

Oligohaline wetlands were found to be subsiding at higher rates compared to mesohaline and tidal freshwater wetlands. To further understand subsidence trends, a second study was done to examine stratigraphic and geomorphic differences among the sites where subsidence was measured. Few clear relationships between subsidence and stratigraphy emerged, though subsidence followed a curvilinear pattern, peaking at intermediate values for organic matter, depth of site, and salinity, unique nature of mid-estuarine sites. Finally, an in situ field experiment was conducted examining the effects of saltwater intrusion on elevation and vegetation dynamics in a tidal freshwater wetland, given that tidal freshwater wetlands may be more susceptible than saline wetlands to loss because of saltwater intrusion associated with sea-level rise. Four years of salt addition treatments did not have significant effects on either elevation or vegetation, indicating resilience of tidal freshwater wetlands to saltwater intrusion. These studies as a whole shed light on the complicated sets of interacting factors affecting surface elevation and the necessity for localized assessments of coastal wetlands to determine vulnerability to sea-level rise.

SUBSIDENCE, ACCRETION, AND ELEVATION TRENDS IN
ESTUARINE WETLANDS AND RELATIONSHIPS TO SALINITY
AND SEDIMENT STRATIGRAPHY

by

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Chapter I: Introduction

Climate change alters the patterns of precipitation, global temperatures, tropical storm and hurricane frequency, intensity and distribution, and sea-level rise (Michener et al. 1997). These factors in turn affect coastal wetlands by altering the hydrology, sediment supply, nutrient supply, disturbance frequency (i.e., storm frequency) and biological processes such as decomposition and primary productivity. Coastal wetlands are vital ecosystems that serve as nurseries for economically important fishes, buffer coastal communities from storm surges, filter excess nutrients from uplands, sequester carbon, and provide habitat for migratory birds (Mitsch and Gosselink 1993). Sea-level rise threatens the health and persistence of coastal wetlands globally.

With increased sea-levels, coastal wetlands are exposed to increases in flooding and salinity, eventually leading to degradation and loss through submergence. Rates of global, eustatic sea-level rise are approximately 2 mm/yr (Gornitz 1995); however, relative sea level rise (eustatic sea level rise + land subsidence) in some areas are at least double global rates. Chesapeake Bay on the U.S. Atlantic coast, one of the largest microtidal estuaries in the world, has current rates that range from 3.08 mm/yr in Baltimore Harbor to 6.08 mm/yr near the Chesapeake Bay tunnel (NOAA 2012) and are predicted to increase over the next century (IPCC 2007).

Soil elevation is an important response variable to consider when evaluating the potential vulnerability of coastal wetlands to sea-level rise. Soil elevation affects the reduction-oxidation state, hydroperiod, sedimentation and productivity of coastal wetlands, and these factors create negative feedback loops altering the elevation (Rybczyk and Callaway 2009). For coastal wetlands to persist, they are dependent on

maintaining positive elevation relative to sea level. The inability to do so results in drowning and conversion to open water.

Soil elevation is determined by many physical and biological processes. Elevation is increased through the processes of accretion (organic and inorganic sediment build up), geologic processes such as uplift, root growth, and soil swelling due to groundwater dynamics. Elevation decreases due to deep geologic subsidence, shallow subsidence, erosion, autocompaction, decomposition, and groundwater withdrawal.

The overarching goal of my dissertation research was to determine whether coastal wetlands of Chesapeake Bay are vulnerable to sea-level rise due to soil elevation deficits, and whether soil elevation dynamics differed among wetlands of varying salinity regime and geomorphic type. I approached this goal through addressing three specific objectives, each of which comprises a chapter in my dissertation.

Objective 1:

To quantify the current surface elevation and accretion trends of marshes of Chesapeake Bay, and to compare coastal wetlands of varying salinity, assessing potential differences among them.

Objective 2:

Determine whether saltwater intrusion would decrease surface elevation and accretion, and cause a shift in vegetation communities to more salt-tolerant species, in a tidal freshwater wetland.

Objective 3:

Characterize Holocene deposit stratigraphic characteristics (i.e., depth, age, organic matter, bulk density) and assess relationships between soil profiles and shallow subsidence, specifically, whether depth and quantity of organic matter would be related to elevation loss.

Chapter II: Coastal Wetland Surface Elevation and Accretion Dynamics along an Estuarine Salinity Gradient in Chesapeake Bay

Abstract

Coastal wetlands are shrinking in area and ailing in health worldwide. Sea-level rise is a major factor in wetland loss along the U.S. Atlantic and Gulf coasts, and in Chesapeake Bay the rate of relative sea-level rise is nearly double the current global rate of 2 mm/yr due to regional subsidence (NOAA 2012). Coastal marshes persist by maintaining surface soil elevations that exceed rates of sea-level rise, thus making surface elevation an important variable to monitor. Soil elevation affects the hydroperiod, reduction-oxidation state, sedimentation and productivity of marshes. The processes controlling soil elevation change are site-specific and are affected by salinity regime. To evaluate the susceptibility of coastal wetlands in Chesapeake Bay to sea-level rise, 15 surface elevation tables (SET) and accretion marker-horizon plots (MH) were installed along an estuarine salinity gradient. Over the course of five years, none of the marshes increased in surface elevation at rates greater than sea-level rise (elevation change rates ranged from -9.8 ± 6.9 to 2.8 ± 3.9 mm/yr), indicating an elevation deficit. Mid-estuary, oligohaline marshes were subsiding the most and had high spatial variability within Sites. Coastal wetlands of Chesapeake Bay are not keeping pace with sea-level rise and may be at risk of drowning under future sea-level rise scenarios.

Introduction

Globally, coastal wetlands, ecologically and economically valuable ecotones, are threatened by sea-level rise. Rising sea-levels cause a host of complications for coastal wetlands, including increases in surface flooding, saltwater intrusion, and the

introduction of sulfates. The ability of coastal marshes (wetlands dominated by herbaceous plants) to persist in conditions of accelerated sea-level rise is dependent upon maintaining positive surface elevation relative to sea level (Stevenson, Ward and Kearney 1985b, Cahoon, Reed and Day 1995, Morris et al. 2002). Failure to stay abreast of sea-level results in drowning and conversion of marshland to open mudflats (van Asselen, Karssenberg and Stouthamer 2011). Relative sea-level rise in Chesapeake Bay, located on the middle Atlantic coast of the United States, is approximately double the current global rate (Lyles et al. 1988) and is rapidly increasing (NOAA 2012). Much of the Bay's shorelines are coastal marshes that are vulnerable to the effects of sea-level rise. Some areas of Chesapeake Bay, for example, Blackwater National Wildlife Refuge, have extensive open-water areas resulting from the loss of marshes by the formation and expansion of internal ponds (Stevenson, Kearney and Pendleton 1985a).

Processes governing surface elevation dynamics including decomposition, vegetation type and productivity and sediment deposition vary by salinity regime. Few, if any, studies compare surface elevation change and accretion at sites encompassing tidal freshwater, oligohaline and mesohaline wetlands. To have a better understanding of marsh vulnerability to sea-level rise, and to determine response differences among marshes of varying salinity, I conducted an evaluation of current soil surface elevation trends along an estuarine gradient in a subestuary of the Bay.

Surface elevation change is controlled by a variety of surface and subsurface processes, including geologic subsidence (deep subsidence) and uplift, autocompaction, compaction from storm events, hydrologic responses such as the shrink-swell of peat layers, decomposition, biomass production, and accretion. Many of these processes occur

in the marsh soil profile within meters of the surface. Accretion--the accumulation of organic and inorganic sediment--contributes positively to wetland vertical development; however, marshes may exhibit elevation loss despite high rates of accretion (Cahoon et al. 1999) due to shallow subsidence processes below the surface. Autogenic belowground and aboveground production contribute the majority of organic accretion. Belowground production through root growth comprises the majority of organic accumulation (Cahoon et al. 2006). In marshes where inorganic sediment is limited, biomass production can be a major mechanism to avoid submergence by positive surface elevation gain, as it is a more dominant contributor to accretion. Production is negatively affected by excessive flooding through waterlogging and salinity as well as herbivory (Odum, Odum and Odum 1995). Increases in salinity caused by saltwater intrusion also cause vegetation stress and mortality in communities not adapted to saline conditions. Additionally, increases in salinity decrease species richness and decrease seedling recruitment and germination following disturbance such as wrack deposition, burning, herbivory, or storms (Baldwin and Mendelssohn 1998b).

Production can be positively influenced through global change factors as well as being negatively impacted. For example, small increases in flooding can initially stimulate primary production by flushing the soil of toxins such as accumulated salts and sulfides, and by contributing nutrients that stimulate plant growth. Additionally, elevated CO₂, an effect of anthropogenic activities, can stimulate belowground biomass production and increase surface elevation of marshes (approximately 4 mm/yr increase) (Langley et al. 2009). It acts as a fertilizer, increasing belowground root production and thus accretion (Cherry, McKee and Grace 2009); however, the effects are more

pronounced in some vegetation types (C_3 plants such as *Scirpus americanus*) compared to others (C_4 plants such as *Spartina patens*) and can alter competition and community composition. Increased competition among species can decrease productivity and mitigate the positive surface elevation effects of elevated CO_2 (Cherry et al. 2009).

In addition to primary production, hydrology plays a major role in surface elevation change. Changes in hydrology can affect surface elevation and cause variability seasonally and interannually. Groundwater withdrawal due to drought and anthropogenic activities such as channelization and ditching can cause elevation losses that may mask long-term elevation trends (Turner 2004, Kroes and Hupp 2010, Cahoon et al. 2011). Dewatering of peat can shrink marsh soils, as pore spaces compact with the loss of porewater, decreasing surface elevation and thus increasing flooding (Kearney and Ward 1986). Flooding has both positive and negative effects on soil elevation. Excessive flooding can decrease organic accretion by negatively impacting primary production, thus lowering the surface elevation further by creating organic accretion deficits. Additionally, compaction due to surface flooding and hydrologically driven erosion can decrease surface elevation (Cahoon 2006). On the flip side, large precipitation events that increase groundwater and river discharge into wetlands and thus flooding, can cause the peaty soils to swell, increasing surface elevations (Whelan et al. 2005).

In addition to production, vegetation communities play a vital role in maintaining positive surface elevation relative to sea-level by exerting strong controls over accretion and subsurface dynamics. Velocities of tidal currents and overbank river discharge are decreased by the stems and roots of wetland plants, decreasing erosion and causing sediment to fall out on the marsh surface, trapping soil particles, and affecting the rates of

inorganic sediment accumulation in the soil and on the marsh surface (Wolanski et al. 2009). Natural levees form along tidal creek banks where stem density is high and creek-edge vegetation slows flow velocities, creates eddies where sediment can fall out, and traps sediments suspended in tidal waters.

Wetland vegetation also controls rates and pathways of decomposition by providing an organic substrate for microbial metabolism and by aerating the soil around root zones during the growing season, controlling elemental cycling such as iron (Neubauer et al. 2005). Nutrient cycling is also mediated by surface elevation. Surface elevation controls hydroperiod, which controls how long soils are inundated. Long inundation periods increases anaerobic conditions in soils, allowing for decomposition to slow as terminal electron acceptors are reduced and methanogenesis becomes more dominant, allowing organic matter to accumulate as microbes utilize less in respiration.

The persistence and health of wetland vegetation determines vulnerability to sea-level rise. Without organic accumulation and sediment accumulation, controlled by vegetation and hydroperiod, the marsh will not be able to accrete vertically at a sufficient rate to avoid submergence. Many effects of sea-level rise such as increased flooding, salinity, and sulfides create vegetation-elevation feedback mechanisms. Flooding creates anoxic conditions in the root zone, stressing vegetation (Cronk and Fennessy 2001), and decreasing primary productivity (Mitsch 1988). Increases in flooding lead to hypoxia, vegetation death, and the conversion of emergent macrophyte communities to unvegetated tidal flats (Morris et al. 2002).

Marshes of differing salinity regime may respond differently to the effects of sea-level rise. Tidal freshwater marshes and oligohaline marshes may not be adapted to

higher levels of salinity, and thus may be more vulnerable to salinity and sulfate intrusion. Because the vegetation is not adapted to saline conditions, it may also exhibit greater vegetation stress and mortality compared to salt and mesohaline marshes. Additionally, soils of tidal freshwater and oligohaline marshes have less refractory organic material compared to salt marshes, and are thus more subject to high rates of decomposition (Neubauer 2008).

Decomposition and elevation loss may also be stimulated by excessive amounts of nitrogen and phosphorus. Eutrophic surface waters high in nitrogen and phosphorus “fertilize” marsh vegetation, causing an increase in aboveground biomass production, and a decrease in belowground biomass production (Turner 2011). Fertilization alters the foraging strategy of plants, causing them to allocate fewer resources into root production, making them more vulnerable to disturbance such as storm surges (Delaune et al. 1990, Darby and Turner 2008a). Without strong anchoring systems, vegetation mats are torn, folded and displaced by storm surges creating internal ponds that cannot be colonized and are thus unvegetated (Kirwan, Murray and Boyd 2008). Internal pond formation and tidal network expansion are key processes in the loss of coastal marshes (Kearney, Grace and Stevenson 1988, Hartig et al. 2002).

Additionally, given that belowground biomass production is key in accretion and thus surface elevation control, with less root production organic accumulation decreases and surface elevation may drop, creating a positive feedback mechanism in which vegetation is exposed to greater increases in flooding and eutrophic conditions. Rates of organic accumulation, particularly important for sediment-starved marshes of Chesapeake Bay (Stevenson et al. 1985a), decrease as nitrogen amounts increase.

Coastal estuaries are becoming increasingly loaded with nutrients, particularly nitrogen (Nixon 1995, Vitousek et al. 1997). Excessive nutrients cause spikes in productivity which result in low amounts of dissolved oxygen as organic matter is decomposed (Nixon 1995). This process is eutrophication, a particularly prominent problem in areas with increased pressures from development and agriculture (sources of nutrient run off). With an increase in flooding of marsh surfaces due to increases in sea-level and storm frequency, comes an increase in inundation by eutrophic surface waters.

Amounts of nitrogen in coastal marshes are also affected by seepage of sewage effluent from rural septic systems. Groundwater becomes loaded with nitrogen and discharges into coastal wetlands, increasing both the NH_4^+ and NO_3^- levels (Valiela et al. 1997, Cole et al. 2006). With increased flooding of eutrophic water (Darby and Turner 2008b), a prominent problem in Chesapeake Bay, and increases from development pressures and higher population densities, amounts of nitrogen will increase and rates of organic accumulation will decrease as belowground mineralization increases (Swarzenski et al. 2008). An increase in decomposition of marsh soils may cause a decrease in surface elevation (Turner 2011), increasing vulnerability to sea-level rise and further flooding.

In addition to increases in salinity and exposure to eutrophic surface waters, sea-level rise delivers sulfates, present in sea water, further up the estuary. The introduction of sulfates may cause a shift in decomposition in marshes of intermediate or fresh salinity from methanogenesis (Weston et al. 2011), a slower mineralization process, to sulfate reduction, a metabolic pathway that mineralizes carbon more efficiently (Sutton-Grier et al. 2011). Increases in sulfate reduction will result in a larger presence of sulfides, an end

product of sulfate reduction, causing more vegetation stress through sulfide toxicity, which inhibits nutrient uptake and primary production (Portnoy and Giblin 1997).

Previous studies have focused on marshes of one salinity regime or type (Childers et al. 1993, Ford, Cahoon and Lynch 1999, Paquette et al. 2004), or have been conducted in different regions of the United States such as Louisiana (Rybczyk and Cahoon 2002, Bjerstedt 2011, Kearney, Riter and Turner 2011). Marshes of differing salinity regime may respond differently to the effects of sea-level rise. Tidal freshwater marshes may be more vulnerable to the effects of sea-level rise given that vegetation communities are not adapted to the presence of salts and are more readily decomposed (Craft et al. 2009). The addition of salinity also changes nutrient and mineral cycling, altering the biogeochemistry of the marsh (Megonigal and Neubauer 2009). Marshes at the salt-fresh boundary (oligohaline) may also exhibit greater vulnerability to increases in ambient salinity. Oligohaline marshes may be subject to some of the same biogeochemical changes as tidal freshwater marshes; however, being at the interface of the estuarine salt wedge, they may be exposed and altered sooner as saline intrusion pushes saltwater further up estuary. Additionally, tidal freshwater marshes may be buffered by fresh river discharge, although, in times of drought, discharge will decrease, pushing saltwater further inland (Megonigal and Neubauer 2009). By examining marshes along an estuarine salinity gradient, an assessment of differential responses and vulnerability can be determined.

In this study I sought to address the question, “How do elevation and accretion dynamics of coastal wetlands vary along estuarine salinity gradients?” To answer this question, surface elevation and accretion data were collected in a sub-estuary of

Chesapeake Bay over a four-year period. The primary objective of this study was to determine rates of elevation change and accretion, and specifically, to investigate whether coastal marshes have positive rates of elevation gain and are keeping pace with sea-level rise. I hypothesized that there would be differential rates of elevation and accretion change along a salinity gradient, and that some marshes would be vulnerable to the effects of sea-level rise by exhibiting overall elevation loss.

Materials and Methods

Site Description

The Nanticoke estuary is a large sub-estuary of Chesapeake Bay. It is microtidal [tidal range of 0.6 m (2 ft) (Fisher 1986)], ebb-dominated (Stevenson et al. 1985a), and extends from the Chesapeake Bay proper in Maryland into Delaware (Fig. 2.1). It is made up of tidal marshes of differing geomorphic types, including submerged-upland marshes in the lower estuary and estuarine-meander marshes in the upper estuary. The salinity ranges from about 15 psu at the mouth to 0 psu north of Sharptown, Maryland.

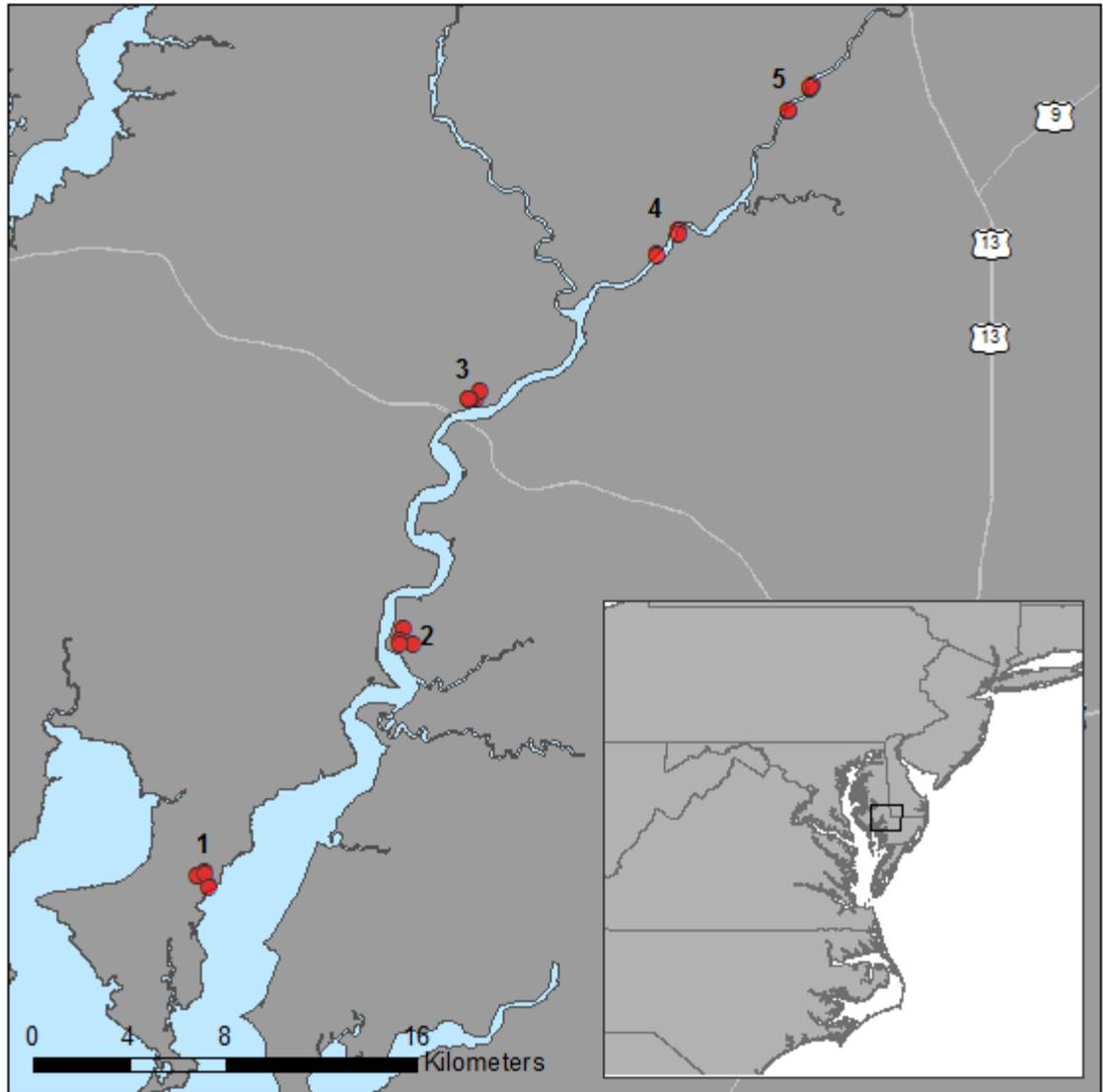


Figure 2.1: Location of study Sites on the Nanticoke River, MD and DE. Each Site is comprised of 3 replicate subsites (Map: Made by Leah Beckett, February 8, 2012. Sources: ESRI, Tele Atlas)

Five sites were established along the salinity gradient of the Nanticoke River (Fig. 2.1). At each of the five sites, three replicate subsites were established. The Sites are approximately 10 km apart and subsites are approximately 1-2 km apart. Subsites were chosen to be in interior marsh sections (e.g., not adjacent to channels) because of the reported vulnerability of interior marshes to internal ponding, salt and sulfide

accumulation, and decreased sediment deposition and loss (Stevenson et al. 1985a, Howard and Mendelssohn 2000, Neubauer et al. 2002). Interior marsh sites may be more vulnerable to loss than natural levee sites due to lower sedimentation (sedimentation is greater along the natural creek bank levee), higher concentrations of sulfides causing sulfide toxicity and vegetation death, and the effects of other factors causing vegetation death, stress, and eventual ponding such as the accumulation of salts due to lack of tidal flushing and lack of nutrient input. Subsites were established utilizing a random numbers table to generate direction and number of meters into marsh from creek or river bank.

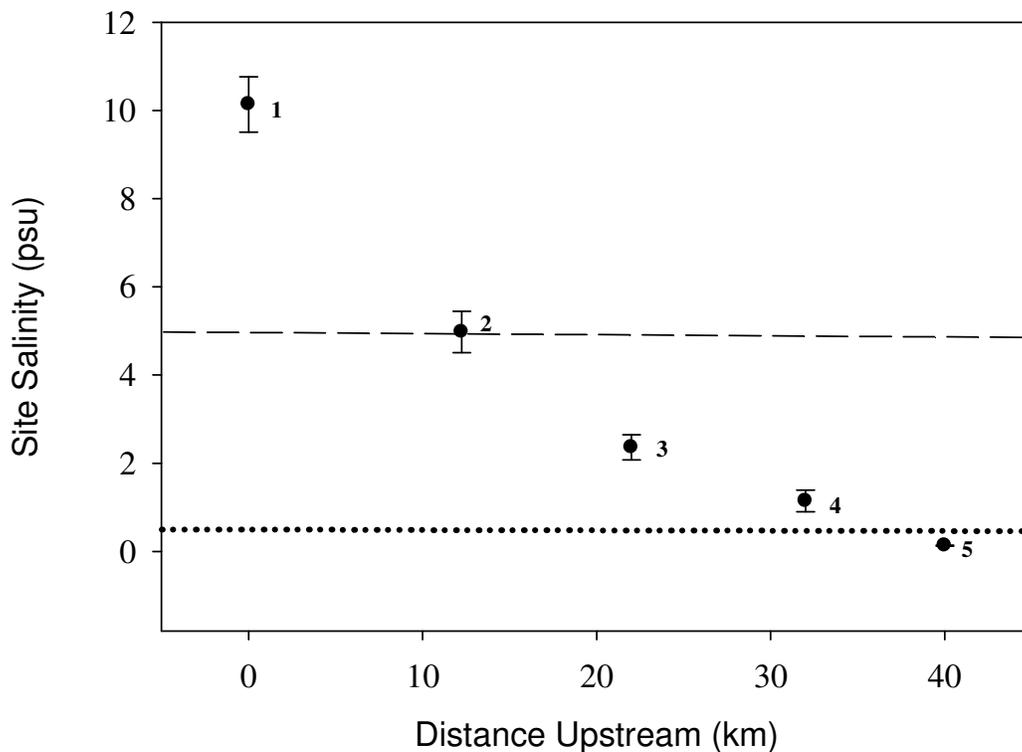


Figure 2.2: Salinity of study sites on the Nanticoke subestuary. Plotted values are mean \pm SE of measurements of marsh surface water and porewater salinity from 2007-2011. Porewater measurements were taken at 10-20 cm depths adjacent to surface elevation tables. Distance upstream is distance from Site 1 (0 km upstream). Dashed line represents

the oligohaline-mesohaline salinity boundary (5 psu) and dotted line represents the fresh-oligohaline boundary (0.5 psu). Numbers indicate Site number.

Site 1 is a submerged-upland marsh, and Sites 2-5 are estuarine-meander marshes.

Additionally, Sites 1, 2, and 3 are in marshes that have extensive tidal creeks. Sites 4 and 5 are interior marshes adjacent to the main river channel (although the river is significantly narrower at these Sites compared to Sites 1 and 2), and are relatively closer to uplands compared to Sites 1, 2, and 3.

Table 2.1: Study site descriptions. Soil series identified through Natural Resources Conservation Service Web Soil Survey (NRCS 2012)

Subsite	Dominant Vegetation	Primary Soil Series	Geomorphic Description or Location
Site 1	Undeveloped peninsula in Bay, dissected by tidal channels		
1A	<i>Spartina patens</i> , <i>Spartina alterniflora</i>	Honga peat (peat)	75 m from tidal creek, submerged upland
1B	<i>Spartina patens</i> , <i>Spartina alterniflora</i>		110 m from tidal creek, submerged upland
1C	<i>Juncus romerianus</i>		30 m from tidal creek, submerged upland
Site 2	Estuarine marsh adjoining upland and bordered by main river channel, dissected by tidal creeks, upland is residential/rural, marsh is 1500 m from upland at narrowest, 2500 m at widest, heavy muskrat activity (many small channels and muskrat lodges), occasionally burned by landowners		
2A	<i>Spartina cynosuroides</i> , <i>Iva frutescens</i> , <i>Spartina alterniflora</i>	Transquaking and Mispillion (mucky peat, muck, silty clay loam)	30 m from interior tidal channel, burned in November 2007
2B	<i>Spartina cynosuroides</i> , <i>Iva frutescens</i> , <i>Spartina alterniflora</i>		30 m from tidal channel that feeds into main river channel
2C	<i>Spartina cynosuroides</i> , <i>Iva frutescens</i> , <i>Spartina alterniflora</i> , <i>Kosteletzkya virginica</i>		30 m from tidal channel, burned every November

Site 3	Estuarine marshes adjacent to Rte. 50 bridge (bridge is elevated above marsh), adjacent to upland, dissected by tidal creeks, subsites are on tidal creeks, marsh is 1000 m wide (upland to main river channel) at widest, 200 m wide at narrowest upland to main river channel		
3A	Polygonum arifolium, Bidens laevis, Hibiscus moscheutos	Nanticoke Silt Loam (silt loam)	350 m from main river channel, 40 m from tidal channel, 475 m from upland
3B	Polygonum arifolium, Typha angustifolia, Bidens laevis		200 m from upland, 30 m from tidal creek, 600 m from main river channel
3C	Polygonum arifolium, Hibiscus moscheutos, Typha latifolia		40 m from tidal creek, 425 m from upland, 350 m from main river channel
Site 4	Narrow band of estuarine marsh on either side of the main river channel, bordered by 500 m of tidal swamp, swamp adjacent to upland, subsites off main river channel		
4A	Polygonum arifolium, Acorus calamus, Leersia oryzoides	Nanticoke and Mannington (silt loam, silty clay loam)	75 m from main river channel, 55 m from swamp, 700 m from upland/rural/residential area
4B	Schoenoplectus fluviatilis, Polygonum arifolium, Impatiens capensis		150 m from main river channel, 100 m from swamp, 700 m from residential/rural upland
4C	Acorus calamus, Polygonum arifolium, Cicuta maculate		500 m from upland (500 m are swamp), 40 m from main river channel, western side of river
Site 5	Narrow band of estuarine marsh bordered by small wooded buffer and large suburban area		
5A	Polygonum arifolium, Bidens laevis, Acorus calamus, Peltandra virginica	Nanticoke and Mannington (silt loam, silty clay loam)	40 m from main river channel, 80 m from non-tidal woods, 250 m from suburban area
5B	Polygonum arifolium, Murdannia keisek, Peltandra virginica		50 m from main river channel, 35 m from non-tidal woods, 250

5C	Acorus calamus, Mentha arvensis, Polygonum arifolium	m from suburban area 50 m from main river channel, 5 m from non-tidal woods, 225 m from suburban area, 400 m from Woodland ferry which crosses main river channel
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Elevation

Deep-Rod Surface Elevation Tables (SET) were utilized (Cahoon et al. 2002) (Appendix 1). SETs were installed at each of the 15 subsites in August 2007. Baseline measurements were taken in October 2007, and subsequent measurements were taken twice annually in April and October through 2011. Surface elevation tables measure elevation change between the surface of the marsh and the bottom of the benchmark rods (a reference plane established during installation). SETs are a portable, mechanical device for measuring elevation change on the marsh surface. It consists of an arm with 9 pins for measuring the height of the marsh surface, and is attached to the benchmark for each measurement, at up to 8 different positions. For each SET, four of the eight positions were used. Pin height differences between the baseline and each subsequent reading are used to calculate elevation change. Processes affecting the thickness of the stratigraphic profile above the basement sediments (surface elevation) such as root growth, swelling and contracting of peat due to water level changes, accretion and surface erosion are accounted for in SET measurements (though the effects of each process independently cannot be distinguished without further modifications of the SET such as a shallow benchmark as well as the Deep-Rod SET or multi-season monitoring, etc.). SETs have been used in a variety of coastal habitats (Cahoon et al. 2006) such as

mangroves, salt marshes, intertidal zones and tidal freshwater marshes (Day et al. 1999, Whelan et al. 2005, Lovelock et al. 2011) and are an accurate [± 1.5 mm (Cahoon et al. 2000)] way to collect localized elevation change data (Cahoon et al. 2002).

Accretion

Accretion was measured using feldspar marker-horizon plots and cryogenic soil cores (Cahoon, Lynch and Knaus 1996, Cahoon et al. 2000) (Appendix 1). Powdered feldspar is laid down and over time as organic and inorganic materials accrete on top, rates of accumulation can be estimated by taking cores and measuring the amount of material from the white feldspar layer to the surface. Three marker horizon plots were laid at each SET subsite (3x15 subsites = 45 feldspar plots). Plots were laid concurrently with baseline SET measurement (October 2007). Cores were taken each April and October concurrently with SET elevation measurements. Cores were taken cryogenically to avoid compaction during core removal by inserting a 0.32 cm diameter copper tube with a bullet welded to the tip into the feldspar plot, pumping liquid nitrogen into the copper tube, and cutting the core out (Cahoon et al. 1996). For each accretion core, the rough sides were shaved off with a knife and two measurements were made on different faces of the core and averaged. Accretion was measured as the distance from the top of the white feldspar layer to the surface of the soil using a Vernier caliper.

Data Analysis

Pin measurements for each of the four positions were averaged. Position averages were averaged for an overall elevation reading for each of the fifteen SETs.

Elevation readings were plotted against time (x =time, y =elevation) for each SET and simple linear regressions were fit to the data using no-intercept models (that is, the

line was forced through the origin). The slopes of the regression lines were used as rates of elevation change and analyses were subsequently conducted on the slopes.

Rates of elevation change of subsites were averaged by site (e.g., 1A, 1B, 1C were averaged for a rate at Site 1) and an analysis of variance (ANOVA) was conducted among sites using Proc Mixed procedure of the SAS system, version 9.1 (SAS Institute, Cary, North Carolina).

Subsites were averaged by salinity regime (1A,1B,1C,2A,2B,2C, mesohaline, 3A,3B,3C,4A,4B,4C, oligohaline, 5A,5B,5C, tidal fresh) and an analysis of variance was conducted among salinity regimes (mesohaline vs oligohaline vs tidal freshwater). Site means were also averaged by salinity regime (1,2 mesohaline, 3,4, oligohaline, 5, tidal fresh), and an analysis of variance was conducted between salinity regime. The latter ANOVA was conducted to account for between-Site variation which may affect whether subsites from different Sites are true replicates of salinity regime, or whether the Sites themselves are the replicates.

Similarly, accretion measurements at each subsite were plotted against time (days) (x =time, y =accretion) and slopes of regression lines fit to each curve were used for average rates of accretion. Slopes were averaged by site (e.g. the average of 1A, 1B, 1C for Site 1) and an ANOVA was conducted to examine the differences among sites using SAS.

Rates of shallow subsidence were calculated for each SET by taking the difference between average annual accretion (mm/yr) and average annual elevation change (mm/yr) (accretion-elevation) (McKee, Cahoon and Feller 2007). These were also compared using ANOVA.

Given the large spatial scale of sites and replicates, a critical level of 0.1 was considered appropriate for assigning statistical significance.

Salinity

Salinity was measured using a conductivity salinity meter (YSI 30-25, YSI Incorporated, Yellow Springs, Ohio). Channel salinity measurements were taken 2-4 times a year for the duration of the study. Measurements were taken by placing the probe in the adjacent river or tidal creek at a depth of approximately 30 cm. Porewater measurements were taken within 4 m of each SET site by placing the probe in a pilot hole (made with a PVC pipe or stick) at a depth of approximately 10 cm. Porewater and channel salinity measurements were averaged together at each SET site across season and year. Site averages are comprised of subsite averages, averaged.

Results

Accretion

Accretion rates were uniformly high across the estuary (~7-19 mm/yr). Rates of accretion did not significantly differ among Sites or salinity regimes; however, accretion was significantly greater than zero at all Sites ($p > 0.05$). Accretion rates ranged from 7.35 mm/yr at subsite 1A to 19.38 mm/yr at subsite 5B (Fig.2.3). Contrary to expectations, accretion rates were not highest at Sites having the greatest elevation gain, but were highest at Sites with varying rates of elevation change, including some Sites that are decreasing in surface elevation (Site 4). There was not a linear relationship between accretion and elevation change. Despite high rates of accretion, Sites are still losing elevation, and thus have shallow subsidence below the surface. Accretion rates were

slightly lower at Site 1, a submerged-upland marsh (8 mm/yr), and Site 3, an estuarine marsh (10 mm/yr) compared to the other Sites, though not statistically significantly so (Fig.2.4). These results indicate that short-term accretion measurements are not always good indicators of overall elevation trends.

Additionally, there were no clear seasonal, spatial or annual patterns of accretion. However, there was an initial effect of laying new feldspar layers. Initial accretion rates (during the first time interval) were higher compared to later accretion rates (during subsequent time intervals) due to a "fertilization" effect of laying the plots (Fig.2.3). This may be because plant growth was stimulated on the new surface created by the plot because of lack of competition, as well as shallow compaction over time.

Surface Elevation

Mid-estuarine, oligohaline marshes are decreasing in surface elevation (Fig.2.3). Tidal freshwater marshes near in the upper reaches of the estuary, and mesohaline marshes at the mouth of the estuary are increasing in surface elevation (Fig.2.3). Rates of surface elevation change differed significantly among marshes of differing salinity regime ($F_{2,2}= 46.00, p=0.02$; $F_{2,12}= 2.82, p=0.09$). Oligohaline marshes (Sites 3 and 4) decreased in surface elevation significantly more compared to mesohaline marshes (Sites 1 and 2) ($p=0.03, p=0.07$) and tidal freshwater marshes (Site 5) (Tukey means comparison test) ($p=0.03, p=0.07$). Oligohaline marshes exhibited rates of surface elevation loss of -8.7 ± 3.81 mm/yr. Surface elevation change in tidal freshwater marshes during the study period was on average 4.48 ± 5.39 mm/yr and was 2.05 ± 3.81 in mesohaline marshes. Rates of surface elevation change did not differ between tidal freshwater marshes and mesohaline marshes.

Despite differing significantly when grouped by salinity regime, as individual Sites, rates of surface elevation change did not differ significantly across the estuary ($P > 0.05$). Additionally, elevation change did not significantly differ from zero ($P > 0.05$) at any Site. Rates of surface elevation change vary dramatically spatially (even by subsite) and temporally (seasonally and annually). Because of the high variation (and thus high standard errors) within Sites, there are no significant differences in rates of surface elevation change among Sites. For example, within Site 5, which has an average rate of surface elevation change of 4.47 ± 4.27 mm/yr, subsite 5A has a positive surface elevation gain from October 2007-October 2011 of 12.8 mm/yr, in contrast, subsites 5B and 5C increased only 1.8 mm/yr (5B) and decreased -1.2 mm/yr (5C) during the same time period (Fig. 2.3). Additionally, subsites 5B and 5C exhibit nearly identical seasonal (April-October) variations in surface elevation change, decreasing in April and October of 2008 and April of 2009 and increasing in April 2010, and 5A, having not only a dissimilar rate of surface elevation change, also reflects different seasonal patterns compared to 5B and 5C (Fig.2.3). Baseline measurements (Day 0) were taken in October 2007, and each subsequent sampling event alternates between April and October (every other sample period is in April and alternates with every other sample period being October, except for October 2010, during which no measurements were taken). A similar discrepancy among subsites can be seen at Site 3 which has an average rate of surface elevation change of -7.59 ± 9.42 mm/yr. Subsite 3A decreased in surface elevation -26.2023 mm/yr. In contrast, subsite 3B decreased -0.80 mm/yr, and 3C increased 4.24 mm/yr. Site 4 ranged from -23.14 mm/yr to -0.19 mm/yr. Site 1, a mesohaline submerged-upland marsh, exhibits very little within-Site variability and has an average

increase of surface elevation of 1.28 ± 1.57 mm/yr. Overall, there are replicate subsites (1A, 1B, 2B, 2C, 3C, 5A, 5B) at each Site that had positive surface elevation gains during the study period; however, only Sites 1,2 and 5 had overall positive average rates of surface elevation change (Fig. 2.4). Site 5, tidal freshwater marshes, exhibited the highest rates of surface elevation gain with an average increase of 4.47 mm.yr (Fig.2.4). Sites 1 and 2 also had positive surface elevation gains of 1.2 ± 1.6 mm/yr and 2.83 ± 3.9 mm/yr, respectively (Fig.2.4). Oligohaline, mid-estuary marsh Sites 3 and 4 had surface elevation decreases during the study period of -7.59 ± 9.4 mm/yr and -9.75 ± 6.89 mm/yr, respectively. Site averages were not significantly different from zero.

There were no clear seasonal patterns. At some subsites, surface elevation increased during October measurements, and at other subsites, it decreased during October measurements. There were also no clear annual trends such as surface elevation increase across the system during one sampling event relative to the previous measurement.

Shallow Subsidence

All Sites and subsites exhibit shallow subsidence, or a decrease in surface elevation due to processes in the top 3-5 m excluding accretionary processes (Cahoon et al. 1995). Oligohaline marshes are subsiding significantly more than marshes of differing salinity ($p=0.09$)(Fig. 2.6). Oligohaline marshes are subsiding 21.64 ± 5.14 mm/yr. Tidal freshwater marshes are subsiding 10.56 ± 4.64 mm/yr and mesohaline marshes 9.84 ± 1.87 mm/yr. Oligohaline marshes exhibit significantly higher rates of shallow subsidence compared to mesohaline marshes ($p=0.04$); however, rates of shallow subsidence do not significantly differ between oligohaline marshes and tidal freshwater marshes (Tukey

test, $p=0.11$). Shallow subsidence ranged from 24.58 ± 7.29 mm/yr at Site 4 to 7.36 ± 1.97 mm/yr at Site 1 (Fig. 2.5). The maximum shallow subsidence was 39.04 mm/yr at subsite 4B and the minimum was 1.76 mm/yr at subsite 5A. Rates of shallow subsidence do not significantly differ by Site; however, rates of shallow subsidence significantly differ from zero at Sites 2 ($p=0.05$), 3($p=0.0075$) and 4 ($p=0.0014$). Subsites that are significantly decreasing in surface elevation, such as 3A and 4B also have high rates of shallow subsidence. Interestingly, though, there is high spatial variability in shallow subsidence rates within Sites and among Sites. For example, subsites 4A and 4B are approximately 100 m apart in the marsh; however, rates of shallow subsidence are dramatically different between the two. Subsite 4A has a shallow subsidence rate of 19.13 mm/yr compared to 4B which has a shallow subsidence rate of 39.04 mm/yr, one of the greatest rates of shallow subsidence along the study gradient. 4A and 4B have very different rates of surface elevation change (-0.19 mm/yr; -23.14 mm/yr; respectively); however, they have nearly the same rates of accretion (18.9 mm/yr; 15.9 mm/yr; respectively), which are also some of the highest rates of accretion along the study gradient. Subsite 4A exhibited low variability in rates of surface elevation change while 4B had high variability.

Salinity

Salinity level of pore, surface, and channel water fluctuate seasonally and annually, as to be expected. In June and July, porewater salinities were higher compared to corresponding channel salinities; however, in August and September, channel salinities were higher than porewater salinities, at a depth of 30 cm, in adjacent marshes. Both porewater and channel salinity levels were higher at all Sites during the summer months

of June, July and August compared to April levels. Site 5, tidal freshwater marshes, with average salinity levels of 0.13 psu, had the most stable salinity and did not exceed 0.3 psu during the study period. Measured channel salinity levels were highest in late October, 2007, and the maximum value was 21.7 psu at subsite 1A, double the average salinity level of 10.14 ± 0.62 at Site 1 during the study period.

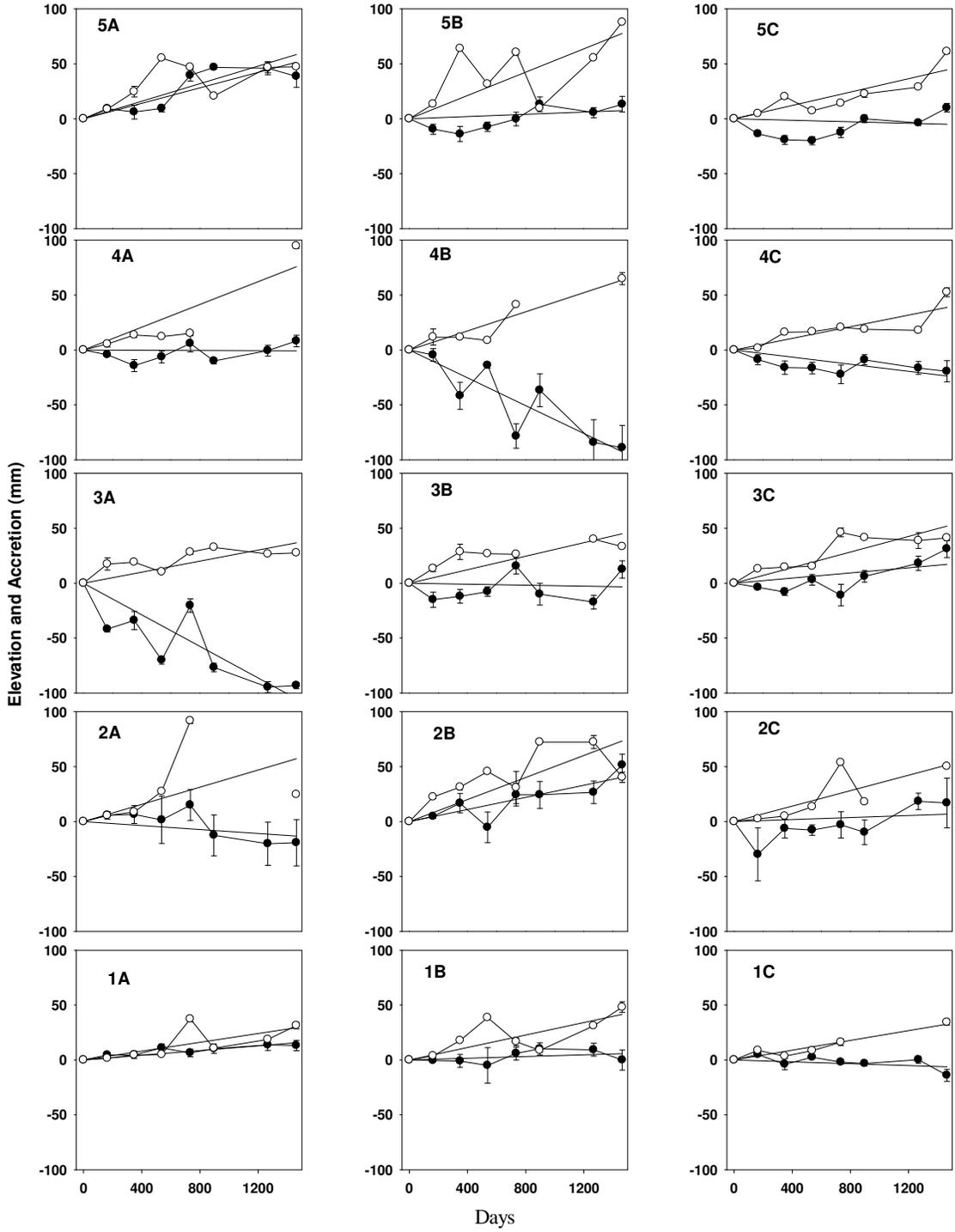


Figure 2.3: Rates of elevation change and accretion for each replicate SET by days from baseline measurement. Day 0 represents baseline measurement in October 2007. Sites 1,

2 are mesohaline, Sites 3, 4 are oligohaline, Site 5 is tidal fresh. (●=elevation
○=accretion)

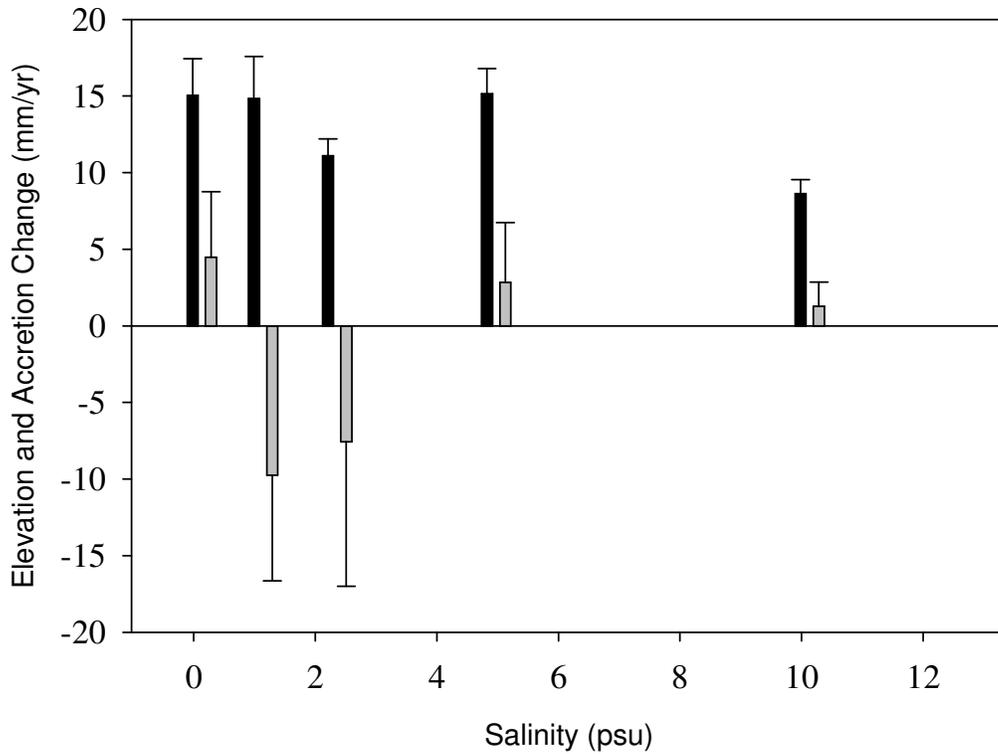


Figure 2.4: Rates of elevation change and accretion + SE along the Nanticoke estuary from October 2007 (baseline) through October 2011. Each site mean is an average of three replicates subsite means (n=15) (■=accretion □=elevation change).

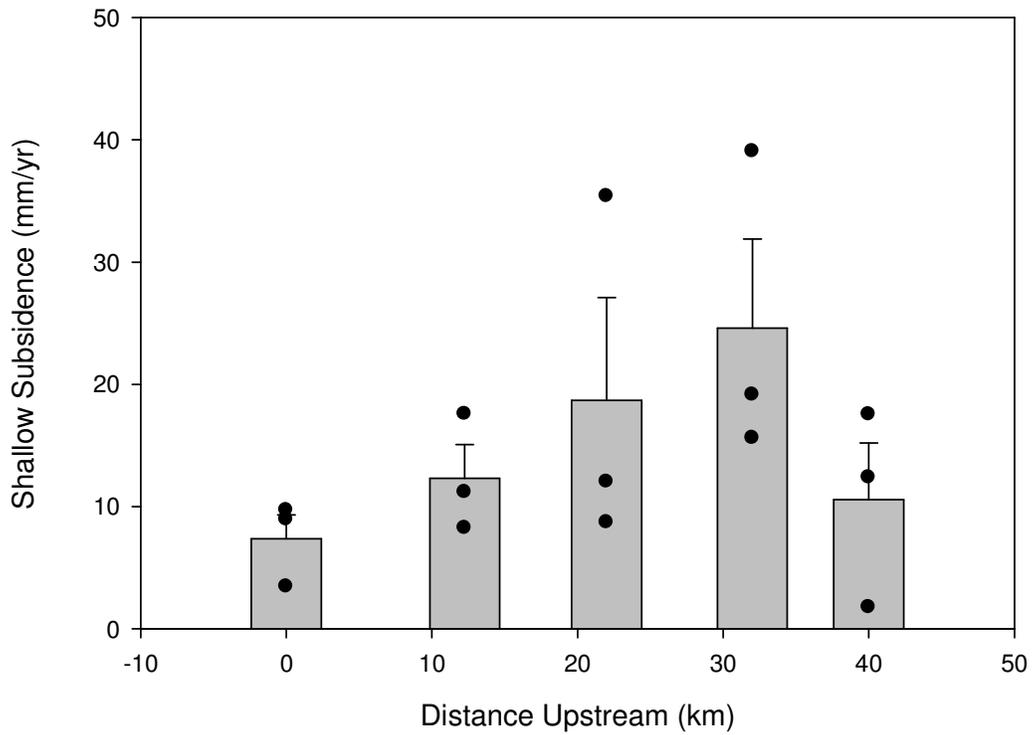


Figure 2.5: Rates of shallow subsidence (accretion-elevation) +SE from 2007-2011. Positive values for subsidence indicate a decrease in elevation. Distance upstream indicates distance from mesohaline Site 1 (0 km) proceeding to tidal freshwater (40 km). Dots (●) are subsite shallow subsidence values. Vertical bars are Site means (mean + SE of three subsite values).

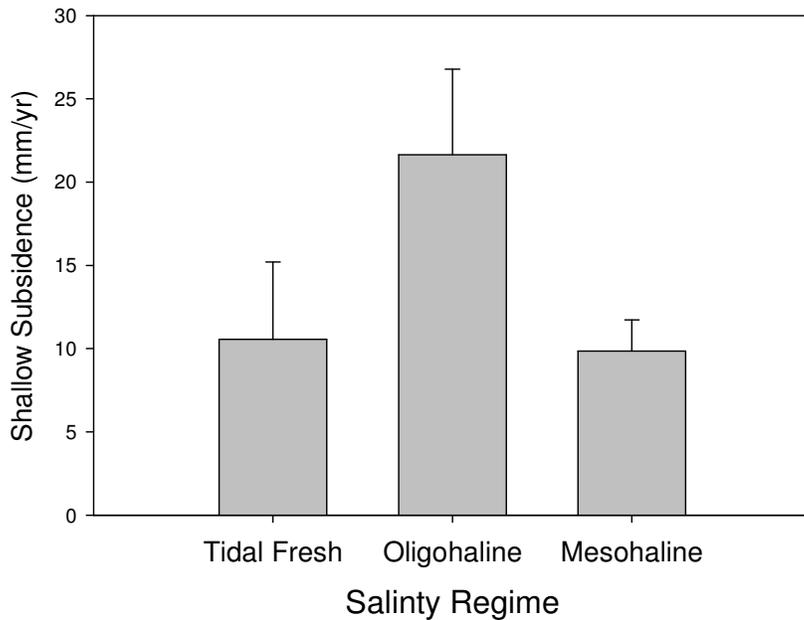


Figure 2.6: Rates of shallow subsidence +SE by salinity regime from 2007-2011.

Discussion

Coastal marshes of the Nanticoke River are not keeping pace with current rates of relative sea-level rise (3-4 mm/yr, Lyles et al. 1988), and mid-estuary sites are losing surface elevation at rates more than six times the rate of eustatic sea-level rise (-26-4 mm/yr elevation change) despite rates accretion of 9-15 mm/yr (Fig. 2.3). Previous studies have examined rates of elevation and accretion in salt marshes and mangroves (Cahoon et al. 1995, Ford et al. 1999, McKee et al. 2007), but few studies have examined surface elevation and accretion in tidal freshwater and oligohaline wetlands, and none have compared rates along salinity gradients. This study found that rates of subsidence are higher in oligohaline marshes compared to tidal freshwater and mesohaline marshes.

Some of the rates of elevation change and accretion are higher than those reported in other systems, even within the watershed of Chesapeake Bay. Accretion rates in the Pocomoke River floodplain, approximately 55 km south east of the Nanticoke, were much lower, ranging from 0.6 to 3.6 mm/yr (Kroes and Hupp 2010). Shallow subsidence at some Nanticoke Sites was more than double the highest rate reported for the Pocomoke River (6.4-11.9 mm/yr) (Kroes and Hupp 2010) despite having greater accretion. In a study reviewing average rates of elevation change and accretion, U.S. Atlantic Coast wetlands increase on average 4 mm/yr, with accretion rates of ~3.5 mm/yr (Cahoon et al. 2006). Of those wetlands, estuarine marshes increased in elevation ~ 3 mm/yr, and accreted ~2.5 mm/yr (Cahoon et al. 2006).

Cahoon et al. (1999) reviewed rates of elevation change and accretion from eighteen studies conducted around the world. The highest reported rate of elevation loss was in a high marsh in Washington State at -6.5 ± 2.6 mm/yr with an accretion rate of 3.7 ± 0.6 mm/yr (Cahoon et al. 1999). Of the eighteen studies, seven reported elevation loss; the other eleven had positive elevation gain. Comparatively, of the six oligohaline marshes examined at the Nanticoke, only one had elevation gain, and the other five lost elevation at rates ranging from -26 mm/yr to -0.2 mm/yr. Erwin et al. (2006) compared rates of elevation and accretion using SETs in vegetated and unvegetated sections of marsh along the U.S. Atlantic Coast and found elevation change values ranging from -5.4 ± 0.1 to 11.5 ± 1.5 mm/yr (both values were in unvegetated marsh ponds). Rates of accretion ranged from 3.1 ± 0.7 in a vegetated portion of a marsh to 24.4 ± 1.8 in an unvegetated portion (Erwin et al. 2006). Globally, reported rates of elevation change and accretion are fairly consistent and are far less variable than along the Nanticoke. For

example, Howe et al. (2009) utilized SETs in salt marshes in southeast Australia and found rates of elevation change that ranged from 2.02 mm/yr to 1.78 mm/yr.

Additionally, their rates of accretion in salt marshes ranged from 1.03 mm/yr to 2.25 mm/yr (Howe, Rodriguez and Saco 2009).

Surface elevation loss may be caused by a variety of factors and their interactions. I speculate that elevation loss is an effect of salinity intrusion due to sea-level rise and changes in precipitation distribution (i.e. droughts) and thus river discharge. As droughts become more common (as predicted by climate change), freshwater inputs into coastal marshes may become more sporadic or may decrease (Neubauer and Craft 2009). With less freshwater inflow, saline water will be pushed further up-estuary, exposing sites to higher salinities. Higher salinities may result in increased rates of decomposition in oligohaline and tidal fresh marshes by inducing a shift in the dominant respiration pathway from methanogenesis to sulfate reduction (Weston, Dixon and Joye 2006). Higher rates of decomposition result in drops in surface elevation, for example, as may be seen at our oligohaline sites (Sites 3 and 4). Based on high rates of elevation loss, and higher bulk densities at Sites 3 and 4 (Beckett, unpublished data), possibly indicating organic matter decomposition, I hypothesize that Sites 3 and 4 are exhibiting higher rates of decomposition compared to Site 5, and potentially compared to Sites 1 and 2 as well. Although Sites 1 and 2 are more saline and thus exposed to more sulfates (more sulfate reduction, a more efficient decomposition pathway), they may be dominated by vegetation that contributes more refractory organic matter to the soil compared to Sites 3 and 4, thus exhibiting lower decomposition rates. Tidal freshwater marshes are

dominated by low-refractory vegetation; however, this may be compensated for by having methanogenesis, a less efficient mineralization process, as the dominant pathway.

A role for nitrogen in elevation loss?

Wastewater discharged into groundwater as seepage from septic tanks in adjacent coastal communities can deliver large amounts of nitrogen (typically in the form of NH_4^+ or NO_3^-) to coastal estuaries through groundwater discharge (Martinetto et al. 2010, Cardoni et al. 2011, Valiela et al. 1997, Cole et al. 2005, Cole et al. 2006). Excessive nitrogen increases rates of mineralization and decomposition and can cause a breakdown of the root network and the organic soil matrix, which in turn would cause a decrease in elevation (Kearney et al. 2011). Sites 3 and 4 may receive excessive amounts of nitrogen from nearby farm fields, neighborhoods, or wastewater effluent.

Spatial variability: a precursor to wetland loss?

Spatial heterogeneity not only causes variability in measurements, but may also be an indicator of degradation at marsh sites. Tidal marshes typically have vegetation-driven microtopographic features such as hummocks and hollows; however, these features form in response to environmental stress in areas of the marsh flushed less frequently with tidal water (such as interior marsh) and have lower redox potentials and higher concentrations of salts and sulfides (Stribling et al. 2006). Hummocks and hollows may be typical features of healthy tidal marshes; however, they may also be indicators of degradation, as in the expansion of tidal creek networks and the formation of internal ponds (Kearney et al. 1988). As unvegetated portions of the marsh such as tidal creeks and ponds increase in area in response to stress, spatial variability and elevation change and accretion variability may increase.

The lack of statistically significant differences of mean elevation change and accretion among the five Sites was due to high within-Site variability. Rates of elevation and accretion change exhibited dramatic variability among subsites, as at Sites 3 and 4. Spatial and temporal variations in accretion rates are common given differences in depositional environments (Kearney, Stevenson and Ward 1994). For example, shoreline marshes receive more allocthonous sediment compared to interior marshes (Kearney et al. 1994, Neubauer et al. 2002) increasing rates of sediment accretion, which is also affected by seasonally dependent hydroperiod and land use changes in adjacent areas (Kearney and Stevenson 1991). Erwin et al. (2006) found significant differences in accretion and elevation change between ponds and vegetated marsh. Comparatively, our rates of elevation change and accretion include both vegetated and unvegetated portions of the marsh given that we had some Sites that exhibited microtopography (Site 2), including vegetated hummocks and unvegetated hollows, which may also explain some portion of our high variability within Sites. Because our SETs were placed randomly, measurements included hummocks and hollows.

The influence of water level on surface elevation measurements

Elevation and accretion measurements were taken at varying tidal stage and season. This may account for some of the variability in measurements within Sites, and at subsites over time given that peaty soils swell with increases in groundwater, as at high tide and during spring measurements, and compress at low tide and during growing season when evapotranspiration rates are high (Cahoon et al. 2011).

In addition to tidal stage, another important hydrologic factor to consider is the level of groundwater. Anthropogenic changes in local hydrology can cause a drop in

elevation in tidal wetlands (Turner 2004). Specifically, groundwater withdrawal due to development pressures, a historical cause of land subsidence in Chesapeake Bay (Kearney and Stevenson 1991), may cause a dewatering of the peat, thus lowering the surface elevation of the marsh (Kennish 2001). Channelization and dredging also impact floodplain and wetland hydrology. The channelization of a river can decrease the level of groundwater in the surrounding floodplain by increasing drainage causing shallow subsidence (Kroes and Hupp 2010). Shallow subsidence can also be escalated in channelized systems through a decrease in floodplain sediment storage (Kroes and Hupp 2010). The Nanticoke River is a thoroughfare for barges carrying sand and gravel, and has a navigation channel maintained through dredging to a depth of 4 m (12 ft) and a width of 30.5 m (100 ft) (United States Army Corps of Engineers 2012). Sediment storage in the watershed of the Nanticoke may be decreased due to dredging and channelization increasing shallow subsidence.

Implications for coastal wetland persistence

In order for coastal marshes to keep pace with sea-level rise they must maintain a surface elevation greater than the rate of relative sea-level rise. Accretion rates are uniformly high across the Nanticoke estuary and the Sites are not increasing in surface elevation proportionally to accretion, therefore shallow subsidence and deep subsidence are important factors. The Chesapeake Bay is subsiding approximately 2 mm/yr (Boon, Brubaker and Forrest 2010), but some subsites have shallow subsidence of up to 39 mm/yr! Given that erosion or lack of accretion is not the primary cause of surface elevation loss in these marshes, identifying and understanding anthropogenic factors contributing to shallow subsidence is key to preventing further loss of Chesapeake Bay

wetlands. Mid-estuary, oligohaline marshes may be more subject to loss compared to brackish and tidal freshwater marshes; however, as sea-level rises and saltwater is pushed further up estuary, tidal freshwater marshes may be threatened as well. Salinity may be a major factor in marsh degradation and loss. Other potential explanatory variables include proximity to development which could increase the amounts of nutrients delivered to coastal waters and marshes and groundwater withdrawal. Increases in decomposition rates resultant of saltwater or sulfate intrusion, or by increased nitrogen from development may affect marshes of differing salinity differently, as well, given that the amount of refractory material differs among marshes of different salinity regime.

Chapter III: Coastal Tidal Marsh Stratigraphy in Submerged Upland and Estuarine Meander Marshes Along a Salinity Gradient

Abstract

Surface and subsurface processes contribute to coastal wetland subsidence.

Decomposition, autocompaction, groundwater withdrawal and erosion can cause soil surface elevation deficits that lead to degradation and drowning. These processes are functions of coastal wetland soil stratigraphy, determined in part by geomorphic type. To evaluate the effects of stratigraphic features on coastal wetland subsidence, cores were taken at 5 Sites (3 cores/Site) where soil elevation was monitored. Sites were established in estuarine meander and submerged upland marshes along a salinity gradient. Cores were sectioned and analyzed for bulk density, porewater salinity, organic matter content and depth to basal peat. Shallow subsidence was highest at Sites with intermediate masses of organic matter, intermediate soil salinities and intermediate depths. Subsidence was related curvilinearly to total mass of organic matter in a core and depth. Masses of organic matter increased with depth, and organic matter content increased with salinity (as a function of distance from sediment source). Estuarine marshes at the mouth of the estuary were older, deeper, and more organic compared to the shallower more mineralogenic freshwater marshes upstream. Submerged upland marshes were the youngest, shallowest and most inorganic and had the lowest rates of subsidence. Stratigraphic features may not be as important as salinity regime in determining rates of shallow subsidence; however, submerged upland marshes may be less subject to subsidence compared to estuarine meander marshes due to high bulk density, upland soils that are less subject to compaction and decomposition than meander marshes.

Introduction

A major cause of coastal wetland loss is the inability of marshes to maintain positive surface elevation relative to mean high tide, resulting in submergence and drowning. Understanding the mechanisms by which marshes fail to do so is essential to predicting and potentially preventing future losses of coastal wetlands. Coastal wetland surfaces are the proverbial tip of the iceberg when considering potential dynamics regulating elevation change. Many processes contribute to subsidence including deep geologic processes (faulting, isostasy, compaction) and shallower processes (groundwater withdrawal, dredging, decomposition) (Meckel, ten Brink and Williams 2006). The underlying substrate -- a stratigraphy of alternating peat, silt and clay layers-- also exerts a large influence on surface elevation change and subsidence dynamics (Allen 2000b), and it is only by considering the stratigraphy in addition to the surface dynamics that a holistic picture can be assembled and the vulnerability of a marsh evaluated.

Stratigraphic patterns are results of geomorphic dynamics. Coastal marshes formed after the last ice age during the advent of the Holocene epoch when river valleys and deltas were drowned by rising sea-levels. The development of coastal marshes closely parallels historical rates of sea-level rise. Mean water level controls the elevation of coastal wetlands (the marsh platform) by determining sediment supply and organic matter build up (McKee and Patrick 1988), and sea-level rise must be slow enough to allow fine sediments to settle out and soils to accumulate organic matter (Ward, Kearney and Stevenson 1998). Holocene estuarine sequences are formed on basal till, bedrock, or glaciofluvial sediments that form undeformable "basement surfaces" (Allen 1999). Sequences of silts (historical mudflats), peats (historical marshes), and some sands

(historical tidal creeks) form meters deep on top of historic deltaic sediments (Allen 1999). Accumulation of sediments is determined by the level of mean high tide (Allen 1999), and, thus, controlled by sea-level. Alternating layers of peats and silts correspond to trends in sea-level rise; silty, clay layers formed during periods of rapid sea-level rise, and peats formed under conditions of stable or falling sea-level (Allen 2000a).

The objectives of this study were to evaluate differences in coastal marsh stratigraphy and geomorphology at sites where surface elevation tables (SET) were installed (Sites) in order to examine the relationship between subsidence and the underlying substrate. Specifically, I hypothesized that Sites with greater masses of organic matter, expressed on an areal basis (deeper, more organic-rich profiles) would exhibit higher rates of subsidence or have an increased vulnerability to the effects of sea-level rise.

The amount and quality of organic matter in coastal wetland stratigraphic profiles affects rates of subsidence. Peat is compressible, has large pore spaces, and has more potential to change in volume proportionally than silty, sandy or clayey soils. The volume of organic-rich stratigraphic layers expands and contracts in response to changes in groundwater hydrology. When groundwater is lowered due to withdrawal or drought, peat shrinks, contracting to a smaller volume and lowering the wetland surface in elevation (Kearney and Stevenson 1991). When the level of groundwater increases, peat swells, expanding pore spaces and increasing wetland surface elevation at the surface of the profile (but not at depth) (Whelan et al. 2005). Additionally, decomposition disproportionately affects organic-rich layers compared to mineral layers. Peaty stratigraphic layers provide more substrate (electron donors) for anaerobic bacteria to

decompose compared to silt and clay layers. The volume of peat is reduced through decomposition, which may lower wetland surface elevation.

A good understanding of the underlying stratigraphy may thus explain elevation and subsidence trends. Understanding if and how variables vary among marshes of differing salinity and geomorphic type may also lend insight into vulnerability to sea-level rise.

Materials and Methods

The Nanticoke River estuary is a microtidal tributary of Chesapeake Bay with a significant riverine sediment supply (Kearney and Ward 1986). Nanticoke marshes are mature with characteristically extensive and dynamic drainage systems that migrate laterally and apically within periods of decades (Ward et al. 1998).

Fifteen stratigraphic cores were taken adjacent to Sites where surface elevation tables (SET) had been installed (See Chapter II). Cores were taken utilizing an Eijkelkamp peat sampler (Eijkelkamp, Giesbeek, Netherlands) (Jowsey 1966). Cores were taken in 50 cm sections beginning from the surface of the marsh to the basement surface (in the case of the Nanticoke River marshes, this was sand). Depths to basal peat are measurements from the soil surface to the basement surface at each subsite. Basement surface was determined as the depth at which core samples became composed primarily of sand. 50-cm cores were divided in half and 25-cm sections were described morphologically in the field, transported to the lab in a cooler, and refrigerated. Soil salinity was determined utilizing a refractometer (Extech Instruments, Nashua, New Hampshire) with an accuracy of approximately 0.5 psu on porewater extracted via plastic syringe and filter paper from 1-3 g subsamples (1 subsample for salinity testing/25 cm

section of core). Soil samples were oven-dried to a constant mass at 70° C and weighed for calculation of bulk density (Klute et al. 1986). Subsamples of 1-3 grams were used to determine organic matter content by loss on ignition (LOI) at 400°C for 16 hr (Sparks et al. 1996). Masses of organic matter in the core across the entire stratigraphic profile were calculated by multiplying the proportion of organic matter determined by LOI by the total dry mass of the 25-cm section, and summing the total masses of organic matter in all 25-cm sections in a core.

Average organic matter content was calculated for each core as the average of percent organic matter of each section in a core. Average bulk density was calculated for each core as the average of bulk densities of each 25-cm section. Average porewater or core salinity was determined by averaging porewater salinity measurements determined for each 50-cm section in a core. Average Site porewater salinity, organic matter content, mass of organic matter, and bulk density were calculated by averaging the three subsite averages for each parameter for a Site average. Methods and results for subsidence, surface elevation change and accretion referenced in this chapter are described in detail in Chapter II.

Samples were judged as basal peat when the sediment core became dominated by coarse sand, the layer directly above the sand layer was collected as basal peat samples. Dates of basal peat were determined by radiocarbon (C_{14}) dating, specifically, accelerator mass spectrometry (AMS). Basal samples were dated by Beta Analytic, Inc. (Miami, FL) in June, 2012. Only 10 of the 15 cores were dated because of cost limitations. Two subsite cores were chosen from each Site for dating, and subsite samples were selected based on unusually high or low rates of shallow subsidence. Net accumulation [net

accumulation = organic matter + sediment accretion - (decomposition + compaction)]
rates were calculated by depth of basal peat (mm)/age (years before 1950).

Data were analyzed using Proc Mixed in SAS 9.1 (SAS Institute, Cary, North Carolina). Data were analyzed by analysis of variance (ANOVA) and Tukey's multiple mean comparison tests. Regression analyses were conducted using Sigma Plot (Systat Software Inc., San Jose, California).

Results

Marsh sites along the Nanticoke estuary differed in depth, age, soil series and organic matter content. The formation processes and depositional environments differed over time as Nanticoke River meanders migrated, swamps were converted to tidal marsh, and sediment loads changed. Each site has a unique set of features and there are few trends that apply to all sites. Counter-intuitively, there are few clear patterns of stratigraphic features directly affecting surface elevation change, accretion or subsidence; however, this may indicate the overriding influence of localized processes on surface elevation change.

Depth to Basal Peat

Depths to basal peat vary among Sites ($p=0.001$) (Fig.3.7). Site 1 is the shallowest Site on average having depths ranging from 103 cm to 150 cm. Site 2 is the deepest Site, having depths ranging from 830 cm to 1550 cm. Sites 3, 4, and 5 have intermediate depths (520 cm-756 cm, 740 cm-800 cm, 191 cm-690 cm, respectively). In estuarine meander marshes, depth increases as salinity increases ($R^2=0.71$); however, this is not due to an effect of salinity on depth or vice versa but is because meander marsh Sites at

the mouth of the river (Site 2) which have higher salinities are older, formed earlier, and have accumulated deeper stratigraphic profiles.

Note on SET Benchmark Depths vs Basal Peat Depths

Basal peat depths are all shallower than surface elevation table (SET) benchmarks (in many cases benchmarks were driven 1-3 m deeper than basal peat), indicating that surface elevation measurements encompass entire stratigraphic profiles. Subsite 2C is the only exception to this. Basal peat depth was 1550 cm, and SET benchmark was installed to 1350 cm. This may have been due to the great resistance of the clay at that depth preventing the SET rods from being driven deeper than basal layer.

Subsidence Varies Nonlinearly by Depth to Basal Peat

Rates of subsidence regressed on depth to basal peat follow a polynomial, quadratic pattern ($R^2=0.73$) (Fig. 3.2). Rates of subsidence are highest at Sites of intermediate depth, and are lower at shallow and very deep Sites. There is a moderate quadratic relationship between rates of surface elevation change and depth to basal peat: surface elevation loss is greatest at Sites of intermediate depth ($R^2=0.43$, quadratic regression).

Total Mass of Organic Matter in a Core Varies with Depth and Site

Total organic matter mass differed significantly among sites ($p=0.009$) (Fig. 3.8). The total mass of organic matter in a core increased as depth to basal peat (core thickness) increased (Fig.3.4).

There was a strong curvilinear relationship between total mass of organic matter, averaged by Site, and rates of subsidence, averaged by Site. Rates of subsidence were highest at Sites with an intermediate total mass of organic matter (380 g) (Fig. 3.6).

Additionally, there was a moderate curvilinear relationship among average rates of surface elevation change at each site and total mass of organic matter at each site. Surface elevation loss was greatest at Sites containing intermediate amounts of organic matter ($R^2=0.58$, quadratic regression). There was little relationship between accretion and total masses of organic matter ($R^2=0.34$, linear regression).

Average Organic Matter Content Averaged Over Depth Differs by Site and Salinity

Average organic matter content increased as average core salinity increased ($R^2=0.80$, linear regression), and was lowest at Sites of intermediate depth ($R^2=0.44$, quadratic regression). Average organic matter content differs among Sites ($p=0.027$) (Fig.4.5). Sites 4 and 5, the furthest upriver Sites have low average organic matter and there is an increase in organic matter content proceeding downriver as the estuary widens.

There is no strong relationship between rates of surface elevation change and average organic matter content ($R^2=0.29$, polynomial quadratic regression), subsidence ($R^2=0.39$, quadratic regression), or accretion and average organic matter content ($R^2=0.13$, linear regression). Additionally, there is no linear relationship between average organic matter content (proportion to total volume) and total mass of organic matter ($R^2=0.09$, linear regression).

Subsidence and Surface Elevation Change vary by Core Salinity

Core salinities differed among Sites ($F=,p=0.0001$) (Fig. 3.9) and with depth. Salinities were higher with depth within a core. There is a moderate relationship between average core salinity (salinity measurements at 25-cm intervals averaged by core), and average subsidence by site ($R^2=0.47$, polynomial, quadratic regression). At intermediate salinities, subsidence is the highest. Similarly, there is a moderate quadratic relationship

among rates of surface elevation change and core salinity: rates of surface elevation loss are greatest at sites with intermediate core salinities ($R^2=0.49$, quadratic regression).

Average core salinity does not relate to short-term (2007-2012) accretion rates ($R^2=0.24$, linear regression).

Average Bulk Density

Bulk Density averaged over stratigraphic cores differed among Sites ($p=0.024$) (Fig. 3.10). As to be expected, bulk density was inversely proportional to average organic matter content ($R^2=0.84$, exponential decay regression) (Fig. 3.3). The higher the bulk density, the lower the average organic matter content. Bulk density has little or no relationship to accretion, surface elevation change, or subsidence, and only a moderate relationship to depth to basal peat. Sites of intermediate depth (800 cm) have the highest bulk densities ($R^2=0.51$, quadratic regression).

Age of Basal Peat

Table 3.1: Radiocarbon dates of basal peat samples from bases of stratigraphic cores and net accumulation rates calculated from age and core depths

Subsite	Age (years before 1950)	Net accumulation (mm/yr)
1	B 3130	0.48
1	C 1480	0.70
2	A 3600	2.31
2	B 6480	2.24
3	A 2820	1.84
3	B 4040	1.87
4	A 2540	3.15
4	B 2110	3.51
5	A 1770	2.74
5	B 1370	1.40

Of the estuarine meander marshes, Site 2 is the oldest and Sites are younger progressing upstream (Table 3.1). Depth increases with age in estuarine meander marshes ($R^2=0.82$).

There is a weaker relationship between depth and age including Site 1, a submerged upland marsh ($R^2=0.66$) since Site 1 is of intermediate age, but as previously noted, is shallower than the deeper Sites, 4 and 5.

Net accumulation rates were highest at Site 4 (Table 3.1) and were lowest at Site 1. Net accumulation rates were not good predictors of short-term accretion rates ($R^2=0.33$), and age had little relationship to either net accumulation rates ($R^2=0.0045$) or short-term accretion rates ($R^2=0.02$).

Stratigraphic Core Descriptions by Site

Site 1

Site 1 is a shallow, submerged upland tidal. Depths to basal peat range from 103 cm (1C) to 150 cm (1A,1B). Sediment stratigraphy has a moderate-high proportion of organic matter (25-30%); however, the actual mass of organic matter (44.9 g at 1C to 59.2 g at 1A) is very low due to the Site's shallowness. Accretion rates are low at this Site (~7-10 mm/yr), and much of the accretion is most likely organic due to lack of sediment inputs. Observationally, Site 1 is flooded less frequently and has monoculture vegetation bands of *Spartina patens* (1A, 1B) and *Juncus roemerianus* (1C), bordered by *Spartina cynosuroides*. These species have high C:N ratios, most likely contributing to the high average organic matter contents of the soil profiles. All three profiles have a dramatic decrease in organic matter content at depths of about 60 cm, at which point bulk densities increase. Bulk density peaks at about a depth of 1 m at each of the three replicate subsites. Additionally, salinity peaks at about 110 cm and ranges from 20-30 psu. At 1B and 1A, salinity peaks correspond to peaks in bulk density. Historical imagery of this Site gives no indication that the marsh changed dramatically during the last 70 years.

Site 2

Site 2 is a deep (830 cm 2A, 1450 cm 2B, 1550 cm, 2C) estuarine meander marsh with average soil salinities around 20 psu. Site 2 is also dominated by *Spartina cynosuroides* and *Iva frutescens*. Because of the depth and the moderate-high organic contents of the soils, Site 2 has the greatest total masses of organic matter in each core compared to the other Sites. 2A is the most mineral, and also has a sharp cut off to less than 10% organic matter content at about 350 cm. At 350 cm, bulk density sharply increases. 2A is also the shallowest of the three subsites at nearly half the depth of 2C, and has the greatest subsidence of Site 2 (17.6 mm/yr), though it has moderate amounts of accretion (14.27 mm/yr). Additionally, 2A is burned to stubble by the landowners each October. 2C has peaty layers, high in organic matter to a depth of about 900 cm, at which point organic matter content drops to less than 10%, and bulk density increases. Subsite 2B is variable in respect to organic matter content and bulk density, and is comprised of alternating layers of peat and silt-clay. At Site 2, porewater salinities vary with depth with higher salinities generally in deeper layers. On average, Site 2 is subsiding less than Sites 1, 3 and 4, and is also gaining elevation at a rate of about 3.8 mm/yr. It has moderate rates of accretion (12.87mm/yr, 2C to 18.3 mm/yr, 2B).

Site 3

Site 3 is subsiding at nearly 19 mm/yr, and has a high rate of surface elevation loss relative to Sites 1,2, and 5. It has relatively high organic matter contents (36.7% at 3A to 11.2% at 3C), but has very high within-Site variability. Subsite 3A is of particular interest. It has the highest rate of subsidence of any subsite except 4B (35.4 mm/yr), and has the highest average organic matter content of any subsite. It has low accretion (9.18

mm/yr) relative to other subsites. 3A has low organic matter content at the surface (top 150 cm, 20% OMC), and has more peaty layers at depths of 200-450 cm (65% OMC). This is the opposite pattern seen in many of the other subsites that have high organic matter content at the surface that tapers off proceeding with depth. Salinity spikes in correspondence with organic matter content, possibly indicating the salts are sequestered in organic matter, or that decomposing peat releases salts. This pattern can be seen at other subsites as well; however 3A has the highest salinity peak, approximately 40 psu (peaks at 3B and 3C are 7-10 psu). Site 3 has intermediate depths from 500-800 cm, and bulk densities are very high at the basal depth due to underlying sand in samples.

Site 4

Site 4 has the highest subsidence rate, although there is high within-site variability, and the high average subsidence rate can primarily be attributed to subsite 4B (subsidence is 39.04 mm/yr, despite having moderate accretion, 15.9 mm/yr). Subsites 4A and 4B are close together (~110 m apart) but have very different subsidence rates (4A is subsiding at 19.1 mm/yr) and accretion (4A has high accretion, 18.9 mm/yr). Additionally, Site 4 has the lowest average organic matter content (9.9%), though the site is of intermediate depth (780 cm), and both subsite 4A and 4B have very low organic matter content (9%) but average accretion. 4A and 4B have similar organic matter content and bulk density trends with higher rates of organic matter at the surface (~50 cm) and sharp declines at around 100 cm (organic matter content <10% from 100 cm to basal peat at both sites). Core salinities decrease with depth, having higher salinities in mid-depths and at the surface. Salinities are slightly higher at 4A (4 psu at depth on average), though it is only a 1 psu discrepancy (4B is 5 psu on average). Site 4 has

relatively high bulk densities (0.65 g/cm^3). 4B, the subsite subsiding the most, has lower organic matter content at the surface compared to 4A and 4C. Additionally, belowground and aboveground biomass production are lower at 4B compare to 4A and 4C (Clark 2011).

Site 5

Site 5 is a estuarine meander tidal freshwater marsh (average core salinity 0 psu). Site 5 has the highest average bulk density (0.562 g/cm^3) and low average organic matter content (12.5%). It has an average depth of 455 cm and is the second shallowest (Site 1 is shallower). All subsites have high organic matter at the surface (<100 cm) and low organic matter contents below 100 cm. Subsite 5A has the highest rate of surface elevation gain of any subsite (12.9 mm/yr), has moderate accretion (14.6 mm/yr) and low subsidence (1.76 mm/yr). 5B is subsiding the most of the subsites at Site 5 (17.5 mm/yr), and is also the shallowest (191 cm). 5B has an unusually high spike in salinity to 5 psu at 50 cm depth, and is also subsiding the most of the subsites at Site 5 (17.5 mm/yr).

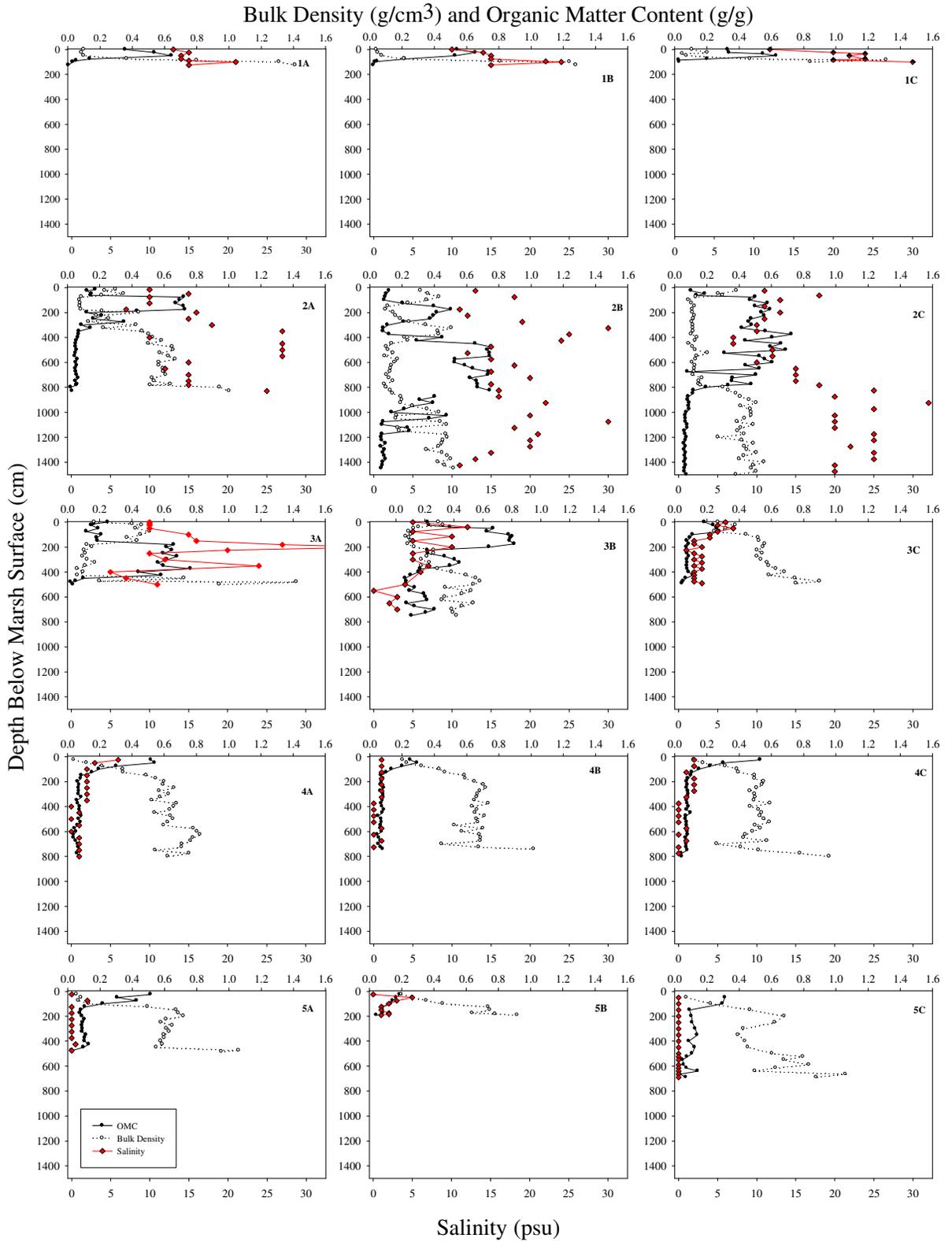


Figure 3.1: Panels 1A-5C: Stratigraphic profiles of 15 subsites, grouped by Site (1-5).

Measurements were taken every 25 cm of bulk density, organic matter content and porewater salinity.

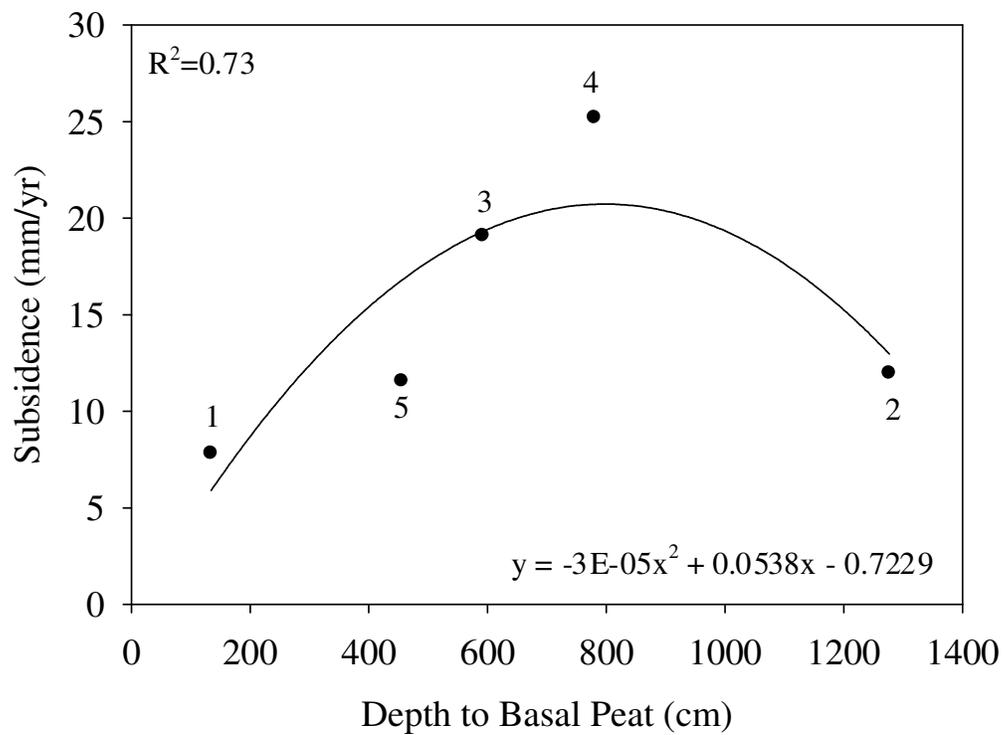


Figure 3.2: Subsidence follows a quadratic trend in relation to depth to basal peat. Sites of average depth have the greatest amount of subsidence.

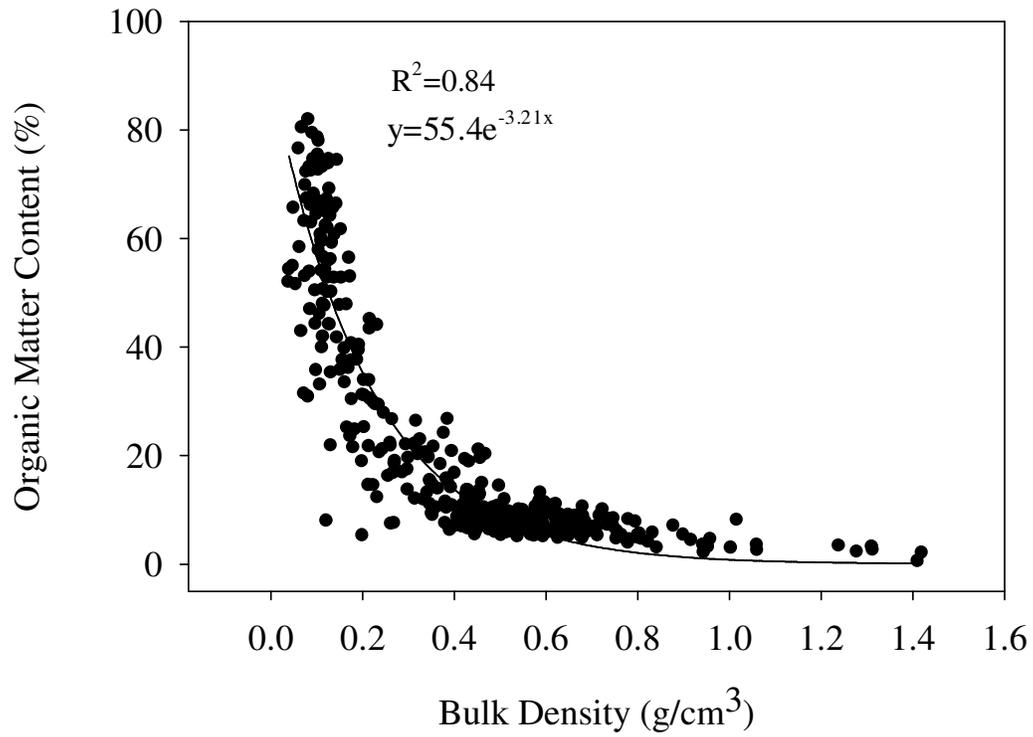


Figure 3.3: Bulk density and organic matter content are inversely related (exponential decay curve). Each point represents values averaged over stratigraphic profiles at each of the 15 subsites.

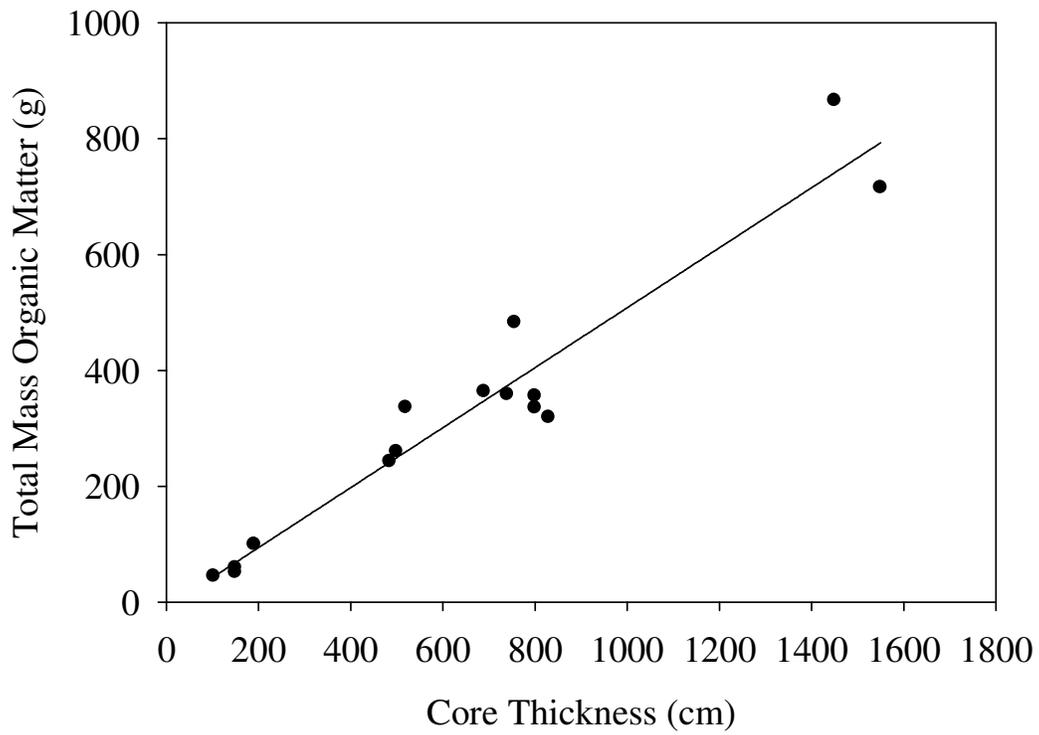


Figure 3.4: Total mass of organic matter increases with core thickness. Points represent totals for each of the 15 subsites.

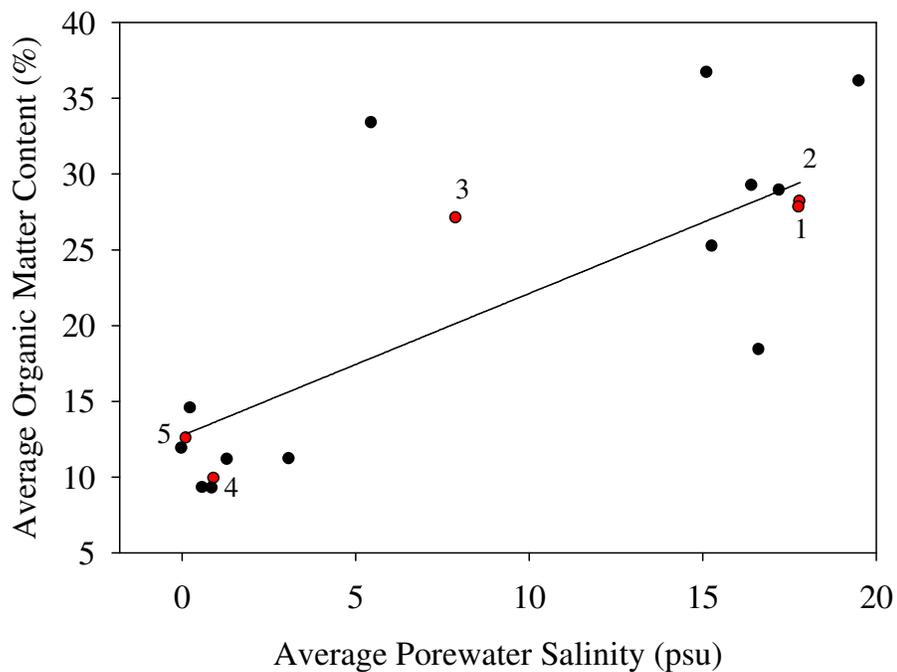


Figure 3.5: Organic matter content (%) averaged over all depths for each core (●) and averaged by Site (●) and its relationship to porewater salinities averaged over all depths of each core and by Site. Site averages are red and numbered by Site ($R^2=0.8$). Individual cores are shown in black ($R^2=0.6$).

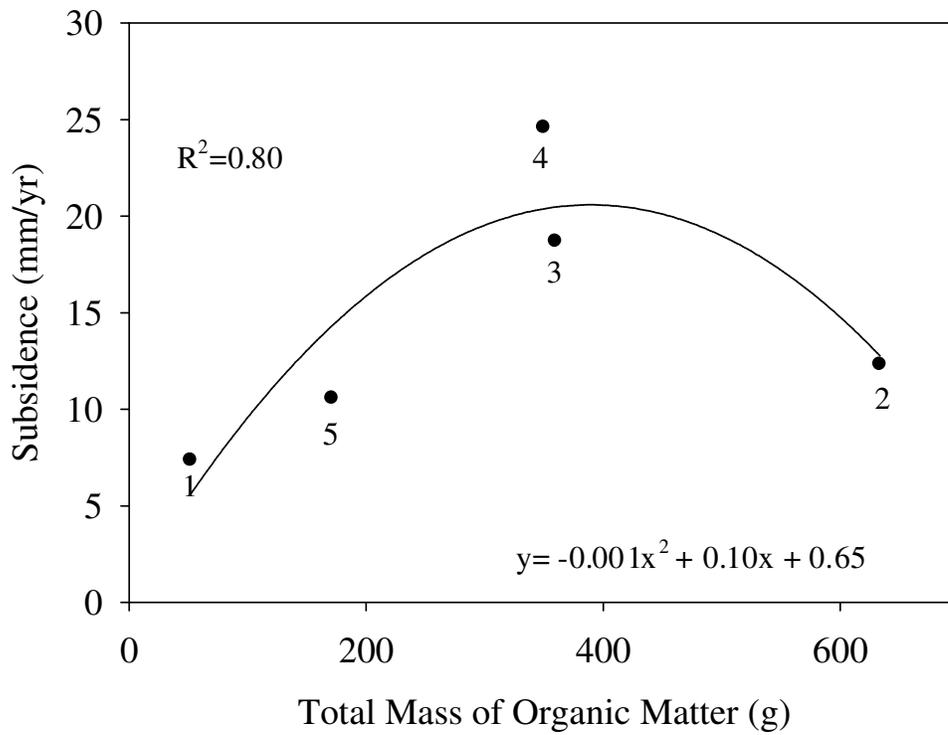


Figure 3.6: Average rates of subsidence by Site and total masses of organic matter (mass of organic matter for each 25 cm depth summed for each subsite and averaged by Site). Regression is quadratic. Sites are labeled by number.

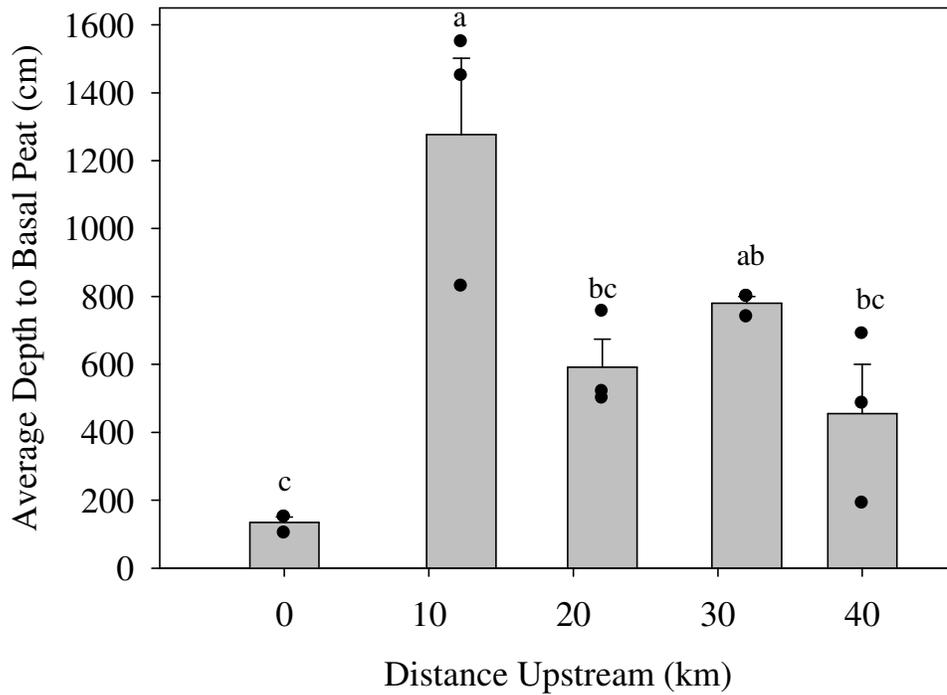


Figure 3.7: Average depth to basal peat by site. Distance upstream is distance from mesohaline Site 1 (0km) proceeding to tidal freshwater Site 5 (40 km). Dots (●) represent subsite depths to basal peat. Vertical bars are Site means (means of three subsites). Letters indicate results of Tukey comparison. Differing letters indicate significant differences among Site means ($p < 0.05$).

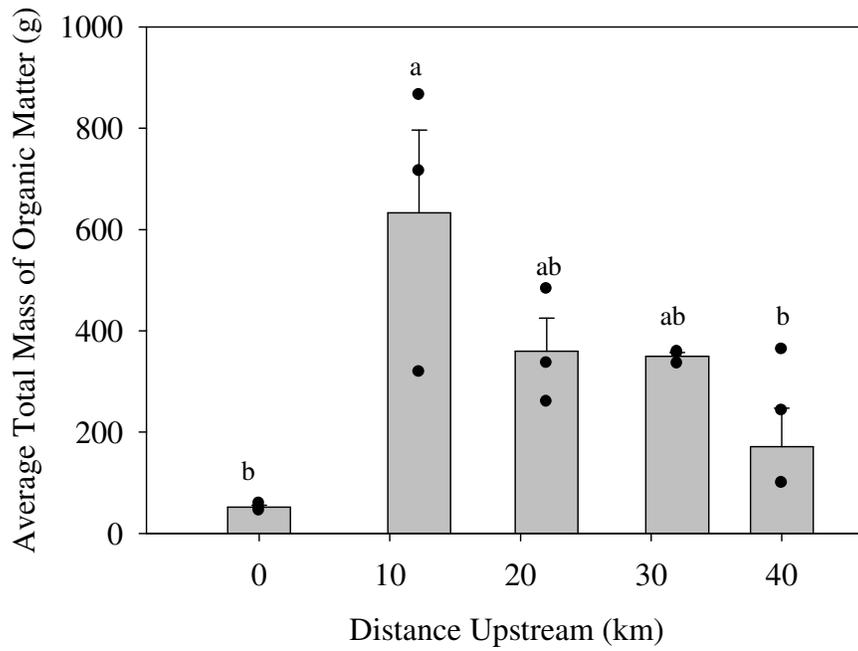


Figure 3.8: Total mass of organic matter, averaged by Site (vertical bars) along an estuarine gradient (distance upstream from Site 1, 0 km). Dots (●) are subsite values of total organic mass of organic matter in a core (profile). Letters indicate results of Tukey's comparison: differing letters indicate significant differences among Site means ($p < 0.05$).

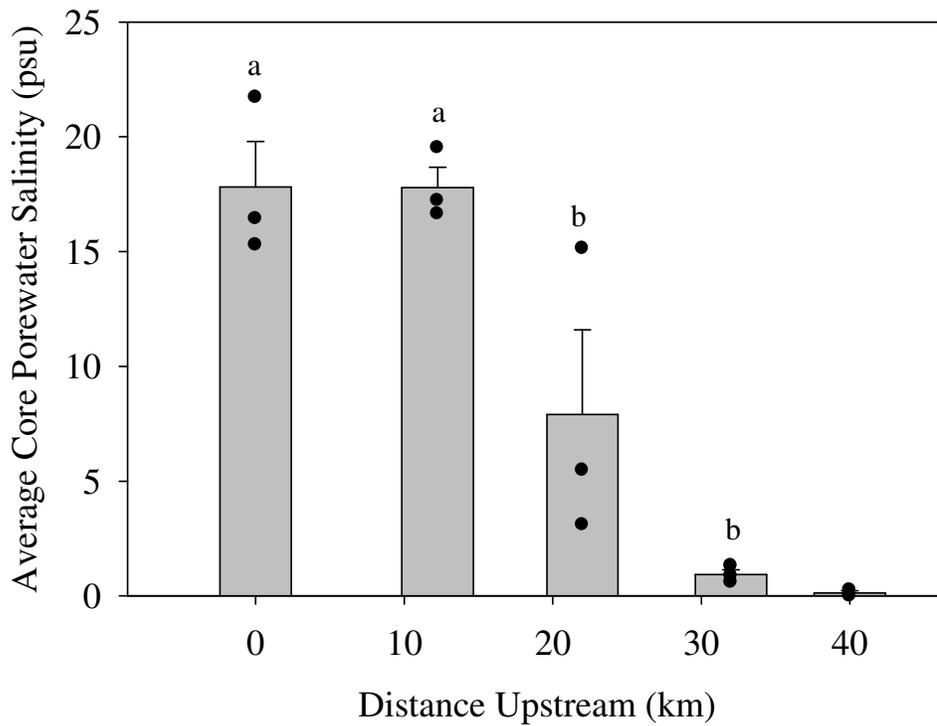


Figure 3.9: Porewater salinity averaged over depth for each core (● represents each core taken at each subsite, vertical bars represent Site averages, the means of the three subsites) and by Site. Distance upstream is the distance proceeding up-estuary from Site 1, 0 km. Letters indicate results of Tukey's comparison. Differing letters indicate significant differences among means ($p < 0.05$).

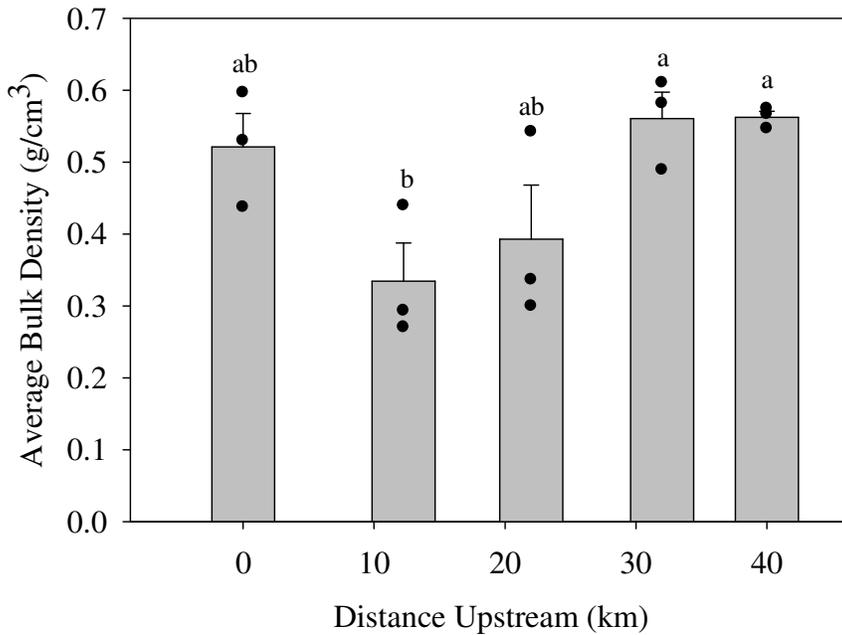


Figure 3.10: Bulk density averaged over depth for each core (● is bulk density for each subsite/core), and averaged by Site (vertical bars are average Site bulk densities, averages of three subsites). Distance upstream from Site 1 (0 km) proceeding to freshwater Site 5. Letters indicate results of Tukey's comparison. Differing letters indicate significant differences among means ($p < 0.05$).

Discussion

Mid-estuarine Sites (3, 4) express intermediate values for many stratigraphic parameters: depth to basal peat, mass of organic matter, bulk density, and salinity; however, they have the highest within-Site variability and include subsites with the highest rates of subsidence (3A and 4B), and the highest total biomass (above + below) production (Site 4) (Clark 2011). Due to these trends, many variables follow a curvilinear (quadratic) pattern, that does not necessarily reflect causation, but rather, it highlights the unusual nature of mid-estuarine, oligohaline meander marshes.

Of the Nanticoke River estuarine marshes, Site 2, the closest estuarine meander marsh to the Chesapeake Bay, most likely formed first given its proximity to the Bay, its

depth, and the large mass of sequestered carbon (largest total masses of organic matter). Older marshes have greater depths of organic matter layers due to being inundated for longer periods of time compared to upstream marshes. Inundation increases carbon sequestration due to slowed decomposition as soils become anaerobic. Additionally, deeper sequences include the early Holocene marshes that were drowned during a period of mid-Holocene rapid sea-level rise (Redfield 1972). As sea-level rose, marshes formed progressively upriver, proceeding from Site 2 to Site 5. These estuarine marshes follow the pattern of depth; Site 2 being deepest, 3 and 4 being about the same depth, and Site 5 being shallower. Site 1 is unique because it is a submerged upland marsh, formed differently and more recently than the other Sites. Site 1 was an upland, with high bulk density, upland soils. During more recent sea-level rise events, the upland was submerged and thus a peat layer began forming over the upland soils. The peat layer is shallow because it has been intertidal for less time compared to older estuarine marshes.

Sedimentation

Differences among Sites may be accounted for in part by sedimentation differences among Sites. Higher bulk densities and lower masses of organic matter and organic matter contents at Sites 4 and 5 are most likely a result of increases in allocthonous inorganic sediment deposition from upland sediment sources. Site 5 marks the beginning of tidal marshes in the Nanticoke estuary. Upriver of Site 5 are few tidal marshes (Sharpe and Baldwin 2009). The river channel itself begins to widen at Site 5, becoming progressively wider proceeding downstream, and tidal marshes and swamps also begin at Site 5 and become progressively larger proceeding downstream. Upland sediments carried by river discharge are most likely deposited in the marshes and swamps

of Sites 5 and 4 as the river widens and velocity slows. The Nanticoke River has significant riverine sediment sources and sedimentation increases from the mouth of the river to the upstream tidal freshwater wetlands (Kearney and Ward 1986). Additionally, the estuarine turbidity maximum (ETM), which resuspends sediments and deposits them likely exists near the oligohaline-fresh boundary at Site 4, which would explain why Site 4 has the lowest average organic matter content, high rates of accretion and the highest bulk density. Site 5 also has high bulk density and low amounts of organic matter compared to the other Sites. Average organic matter content increased as salinity increased. Higher salinity Sites have less inorganic sediment input given that are farther from riverine sediment sources (upstream of mesohaline Sites), and so are more highly organic. Because of this, the marshes at the mouth of the river (Site 1) may rely more on autochthonous organic matter production to accrete rather than inorganic sediment inputs. Another potentially important difference between upriver Sites (4 and 5) and downriver Sites (3,2 and 1) is proximity to the main river channel. Sites 4 and 5 are adjacent to the main Nanticoke River channel. Sites 1, 2 and 3 are adjacent to large tidal creeks. Much of the inorganic sediment may have fallen out at the mouths of the tidal creeks where large sandbars often form, leaving less sediment to be deposited on marsh platforms. In contrast, Sites 4 and 5 are presumably regularly flooded by the main river channel so sediment is most likely falling out on the natural levies bordering these Sites, with some reaching the interior marsh.

Another variable to be considered when evaluating organic matter content is peat quality. Vegetation rich in lignin with high C:N ratios contributes less decomposable peat compared to vegetation with low C:N ratios and an abundance of soluble carbon (Odum

1988). Sites 1 and 2 are dominated by high C:N vegetation (e.g., *Spartina patens*, *Spartina cynosuroides* and *Juncus romerianus*) that may be contributing less degradable peat compared to the vegetation at Sites 4 and 5 (e.g., *Peltandra virginica*, *Acorus calamus*). Additionally, rates of vegetation litter export may differ among Sites, with the majority of autochthonous, organic matter at Site 4, a frequently flooded Site, being exported to Sites downstream such as 3 and 4.

Core Stratigraphy

Organic Matter

In the majority of the cores, organic matter was higher at the surface (this occurred more frequently at Sites 4 and 5), sharply dropped off at some depth and remained low in the deeper layers. Organic matter in deeper layers has been most likely decomposed over the long periods of time since deposition. Additionally, layers represent different depositional environments. As marshes developed from subtidal mudflats, fine sediments (silts and clays) were deposited. As marshes emerged and became intertidal, vegetation colonized, producing autochthonous organic matter and peat layers developed. Sequences alternate as tidal channels migrated, incising into marsh platforms and allowing coarser-textured sands to fall out and more layers of silts and clays. Marsh sequences with thick peat layers near the surface and mid-profile represent young, emergent marshes. Deeper profiles consisting of high bulk density silts and clays represent the compacted sediments of subtidal mudflats at the beginning stages of development (Ward et al. 1998).

Salinity

Unexpectedly, porewater salinities were higher in organic-rich layers compared to more silt-clay rich layers. This could be because either organic matter sequesters or binds salts, or perhaps as organic matter is decomposed it releases salts. This is counterintuitive given that salinity accumulation is determined in part by hydraulic conductivity. Peaty layers would have higher hydraulic conductivity (Hussein and Rabenhorst 2001) and thus the potential to be flushed more frequently. Another hypothesis regarding increasing salinity is that salinity at depth reflects salinity of surface water or vegetation tissue at time of sediment layer deposition. Given that coastal wetland plants sequester salinity in their tissues as a stress tolerance mechanism (Mitsch and Gosselink 1993), peat deposited may have contained higher salinities. Additionally, it is possible that surface water salinity was higher when early marsh sediments were deposited. Given that peat layers formed during periods of sea-level decrease (tidal mud flats were exposed, allowing colonization by emergent vegetation)(Allen 2000a), marsh platforms may have been flushed less frequently by tides, accumulating salts as evapotranspiration by marsh vegetation removed freshwater and concentrated soil salinities. These hypotheses are unlikely, however, and more likely is salinity migration within sediments and accumulation at depth and in organic-rich stratigraphic layers.

Subsidence and Stratigraphic Features

This study was undertaken to explore the hypothesis that Sites with greater depths and quantities of organic matter would have higher rates of subsidence and surface elevation loss. Rationale for these hypotheses was that as sea-level rises, sulfates become more prominent in marsh substrates. Increases in sulfates lead to increases in sulfate

reduction, a decomposition pathway that breaks down organic matter more rapidly compared to methanogenesis, thus leading to organic matter degradation and collapse at Sites with deep, peaty layers. Additionally, increases in salinity lead to increased plant stress and mortality (McKee and Mendelssohn 1989), thus decreasing biomass production and increasing soil nitrogen (Mendelssohn and McKee 1988). As vegetation dies, nitrogen is released from the tissues of the dead vegetation. This nitrogen increases soil decomposition and causes a decrease in belowground biomass production (Valiela, Teal and Persson 1976). Both factors can cause a loss in surface elevation. Subsidence may also be greater at sites with greater masses of organic matter because organic matter mass can be a predictor of depth (the greater the depth, the greater the mass of accumulated organic matter), and coastal wetlands with greater depths have been shown in previous studies to have higher subsidence rates (Penland and Ramsey 1990). These hypotheses were not supported; however, they cannot be easily rejected either given the stratigraphic characteristics of the Sites. Contrasting the two subsites with the greatest surface elevation losses and subsidence; 3A and 4B have nearly opposite trends in organic matter accumulation. 3A has the highest average organic matter content (36.7%) of any of the 15 subsites; in contrast, 4B has the lowest organic matter content (9.2%) of any of the 15 subsites. They are both of intermediate depth and have intermediate total masses of organic matter (most likely a function of age, as discussed above).

High variability in subsidence rates and stratigraphic features reflects that subsidence is caused by a large number of interacting features and causes or rates can only be applied to small, localized areas, and not to whole systems (Meckel et al. 2006).

Autocompaction

Another contributor to subsidence rates is autocompaction. In Holocene sediments, autocompaction can cause at least a few mm/yr of subsidence (Pizzuto and Schwendt 1997), and begins within 10 cm of the marsh surface (Massey et al. 2006). Newly deposited sediments increase in bulk density over time due to autocompaction. Autocompaction occurs when the weight of sediments causes a collapse in deeper layers. Autocompaction can be a large component of subsidence given that consolidation of sediments decreases pore space and causes a decrease in surface elevation (Allen 1999). Autocompaction affects sediments of varying composition differently. Sands and siliceous gravels are nearly incompressible, and therefore hardly subject to autocompaction. Peats, having large pore spaces, are very compactable and silts and clays are moderately compactable (Allen 1999). Holocene deposits high in organic matter retain only 10-20% of their original thickness (thickness at time of deposition) over time (Pizzuto and Schwendt 1997).

There is little relationship between short-term and net accumulation rates because of autocompaction. When evaluating accretion by depth using only basal peat dates, deeper layers will indicate lower accretion rates because the same volume of sediment will reflect higher rates of accretion at the surface than at depth due to autocompaction (Bartholdy, Pedersen and Bartholdy 2010).

Autocompaction does not occur at constant, linear rates, and so it is difficult to predict the magnitude of future surface elevation loss due to autocompaction. In marshes with high organic matter and unconsolidated sediments, surface accretion rates would need to compensate for higher rates of autocompaction. Identifying the stratigraphic

features of a marsh profile may help determine how much autocompaction is to be expected and at what rate.

Submerged upland marshes may be less vulnerable to sea-level rise due to their shallow peat layers underlain by high bulk density upland soils. High bulk density, low-organic matter soils are less subject to surface processes such as compaction, decomposition and changes in hydrology such as the shrinking-swelling of peat due to groundwater withdrawal and dewatering (Kearney and Stevenson 1991, Cahoon et al. 2011). Surface processes have a greater impact on surface elevation change and subsidence compared to deep processes; thus, submerged upland marshes may be more able to maintain positive surface elevation relative to sea-level rise than estuarine meander marshes.

Stratigraphy can provide a geomorphic history of coastal wetlands. Understanding the processes that formed a wetland can guide future restoration efforts. Inputs of sediments similar in nature to formative deposits may have greater positive impacts on subsiding wetlands than sediments from other depositional environments (Flocks et al. 2009).

In this study, I sought to determine relationships between the underlying substrate and shallow subsidence. Few clear relationships were determined among shallow subsidence, elevation, accretion and stratigraphic features such as depth, organic matter content, bulk density, and salinity. There was high variability of stratigraphic features within Sites, and no patterns relating those features to elevation loss (e.g., 3A and 4B had opposite trends in bulk density and organic matter content but are the subsites with subsiding the most). Sites of intermediate characteristics (intermediate quantities of

organic matter, intermediate depths, intermediate salinities, etc.) have the greatest rates of shallow subsidence, highlighting the unique nature of mid-estuarine marshes. Although it is unclear whether many of the curvilinear relationships between subsidence and stratigraphic features have ecological significance, position in the estuary (i.e., mid-estuarine sites) may determine susceptibility to sea-level rise.

Chapter IV: Effects of Salinity Intrusion on Tidal Freshwater Wetland Surface Elevation and Vegetation Community Dynamics: An *in situ* Experiment

Abstract

Tidal freshwater wetlands are herbaceous-dominated, biodiverse ecotones in the upper reaches of estuaries where salinities are less than 0.5 psu but tidal influences persist. Sea-level rise threatens coastal wetlands and may have greater negative effects on tidal freshwater wetlands compared to salt or brackish marshes. Vegetation communities adapted to freshwater conditions may be stressed by increased salinity and decrease in productivity, as well as lose diversity through the increased dominance of salt-tolerant species. Saltwater intrusion and the influx of sulfates may cause a shift from methanogenesis to sulfate reduction, increasing decomposition and altering soil surface elevation dynamics. To simulate sea-level rise in a tidal freshwater wetland, an *in situ* salinity addition experiment was conducted in a tributary of Chesapeake Bay. Vegetation communities and surface elevation and accretion were measured for a period of 4 years during treatment application. Rates of elevation change did not significantly differ between treatments, and were 7.8 ± 4.9 mm/yr in fresh-treated plots and 6.9 ± 2.3 mm/yr in salt-treated plots. Accretion rates also did not significantly differ between treatments (19.3 ± 1.7 mm/yr to 20.3 ± 1.9 mm/yr), nor did species richness and total cover during the experimental period despite persistent oligohaline conditions in the soils of the salt-treated plots. This suggests that tidal freshwater wetlands are more resistant to small increases in salinity and flooding than previously thought.

Introduction

Tidal freshwater wetlands are biodiverse, productive and unique ecosystems (Odum 1988) situated in the upper reaches of estuaries where tidal signals persist but salinities are less than 0.5 psu (Cowardin et al.1979). They are threatened by the effects of anthropogenic activities such as sea-level rise, which results in increased flooding, saltwater and sulfate intrusion (Orson, Panageotou and Leatherman 1985). Tidal freshwater wetlands may be more vulnerable to saltwater and sulfate intrusion in comparison with coastal marshes of differing salinity regimes (i.e., oligohaline, mesohaline and salt marshes) (Doyle et al. 2007, Craft et al. 2009) due to the sensitivity of plant communities to salinity increases, increased rates of decomposition, and the hardening of coastlines and increased upland slope that prevents migration (Desantis et al. 2007). The ability of tidal freshwater wetlands to persist in conditions of accelerated sea-level rise is dependent on maintaining surface elevation that is equal to or greater than the rate of relative sea-level rise (Patrick and Delaune 1990).

The objectives of this study were to experimentally examine the effects of saltwater intrusion on surface elevation dynamics and vegetation communities in a tidal freshwater wetland. Surface elevation change is a primary factor in determining whether coastal wetlands will persist in conditions of rising sea-levels. Many variables such as plant community dynamics, sedimentation, hydrology, nutrient cycling, and herbivore interactions directly affect surface elevation, making it a good predictor of overall wetland health and persistence. If any one factor negatively contributes to surface elevation loss, wetlands will be subject to the negative effects of increased flooding under

strictly fresh regimes, and possibly saltwater and sulfate intrusion under conditions of drought, sea-level rise, or salinity pulses from storm events.

Saltwater intrusion may increase rates of decomposition in tidal freshwater soils by causing a shift from methanogenesis to sulfate reduction, resulting in greater rates of organic matter mineralization (Weston et al. 2006) and decreases in surface elevation. Methanogenesis accounts for the majority of anaerobic respiration in tidal freshwater wetland soils, although other metabolic pathways such as Fe (III) reduction may be dominant in portions of the marsh including the root zone (rhizosphere) at some portions of the year (Neubauer et al. 2005, Weiss et al. 2003). Sulfate reducing bacteria outcompete methanogens for organic matter substrate, preferentially utilizing sulfate as a terminal electron acceptor to break down carbon compounds (Capone and Kiene 1988); however, sulfate reduction accounts for very little anaerobic metabolism in tidal freshwater wetlands due to sulfate limitation. Saltwater intrusion causes an influx of sulfates into tidal freshwater marsh soils potentially leading to a shift in dominant respiration pathways from methanogenesis to sulfate reduction and increasing mineralization rates or organic carbon (Weston et al. 2006). Increases in decomposition may lead to decreases in surface elevation as highly organic soils are broken down more rapidly.

In addition to increasing decomposition, sea-level rise may impair the ability of a wetland to accrete autochthonous organic matter by altering plant community dynamics and physiological processes. Sea-level rise may cause a shift in plant communities to more salt-tolerant species (Warren and Niering 1993), thus converting tidal freshwater wetlands to oligohaline wetlands (Boesch et al. 1994). However, under rapid sea-level

rise, communities cannot adapt, causing rapid dieback and the inability of plant communities to rebound through regeneration and colonization (McKee and Mendelssohn 1989).

Seed banks of coastal marshes, seeds stored in marsh soils that often have greater species richness than emergent plant communities, provide one mechanism of community shift by containing seeds of species with a range of salinity tolerances (Parker and Leck 1985, Baldwin, McKee and Mendelssohn 1996), allowing new salt-tolerant species to emerge after salt-sensitive vegetation has died back. Salt-tolerant species are often present in the seedbanks of tidal freshwater wetlands and may have better regeneration and germination rates compared to tidal freshwater obligate species under conditions of elevated salinity (Middleton 2009). Seed viability is robust under conditions of temporary increases in salinity such as salt pulses. Seedling recruitment, germination and regeneration are negatively affected by sustained increases in flooding and salinity as low as 2-4 psu (Baldwin et al. 1996).

Tidal freshwater wetland plant communities may be more sensitive to saltwater intrusion compared to oligohaline and salt marshes. Wetzel et al. (2004) conducted a reciprocal transplant study and found that freshwater species when transplanted into oligohaline conditions were stressed and experienced high mortality rates (Reed 1995, Wetzel et al. 2004), in contrast, oligohaline and salt marsh species can thrive in freshwater wetlands in the absence of competition. Additionally, oligohaline seed banks have species that have greater ranges of salt tolerance and are therefore more robust in regards to changes in salinity (Baldwin et al. 1996).

Surface elevation is directly affected by plant community dynamics. Belowground biomass production is a large contributor to accretion in coastal wetlands, and thus to the ability of a wetland to maintain positive elevation relative to sea level (Pendelton and Stevenson 1983). Rates of primary production, including belowground biomass production, decrease as a result of vegetation stress from increased salinity and flooding. Additionally, sedimentation is controlled, in part, by vegetation. High stem density and adventitious roots can trap sediment particles, increasing mineral accretion. As vegetation declines due to salt stress and sulfide toxicity, rates of sedimentation also decline, causing accretion deficits, and subsequent losses in surface elevation.

I hypothesized that tidal freshwater wetlands experimentally exposed to saltwater would have higher rates of elevation loss than freshwater locations, and that simulated saltwater intrusion would reduce vegetation species diversity and total cover.

Materials and Methods

Site Description

The study area was located on the Patuxent River, a tributary of Chesapeake Bay in southeastern Maryland, USA. Sites were set up in Jug Bay, a 2,087-acre section of tidal freshwater marsh (i.e., a wetland dominated by herbaceous plants), open water and forested swamp in the middle reaches of the river, at the confluence of the Western Branch and Patuxent (Fig.4.1.a). Average salinity in Jug Bay is <0.5 psu. Jug Bay is a semi-diurnal, microtidal system.



Figure 4.1: Jug Bay study sites at the confluence of Western Branch and the Patuxent River. Patuxent River is a tributary to Chesapeake Bay (a.). Five sites were established in high marsh areas (b.). Map Sources: Google Earth (2012), United States Department of Agriculture (2009).

Experimental Design

Five plots were randomly set up in the tidal freshwater marsh between Western Branch and the primary branch of the Patuxent River (Fig.4.1.b). Of the five plots, three plots were closer together in the interior marsh, adjacent to a tidal creek branching off of the main river channel, and two plots were closer together adjacent to Western Branch. Each plot was divided into paired treatment plots (salt and fresh)(Fig.4.2).

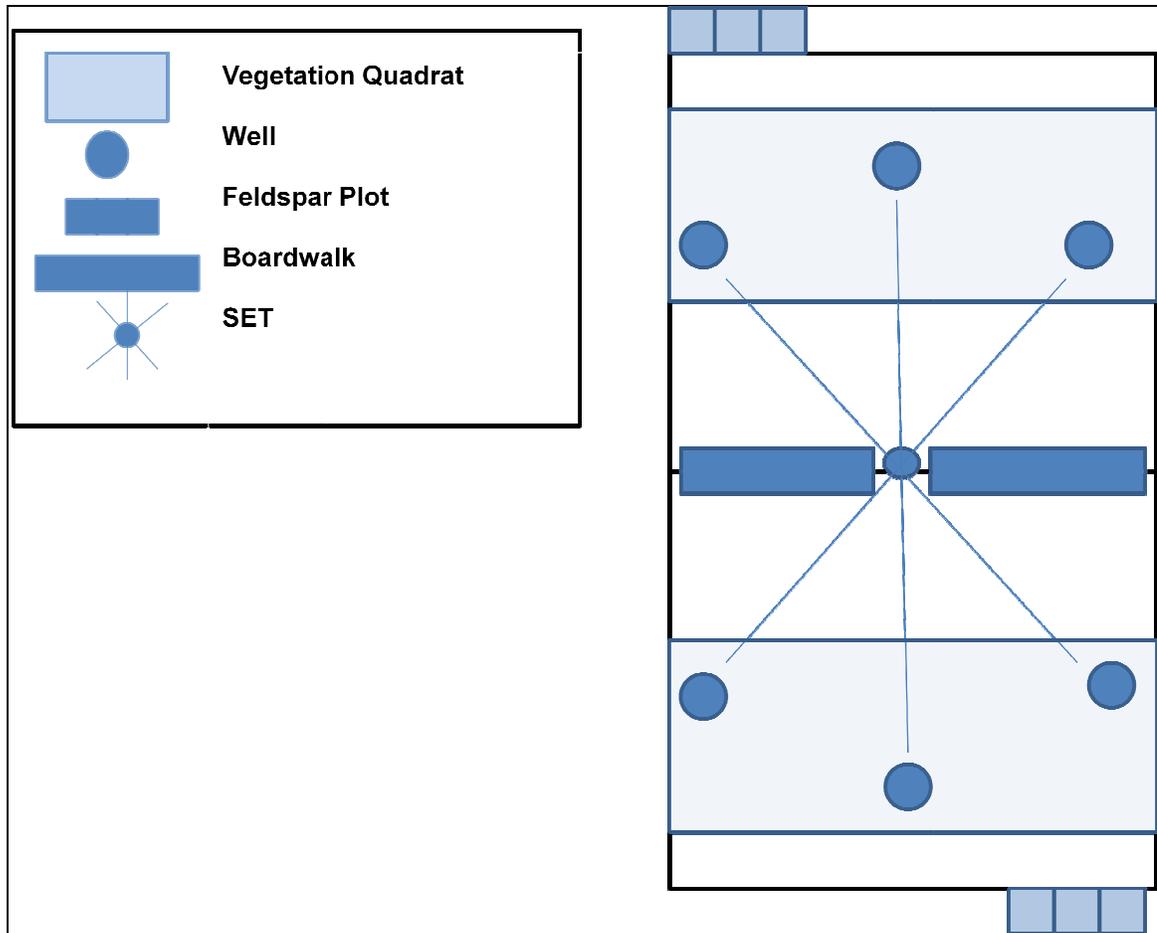


Figure 4.2: Experimental paired plot design: Five SETs were installed and established the centers of 10 plots. For each SET, there was a salt plot and a fresh plot. Each plot had 3 positions of the SET, a vegetation quadrat, and three feldspar plots adjacent to it.

Variables

Factors being studied in this experiment include vegetation community composition, surface elevation and accretion, and porewater salinities.

Plots were developed around deep-rod surface elevation tables (SET). Five SETs were installed, and plots were determined by positions of the SET arm. A SET is comprised of a stable, permanent benchmark, driven to the underlying boundary layer beneath the marsh, and a portable arm that rotates around the benchmark at up to eight possible

positions (each 45° apart). For this study, a modified (longer) arm was used. The SET arm was 1.45 m long to create larger experimental plots compared to the standard SET arm (53.5 cm). At the end of the arm are nine pins that are dropped to the surface of the marsh. Elevation change over time is determined by differences in pin measurements. In this study, six of the eight positions were measured, excluding the two positions parallel to the boardwalk. Plots were developed around 3 SET-arm positions and were randomly assigned fresh or salt treatments. Marker horizon plots are 30 cm x 30 cm and arranged next to each other adjoining the SET measurement area. Three marker horizon plots were laid for each treatment (5 plots, 2 treatments/plot, 3 marker horizon plots/treatment, n=30)(Fig.4.2).

Vegetation plots are 1x2.5 m² quadrats centered over the last 30 cm of the SET arm at three positions (the quadrat is centered over where pin measurements are taken at three positions).

Treatments

Treatments began in June 2008 and continued until December 2011. Treatments were applied every two weeks during the growing season (April-November). In 2008 and 2009, treatments consisted of 375 liters (100 gallons) of river water (fresh) or 375 liters of river water mixed with sea salt (Instant Ocean) sufficient to raise the salinity of the treatment to 10-25 psu (salt). Saltwater dispersion technique was discontinued due to multiple problems with clogging of irrigation tubing that dispersed saltwater. Treatments in 2010 and 2011 consisted of sea salt diffusing from 100 cm PVC wells (7.6 cm diameter) installed to a depth of 50-cm (50 cm above surface, 50 cm below soil surface

with holes along 50 cm belowground length and capped at bottom). Wells were filled with salt every two weeks. Control plots (fresh) also had wells installed, but no further treatment was applied.

Elevation

Pin differences over time were averaged by position, and then by SET to obtain replicate values for analysis. Pin differences from baseline were regressed on time (days since beginning) and slopes constituted daily elevation change. Analyses were conducted on average slopes, or rates of elevation change. Means of salt and fresh treatments were analyzed utilizing a paired t-test [Proc ttest, Statistical Analysis Software (SAS 9.1), SAS Institute, Cary, NC]. Rates of elevation change were also compared between Western Branch sites and tidal creek sites to evaluate potential differences due to location, pin 1 and pin 9, comparing pins closest and furthest from treatment wells, and during different time intervals, evaluating effects of alternate treatment methods, by paired t-tests.

Vegetation

Vegetation plots were consued for percent cover by species using cover classes (Peet, Wentworth and White 1998). Species richness, or the number of species per plot, was determined from cover data. Vegetation was monitored in June 2008 (Day 0), June 2010 (Day 730), June 2011 (Day 1095), and August 2011. Measurements in June 2008 were pre-treatment baseline measurements.

Results

Salinity

Porewater salinities in the salt-treated plots were maintained at oligohaline conditions (0.5-5 psu; Cowardin et al. 1979) during the study period. Porewater salinities ranged with depth from surface and distance from well five days after wells were filled, with highest salinities at the wells (22-27 psu) and slightly increasing with depth (Fig. 4.4). During the course of the study, porewater measurements on average two weeks after treatment were 0.25 ± 0.01 psu (n=95) in the fresh-treated plots and 1.35 ± 0.18 ppt (n=100) in the salt-treated plots. Porewater salinities in the fresh plots did not differ significantly from ambient salinities (0.2 psu) in surrounding areas of the marsh and river. Porewater measurements two weeks after treatment differed significantly ($p=.00000001$) between fresh-treated plots and salt-treated plots (Fig. 4.3).

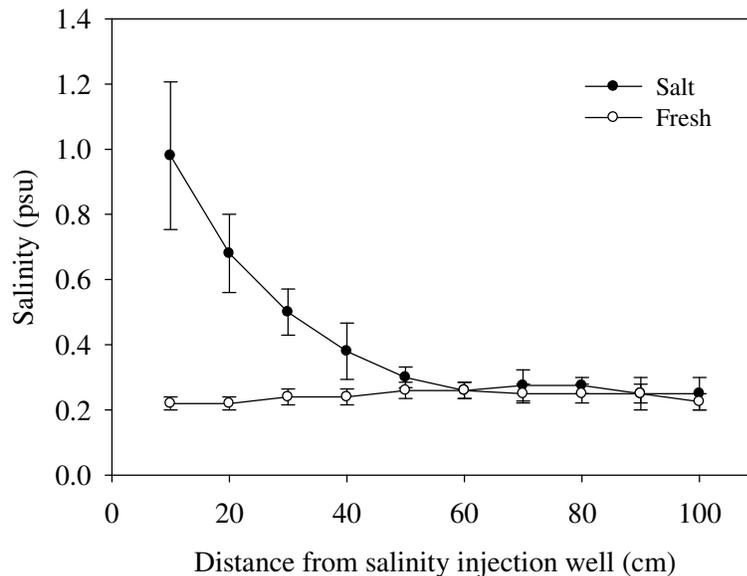


Figure 4.3: Salinity measurements taken at varying distances and depths from well two weeks post treatment. Each point is a mean of 10 wells on one sampling day.

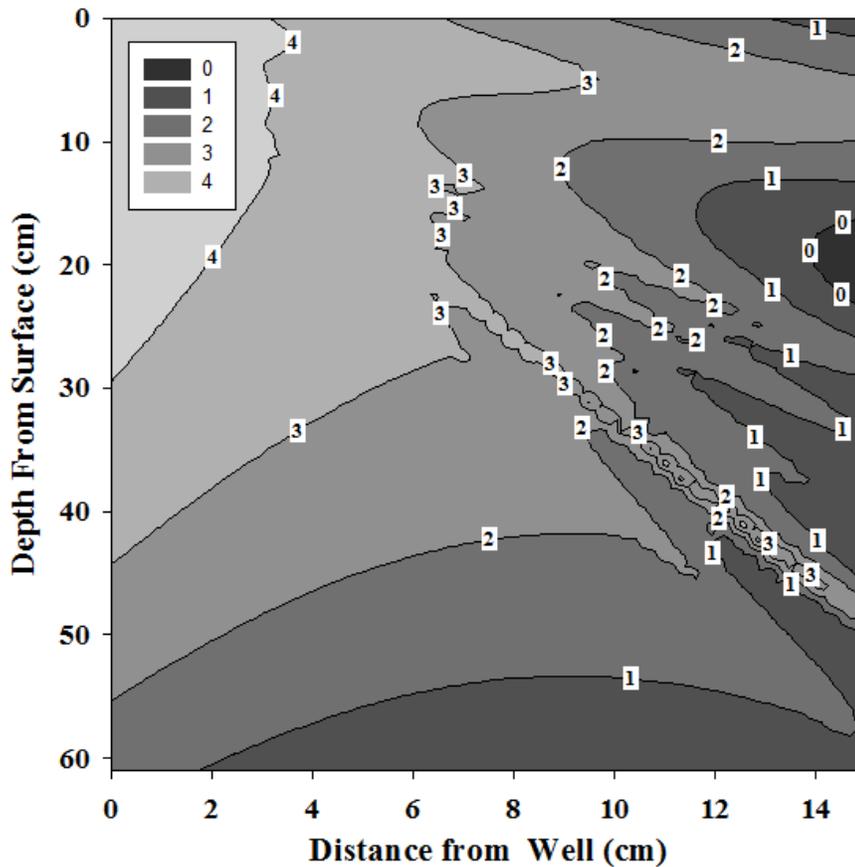


Figure 4.4: Porewater salinity (psu) five days post treatment, measured in and near 10 wells five days after treatment application (wells were filled). Salinities were measured at different depths and distances from wells.

Vegetation

Species richness (treatment averages for all sampling events) did not differ between salt and fresh-treated plots overall ($p=0.24$), or during any one sampling event (Day 0, $p=0.34$; Day 730, $p=0.74$; Day 1095, $p=0.77$; Day 1155, $p=0.24$) (Fig. 4.5).

Despite species richness not differing between treatments, it did differ temporally.

Species richness dropped significantly between June 2008 (Day 0) and June 2010 (Day 730) (Fig. 4.5) at all sites and did, however, differ significantly between Day 0 and Day

730 in both the salt plots (n=5, p=.04) and the fresh plots (n=5, p=.01). Between Day 730 and Day 1095 (June 2011), there was a rebound in species richness in eight of the 10 plots; however, in some cases the rebound constitutes an increase of 1-2 species. Site 3 saw a further decrease in both the salt and fresh plots between June 2010 and June 2011; however, between June 2011 and August 2011, there was a doubling of species richness in those same plots. In eight of the 10 plots, species richness declined between June 2011 and August 2011 (a typical seasonal pattern).

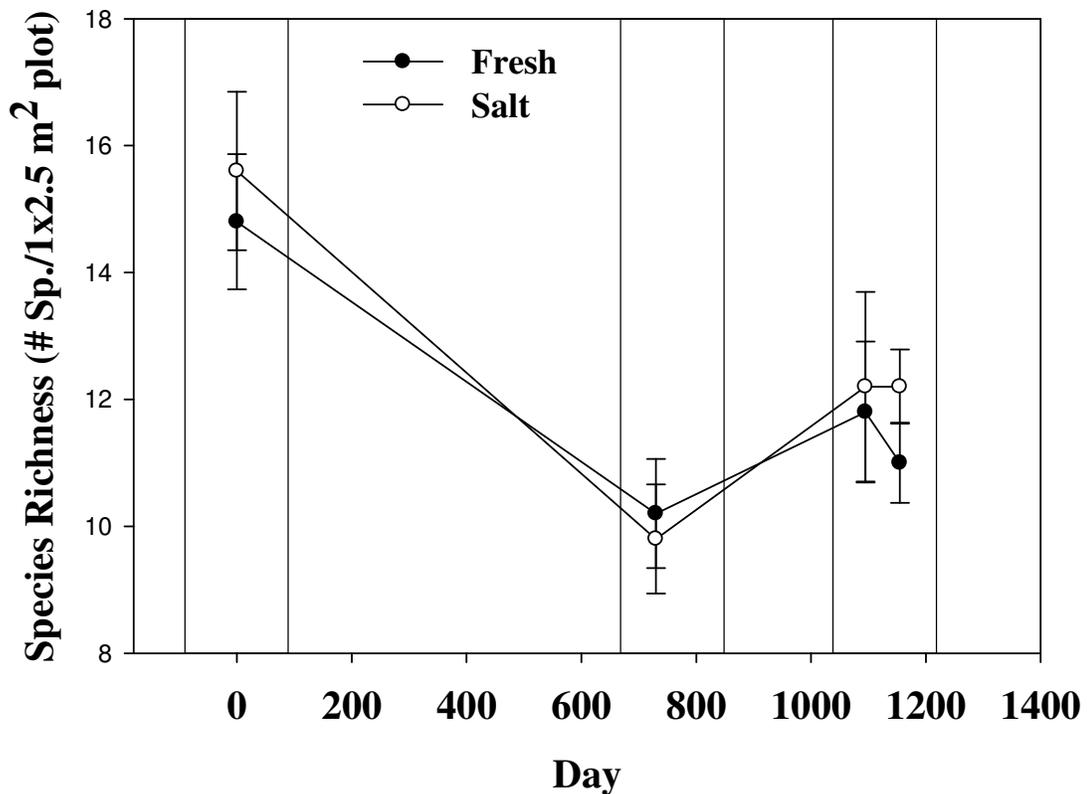


Figure 4.5: Species richness (n=5 plots for each sampling event) in salt and fresh treated plots. Plotted values are mean \pm SE. Boxes are growing season (April-October).

In addition to decreasing in species richness, total cover decreased from June 2008 (Day 0) to June 2010 in eight of the ten treatment plots but, as with species richness, site 3 had an increase in total cover during that time period. Average (n=5 for each treatment) total cover did not significantly differ for either treatment between Day 0 and Day 730 (p=0.1, salt plots; p=0.06, fresh plots). Additionally, total cover did not differ significantly overall between salt and fresh treatments (p=0.9) (Fig. 4.6).

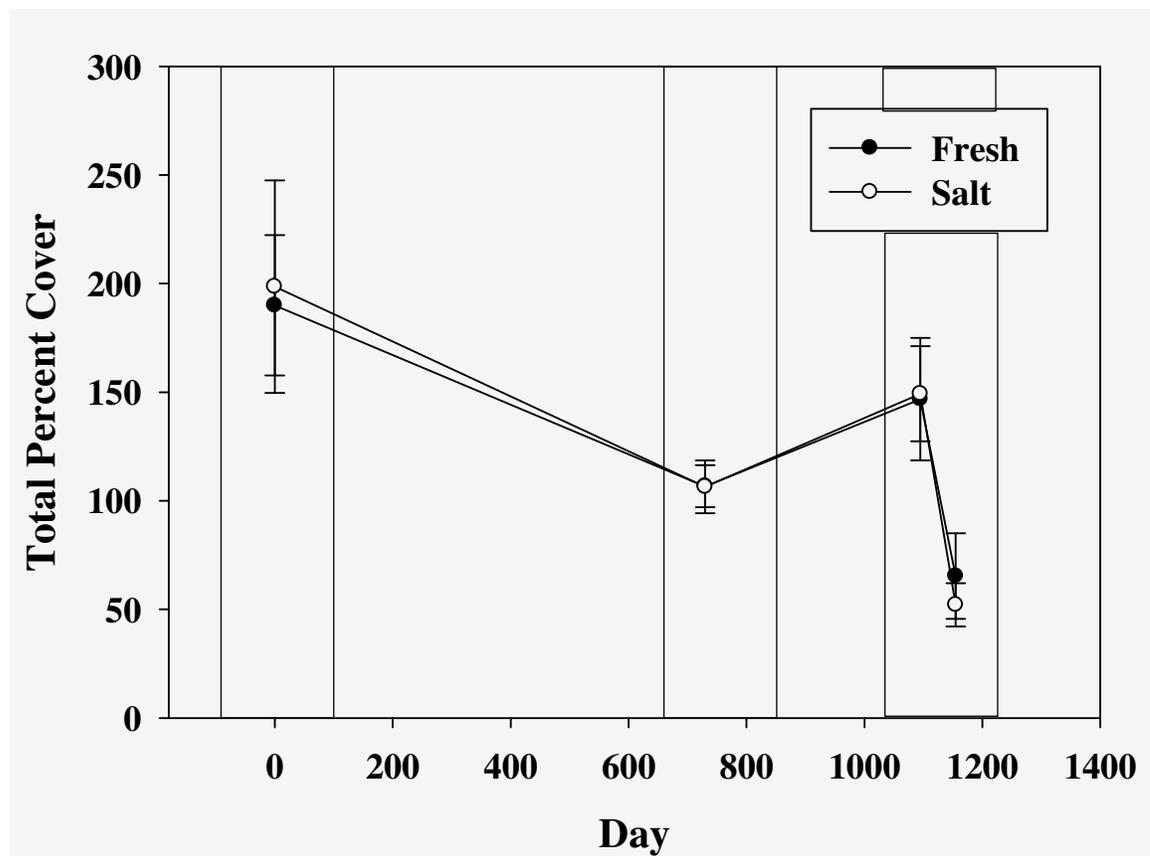


Figure 4.6: Average total percent cover (n=5 plots for each treatment during each sampling event). Day 0 was sampled June 2008. Percent cover was estimated for a 1 x 2.5 m² plot. Boxes are growing season (April-October).

When ranked by cover, the dominant vegetation species did not significantly differ between treatments (Fig. 4.7). *Impatiens capensis*, an annual species, was the

dominant species in both fresh and salt-treated plots followed by *Polygonum arifolium*, also an annual species. Despite having a greater number of perennial species compared to annual species, the majority of the canopy is made up of annuals. In the fresh plots, there were 6 annuals and 12 perennials, and in the salt plots, there were 7 annuals and 14 perennials. The number of annuals and perennial species did not significantly differ by treatment. Most species were present in both salt and fresh plots, and when ranked by abundance, did not deviate from their relative ranking between salt and fresh plots. One exception is *Sparganium eurocarpum* which decreased slightly in cover between the fresh and salt plots (Fig.4.7).

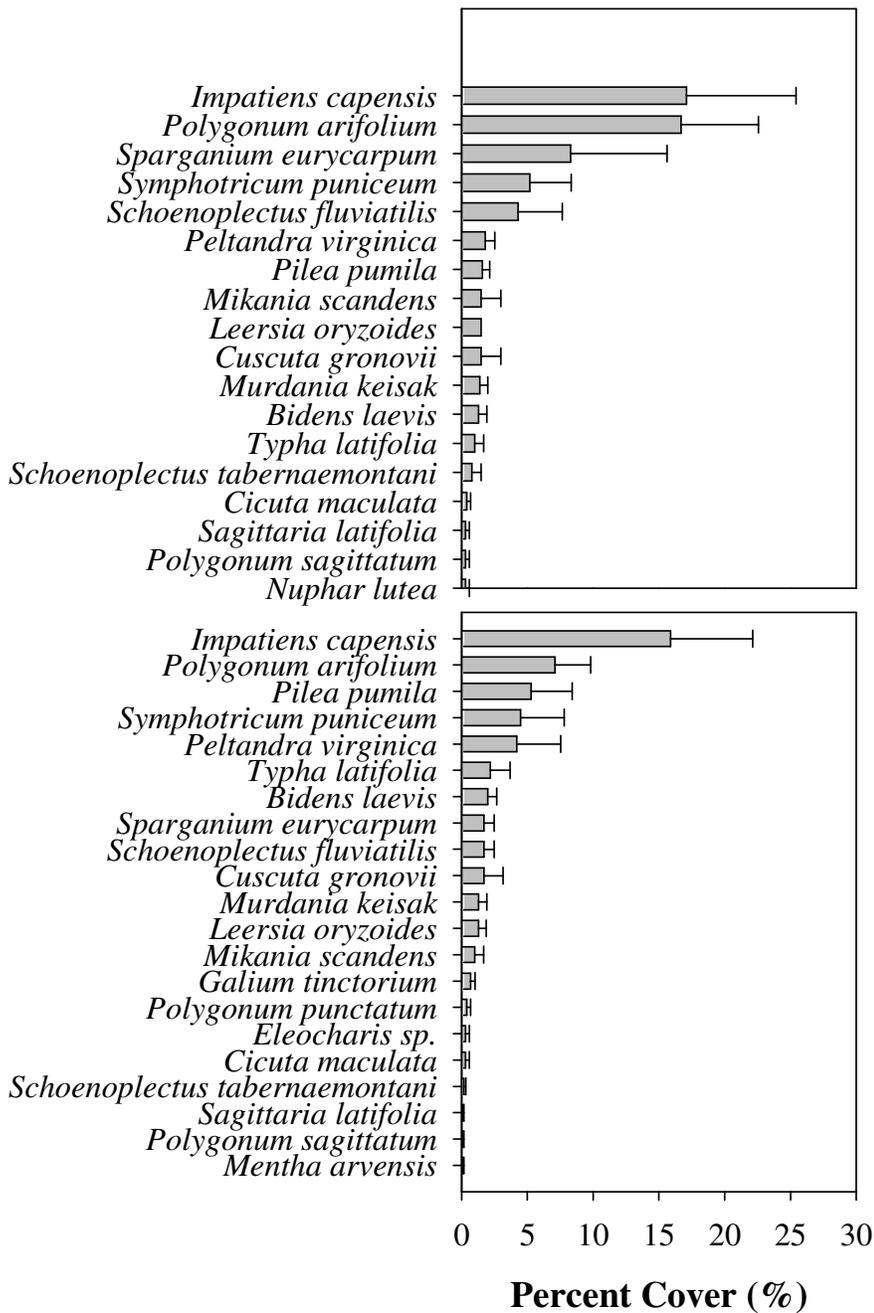


Figure 4.7: Average vegetation percent cover of individual species in fresh (top) and salt (bottom) treatments. Cover for each species is an average of 5 treatment plots. Data are from final sampling date (August 2011) only.

Elevation Change

Rates of elevation change did not differ significantly between salt and fresh treatments ($p=.72$). Elevation increased 7.8 ± 4.9 mm/yr (mean \pm SE) in fresh-treated plots and 6.86 ± 2.3 mm/yr in salt-treated plots from the baseline (Day 0, June 2008) until the most recent sampling date (November 2011) (Table 3.1; Fig. 4.8). On average elevation consistently increased in both treatment plots at each sampling event except for the final sampling period (between Day 1155 and 1245) (Fig. 4.8). During the final sample period, August 2011 to November 2011, Hurricane Irene and Tropical Storm Lee submerged the entire marsh for a period of roughly one week. For this reason, results were analyzed both including the final sampling date and excluding it.

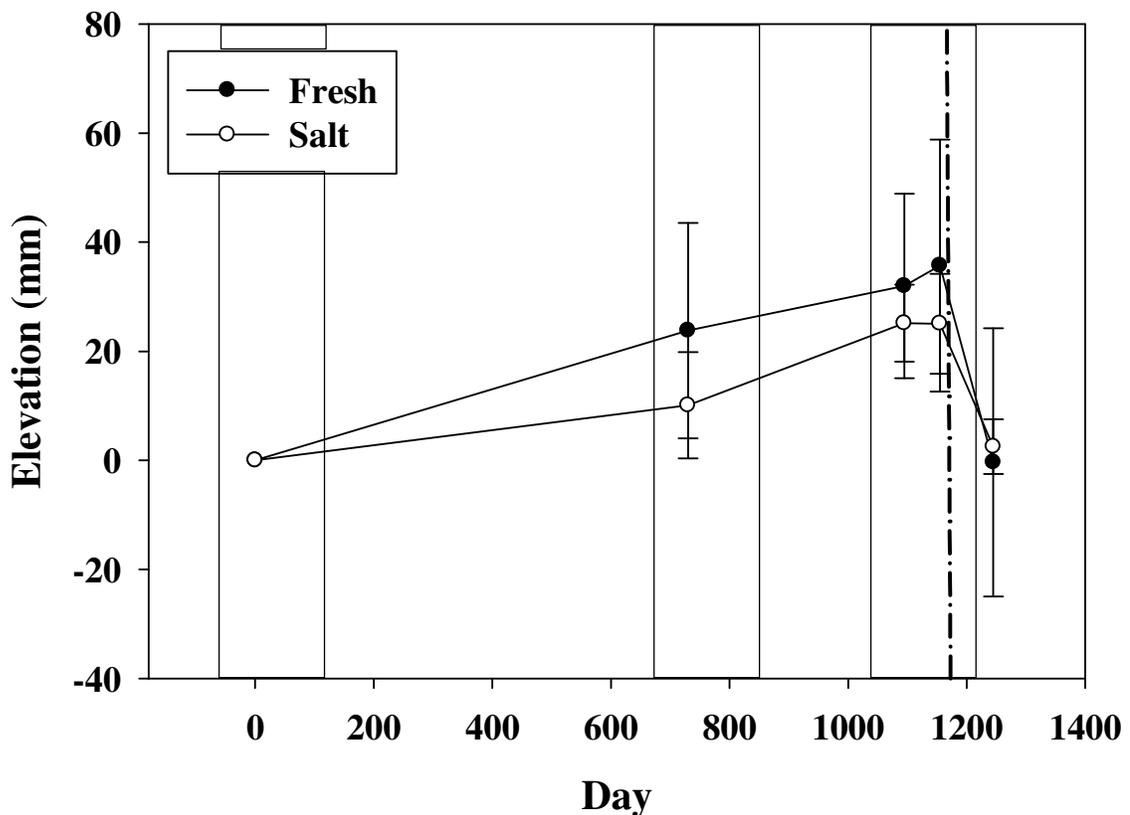


Figure 4.8: Average elevation in salt and fresh plots. Each point is an average of 5 SET averages (3 positions/1 SET). Day 0 is baseline measurement, taken in June 2008. The

final sample date was November 16, 2011. Boxes are growing season (April-October in Maryland). Dot-Dash line represents the date of Hurricane Irene (Day 1172, August 28, 2011).

Rates of elevation change varied little among sites and treatment plots (Appendix II). In fresh treatments, elevation change ranged from -7.5 mm/yr at site 5 to 32.77 mm/yr at site 1, although elevation change at in the fresh-treated portion of site 1 may be an outlier due to observed wrack deposition in the elevation plot. Excluding site 1, elevation change in fresh plots ranged from -7.5mm to 10.47 mm/yr. In salt plots, elevation change ranged from 0.54 mm/yr at site 1 to 12.23 mm/yr at site 3. Elevation change trends were not consistent by treatment (e.g., salt plots did not always exhibit higher or lower rates of elevation change), and differed by site and sampling event. All SETs experienced a significant elevation loss during the last sampling period, possibly linked to previous month's storm. Excluding the final sampling date (Day 1245, November 2011), rates of elevation change from Day 0 to Day 1155 in fresh plots ranged from -2.6 mm/yr at site 5 to 37.08 mm/yr at site 1, and 1.06 mm/yr at site 1 to 15.25 mm/yr at site 3 in salt plots.

Sites 1,2, and 3 were closer together adjacent to a tidal creek that branches from the main Patuxent river channel. Sites 4 and 5 were closer together adjacent to the Western Branch of the Patuxent (Fig. 4.1). When grouped together for each of these locations, surface elevation change at Western Branch plots was 5.09 ± 2.79 mm/yr in salt plots and -3.85 ± 3.66 mm/yr in fresh plots. Sites adjacent to the tidal creek had rates of surface elevation change of 4.71 ± 3.76 mm/yr in salt plots and 15.57 ± 8.84 mm/yr in fresh plots. Salt-treated plots on Western Branch were not significantly different from salt-treated plots on the tidal creek (Table 4.1). Additionally, fresh-treated plots were not significantly different between the two locations (Table 4.1). When treatment-effects

were compared within an area, rates of elevation change significantly differed between salt and fresh plots on the Western Branch side; however, fresh plots had higher rates of elevation loss ($p=0.06$).

Table 4.1: Treatment effects of salinity addition experiment on accretion, shallow subsidence, and surface elevation

Variable		Fresh	Salt	p-value
Accretion Rate (mm/yr)		20.34 ± 1.93	19.29 ± 1.70	0.67
Shallow Subsidence (mm/yr)		12.53 ± 8.62	14.43 ± 3.57	0.85
Elevation (mm/yr)	Overall	7.80 ± 4.86	6.86 ± 2.25	0.72
	Days 0-730	11.89 ± 9.87	5.04 ± 4.88	0.59
	Days 0-1155	11.15 ± 6.91	7.60 ± 2.65	0.67
	Western Branch Creek	-3.85 ± 3.66	5.09 ± 2.79	0.06
	Western Branch vs. Creek	p-value: 0.15	p-value: 0.94	

Rates of surface elevation change were analyzed from Day 0 to Day 730, the first sampling date, to determine whether there was an initial difference that was later muted by vegetation adapting to oligohaline conditions. Surface elevation change did not differ significantly between treatments in the initial sample period (Day 0 to Day 730) (Table 4.1).

To examine the potential differences due to distance of SET pins from treatment wells, individual SET pins were compared. Salt wells were set-up directly adjacent to the ninth pin (pin 9) and measurements at pin 1 were furthest elevation measurements from salt well. Differences of pin measurements from baseline measurements (Day x pin

measurement – baseline measurement for that pin) were compared by individual pin (1 vs 9), or distance from treatment well.

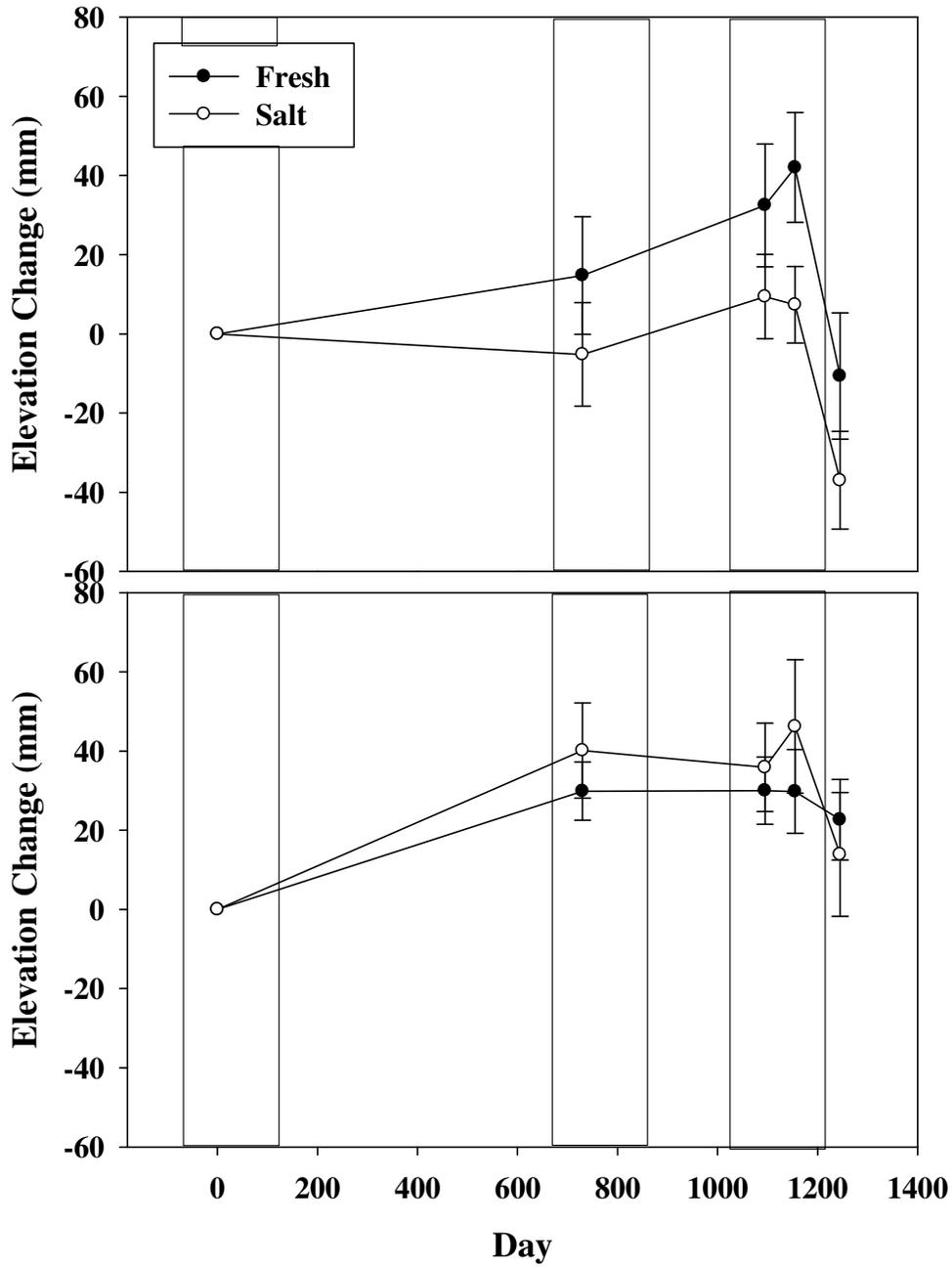


Figure 4.9: Elevation differences between Pins 1 (top) and 9 (bottom), and between treatments. Pin 9 is adjacent to salt well. Pin 1 is 30 cm from Pin 9. Each sampling event

is an average of 5 averages of 3 pins (3 pin 1 or 9/SET, 5 SET per sample point). Boxes indicate growing season (April to October in Maryland).

Elevation differences (difference from baseline) in salt plots only significantly differed between pin 1 and pin 9 on Day 730 ($n=15$, $p=0.03$), Day 1095 ($n=15$, $p=0.03$), and Day 1245 ($n=15$, $p=0.003$). Elevation was significantly lower at Pin 1, the furthest pin from the salt well (approximately 35 cm from salt well). Elevation change measurements did not meaningfully differ between pin 1 and pin 9 (Fig. 4.9).

Accretion and Shallow Subsidence

Accretion rates did not significantly differ between treatments (Table 4.1). Accretion rates in fresh plots ranged from 14.32 mm/yr to 25.72 mm/yr, and in salt plots 13.51 to 22.62 mm/yr. Rates of shallow subsidence did not differ between salt and fresh plots (Table 4.1). Shallow subsidence ranged from -18.54 mm/yr to 29.67 mm/yr in fresh plots and 1.28 mm/yr to 21.08 mm/yr in salt plots.

Discussion

For coastal wetlands to keep pace with rising sea-levels they must increase vertically in surface elevation at rates that equal or exceed relative sea-level rise. One complicating factor affecting the ability of a wetland to maintain positive surface elevation is an increase in decomposition rates due to salinity intrusion. I hypothesized that rates of decomposition in tidal freshwater marshes would increase when affected by salinity increases due to the effects of sulfate reduction and possibly plant dieback, thus causing rapid shallow subsidence. However, despite additions of a cumulative amount of

165 kg of sea salt mix at each SET site (~820 kg total; enough to raise 23,000 L of water to 35 psu), I observed no significant effects on elevation or vegetation that could be attributed to treatments. Salinity additions may have affected decomposition dynamics; however, decomposition rates were not directly measured so no conclusions can be drawn regarding decomposition directly in this study. However, the lack of effects on elevation and plant community may suggest that tidal freshwater wetlands may be more resilient to increases in salinity than previously thought and may withstand oligohaline conditions for some period of time without significant changes in plant community composition or surface elevation and accretion. Although, it may also indicate that studies conducted over short time periods (4 years) do not accurately demonstrate the effects of long-term (decadal) saltwater intrusion. However, there are other factors that affect the surface elevation of tidal freshwater wetlands, and that may exacerbate the effects of saltwater intrusion. Major storm events which may become increasingly more common due to climate change (Michener et al. 1997) may have an array of effects in the event of major flooding. Flooding can compact marsh surfaces, decreasing surface elevation significantly (Cahoon 2006) and affecting the regeneration of perennials and the establishment of annuals, thus potentially influencing an accretion deficit by decreasing belowground biomass production (Nyman et al. 2006). Additionally, large storm events have significant impacts on sediment deposition and erosion.

Sustained oligohaline conditions during the growing season over a four year period did not significantly affect surface elevation, accretion, or vegetation at the surface. There was no appreciable effect of saltwater addition on rates of surface elevation change. Rates of elevation increase in both fresh and salt plots were 7.8 ± 6.9

and 4.9 ± 2.2 mm/yr respectively; sufficiently high to maintain elevation under the current rate of relative sea-level rise in Chesapeake Bay of 3-4 mm/yr.

Porewater salinities increased with depth in salt-treated plots, indicating that salts had greater vertical movement than horizontal. Nuttle (1988) found that horizontal porewater movement is determined by distance from tidal creek or river and tides, and interior marshes (>15 m from tidal creek) experience very little to no horizontal movement of porewater (Nuttle 1988). In support of this, fresh-treated plots though adjacent to salt-treated plots did not exhibit salinities greater than ambient, even at depths below the 30 cm vertical barrier I placed between the plots (indicating that salt did not move horizontally even at depths below 30 cm). Simple possible explanations for this are the fact that saltwater has a higher density than freshwater, and so would sink in saturated soils, and salt wells, though they had holes at the soil surface, would have dispersed salt at the bottom of the capped well for a greater amount of time compared to at the surface (gravity pulls the salts down as the well empties). Additionally, if there was a large amount of horizontal diffusion of salts, there may be a treatment signal that weakened with distance from the well; however, when analyzed by pin position (or distance from treatment well), there were no differences between pin 1, the farthest pin from the salt well, and pin 9. Because of this, it appears that the salts moved downward primarily rather than outward.

Vegetation community dynamics may not have been significantly altered because salt treatments primarily affected the soil below the root zone (Top 30 cm). Flushing by freshwater (i.e., precipitation or surface flooding from increased river stage) may have muted the effects of salt treatments; however, given that there were residual oligohaline

salinities two weeks post treatment, it is possible that vegetation responds primarily to increased salinities at the surface. Initial effects of sea-level rise would result in pulses of higher salinity during storm events and droughts (reduced freshwater input) followed by accumulation of salts in marsh soils not tidally flushed, concentrations of which would depend on rates of evapotranspiration and hydroperiod. Porewater salinity levels can remain elevated for over one year (Blood et al. 1991, Middleton 2009). Over time, elevated levels of salinity would become more consistent as the seasonal salt wedge move up-estuary as sea-level rises. Treatments in this study simulate the initial stages of sea-level rise given that treatments were applied every two weeks and simulate pulses with initial salinity levels higher and diffusing over time. If this study is any indication, continuous elevated levels of salinity in marsh substrate due to the accumulation of salts may be expected under sea-level rise conditions.

The residual treatment effects of low-level salinity increases in the soil may in the long run cause a shift towards more salt-tolerant plant species. Initial community dynamics may shift towards the establishment and regeneration of salt-tolerant species already present in the seed bank (Middleton 2009), and eventually facilitate the establishment of new salt-tolerant species. Baldwin et al. (1996) found that seed viability was not affected by salinity pulses. However, recruitment from seed bank and germination of annual species and regeneration of perennial species are significantly affected by increases in salinity and flooding (Baldwin and Mendelsohn 1998a, Baldwin, Egnotovich and Clarke 2001, Baldwin et al. 1996, Callaway and Sabraw 1994), and increased salinity may limit the ability of plant communities to rebound after salt pulses. Baldwin et al. (1996) found that seedling establishment was affected when

salinity levels were increased to 2 psu and that at salinities of 4 psu, germination rates were low and few seedlings could establish. Species diversity and total cover were not affected by salinity increases in this study; however, one possible explanation is the lack of an salinity treatment effect on seed viability. Study plots were dominated by annuals; germinated by seed each year. If seed viability was unaffected, and treatments were applied after seedling recruitment and establishment, than it would preclude vegetation from the detrimental effects of increased salinity on germination and establishment.

Despite the lack of salinity treatment effect, there was a significant drop in species richness between pre-treatment plant communities and two years into treatments (Day 730). One possible explanation for this is increased flooding and application of saltwater at the surface. Treatments during the first two years consisted of 100 gallons of either fresh or salt water applied via irrigation hoses to the study plots. This increase in surface flooding may have decreased species richness given that seedling germination of many species is negatively affected by flooding. Evidence to support this supposition is the switch from saltwater application to salt-filled wells after the second year of treatment. During that time, species richness rebounded some, but did not reach initial number of species per plot. Additionally, this would support the lack of effect on vegetation due to salt treatments being below the root zone. Initial treatments dispersed saltwater onto the surface of the marsh, which percolated down through the soil layer, thus affecting the root zone as well as settling into the soil below the rhizosphere. In later salt-well treatments, saltwater may have been delivered at deeper depths (diffusing from the bottom of the well at 50 cm, thus excluding the top 30 cm root zone) and thus did not affect the root zone, and there were no further changes in species richness and total

percent cover. In order for vegetation communities to change because of saltwater intrusion, salinity level increases may have to affect the zone where physiological and initial growth dynamics (seedling recruitment and germination and vegetative regeneration) take place: the root zone and seed bank zone near the surface of the marsh. Treatment salinity levels may also have been too low to cause significant changes in vegetation communities.

In this study, I hypothesized that there would be a vegetation shift either to more perennials (regeneration may not be so greatly affected compared to seedling recruitment under higher salinity conditions) compared to annuals or to more salt-tolerant annuals in the salt-treated plots; however, there were no significant shifts in the ratio of annual: perennial species, and no particular species changed in percent cover significantly. As mentioned above, this may be because the salinity increases primarily affected the substrate below the root zone. Another possible explanation of the lack of community change is hydrology. Plants are adapted to combinations of salinity tolerance and flooding regime. Because the hydrology of the marsh was not significantly altered, except some increased soil waterlogging during the initial two seasons, by my study, vegetation communities did not have to respond to sustained increased flooding, which may exert a greater influence on community composition compared to small increases in residual salinity.

Surface elevation is affected by many processes: root growth, decomposition, hydrology, sediment deposition, to name a few. I hypothesized that with the presence of sulfates in salt-treated plots, rapid shallow subsidence would result from increased rates of decomposition in tidal freshwater wetlands. Sulfides, the residual product of sulfate

reduction, were not measured in this study so I am unable to draw definite conclusions regarding which microbial respiration pathway was dominant (i.e., methanogenesis, sulfate reduction); however, decomposition did not have a measurable effect on surface elevation, possibly because sulfate reduction was being inhibited. It may be that there are more prominent microbial respiration processes governing decomposition rates at the surface (root zone) for at least some part of the growing season, and/or that the quality of peat below the root zone is resilient to the effects of sulfate reduction. In many tidal marshes, Fe(III) reduction is a dominant metabolic pathway during the beginning of the growing season, and because it is a more efficient respiration pathway compared to either sulfate reduction or methanogenesis, if it is the dominant pathway at both my fresh and salt sites, it would mask or suppress sulfate reduction and methanogenesis, thus possibly explaining the lack of significant differences between the plots (Neubauer et al. 2005, Sutton-Grier et al. 2011, Kostka et al. 2002, Ma et al. 2008). In addition, there may be Fe-S interactions that affect and interfere with sulfate reduction (Burton et al. 2011). Neubauer et al. (2005) examined rates of anaerobic respiration in Jug Bay and found that in the early part of the growing season (June) Fe(III) reduction accounted for 98% of measured metabolism. Additionally, rates of anaerobic metabolism were significantly lower at the brackish sites downstream from Jug Bay (Neubauer et al. 2005). Fe(III) reduction is limited by the availability of Fe (III) and organic matter substrate. Once reduced to Fe(II), the metabolic process would potentially shift from iron reduction to sulfate reduction, if sulfates and organic matter were available, and methanogenesis if no sulfates are present. Theoretically, then, iron reduction can only occur while Fe(III) is being added to the system; however, in the rhizosphere of tidal wetlands are many Fe(II)-

oxidizing bacteria that create Fe(III) plaques on the roots of tidal marsh plants (Weiss et al. 2003, Weiss, Emerson and Megonigal 2004, Weiss, Emerson and Megonigal 2005), thus creating a cycle of Fe(III)-Fe(II) reduction and oxidation governed by bacterial processes in the rhizosphere. Despite this, Fe (III) reduction may be dominant in the beginning of the growing season, but by late summer may taper off in favor of methanogenesis (Neubauer et al. 2005). Despite this, given that Fe(III) reduction is mediated by vegetation roots, one would still expect a possible shift from methanogenesis to sulfate reduction in the substrate below the root zone, causing a drop in surface elevation.

There may be no significant differences in surface elevation change due to a lack of significant difference in rates of total mineralization by sulfate reducers and methanogens (D'Angelo and Reddy 1999, Sutton-Grier et al. 2011). Anaerobic decomposition in wetland soils is limited by availability of carbon, primary production being the main carbon source (Sutton-Grier and Megonigal 2011). In cases where mineralization is limited by availability of organic matter (electron donor), total mineralization rates would be the same regardless of terminal electron acceptor (i.e., sulfate reducers or methanogens), because whatever organic matter is available to be respired will be, even if it is at different rates (Sutton-Grier et al. 2011). Organic matter substrate may be limited at study plots. Tidal freshwater wetland plant species often have lower C:N ratios and are more easily mineralized. At my study sites, easily-decomposable compounds may have mineralized immediately, or in subsurface soils may be already reduced, leaving behind higher C:N, lignin-rich organic matter that limits rates of total mineralization. Sutton-Grier et al. (2011) found that when tidal freshwater

wetland soils of Jug Bay were transplanted into brackish marshes downstream, they had low rates of mineralization; however, the authors concluded that it was most likely due to the persistence of residual bacteria present in the cores.

Salinity may not be the strongest driver of carbon mineralization. Rather, hydrology, carbon sources (i.e., vegetation type) and soil temperature may exert greater influences. Krauss et al. (2011) found that there were no significant differences in methane and carbon dioxide fluxes (i.e., mineralization) along a salinity gradient from tidal freshwater to oligohaline forested wetlands (Krauss and Whitbeck 2012). These conclusions differ from many previous studies (Chambers, Reddy and Osborne 2011, Weston et al. 2011, Portnoy and Giblin 1997); however many studies have examined the effects of sulfate additions on soil cores incubated in greenhouses and labs. Site-specific, *in situ*, factors including temperature and hydrologic differences, may exert a greater influence on gas fluxes compared to salinity additions.

Storm events, predicted to become more common due to climate change (Michener et al. 1997), can significantly alter the surface elevation of a marsh, at times at a magnitude that trumps autogenic processes such as accretion and shallow subsidence, and causing lasting change. Surface elevation was decreased significantly at all of my plots during the final sampling period possibly due to the effects of sustained flooding from Hurricane Irene and Tropical Storm Lee: back to back storms that lingered over Upper Marlboro, MD (Jug Bay) for an extended amount of time in September 2011. Though storms can often increase elevation by depositing large amounts of sediment scoured from the watershed onto the surface of marshes and swelling peaty soils with increase in flooding, in this case, sediments were either deposited in sand bars in the

channel of Western Branch (personal observation), or eroded by the swiftness of the heightened river stage and discharge. Though it is beyond the scope of this study, many changes may result from the effects of lowered elevation and sediment deposition on the banks of Western Branch and the main river channel of the Patuxent. The sediment may act as a subsidy to marshes in that area, or may impair vegetation communities by lowering seedling recruitment (Peterson and Baldwin 2004). Additionally, a decrease in surface elevation may lead to the increased vulnerability of marshes to flooding, which, as discussed, decreases species diversity, seedling recruitment and vegetative regeneration, thus leading to decreased belowground biomass production, thus leading to further accretion impairments and decreased surface elevation.

Conclusion

Salinity levels were increased to oligohaline conditions in a tidal freshwater wetland over the course of four growing seasons. Initial salinities affecting the surface of the marsh, specifically, the root zone and seed bank, caused a decrease in species richness and total percent cover of vegetation; however, surface application did not significantly affect surface elevation change, most likely due to a lack of a significant effect on microbial respiration rates because of the dominance of Fe (III) reduction. Treatments affecting the marsh soil below the root zone had no significant effects on vegetation communities, and may not have affected decomposition due to lack of organic matter supply and preclusion of vegetation from treatments. Tidal freshwater marshes may be more resilient than expected to the effects of saltwater intrusion given the success of

treatments in raising salinity and the lack of significant differences between salt and fresh-treated plots.

Chapter V: Conclusion

The main questions I sought to answer through my dissertation research were: 1. whether marshes of Chesapeake Bay were keeping pace with sea-level rise, i.e., were they maintaining positive surface elevations relative to sea level, and 2. what were the controlling mechanisms of surface elevation change? To address these questions, I took a multi-faceted approach; first identifying rates of surface elevation change and subsequently investigating the influences of inherent characteristic variations (salinity, stratigraphy, and geomorphic type) on marsh subsidence. My original hypotheses were that tidal freshwater wetlands would have higher rates of surface elevation loss and subsidence under conditions of saltwater and intrusion or, at the very least, that marshes along an estuarine gradient would have different rates of accretion, shallow subsidence and elevation loss, affecting their vulnerability to the high relative rates of sea-level rise in Chesapeake Bay. I tested those hypotheses and found that oligohaline wetlands have higher rates of shallow subsidence, despite accretion being uniformly high across the estuary, but that the spatial variability was huge, even within study sites.

Questions arising from initial studies on salinity intrusion and elevation trends were to explore the causes of shallow subsidence and to identify why oligohaline marshes were subsiding more, and why the marshes of the Nanticoke River are all subsiding since it was not for lack of surface accretion. I hypothesized that the geomorphology was dictating shallow subsidence, specifically that deeper, more organic sites (estuarine meander marshes) were subsiding more than shallow mineral-rich sites (submerged upland marshes) due to the disproportionate effects of decomposition and compaction on

deeper, peaty soils compared to mineral soils or soils with shallower organic layers. These hypotheses were not supported, though the picture is not clearly rejecting these hypotheses. Shallow subsidence has unusual curvilinear relationships to organic matter and depth within a stratigraphic profile, just as it has a curvilinear relationship to salinity and position in the estuary (distance downstream). These curvilinear relationships may not be ecologically meaningful (as in, cannot be applied to other estuaries or used to predict relationships between variables), but rather they simply describe the marshes of the Nanticoke River and highlight the unusual nature of mid-estuarine Sites and the high variability within an estuary.

To further confound the story of sea-level rise and coastal wetland subsidence, my hypothesis that tidal freshwater wetlands would be negatively impacted by saltwater intrusion and sulfate reduction, I found that after dosing a tidal freshwater marsh with salt for four years, there were no clear responses of the substrate or the vegetation. These results differ from many other studies (Delaune, Nyman and Patrick 1994, Baldwin and Mendelsohn 1998a, Weston et al. 2006, Cahoon et al. 2006) that found that vegetation communities shift to more salt-tolerant communities, organic matter mineralization increases with the addition of sulfates, and that peat collapse and loss of surface elevation result from increases in salinity and flooding. My contradictory results may be because salinity levels were not raised high enough in treatment plots, or for a long enough time period annually; however, it may also indicate that coastal wetlands, and tidal freshwater wetlands are dynamic and may be more resilient to stressors than previously thought, at least in some locations.

My goal was to put together a conceptual model, utilizing my data, that could help forecast sea-level rise response processes and feedback loops in marshes of Chesapeake Bay in order to identify potential hotspots for drowning; however, this was not possible given the complex picture my results paint. Instead of conclusions, further questions and hypotheses were raised. Original hypotheses and objectives of my dissertation research are discussed below followed by my best-guess hypotheses for the data trends I observed and possible avenues for further exploration.

Objective 1:

Identify surface elevation, accretion, and subsidence trends along a salinity gradient

Original Hypotheses: Tidal freshwater wetlands will have higher elevation loss due to saltwater intrusion stresses and increased decomposition from sulfate reduction

Result: Marshes of all salinities have insufficient rates of vertical surface elevation gain to keep pace with current and future sea-level rise, and are subsiding. Oligohaline marshes have the highest shallow subsidence in addition to the greatest within-site variability

Explanation and Hypotheses for Further Study

- a. Increased decomposition due to sulfate reduction and eutrophication. To test the hypotheses that excessive nitrogen from rural septic systems or sulfides from salinity pulses are increasing decomposition, porewater analyses along a depth profile for isotopic nitrogen and sulfides could be conducted. Additionally, a cross-estuary decomposition study could aid in identifying differential decomposition.
- b. Groundwater withdrawal and autocompaction are causing shallow subsidence.

Objective 2: Simulate saltwater intrusion in a tidal freshwater marsh to evaluate how vegetation communities and substrate will change

Original Hypotheses: Vegetation will exhibit stress and high mortality in salt-treated plots and more salt-tolerant plants will colonize or increase in cover, and surface elevation will decrease due to increased decomposition from a switch from methanogenesis to sulfate reduction

Results: No significant change in vegetation total cover or cover by species and no significant differences in surface elevation change between salt and fresh plots

Explanation and Hypotheses for Further Study

- a. Salinity levels were not high enough to cause changes in vegetation, or salinity treatments did not affect seedling establishment and germination due to treatments beginning too late in the growing season after seedlings were already established. Additionally, salinity was sequestered below the root zone and so did not affect the vegetation. To test these hypotheses, adding more salt during the growing season, on the surface, and adding salt beginning in January rather than in April, May, or June. Additionally, collecting above and belowground biomass may illuminate differences that were eclipsed by lack of change in total cover
- b. Iron reduction and oxidation, a preferential pathway of respiration by soil anaerobes, is dominant in the root zone, muting any signal by sulfate reduction. Sulfate reduction was occurring, but sulfides were binding with the reduced iron present in the soil and forming immobile iron sulfide complexes. Substrate is carbon limited so total respiration rates did not change, thus causing no change in surface elevation. To examine these dynamics, microbial and biogeochemical

studies could be conducted to characterize microbe communities in experimental plots, test for iron-sulfide complexes, and examine amounts of carbon in the substrate.

Objective 3: Identify relationships between geomorphology and stratigraphy to shallow subsidence

Original Hypotheses: Older, deeper, more organic estuarine meander marshes would have higher subsidence rates compared to younger, shallower, less organic submerged upland marshes

Results: There were no clear relationships between stratigraphy and geomorphic type and shallow subsidence. Upstream Sites had more mineral sediments. Marshes near the mouth are older compared to upstream marshes

Explanation and Hypotheses for Further Study:

a) Upstream marshes (the first marshes along the river, formed as the river widens) on the main river channel receive more riverine sediments compared to marshes on tidal creeks and marshes near to the mouth of the estuary. Marshes at the mouth may receive marine sediments, but sedimentation is less in marshes flushed by tidal creeks compared to bordering the main river channel. To test this, sedimentation tiles could be placed at all Sites and sediment source characterized.

b) Estuarine marshes formed first at the mouth where historic sea-level rise occurred before pushing upriver. Tidal freshwater wetlands are younger and shallower compared to estuarine marshes downstream.

Marshes of the Nanticoke River and Chesapeake Bay are subsiding and are vulnerable to sea-level rise. Identifying the multiple factors that are contributing to subsidence is crucial to preventing the loss and drowning of estuarine wetlands. Each wetland has a unique set of characteristics and most likely, there are unique combinations of factors contributing to subsidence. This suggests that there is no clear silver bullet management approach, but rather the combination of multiple anthropogenic stressors acting synergistically to cause subsidence and degradation should be considered and addressed individually (i.e., mitigate saltwater intrusion, etc.). However, what is also apparent is that coastal wetlands are dynamic and varied and do not respond uniformly. Some coastal wetlands may be more resilient and may be able to persist under conditions of multiple stressors.

Appendix I.

Surface Elevation Table Installation

Set up two stools with a 15-ft long bench (metal bleacher bench ideal) across the stools. This will prevent disturbance of elevation plot. Drive 6” diameter, 24” length PVC pipe, sharpened on one end, into marsh until roughly 5-6” are above marsh surface using a rubber hammer. Connect two stainless steel survey rods with a driving tip, drive them into marsh in center of PVC pipe. Connect additional rod, drive down by hand. Utilize a bubble measure to ensure rods are being driven in straight. Once hand-driving reaches limit, utilize a pounder slammer. Pounder slammer is a reinforced steel cylinder with one closed end and handles on either side of the cylinder. Pounder slammer is placed over end of protruding survey rod, lifted and “slammed” down on the rod, driving it into marsh. Once pounder slammer reaches driving limit, utilize a reinforced steel “cap” placed over end of survey rod and sledge hammer. Continue to hammer until survey rods stop going into marsh surface (it will decrease to a couple of millimeters per hammer hit when point of refusal is nearly reached).

Once rods are driven to point of refusal, use a battery-powered angle grinder (DeWalt) to remove excess rod. Rod should be level with PVC edge. Attach receiver. Fill PVC pipe with quick-dry cement (note that cement will not set if mixed with salt water, it must be mixed with fresh water). Place survey marker in wet cement.

Surface Elevation Table Measurements

Set up stools and bench roughly 1-m from SET, take measurements from bench to ensure elevation plot is not disturbed. Remove cap on SET receiver (capped-threaded,

and notched portion of the permanent SET in concrete). Place notched portion of the SET that has the numbered-hole dial on it into receiver, fitting the notches together. Attach arm, locking pins into place in position. Level arm utilizing turnbuckles (2). Drop pins one pin at a time. To find marsh surface, slowly lower pin until there is resistance. To ensure this is marsh surface and not vegetation stem, rotate pin in fingers. Gently lift and lower pin again until surface is felt. Dead vegetation indistinguishable from marsh surface is included. No detritus was cleared from marsh surface. In inundated soils that have a soupy-saturated silt area, pins rest below level of saturated silt. Whatever method is employed, ensure that it is uniform for all pins, positions, SETs and measurement events. Once pin rests on marsh surface, clip a badge clip on the pin resting on the top of the arm to keep the pin in place.

When all nine pins are in place on marsh surface, measure distance from the top of the arm to the top of the pin. Place the ruler against the badge clip, leaning the top of the pin against the ruler. Read pin measurement. Whatever methodology for holding ruler, utilize identical procedure or ruler position for each pin, position, SET and measurement event.

Baseline measurements are the first set of measurements read (for this study, baseline measurements were taken 2 months after SET installation). Four of the eight positions were measured (evens or odds) for each SET (36 pin measurements/SET).

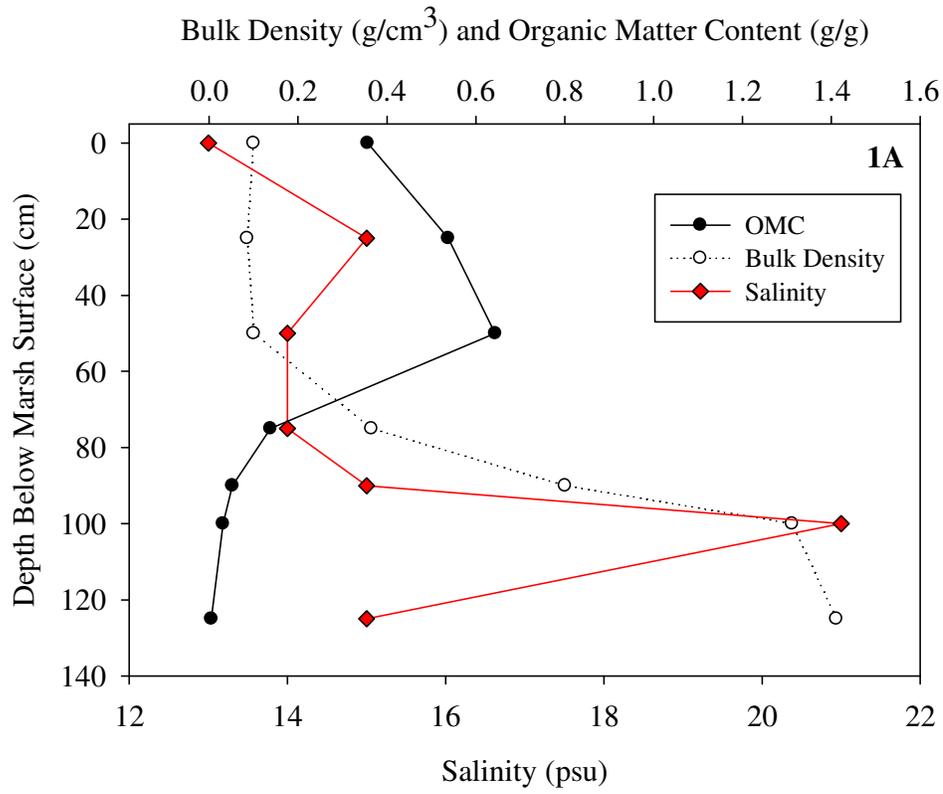
Surface Elevation Data Analysis

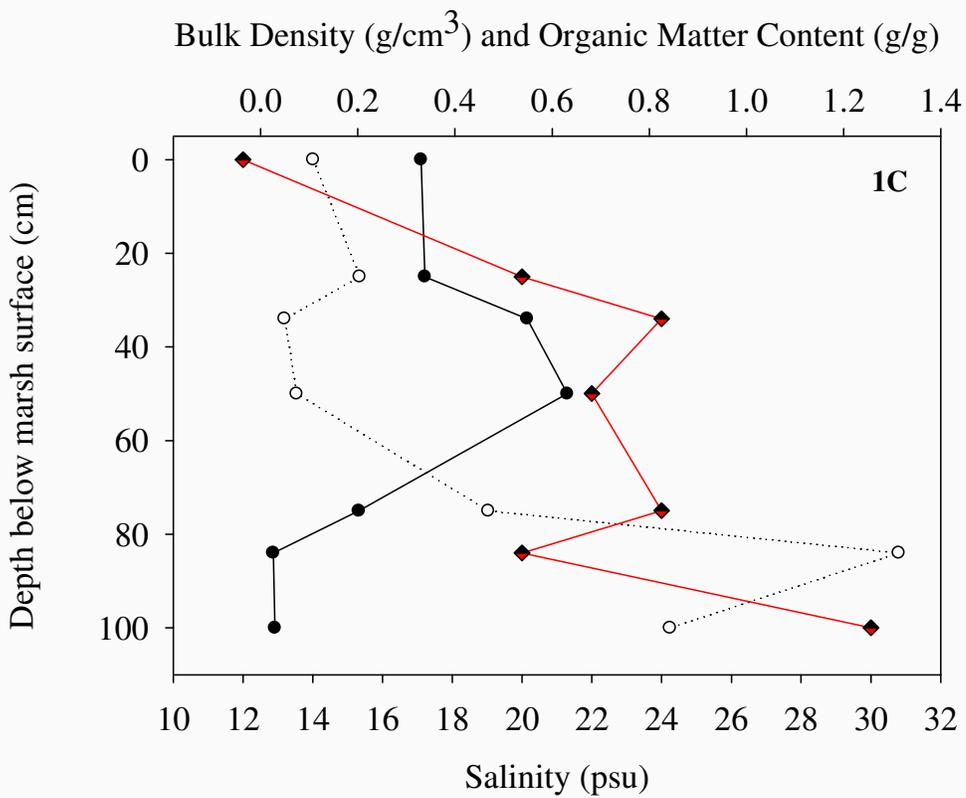
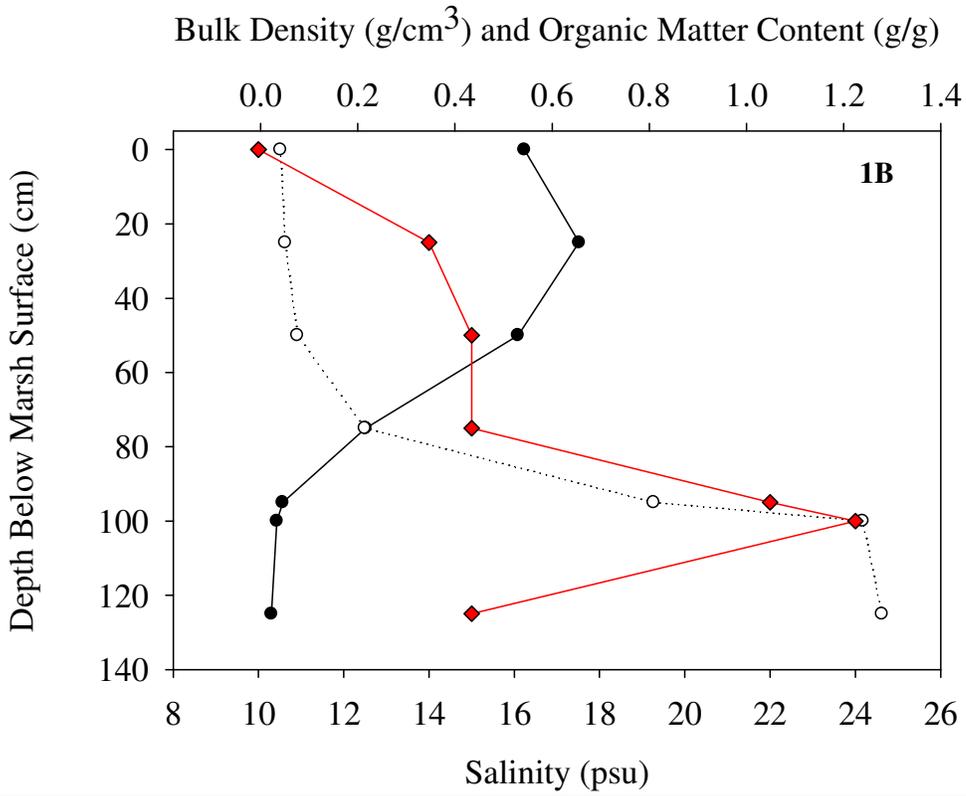
Subtract baseline measurements from recent set of pin measurements. For example, for measurement event two (first after baseline measurements), take the pin value from pin #1 in position 4 and subtract the baseline measurement for pin #1 in

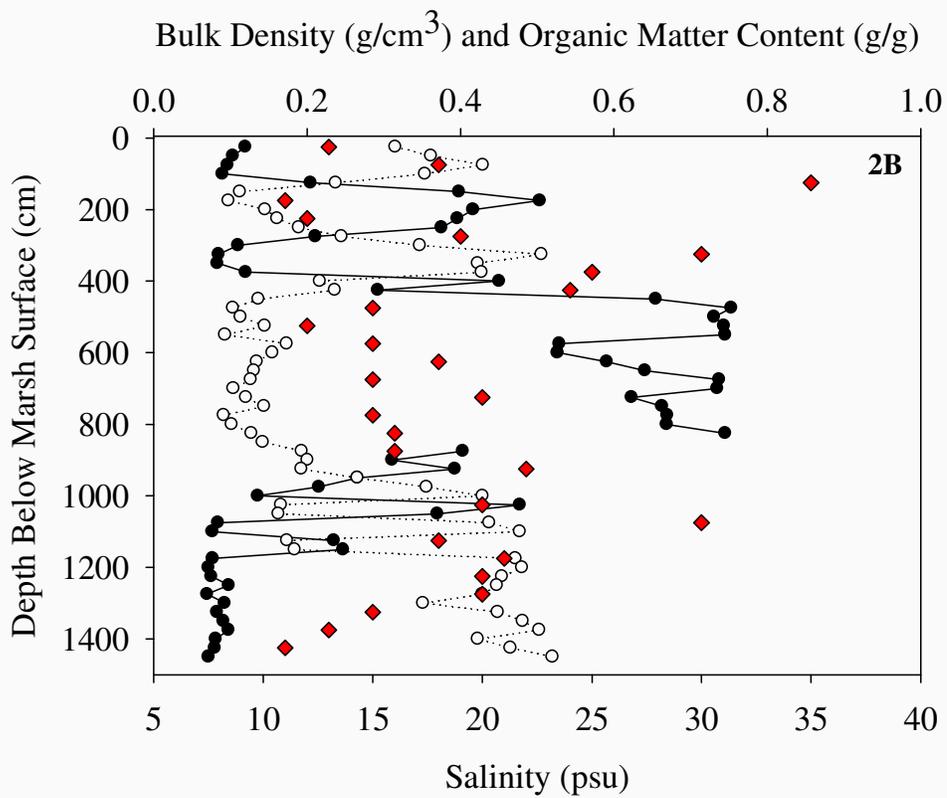
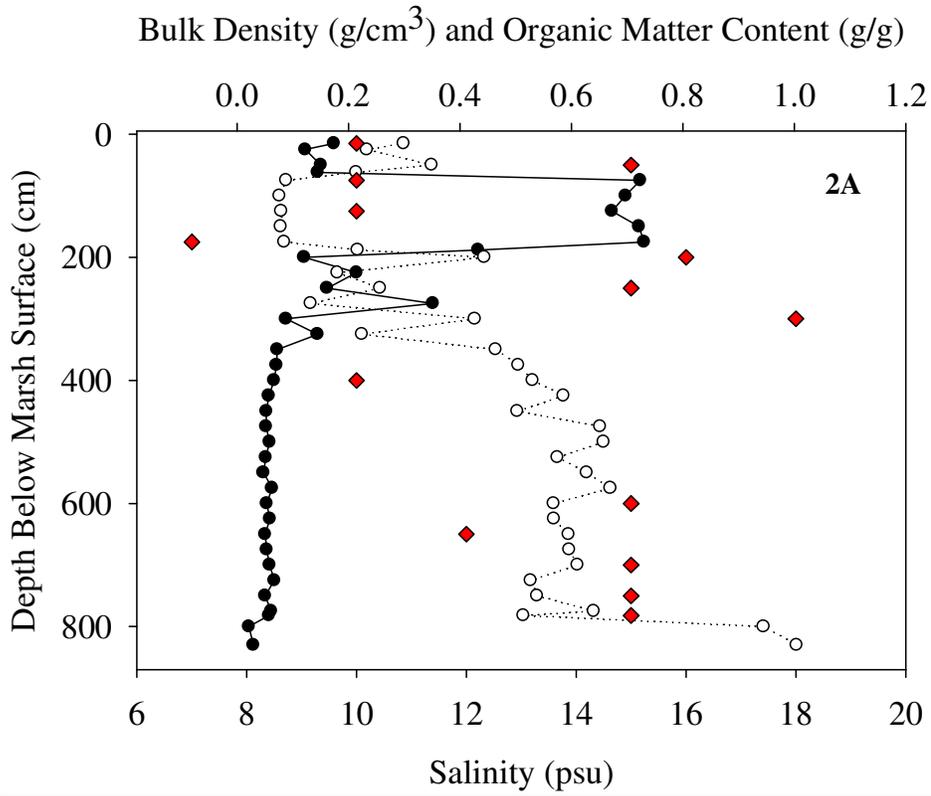
position 4. Do this for all nine pins in a position, average the differences for each position, then average the position averages for an SET average (average pin differences in each position, then average the 4-8 positions for an SET elevation average).

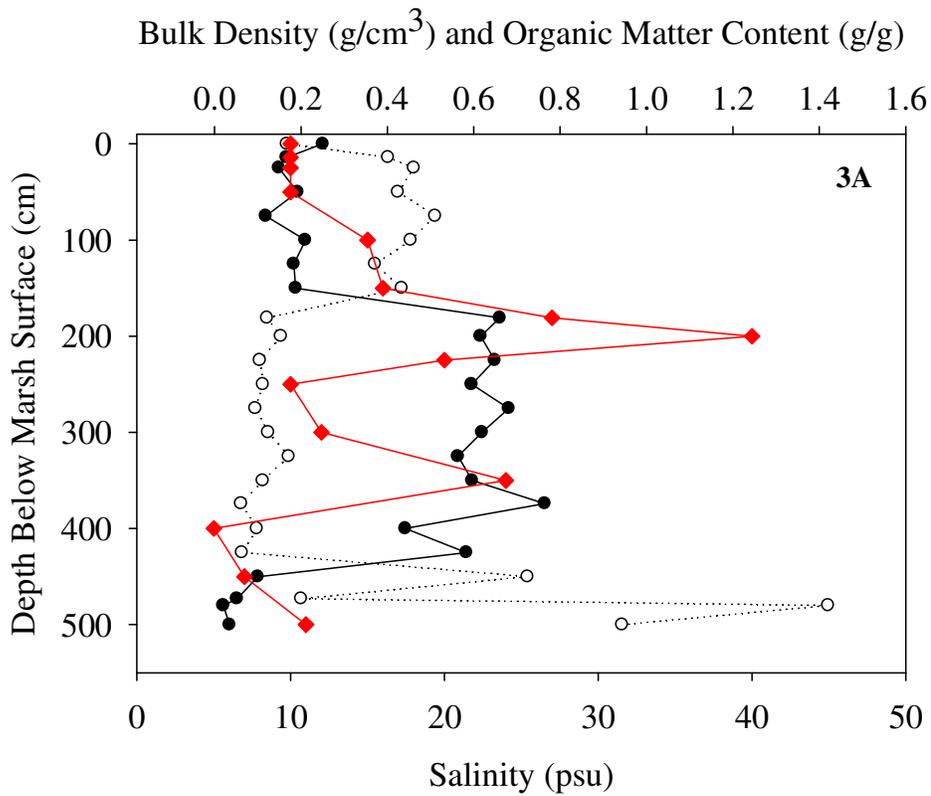
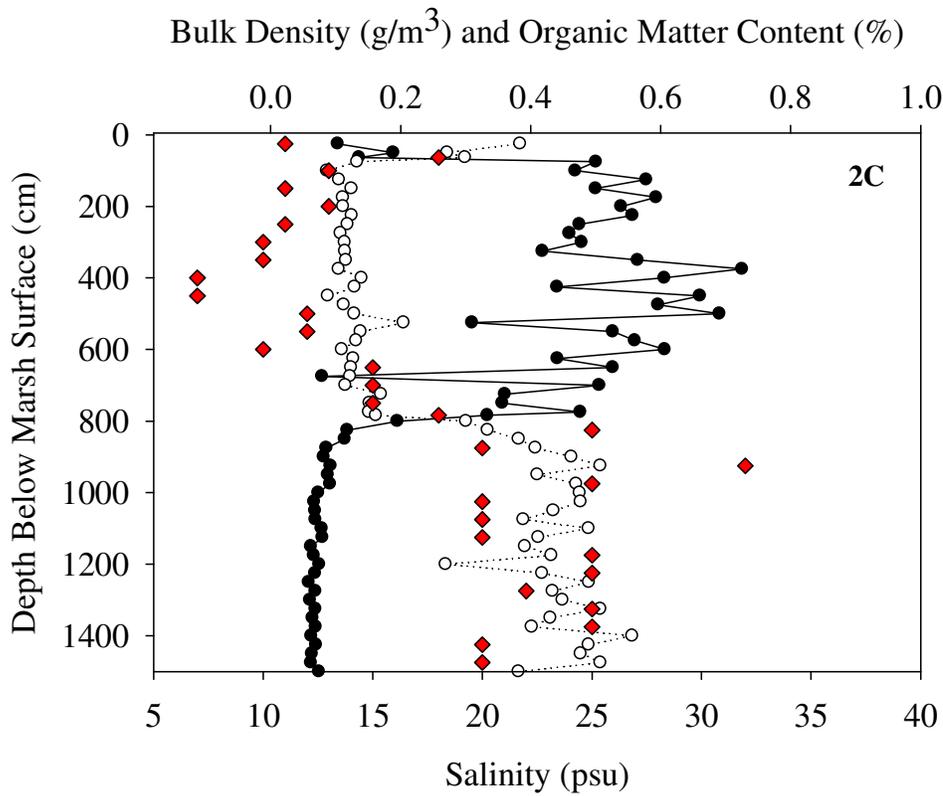
Plot SET average pin differences (y) by day (x) beginning at Day 0 (baseline) with a measurement of 0 (0,0). Add a line of best fit. Slope of this line will be your average elevation change in mm/day. For annual rates, multiply daily rate by 365. To determine whether rates of annual elevation change differ significantly among SET sites, conduct an ANOVA on the mean annual elevation change of each site (for this study, each replicate sub-site (3) was averaged among the five sites and an ANOVA detected significant differences among the means of each site (3 sub-sites/5 Sites, n=15). Add each new measurement event average pin measurements to trend line to readjust annual rate of elevation change.

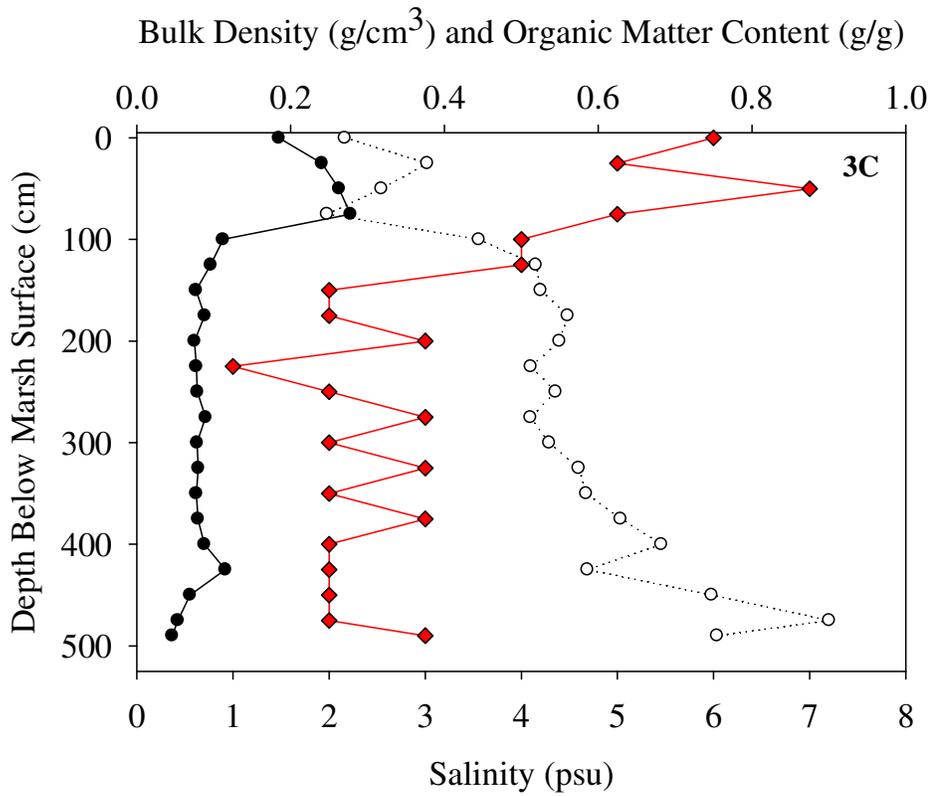
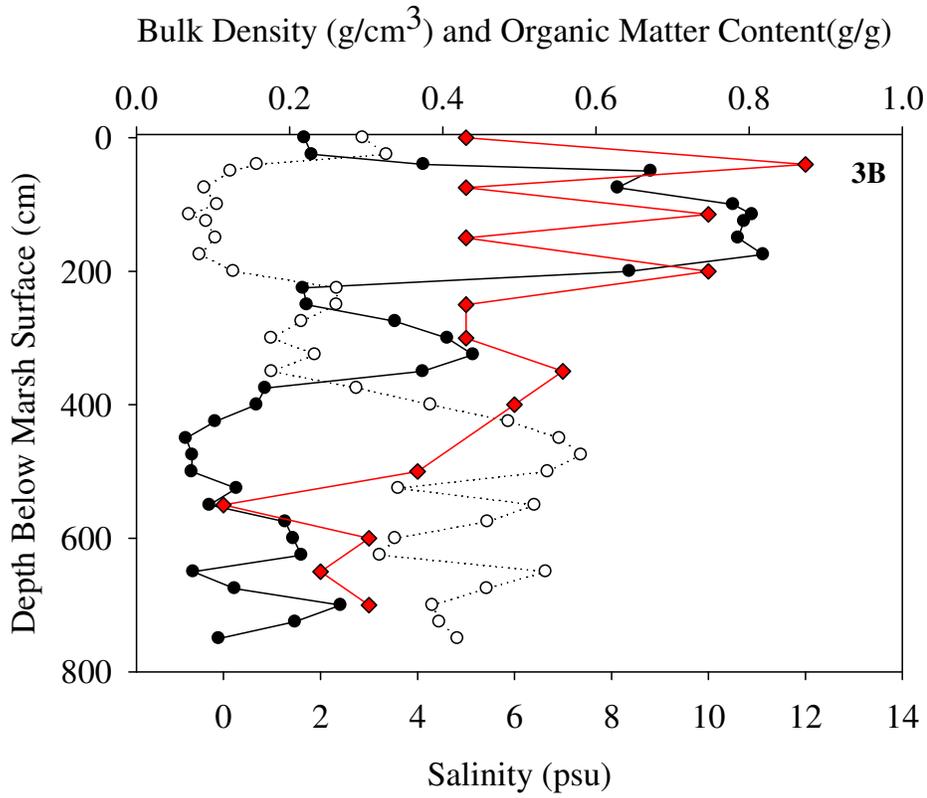
Appendix II:

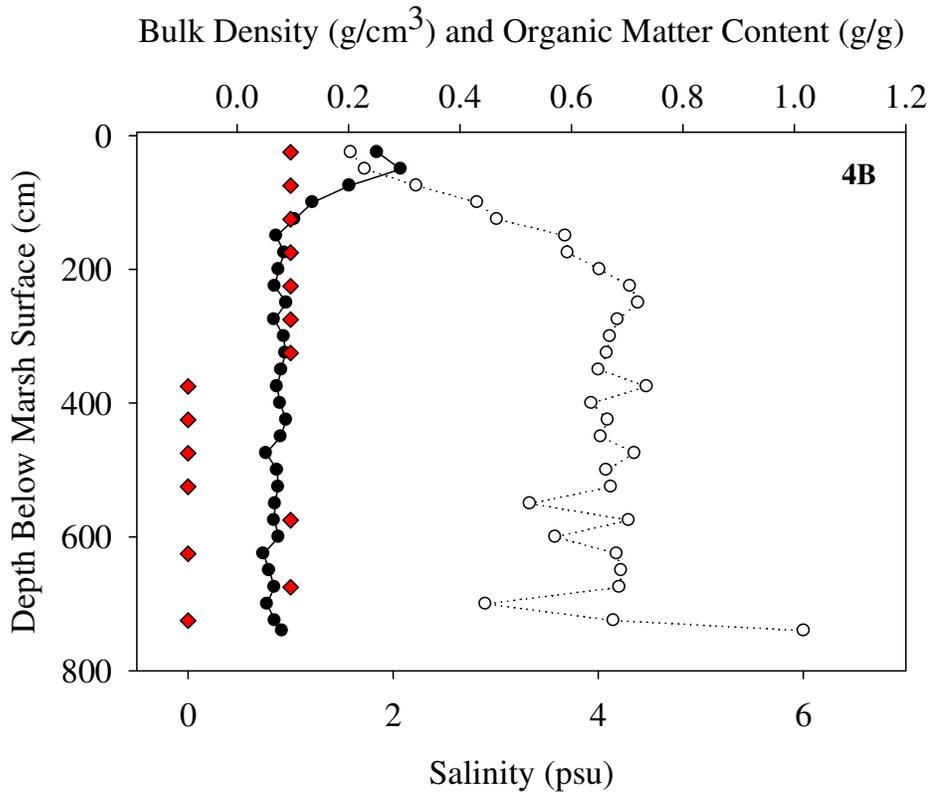
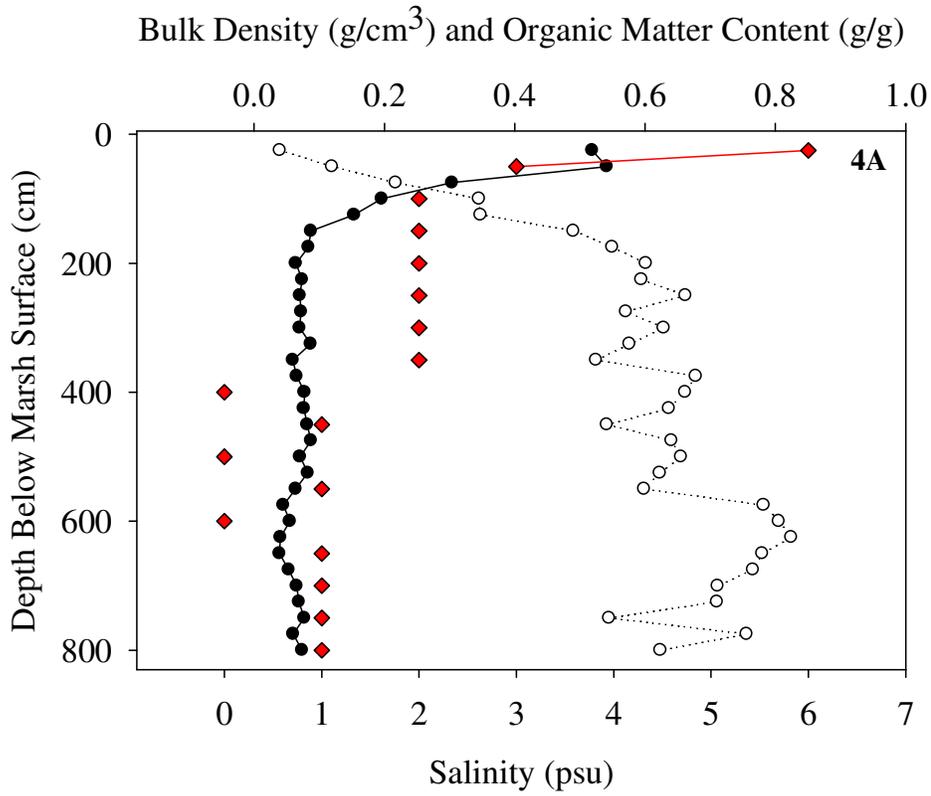


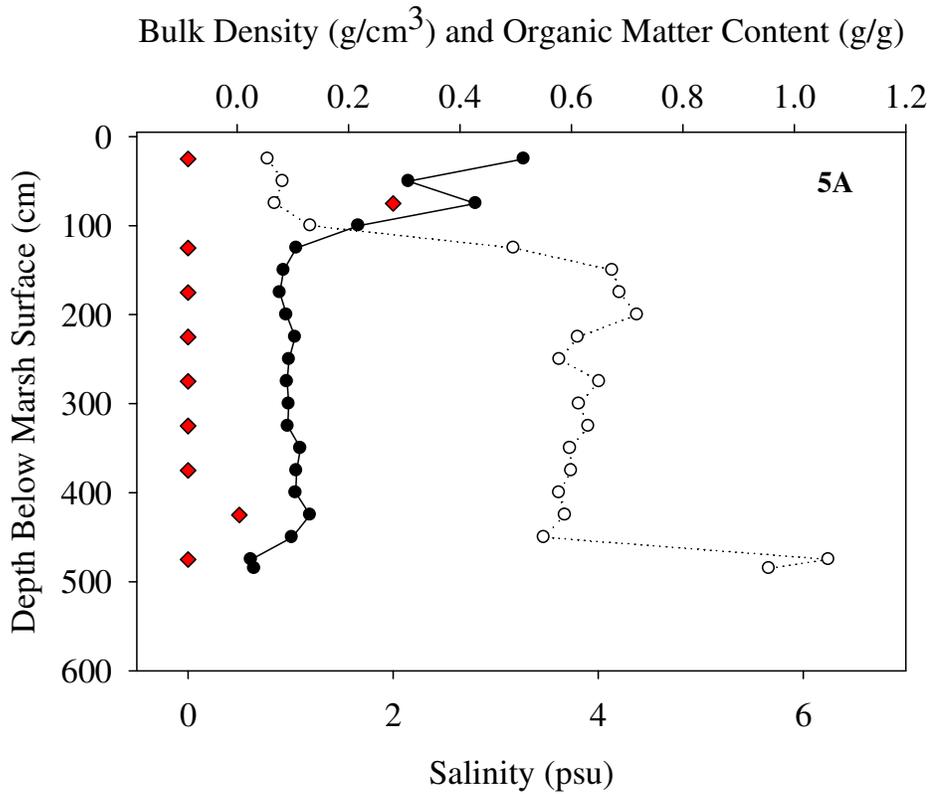
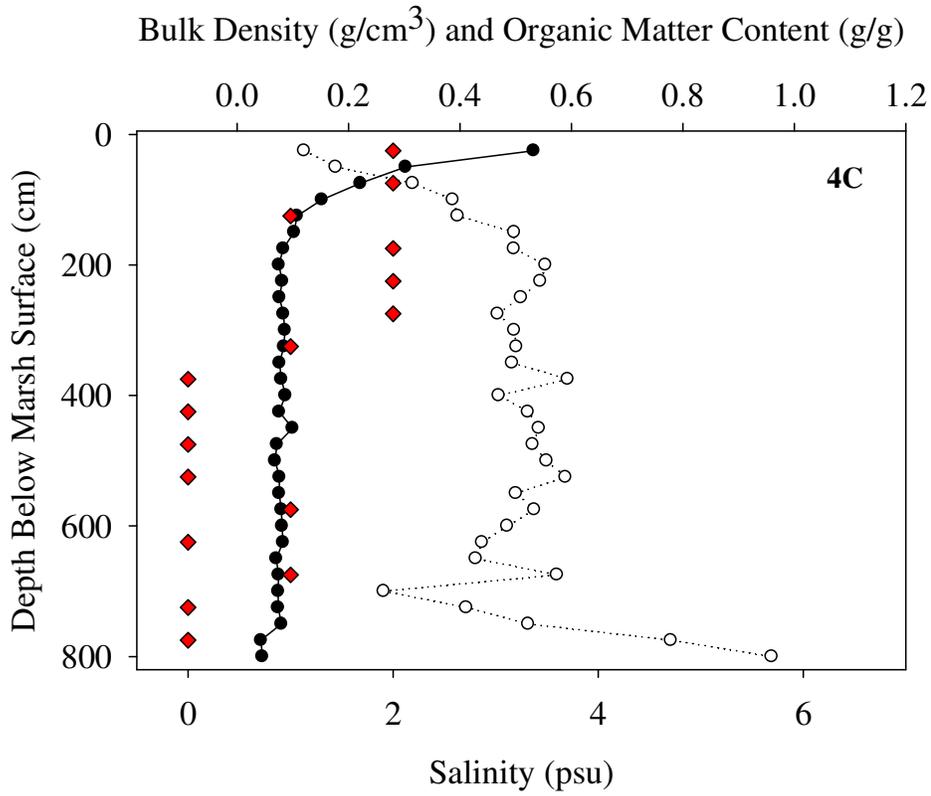












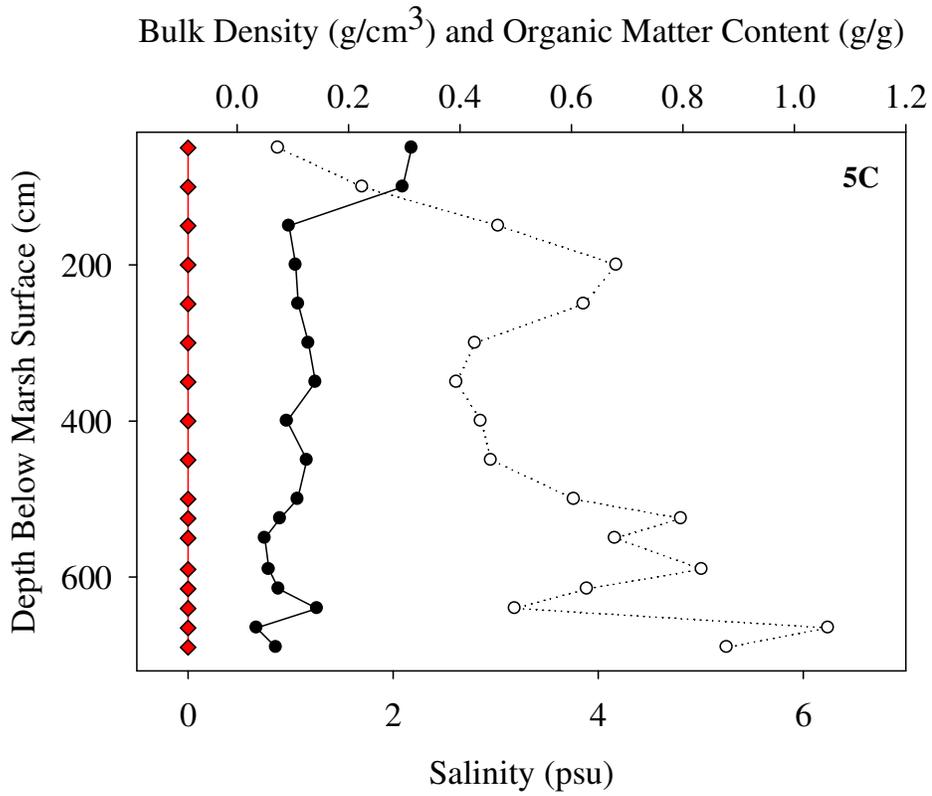
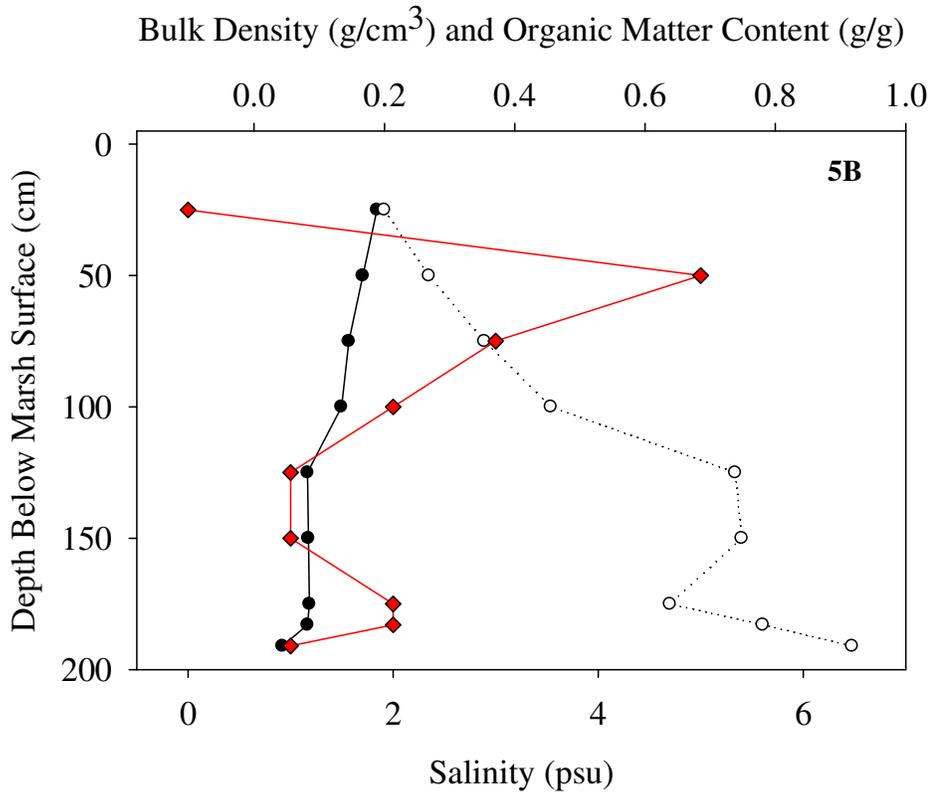


Figure AII.1 (1A-5C): High resolution geomorphic cores for each subsite along the Nanticoke estuary.

Appendix III:

Average Porewater Salinity Two-Weeks Post Treatment

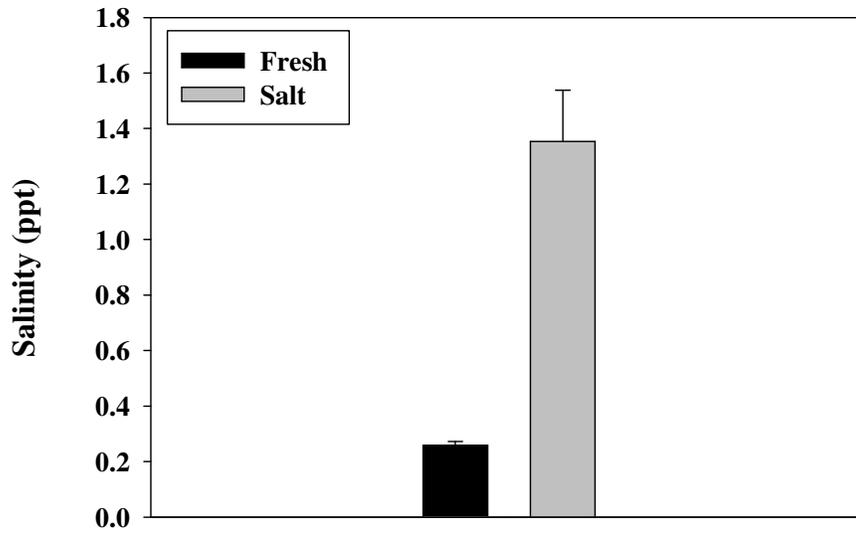


Figure AIII.1: Average porewater salinity at 30 cm depth two weeks post treatment application at Patuxent River salinity-addition experiment plots

Annual Average Accretion by Treatment

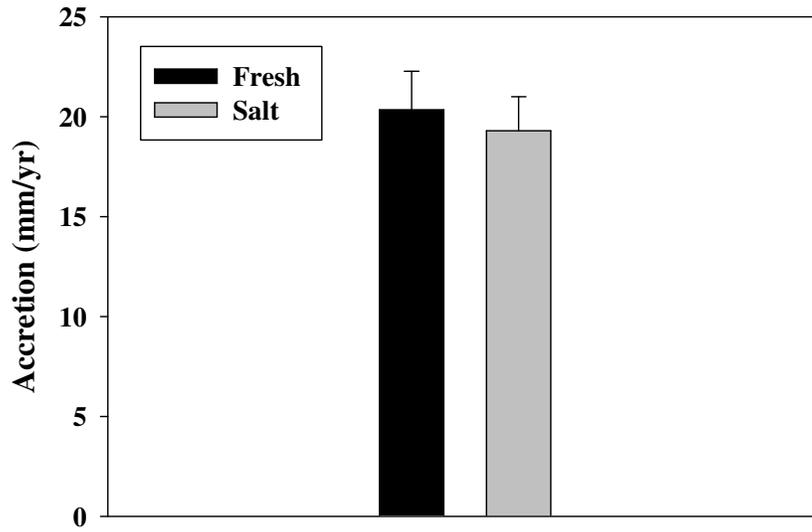
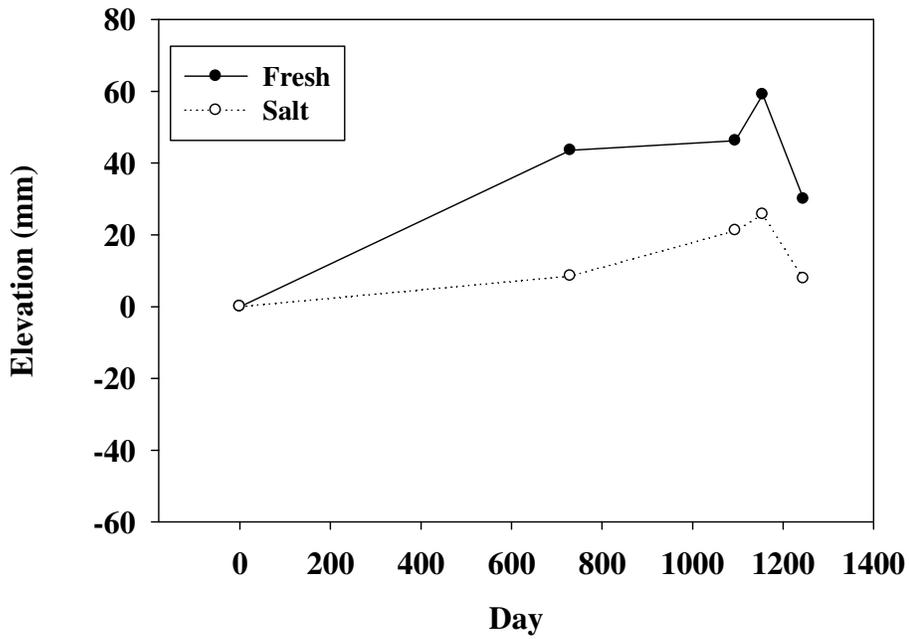


Figure AIII.2: Accretion rates (+ SE) in salt-treated and fresh-treated plots at Patuxent River salinity addition experiment. Accretion rates are averaged over June 2008-November 2011.

Elevation of Tidal Creek Sites (1,2,3) by Treatment



Elevation of Western Branch Sites (4,5)

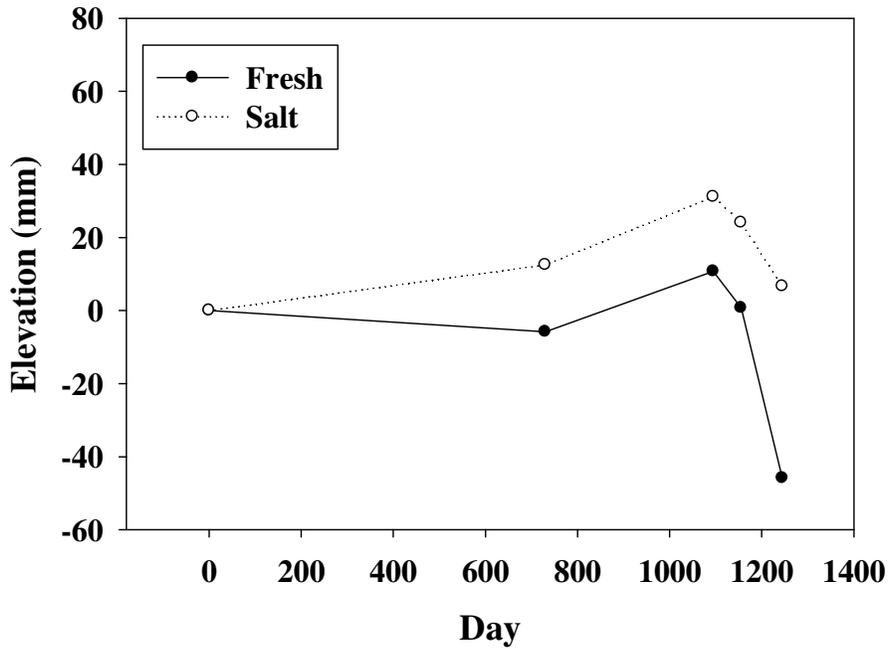


Figure AIII.3: Rates of elevation change averaged by position in the marsh, either adjacent to tidal creek or adjacent to Western Branch at Patuxent salinity addition experimental plots

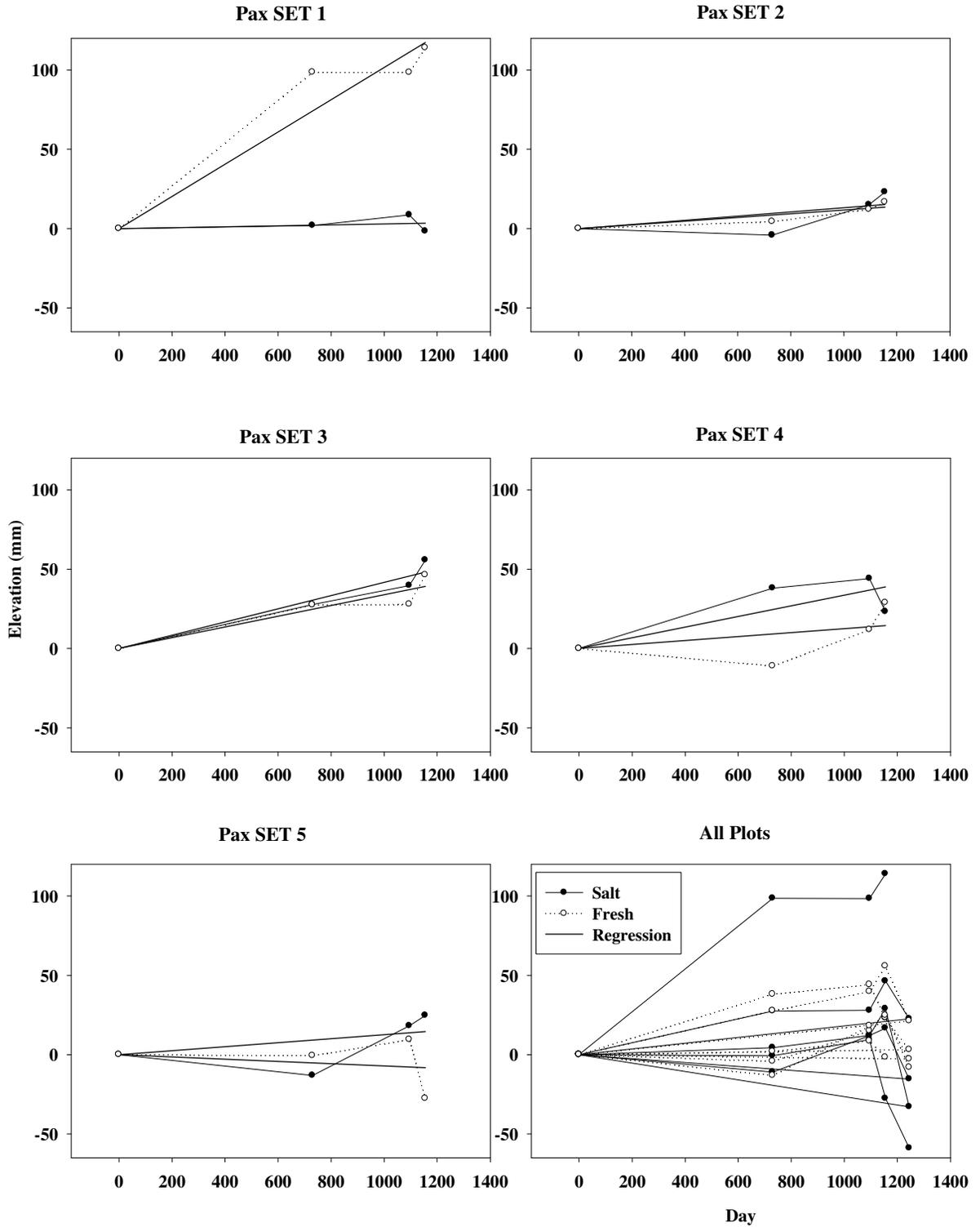


Figure AIII.4: (PaxSET 1-5 and combined): Rates of elevation change in salt-treated and fresh-treated plots at individual surface elevation table plots at Patuxent salinity addition experimental plots. Figures exclude final elevation sampling event, post Hurricane Irene.

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