Sediment accumulation, elevation change, and the vulnerability of tidal marshes in the Delaware Estuary and Barnegat Bay to accelerated sea level rise

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Abstract

Tidal marshes protect coastal communities from the effects of sea level rise and storms, yet they are vulnerable to prolonged inundation and submergence. Uncertainty regarding their vulnerability to sea level rise motivated the establishment of a monitoring network in the Delaware Estuary and Barnegat Bay. Using data collected through these efforts, we determined whether rates of tidal marsh sediment accumulation and elevation change exceeded local sea level rise and how these dynamics varied along geographic and environmental gradients. Marker horizons, surface elevation tables, elevation surveys, water level data, and water column suspended sediment concentrations were used to evaluate sea level rise vulnerability. Of 32 study sites, 75% had elevation change that did not keep pace with long-term rising sea levels (1969–2018) and 94% did not keep pace with recent sea level rise (2000–2018). Mean high water rose most rapidly in the freshwater tidal portion of the Delaware Estuary with rates nearing 1 cm yr^{-1} from 2000–2018. We noted that greater sediment accumulation rates occurred in marshes with large tidal ranges, low elevations, and high water column suspended sediment concentrations. We found correlations between rates of shallow subsidence, increasing salinity, and decreasing tidal range. Marsh elevation and water level surveys revealed significant variability in elevation capital and summer flooding patterns (12–67% inundation). However, rapid increases in mean high water over the past 19 years suggests that all marsh platforms currently sit at or below mean high water. Overall, these data suggest that tidal marshes in the Delaware Estuary and Barnegat Bay are vulnerable to submergence by current rates of sea-level rise. While we observed variability in marsh elevation capital, the absence of strong correlations between elevation trends and environmental parameters makes it difficult to identify clear patterns of sea level rise vulnerability among wetlands.

Introduction

Tidal marshes can moderate some of the impacts of climate change including storm surge and wave attenuation, nutrient uptake and removal through denitrification, and mitigation of greenhouse gas emissions through carbon sequestration [1–3]. An analysis of damage caused by Hurricane Sandy in the U. S. Mid-Atlantic suggested that intact tidal marshes reduced flood damages by more than US $625 million and lower annual flood risks around Barnegat Bay, NJ by up to 70% [4]. Ensuring the existence of tidal marshes in coastal areas bordering high-value infrastructure over the coming decades is an important part of protecting coastal communities from increased flood risk due to sea level rise (SLR) and extreme weather events like hurricanes [4–6].

Despite being assets for coastal community protection, tidal marshes are vulnerable to the impacts of SLR, especially when additional anthropogenic stressors reduce their elevation or accretionary capacity, such as declining sediment inputs and altered hydrology [7–10]. Accumulation of plant material in the soil and sediment accumulation interact to build elevation through dynamic feedbacks with sea level [11–16]. Declines in sediment availability caused by channel dredging, upstream damming, and changes in agricultural practices, coupled with hydrological changes, such as mosquito ditching, attenuate tidal marsh responses to sea level changes by reducing sedimentation rates and increasing inundation [9, 17,18]. Further, as rates of SLR increase, it is not clear whether marshes that may already be vulnerable to inundation and low sediment supply will be able to increase elevation at rates that would prevent submergence.

SLR in the U. S. Mid-Atlantic is increasing faster than the global average due to steric sea level changes and geologic subsidence [19–23]. Long-term observations derived from local National Oceanic and Atmospheric Administration (NOAA) tidal gauges show that SLR in Delaware Estuary (ranging from 2.98 ± 0.19 to 4.63 ± 0.50 mm yr^{-1}) and Barnegat Bay (4.09 ± 0.15 mm yr^{-1}) are nearly twice that of the early 20th century global average (1.7 mm yr^{-1})[23]. Projections of local SLR in Delaware suggest that rates will likely exceed 10 mm y^{-1} by 2100 for intermediate or high emission scenarios [24]. This projected rate approaches a critical threshold for tidal marsh elevation feedbacks, and suggests marsh drowning will occur [16]. Additionally, subsidence driven by local groundwater withdrawal [25] or historical land manipulations [26], such as diking, accelerates SLR locally and further expedites marsh drowning. In fact, tidal marsh loss due to submergence across the U. S. Northeast and Mid-Atlantic is already widespread [27–29].

Extensive disturbances in Barnegat Bay and below average elevations relative to tides in the Delaware Bay suggests that tidal marshes in both estuaries have a high degree of vulnerability to accelerating rates of SLR [18, 30–36]. Tidal marsh losses and increased interior flooding have been documented in both estuaries [7, 26–27, 30–36], but no analyses of elevation dynamics across these areas have been completed to date. Further, elevation change data from other locations in the Mid-Atlantic and Northeast have shown that SLR frequently exceeds marsh accumulation [37–42].

The importance of maintaining Mid-Atlantic tidal marshes for community protection, combined with ongoing evidence of marsh drowning, motivated the establishment of a monitoring network. Two National Estuary Programs in collaboration with the Academy of Natural Sciences of Drexel University established this network to determine how tidal marshes in the Delaware Estuary and Barnegat Bay are responding to accelerated SLR (www.macwa.org). A variety of data were collected as part of these monitoring efforts, but this particular study focuses on surface elevation change and surface accretion rates. Our objectives were to: (1) compare rates of marsh sediment accumulation and vertical elevation change to long-term and contemporary rates of SLR, as well as contemporary rates of rise in mean high water for the Delaware Estuary and Barnegat Bay; (2) to determine how elevation capital, salinity, tidal range, and water column suspended sediments influence elevation dynamics among geographically distinct tidal marshes.

Materials and methods

Study Sites

The Delaware Estuary, located in New Jersey, Delaware, and Pennsylvania, contains over 66,000 ha of tidal marsh [35]. It has a length of approximately 215 km from the fall line at

Trenton, New Jersey to its confluence with the Atlantic Ocean. Upstream of Wilmington, Delaware, the estuary is freshwater (<0.3‰) riverine and has a tidal range between 1.6 and 2.3 m. Downstream of Wilmington, salinities gradually transition to polyhaline (20–25‰) with tidal ranges from 1.2 to 1.7 m. Freshwater, brackish, and saline tidal marshes exist along this estuarine gradient. *Zizania aquatica, Peltandra virginica, Polygonum punctatum,* and *Nuphar advena* are among many herbaceous perennial and annual species that dominate tidal freshwater marshes in the Delaware River. Halophytic grasses, such as *Spartina alterniflora* (syn. *Sporobolus alterniflorus*), *Spartina patens* (syn. *Sporobolus pumilus*) and *Distichlis spicata,* dominate the brackish and saline marshes of the Delaware Bay. Because of heavy development in the Trenton-Philadelphia-Wilmington industrial corridor, only 16.7% of tidal freshwater wetlands remain in Pennsylvania from the pre-colonization extent (500 of ~3,000 ha) [35, 43] and at least six fresh intertidal plant species are extirpated from the tidal Delaware River [44]. Satellite imagery analyses suggest that marsh loss in the Delaware Estuary was ~80 ha y^{-1} from 1996–2010 [35]. Although wetland loss to development has declined, major threats remain. Marsh loss in the Delaware Estuary are occurring due to open water conversion linked to historical manipulations (*e.g.* diking, ditching), pervasive shoreline erosion, and anthropogenic limitations to sediment supplies (*e.g.* dredge spoil disposal, changes in agricultural practices, damming) [9, 26, 35, 44].

Barnegat Bay is a 520-km^2 shallow lagoon bordered to the east by a 65 km long barrier island separating the bay from the Atlantic Ocean. The 9,200 ha of salt marsh [36] fringing the Barnegat Bay are polyhaline (>18‰) and dominated by *S. alterniflora*, *S. patens,* and *D. spicata*. Tidal ranges are approximately <0.2–0.7 m depending on bathymetry and distance from the Bay's two inlets [45]. While northern Barnegat Bay is largely urbanized with extensive shoreline hardening (e.g. bulkheads), the southern region is less developed and contains more protected lands (i.e., the Edwin B. Forsythe National Wildlife Refuge holds ~14,800 ha) [46]. Estimated rates of salt marsh loss in the Barnegat Bay were ~6 ha y^{-1} from 1996–2010 [36]. Tidal marshes manipulation for mosquito population control have been extensive, with many marshes grid ditched and over 7,000 mosquito management ponds excavated [47]. Open marsh water management (OMWM), a mosquito management tactic, increases pond habitat to lure insectivorous fish to the high marsh, but the process also causes vegetation losses and affects elevation, as excavated peat is side-casted onto the marsh [18, 47].

Monitoring protocols

We established eleven monitoring sites in tidal marshes of the Delaware Estuary and Barnegat Bay, which varied in tidal range, salinity, and dominant vegetation type (Fig 1; Table 1). For each site, three deep rod surface elevation tables (SET) were installed [48–49] between 2010 and 2014 to measure marsh elevation change. Three feldspar marker horizon (MH) plots encircle each SET to measure short-term vertical sediment accumulation, or accretion [50]. Combined, the SET-MH technique distinguishes elevation change by surface and subsurface processes, and can provide estimates of resilience to SLR [42, 49, 51]. Here, we calculated shallow subsidence by subtracting elevation change from accretion rates, so that negative values represent declining subsurface elevations [51]. We read SET-MHs twice a year over the course of 5–7 years, depending on installation dates.

Fig 1. Monitoring sites in the tidal marshes of the Delaware Estuary and Barnegat Bay.

Table 1. Site descriptions for the Delaware River, Delaware Bay, and the Barnegat Bay.

Region	Site	Install Year	Location	Latitude, Longitude	Tidal Range (m)	Salinity (‰)	Dominant Vegetation
Delaware River	Crosswicks Creek	2011	Bordentown, NJ	40° 9.76’N, 74° 42.51’W	2.4	0.10	*Z. aquatica, P. virginica, P.*

							punctatum, and *N. advena*
	Tinicum	2010	Philadelphia, PA	39° 53.05'N, 75° 16.49'W	1.7	0.18	*Z. aquatica, P. virginica, P. punctatum,* and *N. advena*
	Christina River	2010	Wilmington, DE	39° 43.29'N, 75° 34.07'W	1.7	0.40	*Typha angustifolia, P. virginica, P. punctatum,* and *N. advena*
Delaware Bay	Dividing Creek	2012	Dividing Creek, NJ	39° 14.14'N, 75° 6.76'W	1.7	17	*S. alterniflora, S. patens,* and *D. spicata*
	Maurice River	2010	Bivalve, NJ	39° 15.95'N, 74° 59.72'W	1.7	11	*S. alterniflora, S. patens,* and *D. spicata*
	Dennis Creek	2011	South Dennis, NJ	39° 10.58'N, 74° 51.74'W	1.6	16	*S. alterniflora, S. patens,* and *D. spicata*
	Broadkill River	2014	Lewes, DE	39° 47.13'N, 75° 10.75'W	1.1	27	*S. alterniflora, S. patens,* and *D. spicata*
Barnegat Bay	Island Beach	2011	Seaside Park, NJ	39° 47.96'N, 74° 6.10'W	0.2	27	*S. alterniflora*
	Dinner Point Creek	2010	West Creek, NJ	39° 37.43'N, 74° 16.20'W	0.6	26	*S. alterniflora*
	Horse Point	2012	West Creek, NJ	39° 37.59'N, 74° 15.43'W	0.6	26	*S. alterniflora*
	Reedy Creek	2010	Brick, NJ	40° 1.74'N, 74° 5.07'W	0.2	20	*S. alterniflora, S. patens,* and *D. spicata*

We conducted topographic surveys of tidal marsh elevations in each location from 2014 to 2018 using real-time kinematic GPS receivers (a Leica Viva GS14 GNSS Receiver and Viva CS15 field controller, or a Trimble R10 GNSS receiver and TSC2 data controller). Data collection followed National Geodetic Survey guidelines for the RT3 accuracy class (0.04–0.06

m horizontal; 0.04–0.08 m vertical precision): baselines <20 km, collection at 1 s intervals for 15s, with a steady fixed height rover pole without use of a bipod [52].

We measured a suite of physicochemical parameters along five water quality stations 2–3 times per year, in spring, fall, and/or summer. For each station, salinity was measured with a YSI (Professional Pro+, Xylem, Yellow Springs, OH) and samples were collected for further laboratory analyses, including total suspended sediment and dissolved nutrient concentrations (not reported). Additionally, we used existing available data and short-term water level logger deployments to determine local tidal datums for each site (Supporting Information).

Calculations and data analysis

Elevation capital, often expressed as marsh elevation relative to a tidal datum such as mean high water (MHW), is a useful proxy for tidal marsh vulnerability to SLR [14]. Tidal marshes with elevations near the upper limit of intertidal plant growth possess greater elevation capital and are more resilient to rising sea levels. Tidal datums relative to the National Tidal Datum Epoch (NTDE; 1983–2001) were computed for eight sites using the modified-range-ratio method on short-term water level measures derived from in-situ loggers (Supporting Information) [53]. This method is generally associated with accuracy of 2–3 cm for a 5-month period [54]. NOAA tide stations of Atlantic City, NJ, Lewes, DE, Cape May, NJ, or Philadelphia, PA served as controls. For Crosswicks Creek and the Christina River, tide ranges matched nearby NOAA tide stations (Newbold, PA and Delaware City, DE). Because these stations lacked datum conversions to the North American Vertical Datum 1988 (NAVD88), we obtained reference water levels by surveying water levels with real time kinematic surveys to permit conversion. We calculated average flooding times (%) for the median marsh elevations using 2016 or 2018 water level data. Elevation capital, expressed relative to MHW, was the difference between median survey elevation (m NAVD88) and the elevation of local MHW (m NAVD88) relative to the National Tidal Datum Epoch (1983–2001).

To estimate rates of tidal marsh elevation change and sediment accumulation across the Delaware Estuary and Barnegat Bay, we constructed regressions of elevation and accretion against time [48, 51, 55–56]. We compared rates of accretion and vertical elevation change with local sea level rise (LSLR) trends derived from the nearest NOAA tide station [57]. We also compared trends in rates of elevation gain and sediment accretion with long-term (LT) (1969–2018) and short-term (ST) (2000–2018) trends in SLR and, as well as ST (2000–2018) rates of increase in mean high water level (ST MHWR). To be conservative, we concluded that a site was gaining elevation or accreting sediment at rates significantly greater than SLR if its rate of rise (±SE) exceeded LSR. Associations were identified between key processes that determine marsh survival (elevation capital, elevation change, accretion, and shallow subsidence) with gradients in environmental conditions (salinity, tidal range, water column dissolved organic carbon, and suspended sediment concentration) using linear regression.

Between 2011 and 2013, pond excavation associated with OMWM affected all of the SET-MHs at Dinner Point Creek in Barnegat Bay. We began monitoring in 2010-2012, prior to pond construction. In 2012, these activities casted ~0.20 m of sediment directly onto SET 1 (Supporting Information). We then installed SET-MHs at Horse Point, an unaffected area adjacent to Dinner Point Creek. We excluded Dinner Point Creek SET 1 data from some analyses, where noted, because they were not representative of natural or ambient processes.

Results

Marsh elevation capital relative to MHW was variable among focal tidal marshes (Fig 2). In the Delaware Estuary, marshes along the Broadkill River, the Christina River, and Dennis

Creek possessed notable elevation capital, whereas Tinicum marshes sat low in the tidal frame. In Barnegat Bay, Reedy Creek and Island Beach had lower elevation capital compared to Dinner Point Creek and Horse Point. We found significant differences between long-term (i.e. NTDE) and current high tide flooding levels (Tables 2 and 3), suggesting most marshes, with the exception of Dinner Point Creek and Horse Point, sit well below the current MHW. Elevation of the marsh relative to MHW influenced accretion (R^2=0.30, p<0.001) and elevation change (R^2=0.13, p<0.05), such that sediment accretion and elevation change rates were greater for marshes sitting low in the tidal frame (Fig 3). However, elevation did not influence rates of shallow subsidence.

Fig 2. Frequency distribution of marsh elevations in Barnegat Bay and the Delaware Estuary relative to local MHW (NTDE) in meters.
Vertical dashed line is 0.0 m MHW.

Table 2. Median tidal marsh elevations relative to NAVD88, local MHW relative to the NTDE, and summer-fall mean high tide levels. Values were derived from 2016 water level data, with the exception of the Broadkill River, where data are from 2018.

Site	Median Elevations (m)			% Time Inundated
	NAVD88	MHW NTDE	MHW 2016	
Crosswicks Creek	1.10	-0.18	-0.29	20%
Tinicum	0.44	-0.37	-0.53	39%
Christina River	0.62	0.07	-0.34	30%
Dividing Creek	0.68	-0.04	-0.29	27%
Maurice River	0.67	-0.05	-0.30	25%
Dennis Creek	0.61	-0.03	-0.21	24%
Broadkill River[1]	0.39	0.02	-0.22	27%[1]
Reedy Creek	0.13	0.05	-0.21	67%
Island Beach	0.19	0.13	-0.09	42%
Horse Point	0.40	0.18	-0.01	12%
Dinner Point Creek	0.44	0.25	0.00	12%

Table 3. Long (1969–2018) and short-term (2000–2018) water level trends at NOAA harmonic tidal datum stations from linear regression of monthly mean tidal data.

Tide gage	Associated monitoring site(s)	Mean sea level rise (mm yr^{-1})		Mean high water rise (mm yr^{-1})
		LT 1969–2018	ST 2000–2018	ST 2000–2018

Philadelphia, PA	Crosswicks Creek, Tinicum	4.03	6.39	9.53
Reedy Point, DE	Christina River	4.16	5.60	7.11
Lewes, DE	Broadkill River	3.96	6.72	7.77
Cape May, NJ	Dividing Creek, Maurice River, Dennis Creek	4.86	6.81	8.53
Atlantic City, NJ	Reedy Creek, Island Beach, Horse Point, Dinner Point Creek	4.69	5.75	7.71

Figure 3. Relationship between elevation capital relative to 2016 MHW and (A) elevation change, (B) accretion, and (C) shallow subsidence.
Solid triangles are freshwater tidal marshes in the Delaware River, solid squares are salt marshes in the Delaware Bay, and hollow circles are salt marshes in the Barnegat Bay. There were no significant differences among estuaries.

Rates of LSLR and MHWR varied depending on the time period under consideration, with lower rates associated with longer time windows, and higher rates recorded over the past 19 years (Table 3). Notably, LSLR was 44% greater over the past 19 years than over the last 50 years, and over the past 19 years, the rise in MHW was significantly greater (30%) than LSLR (Table 3). The most rapid rate of rise was found for tidal marshes in the freshwater tidal portion of the Delaware Estuary, where MHWR has approached 1 cm yr^{-1} (0.95 cm yr^{-1}), values that exceed even the most extreme forecasts for 21st century SLR rates (Najjar et al. 2000).

Comparisons of accretion and elevation change rates with LT LSLR show variability within and across estuaries and sites. Of the 32 surface elevation tables, excluding Dinner Point Creek SET 1, 75% had elevation trends that did not exceed rates of LT LSLR (1969–2018) (Supporting Information). When compared to ST LSLR and MHWR, only 6% and 4% of marsh areas had rates of elevation increase that kept pace, respectively. Accumulation deficits, or the difference between LT LSLR and elevation change, varied from -4.12 to 10.8 mm yr^{-1} in the Delaware Estuary and -6.65 to 1.08 mm yr^{-1} in Barnegat Bay (Table 4) (Supporting Information).

Table 4. Ranges of elevation change, accretion, and shallow subsidence summarized by tidal freshwater marshes in the Delaware River, salt marshes in the Delaware Bay, and salt marshes in Barnegat Bay. Median values are in brackets. In Barnegat Bay, Dinner Point Creek SET 1 was excluded from this table, due to dispersal of sediment from mosquito management practices causing anomalously high annual mean accretion and elevation change rates (>25 mm yr^{-1}). Site-specific information is in Supporting Information.

Marsh type	Rates (mm yr^{-1})		
	Elevation Change	Accretion	Shallow Subsidence
Delaware River, freshwater	2.73 to 14.8 [4.5]	5.19 to 17 [9.5]	-9.41 to 4.54 [-3.8]
Delaware Bay, saltwater	0.74 to 6.89 [4.9]	3.19 to 10.1 [5.5]	-6.54 to 2.07 [-1.4]
Barnegat Bay, saltwater	-1.96 to 5.77 [4.1]	1.91 to 6.72 [5.1]	-5.76 to 3.61 [-1.4]

Processes of elevation change, accretion, and shallow subsidence varied between the two estuaries, as well as along the salinity gradient in the Delaware Estuary (Figs 4 and 5; Table 4). Intra-site variation of elevation change was large at some sites, with the largest range at Tinicum (8.82 mm yr^{-1}). Twenty-four of the 32 (75%) SET-MHs experienced shallow subsidence (Supporting Information; excluding Dinner Point Creek SET 1). Seven (21%) SET-MHs experienced subsurface expansion and only one site (3%) had elevation changes that were loosely equivalent to accretion (Maurice SET 2). Shallow subsurface processes had a substantial role in elevation dynamics, whether it was due to subsurface expansion (presumably linked to belowground production) or consolidation.

Fig 4. Accretion and surface elevation change for (A) the tidal freshwater marshes in the Delaware River and (B) salt marshes in the Delaware Bay.
Horizontal dashed lines are LT SLR (1969–2018), ST SLR (2000–2018), or MHWR derived from the nearest NOAA tide gages (superscripts denote tidal station used for each site, north to south: (a) Philadelphia, (b) Reedy Point, (c) Cape May, and (d) Lewes. Error bars are standard error.

Fig 5. Accretion (MHs) and surface elevation change (SETs) for Barnegat Bay.
Horizontal dashed lines are LT SLR (1969–2018), ST SLR (2000–2018), or MHWR derived from NOAA's Atlantic City tide gage. Dispersal of the sediment across Dinner Point Creek SET

1 caused anomalously high annual mean accretion and elevation change rates (>25 mm yr^{-1}) and so, those data were excluded. Error bars are standard error.

Associations among environmental parameters (salinity, tidal range, surface elevations, and water column suspended sediments) and accretion or elevation change rates were estuary-dependent (Figs 6–8). In the Delaware Estuary, maximum accretion rates occurred in tidal freshwater marshes, where suspended sediment concentrations were low and tidal ranges were large. While accretion rates were positively associated with tidal range ($R^2 = 0.44$, $p<0.05$) and water column suspended sediments ($R^2 = 0.33$, $p<0.05$) in Delaware Estuary salt marshes, this association was not significant for freshwater marshes or salt marshes in Barnegat Bay. In salt marshes of the Delaware Estuary, subsurface process rates were more negative (subsidence) where salinities were lower ($R^2 = 0.42$, $p<0.05$) and tidal ranges greater ($R^2 = 0.42$, $p<0.05$).

Fig 6. Relationship between elevation change and (A) salinity, (B) tidal range, and (C) total suspended solids (TSS).
Solid triangles are freshwater tidal marshes in the Delaware River (DF) and solid squares are salt marshes in the Delaware Bay (DS). Hollow circles are salt marshes in the Barnegat Bay (BB). Dispersal of the sediment across Dinner Point Creek SET 1 as part of construction of mosquito control ponds caused anomalously high (>25 mm yr^{-1}) values for accretion (hollow stars) and so, those data were excluded from analyses.

Fig 7. Relationship between sediment accretion and (A) salinity, (B) tidal range, and (C) total suspended solids (TSS).
Solid triangles are freshwater tidal marshes in the Delaware River (DF) and solid squares are salt marshes in the Delaware Bay (DS). Hollow circles are salt marshes in the Barnegat Bay (BB). Data from Dinner Point Creek SET 1 (hollow stars) were excluded from these analyses.

Fig 8. Relationship between shallow subsidence and A) salinity, (B) tidal range, and (C) total suspended solids (TSS).
Solid triangles are freshwater tidal marshes in the Delaware River (DF), solid squares are salt marshes in the Delaware Bay (DS), and hollow circles are salt marshes in the Barnegat Bay (BB). Solid triangles are freshwater tidal marshes in the Delaware River, solid squares are salt marshes in the Delaware Bay, and hollow circles are salt marshes in the Barnegat Bay. R_D^2 is the Delaware Estuary and R_B^2 is for Barnegat Bay.

Sediment placement by OMWM at Dinner Point Creek SET 1 increased absolute elevations by more than 18 cm, but elevations have declined by 10 cm since 2013 (Supporting

Information). Rates of elevation change were 4.9 mm yr^{-1} (R^2= 0.93, $p < 0.05$) before placement, then declined to -13.5 mm yr^{-1} (R^2=0.83, $p < 0.001$) after placement. These rates are not representative of natural accretion or elevation change so the Dinner Point Creek SET 1 data point was removed from elevation change and accretion analyses. Rates of shallow subsidence at this location, however, were not anomalous so we retained those values.

Discussion

Few of the studied tidal marshes had rates of elevation change equal to or greater than LT or ST LSLR, suggesting that regional marsh loss may be associated with deficits in vertical accretion. Although accretion rates were comparable to LT LSLR, shallow subsidence offset the effects of surface accretion, and thus few sites had elevation change rates that “kept pace” with LT LSLR. From 1996–2010, tidal marsh area in Barnegat Bay and the Delaware Estuary declined by 199 ha yr^{-1} [35] and 194 ha yr^{-1} [36], respectively. Tidal marsh loss in Maryland [40], New York City [58], and Rhode Island [42] have been similarly linked to elevation changes less than LSLR. Losses, coupled with ongoing interior marsh loss and deterioration [27–28, 34, 37], indicate that many marshes in this region are unable to maintain elevations relative to accelerating sea level rise.

Rates of elevation change, accretion, and shallow subsidence varied across our study sites, which included tidal marshes in a coastal lagoon as well as in tidal freshwater and saline portions of a large coastal plain estuary. In the Delaware Estuary, rates of accretion and subsurface processes varied between salt (median values of 5.5 mm yr^{-1} for accretion and -1.4 mm yr^{-1} for shallow subsidence) and freshwater marshes (median values of 9.5 mm yr^{-1} for accretion and -3.8 mm yr^{-1} for shallow subsidence). Differences in dominant vegetation structure and sediment sourcing contribute to higher rates of accretion in tidal freshwater marshes compared to salt marshes, as others have previously discussed [59–61]. In Barnegat Bay and Delaware Estuary salt marshes, magnitudes of elevation change (median values of 4.9 and 4.1 mm yr^{-1}, respectively) and accretion (median values of 5.5 and 5.1 mm yr^{-1}, respectively) were comparable, despite higher salinities, lower suspended sediment concentrations, and smaller tidal ranges in Barnegat Bay. Previous findings suggest that in addition to sedimentation, organic production plays an important role in accretion for Barnegat Bay salt marshes [31].

As rates of elevation change and accretion rates are higher at lower elevations, it can be problematic to discern drowning or vulnerability to open water conversion without added context of elevation capital [58, 62]. For instance, accretion and elevation change rates at Tinicum mostly exceeded ST SLR, yet, these marshes sit low in the tidal frame (-0.47 m MHW), and they flooded 39% of the time during the 2016 growing season. As such, rising flood frequencies may drown vegetation at Tinicum before marshes that have greater elevation capital, but lower accretion rates. Similarly, in Barnegat Bay, Reedy Creek had lower elevation capital (0.05 m MHW) and flooding was frequent (67%), despite elevation change and accretion rates that were approximately equal with LT SLR. For tidal marshes with more elevation capital, like Horse Point (0.18 m MHW), lower elevation change or accretion rates likely reflect less sediment delivery due to less frequent tidal flooding. Thus, high accumulation or high elevation gains – rather than a symptom of high SLR resilience – may instead reflect submergence due to positive feedbacks between flooding frequency, sediment deposition, and high water column suspended sediment concentrations liberated by degrading or fragmenting wetlands.

Elevation processes, specifically accretion, do not wholly capture tidal marsh vulnerability to drowning or loss as other sediment dynamics also contribute to tidal marshes stability. In two Maryland-based studies, Ganju et al. [63–64] found that net sediment export likely drove deterioration and instability of tidal marshes at Blackwater River, despite abundant suspended sediment and seemingly adequate accretion rates relative to sea level rise. Thus, Ganju et al. surmised that the release of organic marsh sediments through the deterioration process might subsidize suspended sediment concentrations and sedimentation rates [64]. In a follow-up study, Ganju et al. monitored sediment dynamics at two sites in our monitoring network: deteriorated Reedy Creek, where marshes are fragmenting, and Dinner Point Creek, which was relatively stable with a larger proportion of vegetated marsh area [65]. Those results similarly found that deterioration at Reedy Creek correlated with larger net export of sediments compared to Dinner Point Creek. From our results, rates of accretion and elevation change at Reedy Creek ranged from 4.52–6.72 mm yr^{-1} and 2.24–5.77 mm yr^{-1}, respectively. Accretion and elevation change rates at Horse Point, located adjacent to Dinner Point Creek yet undisturbed from pond excavation, ranged from 5.39–5.87 mm yr^{-1} and 3.92–4.40 mm yr^{-1}, respectively. Our findings ultimately support previous assertions that high rates of sediment accretion or elevation do not protect against marsh loss if they are a symptom of marsh degradation and fragmentation.

Although it is convention to compare tidal marsh elevation changes to LT LSLR trends to determine if accumulation debts are accruing [51], we find that ST LSLR or MHW trends more adequately represented changing inundation regimes in tidal marshes. Analysis of water levels suggest that SLR in our region was 44% greater during the 2000–2018 period than it was over the last 50 years (1969–2018), and in addition, rates of MHW rise over the past 19 years were nearly double rates of LT SLR (87% greater) (Table 3). In this study, ST LSLR and MHWR did not increase proportionately with LT LSLR across sites, so local conditions may contribute to how inundation patterns change in response to accelerating LSLR. A previous study on tidal datum changes in the U.S. similarly found that mean high water rose faster than mean sea level at tidal gages in Atlantic City, NJ, Cape May, NJ, and Lewes, DE [66]. If local MHW rises faster than LSLR, comparing LT LSLR to elevation change rates could lead to underestimations of tidal marsh SLR vulnerability. Although differences between rates of elevation change and ST SLR or MHW presented in this study were concerning, these differences could also represent lags or shifts in tidal marsh elevation dynamics [16] that were not captured in this 8-year dataset, which only further monitoring might elucidate.

Broadly, our results serve as a baseline to evaluate how patterns between elevation change and inundation become detrimental to tidal wetlands as SLR accelerates. Elevation changes relative to LSLR dynamically associate with tidal marsh elevation capital, and so, accretion is likely to increase with rising water levels. Although increased flooding facilitates greater accretion, temporal lags may exist between increasing depth and duration of floods and cumulative sediment accumulation [16]. Tidal marsh plants are most productive within a specific range of elevations relative to local tidal datum [11], so delays in elevation building by sedimentation leads to reduced plant production as rapidly rising water levels surpass optimal growth elevations. Indeed, many recent studies have attributed marsh losses to plant intolerance to excessive flooding caused by rising water levels [29, 67–70]. Subsequent vegetated area losses cause further destabilization through net sediment export [65]. A portion of our sites had accretion rates congruent with LT SLR and nearly all were well below ST SLR, thus we posit that sedimentation paces dynamically with LT SLR but lags behind ST SLR so that biological limitations associated with plant flood tolerance drives marsh deterioration or drowning. In future studies, it may be more pragmatic to ask whether marshes are “catching up” with SLR, rather than intrinsically “keeping pace.”

Conclusions

Rates of elevation change in most (96%) tidal marshes across the Delaware Estuary and Barnegat Bay have not kept pace with recent (2000–2018) rising water levels. Accumulation deficits, relative to LT LSLR (1969–2018), varied from -4.12 to +10.8 mm yr^{-1} in the Delaware Estuary and -6.59 to +1.08 mm yr^{-1} in Barnegat Bay. Overall, rates of elevation change, sediment accumulation, and shallow subsidence varied between estuaries and across sites. Relationships among these rates and salinity, water column suspended sediments, and tidal range differed by geography. We found that more recent water level information was useful in determining inundation frequency more precisely. Due to the importance of sustained tidal marsh acreage to coastal communities across the U. S. Mid Atlantic, we hope these data provide much-needed context for future intervention efforts focused on preventing losses due to rising sea levels in our region.

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Supporting Information

S1 Table. Data collected for calculation of empirical tidal datums.

S2 Table. Tidal datums calculated from 2000–2018.

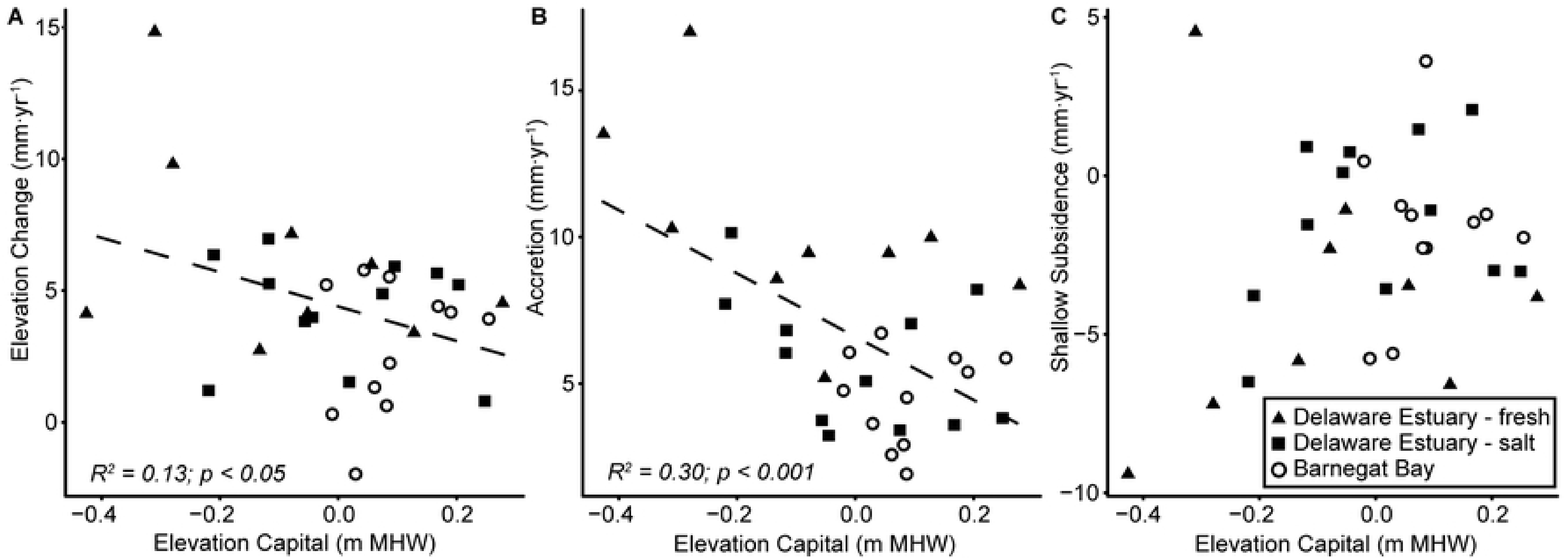
S3 Table. Tidal datums calculated for the National Tidal Datum Epoch (1983–2001).

S4 Table. Tidal datums from NOAA Vdatum.

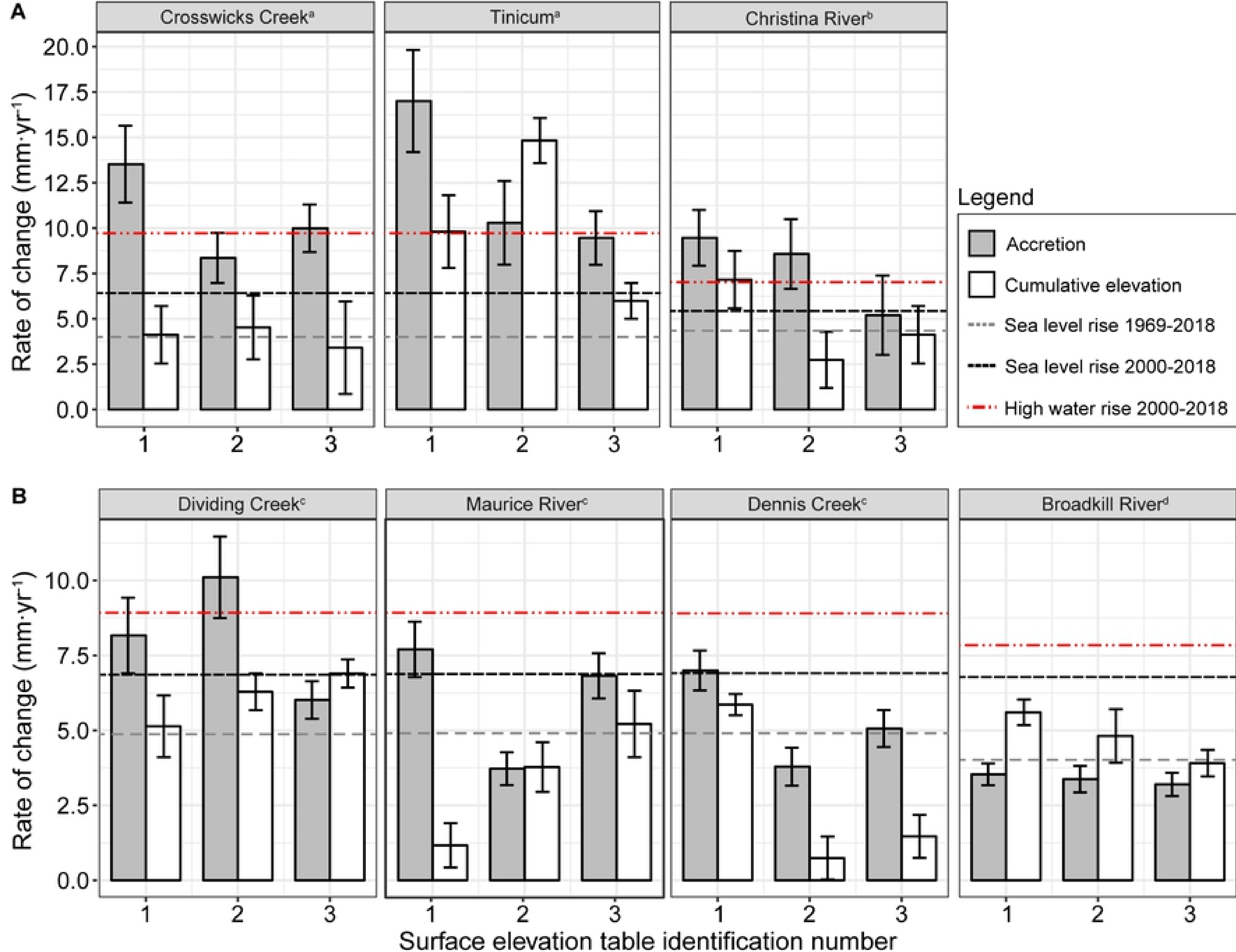
S5 Table. Error between NTDE datum and Vdatum (%).

S6 Table. Values of accretion, surface elevation change, shallow subsidence, and comparisons to long-term (LT; 1969–2018) local sea level rise (LSLR), short-term (ST; 2000–2018) LSLR, and mean high water rise (MHWR; 2000–2018)(see Table 4 for those rate values).
Standard errors are in parenthesis. To be conservative, when rates of accretion or elevation change exceeded water level rise rates (LT LSLR, ST LSLR, or MHWR) by at least one standard error, we assigned it *yes* (bold Y, boxes gray shaded) for keeping pace; otherwise, we assigned it *no* (N) for not keeping pace.

Fig S1. **Trends of elevation change at Dinner Point Creek SET 1 before and after it was abruptly buried with sediment as part of the adjacent construction of a mosquito control pond (i.e. OMWM).**
Between 2011 and 2013, marsh elevation increased at a rate of 4.92 mm y^{-1}; 2013 to the present, elevation decreased at a rate of 13.5 mm y^{-1}. Error bars are standard error.

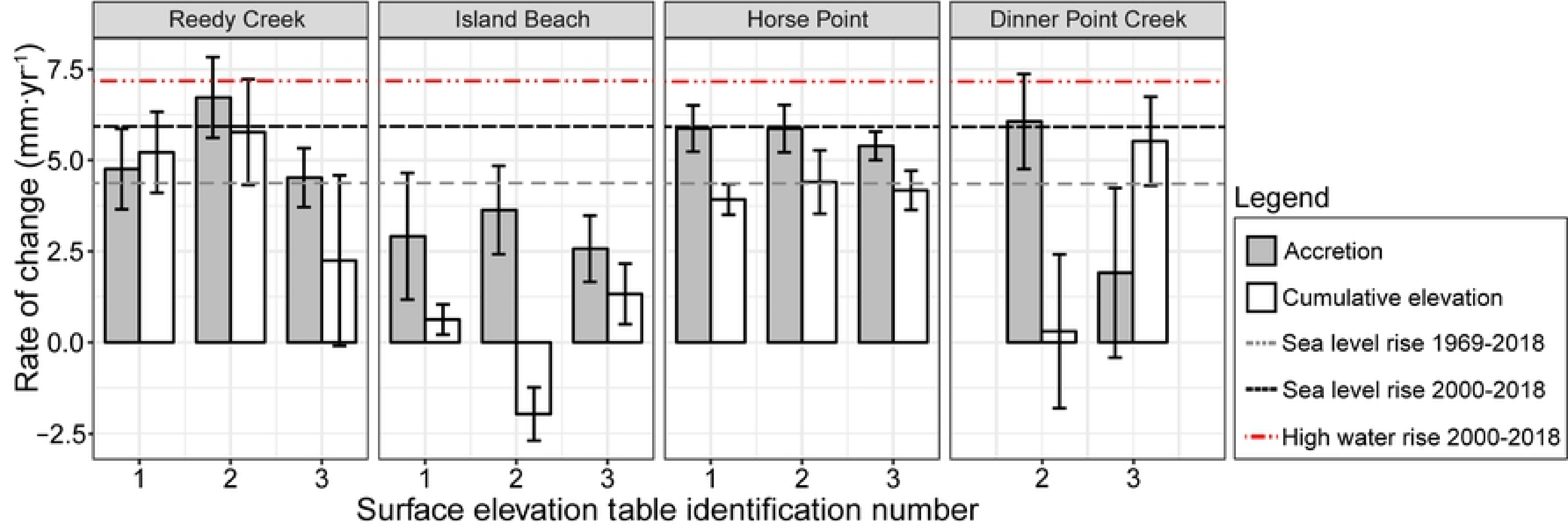

A
B
C
Elevation Change (mm·yr-1)
Accretion (mm·yr-1)
Shallow Subsidence (mm·yr-1)
$R^2 = 0.13; p < 0.05$
$R^2 = 0.30; p < 0.001$
Elevation Capital (m MHW)
Elevation Capital (m MHW)
Elevation Capital (m MHW)
Delaware Estuary - fresh
Delaware Estuary - salt
Barnegat Bay

Figure



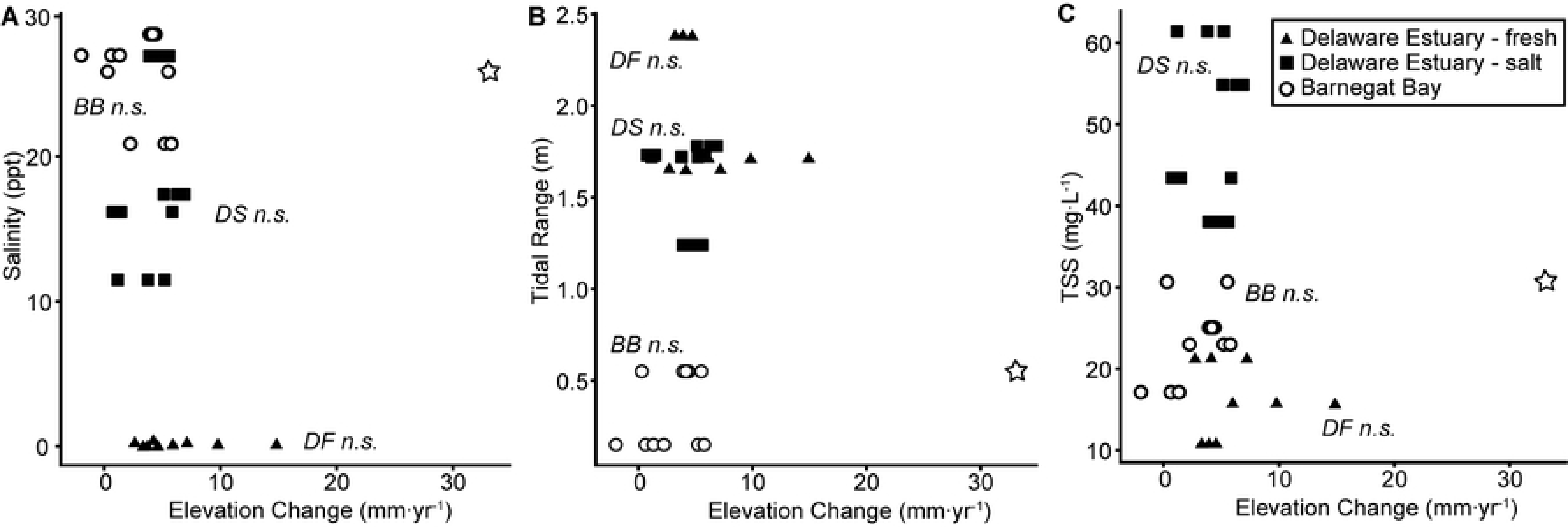
A
Crosswicks Creek[a]
Tinicum[a]
Christina River[b]
Rate of change (mm·yr⁻¹)
20.0
17.5
15.0
12.5
10.0
7.5
5.0
2.5
0.0
1
2
3
Legend
Accretion
Cumulative elevation
Sea level rise 1969-2018
Sea level rise 2000-2018
High water rise 2000-2018
B
Dividing Creek[c]
Maurice River[c]
Dennis Creek[c]
Broadkill River[d]
Rate of change (mm·yr⁻¹)
10.0
7.5
5.0
2.5
0.0
Surface elevation table identification number

Figure



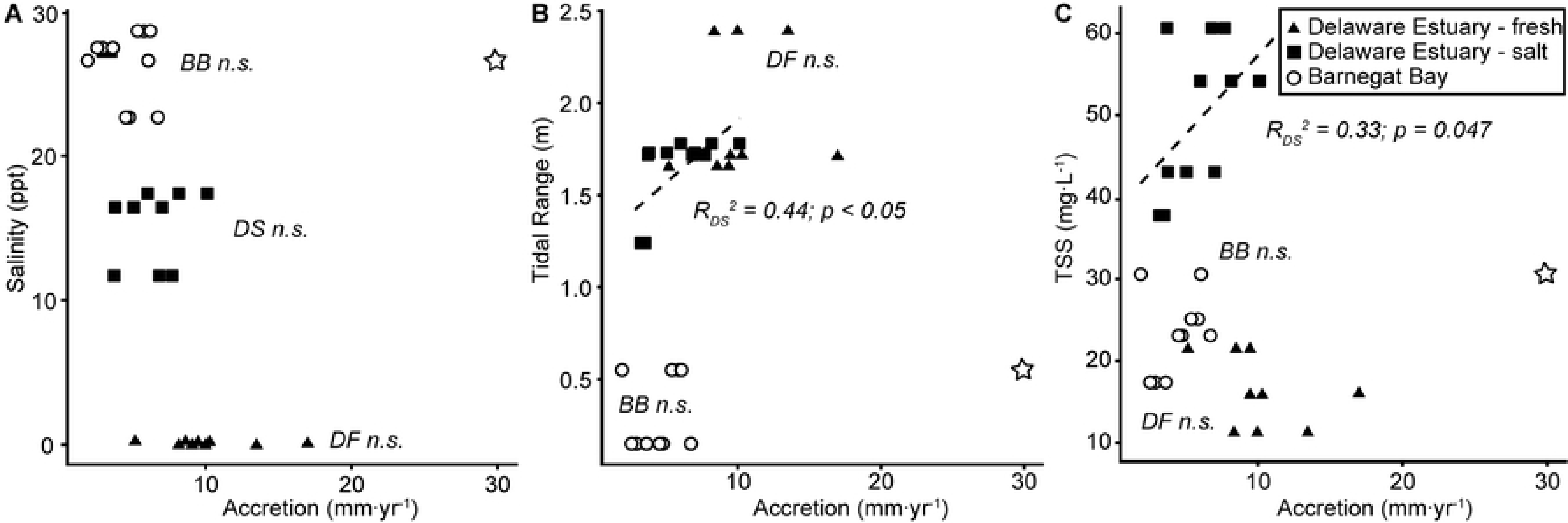
Reedy Creek
Island Beach
Horse Point
Dinner Point Creek
Rate of change (mm·yr⁻¹)
7.5
5.0
2.5
0.0
−2.5
1
2
3
Surface elevation table identification number
Legend
Accretion
Cumulative elevation
Sea level rise 1969-2018
Sea level rise 2000-2018
High water rise 2000-2018

Figure

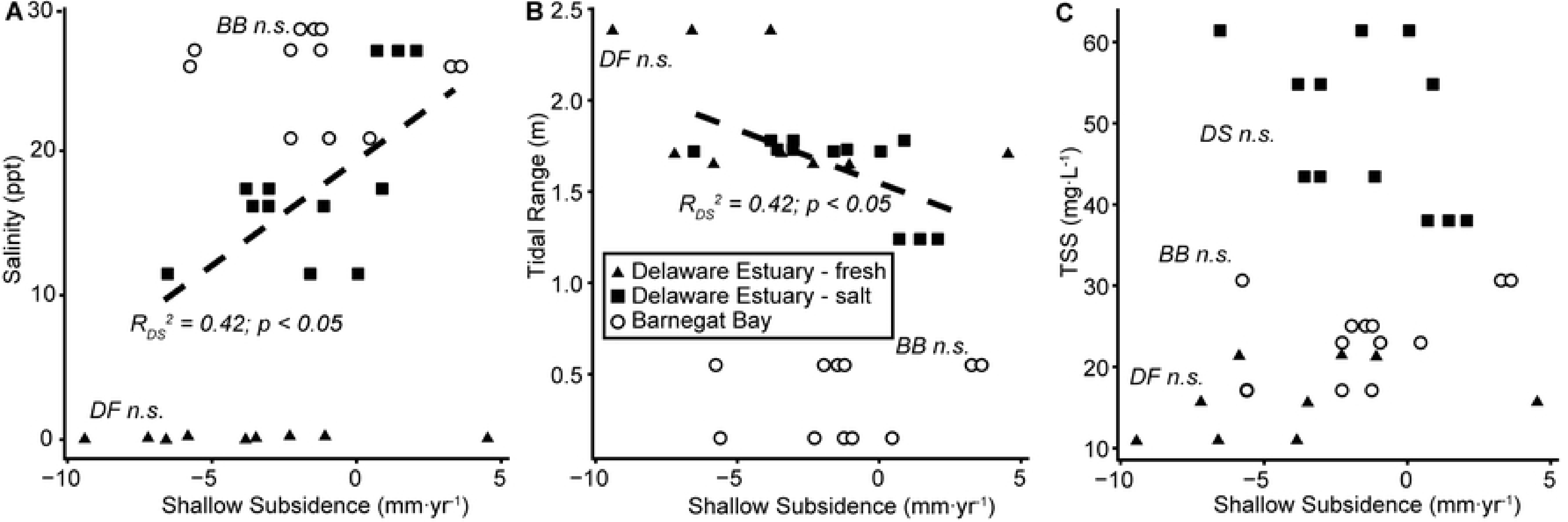


A
Salinity (ppt)
BB n.s.
DS n.s.
DF n.s.
Elevation Change (mm·yr⁻¹)
B
Tidal Range (m)
DF n.s.
DS n.s.
BB n.s.
Elevation Change (mm·yr⁻¹)
C
TSS (mg·L⁻¹)
Delaware Estuary - fresh
Delaware Estuary - salt
Barnegat Bay
DS n.s.
BB n.s.
DF n.s.
Elevation Change (mm·yr⁻¹)

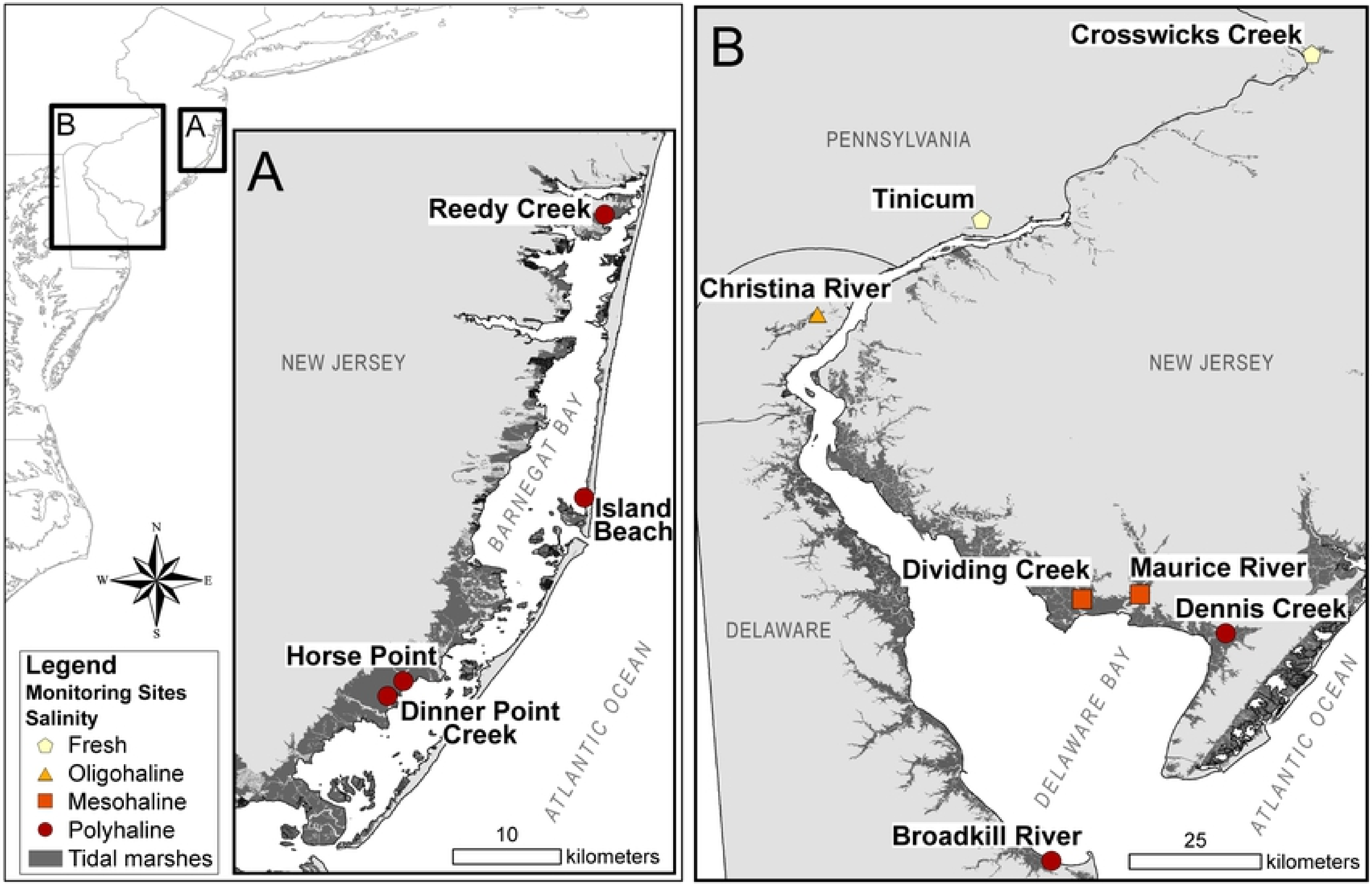
Figure


A
B
C
Salinity (ppt)
Tidal Range (m)
TSS (mg·L⁻¹)
Accretion (mm·yr⁻¹)
BB n.s.
DS n.s.
DF n.s.
$R_{DS}^2 = 0.44; p < 0.05$
$R_{DS}^2 = 0.33; p = 0.047$
Delaware Estuary - fresh
Delaware Estuary - salt
Barnegat Bay

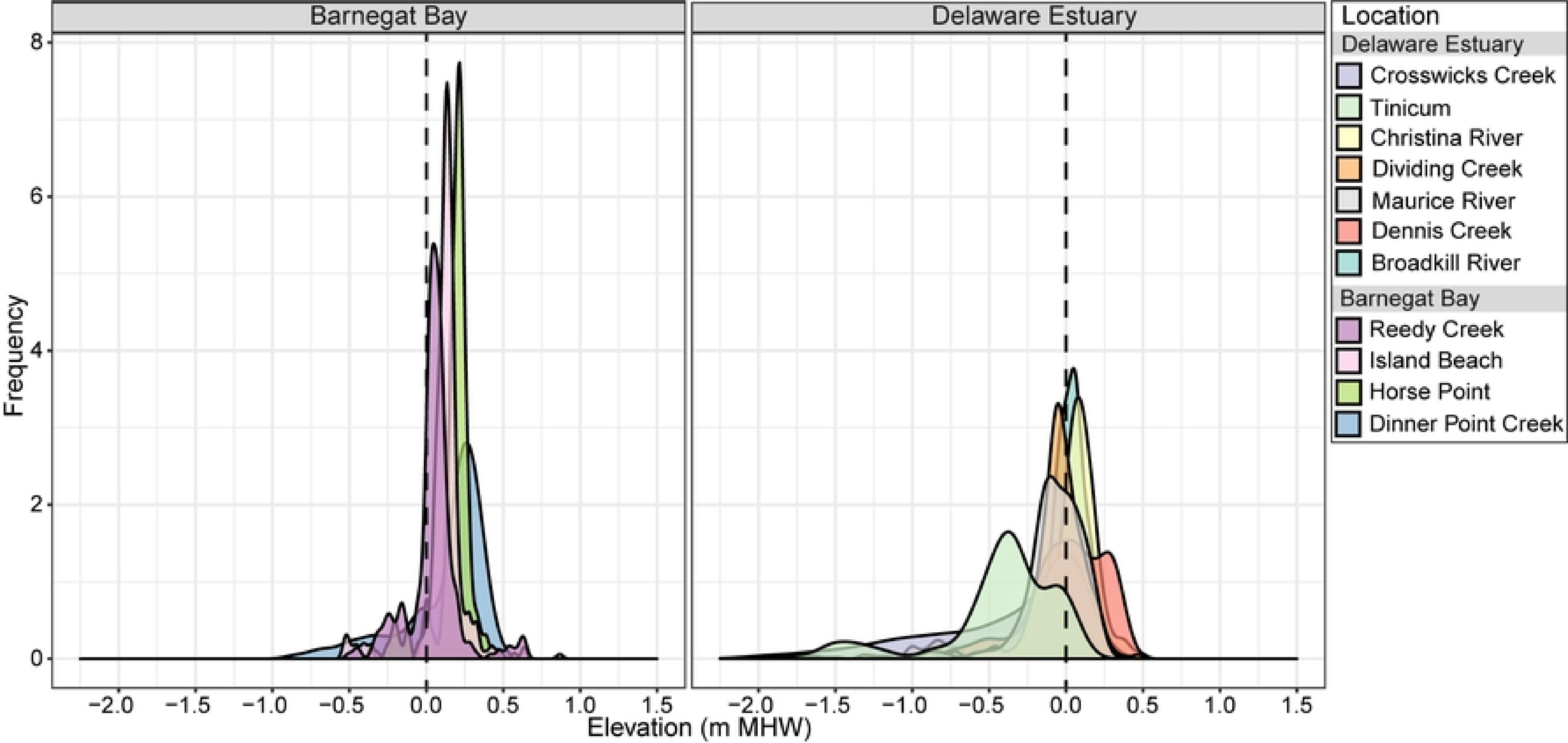
Figure


A
B
C
BB n.s.
DF n.s.
$R_{DS}^2 = 0.42; p < 0.05$
Salinity (ppt)
Shallow Subsidence (mm·yr⁻¹)
DF n.s.
BB n.s.
Tidal Range (m)
Delaware Estuary - fresh
Delaware Estuary - salt
Barnegat Bay
DS n.s.
TSS (mg·L⁻¹)

Figure


B
A
Legend
Monitoring Sites
Salinity
Fresh
Oligohaline
Mesohaline
Polyhaline
Tidal marshes
N
S
E
W
Reedy Creek
NEW JERSEY
BARNEGAT BAY
Island Beach
Horse Point
Dinner Point Creek
ATLANTIC OCEAN
10
kilometers
Crosswicks Creek
PENNSYLVANIA
Tinicum
Christina River
NEW JERSEY
Dividing Creek
Maurice River
Dennis Creek
DELAWARE
DELAWARE BAY
ATLANTIC OCEAN
Broadkill River
25
kilometers

Figure


Barnegat Bay
Delaware Estuary
Frequency
Elevation (m MHW)
8
6
4
2
0
-2.0
-1.5
-1.0
-0.5
0.0
0.5
1.0
1.5
Location
Delaware Estuary
Crosswicks Creek
Tinicum
Christina River
Dividing Creek
Maurice River
Dennis Creek
Broadkill River
Barnegat Bay
Reedy Creek
Island Beach
Horse Point
Dinner Point Creek

Figure