EFFECTS OF BULKHEADS ON SALT MARSH LOSS: A MULTI-DECADAL ASSESSMENT USING REMOTE SENSING

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Executive summary:

Coastal salt marshes and the ecosystem services they provide are on the decline, disappearing more rapidly than any other type of wetland in the United States. Salt marshes provide numerous ecosystem services, including storm protection, improved water quality, carbon sequestration, and critical habitat and nursery areas for commercially and recreationally important fish and shellfish species. Coastal development has risen considerably in the last several decades and has often led to shoreline hardening, whereby shoreline stabilization structures like bulkheads are used to protect against property erosion. Despite the widespread use of bulkheads and a growing body of evidence of their potential negative impacts, little is known about the effects of bulkheads on loss of salt marsh ecosystems.

To inform estuarine shoreline management, this study investigated the long-term effects of bulkheads on salt marsh loss using historic aerial imagery of Bogue, Back, and Core Sounds (Carteret County, North Carolina, USA) from 1981, 1992, 2006, and 2013. In addition to the effect of bulkhead structures, I investigated the role of wave energy on marsh loss in this system. Rates of marsh loss at landward bulkheads (i.e. bulkheads with adjacent salt marsh) were compared to 'background' rates of loss at natural marshes (i.e. non-stabilized controls). A combined wave energy index was developed to assess overall wave energy at a given site, including wind wave energy data from a previous simulation of the National Oceanic and Atmospheric Administration's Wave Exposure Model (WEMo) and distances to commercial and recreational boat channels as proxies for boat wave energy. A two-way analysis of variance was used to determine the impact of shoreline type (bulkhead vs. natural marsh) and wave energy regime (low, medium, and high) on rates of marsh loss from 1981 to 2013. Additionally, a linear mixed effects analysis was used to determine the effect of shoreline type (bulkhead vs. natural marsh), wave energy regime (low, medium, and high), date (1981 to 1992, 1992 to 2006, 2006 to 2013), and their interaction on rates of marsh loss.

The results of this work suggest that rates of marsh loss are higher at bulkheads, as these structures appear to increase outer edge erosion, and they prevent marsh gain through upland migration. Many natural marsh sites experienced upland migration but gains in marsh through this landward expansion were still insufficient to offset marsh loss from erosion of the waterward edge. Additionally, rates of marsh loss from 1981 to 2013 were not significantly different among

wave energy regimes. However, the highest rate of marsh loss occurred at landward bulkheads in high energy regimes. While not statistically significant, this observation supports the idea that the effect of wave energy on marsh loss at bulkheads may be amplified as wave energy increases because of wave reflection. My results also suggest that horizontal erosion rates of salt marsh correlate with rates of sea level rise (SLR), as the lowest marsh loss occurred during the period with the lowest rates of SLR (1992-2006), and the highest marsh loss was observed during the period with the most rapid rate of SLR (2006-2013).

The results of this study are intended to inform estuarine shoreline management. Since the assumption that bulkheads do not negatively affect public trust resources (e.g. salt marshes) is negated by this work, I provide several policy recommendations to begin leveling the playing field for bulkheads and living shorelines, including: 1) develop estuarine setbacks based on long-term erosion rates (as quantified by this study), 2) increase the price of bulkhead permits to incentivize the use of living shorelines, 3) incorporate the Living Shorelines Suitability Tool into the permitting process to help identify a site's suitability for different stabilization techniques, and 4) implement and expand educational programs to inform property owners and the coastal engineer and contractor communities about living shorelines.

This study was the first to investigate multi-decadal effects of bulkhead structures on marsh loss in the Albemarle-Pamlico Estuary and provides useful information for better understanding the effects of shoreline hardening on salt marsh ecosystems. Ultimately, guarding against property erosion should not compromise the integrity of salt marsh ecosystems and the ecosystem services they provide to coastal communities throughout North Carolina.

Introduction:

Coastal salt marshes and the ecosystem services they provide are on the decline, disappearing more rapidly than any other type of wetland in the United States (Dahl 2011). Salt marshes provide numerous ecosystem services, including storm protection, carbon sequestration, and critical habitat and nursery areas for commercially and recreationally important fish and shellfish species. Additionally, salt marshes regulate water quality and provide numerous recreational and economic benefits (Barbier et al. 2011). Coastal development and sea level rise (SLR) place tremendous pressure on salt marshes, threatening their integrity and the important ecosystem services that they provide. The need for protection against storm surges and erosion is exacerbated as coastal development increases, storms intensify, and sea level rises. Coastal development has risen considerably in the last several decades and has often led to 'shoreline armoring,' whereby shoreline stabilization structures are used to protect against property erosion. The most commonly-used stabilization structure in the United States are bulkheads, which cover approximately 14% (22,842 km) of the US coastline and constitute 73% of the hardened structures along North Carolina's estuarine shoreline (714 km) (Gittman et al. 2015, NC DCM 2012). Bulkheads are hardened vertical structures typically made of wood, vinyl, or concrete which are built at or landward of normal high water, or the ordinary extent of high tide at a location (NCNERR 2013). In instances where bulkheads are constructed behind existing salt marsh, these are termed landward bulkheads. The objective of my research was to evaluate whether landward bulkheads intensify the rate of salt marsh loss using three decades of aerial imagery from the Albemarle-Pamlico Estuary, NC, USA.

Several characteristics of bulkheads may negatively affect salt marshes and coastal habitat quality. Bulkheads cut off the upland from the intertidal and subtidal region and have been shown to negatively affect waterbird communities (Gittman et al. 2016, Prosser et al. 2017), bivalve abundance and diversity (Sietz et al. 2006), benthic infauna abundance (Gittman et al. 2016), fish community integrity and species diversity (Bilkovic and Roggero 2008), and resilience of submerged aquatic vegetation (SAV) (Landry and Golden 2017). The vertical, hardened construction of landward bulkheads reflects wave energy onto marshes, potentially leading to scour that can deepen the adjacent water and undercut the roots of marsh grasses, thereby threatening marsh vegetation (Bozek and Budick 2005, NRC 2007). Furthermore, bulkheads alter the hydrodynamics and sediment transport, which may prevent nearby marshes from maintaining surface elevation through sedimentation (Dugan et al. 2011, Fear and Currin 2012).

Salt marshes accumulate elevation and transgress, or migrate, upland in response to rising sea level. A combination of rising sea level and coastal development leads to a process known as coastal squeeze, in which salt marsh is blocked from migrating upland (Pontee 2013). Bulkheads prevent upland migration of marshes by serving as a physical barrier for expansion inland. If there is not enough sediment input for the marsh to build elevation at a rate equal to or above the rate of sea level rise, the marsh becomes inundated and cannot survive (Bozek and Burdick 2005, Pontee 2013). In addition, increasing rates of sea level rise tend to cause increasing rates of wave erosion on marsh edges (Mariotti and Fagherazzi 2013). If a marsh is being inundated, or eroding horizontally, its area will inevitably decrease if the landward margin of the marsh is not able to shift landward. Therefore, shoreline hardening may be a potential mechanism for marsh loss in combination with accelerating rates of sea level rise. This trend is projected to continue, which poses a major threat to salt marsh ecosystems (Kopp et al. 2015). Several studies have investigated relative sea level rise in North Carolina, and projections for 2100 vary from 0.38 meters to 1.4 meters (NC CRC 2010). As sea level rises and shoreline development continues, coastal squeeze threatens the survival of salt marsh ecosystems.

Wave energy may be a major driver of estuarine shoreline and salt marsh erosion (Leonardi et al. 2015, Schwimmer 2001). Wave energy models such as the National Oceanic and Atmospheric Administration's Wave Exposure Model (WEMo) are often employed to simulate wind wave energy in a given area. This model incorporates local bathymetry and exposure to prevailing wind direction and calculates representative wave energy (RWE) values for a given location (Malhotra and Fonseca 2007). WEMo focuses on wind events generating waves within 10-100 kilometers of a point, making it ideal for closed water bodies like estuaries and sounds (Currin et al. 2017). This model has previously been used to forecast the resiliency of marshes to sea level rise and wind wave energy (Cowart et al. 2010, Currin et al. 2015, Currin et al. 2017), and develop a living shoreline suitability tool to inform decision-making for shoreline stabilization (NCCOS and TNC 2017). If bulkheads intensify wave energy through wave reflection, the negative impacts of bulkheads are likely to worsen as wave energy increases.

Despite these findings and suspected impacts of bulkheads on salt marshes, policy regarding shoreline stabilization in North Carolina is based on the assumption that bulkheads do not significantly impact public trust resources including salt marshes (NC DCM 2014). The Coastal Resources Commission has classified salt marshes in this estuary as Areas of Environmental Concern, or areas designated for protection from uncontrolled development (NC DCM 2014). Recent short-term empirical field studies have not conclusively shown that bulkheads increase erosion of salt marshes (Bozek and Burdick 2005, Fear and Currin 2012), but the processes by which bulkheads lead to marsh loss may occur over decadal scales and may go undetected by such short-term studies.

Objective:

The primary objective of this study was to quantify rates of marsh loss at landward bulkhead structures and compare those rates with 'background' rates of loss quantified at natural marshes (i.e. non-stabilized controls). To do so, I used aerial imagery of Bogue, Back, and Core Sounds (Carteret County, North Carolina, USA) taken between 1981 and 2013. I hypothesized that 1) rates of salt marsh loss adjacent to bulkheads would be greater than rates of marsh loss at natural marshes and 2) the negative impact of bulkheads on rates of marsh loss would increase in areas of higher wave energy. This study quantified changes in marsh width at landward bulkheads and natural marsh controls, accounting for upland migration of marshes where present to measure both shoreline change (erosion of waterward edge) and net change in width. The role of wave energy on rates of salt marsh loss was investigated using outputs from simulations of the National Oceanic and Atmospheric Administration's Wave Exposure Model (WEMo) and distances from commercial and recreational channels as a proxy for boat wave energy. Since marsh width (i.e. structure) is directly related to marsh ecosystem services such as nutrient reduction and wave buffering capabilities, I can deduce from my results the effects of reductions in marsh structure on marsh function (Fear and Currin 2012). This research was intended to test the assumption that bulkheads do not significantly impact public trust resources such as salt marshes and, ultimately, to inform policy regarding the bulkhead permitting process in North Carolina and beyond.

Methods:

Study Area

Salt marshes are the most dominant shoreline type along the 12,000 miles of estuarine shoreline in the Albemarle-Pamlico Estuary, comprising a total of 65% of the estuarine shoreline in the second-largest estuarine system in the United States (McVerry 2012). The geographic area of this study is located at the southernmost portion of the Albemarle-Pamlico Estuary, encompassing Bogue Sound and Back Sound, as well as portions of Harlowe Creek and Core Sound in Carteret County, North Carolina (Figure 1). This area of the Albemarle-Pamlico estuarine system is mainly characterized by astronomical tides, while wind-driven tides are more prominent in the northern area of the system (APNEP 2018). This system is fed by the White Oak, Newport, and North Rivers. This area was selected due to the availability of historic aerial imagery and WEMo simulation outputs.

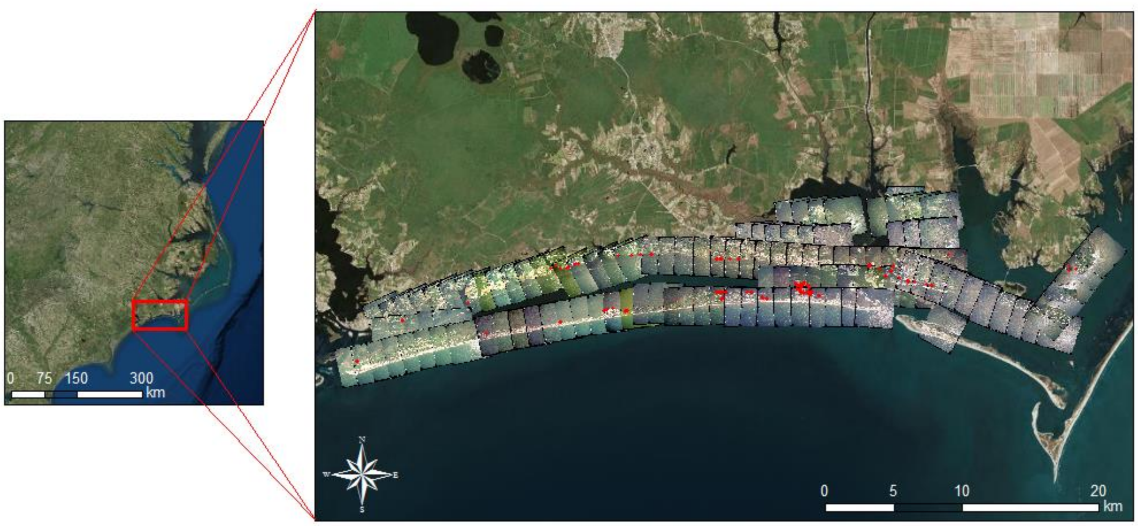

Figure 1: Inset map of the geographic study area: Carteret County, North Carolina with 1981 imagery tiles overlaid onto an ESRI satellite imagery basemap.

Aerial Imagery Acquisition & Georectification

1981 aerial imagery from the North Carolina Department of Transportation (NCDOT) Photogrammetry Unit was acquired to be used as the baseline from which change in marsh width over time was measured. I also acquired three other imagery datasets to observe changes in marsh width across time, including 1992 imagery from NC OneMap, 2006 from the Albemarle Pamlico National Estuary Partnership (APNEP), and 2013-14 imagery from NCDOT. The spatial resolution of the 2006 and 2013-14 imagery was 0.3 m*0.3 m, whereas the resolution of the 1992 imagery ranged from 1 m*1 m to 1.4 m*1.4 m. The 1992, 2006, and 2013-14 imagery datasets were previously georectified, but the 1981 aerial imagery was acquired as digital scans that required georectification. The projection of the 1992 and 2006 imagery was North American Datum (NAD) 1983 UTM Zone 18N, and the 2013-14 imagery was NAD83 North Carolina [feet U.S.]. All four aerial imagery sets were taken at low tide for the primary purpose of surveying Submerged Aquatic Vegetation (SAV), which was optimal for maintaining consistency in observing marsh boundaries.

Using the georeferencing tool in ESRI ArcGIS v. 10.3.1, I georectified the 1981 aerial imagery using the 2006 imagery as a basemap. 10 control points were selected for each image based on visual identification of coincident points between the 1981 and 2006 imagery. Control points such as road intersections and buildings were used so long as there was no apparent change in the structures between 1981 and 2006. The imagery was georectified using a second-order polynomial transformation with a nearest neighbor resampling method, and an output cell size of 0.3 x 0.3 meters. The Root Mean Square Error (RMSE) was maintained no greater than 2 meters to minimize locational error, with an average RMSE of 1.3 meters (Hapke and Henderson 2015).

Bulkhead Site Selection and Analysis

Bulkhead shapefiles from the North Carolina 2010 Estuarine Shoreline Mapping Project (ESMP) were obtained for selecting landward bulkhead sites. The ESMP shapefiles included all bulkheads on the North Carolina coast existing in 2010 (NC DCM 2012). Only bulkheads located in Carteret County were selected to ensure overlap with available aerial imagery. To select sites for this analysis, I overlaid the ESMP bulkhead shapefiles onto the rectified 1981 imagery. The criteria used to select potential sites were as follows: 1) a bulkhead was visibly present in all of the imagery sets, 2) the bulkhead was landward in 1981 (i.e. marsh seaward of the bulkhead was visible in the imagery), 3) marsh boundaries at the water’s edge were visibly apparent, 4) the patch of marsh was larger than 10 m^2, and 5) the marsh seaward of the bulkhead was not constrained by a dock parallel to the shoreline. Additionally, a bulkhead site had to be linear with a single shoreline orientation. Therefore, a single contiguous bulkhead structure could contain multiple sites due to differing shoreline orientations (e.g. Figure 2).

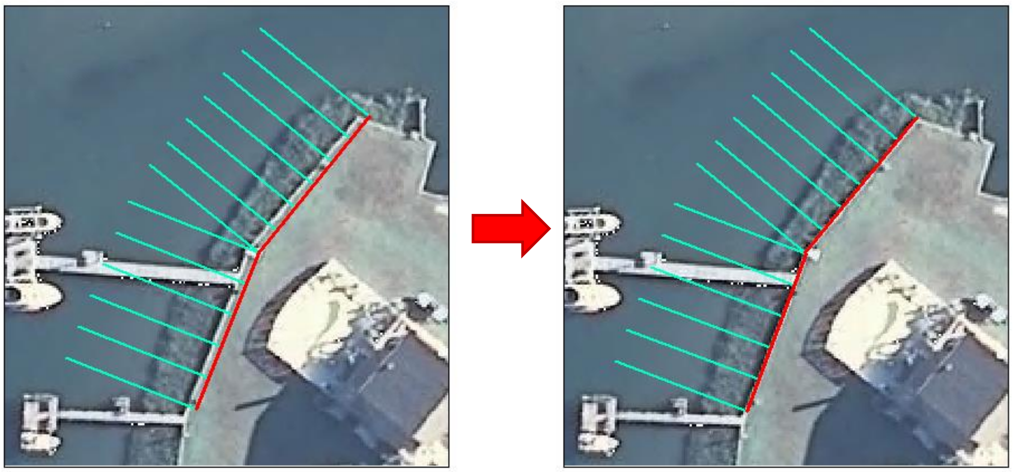

Figure 2: Right image depicts 1981 delineation of two bulkhead sites (red) and corresponding transects (blue) overlaid onto 2013 imagery. Left image shows their appropriate locations in 2013 after shifting. This single bulkhead structure was treated as two separate sites due to different shoreline orientations.

After potential sites were selected from the ESMP shapefiles, I manually digitized the bulkhead sites using the editor toolbar in ESRI ArcGIS v. 10.3.1. Bulkheads were delineated using the 1981 imagery and only landward sections of bulkheads existing in 1981 were delineated. The entire length of each bulkhead site was directly adjacent to salt marsh (landward) in 1981, which served as a baseline to measure change in marsh width in subsequent years. Bulkhead site lengths ranged from 7 meters to 77 meters.

I established equidistant 20-meter transect lines perpendicular to each bulkhead site to measure marsh width (Figure 2). The number of transects per site was proportional to the length of the bulkhead. The number of transects ranged from 3 transects to 14 at each site. These delineated transects were used to mimic typical empirical field methods used to quantify changes in marsh structure such as width and percent vegetation cover. Once delineated, I shifted the bulkhead sites and their corresponding transects in each time period after 1981 to their best approximated location using distance from nearby landmarks. This was done to minimize error due to differences in georectification methods among imagery datasets and their subsequent locational discrepancies (Figure 2).

I delineated the waterward edge of marsh at each bulkhead to calculate marsh area at each site, using the two outer transects of the bulkhead as side boundaries and the bulkhead itself as a landward boundary (Figure 3). Unusually dark or light edges of the waterward marsh boundary were not included in the area delineation and were assumed to be either sand, submerged aquatic vegetation (SAV), macroalgae, or shadows. Additionally, small patches of non-contiguous marsh outside of the marsh boundary were excluded from the digitized polygon.

Figure 3: Image of two bulkhead sites in 2013 with bulkhead delineation in red, transects in blue, and the waterward marsh edge in green.

Width measurements were taken at each transect and mean width for each bulkhead site was calculated by averaging transect widths for 1981, 1992, 2006, and 2013 (Figure 4). Transects crossing over docks were excluded from the mean width calculations. Average marsh width was also calculated by dividing marsh area by the length of the bulkhead site.

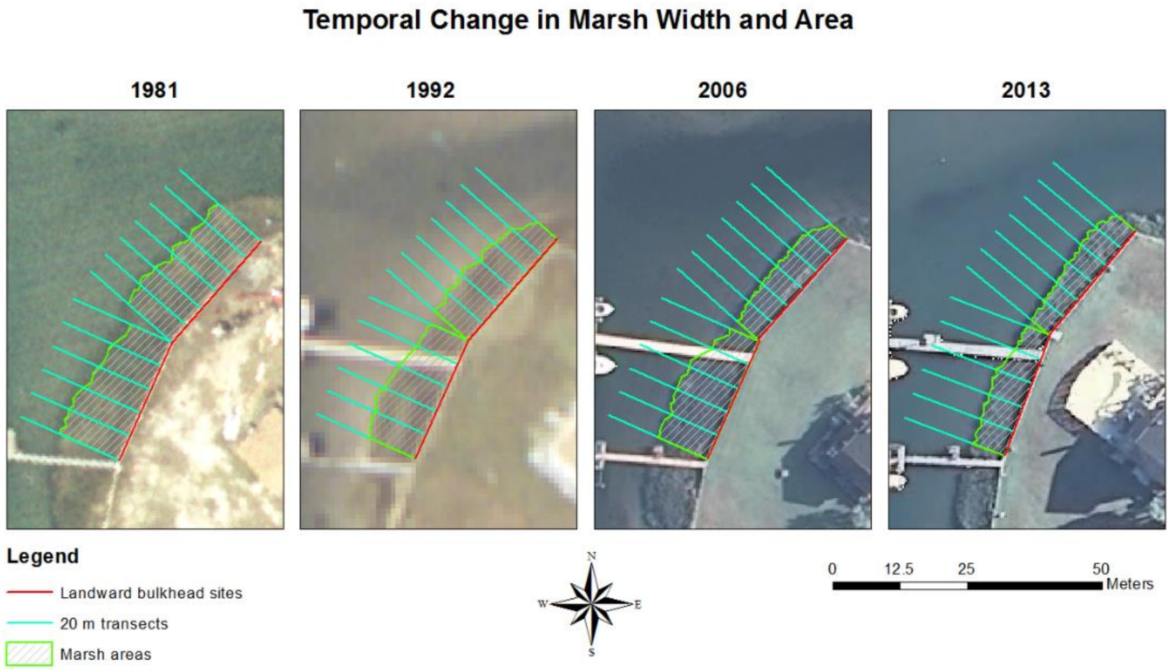

Figure 4: Time series of two bulkhead sites illustrating changes in marsh width from 1981 to 2013.

Wave Energy Index

Wind wave energy data for the study area was utilized from a previous simulation of the National Oceanic and Atmospheric Administration's Wave Exposure Model (WEMo). I used the output values from the model simulation conducted by Currin et al. (2015), in which the model was run on a grid of points spaced at 50-meter intervals 200 meters offshore along the estuarine shoreline within the North Carolina Sentinel Site (NCSS). Hourly wind data was acquired from the NOAA National Data Buoy Center station CLKN7 at Cape Lookout for the period of 2012-2015 and bathymetry data were acquired from several sources, including soundings data generated by the National Ocean Service, LIDAR from the North Carolina Flood Plain Mapping Service, and bathymetric data collected by a small boat. Each point in the grid was assigned an RWE value using the top 20% of wind data (RWE_{20}), and wind-wave energy for each shoreline point was calculated using the average value in a 75-meter radius of each point (NCCOS and TNC 2017). I used the output values of the Currin et al. (2015) WEMo simulation to assign RWE_{20} values to each bulkhead site, selecting the value from the closest shoreline point to the bulkhead site.

In addition to waves generated from wind, boat wake may also be an important component of wave energy experienced by marshes. There are no formally developed boat wave energy models, so I used distances from boat channels as proxies for boat wave energy at a given site. As the distance from boat channels increased, I assumed that boat wave energy would decrease (Currin et al. 2017). Recognizing that vessel size is proportional to boat wave energy, I used distance from commercial boat channels (i.e. large ships) and distance from recreation boat channels (i.e. personal watercraft). The use of distance as a proxy for boat wave energy is supported by evidence of increased erosion rates along boat channels (Browne 2017). To assess overall wave energy, I created a combined wave energy index accounting for wind wave energy and boat wave energy using RWE_{20} values, distance to commercial channels, and distance to recreational channels.

To assign relative weights (i.e. relative importance) of each metric to the wave energy index, I explored the relationships between the wave energy parameters (RWE_{20}, distance to commercial channels, and distance to recreational channels) and data on marsh width and area from the bulkhead sites in this study. These relationships were explored by splitting the data values into eight classes using Jenks optimization in ArcGIS v. 10.3.1, which minimizes variance within groups and maximizes variance between groups (Currrin et al. 2017). RWE_{20} was positively correlated with changes in marsh width over time (Figure 6). However, distances to commercial channels (e.g. the Intracoastal Waterway) and recreational channels were not strongly related to changes in marsh width (Figure 7). A weak positive correlation occurred with changes in marsh width and distance from recreational channels, while a weak negative correlation was observed between total change in marsh width and distance from the Intracoastal Waterway. A negative correlation between both distance values and changes in marsh width was expected, as shorter distances would likely result in higher wave energy. However, the trend observed between distance to commercial channels and RWE may be driven by an outlier (Figure 7a), obscuring the true relationship in my data.

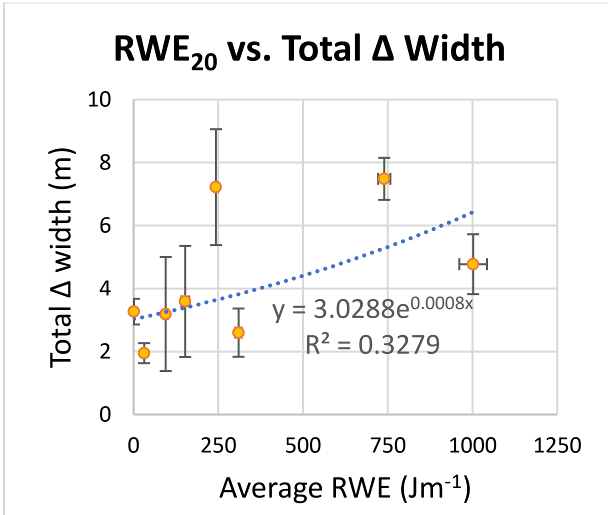

Figure 6: Average RWE_{20} values (Jm^{-1}) plotted against total change in marsh width at bulkhead sites. Data point values were broken up into five classes using Jenks Classification and the average values were calculated within each class.

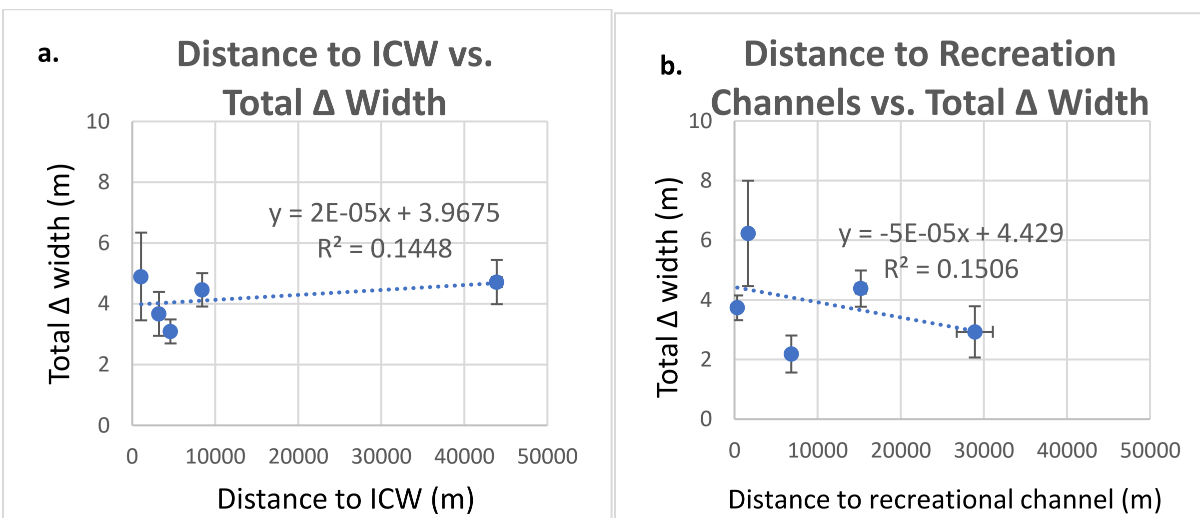

Figure 7: a) distance to the Intracoastal Waterway plotted against total change in marsh width and b) distance to recreational boat channels vs. total change in marsh width at bulkhead sites.

Considering the observed relationship between average RWE_{20} and changes in marsh width, a higher weight was applied to average RWE_{20} factor score values (10:1) in the development of the wave energy index. The RWE_{20} factor scores ranged from 0-20, whereas the distances to commercial and recreational channels were given scores ranging from 0-2 (Table 1).

Since clear trends between change in marsh width and distances to boat channels were not observed from my data, the break values from the development of the Nature Conservancy's Living Shoreline Suitability Tool were applied to this index (NCCOS and TNC 2017). Natural breaks in our data were in line with the RWE_{20} break values from this tool, while exploration of data from landward bulkhead sites did not assist in identifying natural breaks for distance values.

Table 1: Ranges of RWE_{20} values ($J\ m^{-1}$), distance to commercial channels (m), distance to recreational channels (m), and their corresponding energy factor scores.

Energy Factor Scores	
RWE_{20} values ($J\ m^{-1}$)	**Factor score**
<300	0
300-700	10
>700	20
Dist. to commercial channel (m)	
<200	2
200-500	1
>500	0
Dist. to recreational channel (m)	
<100	2
100-200	1
>200	0

Once the combined wave energy index was created, I assigned factor scores to each bulkhead site based on their RWE_{20} and distance values, and a cumulative factor score was calculated. While the maximum potential cumulative factor score was 24, the assigned scores ranged from 0 to 22. The Jenks optimization method was used to split the range of values into three energy regimes: low energy (wave energy index= 0-3) medium energy (wave energy index= 4-12), and high energy (wave energy index= 13-22). The number of bulkhead sites within each energy regime was calculated to select a similar proportion of natural marsh control sites within each energy regime.

Natural Marsh Site Selection and Analysis

I used imagery from 1981 and 2013-14 to identify natural marsh control sites within the study area that were present at the beginning of the study period and remained non-stabilized at the end of the study period. Natural marshes within close proximity to bulkhead sites used in this study were prioritized for selection. Additionally, marsh control sites near clear reference points (e.g. buildings, road intersections) were also given priority to ensure accurate locational shifts between imagery sets. A total of 50 natural marsh control sites were selected and the proportions of control sites within each wave energy regime were nearly equivalent to the proportions of bulkheads sites within each regime (Table 2). Since there were only 6 bulkhead sites in the medium energy category, I chose not to decrease this number when selecting control sites to prevent the sample size from being too small for subsequent analyses.

Table 2: Cumulative factor scores associated with each energy regime and the counts (n) and percentages of bulkhead and control sites within each regime.

Wave Energy Regimes

		Bulkhead sites		**Control sites**	
Regime	*Factor score*	*n*	*Percentage*	*n*	*Percentage*
Low	0-3	68	76.4 %	36	72%
Med	4-12	6	6.7 %	6	12%
High	13-22	15	16.9 %	8	16%

I digitized the 50 natural marsh control sites selected using the tree line landward of the natural marsh in 1981. A straight line was delineated as the site line to maintain consistency with bulkhead sites. The site line length was determined by the orientation of the tree line behind the marsh in 1981; when the tree line orientation changed, the site line was terminated. The control site lengths ranged from 17 meters to 73 meters. The same protocol as used for determining the number of transects per bulkhead site was used to determine the number of transects per natural marsh site, whereby the number of transects was proportional to the site length. For natural marsh control sites, the number of transects ranged from 5 to 14 per site.

Due to discrepancies in georectification of each imagery dataset, I shifted the location of each marsh site according to the locational shift of a chosen reference point such as a road intersection, building, or bulkhead corner (Figure 8). I assigned one reference point to each control site and the shifts in the location of this reference point in the subsequent imagery datasets were used to shift the location of the natural marsh control site lines and transects (Figure 8).

Figure 8: Left image depicts the locations of control site line, transects, and reference point (all in blue), where reference point is offset from corner of the bulkhead. Right image shows result of shifting the site line and transects simultaneously with the reference point, now located at the corner of the bulkhead used as the reference location (red line indicates location of 1981 control site line, blue line is 2013 location).

Marsh areas waterward of the control site line were delineated using the same protocol as used for bulkhead sites. Marsh width at each transect was measured and an average marsh width was calculated for each control site in 1981, 1992, 2006, and 2013. Marsh width was also calculated by dividing the waterward marsh area by the length of the site line. Measurements waterward of a static control line assess erosion of the outer marsh edge, not necessarily net loss or gain of marsh at a given site.

It was clear from the imagery that many of the natural marsh control sites experienced upland migration over the study period. This upland migration was calculated to assess net change of marsh width and area at the control sites. To measure upland migration, 10-meter transects were established on the landward side of the control site lines, directly opposite of the existing waterward transect lines. Landward marsh area, or the area between the site line and the tree line, was delineated for these twenty-one sites in 1981, 2006, and 2013 (Figure 9). Due to lower spatial resolution, the tree line was difficult to distinguish in the 1992 imagery and this imagery dataset was excluded from the upland migration analysis. Landward marsh was delineated at some of the sites in 1981 because of a jagged tree line that was not exactly parallel to the site line (Figure 9). Calculating the difference in marsh area landward of the site line in 1981 from area in subsequent years ensured a more accurate calculation of marsh migration in 2006 and 2013.

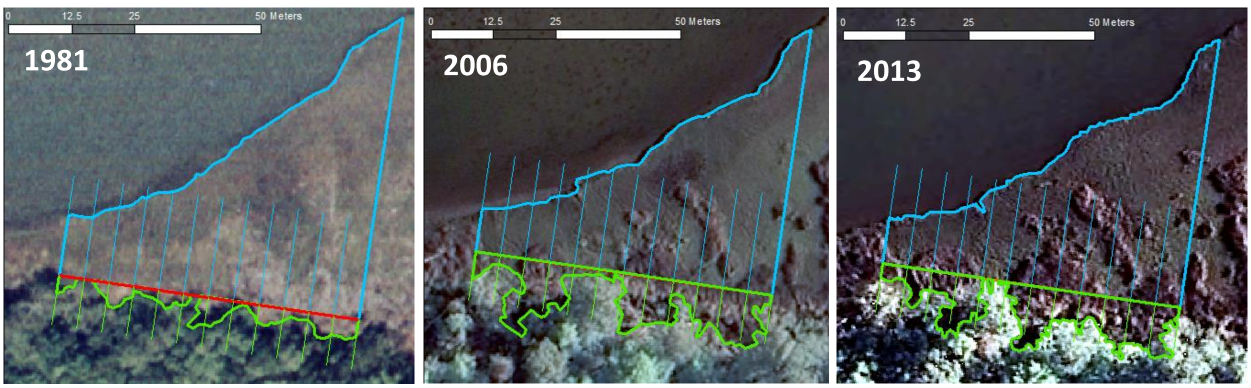

Figure 9: Waterward (blue) and landward (green) marsh areas and transects at a control site in 1981 (left), 2006 (middle) and 2013-14 (right). Controls measured from the site line waterward (blue) evaluate shoreline change, while controls accounting for migration (blue and green) evaluate overall change in marsh loss from shoreline erosion and upland migration.

Widths at each landward transect were taken and an average landward marsh width was calculated for each control site. Landward marsh width was also calculated by dividing the landward area by the length of the site line. Total area at each control site in 1981, 2006, and 2013 was calculated by summing the waterward and landward marsh areas of each the site. Upland migration was not evident at all control sites. Additionally, I calculated total marsh width of each site by averaging the sum of the waterward and landward marsh transect widths. Total marsh width was also calculated by dividing total marsh area by the length of the control site line.

Statistical Analyses

A two-way analysis of variance was used to determine the impact of shoreline type (bulkhead vs. natural marsh) and wave energy regime (low, medium, and high) on rates of marsh loss from 1981 to 2013. For significant factors in the two-way ANOVA, I ran Tukey's post-hoc tests to determine pairwise differences in factors.

A linear mixed effects analysis was used to determine the effect of shoreline type (bulkhead vs. natural marsh), wave energy regime (low, medium, and high), date (1981 to 1992, 1992 to 2006, 2006 to 2013), and their interaction on shoreline change rates. Because the 1992 imagery was excluded from the upland migration analysis, dates used to evaluate shoreline change were from 1981 to 2006 and 2006 to 2013. Site was modeled as a random effect. Visual inspection of residual plots did not reveal any obvious deviations from homoscedasticity or normality. Likelihood ratio tests were used to test for significance of fixed effects and their interactions. Statistical tests were conducted in RStudio 0.Version 1.1.442 using package (RStudio Team 2016) and lme4 (Bates, Maechler and Bolker, 2012).

Results:

The average overall rate of change in marsh width during the study period for all 139 study sites was -0.14 ± 0.02 m/yr when upland migration of natural marsh control sites was not included. When upland migration was considered, the average rate of change in marsh width for all sites was -0.12 ± 0.02 m/yr (i.e. a net loss of marsh area).

Rates of marsh loss by shoreline type

Rates of marsh loss from 1981 to 2013 varied significantly among shoreline type ($F = 7.8$, $df = 2, 180$, $p = 0.0005$; Figure 10). Rates of marsh loss adjacent to bulkheads were significantly higher than rates of marsh loss at natural marshes with fixed site lines ($p = 0.05$) and at natural marshes accounting for marsh migration ($p = 0.005$). Differences in marsh loss rates between fixed and migrating natural marshes were not significant ($p = 0.4$). However, when comparing marsh loss rates across all time intervals studied (1981-1992, 1992-2006, and 2006-2013), there was no significant difference between bulkheads and natural marshes with fixed site lines ($\chi2=1.4$, df=1 p=0.2) largely due to the high rates of marsh loss measured at natural marshes during 2006-2013 (Figure 12). This discrepancy suggests that long-term, multi-decadal assessments are required to detect impacts of bulkheads on salt marshes; shorter 2-10-year studies may be inconclusive. When accounting for migration, rates of marsh loss from 1981-2006 and 2006-2013 remained significantly higher at bulkheads than natural marshes ($\chi2=11.62$, df=2 p=0.016; Figure 10).

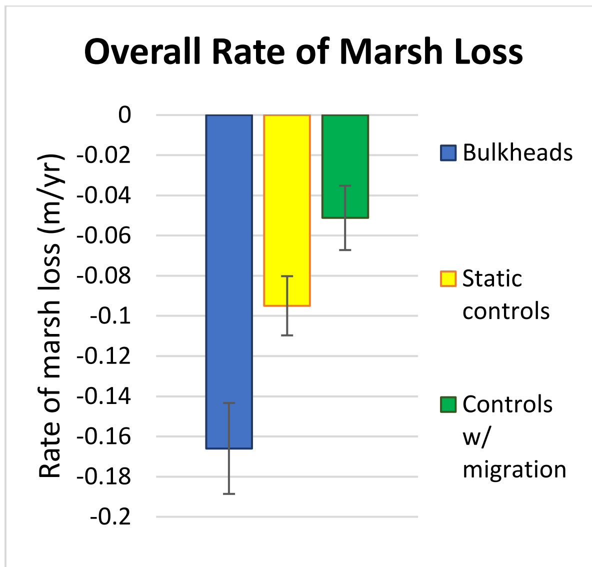

Figure 10: Overall rates of marsh loss from 1981-2013 for bulkhead sites, control sites measured from a static line, and controls accounting for upland migration. Values plotted from transect method for measuring marsh width.

Of the 89 bulkhead sites, 85 sites (95.5%) experienced marsh loss during the study period, while 4 sites (4.5%) accreted marsh. The average total change in marsh width at bulkhead sites using transect calculations was -4.74 ± 0.57 meters, with a rate of marsh loss of -0.17 ± 0.02 m/yr. The average total change in marsh width at bulkhead sites using area to calculate width was -4.84 ± 0.57 meters, with a change rate of -0.17 ± 0.02 m/yr (Table 3). The average percentage of total marsh area lost at bulkhead sites over the time period was 51.8%, and the median percentage loss was 43.7% (Table 4). Additionally, 21 (23.6%) of the bulkhead sites lost the entirety of their marsh area by 2013, 18 of which had lost all of the adjacent marsh by 2006.

Table 3: Summary of the number of sites eroding and accreting, average change in marsh width using transects (T) and area (A) and rates of marsh loss for bulkhead sites, control sites measured from a fixed line, and control sites accounting for upland migration.

	Sites eroding (T)	Sites accreting (T)	Sites eroding (A)	Sites accreting (A)	Avg. Δ marsh width- T (m)	Change rate- T (m/yr)	Avg. Δ marsh width- A (m)	Change rate- A (m/yr)
Bulkheads	85 (95.5%)	4 (4.5%)	85 (95.5%)	4 (4.5%)	-4.74 ± 0.57	-0.17 ± 0.02	-4.84 ± 0.57	-0.17 ± 0.02
Static controls	41 (82%)	9 (18%)	42 (84%)	8 (16%)	-3.04 ± 0.47	-0.09 ± 0.01	-3.16 ± 0.48	-0.1 ± 0.02
Controls w/ migration	32 (64%)	18 (36%)	34 (68%)	16 (32%)	-1.64 ± 0.51	-0.05 ± 0.02	-1.81 ± 0.52	-0.06 ± 0.02

Table 4: Mean and median percentages of marsh area lost from 1981-2013 at bulkhead sites, control sites measured from a static line, and controls accounting for upland migration.

	Mean	Median
Bulkheads	-51.8%	-43.7%
Static controls	-16%	-15.1%
Controls w/ migration	-7.7%	-6.8%

Of the 50 natural marsh control sites, 41 sites (82%) experienced marsh loss waterward of the fixed site line, and 9 sites (18%) accreted marsh waterward of the site line when transects were used to calculate width (Table 3). The average total change in waterward marsh width at control sites using transect calculations was -3.04 ± 0.47 meters, with an average overall rate of 0.09 ± 0.01 m/yr (Figure 10). The average percentage of waterward marsh area lost at control sites over the time period was 16%, and the median percentage loss was 15.1% (Table 4).

Upland migration of marsh landward of the fixed control site line was visible at 21 of the 50 natural marsh control sites. Several additional sites appeared to have also experienced migration, but shading prevented accurate quantification of migration at these sites. Control sites that experienced upland migration gained an average of 3.35 ± 0.41 meters of marsh at a rate of 0.10 ± 0.01 m/yr over the 32-year study period when using transect measurements.

When accounting for upland migration, 32 (64%) of the natural marsh sites experienced net loss of marsh when using transects to measure width, and 18 (36%) sites experience net gain. The average net change in marsh width (using transect calculations) was -1.64 ± 0.51 meters, with an average overall loss rate of -0.05 ± 0.02 m/yr (Figure 10, Table 3). The average percentage of waterward marsh area lost at control sites over the time period was 7.7%, and the median percentage loss was 6.8% (Table 4).

Rates of marsh loss by wave energy regime

Rates of marsh loss from 1981 to 2013 were not significantly different among wave energy regimes ($F = 0.78$, $df = 2, 180$, $p = 0.46$; Figure 11). Likewise, the effect of energy regime on marsh loss rate, across all time intervals studied (1981-1992, 1992-2006, and 2006-2013), was also not significant ($\chi 2=1.3$, $df=2$ $p=0.5$; Figure 12). However, the highest rate of marsh loss occurred at landward bulkheads in high energy regimes, at an average -0.2 ± 0.02 m/yr over the 32-year study period (Figure 11). Bulkhead sites in medium and low energy regimes experienced similar rates of loss, with marsh loss rates of -0.12 ± 0.03 m/yr and -0.14 ± 0.02 m/yr, respectively. In contrast, marsh loss rates were relatively constant across energy regimes for static and migrating marshes. Rates of marsh loss were consistent among energy regimes for control sites with migration, all with an average loss rate of -0.05 ± ≤0.05 m/yr. Control sites measured from a static line experienced the lowest rates of marsh loss in medium energy, at a loss rate of -0.06 ± 0.02 m/yr, while marsh loss was consistent among low and high energy at static control sites, with rates of -0.1± ≤0.04 m/yr.

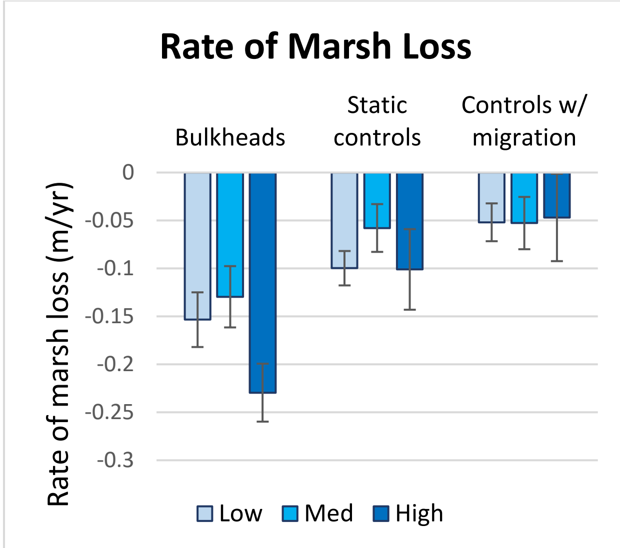

Figure 11: Bar graphs illustrating the rates of marsh loss at bulkheads, control sites from a fixed line, and control sites accounting for migration in low, medium, and high wave energy regimes.

Rates of marsh loss by time interval

Rates of marsh loss were not significantly different across time intervals studied (1981-1992, 1992-2006, and 2006-2013; ($\chi2$=2.1, df=2 p=0.1). The average rate of loss during the 1981-1992 period for all study sites (using static controls) was -0.16 ± 0.03 m/yr, -0.12 ± 0.03 m/yr from 1992-2006 and -0.15 ± 0.02 m/yr from 2006-2013. When accounting for upland migration, the average rate of loss during the 1981-2006 period for all study sites was -0.11 ± 0.02 m/yr, and -0.12 ± 0.02 m/yr for the 2006-2013 period. Rates of marsh loss at a given time interval were generally greatest at bulkheads (Figure 12), apart from the interval from 2006-2013 when rates of marsh loss were highest at natural marshes without accounting for migration.

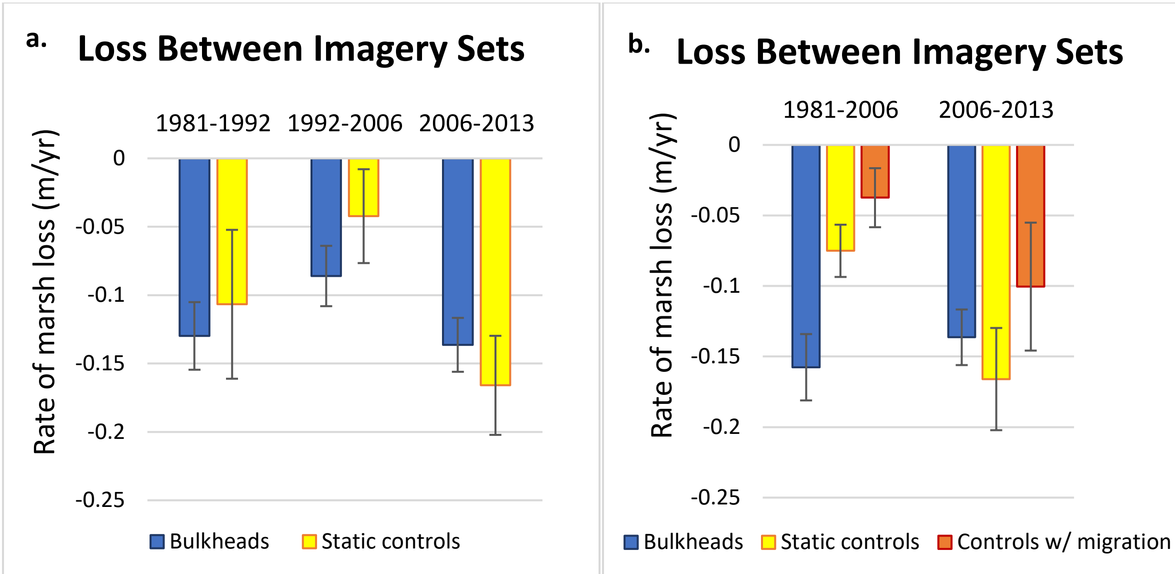

Figure 12: a) Rates of marsh loss from 1981-1992, 1992-2006, and 2006-2013 for bulkhead sites and controls without migration (static controls) and b) rates of marsh loss from 1981-2006 and 2006-2013 for bulkhead sites, controls without migration (static controls), and controls with migration.

Discussion:

Salt marshes are threatened by a multitude of factors and are declining at a rate of about 5,000 acres per year in the US (Dahl 2011). Irrespective of shoreline type, marsh loss was evident at 84% of all sites included in this study over the 32-year time period in the southern portion of the Albemarle-Pamlico Estuary. My results suggest that shoreline hardening approaches, such as bulkheads, that are frequently used to combat property erosion may increase rates of salt marsh loss by up to 300%. The prevention of upland migration by bulkheads as sea level rises is a likely mechanism of marsh loss in this region. Furthermore, it appears that the effects of wave energy reflection may be amplified by bulkheads in high energy settings (Bozek and Budick 2005, NRC 2007). To protect productive salt marsh ecosystems and the many ecosystem services they provide, estuarine shoreline management should account for potential effects of shoreline stabilization practices. Additionally, further research is needed to investigate the effects of shoreline hardening to develop a more robust body of science for invoking changes in policy that support a shift away from estuarine shoreline hardening.

Previous work investigating shoreline change rates in this geographic region have reported a range of loss rates. Currin et al. (2015) observed an average shoreline change rate (SCR) of -0.18 m/yr along marsh shorelines in the New River Estuary, which is comparable to the average shoreline change rate in this study of -0.14 ± 0.02 m/yr when assessing loss through erosion of the marsh edge.

Wave Energy

The effect of wave energy regime on rates of marsh loss from 1981 to 2013 and across all time intervals studied (1981-1992, 1992-2006, and 2006-2013) was not statistically significant. Previous studies investigating the effects of wave energy on salt marsh have reported mixed results. For instance, Cowart et al. (2011) assessed shoreline change in the New River Estuary and found no detectable relationship between erosion rates and the Relative Exposure Index wave energy metric, while Schwimmer (2001) identified a positive relationship between wave power and salt marsh erosion rates in Delaware Bay. Furthermore, research by Currin et al. (2015) identified a positive correlation between RWE and marsh loss in high wave energy areas, with an RWE >300 J/m^{-1}. Recent work suggests that long-term erosion at natural marshes is mainly governed by typical wave conditions, and that large storms only contribute less than 1% to long-term erosion rates (Leonardi et al. 2015). Likewise, peak rates of marsh loss in this study during 2006-2013 did not coincide with the increased occurrence of tropical storms and hurricanes.

Although wave energy regime did not significantly affect marsh loss at all study sites, the highest rates of marsh loss occurred at bulkheads in high wave energy environments. This observation supports the idea that the effect of wave energy on marsh loss may be amplified as wave energy increases. Since a linear relationship between marsh loss and wave energy was not observed, there may be a critical threshold energy value that must be reached for the reflection of wave energy off bulkheads to increase marsh loss. However, there is limited statistical confidence in this relationship, and future research should further investigate wave reflection as a mechanism of marsh loss at landward bulkheads.

Boat wave energy was included in the cumulative wave energy index because previous studies have demonstrated the correlation between boat wake waves and shoreline erosion

(Currin et al. 2017). I used distances to both commercial and recreational navigation channels as proxies, each with varying levels of potential wave energy. In this study, I observed a very weak relationship between marsh loss and distances to commercial and recreational channels, illustrating that these values may not have been the most accurate proxies for boat wave energy in the system. NOAA is currently developing a boat wake model (BoMo) to simulate wave heights generated by boats of various shapes and sizes (Fonseca and Malhotra 2012). Once this model is developed, it could be applied in similar studies to provide a more representative value of boat wave energy in a given area.

Sea Level Rise and Coastal Squeeze

The presence of natural marshes is governed by several dynamic processes. Salt marshes build elevation through sedimentation and transgress upland as sea level rises, but vertical drowning of marsh may occur if sediment input is not sufficient to keep up with sea level rise (Pontee 2013). Without adequate sedimentation to build surface elevation, the survival of salt marshes relies upon marsh gain through upland transgression. In fact, upland migration has enabled the Chesapeake Bay to maintain the spatial extent of its salt marshes since the late nineteenth century through conversion of upland areas to marsh (Schieder et al. 2017). In addition to these processes, marshes often experience horizontal erosion, or marsh loss at the waterward edge. Coastal development, including hardened structures like bulkheads, prevent the process of upland migration of marshes by physically blocking transgression. This, in combination with increasing rates of sea level rise, results in coastal squeeze and subsequent loss of salt marsh.

In this study, I estimated erosion of marsh edge and net loss of marsh at bulkheads and natural marshes. The results of this work suggest that rates of marsh loss are higher at bulkheads, as these structures appear to increase outer edge erosion, and they prevent marsh gain through upland migration. Many natural marsh sites experienced upland migration but gains in marsh through this expansion were still insufficient to offset marsh loss due to erosion of the waterward edge. Predicted increases in the rate of sea level rise will likely exacerbate marsh loss, as increasing water levels drive erosion of the marsh edge and reduce the ability to accrete marsh through changes in sediment deposition (Mariotti and Fagherazzi, 2013). Additionally, marsh will likely be lost through the prevention of upland migration by coastal development and shoreline hardening. This, in conjunction with the inability of the marsh to build up elevation, or with ongoing erosion of the marsh edge, leads to the phenomenon of coastal squeeze.

Patterns of marsh loss between imagery set time intervals were closely related to patterns in rates of sea level rise. Sea level rise in the study area has averaged 3 mm/yr. since 1953, but the rate during time intervals included in this study ranged from 0.84 mm/yr. to 8.87 mm/yr. (Figure 13). The period from 1992-2006 with the lowest rates of sea level rise during the study period (0.84 mm/yr) corresponded with anomalously low marsh loss rates (Figures 12 & 13). Conversely, the period from 2006-2013 with the most rapid rates of sea level rise (8.87 mm/yr.) corresponded with the highest rates of marsh loss. These results suggest that horizontal erosion rates of salt marshes correlate with rates of sea level rise, providing observational support for model predictions (Mariotti and Fagherazzi, 2013).

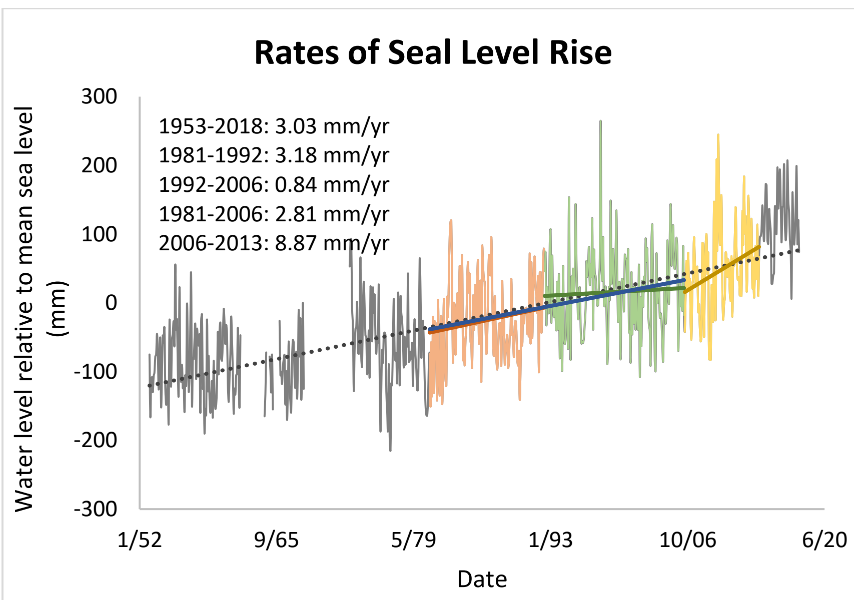

Figure 13: Monthly mean sea level without the regular seasonal fluctuations due to coastal ocean temperatures, salinities, winds, atmospheric pressures, and ocean currents. Linear trends from 1953-2018 (dashed line), 1981-1992 (orange line), 1992-2006 (green line), 1981-2006 (blue line), and 2006-2013 (yellow line) are shown along with their slopes (in mm/yr). The plotted values are relative to the most recent Mean Sea Level datum established by CO-OPS. Data from NOAA tides and currents (https://tidesandcurrents.noaa.gov/sltrends/sltrends_station.shtml?id=8656483).

Policy Recommendations

The state permitting process in North Carolina does not fully account for the potential effects of shoreline stabilization on the ecosystem functions and services of salt marshes. In fact, the stabilization permitting process in North Carolina assumes that bulkheads do not significantly impact public trust resources, including salt marshes (NC DCM 2014). However, recent studies have begun to identify negative impacts of shoreline hardening such as bulkheads on salt marsh ecosystems, including detrimental effects on coastal diversity, species abundance, and habitat quality. The results of this work also provide evidence supporting the coastal squeeze concept, in which bulkheads contribute to marsh loss by preventing upland transgression as sea level rises. It is likely that salt marshes along estuarine shorelines cannot coexist indefinitely with bulkheads and vertical walls. A potential intermediate solution could be to develop estuarine setbacks based on long-term erosion rates (as quantified in this study) similar to how oceanfront development is regulated currently. For instance, bulkheads could be required to be installed a distance from the upland extent of salt marsh equal to 30 times the measured marsh upland migration rate to provide space for marshes to migrate over the span of a typical home mortgage.

There are several estuarine shoreline stabilization techniques available, including a suite of techniques such as vegetation, oyster reefs, and marsh sills that are collectively referred to as 'living shorelines' (NCNERR 2013). Living shorelines typically maintain the connectivity between aquatic, intertidal and terrestrial habitats and minimize negative effects of estuarine shoreline stabilization while maintaining salt marsh (Restore America's Estuaries 2015, Smith et al. in press). Additionally, Gittman et al. (2014) found that bulkheads were less resilient than living shorelines to the impacts of category 1 hurricanes. Yet, the current regulatory framework of shoreline stabilization permitting is designed in a way that potentially incentivizes the use of bulkheads over living shorelines, as bulkhead permits are often issued more quickly than permits for living shorelines and typically require less government oversight (Scyphers et al. 2015). I recommend that the permitting process be restructured in a manner that encourages the use of living shorelines for shoreline stabilization. This could be accomplished by increasing the price of bulkhead permits to incentivize homeowners and businesses to pursue the use of living shorelines. Furthermore, the state permitting process should account for wave energy by examining sites with the Living Shorelines Suitability Tool to investigate wave energy at specific sites and avoid the use of bulkheads where living shorelines would be effective. However, expansion of the Living Shorelines Tool beyond its current geographic reach is needed for this tool to be utilized throughout the state.

In addition to changes in the permitting process, educational programs should be implemented to inform property owners, coastal engineers, and contractors about living shorelines. Courses on estuarine shoreline stabilization should be offered to the public that describe the options at length, allowing them to understand the economic and environmental factors and make more informed decisions. In addition, the North Carolina Coastal Training Program held by the NC Coastal Reserve should continue to provide workshops to coastal engineers and contractors on the implementation of living shorelines (NCNERR 2018). The revenue generated by increased permit prices may be used to fund such programs and provide subsidies for living shorelines projects or fund further research. Policymakers need a more comprehensive understanding of the adverse impacts of bulkheads to better manage estuarine stabilization and maintain coastal resiliency.

Future Research

Several avenues of research should be pursued to expand on the work of this study to better understand the impacts of bulkhead structures in North Carolina. A similar study investigating other areas within the Albemarle-Pamlico Estuary is needed, as they may experience dissimilar impacts due to differences in tidal patterns, salinity, and other variables. Microtidal systems like the Albemarle and Pamlico sounds are likely to have greater impacts of sea level rise, and impacts are likely to be lessened in systems with larger tides like along the Cape Fear River. Distances to commercial and recreational channels may not have been the most appropriate proxies for boat wave energy, and the cumulative wave energy index could be improved with a more representative boat wave energy metric or proxy. The Wave Exposure Model was run using wind energy data from a single period, and it may have been useful to run the model for each of the dates corresponding to the imagery sets for a more precise measure of overall energy during each time period.

Conclusion:

Salt marsh ecosystems are on the decline and are threatened by a multitude of factors. This study investigated the long-term effects of shoreline hardening and wave energy on marsh loss by assessing changes in marsh width from 1981-2013 using historic aerial imagery. The results of this work suggest that bulkheads have a significant effect on marsh loss through erosion of the waterward edge and prevention of upland migration. Wave energy regime did not significantly correlate with marsh loss, but the highest rates of marsh loss occurred at bulkheads in high wave energy environments, potentially due to reflection of wave energy. These results show that bulkheads may negatively impact salt marsh ecosystems, which should be considered in estuarine shoreline management. Only about 8% of North Carolina's of estuarine shoreline has been stabilized, which allows for the opportunity to move away from the use of hardened structures and towards the use of natural methods like living shorelines (Currin et al. 2010, NC DCM 2012). A growing body of evidence has shown that living shorelines can maintain or increase salt marsh ecosystem services and have less of an impact on the surrounding ecosystem than hardened structures. Living shorelines have shown to be more resilient than bulkheads and rip rap revetments during storms (Gittman et al. 2014) and can enhance nursery provision (Gittman et al. 2016) and carbon sequestration (Davis et al. 2015). This study was the first to investigate multi-decadal effects of bulkhead structures on marsh loss and provides useful information for better understanding the effects of shoreline hardening on salt marsh ecosystems. Ultimately, guarding against property erosion should not compromise the integrity of salt marsh ecosystems and the ecosystem services they provide to coastal communities throughout North Carolina.

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disseminated by NOAA or the NC Department of Environmental Quality, and do not represent any agency determination, view, or policy.

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