



Historical Changes in the Vegetated Area of Salt Marshes

Christine J. Burns^{1,2} · Merryl Alber¹ · Clark R. Alexander^{1,2}

Received: 3 April 2019 / Revised: 5 June 2020 / Accepted: 10 June 2020 / Published online: 30 June 2020
© Coastal and Estuarine Research Federation 2020

Abstract

Salt marshes are valuable ecosystems, and there is concern that increases in the rate of sea level rise along with anthropogenic activities are leading to the loss of vegetated habitat. The area of vegetated marsh can change not only through advance and retreat of the open fetch edge, but also due to channel widening and contracting, formation and drainage of interior ponds, formation and revegetation of interior mud flats, and marsh migration onto upland areas, each of which is influenced by different processes. This study used historical aerial photographs to measure changes in the extent of vegetated marsh over approximately 70 years at study marshes located in three long-term ecological research (LTER) sites along the US East coast: Georgia Coastal Ecosystems (GCE), Virginia Coast Reserve (VCR), and Plum Island Ecosystems (PIE). Marsh features were categorized into vegetated marsh, ponds, interior mud flats, and channels for three time periods at each site. The three sites showed different patterns of change in vegetated marsh extent over time. At the GCE study site, losses in vegetated marsh, which were primarily due to channel widening, were largely offset by channel contraction in other areas, such that there was little to no net change over the study period. The study marsh at VCR experienced extensive vegetated marsh loss to interior mud flat expansion, which occurred largely in low-lying areas. However, this loss was counterbalanced by marsh gain due to migration onto the upland, resulting in a net increase in vegetated marsh area over time. Vegetated marsh at PIE decreased over time due to losses from ponding, channel widening, and erosion at the open fetch marsh edge. Digital elevation models revealed that the vegetated areas of the three marshes were positioned at differing elevations relative to the tidal frame, with PIE at the highest and VCR at the lowest elevation. Understanding the patterns of vegetation loss and gain at a given site provides insight into what factors are important in controlling marsh dynamics and serves as a guide to potential management actions for marsh protection.

Keywords Salt marsh · LTER · Image analysis · Ponding · Marsh migration

Introduction

Salt marshes are intertidal wetland habitats found in protected areas in the temperate zone. They provide many important ecosystem services including storm protection, nursery habitat, nutrient cycling, and carbon storage (as reviewed in Barbier et al. 2011; Temmerman et al. 2013).

Marshes are subject to changes in sea level and have historically kept pace with rising seas. However, global rates of sea level rise (SLR) have been increasing over the past century (Church and White 2011), from an estimated 1.4 mm year⁻¹ from 1901 to 1990 to 3.6 mm year⁻¹ for 2006–2015 (Oppenheimer et al. 2019). These rates are expected to increase even further, with some model estimates projecting global sea level rise rates of 15 mm year⁻¹ by 2100 (Oppenheimer et al. 2019). This has raised concerns regarding potential drowning and consequent loss of vegetated marsh area. Human activities that affect sediment delivery to the coast (Fagherazzi et al. 2013; Ganju et al. 2017) or provide barriers to marsh migration (Torio and Chmura 2013) can also have consequences for marsh extent, as can the legacy of direct manipulations such as ditching and draining (Adamowicz and Roman 2005; Coverdale et al. 2013).

Shifts in the area of salt marsh vegetation are manifest as changes in the features of a salt marsh, such that the net change in vegetated area over time is a consequence of

Communicated by Charles Simenstad

Electronic supplementary material The online version of this article (<https://doi.org/10.1007/s12237-020-00781-6>) contains supplementary material, which is available to authorized users.

✉ Merryl Alber
malber@uga.edu

¹ Department of Marine Sciences, University of Georgia, Athens, GA 30602, USA

² Skidaway Institute of Oceanography, University of Georgia, Savannah, GA 31411, USA

advance and retreat of the marsh edge, channel widening and contraction, formation and drainage of ponds, interior mud flat formation and revegetation, and migration of the marsh onto the upland (Fig. 1). Each of these shifts is responding to different processes, and it is therefore instructive to understand their relative importance when evaluating overall changes in vegetated marsh area.

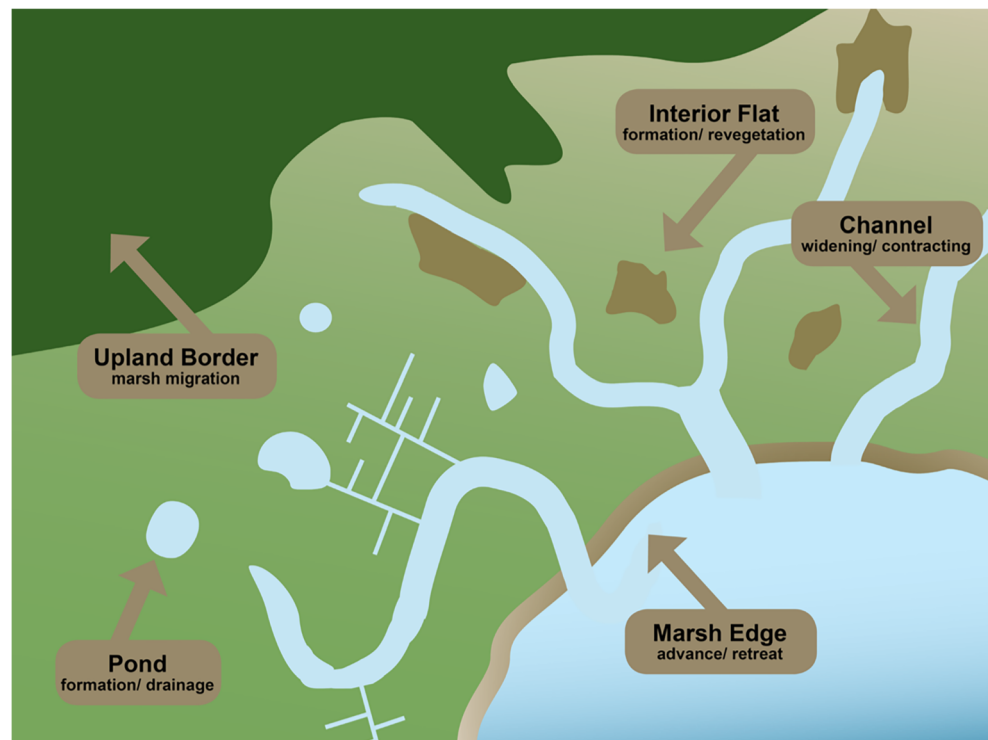
The location of the marsh edge naturally advances and retreats over time (Fagherazzi 2013). In eroding areas, wave action physically removes material from the seaward edge of the marsh, thus leaving no substrate for plants (Schwimmer 2001). Wave-induced erosion is more severe in places with large fetch (Day et al. 1998; Kearney et al. 1988) or those with heavy boat traffic (Browne 2017). SLR exacerbates this type of marsh loss by increasing the depth of the adjacent body of water and allowing for the generation of larger waves and their propagation to shore (Mariotti and Fagherazzi 2013).

Channels often have alternating sections of edge advance and retreat due to internal sediment redistribution and channel current gradients, which leads to erosion on outside bends where the current is stronger and deposition along the inside (Burns 2018; Eisma 1998; Seminara 2006). However, they can also expand or contract due to changes in the tidal prism caused by SLR (D’Alpaos et al. 2010; Hughes et al. 2009). Under conditions of prolonged inundation, the vegetation along the channel edge can become stressed, causing plants to die back (Downs et al. 1994). This has been seen, for example, in Rhode Island, where a historical analysis

indicated that 35 of 36 marshes studied were experiencing channel widening, with higher loss rates in marshes that were at lower elevations (Watson et al. 2017). Herbivory and burrowing by crabs can also affect channels through the removal of stems as well as through the excavation of burrows, which leads to both channel widening and headward erosion (Hughes et al. 2009; Perillo et al. 1996; Vu et al. 2017), and there is evidence that this phenomenon has increased in recent years (Crotty et al. 2017). Changes in drainage patterns can also lead to recolonization of the channel edge and subsequent channel narrowing. This process can be rapid due to positive feedbacks between plants and sediment trapping (Morris et al. 2002).

Vegetation in the marsh interior may convert to interior mud flats (referred to as mud flats, below) or ponds through several different mechanisms (Passeri et al. 2015; Schepers et al. 2017) (Fig. 1). In many cases, these features form in low-lying areas where prolonged inundation causes vegetation stress and organic matter decomposition. Thus, mud flats and ponds observed as horizontal features on the marsh surface can be indicative of a loss of vertical elevation (Downs et al. 1994; Hughes et al. 2009). Wrack deposition (Hartman 1988; Pennings and Richards 1998) and marsh dieback (Alber et al. 2008) are two other phenomena known to result in mud flats. Although plants can recolonize these bare areas, they can also convert to ponds, particularly in places where the marsh cannot keep up vertically with SLR (Downs et al. 1994; Kearney et al. 1988). Ponds and mud flats can also form due

Fig. 1 Conceptual diagram showing the different types of lateral marsh expansion and contraction considered in this study



to physical mechanisms such as creek blockage from slumping and ice rafting (Redfield 1972). In many cases, ponds are natural features, and there are historical records of ponds in mid-Atlantic and northeastern US marshes (Harshberger 1916). However, Wilson et al. (2014) documented an increase in marsh ponding between 1938 and 1994 when ditches originally dug to promote drainage were allowed to fill in.

Once created, ponds and mud flats can follow one of several paths, depending on the conditions. For example, these areas can revegetate, particularly if they have been formed by wrack or another short-term disturbance. Ponds may also revegetate if they connect with a tidal channel that allows them to drain. Both ponds and mud flats can be persistent, with little to no change over decades, or they can expand, getting larger and deeper over time. In some cases, they may enter a “run-away expansion path,” which may occur once these features reach a critical minimum width, and wave erosion takes over as the main driver of expansion (Kearney et al. 2002; Mariotti and Fagherazzi 2013). Whether ponds and flats expand over time is influenced by the rate of external sediment deposition in relation to relative SLR (Mariotti 2016).

Finally, the area of vegetated marsh can expand when plants move upslope, encroaching on areas that were formerly uplands (Fig. 1). Marsh migration generally occurs as a result of increased inundation of the marsh-upland border by salt water due to SLR. Over 10 years in Elkhorn Slough, CA, Wasson et al. (2013) measured upslope migration of the marsh-upland boundary by 20 cm in the vertical direction and 100 cm in the horizontal direction. This migration correlated with an increase in inundation time. A steep slope to the upland (Kirwan et al. 2016) and resistant forest vegetation (Field et al. 2016) can slow down or inhibit this horizontal shift. The presence of static structures at the marsh-upland boundary such as shoreline armoring can also prevent this horizontal migration. This condition of limiting the extent of the intertidal area between rising seas at the marsh edge and human development on the upland is often referred to as “coastal squeeze” (Torio and Chmura 2013). Note that it is also possible for upland border to shift into marsh habitat, which might occur when fill is used to build a bulkhead or other structure that encroaches onto the marsh.

Marshes are naturally dynamic in the horizontal direction, and the types of lateral marsh expansion and contraction shown in Fig. 1 may all occur simultaneously. Schieder et al. (2018) demonstrated that marshes along Chesapeake Bay experienced net erosion at the marsh-water boundary while simultaneously shifting horizontally onto the upland. Observed rates of marsh migration onto the upland were largely equal to or greater than the erosion documented at the marsh-water boundary, such that the net effect on marsh extent was neutral over the study period. However, it is rare to see a study that evaluates each of these processes separately.

By analyzing changes across the entire marsh, one can develop a holistic understanding of which types of vegetation loss and gain are occurring and what processes are important in controlling marsh dynamics at a given site.

We present three case studies of historical change in vegetation extent in salt marshes over the last 70 years: one where marsh vegetation was lost, one where the change was net neutral, and one where vegetated area increased. We quantified historical changes in marsh features (channels, mud flats and ponds, and the location of both the seaward and marsh-upland border) to determine how much they each contributed to observed shifts in the overall area of vegetated marsh. We also evaluated the elevation of the various marsh features in the present day as well as their upland slopes to gain insight into the causes of the changes and the potential of each site to adapt to future conditions. This type of information can serve as a guide to potential management actions for marsh protection.

Methods

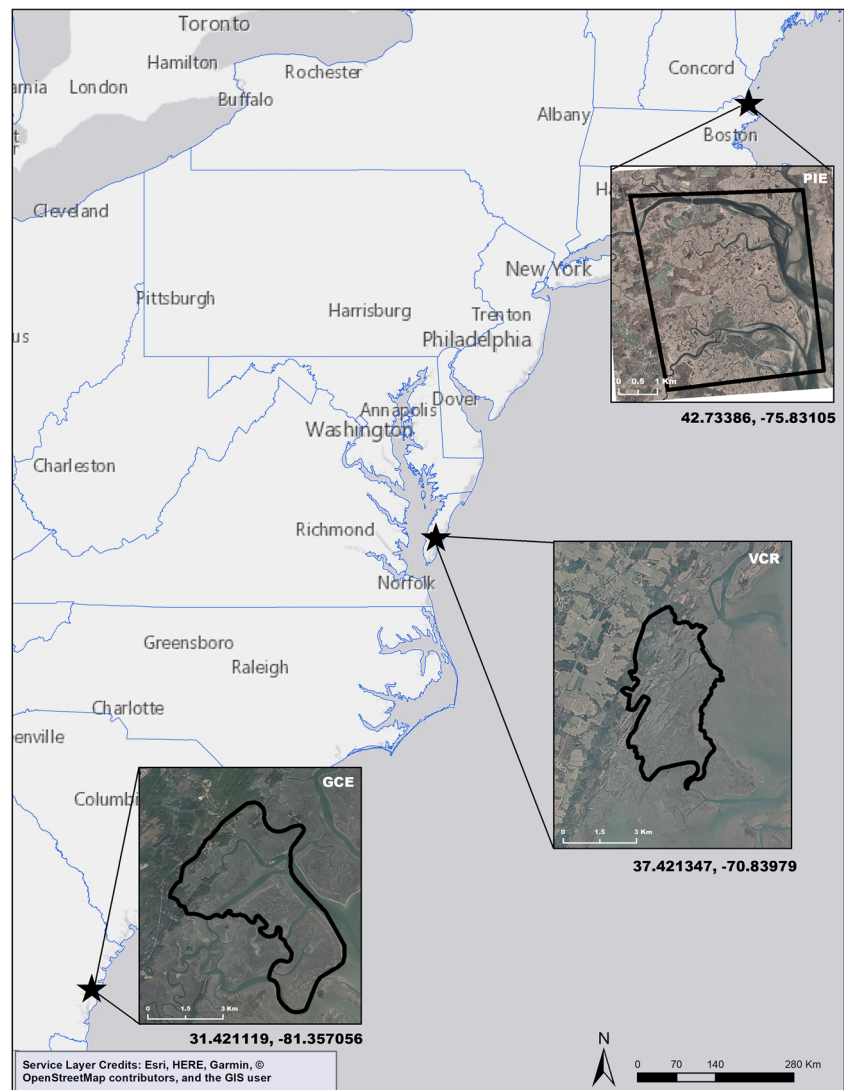
Study Area

Salt marshes located at three long-term ecological research (LTER) sites were used as case studies for this research: Georgia Coastal Ecosystems (GCE), Virginia Coast Reserve (VCR), and Plum Island Ecosystems (PIE). As described below, these marshes vary in tidal range, SLR rate, and sediment supply, and hence we expected that they were likely to exhibit differences in the relative importance of the various types of vegetated marsh change observed over time. We selected a focus area of approximately 18–25 km² for detailed study at each site. The choice of focus area was constrained in part by the boundaries of the LTER site and available aerial imagery, but in each case, it included a portion of the marsh-upland boundary and the main bay or sound.

Georgia Coastal Ecosystems, GA (GCE)

The GCE LTER is situated at the midpoint of the Georgia coastline. The salt marshes within the GCE are typical of those found in the southeastern USA and are dominated by salt marsh cord grass, *Spartina alterniflora*, although *Juncus roemerianus* and other species can be found in the high marsh along the upland transition (Hladik and Alber 2014). This analysis focused on a 25-km² section of largely unmodified marsh adjacent to the mainland behind Sapelo and Wolf Islands that is dominated by *S. alterniflora* (Fig. 2). The suspended sediment concentrations at this site average approximately 40 mg L⁻¹ and range from 3 to 130 mg L⁻¹ (Alber 2018). Tidal and SLR data were derived from the National Oceanic and Atmospheric Administration’s (NOAA) tidal datum for the closest station (Fort Pulaski, Savannah, Station ID 8670870). GCE is considered mesotidal,

Fig. 2 Three LTER study sites with inset maps showing the locations of the Plum Island Ecosystems (PIE), Virginia Coast Reserve (VCR), and Georgia Coastal Ecosystems (GCE), LTER sites along the US east coast. The black line delineates the study area used for this analysis. The PIE study area is 21 km², VCR is 18 km², and GCE is 25 km²



with a mean tidal range of 2.1 m for the 1983–2001 tidal epoch. The relative rate of SLR for the period of this study (1942–2013) was 2.88 ± 0.17 mm year⁻¹, with a more recent (1986–2016) rate of 4.30 ± 0.32 mm year⁻¹ (PMSL 2019).

Virginia Coast Reserve, VA (VCR)

The VCR LTER site is located on the Delmarva Peninsula (Fig. 2) (McLoughlin et al. 2015). We focused on an 18 km² section of salt marsh adjacent to Hog Island Bay, one of a number of shallow bays included in the VCR study area. This site is largely vegetated by *S. alterniflora*. Average suspended sediment concentration is 25 mg L⁻¹ and ranges from 3 to 48 mg L⁻¹ depending on the season (Lawson et al. 2007). This site has a mean tidal range of 1.2 m for the 1983–2001 tidal epoch (Kiptopeke, VA, Station ID 8632200). Relative SLR at this station was 3.53 ± 0.30 mm year⁻¹ for

the period of this study (1949–2013), with a recent rate (1986–2016) of 4.32 ± 0.83 mm year⁻¹ (PMSL 2019).

Plum Island Ecosystems, MA (PIE)

The PIE LTER is located in northeastern Massachusetts. Three major watersheds drain into Plum Island Sound, but upstream dams limit the supply of sediment to the estuary. The mean suspended sediment concentrations are 16 mg L⁻¹ with a range of less than 10 to 40 mg L⁻¹ (Hopkinson et al. 2018). The 21 km² site used in this study is positioned in the salt marshes on the western mainland side of Plum Island Sound (Fig. 2) and is dominated by the high marsh grass, *Spartina patens*. The mean tidal range is 2.9 m for the 1983–2001 tidal epoch (Boston, MA, Station ID #8443970). Relative SLR at this station was 2.47 ± 0.32 mm year⁻¹ for the period of this study (1938 to 2013), with a more recent rate (1986–2016) of 4.67 ± 1.07 mm year⁻¹ (PMSL 2019).

Data Acquisition

Three sets of aerial images were obtained for each site beginning in the late 1930s and 1940s and ending in 2013 (Table 1). GCE imagery was from 1942, 1972, to 2013; VCR imagery was from 1949, 1957, to 2013, and PIE imagery was from 1938, 1971, to 2013. Vegetated marsh as well as other features such as ponds, mud flats, channels, and upland areas was manually digitized in ArcGIS 10.4. Manual delineation of aerial images is a well-established technique of classifying historical marsh extent for long-term studies (Downs et al. 1994; Erwin et al. 2004).

To provide consistency over time and across study sites, one person digitized all of the imagery following a set of digitizing rules. All features were digitized at a scale of 1:500 to 1:1500. We only used low-water imagery to reduce the effects of tidal flooding, and we excluded photographs if reflections made them difficult to interpret. Digitization was also restricted to the center of each aerial photograph where the least amount of warping occurred, and where images overlapped, we only digitized features that were present in both images to ensure that they were stable. Finally, only features that met a 10-m minimum width requirement were included, as those <10 m were too pixelated to consistently delineate in the older imagery.

In addition to the study-wide digitizing rules, adjustments were made as necessary to account for specific circumstances at each site. At GCE, the 1972 dataset was missing one photo along the upland boundary. A 1976 photo was therefore used to fill that gap as it was the closest available imagery in time. This was deemed acceptable as shorelines throughout the rest

of the 1976 photo closely tracked the 1972 shorelines. At VCR, the study area boundary was modified to avoid reflectance and defects in two of the 1949 images which made mud flats impossible to digitize. This resulted in the exclusion of a highly erosional portion of the VCR marsh. At PIE, ponds were often located in groups with a thin, 1 to 2 m wide, barrier between them. As this is not likely to be discernable in the older imagery, these ponds were digitized as single features.

Data Analysis

We included three sources of uncertainty in assessments of the accuracy of the location and size of digitized marsh features: the consistency of the digitizer (D), the pixel resolution (P), and the georectification error (G). These were combined to calculate an error term (E) for the digitized polygons (Eq. 1) (Dolan et al. 1991; Romine et al. 2009).

$$E = \pm\sqrt{D^2 + P^2 + G^2} \quad (1)$$

Digitizer error was assessed by repeat digitization of polygon features, pixel resolution was obtained from the imagery (Table 1), and georectification error was calculated from root mean square error (RMSE) values provided by the ArcGIS software. Since this analysis used polygon rather than line features, the error term was calculated in percent area rather than absolute measurements so that it scaled appropriately with the feature size. Errors for each image ranged from 2 to 5% (Table 1). These terms were used to determine whether errors overlapped when images were compared.

A subtraction analysis was performed using the UNION tool in ArcGIS to evaluate change over time at each site. This tool merges the datasets and creates new polygons, revealing where features have changed. The subtraction analysis was used to show overall changes from the first (1930/40s) to the middle dataset (1950s–1970s), from the middle to the present day (2013), and for the entire time period. The results of the subtraction analysis were used to assess the conversion of vegetated marsh either to or from other features so that we could evaluate net change in vegetated marsh over time. As part of this, we distinguished between changes in vegetated marsh that occurred along interior channels as opposed to those that occurred along the open fetch marsh edge.

DEM Analysis

An existing digital elevation model (DEM) was obtained for each study site to evaluate the elevation of each of the digitized features as well as the slope of the upland transition in the present-day marsh. These DEMs were created from LiDAR elevations obtained for each LTER's specific projects and so were from different years (Table 1). The DEMs for PIE (Hopkinson and Valentine 2017) and VCR (Dewberry and Davis, LLC

Table 1 Description of aerial imagery used for the three time periods at each of the LTER sites considered in this study: Georgia Coastal Ecosystems (GCE), Virginia Coast Reserve (VCR), and Plum Island Ecosystems (PIE). The error term (% Error) is a combination of pixel resolution, georectification, and digitizer error (see text for details). The date of the digital elevation model (DEM) used for each site is also included

Site	Date	Image Type	Scale	% Error
GCE	Nov 28, 1942	Black and white	1:40,000	4%
	Dec 2, 1972	Color aerial photograph	1:20,000	3%
	Early 2013	Orthomosaic	–	3%
VCR	2010	DEM	–	–
	Feb 2, 1949	Black and white	1:20,000	2%
	Oct 20, 1957	Black and white	1:20,000	3%
PIE	Spring 2013	Orthomosaic	–	3%
	April 2015	DEM	–	–
	Nov 1, 1938	Black and white	1:25,000	4%
PIE	June 11, 1971	Black and white	1:20,000	5%
	April 2013	Orthomosaic	–	2%
	April 2005	DEM	–	–

2015) were flown when vegetation was low and had a vertical accuracy of ± 0.20 m at PIE and ± 0.12 m in the vegetated marsh and ± 0.18 m in the unvegetated marsh at VCR. The DEM for the GCE site was corrected to account for the presence of vegetation on the marsh, resulting in a vertical accuracy of ± 0.18 m (Hladik and Alber 2012). The DEMs were all resampled to 10 m, which was the resolution of the coarsest data set.

We used the DEMs to determine the average elevation of the pixels within each of the polygons classified as vegetated marsh, ponds, and mud flats. The elevations were transformed from NAVD88 (m) and standardized relative to a local tidal reference plane based on data from the closest long-term NOAA tide gauge using the equation: (marsh elevation – mean sea level)/(mean higher high water – mean sea level) (Schile et al. 2014). This transformation provides a fractional elevation that can be used for comparisons among sites as it shows where the marsh features fall relative to the upper half of the great diurnal range, which is the difference in height between MHHW and MLLW. With this transformation, a fractional elevation of 1 is equivalent to MHHW and a fractional elevation of 0 is equivalent to MSL. For comparisons with previous studies at these sites, we provide Figs. S1–S4 referenced to NAVD88 in the supplemental materials. We also calculated the slope of the first 50 m of the upland transition along the upland border, as the amount of horizontal area available for marsh expansion onto the adjacent upland increases with decreasing slopes (Kirwan et al. 2016). We considered slopes $<3^\circ$ to be most likely to experience horizontal expansion (Field et al. 2016).

Results

Classification of Marsh Features

The study marshes at GCE, VCR, and PIE all included vegetated marsh, tidal channels, upland area, and either mud flats or ponds. Upland area is in part a function of how the study area boundaries were drawn, and so made up a different proportion of each site. When only the marsh was considered, the vegetated area averaged 64% of the study site at PIE, 74% at GCE, and 80% at VCR. Tidal channel area was a smaller portion of the study area at VCR (14%) than at the other two sites (24% at GCE and 32% at PIE). Mud flats and ponds comprised only a small proportion of the study marshes. GCE had exclusively mud flats (2%), PIE had exclusively ponds (4%), and VCR included both mud flats (6%) and a very small area of ponds ($<1\%$).

Changes at GCE

The areas of the various marsh features remained relatively constant at the GCE site, with overlapping error terms

associated with the values for the earliest (1942) as compared to the latest (2013) datasets (Table 2). The area of vegetated marsh decreased by only 0.23 km^2 (1%) over the study period, with most of that occurring due to a net loss at the edges of interior channels (Table 3, Fig. 3).

Although there was little net change in area, many features within the GCE study site changed their locations over time. Mud flats experienced both expansion and contraction, as some areas converted from vegetated marsh to mud flats while others were colonized by plants. However, the ratio of vegetated marsh:flats remained constant, 41:1, throughout the study, with similar losses and gains within both the early (1942–1972) and late (1972–2013) time intervals examined. The location of the mud flats also shifted between observations, such that only 28% of the mud flat area remained in the same location from 1942 to 2013 (Fig. 3). The open fetch marsh edge along Doboy Sound showed no evidence of erosion. However, interior channels were highly dynamic, experiencing both losses and gains over time. Expansion of interior channels was considerably higher than contraction between 1972 and 2013, leading to a net loss of 0.48 km^2 of vegetated marsh during this later period. In some cases, interior channels extended further into the marsh where they drained existing areas of mud flats. New flats often were then formed, such that the flats maintained their position at the head of the creek (i and ii in Fig. 3).

Change in upland extent was small at GCE, but some interesting changes occurred directly adjacent to the vegetated marsh boundary in an area that was historically farmed (see box in Fig. 3). Between 1942 and 1972, this farm was overtaken by vegetated marsh and 4.7 ha of upland were lost (iii in Fig. 3), and then between 1972 and 2013, 1.3 ha of the recently converted upland area shifted from vegetated marsh to mud flat (iv in Fig. 3). Although it was not a large area, this suggests that marsh migration can and does occur at this site.

Changes at VCR

At the VCR study site, there was no overlap in the error terms associated with any feature categories except ponds when 1949 was compared with 2013 (Table 2). The area of vegetated marsh experienced a net increase due to gains primarily from upland conversion, which outweighed losses resulting from increases in the area of mud flats. When all changes were considered, the net effect was a 0.75 km^2 (7%) increase in vegetated marsh (Table 3, Fig. 4).

The total area of mud flats at the VCR study site had a net increase of 0.48 km^2 (80%) between 1949 and 2013, with most of the change occurring during the later time interval (after 1957) (Table 2). In contrast to GCE, this increase was primarily through the expansion of existing mud flats, as 76% of the area of mud flats remained consistent between 1949 and 2013 (Fig. 4).

Table 2 Area of each feature category for each time period at the three study sites, along with average areas as a percentage of the marsh (the sum of vegetated marsh, tidal channel, interior mud flats and ponds). The

2013 values in bold indicate when the values for the earliest and latest dataset did not have overlapping error terms (see Table 1). NA denotes “not applicable”, as upland was excluded from the percent calculations

Site	Year	Vegetated marsh (km ²)	Tidal channels (km ²)	Mud flats (km ²)	Ponds (km ²)	Upland (km ²)
GCE						
	1942	16.45	5.39	0.40	–	2.81
	1972	16.74	5.21	0.37	–	2.74
	2013	16.22	5.73	0.40	–	2.72
Average % of marsh		74%	24%	2%		NA
VCR						
	1949	10.72	2.14	0.60	0.02	4.33
	1957	11.27	1.97	0.72	0.02	3.76
	2013	11.47	1.85	1.08	0.02	3.28
Average % of marsh		80%	14%	6%	<1%	NA
PIE						
	1938	11.71	5.34	–	0.33	3.55
	1971	11.25	5.44	–	0.71	3.53
	2013	10.55	5.82	–	1.01	3.56
Average % of marsh		64%	32%		4%	NA

Table 3 Losses and gains in vegetated marsh area at the three study sites as determined by subtraction analysis. Results are shown for each time interval along with net change for the entire period. Time period I is the first set of imagery (1942 at GCE, 1949 at VCR and 1938 at PIE), time

period II is the second set of imagery (1972 at GCE, 1957 at VCR, and 1971 at PIE), and time period III is the most recent imagery, which was 2013 for all three sites

Conversion of each feature from (loss) or to (gain) vegetated marsh	Time period I to II			Time period II to III			Time period I to III		
	Loss (km ²)	Gain (km ²)	Net (km ²)	Loss (km ²)	Gain (km ²)	Net (km ²)	Loss (km ²)	Gain (km ²)	Net (km ²)
GCE									
Open fetch edge	–0.01	+0.06	+0.05	–0.04	+0.01	–0.03	–0.03	+0.04	+0.01
Channel edge	–0.72	+0.87	+0.15	–0.72	+0.24	–0.48	–1.15	+0.83	–0.32
Mud flats	–0.24	+0.28	+0.04	–0.25	+0.21	–0.04	–0.25	+0.28	+0.03
Ponds	0	0	0	0	0	0	0	0	0
Upland	–0.13	+0.18	+0.05	–0.07	+0.08	+0.01	–0.12	+0.17	+0.05
GCE total	–1.1	+1.39	+0.29	–1.08	+0.54	–0.54	–1.55	+1.32	–0.23
VCR									
Open fetch edge	–0.06	+0.04	–0.02	–0.05	+0.09	+0.04	–0.06	+0.09	+0.03
Channel edge	–0.23	+0.37	+0.14	–0.31	+0.45	+0.14	–0.30	+0.45	+0.15
Mud flats	–0.28	+0.16	–0.12	–0.51	+0.17	–0.34	–0.59	+0.13	–0.46
Ponds	0	0	0	0	0	0	0	0	0
Upland	–0.09	+0.65	+0.56	–0.07	+0.54	+0.47	–0.02	+1.05	+1.03
VCR total	–0.66	+1.22	+0.56	–0.94	+1.25	+0.31	–0.97	+1.72	+0.75
PIE									
Open fetch edge	–0.02	+0.02	0	–0.08	+0.00	–0.08	–0.09	+0.00	–0.09
Channel edge	–0.40	+0.29	–0.11	–0.50	+0.21	–0.29	–0.51	+0.13	–0.38
Mud flats	0	0	0	0	0	0	0	0	0
Ponds	–0.54	+0.19	–0.35	–0.62	+0.31	–0.31	–0.86	+0.19	–0.67
Upland	–0.29	+0.29	0	–0.24	+0.21	–0.03	–0.21	+0.20	–0.01
PIE total	–1.25	+0.79	–0.46	–1.44	+0.73	–0.71	–1.67	+0.52	–1.15

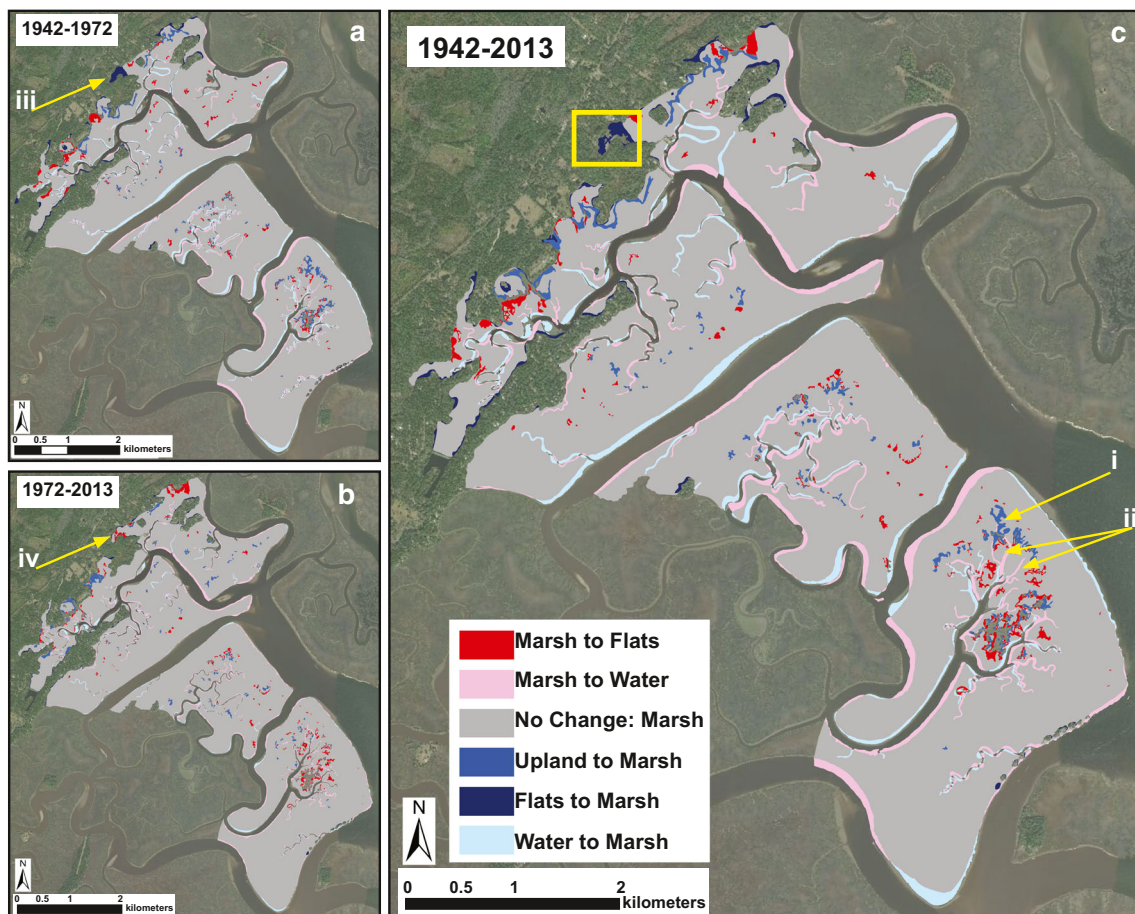


Fig. 3 GCE subtraction analysis showing the change in marsh features between **a** 1942 and 1972, **b** 1972 and 2013, and **c** 1942 and 2013. (i) Indicates the retreat of interior mud flats as the (ii) channels move deeper

into the marsh. The yellow boxes indicate where farmland was converted first to marsh (iii) and then mud flats (iv). Marsh losses and gains refer to changes in the vegetated marsh

Much of the expansion occurred in a few low-lying sections in the interior of the marsh around Castle Ridge Creek and Red Bank Creek; between 1949 and 2013, these two channels narrowed and the surrounding mud flats expanded. Changes in mud flats resulted in a decrease in the ratio of vegetated marsh:flats from 18:1 to 10:1 over time. Concurrent with this expansion in mud flats was a reduction in the area of tidal channels, which were converted into both vegetated marsh and mud flats. Most of this loss occurred in the interior channels of the marsh as opposed to the open fetch edge adjacent to Hog Island Bay.

The VCR study site had large upland islands on the marsh platform in addition to the mainland border (Fig. 4). Between 1949 and 2013, the vegetated marsh had a net increase of 1.03 km² due to horizontal marsh migration (Table 3). Most of this occurred between 1949 and 1957, largely on the borders of the upland islands. Between 1957 and 2013, upland area was lost at the marsh upland boundary as well as at these islands. Over the entire study period, 71% of the marsh migration occurred at the vegetated islands.

Changes at PIE

At the PIE study site, there was no overlap in the error terms associated with 1938 in comparison to 2013 for any feature categories except upland area (Table 2). When all conversions were considered, vegetated marsh had a net decrease of 1.15 km² (10%) over the study period (Table 3), much of which could be attributed to an increase in ponds (Fig. 4). The net increase in ponds was higher between 1938 and 1972 than during the later interval (1972 to 2013). Over the entire period of study, there were net increases in both the number of ponds (1094 to 1484 ponds) and mean pond size (302 to 683 m²). The change detection analysis indicated that 42% of the ponded area in 1938 was still present in 2013, suggesting that pond expansion and initiation of new ponds both played a role in their increase (Fig. 5). The ratio of vegetated marsh:ponds decreased from 35:1 in 1938 to 10:1 in 2013. PIE also experienced vegetated marsh loss as a result of a net increase in tidal channels, with much higher net losses during the second part of the study (−0.11 km² between 1938 and 1972 compared to −0.29 km² between 1972 and 2013).

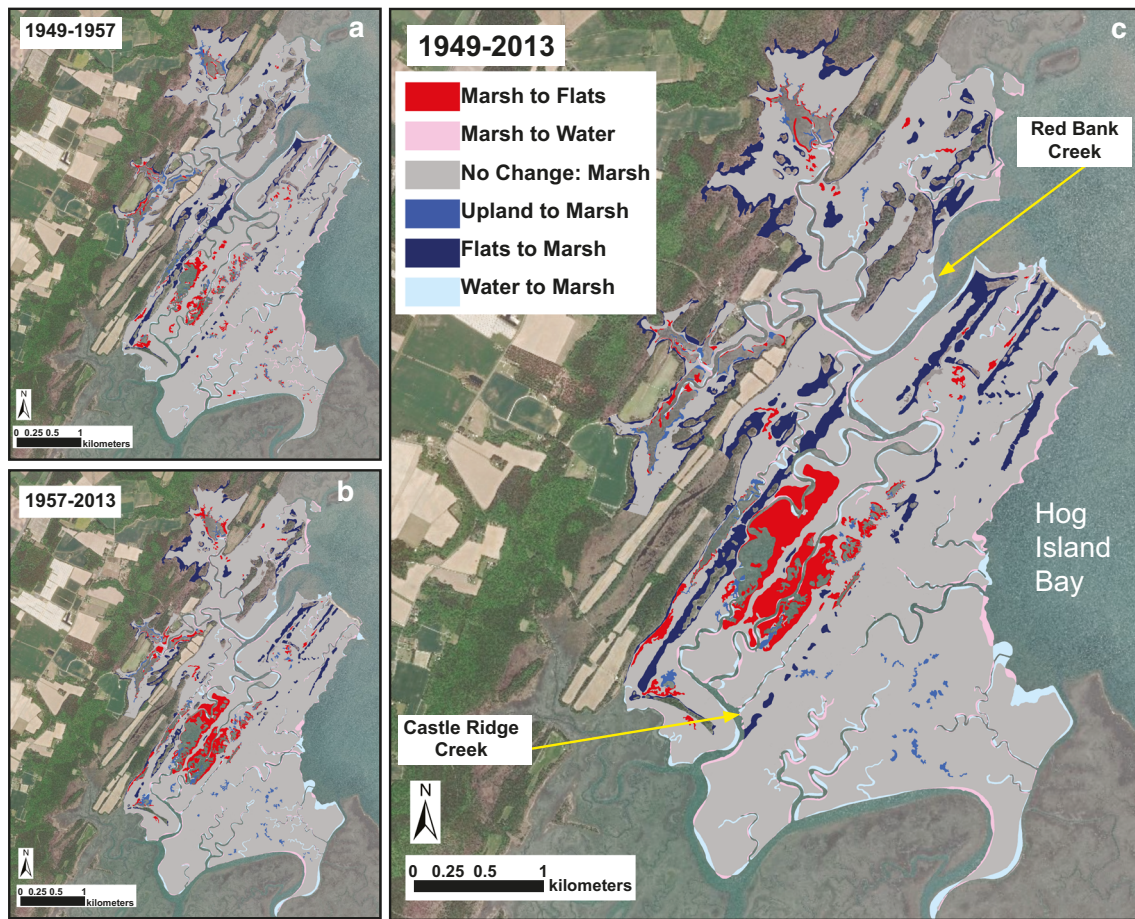


Fig. 4 VCR subtraction analysis showing the changes in marsh features between **a** 1949 and 1957, **b** 1957 and 2013, and **c** 1949 and 2013. Marsh losses and gains refer to changes in the vegetated marsh

Although some of this increase occurred along the open fetch marsh edge of Plum Island Sound, most of the change was due to a slight widening of interior channels. PIE did not experience any change at the marsh-upland border.

DEM Analysis

Vegetated Marsh Elevation

The relative height of the vegetated marsh platform varied considerably among the three sites (Figs. 6 and S1). GCE had the widest distribution of plants within the local tidal frame, ranging from a fractional elevation of -0.4 to 1.4 , with the mode falling between 0.7 and 0.8 and 37% of the marsh located below 0.7 . This value is instructive, as Schile et al. (2014) identified a fractional elevation of 0.65 – 0.75 as the cut-off between high and low marsh habitats in their study sites. The site at VCR sat lower in the tidal frame, with 100% of the marsh falling below 0.7 and the mode occurring between 0.4 and 0.5 . In contrast, the range at PIE was from 0.2 to 1.4 and the mode between 0.9 and 1 , which is just below MHHW, with only 17% of the marsh occurring below 0.7 .

Elevation of Ponds and Flats

At the GCE study site, mud flats occurred at a fractional elevation between -0.3 and 1.3 , which was nearly the full marsh elevation range (Fig. 7a). The biggest mud flats and therefore the largest area occurred below MSL, between -0.2 and -0.3 (Fig. 7b, c). Although the elevation of these large mud flats could not be measured in the historical imagery, they were consistently present in this location across all images. The vertical distribution of mud flats at the VCR study site had a narrower range relative to the local tidal datum (fractional elevations of 0.0 to 0.6) than at GCE, with a peak between 0.1 and 0.2 (Fig. 8a). The mud flats close to MSL (fractional elevation of 0) were much larger than those at higher elevations, and thus the total area exhibited an inverse relationship with elevation (Figs. 8b, c). The distribution of ponds at the PIE study site mirrored the elevation distribution of the vegetated marsh, with the most ponds (number of ponds and total ponded area) falling just below MHHW (fractional elevations of 0.9 – 1.0) (Fig. 9a). Mean pond size peaked around MHHW (Fig. 9b), and total ponded area peaked at just below MHHW (Fig. 9c).

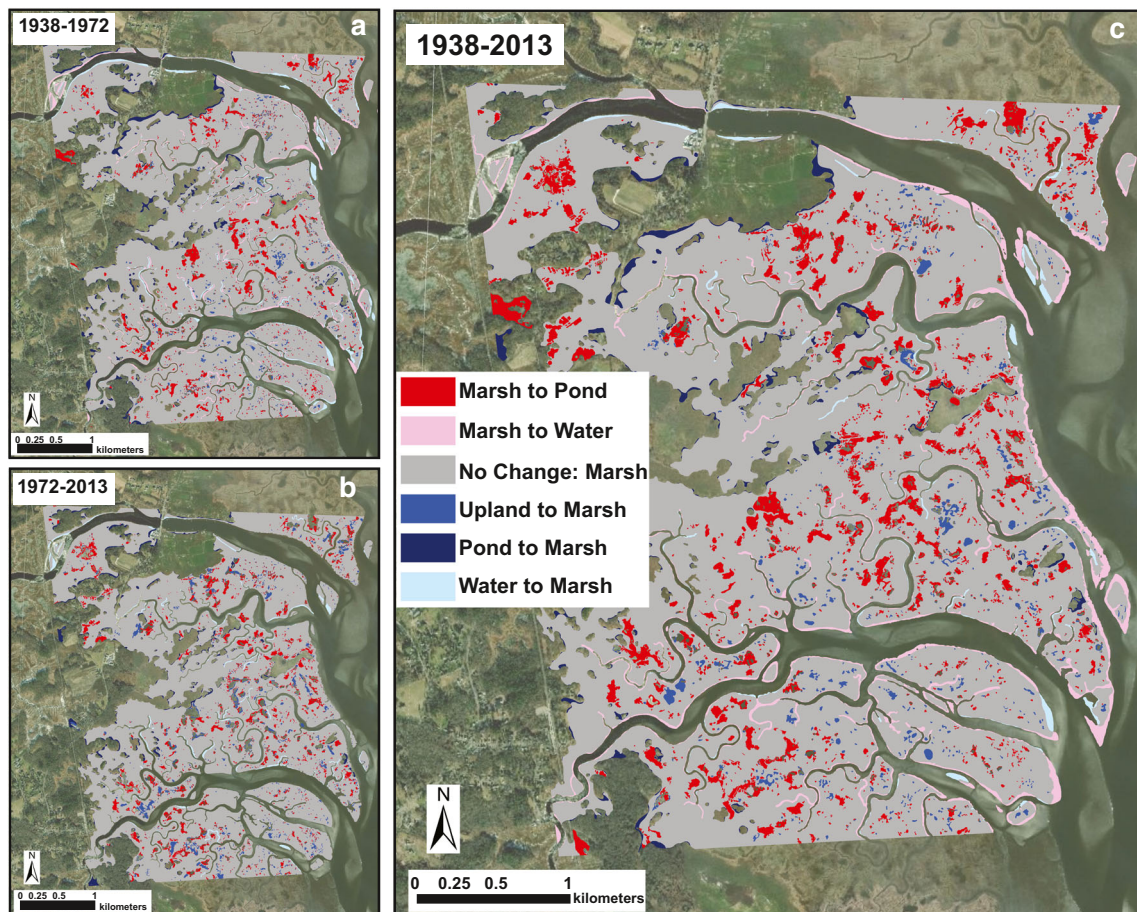


Fig. 5 PIE subtraction analysis showing the changes in marsh features between **a** 1938 and 1972, **b** 1972 and 2013, and **c** 1938 and 2013. Marsh losses and gains refer to changes in the vegetated marsh

Upland Slope

Of the three study sites, VCR had the lowest average upland slope at 0.8° , with a range of 0 to 5° . The slope was $<1^\circ$ along nearly all of the upland border including that of the marsh islands. Slopes at the GCE site averaged 2.24° and ranged from 0 to 18.5° . The upland that had converted to mud flat (box in Fig. 3) had a gentler slope, ranging from $<1^\circ$ to 3° , than the southwestern section, where slopes were $>10^\circ$ and no migration was observed. Slopes at PIE averaged 4.06° and ranged from 0 to 22.1° . At VCR, nearly all (98%) pixels had slopes below 3° , as compared to 75% at GCE and only 46% at PIE.

Discussion

Both the net change in the area of vegetated marsh and the major types of loss and gain over the past 70 years were different for the three marshes we studied: At GCE, there were fairly large shifts over time in the location of mud flats and channels, but little overall net change in vegetated area. At

VCR, there was a loss of vegetated marsh due primarily to mud flat expansion, which was offset by horizontal marsh migration at the upland edge, such that the overall net change was positive. At PIE, there was a net loss in vegetated marsh that was primarily due to an increase in ponding and interior channel expansion. Below, we evaluate the main types of marsh change at these sites and discuss the implications of these observations.

The largest shifts at the GCE were due to changes in the interior channels, which expanded into the vegetated areas (0.83 km^2) in some places over the course of the study period while contracting (1.15 km^2) in others. The changes in interior channels were consistent with our analysis of shoreline change, which showed high rates of both advance and retreat along channel edges (Burns 2018). However, the fact that edge retreat outpaced advances during the second half of the study (1972 to 2013) may be an indication of channel widening due to an increase in the tidal prism as a result of SLR. There were also changes in the locations of mud flats over time at GCE, with more than 70% of the flats present in 1942 switching to vegetated marsh in 2013, which was offset by vegetated marsh converting to mud flats elsewhere. The

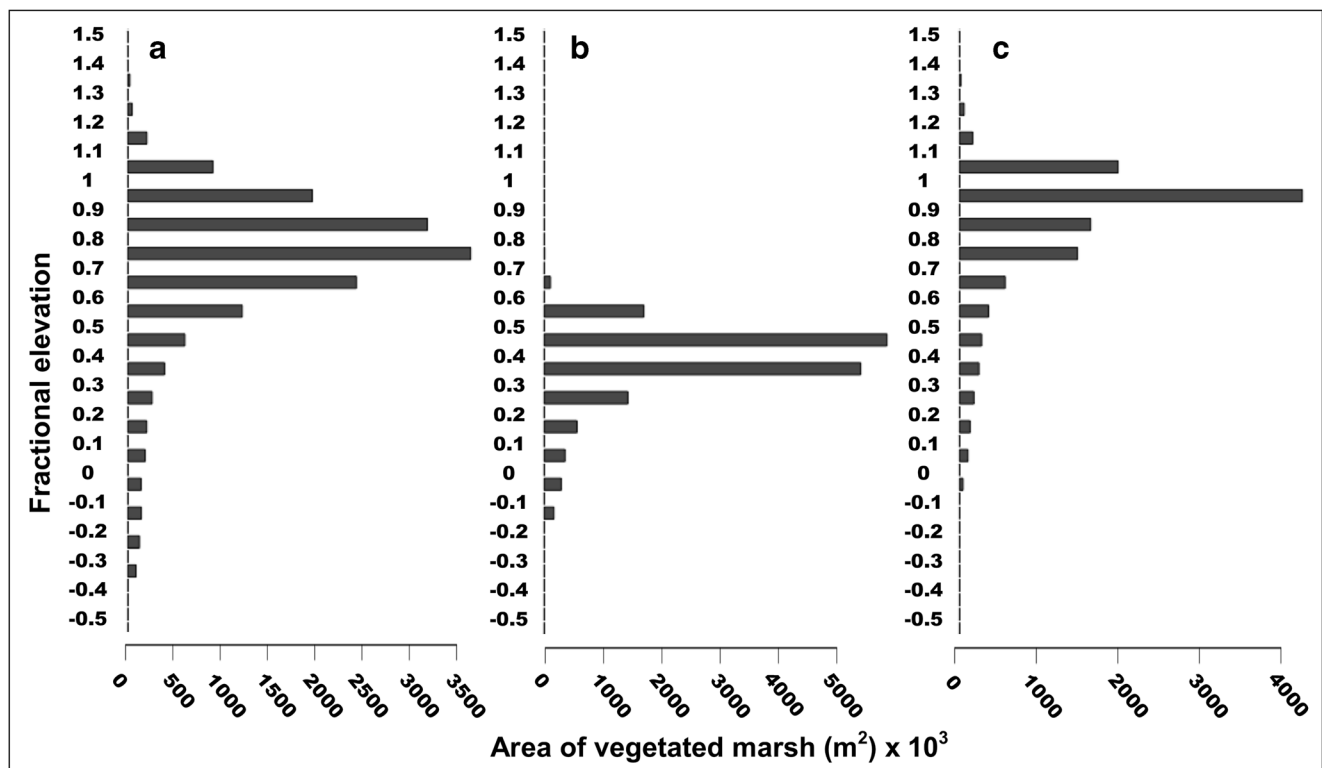


Fig. 6 Frequency distribution of vegetated marsh with respect to fractional elevation at the **a** GCE, **b** VCR, and **c** PIE study sites. Fractional elevation calculated as $(\text{elevation}-\text{MSL})/(\text{MHHW}-\text{MSL})$ in relation to the reference tidal plane at the closest long-term NOAA tide

gauge at each site (see text for details). A fractional elevation value of 1 falls at MHHW and a value of 0 falls at MSL for each site. See Fig. S1 for observations referenced to NAVD 88

fact that these interior flats did not expand over time and occurred across the entire elevation range of the GCE study site suggests they are not likely to have been caused by waterlogging of low-lying areas. Many of the bare areas may have been due to wrack deposition, which can smother plants if left in place for long enough and is known to cause patches within the marsh (Hartman 1988). Wrack is most likely the cause of the mud flats at the marsh-upland boundary, where *J. roemerianus* dominates, as this grass is well known for its wrack-trapping abilities (Pennings and Richards 1998). Other mud flats may have been created by sudden dieback. This phenomenon is not well-understood but it is known to be associated with drought (Alber et al. 2008) and so may be increasing in frequency. Finally, some of the flats located at the heads of creeks are likely evidence of crab activity, which may be facilitating increased drainage of these areas (Hughes et al. 2009; Perillo et al. 1996; Vu et al. 2017).

The total mud flat area at the VCR site nearly doubled from 1949 to 2013. Although some were revegetated over time, those that were low in elevation with respect to MSL showed evidence of marsh waterlogging that is likely linked to SLR. Two of these mud flats expanded to the extent that they achieved the critical minimum width for runaway expansion as modeled by Marriotti and Fagherazzi (2013). Compensating for this loss was a net expansion of vegetated

marsh due to horizontal migration. The VCR study marsh had the lowest mean slope at the upland edge out of the three sites as well as the lowest range in slope. An earlier study at this study site documented low upland slopes and horizontal marsh migration (Kastler and Wiberg 1996), as have other studies in Chesapeake (Schieder et al. 2018) and Delaware Bays (Smith 2013). These studies also note that horizontal migration onto the upland often exceeds marsh loss in other areas. In fact, without marsh migration, the VCR site would have lost 0.28 km² of vegetated marsh rather than gained 0.75 km² over the study period.

Total ponded area at PIE tripled over the 70-year study period. Contrary to the expanding mud flats at VCR, which were positioned just above MSL, the ponds at PIE were positioned closer to MHHW. Although ponding may be indicative of drowning in many marshes (Downs et al. 1994), they are natural in this system (Harshberger 1916) and so understanding the base conditions is critical to determining whether or not ponding is truly a warning sign of degradation (Wilson et al. 2014; Kearney et al. 2002). Many New England marshes, including those at PIE, were heavily ditched for mosquito control from the early 1900s through the 1930s (Adamowicz and Roman 2005), and previous studies suggest this led to over-drainage (Redfield 1972). These ditches are now filling in, creating ponds (Wilson et al. 2014; Adamowicz

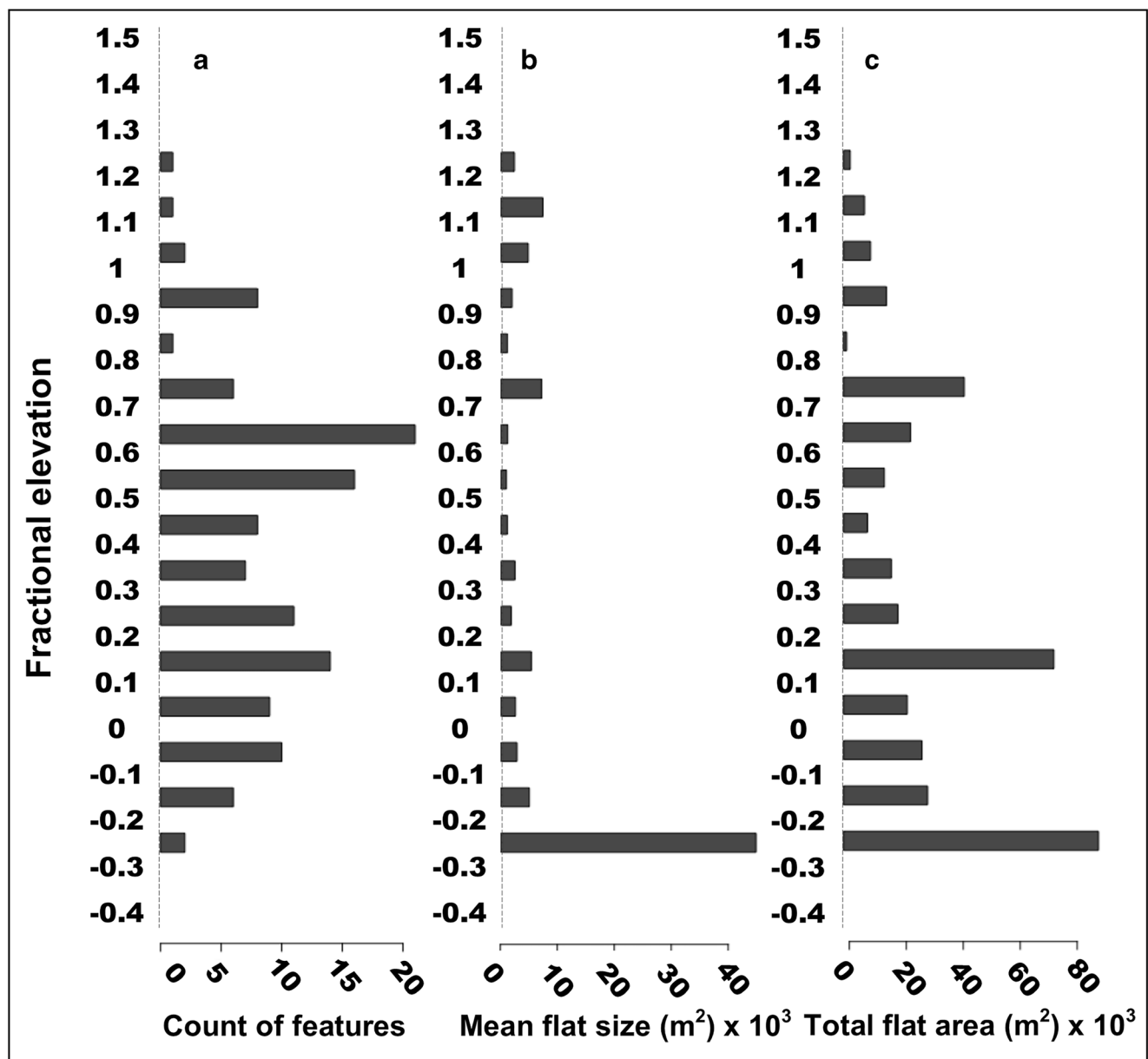


Fig. 7 Characteristics of interior mud flat features as a function of fractional elevation at the GCE study site, calculated as in Fig. 6. Frequency of a number of flats, **b** mean flat size, and **c** total mud flat area per interval. See Fig. S2 for observations referenced to NAVD88

and Roman 2005). It is possible that this process has stabilized: in an analysis of ponds at PIE, Wilson et al. (2014) found that ponding density was $0.12 \text{ m}^2 \text{ m}^{-2}$ marsh in 1994 and $0.10 \text{ m}^2 \text{ m}^{-2}$ in 2008, which was the final year of their study. We measured a ponding density of $0.10 \text{ m}^2 \text{ m}^{-2}$ in 2013 in the same sub-area. Moreover, our observations showed that vegetated marsh had higher losses to ponds during the early part of the study.

Another source of vegetated marsh loss at PIE occurred at the edges of interior channels. This is consistent with our analysis of shoreline change at PIE, which documented low-level erosion along the creek edges throughout the entire marsh (Burns 2018). This loss more than doubled during the

second part of the study and is likely an effect of SLR. Mariotti (2016) suggested that SLR has a unique “widening footprint” in the marsh, which is reflected by spatially uniform widening along the channels at PIE. There was also some loss along the open fetch edge. This may be a result of an increase in the frequency and intensity of storms over time (Hayden and Hayden 2003), which would bring more high energy wind waves to the coast and increase erosion.

Although this was a historical analysis, all three sites have experienced increasing rates of SLR over time and further increases are expected in the future (Oppenheimer et al. 2019). SLR ranged from to 2.5 to 3.5 mm year^{-1} when calculated for the entire study period as compared to more recent rates (1986–

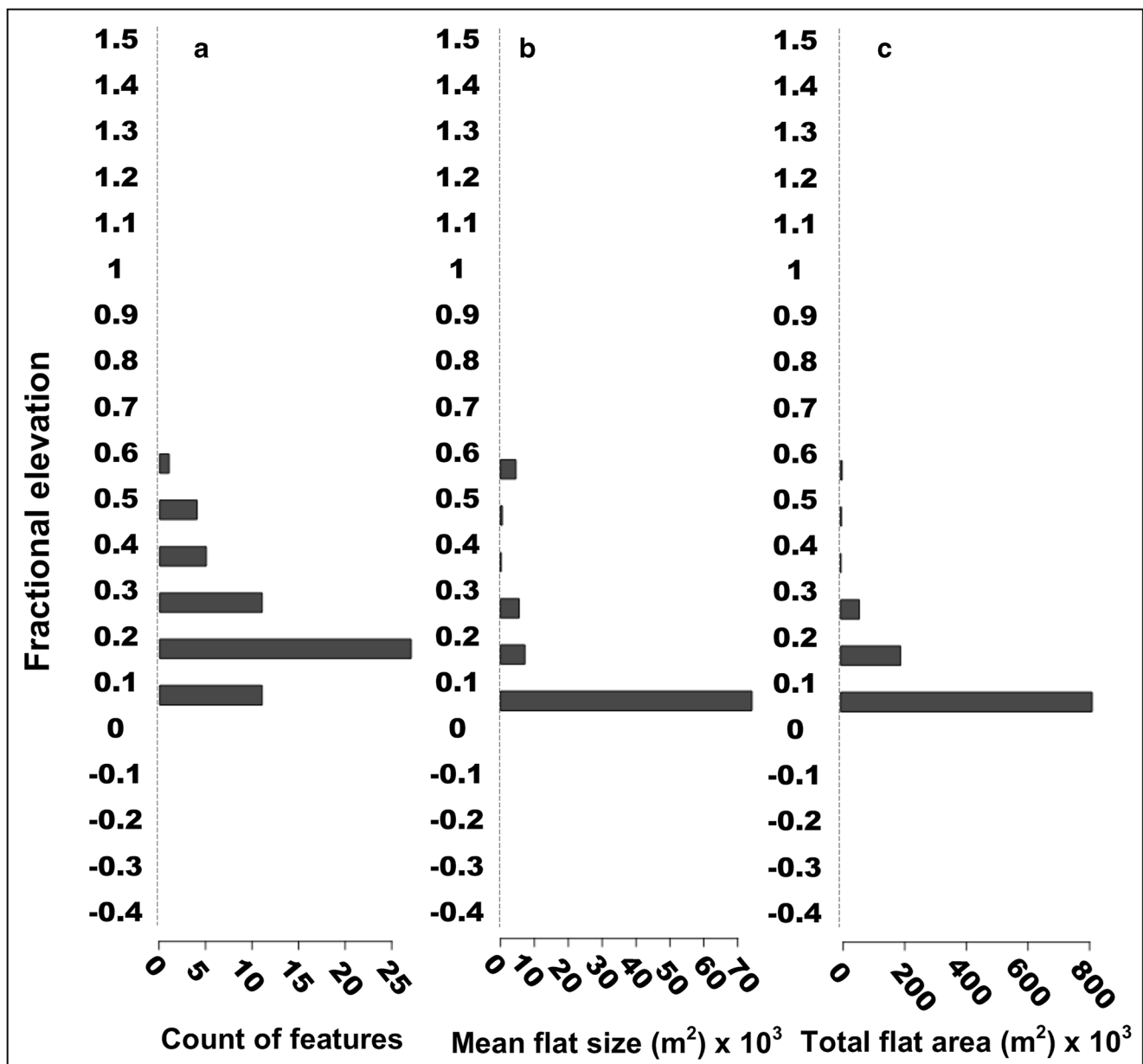


Fig. 8 Characteristics of interior mud flat features as a function of fractional elevation at the VCR study site, calculated as in Fig. 6. Frequency of a number of flats, **b** mean flat size, and **c** total mud flat area per interval. See Fig. S3 for observations referenced to NAVD88

2016) of 4.3 and 4.7 mm year⁻¹ (see site descriptions). As sea levels continue to rise, we would expect to see further losses of vegetated marsh due to increases in the size of their interior channels at both GCE and PIE. At VCR, we would expect to see losses due to increases in interior flats. Much of the vegetated marsh at this site is situated low in the tidal frame, making it vulnerable to future to increased flooding. Moreover, the expansion of mud flats is likely to continue given the potential for wind-wave erosion and the low sediment concentrations. Although the site has low upland slopes that are conducive to horizontal migration, Kirwan et al. (2016) predicted that horizontal marsh expansion will exceed horizontal marsh loss at SLR rates between 8 and 9 mm year⁻¹.

This analysis also provides a context for developing site-specific management options. At GCE, the increase in interior channels is something that should be evaluated regularly. One recommendation here would be to recognize that sediment is being redistributed within the system, such that stabilizing channel edges could result in losses in other areas (Ganju 2019). This would therefore be an important consideration for a potential erosion control project. The GCE site has low enough slopes in some areas for the marsh to migrate horizontally, and in fact, one small section of upland was converted to vegetated marsh during the time period. Much of the upland at this site is currently agricultural and forested land, and so targeting these areas for protection would be another potential strategy.

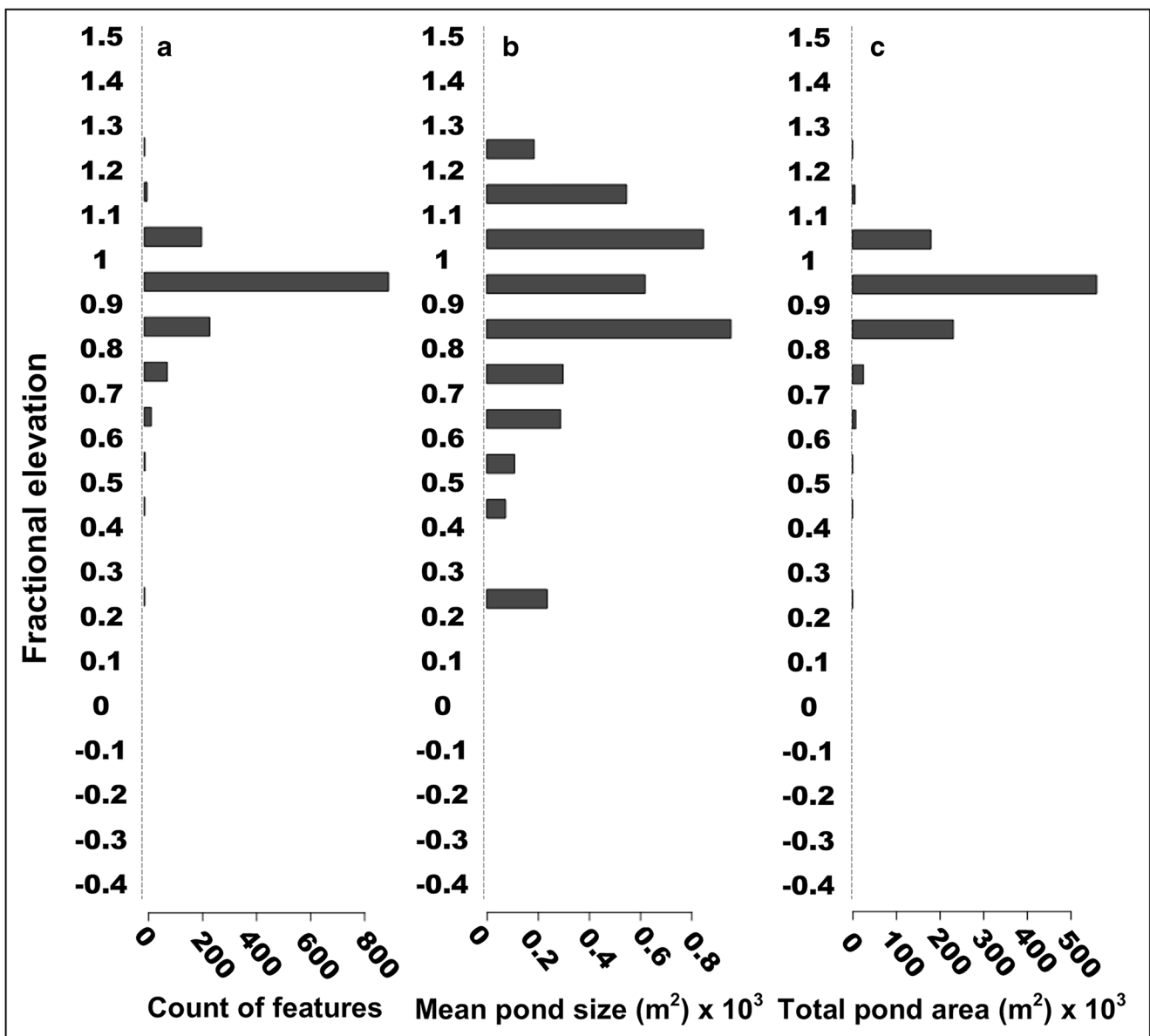


Fig. 9 Characteristics of pond features as a function of fractional elevation at the PIE study site, calculated as in Fig. 6. Frequency of a number of ponds, b mean pond size, and c total ponded area per interval. See Fig. S4 for observations referenced to NAVD88

At VCR, where the net gain in vegetated marsh was almost completely the result of marsh migration onto the upland, preventing barriers such as seawalls at the upland edge is already underway. The marsh to upland boundary at this site is dominated by forest and agriculture, and much of it has been protected under easements (McLoughlin et al. 2015). The expanding mud flats are difficult to address with management actions, given the low sediment concentrations in this area. However, thin layer placement of dredged material has been a successful management strategy in some marshes such as Gulf Rock, NC (Wilber 1992), and Bayou Lafourche, LA (Stagg and Mendelssohn 2011). Although this option has not been discussed within this specific study area, there are active plans to do so in nearby Cedar Island (USACE 2019).

The primary cause of vegetated marsh loss at PIE was due to an increase in ponding, which may have stabilized in this area. However, the study site also lost vegetated marsh due to channel widening, and Hopkinson et al. (2018) suggested that sediment eroded from creek edges serves as an important source for vertical accretion on the marsh platform at PIE. This would be difficult to address through management actions, but it might be possible to increase suspended sediment input by removing some of the hundreds of upstream dams in this area. Dam removal is being pursued, largely to restore access for anadromous fish, but most projects are still in the early stages and few dams have been removed to date (A. Giblin, personal communication).

This study offers a reminder that marshes are dynamic environments, with multiple processes occurring simultaneously that affect the extent of vegetated habitat. The three sites that we examined here all had both losses and gains in all categories over time, with three different net outcomes for marsh vegetation. Identifying changes across the entire marsh, along with information on the distribution of the various features within the tidal frame, provides insight into what factors are important in driving vegetation losses and gains over time. It also provides a useful context for identifying site-specific management options.

Acknowledgements We thank Mike Robinson, John Porter, Anne Giblin, Matt Kirwan, John Porter, and Hap Garritt for providing the imagery and access to the DEM data used in this study, and Marguerite Madden, Chuck Hopkinson, and two anonymous reviewers for comments on the manuscript. We acknowledge the support of the NSF-funded Coastal SEES (NSF14-26308), the Georgia Coastal Ecosystems LTER (OCE18-32178), VCR (DEB 1832221), and PIE (OCE 1637630). The authors declare that they have no conflict of interest. This is contribution number 1086 of the University of Georgia Marine Institute.

References

- Adamowicz, S.C., and C.T. Roman. 2005. New England salt marsh pools: A quantitative analysis of geomorphic and geographic features. *Wetlands* 25 (2): 279–288.
- Alber, M. 2018. Long-term water quality monitoring in the Altamaha, Doboy and Sapelo sounds and the Duplin River near Sapelo Island, Georgia from November 2013 to December 2015. Georgia Coastal Ecosystems LTER Project, University of Georgia, Long Term Ecological Research Network. <https://doi.org/10.6073/pasta/c55e5b0e279ea54151517cabe894a44b>
- Alber, M., E.M. Swenson, S.C. Adamowicz, and I.A. Mendelssohn. 2008. Salt marsh dieback: An overview of recent events in the US. *Estuarine, Coastal, and Shelf Science* 80 (1): 1–11.
- Barbier, E.B., S.D. Hacker, C. Kennedy, E.W. Koch, A.C. Stier, and B.R. Silliman. 2011. The value of estuarine and coastal ecosystem services. *Ecological Monographs* 81 (2): 169–193.
- Browne, J.P. 2017. Long-term erosional trends along channelized salt marsh edges. *Estuaries and Coasts* 40 (6): 1566–1575.
- Burns, C. 2018. Historical analysis of 70 years of salt marsh change at three coastal LTER sites. M.S. Thesis. University of Georgia, Athens, GA.
- Church, J.A., and N.J. White. 2011. Sea-level rise from the late 19th to the early 21st century. *Surveys in Geophysics* 32 (4–5): 585–602.
- Coverdale, T.C., N.C. Herrmann, A.H. Altieri, and M.D. Bertness. 2013. Latent impacts: The role of historical human activity in coastal habitats. *Frontiers in Ecology and the Environment* 11 (2): 69–74.
- Crotty, S.M., C. Angelini, and M.D. Bertness. 2017. Multiple stressors and the potential for synergistic loss of New England salt marshes. *PLoS One* 12 (8): e0183058.
- D'Alpaos, A., S. Lansoni, M. Marani, and A., Rinaldo. 2010. On the tidal prism-channel area relations. *Journal of Geophysical Research* 115(F1).
- Day, J.W., F. Scarton, A. Rismondo, and D. Are. 1998. Rapid deterioration of a salt marsh in Venice lagoon, Italy. *Journal of Coastal Research* 14 (2): 583–590.
- Dewberry and Davis LLC. 2015. Eastern Shore Virginia QL2 LiDAR BBA: Report Produced for US. Geological Society.
- Dolan, R., M. Fenster, and S.J. Home. 1991. Temporal analysis of shoreline recession and accretion. *Journal of Coastal Research* 7(3): 723–744.
- Downs, L.L., R.J. Nicholls, S.P. Leatherman, and J. Hautzenroder. 1994. Historic evolution of a marsh island: Bloodsworth Island, Maryland. *Journal of Coastal Research* 10 (4): 1031–1044.
- Eisma, D. 1998. *Intertidal deposits: River mouths, tidal flats, and coastal lagoons*. Boca Raton: CRC Press.
- Erwin, R.M., G.M. Sanders, and D.J. Prosser. 2004. Changes in lagoonal marsh morphology at selected northeastern Atlantic coast sites of significance to migratory water birds. *Wetlands* 24 (4): 891–903.
- Fagherazzi, S. 2013. The ephemeral life of a salt marsh. *Geology* 41 (8): 943–944.
- Fagherazzi, S., G. Mariotti, P.L. Wiberg, and K.J. McGlathery. 2013. Marsh collapse does not require sea level rise. *Oceanography* 26 (3): 70–77.
- Field, C.R., C. Gjerdrum, and C.S. Elphick. 2016. Forest resistance to sea-level rise prevents landward migration of tidal marsh. *Biological Conservation* 201: 363–369.
- Ganju, N.K. 2019. Marshes are the new beaches: Integrating sediment transport into restoration planning. *Estuaries and Coasts*: 1–10.
- Ganju, N., Z. Defne, M.L. Kirwan, S. Fagherazzi, A. D'Alpaos, and L. Caniello. 2017. Spatially integrative metrics reveal hidden vulnerability of microtidal salt marshes. *Nature Communications* 8 (1): 14156.
- Harshberger, J.W. 1916. The origin and vegetation of salt marsh pools. *Proceedings of the American Philosophical Society*. 55 (6): 481–484.
- Hartman, J.M. 1988. Recolonization of small disturbance patches in a New England salt marsh. *American Journal of Botany* 75 (11): 1625–1631.
- Hayden, B.P., and N.R. Hayden. 2003. Decadal and century-long changes in storminess at long-term ecological research sites. In *Climate variability and ecosystem response at long-term ecological research sites*, ed. D. Greenland, D.G. Goodin, and R.S. Smith, 262–285. New York: Oxford University Press.
- Hladik, C., and M. Alber. 2012. Accuracy assessment and correction of a LIDAR-derived salt marsh digital elevation model. *Remote Sensing of Environment* 121: 224–235.
- Hladik, C., and M. Alber. 2014. Classification of salt marsh vegetation using edaphic and remote sensing-derived variables. *Estuarine, Coastal, and Shelf-Science* 141: 47–57.
- Hopkinson, C., and V. Valentine. 2017. PIE LTER 2005 Digital elevation model for the Plum Island Sound estuary, Massachusetts, filtered grid, last filtered grid - Raster. *Environmental Data Initiative*. <https://doi.org/10.6073/pasta/2a3cdfb8687ff511e599bcaa8e83482d> Dataset accessed 7/23/2018.
- Hopkinson, C., J.T. Morris, S. Fagherazzi, W.M. Wollheim, and P.A. Raymond. 2018. Lateral marsh edge erosion as a source of sediments for vertical marsh accretion. *Journal of Geophysical Research: Biogeoscience* 123: 2444–2465.
- Hughes, Z.J., D.M. FitzGerald, C.A. Wilson, S.C. Pennings, K. Wieszki, and A. Mahadevan. 2009. Rapid headward erosion of marsh creeks in response to relative sea level rise. *Geophysical Research Letters* 36 (3).
- Kastler, J.A., and P.L. Wiberg. 1996. Sedimentation and boundary changes in Virginia salt marshes. *Estuarine, Coastal and Shelf Science* 42 (6): 683–700.
- Keamey, M.S., R.E. Grace, and J.C. Stevenson. 1988. Marsh loss in the Nanticoke estuary, Chesapeake Bay. *Geographical Review* 78 (2): 205–220.

- Kearney, M.S., A.S. Rogers, J.R. Townshend, E. Rizzo, D. Stutzer, J.C. Stevenson, and K. Sundborg. 2002. Landsat imagery shows decline of coastal marshes in Chesapeake and Delaware Bays. *Eos, Transactions American Geophysical Union* 83 (16): 173–178.
- Kirwan, M.L., D.C. Walters, W.G. Reay, and J.A. Carr. 2016. Sea level driven marsh expansion in a coupled model of marsh erosion and migration. *Geophysical Research Letters* 43 (9): 4366–4373.
- Lawson, S.E., P.L. Wiberg, K.J. McGlathery, and D.C. Fugate. 2007. Wind-driven sediment suspension controls light availability in a shallow coastal lagoon. *Estuaries and Coasts* 30 (1): 102–112.
- Mariotti, G. 2016. Revisiting salt marsh resilience to sea level rise: Are ponds responsible for permanent land loss? *Journal of Geophysical Research: Earth Surface* 121 (7): 1391–1407.
- Mariotti, G., and S. Fagherazzi. 2013. Critical width of tidal flats triggers marsh collapse in the absence of sea-level rise. *Proceedings of the National Academy of Sciences* 110 (14): 5353–5356.
- McLoughlin, S.M., P.L. Wiberg, I. Safak, and K.J. McGlathery. 2015. Rates and forcing of marsh edge erosion in a shallow coastal bay. *Estuaries and Coasts* 38 (2): 620–638.
- Morris, J.T., P.V. Sundareshwar, C.T. Nietch, B. Kjerfve, and D.R. Cahoon. 2002. Responses of coastal wetlands to rising sea level. *Ecology* 83 (10): 2869–2877.
- Oppenheimer, M., B.C. Glavovic, J. Hinkel, R. van de Wal, A.K. Magnan, A. Abd-Elgawad, R. Cai, M. Cifuentes-Jara, R.M. DeConto, T. Ghosh, J. Hay, F. Isla, B. Marzeion, B. Meyssignac, and Z. Sebesvari. 2019. Sea level rise and implications for low-lying islands, coasts and communities. In *IPCC Special Report on the Ocean and Cryosphere in a Changing Climate*, ed. H.-O. Pörtner, D.C. Roberts, V. Masson-Delmotte, P. Zhai, M. Tignor, E. Poloczanska, K. Mintenbeck, A. Alegría, M. Nicolai, A. Okem, J. Petzold, B. Rama, and N.M. Weyer. In press.
- Passeri, D.L., S.C. Hagen, S.C. Medeiros, M.V. Bilskie, K. Alizad, and D. Wang. 2015. The dynamic effects of sea level rise on low-gradient coastal landscapes: A review. *Earth's Future* 3 (6): 159–181.
- Pennings, S.C., and C.L. Richards. 1998. Effects of wrack burial in salt-stressed habitats: *Batis maritima* in a Southwest Atlantic salt marsh. *Ecography* 21 (6): 630–638.
- Perillo, G.E., M. Ripley, M.C. Piccolo, and K. Dyer. 1996. The formation of tidal creeks in a salt marsh: New evidence from the Loyala Bay salt marsh, Rio Gallegos estuary, Argentina. *Mangroves and Salt Marshes* 1 (1): 37–46.
- PMSL 2019, Permanent Service for Mean Sea Level, National Oceanography Centre, Liverpool, England. <https://www.psmsl.org/data/> Accessed 10/18/19
- Redfield, A.C. 1972. Development of a New England salt marsh. *Ecological Monographs* 42 (2): 201–237.
- Romine, B.M., C.H. Fletcher, L.N. Frazer, A.S. Genz, M.M. Barbee, and S.C. Lim. 2009. Historical shoreline change, southeast Oahu, Hawaii; applying polynomial models to calculate shoreline change rates. *Journal of Coastal Research* 1236–1253.
- Schepers, L., M. Kirwan, G. Guntenspergen, and S. Temmerman. 2017. Spatio-temporal development of vegetation die-off in a submerging coastal marsh. *Limnology and Oceanography* 62 (1): 137–150.
- Schieder, N.W., D.C. Walters, and M.L. Kirwan. 2018. Massive upland to wetland conversion compensated for historical marsh loss in Chesapeake Bay, USA. *Estuaries and Coasts* 41 (4): 940–951.
- Schile, L.M., J.C. Callaway, J.T. Morris, D. Stralberg, V.T. Parker, and M. Kelly. 2014. Modeling tidal marsh distribution with sea-level rise: Evaluating the role of vegetation, sediment, and upland habitat in marsh resiliency. *PLoS One* 9 (2): e88760.
- Schwimmer, R.A. 2001. Rates and processes of marsh shoreline erosion in Rehoboth Bay, Delaware, USA. *Journal of Coastal Research* 17 (3): 672–683.
- Seminara, G. 2006. Meanders. *Journal of Fluid Mechanics* 554 (1): 271–297.
- Smith, J.A. 2013. The role of *Phragmites australis* in mediating inland salt marsh migration in a mid-Atlantic estuary. *PLoS One* 8 (5): e65091.
- Stagg, C.L., and I.A. Mendelsohn. 2011. Controls on resilience and stability in a sediment-subsidized salt marsh. *Ecological Applications* 21 (5): 1731–1744.
- Temmerman, S., P. Meire, T.J. Bouma, P.M. Herman, T. Ysebaert, and H.J. De Vriend. 2013. Ecosystem-based coastal defense in the face of global change. *Nature* 504 (80): 79–83.
- Torio, D.D., and G.L. Chmura. 2013. Assessing coastal squeeze of tidal wetlands. *Journal of Coastal Research* 29 (5): 1049–1061.
- USACE. 2019. Cedar Island beneficial use of dredged material. US Army Corps of Engineers Norfolk District. <https://www.nao.usace.army.mil/About/Projects/Cedar-Island-CAP-204/>.
- Vu, H.D., K. Więski, and S. Pennings. 2017. Ecosystem engineers drive creek formation in salt marshes. *Ecology* 98 (1): 162–174.
- Wasson, K., A. Woolfolk, and C. Fresquez. 2013. Ecotones as indicators of changing environmental conditions: Rapid migration of salt marsh-upland boundaries. *Estuaries and Coasts* 36 (3): 654–664.
- Watson, E.B., C. Wigland, E.W. Davey, H.M. Andrews, J. Bishop, and K.B. Raposa. 2017. Wetland loss patterns and inundation-productivity relationships prognosticate widespread salt marsh loss for southern New England. *Estuaries and Coasts* 40 (3): 662–681.
- Wilber, P. 1992. Case studies of the thin-layer disposal of dredged material—Gull Rock, North Carolina. *Environmental Effects of Dredging* D-92-3.
- Wilson, C.A., Z.J. Hughes, D.M. FitzGerald, C.S. Hopkinson, V. Valentine, and A.S. Kolker. 2014. Saltmarsh pool and tidal creek morphodynamics: Dynamic equilibrium of northern latitude saltmarshes. *Geomorphology* 213: 99–115.