

Impacts of sea level rise on tidal wetland extent and distribution

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Molly Magruder Mitchell

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APPROVAL PAGE

This dissertation is submitted in partial fulfillment of
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Molly Magruder Mitchell

Approved by the Committee, May 2018

Carlton Hershner, Ph.D.
Committee Chair / Advisor

Donna Marie Bilkovic, Ph.D.
Committee Chair / Advisor

Jian Shen, Ph.D.

Randolph M. Chambers, Ph.D.
College of William & Mary
Williamsburg, VA, USA

Larry P. Atkinson, Ph.D.
Old Dominion University
Norfolk, VA, USA

This work is dedicated to my son, Levi, whose constant interest and engagement in the natural world has been a great inspiration to me.

Table of Contents

ABSTRACT PAGE	xi
Chapter 1 Introduction	2
Chapter 2 Marsh persistence under sea-level rise is controlled by multiple, geologically variable stressors	4
Introduction	5
Methods.....	8
Inventory development	9
Tidal marsh digitization.....	10
Dataset corrections.....	12
Watershed characterization	13
Landuse and shoreline stabilization.....	15
Elevation	15
Statistical analysis	16
Results.....	17
Marsh change.....	17
Partition analysis.....	20
Discussion	26
Spatial differences in marsh response.....	27
Comparison of historic and modern marsh extents	31
Consequences of marsh change on ecosystem health.....	32
Conclusions	34
Chapter 3 Marsh vegetation as an indicator of ecosystem response to sea level rise.....	36
Introduction	37
Methods.....	41
Tidal Marsh Inventory.....	42
Marsh vegetation surveys.....	43
Comparison of historic and current plant communities.....	47
Species matrix driven changes.....	48

Low marsh plant community extent.....	49
Changes in community type	50
Results.....	50
Change in species matrix (York River, VA)	54
Changes in low marsh plant community extent	62
Change in community type (Chesapeake Bay, VA)	65
Discussion	68
Marsh vegetation as an indicator of inundation	68
Marsh vegetation as an indicator of salinization.....	70
Phragmites australis expansion in Chesapeake Bay, VA.....	71
Confounding factors for interpreting community shifts.....	73
The utility of patterns in community change.....	74
Conclusions	76
Chapter 4 Soil sulfides in transitioning headwater-brackish marsh systems	78
Introduction	79
Methods.....	81
Core collection	82
Sulfur analysis	85
Organic matter analysis	86
Data analysis	86
Results and Discussion	87
Organic matter as a moderator of sulfur	88
Linking vegetation to sediment sulfur content.....	90
Seasonal variability of sulfur	92
Conclusions	93
Chapter 5 Evolution of tidal marsh distribution under accelerating sea level rise.....	95
Introduction	96
Methods.....	99
Movement of the tidal frame across the landscape.....	100
Approximating time frames for the projections.....	102
Evaluating the impacts of development on tidal wetland movement	102
Salinity distribution	104

Results.....	104
Tidal elevation range as an indicator of tidal marsh extent	104
Projected changes in marsh area and distribution	105
Impervious surfaces in migration pathways	109
Marsh salinity distributions	112
Discussion	113
Interaction of sea level rise, accretion and erosion.....	114
Management implications	116
Chapter 6 Summary	119
References	120

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List of Tables

Table 2-1. Marsh forms found in the York River, VA.	10
Table 2-2. Summary of results by watershed.....	18
Table 2-3. Subwatershed characteristics from analysis	25
Table 3-1. Years of historic and current TMI surveys.....	45
Table 3-2. Plant communities as identified in the Tidal Marsh Inventories	46
Table 3-3. Changes in community over time identified in by CAP.....	59
Table 3-4. Change in community types in marshes surveyed in both years.....	66
Table 4-1. Site characteristics of the nine headwater-brackish marsh systems used in the study.....	83
Table 4-2. Dates of sampling for each site and transect location	85
Table 5-1. Scenarios used for analysis with their elevations and approximate time frames (based on Boon and Mitchell, 2015).....	101

List of figures

Figure 2-1. Mechanistic drivers of marsh change.....	8
Figure 2-2. York River estuary sub-watershed boundaries and numbers.....	14
Figure 2-3. Wormley Creek, York VA. Example of marsh fragmentation and loss due to shoreline stabilization.	19
Figure 2-4. Catlett Islands, Gloucester VA.....	21
Figure 2-5. Partition analysis results: (a) AIC table, (b) tree diagram and (c) map of leaf group position in the watershed.	22
Figure 2-6. The percent change in marsh area by distance.....	28
Figure 3-1. Marshes with communities surveyed in both time periods.....	44
Figure 3-2. Plant species found from the historic and current TMI surveys in the York River, VA.....	51
Figure 3-3. <i>Phragmites australis</i> community distribution.....	53
Figure 3-4. Changes in common York River plant community species.	55
Figure 3-5. CAP results from community type analysis.....	56
Figure 3-6. CAP results from salinity analysis	57
Figure 3-7. Mapped Community types identified from CAP analysis	60
Figure 3-8. Mapped salinity designations identified from CAP analysis	61
Figure 3-9. Changes in low marsh extent on the York River, VA.....	63
Figure 3-10. % organic matter and the change in low marsh extent.....	64
Figure 3-11. Sea level rise signals indicated by community change	66
Figure 4-1. . Map of sampling sites at three scales.....	84
Figure 4-2. Sulfur concentration at 5 sites along a headwater-salt marsh transect.....	87

Figure 4-3. Organic matter in cores by transect and depth	89
Figure 4-4. Mean sulfur concentration in marsh sites by season box plot.....	93
Figure 5-1. Comparison of modeled and surveyed marsh area (m ²) in 25 watersheds on the mainstem York River, VA.	105
Figure 5-2. Changes in area (m ²) within the tidal marsh elevation range over time.....	107
Figure 5-3. Changes in marsh area over time by locality	108
Figure 5-4. Total projected marsh area over time in two localities (a) Hampton (urban) and (b) Mathews (rural)	110
Figure 5-5. Tidal marsh and impervious surface conflicts by locality	111
Figure 5-6. Changes in marsh area by salinity type over time.....	112

ABSTRACT PAGE

Tidal marshes are a major ecological resource in Virginia and a driver of many estuarine functions. Therefore, the long term sustainability of tidal marsh ecosystems is a question of great interest in the research community. Sea level is rising at an unusually high rate in the Chesapeake Bay relative to most of the Atlantic coastline, putting Bay marshes at high risk from drowning and erosion. Sea level rise-driven salinity changes communities and alters ecosystem services. Understanding the patterns of change and the importance of different drivers of change is critical to tidal marsh sustainability.

The overarching goal of this research is to examine how changes in natural and anthropogenic factors interact to affect tidal wetland distribution, extent and plant composition with the intent of promoting coastal resiliency to sea level rise impacts through informed coastal management. I quantified changes in marsh extent over the past 40 years and related changes in marsh extent to sea level rise and other drivers of change. Then I examined shifts in plant community composition throughout the Chesapeake Bay, VA, looking for signals of increased inundation and salinity. In small headwater systems, I explored the utility of these changes in plant composition for predicting soil sulfur content (an early signal of salinity intrusion). These changes in marshes from the past 40 years were used to elucidate results from an elevation-based model of future marsh persistence under accelerating sea level rise.

Several lessons emerged from this dissertation:

1. Analyses of changes in tidal marsh extent and plant communities are complementary, clarifying vulnerabilities and prognosis under future conditions.
2. Human shoreline use (e.g., development, shoreline hardening, boating activity) can dominate physical processes to alter the marsh response to sea level rise.
3. Defining sediment availability for a given marsh may not be sufficient to determine its potential for expansion or persistence under sea level rise.
5. Marsh plant communities can be an early signal of change, showing shifts in inundation frequency before there is any change in marsh extent.
6. Tidal marshes will continue to decline over the next 100 years. However, most of the loss will be in low salinity, riverine marshes. Some high salinity, Bayfront marshes will expand if the land they need to migrate is preserved.
7. Tidal marsh response to sea level rise has, and will continue to, vary by marsh form, geologic setting, location in the estuary, and surrounding land use decisions.
9. Targeted land use decisions coupled with active restoration may help minimize future marsh loss.

Impacts of sea level rise on tidal wetland extent and distribution

Chapter 1 Introduction

Coastal marsh loss is a significant issue throughout the United States. Tidal marshes are highly productive ecosystems that provide a myriad of services to the human and aquatic system. Services include modification of wave climates to create habitat opportunities (Bruno 2000) and enhance shoreline stabilization (Shepard et al. 2011), provision of refuge habitat translating to enhanced fisheries (Minello et al. 2012), modifiers of nutrient loads from upland (Valiela & Cole 2002) and tidal (Deegan et al. 2007) sources, and a long term carbon sink (Chmura 2003, Bridgham et al. 2006). Their loss has the capacity to dramatically change coastal and estuarine functions and potentially impact global cycles (Coverdale et al 2014, Chmura 2013).

Estimating changes in tidal marsh on a large scale requires an extensive past dataset which can be compared to current marsh distributions and communities. The Tidal Marsh Inventory (TMI; CCRM, VIMS) is a large scale survey of marsh extent and plant community composition covering every tidal marsh in Virginia. The field work for the original inventories was predominately done throughout the 1970s. Recently, this survey has been repeated for large portions of the Virginia coast (2010-present), providing a unique opportunity to look at changes in marsh distribution and community composition.

The range of time between the original and new tidal marsh surveys corresponds to acceleration in the rate of sea level rise in the Mid-Atlantic (Sallenger et al., 2012 Boon 2012, Ezer and Corlett 2012). Relative sea level rise in the Chesapeake Bay since 1970 has averaged (across the Bay) around 5 mm/year (Ezer and Atkinson 2015, Boon and Mitchell 2015), which is commensurate with the maximum rate of accretion theoretically possible for marshes (Morris et al. 2016). The capacity of marshes to adjust to sea level

rise diminishes with high rates of sea level rise, making it likely that there will be measurable signals of marsh loss and community change between the two TMIs.

Sea level rise impacts the tidal marshes in two primary ways:

1) Sea level rise increases tidal inundation frequency, tidal flooding extent and shoreline erosion (due to increased water depth). *Changes in inundation are reflected in the marsh extent and position on the landscape and the plant community composition.*

2) Sea level rise changes the salinity distribution in the estuary, pushing brackish waters up into previously freshwater systems. *Changes in salinity are reflected in the plant community composition.*

In this dissertation, I examine the effects of increased inundation (water depth) and shifting salinity regimes on tidal marshes in the Chesapeake Bay. I quantified changes in tidal marsh extent over the past 40 years and related changes in marsh extent to sea level rise and other drivers of change (**Chapter 2**). Then I examined shifts in tidal marsh plant community composition throughout the Chesapeake Bay, VA, looking for signals of increased inundation and salinity (**Chapter 3**). In small headwater systems, I explored the utility of these changes in plant composition for predicting soil sulfur content (an early signal of salinity intrusion) (**Chapter 4**). These changes in marshes from the past 40 years were used to elucidate results from an elevation-based model of future marsh persistence under accelerating sea level rise (**Chapter 5**). Important lessons from each chapter are synthesized in the summary (**Chapter 6**).

Chapter 2 Marsh persistence under sea-level rise is controlled by multiple, geologically variable stressors

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Abstract

Marshes contribute to habitat and water quality in estuaries and coastal bays. Their importance to continued ecosystem functioning has led to concerns about their persistence. Concurrent with sea-level rise, marshes are eroding and appear to be disappearing through ponding in their interior; in addition, in many places they are being replaced with shoreline stabilization structures. We examined the changes in marsh extent and community over the past 40 years within a subestuary of the largest estuary in the United States, Chesapeake Bay, to better understand the effects of sea-level rise and human pressure on marsh coverage.

Approximately 30 years ago, an inventory of York River estuary marshes documented the historic extent of marshes. Marshes were re-surveyed in 2010 to examine shifts in tidal marsh extent and distribution. Marsh change varied spatially along the estuary, with watershed changes between a 32% loss and an 11% gain in marsh area. Loss of marsh was apparent in the high energy sections of the estuary while there was marsh gain near in the upper/riverine section of the estuary and where forested hummocks on marsh islands have become inundated. Marshes persisted with little change in the small tributary creeks, except in the creeks dominated by fringing marshes and high shoreline

development. Differential resilience to sea-level rise and spatial variations in erosion, sediment supply and human development have resulted in spatially variable changes in specific marsh extents; which are predicted to lead to a redistribution of marshes along the estuarine gradient, with consequences for the unique communities associated with them.

Key Words: Chesapeake Bay; climate change; coastal resilience; marsh change; salt marsh; sea-level rise; tidal wetlands

Introduction

Coastal marsh loss is a significant issue globally (Barbier et al. 2011). Tidal marshes are highly productive ecosystems that provide a myriad of services to the human and aquatic system. Services include modification of wave climates to create habitat opportunities (Bruno 2000) and enhance shoreline stabilization (Shepard et al. 2011), provision of refuge habitat translating to enhanced fisheries (Minello et al. 2012), modifiers of nutrient loads from upland (Valiela & Cole 2002) and tidal (Deegan et al. 2007) sources, and a long term carbon sink (Chmura 2003, Bridgham et al. 2006). Their loss has the capacity to dramatically change coastal and estuarine functions and potentially impact global biogeochemical cycles (Coverdale et al 2014, Chmura 2013). In estuarine systems, their role in mediating water quality, both through sediment removal from tidal waters and precipitation-induced runoff and through the provision of habitat for filter feeding organisms, such as mussels, directly links the abundance of marsh systems to the overall health of the estuary.

Marsh loss has been accelerating over the past century with a total loss greater than 50% of the original tidal salt marsh habitat, due to in part to human activity (Kennish 2001). Concurrently, sea-level rise has been changing tidal regimes, wave energy and other physical characteristics that help define marsh extent and placement on the shoreline. Sea-level rise has been cited as a cause of on-going marsh loss in many estuaries, including the largest estuary in the United States, Chesapeake Bay (e.g., Stevenson et al. 1985, Wray et al. 1995, Beckett et al. 2016) and a potentially increasing threat in the future. Relative sea-level rise in the Chesapeake Bay since 1970 has averaged (across the Bay) around 5 mm/year (Ezer and Atkinson 2015, Boon and Mitchell 2015), which is commiserate with the maximum rate of accretion theoretically possible for marshes (Morris et al. 2016), suggesting that marshes are becoming stressed by increased inundation. Research on the response of marshes to sea-level rise has typically focused on a limited number of discrete marshes, leading to conflicting results, with some studies suggesting that marshes are expanding under sea-level rise (Kirwan et al. 2016) while other suggest marshes are fragmenting and losing extent (Beckett et al. 2016). Both of these processes are likely occurring in the Chesapeake Bay, but the importance of each and an understanding of the role that location, physical changes and human activity play in these changes requires examination of marsh change on an estuarine scale.

Estimating changes in tidal marsh on an estuarine scale requires an extensive historic dataset which can be compared to current marsh distributions and communities. The Tidal Marsh Inventory (CCRM, VIMS 1992) is extensive survey of marsh extent and plant community composition covering every tidal marsh in Virginia. The field work for

the original inventories was predominately done throughout the 1970s. Recently, this survey has been repeated for large portions of the Virginia coast (2010-present), providing a unique opportunity to look at changes in marsh distribution and community composition. The range of time between the original and new tidal marsh surveys corresponds to acceleration in the rate of sea-level rise in the Mid-Atlantic (Sallenger et al., 2012 Boon 2012, Ezer et al. 2013). The capacity of marshes to adjust to sea-level rise diminishes with high rates of sea-level rise, making it likely that there will be measurable signals of marsh loss and community change between the two tidal marsh inventories.

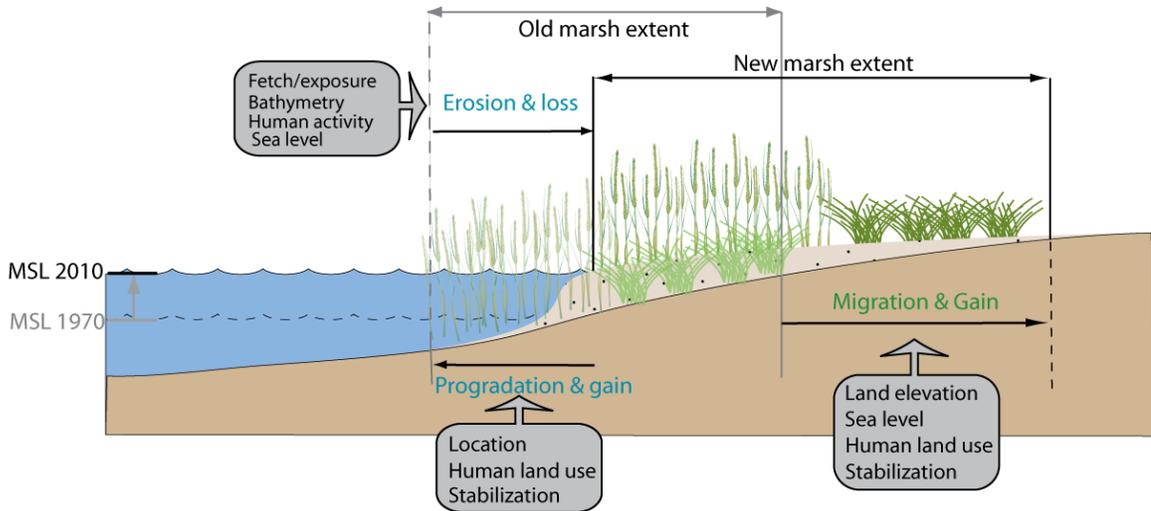
The overarching goal of this research is to examine how changes in natural and anthropogenic factors interact to affect tidal wetland resilience to sea-level rise and how variations in this response may affect marsh extent and distribution. Marshes change through three basic mechanisms: migration, erosion and progradation (Figure 1). The rate at which these mechanism drive change is determined by a variety of factors:

Migration rates are tightly tied to sea-level rise, but also respond to human activities, such as shoreline hardening. Erosion rates are driven by wave energy (a function of fetch, nearshore bathymetry, boating activity or adjacent shoreline stabilization), which increase with sea-level rise due to increased nearshore water depths (Leatherman et al., 2000).

Progradation relies on sediment supply, and so is tied to human landuse and shoreline stabilization, which can reduce or exacerbate sediment supply (depending on the activity). We hypothesized that while the overall extent of marshes is declining, spatial

variations in sea-level rise, erosion, sediment supply and human development will result in spatially variable changes in specific marsh extents over the past 30 years.

Figure 2-1. Mechanistic drivers of marsh change. Mechanisms in grey boxes exacerbate or mitigate the effects of marsh change drivers.



Methods

The York River Estuary, Virginia, USA is the target site for this study. It is one of five major tributary systems in Chesapeake Bay and generally representative of conditions encountered throughout the Bay and similar estuaries (Reay and Moore 2009). The York River estuary is a brackish system approximately 64 km long branching into two smaller tributaries; the Mattaponi and Pamunkey rivers. It possesses a wide range of salinities from approximately 20 ppt near the mouth of the river, to 0 ppt several kilometers upriver of the branch. The estuary has a primary turbidity maximum near the branching point and a secondary turbidity maximum approximately 30 km from the mouth of the estuary (Lin and Kuo 2001). Mean tidal range near the mouth of the York River is 0.7 m and increases to 1.1 m in the upper reaches of the Mattaponi River (Sisson et al. 1997). The estuary

supports a wide range of habitats, including freshwater swamps, tidal freshwater marshes, and salt marshes, and the watershed is dominated by forested (61%) and agricultural (21%) land use, with developed areas near the mouth of the estuary (Reay 2009).

Subsidence varies along the length of the estuary, from approximately 2.8 mm/yr at the mouth of the estuary to approximately 3.8 mm/yr at the branching point (Eggleston and Pope 2013). Marsh cores along the mainstem of the York River show top layer soils to be silt and clay with organic inclusions of *S. alterniflora* (Finkelstein and Hardaway 1988).

Inventory development

The Tidal Marsh Inventory (TMI; CCRM, VIMS 1992) is a geospatial survey of all tidal marshes in Virginia, including their location, extent and plant community; the survey has been done twice, approximately 30 years apart. The surveys involved digitization of marsh extents and locations from maps and aerial imagery. The digitization was field-verified for all mainstem marshes and most creek marshes during the collection of plant community data. Field verification in both surveys involved sending a boat along the entire shoreline of the York River estuary. Every marsh was compared to the digital coverage and marshes were added or altered where necessary. The addition of very narrow (>5m width) fringe marshes, hidden on the aerial photography by overhanging trees was the most common change in both time periods. Marshes were also categorized by their form (i.e., fringe, extensive, embayed, marsh island; see Table 1) following strict definitions established by survey scientists.

Table 2-1. Marsh forms found in the York River, VA.

Forms	Diagram	Characteristics	Water edge:area	Typical wave exposure
Extensive (attached)		large, flat marshes with extensive marsh channels; attached to land on one edge	Small (not including creek edges)	Frequently in high exposure areas, unless riverine
Extensive (marsh island)		large, flat marshes with extensive marsh channels; islands	Small (not including creek edges)	Frequently in high exposure areas
Embayed		v-shaped marshes that form along the edges and tops of creeks; some marsh channels	Moderate	Frequently in sheltered area
Fringing		Long, narrow marshes that form along river and creek edges; few marsh channels	High	Found equally in high energy and sheltered areas

In the York River estuary, the original survey was digitized from USGS topographic maps that were originally mapped in the late 1950s to early 1960s. Field verification was done between 1974 and 1987 (depending on the county), making it difficult to assign a specific year to the data. The second survey, digitized from 2009 aerial imagery (VBMP) was field-verified in 2010.

Tidal marsh digitization

The original survey was digitized at 1:24,000 resolution with a reported horizontal accuracy of +/-12.2 meters. Topographic maps printed on stable based mylar were placed on Numonics 2200 series digitizing tablets and marsh boundaries were hand digitized

using precision cursors. Tablets were interfaced with SUN Unix workstations running the ESRI software ArcInfo®. Mylar maps were geo-registered on the tablet using a quality assurance digitizing standard of RMS = 0.002 inches or better. Other program and computer based standards were put in place to insure accuracy of the digital product, including a node snap tolerance (<0.05 inches) and fuzzy tolerance (0.001 inches = 1.0 meters in UTM), which are procedural standards that control digitizing accuracy and final product quality (Berman et al.1993).

In the recent TMI survey, tidal marshes were digitized off digital high resolution (6-inch) color infrared aerial photography collected in 2009 (VBMP) at 1:1,000 resolution. Heads-up digitizing (capturing vector objects directly from the computer screen using a mouse or cursor) was performed to develop the boundary delineation for current wetland distribution. This method is considered more accurate than traditional tablet digitizing since the user can resolve more features using zoom functions. Photo interpretation techniques were used to identify wetland objects on the screen in ArcMap versions 9.3 and 10.0. Ancillary data sets including the VA Shoreline Inventory (Berman et al. 2013; 2014a, 2014b, 2014c) and National Wetlands Inventory (NWI) were used to help identify narrow fringe marshes masked by tree canopy or visual scale. When digitizing was complete the file was smoothed to improve the cartographic quality. The smoothing algorithm used was PAEK (Polynomial Approximation with Exponential Kernel) using a smoothing tolerance of 5 meters.

Quality control and assurance was by independent staff scientist review and during field work. During field observations, marsh boundaries were added or visually adjusted on rectified image base maps. Digital corrections were made in the lab when community composition data were added to the attribute files. Consistency in identifying and digitizing the marsh boundary was tested using repetitive sampling techniques. Six marshes of varying size and complexity were selected and each digitized three times. Each digitized area was compared to the mean; the average difference in calculation of area for each sample was +/-0.0003 acres.

Dataset corrections

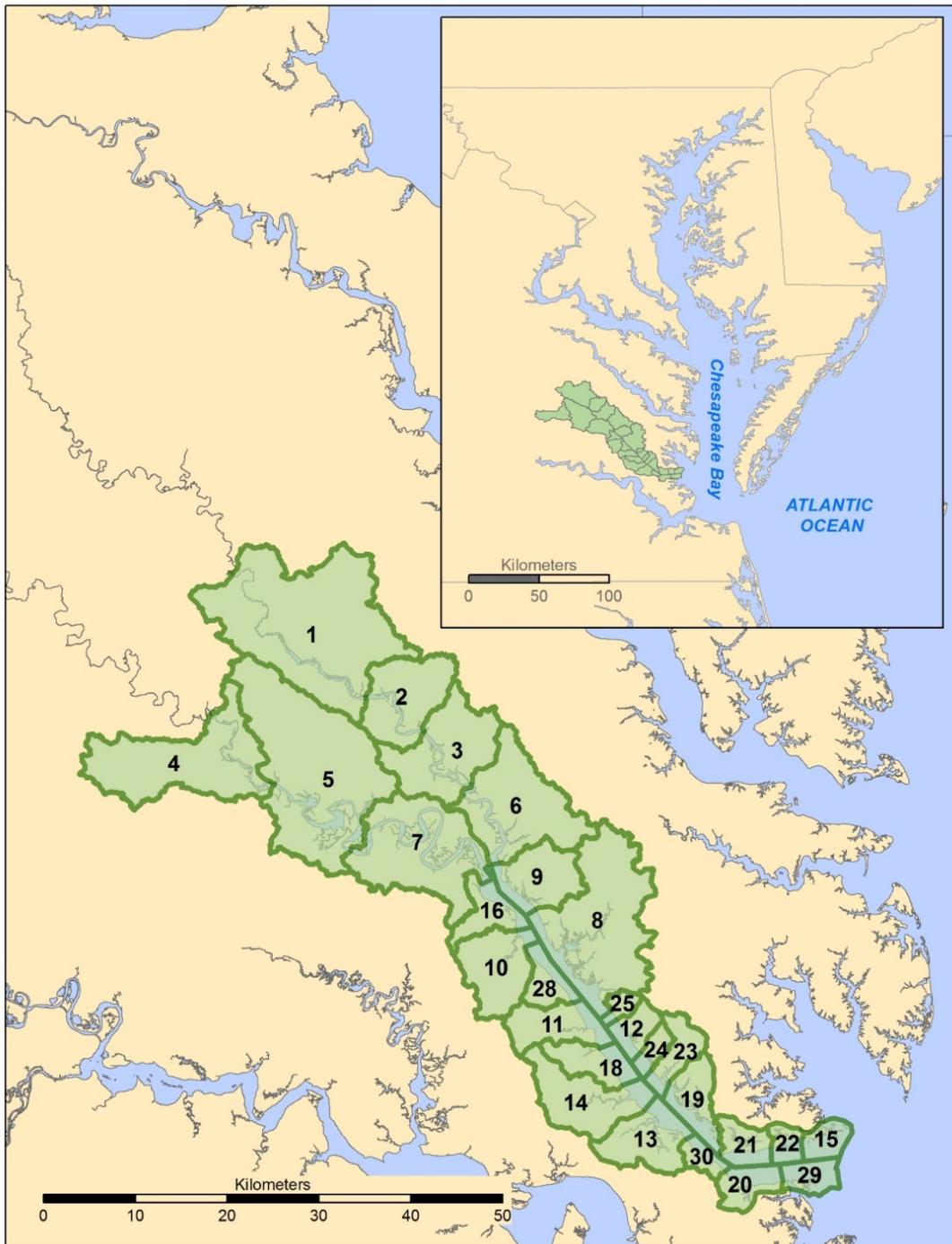
Examination of the old TMI against current elevation data (CoNED TBDEM 2016) showed that there were errors in the landward extent of some marshes, particularly the fringe marshes, leading to overestimation of marsh extent in the original survey. These errors were due to the resolution at which digitization occurred in the original survey and the fact that many fringing marshes were discovered during the field-verification whose exact widths were difficult to determine. To minimize these errors, marshes in the original survey were clipped to an elevation (1m NAVD 88) representing the theoretical maximum elevation of tidal wetlands in 1970. This correction removed 5,988,795 m² of wetlands that were clearly digitized into upland areas. Results were verified against aerial photos from the 1960s where available in the York River estuary.

Watershed characterization

High spatial variability in estuarine characteristics makes it difficult to see patterns in marsh change. Therefore, the York River estuary was divided into subwatersheds based on the broader designations of the NWBD (National Watershed Boundary Database, 2008), split into smaller subwatershed using elevation contours (Figure 2). This kept marshes which would reasonably be responding to similar landuse and water quality measures in a single watershed (e.g., creek marshes and mainstem marshes that were immediately adjacent to the creek mouth, tending to extend further downriver than up), while still minimizing the variability in estuarine characteristics.

Subwatersheds were characterized by location and marsh form. Location of the watershed was measured as the distance from the mouth of the estuary, up the centerline of the estuary, to the center of each subwatershed, using the Measure tool in ESRI ArcMap (10.2). The continuous distances (km) were used for the analysis; however for ease of discussion, marshes are referred to by three location groups with similar hydrodynamic characteristics in the results and discussion section: low estuary (high energy <20km from mouth), mid-estuary (moderate energy in mainstem, low energy in creeks, >20km and <58km from mouth), and upper/riverine (low energy, river-dominated, >58km from mouth).

Figure 2-2. York River estuary sub-watershed boundaries and numbers.



Landuse and shoreline stabilization

Landuse within a 1500m buffer of the shoreline was obtained from the VGIN 1m Land Cover dataset (2016). Landuse was grouped into 3 categories based on similar landcover types: 1. Developed (included landcovers: Impervious (extracted), Impervious (local datasets), Barren), 2. Natural (included landcovers: Forest, Tree, Scrub/shrub, NWI/other) and 3. Agriculture (included landcovers: Harvested/disturbed, Pasture, Cropland). Each category was summed by watershed and percent cover was calculated for each. Shoreline stabilization lengths were obtained from the Shoreline Inventory (Berman et al., 2013, 2014a, 2014b, 2014c). There are multiple categories of shoreline stabilization, but only bulkhead and riprap (“hardening” henceforth) were used since these structures disconnect the tidal marsh from the upland, reducing both function and the ability of the marsh to migrate (Bilkovic and Mitchell 2017). Length of hardening was summed by subwatershed.

Elevation

Low elevations areas adjacent to tidal marshes enhance tidal marsh migration. Areas with very low relief can allow migration to proceed at a pace equal to or greater than marsh erosion, leading to marsh expansion. To see the importance of elevation as a driver of marsh change, a metric of elevation (henceforth, %low) was developed.

Elevation data was obtained from a seamless lidar-derived digital topographic and point-derived bathymetric elevation model (CoNED TBDEM, 2016). Elevations below 1m NAVD88 (tidal marsh elevations) were discarded due to concerns about the accuracy of these elevations in salt marshes (Hladik and Albe, 2012; Wang et al., 2009). Elevations

above 3m NAVD88 were also discarded since they represent lands which are unlikely to be marsh at any time between the start of the survey and 2100 (based on the High scenario projection of mean sea-level; Sweet et al. 2017). Elevations between 1m and 3m are transitional areas with the potential to become tidal marshes by 2100, therefore critical habitat for marsh migration. Elevations between 1m and 3m NAVD88 within a 1500m buffer from the creek were extracted from the DEM. Within each watershed, the percent of land represented by this range in elevation (% low) was calculated for the extracted data.

Statistical analysis

All statistical analyses were done in JMP 10 (SAS). A recursive partition analysis using a decision tree was used to classify percent marsh change according to sub-watershed characteristics: location of the watershed in the estuary, landuse (% developed, % agriculture, % natural), marsh form (%fringing, %embayed, %extensive), shoreline hardening (m) along watershed shorelines, and elevation (%low). Recursive partitioning decision trees are a non-parametric, multivariate, classification and regression tree-type analysis. Decision trees explain the variation in a response variable (in our case, %change in marsh) as a function of multiple explanatory variables, can handle variables with non-linear relationships and are not affected by monotonic transformations (De'ath and Fabricuis, 2000). KFold validation (KFold = 10) was used to select the final model (JMP 10). This process reduces overfitting of the model; however, overfitting of the tree was unlikely given the low complexity of the resulting model (Olden et al., 2008). Splits in continuous data were made on the explanatory variable with the greatest LogWorth at

each step in the tree. Automatic splitting was used, where splitting continues until the KFold validation R^2 exceeds the values that next 10 splits would obtain (JMP 10).

A weakness of decision trees is that correlations between independent variables can complicate the analysis. We know some of our response variables are necessarily related; therefore, we performed a correlation analysis on our explanatory variables to elucidate potentially important variables not explicitly identified in the tree.

Results

Marsh change

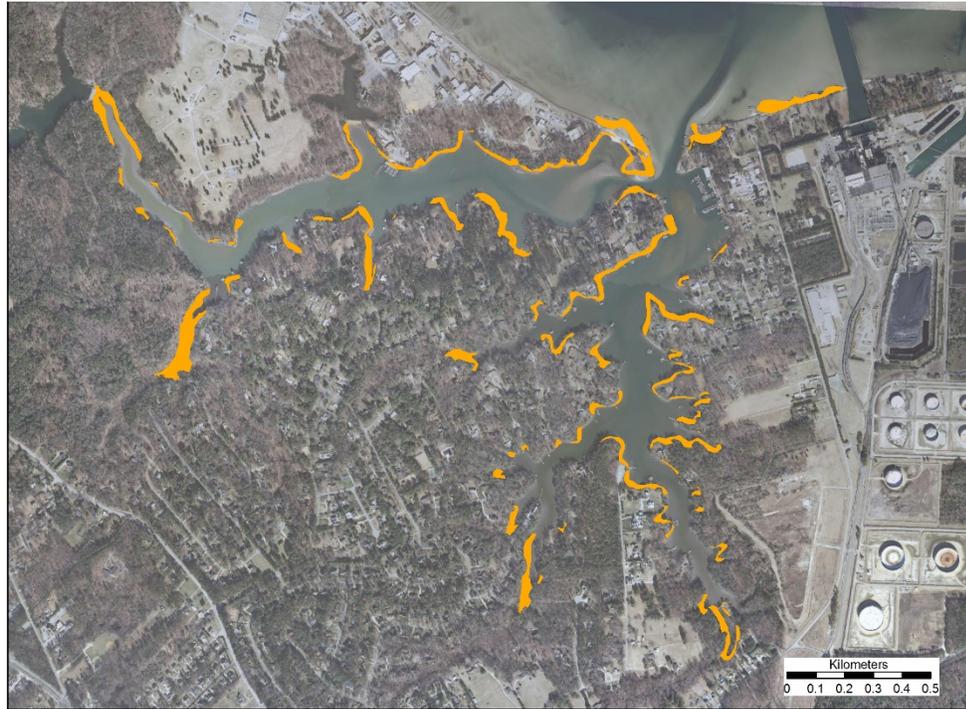
Between the early 1970s and 2009, sea-level rose approximately 20 cm in the York River estuary while concurrent overall marsh change was a loss of approximately 2,187,000 m², or ~2.7% of marsh area from the original survey. Marsh change varied by watershed, with some watersheds showing an increase in marsh area while others showed losses (Table 2). Examination of the marsh change and aerial photography from both time periods indicated that most of the marsh loss is due to edge erosion (reduction in marsh width), with minimal loss of linear marsh extent (reduction in marsh length or marsh loss). However, in subwatersheds 19 and 20 (which are predominantly fringing marsh systems that are developed with extensive shoreline stabilization), there is total loss of multiple marshes. This has resulted in both a loss of area and fragmentation of the marsh system (Figure 3, watershed 19).

Table 2-2. Summary of results by watershed

Watershed	Location	Approximate Distance (km)	Old TMI marsh extent (m²)	New TMI marsh extent (m²)	% Change in marsh extent	Leaf group
1	Upper	83	3204315	3296848	2.89	1
2	Upper	74	1842669	2046628	11.07	1
3	Upper	58	5984716	5957608	-0.45	1
4	Upper	83	1398541	1318269	-5.74	1
5	Upper	74	6063007	6022791	-0.66	1
6	Upper	58	7669161	7722836	0.70	1
7	Upper	58	21740124	21541880	-0.91	1
8	Mid	40	9976593	9904033	-0.73	2
9	Mid	51	2733013	2702113	-1.13	2
10	Mid	45	1879906	1889505	0.51	3
11	Mid	34	2112664	2025131	-4.14	3
12	Mid	30	718054	672257	-6.38	3
13	Low	19	1560796	1672170	7.14	3
14	Mid	24	2483726	2317941	-6.67	4
15	Low	2	994826	744878	-25.12	2
16	Mid	51	2295306	2041473	-11.06	2
18	Mid	29	925482	698189	-24.56	4
19	Low	18	2000614	1466219	-26.71	4
20	Low	10	140239	115403	-17.71	4
21	Low	10	150813	116243	-22.92	4
22	Low	5	595644	406073	-31.83	4
23	Mid	23	815024	706616	-13.30	2
24	Mid	26	296018	254917	-13.88	2
25	Mid	3	303942	302270	-0.55	3
28	Mid	40	725091	649417	-10.44	3
29	Low	4	1187314	1013941	-14.60	4
30	Low	15	155879	161300	3.48	4

Figure 2-3. Wormley Creek, York VA. Example of marsh fragmentation and loss due to shoreline stabilization. (a) Old TMI marsh distribution in orange, on an aerial photo from 2009. (b) New TMI marsh distribution in orange, on an aerial photo from 2009.

a)



b)



Examination of the marsh change and aerial photography from both time periods indicated that most of the marsh gain is due to landward migration, frequently into previously forested hummocks (Figure 4). A couple of subwatersheds in the upper/riverine section of the estuary showed slight marsh expansion through progradation.

Three subwatersheds in the mid-estuary (10, 13, and 30) showed gains in marsh area between the two surveys that were due to apparent upriver migration of tidal influence (i.e., in the original survey the marshes were non-tidal; in the current survey they were tidal). In all cases, the expansion is linked to a barrier (bridge/culvert) and could have been caused by increased culvert size between the two surveys, allowing an expansion of the tidal influence. Unfortunately, these changes could not be verified by aerial photography and therefore the gain shown in these sub-watersheds should be considered uncertain.

Partition analysis

The partition analysis split the subwatersheds into 4 groups (Figure 5) based on (in order of split): development (split at 15%); approximate distance from the mouth of the estuary (split at 58 km); percentage of embayed marshes (split at 61%). R^2 values increased with each split, and by the last split there were no likely candidates for splitting in any of the 4 groups.

Figure 2-4. Catlett Islands, Gloucester VA. Example of marsh migration into forested hummocks. (a) Aerial photo of the site from 1978, showing a large forested marsh hummock. (b) Aerial photo of the site from 2009, showing most of the hummock has converted to marsh.

a)



b)

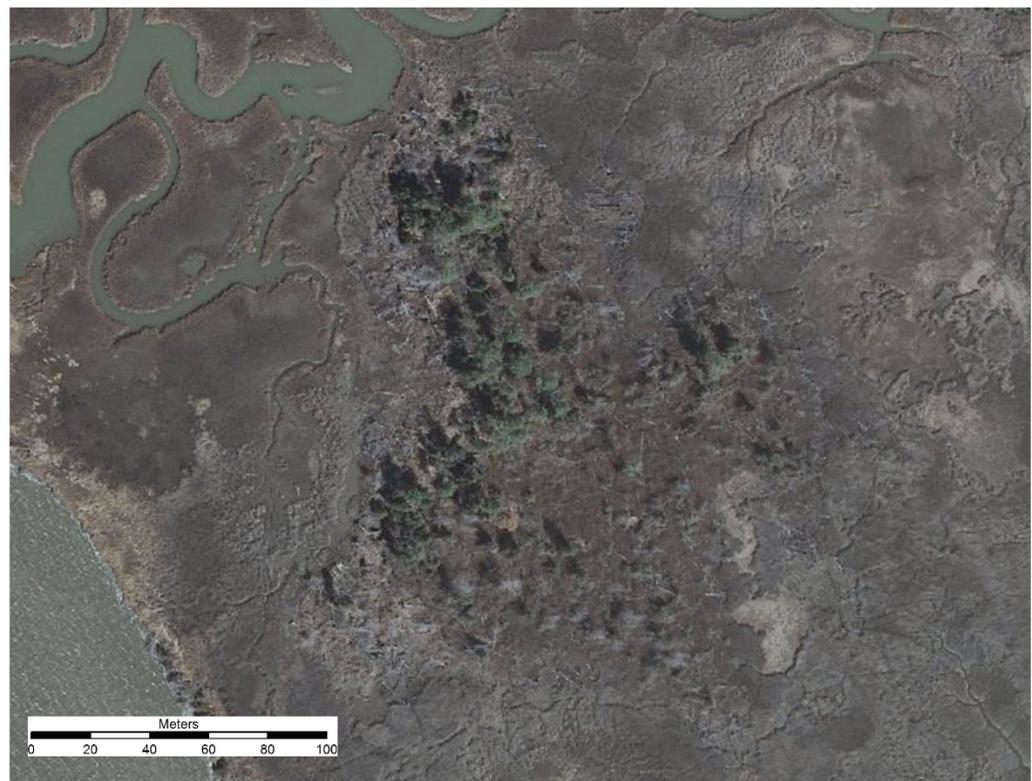
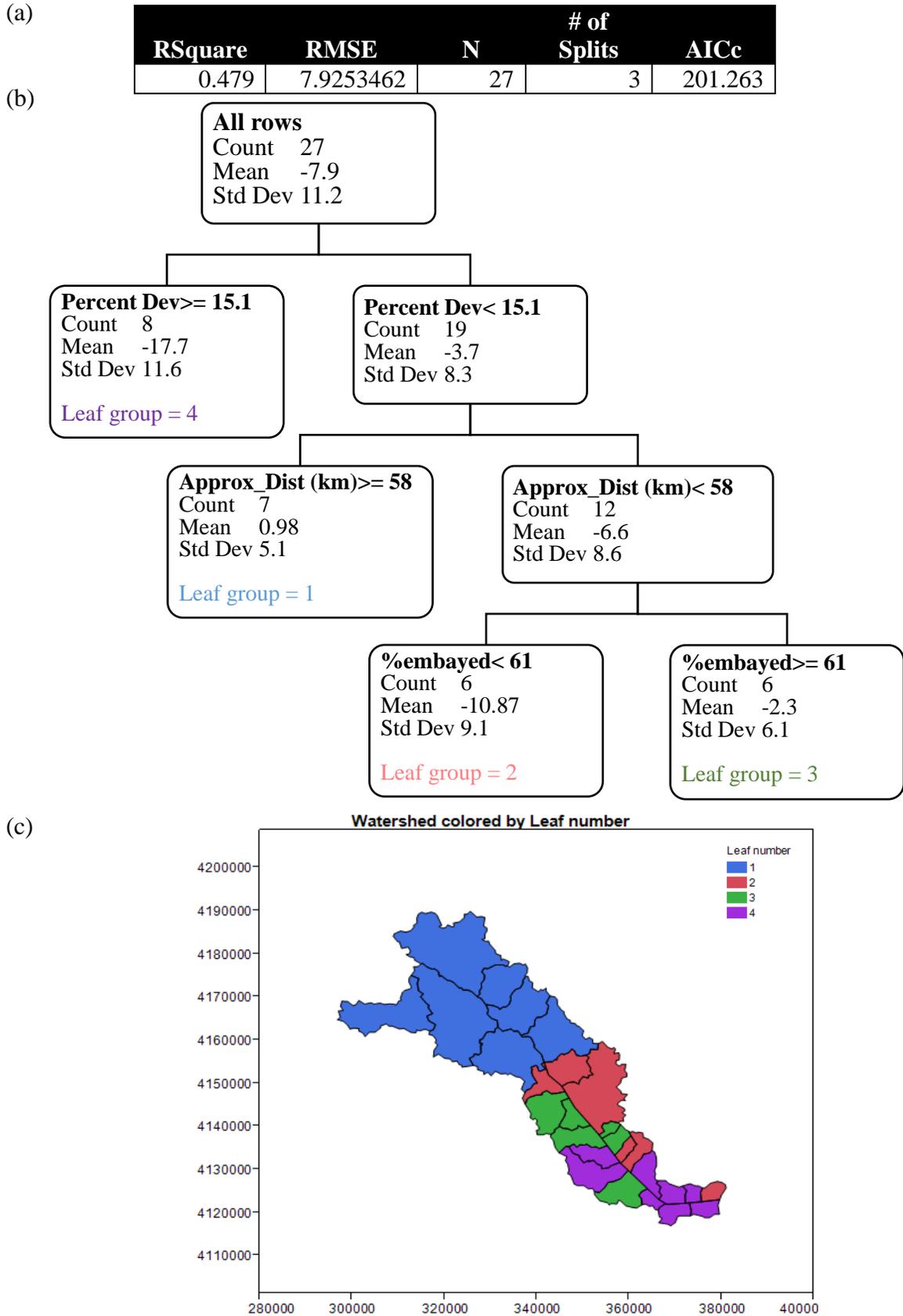


Figure 2-5. Partition analysis results: (a) AIC table, (b) tree diagram and (c) map of leaf group position in the watershed.



Development was the most important predictor of marsh change in the estuary, with areas of higher development having a higher percent loss of marsh. However, land use within a 1500m buffer of the shoreline was predominantly natural (mean % Natural landuse = 75%), with only two subwatersheds having greater than 40% developed land (Appendix 1). % Developed landuse was somewhat negatively correlated with % Natural landuse ($r^2 = -0.62$), so it should be considered that the tree could be splitting on a balance between developed and natural lands within the subwatersheds, but the evidence for this is not strong.

Although a few subwatersheds had high agricultural levels, it was never the dominant landuse in a subwatershed and plays a small role overall in the estuary (mean % Agricultural landuse = 12%). It was only weakly correlated with % Developed landuse ($r^2 = -0.30$) and therefore is not a discriminant factor in the York River estuary.

Interestingly, % Developed lands were highly positively correlated with length of riprap and bulkhead ($r^2 = 0.85$) and % fringe marsh ($r^2 = 0.77$); suggesting these might be important predictors of marsh loss that were not identified in the decision tree.

Shoreline hardening was highest in subwatersheds in the lower section of the estuary, and minimal throughout the rest of the estuary. Three subwatersheds on the southside of the mid-estuary (10, 13, and 30) had no shoreline hardening at all. These are the same subwatersheds where there appeared to be marsh gain through the conversion of upriver migration of tidal influence.

In areas of low development, the distance upstream was the most important factor predicting marsh change. In the low and mid-estuary, there was marsh loss on average, while in the upper estuary there was an average small increase in marsh acreage. Distance upstream was positively correlated with % Agricultural landuse ($r^2 = 0.63$) and negatively correlated with % Developed landuse ($r^2 = -0.59$) and % low ($r^2 = -0.53$). All other correlations were weak ($r^2 < 0.40$).

Land elevation within a 1500m buffer of the shoreline showed a general pattern of lower elevations behind the marshes in the low estuary, with higher elevations on the south side of the river and in the mid-estuary and upper/riverine sections on both sides of the river (Table 3). The analysis does not provide strong evidence for our expectation that marsh gains would be highest where there are the most opportunities for landward migration (highest % low). However, 1) there were gains in some of the low elevation-backed marshes, they were just outweighed by the losses and 2) the high elevation lands on the south side of the estuary include a number of eroding bluffs (Berman et al. 2013, 2014b) which may contribute sediment supply essential for marsh persistence.

Table 2-3. Subwatershed characteristics from analysis

Watershed	% Developed landuse	% Agriculture landuse	% Natural landuse	% Fringing	% Extensive	% Embayed	% Low elevation land	Riprap & Bulkhead length (m)
1	6	30	65	9	78	13	6	1545
2	4	12	84	7	65	27	5	512
3	4	20	77	0	89	10	9	1210
4	4	44	51	5	21	74	6	521
5	4	23	73	11	68	21	7	708
6	11	18	71	1	75	24	16	942
7	8	16	76	1	92	7	11	3068
8	5	13	82	1	71	29	24	2882
9	12	11	77	2	76	22	33	1396
10	2	4	94	6	0	94	7	0
11	9	3	88	2	37	61	7	627
12	6	17	77	1	10	88	11	2140
13	13	1	86	11	24	65	4	0
14	16	1	83	3	74	23	7	759
15	8	0	92	3	97	0	100	630
16	8	10	82	5	65	30	17	1316
18	15	2	83	13	0	87	9	1433
19	29	16	54	10	75	15	7	4082
20	42	6	52	70	0	30	16	8581
21	50	6	45	53	2	45	54	13852
22	19	11	70	26	19	55	100	2868
23	7	22	71	3	45	53	8	890
24	12	28	60	37	4	59	6	2326
25	11	14	74	1	0	99	4	698
28	5	2	93	21	0	79	6	787
29	17	0	82	2	98	0	100	1576
30	19	3	77	15	0	85	8	0

In areas of low development in the low and mid-estuary, the % embayed marsh was the most important factor predicting marsh change. There was more marsh loss in areas with less than 61% embayed marshes. The % embayed marshes is strongly, negatively correlated with % extensive marshes ($r^2 = -0.89$), but weakly ($r^2 < 0.40$) correlated with all other explanatory variables. Extensive marshes might be important predictors of

marsh loss, in this subset of marshes, which were not identified in the decision tree. In general, in this subset of marshes, extensive marshes are found on the mainstem of the estuary, and subject to higher energy, while embayed marshes are found in sheltered tributary creeks.

Discussion

Marsh change along the York River estuary is highly variable and that variability is not primarily explained by differences in erosion rates and migration potential, as would be expected under rising sea levels. Development and marsh form interact with location in the estuary, a surrogate for erosion potential, to modify the marsh response to sea-level rise. Although the marsh change groups into four categories, there is variability in response even within those categories. This calls into question the current practices of evaluating regional marsh change with studies of only one or a few marshes and/or studies limited to only extensive marshes.

Extending the marsh change in one marsh or creek system to an estuarine-scale requires careful understanding of the spatial variability of the drivers and the magnitude of their importance in each setting. Considering only net overall change in estuarine marsh extent does not adequately represent the potential impact to the resource. In this study, marsh change was highly variable across subwatersheds, ranging between a 32% loss and an 11% gain in marsh extent. The importance of the marsh loss to overall estuarine function will depend on the location and type of marsh lost. Loss was focused in the brackish part of the estuarine, compared to the more stable oligohaline areas. In addition,

much of the marsh loss was in fringing marshes which constitute a small part of the total estuarine acreage, but a disproportionately large part of the ecosystem service capacity (Bilkovic et al., 2017; Bilkovic and Mitchell, 2017; Beck et al., 2017).

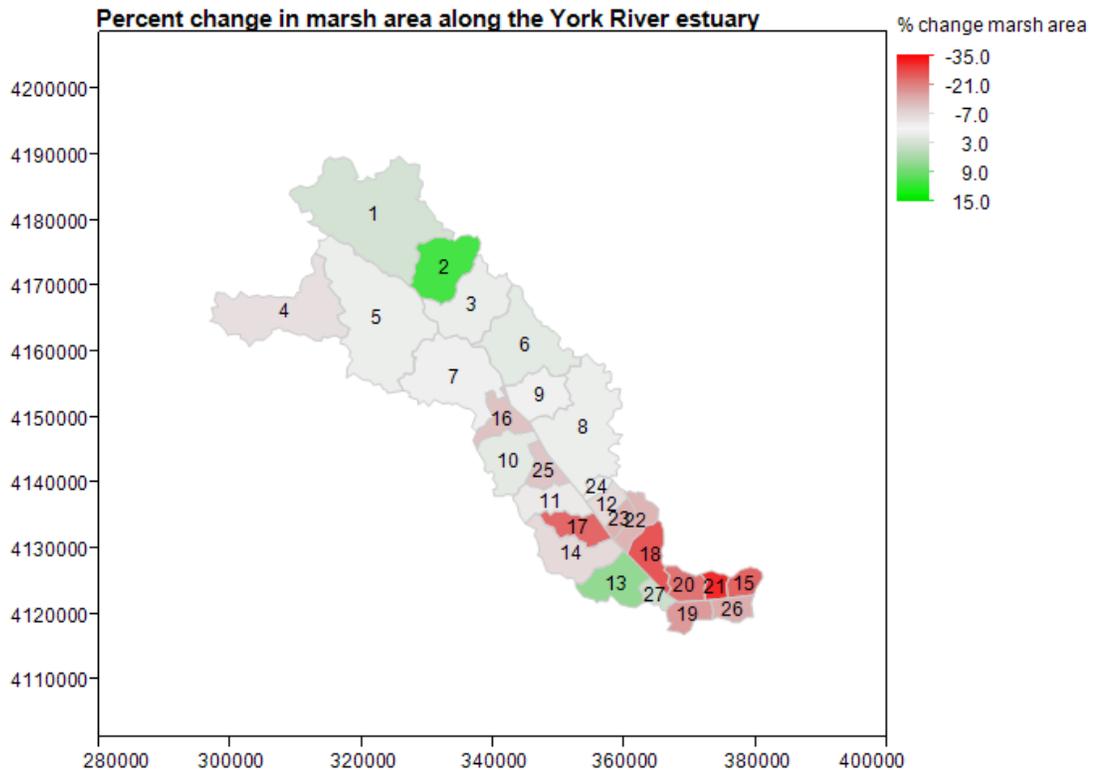
Spatial differences in marsh response

Developed landuse was the most important predictor of marsh loss. Sub-watersheds with high development (Leaf group 4) tend to have extensive creeks edged with fringe marshes. They also tend to have stabilized shorelines, heavy boat traffic and lawns that extend to the water. These three factors may explain the link between development and marsh loss. Boat wakes have been shown to negatively impact shoreline stability in salt marshes (Castillo et al. 2000) and shoreline structures (bulkheads in particular) reflect wave energy, exacerbating erosion. Another link between human development and marsh loss, which might be explained by these patterns, is eutrophication due to fertilization (Deegan et al., 2012). Although it is not clear which of these factors is responsible for the loss in developed creeks, creek systems with lower development (found in Leaf group 3) with lots of natural lands surrounding them and relatively little shoreline stabilization had lower marsh loss.

Fetch decreases with distance up the estuary, and therefore, wind-driven wave energy (the predominant driver of marsh erosion on coastal shorelines; Schwimmer 2001) would also be expected to decrease. In this study, marsh loss generally decreased with distance from the mouth of the estuary (Figure 6), suggesting that wind-wave erosion is an important driver of marsh loss. Within this general trend there is still significant

variability among subwatersheds in the same section of the estuary. This is likely due to high spatial variability in the drivers of marsh change, including sea-level rise, wave energy, land elevation, sediment supply, and shoreline stabilization.

Figure 2-6. The percent change in marsh area by distance from the mouth of the estuary. Numbered regions are subwatersheds used in the analysis. Areas in red (negative values) represent marsh loss. Areas in green (positive values) represent marsh gain.



The magnitude of variations in local sea-level rise is impossible to determine with existing data, but could be an important explanatory variable. As sea-level rises, it increases the depth of inundation on the marsh surface, which triggers responses in vegetation (Morris et al. 2002), sediment accumulation (Kirwan and Murry 2007), and erosion (Mariotti and Fagherazzi 2010). These responses are specific to plant species and marsh position (and may be related to associated fauna, such as ribbed mussel (*Guekensia*

demissa) presence), leading to spatial variability in marsh response to sea-level rise. In addition, subsidence can vary on small spatial scales (Cahoon 2015) causing marshes in neighboring subwatersheds to experience different rates of relative sea-level rise. High rates of sea-level rise can lead to marsh drowning, but in areas with sufficient sediment supply and low elevation adjacent lands it can lead to marsh expansion. In the York River estuary, the highest known rate of subsidence (Eggleston and Pope 2013) is found in the group of subwatersheds (subwatersheds 6,7) located at the estuarine turbidity maximum, suggesting ample sediment supply. Overall, they are showing little change (<1% change) in marsh extent, suggesting that the sediment supply may be compensating for the increased rate of sea-level rise. However, their low elevation adjacent lands suitable for marsh migration are constrained. With continued acceleration in sea-level rise rates, this area may be less resilient than it currently appears.

Land elevation is the dominant factor controlling marsh migration potential although it is moderated by development (which is the most important factor controlling marsh change in the partitioning analysis). Areas with low elevation lands immediately adjacent to wetlands show signs of marsh gain through migration, with marsh gain in the lower estuary primarily seen in extensive marshes as migration into interior forested hummocks (Figure 6), and along the river shoreline as migration into low-lying riparian uplands. The conversion of forest hummocks to marsh is expected to continue with sea-level rise, but represents only a small area of potential future gain relative to upland migration.

Subwatersheds 21 & 22 are areas which would be expected to show marsh gain through migration due to their low riparian elevations. Instead they have had a loss in marsh

extent of 13% and 31% respectively. The shorelines in these subwatersheds are heavily stabilized, blocking upland migration (Figure 5) and potentially impacting sediment availability by trapping sediment landward of the bulkhead (Douglass and Pickel 1999, Griggs 2005).

Progradation, the growth of marshes into the unvegetated intertidal zone, is only seen in the upper/riverine sub-watersheds of the York River system, above the turbidity maximums (e.g. subwatershed 2). There it is a minor process, despite the presence of higher total suspended solids (TSS) (Reay 2009) and eroding bluffs. Progradation is controlled by the balance between nearshore sedimentation and sea-level rise (Schwimmer and Pizzuto 2000), and it is typically favored by low rates of sea-level rise (Mariotti and Fagherazzi 2010) which are not found in the York River. Without sufficient sediment supply, marshes can begin to pond, leading to fragmentation and permanent loss (Mariotti 2016).

Published values for the York River estuary (Reay 2009) suggest that TSS is most likely to contribute to marsh gain and persistence around the two turbidity maximums. The primary turbidity maximum is found in subwatersheds 6, 7, 9 and 16. But, marsh extent change was minimal in these areas. Subwatershed 16 actually shows losses due to marsh fragmentation. Despite the low levels of development and shoreline stabilization, sediment supply is apparently still inadequate to counter the local rates of sea-level rise.

Comparison of historic and modern marsh extents

Comparisons of historic and modern marsh extents should always be approached with caution. Comparison errors are unavoidable, but can be minimized with good digitization and verification processes; allowing accurate determination of past shoreline changes (Crowell et al., 1991). Errors stem from the precision (scale) of the aerial photography used in the marsh delineation and the digitizing technology. In our case, the old aerial photography was the limiting driver of the error, but it was mitigated by the field verification process. Using aerial photography alone (at a scale of 1:24,000) would preclude the inclusion of narrow (<5m wide) marshes in the original survey, and potentially leading to an overestimation of marsh gain. However, these marshes were added following the field surveys, improving the accuracy of the surveys.

In addition to the error due to technological limitations, there is an undefinable interpretation error, both during the digitizing and the field verification. Both the wetland/upland boundary and the water/wetland boundary are subject to this error (Anderson and Roos, 1991; McCrain, 1991). The water/wetland boundary is defined as mean sea level, but aerial photography is seldom tidally coordinated, leaving room for interpretation by the digitizer. We minimized this error through constant definitions of mean sea level signals (e.g., edge of vegetation in *S. alterniflora* marshes) and verification of the digitization (each digitization is verified by two independent reviewers). The wetland/upland boundary can be subject to interpretation, particularly where mowed lawns intersect with marshes. This error was minimized by training on

signals of waterlogged soils, the verification processes and the use of a lidar-based digital elevation model to define elevations above the tidal extent.

Consequences of marsh change on ecosystem health

Percent marsh losses were heaviest in fringing marsh systems, which are ecologically important due to their high edge:area ratio. Despite their small acreages, fringe marshes have been found to have similar wave attenuation, nutrient removal, sediment accretion, and habitat values compared to extensive marshes (Bilkovic et al. 2016). In the original survey, fringe marshes were nearly continuous along the shoreline, while in the current survey they have become fragmented in many creek systems. Fragmentation threatens marsh resilience under sea-level rise, as there is more exposure for erosion. In addition, habitat fragmentation in terrestrial and estuarine systems has been linked with shifts in biodiversity, loss of habitat-specific sensitive or functionally important species, and isolation of populations when connectivity is diminished (Kareiva and Wennergren 1995, Fahrig 2003, Thrush et al. 2008, Collinge 2009).

Marsh losses by area were highest in extensive marshes, particularly marsh islands, which are important habitats for avian species (Wilson et al. 2009). Both fringing marshes and marsh islands have limited potential for migration in this estuary, so loss to erosion cannot be counterbalanced in the long term (e.g., Schile et al 2014). Embayed marshes appear particularly resilient, with small embayed marshes persisting at the tops of creeks where long extents of fringe marsh have been lost.

Ecological concerns with the observed shifts in marsh extent include both loss and redistribution of ecological services provided by marshes, particularly water quality and habitat functions. For both of these functions, location is often as important, if not more important, than total amount of marsh. Fragmentation and relocation risk disconnecting marsh service capacity from landscape-based needs and opportunities.

Marshes are efficient at removing sediments (Fredricks and Perry 2001) and nutrients (Deegan et al. 2007) from the tidal waters and nutrients from groundwater (Tobias et al. 2001). Removal of sediment from the water can benefit light-limited and filter feeding species, while the removal of nutrients reduces algal blooms, contributing to the overall health of the estuary. Fringing marshes may be particularly important for groundwater nutrient removal (Beck et al. 2017) due to their near continuous presence along undisturbed shorelines. In the Chesapeake Bay groundwater discharge of nutrients may be as high as 30% of surface inputs (Libelo et al., 1991), potentially making fringe marshes a critical mediator of estuary water quality.

The loss of marsh in the developed creek systems (>15% developed) suggests that they may be approaching or even have crossed an ecological threshold (breakpoints at which a system or community notably responds, perhaps irreversibly to a disturbance).

Ecological thresholds studies suggest that the relationship between development and ecological function is not a gradual, linear relationship and that alarmingly low levels of development (between 10-25%) can dramatically diminish a multitude of system functions (e.g. Wang et al., 1997; Limburg and Schmidt, 1990; Paul and Meyer, 2001;

DeLuca et al., 2004; Brooks et al., 2006; King et al., 2005; Bilkovic et al. 2006; Lussier et al. 2006).

Migration of tidal marshes into upland habitats is not a dominant process in the estuary, but will mitigate some of the wetlands loss. These new tidal marshes should provide similar habitat and water quality functions. They do not provide the same carbon storage function because wetland soils take many years to develop (Craft et al. 2003). Migration of tidal marsh into previously non-tidal wetlands (as seen in some of the watersheds) may result in some changes in function (non-tidal wetlands provide different types of habitat and have different nutrient cycling pathways), but should have a net neutral impact to water quality.

Conclusions

Within a single estuary, marsh change over time shows high spatial heterogeneity related to the variability in the importance of and interactions between multiple drivers. Erosion rates, migration opportunities, and the rate of sea-level rise all affect marsh persistence. Importantly, human actions are also critical, and frequently less predictable, determinants of how marshes respond through time.

Improving our understanding of marsh change requires examination of change on ecosystem scales. Despite the use of an entire estuarine system in this study, extension of results to characterize an even larger system (e.g. Chesapeake Bay) is probably inappropriate. Forecasts of ecosystem change based on small scale studies often leads to

inaccurate or unsubstantiated conclusions. The processes leading to change are spatially variable and not always predictable.

There are, however, some lessons that can be taken from this study:

1. Human shoreline use (e.g., development, shoreline hardening, boating activity) can dominate physical processes to alter the marsh response to sea-level rise.
2. Defining sediment availability for a given marsh may not be sufficient to determine its potential for expansion or persistence under sea-level rise.
3. Marsh response varies by form as well as setting, and ecologically important fringe marshes may be particularly vulnerable.

Understanding past changes in marsh extent are critical for improved prediction of future change under accelerating sea-level rise. Knowing which marshes are most vulnerable allows us to protect them, minimizing future impacts to estuarine systems.

Chapter 3 Marsh vegetation as an indicator of ecosystem response to sea level rise

Abstract

Tidal marshes are a major ecological resource and a driver of many estuarine functions. Therefore, the long term sustainability of tidal marsh ecosystems is a question of great interest in the research community. Sea level is rising at an unusually high rate in the Chesapeake Bay relative to most of the Atlantic coastline, putting Bay marshes at high risk from drowning and erosion. Understanding the patterns of change and the importance of different drivers of change is critical to tidal marsh sustainability. Tidal marshes plant communities are highly reflective of their environment; tightly reflect inundation period and salinity. Long-term vegetation changes can be an indicator of marsh resilience or response to sea level rise and may help improve predictions about future conditions. Specifically, marsh vegetation can help identify marshes which are not keeping pace with sea level rise (therefore likely to drown and disappear) and marshes which are undergoing salinization, resulting in ecosystem shifts. In this study, we use tidal marsh vegetation surveys from approximately 40 years apart to examine changes in plant communities indicative of stress from salinity and inundation.

Patterns of community change suggested salinity increases near the freshwater-brackish water interfaces on the tributary rivers and some creek systems. In addition, examination of changes in both the extent of low marsh and the change in community type suggested areas of increased inundation and erosion that were fairly consistent between analyses,

with erosion dominating on higher energy river shorelines and inundation dominating in creek systems. Another change between surveys was an expansion of *Phragmites australis*. The highest concentrations of *P. australis* dominated communities are found on the northern peninsula although it is currently found throughout the estuary including on high salinity Bayfront shorelines and lower salinity riverfront and creek shorelines. Changing marsh vegetation is a flexible measure of ecosystem alternation; understanding the patterns of vegetative change should enhance our understanding of future marsh changes and the ecosystem consequences of those shifts.

Introduction

Sea level is rising globally at about 3.2 mm/yr (1993-2009; Church and White 2011) and evidence suggests that this rate is accelerating (Nerem et al. 2018). Although sea level rise-associated increases in water levels have implications for all coastal areas, it is particularly critical in estuarine ecosystems where it changes both intertidal inundation patterns and salinity distribution. Increasing water depths and volumes interact with estuarine morphology to change tidal resonance or the tidal prism in a system; affecting local tide ranges (Pethick 1993). Changing salinity patterns are shaped by movement of the salt wedge up the estuary, counterbalanced by freshwater flows from the rivers. These changes propagate through the ecosystem, changing habitat types and associated communities (e.g., Short and Neckles 1999, Saunders et al. 2013), nutrient storage and cycling (e.g., Weston 2011, Neubauer 2013), and marsh stability. Sea level rise-driven changes can be particularly apparent in intertidal habitats, where changing inundation and salinity patterns visibly shift ecological niches.

Tidal marshes are an important intertidal habitat globally, providing modification of wave climates that reduces erosion (Shepard et al. 2011) and creates habitat opportunities (Bruno 2000). Their complex structure serves as refuge habitat enhancing fish populations (Minello et al. 2012), while their vegetative plant community and soil characteristics modify estuarine nutrient loading from upland (Valiela & Cole 2002) and tidal (Deegan et al. 2007) sources, and act as a long term carbon sink (Chmura 2003, Bridgham et al. 2006). As intertidal communities, tidal marshes are highly reflective of their environment, exhibiting communities that tightly reflect inundation period and salinity. As sea level rises, longer tidal inundation periods increase hypoxia in marsh soils, leading to declines in plant productivity and survival (Morris 2007) at the leading edge of the marsh. Although marshes have mechanisms to cope with rising waters, excessive rates of rise can overwhelm the ecological resilience cycles leading to marsh loss (Schile et al. 2014). Ecological shifts from salinization tend to be less dramatic, leading to changes in community composition rather than loss. However, these community shifts change habitat provision services and may result in a loss of associated species (e.g., change in nesting bird species, Wilson et al. 2009; loss of anadromous fish, Bilkovic et al. 2012). Tidal marsh plant communities respond to sea level rise through dramatic shifts in community composition, which are easily observed, therefore, they can be used as sentinels of change in estuarine ecosystems.

Changing inundation periods in a marsh are easily observed in the landward migration of low intertidal marsh plants, which in microtidal, temperate estuaries are typically found between mean sea level and mean high water. This portion of the tidal range experiences daily inundation and few plants are capable of thriving under those conditions. As high

marsh plants begin to experience daily inundation, their productivity declines and their competitive advantages are lost, leading to landward expansion of the low marsh plants. Unless shifts in the mean sea level boundary are mitigated by changes in the tidal frame (mediated by tidal resonance) or marsh surface accretion (mediated by sediment availability and plant characteristics), the concurrent shift in mean sea level stresses plants at the leading edge of the marsh. Productivity of these plants begins to decline, reducing soil stability, and eventually the area converts to tidal mud flats (Schile et al. 2014). Sea level rise-driven shifts in the tidal frame may have no impact if marsh migration and accretion are sufficient to keep pace, but when these processes fall below the rate of sea level rise the marsh begins to drown leading to marsh loss. Sea level rise is also linked to enhanced erosion due to increased water depth. For a variety of reasons, coastal marsh loss already is a significant issue throughout the United States and has been documented in the Chesapeake Bay (Mitchell et al. 2017, Kearney et al. 1988). Marsh loss results in a total loss of related functions, releases previously stored blue carbon, and reduces global carbon storage capacity, potentially impacting global cycles (Coverdale et al 2014, Chmura 2013).

Salinization can also be observed in changes in the plant community. In the mid-Atlantic, tidal marshes can be broadly categorized into four groups with relatively distinct communities: salt marsh (40-18psu), brackish marsh (18-5psu), oligohaline marsh (5-0.5psu) and freshwater marsh (<0.5psu). In general, the diversity of the community increases as the salinity decreases due to the dual stress of inundation and salinity/sulfur present in salt and brackish marshes, which few plants can successfully manage. Plant growth is generally reduced under increased salinity changing competitive interactions

and resulting in shifting spatial patterns of plant species (Janousek and Mayo 2013). Relatively small increases in salinity can shift any community to a more salt-tolerant community. Tidal freshwater marshes are particularly susceptible to relatively small increases in salinity (Perry and Atkinson 1997, Sutter 2014), resulting in documented shifts in plant communities (e.g., Perry and Hershner 1999). Freshwater marsh plants are typically found below 0.5 psu and are measurably impacted with salinities as low as 1.5 psu (Sutter et al. 2014, 2015). These changes alter the entire ecology of the marsh and all associated services. Salinization of a freshwater marsh to a brackish marsh may result in increased sediment stability (Odum 1988) and resistance to erosion but also altered habitat type and decreased carbon storage (Craft 2007, Herbert et al. 2015). Similar to marsh loss, estuarine freshwater marsh salinization is predicted to alter marsh carbon storage with global implications (Baustian et al. 2017).

The lower portion of the Chesapeake Bay is experiencing the highest rates of relative sea level rise along the Atlantic coast (Boon, 2012; Ezer et al. 2013; Sallenger et al., 2012; Kopp, 2013, Boon and Mitchell 2015). In the Bay, sea level rise rates since the 1980s have been between 3.93 – 5.86 mm/yr (Ezer and Atkinson 2015), outpacing global rates. During that time, sea level has come up approximately 0.15 m vertically along Bay shorelines, which, in flat intertidal areas, can translate to a horizontal shift in the mean high water mark of 1-2 m. This extreme rate of rise has led to a demonstrable increase in flooding (inundation) frequency (Ezer and Atkinson 2014; Sweet and Park 2014). Salinity has also increased concurrent with sea level rise in the Chesapeake Bay, with an increase of 0.5 psu in the mainstem of the Bay since 1949 (Hilton et al. 2008). Documented loss of marshes (Mitchell et al. 2017, Kearney et al. 1988) and salinization

of plant communities (Perry and Hershner 1999) coupled with unusually high rates of sea level rise make the Chesapeake Bay an ideal location to look for estuary-wide impacts of sea level rise. Long-term vegetation changes can be an indicator of marsh resilience or response to sea level rise and may help improve predictions about future conditions. Specifically, marsh vegetation can help identify marshes which are not keeping pace with sea level rise (therefore likely to drown and disappear) and marshes which are undergoing salinization, resulting in ecosystem shifts.

In this study, we used shifts in tidal marsh plant community composition to highlight areas in the Chesapeake Bay, VA that are undergoing change. We categorized the types of change to target early indications of sea level rise stress.

Methods

The Chesapeake Bay is the largest estuary in the northern hemisphere and has intricate shorelines edged by marshes of all shapes and sizes. For this study, we focused on the Virginia portion of the Chesapeake Bay. The Chesapeake Bay, Virginia is generally representative of tidal estuaries, containing a diverse array of tidal marsh types and ecologies, driven by salinity regime and geologic setting. Salinity ranges from approximately 35 psu near the mouth of the Bay to 0 psu in the upper reaches of the tributaries and many of the small tidal creeks along their edges. Currently, there are approximately 761 km² of tidal marshes, consisting of a mix of salinity types consisting of about 25% tidal freshwater marsh, 15% oligohaline marshes, 30% brackish and 30% salt marsh (TMI; CCRM 2017). Fringing marshes are spread extensively along the shoreline, encompassing all salinities and a diverse array of plant communities. Embayed marshes are found near the tops of tidal creeks and are typically oligohaline or freshwater

marshes, although they can be salt marshes in the more saline areas. Extensive salt marsh areas are found in Bay-front localities Accomack and Poquoson, and extensive tidal freshwater marsh areas in York River tributary localities King and Queen, King William and New Kent.

The York River tributary (and sub-estuary) was targeted for a detailed plant composition quantification survey. The York River, Virginia is one of five major tributary systems in Chesapeake Bay and generally representative of conditions encountered throughout the Bay and similar estuaries (Reay and Moore 2009). It is a brackish system approximately 64 km long branching into two smaller tributaries; the Mattaponi and Pamunkey Rivers. It possesses a wide range of salinities, from approximately 20 psu near the mouth of the river to 0 psu several kilometers upriver of the branch, and supports the same diverse habitats found in the Bay, including tidal freshwater, oligohaline and brackish/salt marshes.

Tidal Marsh Inventory

The Tidal Marsh Inventory (TMI; CCRM, VIMS) is a geospatial survey of all tidal marshes in Virginia, including their location, extent and plant community; the survey has been done twice, approximately 30-40 years apart. The surveys involved digitization of marsh extents and locations from maps and aerial imagery. The original inventory of tidal wetlands was developed by VIMS in the early 1970s. This survey represents a quantified baseline for areal and biotic change over a 30+ year period. The recent surveys, were conducted by CCRM, VIMS from 2010 to 2018. High resolution color infra-red imagery was used to generate marsh boundaries using heads-up digitizing techniques at a scale of 1:1,000. Marsh boundaries were verified in the field and vegetative surveys were done as

described below. Marshes were geospatially linked between the two time periods through superposition and cross-walking identification numbers.

Marsh vegetation surveys

Field surveys of tidal marsh vegetation (henceforth referred to as TMIs or Tidal Marsh Inventories) were conducted in the Chesapeake Bay, Virginia, with a particular emphasis on the York River sub-estuary (Figure 1). Surveys were conducted by boat and included all accessible marshes. They were typically done during the summer months (May-August), although a few historic surveys were done in the fall. Historic TMIs were surveyed from 1973-1991 and current TMIs were surveyed from 2010-2018 (dates of surveys depend on the location; see Table 1). The average time between surveys was 32 years.

In the York River, each plant species present in the marsh was identified to genus level and species level when possible. Percent contribution of each species to the overall community was estimated resulting in a species matrix for each marsh and from each survey. Each marsh was also categorized as one of 12 recognized plant community types (Table 2) based on the dominant species mix. Categorizing the marshes into community types can be done relatively rapidly, allowing extensive surveys and circumvents the problem of surveys occurring during different months, when individual species might be more or less visible. Following analysis of the York River survey data, the current TMI was expanded to cover most Chesapeake Bay, VA localities. In the expanded survey, only marsh community type was recorded. For the remainder of the paper, individual plant species will be referred to by their scientific names, and community types will use common names.

Figure 3-1. Marshes with communities surveyed in both time periods used for this study. Summer salinity for the Bay (from the year 2000) is shown for context.

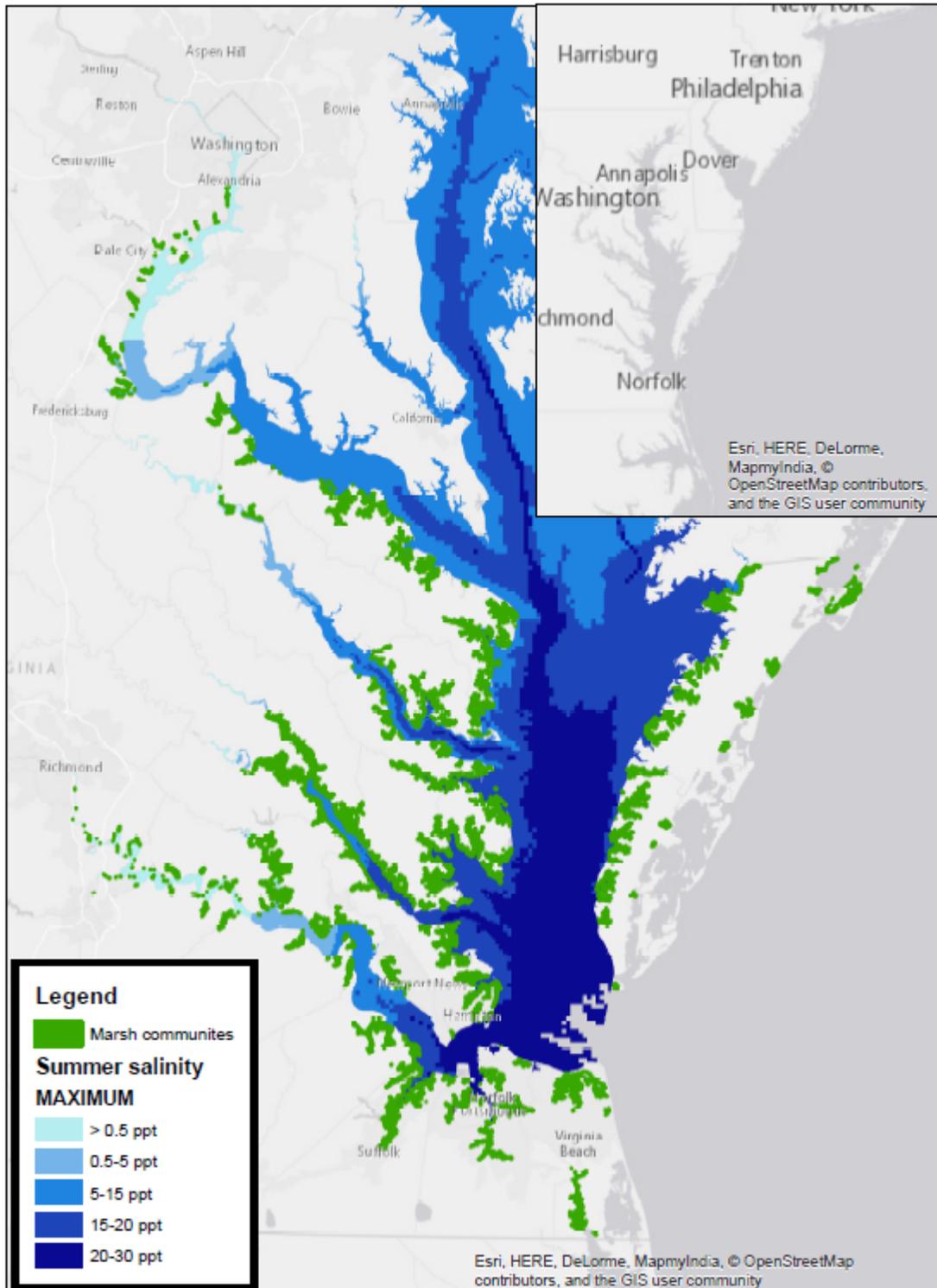


Table 3-1. Years of historic and current TMI surveys. York River localities are indicated by an asterisks (*). Parts of some localities were surveyed in different years; when that happened, all years are listed in the table and the time between surveys is calculated as the shortest time.

Locality	Historic TMI	Current TMI	Years between
Accomack	1977	2016	39
Alexandria	1976	2012	36
Charles City	1990	2013	23
Chesapeake	1991	2016	25
Chesterfield	1991	2017	26
Colonial Heights	1991	2017	26
Essex	1979	2018	39
Fairfax	1976	2012	36
Fredericksburg	1979	2017	38
Gloucester*	1976	2010/2014	34
Hampton	1975	2012	37
Henrico	1991	2017	26
Hopewell	1989	2016	27
Isle of Wight	1981	2017	36
James City*	1980	2010/2014	30
King & Queen*	1987	2010	23
King George	1975	2017	42
King William*	1987	2010	23
Lancaster	1973	2015	42
Mathews	1974	2012	38
Middlesex	1981	2015	34
New Kent*	1979	2010	31
Newport News	1977	2014	37
Northampton	1977	2011	34
Northumberland	1975	2014	39
Norfolk	1987	2014	27
Petersburg	1991	2017	26
Poquoson	1974	2013	39
Portsmouth	1989	2015	26
Prince George	1989	2016	27
Prince William	1975	2013	38
Richmond (city)	1991	2017	26
Richmond (county)	1990	2018	28
Spotsylvania	1979	2017	38
Stafford	1975	2015	40
Suffolk	1991	2013	22
Surry	1981	2017	36
Virginia Beach	1976/1979/1989	2012	23
West Point*	1987	2010	23
Westmoreland	1978	2012	34
York*	1974	2010/2013	36

Table 3-2. Plant communities as identified in the Tidal Marsh Inventories. Habitat niches are approximate and based on typical distribution in Virginia tidal marshes. Mixed community types are marshes where no single species encompassed 50% or more of the plant community.

Plant community types		Dominant species	Habitat niche characteristics (typical)	
			Elevation	Salinity
1	Saltmarsh cordgrass	<i>Spartina alterniflora</i>	Low	salt --> brackish
2	Saltmeadow	<i>Spartina patens, Distichlis spicata</i>	High	salt --> brackish
3	Black needlerush	<i>Juncus roemerianus</i>	Mid	salt --> brackish
4	Saltbush	<i>Iva frutescens, Baccharis hamiflora</i>	Very high	salt --> brackish
5	Big cordgrass	<i>Spartina cynosuroides</i>	High	brackish --> oligohaline
6	Cattail	<i>Typha latifolia, Typha angustifolia</i>	Low	fresh
7	Arrow arum-Pickerel weed	<i>Peltandra virginica, Pontederia cordata</i>	Low	fresh
8	Reed grass	<i>Phragmites australis</i>	Mid/High	oligohaline --> fresh
9	Yellow pond lily	<i>Naphur luteum</i>	Very low	fresh
10	Saltwort	<i>Salicornia sp.</i>	Mid	salt
11	Freshwater mixed	<i>Zizania aquatica, Polygnum sp., Spartina cynosuroides,</i> and freshwater species listed above	Entire	fresh
12	Brackish water mixed	<i>Scirpus robustus, Scirpus olnei,</i> and brackish species listed above	Entire	Salt --> brackish

Elevations: Very low (below Mean Sea Level)
 Low (Mean Sea Level --> Mean High Water)
 Mid (above Mean Sea Level --> below Highest Astronomical Tide)
 High (Mean High Water --> Highest Astronomical Tide)
 Very High (around Highest Astronomical Tide)

In both historic and current surveys, plant identification was performed by experts. In the current survey, extensive QA/QC was implemented in the York River to ensure consistency, including have multiple teams survey a single marsh and bringing in additional experts to re-survey marshes. These tests indicated consistency on community typing and plant identification and a discrepancy of 10-15% in the percent coverage of a given plant species.

Comparison of historic and current plant communities

Species matrices (York River) and community types (Chesapeake Bay, VA) were georeferenced for analysis. Between the two surveys marshes have been created, lost and fragmented. In addition, some of the original survey marshes were not accessible in the current TMI due to infilling of channels. For vegetative community comparison purposes, only marshes with communities surveyed during both time periods have been included in the analysis and marshes lost or gained between the two surveys have been excluded from the datasets.

Marsh plant communities were compared in three ways:

- Change categorization based on a species matrix (applied in the York River and used to target general patterns of change and specific salinization of fresh and brackish water marshes)
- Change in low marsh plant community extent (applied in the York River and used to target areas experiencing increased in inundation)
- Change in community type (applied across the sampling area and used to see differences on a broad scale)

Species matrix driven changes

In the York River, marsh communities at 263 marshes were compared between the two surveys using Canonical Analysis of Principal coordinates (CAP, PEMANOVA+ for PRIMER 2008). CAP is a constrained ordination technique that uses an *a priori* hypothesis (in this case, the marsh plant community types or community salinity designations) to draw an axis through the cloud of points in a way that maximizes the group differences (separates the groups). CAP can be used for classification and those classifications can be used to track changes in a site over time. We used a historic species matrix to create the classifications and those classifications were then used to classify the 2010 species matrix. Two separate CAPs were run. The first analysis was designed to look at overall changes in marsh plant communities; it used a Bray-Curtis resemblance measure on a raw species matrix (to reduce the influence of rare species). The historic data was classified into community types (see Table 2). The second analysis was designed to target changes in plant communities linked to salinity. It used a Bray-Curtis resemblance measure on a square root-transformed species matrix. The square root transformation was used to increase the influence of rare species and enhance the classification of freshwater marshes, since freshwater marshes are typically more diverse than brackish and oligohaline marshes. For this test, the historic data was classified into three broad salinity community types (brackish, oligohaline and fresh) based on dominant plant tolerance. Permutation tests were used in both analyses to indicate significance of the classifications. Changes in plant communities from the first analysis between the two surveys were mapped in the York River (ArcGIS 10.4.1) to look for spatial patterns of change.

Low marsh plant community extent

In the York River system, changes in the low marsh plant community extent were calculated for each marsh. The percent of low marsh plant community is an indication of the extent of flooding experienced by the marsh, with the low-high marsh plant transition at roughly mean high water. Changes in this boundary over time are indicative of changing wetland conditions, with a widening of low marsh extent indicating increased inundation extent and a narrowing of low marsh extent a likely result of erosion.

Changes in inundation extent were compared to location (riverfront or creek) using a one-way ANOVA to see if there was a relationship that might help explain patterns of spatial variation. Riverfront was defined as the margins of the York River up to approximately 58 km from the mouth of the river. The two riverine tributaries (Mattaponi and Pamunkey rivers) were considered “creeks” in this analysis. Marsh erosion is a contributor to marsh loss on the York River, but wave energy tends to decline upstream and marshes in the tributary rivers and creeks tend to show little signs of wind-wave driven erosion (Mitchell et al. 2017).

Changes in inundation extent were also compared to sediment organic matter content. Sediment samples were taken to 5 cm depth from the center of 29 marshes on the York River and its tributary creeks. To reduce confounding effects of community type, samples were only taken from brackish marshes. Samples were dried, weighed, and analyzed for organic matter using loss-on-ignition (Craft et al. 1991). Marsh accretion is a process that counteracts increases in inundation, allowing plants to maintain their elevation in the tidal frame. Accumulation of both organic and inorganic sediment is higher when inundation duration is longer, although the inorganic component seems to

increase faster in relation to flooding duration (Cahoon and Reed 1995). This suggests that in areas experiencing increased inundations, the proportion of inorganic to organic matter in the sediment would increase (% organic matter of sediment would decline). For the 29 sampled marshes, % organic matter was correlated with change in low marsh extent to determine if there is a relationship between the two measures.

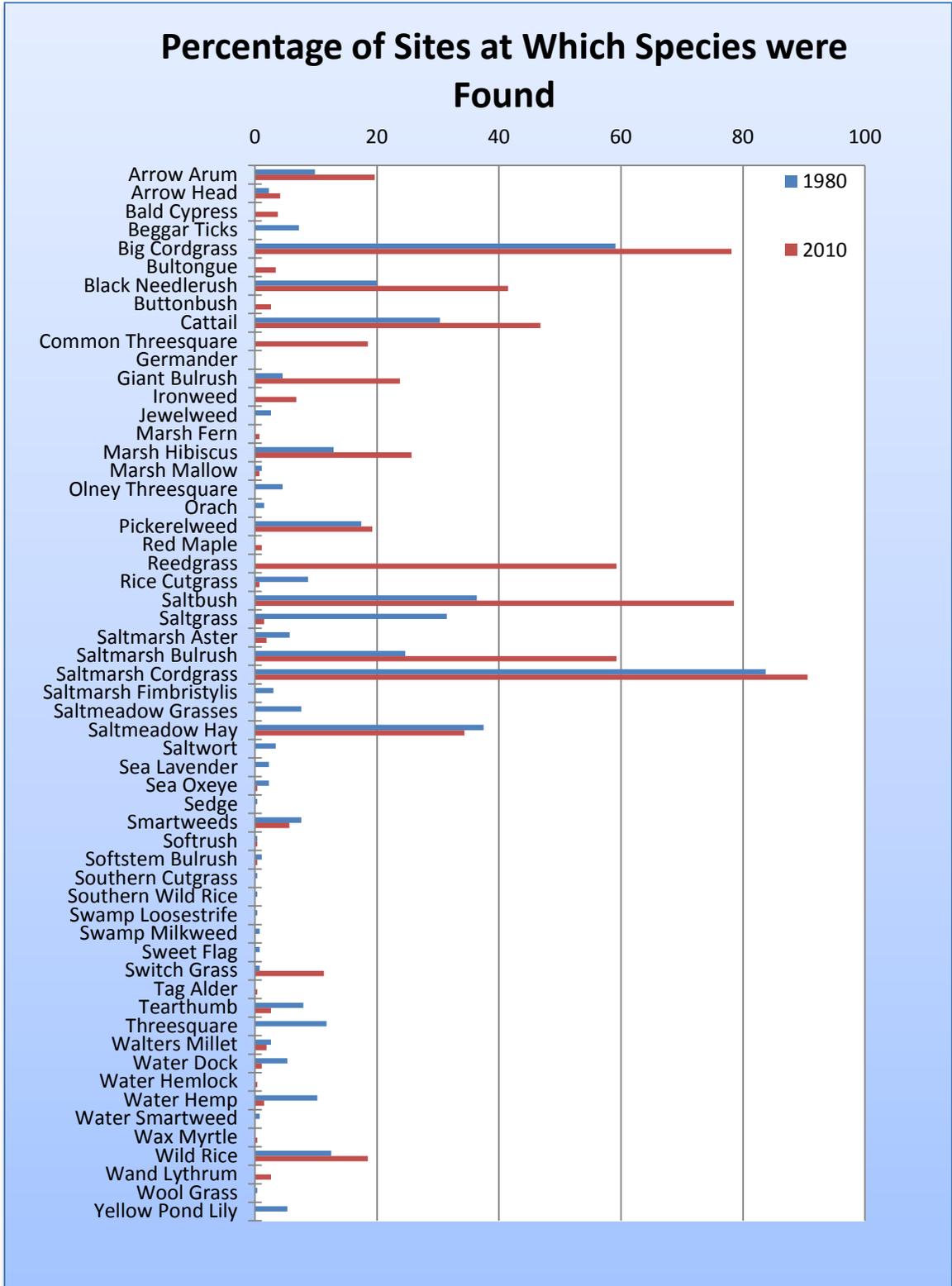
Changes in community type

To look for broad scale patterns of change, marsh plant community types from around the Chesapeake Bay, VA were compared on a marsh-by-marsh basis (using Intersect tool; ARC GIS 10.4.1) and community change was categorized as one of the following: (1) “no change”, (2) “increased inundation” (change to Saltmarsh cordgrass, Arrow Arum/Pickerelweed, or Yellow pond lily community), (3) “increased salinity” (change from community types Cattail, Arrow Arum/Pickerelweed, Yellow pond lily or Freshwater mixed to a non-freshwater, non-Reed grass community type and change from Big cordgrass to Saltmarsh cordgrass, Saltmeadow or Black needlerush), or (4) “*P. australis* invasion” (change from any community type to Reed grass). Changes in plant communities from the first analysis were mapped in the York River (ArcGIS 10.4.1) to look for spatial patterns of change.

Results

In the York River, 263 marshes were surveyed in both time periods. In those marshes, a total of 57 species were identified over the two time periods (Figure 2). Only 11 species were found at more than 20% of sites in either survey and a few of those species were found in many marshes but were always minor components of the plant community (e.g., *Hibiscus moscheutos*).

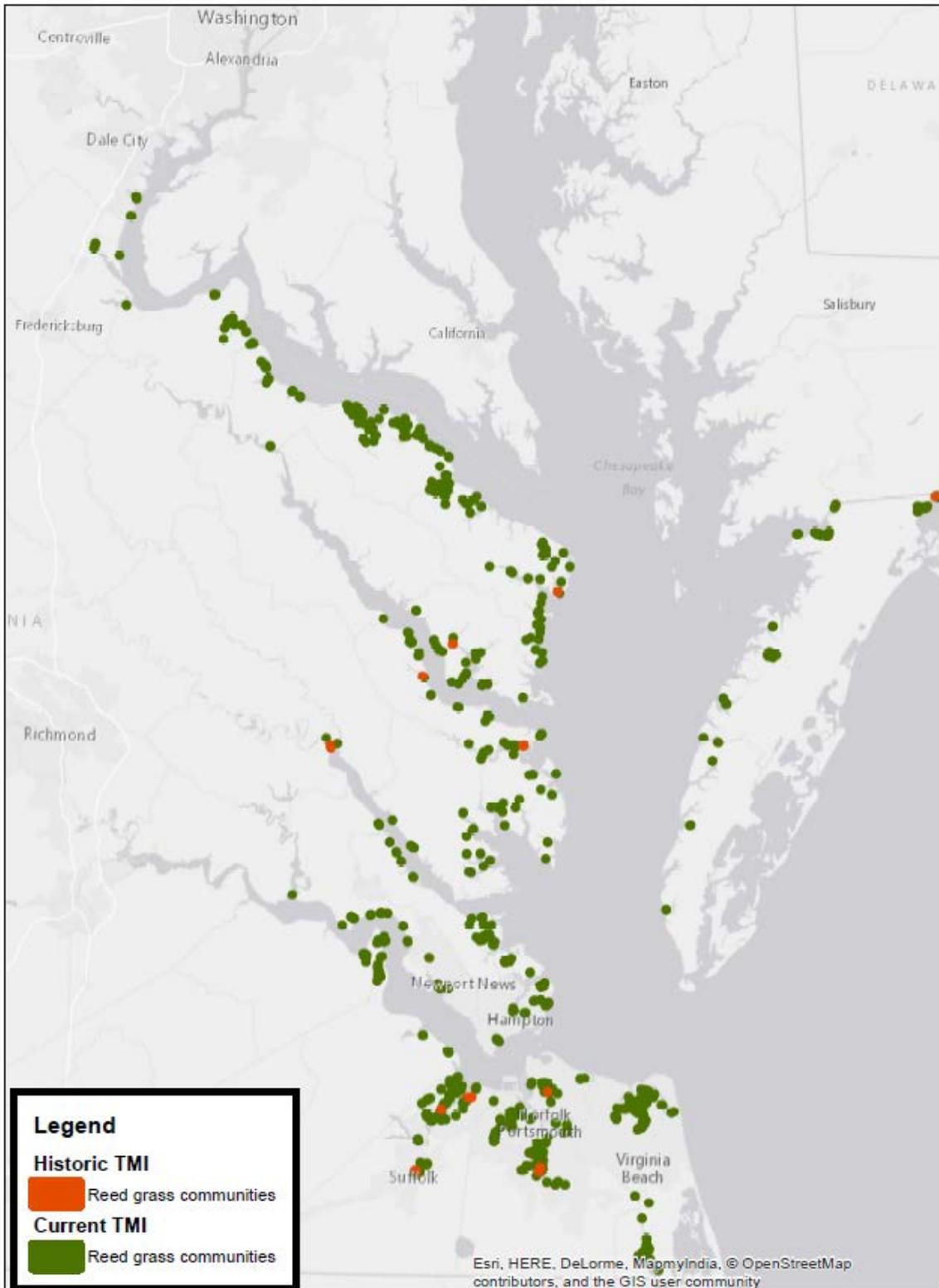
Figure 3-2. Plant species found from the historic and current TMI surveys in the York River, VA.



There were also some less common species (e.g., *Acorus americanus* and *Bidens* sp.) that were extensive in a few marshes in one survey, but missing in the next survey. *P. australis* became an important plant community in the current survey (although completely absent in the historic survey). *P. australis* is now distributed throughout the York River system, and dominant in a number of marshes.

Throughout the Chesapeake Bay, VA, a total of 17,658 marsh plant communities were surveyed in both time periods. In those marshes, 11 of the 12 marsh plant community types (from Table 2) were found in both time periods. The Saltwort-dominated community type was not identified in either the historic or current survey comparison. This is a high salinity, saltpan community that is rarely found in the Chesapeake Bay, although small plots of saltwort are common in Saltmarsh cordgrass and Brackish water mixed communities. The majority of surveyed marsh plant communities in both surveys were Saltmarsh cordgrass, followed by Brackish water mixed (57% and 19% respectively in the historic survey; 43% and 33% respectively in the current survey). The relative importance of other community types differed between the two time periods with a decline in the percentage of Saltmeadow- and Big cordgrass-dominated communities and an increase in the percentage of Arrow-arum Pickerel weed- and Reed grass-dominated communities. Similar to the York River, the distribution of *P. australis* greatly expanded between the two surveys. Reed grass community types were insignificant in the historic survey (22 marsh plant communities, less than 0.5% of total surveyed) but accounted for 733 marsh communities in the current survey, spread throughout Chesapeake Bay, VA shorelines (Figure 3).

Figure 3-3. *Phragmites australis* community distribution. Orange marshes were dominated by *P. australis* at the time of the historic survey, while green marshes were dominated by *P. australis* during the current survey.



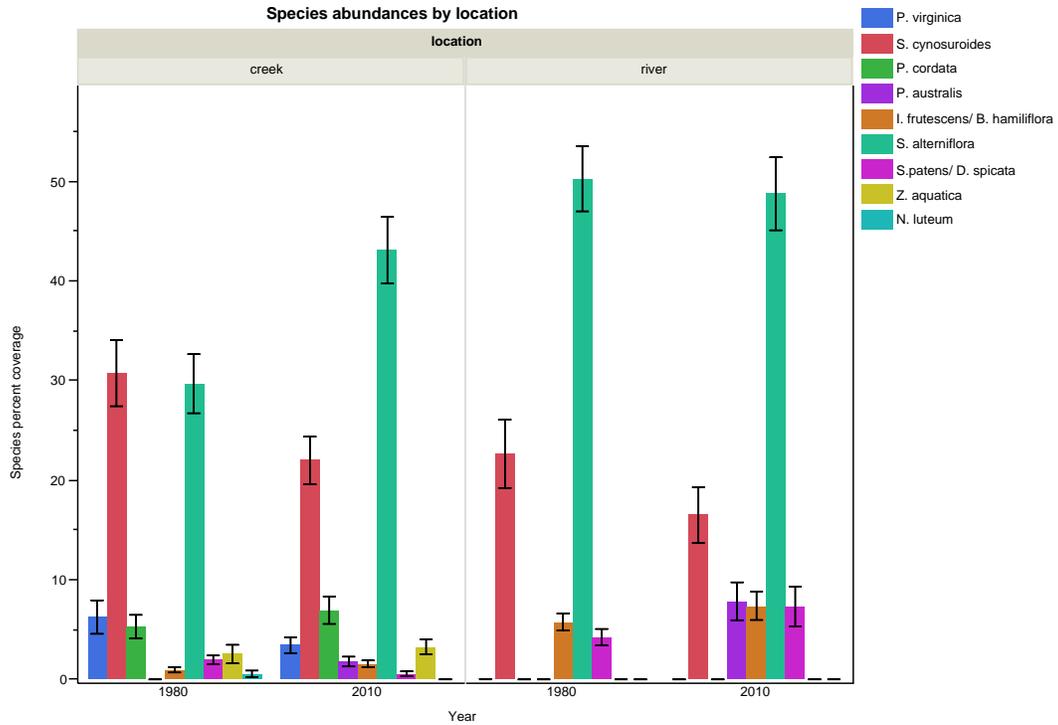
Change in species matrix (York River, VA)

Changes in marsh plant species matrix between surveys varied by energy regime (riverfront vs. creek) and salinity type (Figure 4). In the creeks, two of the three freshwater low marsh species (*Peltandra virginica* and *Nuphar lutea*) declined in the creeks between the two surveys; while a third species (*Pontedaria cordata*) increased slightly. The salt/brackish water low marsh species (*Spartina alterniflora*) increased between the two surveys. This could be due to increased inundation in some creek marshes, or may indicate salinization of the creek system. During the same time period, *S. alterniflora* declined slightly in the riverfront marshes, suggesting erosion of the leading marsh edge. *P. australis* appeared in the current survey (2010) and was found in 14 creek and 19 riverfront marshes. It was more expansive in the riverfront marshes on average (mean = 7.8 %, stdev = 16.1) than the creek marshes (mean = 1.8 %, stdev = 5.4) though it was dominant (>50% cover) in only three marshes.

CAP analysis indicated significantly different groups existed based on the historic species matrix for both the Type and Salinity categories (Figure 5 & 6). Although there were a few historic marshes categorized as type 2 and 3, the distinctions between these groups based on the species matrix were not robust and therefore none of the 2010 marshes were categorized into those groups. The remaining groups (types 1, 12, 4, 5, 7, and 11) were distinct (=> 75% correct categorization). In the salinity categorization test, all three groups (Brackish, Oligohaline, and Fresh) were very distinct (> 90% correct categorization). Examination of plant species correlations shows that splits in the community groups are driven by dominant plant species indicative of brackish (*S.*

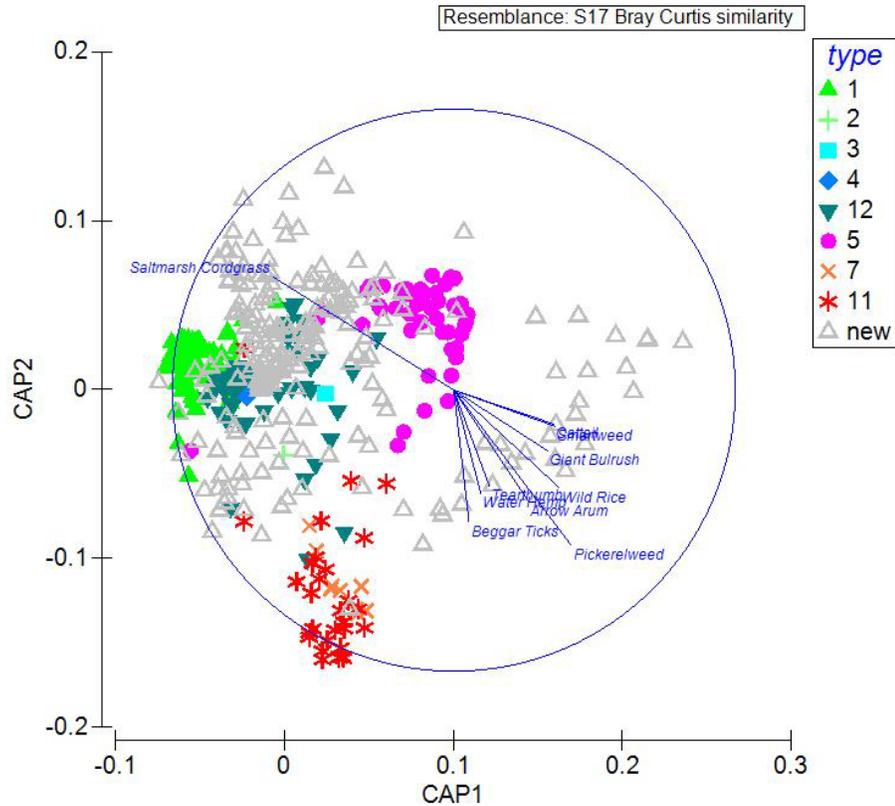
alterniflora, Saltmarsh cordgrass) and fresh (*P. virginica* Arrow arum, *P. cordata* Pickerelweed, *Zizania aquatica* Wild rice) marsh plant communities.

Figure 3-4. Changes in some common York River plant community species.



Each error bar is constructed using 1 standard error from the mean.

Figure 3-5. CAP results from community type analysis. Vectors show species with >0.35 correlation.



Choice of m: 10

Cross-validation showed:

Total correct: 217/264 (82.197%)

Mis-classification error: 17.803%

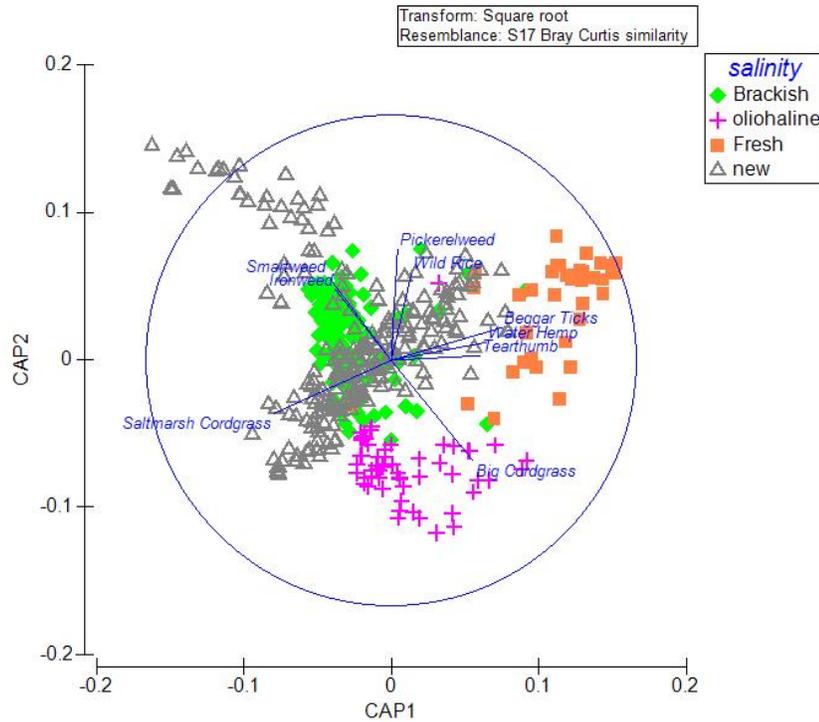
PERMUTATION TEST

trace statistic ($\text{tr}(Q_m'HQ_m)$): 2.88013 P: 0.0001

first squared canonical correlation (δ_1^2): 0.88694 P: 0.0001

No. of permutations used: 9999

Figure 3-6. CAP results from salinity analysis. Vectors show species with >0.35 correlation.



m=8

Cross-validation showed:

Total correct: 248/264 (93.939%)

Mis-classification error: 6.061%

PERMUTATION TEST

trace statistic ($tr(Q_m'HQ_m)$): 1.49476 P: 0.0001

first squared canonical correlation (δ_1^2): 0.78865 P: 0.0001

No. of permutations used: 9999

In the Salinity categorization, *Spartina cynosuroides* (Big cordgrass, dominant in oligohaline communities) was also an important driver. In both categorizations, the less common freshwater species were also important drivers of categorization. Some of these were minor components of the community; for example, *Polygonum virginianum* (Smartweed) and *Vernonia gigantea* (Ironweed) were never more than 5% of the plant community but were likely emphasized because of the square root transformation, which increases the influence of rare species. Similarly, *Amaranthus cannabinus* (Water hemp) and *Polygonum arifolium/sagittatum* (Tearthumb) were never more than 10% of the plant community.

Most marsh plant communities (148 based on type) did not change significantly between the between historic and current surveys (Table 3). However, there were some areas where changes seemed to be concentrated. Indications of increased inundation were more prevalent along the north shore of the river in the brackish region and in the upper portion of the Pamunkey and Mattaponi rivers (Figure 7). Indications of increased salinity were most prevalent in the Pamunkey and Mattaponi rivers, though closer to the mouths of the rivers than the indications of inundation. In this area, Saltmarsh cordgrass appears to have an increasing presence. On the Mattaponi River, the historic data shows little Saltmarsh Cordgrass north of the bridge at West Point, while in the current survey; it can be commonly found almost the entire surveyed length. On the Pamunkey River, marshes which were previously a mix of communities (including freshwater species, *Z. aquatica* and *P. virginica*) are now almost entirely brackish marshes (*S. alterniflora* and *S. cynosuroides*). The salinity analysis identified additional salinity shifts on the upper south side of the York River (Figure 8), where Big Cordgrass communities have been

shifting to Brackish Water Mixed and Saltmarsh Cordgrass communities. Since Brackish Water Mixed communities can include *S. cynosuroides* (Big cordgrass) it is difficult to categorize this change as definitive of a particular process. However, it does indicate some diversification of the plant communities in this area.

Table 3-3. Changes in community over time identified in by CAP. Bolded numbers indicate the number of marshes where community type did not change between surveys. a) from the community type analysis, b) from the salinity analysis.

a)

		Current community type							
		Saltmarsh cordgrass	Saltmeadow	Black needlerush	Saltbush	Big cordgrass	Arrow-arum/ Pickerel weed	Freshwater mixed	Brackish water mixed
Historic community type	Saltmarsh cordgrass	79	0	1	1	2	0	0	23
	Saltmeadow	0	0	0	0	1	0	0	
	Black needlerush	1	0	0	0	0	0	0	
	Saltbush	1	0	0	0	0	0	0	
	Big cordgrass	7	0	0	1	28	1	3	23
	Arrow-arum/Pickerel weed	0	0	0	0	0	7	1	1
	Freshwater mixed	0	0	0	0	2	11	11	5
	Brackish water mixed	25	0	0	0	2	1	0	24

b)

		2010 marsh grouping		
		Brackish	Oligohaline	Fresh
Historic marsh grouping	Brackish	152	8	1
	Oligohaline	30	29	4
	Fresh	4	4	30

Figure 3-7. Mapped Community types identified from CAP analysis. Large dots are Historic, small dots are 2010.

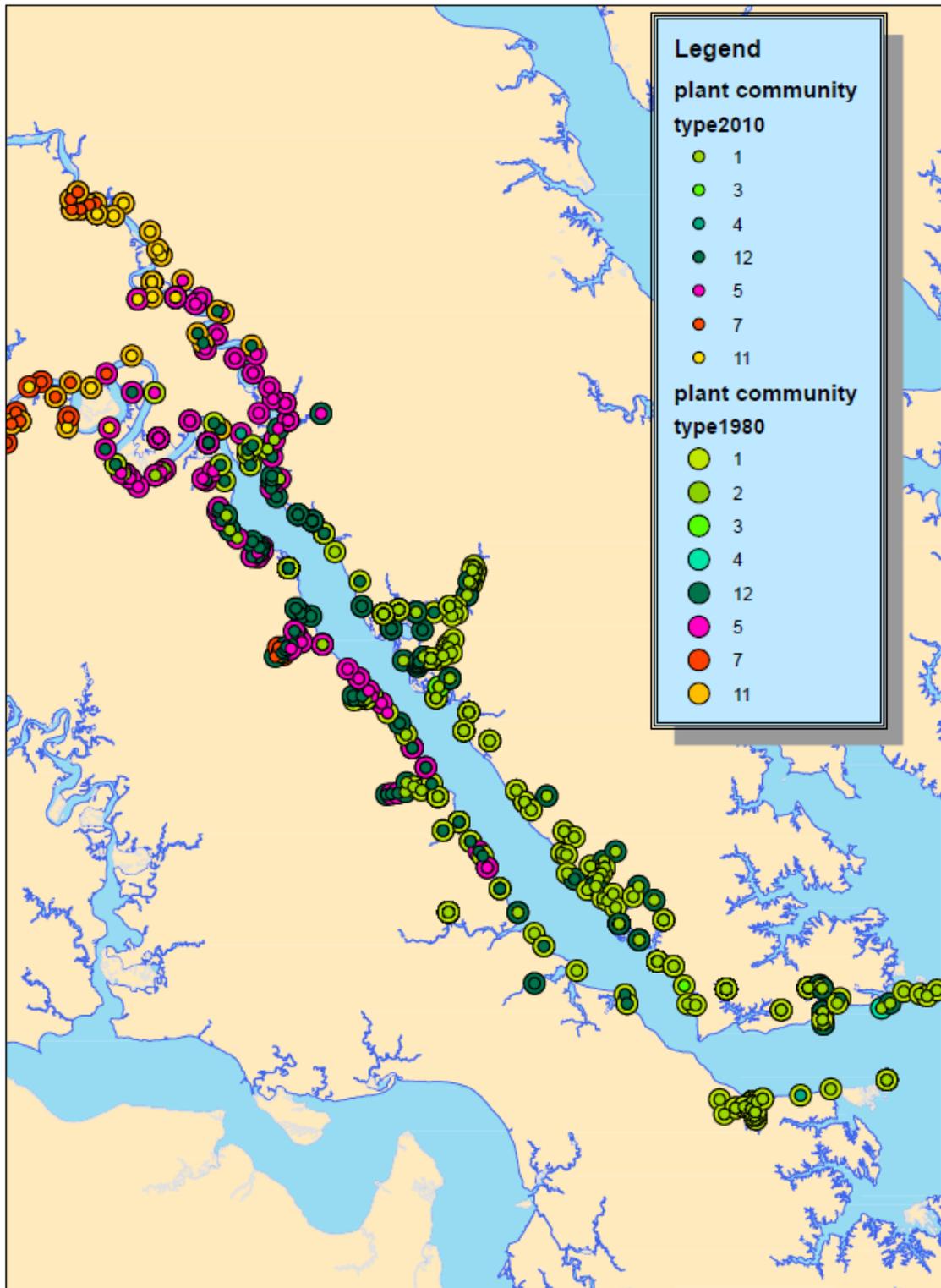
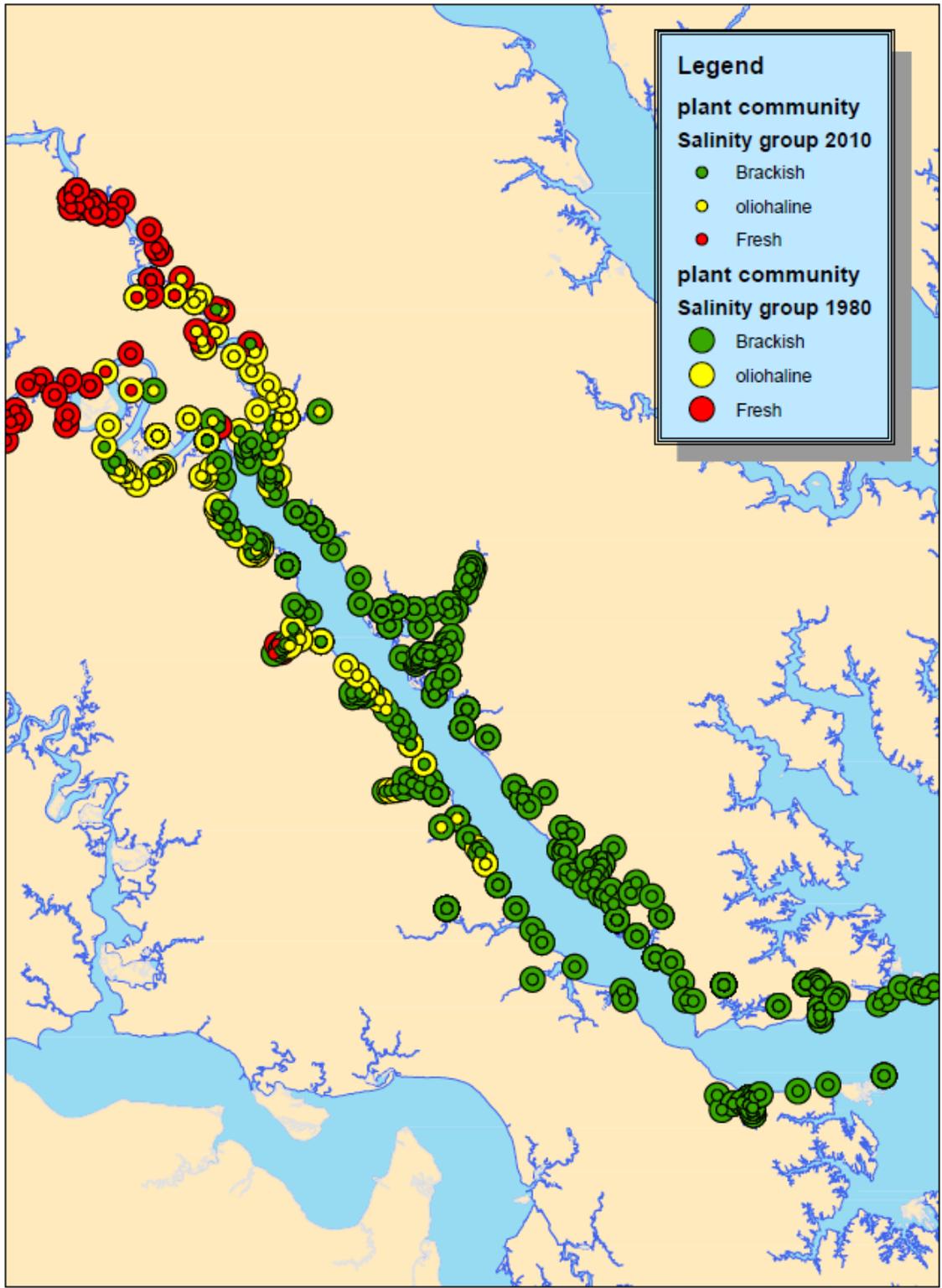


Figure 3-8. Mapped salinity designations identified from CAP analysis. Large dots are Historic, small dots are 2010.



Changes in low marsh plant community extent

Indications of increased marsh inundation can be seen in the York River where areas that historically had significant high marsh community (40-50%) have converted to almost entirely low marsh. The change in low marsh extent was spatially variable, ranging from a large loss of low marsh vegetation (declined by 75%) to a large gain in the percent marsh covered by low marsh vegetation (increased by 100%) (Figure 9). Although variation was high, percent change in low marsh was significantly different (ANOVA, $F=9.0106$, $DF=191$, $p=0.003$) between creek (mean= 11.7% increase, stdev= 25.8) and riverfront marshes (mean= 1.5% decline, stdev= 34.9), with more gain in creek marshes and more loss in riverfront marshes. This suggests that erosion is the driving factor for declines in low marsh extent, since erosive energy tends to be higher on the river than in the creeks. In the low energy creeks, increases in low marsh area suggest that inundation is occurring.

Contrary to our hypothesis, % organic matter was found to be poorly correlated with changing low marsh coverage (Figure 10). The highest values of % organic matter were found in marshes with increasing low marsh widths, although variability was high. The three marshes with the highest % organic matter (29, 35 and 42%) had increases in low marsh extent of 45, 20, and 10%, respectively. However, these three marshes were all adjacent to each other and found in a watershed with high agricultural land use. Therefore, the high organic matter may be reflective of land-based inputs.

Figure 3-9. Changes in low marsh extent on the York River, VA.

Changes in low marsh extent between historic and current surveys

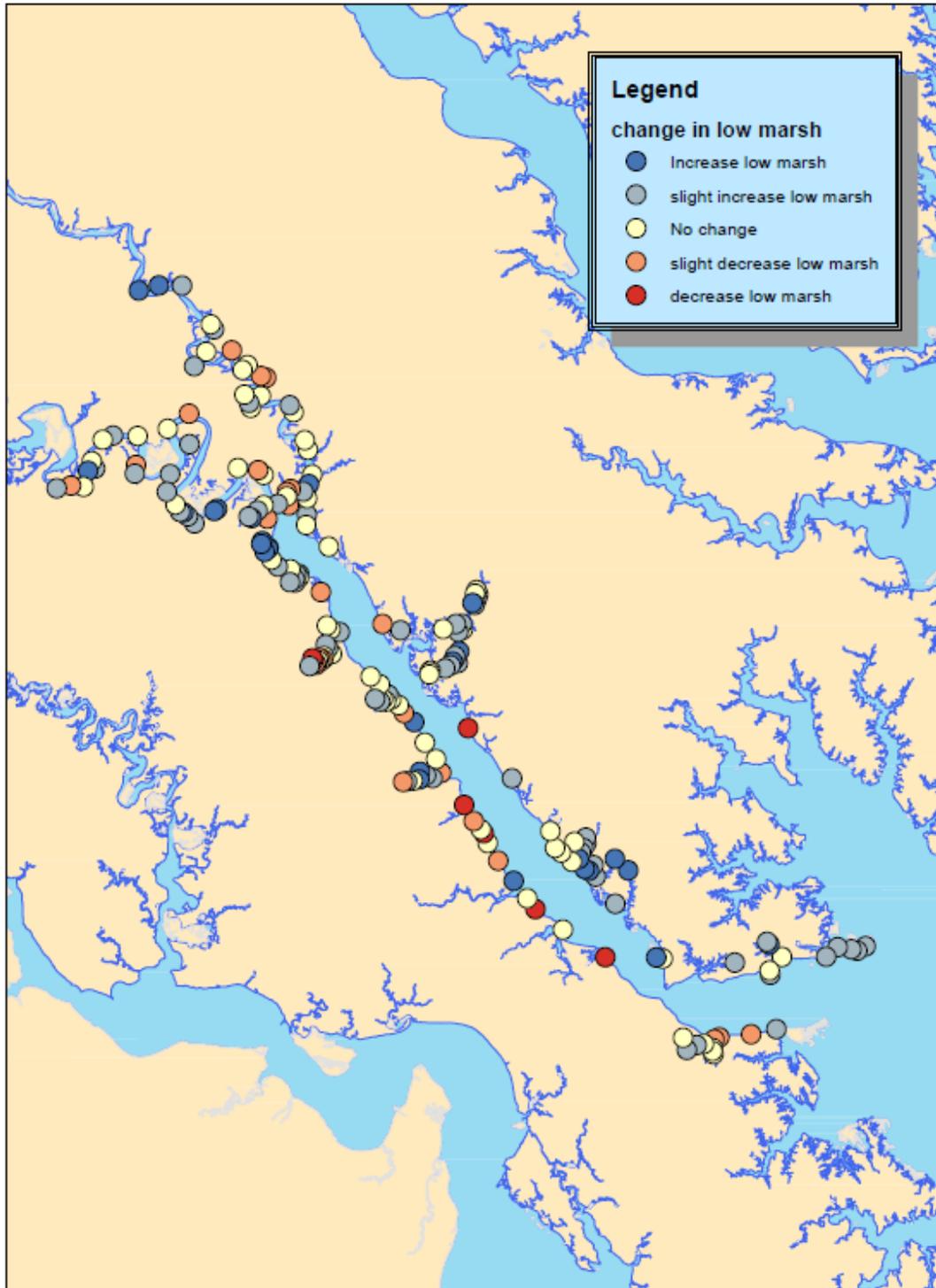
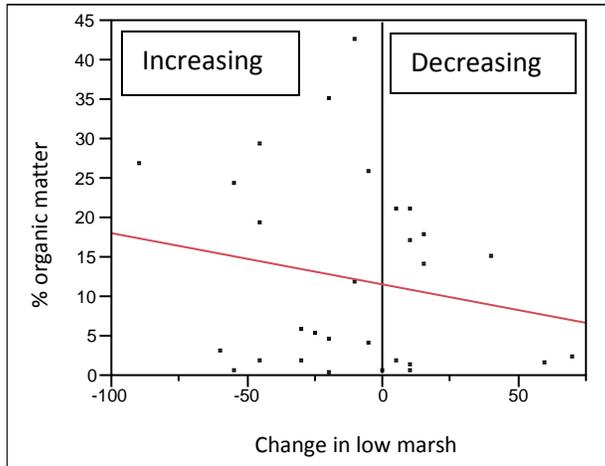


Figure 3-10. Graph of the relationship between % organic matter and the change in low marsh extent. The line indicates the break between increasing and decreasing low marsh extent.



Another possibility is that marsh morphology may be a confounding factor in this analysis. Samples were taken from the center of the marsh, regardless of marsh width. Extensive marshes are less likely to accrete inorganic sediment in the center of the marshes than narrow fringe marshes, so standardizing sampling may improve this metric.

Change in community type (Chesapeake Bay, VA)

Marsh plant community change was seen in 51% of marshes surveyed in both time periods (Table 4). Community changes suggesting increased inundation, increased salinity or *P. australis* invasion were seen in 18% of marshes surveyed in both times. The most common of these changes was increased inundation, which accounted for about 12% of the marshes with altered community types. Signals of increased salinity were rarer, although freshwater marshes at the tops of shallow creeks were under-represented due to the difficulty of accessing them for the surveys. In 34% of marshes, community changes were non-conclusive (not indicative of a particular driver). Only 4% of marshes (724 marsh plant communities) showed a change from a previous community type to *P. australis* (Reed grass), however, there were approximately 1080 marshes in the current survey dominated by *P. australis*. The overall distribution of *P. australis*-dominated marshes is under-represented in the change analysis because approximately one third of them were not associated with a previously typed historic marsh and therefore were excluded from the analysis (per the Methods section, only marshes with surveys done in both time periods are included in the analysis).

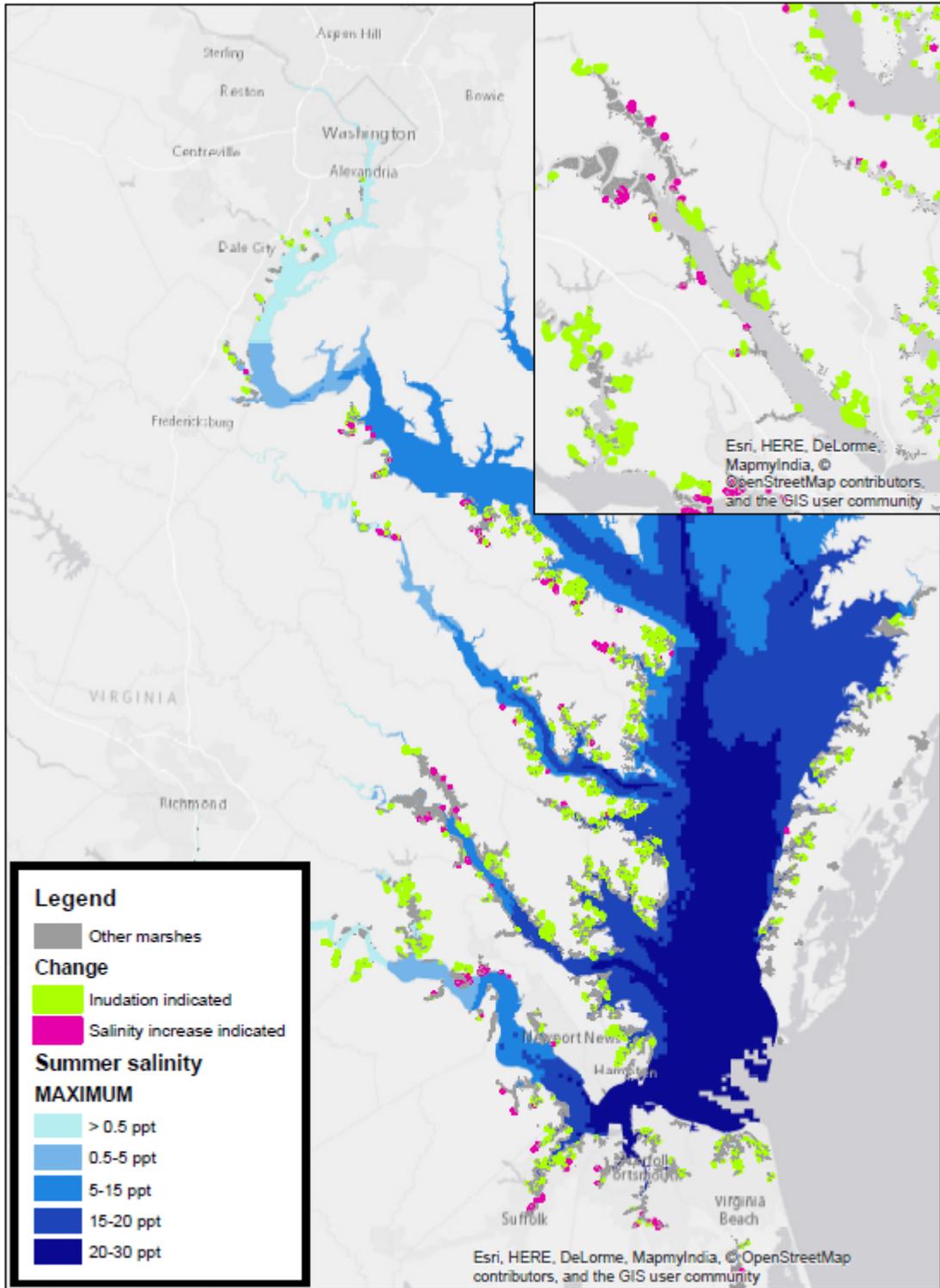
Inspection of the spatial dataset indicates that indications of increased inundation or salinity were mostly seen in tidal creeks, although instances could be found in tributary rivers (Figure 11).

Table 3-4. Change in community types in marshes surveyed in both years. Grey boxes indicate no change in community type.

Historic plant community	Current plant community										
	Saltmarsh Cordgrass	Saltmeadow	Black Needlerush	Saltbush	Brackish mix	Reed Grass	Big Cordgrass	Arrow Arum – Pickerel Weed	Yellow Pond Lily	Cattail	Freshwater mix
Saltmarsh cordgrass	5782	22	362	158	3149	386	103	1		14	10
Saltmeadow	200	4	17	14	236	29	4			2	
Black needlerush	206	8	214	16	368	24	11			6	8
Saltbush	114		8	28	162	14	4			6	
Brackish water mixed	1166	11	246	70	1534	167	169	26		20	29
Reed grass	5				5	9	3				
Big cordgrass	131	2	4	5	319	65	379	4		6	99
Arrow arum-Pickerel weed	3				8	2	15	198	4	7	142
Yellow pond lily								13	24	1	23
Cattail	15		1	4	24	16	18	3	1	19	66
Freshwater mixed	7		3		24	21	49	286	54	25	418

	Total	Percent
Did not change	8609	49%
Salinity increase indicated	308	2%
Inundation indicated	2065	12%
<i>P. australis</i> invasion	724	4%
Non-conclusive change	5952	34%

Figure 3-11. Sea level rise signals indicated by community change throughout the study area. Inset shows same information for the York River.



Increased inundation is indicated in marshes along the length of many tidal creeks and some extensive marshes, particularly on the York River, Chickahominy River, and Back River. Indications of increased salinity were found at the upper reaches of the tidal creeks (where freshwater communities are more common) as well as riverine marshes on the James, Mattaponi, Pamunkey and Rappahannock Rivers near the transition from brackish to freshwaters. Bayfront marshes typically did not show signs of increased inundation or salinity. A change from any community type to a *P. australis*-dominated community (suggesting invasion) was seen throughout the surveyed areas. Although still relatively rare on Bayfront marshes, this change was more likely to be seen than those indicative of increased inundation or salinity.

Discussion

Shifts in vegetation patterns are an early signal of sea level rise-driven impacts to marshes. They can highlight marshes at high risk of drowning and disappearance and show where salinity intrusion is beginning to affect the community. The change in sea level over the period of examination was relatively small, approximately 15-20 cm, but shifts in communities were still evident. The expected rise over the next 30 years is nearly three times that (Boon and Mitchell 2015). The ability of the vegetation to reflect a small shift in sea level suggests that monitoring of vegetation is a useful sentinel of change, allowing for enhanced projections of sea level rise-driven ecosystem shifts.

Marsh vegetation as an indicator of inundation

When a marsh is considered as a whole (rather than as transects or plots) changes in the extent of low marsh are a key indicator of how marshes are responding. Low marsh plant migration upland has been shown to correlate well with accelerating sea level rise in a

New England salt marsh (Donnelly and Bertness 2001). In marshes where migration is a viable response, marsh vegetation will migrate, with little change in the overall balance between low and high marsh. If the marsh cannot migrate landward, the marsh will get smaller and the high marsh will be squeezed against the land. Therefore, in marshes that are diminishing primarily due to sea level rise, the percent of low marsh coverage will increase over time. In marshes diminishing primarily due to erosion, the percent of low marsh should decrease as the low marsh erodes away, leaving only the high marsh platform. In the York River, examination of both the extent of low marsh and the change in community type suggested areas of increased inundation and erosion that were fairly consistent between analyses, with erosion dominating on higher energy river shorelines and inundation dominating in creek systems.

Patterns of change seen in this analysis suggest that some marshes are becoming more inundated, allowing inundation-tolerant plants to become dominant in those marshes. The location of these changes predominantly on tidal creeks (not on the Bay-front and typically not on the riverfronts) may indicate a lack of sediment in the creek systems. Marshes are capable of maintaining their elevation in a changing tidal envelope by capturing sediment used to raise the marsh surface (Van Wijnen and Bakker 2001, Pethick 1991, Kirwan and Murray 2007). When sediment supplies are not adequate to compensate for rate of rise in sea level, increased inundation periods on the marsh surface force shifts in the plant communities and loss of the front edge of the marsh through conversion to unvegetated tidal flats (Schile et al. 2014). A certain portion of the sediment for marsh accretion can come from the erosion of the front edge of the marsh, a process that is likely more important on the high energy Bay-front and river front

marshes. Erosion in tidal creeks is relatively minor and may hamper the capacity of those marshes to keep pace with sea level rise. In addition, some areas may be overwhelmed by a combination of sea level rise and their exposure to Bay winds. Indications of inundation in the marsh plant community at the head of Back River are consistent with an increase of flooding in the adjacent upland areas, where recent FIRM maps show this as 1% Annual Chance Flood Hazard Increase zone (FEMA 2014). Back River is west of the mouth of the Bay; nearly open to the Atlantic Ocean. Along the Mid-Atlantic coast, there is evidence that winds have shifted almost directly westward since 1950, and that this shift is strongly correlated with increasing coastal mean sea level (Woodworth et al. 2014). Ongoing westward winds could cause water to pile up in the Back River, increasing inundation periods in the marsh and leading to plant community shifts. In all tidal creeks and the Back and Chickahominy Rivers, there is also the possibility that some increased inundation is due to increased groundwater and overland flow (freshwater contribution) linked to increased precipitation intensity (NCA; Melillo et al. 2014) and changes in land use patterns. Elucidation of the importance of sea level rise as a factor driving these plant community changes will require additional study in the areas where change has been identified.

Marsh vegetation as an indicator of salinization

The spatial distribution of community shifts indicating salinization are compellingly found predominately around the 0.5 – 5 psu point in the tributary rivers. These are exactly the areas that are hypothesized to be undergoing change from sea level rise, but these changes have only been documented in the York River (Perry and Hershner 1999) where salinization has been exacerbated by high groundwater withdrawal rates. Although

shifting salinity regimes are a direct consequence of sea level rise, they can also be somewhat mitigated and/or exacerbated by anthropogenically-moderated increases or decreases in freshwater flows (e.g., through water withdrawals or increases in impervious surfaces) and changes in precipitation. Alterations to the freshwater flow could potentially mask the signals of salinization, but the presence of change in all tributary rivers suggests that this is a minor concern.

Arguably, the smaller freshwater systems (creeks and small rivers) that come off the mainstem tributaries are at greater risk from shifts in salinity because they have a much smaller freshwater component to mitigate the rise in salinity and smaller watersheds with less capacity to collect precipitation to recharge their groundwater. Although there are some indications that vegetation has shifted in these marshes between the two TMI surveys, the mapping approach used in this analysis was insufficiently resolved to pinpoint the shifts in salinity in communities. These areas are prime targets for enhanced monitoring.

Phragmites australis expansion in Chesapeake Bay, VA

P. australis expansion seen in this study is similar to that seen in the northern portion of the Chesapeake Bay (MD), where it was relatively rare in the 1970s but increased to 25 times the aerial coverage by 2007 (McCormick et al. 2010). However, the increase in the Virginia portion of the Chesapeake Bay has been less extensive than in Maryland, possible due to the higher salinities in the mainstem Bay. In Virginia, *P. australis* is more commonly found on riverfront and creek shorelines. However, it has appeared on some high salinity Bayfront shorelines. The recent spread of *P. australis* into more saline waters has been theorized to be due to increased salt tolerance by a non-native (invasive)

haplotype (Vasquez et al. 2005). In Virginia, the highest concentrations of *P. australis* dominated communities are found on the northern peninsula. It is possible that the spread of the invasive haplotype began in that area and migrated southward; however, the historic data does not support that. It is more likely that lower salinity levels in the northern region are more conducive to its spread. Invasion has also been linked to urbanization (King et al. 2007), which is supported by the distribution of *P. australis* in the southern portion of the Bay, but does not explain the concentration of *P. australis* the relatively rural Westmoreland and Northumberland counties in the northern portion of the Chesapeake Bay, VA.

The primary concern with *P. australis* is the replacement of native habitats (low-moderate diversity plant communities) with a monotypic, species. In this analysis, *P. australis* replaced all community types except Yellow pond lily, which is the most tolerant of long inundation periods. It most commonly replaced Saltmarsh cordgrass and Brackish water mixed community types. Replacement of native community types with *P. australis* has been suggested to negatively impact nekton populations (Able and Hagan 2003), epifaunal species (Robertson and Weis 2005), and birds (Benoit and Askins 1999). In addition, high rates of sediment trapping associated with *P. australis* can fill marsh channels, reducing access to the marsh surface for aquatic organisms (Chambers et al. 1999). Despite being a non-native plant species, invasive *P. australis* still provides many ecosystem services. Although problematic for plant species and certain animals, for many animal species *P. australis* invasion may be considered a neutral habitat shift (blue crabs, Long et al. 2011; general macrofaunal density, Osgood et al. 2003; Posey et al. 2003). Compared to natural marshes, *P. australis* provides equivalent or better water

quality services, removing nutrients, particulates and some heavy metals (Chambers et al. 1999). In addition, it has been suggested that *P. australis* marshes have an enhanced capacity to accrete sediment relative to other marsh plants (Rooth and Stevenson 2000) potentially raising marsh elevations to keep pace with accelerated rates of sea level rise.

Confounding factors for interpreting community shifts

Sea level rise brings simultaneous increases in inundation period and salinity. This analysis addresses them as separate processes because the vegetative responses tend to be more attributable to one of the two drivers of change. This is a simplification since both drivers co-occur. One of the most common changes between TMI surveys was a change to Saltmarsh cordgrass-dominated community from any other community type. This was interpreted as increased inundation when the historic community type was a brackish or oligohaline community type and salinization when the historic community type was fresh. However, *S. alterniflora* (the dominant species in the Saltmarsh cordgrass community type) is particularly tolerant of both long inundation periods and salinity, but requires neither. A change from a Big cordgrass community to a Saltmarsh cordgrass community could indicate either increased inundation or increased salinity (as stated in the analysis) or both simultaneously. Distinguishing between the two drivers is not possible, even with the species matrix, and would require long term monitoring of water levels and salinity.

Freshwater marsh plant community are characterized by high diversity (both annual and perennial species) resulting in high interannual variability in community composition. Although average salinity in these marshes is less than 0.5 psu, annual salinity can shift between years of high and low precipitation, with communities responding quickly to

change (Bilkovic et al. 2012). Perennial species (such as *S. alterniflora*) are less influenced by inter annual variation and area considered to be a good indicator of long-term trends; conversely, annual species can respond within months to an increase in salinity (Wetzel et al 2004), but will also disappear if there is a low salinity year, making them poor indicators of change in a system. *S. alterniflora* that grows in a freshwater marsh during a high salinity year, is likely to persist in low salinity years. Therefore small percentages of *S. alterniflora* may be indicative of change or may be an artifact of past conditions. This highly variable plant communities means that the species matrix in any given year is a questionable indicator of average community composition. In this analysis, we chose to focus on community types. The advantage of using community types in freshwater marshes is that they require substantial shifts in the community before a change is evident. This means that they might be missing subtle changes in community, but they are unlikely to be overestimating salinization.

The utility of patterns in community change

Marsh plant community typing is a quick, high level characterization of the plant community predominately based on dominant vegetation. It distinguishes ecologically different communities rapidly and can easily be applied on extensive spatial scales. CAP analysis shows that it is a robust method for quickly capturing the critical species in a tidal marsh plant community. However, for temporal comparisons, it misses some of the nuanced changes that might be indicative of changing processes. For example, a community that was 60% *S. alterniflora*, 40% *S. patens* in the historic survey would be typed as a Saltmarsh cordgrass-dominated community. If the same marsh was 100% *S. alterniflora* in the second survey, the community type would be the same—despite the

increased inundation indicated by the expansion of *S. alterniflora*. Results from this study suggest that community typing can be used to monitor change on a large scale, but care should be taken when interpreting the results to keep from overextending conclusions.

Although community changes were classified only as those suggesting inundation, salinity increase or *P. australis* invasion, it is important to note the most common community change was a shift from Saltmarsh cordgrass in the historic survey to Brackish water mixed in the current survey. The second most common shift was the opposite pattern, Brackish water mixed in the historic survey to Saltmarsh cordgrass in the current survey. This second shift was considered an indicator of inundation; however, Brackish water mixed communities frequently contain a border of Saltmarsh cordgrass and Saltmarsh cordgrass communities sometimes contain an upper marsh (with a mixture of plants), making the distinction between them reliant on whether *S. alterniflora* is dominant (>50% cover). Visual inspection of the distribution of each change suggests that the Saltmarsh cordgrass to Brackish water mixed change tended to be downstream of the Brackish water mixed to Saltmarsh cordgrass, where the wave energy would be higher, suggesting the possibility that this change is capturing erosion of the *S. alterniflora* boarder, resulting in a decline in dominance.

Despite some limitations, the utility of community typing for elucidating patterns and targeting areas for research is promising. Traditional monitoring techniques (such as tide gauges and groundwater wells) are typically limited in spatial scope, restricting the breadth of inference that can be logically made and potentially missing critical shifts in non-monitored marshes. Patterns of community change, such as the apparent salinity

increases near the freshwater-brackish water interfaces on the tributary rivers, can be used to target areas for monitoring. Detailed monitoring of marsh plant communities in these areas and the marshes immediately upstream may allow early detection of freshwater to brackish marsh and an enhanced understanding of resultant processes (e.g., increased carbon mineralization; Weston et al. 2006; reduced N sequestration and denitrification; Craft et al. 2009). The expansive spatial change analysis means that results from targeted marshes can be more confidently extended to other marshes undergoing the same shifts.

Conclusions

There is a critical need for the ability to observe and predict changes in estuarine ecosystems. Sea level rise causes non-linear changes in hydrodynamics, leading to changes in sediment transport and ecological processes (Passeri et al. 2015), which will affect the signal of change in shoreline systems. This non-linearity means that signals of change may be muted until sea level rise acceleration passes a critical threshold. Marshes (as measured by extent) appear to show a threshold effect related to sediment supply in relation to sea level (Kirwan et al. 2010). Up to some inundation frequency, marshes will accrete sediment to keep pace with sea level (i.e. no discernable signal) and beyond that frequency should begin to drown. Therefore the effect of accelerating sea level will not be apparent until it has crossed the threshold, and then for a short period of time there will be a relationship between changing water levels and marsh loss, followed by total marsh loss. This effect will be more evident in microtidal systems, such as the Chesapeake Bay, because the changes in sea level will be a larger proportion of the tidal range (Friedrichs and Perry 2001). Marsh vegetation is a flexible measure of inundation

and salinization that responds to accelerating sea level rise through shifts in plant distributions (Donnelly and Bertness 2001). Therefore, it should provide an early signal of change, preceding marsh loss. In this study, we observed signals of inundation and erosions reflected in marsh community change that might proceed more extensive marsh loss. We also saw indications of salinization in the transition from salt to freshwater. Understanding the spatial distributions of these patterns of vegetative change should enhance our understanding of future marsh changes and the ecosystem consequences of those shifts.

Chapter 4 Soil sulfides in transitioning headwater-brackish marsh systems

Abstract

In temperate estuarine systems a salinity gradient occurs along the length of each contributing tidal creek culminating in a headwater-brackish marsh complex. These systems are vulnerable to sea level rise and salinity intrusion that will change dominant plant communities, soil characteristics and ecological roles. In this study, we examine the distribution of reduced sulfur compounds in wetland soils occupying these transitioning communities. Sediment sulfur content is higher in salt marshes and may be a primary indicator of marsh community change in response to marsh salinization. Reduced sulfur concentrations varied significantly along the salinity gradient, with similar concentrations in headwater and freshwater marsh locations while brackish marsh locations had higher concentrations.

Phragmites australis patches were found at six of the headwater-brackish marsh systems, sometimes above and sometimes below the emergent freshwater marsh. Reduced sulfur concentrations were high in one of the *P. australis* patches suggesting that neither salinity nor sulfur was a controlling factor in the *P. australis* colonization and persistence. As sea level rise accelerates, these upper creek communities will become increasingly vulnerable to salinity intrusion. This study shows that brackish marsh plant communities are good predictors of salinity intrusion and sulfur sequestration in headwater-marsh complexes. Monitoring these plant communities should allow for detection of headwater vulnerability to sea level rise-driven salinization. However, in areas with significant *P.*

australis communities, the lack of change in the vegetation may mask encroaching salinity, requiring monitoring of sediment sulfur content instead.

Introduction

In temperate estuarine systems a salinity gradient occurs not only along the mainstem of the estuary, but also along the length of each contributing tidal creek. Wetlands along the salinity gradient in those creeks are microcosms of the larger estuarine systems; typically beginning near the headwaters with non-tidal, groundwater-fed, forested wetlands, then transitioning through emergent tidal freshwater marshes and culminating in brackish marsh communities. The headwater-marsh complexes are vulnerable to sea level rise and salinity intrusion that will change dominant plant communities, soil characteristics and ecological roles. Changing salinity results in shifts in plant communities that affect the ecosystem function of marshes, including production and decomposition rates (affecting carbon sequestration), habitat type, and supported fauna (White and Kaplan 2017, Bilkovic et al. 2012). Sea level rise-driven shifts in salinity distributions are predicted to have a bigger immediate impact on tidal marsh community distribution than increased inundation, since even relatively small shifts in salinity can lead to shifts in dominant species, with freshwater marshes being replaced by brackish marshes (Callaway et al. 2007).

Sea level is rising particularly quickly in the Chesapeake Bay (Boon, 2012; Ezer et al. 2013; Sallenger et al., 2012; Kopp, 2013, Boon and Mitchell 2015) resulting in salinity intrusion (Hilton et al. 2008). Models suggest that rising tides and increased salinity will be an increasing problem under projected sea level rise, particularly in drought years (Hong and Shen, 2012; Rice et al. 2011). In the York River, Virginia, tidal influence in

some creek systems appears to have migrated towards the headwaters over the past 40 years, concurrent with rising water levels (Mitchell et al. 2017). Associated salinity intrusion may impact the ability of tidal freshwater wetlands to accrete organic matter, hampering their ability to keep pace with sea level rise (Neubauer 2013). Sediment sulfur content and plant communities are useful indicators that can be monitored to assess the vulnerability of these systems to sea level rise. However, this requires a baseline understanding of sediment sulfur distributions along the headwater-marsh complexes and their relationship to the plant communities.

Salt stress is known to reduce plant productivity through numerous pathways, including reducing nutrient uptake and salt toxicity (Parida et al. 2005, Poljakoff-Mayber 1988, Percy and Ustin 1984, Greenway and Munns 1980). Plant growth is reduced under increased salinity and spatial patterns of plant species shift in response to salinity, increased inundation and a combination of the two stressors (Janousek and Mayo 2013). In the Mid-Atlantic region, freshwater and brackish marsh communities are typically composed of entirely different plant species, allowing them to be distinguished easily. Brackish marshes typically are dominated by *Spartina alterniflora* (Saltmarsh cordgrass), which is tolerant of a wide range of salinity. Freshwater marshes are more diverse, but are often dominated by *Typha latifolia* (Cattail), *Peltandra virginica* (Arrow Arum), *Pontedaria cordata* (Pickerelweed) and *Leersia oryzoides* (Rice cutgrass) none of which have a high tolerance for saline conditions. Salinity intrusion into freshwater areas results in demonstrable community shifts (e.g., Baldwin and Mendelsohn 1998; Perry and Hershner 1999; Wetzel et al. 2004). The accumulation of reduced sulfur compounds in marsh sediments is related to salinity and can be an early indicator of salinity intrusion,

because sulfide can complex with reduced iron in the soil, leaving a signature (pyrite-bound sulfur) even if the intrusion is only periodic. In iron-rich sediments, like those of the mid-Atlantic Coastal Plain, frequent tidal inundation and sulfate-rich seawater increase both organic sulfur and pyrite-bound sulfur sequestration (Hussein and Rabenhorst 1999). Therefore, flooding patterns are expected to be the explanatory variable for variations in reduced sulfur concentrations along a tidal gradient.

In this study, we examine the distribution of reduced sulfur compounds in wetland soils occupying the transitional space between forested headwater systems and brackish estuarine marshes. Sediment sulfur content is higher in salt marshes and may be a primary indicator that the cause of marsh plant community change includes salinization. We hypothesized that the sediment sulfur content would be correlated with the plant community, with higher sediment sulfides in the brackish marsh communities than the freshwater marsh communities. Many of our transitional marsh systems were vegetated by common reed *Phragmites australis* communities, which can be an invasive species. In these marshes we expected lower soil sulfur levels due to the low tolerance of *P. australis* for free sulfide (Chambers et al. 1998).

Methods

The York River estuary is a brackish water tributary to the Chesapeake Bay, Virginia, approximately 64 km long and containing about 38 tributary creek systems along its length. The salinity gradient extends from 20 psu near the mouth of the river to approximately 5 psu at the head of the main estuary (Reay and Moore 2009). In both mainstem and tributary tidal creeks, the York River encompasses a range of wetlands, including freshwater swamps, tidal freshwater marshes, oligohaline marshes and

brackish/salt marshes (CCRM, VIMS 2010). Each marsh type is associated with unique plant communities, driven by salinity and inundation differences. Dominant land cover in York River watershed is natural, typically forested, with some sub-watersheds having high agricultural land use. Developed land use is restricted to the sub-watersheds near the mouth of the estuary and of minor importance overall. Nine tributary creeks exhibiting a gradient from headwater to salt marsh wetland complexes were selected from those along the York River, representing spatial diversity and encompassing representative plant communities and land uses (Table 1).

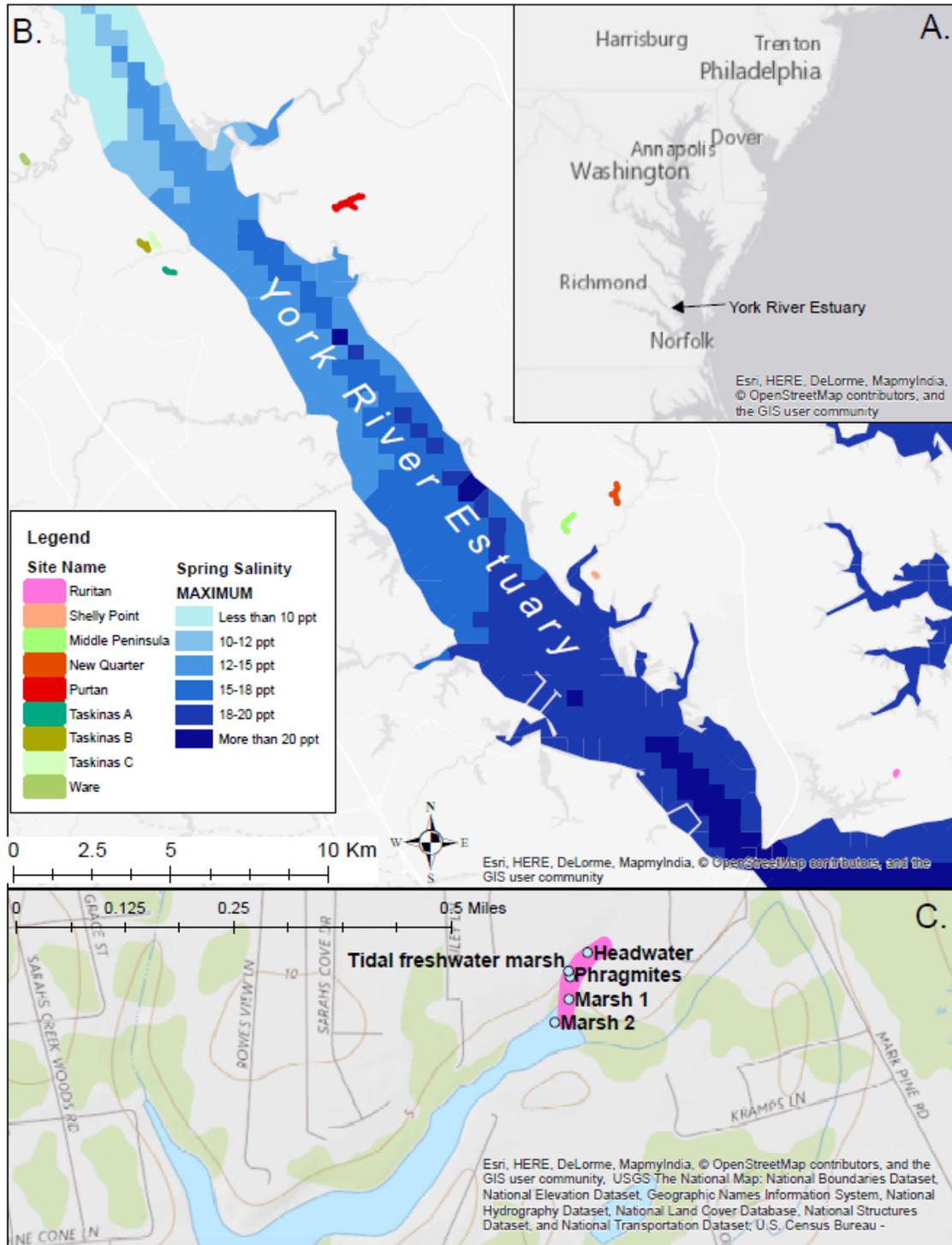
Core collection

Transects were set up at each of the nine sites (Figure 1) extending from the forested headwater through the emergent tidal freshwater marshes and then into the brackish marshes. Cores were taken at four points along each transect: forested headwater, emergent freshwater marsh community, brackish marsh near the freshwater marsh community (marshUP), and brackish marsh several meters downstream (marshDS). Both brackish marsh communities were *S. alterniflora* dominated. At each location, plant community types were identified. A fifth core was taken in *P. australis* community, where present. Cores were taken to 40 cm in depth using a 5cm-diameter PVC corer. Immediately following collection, cores were extruded and subsampled by depth: 0-2 cm, 2-10 cm, 10-20 cm, 20-30 cm, 30-40 cm. Subsamples intended for sulfur analysis were suspended in 1N ZnAc immediately following collection to fix the reduced sulfur compounds (Chambers and Pederson 2006) and then were refrigerated until processing. Subsamples intended for organic matter analysis were thoroughly mixed and refrigerated until processing.

Table 4-1. Site characteristics of the nine headwater-brackish marsh systems used in the study. Dominant land cover is the primary land use in the immediate sub-watershed (Ag = Agriculture, For = Forested).

Site names	Dominant Vegetation				Distance (m)			Dominant land cover
	Headwater	Freshwater marsh	Marsh 1	Marsh 2	Headwater --^	Freshwater Freshwater --^	MarshUP	
Shelly Point	Eastern Red cedar, Southern Red Oak	Lizard tail	Saltmarsh cordgrass	Saltmarsh cordgrass	45	127	Ag	
Middle Peninsula	Eastern Red cedar, Southern Red Oak	Cattail	Saltmarsh cordgrass	Saltmarsh cordgrass	245	195	For	
New Quarter	American Sycamore, Southern Red Oak	Arrow Arum, Cattail, Big cordgrass	Saltmarsh cordgrass, Black needlerush	Saltmarsh cordgrass, Black needlerush	71	63	Ag	
Purtan	Green Ash, Southern Red Oak	Lizard tail, Rice cutgrass, Cattail, Smartweed	Saltmarsh cordgrass	Saltmarsh cordgrass	113	100	Ag	
Ruritan	Green Ash, Southern Red Oak	Cattail	Saltmarsh cordgrass	Saltmarsh cordgrass	41	32	Ag	
Taskinas A	Green Ash, Southern Red Oak, American Elm	Wild rice, Cattail, Smartweed, Wild rice	Saltmarsh cordgrass	Saltmarsh cordgrass	81	48	For	
Taskinas B	Green Ash, Southern Red Oak	Japanese stilt grass, Dog fennel, Big cordgrass	Saltmarsh cordgrass	Saltmarsh cordgrass	124	355	For	
Taskinas C	Pignut Hickory, Southern Red Oak, American Hornbeam	Cattail, Big cordgrass	Saltmarsh cordgrass, Black needlerush	Saltmarsh cordgrass, Black needlerush	328	134	For	
Ware	Green Ash, Southern Red Oak, American Elm	Japanese stilt grass	Saltmarsh cordgrass	Saltmarsh cordgrass	99	2	For	

Figure 4-1. . Map of sampling sites at three scales. A. Map of Mid-Atlantic coastline, showing the location of the York River estuary. B. Map of 9 headwater systems located along the estuary with approximate maximum salinity for the area. Sites in pink/red are surrounded by forested land cover. C. Close-up map of Ruritan site with blue points marking the sampling locations.



Initial sampling was done in May 2017. Samples were taken from the freshwater marsh and two brackish marsh sites (marshUP and marshDS) at three of the creek systems (Shelly Point, Middle Peninsula, and Ruritan). Following analysis of the samples, a second round of sampling in October 2017 was done where headwaters and *P. australis* marshes were added to the headwater-brackish marsh transects. Organic matter samples were collected during the second round of sampling. Potential sampling sites were restricted by availability at some creek systems; therefore six of the nine creeks had *P. australis* marshes and seven of the nine sites were large enough to have two brackish marsh sites (Table 2).

Table 4-2. Dates of sampling for each site and transect location. All locations were sampled in 2017. Locations with “N/A” did not exist at location or were not sampled due to access restrictions.

	Location on transect				
	Headwater	Freshwater marsh	<i>P. australis</i>	MarshUP	MarshDS
Shelly Point	October	May	October	May	May
Middle Peninsula	October	May	October	May	May
New Quarter	October	October	October	October	N/A
Purtan	October	October	N/A	October	
Ruritan	October	May	October	May	May
Taskinas A	October	October	October	October	October
Taskinas B	N/A	October	October	October	October
Taskinas C	N/A	October	N/A	October	N/A
Ware	October	October	N/A	October	October

Sulfur analysis

Total sediment content of reduced sulfur compounds (acid-volatile sulfides and chromium-reducible sulfides) was determined for each subsample using a one-step

extraction sequence (Chambers et al. 1994). Following hot chromium extraction under acidic conditions, sulfides were collected in a 1N NaOH trap, and then analyzed colorimetrically against prepared sulfide standards (Cline 1969). Following analysis at the three sites sampled in May 2017, subsamples were only analyzed to 20 cm in depth (3 subsamples per core), because all cores had high sulfur content below that depth.

Organic matter analysis

Soil subsamples were dried for 48 hours, weighed and analyzed for organic matter using loss-on-ignition (Craft et al. 1991). Total organic carbon (TOC) was determined for the top subsample from each core and was analyzed with a TOC analyzer with a non-dispersive infrared detector (Shimadzu model TOC-5000).

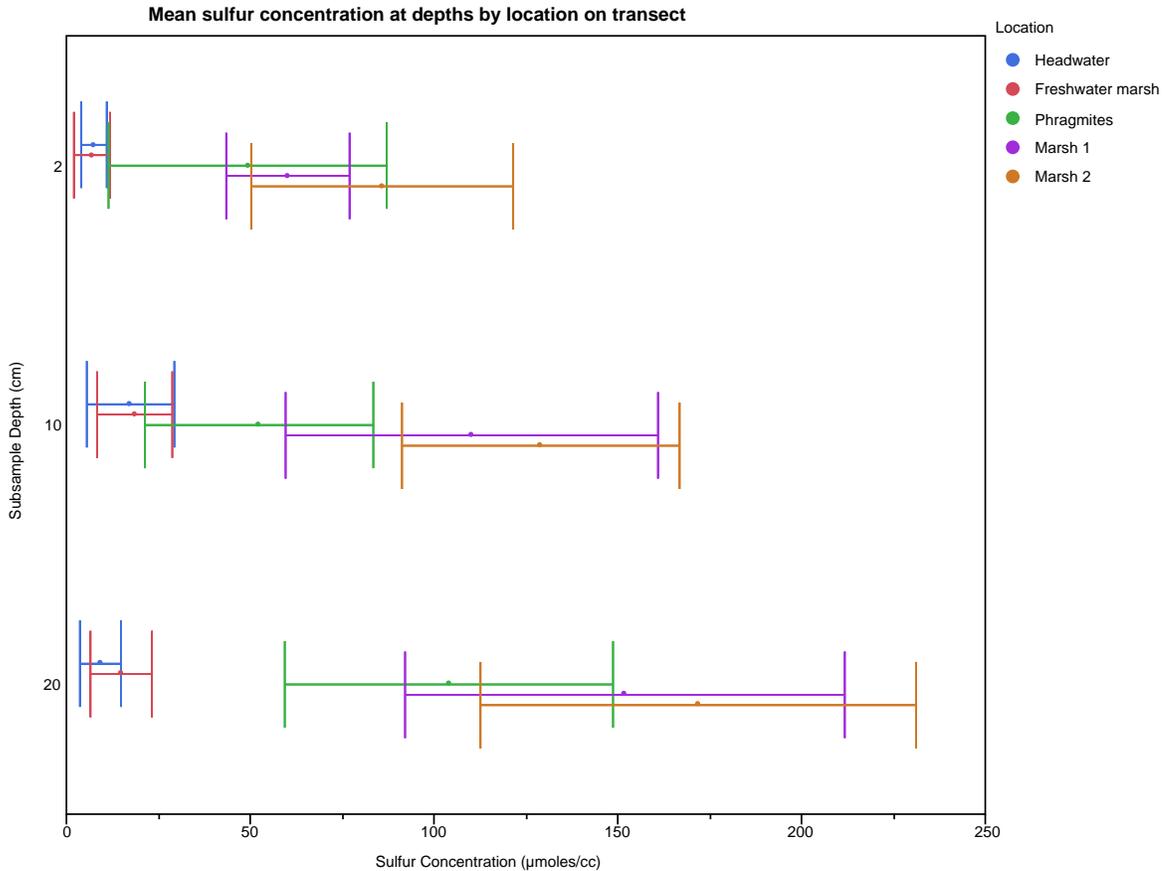
Data analysis

A one-way ANCOVA was used to determine a statistically significant effect of transect location on sediment reduced sulfur content controlling for sampling depth. Type III sums of squares were used to account for uneven sample size. Homogeneity of variances assumption was violated (Levene's Test for Equality of Variances; $p < 0.005$) so sulfur concentrations were transformed using the natural log (\ln). Pairwise comparisons with Bonferroni adjustment for multiple comparisons were used to test between sites. The same analysis was used to compare effect of transect location on sediment organic matter; however, organic matter did not need to be transformed prior to analysis. Sulfur content by sampling month was compared using natural log transformed sulfur content in a two-way ANOVA of location x sampling month. Organic matter was compared to total organic carbon and reduced sediment sulfur concentration using a non-parametric correlation (Spearman's rho). All statistics were performed with IBM SPSS Statistics 24.

Results and Discussion

We hypothesized that the sediment sulfur content would be correlated with the plant community, with higher sediment sulfides in the brackish marsh communities than the freshwater marsh communities. Our results were consistent with this hypothesis; there was a significant effect of transect location on sediment reduced sulfur content after controlling for sampling depth ($F(4,105) = 19.066, p < 0.005$). Headwater and freshwater marsh locations were statistically similar (with very low reduced sulfur content), while *P. australis* and two brackish marsh locations were statistically similar (Figure 2).

Figure 4-2. Sulfur concentration at 5 sites along a headwater-salt marsh transect. Each error bar is constructed using 1 standard error from the mean.



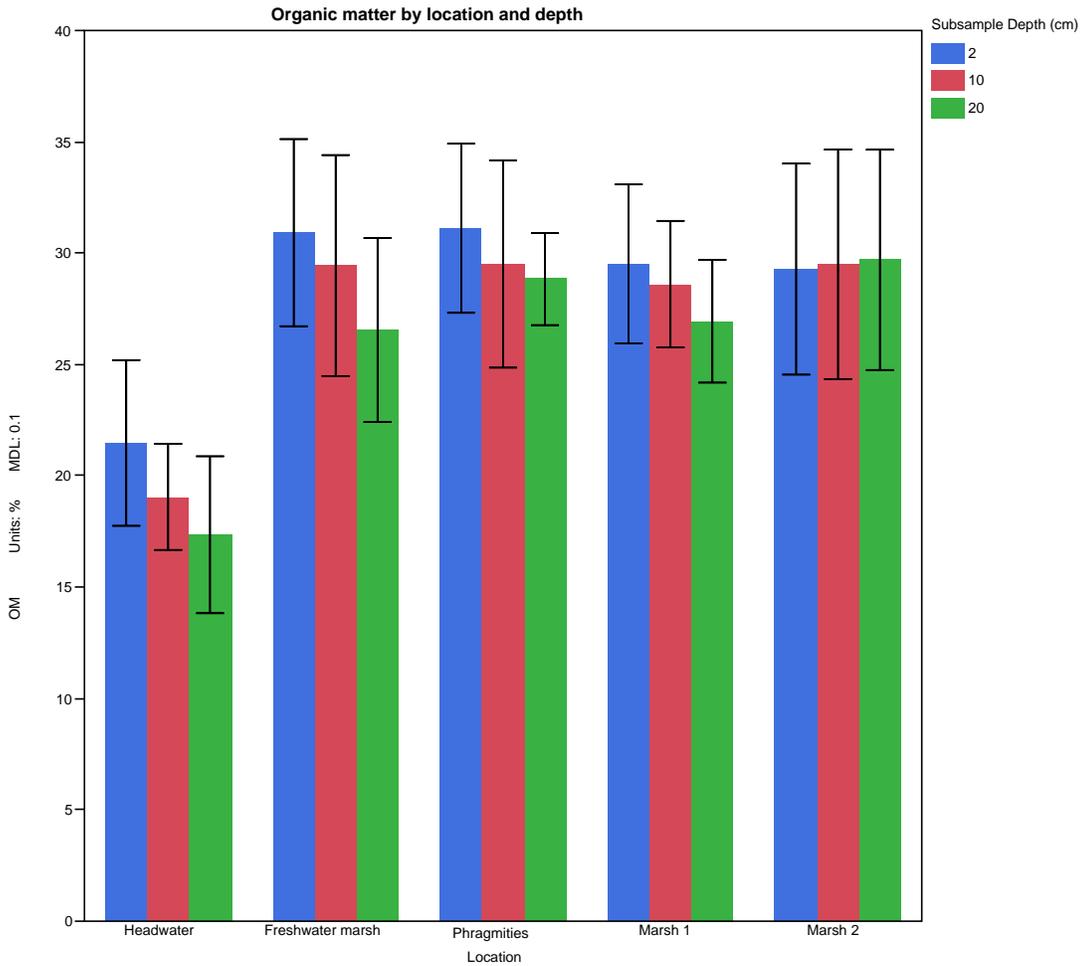
This is consistent with previous studies in tropical estuarine systems (Chambers and Pederson 2006) and with high sulfate content in marine waters which provides the electron sink for bacterial reduction of organic matter in brackish sediments. The covariate, subsample depth, was close to significantly related to the sulfur concentration ($F(1,105) = 3.552, p = 0.062$) in this analysis. Sulfur concentrations generally increased with depth; however, headwater and freshwater sites had low sulfur concentration (i.e., less than $90 \mu\text{mol/cc}$) at all depths. Increases in sulfur concentration with depth might indicate periodic salinity intrusion in deeper layers (with fresh groundwater overlaying the saline waters) but might also result from the decomposition of refractory plant material at depth.

Organic matter as a moderator of sulfur

The organic matter content of a marsh affects the redox potential (Moy and Levin 1991) and microbial remineralization rates in the sediment (Piehler et al. 1998) and may affect sediment sulfur accumulation. Organic matter content was variable both within and between sites. Organic matter soil contribution ranged from 4 – 58% but was typically less than 50%, only exceeding that value at Middle Peninsula, where the whole transect had high organic matter relative to other sites (Figure 3). The mineral-dominated soils are consistent with marsh sediments in the mid-Atlantic coastal plain where organic matter is typically around 30%. The high organic matter at Middle Peninsula is difficult to explain by its surroundings, which are forest-dominated; in contrast to nearby New Quarter and Shelly Point sites that are agriculture-dominated and likely collect organic sediment from adjacent farm fields. In addition, Middle Peninsula is not draining a larger

area than other headwater sites. It is possible that soils are inundated longer at this location, slowing organic matter decomposition.

Figure 4-3. Organic matter in cores by transect and depth. Each error bar is constructed using 1 standard error from the mean.



Organic matter appears to be closely linked to vegetation type (emergent vs. forested).

There was a significant effect of transect location on sediment organic matter after controlling for sampling depth ($F(4,111) = 4.609, p=0.002$). Organic matter was significantly lower in the headwater sites but fairly consistent across all marsh types, regardless of vegetation type. The covariate, subsample depth, was not significantly

related to the organic matter content ($F(1,111) = 1.449, p = 0.231$). Total organic carbon (TOC) was positively correlated to organic matter ($R^2=0.901, P<0.001$), but overall low (mean = 7.5 %) and did not vary significantly with system type. Unlike terrestrial systems, wetlands tend to accumulate large amounts of organic matter (Craft 2000). Along the gradient from headwater wetlands to brackish marshes, the sediment organic matter is expected to increase as the system switches from periodically inundated to a regularly inundated as seen in this study. Areas of emergent freshwater marsh immediately adjacent to the forested headwater systems may be expected to have low sediment organic matter for a marsh system, if it has been migrating into previously forested area and has not yet had an opportunity to build a more organic sediment. In addition, salt water intrusion into tidal freshwater marshes can result in increased microbial decomposition resulting in decreased soil organic carbon (Weston et al. 2011, Chambers et al. 2013, Neubauer et al. 2013). However, in our study, neither organic matter nor TOC was low in the freshwater marshes. This was true even at Ruritan, where the freshwater marsh is visibly invading into forested areas.

Although we hypothesized a link between organic matter and sulfur concentration, they did not vary significantly ($R^2=0.023, P=0.810$) and sulfur content was similar in the headwater and freshwater marsh sites despite the much lower organic matter in the headwater systems. This suggests sulfur availability, not organic matter availability, was the primary driver of sulfur patterns.

Linking vegetation to sediment sulfur content

Our results indicate that vegetative cover (with the exception of *P. australis*-dominated areas) is a good indicator of soil sulfur content, suggesting that the migration of the

brackish marsh vegetation closely follows salinity intrusion, because sulfur content was similar between both the upper and lower brackish marsh community sites, but lower at all freshwater community sites. The brackish plant community at these sites is dominated by perennial species (*S. alterniflora* dominated at most creek sites except Taskinas A & B and Ware, which had mixed communities with no dominant species). Perennial species (such as *S. alterniflora*) are considered to be a good indicator of long-term trends while annuals can respond within months to an increase in salinity (Wetzel et al 2004), but will also disappear if there is a low salinity year, making them poor indicators of change in a system.

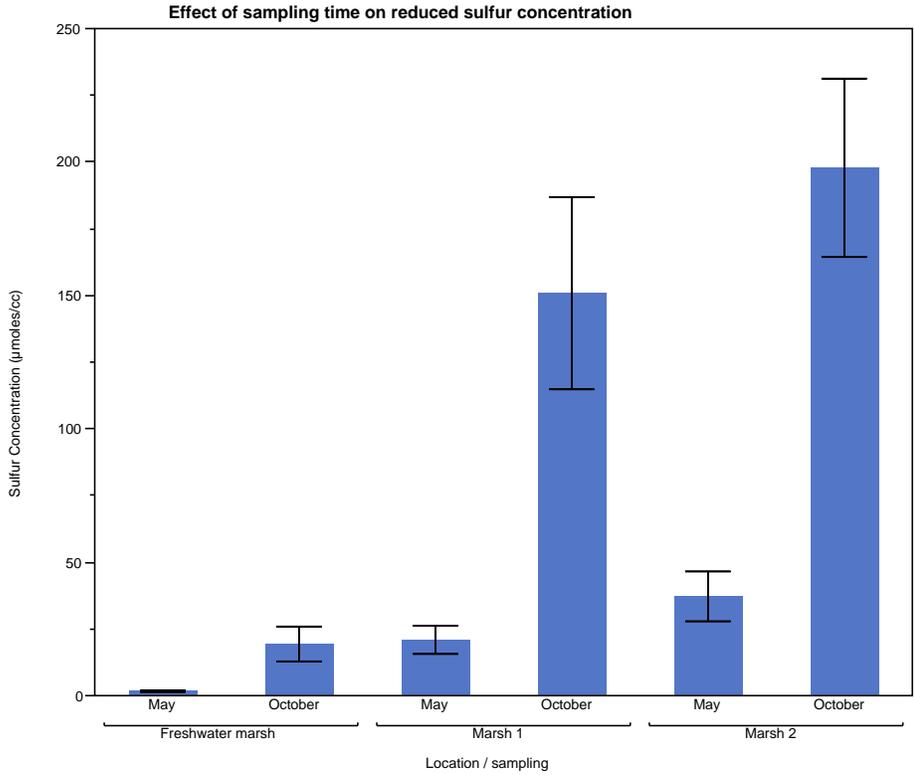
Neither salinity nor sediment sulfur content were good predictors of *P. australis* presence. *P. australis* patches were found between 20m upstream and 140 m downstream of the emergent freshwater marsh, suggesting that salinity was not a controlling factor its colonization. Typically, sulfur content in the *P. australis* areas was very low. The exception is the Ruritan creek where values at all depths were high. Removing that site dropped the mean concentration in the top subsample from 49 $\mu\text{mol/cc}$ to 12 $\mu\text{mol/cc}$, which is only slightly above the transitional marsh concentration average of 7 $\mu\text{mol/cc}$. The Ruritan site is located at the mouth of the York River and the distance from the river to the headwater system is short, therefore it is possible that the *P. australis* marsh is subject to frequent salinity intrusion which may be driving the high sulfur concentration. One study suggested that the invasive *P. australis* haplotype is particularly tolerant of saltwater (Vasquez et al. 2005). It is possible that our headwater systems contained different haplotypes, with different tolerances to salinity. However, the high sediment sulfur at that site may be an artifact. Coastal plain soils tend to be iron-rich, providing the

opportunity for sulfide precipitation as pyrite without any accumulation of toxic sulfide. Therefore, sediment sulfur may be high but non-toxic, since it is bound to iron. Low levels of salinity intrusion alone is not enough to stress *P. australis* which can tolerate moderate salinity levels (Farnsworth and Meyerson 2003) in contrast to freshwater marsh species such as *Panicum hemitomon* (Maidencane), *Leersia oryzoides* (Rice cutgrass) and *Sagittaria lancifolia* (Bulltongue arrowhead) which have very low tolerance for salinity intrusion (McKee and Mendelssohn 1989). This explains why high sulfur content was never found in freshwater communities—even if pyritization ensured that the sulfur was non-toxic, the plant communities would not be able to tolerate the accompanying salt concentration.

Seasonal variability of sulfur

Taking sulfur samples during two different seasons (the beginning, May, and end, October, of the growing season) was confounding and likely reduced the significance of our results. Comparing between seasons, sulfur was significantly higher in the fall samples than the summer samples at freshwater and brackish marsh sites (Figure 3, $F(1, 69) = 46.389, p < 0.005$). Temporal variability in sediment sulfur has been observed in other marshes and may be due to seasonal patterns of pyrite cycling, where oxidation of the plant rhizosphere during the growing season results in pyrite oxidation and release of sulfur compounds (Luther and Church 1988, Stribling and Cornwell 2001). This is consistent with our study, and suggests that fall sampling may enhance differences between sites. However, removing the spring samples from the analysis raised the mean sulfur content in the marsh sites, but did not change the overall trajectory of sediment sulfur content along the headwater-brackish marsh transect.

Figure 4-4. Mean sulfur concentration in marsh sites by season box plot. Each error bar is constructed using 1 standard error from the mean.



Conclusions

Shifting salinity regimes are a direct consequence of sea level rise but can also be somewhat mitigated and/or exacerbated by anthropogenically-moderated increases or decreases in freshwater flows (e.g., through water withdrawals or increases in impervious surfaces) and changes in precipitation. Shifts have been documented in extensive riverine marshes along the mainstem York River estuary (Perry and Hershner 1999) as freshwater plant communities become more brackish over time. Arguably, the smaller freshwater systems that feed into mainstem tributaries are at greater risk from shifts in sea level rise-driven salinization than the riverine marshes. These headwater systems have smaller watersheds and therefore less capacity to collect precipitation, recharge their

groundwater, and mitigate the influence of tidal water salinity intrusion. This study shows that brackish marsh plant communities are good predictors of salinity intrusion and sulfur sequestration in headwater-marsh complexes. Monitoring these plant communities should allow for detection of headwater vulnerability to sea level rise-driven salinization. However, in areas with significant *P. australis* communities, the lack of change in the vegetation may mask encroaching salinity, requiring monitoring of sediment sulfur content instead.

Chapter 5 Evolution of tidal marsh distribution under accelerating sea level rise

Abstract

Tidal marshes are important ecological components of the coastal system that are currently responding to sea level rise-driven changes in tidal regimes. These changes will affect future tidal marsh distribution, connectivity and role in estuarine systems.

Concurrently, human development along the coastline is creating barriers to marsh migration that will also be an important moderator of future tidal marsh distributions. Sea level rise is creating pressures for coastal areas to defend their infrastructure, leading to conflict between human and natural landscapes as tidal marshes attempt to migrate inland.

This study shows that in the Chesapeake Bay, an estuarine system with a range of shoreline elevations and development characteristics, overall estuarine tidal marshes are projected to decline by approximately half over the next century. Tidal freshwater habitats, which are found in the upper reaches of the estuary, typically backed by high elevation shorelines are particularly vulnerable. Due to their geological setting, losses of large extents of tidal freshwater habitat seem inevitable under sea level rise. However, in high salinity, low elevation, Bay-front localities, tidal marshes are capable of undergoing significant expansion. These areas should be prime management targets to maximize future tidal marsh extent. Redirecting new development to areas above 3m in elevation and actively removing impervious surfaces as they become tidally inundated results in a

best possible future. Under rising sea levels and increased flooding, the future of tidal marshes will rely heavily on the policy decisions made and the balance of human and natural landscapes in the consideration of future development.

Introduction

Tidal marsh loss is a significant issue throughout the United States and there is growing concern about accelerating sea level rise and the impact it will have on marsh persistence. Significant marsh loss may dramatically change coastal and estuarine functions and potentially impact global cycles (Coverdale et al. 2014; Chmura 2013). Marsh loss associated with sea level rise, erosion and human activity has been documented throughout the United States (e.g. DeLaune et al. 1994; Hartig et al. 2002; Bromberg and Bertness 2005; Mitchell et al. 2017).

Tidal marsh extents are defined by the interaction of landscape elevations and tidal regime. As sea levels rise and the maximum extent of tidal inundation reaches higher elevations, tidal marshes can migrate inland to maintain their place in the tidal frame. In areas with low coastal elevations, tidal marshes can expand or maintain their size as they move across the landscape, resulting in a potential future gain of tidal marshes (e.g., Kirwan et al. 2016). However, in areas with higher elevations or where migration paths are blocked by shoreline structures or impervious surfaces, marsh loss has been documented (Torio and Chmura 2013, Mitchell et al. 2017). Tidal marshes along shorelines with high banks, stabilized shorelines and marsh islands have limited migration potential and are at particular risk of reduction under sea level rise. Although elevation is the primary control on marsh migration potential, as marshes migrate inland they also conflict with development, particularly impervious surfaces. This conflict is

likely to increase in importance since coastal zones are not only more densely populated than inland areas but also show a trend of increasing population growth and urbanization (Neumann et al. 2015). Within the coastal zone, populations tend to be clustered in the lowest elevation areas (Small and Nicholls 2003), which are prime areas for marsh migration. Development patterns in urbanizing areas are a controlling factor in habitat loss (Bierwagen et al. 2010), and in coastal areas will be critical to the persistence of tidal marsh ecosystems. An understanding of future patterns from the intersection of tidal marsh distribution and development is required to maximize marsh persistence.

Concurrent with human landuse, erosion rates complicate the issue of marsh persistence. Long fetches lead to high erosion rates, even within the relative shelter of an estuary. Erosion rates are predicted to increase with sea level rise, exacerbating marsh loss (Leatherman et al. 2000). On high energy, moderate gradient slopes, high erosion rates have the potential to outpace landward migration, resulting in shrinking marsh extent. High erosion rates are also associated with proliferation of shoreline stabilization structures designed to protect developed areas but often actively block marsh migration. Shoreline hardening currently occurs on 14% of the U.S. coastline (Gittman et al. 2015) and in the Chesapeake Bay, approximately 18% of all tidal shorelines are already hardened (Bilkovic and Mitchell, 2017).

The question of marsh persistence is incomplete without consideration of the types of marshes and their position in the landscape. Many marsh functions (e.g., enhanced shoreline stabilization, Shepard et al. 2011; provision of refuge habitat, Minello et al. 2012) are reliant on a wide-spread distribution of marshes along shorelines, while some (e.g., modifiers of nutrient loads from upland, Valiela and Cole 2002; Valiela et al. 2002)

require their persistence in the upper portion of the estuary where they can effectively intercept groundwater and overland flow (Arheimer et al. 2004). Furthermore, freshwater marshes support unique floral and faunal communities that are not replicated in other marshes.

The Chesapeake Bay is the largest estuary in the northern hemisphere. Its long, crenulated shoreline means there are marshes of all shapes and sizes along the edges of the Bay and its tributaries. With a population of ~7 million people (Lotze et al. 2006), Bay shorelines vary from highly developed to rural settings and cover a wide range of erosive energy and topographic settings. Currently undergoing the highest rates of relative sea level rise along the Atlantic coast (Boon, 2012; Ezer et al. 2013; Sallenger et al., 2012; Kopp, 2013) and with evidence that those rates are accelerating (Boon and Mitchell 2015, Boon et al. 2017), the Chesapeake Bay is a perfect laboratory for investigating the balance between forces promoting and restricting marsh persistence into the future.

Marshes have the capacity to migrate landward with rising sea levels; however, the capacity of an individual marsh system is affected by their morphology and position in the landscape, their surrounding topography and adjacent human land use. It is likely that sea level rise will result in a change in marsh distribution driven by a combination of natural and human factors. In this paper, we move a theoretical future tidal frame across the landscape, allowing examination of how different factors impact future marsh distributions.

Methods

The Chesapeake Bay, Virginia is generally representative of tidal estuaries, containing a diverse array of tidal marsh types and ecologies, geologic settings, and human settlements. In Virginia, the Bay estuary consists of the mainstem Bay (with long fetches and flat, coastal plain shorelines) and estuarine rivers (with variable topography and fetches). It possesses a wide range of salinities from approximately 35 psu near the mouth of the Bay, to 0 psu in the upper reaches of the estuaries and in the small tributary creeks found along their edges. Currently, there are approximately 761 km² of tidal marshes, consisting of a mix of salinity types consisting of about 25% tidal freshwater marsh, 15% oligohaline marshes, 30% brackish and 30% salt marsh (TMI; CCRM 2017). Marshes are spread extensively along the shoreline, with pockets of concentrated salt marsh areas in Bay-front localities Accomack and Poquoson, and tidal freshwater marsh areas in York River tributary localities King and Queen, King William and New Kent. The tributary rivers split the Bay landscape into 4 peninsulas, creating corridors of development from old harbors. Because of this, areas of concentrated development are predominately on the Peninsula (Newport News, Hampton) and Southside (Norfolk, Virginia Beach, Chesapeake, Portsmouth), with a pocket of heavy development at the upper reaches of the Northern Neck (Alexandria, Arlington, Fairfax). Future development is expected to continue in these areas and the nearby areas; sprawling north and west in the southern part of the Bay and south in the northern part of the Bay (U.S. EPA 2010).

On average, the Chesapeake Bay, Virginia is experiencing the highest rates of relative sea level rise along the Atlantic coast (Boon 2012; Ezer and Corlett 2012; Ezer 2013;

Sallenger et al. 2012; Kopp 2013; Boon and Mitchell 2015). Recent rates from around the Bay are in the range of 4-6 mm/yr and appear to be accelerating (Ezer and Atkinson 2015; Boon and Mitchell 2015) while the rate of recent global sea level rise (based on satellite altimetry) is around 3.2mm/yr (Church and White 2011; Ezer 2013). This extreme rate is attributed to multiple factors including changes in global sea level in combination with regional and local land subsidence (Boon 2012; Eggleston and Pope 2013) and shifts in the Gulf Stream Current location and speed (Ezer 2013). This has led to an increase in flooding (Ezer and Atkinson 2014; Sweet and Park 2014) and an interest in flooding adaptations that reduce impacts to human infrastructure.

Movement of the tidal frame across the landscape

Modeling of the tidal marsh extent was based on elevation in a high-resolution lidar data set of the Chesapeake Bay, VA localities (CCRM, 2015) using ArcGIS software (ESRI, v 10.4.1). Elevations in the dataset are given as NAVD88 and therefore we have used those elevations throughout the study, rather than refer to a tidal datum. Vertical resolution is 0.15 m and horizontal resolution is 0.30 m. The vegetated tidal marsh frame in the Chesapeake Bay falls in the elevation range between Mean Sea Level (MSL) and the Highest Astronomical Tide (HAT). The exact elevations vary somewhat around the Bay, but always fall into the micro-tidal category. For example, at the Yorktown Station (the mouth of the York River), the difference from MSL (1.96 m) and HAT (2.66 m) is 0.69 m in the current epoch (NOAA, Datum for 8637689, Yorktown USCG Training Center VA). At the Lester Manor station (a freshwater tributary to the York River), the difference from MSL (-0.05 m) to HAT (0.58 m) is 0.63 m in the current epoch (NOAA, Datum for 8636653, Lester Manor VA). For convenience, and since the differences

between sites are frequently within the margin of error in the lidar data, the vegetated tidal marsh frame was considered to be 0.61 m (four times the vertical resolution of the lidar) in this analysis across the Chesapeake Bay, Virginia. In this vulnerability matrix, the vegetated tidal marsh frame (as described above) was moved across a lidar-based digital elevation model (DEM) land surface in overlapping 0.15 m (the vertical resolution of the lidar data) elevational increments (Table 1). This gives an estimate of the tidal wetland appropriate elevations in each step. For each elevation step, area of tidal wetland was calculated for each locality, giving a measure of how tidal wetland distribution is likely to change throughout Virginia, based solely on elevation. Starting elevations were 0 m – 0.61 m NAVD88, which was considered to be the approximate tidal frame for 2010 (see the next section).

Table 5-1. Scenarios used for analysis with their elevations and approximate time frames (based on Boon and Mitchell, 2015).

Scenario step number	Elevations (NAVD88)	Approximate year
1	0 m - 0.61m	2010
2	0.15 m – 0.46 m	2020
3	0.30 m – 0.91 m	2030
4	0.46 m – 1.07 m	2040
5	0.61 m – 1.22 m	2050
6	0.76 m – 1.37 m	2058
7	0.91 m – 1.52 m	2062
8	1.07 m – 1.68 m	2070
9	1.22 m – 1.83 m	2078
10	1.37 m – 1.98 m	2082
11	1.52 m – 2.13 m	2090
12	1.68 m – 2.29 m	2095
13	1.83 m – 2.44 m	2100
14	1.98 m – 2.59 m	2105
15	2.13 m – 2.74 m	2110
16	2.29 m – 2.90 m	2115
17	2.44 m – 3.05 m	2118
18	2.59 m – 3.20 m	2121
19	2.74 m – 3.35 m	2124
20	2.90 m – 3.51 m	2127
21	3.05 m – 3.66 m	2130

To test that a 0.61 m tidal frame is a reasonable approximation of tidal marsh area, predicted 2010 modeled tidal marsh areas (elevation 1, 0 m – 0.61 m NAVD88) were extracted from 25 watersheds along the mainstem York River, VA. These areas were compared to a survey of tidal marshes conducted in 2010 in the same watersheds (Mitchell et al. 2017) using a regression (JMP 10).

Approximating time frames for the projections

To set a timeframe for shifts in elevation in the tidal frame, a sea level rise projection curve based on data from Sewell’s Point, Virginia tide gauge was used (Boon and Mitchell 2015), which project that the tidal frame in 2050 will be 0.61 m – 1.22 m and can be extrapolated to projections in 2100 of approximately 1.83 m – 2.44 m. Sea level rise projections vary minimally across the Virginia portion of the Chesapeake Bay (Ezer and Atkinson 2015), and Sewell’s Point is considered representative. The time for each 0.15 m elevation shift was obtained from the curve. Since the curve begins in 1992 (the center point of the current tidal epoch), it was necessary to estimate a starting elevation for 2010. The MSL point 0 m NAVD88 was chosen from a historic sea level rise curve (Boon and Mitchell 2015) as an approximate MSL for 2010.

Evaluating the impacts of development on tidal wetland movement

Developed/impervious areas cannot convert to wetland without either 1) removal of the impervious surface, or 2) significant burial of the impervious surface by sediment. In addition developed areas have economic value, making them likely areas for protection measures that would prevent wetland migration. To examine the importance of developed areas on future marsh persistence, current impervious surfaces that are in the migration pathway were identified at each time step. This gives a “best case scenario”, assuming

no future development into coastal areas. Impervious surface projections for 2050 and 2100 within the migration pathway were also identified for the appropriate time steps. This gives a “projected scenario” which assumes continued patterns of development into the coastal zone.

For current development, the VGIN 1m Land Cover dataset (2016) was used to categorize the type of land in the tidal frame for each step with impervious, turf grass and barren areas considered “Developed” and all other categories (e.g., wetland, pasture, forest, agricultural) considered “Undeveloped”. Acres of land in each type were summed for locality, and the percent of developed land within the tidal frame was calculated for each time step. Localities can be compared based on the importance of their developed lands to marsh migration and the timeframe in which the conflict between marsh migration and human development will become pronounced.

Future development scenarios were analyzed using impervious surface projections based on housing density growth models (U.S. EPA 2010). The baseline scenario impervious surface of 20% or greater for 2050 and 2100 were extracted and spatially intersected with projected tidal frames for 2050 and 2100, respectively. Ecological thresholds studies suggest that levels of development between 10-25% can impact ecosystem system functions (e.g. Wang et al. 1997; Limburg and Schmidt 1990; Paul and Meyer 2001; DeLuca et al. 2004; Brooks et al. 2006; King et al. 2005; Bilkovic et al. 2006; Lussier et al. 2006) and previous work has shown accelerated loss of marshes with greater than 15% development (Mitchell et al. 2017). These results were compared with the total area in the projected tidal frames in 2050 and 2100 to elucidate the difference in tidal marsh

migration patterns under management decisions that allow growth to continue in current patterns and those that direct housing growth away from coastal areas.

Salinity distribution

Localities were assigned a dominant marsh ecotype (salt marsh, brackish marsh, freshwater) based on the plant communities identified in the Tidal Marsh Inventory (CCRM 2017). Many localities that are dominated by salt marsh currently have creeks with tidal freshwater marshes near their headwaters. However, these marshes are a minor component under current salinity regimes and will be stressed further by sea level rise-induced increases in salinity. Therefore, only the current dominant community type was used for categorization. No attempt was made to project changes in salinity due to the difficulty of balancing sea level rise-induced upstream salinity migration with the potential increases in river flow due to changing precipitation under current projections.

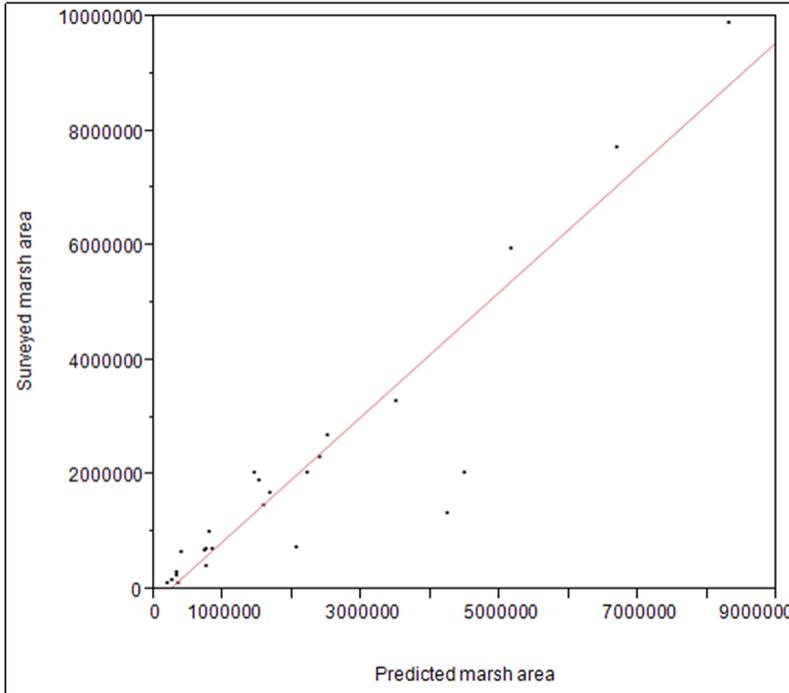
Results

Tidal elevation range as an indicator of tidal marsh extent

A comparison of the 2010 modeled tidal marsh areas (elevation 1, 0 m – 0.61 m NAVD88) with survey tidal marshes (Mitchell et al. 2017) showed that the model did a good job of identifying tidal marshes (Figure 1, $R^2=0.89$), with small overestimation in some watersheds and small underestimation in other watersheds. Examination of mapped extents show that, in general, the model seemed to slightly underestimate marsh extents in extensive marshes. This is not unexpected, since in the York River, HAT is 0.69 m above MSL. This issue should be minimal in the lower parts of the Bay, where the tidal marsh envelope is closer to the 0.61 m used in the model. The model also seemed to slightly overestimate marsh extents at the tops of some creeks. In these cases, landuse

frequently indicated that the areas were treed/forested—suggesting that these might be tidal swamp areas (which would not be captured in the TMI dataset) or forested areas transitioning to tidal marsh.

Figure 5-1. Comparison of modeled and surveyed marsh area (m²) in 25 watersheds on the mainstem York River, VA.



Projected changes in marsh area and distribution

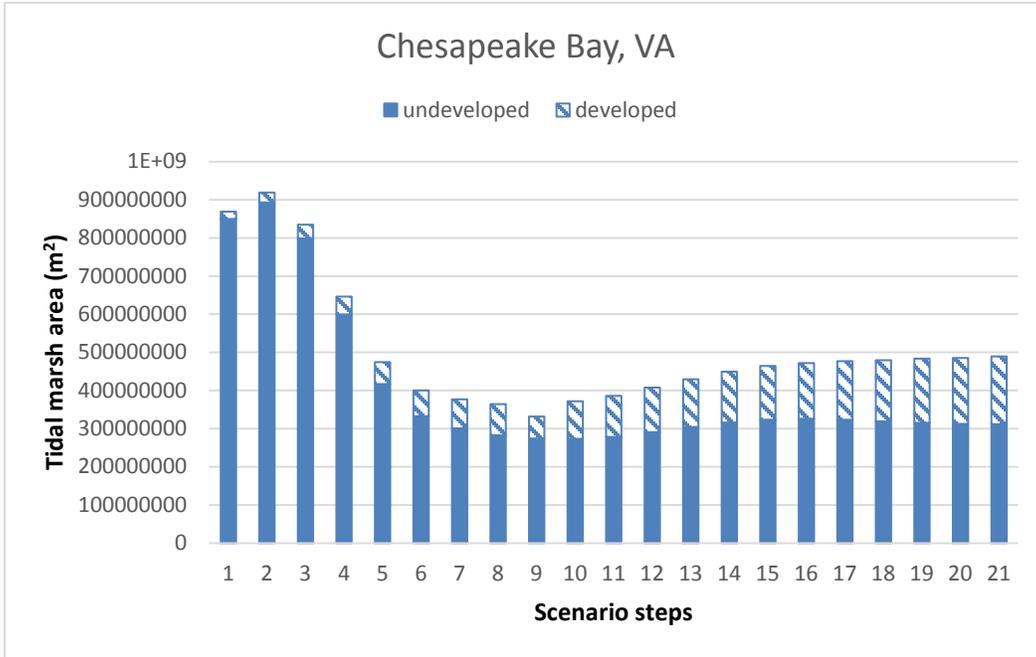
In the 2010 tidal frame elevation range there were 850 km² of potential tidal marsh in Chesapeake Bay, Virginia. This number declines slowly over time steps to a minimum of 331 km² at time step 9 (approximately 2070; Figure 2a, entire bars). The tidal area then recovers slightly, ending with a net loss of 379 km² of tidal marshes, or 43% of the starting tidal marsh area. Most of the tidal marsh loss will be realized relatively early, by 2050-2070. Following that time period, total tidal marsh extent should remain fairly constant or even expand slightly. However, the distribution of tidal marsh loss varies by

location in the Bay, with some areas showing essentially continual decline while others expand in the post-2070 time frame (Figure 2b). The greatest area of tidal marsh in 2010 is found in the Eastern Shore, but this is the region that sees the greatest percent reduction (84%) in tidal marsh area over time. In contrast, the Southside (with the second highest starting tidal marsh area) shows a rapid loss of tidal marsh (<50%) by 2050, followed by a significant recovery of marsh area as the marshes migrate inland. By the final time step, there is a slight (10%) gain in tidal marsh. The most northern regions, the Northern Neck, had the fewest marshes in the early time frame but also shows a pattern of slight gain (3%) in the later time steps. The Middle Peninsula and Peninsula both show a pattern of tidal marsh loss by 2050, followed by a period of recovery, resulting in a 47% and 62% loss of marsh area, respectively.

Not only is the area of tidal marsh projected to decline over time due to rising sea level, but the way in which the remaining area of marsh is distributed will change (Figure 2b). In the 2010 time frame, 38% of total tidal marsh area is in the Eastern Shore region and only 27% of tidal marshes are found in the Southside region. By the final time step, this has shifted so that the Southside region has 53% of all tidal marshes, while the Eastern Shore region has only 11% of the remaining tidal marshes. In the Southside region, most of the marshes will be in Chesapeake and Virginia Beach (Figure 3). This means that lands in the Southside region, particularly in those two localities, are the most critical for preservation to ensure marsh migration while Eastern Shore and Peninsula regions have limited opportunity for marsh migration based on elevation.

Figure 5-2. Changes in area (m2) within the tidal marsh elevation range over time. Scenario steps are 0.61m in range and move up 0.15m in elevation with each step. a) Total tidal marsh area in the Virginia Chesapeake Bay. b) Tidal marsh area split by region.

a)



b)

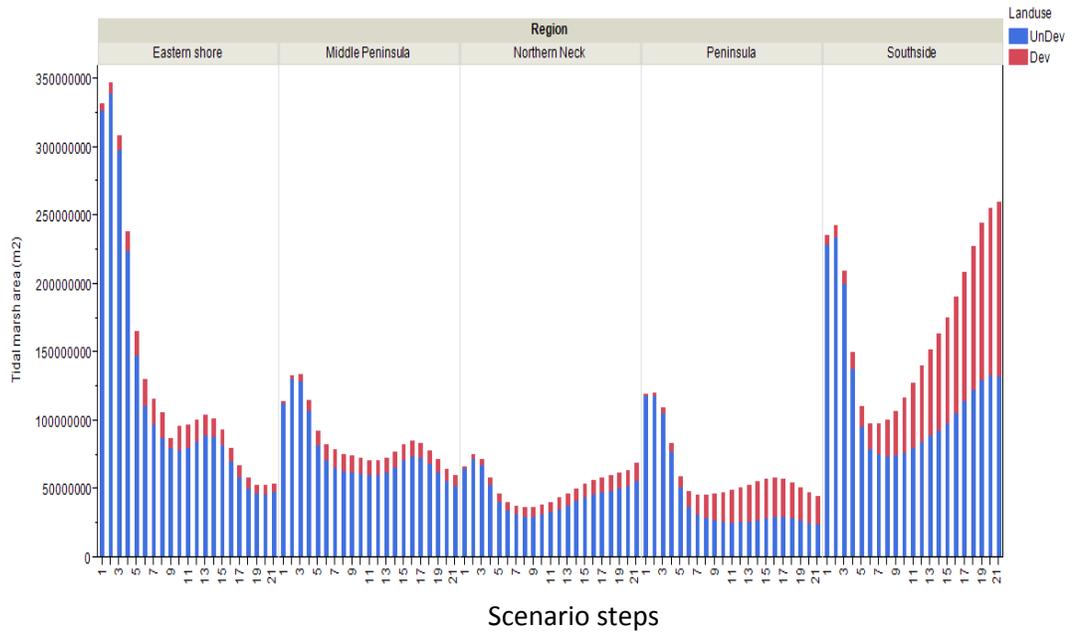
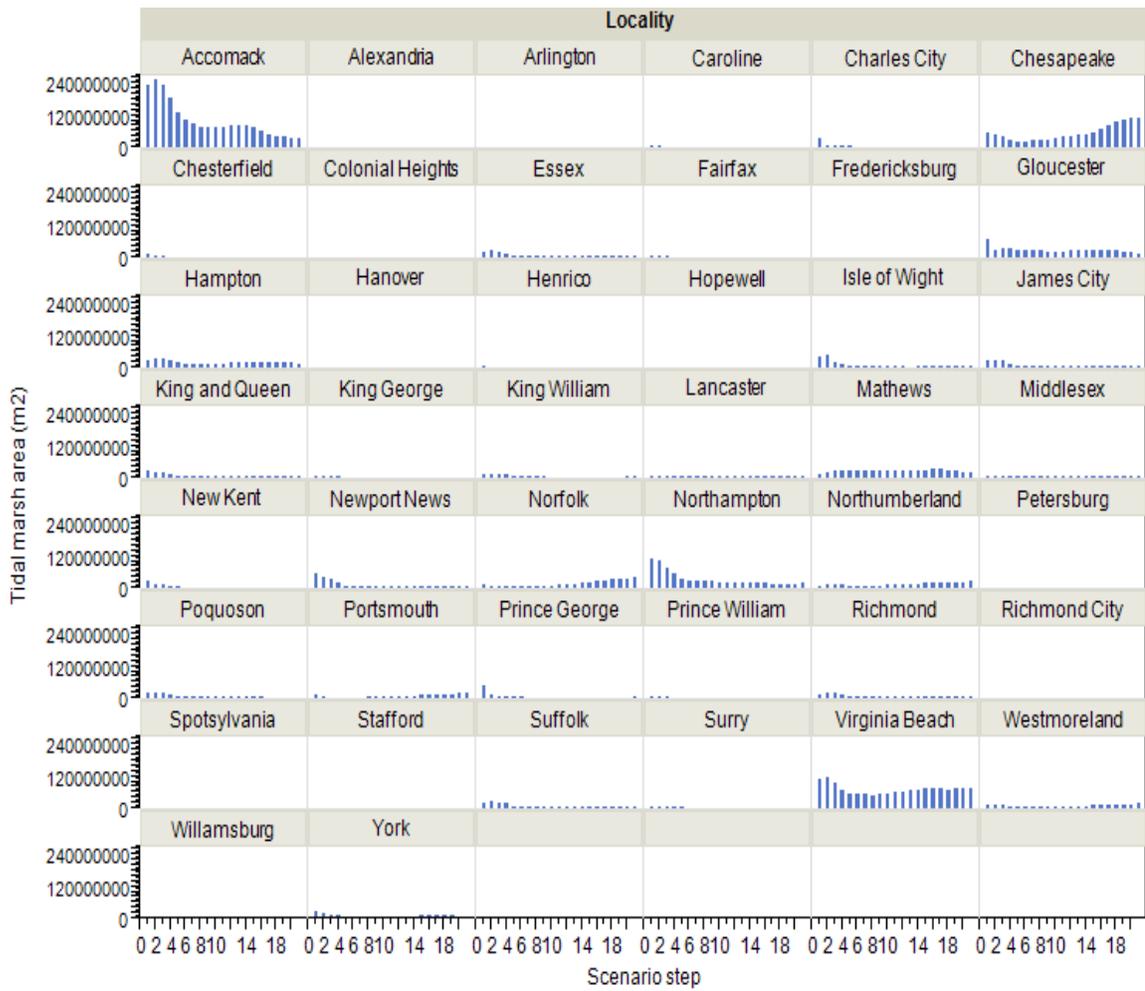


Figure 5-3. Changes in marsh area over time by locality.



Impervious surfaces in migration pathways

Under current development conditions, 2-36% of the area in each time step's tidal elevation range is developed (Figure 2a, hatched portion of bars). The proportion of developed area in the tidal frame increases over time as the tidal frame migrates upland, limiting the likely area of tidal marsh. The proportion of impervious surface varies by locality as well as through time (Figure 4a and b). In the low elevation urban localities (e.g., Hampton), there are ample lands in the future tidal elevation range for marsh migration. However, the majority of those lands are already developed. Only a small fraction of the appropriate elevations are currently natural lands. In the low elevation rural localities, (e.g., Mathews) the percentage of impervious surface currently in the projected tidal elevation ranges is low. If future coastal development is discouraged, tidal marsh areas will be essentially consistent over time in these localities.

When projected future impervious surface is included in the analysis (Figure 5), it is clear that there are only a few localities where targeting future development patterns will substantially increase projected marsh area in 2050 or 2100; namely Chesapeake and Virginia Beach. However, there are a number of localities where current development may impact a high percentage of future marsh migration and could be targeted for mitigation strategies, including Fairfax, Hampton, Newport News and Portsmouth.

Figure 5-4. Total projected marsh area over time in two localities (a) Hampton (urban) and (b) Mathews (rural). Solid portions of the bars indicate areas that are pervious (natural lands) in the projected tidal elevations. Hatched portions of the bars indicate areas that are currently impervious surfaces. These areas would have to be removed to allow tidal marshes to establish through migration.

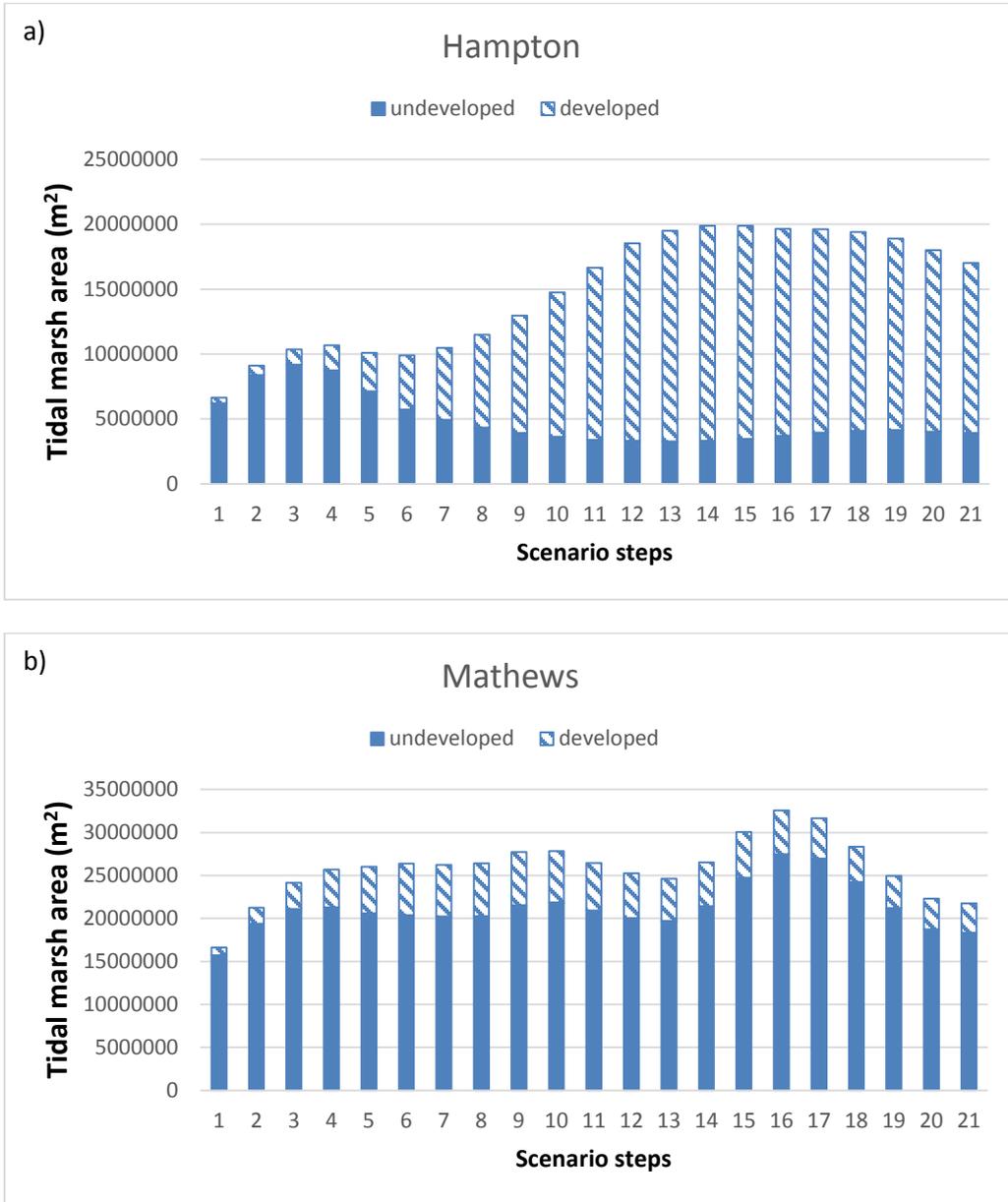
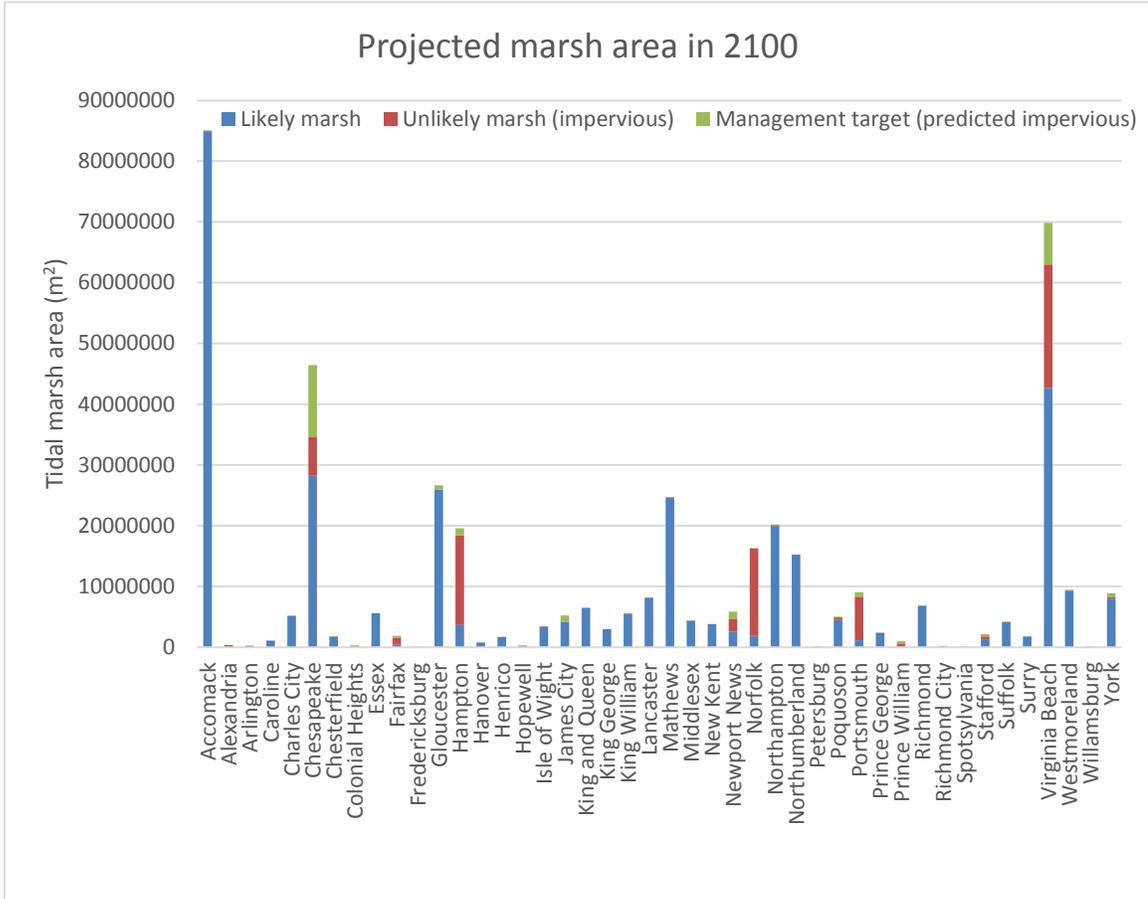


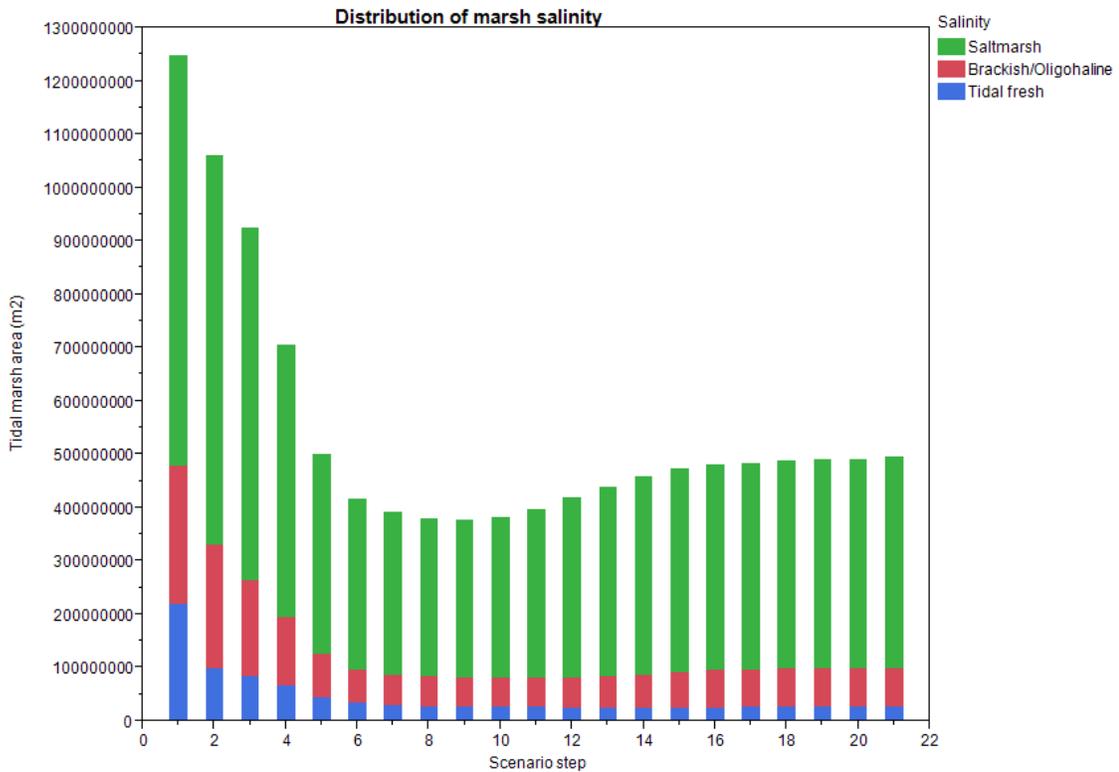
Figure 5-5. Tidal marsh and impervious surface conflicts by locality. “Likely marsh” is natural lands in the 2100 tidal range elevations, which is projected to become marsh. “Unlikely marsh” is currently impervious surfaces in the 2100 tidal range elevations, which would have to be actively removed for marsh to establish. “Management target” is the additional projected impervious surface in the 2100 tidal range elevations, which will occur if development patterns continue to follow their current trends.



Marsh salinity distributions

Concurrent with an overall decline in marsh area, there is an increase in the dominance of salt marsh types and a reduction in the proportion of brackish/oligohaline and freshwater marshes (Figure 6). In the first time step (i.e., 2010), 18% of marsh acreage is tidal freshwater, 21% is brackish/oligohaline, and 62% is salt marsh. This shifts rapidly and by 2050, only 6% of marsh acreage is tidal freshwater, while 81% of marsh acreage is salt marsh. Because this study did not include upstream salinity migration, this shift is entirely driven by the expansion/enhanced persistence of Bayfront marshes (which are dominated by saltmarsh plants) and the loss of tributary marshes (dominated by brackish/oligohaline and tidal fresh marshes).

Figure 5-6. Changes in marsh area by salinity type over time.



Discussion

When planning for the future, it is important to understand the distribution of natural resources, how they will change and which changes will be affected by management decisions. It is clear from this analysis that tidal marsh area in the Chesapeake Bay, Virginia will inevitably decline over time, and that much of this decline is likely to occur in the relatively near future (by 2100). In addition, there will be shifts in the distribution of tidal marshes leading to an increase in salt marshes and a decline in the oligohaline and tidal freshwater marshes that will alter ecological connections and functions. However, management decisions, particularly in the low elevation areas can maximize future tidal marsh extent. Although this study was conducted in the Chesapeake Bay, Virginia, it results should be applicable to many estuarine systems, where elevations rise and salinities decline with distance from the coast.

Our study suggests a highly geographically controlled outcome to future marsh persistence, similar to observed past changes (Mitchell et al. 2017). Although, this study shows an overall decrease in tidal marsh extent throughout the Bay, marsh extents in Bay-front localities will increase. This is due to the low elevations in these areas which provide ample land for marsh expansion, coupled with the currently low human development in many of these areas. Hampton is an exception in its high development, and the cost of this development is evident in the low amount of natural lands available for marsh migration.

In addition to changes in the distribution of marsh extent, the pattern of topography in the Chesapeake Bay region drives a shift in the distribution of marsh ecotypes over time. As Bay-front marshes expand, oligohaline and tidal freshwater marshes (particularly those in

headwater systems) contract. This is likely to have significant ecological impacts due to a decline in important tidal marsh habitats and a reduced potential for groundwater interception and filtering at the heads of the estuaries as marsh acreage in these areas declines. This study did not attempt to project sea level rise-induced changes in salinity; however, it is important to note that upstream migration of salinity is predicted in the Chesapeake Bay (Hong and Shen 2012) and that this will further reduce the proportion of tidal freshwater marshes in projected distributions unless increased precipitation is sufficient to counter the salinity migration.

Interaction of sea level rise, accretion and erosion

Factors not explicitly considered in this analysis that can impact marsh persistence include marsh accretion and erosion rates.

The contribution of marsh accretion to future marsh extent is still an open question. Marsh accretion is a factor of both *in situ* organic production rates and allochthonous sediment retention. It is the hardest variable to project into the future, since climatic shifts can affect plant productivity (e.g. C3 plant production under increased CO₂; Drake 2014) and sediment supply (e.g. sediment erosion under increased precipitation intensity; Williams et al. 2017). Marsh plant production rates are highly variable, but a geographically expansive survey suggests that there is a theoretical limit to *in situ* organic sediment accretion of 5mm/yr (Morris et al. 2016). Sea level rise has exceeded this rate in the Chesapeake Bay over the past 30 years (5.86 mm/yr at the mouth of the Bay; Ezer and Atkinson 2015) and is predicted to accelerate (Boon and Mitchell 2015). During the same time period, sediment loads to the Bay (a potential source of allochthonous sediment contribution to marshes) have declined due to management actions (Gellis et al.

2004). Explicit TSS reduction goals for the Bay (<http://www.epa.gov/chesapeake-bay-tmdl>) are designed to continue aggressive sediment management into the future. These reductions in sediment supply coupled with the predicted acceleration in sea level rise reduce the importance of marsh accretion for future marsh persistence. Even in areas with high sediment supply, rates of RSLR above 10.2 mm/yr are predicted to be unsustainable for marshes (Morris et al. 2002). Under current rates of acceleration (0.119 mm/yr²; Boon et al. 2017), RSLR in the Bay will exceed those values within 60 years. However, previous studies in the Chesapeake Bay have shown a time lag between the time sea level rise rates exceeded local accretion rates and the subsequent marsh loss (Kearney et al. 2002) that may mean tidal marsh loss in the next couple decades is controlled more by erosion rates than sea level drowning.

Erosion rates are highly variable along Chesapeake Bay shorelines, even sometimes within close geographic proximity. Although relatively stable over the recent past (Kirwan et al. 2016), erosion rates are predicted to increase with accelerating sea level rise, potentially resulting in huge coastal losses (Leatherman et al. 2000; Mariotti and Fagherazzi 2010). On average, Bayfront locality shorefronts experience low to moderate erosion on 30% of their shorelines (Milligan et al. 2012). Exceptions are heavily stabilized shorelines such as those in Norfolk. Bayfront marshes are considered one of the more stable Bay shoreline environments, eroding at 0.54 – 0.66 m/yr, depending on the underlying substrate (Rosen 1980). Rates on the tributaries are generally lower (e.g., York River marshes are eroding at 0.21 m/yr; Byrne and Anderson, 1978) and erosion in the creeks is generally negligible. Given these rates, the marshes where erosion rates will most affect marsh acreage are located in the same localities where much of the marsh

expansion is projected (e.g., Gloucester, Mathews). The balance between marsh erosion and marsh migration will vary over time depending on their relative trends (i.e., linear vs. accelerating rise), and the impact to marsh acreage will be highly dependent on the slope of the shoreline (Figure 8). However, it is expected that erosion will result in the loss of some of the projected marsh acreage; therefore, the numbers in the study may be overestimating marsh extent, particularly where there are narrow, fringing marshes that could erode before having the opportunity migrate significantly.

Shoreline stabilization placed at the front edge of a marsh will reduce or eliminate erosion, while allowing marsh migration. However, where shoreline stabilization is placed landward of the marsh, erosion of the marsh will continue while marsh migration will be blocked until the elevation of the stabilizing structure is topped. This may lead to a temporary loss of marsh in heavily stabilized areas, even with low gradient shorelines, or longer term loss if stabilization structures are high. Tidal marshes should re-establish following overtopping of stabilization structures by the tidal envelope, but ecological services associated with those marshes may be difficult to re-establish, particularly if the new plant community differs from the original.

Management implications

Maximizing future tidal marsh extent will require prioritize of natural land preservation in low elevation lands contiguous to the shoreline. A clear policy consideration resulting from this study is that a uniform state-wide management policy will not maximize future tidal marsh extent unless that policy is specifically tied to elevations (e.g., minimizing development in lands below 3 ft NAVD88). In localities with shallow shoreline elevation gradients, passive measures (such as the preservations of natural lands) can be a powerful

management action, assuming that extensive natural lands exist. However, in localities with steep shoreline gradients, tidal marsh persistence will require more active measures and may eventually be futile. Active management in these areas may include the construction of “living shorelines” to replace or expand dwindling marsh extents or thin-layer deposition to help existing marshes maintain elevation in the tidal frame (Wigard et al. 2017).

In highly developed/urban areas, tidal marshes may be of particular ecological importance since they are often scarce and therefore the remaining marshes represent critical refuges for faunal marsh residents. In the Chesapeake Bay, many of the localities with shallow shoreline elevation gradients are also highly urbanized and expanding. In these localities, tidal marshes have the capacity to expand and become less fragmented under sea level rise. However, that endpoint requires aggressive preservation of remaining natural lands in tidal marsh migration corridors and consideration of the active removal of impervious surfaces as they become inundated to allow marsh development. This type of activity is contrary to the actions taken by many urban areas under pressure from flooding and sea level rise. Rising water levels are frequently met with shoreline hardening and coastal barriers, which can preserve or improve property values (Jin et al. 2015). Less frequently used, managed retreat/realignment and rolling easements, where development is gradually moved out of the water’s path, is the adaptation that is most in line with the goal of maximizing future tidal marsh extents. Other adaptations that allow a balance between protection of human infrastructure and tidal marshes include storm surge barriers (which allow natural tidal action except during storm events) and the use of

natural features (such as beach nourishment or marsh creation) to alleviate storm-associated flooding.

In conclusion, the future of tidal marsh complexes is highly dependent on their location within the geological (elevation) and human (impervious surface) landscape. Under a scenario of “no change” in landuse, tidal marshes will expand in some locations and contract in others, resulting in a net loss of approximately half the tidal marshes in the Chesapeake Bay, VA. About another third of the marsh extent will conflict with current impervious surfaces. Rising sea levels and increased flooding create additional pressures to shoreline systems as urban areas prepare to protect their infrastructure. The future of tidal marshes will rely heavily on the policy decisions made and the balance of human and natural landscapes in the consideration of future development.

Chapter 6 Summary

1. Analyses of changes in tidal marsh extent and plant communities are complementary, clarifying vulnerabilities and prognosis under future conditions.
2. Human shoreline use (e.g., development, shoreline hardening, and boating activity) can dominate physical processes to alter the marsh response to sea level rise.
3. Defining sediment availability for a given marsh may not be sufficient to determine its potential for expansion or persistence under sea level rise.
4. Marsh plant communities have been changing throughout the Chesapeake Bay, indicative of inundation, salinization, erosion and non-native plant invasion.
5. Marsh plant communities can be an early signal of change, showing shifts in inundation frequency before there is any change in marsh extent. Monitoring plant communities in areas already showing change will allow us to track the trajectory of change throughout the Bay.
6. Tidal marshes will continue to decline over the next 100 years. However, most of the loss will be in low salinity, riverine marshes. Some high salinity, Bayfront marshes will expand if the land they need to migrate is preserved.
7. Tidal marsh response to sea level rise has, and will continue to, vary by marsh form, geologic setting, location in the estuary, and surrounding land use decisions.
8. The variability of marsh response emphasizes the issues with the current practices of evaluating regional marsh change with studies of only one or a few marshes and/or studies limited to only extensive marshes.
9. Preservation of marsh migration corridors in Bayfront localities coupled with marsh creation in tributaries may help minimize future marsh loss.

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