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Bulkheads Reduce Salt Marsh Extent: A Multidecadal Assessment Using Remote Sensing

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ABSTRACT

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Coastal development and shoreline armoring have contributed to rapid declines of salt marsh ecosystems. This study investigates multidecadal effects of bulkheads, a common shoreline armoring technique, on marsh extent in microtidal salt marshes. Aerial imagery of Bogue and Back sounds and Newport and North rivers (North Carolina, U.S.A) from 1981, 1992, 2006, and 2013 was used to measure changes in marsh extent at 45 sites with bulkheads landward of marsh and 45 control sites without bulkheads. At each site, change in marsh shoreline position (*i.e.* erosion or accretion) as well as landward marsh migration were measured. Rates of shoreline change and net change, the sum of shoreline change and landward migration, were compared among bulkhead and control sites. Over a 32-year period, salt marshes with landward bulkheads experienced higher mean rates of shoreline erosion than marshes without bulkheads (−0.14 ± 0.01 m/y *vs.* −0.09 ± 0.01 m/y). Sites without bulkheads as a barrier were able to offset some shoreline erosion through landward migration (mean migration rate = 0.05 ± 0.01 m/y). All bulkhead sites experienced net marsh loss, whereas 36% of control sites experienced net marsh gain. Net marsh loss was nearly three times higher at sites with bulkheads over the study period (−0.14 ± 0.01 m/y *vs.* −0.05 ± 0.01 m/y). Our results suggest that bulkheads can have a significant negative effect on marsh extent through increased erosion of the waterward edge and prevention of landward migration with sea-level rise (*i.e.* coastal squeeze). Land-use planning and conservation efforts protecting marsh migration corridors, combined with living shoreline strategies to reduce shoreline erosion, will be critical in protecting productive salt marsh ecosystems and the vital ecosystem services they provide.

ADDITIONAL INDEX WORDS: *Shoreline erosion, marsh loss, marsh migration, shoreline armoring, coastal squeeze.*

INTRODUCTION

Coastal salt marshes are on the decline worldwide and disappearing more rapidly than any other type of wetland in the United States (Dahl, 2011). Concurrently, the many ecosystem services salt marshes provide, such as storm surge protection, carbon sequestration, improved water quality, and nursery habitat, are also diminishing (Barbier *et al.*, 2011). Among a multitude of factors, sea-level rise (SLR) and coastal development contribute to the deterioration of salt marsh habitats. Coastal development often results in shoreline hardening, or armoring, whereby hardened shoreline stabilization structures like bulkheads or seawalls are used to protect against coastal hazards such as erosion, flooding, and subsequent property damage. As human populations have increased in coastal areas, shoreline hardening has become an increasingly popular practice globally.

A growing body of research has identified numerous detrimental impacts of shoreline armoring on adjacent sediments, vegetation, and fauna, as well as cumulative impacts at the watershed scale (Braswell and Heffernan, 2019; Currin, 2019; Dugan *et al.*, 2018; Kornis *et al.*, 2017). Hardened, vertical bulkheads reflect wave energy onto adjacent marshes, often leading to scour that can deepen the adjacent water and undercut the roots of marsh grasses, thereby threatening the survival of marsh vegetation (Bozek and Burdick, 2005; NRC, 2007). Shoreline hardening also alters the hydrodynamics and sediment transport within a system (Martin, Cable, and Jaeger, 2005; Miles, Russel, and Huntley, 2001), which may prevent nearby marshes from maintaining surface elevation through sedimentation (Dugan *et al.*, 2011). Bulkheads physically cut off the upland from the intertidal and subtidal regions and, in doing so, have shown to adversely affect waterbird communities, bivalve abundance and diversity, benthic infauna abundance, fish community integrity and species diversity, and resilience of submerged aquatic vegetation (SAV) (Bilkovic and Roggero, 2008; Gittman *et al.*, 2016; Landry and Golden, 2018; Prosser *et al.*, 2018; Sietz *et al.*, 2006). Furthermore, bulkheads serve as physical barriers that prevent landward migration of marshes as sea level rises.

In response to rising sea level, salt marshes must either “keep up” by increasing surface elevation or “move up” via landward migration to persist (Brinson, Christian, and Blum, 1995). Recent modeling efforts have demonstrated that in many areas, landward migration of marsh will be the primary

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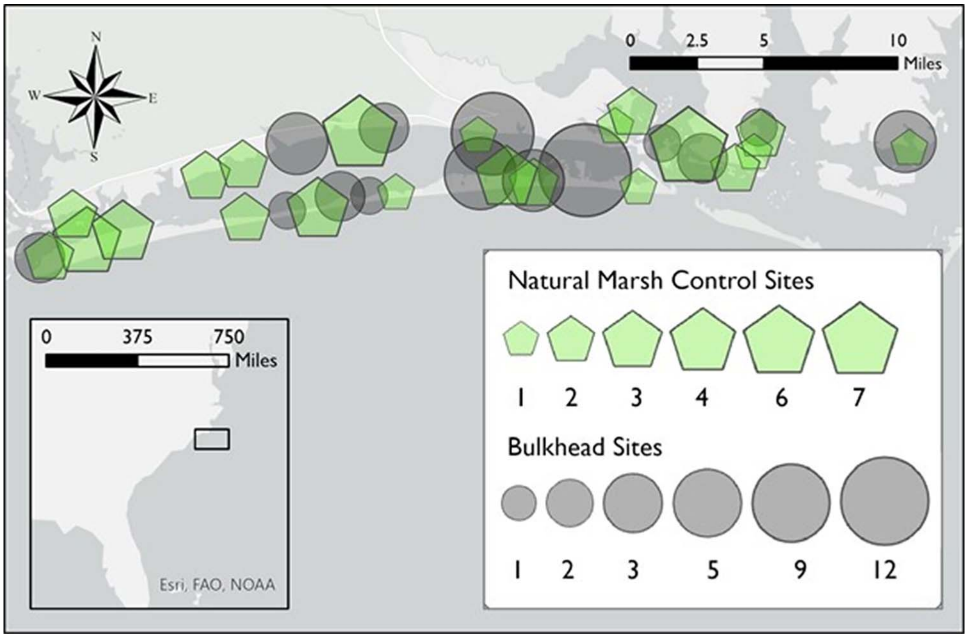

Figure 1. Map of the study area, including Bogue Sound, Back Sound, Newport River, and North River in Carteret County, North Carolina. Symbols represent location and concentration of natural marsh controls and bulkhead sites.

mechanism to maintain marsh extent (Kirwan *et al.*, 2016). Marsh migration is highly dependent on several factors, including tidal amplitude, suspended sediment concentrations, suitable topography, and undeveloped space (Nunez *et al.*, 2021; Torio and Chmura, 2013; Warnell, Olander, and Currin, 2022).

The combination of rising sea level and coastal development can lead to a process known as coastal squeeze. Development and hardened structures limit the ability of marshes to naturally migrate landward in response to SLR (Doody, 2013; Pontee, 2013). Further, bulkheads can reflect wave energy, potentially leading to increased wave height, sediment resuspension, and scour, which can threaten the persistence of marsh vegetation (Bozek and Burdick, 2005; Miles, Russel and Huntley, 2001; NRC, 2007). When marshes cannot increase surface elevation fast enough to keep pace with SLR, the combination of erosion at a marsh's waterward edge and inability to migrate landward will inevitably result in a net loss of salt marsh extent.

The primary objectives of this study were to compare rates of change in extent at salt marshes with and without adjacent landward bulkheads over a 32-year time series (1981, 1992, 2006, and 2013–14) of aerial imagery collected in Bogue and Back sounds and Newport and North rivers in North Carolina, U.S.A. Change in marsh extent was measured in two ways: (1) as the landward (−) or waterward (+) movement of the shoreline, termed shoreline change, and (2) as net change, the sum of shoreline change and migration of the landward marsh edge (+). Shoreline change and net change rates were annualized to estimate change rates. We hypothesized that rates of marsh shoreline erosion and net marsh loss would be greater where bulkheads were landward of the salt marsh.

METHODS

Salt marshes are the most dominant shoreline type along the 10,657 miles of estuarine shoreline in North Carolina (McVerry, 2012). The geographic area of this study encompasses Bogue Sound and Back Sound, as well as portions of Newport and North rivers in Carteret County, North Carolina (Figure 1). Salt marsh accounts for approximately 83% (1,270 miles) of the 1,530 miles of estuarine shoreline in Carteret County, whereas roughly 6% (87 miles) of the shoreline is hardened with bulkheads (McVerry, 2012). The remaining 11% of the shoreline is either a different shoreline type (*e.g.*, sediment bank) or hardened with structures other than bulkheads (*e.g.*, riprap). This geographic scope was selected because of the availability of historic aerial imagery (see below). Circulation in Bogue, Back, and Core sounds is primarily driven by astronomical tides and freshwater input comes from the White Oak, Newport, and North rivers (Churchill *et al.*, 1999 and references therein). Mean range of tide across the study system varied from 0.41 m to 0.97 m.

Aerial Imagery Acquisition and Georectification

Aerial imagery taken in 1981 was acquired from the North Carolina Department of Transportation (NCDOT) Photogrammetry Unit (Carraway and Priddy, 1983) and used as the baseline against which to assess change in the position of the waterward and landward marsh edges over time. Subsequent imagery sets used in this analysis were taken in 1992, 2006, and 2013–14. All four aerial imagery sets were acquired to map SAV, and hence were captured at low tide, which also ensured consistency for delineating the waterward extent of marsh shorelines. The spatial resolution of

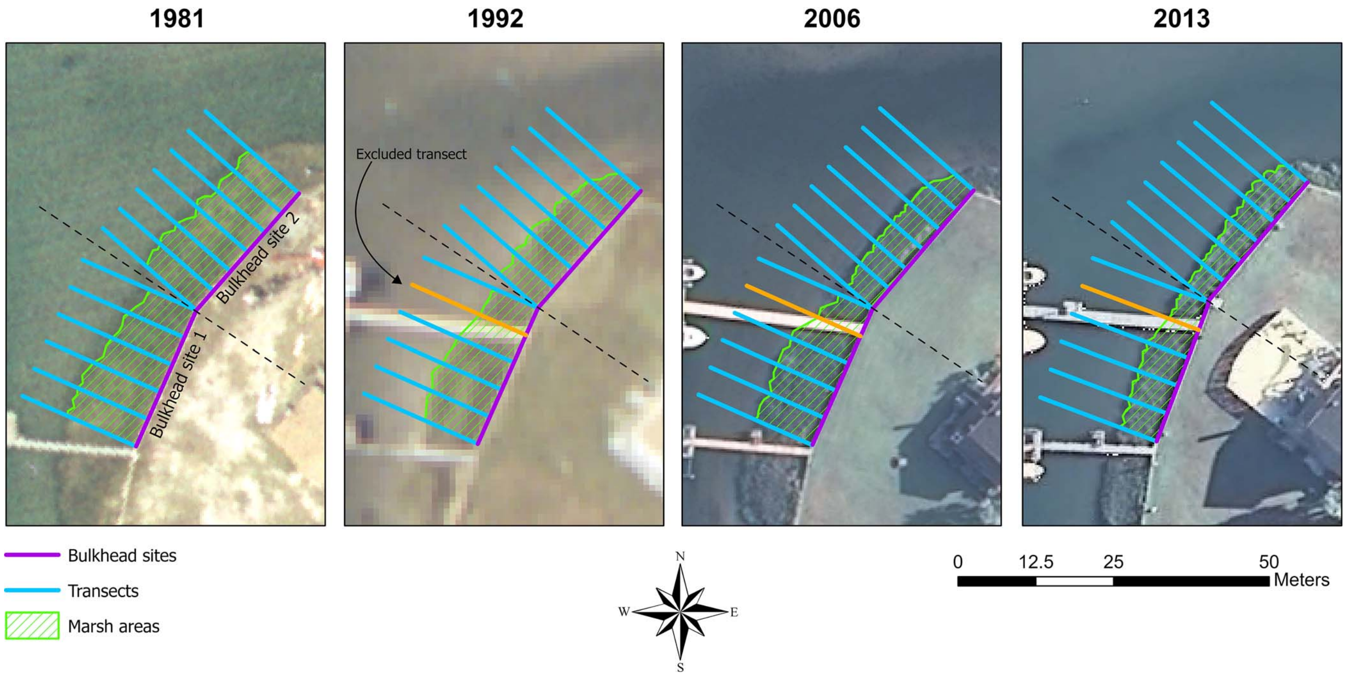

Figure 2. Time series of two bulkhead sites illustrating shoreline erosion (*i.e.* reductions in marsh width). Bulkhead sites were divided into two sites (separated by black dashed line) because of differing shoreline orientation. Transects are shown in blue and waterward marsh areas in green. Transects intersecting dock before reaching the waterward marsh edge (shown in orange) were excluded from the calculations.

the 1981, 2006, and 2013–14 imagery was 0.3 × 0.3 m, whereas the resolution of the 1992 imagery ranged from 1 × 1 m to 1.4 × 1.4 m.

The 1992, 2006, and 2013–14 imagery data sets were previously georectified, but the 1981 aerial imagery was acquired as digital scans that required georectification (242 tiles). 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.).

Using the georeferencing tool in ESRI ArcGIS v. 10.3.1 the 1981 aerial imagery was georectified using the 2006 imagery as a base map. Ten ground control points were selected for each image on the basis of 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 × 0.3 m. The root mean square error (RMSE), an assessment of the transformation's accuracy, was maintained below 2 m, with an average RMSE of 1.3 m (Hapke and Henderson, 2015).

Bulkhead Site Selection and Analysis

To select landward bulkhead sites for this analysis, bulkhead shapefiles obtained from the 2010 North Carolina Estuarine Shoreline Mapping Project (McVerry, 2012) were overlaid onto the georectified 1981 imagery from Carteret County. The criteria used to select potential bulkhead sites were: (1) the bulkhead was visibly present in the entire time series of imagery; (2) the marsh was waterward of the bulkhead and visible in the imagery in 1981 (*i.e.* the bulkhead was landward of the marsh); (3) the waterward extent of the marsh shoreline was visibly apparent; (4) the patch of marsh was larger than 10 m^2; and (5) the marsh waterward of the bulkhead was not obstructed by a dock running parallel to the shoreline. Meeting those criteria, a bulkhead site was constrained to a linear stretch with a single shoreline orientation. In cases where a single contiguous bulkhead structure had differing shoreline orientations, the bulkhead structure was divided into multiple sites (Figure 2). Using these criteria, 45 bulkhead sites were selected for the analysis.

Bulkhead sites were manually digitized using the editor toolbar in ESRI ArcGIS v. 10.3.1. Delineated bulkheads ranged from 10 m to 59 m in length. The waterward edge of the marsh was also delineated at each bulkhead site. Unusually dark or light edges of the waterward marsh boundary were not included in the waterward edge delineation and were assumed to be either sand, SAV, macroalgae, or shadows. Additionally, small patches of noncontiguous marsh outside of the marsh waterward edge were excluded.

To measure marsh width at each site, transect lines were established perpendicular to each delineated bulkhead that extended from the bulkhead to beyond the waterward marsh edge (Figure 2). The number of transects per site was proportional to the length of the delineated bulkhead, with 3 to 13 transects at each site. Marsh width was measured from each bulkhead to the waterward marsh edge along each transect in each imagery set. An average marsh width was calculated at each bulkhead site for each imagery set. Transects

Table 1. *Mean (±standard error [SE]) and range of proxies of wind wave and boat wave energy at bulkhead and natural control sites. Wind wave energy for the study area was modeled using the National Oceanic and Atmospheric Administration (NOAA)'s wave exposure model (WEMo; Malhotra and Fonseca, 2007; Currin et al., 2017). Hourly wind data for the model were acquired from the NOAA National Data Buoy Center station CLKN7 at Cape Lookout (~10+ km from most of our study sites). Representative wave energy (RWE) was calculated using the top 20% of wind data (RWE_{20}; sensu Currin et al., 2017).*

Shoreline Type	RWE_{20} ($J\ m^{-1}$) Mean ± SE; Range	Distance from Recreational Channels (km) Mean ± SE; Range	Distance from Commercial Channels (km) Mean ± SE; Range
Bulkheaded marsh	274.3 ± 45.8; 0–1197.8	1.4 ± 0.3; 0.02–10.0	2.2 ± 0.5; 0.2–13.7
Control marsh	244.8 ± 49.3; 8–1259.9	3.0 ± 0.4; 0.05–10.6	1.6 ± 0.2; 0.2–4.6

that crossed over docks before reaching the waterward marsh edge were excluded from mean width calculations (*e.g.*, orange transects in 1992, 2006, 2013 imagery in Figure 2).

Control Site Selection and Analysis

Imagery from 1981 and 2013–14 was used to identify salt marsh control sites within the study area that were not hardened with a bulkhead throughout the study period. Selection of control sites within 500 m of one of the bulkhead sites used in this study was prioritized. Additionally, marsh control sites near clear reference points were prioritized to aid georectification and site alignment in all imagery sets. Using estimates of wind wave energy modeled using WEMo (Malhotra and Fonseca, 2007), as well as distances from recreational and commercial boat channels as a proxy for boat wave energy, control sites were selected in energy environments similar to selected bulkhead sites (Table 1). A total of 45 control sites was selected.

Selected natural marsh control sites were manually digitized using the editor toolbar in ESRI ArcGIS v. 10.3.1. At each site, the landward marsh boundary was digitized for each imagery set. A straight site line at the landward marsh boundary was also delineated (Figure 3). Shoreline change and landward marsh migration were measured relative to the site line. The site line was terminated when the marsh or upland boundary changed orientation. The control site line lengths ranged from 17 to 73 m. The waterward edge of the marsh at each control site was delineated using the same protocol as bulkhead sites. Because of lower spatial resolution, the landward marsh boundary was not easily distinguishable

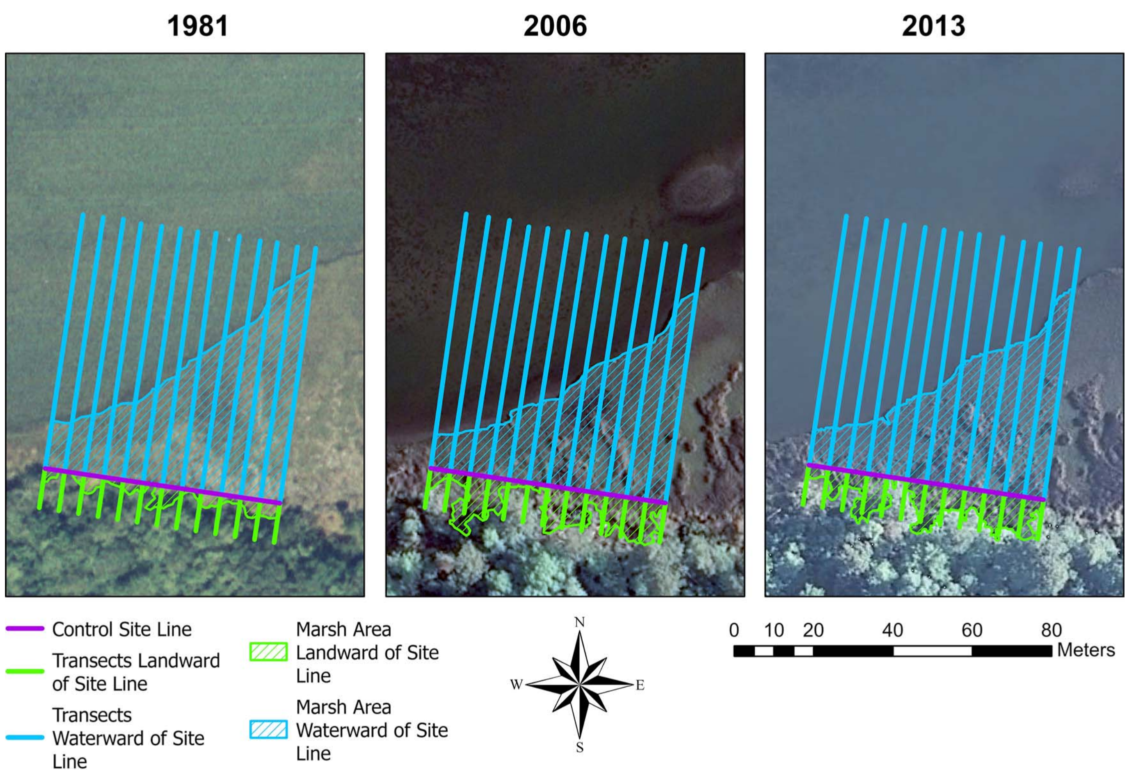

Figure 3. Time series of a natural marsh control site illustrating shoreline erosion (*i.e.* reductions in marsh width) but net marsh gain through landward marsh migration. Transects and marsh area waterward of the site line are shown in blue. Transects and marsh area landward of the site line are shown in green. Measurements from the site line waterward (blue) evaluate shoreline change, whereas controls accounting for migration (green) evaluate net change, the sum of shoreline change and landward migration.

Table 2. *Summary of the number of sites within each shoreline type that eroded or accreted between 1981 and 2013, as well as average (±SE) change in marsh width and rates of marsh width change over that time. Negative values represent reductions in marsh width.*

Shoreline Type	Sites Eroding	Sites Accreting	Avg. Δ Marsh Width (m)	Change Rate (m/y)
Bulkheads	45 (100%)	0 (0%)	−4.45 ± 0.27	−0.14 ± 0.01
Static controls	38 (85%)	7 (15%)	−3.29 ± 0.38	−0.09 ± 0.01
Controls w/migration	25 (55%)	20 (45%)	−1.59 ± 0.44	−0.05 ± 0.01

in the 1992 imagery, so delineations in this imagery only included the site line and waterward marsh edge (*i.e.* 1992 was not used in analyses of landward migration).

Marsh width was measured by establishing transects following similar protocols used for bulkhead sites, whereby the number of transects was proportional to the site line length (Figure 3). The number of transects ranged from 5 to 14 per site. To account for marsh landward migration, transects were also established perpendicular to the landward side of the site line (Figure 3). To assess changes in marsh width due to shoreline change, transect lengths were measured from site line to waterward marsh edge along each transect. To measure marsh migration, transect lengths were measured from the site line to the upland transition (*i.e.* edge of the tree line). At control marsh sites where migration occurred, transect length measurements both waterward and landward of the site line were totaled and averaged to calculate net change in marsh extent. Transects crossing over docks were excluded from calculations.

Statistical Analyses

All statistical analyses were conducted in R v. 4.3.2 (R Core Team, 2023). Rates of mean shoreline change, marsh migration, and net marsh change were compared with t tests and analysis of variance (ANOVA) using the rstatix package in R (v0.7.2; Kassambara, 2023). Normality was assessed using the Shapiro–Wilk test and quantile–quantile plots. Levene's test was used to test for homogeneity of variance. In instances where data transformation did not result in homogeneity of variance, Welch's t test was used. Unpaired t tests were used to compare shoreline change rate and net marsh change rate among shoreline types over the entire 32-year study period using only the 1981 and 2013 imagery. A paired t test was used to compare mean rates of shoreline erosion and migration rates from 1981 to 2013 at a subset of natural marsh control sites ($n = 20$) where shoreline erosion and migration were evident. For this comparison, the absolute values of shoreline erosion rates were used. To compare shoreline change and net marsh change rates at finer temporal scales, a two-way mixed-measures ANOVA was used with aerial imagery time periods (1981–92, 1992–2006, 2006–13) as within-subject factors and shoreline type (bulkhead and natural marsh controls) as between-subject factors. For multiple pairwise comparisons, *p* values were adjusted using the Bonferroni multiple testing correction method.

Total Uncertainty

Error in net change of marsh width was estimated as described in Currin *et al*. (2015) and adapted from Fletcher *et al*. (2003) for estuarine shorelines by Cowart, Walsh, and Corbett (2010). The rectification error was estimated at 1.3 m and tidal stage uncertainty was estimated at 0.5 m. A single individual conducted the heads-up digitizing, so there is not an estimate for digitization error. A reported digitization error of 0.55 m from similar studies was applied (Cowart, Walsh and Corbett, 2010; Currin *et al.*, 2015), which results in an estimated total uncertainty (U_t) of ±1.5 m, which when annualized over the study period is ±0.04 m/y.

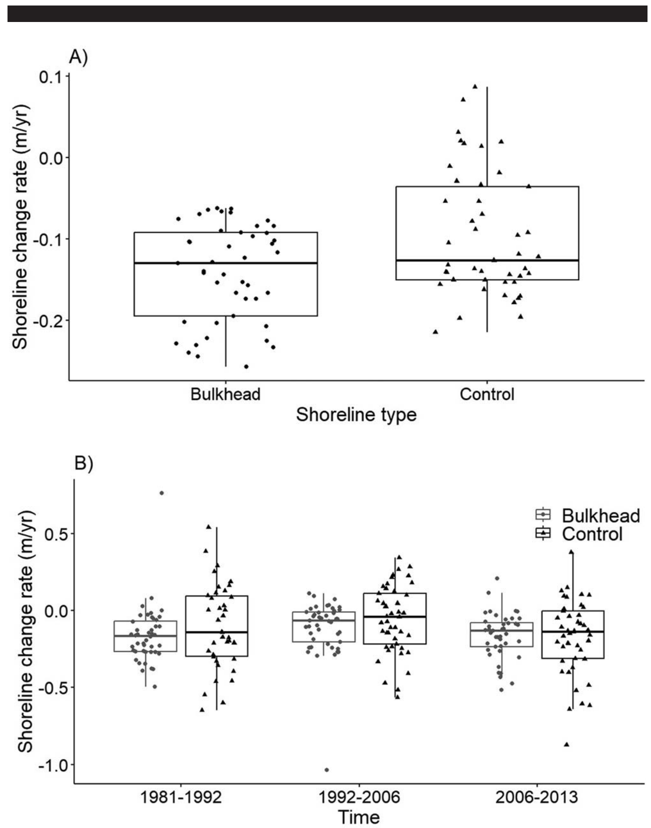

Figure 4. Mean shoreline change rates (A) from 1981 to 2013 for bulkhead and natural marsh control sites and (B) 1981–92, 1992–2006, 2006–13. Negative values represent shoreline erosion. Error bars represent standard error of means.

RESULTS

All 45 bulkhead sites experienced marsh shoreline erosion during the 32-year study period, with complete marsh loss occurring at 11% of bulkhead sites. Over 80% of the 45 natural marsh control sites experienced shoreline erosion, whereas the marsh shoreline at 7 sites (15%) accreted waterward (Table 2). None of the control sites experienced complete marsh loss.

Marsh Shoreline Change

Rates of shoreline erosion from 1981 to 2013 were significantly higher at bulkheads than at natural control marshes ($t = -3.10$, df = 88, $p = 0.003$; Figure 4A). The average rate of shoreline erosion was −0.14 ± 0.01 m/y at bulkhead sites and −0.09 ± 0.01 m/y at natural control marshes (Figure 4A; Table 2). Over the study period, an average of 4.45 ± 0.27 m

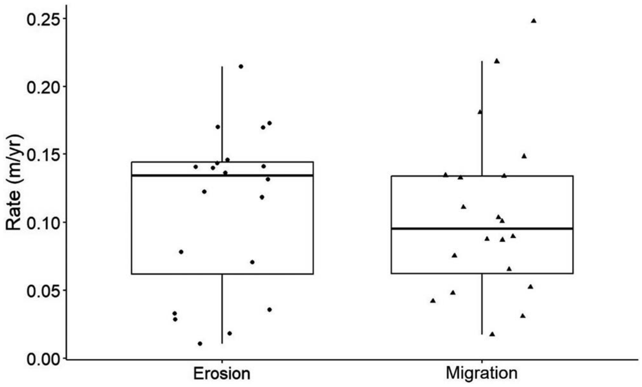

Figure 5. Rate of erosion and migration at natural marsh control sites from 1981 to 2013. Erosion rates are shown as the absolute value of erosion. Only sites where migration was quantifiable and erosion occurred are presented here ($n = 20$).

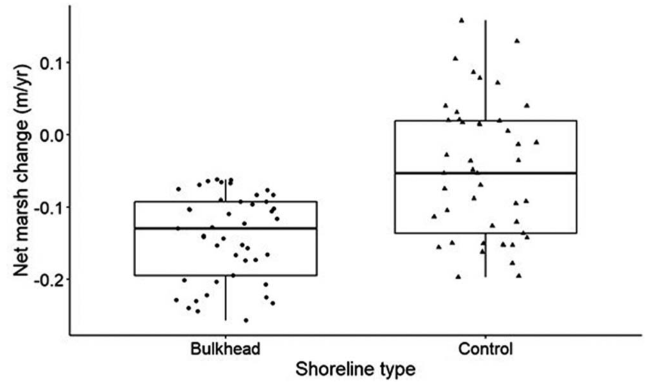

Figure 6. Net rate of marsh shoreline change from 1981 to 2013 for bulkhead and natural marsh control sites.

of marsh shoreline eroded at bulkhead sites and 3.29 ± 0.38 eroded at control sites. Shoreline erosion reduced marsh width by an average of 45% at bulkhead sites and 17% at control sites.

Over the shorter 7- to 14-year time intervals between imagery sets, rates of shoreline erosion were not significantly different between bulkheads and natural control marshes ($F_{1,78} = 3.8$, $p = 0.06$). However, rates of shoreline erosion did vary significantly between time intervals, irrespective of shoreline type ($F_{2,156} = 4.4$, $p = 0.01$; Figure 4B). Shoreline erosion rates were highest during 2006–13 (−0.16 m/y) and significantly higher than in the preceding period from 1992 to 2006 (−0.08 m/y; $t = 2.6$, df = 87, $p = 0.03$).

Landward Migration and Net Marsh Change

Migration of the landward marsh edge did not occur at bulkhead sites. Landward migration was measured at 20 of the 45 natural marsh control sites. Migration appeared to occur at several additional control sites, but shading in the imagery from upland vegetation prevented accurate quantification of migration. At these sites, migration was assumed to equal zero. Average rate of migration at natural control marshes, where quantifiable in imagery, was 0.10 ± 0.01 m/y (0.05 ± 0.01 m/y when averaged across all 45 sites; Table 2). Over the 32-year study period, rates of shoreline erosion and landward migration were not significantly different among natural control marshes ($t = -0.3$, df = 19, $p = 0.7$; Figure 5).

Net marsh change, the sum of shoreline change and migration of the landward marsh edge, was generally negative (*i.e.* net loss of marsh). All the bulkhead sites experienced net marsh loss (mean = −0.14 ± 0.01 m/y). The majority (64%) of the natural marsh control sites experienced net loss of marsh, although 16 (36%) sites experienced net marsh gain (mean = −0.05 ± 0.01 m/y; Figure 6). From 1981 to 2013, bulkhead sites experienced significantly greater net marsh loss than natural control marshes ($t = -5.6$, df = 75, $p < 0.001$). Net loss of marsh width over the study period averaged −1.59 ± 0.44 m (6% loss) at natural marsh control sites and −4.45 ± 0.27 m (48% loss) at bulkhead sites.

Over shorter time intervals, from 1981 to 2006 and 2006 to 2013, net marsh loss was significantly greater at bulkhead sites than at natural marsh control sites ($F_{1,86} = 8.9$, $p = 0.004$). Net marsh loss rates at bulkhead and control sites were not significantly different among time periods ($F_{1,86} = 2.2$, $p = 0.1$).

DISCUSSION

The results suggest that bulkheads landward of salt marshes accelerate shoreline erosion and loss of marsh extent. Salt marsh erosion, or landward movement of the shoreline edge, was greater at sites with bulkheads than sites without bulkheads (Figure 4A). Furthermore, the prevention of marsh landward migration by bulkheads contributed significantly to net marsh loss in this study region (Figure 6). Marshes without bulkheads experienced smaller net losses in overall extent because of expansion of the landward edge through marsh migration, and some sites even experienced net marsh gain.

Marsh functions such as nutrient reduction and wave buffering capabilities are directly related to marsh width (O'Meara, Thompson, and Piehler, 2015; Yang *et al.*, 2012). Hence, reduction of salt marsh extent can significantly reduce marsh ecosystem service provision. To protect productive salt marsh ecosystems and the many ecosystem services they provide, estuarine shoreline management should consider stricter policies on shoreline hardening to account for potential negative effects of shoreline hardening on salt marshes, a vital public trust resource. Parts of the U.S. coastline, such as much of the Southeast, are largely unarmored (Gittman *et al.*, 2015), so there is potential to be proactive and develop policies that account for the negative impacts of bulkheads before their further proliferation on the basis of predicted increases in population growth, coastal development, and SLR.

The environmental impacts of bulkheads can be subtle (Gehman *et al.*, 2018). In a meta-analysis of 25 studies investigating the impact of bulkheads and seawalls on organism abundance and biodiversity, Gittman *et al.* (2016) reported that less than half of the studies found significant negative effects, although when analyzed collectively, biodiversity and abundance of marine fauna were negatively affected by bulkheads and seawalls. Bozek and Burdick (2005) identified a significant reduction in plant diversity at the marsh–upland boundary where bulkheads were present but found no significant effect of bulkheads on a suite of other processes related to marsh health and stability. In this study, the rate of salt marsh shoreline erosion adjacent to bulkheads was not significantly different from that at natural control marshes during 7- to 14-year intervals (*e.g.*, 1981–92) but was significantly different over the 32-year study period. To a large degree, the lack of significant differences in erosion associated with bulkheads over relatively short periods may be due to the error (±0.6 m) associated with measuring shoreline change with aerial photography. The pervasiveness of shoreline erosion in our study system (shoreline erosion at 92% of sites) and other estuarine shorelines, hardened or not, may also complicate the ability to detect significant impacts of shoreline hardening. As such, long-term multidecadal assessments may be required to detect the subtle accumulation of impacts from bulkheads on salt marshes.

The shoreline erosion and migration rates reported in this study were similar to, albeit slightly lower than, those reported along other marsh shorelines, including those in the same geographic region (Cowart, Corbett, and Walsh, 2011; Currin *et al.*, 2015). In this study, average shoreline erosion rates ranged from −0.09 ± 0.01 m/y at control sites to −0.14 ± 0.01 m/y at sites with landward bulkheads. Previous work investigating change along marsh shorelines in southern Pamlico Sound and in the New River estuary have reported rates of −0.22 m/y (Cowart, Corbett, and Walsh, 2011) and −0.18 m/y (Currin *et al.*, 2015), respectively. Average rate of marsh migration at natural control sites, where quantifiable in imagery, was 0.10 ± 0.01 m/y in this study. We are not aware of published rates of marsh migration near the project geography, but rates reported along the East, Gulf, and West coasts of the United States ranged from 0.1 m/y to 6.7 m/y (Flester and Blum., 2020 and references therein).

Shoreline erosion rates varied over time. Time serves as a proxy for important events (*e.g.*, tropical storms and hurricanes) and processes (*e.g.*, SLR) that play a role in marsh shoreline erosion (Fagherazzi *et al.*, 2019). During the time intervals analyzed, the number of named storms within a 50-nautical-mile radius of the study area ranged from ∼1 every 3 years (1981–92) to ∼1 per year (1992–2006). The relative rate of SLR was also highly variable over time intervals, ranging from 0.84 mm/y (1992–2006) to 8.87 mm/y (2006–13). The lack of increased marsh edge erosion during periods of more frequent and intense storms is not all that surprising given that long-term erosion of marsh shorelines may be mainly governed by average wave conditions rather than large storms (Leonardi, Ganju, and Fagherazzi, 2016). The highest and lowest rates of shoreline erosion were observed during periods with the highest and lowest SLR, respectively. Increased inundation frequency and duration during the 2006–13 period were likely detrimental to marsh vegetation and edge stabilization.

Marsh migration can play a pivotal role in maintaining salt marsh extent, particularly under increasing rates of SLR (Kirwan *et al.*, 2016; Schieder, Walters, and Kirwan, 2018; Warnell, Olander, and Currin, 2022). Yet, the rate of landward migration measured in this study did not increase significantly during periods (∼decade) of increased rates of SLR. The lack of a response could be due to (1) resolution of our approach (see suggestions for future research below), (2) a time lag between increased SLR and landward migration (Kirwan and Murray, 2008), and (3) the combination of storms and SLR during time intervals whereby the stormiest interval had the lowest SLR and vice versa. Fagherazzi *et al*. (2019) suggested that SLR and storms combine to regulate landward migration.

Coastal development, including hardened structures like bulkheads, prevent the process of landward migration of marshes by forming a physical barrier (Pontee, 2013), a phenomenon known as coastal squeeze. In the absence of bulkheads, 20 of the 45 control sites exhibited landward marsh migration at rates similar to shoreline erosion at these sites (*i.e.* no net loss). Land-use planning and conservation efforts protecting marsh migration corridors, combined with living shoreline strategies to reduce shoreline erosion, will be critical in protecting marsh structure and, thus, function.

Study Limitations and Future Research

Several characteristics of this study pose limitations that warrant further research to better understand the complexities associated with loss of salt marsh. Factors important to marsh resilience, such as elevation and sediment supply, were not considered in this study. Wind and wave energy proxies were used to select bulkhead and natural control marsh sites exposed to similar wave energy, but wind wave energy modeling was based on dated bathymetry and there were no estimates of boat traffic in navigation channels. The development of a WEMo analog for boat wake energy and quantifying the relative contribution of wind and boat wave energy to shoreline erosion should be research priorities considering that boat wakes can potentially be managed with actions such as no-wake zones (or moving navigational channels farther from shore). Furthermore, our observations from the 1992 imagery were limited because of a coarse resolution, and future historic aerial analyses of salt marshes should use higher-resolution imagery to more accurately delineate marsh boundaries. Last, the study focused on a fairly small geographic area. Outside of our study area where variables such as tidal amplitude, wave energy, SLR, and sediment delivery differ, so too may the impacts of bulkheads on marsh loss. Other areas within the Albemarle–Pamlico Estuary may experience dissimilar impacts due to differences in tidal patterns, salinity, wave energy, and other variables, so future research should expand outside of the geographic scope of this work to investigate impacts of bulkhead structures on salt marshes in other settings.

CONCLUSION

Salt marsh ecosystems are threatened by an array of factors including coastal development and SLR. This study was the first to investigate the long-term impacts of bulkhead structures on loss of salt marsh extent and provide useful information for better understanding the effects of shoreline hardening on salt marsh ecosystems. Our results suggest that bulkheads can have a significant negative effect on marsh extent through increased erosion of the waterward edge and prevention of landward migration with SLR. Ultimately, estuarine shoreline management and policy must consider potential effects of shoreline stabilization practices, such as those illustrated here, to preserve the integrity of productive salt marsh ecosystems and the vital ecosystem services they provide.

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