attempt to reduce the occurrence of this behavior.

General Introduction

Diamondback terrapins (*Malaclemys terrapin*) are the only exclusively estuarine turtle in North America (Wood 1977) and are distributed between Massachusetts and Texas. Maximum carapace length is <30 cm (Ernst et al. 1994) and morphological differences between larger females and smaller males (Ernst et al. 1994) minimize foraging niche overlap between the sexes (Tucker et al. 1995; Levesque 2000). In the late 1800's, diamondback terrapins were prized table fare; however, that commercial market crashed within a few decades following overharvesting (Carr 1952). This long-lived species (Hildebrand 1932) remains globally listed at low but near threatened risk (IUCN; http://www.iucnredlist.org/details/12695/0). In South Carolina, diamondback terrapins are one of 52 species in the reptile/amphibian guild that is listed as a species of concern (SC CWCS 2005, p. 2-11).

No commercial terrapin fishery exists in South Carolina, but recreational "possession" of two terrapins per person is authorized (SC Code, Chapter 5, Article 23, Section 50-5-2300A). Despite no legal harvest, fisheries mortality continues to occur annually in South Carolina as a result of incidental capture and subsequent drowning in crab traps (Roosenburg et al. 1997; Dorcas et al. 2007; Grosse et al. 2011). Incidental capture rates for diamondback terrapins in South Carolina waters are unknown, but crab traps are reported to capture up to 78% of local populations elsewhere (Roosenburg et al., 1997). Despite this observation, measures to minimize diamondback terrapin entry into crab traps have not been mandated in South Carolina (in contrast to some mid-Atlantic states) due to associated reductions in absolute crab catches (despite statistically similar catch rates) based on SC studies to date (Powers et al. 2009a).

The development and implementation of "long-term coastwide standardized surveys to estimate the abundance and distribution of South Carolina's terrapin population" is a top Conservation Action for this species of concern in the South Carolina Wildlife Action Plan (SC SWAP 2015). Diamondback terrapins are the fifth most frequently encountered species captured in a coastwide trammel net survey designed to monitor inshore fisheries (Arnott et al. 2013), and more than 18,000 diamondback terrapin collections have occurred in this survey since 1995[1]. Pronounced seasonal variability in catch rates is noted, with peak captures in the spring (Arnott et al. 2013) concurrent with peak incidental catch rates in commercial crab traps (Powers et al. 2009a). Spatial and temporal disparity in catch rates was also reported, with anomalously high catch rates in the Ashley River relative to other water bodies; however, even in this river system a decline in catch rates has occurred since 1995, consistent with the statewide trend (Arnott et al. 2013).

This report summarizes research activities conducted between 2013 and 2015. In all years, diamondback terrapins captured by the trammel net survey in the Ashley River (Figure 1) were examined, measured, and tagged prior to release; a demographic characterization of captured terrapins is presented in Chapter 1. A subset of captured terrapins were tagged with acoustic transmitters and data were collected remotely using a network of acoustic receivers; distribution patterns and implications for analyzing trammel net catch rates are presented in Chapter 2. Beginning in 2014, several design changes to a commercially available by-catch excluder device (BRD) were investigated; implications for reducing terrapin catch without detrimental impacts to catch of legal-sized blue crabs appear in Chapter 3.

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[1]Unpublished data. Inshore Fisheries Section, Marine Resources Research Institute, South Carolina Department of Natural Resources. Data provided by Dr. Steve Arnott, Principal Investigator.

Figure 1. Between 2013 and 2015, diamondback terrapins were captured by trammel netting (between yellow cross hatches) in the Ashley River (red line), predominantly near Duck Island (yellow star). During 2014–2015, field testing (green dots) of modified by-catch excluder devices (BRDs) primarily occurred in the Stono River (blue line) and in Orange Grove and Old Towne Creeks adjacent to the Ashley River as well as at Duck Island in the Ashley River.

Chapter 1. Population and demographic structure of diamondback terrapins in the Ashley River.

Introduction
Reliable assessment of the number of individuals that exists in a population is perhaps the most basic data need for managing wildlife populations; however, estimating population size is rarely a straight-forward task. Diamondback terrapins return to the water surface to breathe about once every 10 minutes (Baker et al. 2013); thus, counting individuals at the water surface is one method that can be used to estimate population sizes over large areas, particularly with the assistance of citizen scientists (Harden et al. 2009). However, terrapin behavior such as basking or burying in mud varies seasonally (Akins et al. 2014), which in addition to water temperature (Butler 2002), can affect the probability that terrapins will be seen at the water surface. Consequently, visual surveys in South Carolina have primarily been conducted in localized areas with limited total annual observation effort (Harden et al. 2009; Lanzieri 2012).

In contrast to visual counts, physical capture of terrapins ensures accurate counts of individuals as well as confident distinction between male and female terrapins which are sexually dimorphic as adults but not also easy to distinguish at overlapping sizes as juveniles. With the exception of study sites where population declines have been attributed to crab trap mortality due to selective removal of smaller males (Dorcas et al. 2007), male diamondback terrapins are typically captured twice as often as females in South Carolina estuaries (Bishop 1983; Lovich and Gibbons 1990; Gibbons et al. 2001; Broyles 2010; King and Ludlam 2014). In addition to monitoring potential population change as a result of differential survival between males and females, examination of sex ratio data also provides insight into potential mating areas given seasonal aggregation of males for breeding (Estep 2005).

The Inshore Fisheries Research Section of the SCDNR MRD's Marine Resources Research Institute (MRRI) maintains the most comprehensive state-wide database on diamondback terrapin catch rates, which have been recorded monthly since 1995 (Arnott et al. 2013). Diamondback terrapin catch rates in the Ashley River are significantly greater than all other estuarine systems sampled by this survey; thus, high catch rates coupled with proximity to the MRRI have resulted in a considerable number of diamondback terrapins tagged in the Ashley River by graduate students over the past 20 years. As such, continued demographic assessments in the Ashley River are conducive to monitoring temporal variability in population structure to complement catch rate data for this system. Therefore, the first objective of this study was to assess temporal variability in female to male sex ratio in the Ashley River during 2013–2015 as well as during this timeframe relative to 2008–2009 (Broyles 2010). The second objective was to assess temporal variability in size distribution, partitioned by sex, during 2013–2015 as well as during this timeframe relative to 2008–2009. The third objective was to assess temporal variability in recapture rates during 2013–2015 as well as during this timeframe vs. 2008–2009; because selective tagging of diamondback terrapins occurred prior to 2015, formal population modeling was not possible with this data set but will be conducted in the near future given that all terrapins captured during each monthly survey have been tagged since 2015.

Methods

Study site description

Diamondback terrapins were captured in the Ashley River, the southernmost tributary of Charleston Harbor. The Ashley River watershed supports 3,017 acres of estuarine habitat, and surface water quality is most often reported as "SA"[2].

In 2013, diamondback terrapins were predominantly captured in river and creek habitats near Duck Island, a large hummock island located on the south/west side of the river and nearly equidistant between the freshwater/saltwater dividing line and the Atlantic Ocean (Figure 1). Watershed development on the south/west side of the Ashley River near Duck Island is predominantly residential versus industrial development on the opposite side of the river.

In 2014, diamondback terrapin capture locations were periodically expanded to include all sites in the Ashley River (Appendix 1) to increase overall sample size and spatial distribution of marked individuals. Spatial expansion of capture locations also slightly increased the probability of recapturing any of the >1,200 terrapins tagged or marked in conjunction with six graduate studies (Levesque 2000; Lee 2003; Hauswaldt 2004; Estep 2005; Schwenter 2007; Broyles 2010) between 2000 and 2009. Beginning in 2015, diamondback terrapins captured at all monthly trammel stations were examined and tagged prior to release to ensure standardization for repeating the 2008–2009 population assessment by Broyles (2010).

Capture and general processing
A 600 ft. monofilament trammel net (Figure 2a) consisting of multiple panels with a minimum mesh size of 4 in. (stretch) provided the primary means for capturing diamondback terrapins. As described by Arnott et al. (2013), the trammel net was rapidly deployed from the stern of a skiff along ~450 ft. of shoreline; soaked for five minutes, during which time noise was used to 'spook' fish and terrapins from the marsh grass into the net; and then steadily retrieved for 20 minutes. Additional capture gears included 8 ft.[3] wire mesh crab traps (Figure 2b) baited with a single menhaden and soaked for several hours each day but checked hourly. A 15 ft. (head rope) otter trawl (Figure 2c) deployed from a 20 ft. Privateer (125 HP, speed ~1000 rpm) and a reduced length (200 ft.) trammel net were also evaluated as potential sampling gears in 2013.

Figure 2. Diamondback terrapins were primarily captured via a standard trammel net (A) and 8 ft.[3] wire mesh crab traps (B); however, experimental fishing with an otter trawl (C) and a reduced length trammel net was also briefly evaluated in 2013.

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[2] South Carolina Department of Health and Environmental Control (SCDHEC); https://www.scdhec.gov/homeandenvironment/docs/03050201-06.pdf (Accessed 10 March 2016)

Captured terrapins were externally marked with a grease pencil to differentiate individuals while held in aggregate in large, ventilated plastic bins which were partially covered to provide shade. In 2013 and 2014, data were collected after terrapins were transported to a climate-controlled, shore-based facility at Fort Johnson; however, in 2015, terrapins were processed to the fullest extent in a shaded environment at Northbridge Park prior to transport to the shore-based facility at Fort Johnson. Diamondback terrapins were typically held overnight in individualized and ventilated containers which included moist towels for thermal regulation and then released as close as possible to their capture locations the next day. Standard data collection included:

- Visual inspection of terrapins to note pre-existing marks and injuries
- Assessment of sex based on dimorphism (head/body size, cloaca position/tail length)
- Scanning of all soft tissue for the presence of a PIT tag
- Straight-line measurements (cm) of carapace length (CL), width (CW), and body depth (BD) with calipers (Haglof, Sweden)
- Measuring body mass (kg) with a digital spring-scale
- Photographing dorsal, ventral, and lateral perspectives
- Drilling small holes in marginal scutes using a multi-letter combination coding system; this practice was discontinued in 2015 and replaced with PIT tags (Biomark, HPT12) inserted (N125 injector needle, 12-ga) into the left hind limb after soaking PIT tags in 70% Isopropenol and scrubbing the hind limb injection site with Betadine.
- Attaching flexible shellfish tags (Floy Tag, Inc.) to the carapace using epoxy
- Assessing reproductive condition of females using ultrasound (Sonosite 180+)
- Opportunistically collecting and examining fecal matter to identify forage items (2013)
- Opportunistically collecting barnacles (2013, 2014) to support dissertation research at the University of Georgia; in 2014 we also attempted to collect blood and/or scute keratin for researchers at the US Geological Service (Davie, FL) but were unsuccessful.

Statistical analysis
Statistical analyses were performed in Minitab 15® (Minitab, Inc.; State College, Pennsylvania). A significance level of $\alpha = 0.05$ was assumed for all statistical tests, which were selected after first testing the underlying data distributions for normality (Anderson-Darling test).

Chi-square contingency tests were used to assess differences in the ratio of females to males with respect to capture habitats (river, creek) after first determining whether significant differences existed among gear types (trammel net, trawl, and crab trap) within capture habitats. Chi-square contingency tests were also used to determine if pooled (2013–2015) sex ratio differed from data collected in the Ashley River during 2008–2009 by Broyles (2010).

Significant differences in terrapin size (partitioned by terrapin sex) across years were assessed using Analysis of Variance (ANOVA) or Kruskal-Wallis (KW) tests depending on data distribution. ANOVA or KW tests were also used to determine if significant differences in size distribution existed between the 2013–2015 study and that of Broyles (2010).

Chi-square contingency tests were used to assess statistical differences in annual tagging and recapture rates during 2013–2015 and in the present study vs. Broyles (2010). When expected cell counts were <5, a one-tailed Fisher's exact test (http://vassarstats.net/tab2x2.html) was used.

Results

A total of 743 diamondback terrapin collections were recorded from 231 gear deployments (358.4 gear soak hours) during 2013–2015 (Table 1). Sixty percent ($n = 445$) of diamondback terrapins were processed in 2015, with the remainder nearly evenly processed in 2013 ($n = 128$) and 2014 ($n = 170$). Eighty-five percent of gear soak time was associated with crab trap deployments, but this gear only captured 10% ($n = 73$) of all diamondback terrapins (Table 1). In contrast, 14% of gear soak was associated with trammel net deployments, which captured 90% ($n = 666$) of all diamondback terrapins (Table 1). Short trammel net soaks and otter trawling was associated with <1% of overall sampling effort and terrapin catch and was only evaluated in 2013 (Table 1).

Ninety-three percent of diamondback terrapin collections occurred in river habitats; however, only 79% of gear soak time occurred in river habitats (Table 1). Sampling in creek habitats occurred across gear types in 2013, but was restricted to crab traps only in 2014 and 2015. Sixty-two percent ($n = 62$) in 2014 and 56% ($n = 115$) in 2015 of crab trap fishing effort (hours) was concentrated in creek habitats. All but one diamondback terrapin captured by a crab trap was encountered in April or May, despite only 23% ($n = 69$ soak hours) of crab trap fishing effort occurring during these two months and split evenly between creek ($n = 33$ soak hours) and river ($n = 36$ soak hours) habitats. Between June and November, 72% ($n = 170$ soak hours) of crab trap fishing effort occurred in river habitats, which may have potentially reduced the probability of interactions with diamondback terrapins during the summer and fall.

Across all gear types, males comprised 69% ($n = 512$) of all diamondback terrapins processed during 2013–2015 and only 1% ($n = 7$) of diamondback terrapins were not externally sexed. No significant difference ($\chi^2_1 = 2.64$, P = 0.104) was detected in the ratio of female to male terrapins captured by trammel net (208F; 452M) vs. by crab trap (16F; 56M). Within the trammel net, a significant difference was noted in the female proportion among years ($\chi^2_2 = 7.731$, P = 0.028), which ranged from a low of 25% in 2013 (31F; 92M) to a high of 39% in 2014 (62F; 95M). Temporal variability in the proportion of females captured by crab traps could not be statistically assessed due to small sample size ($n = 11$) and exclusive capture of males in 2014; however, no significant difference was detected ($\chi^2_1 = 0.410$, P = 0.522) in the ratio of female to male terrapins captured in 2015 by trammel net (115F; 265M) vs. crab trap (16F; 45M). The overall ratio of female to male terrapins captured across sampling gears during 2013–2015 (224F; 512M) was significantly different ($\chi^2_1 = 8.039$, P = 0.005) than during 2008–2009 (303F; 510M) due to a 26% reduction in the relative occurrence of females relative to nearly identical male captures. During 2008–2009, twice as many ($n = 13$) small terrapins that were not able to be sexed were also collected relative to 2013–2015.

Table 1. A total of 743 diamondback terrapin collections were recorded during 2013–2015, of which 90% were captured by trammel nets, 10% by crab traps, and <1% by other gear types.

| Gear type | *Creek* | | | | *River* | | | |
	N Terrapins	Gear sets	Soak (hr)	CPUE	N Terrapins	Gear sets	Soak (hr)	CPUE
Trammel net	25	18	6.0	4.2	641	133	44.3	14.5
Crab trap	28	23	99.9	0.3	45	41	205.6	0.2
Short trammel net	1	5	1.7	0.6	0	0	0.0	0.0
Otter trawl	1	6	0.5	2.0	2	5	0.4	4.8

Female diamondback terrapins ($n = 224$) captured during 2013–2015 measured 8.5 to 20.3 cm CL (median = 15.8 cm, inter-quartile range (IQR) = 4.8 cm). A significant difference was detected ($H_1 = 18.14$, P<0.001) in female terrapin size among gear types, with a smaller maximum size (15.2 cm CL) associated with crab traps ($n = 15$) than trammel nets ($n = 208$). Irrespective of gear type, a significant decrease in the size of diamondback terrapins captured in the Ashley River was detected ($H_2 = 31.69$, P<0.001) and was characterized by a steady decline in the size (concurrent with increasing sample size) of females during 2013–2015 (Figure 3). Female size distribution was not significantly different in 2008 vs. 2009 ($H_1 = 0.01$, P = 0.917) nor during 2008–2009 vs. 2013–2015 ($H_1 = 0.02$, P = 0.653).

Male diamondback terrapins ($n = 512$) captured during 2013–2015 measured 8.7 to 13.9 cm CL (median = 11.6 cm, IQR = 1.0 cm). A significant difference was detected ($H_1 = 6.49$, P = 0.011) in male terrapin size by gear type, with a slightly narrower distribution range associated with crab traps ($n = 56$; 10.0 to 13.1 cm) than trammel nets ($n = 452$; 8.7 to 13.9 cm). Independent of gear type, no significant difference was detected ($H_1 = 4.41$, P = 0.110) in the size of male terrapins captured annually ($n = 92$ to 310) during 2013–2015. Likely due to large sample sizes, a significant difference in male size distribution was detected during 2008–2009 vs. 2013–2015; however, median (11.8 vs. 11.6 cm), maximum (14.1 vs. 13.9 cm), and IQR (1.1 vs. 1.0 cm) sizes were nearly identical, with the most notable difference in minimum size (9.6 vs. 8.7 cm).
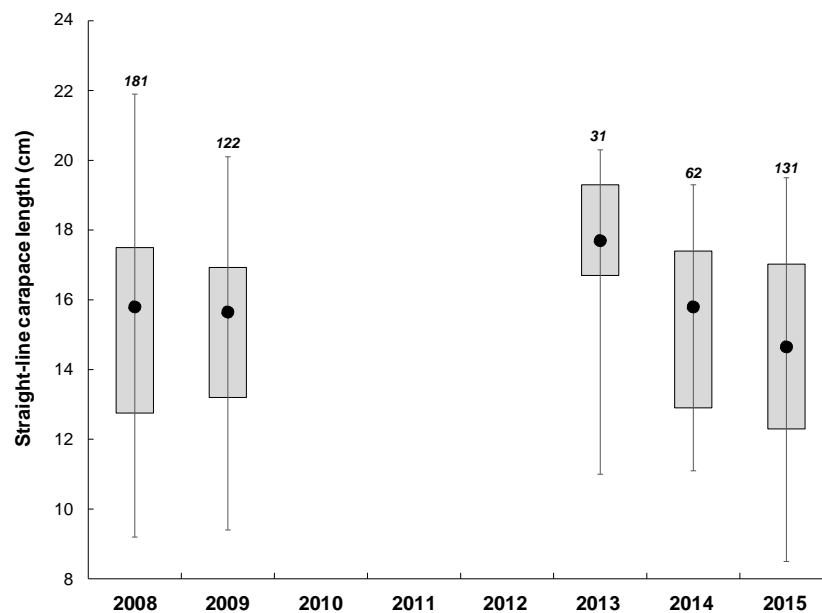
Figure 3. Size distribution of female terrapins captured in the Ashley River during the present study (2013–2015) was not significantly different from 2008–2009 (Broyles 2010). Median size size is denoted by circles; gray boxes depict inter-quartile range; error bars indicate annual minimum and maximum sizes; and annual sample size is italicized above maximum size.

Thirty-eight of 698 (5.4%) unique diamondback terrapins captured during the present study were recaptured during 2013–2015, one of which was recaptured twice. A significant difference was not detected (P = 0.528) in the recapture rate of diamondback terrapins originally encountered in crab traps (4 of 69 unique terrapins, 5.8%) vs. trammel nets (34 of 632 unique terrapins, 5.4%).

Time elapsed between tag-release and recapture ranged from 20 to 937 days (median = 370 days, IQR = 348 days). Eight of 39 recapture events (21%) involved movement of diamondback terrapins between creek and river habitats, and time elapsed between tag-release and recapture was not significantly different (H_1 = 1.70, P = 0.192) between terrapins associated with movement between creek and river habitats vs. capture and recapture within the same habitat.

Relative recapture rates were not significantly different (χ^2_1 = 0.321, P = 0.571) between males (29 events, 483 unique terrapins) and females (10 events, 214 unique terrapins). Capture and recapture sites were located 0 to 4.1 km (median = 0.4 km, IQR = 1.1 km) apart, but distance between capture and recapture sites was not significantly different by sex (H_1 = 0.22, P = 0.636).

Twenty of 812 unique (2.5%) diamondback terrapins tagged in the Ashley River by Broyles (2010) during 2008–2009 were recaptured (3.8 to 7.5 years later) in the Ashley River during 2013–2015, including two diamondback terrapins recaptured a second time during 2013–2015. Similar to the present study, capture and recapture sites for diamondback terrapins originally tagged by Broyles (2010) were located 0 to 3.3 km apart (median = 0.1 km, IQR = 0.5 km). No significant difference was detected (H_1 = 1.42, P = 0.233) between the recapture distance moved for diamondback terrapins tagged by Broyles (2010) and recaptured in the present study vs. diamondback terrapins tagged and recaptured during the present study. However, independent of tagging year, during 2013–2015 diamondback terrapins (*n* = 59) were recaptured significantly (H_1 = 14.24, P<0.001) further (median = 0.2 km, IQR = 0.9 km) from where captured than during 2008–2009 (*n* = 55; median = 0.0 km; IQR = 0.2 km; Broyles, 2010). No obvious temporal-spatial variation in location origin for recaptured terrapins was noted (Figure 4).
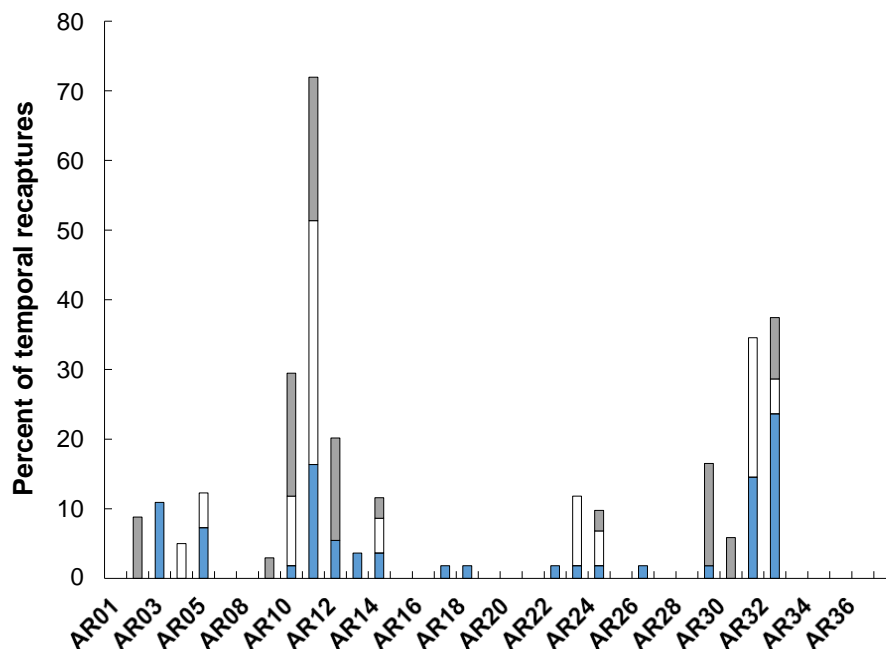
Figure 4. Temporal stability was noted for the spatial origin of diamondback terrapins tagged and recaptured during 2008 –2009 (blue bar); tagged during 2008 –2009 and recaptured during 2013 –2015 (white bar); and tagged and recaptured during 2013–2015 (gray bar).

Chapter 2. Temporal and spatial distribution patterns determined from acoustic telemetry

Introduction

Assessment of wildlife population trends requires reliable abundance metrics that can be monitored (holistically or partitioned by demographic structure) through time. In terrestrial environments, direct observation is conducive to census surveys (Jolly 1969), particularly when measures are taken to scale abundance relative to animal 'detectability' (Anderson 2001). However, in turbid aquatic habitats where direct observation is not effective for estimating abundance, physical capture is often necessary to generate abundance estimates for a subset of the true density of animals in the sampling area. Furthermore, interpretation of abundance trends from physical sampling presents a host of challenges given gear selectivity and catchability which varies as a function of animal size, as well as ingress and egress of animals from study areas which varies in scale from sub-daily to multi-annual (Murphy and Willis 1996).

As noted in Chapter 1, the most comprehensive statewide data set for monitoring population trends for diamondback terrapins in South Carolina since 1995 is a monthly trammel net survey. Temporal differences in catch rate trends and magnitude differences across estuaries are reported in this long-term survey (Arnott et al. 2013); however, the extent to which these differences may be influenced by susceptibility to capture vs. true changes in abundance is not well understood. Across South Carolina estuaries since 1995, peak annual catch rates occur in the spring and early summer, followed by a secondary (but smaller) surge in catch in autumn and rare catch in winter (Arnott et al. 2013); similar seasonal patterns in Delaware Bay in 1975 (Hurd et al. 1979) suggests that this pattern of spring aggregation followed by dispersal is likely inherent to this species throughout its range. Small home ranges and high fidelity to in-water and terrestrial capture sites is also reported for diamondback terrapins from mark- and tag-recapture studies (Gibbons et al. 2001; Szerlag-Egger and McRobert 2007; Sheridan et al. 2010), which is conducive for inferring population trends from catch rates and has also prompted the suggestion for use as a sentinel species for contaminant studies (Blanvillain et al. 2007). However, radio and acoustic telemetry studies reveal that localized within-season movements occur more frequently than suggested by recapture data sets (Spivey 1998; Butler 2002; Estep 2005; Harden and Southwood Williard 2012). Within-season movements likely reflect foraging forays given the diversity and tidal-mediation of prey consumed by this species (Tucker et al. 1995).

Tidally-mediated foraging suggests a high degree of variability in detectability of diamondback terrapins across habitat types, and therefore should be evaluated on a survey-specific basis. Tucker et al. (1995) reported that six radio-tagged female diamondback terrapins "…used creeks and tidal drainages as primary travel corridors both with and against currents during all tide levels." Foraging in the marsh consistently occurred when the marsh was flooded, but variable strategies such as burying in the mud or returning to main channels was observed as the waters receded (Tucker et al. 1995). These observations suggest that diamondback terrapins may only infrequently occur in river edge habitats sampled by the trammel net survey, which are located adjacent to numerous creeks and tidal drainages as described by Tucker et al. (1995). Therefore, in spring 2013, we initiated an acoustic telemetry study to compare residence and movement patterns between river and creek habitats, across a diverse range of temporal scales, in order to evaluate the validity of using the trammel net data set as a means for monitoring long-term relative abundance trends for diamondback terrapin populations in South Carolina waters.

Methods

Acoustic transmitter attachment

Because sexual dimorphism reduces foraging niche overlap (Tucker et al. 1995) and because prior to this study very little multi-season habitat use data were available for male diamondback terrapins (Harden and Southwood Williard 2012; Tulipani 2013), we wished to study males as well as females. The target sampling design in 2013 included six males and six females captured in each of creek and river habitats near Duck Island. In 2014 and 2015, Duck Island remained the primary capture focus for tagging diamondback terrapins with acoustic transmitters, but this data collection technique was also extended to diamondback terrapins captured elsewhere in the Ashley River, preferably in the lower Ashley near Orange Grove Creek (Appendix 1).

Prior to attachment, barnacles and encrusting bryozoans were gently scraped from the vertebral and costal scutes while fine-scale organic matter such as algae was scrubbed off using alcohol-soaked gauze pads. Loose keratin scutes were carefully peeled away and 100-grit sandpaper was lightly applied to ensure no loose keratin remained.

Acoustic transmitters (V9-2H; Amirix Systems, Inc.) were attached to the second vertebral scute, but offset from center due to the vertical relief associated with vertebral scutes. A base layer of epoxy was first pressed to the carapace across at least three scute seams to minimize the risk of transmitter loss due to detachment of a single scute. The epoxy was molded into a shape that had a centralized dome. Next, the transmitter was pressed onto the epoxy dome with the transducer end of the transmitter facing towards the rear of the animal. In "pig in a blanket" fashion, epoxy was then applied over top of the transmitter and blended evenly (Figure 5).

Figure 5. Acoustic transmitters were attached to the carapace of diamondback terrapins using a two-part putty epoxy spread across multiple scutes to increase retention duration.

Acoustic transmitters measured 9 mm (diameter) by 29 mm (length) and weighed 4.7 g in air (2.9 g in water). Approximately 9.5 g of a two-part epoxy putty (SonicWeld™; Ed Greene and Company) was used to secure these transmitters to the carapace. This use of a quick-setting epoxy to attach transmitters is standard for most diamondback terrapin telemetry studies conducted to date (Spivey 1998; Butler 2002; Harden and Southwood Williard 2012), and has

resulted in transmitter retention for nearly a year. For male diamondback terrapins, the total weight associated with the transmitter plus epoxy exceeded the 2% of body weight rule suggested by Winter (1996), but was comparable to transmitter package weights (4 to 39 g) previously reported for this species (Spivey 1998; Harden and Southwood Williard 2012). To minimize excessive drag, transmitters were only attached to males ≥300 g (Tulipani 2013).

Acoustic telemetry data collection
Acoustic transmitters emitted coded signals (which distinguished individual terrapins) on a frequency of 69.0 kHz at random intervals between 180 and 300 seconds.

VR2W receivers (Amirix Systems, Inc.) were deployed near the water surface in a PVC (Sch 40) housing that floated, with the hydrophone end of the receiver facing downward (Figure 6a). This housing slid up and down a 3.8 cm diameter galvanized pole that measured 4.3 to 6.4 m in length depending on water depth and sediment consistency. Housing buoyancy was provided by an air and foam-filled PVC base and later augmented by surface-oriented air-filled PVC tubes. A 5.1 cm diameter PVC pole that measured 4.6 to 6.1 m in length (<1.5 m exposed at high tide) was positioned behind each galvanized pole to restrict the receiver housing from swaying during water level changes, as well as to mark the site during the highest water levels.

Acoustically-tagged terrapins were also detectable from research vessels using a VR60 and later a VR100 receiver with omni and directional hydrophones (Figure 6b). This boat-based system enabled searches for acoustically-tagged terrapins in areas outside of fixed site VR2W receiver reception range (estimated to be <200 m), such as in the upper reaches and bends of creeks where line-of-sight fetch was limited; however, due to difficulty manually tracking transmitters with a 4-minute repeat interval, this data collection technique was used sparingly.
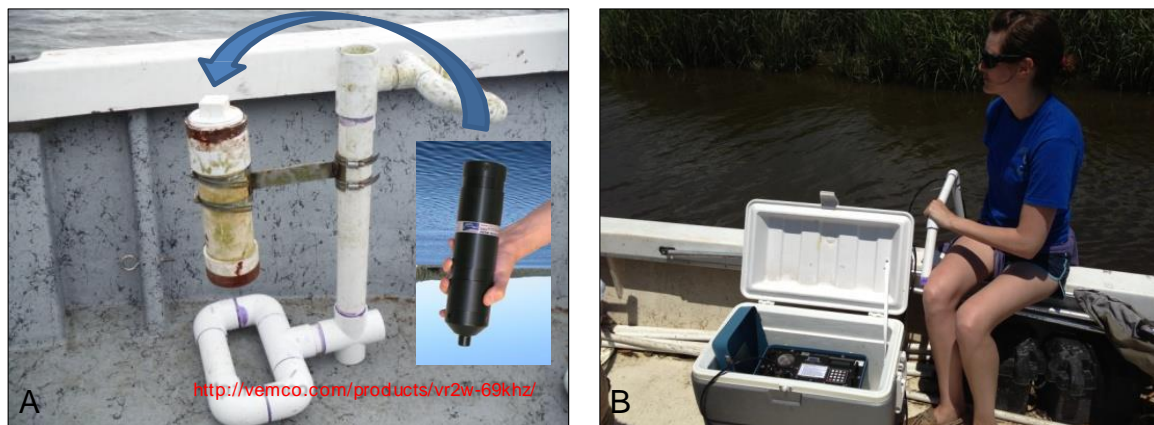
Figure 6. Acoustic detections were recorded by VR2W receivers deployed in buoyant PVC housings (A), as well as opportunistically by a VR60 receiver and hydrophone system (B) during visits to the study area.

Acoustic receivers provided continuous monitoring capability when submerged (Figure 7a); however, because of the tidal nature of this study area, all receivers were exposed at low tide (Figure 7b). Nonetheless, trammel nets can only be deployed when sufficient water levels are present; thus, the VR2W receivers monitored the entire tidal window of opportunity that trammel

net sampling could occur each day of the study. VR2W receivers were removed from PVC housings every four to six weeks and the data were uploaded to a laptop computer using the VUE software and a Bluetooth USB connection. Concurrent with data uploading, biogenic fouling was removed from the PVC housing. Prior to redeployment ~10 minutes later, lithium grease was re-applied to the threaded fittings for the PVC housing cap and the stopper coupling on the galvanized pole, to assist with future data uploading missions.
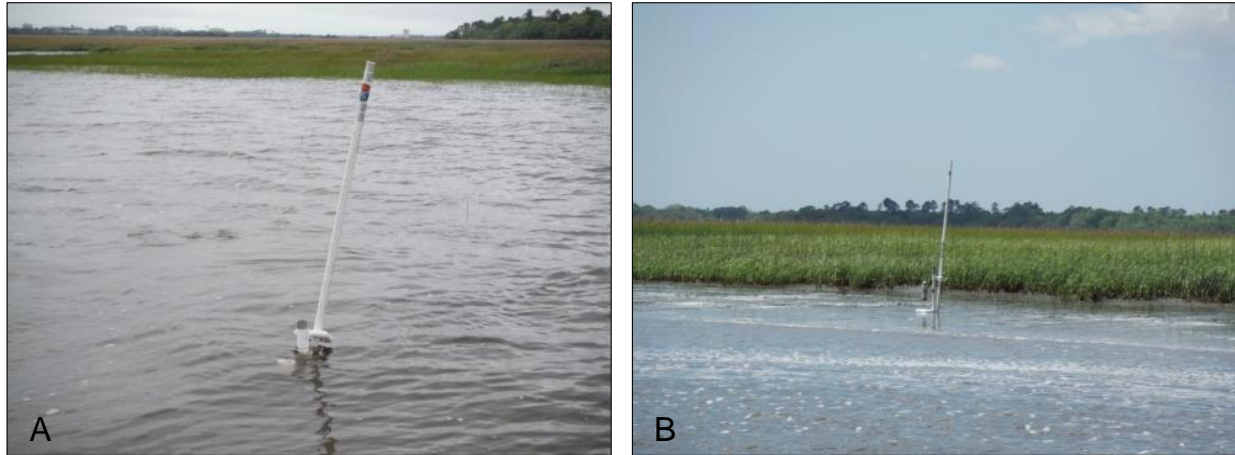
Figure 7. VR2W receivers provided continuous monitoring coverage when submerged (A), but were exposed at low tide (B) due to large tidal amplitudes typical of estuaries in this region.

On 09 April 2013 (Figure 8), VR2W acoustic receivers were deployed at 12 locations encompassing five trammel net stations in the Ashley River near Duck Island, five locations within two creek systems behind Duck Island, two river locations bracketing the mouth of a tidal creek not adjacent to trammel net stations; a US Coast Guard (USGS) Daymarker receiver site established by the SCDNR Diadromous Research Section near the I-526 bridge in 2010 was also maintained by the present study in 2013. In 2014 (Figure 8), the original 13 receiver sites were maintained but expanded (14 March) to include five locations associated with Orange Grove Creek (OGC); two locations near the RT-7 (Cosgrove) bridge across the river from Duck Island (02 May); and USGS Daymarker 13 in the Ashley River between Duck Island and OGC (02 May). Three OGC receivers were lost (and never recovered despite efforts) prior to 04 April due to defective stainless steel hose clamps, and the surviving receivers were pulled as a precaution; monitoring at all five OGC sites resumed on 30 May. During 09-10 March 2015, five VR2W receivers in both tidal creeks behind Duck Island and a sixth receiver site established near the Cosgrove Bridge in 2014 were relocated to various locations throughout the Ashley River, mostly upriver from the Cosgrove Bridge (Figure 8). Five VR2W receivers were added to two tidal creek headwaters in OGC on 02 September 2015 and six VR2W receivers located upstream of the Cosgrove Bridge were pulled on 09 December 2015 in preparation for a shift in telemetry focus to OGC in 2016 (Figure 8).
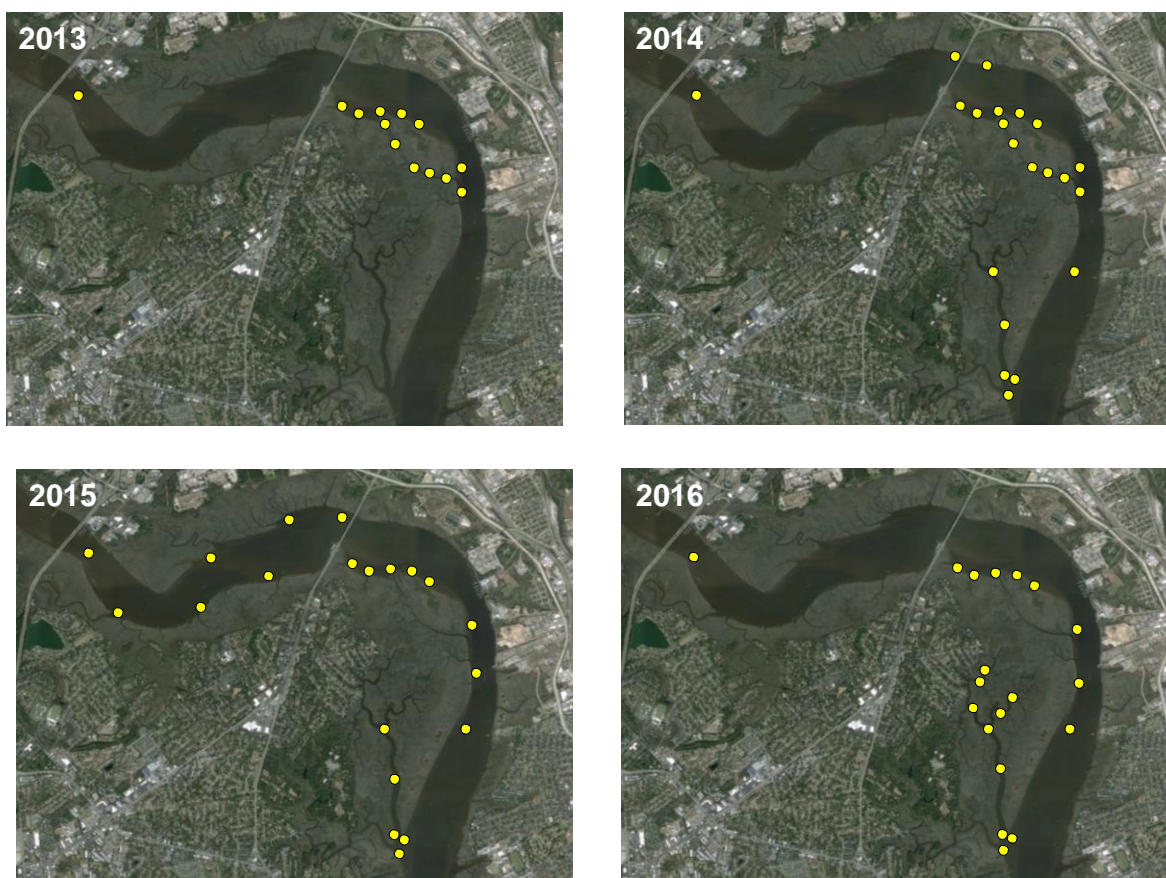
Figure 8. Between 9 April 2013 and 9 December 2015, partial annual acoustic telemetry data were collected by VR2W acoustic receivers at 31 different listening stations in the Ashley River and adjacent tidal creeks; 13 monitored creek habitats and 18 monitored river habitats, with between 13 and 25 maintained at any given time.

CTD data

On 09 April, a Levelogger Junior (Solinst, Inc.) CTD was deployed at the central VR2W receiver location in the South Creek to record water temperature (°C) and conductivity (uS cm^{-1}) data at 15-minute intervals. This location was maintained through 09 December 2015, with a slight (3 ft.) adjustment in position in spring 2014 to reduce the probability of air exposure at low tide.

Conductivity was converted to salinity (ppt) with the "Stevens EC to Salinity" spreadsheet provided by D. Sanger (SCDNR/MRD/MRRI; sangerd@dnr.sc.gov) that includes the following methods description: "Salinity is calculated from the un-normalised or normalised conductivity according to the algorithm outlined in Standard Methods for the Examination of Water and Wastewater, 18th Edition, p. 2-47. The equation for Rt was taken from "Specific Conductance: Theoretical considerations and application to analytical quality control" by R.L. Miller, W.L. Bradford, and N.E. Peters. United States Geological Survey Water-Supply Paper 2311. 1988. USGS, Federal Center Box 25425, Denver, CO 80225. A value of 1.84 %/ deg C is used as the un-normalising factor (the same value used in the EC200 to normalise)."

Statistical analyses
Statistical analyses were performed in Minitab 15® (Minitab, Inc.; State College, Pennsylvania). A significance level of $\alpha = 0.05$ was assumed for all statistical tests, which were selected after first testing the underlying data distributions for normality (Anderson-Darling test).

Hierarchical cluster analysis was used to evaluate similarity between detection histories for acoustically-tagged diamondback terrapins in order to assess gross influences on variability among individuals (by variable) as well as to assess whether excessive variability would preclude pooling data among individuals (by observation) to examine detection trends. Cluster analysis source data for each acoustically-tagged diamondback terrapin included sex (0 = female, 1 = male); capture month; capture year; spatial capture zone (Appendix 1: 0 = lower Ashley; 1 = tidal creek; 2 = Duck Island; 3 = upriver from the Cosgrove Bridge); number of days with detection data; minimum detections per day; maximum detections per day; mean detections per day; standard deviation (SD) among daily detections; and total detections.

Kruskal-Wallis (KW) tests were used to assess male vs. female differences in the number of days detected and total detections recorded.

A Chi-square contingency test was used to evaluate habitat (creek vs. river) detections relative to the cumulative number of days of monitoring in each habitat across receivers. KW tests were then used to test for differences in the percent distribution of detection bins among diamondback terrapins in these two broad habitat types overall and partitioned by terrapin sex.

Detection bins among diamondback terrapins were partitioned temporally (year-month) with respect to creek and river habitats, and the collective number of bins was divided by the number of unique terrapins detected in each habitat type during each year-month observation period. Correlation testing was then used to assess relationships between monthly detection bins per terrapin and (a) mean monthly water temperature and (b) mean monthly salinity. A percent distribution of hourly detection bins with respect to local standard time (LST) was computed independently for creek and river habitats; correlation testing was then used to evaluate a potential relationship in time of day detection among these two habitat types.

Tidal influence was evaluated by establishing frequency distributions for water depth at 0.5 m increments and then evaluating correlation strength between various partitioned data groups: available CTD data (flood vs. ebb); river habitat (flood vs. ebb); creek habitat (flood vs. ebb); pooled river vs. pooled creek habitat; and pooled water level at detection vs. available CTD data.

Gross detections per hour per transmitter per day were computed to provide a metric to gauge temporal variability in diamondback terrapin residence patterns, with the presumption that greater detection rates indicated a higher degree of localization in proximity to receivers. Seasonal (Jan–Mar; Apr–Jun; Jul–Sep; Oct–Dec) and annual (2013, 2014, 2015) differences in gross detection rates were evaluated using KW tests and Bonferroni multiple comparisons. The potential influence of predicted (a cumulative function of deployment date and life expectancy) daily active transmitters and daily gross detection rates was evaluated with correlation testing.

All unique combinations of linear receiver spacing (km), the sum of (n receivers)$^2 - n$ receivers, were computed in MS Excel as the square root of squared latitude differences plus squared longitude differences (i.e., Pythagorean Theorem), after these differences were multiplied by 110.85 km and 96.49 km per degree of latitude and longitude, respectively. Annual receiver spacing distributions were established based on the maximum number of receivers deployed in each calendar year, and a KW test and Bonferroni multiple comparisons were then used to evaluate potential differences in the distribution of annual receiver spacing among years. Distance (km) from capture location for each acoustic detection of diamondback terrapins was calculated in the same manner. Frequency distributions (%) for overall receiver spacing and overall linear detection distance (km) from capture site were then computed and evaluated for similarity using correlation analysis. Cluster analysis was also used to evaluate the association between mean displacement distance for each diamondback terrapin and (a) year of capture, (b) month of capture, (c) terrapin sex, and (d) spatial capture zone, partitioned as in a previous test.

Daily mean and maximum linear terrapin detection distance (km) with respect to the number of days elapsed since tag and release was computed, and correlation testing was then used to evaluate potential temporal influence on linear detection distance trends.

Descriptive statistics were used to characterize detection patterns for acoustically-tagged diamondback terrapins that were (a) detected by receivers maintained by other SCDNR groups, (b) potentially identified as outliers following initial cluster analysis, and (c) represented outliers with respect to handling times between initial capture and subsequent tag and release. Visualization of movement patterns for these diamondback terrapins was performed using the Earth Point tool for Google Earth (https://www.earthpoint.us/ExcelToKml.aspx).

Results

Detection overview
Forty-four diamondback terrapins (26 female, 18 male) were tagged with acoustic transmitters following capture by trammel net (39), crab trap (3), trawl (1), and by hand (1) in the Ashley River or adjacent tidal creeks between 09 April 2013 and 01 June 2015. With the exception of a female terrapin (MT0495) temporarily housed (99 days) at the South Carolina Aquarium Sea (SCA) Turtle Hospital following re-attachment of the left forelimb in 2015, acoustically-tagged diamondback terrapins were generally released where captured 24 to 48 hours earlier. Two additional diamondback terrapins (11.5 cm female, 12.1 cm male) reared at the SCA for several years were also tagged with acoustic transmitters and released in the upper reaches of OGC on 08 September 2015, to evaluate post-release acclimation behavior; data for these two terrapins were analyzed separately because of the different hypothesis of interest.

Between 11 April 2013 and 09 December 2015 (last receiver upload), a total of 146,782 detections of 44 wild terrapins were recorded by 13 to 25 acoustic receivers in the Ashley River and adjacent tidal creeks (Figure 8). All diamondback terrapins tagged with acoustic transmitters were subsequently detected by receivers; however, a high degree of individual variation in the frequency and abundance of detections was observed (Appendix 2). Hierarchical cluster analysis revealed 94% similarity among three daily detection metrics (mean, max, and SD), but 57–60% similarity between these metrics and terrapin sex, month, year, or spatial capture zone within the Ashley River study system. Hierarchical cluster analysis also revealed 98–100% similarity in

detection histories among 42 of 44 acoustically-tagged diamondback terrapins, suggesting that despite individual variability, analysis of broad trends was a reasonable pursuit. Two diamondback terrapins collectively accounted for 36% ($n = 53,505$) of total detections and were associated with 89% similarity to each other but 30% similarity to other terrapins; thus, detection histories for these two outlier diamondback terrapins were analyzed independently.

No significant difference was detected ($H_1 = 0.05$, $P = 0.828$) in the distribution of detection days for male ($n = 17$) and female ($n = 25$) terrapins, which ranged from a minimum of five to eight, a maximum of 232, and identical (81) median days of observation (Appendix 2). No significant difference was detected ($H_1 = 0.87$, $P = 0.350$) in the distribution of total detections with respect to terrapin sex; individual males were detected 49 to 5,629 times (median = 3,251; IQR = 3,422) and individual females were detected 110 to 5,275 times (median = 1,919; IQR = 2,450).

River vs. creek detections
Across 31 unique receiver locations, a total of 17,181 monitoring days (7,335 creek; 9,846 river) were conducted between 11 April 2013 and 09 December 2015, which recorded 93,277 detections for 42 diamondback terrapins that were in turn partitioned into 21,610 hourly bins. A significant difference was noted ($\chi^2_1 = 5,504$, $P<0.001$) in the frequency of hourly detection bins in river (90%; $n = 19,434$ bins) vs. creek (10%; $n = 2,176$ bins) habitats and the distribution of receiver monitoring effort in those habitats. Hourly bin distribution between creek and river habitats was also significantly different among terrapins ($H_1 = 63.07$, $P<0.001$), irrespective of sex ($H_1 = 0.46$, $P = 0.496$), with 99.5% (median) of hourly bins associated with river habitats, but only 0.5% (median) of hourly bins associated with creek habitats (Figure 9).
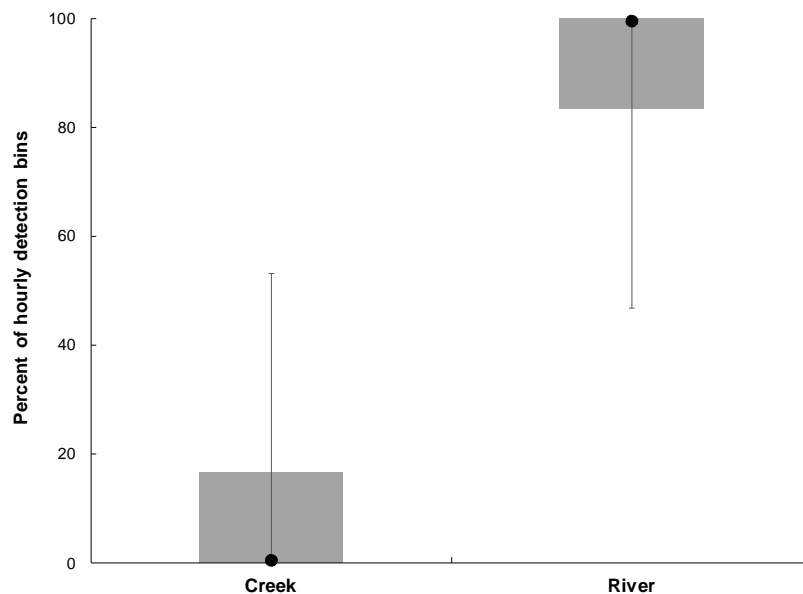
Figure 9. Among 42 diamondback terrapins with <6k detections, 99.5% (median, circle) of hourly bins were associated with river habitats (0.5% with creek habitats); gray bars indicate inter-quartile range and error bars denote minimum and maximum percentages for terrapins.

Diamondback terrapins were detected by river receivers in all 33 months of observation in the present study, during which time gross monthly detections per hour per terrapin (Figure 10a) ranged from three (January 2014) to 259 (November 2015). CTD data were collected during 85% (n = 826) of monitoring days, but were unavailable for two temporal blocks (29 August to 13 October 2014; 02 September to 09 December 2015) consequent to flooding of the CTD unit and subsequent device failure. During 29 months of river detections with corresponding CTD data, gross detections per hour per terrapin were significantly correlated with mean monthly water temperature (r = 0.46, P = 0.012) but not with salinity (r = 0.14, P = 0.458).

Diamondback terrapins were detected by creek receivers during 19 of 33 months, with gross monthly detections per hour per terrapin (Figure 10b) that ranged from one (November 2013) to 90 (June 2015). CTD data were available for all but one month with creek detections, but the frequency of hourly bin detections in creek habitats was not significantly correlated with either water temperature (r = 0.39, P = 0.106) or salinity (r = 0.24, P = 0.329).
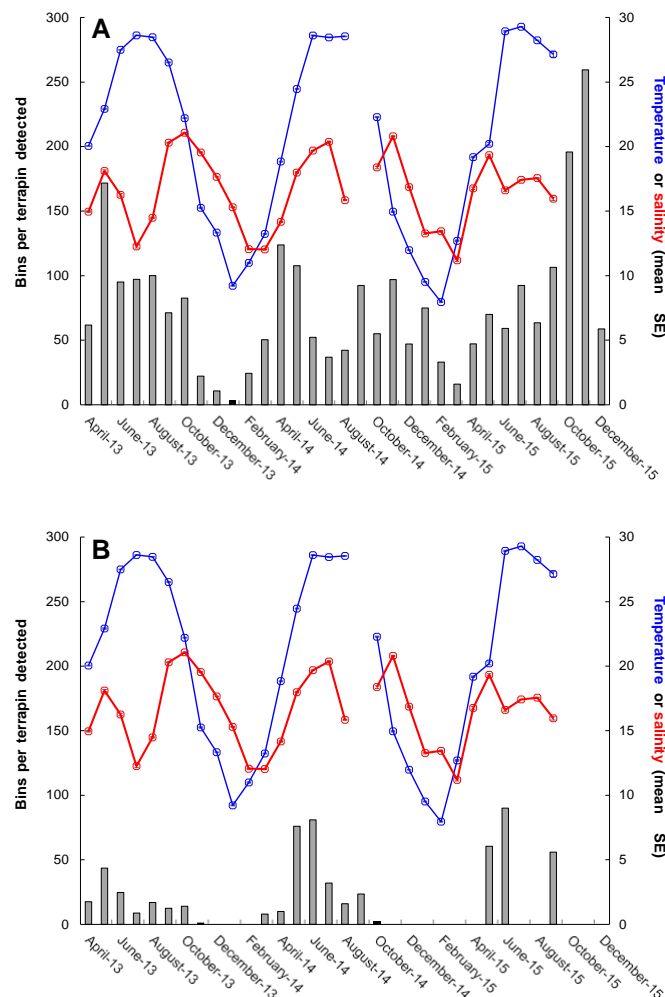
Figure 10. Diamondback terrapins were detected in all months in river habitats (A), but were only detected in creek habitats (B) during 19 of 33 observation months. A significant correlation between mean water temperature and monthly detections per hour per terrapin was noted for river receivers; however, such correlations with salinity were not significant in either habitat.

A significant negative correlation was detected (P<0.001, r = -0.85) in the distribution of hourly detection bins with respect to time of day among receiver habitats (Figure 11); between 06:00 and 17:59 hours LST, 67% of hourly detection bins occurred in creek habitats compared to just 42% of hourly detection bins in river habitats.

Tidal associations with the frequency of hourly detection bins reflected observed water levels, with significant correlation between hourly detection bin frequency during ebb ($n = 9,296$ bins) and flood ($n = 9,343$ bins) tide stages in both creek and river habitats (Table 2).
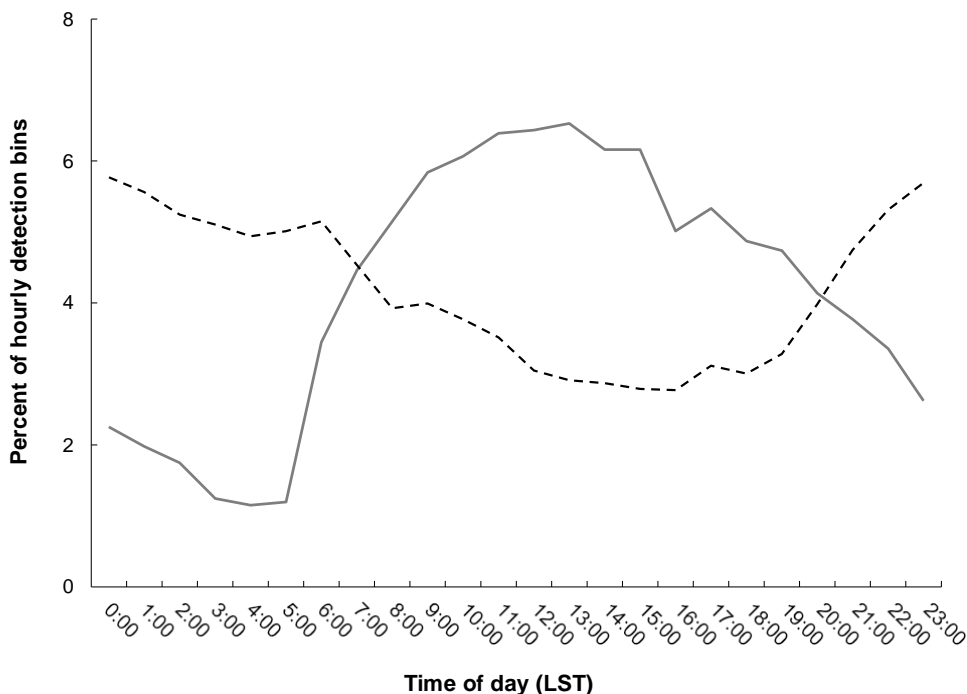
Figure 11. A significant inverse correlation was noted in the diel distribution of hourly detection bins between creek (solid gray line) and river (dashed black line) habitats; 67% of hourly detection bins in creek habitats occurred between 06:00 and 17:59 hours LST, compared to just 42% of hourly detection bins in river habitats during the same timeframe.

Table 2. Correlation testing revealed significant similarity between water level and tide phase; detection frequency and tide phase within and among habitats; and between detection frequency and available water level data.

Correlation test	P-value	Co-efficient (r)
Available CTD, flood vs. ebb	0.001	0.94
River, flood vs. ebb	<0.001	1.00
Creek, flood vs. ebb	0.001	0.96
Pooled River vs. Pooled Creek	<0.001	0.96
Pooled Habitat vs. Pooled CTD	0.008	0.85

Activity and movement patterns

Across habitats, gross detections per terrapin per hour per day between 11 April 2013 and 09 December 2015 ranged from zero (132 days, 14% of observations) to 10 (Figure 12), the upper limit representing two-thirds of maximum detection potential with a four-minute repeat interval. During the present study, the number of diamondback terrapin transmitters actively deployed daily ranged from one to 21 with a median of 11 active transmitters per day (Figure 12). Despite significantly more ($H_2 = 354$, P<0.001) active transmitters daily in 2013 than in 2014 or 2015, a significant inverse correlation with gross detections per hour was detected (P<0.001, r = -0.107).

A significant difference was detected ($H_3 = 277$, P<0.001) in the seasonal distribution of gross detections per hour, with greatest detection frequency observed during April–June ($n = 263$; median = 3.9; IQR = 1.7), the fewest detections during January–March ($n = 180$; median = 1.0; IQR = 1.7), and similar detection frequencies between July–September ($n = 276$; median = 3.0; IQR = 1.8) and October–December ($n = 254$; median = 3.2; IQR = 3.5). A significant difference was also detected ($H_1 = 64.96$, P<0.001) among years, but greatest detection frequency in 2013 ($n = 265$; median = 3.8; IQR = 2.1) likely stemmed from the lack of January–March observations given that annual differences between 2014 and 2015 were not significant.
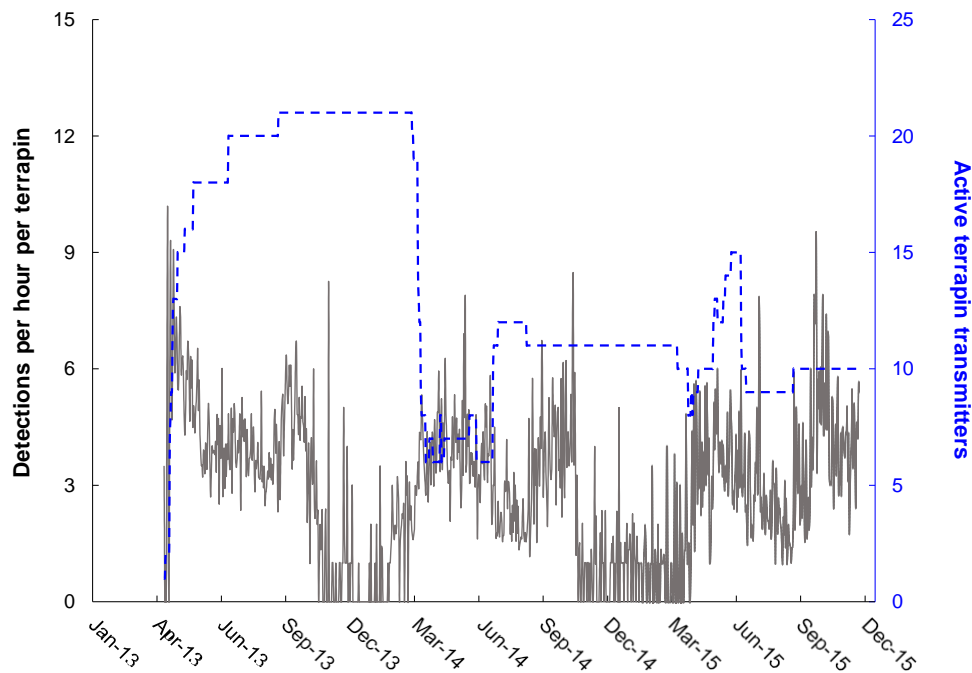
Figure 12. Greatest gross detections per diamondback terrapin per hour per day (gray line) occurred during April–June, independent of the number of acoustically-tagged diamondback terrapins available to be detected on any given day (blue line).

Distance between 31 acoustic receiver locations ($n = 930$ unique combinations) monitored between 11 April 2013 and 09 December 2015 ranged from 0.0 to 6.2 km (median = 1.9 km). Due to expanding and re-positioning receiver coverage across years, a significant difference in the spacing of receivers was detected across years ($H_2 = 21.62$, P<0.001), concurrent with a steady increase in median spacing between 2013 (1.6 km), 2014 (1.9 km), and 2015 (2.1 km).

Eighty-seven percent of detections occurred at receivers located ≤1 km from capture sites (Figure 13), with a maximum distance of 4.7 km (median = 0.3 km, n = 93,277). Conversely, only 19% of receiver spacing transects were located ≤1 km apart (Figure 13); consequently, these two spatial distributions were not significantly correlated (r = -0.11, P = 0.715) with each other. Hierarchical cluster analysis revealed the greatest attribute association for mean detection distance from capture site (range = 0 to 3 km; n = 42) with year of capture (78% similarity); significantly greater (H_2 = 9.30, P = 0.010) mean movement for ten acoustically-tagged diamondback terrapins in 2015 (median = 1.3 km; IQR = 1.9 km) was observed relative to mean movements for 21 acoustically-tagged terrapins in 2013 (median = 0.4 km; IQR = 0.4) and 11 acoustically-tagged diamondback terrapins in 2014 (median = 0.4 km; IQR = 0.7 km). Capture month (63%) and sex and spatial capture zone (52% similarity) did not influence movements.
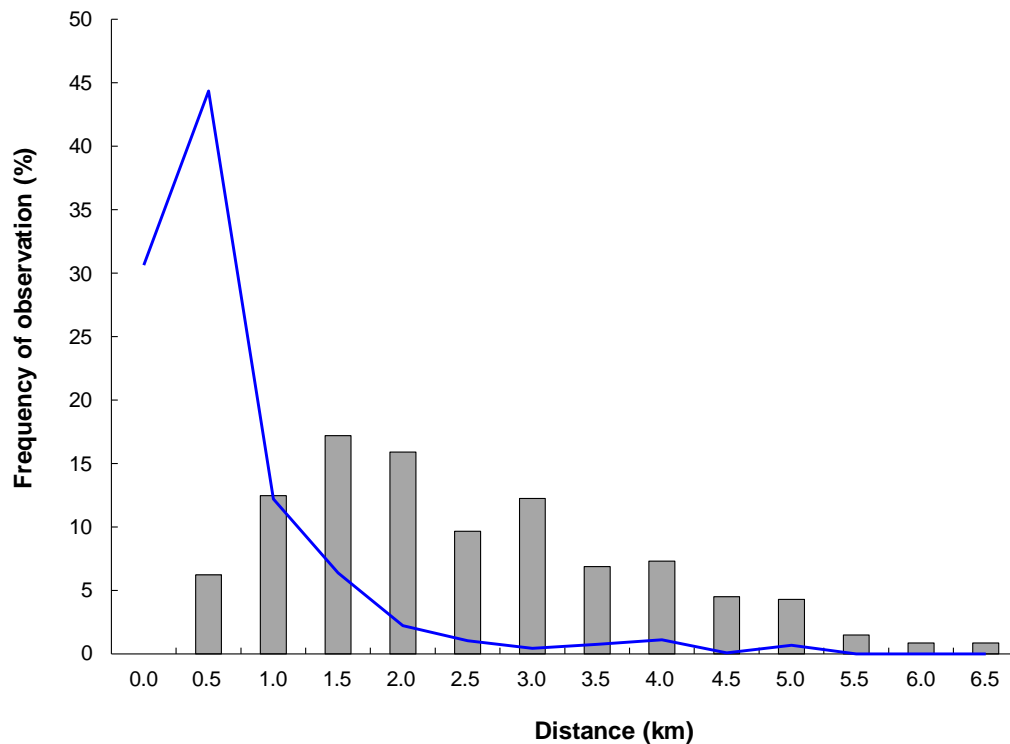
Figure 13. Distance from capture site (blue line) associated with 93,277 detections for 42 acoustically-tagged diamondback terrapins during 2013–2015 was not significantly correlated with 930 unique combinations of spacing between 31 receiver locations (gray bar).

Detection up to 4.7 km from capture sites occurred irrespective of days post-release, but was most frequently documented within 100 days after capture (Figure 14). Conversely, greatest mean detection distance from where released occurred after 200 days at large, generally concurrent with the least difference between mean and maximum distance metrics (Figure 14). A significant (P<0.001) correlation between mean and maximum distance metrics was observed before and after the 200-day benchmark; however, correlation co-efficient strength between these two data series improved by nearly 50% between 0 and 199 days (r = 0.47) and 200 to 378 days (r = 0.68) following capture, tag, and release.
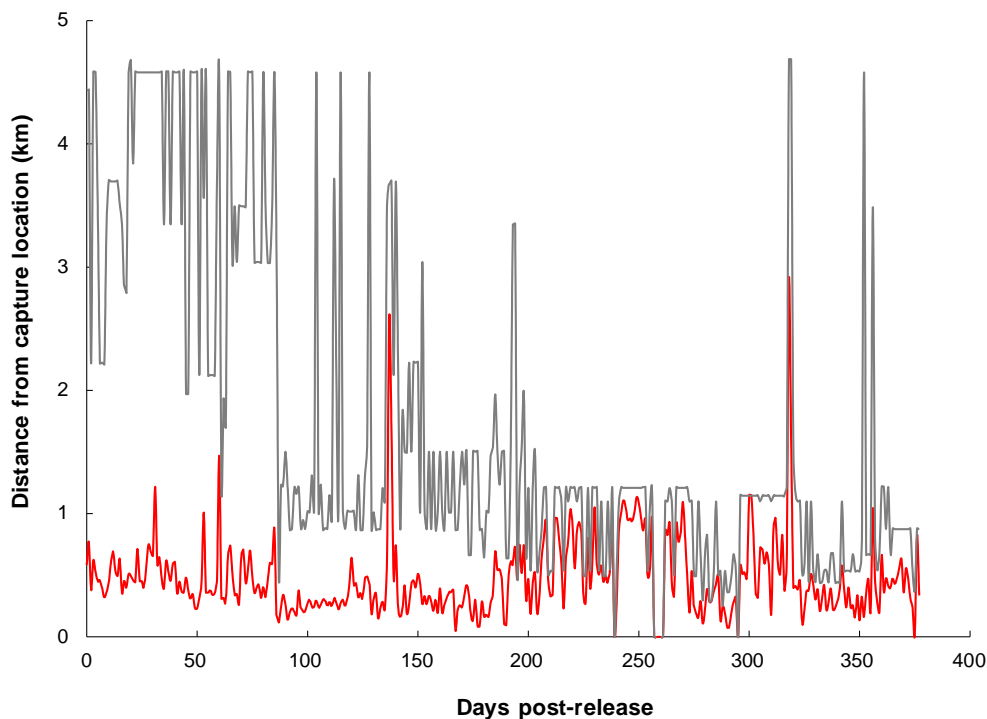
Figure 14. Maximum (gray line) detection distance (y-axis) of acoustically-tagged diamondback terrapins away from capture location generally occurred shortly after tag and release; however, greatest mean detection distance (red line) typically occurred after 200 days post-release (x-axis).

Four of 42 diamondback terrapins were also detected ($n = 102$) by eight fixed location acoustic receivers maintained by two other research groups within the SCDNR, all but one of which were located outside of the spatial receiver boundaries of the present study. Seventy-five percent of non-study detections were associated with a single acoustically-tagged terrapin detected by the within study boundary receiver maintained by the SCDNR Inshore Fisheries group, consistent with the observation of predominantly localized movements within the monitored study area.

A female diamondback terrapin (A69-1303-25234) captured in the North Creek of Duck Island was detected 494 times across nine of 13 receivers between 30 April and 11 May 2013; however, a day later, six terminal detections were recorded by two receivers maintained by the SCDNR Diadromous Fisheries group located another 6 to 9 km from where last detected (Figure 15a).

A single (and terminal) detection for a male terrapin (A69-1303-218) captured just north of the Rt-7 bridge (AR23) on 11 July 2014 was recorded by a receiver at the Breakwater Marina maintained by the SCDNR Diadromous Fisheries group; this detection occurred 11 days after this terrapin had moved 3.3 km downriver from where it was captured, and if valid, represents an additional concentric displacement of 6.1 km and movement into a different river (Figure 15b).

A male terrapin (A69-1303-225) captured at Duck Island (AR11) on 20 April 2015 immediately emigrated away (four detections) from the capture site upon release and for the next five days was detected 229 times across five receivers located 1.6 to 4.4 km upriver from where captured but returned (five detections) to within 1 km of where captured before moving 2 to 3.5 km

downriver by 26 April (75 detections). On 30 April, five detections were recorded by a receiver maintained by the SCDNR Diadromous Fisheries group located 7.3 km from where captured; this terrapin was again detected (four times) at this and another nearby (0.2 km) Diadromous receiver on 14 May. Ten terminal detections across three additional Diadromous receivers (concentrically located 5.8 to 6.8 km from where this terrapin was captured) were recorded during 15–19 May; however, some aspects of this detection track are questionable (Figure 15c).

Long-distance movement of diamondback terrapins within the range suggested by telemetry data is corroborated by the capture and re-sighting history of female terrapin (MT0152; Figure 15d). Originally captured near Duck Island (AR10) on 14 April 2014, this female terrapin was held in captivity at the South Carolina Aquarium for 163 days as part of a behavioral study to evaluate modifications to by-catch excluder devices (see Chapter 3). On 24 September, MT0152 was released at the Dolphin Cove Marina approximately 0.4 km from where captured; inclement weather necessitated shore-based release, hence why release was not 0.0 km from capture site. On 27 October, MT0152 was recaptured at the James Island Yacht Club, the same location where A69-1303-25234 was detected on 12 May 2013, 10.2 km from where originally captured. On 21 May 2015, MT0152 was re-sighted attempting to climb concrete stairs (Figure 16) at an office complex located near the furthest upriver receiver maintained in the present study and approximately 15 km upriver from where re-sighted on 27 October. Eleven days later, MT0152 was recaptured in the trammel net survey at site AR 12, 1.2 km from where captured in 2014.
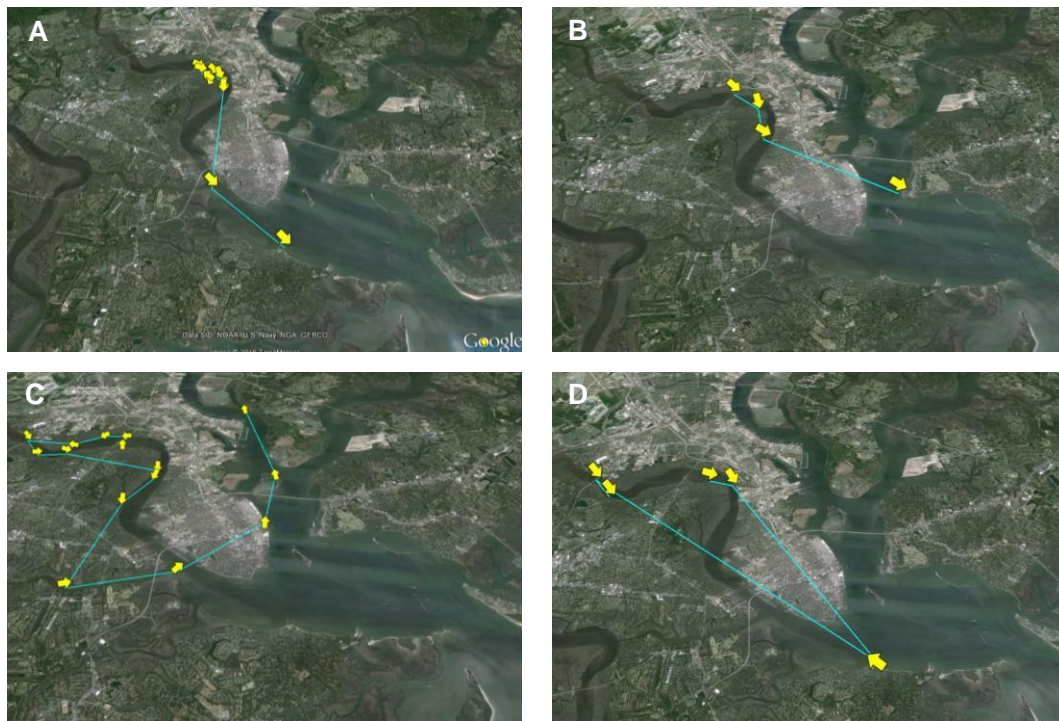
Figure 15. Linear (blue line) movement direction (arrows) of individual acoustically-tagged diamondback terrapins detected by receivers maintained by other SCDNR research groups in 2013 (A), 2014 (B), and 2015 (C). Three re-sighting and recapture events of MT0152 (D) between 27 October 2014 and 1 June 2015 corroborate movement magnitude and direction.

Figure 16. On 21 May 2015, seven months after being recaptured in a cast net at the James Island Yacht Club, MT0152 was observed climbing stairs at an office building located not far from the most up-river acoustic receiver near the I-526 Bridge. Photo courtesy of Ashley Moore.

A total of 28,230 detections were recorded for male terrapin (A69-1303-221), 96% of which occurred at the closest (0.4 km) receiver (Cosgrove West) to where this terrapin was captured (AR23) on 14 July 2014; three percent of detections for this terrapin were recorded by an equidistant receiver site (AR28) established in March 2015. Given the exceptionally limited movement exhibited by this terrapin, temporal variability in detection (Figure 17) likely reflects a high proportion of time spent in an adjacent and extensive tidal creek, or possibly transmitter loss that produced tidally-mediated variation in signal reception by these receivers, which would also suggest that the remaining 54 detections across four Duck Island receivers were aberrant.
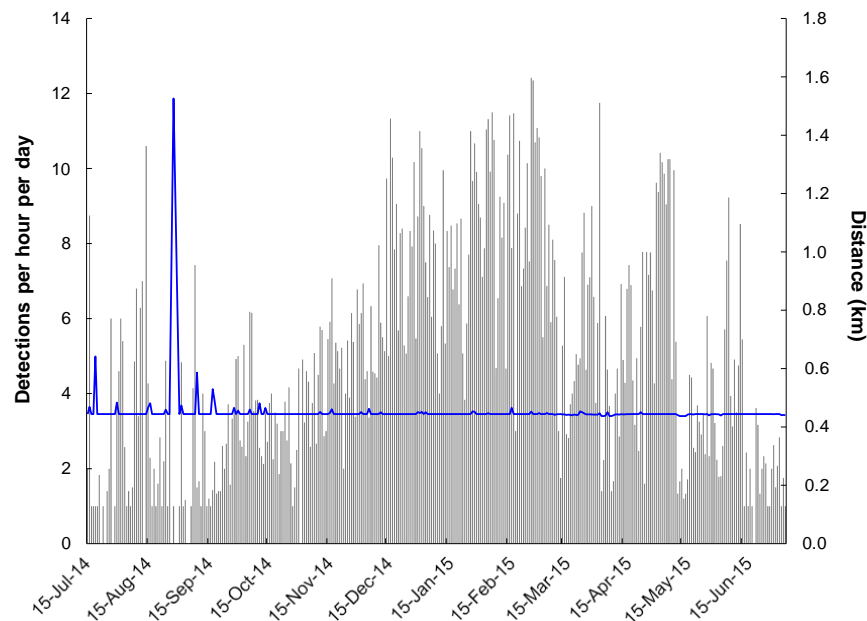
Figure 17. All but 54 of 28,230 detections for a male terrapin were recorded by two receivers located 0.4 km to either side of a creek inlet, indicating extreme residence or transmitter loss.

A total of 25,280 detections were recorded for female terrapin (A69-1303-224), 90% of which were recorded by the middle receiver in the main stem of OGC which was located 2.8 km (linear) from AR30 where this terrapin was captured on 17 April 2015, but which would have required a transit more than three times as great to reach. The remaining 10% of detections were associated with four transit events past 14 different receivers located in the river between Duck Island and OGC (or within OGC) through 26 August, and nine potentially aberrant detections between 30 September and 9 December (Figure 18).
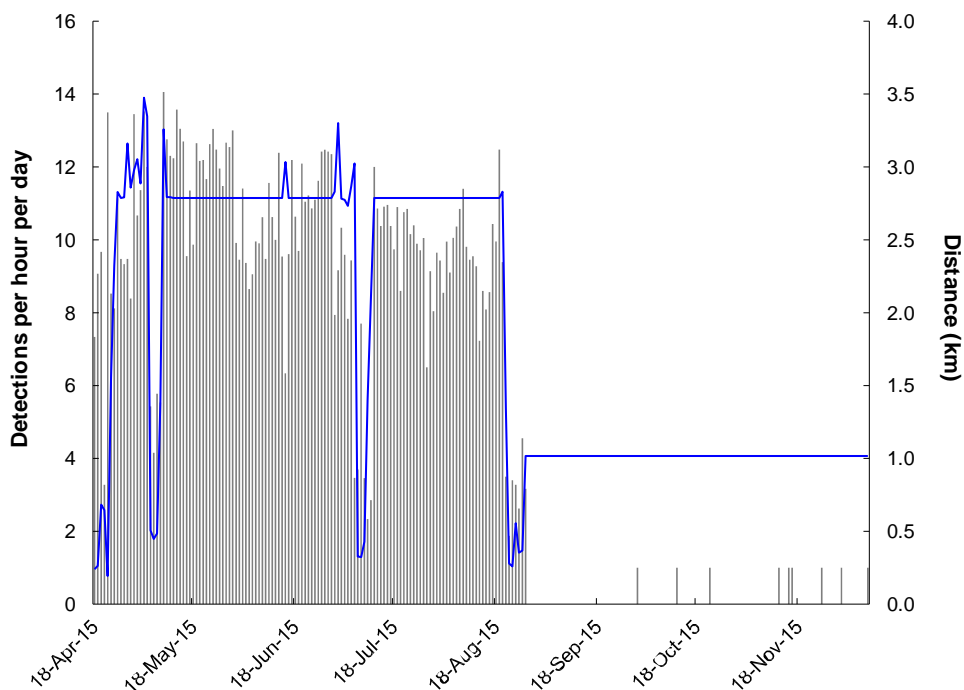
Figure 18. Ninety percent of 25,280 detections for a female terrapin were recorded by a single receiver in Orange Grove Creek, 2.8 km from where this terrapin was captured; however, detection across 14 different receivers during four transit events do not suggest transmitter loss.

In collaboration with the South Carolina Aquarium Sea Turtle Hospital, five captive-reared (from hatchling stage with unknown origin) diamondback terrapins (11.0 to 12.8 cm SCL) were released in a headwater tidal creek of OGC on 08 September 2015; all five terrapins received PIT tags prior to release, and one of three males (11.5 cm) and one of two females (12.1 cm) were also tagged with acoustic transmitters.

A total of 338 detections across 16 receivers (nine maintained by other SCDNR programs) were recorded for the male terrapin (A69-1303-33659) through 07 October. Fifty-two percent (*n* = 176) of detections were recorded in the upper reaches of OGC during the first two days post-release; however, on 11 September this terrapin departed OGC and headed towards Charleston Harbor, reaching the mouth of the harbor the next day and detected in this general vicinity through 14 September (Figure 19a). During 20–21 and 27 September, this terrapin was detected at two locations in the Cooper River, followed by the final 68 detections during 4–7 October at a receiver located near Drum Island in the Wando River (Figure 19a).

A total of 119 detections across 16 receivers (five maintained by other SCDNR programs) were recorded for the female terrapin (A69-1303-33660) through 1 October, with two additional single detection events on 7 and 10 October. In contrast to the captive-reared male, this female terrapin was only detected five times in the upper reaches of OGC on the day of release before disappearing from detection for six days, then re-appearing in the Ashley River between Duck Island and the entrance to OGC. Between 14 September and 1 October, 114 detections were recorded in the lower Ashley River between Bull Creek and Wappoo Creek, with multiple passes through this stretch of river (Figure 19b). Six days after last being detected in the same general vicinity of the Ashley River where first detected after emerging from OGC, two single detections were recorded at the entrance of Charleston Harbor and later at the Folly Beach Pier, suggesting that this terrapin may have become entrained in a freshwater plume that stemmed from historic flooding associated with Hurricane Joaquin.

Figure 19. One captive-reared male (A) and one captive-reared female (B) diamondback terrapin raised from hatchlings at the South Carolina Aquarium were tagged with acoustic transmitters before release on 08 September 2015. Both were highly mobile during nearly one month of data collection, but also exhibited movement patterns that for the most part were quite similar to the most mobile of 44 wild-captured terrapins monitored in the present study during 2013–2015. Arrows indicate direction of movement and blue lines indicate linear movement magnitude.

Chapter 3. Reducing diamondback terrapin mortality in crab traps.

Introduction

Diamondback terrapin populations were overfished for human consumption in the late 1880s (Carr 1952), but today the top conservation threat for this long-lived (~40 years) species (Hildebrand 1932) is purported to be drowning in crab traps (Gibbons et al. 2001; Butler and Heinrich 2007; Dorcas et al. 2007; Grosse et al. 2011; Isdell et al. 2015). Annual mortality of terrapins due to drowning in crab traps is not empirically known, but in one study in a tidal creek in Georgia 94 terrapins were observed dead in a single crab trap (Grosse et al., 2009), highlighting the magnitude of damage that is possible in the blue crab trap fishery. Crab trap capture rates for diamondback terrapins were first reported in the Charleston Harbor, SC estuary by Bishop (1983), who provided two key observations of management interest. First, 87% of captures occurred in spring when diamondback terrapins aggregate to mate (Lee 2003), with the remainder of entrainment in late spring through summer. Second, male terrapins were captured twice as often as females, which likely reflects the smaller size at maturity of males (Ernst et al. 1994) and their subsequent susceptibility to capture. Although females may outnumber males at breeding sites (Coker 1920), high crab trap mortality rates for males could eventually lead to altered demographic structure (Dorcas et al. 2007).

In the past 20 years, a considerable amount of effort has been expended to reduce the drowning of diamondback terrapins in crab traps through variations on a bycatch reduction device (BRD) first introduced by Wood (1997). Results vary across studies, but generally indicate favorable reductions in terrapin entrainment while rarely demonstrating no negative impacts to crab catch; consequently, few states have adopted legislation requiring BRDs and where such laws do exist, compliance is generally quite low (Radzio et al. 2013). In South Carolina, commercial fishers confirm that terrapins are killed in crab traps during the spring when aggregation behavior is most prolific; however, collaborative research conducted during 2006–2009 suggested that commercially-available BRDs could be economically costly to crabbers (Powers et al. 2009a).

Disproportionate detection of acoustically-tagged diamondback terrapins in river vs. tidal creek habitats in the Ashley River (Chapter Two) suggests that diamondback terrapins may be more vulnerable to drowning in commercial crab traps than previously appreciated; thus, additional research into this important topic was conducted during the final two years of the present study. The first objective was to conduct a meta-analysis of existing BRD studies to identify potential relationships between legal blue crab catch (and terrapin exclusion) and different BRD designs. The second objective was to conduct fishery-independent field data collection to evaluate catch rates, size distribution, and sex ratio of blue crabs (*Callinectes sapidus*) and diamondback terrapins captured in traps fished without BRDs, traps fished with the mostly widely-tested BRD, and two modifications to this BRD reflecting change in horizontal and vertical dimension size. The third objective was to analyze catch and morphometric data collected in year one to refine and test an improved BRD design in year two. The fourth objective was to visually observe blue crab and terrapin interactions with trap funnels outfitted with and without BRDs to assist in the interpretation of potential catch rate, size, and sex ratio differences among traps.

Methods

Objective 1: Meta-analysis

A literature review was conducted in Google Scholar to identify published and un-published data sets that enabled comparisons between crab traps fished with (treatment) and without (control) various BRDs. Relative change in species-specific catch in treatment traps was expressed as a percent of control trap catch for each design tested in each study. Percent change distributions for each species were first evaluated with respect to relative abundance for each target species, determined based on *ad hoc* control sample size distribution; Kruskal-Wallis (KW) tests with Bonferroni multiple comparisons were used to evaluate potential influence of control trap sample size on observed BRD efficacy. Correlation testing was used to evaluate potential influence between diagonal BRD dimension, the maximum aperture which was calculated using Pythagorean Theorem, and percent change in species catch in BRD vs. control traps.

Objective 2: BRD design testing, Year 1
Four crab trap colors (black, red, yellow, bright green) that measured 61 cm (2 ft.) on top, bottom, and sides were manufactured by Beaufort Marine Supply in Burton, SC, and were selected to represent the four most commonly-sold trap colors by the manufacturer. Control trap funnel openings (Figure 20a) were oval-shaped and measured 5.1 cm (2 in.) vertically and 17.8 cm (7 in.) horizontally. The first BRD design (Figure 20b) measured 5.1 cm vertically and 15.2 cm (6 in.) horizontally; was made of a hard, orange plastic that was secured to the inside of the crab trap funnel using plastic zip ties; was manufactured by Top-ME (Topsham, ME); and is herein referred to as the "standard" BRD. The second BRD design (Figure 20c) represented a 1 cm (3/8th in.) reduction in the vertical opening size relative to the standard BRD, which was accomplished via a 0.3 cm (1/8th in.) brass bar passed through the BRD and secured with epoxy (Powers T-308; Brewster, NY) on the non-entry side of the BRD; this design is herein referred to as the "reduced height" BRD. The third BRD design (Figure 20d) represented a 3.2 cm (1.25 in.) reduction in horizontal opening size relative to the standard BRD to achieve the same horizontal dimension (but larger vertical dimension) as an alternative Top-ME BRD design. The reduction in horizontal dimension for the BRD design was accomplished by making a transverse cut across the BRD and then splicing the two pieces back together by securing them with plastic zip ties. This BRD was also positioned with the 5.1 cm dimension horizontally and the 12.1 cm dimension vertically; thus, this BRD is herein referred to as the "vertical" BRD.

Crab trap fishing was conducted at 24 locations in the greater Charleston, SC area (Figure 21) in water body sizes ranging from river to small tidal creek. Given tidal amplitudes associated with these creeks and a desire to not leave traps unattended, traps were only fished in small tidal creeks during periods with sufficient water volume for boat access at low tide. Trap fishing occurred between April and November 2014.

Traps were baited with a single Menhaden and fished in clusters of four that included a control trap and each of three treatments described above. Initially, all four trap colors were fished as a cluster, but this practice was later changed to ensure that each cluster of four traps included four different colors to eliminate auto-correlation between color and fishing location. Each trap constituted a sampling event, and each trap retrieval time was denoted with a sub-level code; this practice allowed for later examination of within-event variation of catch rates as so desired. Traps in each cluster were spaced approximately 20 m apart and fished as close to the edge of the marsh as possible, based on anecdotal reports from several crabbers that this practice is associated with increased terrapin catch. Target soak time was one hour before pulling traps for

examination; this threshold was considered a compromise between trap disturbances that could discourage crab entry while simultaneously decreasing the risk of terrapin drowning.
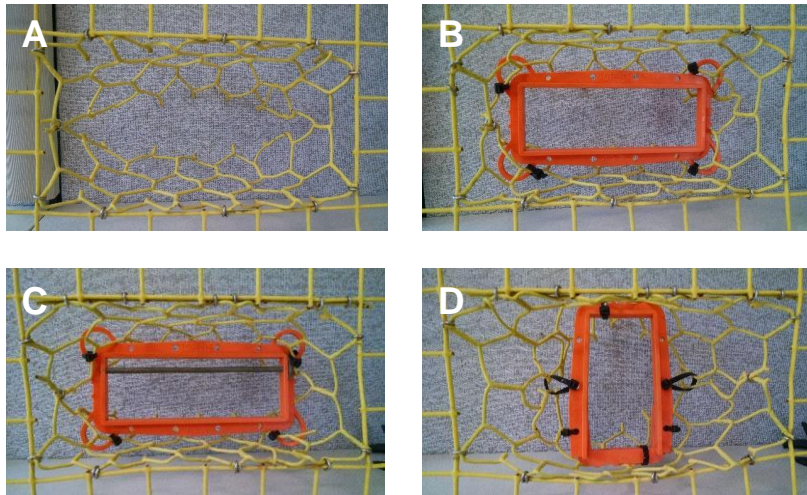
Figure 20. Crab trap funnels without BRDs (A) were used to evaluate changes in catch rates, size distribution, and sex ratio of blue crabs and diamondback terrapins captured in crab traps where funnels were outfitted with a standard BRD (B), a 1 cm reduction in the vertical opening size of this BRD (C), and a 3.1 cm reduction in the horizontal opening size of this BRD that was then also rotated 90° (D) to determine potential impacts to capture rates and catch composition.

Figure 21. Crab traps were fished in clusters of four (control plus three treatments) at 24 different locations in three areas: Ashley River near Duck Island (yellow fill); Orange Grove and Old Towne Creeks (orange fill); and the Stono River (green fill) within 2 km of Elliott's Cut.

When captured, the number of crabs and terrapins in each trap was recorded as well as their relative position within the trap (i.e. in the lower or upper chamber). Each crab and terrapin was assigned a sequential project identification number. Terrapin data collection was the same as described in Chapter 1. Three caliper measurements (to the nearest 0.1 cm) were recorded for crabs: carapace width (CW); carapace length (CL); and body depth (BD).

KW tests were used to statistically compare hourly catch rates among trap types (0 = control; 1 = standard BRD; 2 = reduced height BRD; 3 = vertical BRD). Chi-square contingency tests were used to compare the proportion of males and females among trap types. Cumulative percentile distributions for crab size were computed independently for each trap type group and compared statistically using correlation analysis.

Objective 3: BRD design testing, Year 2
Size frequency distributions were computed for blue crabs captured by trap fishing in 2014 and for all diamondback terrapins captured across sampling gears in 2013 and 2014 to determine optimal BRD dimensions and geometric shape for excluding the maximum number of small terrapins while simultaneously retaining the maximum number of large blue crabs. A refined BRD design was then commercially produced by Voss Signs (Syracuse, New York) and consisted of 0.125-gauge plastic that was router-cut to specifications. Crab traps were fished in clusters of three in 2015, corresponding to a control trap plus a single BRD design fished in two traps, where the BRD was oriented horizontally in one trap but vertically in the remaining trap. All other trap fishing methods and statistical analysis in 2015 were consistent with 2014.

Objective 4: Visual observations
Crab trap funnel entry behavior for diamondback terrapins and blue crabs was evaluated by a graduate student (Janelle Johnson; College of Charleston) in a controlled environment at the South Carolina Aquarium and/or the College of Charleston's Grice Marine Biology Laboratory. Detailed methodology is provided in Johnson (In revision), but briefly, crabs and terrapins were housed separately in holding tanks and fasted prior to being transferred to an experimental tank where they were placed (one subject at a time) on one side of a trap panel outfitted with one of four funnel types (control vs. each of three BRDs) through which they were enticed to pass through to reach bait located on the other side (Figure 22). Statistical metrics included percent entry success by funnel type and, for crabs, entry time.
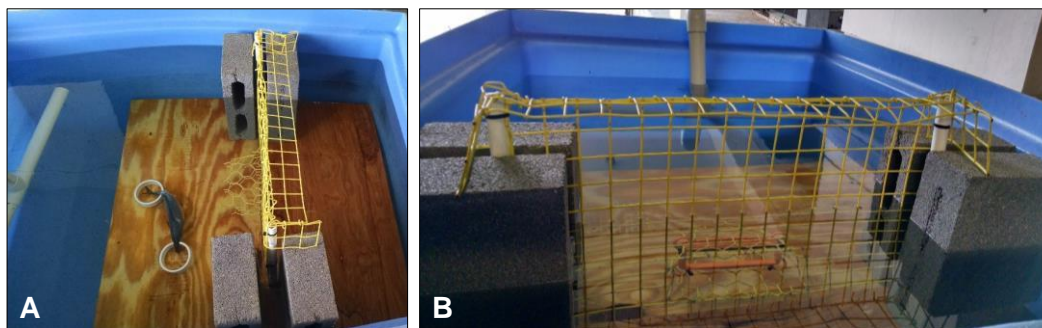
Figure 22. Direct observation of BRD efficacy was conducted in an experimental tank that encouraged terrapin investigation of a crab trap funnel in order to reach bait on the other side. Top-down/lateral (A) and longitudinal (B) views of this experimental tank are shown here.

Results

Objective 1

Eighteen studies conducted across all 12 states between Texas and New Jersey were identified during literature review, which produced 38 control-treatment trap comparisons (Table 3). Quantitative data on diamondback terrapin and blue crab catch were readily available for 29 of these comparisons, of which one involved re-orienting the funnel vertically (Belcher et al., 2008) and a second added plastic cable ties to the entry funnel to reduce the diagonal aperture to 9.3 cm (Hart and Crowder 2011). Among the remaining 27 comparisons, 74% ($n = 20$) involved wire BRDs with only seven comparisons between control traps and plastic BRDs (Table 3). Fifteen different wire BRD designs were evaluated with diagonal apertures ranging from 8.9 to 16.6 cm. Conversely, two plastic BRD designs (diagonal apertures of 12.9 and 16.0 cm) were evaluated.

A total of 1,666 diamondback terrapins were captured between control and treatment traps across 16 studies, with between zero and 584 (median = 45; IQR = 132) captured in each study, of which two to 215 (median = 55; IQR = 69) were captured in control traps. Relative to control traps, BRDs reduced terrapin catch by 12% to 100% (median = 95%; IQR = 14%). A significant difference was detected ($H_2 = 8.42$, P = 0.015) in the percent reduction of diamondback terrapin catch in BRD traps vs. control traps with respect to three control trap sample size groups, with greatest (median = 100%; IQR = 3%) reduction in catch where the number of diamondback terrapins captured in control traps ranged from two to nine but the least reduction in catch (median = 70%; IQR = 45%) when the number of diamondback terrapins captured in control traps ranged from 97 to 215. A non-significant correlation (r = -0.35, P = 0.057) was noted between diagonal aperture and the reduction (%) in diamondback terrapin catch in BRD traps.

A total of 75,416 blue crabs were captured between control and treatment traps across 14 studies, with between 307 and 35,807 (median = 2,489; IQR = 4,792) captured in each study, of which 45 to 10,873 (median = 630; IQR = 566) were captured in control traps. Relative to control traps, crab catch in BRD traps ranged from -60% to +53% (median = -10%; IQR = 27%). A significant difference was not detected ($H_2 = 0.26$, P = 0.877) in the percent reduction of blue crab catch in BRD traps vs. control traps with respect to three control trap crab sample size groups (45 to 168; 202 to 821; 938 to 10,873). A non-significant correlation (r = -0.21, P = 0.264) was noted between diagonal aperture and the change (%) in blue crab catch in BRD traps.

Objective 2

Thirty-one crab trap fishing days (and 362 trap sets) occurred between 06 May and 05 November. Between 06 May and 17 June, crab trap fishing efforts (54 sets, 15% of total) were primarily focused on the Ashley River and creeks in the vicinity of Duck Island. Crab trap fishing efforts further down river in Orange Grove and Old Towne Creeks were initiated on 30 May and continued extensively through 02 July (plus two additional days in September and October), during which time 116 sets (32% of total) were expended. More than half of all crab trap fishing effort (192 sets) occurred in the Stono River between 17 July and 05 November.

Table 3. Eighteen published and unpublished (*) BRD studies conducted in all 12 states between Texas and New Jersey were identified during literature review, representing 38 comparisons between control and treatment traps, of which data for 29 comparisons were analyzed.

Study	Where	BRD material	Comparisons	N terrapins	N legal blue crab	Terrapins per legal crab
Mazzarella 1994*	New Jersey	wire	1			
Wood 1997	New Jersey	wire	3	178	35,807	0.5
Cole and Helser 2001	Delaware	wire	4	584	4,532	12.9
Roosenburg and Greene 2000	Maryland	wire	3	201	4,767	4.2
Rook et al 2010	Virginia	plastic	1	48	348	13.8
Morris et al. 2011	Virginia	plastic	1	51	679	7.5
Upperman et al. 2014	Virginia	plastic	2	71	373	19.0
Grant 1997*	North Carolina	wire	2			
Hart and Crowder 2011	North Carolina	mixed	5	26	6,349	0.4
Chavez and Southwood-Williard 2013*	North Carolina	wire	2	19	1,860	1.0
Powers et al. 2009a*	South Carolina	plastic	1	30	11,472	0.3
Powers et al. 2009b*	South Carolina	plastic	2	76	2,225	3.4
Belcher et al. 2008*	Georgia	mixed	6	315	3,579	8.8
Butler and Heinrich 2007	Florida	wire	1	41	2,753	1.5
Coleman et al. 2011	Alabama	wire	1	24	365	6.6
Cuevas et al. 2000*	Mississippi	wire	1	0		
Guillory and Prejean 1998	Louisiana	wire	1	0		
Baxter 2014*	Texas	plastic	1	2	307	0.7

A total of 1,617 hours was recorded for five trap designs as follows: control traps (91 sets, 406.6 hours); standard BRD (90 sets, 404.1 hours); reduced height BRD (90 sets, 401 hours); vertical orientation BRD (86 sets, 399.3 hours); and "toothed" BRD (five sets, six hours). Given great similarity with the standard BRD, testing of the "toothed" design was quickly discontinued.

A total of 1,341 blue crabs (*Callinectes sapidus*) and four stone crabs (*Menippe mercenaria*) were captured in 290 of 357 non-"toothed" BRD trap sets, of which morphometric data for 1,329 blue crabs were complete. Two stone crabs (7.8, 9.7 cm CW) were captured in control traps, the third (8.4 cm CW) was captured in a standard BRD trap, and the fourth (5.1 cm CW) was captured in a reduced height BRD trap.

Blue crabs measured 6.5 to 18.7 cm CW, of which 37% (n = 494) were classified as legal-sized (≥12.7 cm). Legal-size crabs were not captured in 37% of control trap sets, a nearly identical frequency (36%) as the vertical BRD (Figure 23). Positive catch rates for legal-sized blue crabs ranged from 0.16 to 1.67 crabs per soak hour. A significant difference (H_3 = 10.74, p = 0.013) in legal-sized blue crab catch rates was detected among trap types, with the lowest catch rate (median = 0.00 legal crabs per soak hour) associated with the reduced height BRD and the highest catch rates (median = 0.23 legal crabs per soak hour) associated with control traps.

Captures of 0.16 to 1.00 legal-sized blue crabs per hour comprised 53% of control trap sets, identical to the standard BRD. Due to this catch rate range occurring more often in traps fished with the vertical BRD than among control traps, the vertical BRD was also associated with lower overall occurrence of catch rates >1.00 legal-sized blue crabs per hour (5% of all 357 trap sets) than control traps. Only three control trap sets with catch rates of 1.53 to 1.67 legal blue crabs per hour exceeded the maximum catch rate (1.43 legal blue crabs per hour) for the vertical BRD.
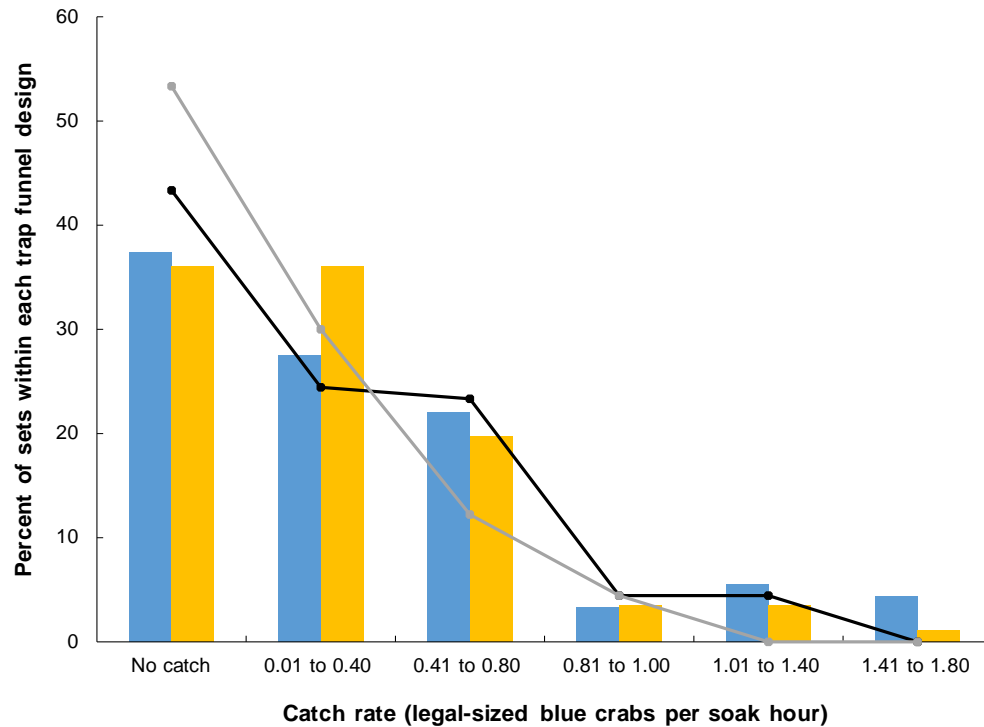
Figure 23. Legal-sized (≥5", 12.7 cm) blue crab catch rates for control traps (blue bar) were significantly different from traps outfitted with reduced height BRDs (gray line), but were not significantly different from standard BRD (black line) or vertical BRD (orange bar) traps.

No significant difference was detected ($H_3 = 4.57$, $P = 0.206$) in blue crab CW percentile distributions among the four trap types; however, legal-sized crabs were associated with the 60th to 63rd percentile for control, standard BRD, and vertical BRD traps but the 71st percentile for the reduced height BRD. Similarly, a slightly smaller (17.0 cm) maximum CW was associated with the reduced height BRD than control or either of the other two BRD traps (18.1 to 18.7 cm CW). No significant difference was detected ($X^2_3 = 3.821$, $P = 0.281$) in the ratio of female to male crabs across trap types, which was 46% female (range = 41% to 48% by trap type) overall.

Eleven male terrapins (10.9 to 13.1 cm) were also captured in crab traps, with all but one captured in the vicinity of Duck Island on 06 May and 20–21 May; the 11th terrapin, a 10.8 cm male, was captured in the Stono River on 30 July. Five of these terrapins were captured in control traps, three in standard BRD traps, and three in the discontinued "toothed" BRD traps. On 5 June, a 12th terrapin (an 11.6 cm male) was acquired from a crabber in Orange Grove Creek who indicated that other terrapins had also been captured in traps (and died) set there recently; this terrapin (MT0187) was subsequently recaptured (trammel net) on 17 April 2015 at AR30.

Two of three male terrapins captured in crab traps on 06 May after a one hour soak mysteriously died 24 and 48 hours later. Both terrapins were examined by Dr. Shane Boylan, DVM, at the South Carolina Aquarium; however, a definitive cause of death was not indicated. The third male terrapin captured in the same trap set received an acoustic tag (ID 214) and was detected 3,251 times between 08 May and 15 July, predominantly in the North Creek behind Duck Island (where it was captured) and at AR11 just outside of this creek before detections abruptly ceased.

Objective 3

Diagonal measurements based on body depth and width during entry through crab trap funnels were computed for 312 diamondback terrapins captured during 2013–2014 and 1,337 blue crabs captured in 2014. Only blue crabs ≥92nd percentile had diagonal body measurements larger than the smallest diamondback terrapin (8.2 cm diagonal); however, twice as many diamondback terrapins (i.e., ≤16th percentile) were smaller than the diagonal measurement (9.4 cm) for the largest blue crab (Figure 24). Presuming that these percentile distributions are representative of size distributions for both species, approximately two small terrapins would be conserved for every large crab lost as optimal diagonal aperture is decreased from 9.4 to 8.2 cm. However, because of the cross-sectional geometry of diamondback terrapins (Figure 25a) compared to blue crabs (Figure 25b), it is nearly impossible to exclude small diamondback terrapins without also excluding large blue crabs (which are most economically valuable) from crab traps. Maximum (but independently measured) blue crab body depth and carapace lengths were 5.0 and 8.3 cm, respectively (9.7 cm diagonal); however, this diagonal BRD aperture size would have allowed 29% of diamondback terrapins to enter traps. As such, the BRD design tested in 2015 measured 5.1 x 7.3 cm (8.9 cm diagonal); based on percentile distributions computed through 2014, this diagonal aperture should have only allowed the smallest 5% of diamondback terrapins to enter traps while simultaneously excluding only the largest 1% of blue crabs from the same traps.
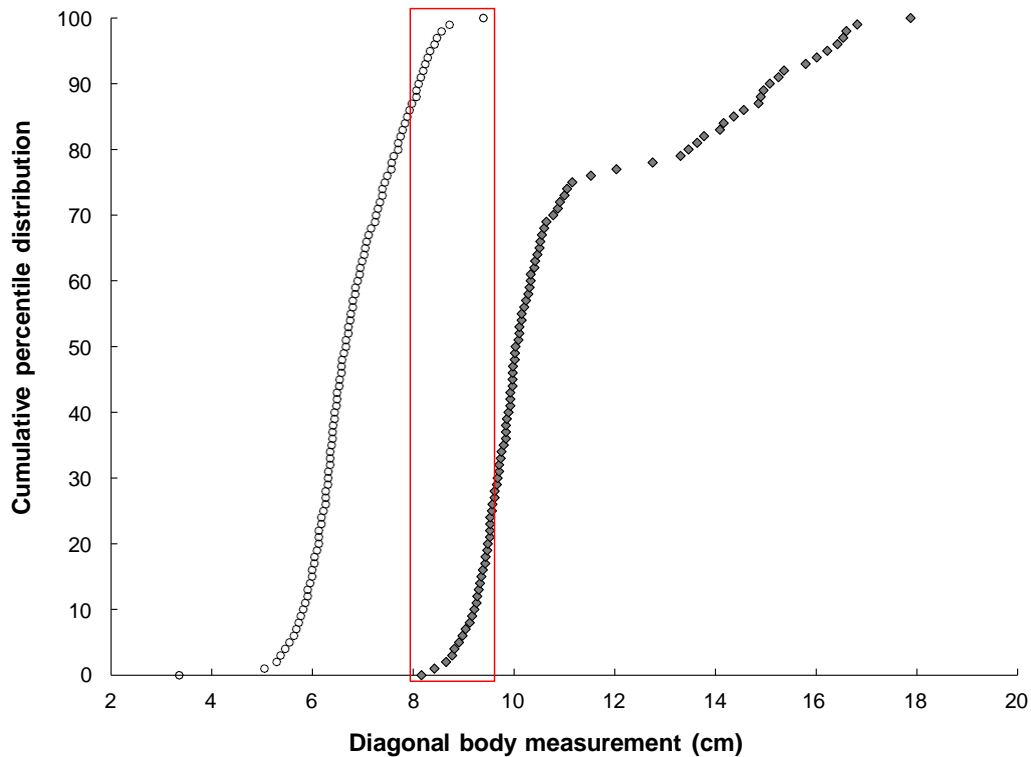
Figure 24. Percentile distribution (y-axis) relative to diagonal body size (x-axis) for 1,337 blue crabs (open circles) and 312 diamondback terrapins (gray diamonds) measured through 2014. Red box denotes optimal diagonal BRD aperture between 8.2 cm (the smallest terrapin diagonal) and 9.4 cm (the largest blue crab diagonal).
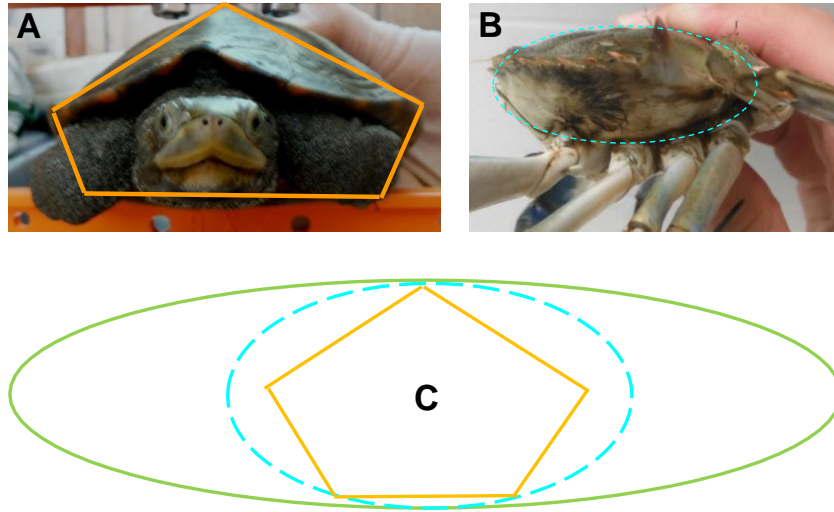
Figure 25. Diamondback terrapins (A) were associated with pentagonal cross-sectional body areas during trap funnel entry as opposed to oval cross-sectional body areas for blue crabs (B); thus, there does not appear to be a geometric shape solution (C) to exclude the smallest terrapins while simultaneously retaining the largest blue crabs in crab trap funnels.

Twenty-three crab trap fishing days (327 trap sets) to test BRD design occurred between 6 April and 6 November 2015; three additional days (18 trap sets, 56.2 soak hours) of crab trap fishing to intentionally (but unsuccessfully) capture diamondback terrapins in Orange Grove Creek in August and September were excluded. Five fishing days (81 trap sets) were conducted near Duck Island between 06 April and 11 May. Three fishing days (39 trap sets) were conducted in Orange Grove Creek and/or in Old Towne Creek between 13 April and 07 July. The remaining 201 trap sets (16 days) occurred in the Stono River. Across fishing areas, a total of 1,445 hours was recorded for three trap designs: control traps (109 sets, 482 hours); horizontal BRD traps (109 sets, 477 hours); and vertical BRD traps (109 sets, 486 hours).

A total of 1,843 blue crabs and three stone crabs were captured in 267 of 327 crab trap sets, of which carapace width and sex were recorded for 1,838 and 1,842 blue crabs, respectively. Blue crabs measured 6.5 to 18.1 cm CW, with 27% ($n = 503$) classified as legal-sized ($\geq$12.7 cm). Legal-size crabs were not captured in 31% of control trap sets and 39–43% of BRD trap sets (Figure 26). Positive catch rates for legal-sized blue crabs ranged from 0.15 to 2.80 crabs per soak hour. Despite BRD traps not capturing legal-sized crabs 8–12% more often than controls as well as a 3% reduction in the frequency of capturing >1.50 legal blue crabs per soak hour, a significant difference was not detected ($H_2 = 4.00$, $P = 0.136$) in legal-sized blue crab catch rates between control and BRD traps.

Blue crabs captured in 2015 ranged in size from 6.5 to 18.1 cm CW, but a significant difference was not detected ($H_2 = 0.72$, $P = 0.698$) in CW percentile distributions among three trap types. Similarly, no significant difference was detected ($X^2_2 = 5.429$, $P = 0.066$) in the ratio of female to male crabs across trap types, which ranged from 32% female for vertical BRD traps ($n = 572$) to 37% for both control ($n = 683$) and horizontal ($n = 587$) BRD traps.
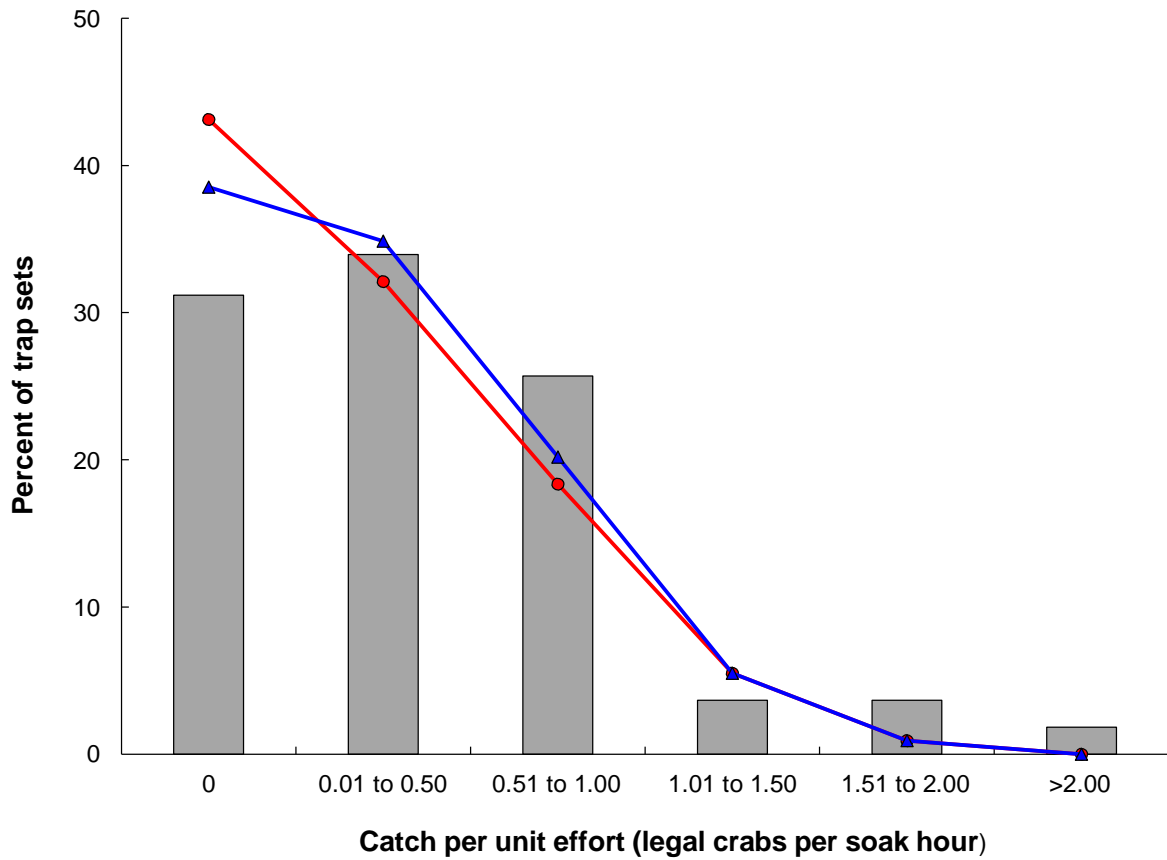
Figure 26. A significant difference in the distribution of catch rates for legal-sized crabs (x-axis) was not detected between control traps (gray bars), horizontal BRD traps (red line), and vertical BRD traps (blue line); however, non-capture events were 6–12% more common in BRD traps.

Fifty-seven diamondback terrapins were captured in crab traps (1 to 11 per trap) set between 06 April and 11 May, 18 (32%) of which were captured in eight of 15 (53%) trap sets on 06 April, prior to reducing BRD diagonal opening from 9.7 cm (8.3 cm width) to 8.9 cm (7.3 cm width). No significant difference was detected ($H_2 = 3.32$, $P = 0.190$) in diamondback terrapin catch rates among crab trap designs (11 control, 4 horizontal, 3 vertical) on 06 April, likely due to the large IQR (0.594 terrapins per soak hour) associated with control traps (Figure 27).

Thirty-six diamondback terrapins were captured in 18 of 66 (27%) trap sets at Duck Island on four fishing days between 20 April and 11 May, and three diamondback terrapins were captured in three of 15 (20%) trap sets in Orange Grove Creek on 13 April. Across these 81 trap sets, a significant difference was detected ($H_2 = 19.43$, $P<0.001$) among trap types, with 82% (32) of diamondback terrapins captured in control traps but only 8–10% (3–4) captured in BRD traps (Figure 27). Diagonal body size of diamondback terrapins captured in BRD traps (8.0 to 9.2; median = 9.0; IQR = 0.5) was significantly less ($H_1 = 10.73$, $P = 0.001$) than diagonal body size for diamondback terrapins captured in control traps (8.5 to 12.8; median = 9.8; IQR = 1.5).
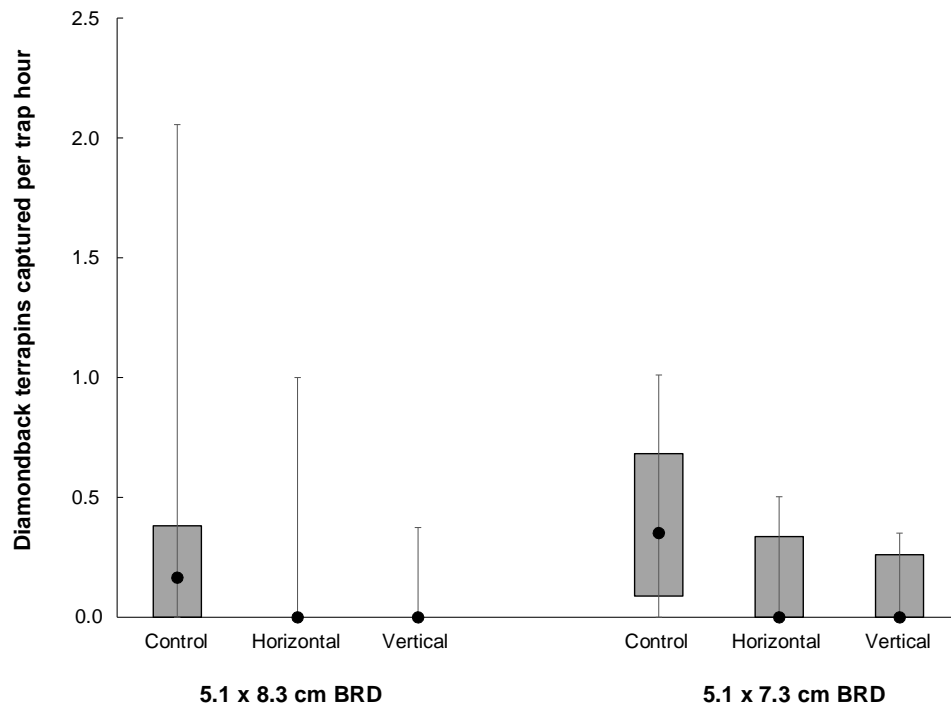
Figure 27. Diamondback terrapin captures per trap soak hour were not significantly different among trap types with the 5.1 x 8.3 cm BRD (06 April, 5 replicates per design), but a significant difference was detected among trap types with the 5.1 x 7.3 cm BRD during 27 replicate sets per design between 13 April and 11 May 2015. Black circles denote median catch rate; gray boxes and error bars denote inter-quartile range and minimum and maximum catch rates, respectively.

Objective 4

Detailed results and Discussion for this objective are provided in Johnson (2016) which should be available by May 2016; however, key findings related to BRD efficacy are highlighted here.

Forty-eight diamondback terrapins were tested in 2014 (23 male, 25 female; CL = 11 to 22 cm) and an additional 40 males were tested in 2015 (9.7 to 12.8 cm CL). In 2014, between seven and 29 of the 48 terrapins attempted entry through the control and three BRD funnels. Terrapin exclusion success was distributed (from best to worst) as follows: 100% (reduced height BRD); 72% (vertical BRD); 35% (standard BRD); 19% (control). In 2015, between 25 and 29 male terrapins attempted to enter BRD-equipped funnels, of which 84% were excluded by the vertically-oriented BRD and 76% were excluded by the horizontally-oriented BRD.

Thirty-six legal-sized blue crabs were tested in 2014 (22 male, 14 female; CW = 13.2 to 18.7 cm) and 46 additional legal-sized (12.7 to 19.3 cm CW) male blue crabs were tested in 2015. In 2014, between eight and 12 blue crabs attempted to enter funnels, of which 95% passed through control funnels, 70–75% passed through standard and vertical BRD funnels, but only 17% passed through funnels with the reduced height BRD. In 2015, 32 blue crabs attempted entry through BRD-equipped funnels, of which 88% (vertical) to 97% (horizontal) succeeded, with 59–88% succeeding on the first pass attempt. Time required for successful crab entry ranged from 1.5 to 26.5 seconds (mean = 8.6 seconds).

General Discussion

The use of automated acoustic monitoring resulted in a large (26 female, 18 male) and potentially unprecedented number of total detections (>146k) for this species from which temporal and spatial distribution patterns were evaluated. Prior to the present study, we are aware of only five other studies that have used acoustic telemetry to collect similar data for diamondback terrapins elsewhere in their range. Estep (2005) deployed 13 acoustic transmitters on adult female diamondback terrapins and remotely collected 21,848 detections over a 16-month period using four acoustic receivers that provided episodic bouts of monitoring at 10 locations in a cove and creek system located near the mouth of Charleston Harbor, SC. Clarkson (2012) deployed acoustic transmitters on five male and three female diamondback terrapins and tracked their movements around south Deer Island in Galveston Bay, TX using four receivers. In Virginia, Tulipani (2013) affixed acoustic transmitters to 10 male and six female (including two presumably immature individuals) diamondback terrapins in 2011 and 2012 and recorded ~26,000 detections using five acoustic receivers for the purpose of developing a model for predicting residence vs. emigration. In New Jersey, nine acoustic receivers deployed at selected creek mouths in the salt marshes of the Cape May, NJ peninsula during 2005–2009 generated an undisclosed amount of acoustic telemetry data for 65 diamondback terrapins.[3] This technique has also been used to investigate terrestrial emergence behavior for 78 adult female diamondback terrapins in Barnegat Bay, New Jersey (Winters et al. 2015).

Prior studies, which have utilized manual radio telemetry, produced vastly smaller data sets on order with acoustic detection data generated during mobile searches employed in the present study, reinforcing the need for automated data collection for this cryptic species. Estep (2005) reported 348 detections across 10 adult female diamondback terrapins between May and December 2001, with mean detection for individual terrapins spanning 10.1 days that occurred 78.9 days apart. Perhaps related to nesting activity, radio transmitter detections were greatest during June and August (Estep 2005). Harden and Southwood Williard (2012) reported 362 detections for 24 female and five male diamondback terrapins in southeastern North Carolina based on one to three search efforts per week between June 2008 and May 2009. Lowest radio detection frequency was reported when tagged terrapins were dormant and buried in mud (Harden and Southwood Williard 2012), a behavior that is also reported to occur in summer (Spivey 1998; Tucker et al. 1995; Butler 2002).

Automated data collection also revealed greater localized movement patterns for this species than is reported in the published literature. These findings were consistent with a high degree of individual variability in residence and localized movement patterns reported for this species in previous telemetry studies (Estep, 2005; Tulipani 2013), and suggest the need to reevaluate the mantra of strictly localized populations (Tucker et al. 1995). Although documented movement out of the Ashley River system was rare and genetic diversity and divergence for this species is "exceptionally low" (Lamb and Avise 1992), we hypothesize that movement within and among waterways may have contributed to the inability of Hauswaldt and Glenn (2005) to differentiate

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[3] Owens, J.R. and R.C. Wood. 2007 "Tracking northern diamondback terrapins (*Malaclemys terrapin terrapin*) in a southern New Jersey (USA) salt marsh using sonic telemetry." 4th Symposium on the Ecology, Status and Conservation of Diamondback Terrapins, 10–12 August, Millersville, Maryland. http://wetlandsinstitute.org/research/ccrp/presentations-publications/ (Accessed 10 March 2016)

population structure within the Charleston Harbor estuary. Individual variability in distribution patterns contributes to inter-annual variability in detection patterns (Tulipani 2013), and has considerable ramifications for analyzing catch rate data for this species. For instance, Tulipani (2013) reported tidally-mediated movement far from shore; however, trammel net surveys are designed to fish very close to shore in order to capture animals after they are 'spooked out' of the marsh (Arnott et al. 2013). Accordingly, we recommend future telemetry studies in other areas where the trammel net survey operates, particularly given anomalously high catch rates in the Ashley River relative to other estuarine survey areas throughout the South Carolina coast.

Movement throughout the study area was facilitated using both creek and marsh corridors as well as along-shore movement in the river. Although Estep (2005) reported routine tidally-mediated movements between creek and cove habitats near the entrance to Charleston Harbor, when data were standardized to a frequency of hourly bins (vs. more individualized hourly detections), time of day exerted the greatest influence on the probability of detecting diamondback terrapins in creek vs. river habitats. Clarkson (2012) also reported significant variation in diel activity, with the greatest probability of nocturnal activity during the mating season. Given seasonal detection patterns in the present study, it is possible that some portion of diel difference in habitat use in the present study was related to an underlying seasonal influence; thus, before publication, the data should be re-examined to test for this potential interaction.

Although the distribution of hourly detection bins across water levels closely mirrored the distribution of available water levels, tidally-mediated influences on a range of behaviors have been reported across numerous studies, and may help to explain the whereabouts of acoustically-tagged terrapins in the present study during periods of absence from detection by receivers. Estep (2005) also reported that a greater proportion of radio detections occurred on ebb tide stages, presumably due to improved signal reception as tagged terrapins moved away from marsh grass and into open water systems. Tucker et al. (1995) also reported directional movement away from flooded marsh grass habitats as waters receded, with less predictability associated with habitat use during low water. Conversely, terrapins have also been observed walking in grass habitats (Butler 2002; Clarkson 2012) as well as burrowing in mud (Clarkson 2012; Akins et al. 2014); thus, habitat use likely reflects time of year, sex, and reproductive condition. Tulipani (2013) also reported tidally-mediated movement, especially by large females, between shallow near-shore waters at high water levels and egress to deeper waters further from shore at low water levels. Tulipani (2013) also suggested that these localized movements serve as vectors for eel grass (*Zostera marina*) seed dispersal, an ecological role akin to serving as vector for Digenean trematode life cycles (Byers et al. 2013).

A high degree of similarity between seasonal detection patterns and seasonal capture rates in the trammel net survey bode well for use of this data set for monitoring population trends for this species. Within-season spatial distribution patterns largely reflected variability among individual diamondback terrapins; however, greatest detection frequency during April and May suggests aggregation behavior and therefore potentially more consistent behavior among individuals. Accordingly, it may be most appropriate to restrict temporal analyses to these months, with the caveat that temporal trends reflect variation in the size of the annual spring aggregation. Given the importance of time of day, future analysis of historical trammel net catch rates should also include this parameter in the analytical model. Use of the trammel net to monitor temporal

trends for diamondback terrapins in river habitats is also supported by the greatest catch rates among techniques employed in the present study. Seining and trammel netting (when set perpendicular to water mass movement) are effective for sampling tidal creeks larger than those found in our study area, when sufficient water levels for concentrating diamondback terrapins were present at current speeds that were conducive to sampling (Gibbons et al. 2001), but did not prove to be lucrative sampling techniques in the present study. Similarly, incidental capture of diamondback terrapins in crab traps only accounted for nine percent of all diamondback terrapin captures, 99% of which were only captured in April or May. Granted, one in four (2014) and one in three (2015) crab traps used for fishery-independent testing were devoid of BRDs designed to minimize diamondback terrapin capture; however, even deliberate attempts to capture diamondback terrapins using crab traps with enlarged funnel openings in tidal creek habitats in August and September 2015 were completely unsuccessful, in stark contrast to the ability of trammel nets to capture diamondback terrapins during the same months.

Despite low overall capture rates of diamondback terrapins in crab traps in the present study, the documented capability of crab traps, particularly when left unattended, to capture a large number of diamondback terrapins at certain times of the year (Grosse et al. 2009) remains alarming. During the present study, a maximum catch rate of 2.1 diamondback terrapins per trap soak hour was observed, but ≤3 diamondback terrapins were captured during each one hour soak period. However, extrapolation of these fishery-independent catch rates for upwards to five non-BRD crab traps per fishing day to the >3 million trap days estimated to be fished in South Carolina annually across 300 licensed commercial crabbers[4] reveals precisely how potentially devastating this fishery could be to diamondback terrapin populations. As such, considerable emphasis in the final two years of the present study was placed on designing a more appropriately-sized BRD based on the morphometric dimensions of blue crabs and diamondback terrapins, as well as behaviors that influence their probability of capture and/or escapement.

Considerable progress was made towards developing a superior BRD design for use by South Carolina crabbers than the designs tested by Powers et al. (2009a,b). A crucial finding in 2014 was the significant reduction in catch rates for and size of legal-sized crabs when a maximum opening size of 4.1 cm (1.625 in.) was used to accommodate the body depth of blue crabs, which was supported by behavioral data demonstrating increased difficulty of large crabs to negotiate that opening size. Another important discovery in 2014 was similarity in legal-sized crab catch and size distribution between the standard BRD and a narrower modification that was oriented vertically. Because diamondback terrapins readily rotate their bodies as needed to enter crab trap funnels, vertical rotation did not enable segregation of species based on size selectivity; however, similarity in catch between both BRD designs capable of accommodating crabs with a 5.1 cm body depth did suggest that this body dimension was most crucial for improving crab entry. Validation of the importance of body depth as the most limiting dimension was affirmed in 2015, when horizontal BRD width was reduced even further (to 8.3 cm), which not only resulted in superior diamondback terrapin exclusion, but also produced catch rates for legal-sized blue crabs that were not significantly different from non-BRD traps. Although work remains to reduce the rate of non-capture of legal-sized blue crabs in BRD traps, results to date are quite encouraging.

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[4]Arendt, M.D., E. Waldrop, J. Heinsohn, J. Schwenter, and P. Kingsley-Smith. 2015. Evaluation of Techniques to Reduce Crab Trap Float Loss in South Carolina. Final Programmatic Report to the National Fish and Wildlife Foundation, Project ID 0304.13.041037, 30 p.

Presentations and Outreach

Professional Meetings and Workshops

- September 2013. 6th Symposium on the Status, Ecology & Conservation of Diamondback Terrapins, Seabrook Island, SC. Poster. Arendt, M.D., J.A. Schwenter, E. Levesque, A.M. Grosse, J.D. Whitaker, D.W. Owens, J. Zalabak, C. Hughes, K. Thorvalson, and S. Boylan. "Validation of trammel netting as a means for monitoring population trends for diamondback terrapins (*Malaclemys terrapin*) in the Charleston harbor estuary."
- March 2014. SCDNR Marine Resources Division Annual Conference, Charleston, SC. Presentation. Schwenter, J., M. Arendt, E. Levesque, A. Grosse, and D. Whitaker. "Mysterious turtles of the salt marsh: Addressing research and management needs to protect the diamondback terrapin (Malaclemys terrapin) in South Carolina."
- March 2015. Southeast Regional Meeting of the Diamondback Terrapin Working Group, Brunswick, GA. Presentation. Arendt, M., J. Schwenter, E. Waldrop, E. Levesque, A. Grosse, and J.D. Whitaker. "Validation of trammel netting as a means for monitoring population trends for diamondback terrapins (*Malaclemys terrapin*) in the Charleston harbor complex."
- March 2015. Southeast Regional Meeting of the Diamondback Terrapin Working Group, Brunswick, GA. Presentation. Arendt, M., J. Schwenter, E. Waldrop, E. Levesque, A. Grosse, and J.D. Whitaker. "BBRD: Better by-catch reduction devices for diamondback terrapins in crab traps."
- March 2015. SCDNR Marine Resources Division Annual Conference, Charleston, SC. Presentation. Waldrop, E., Arendt, M., J. Schwenter, E. Levesque, A. Grosse, and J.D. Whitaker. "BBRD: Better by-catch reduction devices for diamondback terrapins in crab traps."
- November 2015. ASMFC–SEAMAP Crustacean Workgroup Meeting, St. Augustine, FL. Presentation. Fowler, A., Arendt, M., J. Schwenter, E. Waldrop, E. Levesque, A. Grosse, and J.D. Whitaker. "BBRD: Better by-catch reduction devices for diamondback terrapins in crab traps."

General

- January 2013, 2014, and 2015: Scientist Day, James Island Middle School; the main focal points of this grant were introduced to approximately 100 sixth grade students annually.
- October 2014: Discussed preliminary BRD research with a diverse stakeholder group including biologists from the Town of Kiawah and the Kiawah Development Foundation.
- October 2015: SCDNR Marine Resources Division Open House, Charleston, SC. Conveyed terrapin research findings and data needs with approximately 100 citizens.
- Received 40 citizen-scientist reports regarding diamondback terrapins including terrestrial observations of adult females (crawling, nesting, or killed on roadways); hatchlings in swimming pools; fisheries interactions (cast net, crab trap, hook-and-line); and re-sightings of tagged terrapins (alive as well as dead).
- During crab trap fishing and other terrapin field days (especially at Northbridge Park), we discussed this research with approximately a dozen commercial crabbers and roughly 50 members of the public; these interactions led to several follow-up terrapin sightings.

Collaboration

The most prominent collaboration during 2013–2015 involved multiple staff from the SCA Sea Turtle Hospital (STH), a collaboration that actually began in October 2012 when a transmitter-retention study was initiated that continued through winter 2014. Between spring and fall 2014 and 2015, the SCA STH provided tank space to house diamondback terrapins and blue crabs for efforts to visually observe crab trap funnel entry (and potential egress) behavior. In late summer 2015, the scope of the partnership was extended to include PIT-tagging of five and acoustic tagging of two diamondback terrapins reared at the SCA STH from hatchling stage.

The partnership at the SCA STH to evaluate interactions of diamondback terrapins with crab traps also led to opportunistic data collection to assist a second diamondback terrapin study funded by the US Fish and Wildlife Service (State Wildlife Grant). Specifically, terrapin mating in the SCA STH holding tank occurred shortly after transport of 10 male and 10 female terrapins collected from the Ashley River in April 2014, of which 9 of 10 female terrapins ultimately laid 81 eggs that were incorporated into a companion study to evaluate growth rates of young of the year hatchlings reared under various diets (Levesque and Grosse – PIs).

In addition to acoustically-tagged diamondback terrapins, receivers deployed in the Ashley River also recorded 35 detections for an Atlantic sturgeon tagged in a S. Carolina study managed by B. Post (SCDNR); 89 detections for a southern flounder tagged in a N. Carolina study managed by F. Scharf (UNC Wilmington); and 28,561 detections for 34 southern flounder tagged in the Ashley River by M. Hart (CofC) in fall 2015. Detection data for all transmitters tagged by other researchers were promptly provided to these researchers after data were uploaded.

Barnacles were infrequently (88 of 745 collections, 12%) observed attached to diamondback terrapins, but barnacles were collected from ~15 diamondback terrapins and saved for a Ph.D. student (C. Ewers-Saucedo) at the University of Georgia in 2013 and 2014.

Following discussion with Dr. Kristen Hart of the US Geological Survey (USGS; Davie, FL) at the 6th Symposium on the Status, Ecology & Conservation of Diamondback Terrapins on Seabrook Island in September 2013, we unsuccessfully attempted to collect blood samples and keratin scrapings from ~10 diamondback terrapins for USGS evaluation of stable isotope signals. However, on 21 March 2016, Mr. Mat Denton of the USGS will attempt to collect these samples using the same techniques that have worked with diamondback terrapins captured in south Florida; thus, we are hopeful that this collaboration will finally have an opportunity to develop.

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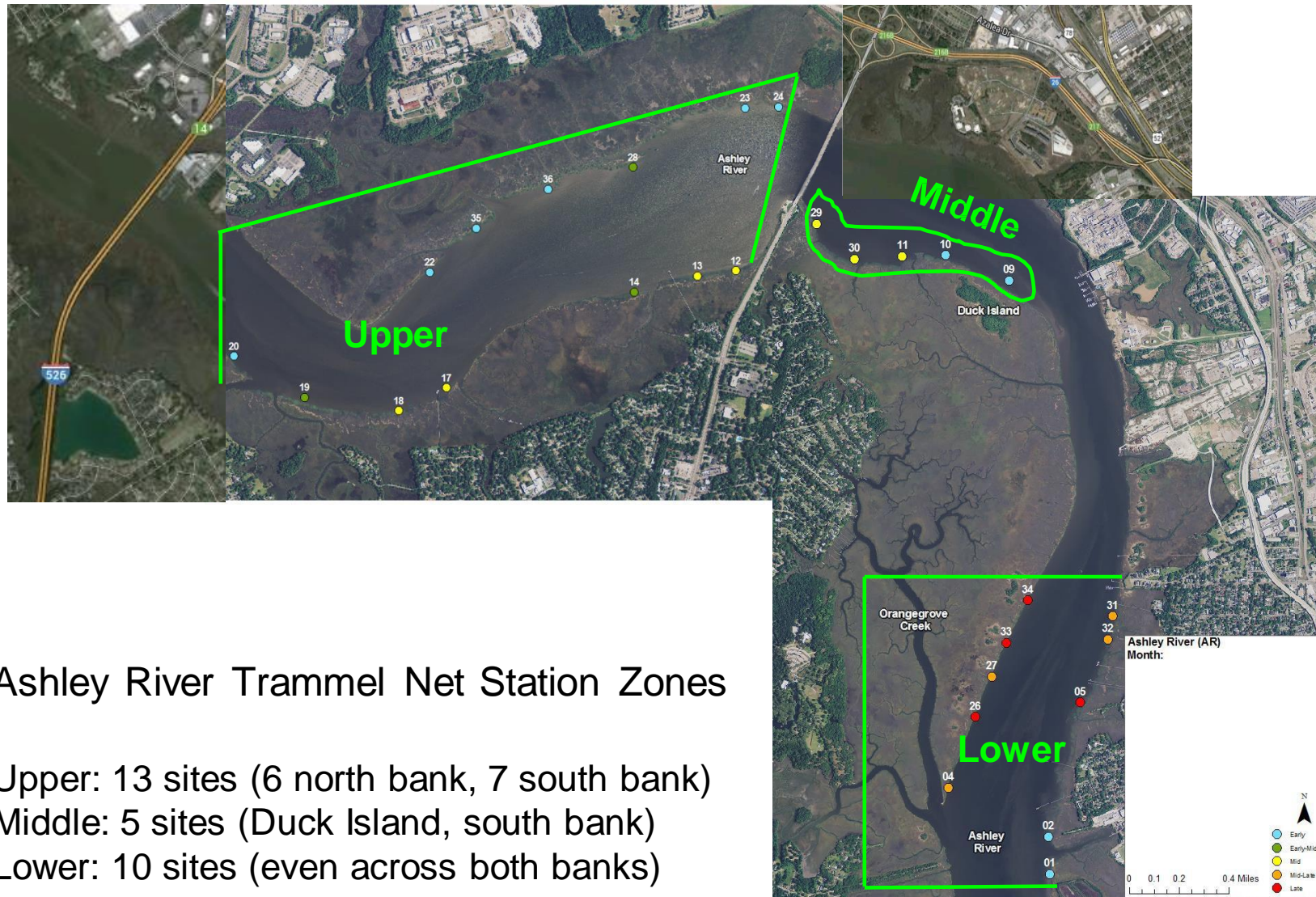
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Appendix 1. Ashley River Trammel Net Station Zones (courtesy of A. Grosse).

Ashley River Trammel Net Station Zones

Upper: 13 sites (6 north bank, 7 south bank)
Middle: 5 sites (Duck Island, south bank)
Lower: 10 sites (even across both banks)

Appendix 2. Variability in detection of acoustically-tagged diamondback terrapins was not explained (57–60% similarity) by sex, spatial capture zone (0 = lower Ashley; 1 = creeks; 2 = Duck Island; 3 = above Rt. 7 Bridge), capture month, or capture year; however, 98–100% similarity in detection trends for terrapins with <6k total detections (hits) warranted further trends analysis. Data are sorted by terrapin sex and increasing number of detection days.

Transmitter	Sex	Capture	Zone	Data days	Daily hits, min	Daily hits, max	Daily hits, mean	Daily hits, SD	Total hits
A69-1601-33657	F	01-Jun-15	0	8	4	53	19	19	153
A69-1303-25234	F	29-Apr-13	1	12	12	100	41	28	494
A69-1303-216	F	11-Jul-14	3	16	1	24	7	7	110
A69-1303-220	F	11-Jul-14	2	16	1	37	9	10	149
A69-1303-228	F	01-Jun-15	2	24	1	77	17	20	409
A69-1303-25239	F	01-May-14	2	31	1	162	43	53	1329
A69-1303-219	F	11-Jul-14	3	39	1	92	15	18	599
A69-1303-223	F	17-Apr-15	2	44	1	51	14	12	619
A69-1303-217	F	11-Jul-14	2	47	1	133	27	32	1266
A69-1303-226	F	27-Apr-15	2	57	1	199	36	41	2068
A69-1303-25222	F	18-Apr-13	2	76	1	164	20	26	1556
A69-1303-25233	F	29-Apr-13	1	76	1	151	25	28	1919
A69-1303-25229	F	18-Apr-13	1	81	1	173	31	42	2492
A69-1601-33658	F	08-Sep-15	0	85	1	207	48	46	4104
A69-1303-25232	F	18-Apr-13	1	89	1	168	42	42	3714
A69-1303-215	F	11-Jun-14	3	98	1	44	10	9	1020
A69-1303-25221	F	18-Apr-13	2	117	1	137	15	18	1700
A69-1303-25231	F	09-May-13	1	119	1	218	24	36	2910
A69-1303-25218	F	09-Apr-13	2	123	1	108	18	21	2228
A69-1303-25217	F	09-Apr-13	2	132	1	113	21	22	2808
A69-1303-25220	F	18-Apr-13	2	138	1	143	25	30	3468
A69-1303-224	F	17-Apr-15	2	141	1	313	179	87	25275
A69-1303-229	F	01-Jun-15	2	178	1	100	18	17	3208
A69-1303-25219	F	09-Apr-13	2	196	1	111	27	25	5275
A69-1303-222	F	11-Jul-14	2	224	1	109	10	13	2167
A69-1303-25230	F	18-Apr-13	1	232	1	217	16	27	3693
A69-1303-225	M	20-Apr-15	2	5	14	136	63	50	313
A69-1303-218	M	11-Jul-14	3	6	1	31	13	11	80
A69-1303-25240	M	01-May-14	2	6	1	35	8	13	49
A69-1601-33655	M	18-May-15	0	17	1	30	10	9	176
A69-1303-25236	M	09-May-13	1	42	1	264	78	66	3262
A69-1303-25235	M	18-Apr-13	1	59	1	93	21	23	1263
A69-1601-33656	M	18-May-15	0	66	1	60	14	17	938
A69-1303-214	M	06-May-14	2	69	6	117	47	25	3251
A69-1303-25227	M	09-Jul-13	2	81	1	104	32	24	2558
A69-1303-25228	M	09-Jul-13	2	108	1	125	52	33	5629
A69-1303-25238	M	14-Apr-14	2	134	1	130	15	25	1949
A69-1303-25223	M	18-Apr-13	2	137	1	152	36	31	4981
A69-1303-25237	M	17-Sep-13	1	155	1	143	21	21	3276
A69-1303-25226	M	21-May-13	2	169	1	111	21	23	3623
A69-1303-227	M	18-May-15	2	195	1	124	23	25	4541
A69-1303-25224	M	18-Apr-13	2	218	1	115	16	20	3459
A69-1303-25225	M	21-May-13	2	232	1	148	19	19	4471
A69-1303-221	M	11-Jul-14	3	347	1	298	81	76	28230