

Using Long-term Mark-recapture Data to Quantitatively Assess Conservation Strategies  
for the Diamondback Terrapin, *Malaclemys terrapin*

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This thesis titled  
Using Long-term Mark-recapture Data to Quantitatively Assess Conservation Strategies  
for the Diamondback Terrapin, *Malaclemys terrapin*

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## ABSTRACT

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Director of Thesis: Willem M. Roosenburg

Successful conservation strategies require thorough understanding of the life history traits and dynamics of the population of interest. Head-starting, captive breeding, and other methods of increasing recruitment and juvenile survivorship are frequently implemented in management plans; however, there is a large body of evidence to suggest that these strategies may not be as effective for long-lived “bet hedging” species. One such species, the diamondback terrapin, experiences a variety of anthropogenic threats, particularly bycatch in Atlantic Blue Crab fisheries, that effect survival at post-hatchling life stages. Despite these potential threats, a diamondback terrapin population at Poplar Island, MD has been undergoing a head-start program since 2005. We used matrix population models parameterized from mark-recapture data to demonstrate that terrapin population growth rates are most sensitive to changes in sub-adult life stages, and that head-starting has little effect on growth rate. We found that post-release survivorship of head-started and natural emergence hatchlings do not significantly differ and find head-starting to be a half-way technology for terrapin conservation. We demonstrated the BRD to be an appropriate technology and stress that proactive management with regards to BRDs and crab pots are the best solution for terrapin conservation by maintaining higher sub-adult and adult survival.

## DEDICATION

*To my family, by blood or by bond: for enduring me and all my faults over the years. I  
owe you all more than I will ever be able to repay.*

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## CHAPTER 1: POPULATION DYNAMICS AND EVALUTION OF CONSERVATION STRATEGIES

### Abstract

We conducted mark-recapture and nest surveys of diamondback terrapins at Poplar Island, MD from 2005-2017. We estimated apparent survival probabilities for wild female diamondback terrapins, as well as post-release apparent survival of head-started and natural emergence (NE) terrapins that were marked at birth. Apparent survival was estimated to be  $0.64 \pm 0.02$  (estimate  $\pm$  SE) for sub-adults (3-7 years of age) and  $0.80 \pm 0.01$  for adults (aged 8+). Juvenile survival (ages 1 and 2) was numerically approximated to be 0.55. Differences in survival between natural emergence and head-started hatchlings were subtle, but survival of NE hatchlings was slightly higher than that of head-starts. We constructed a matrix population model by numerically approximating juvenile survival, based on realized growth rate from abundance estimates. We then perturbed vital rates within the model and observed the impact on growth rate. Sub-adult survival had the largest impact on population growth, followed by adult survival, juvenile survival, and then fertility. We also modeled various levels of head-starting and saw little effect on growth rate compared to survival perturbations. We recommend emphasizing conservation strategies that mitigate mortality sources in later life stages, particularly sub-adult, and caution against the use of head-starting as a means to bolster declining terrapin populations.

## Introduction

Successful conservation requires a large amount of data, and effective understanding of the life history traits of the species one is trying to conserve. Resources are almost always finite; thus, decisions must be made about how to spend those resources to have the greatest impact on the species. Understanding the life histories of species can inform conservationists on where in the life cycle of a species their resources and efforts can have the most benefit. For species of great conservation need, demographic data can provide valuable insight into their life histories, and thereby provide insight into where conservation efforts should be focused. Turtles represent one such group of great conservation need with 48-54% of all turtle species considered threatened (Rhodin et al. 2011). Most turtles can be described as a “bet hedging” species, meaning they typically have low recruitment, delayed sexual maturity, and high mortality in early stages of life, which is compensated for by relatively high survival in later life stages, and iteroparity over long lifespans (Stearns 1976; Heppel 1998).

Because of their characteristically low recruitment, head starting has often been proposed, or performed, for turtle populations. Head-starting is a process by which newly hatched or born individuals are bred in captivity until they have achieved an age/body size where their probability of survival is supposedly higher than that of smaller individuals, at which point they are released (Heppel et al. 1996). While increasing recruitment and supplementing population size should theoretically improve population health, the efficacy of such methods is a function of the quality of the habitat in which these individuals are being released (Frazer 1992). This has led to the criticism of head-

starting as a “half-way” technology i.e. one that treats only symptoms of population decline rather than causes, especially for long-lived organisms such as turtles (Frazer 1992).

It is difficult to evaluate head-start programs due to a poorly defined metric of success (Burke 2015). Some define successful head-starting when head-started individuals are more likely to become part of the breeding population than those who were not head-started, or if head-started individuals are able to reach sexual maturity sooner (Pritchard 1981). Others have suggested that head-started individuals need not become reproductive for the program to be considered successful *per se*, but examine programs on more short-term metrics, such as head-started individuals must grow and survive in the wild post-release at or above the rate at which non-head-started individuals do (Eckert et al. 1994). Head-start programs also provide outreach and public appeal which can increase public activity and interest in conservation at other life stages that may benefit the population in some way, even if the head-starting itself has little demographic benefit (Burke 2015). Outreach benefits, while certainly present, are difficult to quantify; and thus, serve as a complicated metric to evaluate program success.

Head-starting is commonly implemented in conservation, especially in turtles, because they are particularly vulnerable in early life stages, and because eggs and hatchlings are particularly easy to rear in captivity (Burke 2015). Additionally, efforts to assist adults can be complex and expensive (Galbraith et al. 1997). Furthermore, hatchlings and neonates are highly charismatic, and serve as a gateway for the public into conservation efforts (Burke 2015). Eggs are also fairly easy to obtain as few turtle species

provide any parental care and turtle nests can be easy to find. Some head-start programs have even obtained eggs from road-killed females and raised them to successful hatching (Wood and Herlands 1997; Herlands et al. 2004). Head-starting has also been effective at restoring species where they were once extirpated, by taking hatchlings from other areas and reintroducing them to regions they either historically occurred, or into suitable habitat in which they did not previously occur (Griffiths et al. 2012; Hunter and Gibbs 2013).

Criticism of head-starting, particularly for turtles, is not a novel concept (Woody 1991; Frazer 1992; Klemens 2000; Siegel and Dodd 2000). Many population models for sea turtles and other aquatic turtles have shown that survival of older, sub-adult, and adult individuals are far more valuable to population growth (Crouse et al. 1987, Heppel 1998), as turtles have evolved reproductive characteristics that compensate for their vulnerability as eggs and hatchlings (Congdon et al. 1993; Congdon et al. 1994). This leads into Frazer's (1992) criticism of head-starting as a "half-way" technology that does not address the causes of conservation woes, merely the symptoms of it. By spending resources where they are not most impactful, head-starting can cause a worthless, or perhaps even detrimental, distraction from the true problems the species of interest faces (Frazer 1992). Furthermore, head-starting can be costly, and if neonates are behaviorally impaired from captivity, or diseased, survival of head-starts can end up being too low to justify the cost and effort (Dodd and Seidel 1991).

A classic example of head-starting is that of Kemp's Ridley sea turtles (*Lepidochelys kempii*) which began in 1978. Sea turtle populations were suffering heavily

from bycatch mortality in commercial fisheries, with estimated annual deaths of Kemp's Ridley sea turtles reaching 4300 turtles before 2007 (Finkbeiner et al. 2011). While the aforementioned head-starting released approximately 18,000 turtles by 1992, the success of the program was never able to be evaluated properly due to very low recapture rates of head-started turtles (Bowen et al. 1994). Sea turtle populations continued to decline until the United States required turtle excluder devices (TEDs) to be affixed to the trawl nets of commercial fishers (Finkbeiner et al. 2011). Bycatch mortality estimates of sea turtles decreased by ~62% to 2700 annual deaths post regulation of fisheries (Finkbeiner et al. 2011). Population models have shown that population growth rates for Kemp's Ridley and Loggerhead sea turtles are more sensitive to changes in adult and sub-adult survivorship than juvenile survivorship, and head-starting into an environment where risks to these ages are high will have little effect (Crouse et al. 1987, Heppel et al. 1994). Similar models have provided similar results for terrapins and other aquatic turtles (Cunnington and Brooks 1996; Mitro 2003; Enneson and Litzgus 2008).

Here we analyzed long-term demographic data to carefully understand the life histories and vital rates for a population of diamondback terrapins in Chesapeake Bay. We used mark-recapture methods to estimate apparent survival parameters and constructed an age classified matrix population model and then conducted a perturbation analysis to determine the most sensitive life stages. We also used mark-recapture methods to compare the vital rates of head-started and non-head-started individuals to examine the efficacy of the ongoing head-start program.

## Methods

### *Study Species*

Diamondback terrapins are small emydid turtles inhabiting brackish salt marshes of the east coast of the United States from Cape Cod MA, to Galveston TX. They are a sexually dimorphic species, with females achieving straight-line carapace lengths up to 365mm while males are significantly smaller at 160mm maximum carapace length. They are primarily molluscivorous, feeding on gastropods such as periwinkle snails (*Littorina littorea*) and bivalves; however, they also feed on fish, crabs, and plants as well (Ehret and Werner 2004; Ernst and Lovich 2009).

Like most turtle species, diamondback terrapins are long-lived, with long generation times, although there is variation in life history traits with latitude. Female terrapins in Chesapeake Bay reach sexual maturity at approximately eight years of age, while males are sexually mature as early as four years of age (Roosenburg 1991). Age of first reproduction varies across their range with southern populations reproducing at smaller body sizes, with smaller clutches of larger eggs (Allman et al. 2012). Low recruitment in terrapin populations is largely due to predation of nests by mesopredators; mainly raccoons (*Procyon lotor*) but also foxes (*Vulpes vulpes*) (Feinburg and Burke 2003; Butler et al. 2004). These predators are often subsidized by human activity and their increase in both range and abundance has been well documented (Gehrt et al. 2002; Larivierre 2004). For many turtle species and other long-lived organisms, these co-evolved life history traits show an inability of populations to sustain chronic mortality, particularly in adult and sub-adult life stages (Congdon et al. 1993, Congdon et al. 1994).

### *Study Site*

The Paul S. Sarbanes Ecosystem Restoration Project at Poplar Island is a 478 hectare restoration of an island in Chesapeake Bay located near Sherwood, MD (Roosenburg et al. 2014). Poplar island once encompassed approximately 450 hectares in 1850; however, due to erosion, there were only three small remnant islands (less than four hectares in total) remaining by 1998. A large-scale restoration effort by the United States Army Corps of Engineers and the Maryland Port Administration of the Maryland Department of Transportation began in 2000 using dredge material from the approach channels to Baltimore Harbor (Roosenburg et al. 2014). By 2002 the perimeter dike was completed and by 2027 the island will consist of ~50% wetland cells and ~50% coastal plains forest. Poplar Island is segmented into six cells, with cells one, three, four, and five composing the wetland portion. (Figure 1).



Figure 1. Orientation map of Poplar Island. Yellow dots indicate crab pot locations (Chapter 2).

Diamondback terrapins began nesting on Poplar Island in 2002 on the perimeter dike to open sandy areas near Cell 3 and the Notch, and later within the footprint of the island when gaps in the perimeter dike allowed access to the island interior via completed wetland cells, an until 2008 into Cell 6 (Roosenburg et al. 2014; Figure 1). Currently, nesting is most concentrated in the Notch where the absence of rocky rip-rap allows ease of terrapin access to the shoreline, as well as a small sandy beach near cell three (Figure 1). Terrapins nesting on Poplar Island experience high levels of nest success due to the absence of mammalian predators such as foxes (*Vulpes vulpes*) and raccoons (*Procyon lotor*). Predation of nests, particularly in the Notch, do occur, primarily from eastern kingsnakes (*Lampropeltis getula*), and black rat snakes (*Pantherophis alleghaniensis*). Four red foxes traversed the frozen channels to the island in 2015, and subsequently nest success was lower during 2015 and 2016; however, they have since been culled by the U.S. Fish and Wildlife Service. As such, recruitment has remained relatively high on the island (60%-85% nest success among years, Roosenburg et al. 2014).

#### *Mark-Recapture*

A mark-recapture study of diamondback terrapins at Poplar Island has been ongoing since 2004, in which we used a combination of fyke nets, baited traps, and crab pots to capture terrapins. Two to four fyke nets were placed in the wetland cells of the island every Monday and removed every Friday due to limited access to the site on weekends. Two to four cylindrical baited traps with large entry funnels at the base of the trap were also set in the wetland cells during the 2005-2017 seasons, and fifteen crab pots were fished in the channel separating Poplar Island and Coach's Island (Figure 1). Nets

and traps were checked daily to minimize stress and avoid mortality, and all captured terrapins were removed and taken to the on-site lab for processing.

For each terrapin captured we recorded: age, sex, weight, reproductive condition, plastron length, straight-line carapace length, shell width, shell height, width of right pectoral scute, and head width. Mitutoyo digital calipers measured all morphological traits ( $\pm 1$  mm) except for head width ( $\pm 0.01$  mm). An Ohaus Scout, top loading digital scale determined mass ( $\pm 1$  gram). We recorded any shell abnormalities or injuries observed during processing. We determined age by counting seasonal growth annuli on plastron scutes which are indicative of periods of rapid growth during the active season (Cagle 1946). A second crew member always verified ages to assure data quality. When no age could be determined due to excessive wear on the shell, age was either determined by a previous capture when possible, or not recorded. We verified this method by comparing annuli accumulation of recaptures in subsequent years and confirmed its accuracy annually and with NE hatchlings recaptured several years later. Sex was determined by examining cloacal positioning on the tail of the animal. Male terrapins have wider, thicker tails that, when extended, have cloacae positioned farther from the posterior edge of the carapace. Female terrapins have narrower tails with cloacae that do not extend much past the posterior edge of the carapace. Any individual that was too young or too small to sex was recorded as a juvenile. All female terrapins of reproductive size were palpated to determine if they were gravid.

All newly captured terrapins were marked with two uniquely identifying tags. A hole was drilled in the ninth right marginal scute, where we attached a Monel tag with a

unique four-digit code. The shavings from the drill were then collected as a tissue sample and preserved in 95% ethanol. Each turtle was also marked in the hind-right inguinal region with a ten or fifteen-digit alphanumeric Passive Integrated Transponder (PIT; Biomark, Inc., Boise, ID) using a 12-gauge sterilized needle.

### *Nesting Surveys*

Nesting surveys were also conducted at Poplar Island beginning in 2002. Areas where terrapin nesting was known to occur, or areas that were suspected to be potential nesting habitat, were surveyed twice daily. When a nest was found, the eggs were excavated and weighed, and clutch size was recorded. Depth to the top and bottom of the nest cavity were measured to the nearest centimeter, and a GPS point was recorded in decimal degrees. After processing a nest, we would rebury it, place a metal screen made from 1.25cm<sup>2</sup> mesh over the top to prevent invasion by avian predators (Roosenburg et al. 2014). Approximately fifty days from the time the nest was discovered, we placed aluminum flashing around the nest to prevent hatchling escape, and surveyed nests at least three times per day for the emergence of hatchlings. Hatchlings, once found, were taken back to the lab for processing. We also documented the number of successful hatchlings from each nest.

For each hatchling, plastron length, straight-line carapace length, shell height, and shell width were recorded to the nearest tenth of a millimeter using Mitutoyo digital calipers. Mass ( $\pm 0.1$  g) was determined using a digital palm scale. Using a scalpel, we marked all hatchlings with a notch code identifying the year and season in which they emerged (spring or fall). Additionally, each hatchling was implanted in the hind-right

limb with a coded wire tag (Northwest Marine Technology, Inc., Shaw Is. WA). After being tagged hatchlings were held overnight to ensure tag retention, and then released into one of the wetland cells. Hatchlings entering the head-start program were given a separate notch code, unique to their nest.

Starting in 2005, we collaborated with Arlington Echo Outdoor Education center to begin a head-start program in which 150-250 hatchlings recovered from fall-emergent cohorts on Poplar Island are reared in classrooms throughout the state of Maryland annually. In 2008 the National Aquarium in Baltimore began participating, followed by the Maryland Environmental Service in 2011. Participating teachers are responsible for taking regular measurements on captive turtles for the 8-9 months. At this point, head-started turtles have achieved the body size of a 3-5-year-old wild counterpart. Prior to release, all head-started turtles are PIT tagged via the same procedure outlined above, so that they could be individually identified.

#### *Data Analysis*

We used Cormack-Jolly-Seber (CJS) models implemented in MARK via the RMark interface (Laake 2013) to estimate apparent survival and recapture probability for wild female terrapins captured at Poplar Island, as well as natural emergence and head-started marked hatchlings. Models were chosen based on Akaike Information Criteria corrected (AICc). Apparent survival ( $\phi$ ) was modeled as a constant or as a function of age for wild turtles. Recapture probability ( $p$ ) was modelled as a constant, age-dependent, or age and time dependent. For hatchlings, survival was modeled as a function of all combinations of: age, type (head-start or natural emergence), and time, as well as

recapture probability. For age structured models of wild females, ages 1 and 2 were binned together as the juvenile age class, ages 3-7 were binned as sub-adults, and all terrapins aged 8 or older were considered sexually mature adults in concordance with the estimated age of first reproduction for female terrapins in Chesapeake Bay (Roosenburg 1990). Hatchlings were grouped as 1-3 year-olds, 4-7 year-olds, and those aged 8-12, with 12 being the maximum possible age for a marked hatchling since head-starting began.

Survival estimates from CJS models were then used to parameterize a matrix population model for wild female terrapins (Leslie 1945). Where  $S_J$  is survival of juveniles,  $S_{SA}$  is survival of sub-adults, and  $P_A$  is survival of adults given as the equation:

$$P_A = \frac{1 - S_{SA}^{d_i - 1}}{1 - S_A^{d_i}} S_A \text{ (Caswell 2001),}$$

where  $d_i$  is the duration of the stage and, which in the case of adults is longevity minus the age of entry to the stage, and the apparent survival estimate obtained from CJS modelling was used as  $S_A$ . For our population, longevity was assumed to be 40 years (Hildebrand 1932) thus stage duration was 32 years. The fertility parameter ( $F$ ), was the product of mean clutch frequency, mean clutch size, probability of a successful clutch, the proportion of offspring that will be female, and the survival of hatchlings in the remaining 9 months of the year or:  $S_J^{9/12}$

The matrix population model ( $A$ ; Figure 2) was created using the apparent survival estimate for sub-adults as  $S_{SA}$ , and  $P_A$ , as calculated above; however, recaptures of juveniles (hatchlings, one, and two-year olds) were insufficient to estimate apparent

survival. We mathematically derived juvenile survivorship by setting projected population growth rates equal to realized growth rates and numerically solving for juvenile survivorship (see Mitro 2003).

$$A = \begin{bmatrix} 0 & 0 & 0 & 0 & 0 & 0 & F_7 & F \\ S_J & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & S_J & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & S_{SA} & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & S_{SA} & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & S_{SA} & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & S_{SA} & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & S_{SA} & P_A \end{bmatrix}$$

Figure 2. Structure of matrix model used in perturbation analysis and head-start

simulations  $F_7 = 0$  in all cases except for head-start modelling (see Figure 6).

Projected population growth rate ( $\lambda_{projected}$ ) is defined as the dominant eigenvalue of the projection matrix outlined above (Caswell 2001). Realized population growth rate ( $\lambda_{real}$ ) can be described as the geometric mean of the observed annual growth rates of the population among years ( $\lambda_i$ ). We used a POPAN parameterization of Jolly-Seber models to estimate abundance at each sampling occasion ( $N_i$ ) at Poplar Island. The best model was chosen on the basis of AICc, and all parameters were modelled as constant or a function of time. After estimating abundance, we then calculated  $\lambda_i$  as:

$$\lambda_i = \frac{N_{i+1}}{N_i}.$$

We then took the geometric mean of all  $\lambda_i$  to obtain the realized population growth rate. We then set realized growth rate equal to the dominant eigen value of matrix  $A$  and used a bisection algorithm to numerically approximate  $S_J$  with a convergence error tolerance of  $10^{-8}$ . The fertility parameter ( $F$ ) was the product of mean clutch frequency, mean clutch size, probability of a successful clutch, the proportion of offspring that will be female, and the survival of hatchlings in the remaining 9 months of the year after emergence or:  $S_J^{9/12}$ . Terrapins in Chesapeake Bay have been observed depositing 1-3 clutches per year (Roosenburg and Dunham 1997), so a value of 2 was used for clutch frequency, mean clutch size was taken to be 13.69 per observations from nesting surveys, probability of successful clutches was observed to be 0.746, and sex ratios were heavily biased toward females with 82% of offspring being female (Roosenburg 1990; Roosenburg et al. 2013; Converse 2016).

We calculated eigenvalue elasticities of parameters in our model, which are defined as proportional changes in  $\lambda$ , as a function of infinitesimally small changes in vital rates. However, due to a mathematical identity, all off diagonal elasticities are equal, thus in our case  $eF = eS_J = eS_{SA}$  by definition (Caswell 2001). Because of this property, we also perturbed parameters in meaningful proportions that reflect plausible biological scenarios (-10% to +20%). We also modeled different levels of head-starting in the population by inflating the  $S_J^{9/12}$  term in our calculation of  $F$  to reflect various proportions of hatchlings surviving at a rate of 100%. We considered two cases of head-starting: one where age of first reproduction was the same among head-started and wild

individuals, and one where head-started individuals first reproduced with the same parameters as wild counterparts, but one year earlier (Figure 2).

All analyses were conducted in R (R Core Team 2017). MARK analyses were performed using the package “RMark” (Laake 2013), and matrix evaluation was done using package “popbio” (Stubben and Milligan 2007).

### Results

For wild female turtles, the data best supported the model where apparent survival was a function of age, and where recapture probability was a function of age and time (Table 1). Apparent survival was estimated to be  $0.64 \pm 0.02$  (estimate  $\pm$  SE) for sub-adults and  $0.80 \pm 0.01$  for adults. Recapture probabilities varied with time and increased overall with age. For hatchlings, the most supported model included age, type, and time for both apparent survival and recapture probability (Table 2). Apparent survival estimates for natural emergence hatchlings and head-started hatchlings varied with time (Figure 3). Mean apparent survival for NE hatchlings in age groups [1,4), [4,8), [8,12] was  $0.41 \pm 0.01$ ,  $0.71 \pm 0.01$ , and  $0.76 \pm 0.01$  respectively. Mean apparent survival for head-started hatchlings in age groups [1,4), [4,8), [8,12] was  $0.31 \pm 0.01$ ,  $0.62 \pm 0.01$ , and  $0.68 \pm 0.01$  respectively. Recapture probabilities varied by year and were consistently lower for the [8,12] age class than the other two.

Table 1. Model selection of CJS models of apparent survival and recapture probability for wild females.  $\phi$  indicates apparent survival and  $p$  indicates recapture probability. Both parameters were modeled as a combination of age, time, or a constant

<b>Model</b>	<b># of Parameters</b>	<b>AICc</b>	<b><math>\Delta</math>AICc</b>	<b>Weight</b>	<b>Deviance</b>
$\phi(\text{age})p(\text{age} + \text{time})$	42	6468.11	0	0.99	1300.0
$\phi(\text{age})p(\text{time})$	16	6480.13	12.0152	0.01	1365.0
$\phi(1)p(\text{age} + \text{time})$	40	6524.07	55.9543	0	1360.1
$\phi(1)p(\text{time})$	14	6531	62.8929	0	1420.0
$\phi(\text{age})p(\text{age})$	6	6557.01	88.8991	0	1462.1
$\phi(\text{age})p(1)$	4	6569.52	101.408	0	1478.6
$\phi(1)p(1)$	2	6607.73	139.615	0	1520.8
$\phi(1)p(\text{age})$	4	6609.75	141.636	0	1518.8

Table 2. Model selection of CJS models of apparent survival and recapture probability for natural emergence and head-started hatchlings (type).  $\Phi$  indicates apparent survival and  $p$  indicates recapture probability. Both parameters were modeled as combinations of age, NE or HS (type), and time.

Model	# of Parameters	AICc	$\Delta$ AICc	Weight	Deviance
$\phi(\text{age} + \text{type} + \text{time})p(\text{age} + \text{type} + \text{time})$	96	13186.1	0	1	3913.6
$\phi(\text{age} + \text{type})p(\text{age} + \text{type} + \text{time})$	85	13319.8	133.75	0	4069.6
$\phi(\text{age})p(\text{age} + \text{type} + \text{time})$	84	13409.2	223.12	0	4161.0
$\phi(\text{age} + \text{type} + \text{time})p(\text{age} + \text{type})$	24	13742	555.91	0	4614.5
$\phi(\text{age} + \text{type})p(\text{age} + \text{type})$	13	13969.5	783.48	0	4864.1
$\phi(\text{age})p(\text{age} + \text{type})$	12	14028.5	842.40	0	4925.0
$\phi(\text{type})p(\text{age} + \text{type} + \text{time})$	82	16480	3293.9	0	7235.8
$\phi(1)p(\text{age} + \text{type} + \text{time})$	81	16558.3	3372.2	0	7316.1
$\phi(\text{type})p(\text{age} + \text{type})$	10	17132	3945.9	0	8032.6
$\phi(1)p(\text{age} + \text{type})$	9	17213.9	4027.8	0	8116.5
$\phi(\text{age} + \text{type} + \text{time})p(\text{age} + \text{time})$	56	19710.4	6524.3	0	10518.7
$\phi(\text{age} + \text{type})p(\text{age} + \text{time})$	45	19979.1	6793.0	0	10809.5
$\phi(\text{age} + \text{type} + \text{time})p(\text{time})$	28	22615.7	9429.6	0	13480.2
$\phi(\text{age} + \text{type} + \text{time})p(1)$	17	22739.5	9553.4	0	13626.0
$\phi(\text{age} + \text{type})p(\text{time})$	17	22833.3	9647.2	0	13719.8
$\phi(\text{age} + \text{type})p(1)$	6	23105.4	9919.3	0	14014.0
$\phi(\text{type})p(\text{age} + \text{time})$	42	23204.5	10018.5	0	14040.9
$\phi(\text{type})p(\text{time})$	14	23255	10068.9	0	14147.6
$\phi(\text{type})p(1)$	3	23661.5	10475.4	0	14576.1

$\phi(\text{age})p(\text{age} + \text{time})$	44	24318. 4	11132. 3	0	15150.7
$\phi(\text{age})p(\text{time})$	16	24538. 2	11352. 1	0	15426.7
$\phi(1)p(\text{age} + \text{time})$	41	24917. 5	11731. 5	0	15755.9
$\phi(1)p(\text{time})$	13	25035. 8	11849. 8	0	15930.4
$\phi(\text{age})p(1)$	5	25285. 5	12099. 4	0	16196.1
$\phi(1)p(1)$	2	25715. 4	12529. 4	0	16632.0

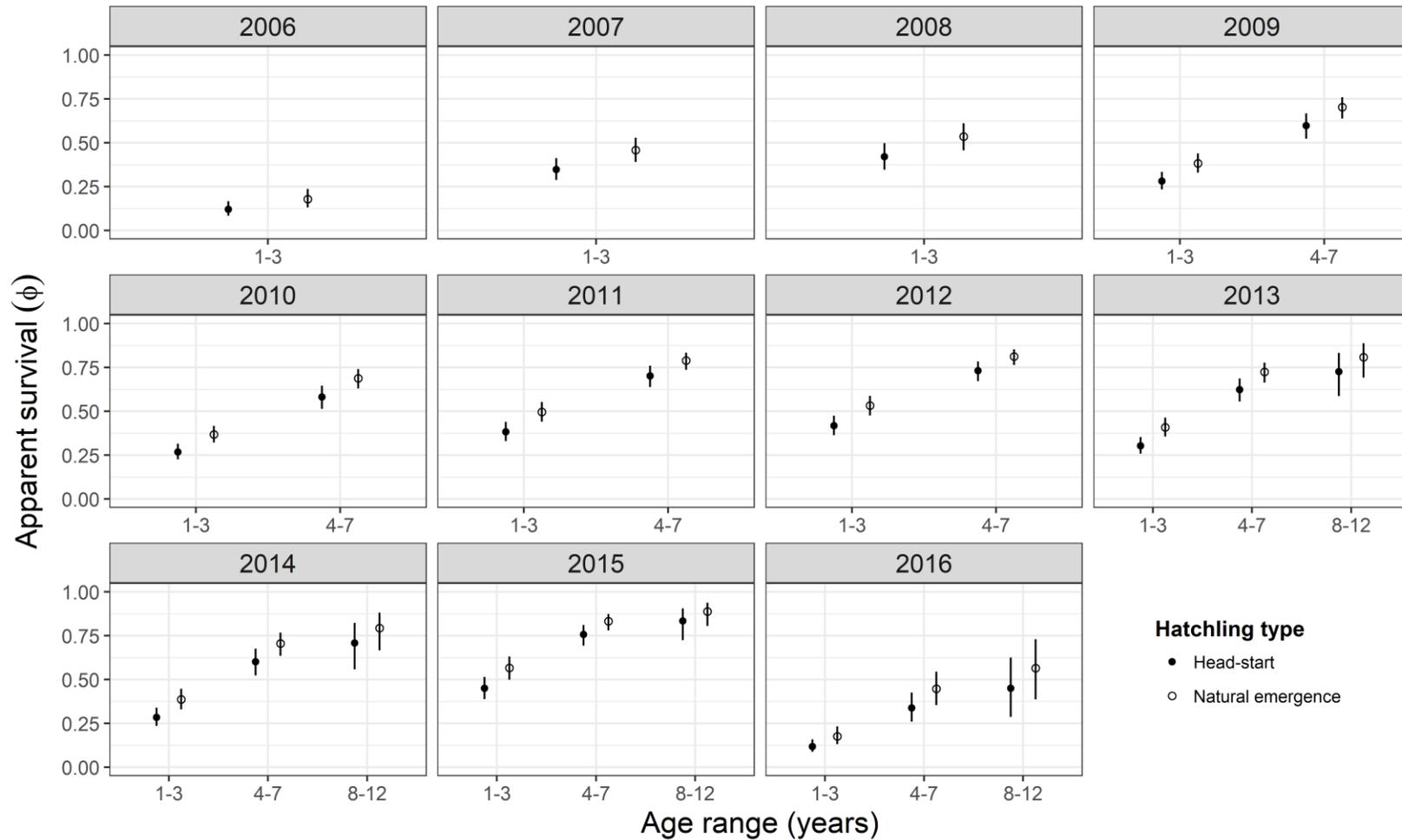


Figure 3. Apparent survival estimates with 95% confidence intervals for natural emergence and head-started hatchlings. Estimates varied with time and for three different age groups (1-3, 4-7, 8+).

Our mark-recapture data supported a POPAN abundance model with all parameters time-dependent (Table 4). Using abundance estimates from POPAN models (Table 3), we calculated  $\lambda_{real}$  to be 1.047, which lead to our bisection algorithm estimating juvenile survivorship ( $S_J$ ) to be  $\sim 0.55$ . This value for juvenile survivorship is comparable to the one obtained by Mitro (2003) using similar methods. Our estimate of  $S_J$  lead to a fertility estimate ( $F$ ) of 10.65 offspring per adult female per year. Due to the large stage duration in calculation of  $P_A$ , differences in  $S_A$  and our calculations of  $P_A$  were negligible ( $<.001$ ).

Table 3. Abundance estimations from POPAN models and calculated realized  $\lambda$ . Geometric mean of realized  $\lambda$  indicates the overall growth rate over the study period.

Year	$N_i$	$\lambda_i$
2005	275	0.55
2006	154	1.86
2007	286	0.84
2008	240	4.10
2009	986	1.37
2010	1347	0.85
2011	1151	1.62
2012	1874	0.87
2013	1642	0.78
2014	1289	0.75
2015	978	1.45
2016	1418	N/A
<b>GeoMean:</b>		<b>1.047</b>

Table 4. Model selection of POPAN abundance models.  $\Phi$  indicates apparent survival,  $p$  indicates recapture probability,  $pent$  indicates probability of entrance into the population, and  $N$  is the abundance. All parameters in all models were either functions of time or constant.

Model	# of Parameters	AICc	$\Delta$ AICc	Weight	Deviance
$\phi(\text{time})p(\text{time})pent(\text{time})N(1)$	41	8896.38	0	1	-5689.03
$\phi(\text{time})p(\text{time})pent(1)N(1)$	29	8925.82	29.44192	0	-5635.1
$\phi(\text{time})p(1)pent(\text{time})N(1)$	28	8985.84	89.45708	0	-5573.05
$\phi(1)p(\text{time})pent(\text{time})N(1)$	29	9025.93	129.55182	0	-5535
$\phi(1)p(1)pent(\text{time})N(1)$	16	9085.85	189.46432	0	-5448.7
$\phi(1)p(\text{time})pent(1)N(1)$	17	9285.55	389.17231	0	-5251.1
$\phi(\text{time})p(1)pent(1)N(1)$	16	9868.5	972.12142	0	-4666.08
$\phi(1)p(1)pent(1)N(1)$	4	10166.4	1270.03978	0	-4344.01

Eigenvalue elasticities were calculated as 0.282 for  $P_A$  and 0.090 for all other parameters ( $S_J$ ,  $S_{SA}$ ,  $F$ ). When parameters were perturbed, changes in  $S_{SA}$  effected the most change in the population growth rate. Changes in growth rate ranged from -0.05 with a 10% decrease in sub-adult survival to +0.09 with a 20% increase in sub-adult survival, -0.03 to +0.07 with changes in adult survival, -0.02 to +0.03 with changes in juvenile survival, and from -0.01 to +0.02 with changes in fertility (Figure 4). Simulated levels of head-starting had little effect on growth rate, with increases ranging between .001 to .01 when age at first reproduction was not altered. (Figure 5). When head-starts were able to reproduce at age 7, increases in growth rate ranged from 0.001 to 0.02 (Figure 5).

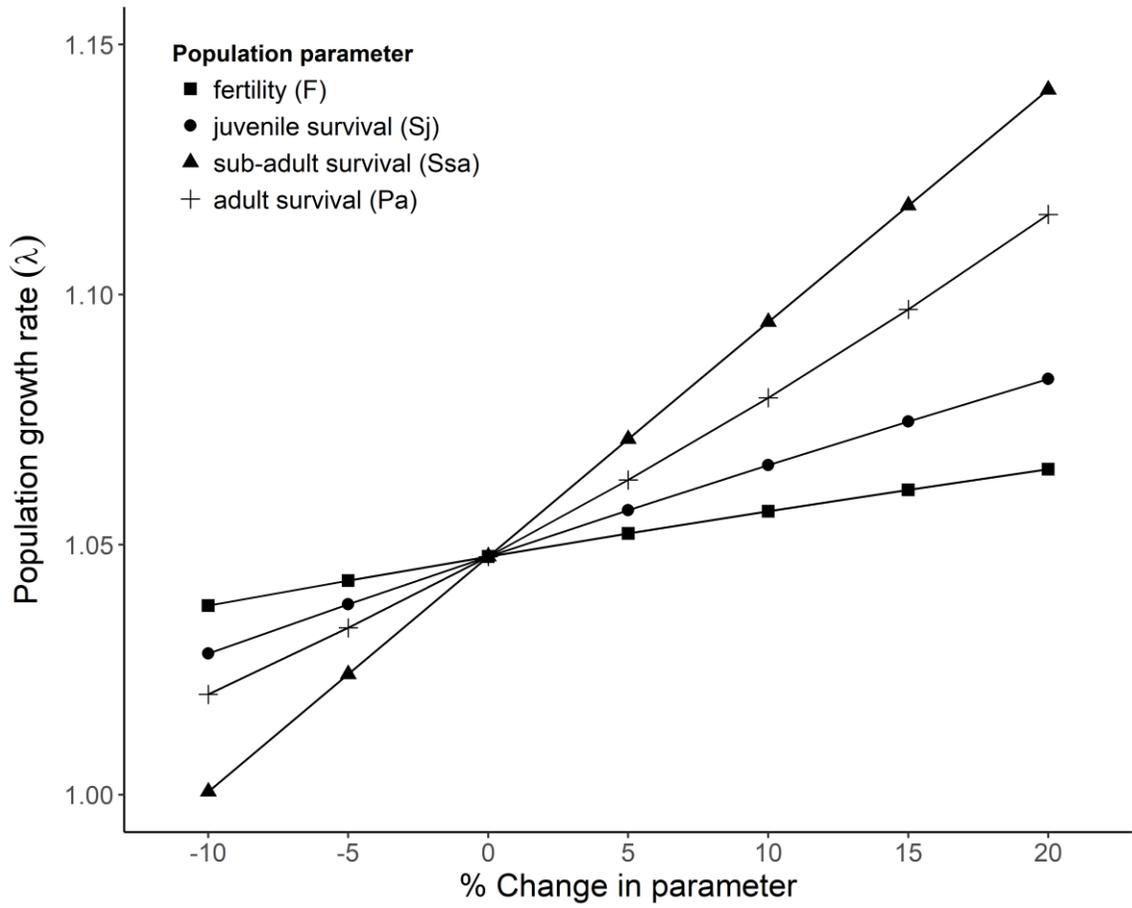


Figure 4. Response of population growth rate to proportional changes in parameters ranging from -10% to +20%.  $F$  represents per capita fertility,  $S_j$  is survival of juveniles,  $S_{SA}$  is survival of sub-adults, and  $P_A$  is survival of adults.

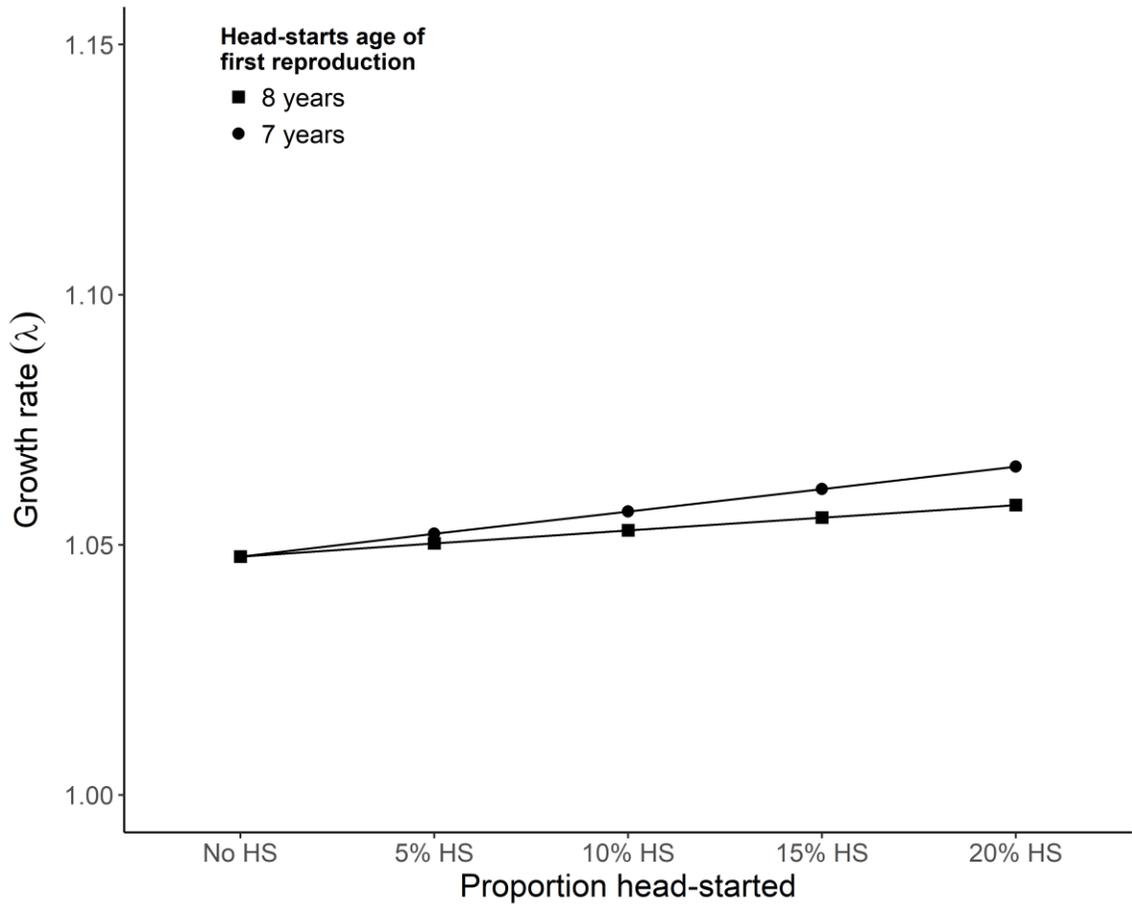


Figure 5. Response of population growth rate to levels of head-starting ranging from no hatchlings head-started to 20% of hatchlings head-started. Two cases were considered: one with reproduction beginning at 7 for head-starts, and one where all individuals reproduce at the same age (8).

$$A = \begin{bmatrix} 0 & 0 & 0 & 0 & 0 & 0 & F_7 & F \\ 0.55 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0.55 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0.64 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0.64 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0.64 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0.64 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0.64 & 0.80 \end{bmatrix}$$

Figure 6. Matrix model with parameter values.  $F = \{10.65, 10.96, 11.26, 11.56, 11.87\}$  for 0, 5%, 10%, 15% or 20% head-started respectively.  $F_7 = \{0, 0.53, 1.10, 1.70, 2.31\}$  for 0, 5%, 10%, 15% or 20% head-started respectively.

## Discussion

The best fit model for apparent survival and recapture modeled survival as a function of age and recapture as a function of age and time. Variation in recapture probabilities is likely attributed to changing crew members between years of data collection and somewhat inconsistent sampling effort as a result. Recapture varied with age, as expected, as captures of juvenile terrapins were limited. Survival estimates varied with age, which indicates that survival in terrapins does differ between sub-adults and adults which previous studies have failed to account for.

Apparent survival estimates for adult females from Poplar Island were similar to those estimated in the literature, (0.61-0.95) and to our knowledge there are no published studies which have estimated survival of sub-adults separate from adults at the time of this writing (Tucker et al. 2001; Mitro 2003, Witczak et al. 2014). Apparent survival, the survival parameter estimated in CJS models, does not distinguish between emigration and

death, so it is almost certain that “true” survival is higher than what we estimated. Additionally, the possibility exists that some adult female terrapins at Poplar Island are transients, coming only to the island to nest and thus reducing our apparent survival estimates. The resource environment at Poplar Island is not well studied; however, periwinkle snails, a major food source for terrapins are notably absent in the wetland cells of the island, and present in relatively low abundance on the perimeter (personal observation). Females may be spending time elsewhere to feed, bask, and produce a clutch, and then taking advantage of the quality nesting habitat at Poplar Island. This transient behavior could result in apparent survival estimates that are lower than the actual survival of adult females, if large amounts of emigration are occurring due to a poor resource environment. Our estimate for sub-adult survival was on the low end of estimates that did not discriminate by age or were for adults only. Derelict crab pots containing as many as 40 dead terrapins have been observed at Poplar Island, so it is possible that our survival estimates may be accurately reflecting substantial bycatch mortality. Further study into the resource environment on the island, and the movement of terrapins is needed to answer these questions.

Our results indicate that survival of sub-adults, and adults to a lesser extent, is a larger component of population growth than survival in the first three years, or reproductive success. Eigenvalue elasticities calculated for adult survival were higher than those for sub-adult survival, however these values treat adults as a stage (ages 8-40) and calculate changes in growth rate due to infinitesimally small changes in sub-adult survival at each age in the stage independently (3 through 7). When perturbing vital rates

on a stage by stage basis (Figure 4), we see that growth is most sensitive to changes in the sub-adult age class.

Head-starting also proved to be largely ineffective at fortifying population growth, especially in the presence of factors reducing survival of sub-adults and adults. Differences between survival of head-started hatchlings and natural emergence hatchlings were slight; however, survival of natural emergence hatchlings was slightly higher than that of head-started hatchlings, suggesting there could be negative behavioral consequences to keeping hatchlings in captivity after emergence. This difference is however subtle and may not be significant in the presence of demographic stochasticity (Caswell 2001). Further stochastic modelling should be done to examine significance of post-release survival differences.

The life history traits of the diamondback terrapin suggest that harvest, be it intentional or incidental, is not a process which can be sustained. Long-lived organisms such as terrapins possess a suite of coevolved life history traits, allowing them to compensate for low fecundity, delayed maturity, and low neonate survival, at the cost of very little tolerance to chronic mortality at older life stages (Congdon et al. 1994). Terrapins face a number of additive mortality sources including road mortality and bycatch in crab pots intended for the Atlantic blue crab (Bishop 1983, Crawford et al. 2014). Understanding the life history traits of the population are vital to devising the most cost-efficient and effective management strategy for the population. Even simple demographic data can be used to evaluate tradeoffs among various conservation strategies (Enneson and Litzgus 2008, Kissel et al. 2014).

Acting blindly to conserve biodiversity should be avoided wherever possible, especially where the data exist or are easily attainable to devise the proper plan. Wildlife managers are however, often faced with mandatory action and may choose to pursue the easiest route in the absence of scientific data or thorough understanding of life histories. Furthermore, conservation efforts are often too concerned with the number of animals in a population, and thus target their solutions toward increasing that number with half-way technologies, all while ignoring appropriate technologies that seek to fix why abundances are low in the first place (Frazer 1992). In fact, for long-lived organisms, there is evidence to suggest that, despite low abundances, populations can be stable and viable, provided vital rates remain above certain thresholds (Shoemaker et al. 2013). For the diamondback terrapin, head-starting is likely a half-way technology, and proactive management with regards to additive mortality sources is needed, particularly those that are biased toward younger sub-adult turtles such as bycatch in crab pots (Bishop 1983; Roosenburg et al. 1997). Head-starting may still prove a useful tool with regards to raising awareness for conservation, changing public perceptions, and serving as a potential segue into strategies that resemble more appropriate technologies, particularly when appropriate technologies are difficult or impossible to implement.

CHAPTER 2: AN APPROPRIATE TECHNOLOGY FOR TERRAPIN  
CONSERVATION

Abstract

We evaluated the effects of two different Bycatch Reduction Devices (BRDs) on crab and terrapin captures at Poplar Island in Talbot County, MD. Crab pots were fished in five sets of three, with each set containing a control pot with no BRD, a pot affixed with the BRD required in the Maryland recreational fishery, and a new BRD developed by South Carolina Department of Natural Resources. We found no significant difference in the number of legal crabs captured in pots ( $\chi^2 = 2.8$ ,  $df = 2$ ,  $p = 0.25$ ) and no significant difference in the width of crabs caught ( $F_{2,128} = 2.17$ ,  $p = 0.12$ ). We found significantly more terrapins captured in the control pots than either of the two BRD types ( $\chi^2 = 44.11$ ,  $df = 2$ ,  $p < 0.001$ ). Analysis of height to width ratios of terrapins at Poplar Island indicate that the South Carolina BRD may be suboptimal for northern diamondback terrapins despite performing well.

## Introduction

Diamondback terrapins have had a complicated relationship with humans. They were once used as a cheap protein source for slaves by Americans in the 19<sup>th</sup> century, only to be later harvested in large quantities as a delicacy post-civil war (Carr 1952; Burke and Francouer 2014). The market for terrapins eventually crashed due to a combination of factors including overharvesting and prohibition (sherry wine was a primary ingredient of terrapin soup) and populations were given some time to recover. Currently, diamondback terrapins are now a species of “lower but near threatened risk” as assessed by the International Union for the Conservation of Nature (IUCN) in 1996. Regulations regarding terrapins vary by state. Commercial harvest of terrapins is legal in 2 of 16 states in which they occur, with limited non-commercial possession legal in 8 of 16 states (Kennedy, in press). The pet trade and illegal harvesting persist, resulting in the U.S. Fish and Wildlife Service successfully listing diamondback terrapins as an Appendix II species under the Convention on International Trade in Endangered Species (CITES).

Despite overcoming their past threats, terrapins now face new threats brought on by humans. Road mortality, loss of nesting habitat, and bycatch in crab pots are all factors harming terrapin populations (Roosenburg et al. 1997, Crawford et al. 2014). The entire range of diamondback terrapins coincides with the range of the Atlantic blue crab (*Callinectes sapidus*) and is frequently fished, both recreationally and commercially. With the exception rare instances where road mortality exceeds 10% of nesting females, bycatch in crab pots intended for *C. sapidus* is the greatest anthropogenic facing diamondback terrapins (Crawford et al. 2014). Crab pots are frequently baited with clams

or oily fish such as menhaden (*Brevoortia tyrannus*), which are also effective at attracting terrapins (Roosenburg et al. 1997). Sub-adult terrapins, along with sexually mature males in many cases, are most likely to become trapped, and eventually drown in crab pots due to the size of the entry funnels on the traps (Roosenburg et al. 1997). This is concerning as sub-adults are often shown to be essential for population viability in turtles (Crouse et al. 1987; Mitro 2003; Enneson and Litzgus 2008; Chapter 1). Perhaps even more concerning is the notion of “ghost” or derelict crab pots; pots which are either abandoned or lost by the waterman, which can accumulate bycatch mortality over years before they are either washed ashore or removed (Bishop 1983; Guillory et al. 2001, Grosse et al. 2009). Long-term demographic studies have shown altered age structure and population declines in areas with high degrees of crab fishing (Dorcas et al. 2007). Due to the sexual dimorphism exhibited by terrapins, biased sex ratios can also be a consequence of bycatch because females eventually grow to be too large to enter some crab pot funnels (Roosenburg et al. 1997; Grubbs et al. 2018).

Several bycatch mitigation measures have been proposed, the most feasible of which is to equip crab pots with bycatch reduction devices (BRDs) where terrapins and crab fisheries overlap. BRDs are inexpensive and have been demonstrated to be highly effective in mitigating terrapin bycatch in most cases (Roosenburg 2004; Rook et al. 2010; Chavez and Williard 2016). The effects of BRDs on crab catch appear to be minimal in the majority of cases; however, there is some discrepancy among experimental bycatch studies (Roosenburg 2004, Chambers and Maerz, in press). BRD requirements on crab pots vary by state with twelve of the sixteen states inhabited by

terrapins not requiring the use of BRDs whatsoever (Hackney 2010, Chambers and Maerz, in press). New York and New Jersey require the use of BRDs in both recreational and commercial fisheries, while Maryland and Delaware only require BRDs be affixed to recreational crab pots; although compliance estimates in Maryland are concerningly low (Hackney 2010, Radzio et al. 2013, Chambers and Maerz, in press).

In this study we quantify the catch rate of terrapins in the form of catch per unit effort (CPUE) measured in terrapins·crabpot<sup>-1</sup>·day<sup>-1</sup>. We also examine the number and size of legal crabs to look for differences between two BRD pots, and those with no BRD affixed. Terrapin size and activity vary with their range; thus it is important to test different BRD sizes throughout the terrapin's range as different BRD dimensions will likely be more effective in different regions.

## Methods

### *Study Site*

The Paul S. Sarbanes Ecosystem Restoration Project at Poplar Island is a 478 hectare restoration of an island in Chesapeake Bay located near Sherwood, MD (Roosenburg et al. 2014). Poplar island once encompassed approximately 450 hectares in 1850; however due to erosion, there were only three small remnant islands (less than four hectares in total) remaining by 1998. A large-scale restoration effort by the United States Army Corps of Engineers and the Maryland Port Authority began in 2000 using dredged material from the approach channels to Baltimore Harbor (Roosenburg et al. 2014). By 2002 the perimeter dike was completed and by 2027 the island will consist of ~50%

wetland and ~50% coastal plains forest. Poplar island is segmented into six cells, with cells one, three, four, and five composing the wetland portion (Figure 1).

There is extensive commercial crabbing conducted in the harbor and waters surrounding the island. “Ghost pots” have washed up on the island on three occasions since 2015, one of which contained over forty dead terrapins (personal observation). Smith (2012) observed 75-228 actively fished crab pots from June through August. This number does not include any derelict or “ghost pots” out in the harbor.

### *Field Methods*

We fished crab pots in the channel between Poplar Island and Coach’s Island (Figure 1) from June to August 2017. Crab pots were constructed from 1.5” coated crab trap wire and measured 5’ x 2’ x 2’ as opposed to the standard 2’ x 2’ x 2’ crab pot to avoid mortality (see Roosenburg et al. 1997). A total of fifteen crab pots were affixed with either the 1.75” x 4.75” BRD required by the state of Maryland, a new 2.875” x 2” BRD developed by the South Carolina Department of Natural Resources, or no BRD. Crab pots were arranged in five sets of three, with each set forming approximately a 3m equilateral triangle with one of each BRD type present. We set crab pots every Monday, and checked them every 24 hours until Friday, when they were moved to the bank and left open. We measured length, width, and height ( $\pm 1$ mm), sexed, and assigned a molting status (peeler or hard shell) all crabs captured in all pot types.

We recorded the following measurements for each terrapin captured: age, sex, weight, reproductive condition, plastron length, straight-line carapace length, shell width, shell height, width of right pectoral scute, and head width. We used Mitutoyo digital

calipers to take all measurements ( $\pm 1$  mm) except for head width ( $\pm 0.1$  mm). We measured mass using an Ohaus Scout, top loading digital scale ( $\pm 1$  gram). We recorded any shell abnormalities or injuries observed during processing. To determine age, we counted seasonal growth annuli on plastron scutes which are indicative of periods of rapid growth during the active season (Cagle 1946). A second crew member always verified ages to assure data quality. Animals with excessive shell wear that were unable to age were assumed to be 8+ years of age. Sex was determined by examining cloacal positioning on the tail of the animal. Male terrapins have wider, thicker tails, which when extended have cloacae positioned farther away from the posterior edge of the carapace. Female terrapins have narrower tails with cloacae that do not extend much past the posterior edge of the carapace. We considered any individual too small or too young to sex a juvenile.

#### *Data Analysis*

We compared the width of all legal crabs captured in crab pots using a Two-way ANOVA with location as a random effect and BRD type as a fixed effect. The number of legal crabs captured, as well as number of terrapins in pot type was compared using a Chi-square goodness of fit test, expecting equal proportions in all pot types. We calculated catch per unit effort (CPUE) for legal crabs captured as well as terrapins as  $\text{animals} \cdot \text{crab pot}^{-1} \cdot \text{day}^{-1}$ . We also plotted shell height and width of turtles captured in crab pots (as well as the height and width of all turtles captured during the ongoing mark-recapture study at Poplar Island) to the maximum possible heights and widths allowed to pass through the two BRD types, as well as the height and width of all turtles captured

during the ongoing mark-recapture study at Poplar Island. All analyses were conducted in R (R Core Team 2017).

### Results

Neither the number ( $\chi^2 = 2.8$ ,  $df = 2$ ,  $p = 0.25$ ), nor width ( $F_{2,128} = 2.17$ ,  $p = 0.12$ ) of legal crabs captured among BRD types differed significantly (Table 5; Figure 7). Significantly more terrapins were captured in control pots than in either of the two BRD types ( $\chi^2 = 44.11$ ,  $df = 2$ ,  $p < 0.001$ ; Table 6). The majority of the turtles captured were of pre-reproductive age; however, 10 large female terrapins were captured in control pots and deemed sexually mature adults of age 8+ (Figure 8). Of all turtles captured through the course of the study, only one was of the right height to width ratio where it could fit through the South Carolina BRD but was excluded by height from the Maryland BRD (Figure 9). No terrapins were captured that could pass through the Maryland BRD but were excluded from the South Carolina BRD. When comparing BRD dimensions to terrapin dimensions across all captures at Poplar Island, a larger portion of terrapins are able to fit the South Carolina BRD but excluded by the Maryland BRD, than vice versa (Figure 10).

Table 5. Number and catch rate (crabs·crab pot<sup>-1</sup>·day<sup>-1</sup>) of legal crabs. No significant difference in catch (Chi-Square Goodness of fit,  $\chi^2 = 2.8$ ,  $df = 2$ ,  $p = 0.25$ )

BRD Type	Catch	CPUE
Control	48	0.22
Maryland	51	0.24
South Carolina	36	0.17

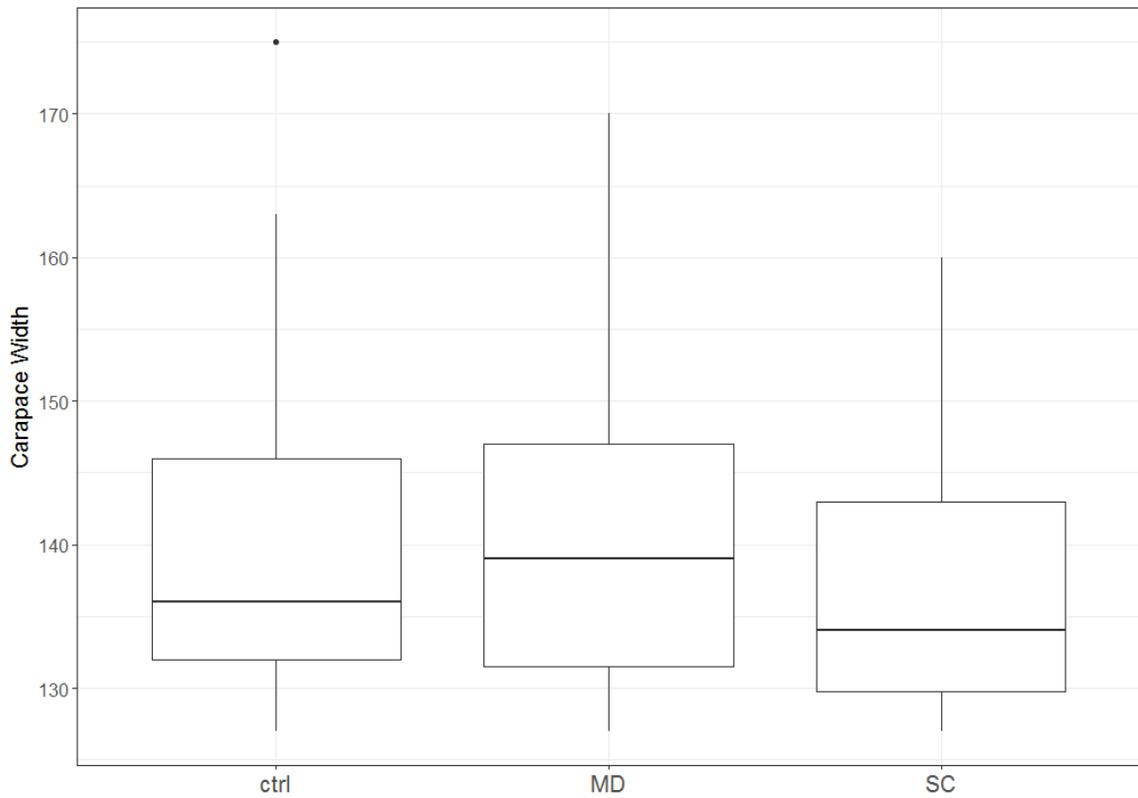


Figure 7. Width (mm) of legal crabs captured in all crab pots by BRD type. No significant difference among BRD types (Two-way ANOVA,  $F_{2,128} = 2.17$ ,  $p = 0.12$ ).

Table 6. Number and catch rate (terrapins·crab pot<sup>-1</sup>·day<sup>-1</sup>) of terrapins. Control pots caught significantly more (Chi-Square Goodness of fit,  $\chi^2 = 44.11$ ,  $df = 2$ ,  $p < 0.001$ )

<b>BRD</b>	<b>Catch</b>	<b>CPUE</b>
Control	41	0.19
MD	7	0.03
SC	6	0.03

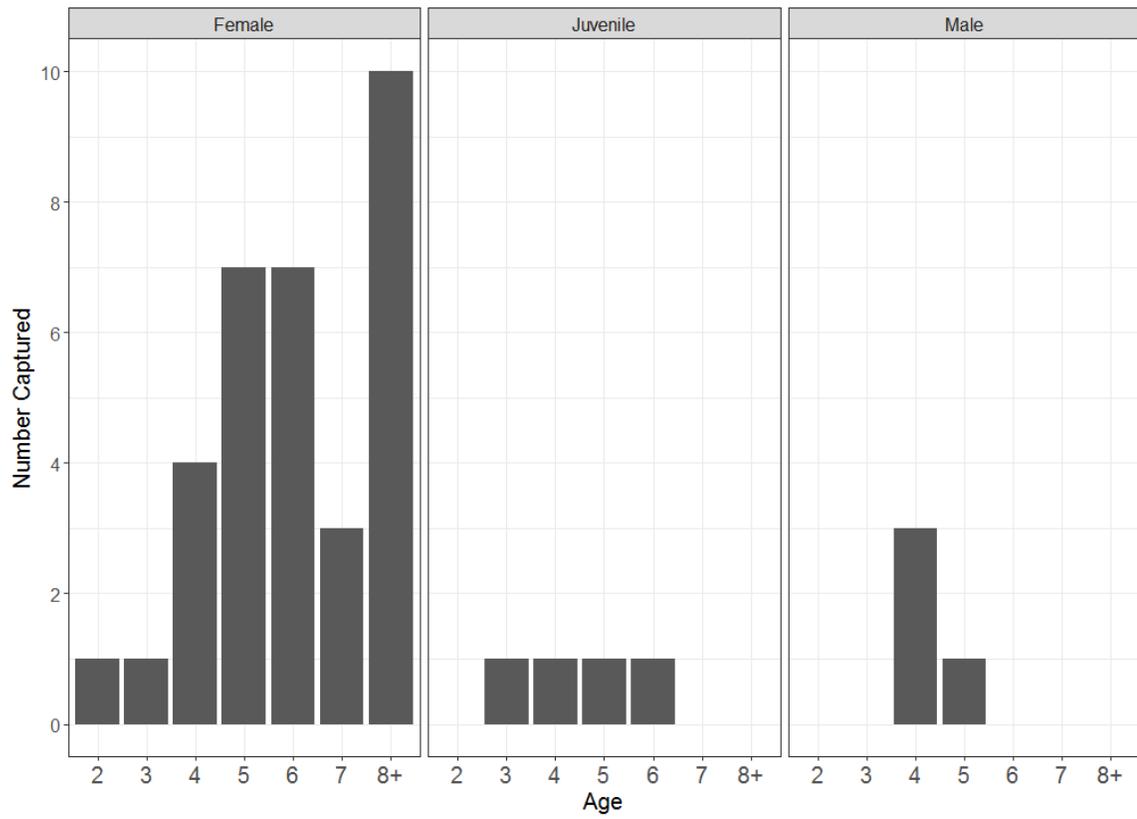


Figure 8. Number of terrapins captured of each sex in control pots, distributed by age.

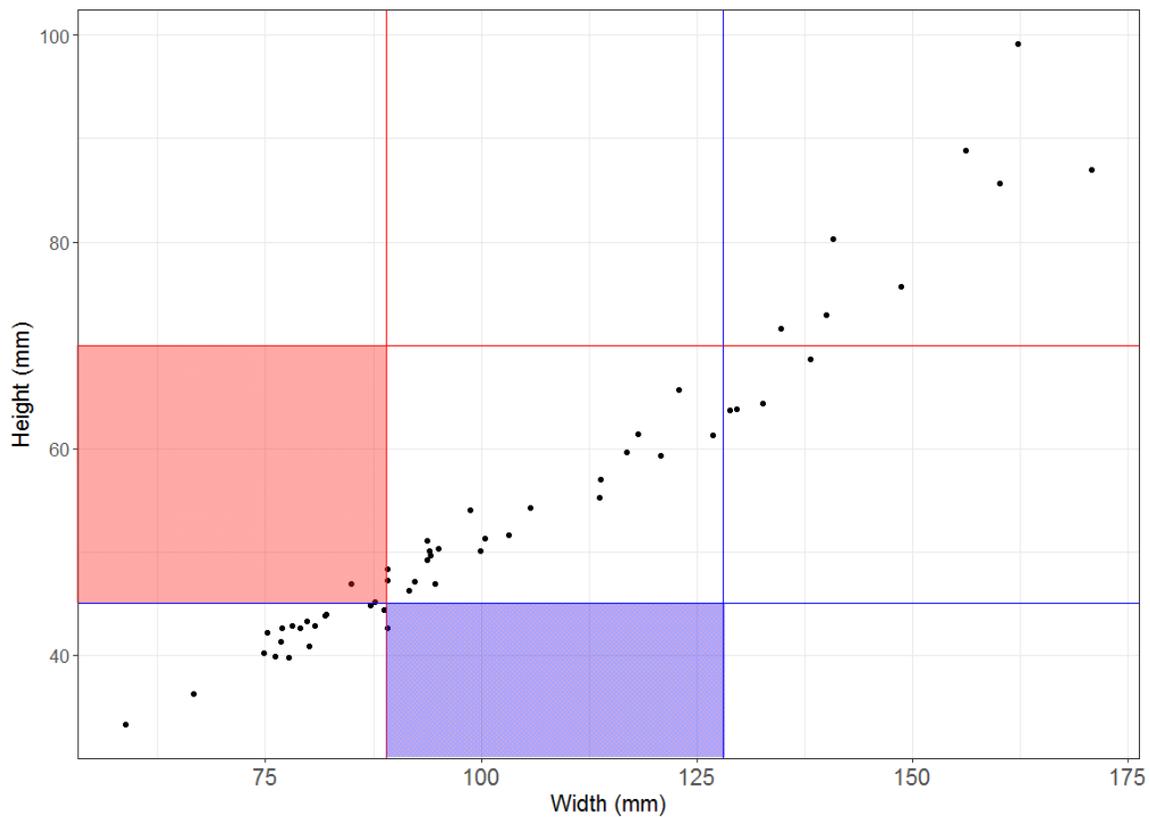


Figure 9. Dimensions of terrapins captured in crab pots compared to dimensions of BRDs. Red lines indicate height and width of South Carolina BRD. Blue lines indicate height and width of Maryland BRD. Red square indicates region where turtles fit through South Carolina BRD but not the Maryland BRD. Blue square indicates region where terrapins fit through Maryland BRD but not South Carolina BRD.

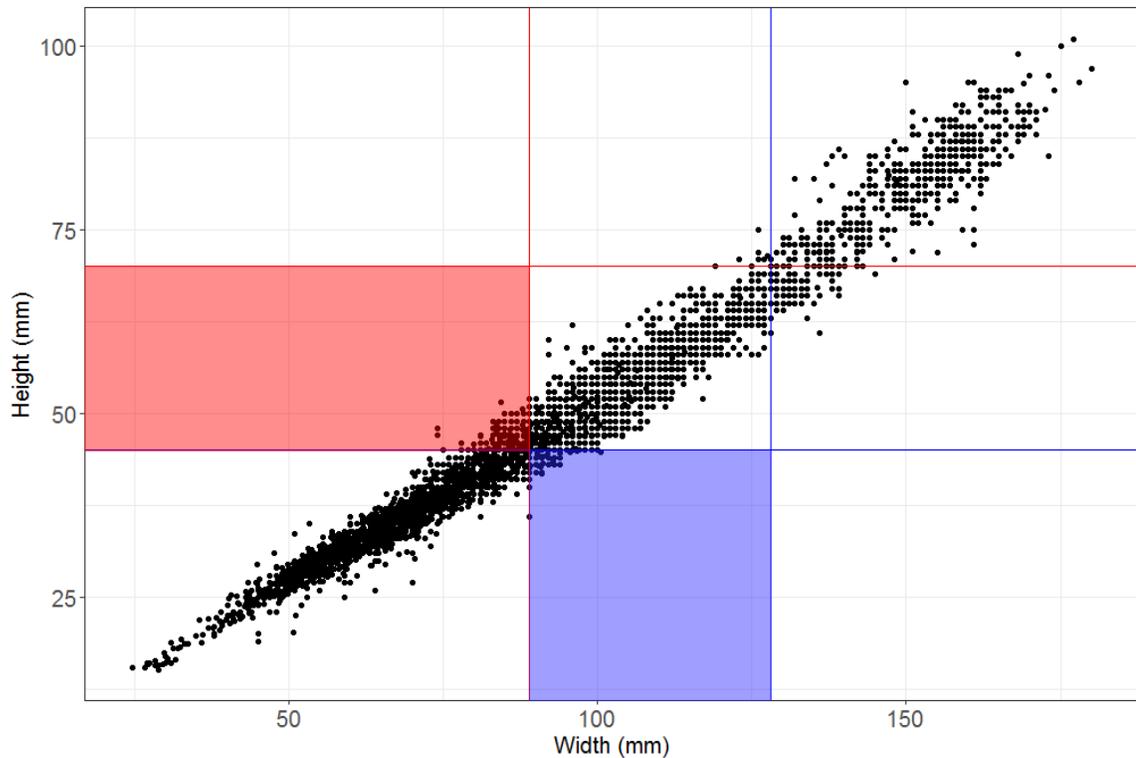


Figure 10. Dimensions of all terrapins captured at Poplar Island compared to dimensions of BRDs. Red lines indicate height and width of the South Carolina BRD. Blue lines indicate the height and width of the Maryland BRD. The Red square indicates region where turtles fit through South Carolina BRD but not the Maryland BRD. Blue square indicates region where terrapins fit through Maryland BRD but not South Carolina BRD.

## Discussion

We found no significant difference in size or number of legal crabs captured among crab pots with and without BRDs, which is consistent with the majority of BRD studies (Roosenburg et al. 1997; Roosenburg 2004; Rook et al. 2010; Chavez and Williard 2016, Chambers and Maerz, in press). Most BRD studies have reported little to no change in size of crabs captured in pots with BRDs, with decreases in number of legal

crabs ranging from increases of 49% to decreases of 47% (Wood 1997; Morris et al. 2011; Chambers and Maerz, in press).

Both BRDs in our study were similarly effective at reducing bycatch (CPUE 0.03) compared to pots with no BRD (CPUE 0.19). This is consistent with previous BRD studies, which all demonstrate significantly fewer terrapins in BRD pots than those without BRDs (Chambers and Maerz, in press). Most terrapin captures were female despite documented bycatch biases toward males; however, sex ratios at Poplar Island are largely skewed with about 4.5 females per male (Converse 2016). Because terrapins exhibit temperature-dependent sex determination, and Poplar Island is a new, artificial environment with little shaded nesting habitat, this bias in sex ratio could simply be a product of the environment, and not indicative of sex ratio bias due to bycatch mortality. Due to female biases in crab pot captures, as well as within the population, we believe this explanation to be more likely.

In a previously published assessment of the South Carolina BRD, differences in terrapin captures between the SC BRD, and one used in Virginia were non-significant, however legal crabs captured were significantly smaller (-2mm) in the SC BRD pots than VA BRD pots and controls (Grubbs et al. 2018). It has been suggested that statistically significant differences of 1-2 mm in crab size may not translate into economic significance for crabbers (Corso et al. 2017, Grubbs et al. 2018). Further study of the economic impacts of BRDs on watermen should be conducted if further enforcement of BRDs is to be asked of crabbers.

The capture of mature adult females was previously undocumented in crab pots. It is likely that this observation is due to the method of funnel construction. Entry funnels for this study were constructed in a way consistent with commercial crab pot construction, while previous studies used crab pots consistent with recreational crab pot specifications. Commercial crab pots commonly have entry funnels which are slightly larger than those of recreational pots, thus the capture of adult females in our control pots indicates that funnel size is a key component in terrapin bycatch, and that there are crab pots in fisheries with funnel design capable of capturing adult females. Further sampling is needed to see if these captures were anomalous or are a consistent trend.

Our results indicate that the BRD is an appropriate technology for mitigating bycatch mortality in crab pots for terrapins; however, it is likely that not all BRDs are created equal. When comparing height to width ratios of terrapins captured as part of an ongoing mark-recapture study at Poplar Island, we observed a larger number of terrapins whose ratios suggest they would be excluded via the Maryland BRD, however they would be able to fit through the South Carolina BRD. Diamondback terrapins can vary significantly with latitude, as such it is unlikely that a “one size fits all” approach to BRDs will be effective. While the BRD currently in use in Maryland appears to be a good fit for terrapins in Chesapeake Bay, it is possibly not a good fit for terrapins elsewhere, thus we encourage robust examination of terrapin height to width ratios throughout its range.

Several options exist for the implementation of BRDs in fisheries. Bait shops which commonly sell recreation crab pots, could begin selling crab pots already affixed

with BRDs appropriate for that region, thus creating an extra step for those who wish to fish without them, rather than creating an extra step for those who wish to fish *with* BRDs. Certain states such as New Jersey require crab pots to be affixed with BRDs only when within 23m of shore, which may represent a regulation watermen find easier to swallow, and could present a reasonable compromise, particularly in areas where fisheries and terrapin home ranges have a high degree of overlap (Harden and Williard 2012). The most extreme solution would be to create a federal regulation as opposed to state-by-state regulations on BRD use; however, this option, while perhaps best for terrapins, would likely be met with popular disapproval.

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