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Anthropogenic Modifications of Connectivity at the Aquatic-Terrestrial Ecotone in the Chesapeake Bay

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ABSTRACT

Connectivity at the aquatic-terrestrial ecotone is essential for maintaining the delicate balances between biotic and abiotic factors. However, humans have been modifying and disrupting this connectivity through activities such as land-use changes, shoreline development, and resource extraction for centuries. In order to assess how these modifications are affecting connectivity, we conducted two studies within the Chesapeake Bay. The first study focused on identifying socio-economic and landscape factors that affect the rates and distribution of shoreline development along Virginia's coastline while the second study examined how the spatial distribution of diamondback terrapins (*Malaclemys terrapin*) responded to human alterations to the aquatic-terrestrial ecotone of the lower Chesapeake Bay. For the shoreline development study, we mapped out changes in two forms of shoreline development (docks and shoreline armoring) from 2002 to 2009 within 83 sites placed along Virginia's coastline. Overall, we documented 1093 new docks and 53.75 km of new shoreline armoring within our study sites. For a fine-scale spatial assessment of shoreline development distribution, we also conducted occupancy surveys for both docks and shoreline armoring in 1250 sites using aerial imagery from 2009, of which, 25.9% of sites had docks, and 15.1% of sites had shoreline armoring. Model results revealed that both rates and distribution were positively affected by human development and negatively affected by large areas of marsh.

To examine the spatial distribution of diamondback terrapins, we conducted repeated occupancy surveys at 165 sites in 2012 and 2013. We modeled potential relationships between occupancy and local and spatial factors related to human modifications to the terrestrial-aquatic ecotone. Tidal salt-marsh was the most important positive predictor of diamondback terrapin occurrence, while agriculture, crab pots, armored shoreline, and low urban were important negative predictors. We also identified thresholds for the major predictive factors which indicated that diamondback terrapins have a low sensitivity to anthropogenic alterations.

Overall, our results indicate that humans have extensively developed the shoreline and are continuing to do so at high rates, and that those modifications are having detectable effects on diamondback terrapin distribution over large spatial scales. The ability to predict shoreline development growth and how it will affect connectivity throughout the Chesapeake Bay provides important information to resource managers about restoration and conservation targets

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CHAPTER 1

Examining shoreline development growth of Virginia's coastline: a socio-economic and landscape level analysis

Abstract

As sea level is rising and is predicted to continue to rise over the next century, the effects of shoreline development on the marine ecosystems must be understood. However, in order to do that, we first need to be able to spatially predict where shoreline development is most likely to occur, and what factors are driving its growth. For this study, we used aerial imagery to map changes in two forms of shoreline development (docks and shoreline armoring) from 2002 to 2009 within 83 2.5 km radius sites placed at random along Virginia's coastline. Overall, we documented 1093 new docks and 53.75 km of new shoreline armoring within our study sites. Low urban land cover, shoreline length, and housing density change were all positively correlated with shoreline development change, while marsh was negatively correlated. When applied at the county level, the counties closest to the Chesapeake Bay had the highest predicted rates of development, and predicted rates declined with decreasing longitude. For a fine-scale spatial assessment of shoreline development distribution, we also conducted presence/absence surveys for both docks and shoreline armoring in 1250 90-m radius sites using aerial imagery from 2009. After adjusting for observer bias, 25.9% of sites had docks, and 15.1% of sites had shoreline armoring. Model results revealed that housing density and agriculture were stronger predictors of docks than of shoreline

armoring, and that low urban land cover was a very strong positive predictor for both features. Marsh was also negatively correlated with shoreline development. In Virginia, shoreline armoring appears to be more closely associated with urban areas than docks, which are more influenced by local factors. As sea levels continue to rise, the need to protect shorelines is expected to increase as the amount of marsh (the only negative predictor) is expected to decrease. Therefore, it is increasingly important to develop and implement management strategies that address both ecosystem health and functioning as well as human needs.

Introduction

Humans have a long and complex relationship with coastal areas of the world. Historically humans were drawn to coastlines because of the ecosystem services these areas provided—especially the provision of raw materials, food, and commercial fish harvests. Unfortunately, concentrated human populations along coasts have diminished these services through direct and indirect alterations to natural ecosystems. Loss of coastal habitats including salt marshes, mangroves, and seagrasses from human activities has been linked to declines in fisheries and natural shoreline protection from flooding and storm events (e.g., (Worm et al. 2006, Cochard et al. 2008, Koch et al. 2009)).

Currently, about 10% of the world's populations live in the low-elevation coastal zone (McGranahan et al. 2007), and societal demands to protect human infrastructure from environmental pressures, such as wind and tide-driven shoreline erosion or encroachment of the sea, have led to a global practice of armoring shorelines, primarily using seawalls, bulkheads (i.e., vertical retaining wall made of concrete, steel, wood or plastic), or riprap revetment structures (i.e., sloped retaining wall made of loose rock, crushed concrete, or other material) (Dugan et al. 2011). The extent of coastal armoring typically increases with increasing population density and development and in many areas covers more than half of the coastline (Dugan et al. 2011). For example, armoring and coastal infrastructure dominates coastlines in the Mediterranean (EEA 1999), several subwatersheds within Chesapeake Bay, USA (Center for Coastal Resources Management 2012a), and Sydney Harbor, Australia (Chapman 2003). Besides shoreline armoring, another integral component of shoreline development is the construction of piers (also referred to as docks) to allow for a variety of recreational or commercial resource uses such as fishing or water-access. Overall, in the United States permit requests for private docks have increased in the past several decades and coastal residents generally perceive it as their right to construct a dock on their property that extends into common waters (Kelty and Bliven 2003).

Proliferation of artificial structures to protect shorelines or enhance water-access has introduced novel habitat to most coastal environments and fragmented natural habitats. Shoreline development alters estuarine landscapes by disrupting connectivity and homogenizing habitat at the land-water interface (Peterson and Lowe 2009), which has been increasingly recognized as an important conservation consideration for coastal areas (Talley et al. 2006). The documented adverse effects of shoreline development are numerous, and include preventing access to nesting sites for estuarine turtles (Roosenburg 1994), altering patterns of accretion and erosion (Ells and Murray 2012), a loss of diversity and reduction of benthic and nekton community structure and integrity (Isdell et al. In Review, Bilkovic et al. 2006, Seitz et al. 2006b, Bilkovic and Roggero

2008), and the facilitation of invasive species (Silliman and Bertness 2004). Coastal development structures have also recently been proposed as a source of coastal jellyfish blooms (Duarte et al. 2012). Additionally, studies have shown that maintaining connectivity and the abiotic environments will be essential to species survival in the future as the climate changes (Talley et al. 2006, Gorman et al. 2009, Schloss et al. 2011). As the climate changes and sea levels rise, the rate of shoreline armoring is likely to increase (Thompson et al. 2002).

While docks may seem relatively unobtrusive, they can alter near shore habitats through a variety of mechanisms. Shading effects of docks have been linked to the fragmentation or elimination of submerged aquatic vegetation (SAV) beds including *Zostera marina* and *Halodule wrightii* (Burdick and Short 1999, Shafer 1999), and vessel activity associated with docks can disturb or remove plants (Haslam 1978, Liddle and Scorgie 1980, Asplund and Cook 1997) and reduce foraging activity among wading shorebirds (McKinney et al. 2010). There is also evidence of reduced fish species abundance, diversity, and feeding and growth rates underneath docks (Duffy-Anderson and Able 2001, Scheuerell and Schindler 2004, Able and Duffy-Anderson 2006). Additionally, increasing dock densities directly resulted in concomitant decrease of marshes (Altieri et al. 2012). Dock presence significantly increased abundance and species richness of gull and terns (e.g., laughing gull [*Leucophaeus atricilla*], and common tern [*Sterna hirundo*]), had no effect on facultative marsh birds (e.g., fish crow [*Corvus ossifragus*], boat-tailed grackle [*Quiscalus major*], and red-winged blackbird [(*Agelaius phoeniceus*]), but had negative effects on obligate marsh birds (e.g., clapper rail [*Rallus longirostris*] and seaside sparrow [*Ammodramus maritimus*]) (Banning and Bowman 2009).

The Chesapeake Bay exemplifies a coastal system that has been extensively modified and is experiencing intensifying pressures as both human occupation and sea level rises. From navigation and resource extraction to recreation, humans have been interacting with and modifying the coastal areas of the Chesapeake Bay for centuries (Brooks 1893). Coastal areas contain a wide range of habitats from the highly productive salt marshes (Gedan et al. 2009b) and beaches to the highly modified urban, residential, and agricultural land. Much of the Chesapeake Bay tidal shoreline has been modified with an estimated 18% armored with engineered structures (bulkhead or riprap revetment), 32% of the riparian zone land use converted to residential or commercial development, and over 47,000 docks built (Titus et al. 2009).

Although the adverse effects of shoreline development are well-documented, efforts to forecast where shoreline development will occur, to our knowledge, have been rare. In the one known previous effort, the authors' primary focus was to assess where sea level rise would have the worst impact (Titus et al. 2009). Here, we propose to use proxies of human land use and extent of available land to assess factors influencing shoreline development. To investigate the effects of human land use on shoreline development, we used housing density, human population density, mean individual income, and land cover classes, including low urban, high urban, and agriculture. Because shoreline development is largely influenced by available shoreline, we also included emergent wetland, vegetated land cover, and shoreline length, to assess factors influencing shoreline development. Identifying estuarine and coastal shoreline reaches that will likely be developed can inform conservation/restoration targeting, resource management, and local planning decisions.

Using Virginia's coastline as a representative coastal environment, our objectives were to (1) delineate change in shoreline development from 2002 to 2009 based on analyses of aerial imagery, (2) identify primary economic, demographic, and/or land use factors driving shoreline development changes, and (3) determine areas most susceptible to shoreline development in the future. We hypothesized that housing density and/or population density will relate positively to shoreline development. We also predicted that the extent of emergent wetland relates negatively to shoreline development as wetlands are protected and managed in Virginia under the Nontidal Wetlands Act (administered by the Department of Environmental Quality) and the Tidal Wetlands Act (administered by the Virginia Marine Resources Commission) (Environmental Law Institute 2008).

Methods and Materials

The coastal areas of Virginia contain a variety of natural and human landscapes ranging from natural preserves to urban centers. Population levels within the lower Chesapeake Bay range from the sparsely populated Eastern Shore of Virginia (Virginia's portion of the Delmarva Peninsula) with fewer than 50,000 inhabitants (US Census Bureau 2012) to the densely populated Norfolk Metropolitan Statistical Area (NMSA) located around the mouth of the James River with more than 1.66 million inhabitants (Norfolk Department of Development 2012) (Figure 1). Additionally, in the Middle Atlantic Coastal Plain ecoregion, which includes the Chesapeake Bay, urban areas increased by 2.5% and forest land cover decreased by 3.3% between 1973 and 2000 (Brown et al. 2005).

All study sites were either on the Chesapeake Bay, along one of its numerous tidal tributaries, or on the seaside of Virginia's Eastern Shore (Figure 1). To assess changes in shoreline development (docks and shoreline armoring), we first delineated losses and gains of shoreline development within 2.5-km radius circles centered on the shoreline. We then identified factors influencing shoreline development using count-based regression analyses to assess shoreline development change between 2002 and 2009, and presence/absence data to assess heterogeneity in shoreline development in 2009.

Delineating areas of shoreline change

To select study sites, we randomly placed 110 points along the shoreline (shoreline data obtained from (Center for Coastal Resources Management 2011)) at least 5 km apart using the Generate Random Points tool from Hawth's Analysis Tools v. 3.27 in ArcMap 9.3.1 (ESRI 2009). Points were buffered by 2.5 km to provide a sufficient area to detect temporal changes in dock and shoreline armoring distribution. All overlapping buffers were eliminated to provide a final count of 83 non-overlapping, independent areas of shoreline for analysis (Figure 1).

We obtained armored shoreline and dock spatial locations by hand digitizing 2002 and 2009 Virginia Base Map imagery at a 1:2000 scale. Armored shoreline consisted of bulkhead or riprap revetment structures and docks refer to any structure extending from the shore to the water, including docks and piers. We selected years 2002 and 2009 to provide the greatest time span with the available aerial imagery. Imagery was unavailable before 2002 and after 2009 for the entire study area. All of the shoreline within each buffer was visually scanned for the presence of obviously armored shoreline and docks in 2002. Any shoreline that was too ambiguous or hidden by trees was not included as armored. The resulting shapefiles of armored shoreline and docks were then copied and modified to fit the 2009 imagery by adding any new shoreline or docks and deleting any that were lost. We converted shapefiles for armoring and docks to rasters where NoData equaled "0" and armoring and docks equaled either "1" or "2" for 2002 and 2009, respectively. The 2002 and 2009 rasters were then summed using Raster Calculator for both armoring and docks resulting in the following classifications: 0 = NoData, 1 = Present in 2002 but absent in 2009, 2 = Present in 2009 but absent in 2002, and 3 = Present in 2002 and 2009. We then used Zonal Statistics to calculate the total length of shoreline armoring and number of docks lost, gained, and held constant from 2002 to 2009 for each study site. All spatial analyses were performed in ArcGIS 10.0 (ESRI 2011).

Factors explaining shoreline change

We used several landscape and demographic factors, all evaluated within 2.5-km radii circles centered on study sites, to identify which best explain the heterogeneity in the change in shoreline armoring and docks between 2002 and 2009. To investigate how human land use influences shoreline development, we related demographic and economic factors to shoreline development change, such as housing and population densities, and mean individual income. We predicted that these factors, either as single or interactive terms, will relate positively to shoreline development. To assess how available land cover relates to shoreline development, we included land cover classes such as low urban, high urban, agriculture, emergent wetland, and vegetated land cover (any vegetated class that was not marsh or agriculture) in 2001. We also included total shoreline length in our analyses as more available shoreline could potentially result in increased shoreline development. We predicted that agriculture, low urban and vegetated land covers as well as shoreline length relate positively to shoreline development whereas high urban land cover, because shorelines have already been developed, and emergent wetlands, because State and Federal laws prohibit alteration of wetlands (Votteler and Muir 2002), relate negatively to shoreline development.

Housing and population density data were gathered from the 1990-2000 and 2010 Wildland Urban Interface (WUI) datasets provided publicly (Radeloff et al. 2005). We derived mean individual income from the US Census Bureau American FactFinder for all census tracts in Virginia. The resulting table was then joined to the US Census Bureau TIGER/Line Shapefile of the 2010 census tracts in ArcGIS (ESRI 2011). Including housing density as a proxy of human land use seems reasonable as (Brown et al. 2005) demonstrated this factor to be important when investigating temporal change of human land use in terrestrial ecosystems.

We obtained percent vegetation land cover in 2001, defined as any vegetated land cover class that was not marsh or agriculture from the WUI data sets [20]. The rest of our land cover data were obtained from the USGS Chesapeake Bay Program's Chesapeake Bay Watershed Land Cover Data Series from 2006. The land cover raster was reclassified into low urban, high urban, agriculture, and emergent wetland (marsh) land cover based on the following crosswalk. Low urban land cover included "Developed Open Space" and "Low Intensity Urban" where impervious surface accounted for 0-49% of the total land cover and consisted mainly of single-family housing and recreational areas. High urban land cover included "Medium Intensity Urban" and "High Intensity Urban" where impervious surface accounted for 50-100% of the total land cover and was typically associated with dense housing and commercial/industrial use. Agriculture consisted of all "Pasture Hay" and "Cultivated Crop". Agricultural land extent, particularly cropland, has been used to analyze temporal changes in human land use for the conterminous U.S. (Brown et al. 2005). Emergent wetland was kept the same from the original raster. Shoreline length was included in the analyses to correct for variation in area available for shoreline development. We calculated shoreline length based on the spatial data set obtained from (Center for Coastal Resources Management 2011).

In Virginia, local citizen wetlands boards with oversight from the Virginia Marine Resources Commission manage tidal wetlands through the implementation of the Tidal Wetlands Act (Va. Code §28.2-1300 et seq.). Under this Act, wetlands boards are given authority for making shoreline development permitting decisions. Technical guidance (developed by VIMS) was provided to the boards for each permit application reviewed. The purpose of this guidance was to assist the boards in making permit decisions that meet the intent and goals of the Tidal Wetlands Act, including no-net loss of wetlands. This guidance is not mandatory, and the final decision is left up to each individual county. The policies of the local wetlands boards can greatly vary from county to county. To address county-level differences in management, we used the percent compliance with guidance provided by VIMS which was obtained from figure 11 of (Center for Coastal Resources Management 2012b). The percent compliance for each county was joined to each site based on its nearest county.

All statistical analyses and modeling were conducted using the R statistical language v. 2.13.2 (R Development Core Team 2011). We used the same methods to analyze both dock and shoreline armoring change, and so will be referring to both as features in the remainder of this section. Rather than looking at features gained and lost separately, the two were combined to obtain a net change from 2002 to 2009 by subtracting features lost from features gained. To allow us to assess multiple count-based regression model structures, we transformed the net feature change by adding the most negative value from each dataset (3 for docks and 20 for shoreline armoring) to all sites so that the lowest amount of change was 0, rather than a negative value for those sites that lost more than they gained. To facilitate comparisons among parameter estimates, all variables were centered and scaled by standard deviation. All potential predictor variables were checked for autocorrelation using a Pearson's correlation matrix. Any variables with a correlation coefficient ≥ 0.7 were considered autocorrelated and only one of any set of correlated variables was included in the model.

To select the best model structure, we used Vuong's non-nested hypothesis test (Vuong 1989) to compare a negative-binomial, Poisson, zero-inflated negative-binomial, and zero-inflated Poisson distribution for the model. Once the appropriate model structure had been determined, a univariate model was run for each variable, and then compared to the null model. All variables that decreased the deviance and resulted in an AIC score lower than the null model were included in a global model. All possible combinations of the variables were analyzed, and the top models with a cumulative AIC weight of 0.95 were selected for model averaging to generate the final model (Burnham and Anderson 2002).

We spatially applied the final model at the county level for dock and shoreline armoring change. The county level was chosen as the most appropriate level for communicating the findings of this study to local governments. County-level predictions were made using the Raster Calculator in ArcGIS 10.0 (ESRI 2011) by spatially applying model averaged slope values to the variables at the 2.5-km extent. The output (predicted amount of shoreline development within 2.5-km of each cell) was then restricted to the shoreline (10-m resolution), and divided by the total number of cells of shoreline within 2.5-km to obtain a proportion developed within 2.5-km. Zonal Statistics (using the counties as zones) then provided the county-level proportion developed. Total shoreline development change from 2002-2009 was then estimated by multiplying the county-level proportion developed by total shoreline in the county. Average yearly rate of change was calculated by dividing the total change by 8 years.

Factors explaining shoreline development based on presence/absence data

To address the inherent ambiguity in delineating shoreline features from aerial imagery, we also conducted a presence/absence survey. A total of 1250 points were randomly placed along the shoreline and buffered by 90m. Within each buffer, the shoreline was scanned for the presence/absence (P/A) of docks and shoreline armoring. Four people independently completed the P/A surveys. To account for the uncertainty in our classifications, only those sites where all four observers agreed on the P/A classification were used in the analysis. A total of 1088 (87.0% agreement) sites for docks and 1033 sites (82.6% agreement) for shoreline armoring were classified consistently across observers. Of those, 188 (17.2%) and 183 (17.7%) sites from docks and shoreline armoring, respectively, were retained for model evaluation.

Site-specific variables were extracted from ArcGIS 10.0 (ESRI 2011) using the same methods as above. However, since the sites for this method were surveyed within 90m rather than 2.5km, multiple scales (extents of analysis) for each variable were used in the analysis. Housing density change, and proportion of agriculture, marsh, low urban, and vegetation in 2001 were extracted within buffers centered on sampling locations ranging in radii from 90 – 990 m in 90 m increments. We decided to include concentric-buffer analyses in our study as factors beyond the sampling extent (90-m radius) may influence P/A of docks and shoreline armoring and a buffer of 1 km was thought to be sufficiently large to capture local heterogeneity in predictor variables.

The data for each feature were analyzed using a generalized linear model with a logit-link (Hosmer Jr and Lemeshow 2004). As we used the count-based regression modeling approach, we centered and scaled each predictor variable by standard deviation and evaluated autocorrelations among predictor variables. We then evaluated each variable at each scale using univariate models, where the best scale for each variable was selected based on the lowest AIC score. The best scale for each variable was then compared to the null model, and those with a lower AIC score were included in a global model. All possible combinations of the variables were analyzed, and the top models with a cumulative AIC weight of 0.95 were selected for final model averaging (Burnham and Anderson 2002). The final models were then spatially applied using the raster calculator function in ArcGIS 10.0 (ESRI 2011).

The final models were then used to predict the probability of occurrence for the test sites from the dock and shoreline armoring data. To assess the predictive ability of the model, a generalized linear model using a logit-link was used to compare observed (P/A dock or shoreline armoring) to predicted values (probability of dock or shoreline armoring from final averaged models). If a spatial model is highly predictive, observed values should relate significantly positive to expected values.

Results

Delineating areas of shoreline change

Between 2002 and 2009, 1093 docks and 53.75 km of shoreline armoring were added within our study sites ($n = 83$) in the Chesapeake Bay. Shoreline development loss was only found in a few areas. Percent shoreline development loss was higher for docks (2.33%, 151 lost/6485 present in both years) than for shoreline armoring (0.49%, 1.21 km lost/247.28 km present in both years).

Factors explaining shoreline change

We identified two pairs of autocorrelated variables. Housing density change was correlated with population density change ($r = 0.828$, $p < 0.001$), and % low urban was correlated with % high urban ($r = 0.878$, $p < 0.001$). Housing density change was selected over population density change because residences were thought to more intuitively explain the change in shoreline features. This is in line with (Brown et al. 2005) who argued that population density is underestimating rural development as the U.S. population census is tied to location of residence and not where people recreate. This is important as shoreline development is associated, in part, with vacation homes whose use is not captured with population density estimates but with housing density. Percent low urban was selected over percent high urban because high urban areas (such as cities and industrial areas) are more likely saturated with developed shoreline features than low urban (such as residential and recreational areas).

The negative-binomial model structure provided the best fit for both the dock data and the shoreline armoring data, based on the Vuong non-nested hypothesis statistic (Table 1). Using a negative-binomial model structure for dock data set with shoreline length as a covariate, only the models including housing change from 2000-2010, % marsh, and % low urban performed better than the null model (Table 2). Competing, all possible combinations of those variables yielded 16 models. Using a 0.95 cumulative sum of AIC weights cutoff, the top 4 models were selected (Table 3) and accounted for >99.99% of the total weight. The final model derived from model-averaged estimates is shown in the following equation

$$Docks = e^{(2.47+0.13x_1+0.59x_2+0.02x_3-0.74x_4)}$$

where *Docks* = the predicted change in number of docks from 2002-2009, x_1 = housing change, x_2 = shoreline length, x_3 = % low urban, and x_4 = % marsh. Rates of new dock construction by county were estimated to range from 0.08-38.30 docks · yr^{-1}. Highest rates of dock construction were predicted for counties near open waters of the Chesapeake Bay, particularly the Eastern Shore of Virginia and the middle and upper peninsulas (Figure 2).

Using the negative-binomial structure for the shoreline armoring data with shoreline length as a covariate, only the models including housing change from 2000-2010, % marsh, and low urban performed better than the null model (Table 2). Using a 0.95 cumulative sum of AIC weights cutoff, the top 4 models were selected (Table 3) and accounted for 100% of the total weight. The final model derived from model-averaged estimates was

$$ShorelineHardening = e^{(4.27+0.15x_1+0.39x_2-0.23x_3+0.0008x_4-0.57x_5)}$$

where *Shoreline Armoring* = the predicted change in the number of cells of shoreline armoring from 2002 to 2009, x_1 = housing change, x_2 = shoreline length, x_3 = shoreline length^2, x_4 = low urban, x_5 = % marsh. Rates of shoreline armoring by county were estimated to range from 0.02-8.04 km · yr^{-1}. Like the docks, highest rates of shoreline armoring were predicted in counties near the open waters of the Chesapeake Bay, especially the Eastern Shore of Virginia and the middle and upper peninsulas (Figure 2).

Factors explaining shoreline change based on presence/absence data

Local land use, housing density, and marsh (90-270m) were important factors explaining the presence/absence (P/A) of both docks and shoreline armoring. The proportion of vegetated land in 2001 at a moderate scale (810m) was also important for predicting dock P/A but not shoreline armoring P/A (Table 4), while housing density was important for predicting shoreline armoring P/A at an intermediate scale (450 m). Using a 0.95 cumulative sum of AIC weights cutoff, the top 2 dock models and top 4 shoreline armoring models were selected from the all possible combinations (Table 5) and accounted for >99.9% of the total weight for docks, and >99.9% for shoreline armoring. The model-averaged final model for the probability of occurrence of docks (*p*(*Dock*)) was

$$p(Dock) = \frac{1}{\left(1 + e^{-(-1.52+0.003x_1+1.95x_2+3.79x_3-4.38x_4+0.95x_5)}\right)}$$

where x_1 = housing density$_{90\text{ m}}$, x_2 = proportion of agriculture$_{180\text{ m}}$, x_3 = proportion of low urban$_{180\text{ m}}$, x_4 = proportion of marsh$_{270\text{ m}}$, and x_5 = proportion vegetated in 2001$_{810\text{ m}}$, and for the probability of occurrence of shoreline armoring (*p*(*Armoring*)) was

$$p(Hardening) = \frac{1}{\left(1 + e^{-(-1.50 + 0.0004x_1 + 0.41x_2 + 4.10x_3 - 6.69x_4)}\right)},$$

where x_1 = housing density$_{90\ m}$, x_2 = proportion of agriculture$_{180\ m}$, x_3 = proportion of low urban$_{180\ m}$, and x_4 = proportion of marsh$_{270\ m}$. The models were then applied to the evaluation data set. Using the logistic regression to assess the predictability of the models, the predicted values for the docks were found to be significantly positively related to the observed values (β_{pred} = 6.55, SE = 1.01, z-value = 6.49, $p<0.001$) and reduced the deviance from 229.00 for the null model to 159.20. The predicted values for the shoreline armoring were found to be significantly positively related to the observed values (β_{pred} = 9.14, SE = 1.65, z-value = 5.538, $p<0.001$) and reduced the deviance from 172.71 for the null model to 114.22. When spatially applied, the models show a predicted probability of occurrence of a dock or shoreline armoring for each segment of shoreline (Figure 5). Shoreline development probability of occurrence was heterogeneously distributed throughout the study area. Overall, docks were far more widespread than armoring, though nearly all of the southern shoreline of the Potomac River was predicted to have a high probability of occurrence for both docks and armoring.

Discussion

Shoreline development growth greatly outpaced loss for both docks and shoreline armoring within our study area. This net growth is in line with our expectations on the basis of an ever increasing human population in this region (Theobald 2010). The slight difference between the higher amount of loss for docks than shoreline armoring is likely primarily due to higher susceptibility of docks to storm damage, and to a lesser degree, our inability to assess the state (dilapidated vs. good condition) of shoreline armoring from the aerial imagery. The similarities between dock and shoreline armoring growth and distribution illustrated in this study demonstrate a continued and persistent effort to develop the shoreline and to reduce the dangers of stochastic events to private property.

Dock and shoreline armoring change were positively correlated with housing density change, shoreline length, and low urban development, while negatively correlated with percent marsh. This indicates that areas most likely to see new shoreline armoring are new residential areas (increased housing density and low urban) near small creeks (greater shoreline length) with little marsh. Shoreline armoring, specifically, had a quadratic relationship with shoreline length, which was likely due to the areas of greatest shoreline length occurring in areas with extensive marsh. Since marshes are a form of shoreline protection and are protected by law, there should be less armoring in those areas. In both shoreline development change models, shoreline length was the strongest predictor of new development, followed by the change in housing density and % low urban. It stands to reason that the more shoreline in an area, the more shoreline development there can be (with the exception of large, embayed marshes). The positive relationship with housing density change and low urban provides an interesting management consideration. According to (Radeloff et al. 2010), growth near conservation areas outpaced the national average. As housing development and low urban land cover continues to expand (Theobald 2001), protected areas like marshes and wetlands will likely continue to see shoreline development related habitat degradation (Bilkovic and Roggero 2008) at their fringes. The effects of shoreline development have been observed at relatively low levels of disturbance and have been shown to affect the near-shore ecosystem from the local invertebrate community to the distribution of a wide ranging aquatic vertebrate (Isdell et al. In Review, Silliman and Bertness 2004, Bilkovic et al. 2006, Seitz et al. 2006b).

In addition to the explanatory variables, the variables that did not perform better than the null model also provide some valuable information. Mean income, vegetation in 2001, agriculture, and low urban change did not sufficiently help to explain the change in shoreline development. For mean income, this indicates that shoreline development occurs regardless of what income bracket the property owner is in. Waterfront property owners range from the lower-income crabbers and fishermen who may use their own docks for business, to the extremely wealthy that may only use their shoreline structures for recreation. Since vegetation in 2001 was not important, we can infer that new development wasn't exclusively occurring in previously undeveloped areas. If we equate agricultural with rural areas, then we can see that shoreline development was not occurring in rural or urban areas, but rather somewhere in between the two. Finally, since low urban change wasn't important, it also indicates that shoreline development wasn't occurring solely in areas of new development. By examining all of the variables together, it shows that most new shoreline development is occurring in mixed-use areas.

The top 5 counties with the most predicted change in docks are (in descending order): Northumberland, Accomack, Northampton, Lancaster, and Gloucester. Northumberland and Accomack counties changed positions from the dock predictions to the armoring predictions. All of these counties are relatively rural with a large amount of tidal shoreline. In contrast, other counties with substantial amounts of shoreline in the Norfolk MSA have much lower predicted levels. This is likely due to the shorelines in many urbanized areas already being close to saturation of shoreline development. However, many areas with large amounts of marsh buffering the upland from waves and storm surge are likely to face a greater threat as climate change continues to cause sea-level rise (Titus et al. 2009). As sea-level rises and marshes flood, the quadratic relationship seen in the shoreline armoring model with shoreline length will likely disappear. Without the marsh and its added shoreline, areas that were previously unlikely to be armored will become prime candidates for new development. Counties likely to be affected by this are Accomack, Northampton, Matthews, Gloucester (which already have high rates of development) and, Poquoson, James City County, Newport News City, Isle of Wight, Suffolk City, and Virginia Beach City (which currently have relatively low rates of development). Additionally, as discussed by (Titus et al. 2009), there is a feedback loop between sea-level rise and shoreline armoring. As sea-level rises, more shoreline will be armored. The more shoreline that is armored, the greater the impact on un-armored shoreline, thereby creating a greater need for more protection.

Although county-level compliance was not important in the models, county-level management decisions could also play an important role in determining the rates of development. We were surprised to see that the rates of development derived from our models didn't correspond with the number of linear feet of shoreline development approved in each county from 2009-2011 (Center for Coastal Resources Management 2012b). This could be due to a disparity in the amount of development approved, and the amount of development realized. Either less development is actually built than is approved, more development is built than approved, or some combination of the two.

The P/A approach allowed us to conduct a fine-scale analysis of factors affecting the distribution of shoreline development. All of the variables, except for the proportion vegetated in 2001 (docks) and housing density (shoreline armoring only), were important at local scales. This indicates that the distribution of shoreline development is primarily based on factors within a local proximity to a given segment of shoreline, rather than within the neighborhood or beyond. Housing density was important at different scales for docks (90 m) and shoreline armoring (450 m). This could be due to the differences in purpose for the two features. Docks are used for water access and recreation whereas armoring provides stabilization and erosion control. Armoring may have a stronger association with urban areas as many of the coastal cities shorelines are nearly saturated with shoreline armoring. This could be due to the higher property values associated with urban areas and therefore a higher potential cost of not protecting the shoreline. Only in the dock model did the proportion vegetated in 2001 play a role at a larger scale (810 m).

The primary result of the P/A approach is the ability to spatially predict where along the shoreline of the CB that shoreline development is likely to occur. Dock occurrence is far more prevalent than shoreline armoring, which is consistent with our rate of change observations. While both models had nearly all of the same explanatory variables, the relative strengths of each variable differed between them. Housing $\text{density}_{90\text{ m}}$ and proportion of $\text{agriculture}_{180\text{ m}}$, in particular, were much stronger positive predictors of dock occurrence than of armoring. One possible explanation for the difference is that docks are viewed as critical to provide water access whereas erosion control is only required for properties with eroding banks. The positive correlation between shoreline development and housing density, agriculture, and low urban indicates that development occurs along a gradient of mixed use areas. Since low urban is a stronger positive predictor of occurrence than agriculture in both docks and shoreline armoring, as areas become more developed, there is a greater chance of development. Agriculture is most likely a positive predictor because it is frequently converted to housing development in the eastern United States (Maizel et al. 1998, Brown et al. 2005).

Shoreline development interrupts connectivity at the terrestrial-aquatic interface (Talley et al. 2006) and has the potential to alter community composition and distribution at relatively large scales (Isdell et al. In Review, Bilkovic et al. 2006). While the model variables and strength of the effects may be unique to the Chesapeake Bay, the strategies used in this study could provide a framework for other coastal areas throughout the world. The ability to predict both change and occurrence of shoreline development will be critical to effect management in a future with elevated sea levels and storm surge risk. Continued coastal development and sea-level rise will almost guarantee further widespread development throughout the CB (Titus et al. 2009), which will only exacerbate many of the problems already prevalent in the region. However, we understand the need to protect personal property and to utilize the Bay for personal enjoyment and its resources. Adger et al. (Adger et al. 2005) stress the importance of social-ecological resilience in coastal systems, and suggests that rather than attempting to control the environment, communities focus on adaptability and sustainability of social and ecological systems. Therefore, we recommend moving away from individual property decisions to comprehensive geomorphic-based community planned shoreline protection that truly minimizes cumulative impacts and encourages the use of shore protection techniques that preserve and create wetlands.

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Figure Captions:

Figure 1. Virginia's Chesapeake Bay and shoreline development study locations. Map of the study area showing the shoreline development change study sites (circles) and their locations along the Virginia coastline. Each of the four major rivers is labeled in blue.

Figure 2. Annual rate of change for Virginia's coastal counties. The predicted annual rate of change from 2002-2009 for each county is shown for both docks (left) and shoreline armoring (right). Dock change is shown as number of docks per year, and shoreline armoring is shown as kilometers per year. The numbers in each county correspond to the county names listed in S1.

Figure 3. Predicted probability of occurrence of shoreline development. Representative areas of high (top) and low (bottom) probability of occurrence of both docks (left) and shoreline armoring (right) are shown for comparison.

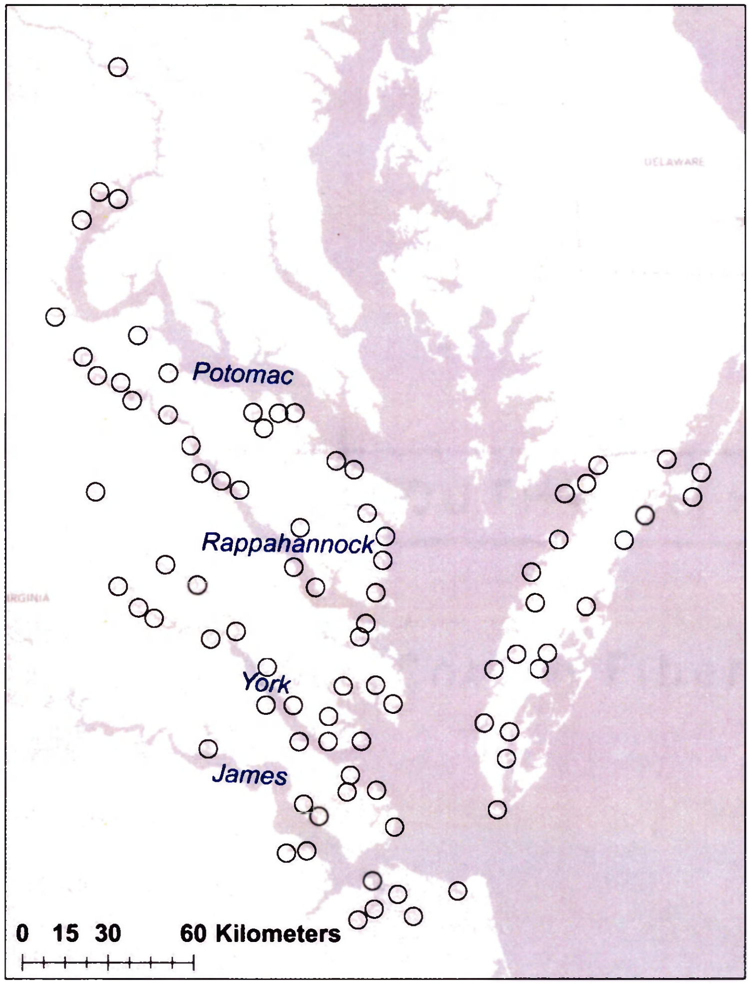

Figure 1

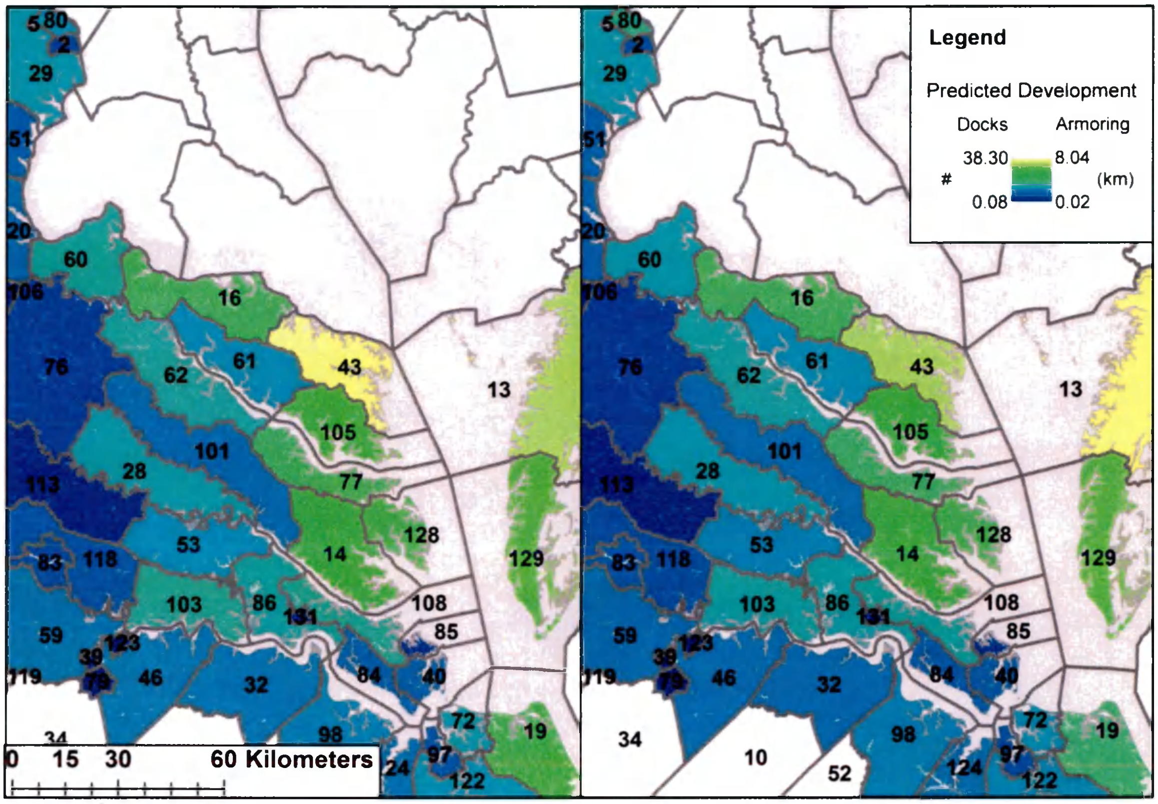
Legend
Predicted Development
Docks
Armoring
38.30
8.04
#
(km)
0.08
0.02
0 15 30 60 Kilometers

Figure 2

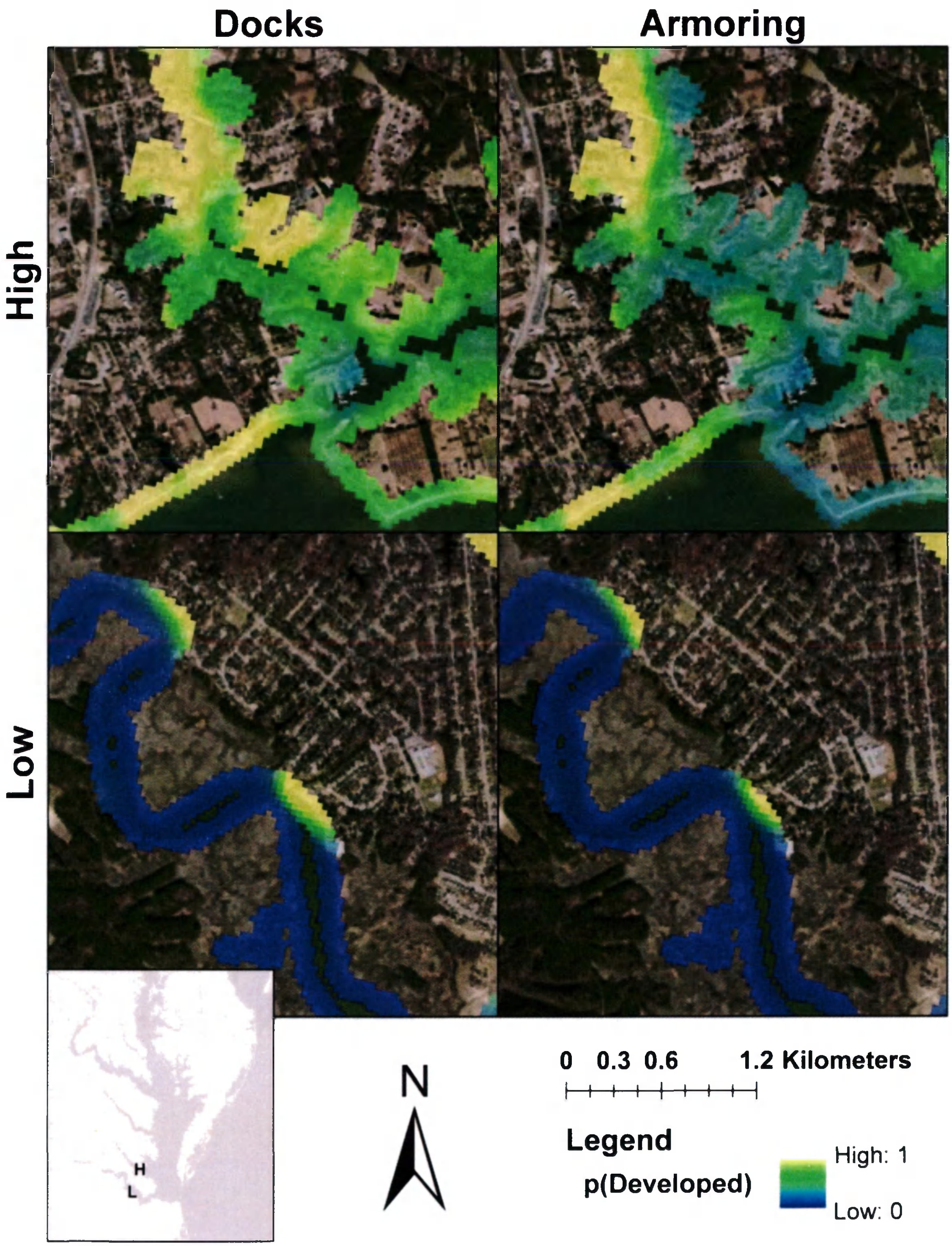
Docks
Armoring
High
Low
H
L
N
0 0.3 0.6 1.2 Kilometers
Legend
p(Developed)
High: 1
Low: 0

Figure 3

Table Headings:

Table 1. Shoreline development change univariate results. The univariate models that performed better than the null model adjusted for length ($Null_l$) based on AIC scores were selected for a global model.

Table 2. Shoreline development change top models. The top models with a cumulative sum $\geq$0.95 were selected for model averaging. Standard errors for each variable are shown in parentheses.

Table 3. Shoreline development probability of occurrence univariate results. The best scale for each variable was selected based on the lowest AIC value. If the best scale for a variable performed better than the null model, the variables were included in the global model.

Table 4 – Shoreline development probability of occurrence top models. The top models for the Presence/Absence data for both docks and shoreline armoring. These models were then averaged to provide a final model. Standard errors for each variable are shown in parentheses.

Table 1

	Docks		Armoring	
Model	AIC	AIC < Null	AIC	AIC < Null
$Null_0$	616.4237	FALSE	904.4629	FALSE
$Null_1$	615.0427	FALSE	899.7268	FALSE
Income	616.9324	FALSE	901.0970	FALSE
Housing Change	609.5250	TRUE	894.1209	TRUE
% Vegetation '01	616.4760	FALSE	901.6637	FALSE
% Agriculture	616.4777	FALSE	900.3535	FALSE
% Low Urban	607.9591	TRUE	897.9379	TRUE
% Marsh	582.0539	TRUE	881.0582	TRUE
Low Urban Change	616.8959	FALSE	901.7266	FALSE

Table 2

Structure	model	Intercept	ΔHousing	length	length^2	low urban	marsh	K	logLik	AICc	ΔAICc	weight
Docks	1	2.46 (0.085)	0.20 (0.082)	0.59 (0.12)			-0.74 (0.13)	5	-285.06	580.91	0.00	0.50
	2	2.48 (0.086)		0.61 (0.12)			-0.78 (0.13)	4	-287.02	582.57	1.66	0.22
	3	2.47 (0.085)	0.19 (0.086)	0.57 (0.12)		0.041 (0.093)	-0.72 (0.14)	6	-284.95	583.02	2.11	0.17
	4	2.48 (0.086)		0.57 (0.13)		0.083 (0.090)	-0.73 (0.14)	5	-286.59	583.97	3.06	0.11
Armoring	1	4.26 (0.088)	0.20 (0.090)	0.39 (0.12)	-0.24 (0.11)		-0.56 (0.13)	6	-433.17	879.46	0.00	0.58
	2	4.26 (0.088)	0.20 (0.094)	0.40 (0.13)	-0.24 (0.11)	-0.013 (0.10)	-0.56 (0.13)	7	-433.16	881.83	2.37	0.18
	3	4.28 (0.090)		0.39 (0.13)	-0.22 (0.11)		-0.60 (0.13)	5	-435.52	881.84	2.38	0.18
	4	4.28 (0.090)		0.37 (0.13)	-0.21 (0.11)	0.051 (0.097)	-0.58 (0.13)	6	-435.40	883.92	4.46	0.06

Table 3

	Docks			Armoring		
Variable	Best Scale	AIC	AIC > Null	Best Scale	AIC	AIC > Null
Null		1016.40			704.81	
Housing Density	90 m	936.92	TRUE	450 m	648.24	TRUE
Agriculture	180 m	986.91	TRUE	180 m	699.92	TRUE
Low Urban	180 m	894.33	TRUE	180 m	587.26	TRUE
Marsh	270 m	793.53	TRUE	270 m	566.92	TRUE
Vegetation '01	810 m	991.15	TRUE	90 m	705.58	FALSE

Table 4

Structure	Model	Intercept	Housing Density	Agriculture	Low Urban	Marsh	Vegetation '01	K	logLik	AICc	ΔAICc	weight
Docks	1	-1.66 (0.36)	0.0026 (0.0009)	2.04 (0.49)	3.92 (0.68)	-4.27 (0.76)	1.17 (0.54)	6	-353.14	718.37	0	0.81
	2	-0.95 (0.16)	0.0024 (0.0009)	1.57 (0.44)	3.25 (0.59)	-4.82 (0.71)		5	-355.59	721.25	2.88	0.19
Armoring	1	-1.56 (0.18)		0.78 (0.54)	4.40 (0.58)	-6.56 (1.37)		4	-246.82	501.69	0	0.27
	2	-1.42 (0.15)			4.18 (0.55)	-6.83 (1.37)		3	-247.84	501.71	0.02	0.27
	3	-1.59 (0.19)	0.0009 (0.0007)	0.82 (0.54)	3.98 (0.66)	-6.56 (1.38)		5	-245.93	501.93	0.24	0.24
	4	-1.44 (0.15)	0.0009 (0.0007)		3.77 (0.64)	-6.84 (1.37)		4	-247.05	502.15	0.46	0.22

CHAPTER 2

Assessment of Landscape-Seascape Connectivity in a Developed Estuary

ABSTRACT

Connectivity at the terrestrial-aquatic ecotone is critical for ecosystem health and functioning. Humans have a long history of modifying this ecotone around the world through activities such as land use changes, shoreline development, and resource extraction. In order to assess the effects of human alterations along terrestrial-aquatic ecotone, we studied the distribution of a small, emydid turtle, the diamondback terrapin (*Malaclemys terrapin*) in the lower Chesapeake Bay, USA. We conducted repeated occupancy surveys at 165 sites from late spring to mid-summer in 2012 and 2013. We used an occupancy modeling approach to evaluate potential relationships between the occurrence data and local and spatial factors related to human modifications to the terrestrial-aquatic ecotone. Diamondback terrapin distribution was affected by features from the home-range scale (≥ 750 m) down to the local scale (270 m). Total area of tidal salt-marsh was the most important positive predictor of diamondback terrapin occurrence, while agriculture land area, the abundance of crab pots, the proportion of armored shoreline, and rural development were all negatively correlated. Thresholds for the major predictive factors indicated that diamondback terrapins require a minimum of 17.6 ha of marsh at the 750-m scale, and no more than 17% armored shoreline at the 1-km scale, 15.4 ha of agriculture at the 500-m scale, or 33% low density urban land use at the 270-m scale. Our study builds upon an increasing body of evidence that estuarine

function is influenced both by terrestrial human land use as well as by human modifications to the aquatic ecosystem. Due to similarities between diamondback terrapin responses and other important species in the Chesapeake Bay, we suggest that the thresholds identified in this study be used to develop management approaches throughout the region.

INTRODUCTION

The spatial interaction of ecological processes is critical to ecosystem function and must be considered to provide accurate assessments of ecosystem health (Jules and Shahani 2003, Wiens 2006). While the majority of research has focused on the connectivity of ecological processes within separate terrestrial or aquatic systems, the connectivity of the terrestrial-aquatic interface is poorly understood (Talley et al. 2006). Terrestrial-aquatic connectivity includes physical (Ells and Murray 2012), biogeochemical (Carpenter et al. 1999), and biological interactions (Nakano et al. 1999, Nakano and Murakami 2001, Cristol et al. 2008). Because ecological processes at the terrestrial-aquatic ecotone are often dependent upon the processes and characteristics of the bordering aquatic and terrestrial systems, any disturbance in either system has the potential to disrupt connectivity. Fragmentation due to anthropogenic activities is the major source of disruption to connectivity (Rizkalla and Swihart 2006). Nowhere, perhaps, is this more evident than throughout the largest estuary in the United States, the Chesapeake Bay (CB). For centuries, humans have modified both terrestrial and aquatic systems within this region by extracting resources for sustenance, defense, and socioeconomic gain (Brooks 1893, Bradley 2011). Today, the connectivity of ecological process within CB is severely disrupted.

Estuaries are among the most productive ecosystems on the planet (Lieth 1972), and the CB is no exception (Kemp et al. 2005). In addition to numerous and critical ecosystem services (Barbier et al. 2010), the CB is home to a wide diversity of species, many of which are commercially important (Boesch and Turner 1984). Currently, few studies have examined the connectivity of terrestrial and aquatic systems along the terrestrial-aquatic ecotone within the CB. Therefore, to increase our understanding of the interaction between terrestrial and aquatic ecosystems and how human stressors influence ecological processes along this ecotone, we investigated the spatial distribution of the diamondback terrapin (*Malaclemys terrapin*) throughout the lower CB. We chose the diamondback terrapin because the uniquely estuarine reptile feeds, matures, and mates in brackish waters, but like all other reptiles, it must lay its eggs on dry land (Brennessel 2006). Species habitat requirements tie terrapins to both terrestrial and aquatic ecosystems. Human activities can disrupt the connection between these ecosystems by altering the shoreline and near-shore areas. Shoreline modification, such as shoreline armoring, is prolific throughout much of the CB (Isdell et al. In Review), and will prevent terrapins from crossing onto land to nest (Roosenburg 1994, Butler et al. 2006). Near-shore terrestrial development increases the abundance of synanthropic terrapin nest predators, such as raccoons (*Procyon lotor*) and crows (*Corvus* spp) (Hart and Lee 2006, Ernst and Lovich 2009). Terrapins also face the threat of drowning in crab-pots that have been placed within their aquatic home ranges (Roosenburg et al. 1997, Rook et al. 2010). Ultimately, diamondback terrapins rely on tidal salt marshes for both food and refuge (Brennessel 2006), and transport nutrients to and from the marine environment. Therefore, factors that negatively affect terrapin occupancy would also represent breaks in estuarine connectivity.

Our objectives for this study were to use occupancy modeling to (1) determine the distribution of diamondback terrapins throughout the lower CB, (2) assess which habitat and human variables affected their distribution, (3) examine possible linkages between terrapin distribution and ecosystem functioning, and (4) identify management targets for restoration and conservation in Virginia. We hypothesized that diamondback terrapins would be found in proximity to their primary habitat (tidal salt-marsh) and would be negatively affected by human stressors such as crabbing, shoreline armoring, and near shore development.

METHODS

Study area

The Chesapeake Bay (CB) is the largest estuary in the United States and is located between the eastern and western shores of Maryland and Virginia. Virginia's portion of CB consists of approximately 15,000 km of shoreline (NOAA National Geophysical Data Center 2013) exposed to a wide variety of land uses ranging from federally managed wildlife refuges to agriculture, residential, commercial, and industrial. An estimated 18% of the shoreline has been armored in some way to prevent erosion, 32% of the riparian zone land use has been converted to residential or commercial development, and approximately 47,000 docks have been constructed (Titus et al. 2009, CCRM 2011). Commercial crabbing in the near shore waters is extensive throughout CB, with over 385,000 pots permitted in 2013. However, it is unlikely that all of those pots will be fished. Annually, approximately 20% of pots are lost (Havens et al. 2008), meaning that a conservative estimate of 50,000 derelict crab pots can be added to the CB each year. Synanthropic predators (predators that are symbionts of human land use (Johnston 2001)) such as raccoons and crows are the major nest predators for diamondback terrapins in Virginia's CB (Ruzicka 2007) as well as in Maryland (Roosenburg 1990). Statewide diamondback terrapin population estimates are currently unavailable in Virginia.

Survey Point Selection

We sampled diamondback terrapin occupancy across gradients of variables known to influence occupancy either positively (marsh (Roosenburg 1990)) or negatively (shoreline armoring [Roosenburg 1990] and crabbing pressure [Rook et al. 2010]). Each variable was obtained from spatial data sets provided by the Center for Coastal Resources Management (CCRM) at the Virginia Institute of Marine Science (CCRM 2011) and reclassified into three levels (none, low, and high) based on an equal areas approach in ArcGIS 10.0 (ESRI 2011). We then combined the three variables and three classifications (none, low, and high) into a spatial 3x3x3 matrix of habitat and disturbance gradients. All variables were assessed within a 270-m neighborhood using the Neighborhood Statistics tool in ArcGIS. We selected 270 m to provide a fine-scale, sub-homerange (Spivey 1998) assessment of diamondback terrapin habitat requirements.

Shoreline armoring was included as percent of available shoreline within a 270-m neighborhood that was armored with bulkhead (i.e., vertical retaining wall made of concrete, steel, wood, or plastic), rip-rap revetment (i.e., sloped retaining wall made of loose rock, crushed concrete, or other material), and/or seawall (all were lumped together as "armored"). The area-wide percentages were then reclassified at 0% armored (none; cell value = 100), 1-32% armored (low; cell value = 200), and ≥ 33% armored (high; cell value = 300). Marsh was included using a two-stage method. First, marsh was coded as present or absent within a 270-m neighborhood. Then, any of the cells that had marsh were separated into cells with marsh but no beach present within 2 km, and cells with marsh and beach present within 2-km. Using these classes, the variable was reclassified as absent = 10, present = 20, and present with beach = 30. Finally, crabbing pressure was included using the number of derelict pots within 270-m as a proxy for historic crabbing pressure in an area. The equal area approach provided cutoffs at 0 pots (none = 1), 1-2 pots (low = 2), and >2 pots (high = 3).

We then summed the three spatial data sets to produce a sampling raster with 27 possible habitat types ranging from 111 (no armoring, no marsh, and no crabbing pressure) to 333 (high armoring, marsh with beach, and high crabbing pressure). This sampling raster was then converted to a shape file and clipped to a 1-km radius buffer placed around each water access point. Access points were selected from a combination of public and private boat ramps, docks, and beaches. Access points ranged from the south-side of the Rappahannock River to Virginia Beach to the Virginia-Maryland border on the bayside of the Eastern Shore of Virginia. Within the 1-km radius buffers around each access location, up to 300 random survey points were placed within each of the 27 habitat types using the Geospatial Modeling Environment's "genrandpoint" function (Beyer 2012). We randomly placed an equal number of survey points (at least 270 m apart to ensure independence among survey points) in each of the 27 habitat types to equalize sampling effort along habitat-disturbance gradients. If site visits revealed that a random point was inaccessible, an alternate corresponding survey point was then selected. We selected a different subset of survey points in 2012 and 2013 to increase sample size.

Survey Design

Each survey point was visited three times over the course of the field season. Surveys were conducted in early May and went through the end of July (2012) and beginning of August (2013); we had to extend the field season in 2013 because of above normal rainfall during May and June. Field work was conducted every day that weather and logistics permitted. Each survey point was accessed by either kayak or canoe. We used a GPS to navigate to each survey point, at which, the boat was anchored 50m ±5m from the shoreline. Environmental and survey specific variables were then recorded prior to the start of each survey. Environmental variables were measured with a portable weather station (Kestrel 2000 Wind Meter) and included air temperature (°C), wind speed (m/s), and Beaufort index. We also noted glare ("yes" or "no"), cloud cover (0, 25, 50, 75, or 100%), precipitation ("yes" or "no"), wave height (in), date and start time of the survey. From the start of the survey, the water between the boat and the shoreline was continuously scanned for any diamondback terrapin heads. We used an 8x monocular laser-rangefinder (Zeiss Victory PRF) to identify objects and to estimate distance between observer and object. All terrapin sightings were recorded, along with the size (small or large), color (black or gray), distance, and time of sighting. Our survey period was 15 minutes with the expectation that any submerged turtle would most likely surface prior to the end of the survey. At the end of each survey, the distance to each active crab pot buoy within range-finder range was also recorded.

Spatial Variables

We selected different landscape and seascape features for their perceived potential impact on diamondback terrapin distribution. All landscape features were derived from the Southeast Gap Analysis Project (SEGAP) dataset (http://www.basic.ncsu.edu/segap/datazip/state/va/ lc_segap_va.zip). First, since diamondback terrapins are known prey of synanthropic predators, we predicted that development would have a negative effect on diamondback terrapin distribution. We reclassified all areas in the SEGAP dataset classified as "developed open space" and "low intensity developed" as low urban.

Because terrapins have also been observed nesting in agricultural fields (Roosenburg 1994, Feinberg and Burke 2003), agriculture was also included as a landscape feature. We reclassified "row crop" or "pasture/hay" land cover types in SEGAP as agriculture. We combined these two land cover types as row crops and hayfields may alternate at different points in time. We hypothesized that as the amount of suitable nesting beach in an area declined due to human development, terrapins may turn to agricultural fields as an alternate nesting substrate. Additionally, clearing land for agriculture opens up former forest edge that otherwise would not be suitable for nesting. Therefore, we predicted that agriculture may have a positive effect on diamondback terrapin distribution.

The Atlantic coast subspecies of diamondback terrapins have all been shown to rely on tidal marshes for both food and shelter (Roosenburg et al. 1999, Brennessel 2006, Butler et al. 2006). We extracted the tidal marsh land cover from the SEGAP dataset as the primary habitat variable for diamondback terrapins. Any areas in the SEGAP classified as "emergent wetland" were reclassified as marsh. We hypothesized that terrapin occupancy would be positively related to the amount of tidal marsh in an area.

Shoreline armoring was also included as a spatial variable because it prevents terrapins from successfully moving from the water to land above the tide line (Roosenburg 1990, Winters 2013). Additionally, shoreline armoring has been shown to alter community composition and structure, and to reduce the associated biodiversity in the areas surrounding it (Seitz et al. 2006a, Bilkovic and Roggero 2008). The spatial data for shoreline armoring was obtained from the CCRM Shoreline Inventory (CCRM 2012a). All sections classified as "rip-rap," "bulkead," or "seawall" were reclassified as armored, and converted from a polyline shape file to a raster of 30-m resolution. We used Focal Statistics and Raster Calculator in ArcGIS 10.0 (ESRI 2011) to calculate the proportion of armored cells for the total shoreline length in a given area. Additionally, to account for the heterogeneous distribution of armored shoreline within an area, the distance from each survey point along the shoreline to the nearest section of armoring was generated with the Cost Distance and Cost Path tools in ArcGIS 10.0.

We included derelict crab pots lost or abandoned by fishermen in our analyses because crab pots, both active and derelict, are a major threat to diamondback terrapins. Numerous studies have shown that whether baited or unbaited, terrapins enter the pots and if left unchecked for too long, drown, sometimes in massive numbers (Grosse et al. 2011, Morris et al. 2011). A Marine Debris Location and Removal Program conducted in Virginia for the past 4 years (2008-2012) has recovered more than 30,000 derelict pots using side-scan sonar (Bilkovic et al. in review). The location of each pot was recorded with a GPS and entered into a database. We used this dataset as both a way to assess the potential impact of derelict crab pots on terrapin distribution as well as a proxy for crabbing pressure prior to the beginning of our current study. We converted the point shapefile to a binary raster of 30-m resolution where any cell containing a pot received a "1" and any cell without received a "0."

Once all variables were selected, we used focal statistics to calculate intensity for each feature within scales ranging from 270-m to 2-km with intervals of ~250-m. Each feature and scale were then extracted to the survey points using Geospatial Modeling Environment's "isectpntrst" tool (Beyer 2012).

Local Variables

We assessed active crabbing pressure by counting crab pot buoys within 250-m at each survey point during each of three annual surveys. We determined the distance to each pot using an 8x monocular laser rangefinder (Zeiss Victory PRF). Because crabbing pressure varies throughout the season, we decided to use the mean number of pots per site, divided by the standard error + 1 (adding 1 to the SE prevented dividing by 0 in some cases). This provided a more comprehensive assessment of seasonal crabbing pressure as it weighted sites that were consistently crabbed with chronically high crabbing pressure higher than sites with sporadic crabbing pressure.

Model Development

We first evaluated all occupancy and detection probability variables in a univariate single-season occupancy model framework (MacKenzie et al. 2002) using package "unmarked" (Fiske et al. 2013) in R (R Development Core Team 2011). Any variable that received a lower Akaike's Information Criterion (AIC) value than the null model was selected for inclusion in a global model. For the occupancy variables, each scale was also run as a univariate to determine the "best" scale for that variable, which was selected as the scale with the lowest AIC value. If the AIC value for the best scale of a variable was lower than the null model, then that variable was included in the global model.

We checked all selected variables, including spatial scale, for autocorrelation using Pearson's correlation matrix. If two or more variables received a correlation coefficient greater than 0.7, then only one was included in the global model based on the a priori hypothesized greater potential effect of the variable.

We then placed all statistically independent variables in a global model, and all possible combinations were run (Doherty et al. 2012). We centered and scaled all of the variables using study area-wide means and standard deviations to enhance our ability to make direct comparisons of relative variable strength. Although we did account for detection probability through the inclusion of survey-specific covariates, the focus of this project was not to identify factors affecting our ability to detect diamondback terrapins. Therefore, we only report results pertaining to the occupancy portion of the model. A model selection table was generated using the "MuMIn" package (Bartoń 2013) in R, and all models whose AIC weights totaled to 0.95 of the cumulative weight were selected for model averaging (Burnham and Anderson 2002). The model averaged beta values were then used in a final model to generate predicted probability of occurrence for each survey point. We then used the predicted vs. observed values to generate a sensitivity/specificity curve in package "ROCR" (Sing et al. 2013) in R. To assess model fit, the package was also used to compute the area under the curve and to derive the cutoff value, the predicted probability of occurrence at which the model equally predicts false positives and false negatives (Metz 1978). Although Lobo et al. 2008 questioned the use of the area under the ROC curve (AUC) to accurately assess model fit, a high AUC in combination with model evaluation using an independent data set, is expected to add confidence to the predictive capabilities of a model.

Once model fit was deemed acceptable, we spatially applied the model using the Raster Calculator tool in ArcGIS to create a predictive surface map. All predictions were restricted to within 1500 m of the shoreline based on radio telemetry estimates of diamondback terrapin movement patterns (Tulipani 2013).

Evaluation of Model Predictions

To determine whether the final model effectively predicted occurrence beyond the original dataset, we used an independent dataset of presence locations collected by boat surveys during summer 2011 (Bilkovic et al. 2012). Surveys were conducted across a large portion of Virginia's middle peninsula (Figure 1) where each observed turtle (N = 174) was georeferenced with a GPS (Garmin GPSmap 60Cx). We overlaid presence locations on the predictive surface map generated by the final model and extracted predicted occupancy values. We binned the extracted and study area-wide occupancy probabilities into 10% increments (0-10%, 11-20%, etc.) and calculated the proportion of the study area in each bin. To get the expected number of observations in each bin, adjusted for probability of occurrence, we used the following equation:

$$\frac{B_i N^2 \cdot \bar{\Psi}_i}{\sum(B_i N \bar{\Psi}_i)} \qquad \textit{Eq. 1}$$

where B_i = the proportion of the study area in the ith bin, N = the sample size of the validation dataset (N=174), and $\bar{\Psi}_i$ = the median value of the predicted probability of occurrence for the ith bin.

We ran a linear regression on the 2011 observed versus modeled expected occupancy values to assess how well our model predicted the presence locations for the independent dataset. A perfect model fit would result in a slope of 1 and an intercept of 0.

Thresholds

We identified thresholds for spatial predictor variables above or below which terrapins were present using the Dose Response Calculator for ArcGIS (Hanser et al. 2011). Thresholds for spatial variables were identified by comparing the intercept between the predicted occupancy curve, evaluated across the range of values for each spatial variable in the predictive model, and the predicted occupancy cutoff value derived from the sensitivity/specificity analysis (Liu et al. 2005). The intercept point was then used to estimate the threshold for each spatial predictor variable. We fit a loess curve to the predicted probability of occurrence and its upper and lower 95% confidence intervals.

RESULTS

Survey and Modeling

Over the two years of the study, we surveyed a total of 165 sites (2012 = 85; 2013 = 80) with a mean distance between sites of 1011 m (SE = 158.8 m). Diamondback terrapins were observed at 55 of the 165 sites (33.3% naïve occupancy).

Univariate analyses revealed two detection covariates with lower AIC values than the null model, and six occupancy covariates with lower AIC values than the null model. As the detection portion of the analysis is not the focus of this study, those results will not be discussed, but we did account for them in the final model. The occupancy covariates, and their best scales, that had lower AIC values than the null model (AIC = 423.13) were marsh$_{750\text{ m}}$ (AIC = 335.20), agriculture$_{500\text{ m}}$ (AIC = 407.00), derelict crab pots$_{500\text{ m}}$ (AIC = 419.93), proportion of armored shoreline$_{1\text{ km}}$ (AIC = 377.71), low urban$_{270\text{ m}}$ (AIC = 415.85), and current crab pots (local counts, not a spatial variable; AIC = 414.41). The cost-path distance from the site to shoreline armoring had a higher AIC value than the null model (AIC = 424.59). We found no autocorrelations among variables that performed better than the null model.

A total of eight variables (including the two detection covariates—start time and precipitation) were used in the global model with all possible combinations of those variables resulting in 256 models. The top 45 models accounted for 95% of the total AIC weight, and were model averaged (Appendix 1) to give the following final occupancy model:

$$\frac{1}{1+e^{-(-3.67+2.64x_1-1.37x_2-0.90x_3-0.80x_4-0.48x_5+0.11x_6)}} \qquad \textit{Eq. 2}$$

where x_1 = proportion of marsh$_{750\ m}$, x_2 = proportion of agriculture$_{500\ m}$, x_3 = current crab pot density, x_4 = proportion armored shoreline$_{1\ km}$, x_5 = proportion low urban$_{270\ m}$, and x_6 = derelict crab pot density$_{500\ m}$. Proportion of marsh$_{750\ m}$ had the highest positive effect and proportion of agriculture$_{500\ m}$ had the largest negative effect on diamondback terrapin occurrence (Table 1). The sensitivity/specificity results indicated that the optimal cutoff value was 0.388 and yielded an area under the curve (AUC) of 0.91, indicating a robust predictive model. Model-based conditional estimates of occupancy (sites where terrapins were observed or predicted probability of occurrence was above the 0.388 cutoff) indicated 73 out of 165 sites as occupied (44.2% occupancy).

Spatial application of the model revealed heterogeneity in occupancy across the study area. High occupancy areas (those above the 0.388 cutoff) typically occurred within expansive marsh embayments. Our model delineated some of the best areas along Accomack County's bayside (Figure 2A) and along the western shore near the open waters of the CB. All of these areas are well away from urban development in the CB. High occupancy areas (areas above the cutoff) accounted for 26.4% of the total predicted area.

Evaluation of Model Predictions

We found that the model predicted well when evaluated with independent diamondback terrapin presence data. Predicted and observed values were positively correlated ($p < 0.001$). Neither the intercept ($\beta_0 = 2.59$, SE = 2.42) nor the slope ($\beta_1 = 0.85$, SE = 0.081) varied significantly ($\alpha = 0.05$) from the expected values of 0 and 1, respectively. The adjusted R^2 for the model was 0.92.

Thresholds

We ran dose-response calculations for marsh, agriculture, low urban, and proportion armored (Fig. 3). For all variables, we used the point where the 95% upper confidence limit (UCL) crossed the sensitivity/specificity cutoff as a conservative threshold. This results in lower environmental thresholds for positively related variables, and higher thresholds for negatively related variables. Marsh showed the clearest positive relationship with terrapin occupancy. Areas with < 17.6 ha of marsh at the 750-m scale were unlikely to have diamondback terrapins present (Fig. 3a). Based on this threshold, ~72% of all shoreline in the study area is unsuitable for diamondback terrapins. Additionally, the thresholds for the other variables were ≥ 17% for the % armored shoreline at the 1-km scale, ≥ 15.4 ha of agriculture at the 500-m scale, and ≥ 33% of low urban at the 270-m scale (Figures 3b, c, & d, respectively). For areas above these negative thresholds, it is statistically unlikely that terrapins would be present.

DISCUSSION

Our study identified a number of scale-dependent factors that influence the distribution of diamondback terrapins. To provide a scale of reference, current estimates of terrapin home range sizes are from ~50-250 ha (Butler 2002), or approximately 400 m – 890 m radii circular home ranges. This indicates that both % armored at the 1-km scale and the amount of marsh at the 750-m scale were important factors explaining heterogeneity of diamondback terrapin occupancy at or beyond the home range scale. Just below the home range scale are agricultural land cover and derelict crab pots. Both of these affected distribution at the 500-m scale, which provides an intermediate scale for terrapin use within a home range. Finally, low urban land cover at the 270-m scale affected local distribution—likely determining whether a section of shoreline within the terrapin's home range was unsuitable. These results are consistent with other spatial scale studies of turtles where habitat variables were most important at the home range level (Rasmussen and Litzgus 2010).

Our model identified tidal marsh as the most important predictor of diamondback terrapin occurrence. Tidal salt marshes are known to be important to Atlantic coast terrapins for foraging and shelter (Roosenburg 1990, Brennessel 2006, Butler et al. 2006). Through our threshold analysis, we were also able to identify the minimum amount of marsh that was required for a terrapin to be present within a 750-m radius circle (176.7 ha) was 17.6 ha (Figure 3a). This threshold effectively rules out narrow fringing marshes as important terrapin habitat and indicates that terrapins utilize more expansive, embayed marshes as habitat. Unfortunately, this outcome provides conservationists with a real management problem. As the climate continues to warm and sea level continues to rise, studies have shown that tidal salt marshes along the Atlantic coast will drastically decrease in size (Titus et al. 2009). A combination of an increasing slope of the land as the water moves inland and shoreline armoring will prevent the inland migration of marshes. Further, more homeowners are likely to protect their property with shoreline armoring structures, resulting in a positive feedback loop for armoring, and a negative feedback loop for marshes unable to accrete soil at a rate to match sea level rise. Although the % armored at the 1-km scale was only the 4^{th} most important variable predicting diamondback terrapins, its impact is likely to increase in the future given the continued development pressure in the Chesapeake Bay (Isdell et al. In Review).

Shoreline armoring has the obvious direct effect of preventing female diamondback terrapins from moving from the water onto land to nest (Roosenburg 1994), but also has been shown to affect near shore biotic communities. Shoreline armoring reduces nekton and benthic diversity (Seitz et al. 2006a, Bilkovic et al. 2006, Bilkovic and Roggero 2008), and estuarine habitat quality is reduced where 10-25% of the shoreline is developed (Silliman and Bertness 2004, Bilkovic et al. 2006, Bilkovic and Roggero 2008). Our threshold of 17% (Figure 3d) falls within the ranges documented by these other studies. This, in combination with its importance at the 1-km scale, indicates that while shoreline armoring may have a local effect on whether a terrapin can access the land in a given location, its primary effect on distribution is likely a result of the reduction in habitat quality at a larger spatial scale. As such, we are the first to demonstrate a link to system-level effects of shoreline armoring on diamondback terrapins.

We were surprised to see that agriculture was the strongest negative predictor of diamondback terrapin occurrence, given our hypothesis of a possible positive relationship. While we knew that synanthropic predators such as crows (American crow [*Corvus brachyrhynchos*] and the fish crow [*Corvus ossifragus*]), raccoons, and red foxes (*Vulpes vulpes*) thrive in landscapes dominated by agriculture (Johnston 2001, Graser et al. 2012), and that terrapins do occasionally nest in these areas (Roosenburg 1994), we did not predict the effect to be so strongly negative. Although no studies have been conducted to show a causal link between decreased terrapin occurrence and agriculture, we suggest the following hypotheses that need further investigation. First, synanthropic predators may have a greater impact on adult survival than expected. For example, over the past 40 years, the Bald Eagle (*Haliaeetus leucocephalus*) populations in the Chesapeake Bay have grown exponentially (Watts et al. 2008). While Bald Eagles are not typically thought of as synanthropic predators, they tend to nest in remnant tall trees along forest-agriculture or forest-rural development ecotones (Courtney Turrin pers. comm.). Numerous diamondback terrapin shells have been recovered from eagle nests along the Bay (Chambers unpl. data), indicating that they may provide a significant source of adult diamondback terrapin mortality. In addition to bald eagles, other synanthropic predator populations are experiencing a concomitant increase in response to increasing urbanization (Marzluff et al. 2001). When the adult mortality is combined with the nest and juvenile mortality from synanthropic predators, the agricultural-related mortality could be driving the negative correlation with occurrence. Second, agricultural land cover may be an ecological trap. Numerous studies have documented that terrapins nest in agricultural land cover (Roosenburg 1994, Feinberg and Burke 2003). Because many agricultural fields are plowed in spring—thereby providing open, loose soil near the beginning of the nesting season, terrapins may nest in in these areas perceiving agricultural fields as suitable nesting habitat. However, as crops begin to grow, roots (a common source of nest mortality on beaches (Brennessel 2006)) and agricultural activities (pesticides, herbicides, cultivation, etc.) may destroy nests. Agricultural land cover is an ecological trap for other species (Northrup et al. 2012, Hiron et al. 2012), and some species of turtles are known to avoid agriculture (Bodie and Semlitsch 2000, Rizkalla and Swihart 2006). A third hypothesis is that runoff from agricultural land is reducing estuarine water quality. We believe that this is unlikely since runoff from these systems often results in eutrophication, and ultimately algal blooms (Ingrid et al. 1996). Because terrapins feed on animals that rely on algae (bivalves and gastropods), this would likely result in an abundance of food (Worm and Lotze 2006) which would be beneficial to terrapins.

Low urban land cover may have many of the same effects on diamondback terrapin distribution as agriculture. Terrapins are likely subjected to higher densities of synanthropic predators, and also lay their eggs in the sub-optimal habitat found around houses. Some homeowners who we encountered during this study said that terrapins came up into their yards and nested in their flowerbeds each year, but they never observed a successful emergence (Isdell pers. obs.). In addition to similarities to agriculture, low urban areas are also known to have a higher occurrence of recreational docks and piers (Isdell et al. In Review). Several studies have demonstrated that there is a reduced diversity of nekton (Duffy-Anderson and Able 2001, Scheuerell and Schindler 2004, Able and Duffy-Anderson 2006), less submerged aquatic vegetation (Burdick and

Short 1999, Shafer 1999), and increased boat traffic associated with docks (Haslam 1978, Liddle and Scorgie 1980, Asplund and Cook 1997), all of which have the potential to negatively affect diamondback terrapins. Many waterfront property owners also deploy recreational crab pots from their docks. These pots, like the commercial pots (identical in construction), can provide a significant source of mortality to diamondback terrapins (Dorcas et al. 2007). In Virginia, state law allows each waterfront property to have 2 pots/person/household (VA §28.2-262). With more than 47,000 docks along Virginia's coastline, recreational pots may be a significant source of mortality terrapin each year. Low urban was most important at the 270-m scale, suggesting these effects can be observed on a localized scale.

In addition to the landscape features, commercial crab pots were also shown to have an impact on diamondback terrapin distribution. Numerous studies have documented that diamondback terrapins drown in crab pots, with some pots containing > 40 carcasses (Roosenburg et al. 1997, Grosse et al. 2011, Morris et al. 2011). Crab pots not only decrease terrapin populations, but as we show in this study, they also influence terrapin distribution. Modeled terrapin occupancy is likely the outcome of the cumulative effects of years of crabbing having extirpated local terrapin populations. One potentially conflicting result was the positive correlation of derelict crab pots to terrapin occurrence which had the smallest relative effect of all variables. One possible explanation for this result comes from the methods used to collect the derelict pot data. Because each record in the dataset is for a pot that was removed from the water, there may be a slight increase in habitat quality due to the removal of those potentially deadly traps. The small positive result could be a statistical anomaly, but because the models that included derelict pots did receive a lower AIC score, we have decided to leave the variable in the model and report the finding in hopes that others may follow up on the study and attempt a more in-depth assessment of how derelict pots affect terrapin distribution.

Using occupancy modeling to identify factors affecting diamondback terrapin distribution has allowed us to identify areas where terrapins are involved in aquatic-terrestrial connectivity. Through our threshold analyses, we can be reasonably certain that these areas overlap tidal salt marsh embayments. The spatial distribution of tidal salt marsh embayments is primarily affected by numerous and complex physical and biological processes, and to a lesser degree by human activity (Bertness et al. 2002). As such, much of the shoreline predicted to have a low probability of diamondback terrapin occurrence has likely never been suitable habitat for terrapins. Assessments of connectivity in areas outside of potentially suitable diamondback terrapin habitat should therefore utilize alternate methods and species. However, our approach and results provide useful information about the overall ecosystem health and functioning in tidal salt marshes. Tidal salt marshes provide economically and ecologically valuable services (Gedan et al. 2009b), including refuge, foraging, and nursery grounds for a plethora invertebrates, fishes, reptiles, birds, and mammals—many of which are of special commercial or conservation interest (Hines et al. 1987, Tupper and Able 2000, Shriver et al. 2004, Gedan et al. 2009a).

Our study builds upon an increasing body of evidence that the function of estuarine ecosystems is influenced by human modification of both terrestrial landscapes and marine seascapes. From facilitating invasives (Silliman and Bertness 2004) and synanthropic predators (Marzluff et al. 2001, Silliman and Bertness 2004, Duarte et al.

2012) to altering nutrient flows due to runoff (Kemp et al. 2005) and physical barriers (Bouchard and Bjorndal 2000), anthropogenic alterations to connectivity in the aquatic-terrestrial ecotone in the CB have had major and lasting impacts on the entire ecosystem (Kemp et al. 2005). Diamondback terrapins are clearly sensitive to disruptions in connectivity along the aquatic-terrestrial ecotone, and our work shows that terrapins respond in predictable ways. Because of their relative ease to complete, occupancy surveys for diamondback terrapins are a rapid and effective method for assessing the overall health and functioning of tidal salt marsh ecosystems within the CB. Other studies have shown that aquatic turtles respond to anthropogenic disturbance levels that are consistent with responses by several other species (Burke and Gibbons 1995, Semlitsch and Bodie 2003, King et al. 2005). Because so many other species rely on the same habitats that diamondback terrapins require (Boesch and Turner 1984, Brennessel 2006) and respond to similar disturbance levels, we suggest that the thresholds identified in this study be used in development of management targets for the entire CB.

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Table 1 – Model-averaged regression coefficients and standard errors for variables included in the top models whose cumulative AIC weights summed to 0.95 (N = 45). Variables were centered and scaled to make the β-values directly comparable.

Variable	β-value	Std. Err.
Marsh	**2.64**	**1.05**
Agriculture	-1.37	0.67
Current Pots	**-0.90**	**0.40**
% Armored	-0.80	0.49
Low Urban	**-0.48**	**0.37**
Derelict Pots	0.11	0.06

Figure. 1. The study area was located in the lower Chesapeake Bay. The location of each study site (N = 165) is shown with a black point, and each evaluation point is shown with a white triangle.

Figure. 2. Predicted probability of occurrence (Ψ) was spatially applied throughout the study area using the final averaged model. Areas of highest Ψ are shown in yellow and areas of lowest Ψ are shown in blue. A section of shoreline dominated by a high probability of occurrence (inset A) is contrasted with a section of shoreline dominated by a low probability of occurrence (inset B).

Figure. 3. Dose-response relationships for each important predictor variable is shown as the solid black line representing predicted probability of occurrence, the short-dashed lines showing the upper and lower confidence intervals, and the dotted line showing the point where the upper 95% confidence interval crosses the ROC-derived cutoff value shown as the horizontal long-dashed line. The histogram in the background indicates the proportion of the study area that falls into each bin class.

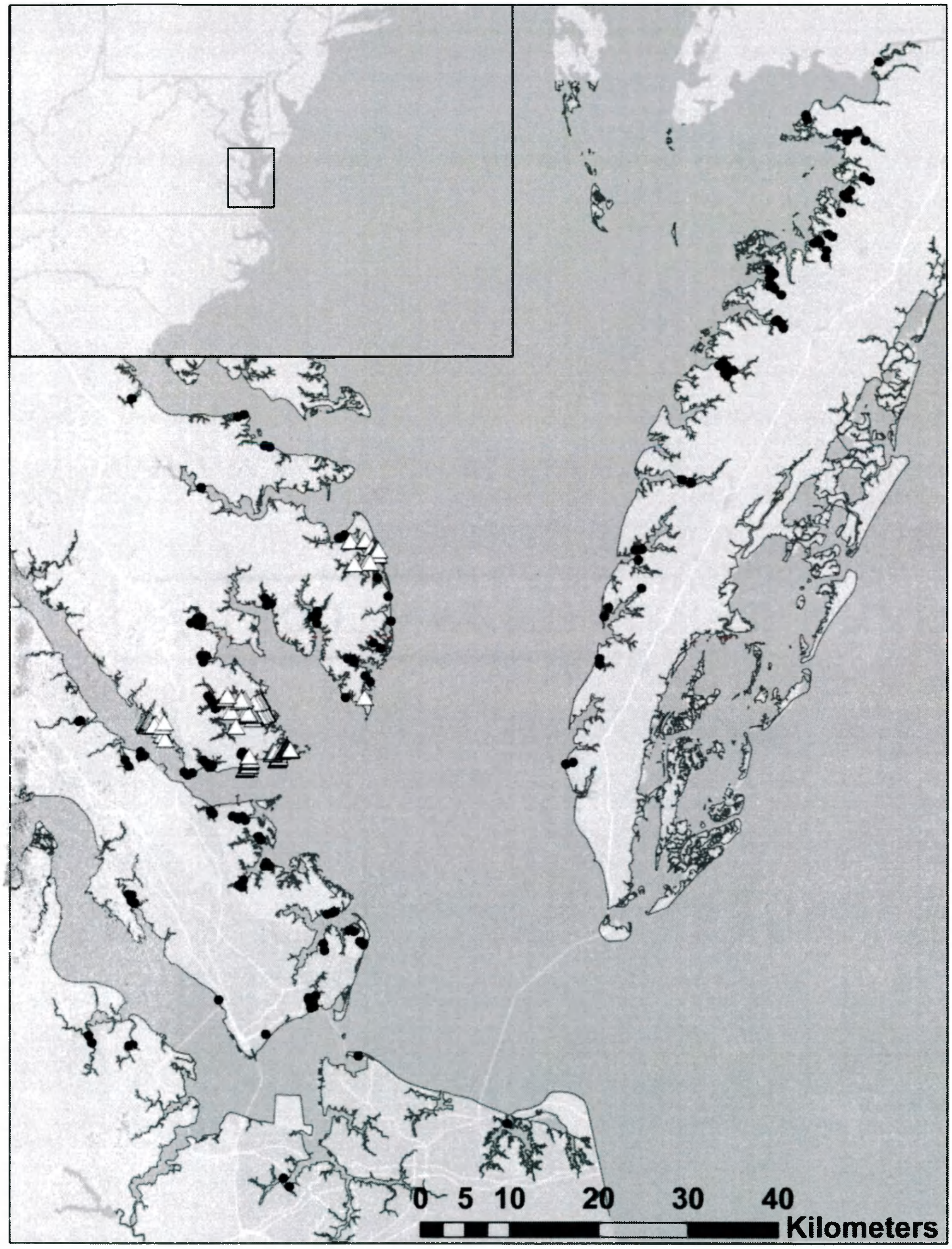

Figure 1

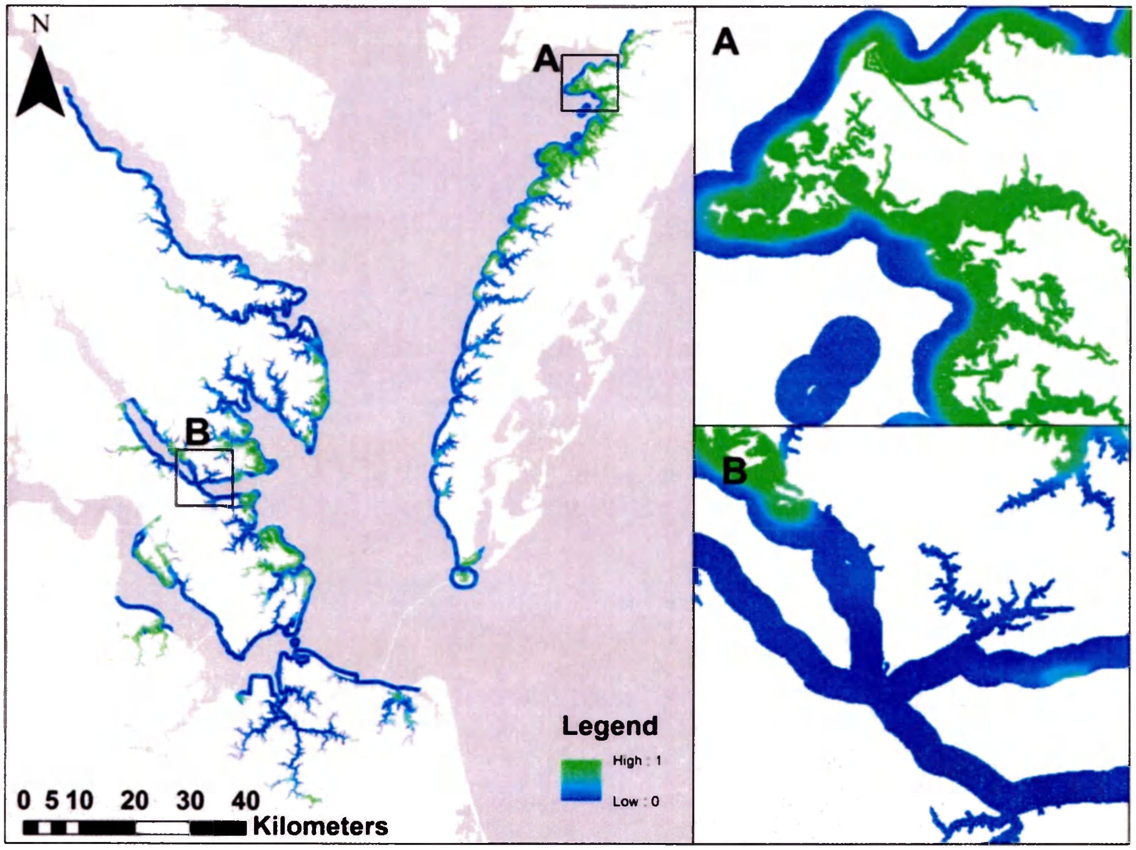

Figure 2

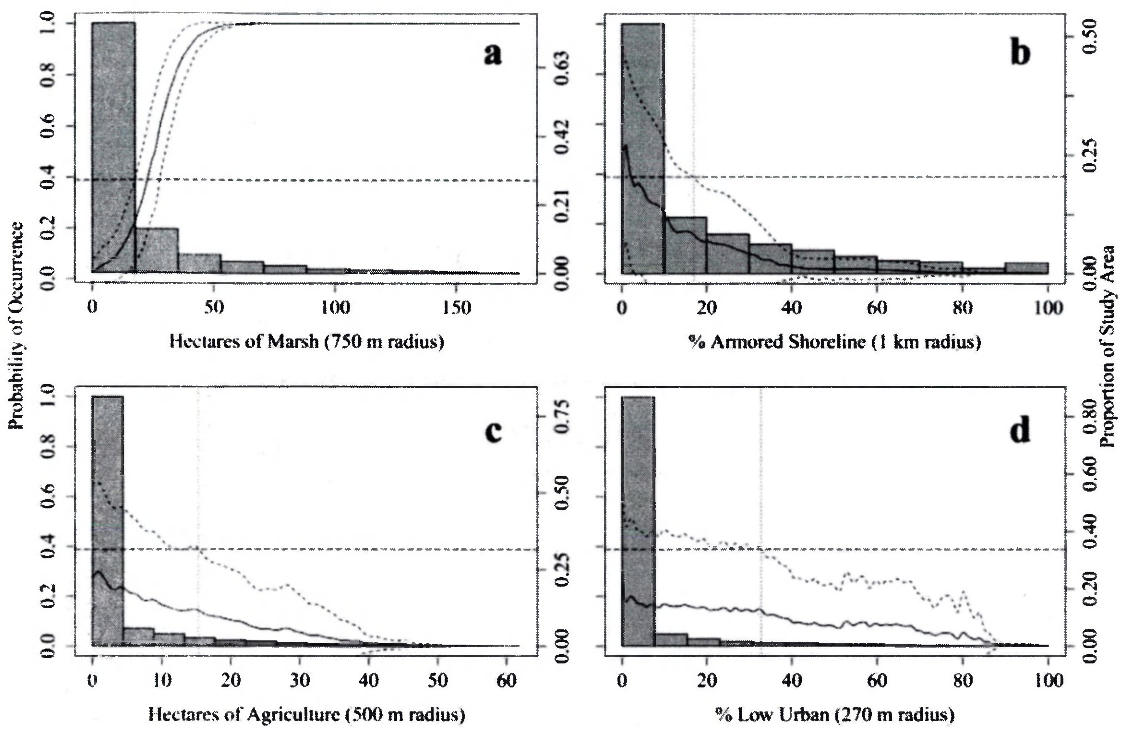
Probability of Occurrence
Proportion of Study Area
a
b
c
d
Hectares of Marsh (750 m radius)
% Armored Shoreline (1 km radius)
Hectares of Agriculture (500 m radius)
% Low Urban (270 m radius)

Figure 3