UTILIZING WETLAND ANALYSIS AND DETAILED ELEVATION SURVEYS TO EVALUATE SPATIAL PATTERNS OF SEDIMENTATION AND POTENTIAL MARSH SUSTAINABILITY IN DELAWARE'S TIDAL AND IMPOUNDED WETLANDS

by

Bartholomew Wilson

A dissertation submitted to the Faculty of the University of Delaware in partial fulfillment of the requirements for the degree of Doctor of Philosophy in Geology

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Bartholomew Wilson

Approved: ____
Neil C. Sturchio, Ph.D.
Chair of the Department of Geological Sciences

Approved: ____
Mohsen Badiey, Ph.D.
Acting Dean of the College of Earth, Ocean and Environment

Approved: ____
Ann L. Ardis, Ph.D.
Senior Vice Provost for Graduate and Professional Education

I certify that I have read this dissertation and that in my opinion it meets the academic and professional standard required by the University as a dissertation for the degree of Doctor of Philosophy.

Signed: ______________________________

John Madsen, Ph.D.
Professor in charge of dissertation

I certify that I have read this dissertation and that in my opinion it meets the academic and professional standard required by the University as a dissertation for the degree of Doctor of Philosophy.

Signed: ______________________________

Holly Michael, Ph.D.
Member of dissertation committee

I certify that I have read this dissertation and that in my opinion it meets the academic and professional standard required by the University as a dissertation for the degree of Doctor of Philosophy.

Signed: ______________________________

Susan McGeary, Ph.D.
Member of dissertation committee

I certify that I have read this dissertation and that in my opinion it meets the academic and professional standard required by the University as a dissertation for the degree of Doctor of Philosophy.

Signed: ______________________________

Susan C. Adamowicz, Ph.D.
Member of dissertation committee

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ABSTRACT

The purpose of this dissertation is to provide better understanding of the processes that control the evolution of natural tidal and impounded wetlands. The evaluation of coastal resources is necessary for conservation and management efforts, especially as sea-level rise and anthropogenic alteration impact tidal wetlands.

This research project consists of three separate studies in the vicinity of the Delaware River and Bay coastline within the Delaware Estuary. They include:

- The determination of the optimal ("Goldilocks") vertical growth range and above-ground and below-ground biomass production of *Spartina alterniflora* (*S. alterniflora)* with respect to mean low water (MLW) and mean high water (MHW) tidal datums within six watersheds (Blackbird Creek, Bombay Hook Complex, St. Jones River, Murderkill, River, Prime Hook Creek, and Broadkill River). In these watersheds, *S. alterniflora* has an optimal growth range between -0.07 and 0.18 m relative to MHW elevation and between 1.25 and 1.72 m, relative to MLW. The results can be used to assess the ability of a marsh to combat changing conditions associated with sea-level rise, by determining whether or not the marsh platform is within the optimal growth range of *S. alterniflora*. This assessment method can be used to determine

whether a tidal wetland area is in need of restoration for longer-term sustainability, through the optimization of below-ground biomass production.

- An assessment of water level management actions on accretion rates and wetland platform elevations of impoundment marshes using neighboring un-impounded tidal wetlands as reference sites. Nine impoundments (three in northern Delaware (in the vicinity of New Castle, Delaware), four in Central Delaware Bay, and two in lower Delaware Bay) and four reference tidal wetlands (two in Northern Delaware, one in Central Delaware Bay, and one in the lower Delaware Bay) were studied. Regular tidal inundation is vital in the development and evolution of natural marsh platforms. The study results show that only three of the nine impoundments having statistically significant differences (at the $p<0.05$ and/or $p<0.01$ levels) in mean accretion rate. Only two of the nine impoundments had significant differences (at the $p<0.05$ level) in mineral mass accumulation rates. Seven of the nine impoundments have significant differences (at the $p<0.05$ and/or $p<0.01$ levels) in organic mass accumulation rates. The effect of water level management on a wetland's ability to produce below-ground biomass and retain organic material appears to drive the differences in accretion and marsh platform elevation between impoundments and natural wetlands. It is paramount to manage water levels for species of concern, but in a way that below-ground biomass is optimized and decomposition rates are not enhanced through long-term marsh platform exposures.

- The utilization of surface elevation table (SET) monitoring within two watersheds, one relatively un-impacted (Blackbird Creek) and the other a heavily impacted mixed urban, suburban, and agricultural watershed (St. Jones River), to evaluate recent trends in tidal marsh surface elevation and short-term vertical accretion. The monitoring revealed that six of the eight sites showed a loss in marsh surface elevation (both shallow and deep) that has not recovered. Deep and shallow elevation changes occurred independent of accretion. It was observed that two (Eagles Nest and Beaver Branch) of the eight monitoring sites experienced rapid gains in elevation (and losses not experienced at the other six sites). The exact cause of large- scale wetland elevation losses that did not recover as of the end of the sampling are difficult to determine. However, moderate El Niño and high sea-level anomalies occurred along the East Coast during the sampling period. The lack of recovery may denote the overall stress of sea-level rise on the tidal wetlands and, as higher-level anomalies occur, vegetation cannot rebound in a manner that some marshes have demonstrated or been theorized to do under un-impacted conditions.

Chapter 1

INTRODUCTION

The evaluation of coastal resources is necessary for conservation and management efforts especially as sea-level rise and anthropogenic alteration impact tidal wetlands along the East Coast of the United States including the State of Delaware. The rate of relative local sea-level rise is currently estimated to be 3.14 to 3.20 millimeters per year (mm/yr) in the Delaware Bay, and it is projected to accelerate at rates that would produce an increase in sea level of 1 meter (m) over the next 100 years (Gornitz and Lebedeff, 1987; Nikitina et al., 2000; Intergovernmental Panel on Climate Change (IPCC), 2001; NOAA, 2014).

Temporal variation in the rate of sea-level rise is one of the main controls on marsh development or destruction (Rampino and Sanders, 1981; Stevenson et al., 1986; Patrick and DeLaune, 1990; Ward et al., 1998). A marsh's ability to keep pace with sea-level rise is directly correlated to the rate of elevation change along the marsh surface. The rate of sea-level rise has to be slow enough that appreciable quantities of mineral matter or organic materials can be deposited to build (and maintain) the marsh surface to an elevation near mean water level (Frey and Basan, 1985; McKee and Patrick, 1988; and Ward et al., 1998). If sea-level rise outpaces vertical sediment accretion, wetland loss will result everywhere that lateral wetland migration is not possible. In addition, the increased presence of salt water will cause shifts in plant community composition (Callaway et al., 1996; Burkett and Kusler, 2000; Winter, 2000; Darke and Megonigal, 2003).

Anthropogenic alterations such as development of coastal areas, water-withdrawal projects, dams, and reservoirs can affect the complex interactions of sediment deposition, erosion, and subsidence that determine wetland surface elevation (Davis, 1997; Najjar et al., 2000; Darke and Megonigal, 2003). Within the Delaware Estuary, intensive human alterations to wetlands have occurred in varying intensities over the course of the last several hundred years. Alterations of the wetlands have included: the impoundment of wetlands for farming (1700's and 1800's) and wildlife enhancement (mid to late 1900's), the dredging of channels for navigational purposes (1700's through the early 1900's), and the ditched and dredging of channels to reduce the prevalence of mosquitos (1930's through 1980's). These alterations have all affected the hydrology of the tidal marshes.

Understanding how wetlands and their vegetation will respond is necessary to constrain the sustainability of tidal wetlands. Current dynamic trends (surface elevation, sedimentation, subsidence, and vegetation parameters) provide information that can be used to predict how wetlands will respond in the future to natural and anthropogenic stressors. The relationships between inorganic and organic sedimentation, subsidence, vegetation composition, health and density, and their role in governing changes in marsh surface elevation are key parameters.

In a final report by the Coastal States Organization (CSO) Climate Change Work Group ("The Role of Coastal Zone Management Programs in Adaptation to Climate Change" (2007)), several critical research needs are identified relative to the impact of accelerated sea-level rise on coastal habitats. These include: “natural sediment sources, the movement of sediment within the system, and the locations and rates of sediment deposition need to be quantified for discrete shoreline reaches in order for predictive capabilities to be developed (CSO, 2007).” In addition, the report indicates that there “continues to be a need for improved models that predict the migration and/or vertical accretion of coastal wetlands in response to accelerated sea-level rise, information on the costs of response options, and the consequence of taking no action.”

The effects of sea-level rise and/or water level management upon natural and impounded marsh platform development and the effect of changing tidal levels upon wetland vegetation has not been studied in detail within the Mid-Atlantic region of the United States. Establishing baselines for marsh surface elevations is an important first-step in determining the impacts of sea-level rise and anthropogenic alterations on wetlands and the ecosystem services they provide.

In this dissertation, three separate wetland monitoring and research studies are presented in the vicinity of the Delaware River and Bay coastline within the Delaware Estuary. The specific studies are:

- The determination of an optimal (or “Goldilocks”) marsh platform elevation over which biomass production of *Spartina alterniflora* (*S. alterniflora*) is maximized. This study examines the vertical growth range including above-ground and below-ground biomass production of *S. alterniflora* with respect to mean low water (MLW) and mean high water (MHW) tidal datums. The results are used to examine how the growth range for *S. alterniflora* can be used to assess the sustainability and restoration potential for wetlands, and the effects of altered hydrology on biomass production. *S. alterniflora* is the dominant halophyte within Delaware's marshes and is a key component in need of better understanding to evaluate the resiliency of tidal wetlands in response to rising sea level (DNREC, 2013).

- The evaluation of water level management actions on the rate of accretion and wetland platform elevation of impoundment marshes in comparison to neighboring un-impounded tidal wetlands. Nine impoundments (three in northern Delaware (in the vicinity of New Castle, Delaware), four in Central Delaware Bay, and two in lower Delaware Bay) and four reference tidal wetlands were sampled to determine their rates of accretion over the past 60 years and their wetland platform elevations relative to local tidal datums. The duration of impoundment and water level management goals vary between these sites. This allows for a unique opportunity to evaluate the short-term and long-term implications of water-level control on the evolution of marshes associated with impoundments.
- The utilization of surface elevation table (SET) monitoring including the use of marker horizons at sites within two Delaware watersheds, Blackbird Creek (relatively un-impacted watershed) and St. Jones River (a heavily impacted mixed urban- suburban, and agricultural watershed), to evaluate recent trends in tidal marsh surface elevation and short-term vertical accretion. The elevation and accretion records provide a means of monitoring the response of a marsh to seasonal, annual, and decadal changes in tidal levels. SET and marker horizon

data also provide information on surficial and below-ground processes. For example, differences between rates of vertical accretion and elevation change may be attributed to dewatering, enhanced root growth, decomposition, and/or root growth collapse.

These studies including the methods used, results, and discussions of the results are described in Chapters 2 through 4 of this dissertation, respectively. A summary of the major conclusions from these studies is presented in Chapter 5.

Chapter 2

DEVELOPING THE GOLDILOCKS ELEVATION FOR *SPARTINA ALTERNIFLORA*: IMPLICATIONS FOR SUSTAINABILITY AND RESTORATION POTENTIAL OF WETLANDS

2.1 Introduction

The role of marsh surface elevation on the growth of halophytes, especially *Spartina alterniflora* (*S. alterniflora),* is documented in many studies (McKee and Mendelssohn, 1989; McKee and Patrick, 1998; Gough and Grace, 1998; Baldwin et al., 2001; Morris et al., 2002; Konisky and Burdick, 2004). Of particular interest is the role of marsh surface elevation on the below-ground biomass production of *S. alterniflora*. Below-ground biomass production plays a large role in enabling tidal wetlands to vertically accrete to keep pace with rising sea levels (Reed, 1995; Cahoon et al., 2006). The total below-ground biomass produced by halophytes is a factor in dictating the elevation change of the marsh platform, and the sustainability of the marsh with respect to long-term coupling of the marsh surface with sea level (Stevenson et al., 1986; Reed, 1995). McKee and Patrick (1998) and Morris et al. (2002) find that natural *S. alterniflora* platform elevation strongly correlates with the elevation of mean high water (MHW).

The growth of most salt marsh species is negatively affected by increased flooding (Kirwan and Guntensbergen, 2012; Janousek and Mayo, 2013; Voss et al., 2013). Increased flooding upon a vegetated wetland platform lowers the redox potential, which in conjunction with more saline conditions, can generate elevated sulfide concentrations (McKee and Mendelssohn 1989). Higher sulfide conditions negatively affect wetland plants and can result in lower biomass production (Koch et al., 1990). Changes in flooding intensity, duration, and salinity may differentially affect above-ground and below-ground production, several studies suggest that flooding disproportionally adversely affects below-ground biomass root production (Rozema and Bloom, 1977; Langley et al., 2013; Janousek and Mayo, 2013). Wetland plants allocate growth in response to the availability of resources such as nutrients and light, with abiotic factors, flooding, pH, etc., also affecting their growth (Tilman 1988; Janousek and Mayo, 2013).

Primary productivity of salt marsh vegetation is regulated by changes in sea level (land subsidence plus eustatic change in sea level), sediment supply, and tidal range. The vegetation, as a result, constantly modifies elevation of the platform toward an equilibrium with sea level (Stevenson et al., 1986; Reed, 1995). *S. alterniflora* is considered an indicator and foundation species because of its ability to modify the physical environment to optimize its growth within tidal marshes (Pennings and Bertness, 2001).

McKee and Patrick (1988) find that the occurrence and growth range of *S. alterniflora* correlates with the local tidal range of a watershed. Although primarily confined to the intertidal zone, they propose that the occurrence limits do not directly correspond to a consistent elevation relative to a tidal datum in all locations (McKee and Patrick, 1988). The disparity in the vertical distribution of *S. alterniflora*, between different watersheds, is chiefly due to the differences in mean tide range (MTR). A positive correlation between MTR and growth range exists, but it is limited to the local tidal conditions within a study region (McKee and Patrick, 1988; Morris et al., 2002).

Morris et al. (2002) builds upon McKee and Patrick's (1988) findings and proposes that every marsh has: a theoretical optimum rate of relative sea-level rise (RSLR), an optimum depth below MHW at which the marsh community is most productive, and an equilibrium depth of inundation below MHW that can be greater or less than the optimum (Morris et al., 2002). The elevation of a marsh platform relative to mean sea level (MSL) is primarily one of the dominant factors in biomass productivity (Mendelssohn and Morris, 2000). *S. alterniflora*, primary production (both above- and below-ground biomass) varies throughout the marsh itself and within the tidal range and is found to be highest at the lower elevations of its vertical growth range (Pomeroy et al., 1981).

At lower surface elevations (and thus greater extent of inundation), growth of *S. alterniflora* is likely limited and the marsh plant community is replaced by un-vegetated tidal mudflats (Morris et al., 2002). A marsh platform positioned above its ideal elevation for biomass production is more sustainable because, in the future, it will endure a higher RSLR (Morris et al., 2002). There is an ideal marsh platform elevation for tidal wetland vegetation productivity, though it can differ by study area as a function of tidal range and other factors (McKee and Patrick, 1988). The constraint on productivity is an important factor in maintaining elevation in response to increased flooding and inundation. The evolution of the marsh platform could bring about increases in production and biomass density that could enhance sediment deposition by increasing the efficiency of sediment trapping (Gleason et al., 1979; Leonard and

Luther, 1995; Yang, 1998), and increasing below-ground biomass production, which increases the elevation from the bottom up (Reed, 1995).

Questions with respect to the growth range of *S. alterniflora* include: can calculated growth ranges be correlated across different watersheds within a region? and are the effects of elevation on below-ground biomass production consistent throughout a region? Being able to apply a standardized *S. alterniflora* growth range across different watersheds within a region allows for monitoring of below-ground conditions within a marsh and the potential sustainability of the marsh platform in response to changes in sea level. A greater understanding of the equilibrium between *S. alterniflora* elevation and the tidal datum increases the capability to assess platform stability. This can then be used in thin-layer application of sediment restoration efforts to optimize the quantity (i.e. thickness or depth) of sediment that could be added to a marsh to enhance *S. alterniflora* growth and long-term sustainability.

One of the main goals of this study is to examine the vertical growth range of *S. alterniflora,* with respect to MLW and MHW, and the effect of that range on above-ground and below-ground biomass production. This study is limited in that marsh platform elevation is only one of many factors that dictates the growth and health of *S. alterniflora*. However, it is one of the most important factors impacting the halophyte (Reed, 1995; Cahoon et al., 2006). Marsh elevation is well-documented as a first-order control in dictating wetland flooding frequency, length of inundation, available suspended sediment concentration, and type and density of vegetative cover (Morris et al., 2002). Because of its limited nature, this analysis is not expected to completely account for all above-ground and/or below-ground biomass variability or deviation in growth elevation with respect to MHW or MLW. It is theorized that altered hydrology (i.e. mosquito ditching, channelization, impounding, and water level management) affects the health of *S. alterniflora* wetlands in Delaware, and the data collected in this study within two heavily impacted watersheds, in addition to four watersheds with significantly lower levels of alteration, assist in determining if changes in hydrology directly impact *S. alterniflora* growth.

This study presents the correlation of tidal datum elevations within each of the study areas to the above-ground and below-ground biomass of *S. alterniflora*. The relationships are used to examine the effects of altered hydrology, and how the growth range for *S. alterniflora* can be used to assess the sustainability and restoration potential for wetlands. *S. alterniflora* is the dominant halophyte within Delaware's marshes and is considered by natural resources managers to be a key component in need of better understanding in evaluating the resiliency of tidal wetlands to rising sea level (DNREC, 2013).

2.2 Location

The biomass and elevation data were collected from six sub-estuaries (St Jones River, Blackbird Creek, Prime Hook Creek, Murderkill River, Broadkill River, and Bombay Hook Estuarine Complex (including the Leipsic, Mahon, and Simon's Rivers)) within the Delaware River and Bay Estuary (Figure 2.1). All of these sub-estuaries are microtidal (< 2 m), with ranges at their mouths of 1.28 to 1.75 m (Table 2.1). The tidal range increases north along the Delaware coast due to the influence of the constriction of the estuary (Sommerfield and Wong, 2011). The furthest upstream occurrence of short-form *S. alterniflora* varies between the sub-estuaries with in-channel distances ranging from 6,990 to 10,450 m (Table 2.1). The five more southern sub-estuaries are seasonally polyhaline, while the most northern sub-estuary can be polyhaline in the fall with spring salinity values of a more mesohaline range (Figure 2.1). All of these systems are heavily discharge dependent in their salinity values.

Biomass data for each study area were collected in mid-August to mid-September at the peak of biomass production. The six sub-estuaries were sampled over the course of four sampling seasons. Samples were collected at the St Jones River in 2008, 2009, and 2010; Blackbird Creek in 2009; Bombay Hook in 2010; Prime Hook Creek in 2011; Murderkill River in 2008 and 2009; and Broadkill River in 2011. The total number of sampling sites per sub-estuary was largely dependent upon the continuity of the short-form *S. alterniflora* platform and the maximum upstream distance of short-form *S. alterniflora* within the sub-estuary. The St. Jones River sub-estuary contained the most sampling sites with thirty-one, and many of these were revisited over the course of the three sampling seasons. The Bombay Hook Complex had fourteen, Broadkill River eleven, Murderkill River ten, Blackbird Creek eight, and Prime Hook Creek five sampling sites, respectively (Figure 2.2).

Table 2.1 Mean range of tides at their confluence with the Delaware Bay (mouth), the tide range at the maximum upstream extent of short-form *S. alterniflora,* and the in-channel distance of the maximum or last occurrence of short-form *S. alterniflora* for the six sampled sub-estuaries.

	Mean Range of Tide at Mouth of Estuary	**Tide Range at Maximum Upstream Extent of *S. alterniflora***	**Maximum In-Channel Distance Upstream of Last *S. alterniflora* occurrence**
Blackbird Creek	**1.75 m**	**0.84 m**	**8,750 m**
Bombay Hook Complex	**1.66 m**	**1.49 m**	**8,190 m**
St. Jones River	**1.57 m**	**0.75 m**	**10,450 m**
Murderkill River	**1.56 m**	**1.20 m**	**9,140 m**
Prime Hook Creek	**1.41 m**	**0.29 m**	**9,400 m**
Broadkill River	**1.28 m**	**1.13 m**	**6,990 m**

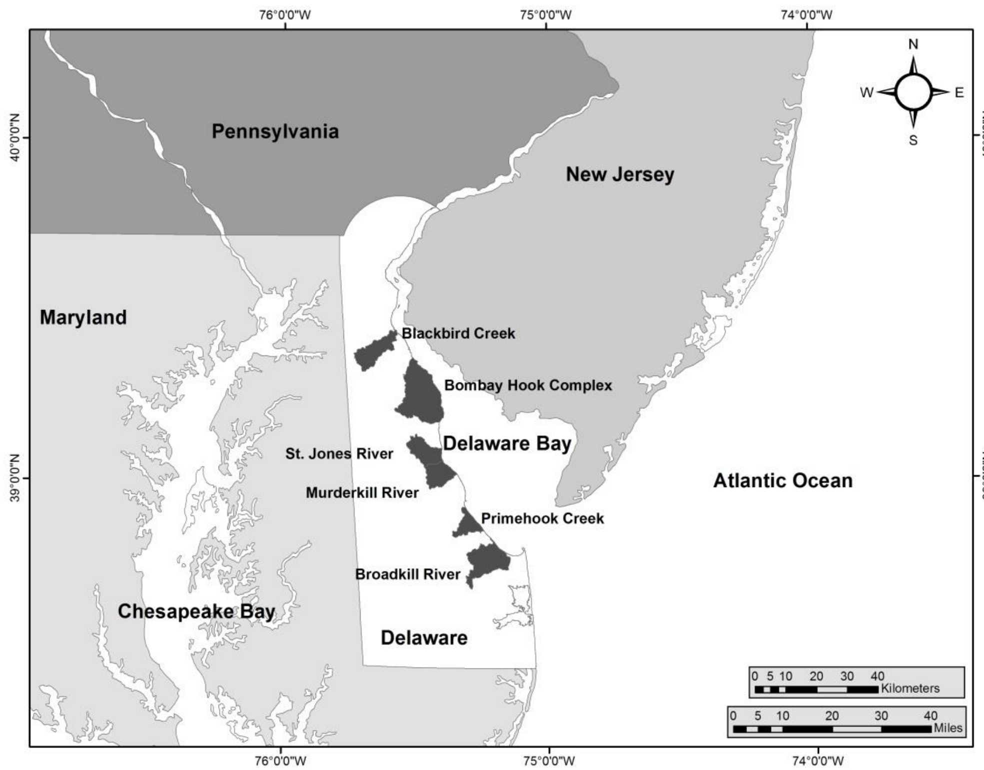


Figure 2.1 Location map for the six sub-estuaries sampled for short-form *S. alterniflora* biomass.

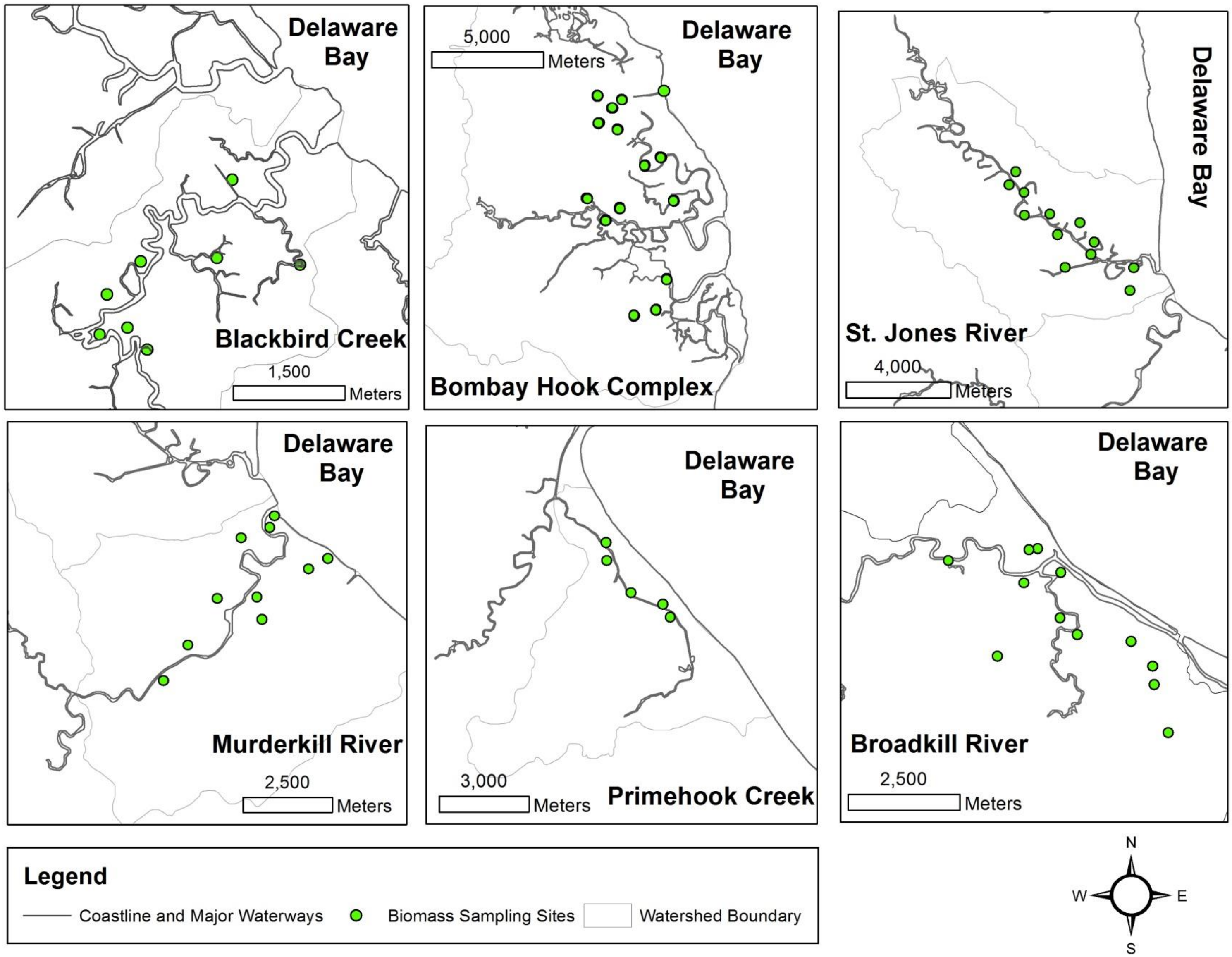


Figure 2.2 Locations of biomass collection sites within the six sub-estuaries sampled.

2.3 Methods

2.3.1 Biomass

Collection sites within each sub-estuary were chosen randomly in areas of the tidal marsh in which short-form *S. alterniflora* was dominant (greater than 80% percent) by the use of random point selection within ArcMAP (Hawth's Tools for Analysis; SpatialEcology, 2007). All sites were visited prior to sampling to verify the presence and percentage cover of *S. alterniflora.* Sites that did not contain the necessary cover were excluded, and the next random point was selected off of the generated numerical point list. Each sampled sub-estuary contained a minimum of five collection sites.

Biomass samples of *S. alterniflora* were collected by adapting procedures outlined in the Mid-Atlantic Tidal Rapid Assessment Method (MidTRAM; Jacobs and Bleil, 2008; Jacobs, 2010). At each selected site, the center point was defined based upon the presence of *S. alterniflora*; then three sample sites were selected 25 m from the center point along three of the cardinal directions using the closest body of water as the starting A direction, and then moving clockwise every 90° to select B and C directions. At each sampling location, a metal 10.2 cm (4 inch) diameter ring (with an area of 80.7 cm^2) was placed over the above-ground biomass portions and slid to the ground. All vegetation in the ring was clipped, bagged, later sorted based on whether the vegetation was alive or dead, and then placed in separately labeled bags. Dead stems on live plants were cut off and counted as dead. A 10.2 cm (4 inch) diameter core (30 cm in length, with a volume of 2,432.2 cm^3) was placed at the exact location of the ring, and pushed into the substrate. A shovel was used to dig an access hole next to the core and the substrate was cut flush with the bottom of the corer. The core was extracted from the ground and the sample was extruded from the core by pushing the core out from the top through the bottom. Upon extrusion, the core was cut to equal 15 cm length pieces (measured down from the top of the core). The root biomass samples were then rinsed in a 2,000 micrometer (μm) screen sieve bucket to separate the root material from sediments (i.e. clay, silt, and sand particles). The root material was then placed into labeled sample bags.

Below-ground biomass samples were washed with fresh water to remove any remnant sediments. Each sample was examined for live roots and rhizomes, and separated from the dead vegetation. Once live and dead roots were separated, they were rinsed with freshwater over a 2,000 μm filter and placed into separately labeled plastic bags. The samples were then transferred to labeled paper bags and placed in a 75 degree Celsius oven for 72 hours. At the end of this time period, samples were weighed and returned to the oven for approximately 6 hours and weighed again to confirm samples were completely dry. Biomass samples were then removed from the bags and re-weighed (Jacobs and Bleil, 2008; Jacobs, 2010). Each collection site's above-ground and below-ground biomass data were defined as an aggregate of the three subsamples for each collection site. Mean values were used to reduce variance due to spatial variability in biomass production along the marsh platform (Jacobs and Bleil, 2008).

2.3.2 Elevation Surveys

At each biomass collection site, marsh surface elevations were recorded using a Real-Time Kinematic (RTK) Global Navigation Satellite System (GNSS). All field elevation and position surveying were conducted using a Trimble 5700 RTK- receiver system. All surveyed points were referenced to the 1983 North American Datum (NAD 83), which uses the Geodetic Reference System of 1980 (GRS 80) ellipsoid as the reference surface for three-dimensional positions. Vertical position was referenced to the 1988 North American Vertical Datum (NAVD 88). At each of the three subsampling locations at each of the biomass collection sites, fifteen elevation measurements were recorded (Figure 2.3). Using the forty-five data points (fifteen from each of the three subsampling locations), a mean elevation for each biomass collection site was then calculated. Minimum, maximum, and the calculated mean elevations were tabulated for each biomass collection site.

2.3.3 Tidal Datum Calculations

The growth elevation of a wetland is controlled by the tidal elevation and range; hence, the elevation at all sites was correlated to the tidal datum elevations MLW, mean tidal level (MTL), and MHW. In addition to a USGS tide gauge and a

NOAA tide gauge, twelve water level recorders were located throughout the study areas (two in the St. Jones River, three in the Blackbird Creek, four in the Bombay Hook Complex, and one each in the Murderkill River, Prime Hook Creek, and Broadkill River sub-estuaries, respectively). Water level data were collected every 15 minutes at these locations using YSI 6000 data sondes (six sites) and OnSet HOBO water level recorders (six sites), and logged into year-long databases. All water level sondes were surveyed to NAVD88 through static RTK surveys. MLW, MTL, and MHW levels were calculated for each site using all available time-series data and the California Department of Water Resource's Delta Simulation Model II (DSM2). All other watershed and Delaware Bay tidal datums needed for the analysis were calculated using a combination of NOAA's VDatum (http://vdatum.noaa.gov/) for tidal elevations in the Delaware Bay, and previous hydrologic and Total Maximum Daily Load (TMDL) studies on the Murderkill River (HydroQual, 2013) and Broadkill River (HydroQual, 2006; Li, 2006).

The spot tidal datum calculations were used to create tidal datum curves along the axis of each sub-estuaries' short-form *S. alterniflora* distribution range. The axis started at the mouth of the sub-estuary at the Delaware Bay and extended up-estuary, within the main channel, to the furthest extent of *S. alterniflora* (Figure 2.4). Best-fit tidal curves were constructed for each sub-estuary; all sample sites were plotted on these curves in accordance with their distance up-estuary, to determine their MHW, MTL, and MLW elevations at each sampling site (Figure 2.4; Appendix A). Due to the

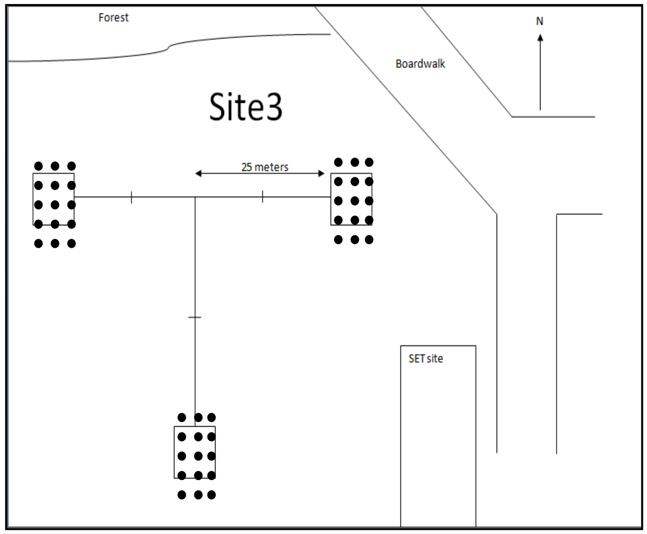


Figure 2.3 Example schematic of biomass site collection overlain with RTK elevation collection points.

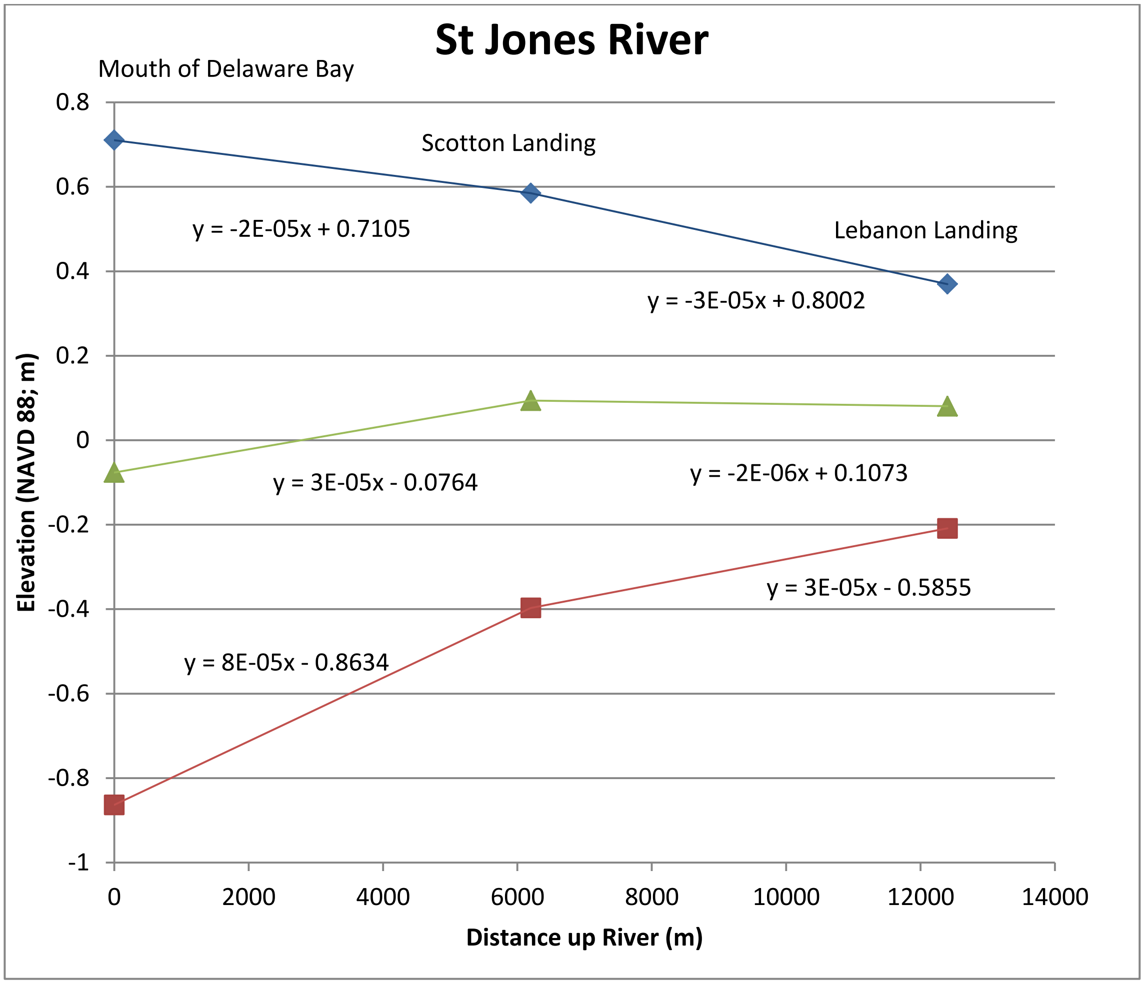


Figure 2.4 Tidal elevations along the St Jones River. The 0 m distance marks tidal elevations at the mouth of the river at the Delaware Bay, the 6,250 m distance tidal elevations at Scotton Landing, and the 12,650 m distance tidal elevations at Lebanon Landing. Blue line represents MHW; green line MTL, and red line MLW tidal elevations, respectively.

limitations of the spatial distribution of the tidal data collection points, the tidal curves between sampling points are linear. This is a simplistic approximation for the likely much more complicated tidal variations within the sub-estuaries sampled. However, the MLW, MTL, and MHW heights as predicted by the simple linear tidal curves provide first-order constraints on the limits of vegetation growth relative to tidal levels.

Of the six estuaries studied, three are typical valley-fill estuaries (St Jones River, Blackbird Creek, and Murderkill River). Two have complex multiple tidal water sources flowing into large fringing marshes (Bombay Hook Complex and Broadkill River), and one is characterized by a back-barrier tidal marsh with two tidal sources into the marsh but only one exit point due to a unidirectional water control structure (Prime Hook Creek).

2.4 Results

The above-ground and below-ground biomass data for all sites were compared to their marsh elevation relative to the MLW and MHW tidal datums to determine if there is a correlation between marsh platform elevation and the amount of biomass produced (Figures 2.5 – 2.9).

2.4.1 Above-ground Biomass

Above-ground biomass data were parsed into live versus dead components and correlated with the MLW and MHW tidal datums at each site. The live above-ground biomass values varied between 3.4 and 15.9 grams (g). The live fraction of the total above-ground biomass showed no apparent correlation to the MLW tidal datum. The live biomass data were spread over the entire tidal range (1.78 m), with the highest occurrences between 1.78 and 0.65 m above MLW, and 0.22 above and -0.18 m below MHW. The above-ground dead biomass data were more closely clustered with respect to the range of masses from 1.8 to 19.4 g, with the highest number of values clustering between 1.8 and 8.9 g. The dead biomass also showed no direct correlation to site elevation with respect to MLW.

The total above-ground biomass (sum of live and dead) also show no relation to the marsh elevation platform relative to MHW or MLW elevations (Figures 2.5 and 2.6). The range of biomass values in relation to MHW is highly variable with no apparent trends or clustering of values (Figure 2.5). The distribution of biomass values in relation to MLW is not as variable as MHW, but no obvious trends in the data are apparent (Figure 2.6).

2.4.2 Below-ground Biomass

Much like the above-ground biomass, the live, dead, and total below-ground biomass showed no direct correlation to the MHW and the MLW datums. This is

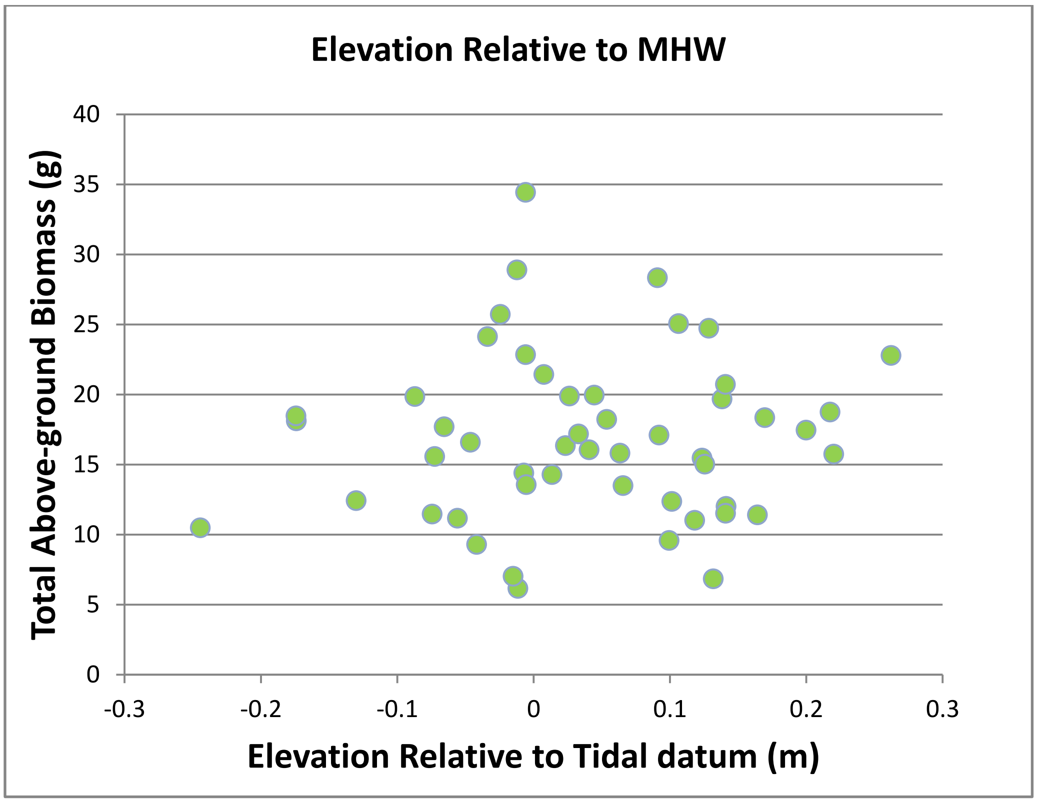


Figure 2.5 Total above-ground biomass versus mean site elevations relative to MHW.

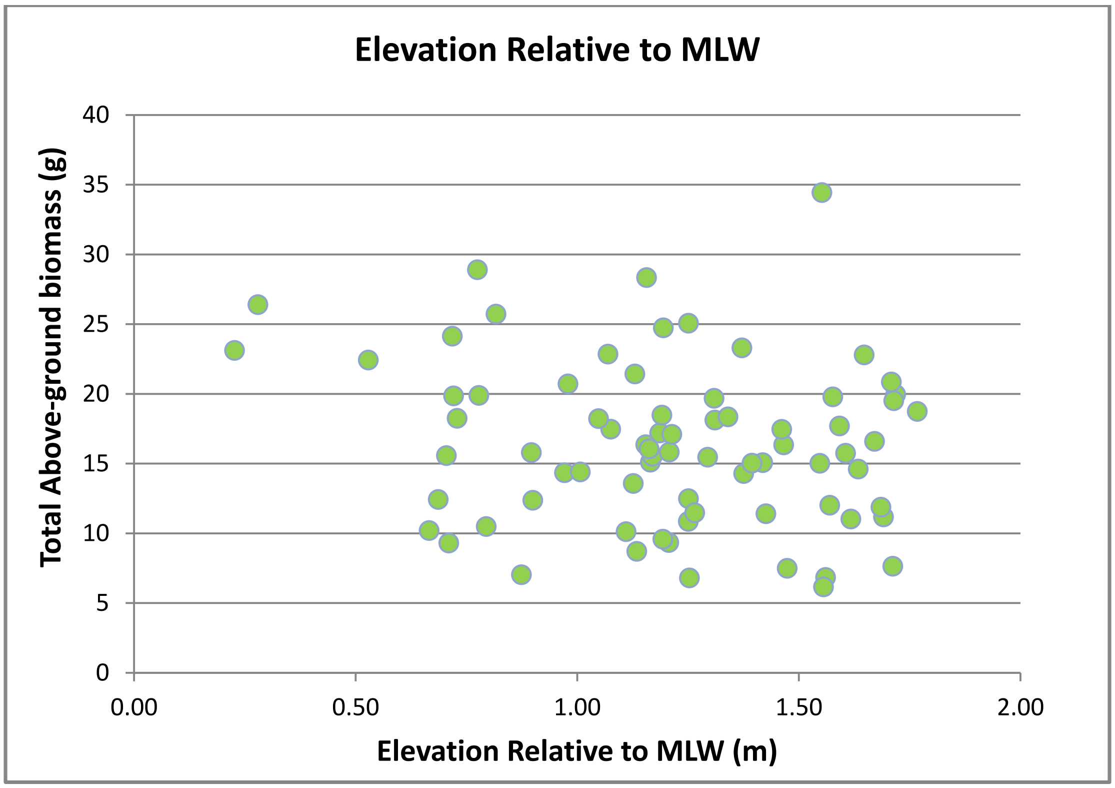


Figure 2.6 Total above-ground biomass versus mean site elevations relative to MLW.

somewhat of an unexpected result as previous research showed a normal to skewed relationship between biomass and MHW elevation (Morris et al., 2002). Morris et al. (2002), based on biomass data from North Inlet in South Carolina, found that there is an optimal depth of inundation, with maximum biomass production around MHW.

In this dissertation project, the below-ground biomass relative to MHW and MLW (for the relatively un-impacted sub-estuaries) plot within envelopes of distribution (Figures 2.7 and 2.8). The upper limit and lower limits of the envelopes define maximum and minimum growth potentials as a function of tidal elevation, respectively (Figures 2.7 and 2.8). The lower growth limits of biomass relative to MHW are on the order of -0.3 m (Figure 2.7); relative to MLW, the lower limit is on the order of 0.7 m (Figure 2.8). Relative to MHW, the largest amounts (>200 g per core) of below-ground biomass occur within elevations between -0.006 and 0.13 m (Figure 2.7). Relative to MLW, the largest biomass amounts (>200 g per core) occur within sites at elevations between 1.25 and 1.72 m (Figure 2.8).

When the total below-ground biomass from the impacted Broadkill River and Bombay Hook Complex sub-estuaries are plotted with data from the four relatively un-impacted watersheds, large differences are observed. It is hypothesized that watersheds should have similar normal distribution patterns with respect to below-ground biomass and MHW (Morris et al., 2002). Instead, the Broadkill River data show a bimodal distribution in total below-ground biomass with one group of data points with lower biomass values ranging between 170 and 210 g, and a second group with much higher biomass values ranging between 260 and 285 g (Figure 2.9). This bimodal distribution is not related to marsh elevations relative to tidal datum, as the two groups have similar elevations (ranging between -0.11 and 0.08 m) relative to the MHW tidal datum (Figure 2.9).

The Bombay Hook Complex total below-ground biomass data are characterized by a wide range in values from 100 to 230 g, with eleven of the fourteen sampling stations having elevations below the MHW datum elevation (0.00 m; Figure 2.9). The data roughly aggregate with respect to their study areas (i.e., North, Central, and South within the complex; Figure 2.9). In the North Bombay Hook portion of the complex, sample locations 20, 21, 22, 23, and 24 cluster having a mean total below-ground biomass of 139.4 g (SD = 8.20) with values ranging between 125 and 145 g (Figure 2.9). The wetland platform elevations at these locations are all negative relative to MHW, although the relative elevations have a wide range between -0.27 and -0.06 m, with a mean value of -0.13 m (SD of 0.09; Figure 2.9). The biomass and elevation data for this study area generally fit within the values observed for the St. Jones River, Blackbird Creek, Murderkill River, and Prime Hook Creek sub-estuaries, which are not significantly altered by anthropogenic activities.

Sample locations 1, 13, 14, 15, 16, and 17 are located in the central portion of the Bombay Hook Complex along the Leipsic River and Lower Duck Creek (Figure 2.10). Sample location 14, which is the only site south of the Leipsic River in this portion of the study area, has a total below-ground biomass value (104 g) that is much less than the other five central sampling locations to the north (Figure 2.9).

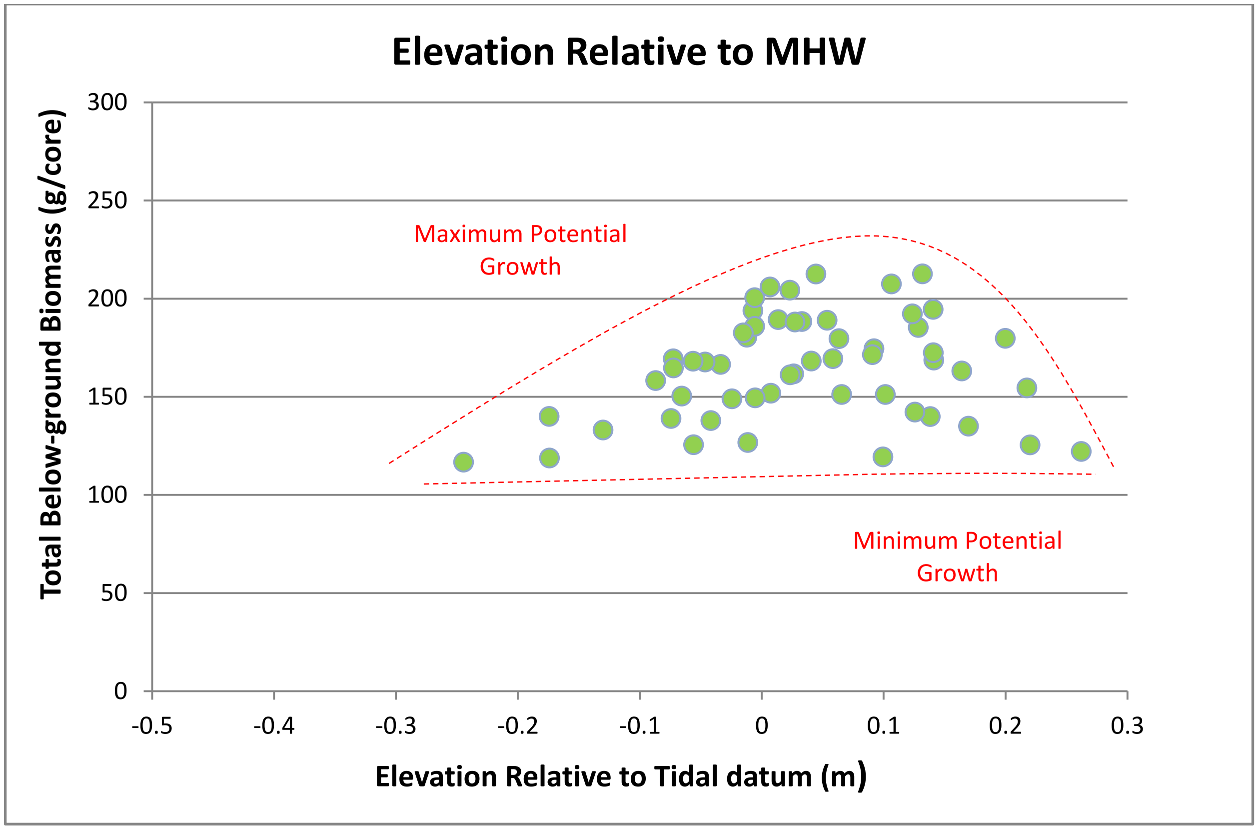


Figure 2.7 Total below-ground biomass for the four relatively un-impacted watersheds (St. Jones River, Blackbird Creek, Murderkill River, and Prime Hook Creek) versus mean site elevations relative to MHW.

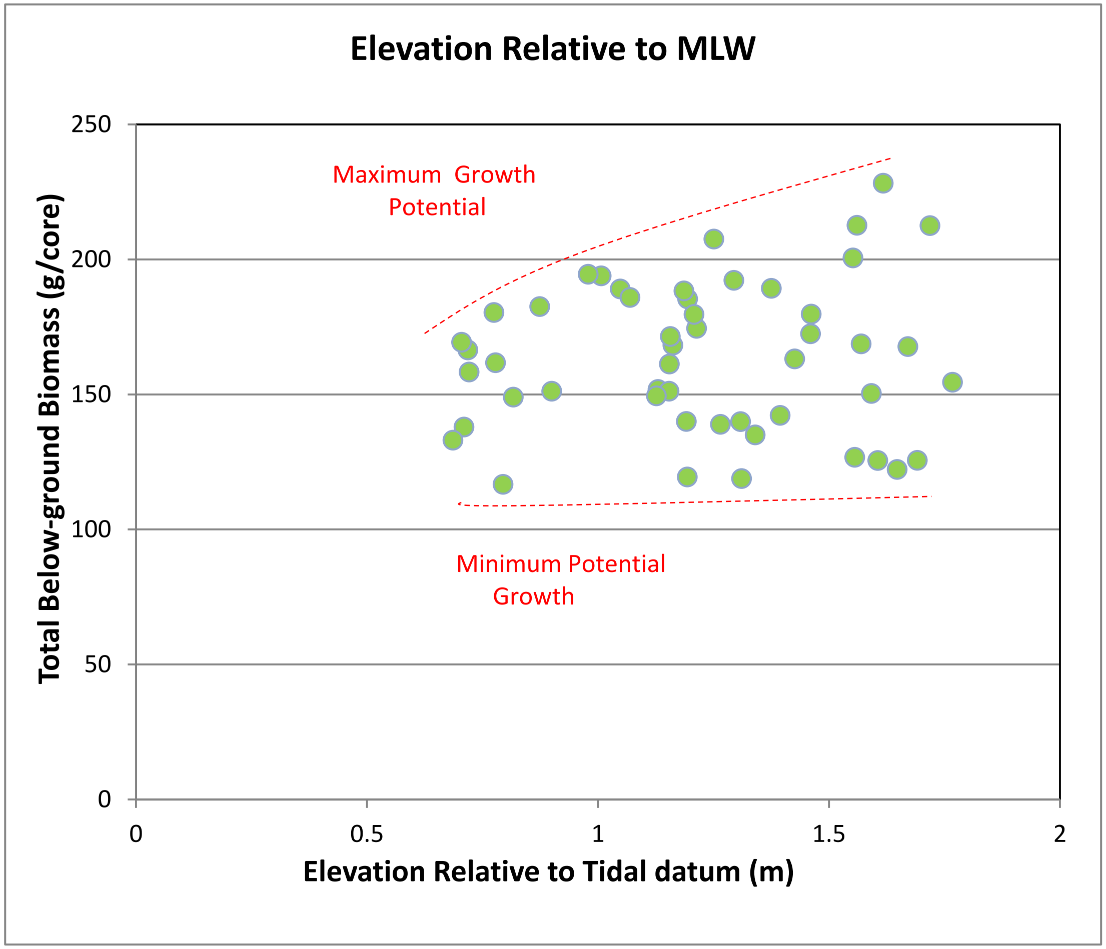


Figure 2.8 Total below-ground biomass for the four relatively un-impacted sub-estuaries (St. Jones River, Blackbird Creek, Murderkill River, and Prime Hook Creek) versus mean site elevations relative to MLW.

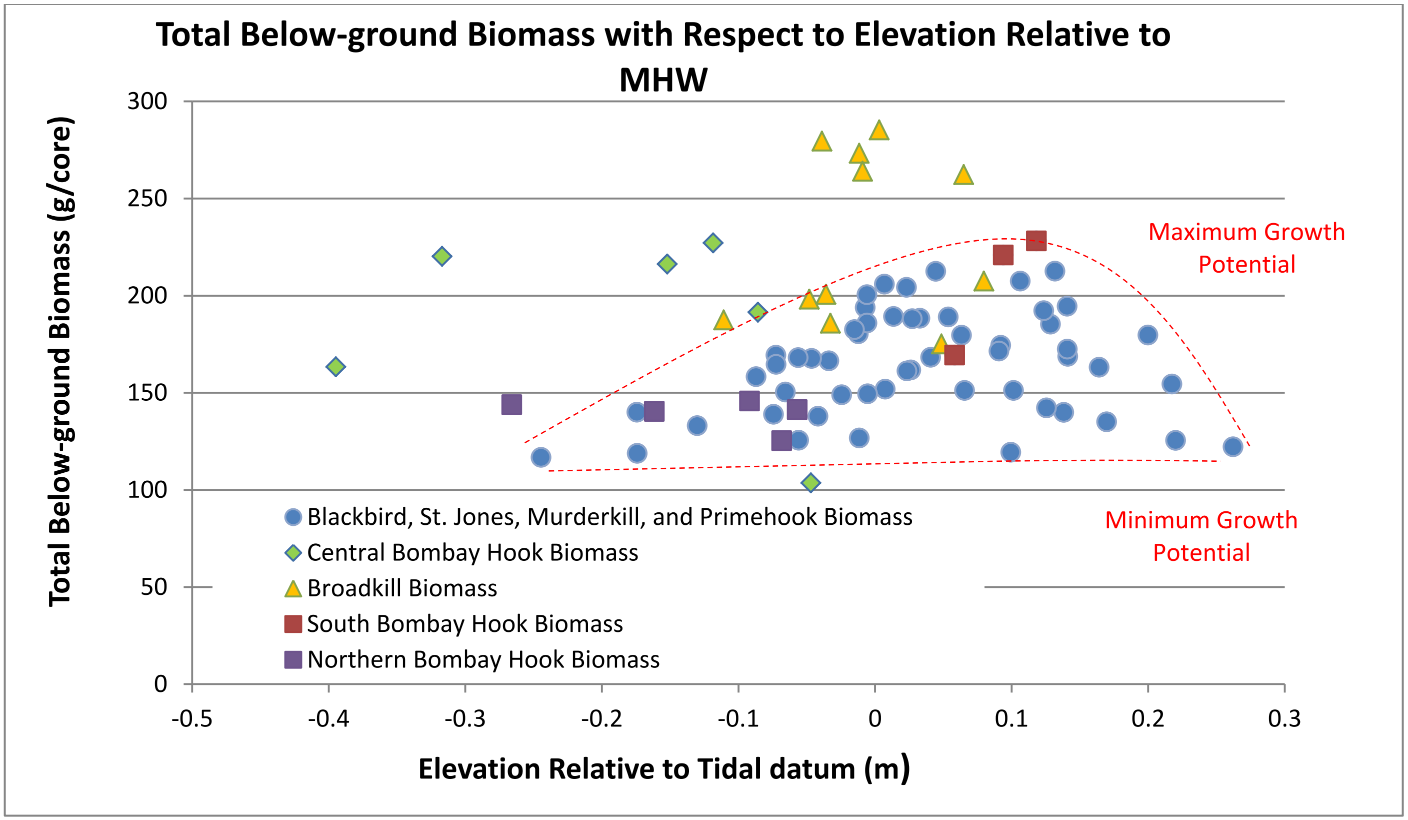


Figure 2.9 Comparison of the total below-ground biomass of the St. Jones River, Blackbird Creek, Murderkill River, and Prime Hook Creek (blue circles) to the biomass from the Broadkill (orange triangles), North Bombay Hook (purple squares), Central Bombay Hook (green diamonds), and South Bombay Hook (red squares) study areas versus mean site elevations relative to MHW.

Sample locations 10, 11, and 12 (which are located in the South Bombay Hook portion of the complex; Figure 2.10) are located in an intensively mosquito-ditched marsh. These sites are the only Bombay Hook Complex locations with net positive (above MHW) wetland platform elevations. Their mean elevation relative to MHW is 0.09 m (SD = 0.03). These three sites are characterized by relatively high mean biomass of 206.2 g (SD=32.03; Figure 2.9).

Sampling locations 1, 13, 15, 16, and 17 all contain below-ground biomass values that are greater than the other four sampled sub-estuaries, and the northern portion of the Bombay Hook Complex relative to MHW elevations (Figure 2.9). These five sampling sites have a mean total below-ground biomass of 203.7 g (SD =26.2), with a range between 163 and 227 g. These moderate to relatively higher biomass values are associated with negative elevations relative to MHW, with a mean elevation of -0.21 m (SD = 0.14) and ranging from -0.39 to -0.09 m (Figure 2.9).

2.4.3 Above-ground versus Below-ground Biomass

Earlier studies suggest that the total below-ground biomass and the total above-ground biomass are related in their content and proportion (McKee and Patrick, 1988; Reed, 1995; Mendelssohn and Morris, 2000; Pennings and Bertness, 2001). However in this study, no apparent trend or relationship is observed (Figures 2.11 and 2.12). The live above-ground to live below-ground biomass comparison shows a scattered

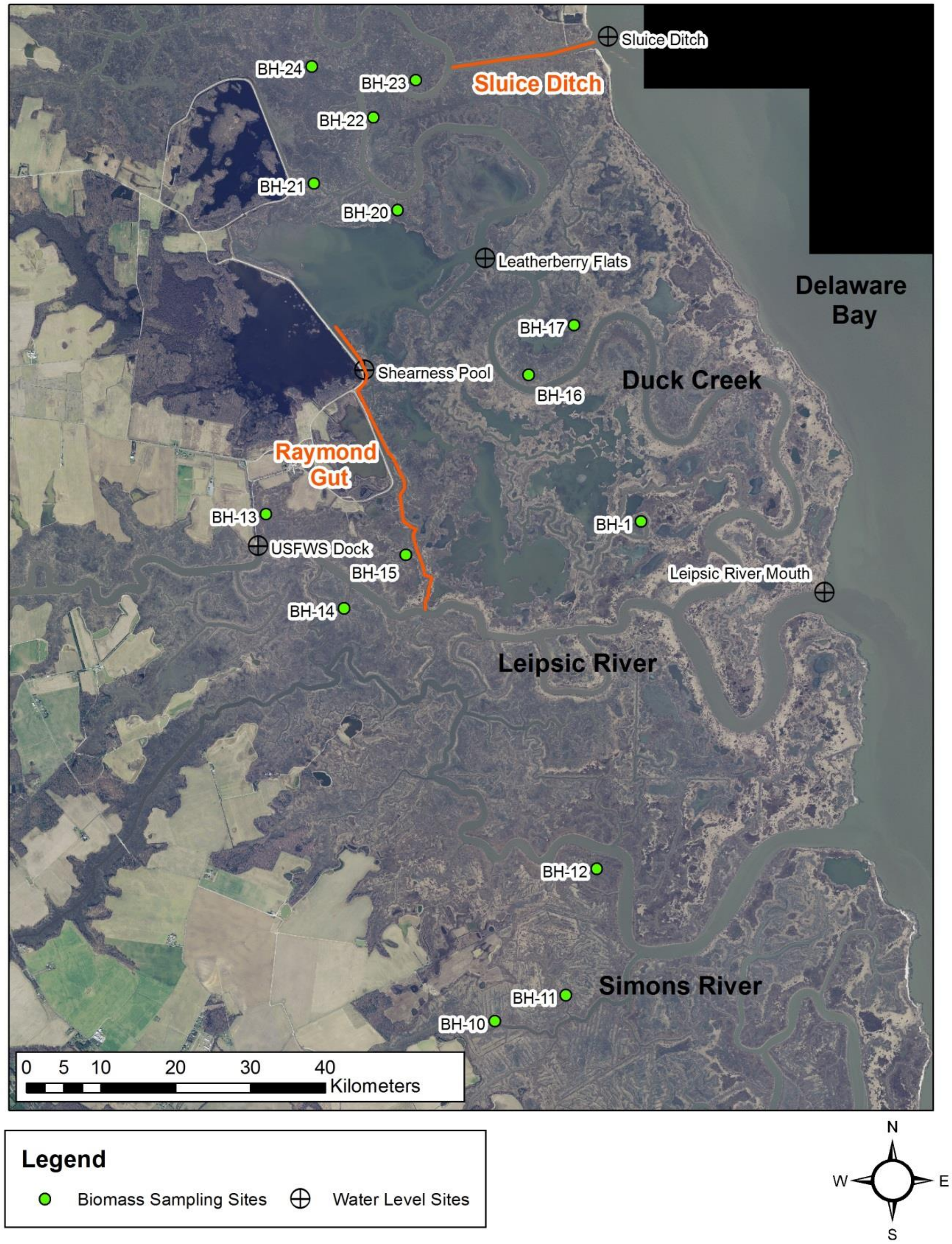


Figure 2.10 Map of Bombay Hook Complex biomass sample locations.

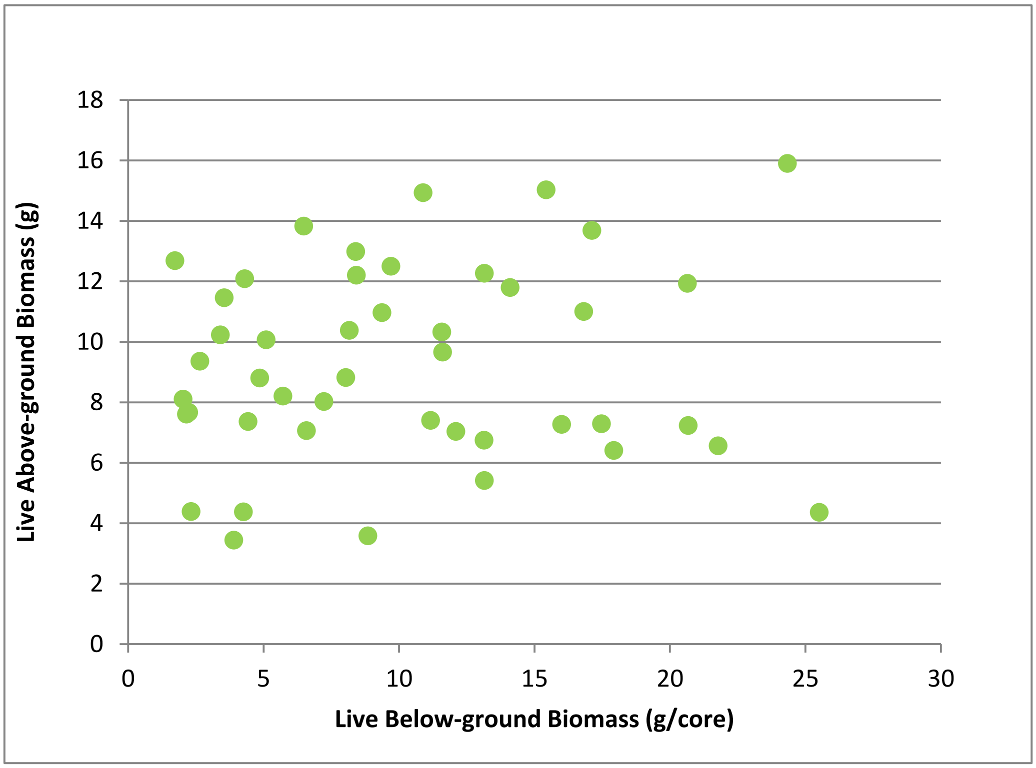


Figure 2.11 Live above-ground versus live below-ground biomass at all sampled sites.

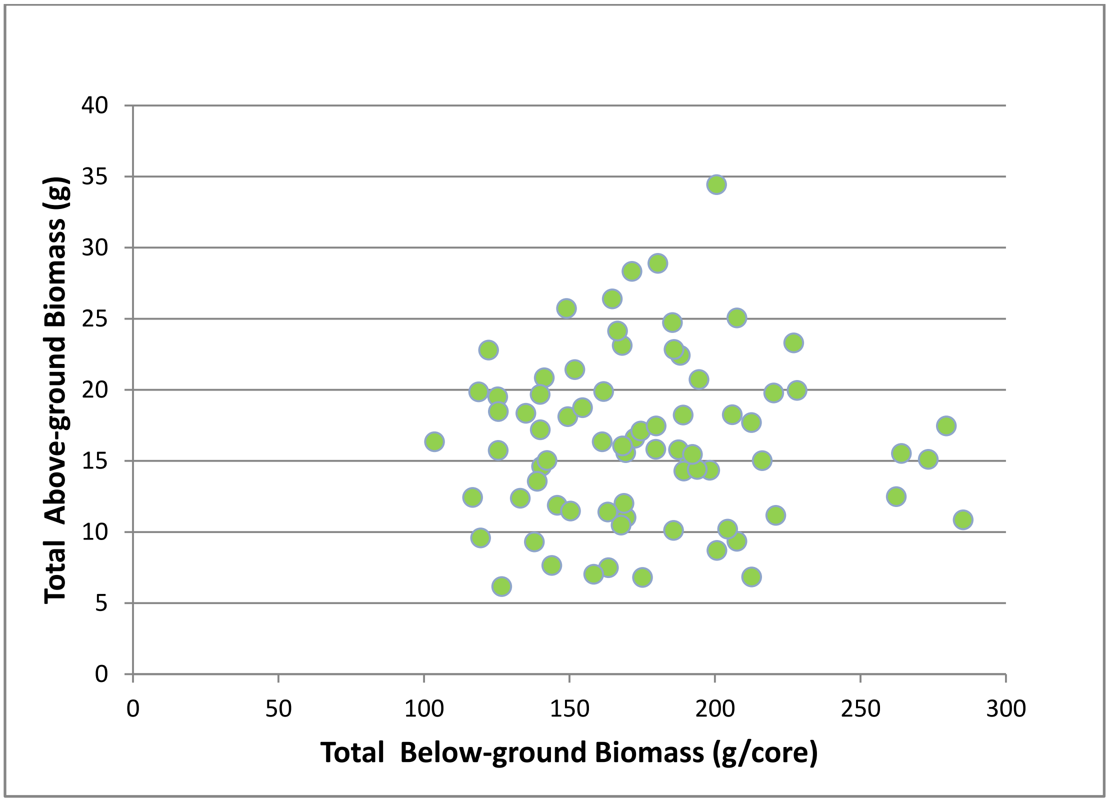


Figure 2.12 Total above-ground versus total below-ground biomass at all sampled sites.

distribution over ranges of biomass for above-ground from 3.4 to 15.9 g and below-ground from 1.7 to 25.5 g (Figure 2.11). Similarly, there is no discernable correlation between total above-ground and total below-ground biomass (Figure 2.12). The above-ground biomass varies between a minimum of 6.0 and a maximum of 34.0 g; the total below-ground biomass ranges between 104.0 and 285.0 g (Figure 2.12).

2.4.4 Tidal Range

Five of the six watersheds studied (Blackbird Creek, Broadkill River, Murderkill River, Prime Hook Creek, and St. Jones River) show a general relationship of increasing marsh platform elevation relative to MHW as the tidal range increases (Figure 2.13). Of these five, Prime Hook Creek and St Jones River show a slight lowering in their platform elevation relative to MHW near their maximum tidal range (Figure 2.13). These slight dips in platform elevation relative to MHW mark the upper elevational extent of *S. alterniflora* growth ranges in these sub-estuaries. Elevations vary between the two sub-estuaries due to the differences in tidal range between Prime Hook Creek (maximum of 0.72 m) and the St Jones River (maximum of 1.55 m; Figure 2.13). The optimal *S. alterniflora* platform elevation is a function of the tidal range within a watershed, as larger tidal ranges will result in a higher upper limitation for the optimal platform elevation.

The Broadkill River and Bombay Hook sub-estuaries have anomalous trends. Bombay Hook shows a negative relationship of decreasing marsh platform elevation relative to MHW as the tidal range increases. The Broadkill River sub-estuary shows a

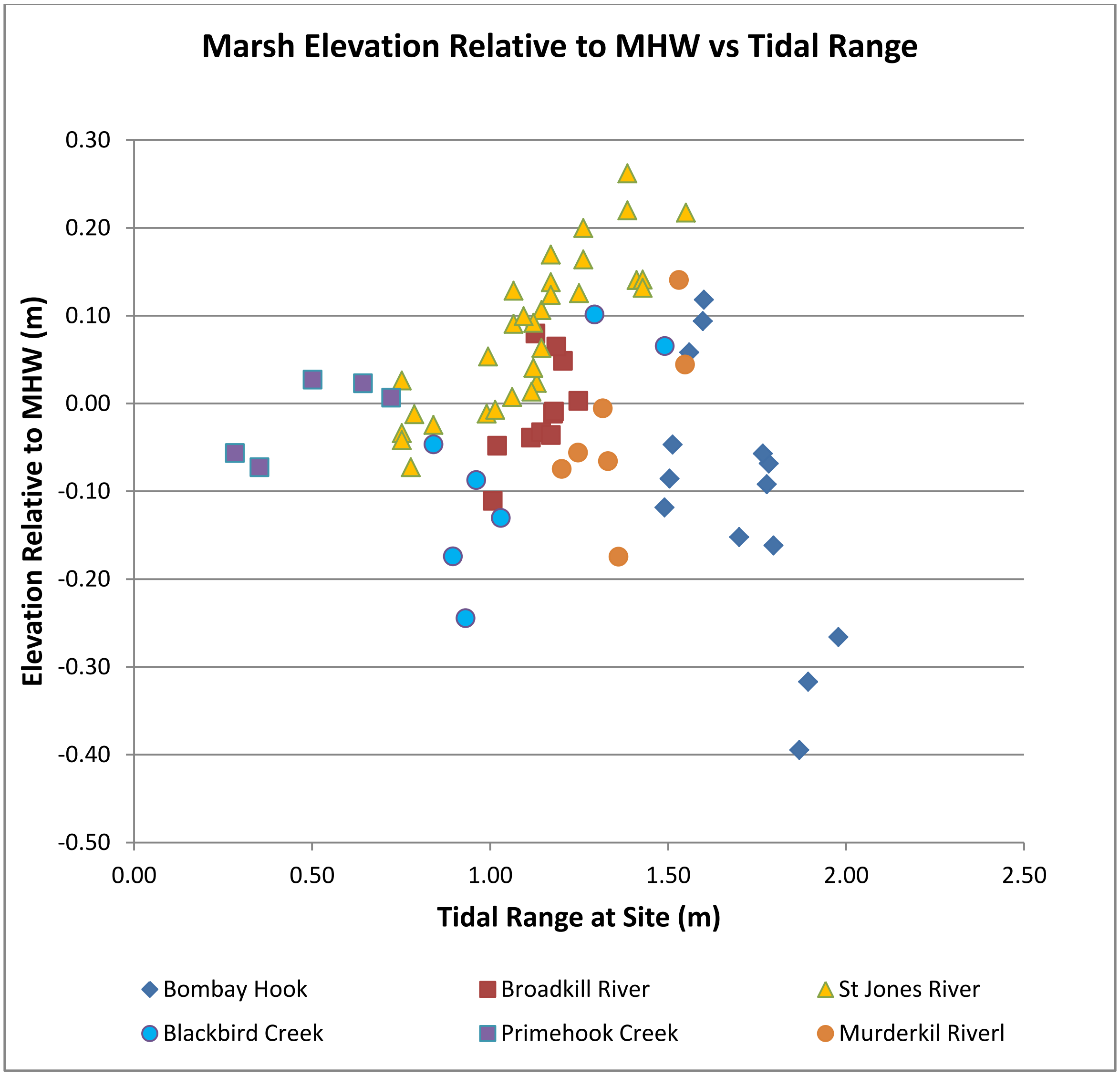


Figure 2.13 Elevation relative to MHW of the *S. alterniflora* sample sites versus calculated tidal range for each site.

wide range with overlapping values of elevations relative to MHW at the same or similiar tidal ranges (above 1.14 m).

In contrast to the MHW relationships, the tidal range and marsh platform elevation relative to MLW exhibit a linear relationship. It is not surprising that as the tidal range increases, the elevation relative to MLW increases above the MLW datum (0.0 m). This trend illustrates that *S. alterniflora* growsat higher elevations relative to the MLW datum as the tidal range increases. This trend is most likely due to the limitations on growth caused by the increasing range of inundation. At tidal ranges above ~1.55 m, the data suggest that marsh platform elevations plateau at ~1.70 m and no longer increase with increasing tidal range (Figure 2.14).

2.5 Discussion

A variety of data have been analyzed to determine the relationships between marsh platform elevation (relative to tidal datums) and biomass production for short-form *S. alterniflora*. The determination of the elevation growth range for *S. alterniflora*, allows for a way to expediently assess the condition and future sustainability of a *S. alterniflora* marsh platform. The growth range determinations provide a more precise means of targeting a desired elevation to optimize restoration of *S. alterniflora*. The restoration can be accomplished by improving below-ground biomass production thereby enabling marshes to better keep pace with rising sea levels.

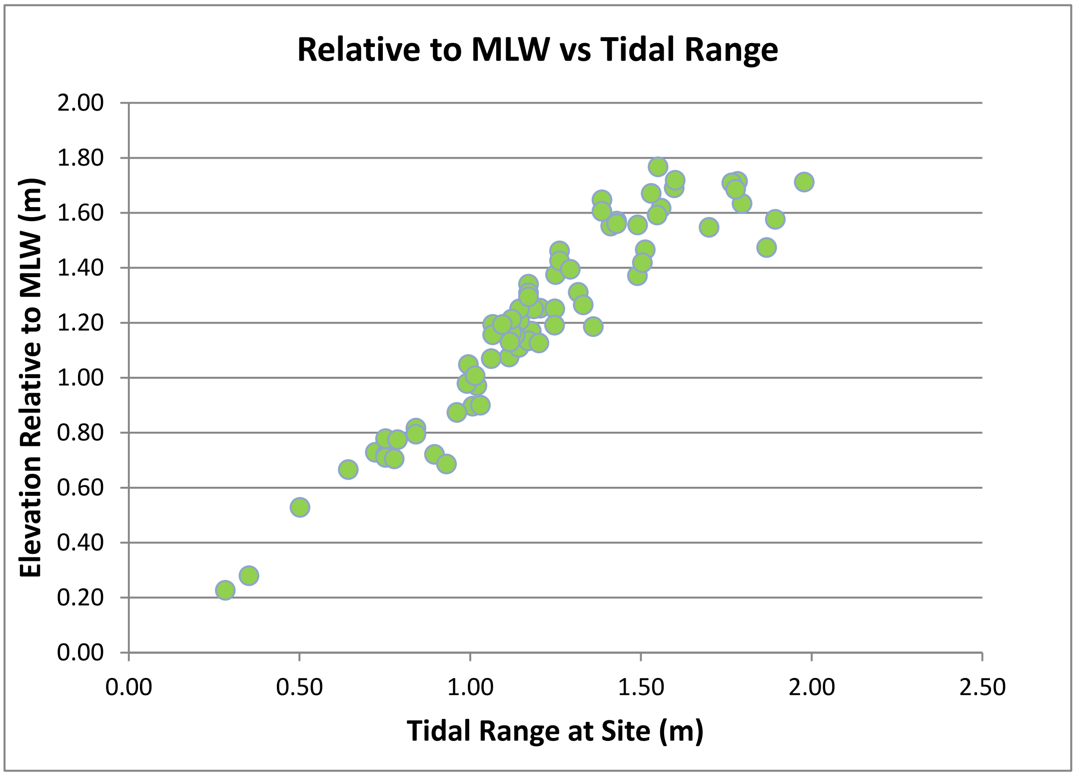


Figure 2.14 Marsh elevation relative to MLW (m) for the *S. alterniflora* sample sites versus the calculated tidal range for each site.

2.5.1 Above-ground Biomass

The collected and analyzed above-ground biomass (both live and dead) show no obvious correlation with marsh elevation relative to MHW and MLW (Figures 2.5 and 2.6). The above-ground production and biomass do not appear to be influenced by the elevation of the site, in relation to the tidal prism. This is not a unique result based upon previous studies that indicate that above-ground biomass production is largely a product of available nutrient resources and the overall stress on the halophytes (Valiela and Teal, 1974; Morris, 1995; Blum, 1993; Visser et al., 2006; Dary and Turner, 2008). The presence of above-ground biomass may appear to show a healthy vegetative platform, but it could also represent an over-eutrophied marsh that is expanding on the surface rather than increasing its vital below-ground biomass (Morris, 2002; Darby and Turner, 2008).

2.5.2 Below-ground Biomass

The relationship of below-ground biomass to marsh elevation platform (relative to a site's tidal datum) shows the most significant correlation. The total below-ground biomass data from the four sub-estuaries (St. Jones River, Blackbird Creek, Murderkill River, and Prime Hook Creek) that are characterized by minor human-induced alterations that could affect the wetland hydrology (e.g., localized mosquito ditching) fit within defined minimum and maximum growth potential curves (Figure 2.7). In contrast, the two sub-estuaries (Broadkill River and Bombay Hook Complex) that contain intensive human alteration through mosquito-ditched and dredging of new waterways are characterized by biomass values that are much more scattered and tend to show much higher below-ground biomass as a function of elevation relative to tidal datums than predicted by the maximum growth potential curve (Figure 2.9).

2.5.3 Effect of Anthropogenic Alterations on Hydrology

As Figures 2.7 and 2.9 illustrate, the total below-ground biomass as a function of marsh platform elevation relative to MHW shows a stronger correlation within the four sub-estuaries (St. Jones River, Blackbird Creek, Murderkill River, and Prime Hook Creek) that have experienced less anthropogenic alteration than the *S. alterniflora* wetlands in the Bombay Hook Complex and the Broadkill River sub-estuaries. The Bombay Hook Complex and the Broadkill River *S. alterniflora* wetlands have a higher prevalence of human-induced alteration that translates into noticeable effects upon the *S. alterniflora* below-ground biomass production. The effect on below-ground biomass production is likely the result of changes in hydrology in the area. The Broadkill River below-ground biomass data has a distinct split into two modes. The upper or higher value data points (with total below-ground biomass values above 260 g) were sampled in areas of intensive mosquito-ditching. The remaining six sampling locations were either distal to the intensive ditching work, or at a proximal location to the headlands at the upper extent of the sampled portion of the sub-estuary (Figure 2.9).

The cause of the higher than normal below-ground biomass production which coincides with areas of intensive ditching, is unknown and unfortunately the level of data collected for this study cannot adequately address this question. The Broadkill River sub-estuary is highly eutrophied with an excess nitrogen and phosphorus load that has been proposed to be cut by up to 40% through the 2006 TMDL (HydroQual, 2006). Previous studies document that excess phosphorus, not nitrogen, appears to induce a response of decreased production of below-ground plant biomass as more phosphorus becomes readily available (Darby and Turner, 2008). Nutrient sampling from 2001 to 2011 at the two main tributaries into the Broadkill River marsh system (Broadkill River at the Route 1 Bridge and Red Mill Pond), shows no statistical change with respect to dissolved inorganic nitrogen (DIN) or dissolved inorganic phosphorus (DIP) concentrations (DEOS, 2013). The total loads remain elevated above the proposed TMDL reduction levels (DEOS, 2013). This negates any hypotheses that lower or optimal nutrient levels could exist in the Broadkill River sub-estuary, allowing the system to optimize below-ground biomass production. It is likely that localized ditching has increased the marsh surface elevation due to the spoils being side-cast along the ditches, increased the rate of flushing, and reduced the effects of flooding toxicity on the adjacent wetlands. It is unknown if these conditions are sustainable.

The central biomass samples from the Bombay Hook Complex contrast with the other four relatively un-impacted sub-estuaries and the Broadkill River (non-interior samples) by having higher below-ground biomass values produced at significantly lower marsh surface elevations below MHW (Figure 2.9). This may be an indication that a different hydrology, resulting in a larger tidal prism and higher MHW levels, may exist for this central region. It is of note that the northern and southern sites do plot in closer proximity to the other sub-estuaries, with the southern sites exhibiting higher biomass production and marsh platform elevations than the northern sites (Figure 2.9).

The data for the Bombay Hook Complex indicate that distinctly different wetland conditions, from north to the south, are affecting marsh platform elevations and overall below-ground biomass production. The central section of the complex appears to be a transition from the lower less productive *S. alterniflora* marshes in the northern section of the complex to the highly productive and subsequently higher elevation *S. alterniflora* marshes in the southern portion (Figure 2.9). The negative effects or factors that are reducing productivity to the north, are not yet pervasive enough to be influencing the southern sites.

Tidal circulation is primarily brought into the interior wetlands of the Bombay Hook Complex by two major creek systems, the Leipsic River (running roughly east to west) and Duck Creek (running northwest to northeast; Figure 2.10). The interior wetland hydrology was likely altered when Raymond Gut was dredged prior to 1926 (likely in the late 1800's) and Sluice Gut was dredged prior to the 1890's (Figure 2.10; USFWS, 2000). Aerial imagery from 1937 shows numerous large interior pools that continue to expand during subsequent imagery surveys (i.e. 1954, 1961, 1968, 1992, etc.). Raymond Gut connects the tidal flow from the Leipsic River to the wetland interior areas near Sheerness Pool. Sluice Gut has been added to connect Duck Creek to open tidal waters at three points (two to the Delaware Bay and one to the Leipsic River). The tidal datums for the Bombay Hook Complex indicate that these two dredged guts increase the interior MHW elevation of Leatherberry Flats to 0.34 m higher than at the mouth of the Leipsic River, 0.44 m higher than the dock on the Leipsic River, 0.44 m higher than the mouth of Sluice Ditch, and 0.37 m higher than the MHW level at the mouth of Sheerness Gut (Table 2.2).

This anthropogenic alteration of tidal datums is significant as the short-form *S. alterniflora* in the complex has optimal growth and maximum below-ground biomass production at a level of 0.04 m (± 0.1 m) above MHW (Figure 2.9). Based upon independent spot elevation surveys of the interior vegetative marsh platform (between the Leipsic River to the south and Sluice Ditch to the north), the mean elevation is 0.16 m (SD 0.12 m) below the MHW elevation. For the interior wetland areas experiencing the most widespread degradation and loss (adjacent to Leatherberry Flats and south toward Duck Creek), the vegetative marsh platform has a mean elevation of 0.27 m (+/- 0.11 m) below MHW.

2.5.4 Optimal Growth Range for Below-ground Biomass

The main focus of this study is to calculate the optimal growth range for *S. alterniflora* where biomass production is maximized. A histogram of the below-ground biomass data from the four sub-estuaries (Blackbird, St. Jones, Murderkill, and Prime Hook) that are minimally altered, with a probability density function, was used to determine the range. The below-ground biomass data were aggregated into 0.08 m elevation bins relative to MHW to determine the frequency or count of values within each bin. The use of binning allows for smoother distribution of the biomass values analyzed and data breaks to be more readily observed. The higher frequency of values in a bin

Table 2.2 Bombay Hook Complex tidal datum elevations (meters NAVD 88) calculated using YSI 5500 data sondes (15 minutes measurement interval) water level and Vdatum. Leipsic River, USFWS Dock, Shearness Pool, and Leatherberry Flats datum elevations were based upon 15 minute continuous data collected from June 2009 to September 2011. Sluice Ditch tidal datum elevations were calculated using Vdatum.

	Leipsic River Mouth	USFWS Dock	Shearness Pool	Leatherberry Flats	Sluice Ditch
MHHW	1.02 m	0.88 m	0.96 m	1.35 m	0.92 m
MHW	0.90 m	0.80 m	0.87 m	1.24 m	0.80 m
MTL	0.09 m	0.14 m	-0.01 m	0.34 m	-0.05 m
MLW	-0.77 m	-0.62 m	-0.58 m	-0.63 m	-0.93 m
MLLW	-0.82 m	-0.70 m	-0.62 m	-0.70 m	-0.98 m

represents a higher prevalence of biomass values collected within that elevation range. A higher prevalence could represent that the elevation range was sampled at a higher frequency, which could be the result of a sampling bias. A lower prevalence, could also represent a sampling bias, or a lower occurrence of that range of elevations within the marsh.   The random sampling employed in the study lowers the probability of an elevation sampling bias, thus the occurrence of the elevation ranges is likely the result of actual elevation ranges within the marsh.

The histogram exhibits a skewed left or negative distribution (skewness -0.194) with the highest proportion of data between –0.07 and 0.09 m, and high values between 0.09 and 0.18 m (Figure 2.15). A fitted 2-parameter Weibull distribution probability density function shows a mean value in relation to MHW distribution of 0.036 m (SD=0.107 m) with a mode of 0.052 m (Figure 2.15). The Weibull slope shape indicates the rate of failure (which for this study would be the death of the plants) with respect to the parameter of interest (biomass growth); for this distribution, $\beta = 0.473$. When β is greater than 1, the failure rate decreases with increasing elevation relative to MHW (ReliaSoft, 2015). The Weibull distribution calculations have a ý-value of -0.396, which indicates that at an elevation of -0.396 m below MHW, the marsh platform has a 63.2% probability of complete vegetative collapse. This statistical threshold represents an elevation where the marsh would not be able to sustain itself or likely recover, and it would be expected to appear as a bare mudflat or permanently inundated shallow flat.

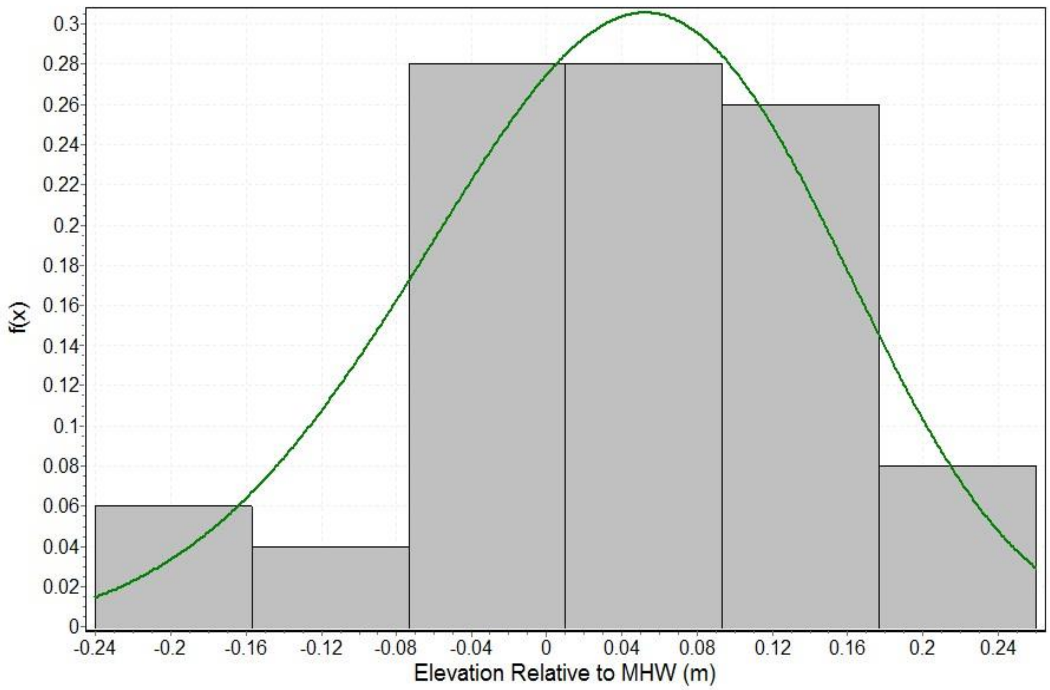


Figure 2.15 Histogram with a fitted 2-parameter Weibull probability density function, for the frequency of below-ground biomass for the elevations relative to MHW (m).

2.5.5 Effect of Tidal Range on Biomass

Four of the sub-estuaries (St. Jones River, Broadkill River, Murderkill River, and Blackbird Creek) show a general positive correlation of marsh elevation increasing with increasing tidal range (Figure 2.13). This relationship is somewhat surprising in that the elevation relative to MHW is not a constant for a given tidal range within an estuary or marsh (Figure 2.13). The growth range only increases above MHW when the tidal range exceeds 1.0 to 1.4 m (Figure 2.13). For tidal ranges less than 1.0 m, nearly all marsh elevations relative to MHW are near (within 0.03 m) or below MHW, for all sampled watersheds (Figure 2.13). This elevation range in the distribution of below-ground biomass with respect to MHW is important in assessing the sustainability of wetlands and/or planning a design elevation for restoration purposes. As such, wetlands at the MHW elevation could be viewed as being in relatively stable conditions. Through the addition of a thin layer of sediment, the overall biomass production could decrease, and open the wetlands up to loss of below-ground biomass through time and potential colonization of an opportunistic invasive species, such as *Phragmites australis (P. australis)*.

Wetlands with tidal ranges above 1.0 m generally show increasing optimal elevation ranges relative to height above MHW with maximum elevation values of 0.26 m for the St. Jones River, 0.10 m for the Blackbird Creek, 0.14 m for the Murderkill River, and 0.08 m for the Broadkill River (Figure 2.13). The Bombay Hook Complex sites show a dramatic increase in marsh elevations from -0.12 to 0.12 m over a very narrow tidal range of 1.49 to 1.60 m. In contrast, above the 1.6 m tidal range, the data for the complex show a strong negative correlation, with a precipitous decline in marsh elevation reaching -0.27 to -0.39 m relative to MHW at tidal ranges approaching 2 m (Figure 2.13).

While the upper limit can vary considerably, the lower growth limit of *S. alterniflora* demonstrates a strong relationship to the tidal range of a site. The lower growth range is tightly constrained by the depth above MLW (Figure 2.14). *S. alterniflora* does not occur below 0.53 m above MLW for tidal ranges above 0.50 m (Figure 2.14). Below a value of 0.50 m, the small tidal range forces *S. alterniflora* to occur at an elevation very close to MLW. The positive correlation shown in Figure 2.14 indicates that as tidal range increases, the *S. alterniflora* platform grows at greater heights above MLW. The amount of total below-ground biomass tends to have greater variability at higher MLW elevations (Figures 2.7 and 2.9). This may hint at other factors outside of elevation as having a greater effect on the below-ground biomass production of *S. alterniflora* as the marsh platform elevation increases above MLW and approaches or exceeds MHW.

2.5.6 Potential Effects of Sea-level Rise

As sea level continues to rise in the Delaware Estuary (current rate of 3.20 mm/yr $\pm$ 0.28 (NOAA, 2014)), increasing tidal water levels will be further amplified through the existing drainage network and will cause longer periods of inundation of the wetlands. Tidal marshes that are already showing signs of stressed and underperforming vegetation will suffer further catastrophic losses, such as the Bombay Hook Complex. *S. alterniflora* wetland losses will only increase as longer durations of inundation will result in decreased productivity, and a continued lowering of the marsh platform through increased water levels and less accretion. This will eventually lead to complete failure of the marsh platform. A catastrophic loss in vegetation will ultimately result in a lowered platform and exposure of fine-grained mineral sediments to wind and tidal current-induced erosion. This cycle will amplify wetland loss as open accommodation space (the space available for potential sediment accumulation resulting from platform lowering through erosion and rising water levels), will be filled with tidal waters. This will result in an increased rate of shallow subsidence, sediment erosion, and further alterations to the tidal prism through the volumetric increase in ebbing and flooding waters.

2.6 Conclusions

Results from this study show that *S. alterniflora* has an optimal below-ground biomass growth between –0.07 and 0.09 m (with high growth from 0.09 to 0.18 m) relative to the elevation of MHW within the sub-estuaries that were sampled. Below-ground biomass data plotted in relation to MHW elevation at each site, the distribution of points and frequency of biomass (within elevation bins) used to determine the tidal elevations that corresponded to the maximum and minimum *S. alterniflora* biomass values. Maximum biomass growth is determined to be around MHW elevation (-0.07 to 0.09 m), while the growth of below-ground biomass is greatly reduced at elevation below (< 0.07 m) and above (> 0.18 m) the MHW datum.

However, data from two of the sub-estuaries did not fit within the maximum and minimum potential growth limits. The Bombay Hook Complex appears to be influenced by altered hydrology within the sub-estuary. Two ditches (Raymond and Sluice Guts) were added within the complex artificially raising MHW near Leatherberry Flats, and not allowing the region to naturally drain. The altered hydrology increases the interior tidal range at a rate quicker than accretion can build the marsh platform. The higher water levels and lower elevation of the *S. alterniflora* relative to MHW stunt a plant's biomass growth causing further degradation of the platform. The Northern and Central biomass sample elevations has values ranges below the optimal range of -0.07 to 0.09 m, with values ranges of -0.06 to -0.27 m and -0.05 to -0.40 m (respectively).

The biomass data relative to MHW elevations from the Broadkill River sub-estuary are also outside of the derived maximum/minimum potential growth limits (Figure 2.9). The range in Broadkill River biomass elevations is between -0.11 and 0.08 m, relative to MHW, with 8 of the 11 sample sites having an elevation at or below MHW (i.e. 0.0 m). This sub-estuary has a history of intensive mosquito-ditching; however, more investigation is needed in order to verify the effect of altered hydrology on the duration and depth of flooding on this platform. Mosquito ditches could result in a faster inundation of the marsh, which could lead to a longer period of inundation. .

The data in this study can be used to assess a marsh's ability to combat changing conditions associated with sea-level rise, by determining whether or not the platform elevation is at the optimal -0.07 to 0.18 m relative to MHW, for maximum below-ground biomass production. This assessment method and the derived relationships to MHW (and

MLW) also greatly assist in developing a better means of determining if a tidal wetland is in need of restoration. Due to the correlation of marsh platform elevation with MHW tidal datum elevation, *S. alterniflora* will attempt to maintain its relationship with water levels. If in need of restoration, it could be accomplished through thin-layer application of material, or alterations to hydrology, to optimize below-ground biomass production, thus building a more sustainable marsh platform.

Chapter 3

THE EFFECTS OF LONG-TERM WATER LEVEL MANAGEMENT UPON ACCRETION AND WETLAND ELEVATIONS IN THE COASTAL IMPOUNDMENTS OF DELAWARE.

3.1 Introduction

Tidal marsh impoundments in Delaware occupy over 11% of the state's approximately 36,500 hectares (90,000 acres) of tidal marsh. While these impoundments are not naturally occurring features, due to their longevity they have become an important habitat for migrating shorebirds and waterfowl. As such, they are protected for wildlife use while other marsh areas have been developed or lost to sea-level rise. With the rapid onset of climate change, these impoundments are increasingly in danger of catastrophic failure.

Delaware's tidal marsh impoundments are located on federal, state, and private lands. They range in size from less than a hectare (ha) to several hundred hectares. The oldest impoundments date back to Swedish and Dutch settlements in the 1600s and were typically built for agricultural purposes or to control flooding (Weslager, 1987). More recent impoundments were developed during the late 1930's by the Civilian Conservation Corps (CCC) to control mosquito breeding. The negative impact of mosquito ditching became apparent by the 1940s as the natural vegetation in the marsh began to perish due to altered flooding and extreme soil salinities (Whitman and Cole, 1987).

To aid in correcting these actions, marsh management entered a new era (during the 1950s and early 1960s) creating coastal impoundments to control mosquito breeding and improve wetlands for waterfowl and other wildlife (Catts et al., 1963; Whitman and Cole, 1987). The immediate short-term success of these coastal impoundments prompted the construction of additional ones along the coast from Port Penn to Little Assawoman Bay (Whitman and Cole, 1987). Most of these were established in the central Delaware Bay region, currently consisting of the Bombay Hook and Prime Hook National Wildlife Refuges (NWRs) and five state wildlife areas. By the mid-1970s, impounded wetlands began to lose favor as a management option for wetland habitat due to the restriction of nutrient exchange with the bay and loss of high and low tidal marsh vegetation (Whitman and Cole, 1987).

The existing impoundments may be operated as freshwater (primarily the NWR impoundments) or brackish water (state wildlife areas) units. Surface exposure in individual impoundments fluctuates from exposed mudflats to waters up to 10 cm (several inches) deep, with changes in water level dictated by rainfall, evaporation, outflow or water control structures, and controlled tidal flooding. The differing levels are adjusted for habitat use, typically flooded in the fall (for waterbirds), and maintained at lower levels in the summer for optimum vegetative growth (both annual and perennial plants) (Meredith et al., 2004). A diversity of habitat types (i.e. open water, wetlands, mudflats) during the seasons is ideal for managing the needs of multiple species of concern. The impoundments are managed to meet these needs in a regional context, allowing for multiple habitat types along the coast (Meredith et al., 2004).

The fate of the impoundments is a major concern and is addressed in several planning studies such as: “Ecological Conditions and Management for Coastal Impoundments in Delaware” (Whitman and Cole, 1987), “The Northern Delaware Wetlands Rehabilitation Plan” (Hossler, 1994), “Comprehensive Conservation and Management Plan (CCMP) for Delaware’s Tidal Wetlands” (Meredith and Whitman, 1994), “Prime Hook National Wildlife Refuge Draft Comprehensive Conservation Plan (CCP) and Environmental Impact Statement” (United States Fisheries and Wildlife Service (USFWS), 2012), and “Preparing for Tomorrow’s High Tide: Recommendations for Adapting to Sea-Level Rise in Delaware” (DNREC, 2013). The 1994 tidal wetlands CCMP has as one of its action steps “develop a management policy that determines under what conditions or at what locations, and for what purposes, should damaged impoundment levees be routinely repaired or modified; and under what circumstances or at what locations should no actions be taken to repair a damaged levee.” (Meredith and Whitman, 1994) A major aspect of impoundment management is the long-term sustainability of the created habitats. Including what changes should be implemented to restore impoundment wetlands to natural tidal wetlands, once it is determined that it is unsustainable to upkeep the infrastructure in response to sea-level rise. It is important to understand how impoundment management affects overall accretion within the wetlands, and how water-level management can be modified to enhance accretion.

The recent freshwater wetland collapse from the levee breaches at the Prime Hook NWR converted ~809 ha (2,000 acres) of freshwater and brackish wetlands to open water (USFWS, 2012). If the impounded wetland platform is not managed at a consistent elevation with the adjacent natural wetland platforms, then catastrophic losses of wetlands can occur. It is the opinion of many, which is shared by the author, that the lack of statewide comprehensive freshwater wetlands protection regulations result in a loss of inland and near-coast wetland birds and waterfowl habitat (CSO, 2007; DNREC, 2013). The largest and most continuous available feeding and roosting habitat is now the managed coastal impoundments on the Delaware River and Bay coast (USFWS, 2012; DNREC, 2013).

With sea-level rise, adaptive and comprehensive management of impounded wetlands becomes imperative to maintain sustainable waterfowl and migratory bird habitat (USFWS, 2012; DNREC, 2013). Water-level management to create specific seasonal habitats, however, could have consequences on the platform elevation of the impoundment wetlands, and therefore, the resilience and sustainability of the intertidal habitat. Sustainability is also threatened by potential failures of levee and water control structures which would inhibit normal water-level management.

This study evaluates the effects of long-term impoundment management on the rate of accretion and subsequent decreases in wetland platform elevation compared to neighboring un-impounded tidal wetlands. Nine impoundments (three in the City of New Castle, four in Central Delaware Bay, and two in lower Delaware Bay) and four reference tidal wetlands were sampled to evaluate their rates of accretion over the past 60 years, and their wetland platform elevations relative to local tidal datums. The duration of impoundment and water level management goals varies between impoundments. This allows for a unique opportunity to evaluate the short-term and long-term implications of water-level control on the impoundments.

3.1.1 Accretion and Wetland Build-up Background

Vertical accretion within tidal wetlands occurs when mineral and organic matter accumulation building the marsh platform happens at a rate faster than the rate of submergence of the platform (due to sea-level rise, subsidence, sediment compaction, and organic matter decay; Redfield, 1972; Warren and Niering, 1993; Morris et al., 2002; Neubauer et al., 2002; Rooth et al., 2003; Nyman et al., 2006). What is not as apparent is the effect of water level manipulations on the rate of vertical accretion in impounded marshes, and what role limited tidal flushing has upon input of mineral sediments and rates of organic decay (both aerobic and anaerobic). Several studies conclude that many coastal marshes accrete primarily through vegetative growth rather than by accumulation of inorganic or mineral sediments (McCaffrey and Thomson, 1980; Hatton et al., 1983; Bricker-Urso et al., 1989; Nyman et al., 1993; Anisfeld et al., 1999; Turner et al., 2000; Chmura and Hung, 2004; Nyman et al., 2006).

A major point of consideration in developing this study is whether impounded wetlands vertically accrete in a fashion that more closely aligns with tidal freshwater wetlands, or if they are analogous to adjacent reference salt marshes. The vertical accretion of the marsh platform is a predictor for wetland sustainability in the face of sea-level rise, but also highlights the effect that water manipulations have had on the marsh platform elevation in comparison to naturally accreting tidal wetlands. The marshes that are not accreting rapidly enough to keep the marsh platform in equilibrium with local tidal levels continue to decrease in elevation with subsequent increased flooding stressing the existing vegetation (Nyman et al., 1993; Nyman et al., 2006).

There is an ideal flooding frequency on marsh vegetation at which the rate of productivity (especially below-ground biomass) is optimized; where flooding can flush out salts, deliver nutrients, and increase biomass production (Morris et al., 2002; Kirwan and Guntenspergen, 2012; Kirwan et al., 2012). At increasing rates of sea-level rise or using particular water level management schemes, anaerobic conditions can negatively impact the productivity of wetland vegetation through the accumulation of sulfides (Morris et al., 2002; Kirwan et al., 2012). It is the seasonal manipulation of water levels in impoundments which makes it hard to predict how these managed wetlands vertically accrete as compared to natural tidal wetlands, and whether these manipulated water levels may affect the rate of decomposition.

In freshwater wetlands, flooding frequency and duration cause an increase in the rates of decomposition in the soil (Ewel and Odum, 1978; Mendessohn et al., 1999), while in salt marshes it is more variable, and not likely related to flood duration and redox potential (Valiela et al., 1982; Hackney, 1987; Blum, 1993; Blum and Christian, 2004). In lower salinity wetlands (such as impounded wetlands), the ambient and pore water salinity may dictate the rate of organic matter decay (Sutton-Grier et al., 2011; Weston et al., 2011) which can then drive elevation changes in fresh and brackish marshes through these root zone processes and interactions (Craft, 2007; Kirwan et al., 2012).

Intermittent flooding typically enhances the decomposition of organic matter in ecosystems with mostly aerobic soil conditions such as riparian forests, prairie wetlands, and freshwater marshes (Brinson, 1977; Ewel and Odum, 1978; Day, 1979; Maltby, 1988; Neckles and Neill, 1994; Mendelssohn et al., 1999; Kirwan et al., 2012). Kirwan et al. (2012) theorizes that flooding may accelerate the decomposition of organic matterby providing moisture and nutrients to microbial and fungal communities (Neckles and Neill, 1994; Bragazza et al., 2012). Decomposition generally occurs more rapidly in aerobic soils than anaerobic soils, with .this mainly occurring near the soil surface with decay rates decreasing with increasing soil depth and decreased oxygen availability(e.g., Neckles and Neill, 1994; Mendelssohn et al., 1999).

3.2 Location

The study areas are split into three provinces or regions based upon their physiographic distribution along the Delaware Coast. The Delaware River study area, along the New Castle County coastline, is the most northern location and includes the Lukens and Rivers Edge reference marshes, and the Buttonwood, Broad Dike, and Gambacorta Impoundments (Figures 3.1 and 3.2). The Central Delaware Bay study area, consisting of the Pickering Beach reference marsh, and the Port Mahon, Little Creek, Logan Lane North and South Impoundments, is positioned along the coast in Central Kent County (Figures 3.1 and 3.3). The Lower Delaware Bay study area, consisting of Prime Hook Unit I and IV reference marshes, and Prime Hook Unit II and III

Impoundments, is located in the Prime Hook NWR in coastal Sussex County (Figures 3.1 and 3.4).

The middle to lower Delaware Bay tidal wetlands (Prime Hook Unit I and IV and Pickering Beach) and impoundments (Port Mahon, Little Creek, Logan Lane North and South, and Prime Hook Unit II and Unit III) are seasonally polyhaline. The northern tidal wetlands (Lukens and Rivers Edge) and impoundments (Buttonwood, Broad Dike, and Gambacorta) can be mesohaline in the summer and fall with spring salinity values of a more oligohaline range (Figure 3.1).

3.3 Impoundment Background

3.3.1 Delaware River Impoundments

In Northern Delaware, the degradation of the wetlands began as early as the mid-1600s. Dutch and Swedish settlers extensively diked and drained tidal freshwater marshes along the Christina and Delaware Rivers to accommodate agriculture and development of adjacent upland areas (Weslager, 1987). This extensive system of dikes and tide gates has, for the most part, been maintained un-changed since its initial construction. This practice essentially prohibited several thousand hectares of tidal wetlands from receiving normal tidal exchange with the Delaware Estuary for up to 340 years (Carter, 1991). The dike system promoted the filling and additional draining of many wetlands for industrial, maritime, and residential development during the Industrial Revolution of the late 1800s

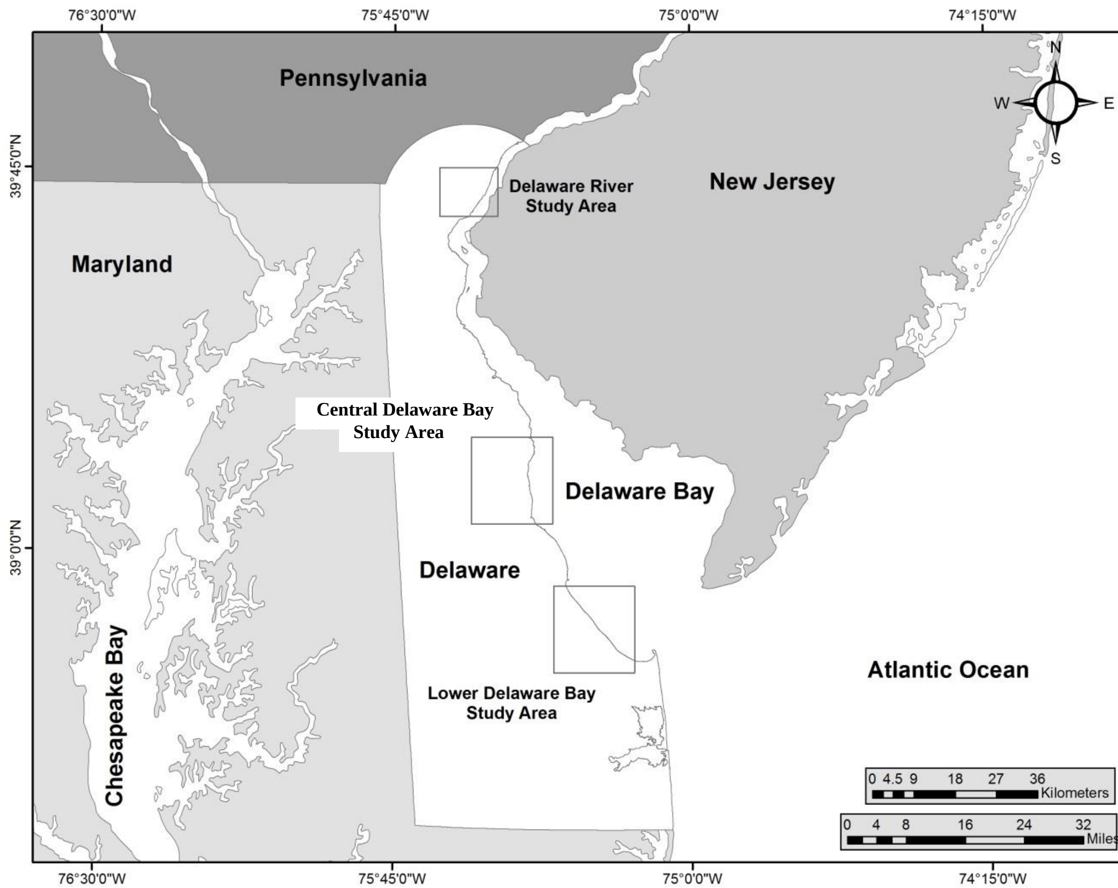


Figure 3.1 Map of the three impoundment and reference marsh study areas along the Delaware Coast.

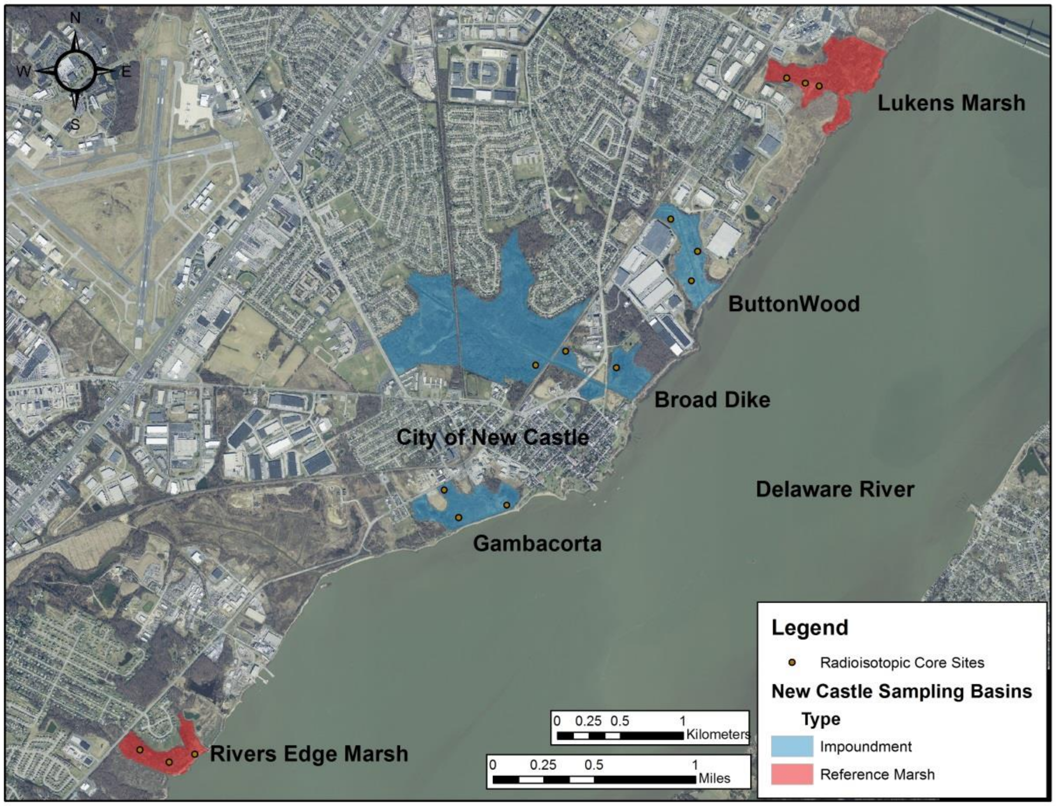


Figure 3.2 Locations of three impoundments and two reference marshes, and their radiometric cores, within the Delaware River study area.

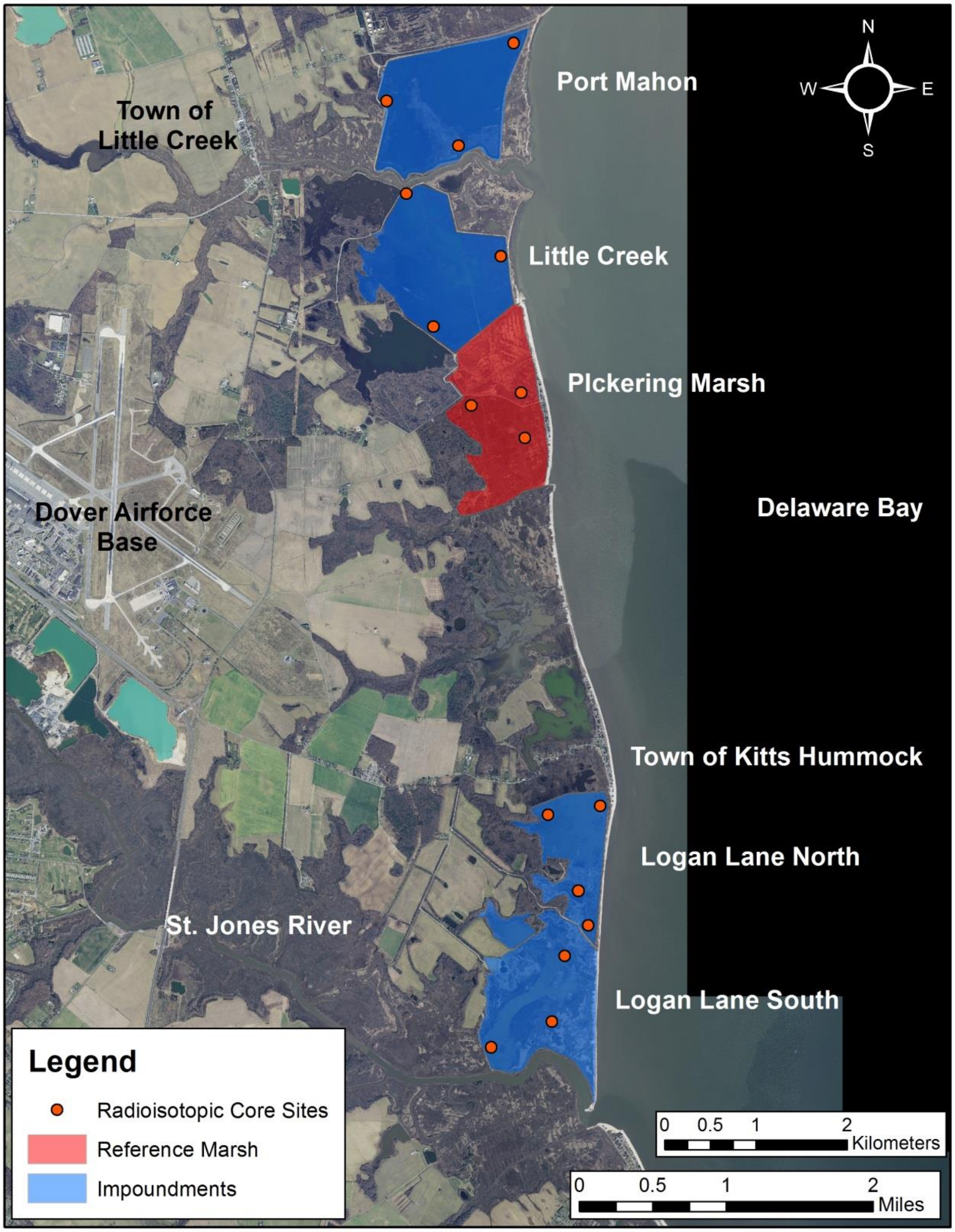


Figure 3.3 Locations of four impoundments and one reference marsh, and their radiometric cores, within the Central Delaware Bay study area.

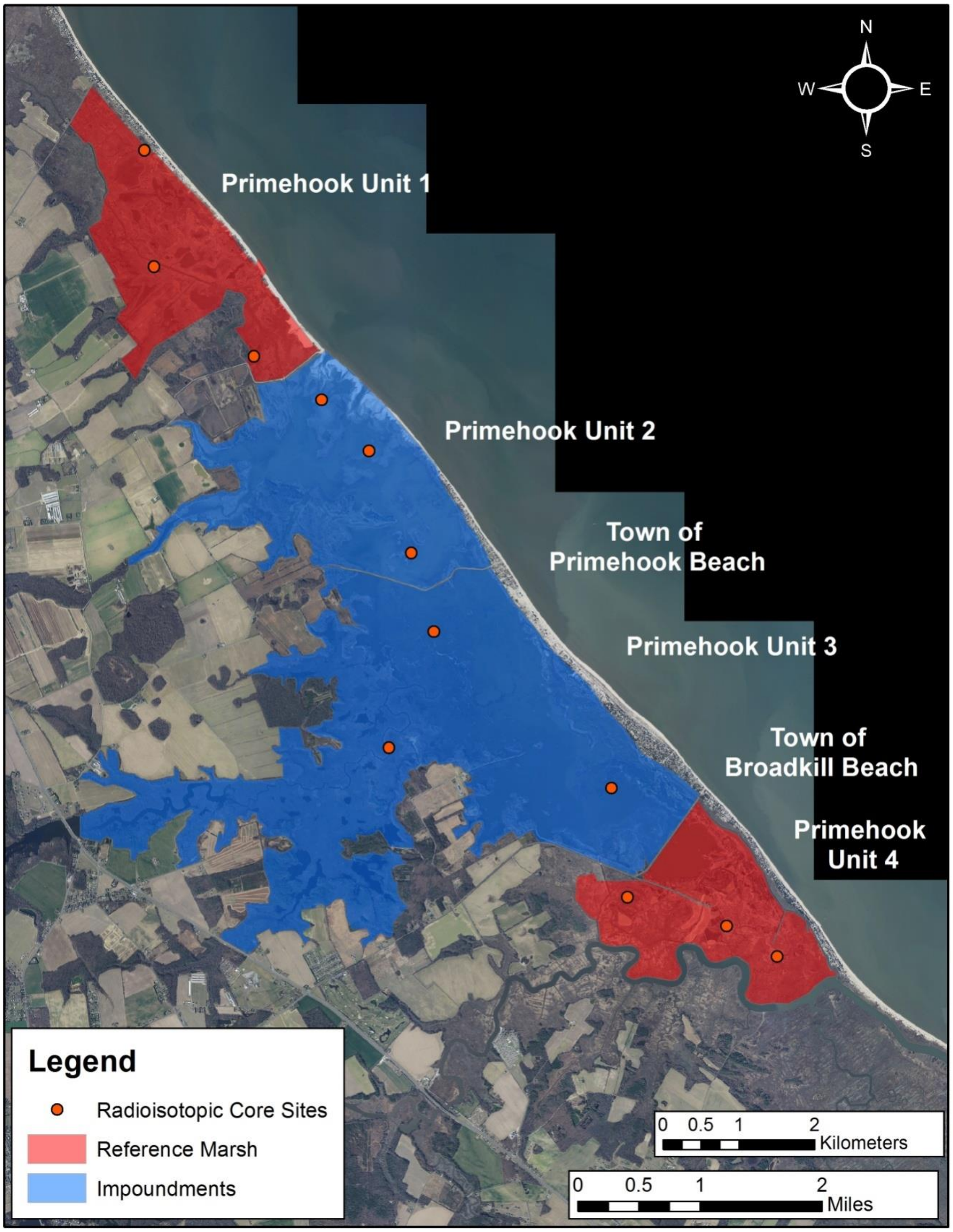


Figure 3.4 Locations of two impoundments and two reference marshes, and their radiometric cores, within the Lower Delaware Bay study area.

and early 1900s (Catts and Mancl, 2013). The Buttonwood, Broad, and Gambacorta Dikes are historic landmarks, which are considered to be some of the oldest dikes in the United States. Their origins are documented as far back as the late 1600s to early 1700s (Catts and Mancl, 2013; Mancl et al., 2013).

Buttonwood Dike was constructed in 1786 by a newly formed marsh company that was responsible for the construction and care of dikes to enable reclamation of the marsh for agricultural purposes and provide foot traffic access (Catts and Mancl, 2013; Mancl et al., 2013). The associated marsh (Swanwyck Marsh) behind the dike, currently 17 ha (42 acres), has been cut off from regular flooding for over 228 years. The Buttonwood Marsh has a drainage area of ~348 ha (860 acres), which is dominated by residential, commercial, and industrial land-use.

Broad Dike was constructed in 1675, when the authorities in the village of New Amstel (today's New Castle), ordered the construction of two dikes to cross marshland north of the town: one in the footprint of the current dike for foot-traffic and a second that was for vehicular traffic (historically called the Cart Dike, which is currently the crossing of Route 13) (Mancl et al., 2013). In 1681, a Dutch merchant ditched and drained the Broad Dike Marsh (Wacker and Clemens, 1995; Mancl et al., 2013). The Broad Dike Marsh is currently an 85 ha (210 acre) freshwater tidal wetland, which has been cut off from normal tidal inundation for over 339 years. The Broad Dike Marsh has a drainage area of 733 ha (1,811 acres), primarily consisting of residential and urban land-uses.

The Gambacorta Dike was first documented in 1706 when the property was confiscated from Peter Alrichs, who had failed to drain the marsh, and was granted to George Deakyne, "who drained the marsh, and built a dike along the river…." (Eckman, 1947). The current 17 ha (41 acre) tidal freshwater wetland has been cut off from normal or regular tidal inundation for 308 years. The marsh's watershed encompasses 104 ha (8,258 acres) of urban and commercial development.

These marshes were once a lush mosaic of rushes, sedges, cattails, and smartweeds (Hossler, 1994). They contained a high diversity of water birds and other wildlife (Hossler, 1994). By the 1700s, they were all diked and drained to accommodate agriculture and settlement of adjacent upland areas (Mancl et al., 2013). These practices continue to the present, resulting in lower biodiversity marshes, which until the mid-to late 1990s were dominated by the nuisance plant *P. australis* (Hossler, 1994; Mancl et al., 2013).

The water management plan for the Buttonwood, Broad Dike, and Gambacorta Marshes has been the same for four decades with a single purpose to prevent flooding of the properties behind the levees, due to high tides, storm surges, or from upland storm runoff accumulating in the marsh (Hossler, 1994; Meredith et al., 2004). To accomplish this flood mitigation, several iterations of water control structures (i.e. flap gates) were installed to allow one-way flow out of the marsh (Hossler, 1994; Meredith et al., 2004). This allows storm run-off to flow out while preventing Delaware River water from inundating the marsh during higher water level events.   However, because of the increase in upland runoff due to the increased impervious surface and decreased infiltration, and a significantly lower elevation of the marsh surface relative to the Delaware River, the existing structures have been found to be inadequate to handle storm runoff from severe rain events (Hossler, 1994; Meredith et al., 2004). In the mid-1990s, new water control structures were installed in the New Castle impoundments to allow daily tidal exchange (Meredith et al., 2004).

The main goals of the new water control structures and updated water management plan are to: improve the wetland habitat and condition, improve water quality of both the marsh and river through daily tidal exchange, reduce the transportation of potential upland pollutants conveyed into the wetland by stormwater runoff, provide nutrient and organism exchange between the water bodies, and increase the volume of water exposed to wetland filtering benefits and nutrient uptake (Meredith et al., 2004). The habitat quality of the impounded wetlands suffered significantly due to reduced tidal flow, and by the mid-1980s were 70% - 90% dominated by large stands of *P. australis* (Meredith and Whitman, 1994). The hope is that increased daily tidal flushing (in addition to several implemented management actions) will increase the percentage and diversity of emergent vegetation and open water habitats (Meredith et al., 2004).

The anthropogenic impacts on the Gambacorta Marsh are more widespread than those experienced by the Buttonwood and Broad Dike Marshes, as this wetland was drained and filled with hazardous industrial waste from Deemer Steel, the Abex Corp., and Wilmington Fiber Co in the early 1900s (Hossler, 1994). In the mid-1980s, the waste was excavated and the former disposal site within the wetland was capped with clean material (Catts and Mancl, 2013). The excavated and capped locations are at the western edge of the impoundment, adjacent to State Route 9.

3.3.2 Central Delaware Bay Impoundments

Currently, the Delaware Division of Fish and Wildlife maintains and manages about 970 ha (2,400 acres) of coastal wetlands in fourteen Kent and Sussex County impoundments situated from the Port Mahon/Little Creek area to Little Assawoman Bay. This study focuses on four of these impoundments: Port Mahon, Little Creek, Logan Lane North, and Logan Lane South.

Covering 275 ha (680 acres), Port Mahon is the largest impoundment in the Central Delaware Bay study area (Figure 3.3). The impoundment was created in 1967 by encircling a tidal marsh with low earthen dikes (Whitman and Cole, 1987). The initial wetland, prior to diking, was dominated by *S. alterniflora*, and *Spartina patens* (*S. pat*ens) (Whitman, 1995).

The Little Creek Impoundment consisted of four separate units, constructed in 1959, but due to the erosion of an earthen dike, only three exist today (Whitman and Cole, 1987). The largest impoundment within this complex, 243 ha (600 acres) in size, is used in this study (Figure 3.3). Before the area was impounded, it was dominated by dense stands of *S. alterniflora*, *S. patens* and *Distichlis spicata (D. spicata)* (Whitman and Cole, 1987; Meredith and Whitman, 1994). Water levels were managed by pumping tidal water into the impoundment, but by the 1980s this management scheme resulted in it being devoid of all emergent vegetation (Whitman and Cole, 1987). The only vegetation was found at the fringing wetlands, and was dominated by *P. australis, Typha latifolia*, and *Iva fructescens* (Whitman and Cole, 1987).

The Logan Lane Impoundments are divided into northern (89 ha (221 acres)) and southern (173 ha (428 acres)) units (Figure 3.3). They were created in the early 1960s by diking around a back-barrier tidal marsh (Whitman and Cole 1987). Prior to ditching, the area was dominated by *S. patens, D. spicata*, and *S. alterniflora* (Whitman and Cole 1987). As with the Port Mahon and Little Creek Impoundments, habitat conditions deteriorated in both the northern and southern Logan Lane units by the mid-1980s (Whitman and Cole 1987; Meredith and Whitman, 1994). In the late 1980s to mid-1990s, water control structures were updated to allow for more effective water level management through the introduction of enhanced tidal flushing (Meredith et al., 2004). This updated water level management has resulted in a more robust habitat (Meredith et al., 2004).

The Logan Lane North Impoundment has limited tidal exchange, through one water control structure into Logan Lane South. Logan Lane North is dominated by open water and mudflat habitat (~54% combined of total area), with vegetated wetlands consisting of a mix of annual vegetation, tall-form *S. alterniflora*, and *S. patens* (Coxe, 2012). The Logan Lane South Impoundment has a large tidal exchange, through a water control structure to the St. Jones River and an emergency spillway (Whitman and Cole, 1987). It is dominated by mixed mudflat, open water habitat (~ 35% of total area), and wetlands consisting of tall-form *S. alterniflora, S. patens*, and *P. australis* (Coxe, 2012).

3.3.3 Lower Delaware Bay Impoundments

The impoundments in the Lower Delaware Bay study area contain about 1,700 ha (4,200 acres) of marshes at the Prime Hook NWR, with federal management interests focusing primarily upon creating and maintaining quality habitats for migratory waterfowl (USFWS, 2012). In the 1980s, the USFWS impounded 81 ha (200 acres) of salt marsh in Prime Hook Unit IV (1981), and converted approximately 607 ha (1,500 acres) of salt marsh in Prime Hook Unit II (1986) and 1,012 ha (2,500 acres) of salt marsh and transition marsh vegetation in Prime Hook Unit III (1984) into brackish and freshwater wetland plant communities (USFWS, 2012). The impounding was done by building a large berm and dune along the eastern side of the refuge (along the shore of the Delaware Bay), installing three concrete water control structures, and using the existing road infrastructure to barrier island communities as breaks between management units (USFWS, 2012). The USFWS (2012) believes these impoundments are a cost-effective opportunity to provide habitat to important migratory waterbirds, and to control *P. australis,* which had been dominating the irregularly inundated back barrier marshes in Prime Hook Units II and III (Figure 3.4).

During the freshwater management period from 1984 to 2006, vegetation of Prime Hook Units II and III was dominated by freshwater annuals, perennials, and *P. australis* (USFWS, 2012). In part, these larger low-salinity impounded wetlands help to compensate for losses of low-salinity wetland habitats (USFWS, 2012). The main cause for these losses is human intervention to keep coastal inlets open through dredging or construction of jetties, or by other types of impacts detrimental to these habitats (USFWS, 2012).

3.4 Water level Management

3.4.1 Delaware River Impoundment Management

As the past main goal of the Buttonwood, Broad Dike, and Gambacorta Impoundments was primarily flood protection, the water level management prior to the mid-1990s was for no tidal exchange with the Delaware River (Hossler, 1994; Meredith et al., 2004). The lower water levels in the wetlands reduced mosquito breeding (by only flooding the pools and ditches to a level to allow access for fish, but not flood the marsh surface; Meredith et al., 1985). During the mid- to late-1990s, tidal gates were installed on these impoundments to allow for two-way tidal exchange to increase flushing and water exchange to provide better habitat and biota access (Meredith et al., 2004).

Since the mid- to late-1990s, the impoundments have been managed to maximize fish exchange and plant regrowth (Meredith et al., 2004). Under this water level management scheme, salinities of the wetlands vary between mesohaline in the summer and fall to oligohaline in the winter and spring (Hossler, 1994). The effectiveness of the water level management, to provide two-way flow, was adequate until the late 2000's (Catts and Mancl, 2013). Since that time, the cumulative effect of rising tidal levels in the Delaware River has made it difficult to remove water from the impounded wetlands during storm events or control the interior water levels under the normal tidal range (Catts and Mancl, 2013). In response to the rising water levels in the Delaware River and the increasing impervious area within the drainage basins, recent water level planning and water control structure upgrades (early-2010's) have been implemented to address these issues (Catts and Mancl, 2013).

3.4.2 Central Delaware Bay Impoundment Management

Past water level management of the four Central Delaware Bay impoundments entirely cut-off water exchange with the adjacent Delaware Bay, except for once yearly flooding (Whitmore and Cole, 1987). Over the years, salinity in the impoundments increased; by the early 1980s, salinities were observed to be as high as 100 ppt. causing the vegetation to die back (Clark, 1995). Use by waterfowl had also drastically declined (Whitman and Cole, 1987; Stocks and Grassle, 2003).Water level management strategy changed by the mid-1980s with an eye toward increasing marsh productivity (Meredith and Whitman, 1994). The main focus of the improved management is to allow a fall flooding (beginning in October), and maintain a higher water level (greater than 61 cm (24 inches) on the marsh surface) through the winter to attract waterfowl (Whitmore and Cole, 1987). Tidal exchange has been eliminated, or is very limited, during the late fall and winter to allow maintenance of high water levels (Whitmore and Cole, 1987). Periodic drawdowns and re-floods are conducted from January through March to prevent ice formation (Whitmore and Cole, 1987). In March, a drawdown to a 50% pool (50% surface inundation) is conducted to expose the marsh surface to allow for shorebird feeding and vegetation to germinate (Whitmore and Cole, 1987). During times when sediments are exposed, frequent tidal exchange is prescribed to prevent soil salinities from climbing (Whitmore and Cole, 1987). From April to August, water levels are maintained at 50% pool (Whitmore and Cole, 1987). In September and October, big drawdowns and re-floods occur to flush the impoundments prior to the traditional fall re-flood (Whitmore and Cole, 1987).

The current water level management plan results in mean salinity levels of 14 to 18 ppt (Meredith et al., 2004). Control of water levels within the impoundments has become more difficult as the result of sea-level rise and subsidence of the water control structures (Meredith et al., 2004). This has resulted in lowering of the openings of the structures within the tidal range of the St. Jones River (exchange with Logan Lane North and South Impoundments), Little River (exchange with Port Mahon and Little Creek Impoundments), and Pickering Beach Marsh (exchange with Little Creek Impoundment) (Meredith et al., 2004). Loss of control in the water levels is particularly evident in the Logan Lane South Impoundment, where an emergency spillway has subsided (relative to sea level and in conjunction with rising water levels), and now only allows a tidal inflow through the upper half of a normal neap tidal cycle (Meredith et al., 2004).

3.4.3 Lower Delaware Bay Impoundment Management

Units II and III of Prime Hook NWR were managed as freshwater impoundments from 1984 to 2006 through a typical moist soil management (USFWS, 2012). This resulted in spring (March through mid-April) drawdown to 50% pool level to expose the mudflats and marsh surface to allow for annual and perennial plant germination (USFWS, 2012). The water level was further lowered to 0% pool level from mid-April through July to expose 100% of the surface to encourage annual plant germination (USFWS, 2012). The water levels would then be raised for the fall flooding in August, reaching 100% pool level by the end of October (USFWS, 2012). The impoundments would stay at 100% pool level until March to provide habitat for overwintering waterbirds and waterfowl (USFWS, 2012).

From the 1990s to 2005 after the impoundment infrastructure was established, salinities ranged from 0 to 5 ppt (USFWS, 2012). Since 2006, problems with the design of the stop-log flap gates and rising water levels have resulted in a loss of much of the capabilities to conduct the same water level management as prior years (USFWS, 2012). Salinity values in the impoundment ranged from 2 to 25 ppt from 2006 to 2009 until multiple breaches in the dunes along the Delaware Bay converted Prime Hook Unit II and the eastern portion of Prime Hook Unit III into saline and brackish open tidal water bodies (USFWS, 2012). After several breaches between 2009 and 2011, rapid and expansive losses of wetland vegetation resulted in conversion of Unit II and the eastern portion of Unit III into open water embayments (USFWS, 2012).

3.5 Methods

^{137}Cs (cesium) radiometric cores and elevation data were collected from nine impoundments (Buttonwood, Broad Dike, Gambacorta, Port Mahon, Little Creek, Logan Lane North and South, and Prime Hook Unit II and Unit III), and four un-impounded tidal marshes (Lukens, Rivers Edge, Pickering Beach, and Prime Hook Unit I) within the Delaware River and Bay Estuary (Figures 3.1-3.4). All of the un-impounded tidal marshes and exterior setting of the impoundments are microtidal (< 2 m), with ranges at their mouths of 1.59 to 1.79 m.

The radiometric cores and elevation data for each study area were collected over the course of three sampling seasons with core samples collected at: Prime Hook Unit I, II, and III in 2009, 2010, and 2011; and Buttonwood, Broad Dike, Gambacorta, Lukens Marsh, and Rivers Edge Marsh in 2010. In addition, samples were collected from Port Mahon, Little Creek, Logan Lane North and South, and Pickering Beach in 2011 (Figures 3.2-3.4; Appendix B). RTK elevation surveys were conducted over the course of three field seasons with elevation data collected at: Prime Hook Unit I, II, and III in 2009; Buttonwood, Broad Dike, Gambacorta, Lukens Marsh, and Rivers Edge Marsh in 2010; Port Mahon, Little Creek, Logan Lane North and South, and Pickering Beach in 2011 (Appendix B). Three radiometric cores were collected from each impoundment and reference tidal wetland. These were distributed throughout each sampling area to provide adequate coverage and variation in potential accretion. Each wetland also had elevation transect surveys conducted within them (Figures 3.2-3.4; Appendix B).

3.5.1 Core Collection

Three soil cores were collected at each of the nine impounded wetland units and at each of the four reference wetlands, for a total of thirty-nine cores. The core barrel consisted of a 10.2 cm (4 inch) PVC pipe cut to 1.5 m lengths. Cores were collected by driving the PVC barrel into the marsh using a mallet. An internal plunger was suspended in the core barrel (parallel to the marsh surface) during the process. This was done to reduce the amount of compaction or rodding that occurred. If a collected core had a rate of compaction or rodding that exceeded 8% of the total length, then the sample was discarded and the core collection was repeated. Cores were pulled out of the marsh subsurface using a winch suspended off of a tripod. The retrieved cores were then capped, and stored vertically in a dark, climate-controlled storeroom until they could be sub-sampled and analyzed.

After cores were collected, the outer PVC barrel was removed with a circular saw. This method of sampling was preferred over vertical extrusion (through a plunger pushing the core out), as the organic-rich sediments would become highly compacted during pushing and could skew the data analysis. Soil cores were vertically sectioned into 2 cm thick slices. They were then placed in sample bags and frozen until they could be analyzed. The 2 cm sediment samples were weighed, dried in a laboratory oven, and re-weighed to calculate the water content of each 2 cm interval. The samples were pulverized and sent for loss-on-ignition and radionuclide geochronology analysis.

3.5.2 ^{210}Pb and ^{137}Cs Radioisotopic Dating

For radioisotope analysis, 25 to 50 g of dry powdered sediment was placed in 70 mm plastic jars and placed in a Canberra GL2020R gamma detector for 24 to 48 hours. The total ^{210}Pb (lead) and ^{137}Cs concentrations were computed spectroscopically from the 46.5 keV and 661.7 keV photopeaks, respectively, whereas excess ^{210}Pb was determined by subtracting the supported activity (determined from the ^{214}Bi (bismuth) photopeak) from total activity. Detector efficiencies were determined using National Institute of

Standards and Technology standard reference material 4357 (Inn et al., 2001; Sommerfield, 2005 and 2012).

Measurements of ^{137}Cs, with a half-life of 30 years, and ^{210}Pb, with a half-life of 22.3 years, were calculated on sectioned cores following methods described in previous studies (Cutshall et al., 1983; Nittrouer et al., 1984; Sommerfield, 2005). Profiles of excess ^{210}Pb activity were used to derive linear sediment accumulation and mass accumulation rates averaged over the past ~100 years, whereas ^{137}Cs profiles provided the rates subsequent to 1954. The ^{137}Cs data were used further to validate the ^{210}Pb-based rates. A natural radioisotope of the ^{238}U (uranium) decay series, ^{210}Pb is produced via ^{222}Rn (radon) decay in the atmosphere and is deposited on the continents and surface waters through wet and dry deposition (Sommerfield, 2005 and 2012). Because ^{238}U is enriched in sea water relative to freshwater, the standing amount of ^{210}Pb in estuarine waters is typically larger than in rivers, but less than that of the open coastal ocean (Sommerfield, 2005 and 2012). Background levels of ^{210}Pb are produced in the sediment column via decay of ^{222}Rn, a source known as “supported” activity (Sommerfield, 2005 and 2012). In the estuarine water column, dissolved ^{210}Pb is scavenged by fine-grained particles and delivered to the seabed by sediment deposition (Sommerfield, 2005 and 2012). If the deposition rate is high relative to the mean life of ^{210}Pb, activity is concentrated in the upper sediment column above that supported by *in situ* production—this “excess” activity is what allows for estimation of sediment accumulation rates (Sommerfield, 2005 and 2012). Excess ^{210}Pb activity eventually decays to background levels at a rate constrained by the half-life of 22.3 years

(Sommerfield, 2005 and 2012). At steady state, the excess activity-depth profile reflects a balance between ^{210}Pb burial (gain) and radioactive decay (loss) (Sommerfield, 2005 and 2012).

^{137}Cs, a product of nuclear fission, was first introduced to the environment by means of nuclear weapons testing and reactor releases around 1954 (Sommerfield, 2005 and 2012). Atmospheric fallout of ^{137}Cs peaked in 1963−64, dropping thereafter to insignificant levels by about 1980 (Sommerfield, 2005 and 2012). Sub-areal marsh surfaces sequester ^{137}Cs activity (and ^{210}Pb) directly through wet and dry deposition, whereas tidal waters provide another source of particulate and dissolved-phase ^{137}Cs (and ^{210}Pb) (Sommerfield, 2005 and 2012). It is important to note that, because the atmospheric flux of ^{137}Cs has been negligible for the past several decades, ^{137}Cs present in the post-1980 sediment column represents activity redistributed from upland sources or surrounding estuarine deposits (Sommerfield, 2005 and 2012). In this manner, ^{137}Cs distributions in estuarine sediments reflect the regional atmospheric source function as well as more localized sedimentary processes (Sommerfield, 2005 and 2012).

The 1954 surface was the only identified radioisotopic surface that was present in all but one of the forty-two collected cores (PK2 did not have an identified 1954 or 1963 surface and was not used in the analysis, Figure 3.3; Sommerfield, 2012). Two additional cores were excluded from analysis due to the following concerns: first, excavation of hazardous waste and filling of those areas in the Gambacorta Impoundment (Core NCGB3; Figure 3.2; Boyd, 2012); second, the collapse of the wetland platform, initiated prior to PM9 core collection, resulting from the 2009 breaching and salt water intrusion in the Prime Hook NWR (Core PM9; Figure 3.4; Sommerfield, 2012). Mean rates of accretion, bulk mass accumulation, mineral mass accumulation, and organic mass accumulation were calculated based upon the depth of occurrence of the 1954 surface obtained from the ^{137}Cs data.

3.5.3 Elevation Surveys

Each impoundment and reference tidal wetland was surveyed using Real-Time Kinematic (RTK) Global Navigation Satellite System (GNSS) to collect the elevation of each wetland basin. All field elevation and position surveying were conducted using a Trimble 5700 RTK- receiver system. All surveyed points were referenced to the North American Datum of 1983 (NAD 83; which uses the Geodetic Reference System of 1980 (GRS 80) ellipsoid as the reference surface for three-dimensional positions). Vertical positioning was referenced to the North American Vertical Datum of 1988 (NAVD 88).

Transects of elevation were collected over the entirety of each wetland site. The transects were ~150 m apart, with points along each transect collected every 15 m. The elevation data were downloaded and pre-processed using Trimble Business Center, and then converted to ArcMAP shapefiles for further analysis. The transect elevation data were used to calculate the mean platform elevation for each reference (natural) and impounded marsh (Table 3.1).

3.5.4 Tidal Datum Calculations

Tidal datums for the reference and impoundment wetland sites were determined from several data sources. Three NOAA tide gauges and one USGS flow gauge were located in proximity to several of the study sites. Datums for all other sites were calculated using NOAA's VDatum (http://vdatum.noaa.gov/) for tidal elevations in the Delaware Bay. A mean value was determined for each reference and impoundment marsh based upon the available data (Table 3.1). A standardized wetland elevation was calculated for each reference and impoundment marsh by subtracting the site's MHW elevation from the mean platform elevation (calculated from the transect elevation surveys) (Table 3.1). This allowed the reference and impounded marshes to be compared across different tidal ranges and absolute tidal datum elevations.

3.5.5 Nonmetric MDS and PCA Statistical Analyses

As a first-order comparison between data collected from the three different study areas, and to compare data collected between reference marshes and impoundments within each study area, nonmetric multidimensional scaling (MDS) and principle component analysis (PCA) statistical analyses were performed. Nonmetric MDS is a means of visualizing the similarity of individual parameters in a dataset. MDS normalizes the data through the use of an algorithm aimed to place each object or parameter in a dimensionless distance matrix, so the values can then be compared within an ordination coordinate frame (Wickelmaier, 2003; Holland, 2008a). MDS is not an eigenvalue-

Table 3.1 Mean range of tides, mean high water (MHW) elevation, mean wetland platform elevation (with standard deviation), and standardized wetland elevation for the fourteen sampled mashes. Reference marshes are shown by shading. Standardized elevation = mean platform elevation – MHW elevation. All elevation data were recorded in meters, North Atlantic Vertical Datum 1988 (NAVD 88). Mean platform elevation was not determined at Prime Hook Unit 4.

	Mean Range of Tides	MHW Elevation	Mean Platform Elevation	Standardized Elevation
Lukens Marsh	1.79 m	0.83 m	0.94 m (0.08)	0.11 m
Buttonwood	1.79 m	0.82 m	-0.07 m (0.18)	-0.89 m
Broad Dike	1.78 m	0.81 m	-0.42 m (0.10)	-1.23 m
Gambacorta	1.77 m	0.80 m	0.25 m (0.10)	-0.55 m
Rivers Edge Marsh	1.78 m	0.80 m	0.92 m (0.12)	0.12 m
Port Mahon	1.82 m	0.74 m	0.47 m (0.18)	-0.27 m
Little Creek	1.82 m	0.74 m	0.57 m (0.21)	-0.17 m
Pickering Marsh	1.80 m	0.73 m	1.07 m (0.09)	0.34 m
Logan Lane North	1.78 m	0.72 m	0.74 m (0.20)	0.02 m
Logan Lane South	1.75 m	0.71 m	0.62 m (0.16)	-0.09 m
Prime Hook Unit 1	1.59 m	0.60 m	0.51 m (0.11)	-0.09 m
Prime Hook Unit 2	1.59 m	0.60 m	0.41 m (0.07)	-0.19 m
Prime Hook Unit 3	1.59 m	0.60 m	0.33 m (0.08)	-0.27 m
Prime Hook Unit 4	1.59 m	0.60 m	-	-

eigenvector technique that ordinates the data such that one axis explains the greatest amount of variance, and the second axis explains the next greatest amount of variance (Holland, 2008a). Rather in MDS, values are arranged in such a way that their separation corresponds to their degree of similarity; similar objects occur closer to each other, dissimilar objects are farther apart. (Wickelmaier, 2003).

PCA is a technique used to evaluate the variability of interrelated parameters by transforming them into dimensionless values, while retaining their initial variation (Jolliffe, 2002). The transformation is accomplished by generating values, or principal components, which are uncorrelated aggregates of the variation within the original data (Jolliffe, 2002; Holland, 2008b). The principal components are systematically generated so that the top few retain most of the variation of the original parameters (Jolliffe, 2002). The principal components are reduced to a matrix of eigenvalues and eigenvectors (Jolliffe, 2002; Holland, 2008b). The eigenvectors denote the directions of the variance (either positive or negative), and the eigenvalues are numerical values, which relay the variance present in that direction (Holland, 2008b). The eigenvector with the corresponding highest eigenvalue is the principal component (Holland, 2008b).

Data from a total of thirty-six cores were used in the nonmetric MDS and PCA analyses (Table 3.2). Fourteen of the cores were from the Delaware River Study Area. Of these cores, six were from reference marshes and eight from impoundment wetlands (see “n” value shown in Table 3.2 for specific number of cores from a given study site). A similar number of cores were available from the Central Delaware Bay Study Area. Of

Table 3.2 Mean, standard deviation (in parentheses) and results of Tukey's test (*p*-values) for all of the impounded and reference (shaded) wetlands within the Delaware River, Central Delaware Bay, and Lower Delaware Bay study areas.

Study Area Site (n=number of cores)	Mean Accretion (^{137}Cs, cm/yr)	Bulk Mass Accumulation (g/cm^2/yr)	Mineral mass accumulation (g/cm^2/yr)	Organic Mass Accumulation (g/cm^2/yr)
Delaware River **Lukens Marsh (n=3)**	0.83 (0.09)	0.54 (0.26)	0.45 (0.11)	0.09 (0.01)
Delaware River **Buttonwood (n=3)**	0.44 (0.05) $p<0.01$	0.38 (0.19) $p<0.05$	0.32 (0.01) $p=0.078$	0.053 (0.015) $p<0.05$
Delaware River **Broad Dike (n=3)**	0.56 (0.15) $p<0.05$	0.23 (0.07) $p<0.01$	0.167 (0.099) $p<0.05$	0.063 (0.012) $p<0.05$
Delaware River **Gambacorta (n=2)**	0.55 (0.35) $p=0.169$	0.18 (0.06) $p<0.01$	0.13 (0.028) $p<0.05$	0.050 (0.028) $p=0.119$
Delaware River **Rivers Edge Marsh (n=3)**	1.00 (0.15)	0.57 (0.26)	0.470 (0.118)	0.103 (0.012)
Central Delaware Bay **Port Mahon (n=3)**	0.30 (0.07) $p=0.077$	0.18 (0.09) $p=0.179$	0.157 (0.151) $p=0.260$	0.027 (0.012) $p<0.01$
Central Delaware Bay **Little Creek (n=3)**	0.59 (0.09) $p=0.195$	0.23 (0.10) $p=0.140$	0.187 (0.012) $p=0.238$	0.047 (0.006) $p<0.05$
Central Delaware Bay **Pickering Marsh (n=3)**	0.85 (0.20)	0.40 (0.16)	0.313 (0.157)	0.083 (0.015)
Central Delaware Bay **Logan Lane North (n=3)**	0.35 (0.08) $p=0.087$	0.20 (0.09) p=0.099	0.163 (0.021) $p=0.172$	0.033 (0.006) $p<0.01$
Central Delaware Bay **Logan Lane South (n=3)**	0.32 (0.04) p=0.078	0.17 (0.08) $p<0.05$	0.140 (0.070) $p=0.105$	0.027 (0.006) $p<0.01$
Lower Delaware Bay **Prime Hook Unit I (n=3)**	0.58 (0.18)	0.24 (0.09)	0.183 (0.111)	0.053 (0.006)
Lower Delaware Bay **Prime Hook Unit II (n=3)**	0.24 (0.01) $p<0.05$	0.20 (0.10) $p=0.321$	0.173 (0.023) $p=0.446$	0.027 (0.006) $p<0.01$
Lower Delaware Bay **Prime Hook Unit III (n=2)**	0.35 (0.11) $p= 0.093$	0.12 (0.04) $p=0.301$	0.085 (0.078) $p=0.262$	0.035 (0.007) $p=0.196$
Lower Delaware Bay **Prime Hook Unit IV (n=3)**	0.47 (0.06)	0.077 (0.007)	0.033 (0.015)	0.043 (0.012)

these, two were from the reference Pickering Marsh and twelve were from the study area's impoundment marshes (Table 3.2). Eight cores were used from the Lower Delaware Bay Study Area, three from the Prime Hook Unit I reference marsh and five from the Prime Hook Unit II and III impoundments. Core data from the Prime Hook Unit IV reference marsh were not used, since elevation data was not available for this site.

In the nonmetric MDS analysis, the variables (rates of accretion, bulk mass accumulation, mineral mass accumulation, organic mass accumulation, and ^{137}Cs and ^{210}Pb inventories) were normalized by assigning each a number between 0 and 1 corresponding to its rank (i.e., 0 = minimum; 1 = maximum value measured for that parameter). Normalization is done to allow for all the data to be scaled to allow for an easier comparison of relative values between parameters (Clarke and Gorley, 2006). The normalized parameters were aggregated and compared by evaluating the Euclidean distances between cores based on the normalization. The more similar the cores in their normalized values the closer their Euclidean distances, the more dissimilar the values the farther the distance between the cores. The ^{137}Cs and ^{210}Pb inventories were not discussed in detail previously, but for this analysis of variability, they were included as an additional parameter to be used in distinguishing between cores. Inventories of ^{137}Cs and ^{210}Pb are the concentration of the nucleotides normalized to the bulk density of the sediment and the thickness or depth of the sample (mm).

An examination of the nonmetric MDS plot indicates that the Central and Lower Delaware Bay Study Areas are more similar in their parameters in comparison to the

Delaware River Study Area. The Central and Lower Delaware Bay core data points tend to cluster in the same general ordinate location (Figure 3.5). The Delaware River Study Area data are much more widely distributed and do not occur, with the exception of the two cores from the Gambacorta impoundment wetland, in close proximity to the Central and Lower Delaware Bay data points (Figure 3.5).

For the Delaware River and Central Bay Study Areas, the nonmetric MDS plot indicates a distinct difference between the parameters of the impoundment wetlands and their associated reference marshes. The dissimilarity is most pronounced for the Delaware River Study Area where the reference Rivers Edge and Lukens Marsh core data points generally cluster away from the more widely spaced impoundment wetland data (Figure 3.5). The Central Delaware Bay reference Pickering Marsh core data points are located relatively near the Delaware River reference marshes and are separated from their impoundment wetlands. The Lower Delaware Bay Prime Hook Unit I reference marsh core data points are not clustered, and there is no clearly apparent dissimilarity between the reference marsh and impoundment wetlands core data points for this study area.

The clustering of the Lower and Central Delaware Bay cores as shown in the nonmetric MDS analysis is expected, as their conditions of deposition and water-level management have most likely led to values (rates of accretion, bulk mass accumulation, mineral mass accumulation, organic mass accumulation, and ^{137}Cs and ^{210}Pb inventories) that are more similar, than those of the Delaware River cores. For the Delaware River study area, the separation in Euclidean distances of the Buttonwood and Broad Dike cores from the Gambacorta, Rivers Edge, and Lukens Marsh cores, is likely driven by the

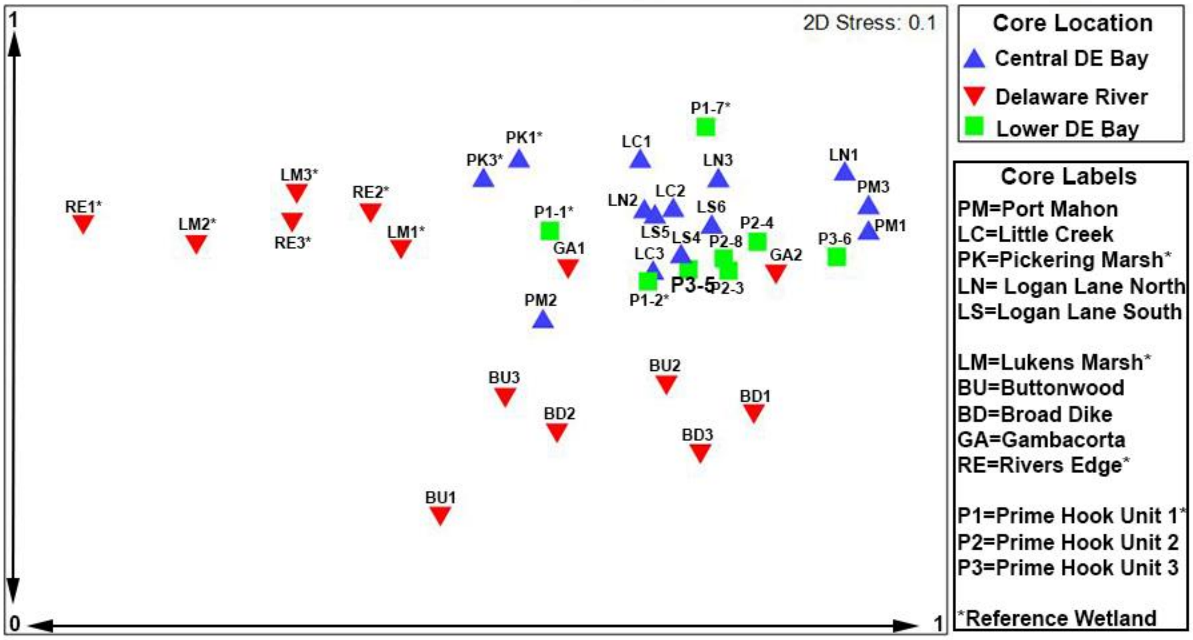


Figure 3.5 Nonmetric multidimensional scaling (MDS) plot for radiometric core parameters. X- and Y-axes are unit-less allowing for comparison of different parameters.

standardized elevation variations between the core sites (i.e., the standardized elevations of the Buttonwood and Broad Dike sampling sites are much lower (-0.89 and -1.23 m, respectively) compared to Gambacorta and the Lukens Marsh and Rivers Edge Marsh reference wetlands (-0.55, 0.11, and 0.12 m respectively; Table 3.1).

The results from the PCA analyses are similar to the nonmetric MDS. As shown in Figure 3.6, the Central and Lower Delaware Bay Study Area core data points cluster closer together and the Delaware River Study Area data are more broadly distributed. The PCA plot also shows dissimilarity between the reference marshes and impoundment wetlands for the Delaware River and Central Delaware Bay Study Areas (Figure 3.6). In contrast to the nonmetric MDS analysis, the PCA is evaluating the percentage of variance that can be explained by the differences in the parameters of interest (rates of accretion, bulk mass accumulation, mineral mass accumulation, organic mass accumulation, and ^{137}Cs and ^{210}Pb inventories).

The principal components that show the greatest effect on variance (principal components (PC) 1 and 2) are the axes on the PCA plot (Figure 3.6). Eigenvalues show that PC1 and PC2, account for 80.3% of the cumulative variation, with PC1 accounting for 66.5% (Table 3.3). The eigenvectors for PC1 indicate that the variation within this component is dominated by the rates of bulk mass accumulation, mineral mass accumulation, organic mass accumulation, and accretion, respectively (Table 3.3). As these parameters are all closely related, it stands to reason that changes in the rates of mineral mass and organic mass accumulation would affect the rates of accretion and/or bulk mass accumulation. For PC1, as shown in Table 3.3 and the rosette in Figure 3.6, all of the variables impact the positions of the core data points in a negative direction.

The eigenvectors for PC 2 show that the variation within this component is dominated by mean elevation of the site (in a positive direction) and ^{210}Pb inventory in a negative direction (Table 3.3; also shown in the rosette in Figure 3.6). ^{210}Pb inventory was not examined in detail within this analysis as it was a parameter that was not present in appreciable quantities in all of the collected cores. The Buttonwood and Broad Dike cores, as compared to the other data collection sites, have the lowest, and largest deviations between their, elevations. These relationships are shown in Figure 3.6 by their lower position (relative to the PC2 axis) and broader distribution for a given core collection site.

3.5.6 ANOVA Statistical Analysis

To examine in further detail the differences between the three study areas, and within each of the study areas the variability between reference marshes and impoundments, one-way analysis of variance (ANOVA) techniques were employed. The technique was used to determine if the means of the rates of accretion, bulk mass accumulation, mineral mass accumulation, and organic mass accumulation within a given study area (e.g., Delaware River sites) were statistically different than the rates of the other study areas (e.g., Central Delaware Bay and Lower Delaware Bay sites). The one-way ANOVA statistical determination was an important component of this project

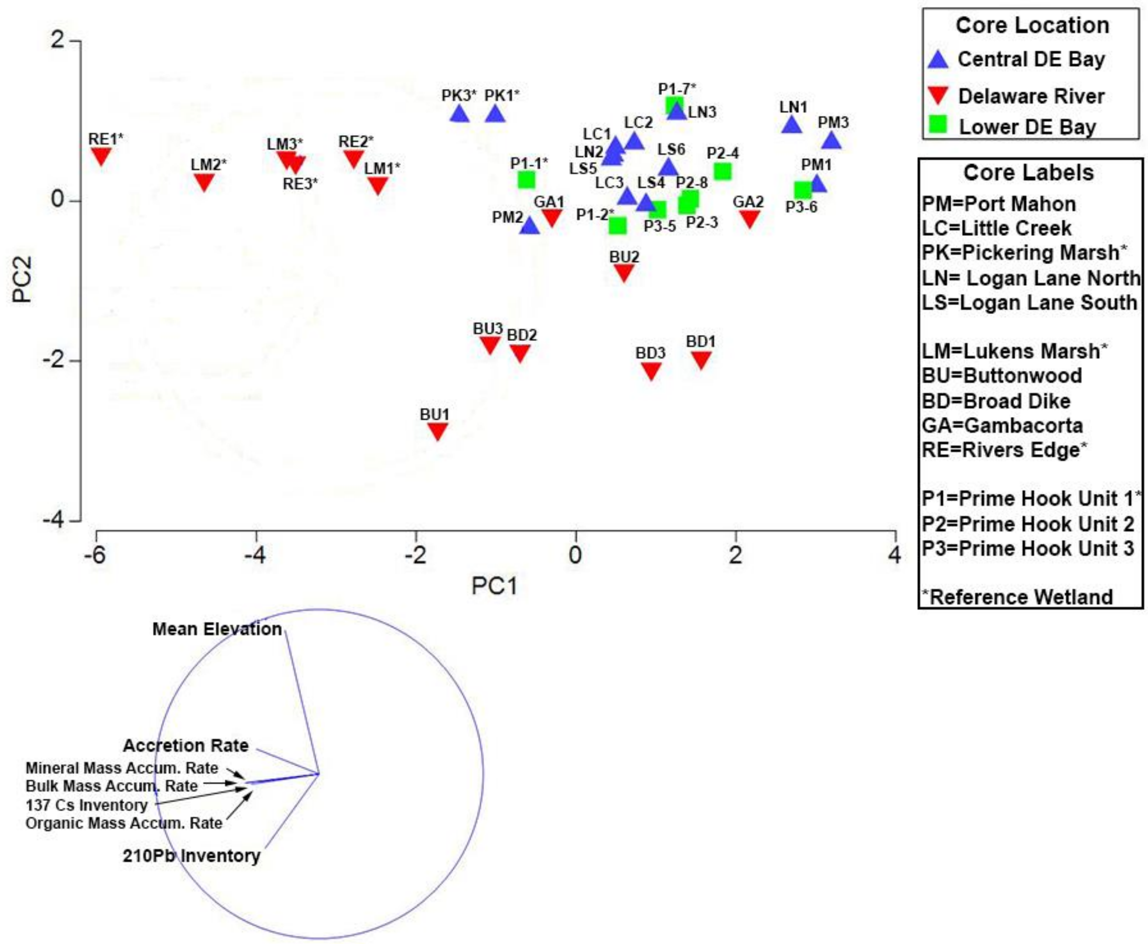


Figure 3.6 Principal Component Analysis (PCA) plot for radiometric core parameters. The rosette of the principal components is displayed beneath the PCA plot.

Table 3.3 Principal Component Analysis (PCA) eigenvalue and eigenvector outputs for the radiometric core parameters.

Eigenvalues

PC	Eigenvalues	%Variation	Cumulative % Variation
1	4.66	66.5	66.5
2	0.962	13.7	80.3
3	0.584	8.3	88.6
4	0.39	5.6	94.2
5	0.36	5.1	99.3

Eigenvectors

Variable	PC1	PC2	PC3	PC4	PC5
Accretion Rate	-0.392	0.102	0.658	0.008	-0.082
Mineral Mass Accumulation Rate	-0.425	0.009	-0.289	0.155	0.528
Organic Mass Accumulation Rate	-0.422	-0.090	0.473	-0.065	-0.071
^{137}Cs Inventory	-0.375	-0.059	-0.327	0.568	-0.652
^{210}Pb Inventory	-0.338	-0.447	-0.293	-0.721	-0.242
Bulk Mass Accumulation Rate	-0.441	-0.008	-0.170	0.123	0.446
Mean Elevation	-0.194	0.882	-0.196	-0.336	-0.165

because the study areas are geographically separated, and their impoundment management strategies correspond to their different geographic locations. Statistically different results support the hypothesis that geographical position and thus impoundment management have resulted in differences between the marshes in the three study areas.

Tukey's honest significant difference (HSD) test was conducted after the ANOVA analysis (post-hoc) to determine if the differences in the means of the marsh properties measured (rates of accretion, bulk mass accumulation, mineral mass accumulation, and organic cumulative mass) were statistically significant between impoundment wetlands and their corresponding reference marshes. A significant difference was defined by significance level (*p*-values) less than 0.05, implying that there was less than a 5% probability that there were no differences between the impoundment and reference marshes. The descriptive statistics (i.e. mean and standard deviation (in parenthesis)) and the *p*-values (i.e., significance levels as determined from Tukey's HSD test) for each of the wetlands within each of the study areas are summarized in Table 3.2. The number of cores from each of the sites from which the mean rates were determined is also included (Table 3.2).

The results from the one-way ANOVA determination indicate that there were statistically significant differences in the mean rates of accretion, bulk mass accumulation, mineral mass accumulation, and organic mass accumulation between the three study areas. All of the *p*-values for these mean rates are much less than 0.05, ranging between a maximum of p=1.4x10^{-5} and a minimum of p=2.2x10^{-7} for the mean rates of accretion and organic mass accumulation, respectively. The three study areas span ~100 km along the Delaware Bay and River. The Delaware River and Central Delaware Bay study areas are ~72 km apart, while the Central and Lower Delaware Bay areas are ~27 km apart. The ANOVA results support the hypothesis that the different geographical locations (and corresponding differences in management associated with their locations) are factors that are responsible for the variations in the characteristics (i.e., rates of accretion, bulk mass accumulation, mineral mass accumulation, and organic mass accumulation) of the marshes in the different study areas. When examining the differences between the reference and impoundment marshes within each of the study areas as determined from the Tukey's HSD test, there is variability in results between the factors studied, as such results from each of the rates will be described individually.

3.6 Results

3.6.1 Mean Accretion Rates

Although it needs to be taken in context with the results of statistical significance tests that are described below, there is a pronounced difference in mean accretion rate between the impoundment wetlands and the reference marshes. The mean accretion rate (0.82 cm/yr; SD=0.17)) of the four reference marshes (Lukens, Rivers Edge, Pickering, and Prime Hook Unit I) is nearly twice as much as the mean rate (0.41 cm/yr; SD=0.15) of the nine impoundment wetlands. Prime Hook Unit IV Marsh is not included in the reference mean calculation as platform elevation data (used in comparing accretion, bulk mass accumulation, mineral mass accumulation, and organic mass accumulation rates relative to standardized wetland elevations) were not obtained for this site.

Even though the mean accretion rate between the reference marshes and impoundment wetlands is markedly different, only three of the nine impoundment wetlands have mean accretion rates that are statistically significantly different ($p<0.05$) when compared to their specific reference marshes (Table 3.2). In the Delaware River study area, the mean accretion rates of the Buttonwood and Broad Dike wetlands are significantly different from the reference Lukens Marsh. In the Lower Delaware Bay study area, the Prime Hook Unit II Marsh mean accretion rate is significantly different from its reference Prime Hook Unit I Marsh. In the Central Delaware Bay study area, none of the impoundment wetlands are statistically different at the $p<0.05$ confidence level with respect to the mean accretion rate of their reference Pickering Marsh (Table 3.2).

The differences in mean accretion rates between the reference marshes and the impoundment wetlands are shown in a plot of mean accretion rates versus standardized marsh/wetland elevations (Figure 3.7). The reference marsh mean accretion rates cluster in the higher accretion rate, higher standardized wetland elevation portion of the plot. The impoundment wetlands mean accretion rates are less than the reference marshes and plot in the moderate to lower accretion rate (<0.6 cm/yr) portion of the diagram (Figure 3.7). The reference marsh mean accretion rates relative to standardized wetland elevations show increasing accretion rates as a function of increasing wetland elevations (Figure 3.7). In contrast, the impoundment wetlands exhibit generally decreasing mean accretion rates as wetland elevations increase (Figure 3.7). Of the impoundments, the Delaware River study area wetlands cluster in the lower elevations, moderate accretion rates (0.4 to 0.6 cm/yr) portion of the diagram in comparison to the Central and Lower Delaware Study area wetlands that are mainly located in the moderate to higher elevation (-0.4 to 0.1 m relative to MHW) and lower accretion rate (<0.4 cm/yr) area of the plot (Figure 3.7).

3.6.2 Mean Bulk Mass Accumulation Rates

Four of the nine impoundment wetlands have mean bulk mass accumulation rates that are statistically significantly different (p<0.5) in comparison to their reference marshes (Table 3.2). In the Delaware River study area, all three of the impoundment wetlands (Buttonwood, Broad Dike, and Gambacorta) have mean bulk mass accumulation rates that are statistically significantly different from their reference marshes. In the Central Delaware Bay study area, the Logan Lane South Marsh is the only one with a statistically significant difference in comparison to its reference Pickering Marsh. In the Lower Delaware Bay study area, none of the impoundment wetlands are statistically different at the p<0.05 confidence level with respect to the mean bulk mass accumulation rate of the reference Prime Hook Unit I Marsh (Table 3.2).

The difference in the bulk mass accumulation since 1954 in the impounded and natural reference marshes demonstrates that the variance is not only the function of water level management, but also the position along the coast and impoundment age (when the age of the impoundments are taken into consideration based upon previous discussions;

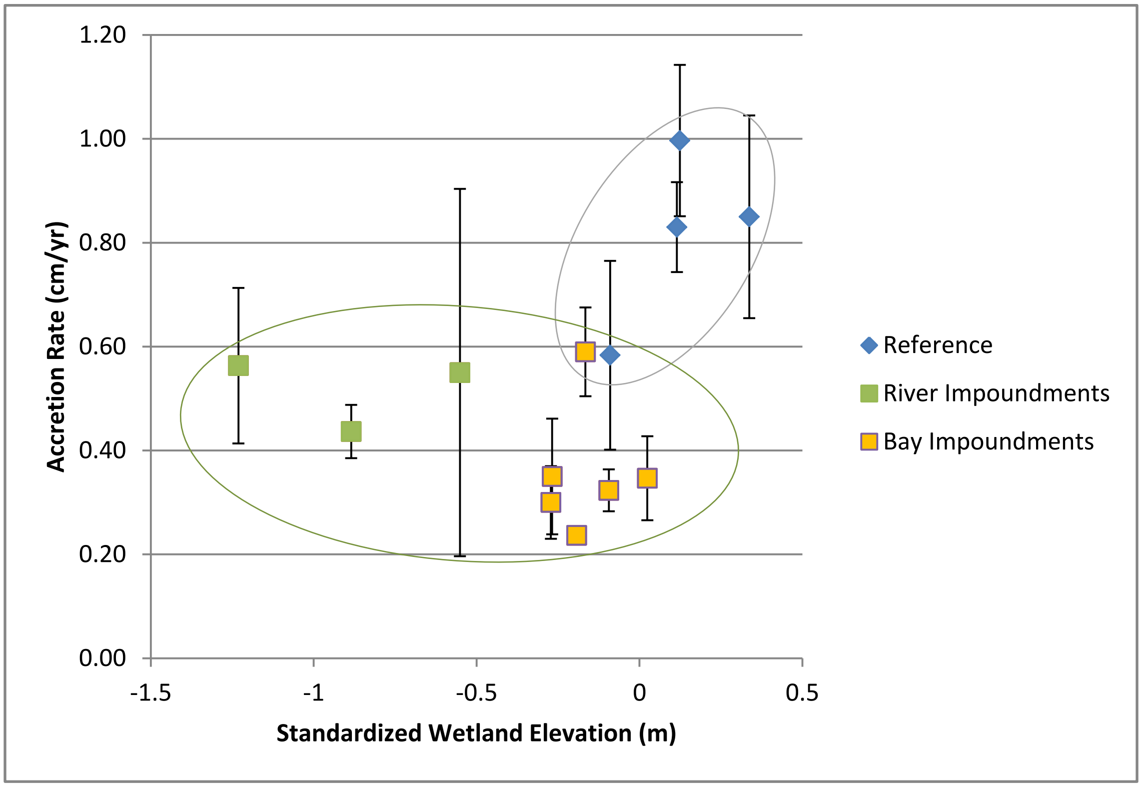


Figure 3.7 Rate of accretion of impounded and natural reference wetlands versus their standardized wetland elevation (MHW elevation subtracted from mean platform elevation; m NAVD 88). Standard deviation ranges for each data point are shown.

Figure 3.8). The impounded marshes exhibit a negative trend between bulk mass accumulation rate and standardized wetland elevation, with accumulation rates decreasing as the standardized wetland elevation increases toward 0 m (MHW) (Figure 3.8). In contrast, the natural or reference marshes exhibit increasing mass accumulation rates as the standardized marsh elevations increase above 0 m (above MHW).

3.6.3 Mean Mineral Mass Accumulation Rates

Only two of the nine impoundment wetlands have mean rates of mineral mass accumulation that are statistically significantly different (p<0.5) in comparison to their reference marshes (Table 3.2). These two wetlands are the Broad Dike and Gambacorta Marshes located in the Delaware River Study Area. As shown in Figure 3.9 and similar to that observed in plots of the mean accretion and mean bulk mass accumulation rates versus standardized marsh elevation (Figures 3.9 and 3.10, respectively), the mean rates of mineral mass accumulation for the impounded wetlands exhibit a general decrease as elevations increase towards MHW (0 m), while the reference wetlands show increasing rates as standardized marsh elevations increase above MHW (0 m elevation).

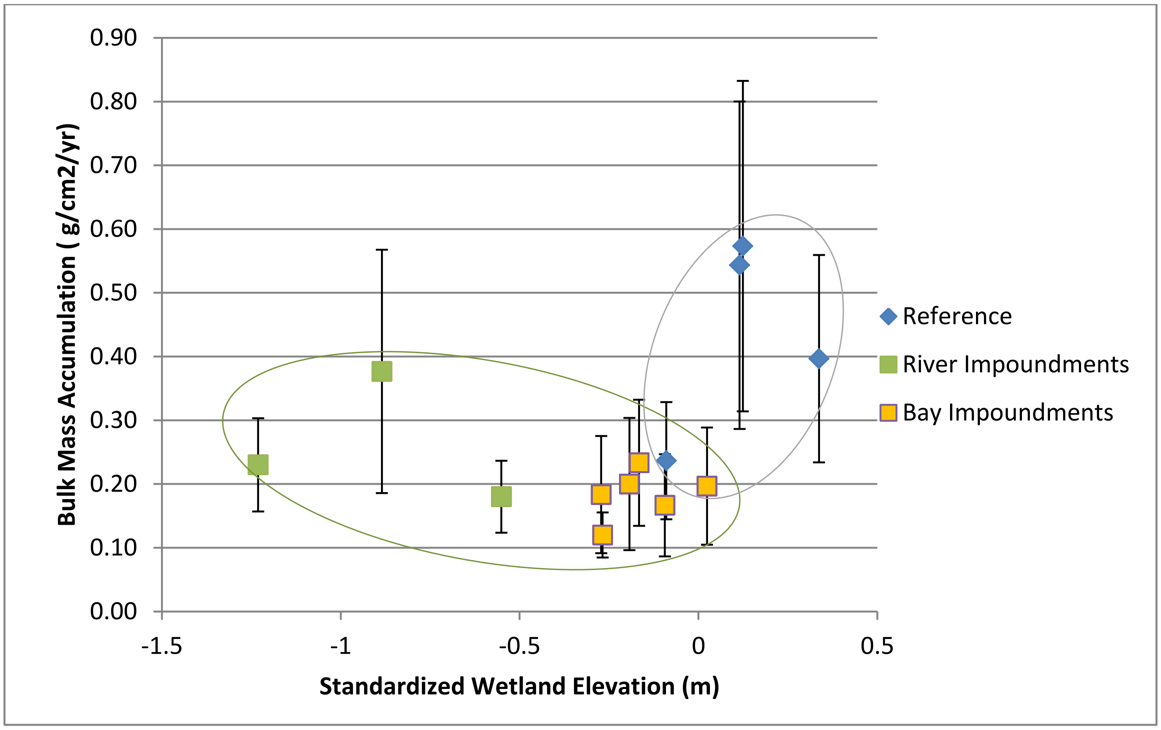


Figure 3.8 Bulk mass accumulation rates of impounded and natural reference wetlands versus their standardized wetland elevation (MHW elevation subtracted from mean platform elevation; m NAVD 88). Standard deviation ranges for each data point are shown.

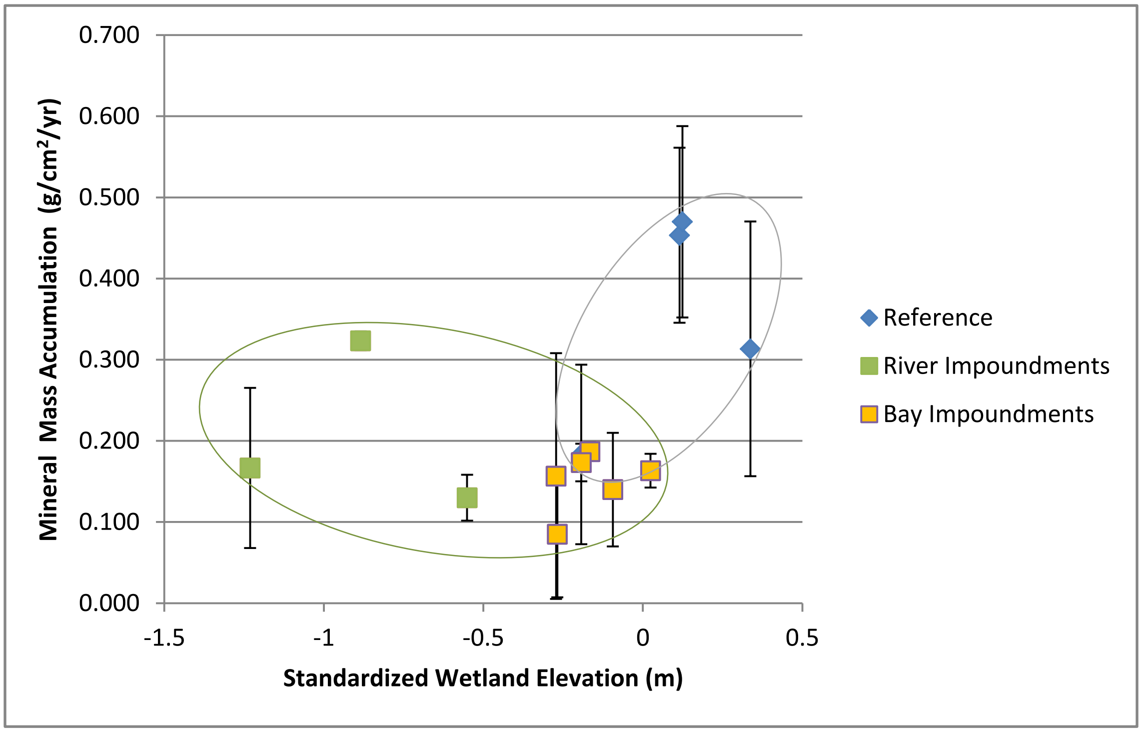


Figure 3.9 Mineral mass accumulation rates of impounded and natural reference wetlands versus their standardized wetland elevation (MHW elevation subtracted from mean platform elevation; m NAVD 88). Standard deviation ranges for each data point are shown.

3.6.4 Mean Organic Mass Accumulation Rates

In contrast to mean rates of accretion, bulk mass accumulation, and mineral mass accumulation, the mean rates of organic mass accumulation are statistically significantly different (p<0.5) for seven of the nine impoundment wetlands in comparison to their reference marshes (Table 3.2). In the Delaware River Study Area, two of the three impoundments (Buttonwood and Broad Dike) are statistically significantly different in mean rates of organic mass accumulation in comparison to their Lukens Marsh and Rivers Edge Marsh reference wetlands. All four of the impoundment marshes (Port Mahon, Little Creek, Logan Lane North, and Logan Lane South) in the Central Delaware Bay Study Area are statistically different in organic mass accumulation rates in comparison to their Lukens Marsh and Rivers Edge Marsh reference wetlands. All four of the impoundment marshes (Port Mahon, Little Creek, Logan Lane North, and Logan Lane South) in the Central Delaware Bay Study Area are statistically different in organic mass accumulation rates in comparison to their Pickering Marsh reference wetland. In the Lower Delaware River Study Area, one of the two impoundments (Prime Hook Unit II) is statistically significantly different when compared to its Prime Hook Unit I reference wetland.

A plot of mean organic mass accumulation rates versus standardized marsh elevations for the impoundment and reference marshes shows similar relationships to that observed in plots of mean accretion, bulk mass accumulation, and mineral mass accumulation rates versus standardized marsh elevations (Figure 3.10). The impounded wetlands organic mass accumulation rates appear to have a weak negative relationship with respect to the standardized wetland elevation. The similar apparent positive relationship between organic mass accumulation rates and standardized wetlands elevation is observed for the reference wetlands (Figure 3.10).

The rate of organic matter accumulation at impounded and natural (reference) wetlands reflects the difference in the water level and salinity management of each site. The influence of water level is shown by the three Delaware River Study Area impoundments, with the lowest standardized wetland elevations of all the sampled marshes, having the three highest impoundment mean rates of organic mass accumulation (i.e., Broad Dike 0.063 g/cm^2/yr, Buttonwood 0.053 g/cm^2/yr, and Gambacorta 0.050 g/cm^2/yr; Table 3.2). The Delaware River Study Area reference wetlands also contain the highest mean organic mass accumulation rates of the reference wetlands (Lukens Marsh 0.09 g/cm^2/yr, and Rivers Edge 0.103 g/cm^2/yr; Table 3.2). The six impounded wetlands from the Central and Lower Delaware Bay Study Areas have organic mass accumulation rates that generally cluster, with values between 0.027 and 0.047 g/cm^2/yr (Mahon 0.027 g/cm^2/yr, Little Creek 0.047 g/cm^2/yr, Logan Lane North 0.033 g/cm^2/yr, Logan Lane South 0.027 g/cm^2/yr, Prime Hook Unit 2 0.027 g/cm^2/yr, and Prime Hook Unit 3 0.035 g/cm^2/yr; Table 3.2; Figure 3.10). Prime Hook Unit I, the reference wetland for Prime Hook Units II and III, has the lowest reference organic mass accumulation rate of 0.053 g/cm^2/yr (Figure 3.10). The Pickering Marsh wetland (reference for Port Mahon, Little Creek, Logan Lane North, and Logan Lane South) has an organic mass accumulation rate that is nearly one and a half to two times that of its neighboring impounded wetlands (0.083 g/cm^2/yr; Table 3.2; Figure 3.10).

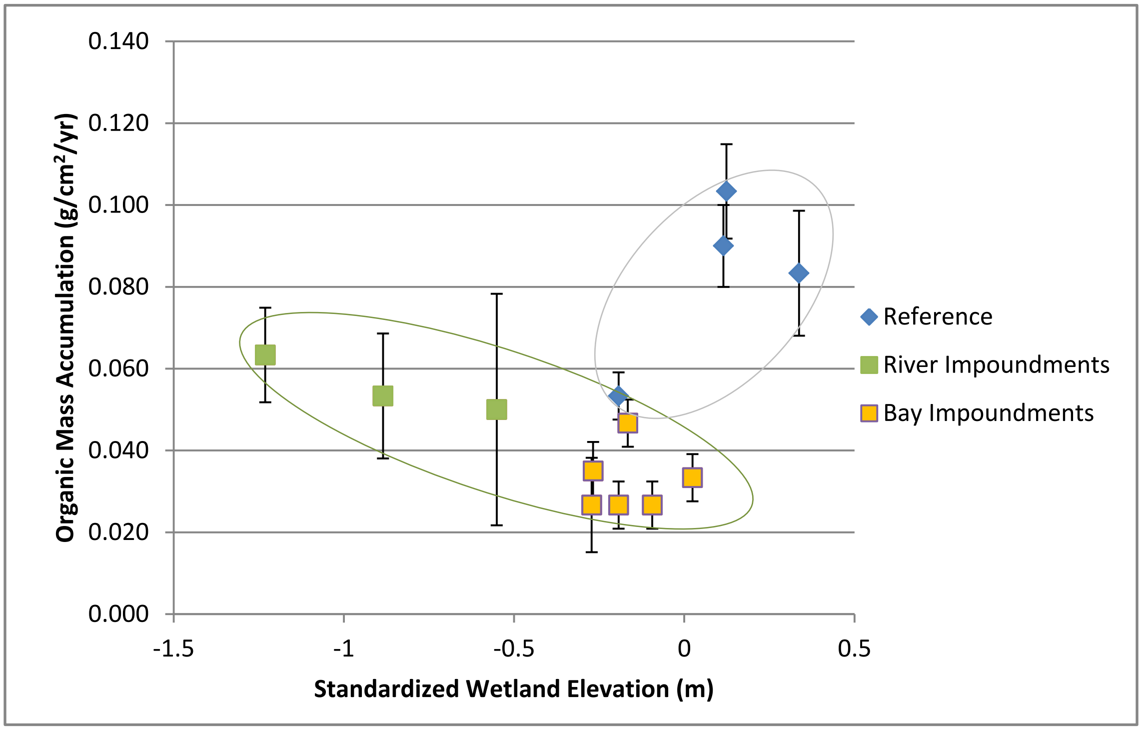


Figure 3.10 Organic mass accumulation rates of impounded and natural reference wetlands versus their standardized wetland elevation (MHW elevation subtracted from mean platform elevation; m NAVD 88). Standard deviation ranges for each data point are shown.

The three highest organic mass accumulation rates for impoundments and the two highest for reference wetlands are all from wetlands that experience lower mean salinity values, as the result of their position along the coast, their overall management scheme, or size of their drainage basin. The six clustered lowest rates of mean organic mass accumulation for impoundments, and the two lowest organic mass accumulation rates for reference wetlands all have higher mean salinity values given their location in the middle to lower portions of the Delaware Bay.

The importance of the interrelationship of mineral (inorganic) and organic mass accumulation, and the effect of water-level manipulations upon them in the building of a marsh platform is illustrated in Figure 3.11 through a strong linear relationship (R^2=0.905), with corresponding increases in both organic and mineral mass accumulation rates. This increasing relationship is reflective of the reference wetlands positions along the coast, with Prime Hook Unit IV containing the lowest organic and inorganic mass. Pickering Beach contains a moderate accumulation in both constituents, and finally the Delaware River reference marshes with the highest accumulation rates in both mineral and organic mass (Table 3.2; Figure 3.11).

In contrast, the impounded marshes show no distinguishable correlation between mineral and organic mass accumulation rates, with a very weak relationship (R^2=0.145: Figure 3.11) observed between increasing rates of mineral mass accumulation and corresponding increases in rates of organic mass accumulation. The highest rate of mineral mass accumulation is measured at the Buttonwood Impoundment (mean 0.32 g/cm^2/yr, SD 0.01), with the remaining impoundments having relatively low

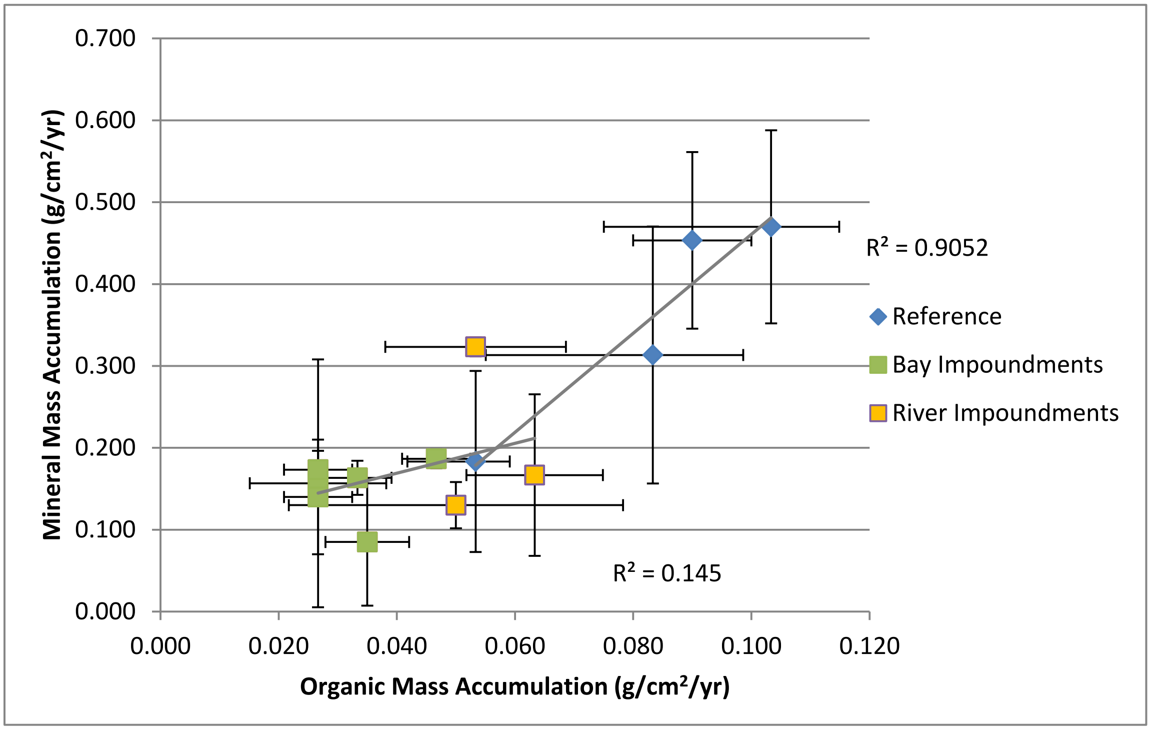


Figure 3.11 Mineral mass accumulation rates of impounded and natural reference wetlands versus their rates of organic mass accumulation. Standard deviation ranges for each data point, trendlines, and goodness of fit (R^2) also shown.

(<0.2 g/cm^2/yr) mineral mass accumulation rates (Figure 3.11). The two lowest organic mass accumulation rates for reference wetlands all have higher mean salinity values being located in the middle to lower portions of the Delaware Bay.

3.7 Discussion

Evaluating a wetland's ability to naturally evolve within its regional setting, before anthropogenic intervention, may be paramount in developing an understanding of its future response. Recovery is commonly thought of as a restoration of the system to the conditions before the area was disturbed, thus achieving a return to natural equilibrium (Magilligan and Stamp, 1997). Natural equilibrium is thought to be a relatively continuous relationship between the scale and degree of the inputs and outputs that affect the site's evolution, with an expectation that the relationship naturally changes through time, but ultimately maintains some longer-term median condition (Mackin, 1948; Hack, 1960; Langbein and Leopold, 1964; Renwick, 1992; Magilligan and Stamp, 1997). In many cases, recovery to a condition that closely approximates pre-disturbance conditions may take long, geologic-scale, intervals of time (Magilligan and Stamp, 1997). The recovery of tidally-influenced systems is not well understood and far less studied than upland systems. It is not known, definitively, if natural recovery can be obtained in tidal wetlands, let alone determining/estimating a timescale for it to fully occur.

Lasting effects of anthropogenic disturbance may alter a system beyond normal restoration activities (Magilligan and Stamp, 1997). Sediment storage in fluvial to tidal marsh systems increases with increased drainage area (Renfroe, 1975), and sediment pulses of material may require tens to hundreds of years to migrate through a watershed (James, 1989; Magilligan and Stamp, 1997). Therefore, land-use alterations which have affected upland sediment input over the past ~250 years, could possibly affect wetlands and watersheds in the future even after stressors have been alleviated.

The lack of mineral sediment input, due to the constriction of water control structures, water level manipulation, and reduced regular tidal flushing, has long been suspected of impacting impounded wetlands (Hossler, 1994; Meredith and Whitman, 1994). To shed further light on this hypothesis, earlier results in this dissertation suggest that plant biomass production plays the largest role in accretion. Specifically, it is the below-ground biomass production that appears to dictate overall platform elevation in relation to MHW elevation (see Chapter 2). The interrelation of vegetative biomass production and perceived organic decomposition can been shown, through the Tukey's HSD tests, to account for a significant difference between the impounded and reference wetlands. The statistical differences in the organic mass accumulation is irrespective of the length of impoundment (years) and/or overall water level management scheme (Table 3.2).

It is noteworthy that only two impoundments contained a statistically significant difference (since 1954) in mineral mass accumulation rates in comparison to their reference wetlands. This illustrates that the long hypothesized notion that mineral accumulation dominantly dictates trends in platform elevation (Hossler, 1994; Meredith and Whitman, 1994) is not a ubiquitous theory that can be applied to all tidal wetland and adjacent impoundments. Water control structures and/or manipulations could affect the overall volume of suspended mineral mass that could enter an impounded wetland, as compared to a regularly flooded reference wetland. Even natural wetlands commonly experience a lack of vertical accretion, usually in the marsh interior, in locations that are the greatest distance from the flooding source.

The Delaware River study area impoundments and reference wetlands are in close proximity to the turbidity maximum of the Delaware Estuary (Sommerfield and Wong, 2011), yet two of these three impoundments (i.e. Broad Dike and Gambacorta) show a statistically significant greater difference (at the $p<0.05$ level) in mineral mass accumulation rates compared to their reference wetlands. The Delaware River study area impoundments have improved regular tidal flushing, and subsequent mineral mass input, since the mid-1990's, but over the past 60 years the time average or long-term trend is that the mineral mass accumulation rate differences have been relatively significant (in relation to the reference wetlands). In comparison, the Central and Lower Delaware Bay study area impoundments are in a setting of lower suspended sediment concentrations (not being in close proximity to the turbidity maximum). The suspended sediment concentrations lower in the estuary are significantly different between tidal levels, and are commonly fed by the cannibalization of existing marsh and the re-circulation of re-suspended sediments from the bay bottom and wetlands (Cahoon and Reed, 1995; Reed, 1995).

The adaptability of coastal wetlands to sea-level rise has been partly attributed to flooding that will slow rates of organic matter decay (under anaerobic condition) and facilitate more rapid organic matter accumulation (Nyman and DeLaune, 1991; Reed, 1995; Miller et al., 2001). However in a study by Kirwan et al. (2013), it was found that enhanced flooding in brackish marshes led to little change in the rate of organic matter decay. Instead, Kirwan et al. (2013) suggested that enhanced root production (e.g. Langley et al., 2009; Kirwan and Guntenspergen, 2012) and/or mineral sedimentation were the principal mechanisms by which marshes could maintain their equilibrium in response to sea-level rise (e.g. Morris et al., 2002).

Although the Delaware River, Central Delaware Bay, and Lower Delaware Bay study area impoundments all have distinctly different water level management schemes and goals for management, the Tukey's HSD analysis shows that they all (except Prime Hook Unit III and Gambacorta; Table 3.2) had statistically significant lower organic mass accumulation rates, as compared to their reference wetlands. The water level management schemes vary in time, duration, and amount of drawdown, but what is consistent are drawdowns in the spring/summer to reduce mosquito breeding and promote annual vegetation growth. During these drawdowns, the Delaware River and Central Delaware Bay study area impoundment wetlands are exposed, with flushing (in the ditches and pools), allowing for annual and perennial vegetation growth.

The marsh surfaces are fully exposed in the Delaware Bay study area impoundments only from March 1 to April 30, with pool levels of 50% from May 1 to 30, and then 75% levels from June 1 through July 31 (Meredith and Whitman, 1994). Thus, aerobic exposure of the marsh flats is up to 100% from March 1 to April 30, 50% to 55% from May 1 to 30, and 25% to 35% from June 1 to July 31 (Meredith and Whitman, 1994). The Central Delaware Bay study area impoundments have a drawdown on March 28 to a 50% pool, which is maintained until August (Meredith and Whitman, 1994). This 50% pool level also incorporates a prescribed maximum tidal exchange through the water control structures to prevent soil salinity from becoming too high (Meredith and Whitman, 1994).

A 50% marsh surface exposure may or may not always be possible in several of the Central Delaware Bay study area impoundments. The water control structures in these impoundments have experienced subsidence, in conjunction with rising sea level, and now have difficulties with moving water adequately in and out (Meredith and Whitman, 1994; Meredith et al., 2004). It is common for these impoundments (especially Port Mahon and Logan Lane North) to develop extensive desiccation cracks along their marsh surface (Figure 3.12). The desiccation cracks are indicative of long-term exposure, and therefore aerobic soil conditions (Kirwan et al., 2012). Evidence of desiccation is not as common in the Delaware River study area impoundments. This may be a reflection of the ability to better control these impoundments to meet water level management goals.

The Lower Delaware Bay study area impoundments are managed under freshwater moist soil management, with a 50% pool from March to mid-April, then a 0% pool from mid-April to July (USFWS, 2012). The marsh flats are 50% exposed from March to mid-April and then 100% exposed mid-April to July (USFWS, 2012). Extensive exposure conditions (with limited or no punctuated anaerobic conditions via regular inundation) could drive organic decomposition rates to be extremely high (Neckles and Neill, 1994; Turner et al., 2000; Kirwan et al., 2012). Especially when compared to natural inundating of reference wetlands and controlled flooding and

Figure 3.12 Images of desiccation cracks that formed on the marsh surface of the Port Mahon Impoundment (Images taken 07/26/2011).

draining of impoundments under prescribed water level management conditions (Kirwan and Guntenspergen, 2012). High water levels from August to mid-October would also likely reduce late summer and fall growth of wetland vegetation, thereby reducing the overall opportunity for vegetative growth and production of below-ground biomass.

Given the current water management, it is not surprising that the Port Mahon, Logan Lane North, Logan Lane South, Prime Hook Unit II, and Prime Hook Unit III wetlands contain the lowest rates of organic mass accumulation (Table 3.2; Figure 3.10). In comparison, the Delaware River study area Buttonwood, Broad Dike, and Gambacorta Impoundments have the highest organic mass accumulation rates of all impoundments (Table 3.2). These higher rates could be the result of the lower salinity conditions in the Delaware River study area promoting a more stable vegetative community (and thereby a more robust below-ground biomass production). However, this is not supported by the data from the higher salinity Central and Lower Delaware Bay study areas where the reference Pickering Beach and Prime Hook Unit I Marshes had rates of organic mass accumulation that were higher than, and comparable to, the Delaware River study area impoundments, respectively (Table 3.2).

A more appropriate assessment may be that water level management in the Delaware River study area impoundments is conducive to below-ground biomass production, and promotes a lower rate of organic decomposition with shorter punctuated aerobic exposure of the marsh platforms. A natural marsh platform is exposed to regular tidal inundation causing a shift in the decomposition setting (Kirwan et al., 2013). At lower tides, the marsh platform will become exposed and dry causing the rate of aerobic decomposition to increase (also through oxygen pumping into the soil through the roots of halophytes such as *S. alterniflora*) (Kirwan and Guntenspergen, 2012; Kirwan et al., 2012). As the marsh surface becomes inundated, the available oxygen is reduced and anaerobic decomposition rates increase (Kirwan et al., 2012 and 2013). Anaerobic decomposition is slow and under long-term conditions results in the accumulation of large quantities of organic material (e.g., peat in bogs and wetlands) (Kirwan et al., 2012 and 2013).

It can be theorized that the most favorable means of managing impounded wetlands so that they vertically accrete is through enhanced below-ground biomass growth and retention. To do this, the impoundments in the Central and Lower Delaware Bay study areas (especially Port Mahon, Logan Lane North, Logan Lane South, and Prime Hook Unit III) need to have reduced long-term drawdowns, and increased control of water levels to allow for more short-term marsh platform inundations. The Delaware River study area impoundments have a period from June 1 to July 31 when the marsh platforms are at a 75% pool (Meredith and Whitman, 1994). With the addition of retrofit water control structures, these impoundments now appear to have better short-term management thereby reducing the time periods of aerobic decomposition during exposure of the marsh platforms. This enhanced water management provides an explanation for the relatively higher organic mass accumulation rates for these impoundments. However, it should be noted that for two of the three Delaware River study area impoundments (Buttonwood Dike and Broad Dike), their rates, although higher, are still statistically significantly lower than the organic mass accumulation rates of their comparison, natural-setting, reference marshes.

3.8 Conclusions

Regular tidal inundation is vital in the development and evolution of natural marsh platforms. Manipulations of water levels, through management and impoundment, can extensively alter long-term vertical accretion. This study evaluated the differences in accretion between natural and impounded wetlands. While there was a nearly two times greater difference in the mean rate of accretion between the reference (i.e., natural) marshes (0.82 cm/yr) and the impoundment wetlands (0.41 cm/yr), only three of the nine impoundments had statistically significant differences (at the $p<0.05$ level) in their mean accretion rates, and only four of the nine contain statistically significant differences (at the $p<0.05$ level) in bulk mass accumulation rates in comparison to their reference marshes. Long-term bulk mass accumulation rates, determined using radioisotopic dating (^{137}Cs), encompass deposition over the past ~60 years. As such, the impact of water level management changes and water control structure upgrades, or lack thereof, are aggregated into the results over the time interval of measurement (i.e., 1954 – present).

One of the major results is that only two of the nine impoundments studied were characterized by statistically significant differences (at the $p<0.05$ level) in mineral mass accumulation rates when compared to their reference wetlands. These results contradict the generally accepted hypothesis that mineral mass accumulation is the largest difference between natural and impounded wetlands. For example, it is commonly discussed in coastal resilience outreach, and within the management agencies for impoundments in Delaware, that the lack of sediment deposition is a leading factor in the differences in accretion rates and overall marsh platform elevation between impounded and natural wetlands (Hossler, 1994; Meredith and Whitman, 1994).

Although the highest priority might be to control water levels for wildlife of concern (i.e. waterfowl, shore birds, mosquito control, etc.), it is also important to manage an impoundment in a way that below-ground biomass production is optimized and decomposition rates are not enhanced through long-term platform exposure. A lower marsh platform, with respect to MHW, makes water level management even more difficult, especially facing accelerating rates of sea-level rise. If catastrophic levee breaches occur, as witnessed at Prime Hook Unit II in 2008, then the lowered platform will be permanently flooded (due to disequilibrium with the tidal prism), and a total loss of the resource for desired habitat management goals would occur. For example, the Prime Hook Unit II Impoundment is now a lagoon rather than a tidally-flooded back barrier wetland. This loss of a managed wetland serves as an indicator of the importance that water level management can have upon the long-term sustainability of impounded wetlands, and the importance of managing marsh platform elevation, especially in the face of sea-level rise.

Chapter 4

UTILIZING SURFACE ELEVATION TABLES (SETS) TO EVALUATE MARSH SURFACE ELEVATION AND ACCRETION TRENDS ALONG THE ST. JONES RIVER AND BLACKBIRD CREEK

4.1 Introduction

The evaluation of coastal resources is a necessity for resource managers and conservation organizations, especially as sea-level rise and anthropogenic alterations have notably impacted tidal wetlands. Assessing marsh surface elevation trends are useful in determining impacts of sea level on wetlands and the ecosystem services that they provide. Local rates of relative sea-level rise of 3.14 to 3.20 mm/yr have been observed in the Delaware Bay, and they are projected to increase to as high as 10 mm/yr within the next 100 years (Gornitz and Lebedeff, 1987; Nikitina et al., 2000; IPCC, 2001; NOAA, 2014). Temporal variations in the rate of sea-level rise are one of the main controls on marsh development (Rampino and Sanders, 1981; Stevenson et al., 1986; Patrick and DeLaune, 1990; Ward et al., 1998). A marsh's ability to keep pace with sea-level rise is directly correlated with the rate of elevation change along the marsh surface. The rate of sea-level rise has to be slow enough so that appreciable quantities of mineral matter or organic materials can be deposited to build and maintain the level of the marsh to at least approximately the mean water level (Frey and Basan, 1985; McKee and Patrick, 1988; and Ward et al., 1998).

The equilibrium between natural sedimentation and sea level reflects how well a marsh will react to long-term and rapid shifts in the rate of sea-level change. If sea-level rise outpaces vertical sediment accretion, wetland loss will result where lateral migration is not possible, and the change in location of the fresh water/ salt water interface will cause shifts in plant community composition (Callaway et al., 1996; Burkett and Kusler, 2000; Winter, 2000; Darke and Megonigal, 2003). On a local scale, anthropogenic activities such as development of coastal areas, reservoirs, water-withdrawal projects, and dams can affect the complex interactions of sediment deposition, erosion, and subsidence that control wetland surface elevation (Nittrouer et al., 1984; Davis, 1997; Darke and Megonigal, 2003).

In order to better understand the relationship between marsh development processes and control factors, such as sea-level rise, the Delaware Department of Natural Resources and Environmental Control (DNREC) deployed surface elevation table (SET) networks along the St. Jones River and Blackbird Creek. Blackbird Creek is a relatively un-impacted watershed while the St. Jones River is a heavily impacted mixed urban, suburban, and agricultural watershed. Both of these locations are components of the Delaware National Estuarine Research Reserve (DNERR). The SET networks monitor marsh surface elevation, subsidence, and sedimentation trends. They extend across the spectrum of coastal marsh environments and salinity (i.e., fringing, bay mouth bar, valley-fill, polyhaline, mesohaline, oligohaline, and freshwater [transitional]) with an approximate equal spacing.

4.2 Location

The DNREC fixed SET monitoring sites in this study are located along the St. Jones River (Central Kent County) and Blackbird Creek (Southern New Castle County) within the Delaware River and Bay Estuary (Figures 4.1, 4.2 and 4.3). The St. Jones River and Blackbird Creek are microtidal with tidal ranges at their mouths of 1.57 and 1.75 m respectively, and 0.58 (at Isaac's Branch along the St. Jones River) and 1.01 m (at Blackbird Landing along Blackbird Creek; Figures 4.2 and 4.3). The St. Jones River is seasonally polyhaline (salinity ranging between 18 and 30 ppt). The more northern Blackbird Creek can be polyhaline in the fall with spring salinity values of a more mesohaline (salinity ranging between 5 and 18 ppt) range. Both of their salinity values are heavily discharge dependent.

4.3 Methods

A surface elevation table (SET) is a counter balanced arm that has nine measuring pins, adjustable in their length, that are located at one of the distal ends (Cahoon et al., 2002 a and b). The arm sits on a collar and pivot joint, and can be rotated 360º (Figure 4.4). When attached to a benchmark platform or rod, it provides a constant reference plane from which the distance to the sediment surface can be measured. SETs provide a nondestructive method for making highly accurate and precise measurements of sediment elevation in intertidal and subtidal wetlands over long periods of time, relative to a fixed

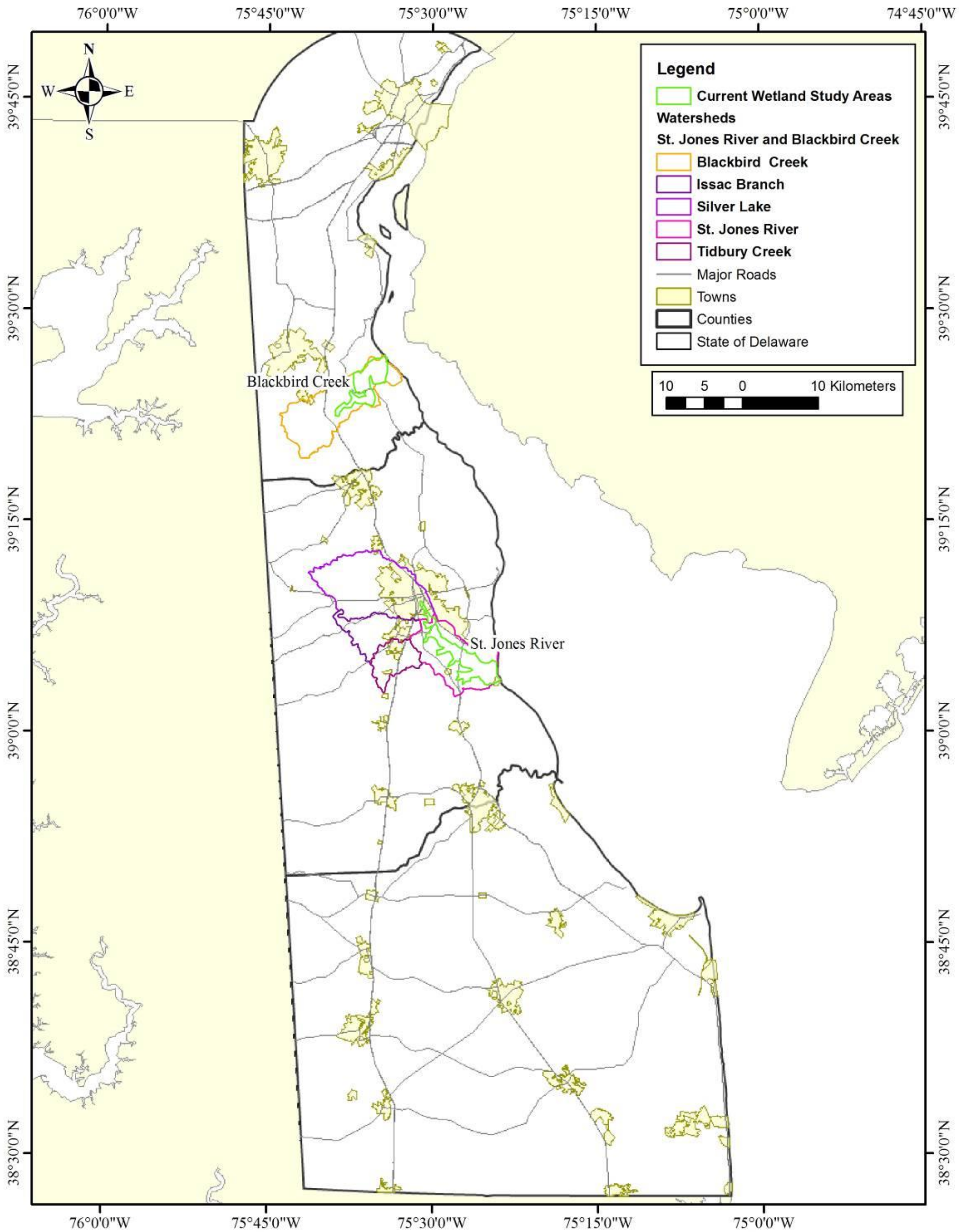


Figure 4.1 Location of the St. Jones River and Blackbird Creek watersheds (highlighted by separate colors) and current wetland study areas (highlighted in green).

Figure 4.2 Map showing the location of the four SET monitoring sites within the St. Jones River sub-estuary.

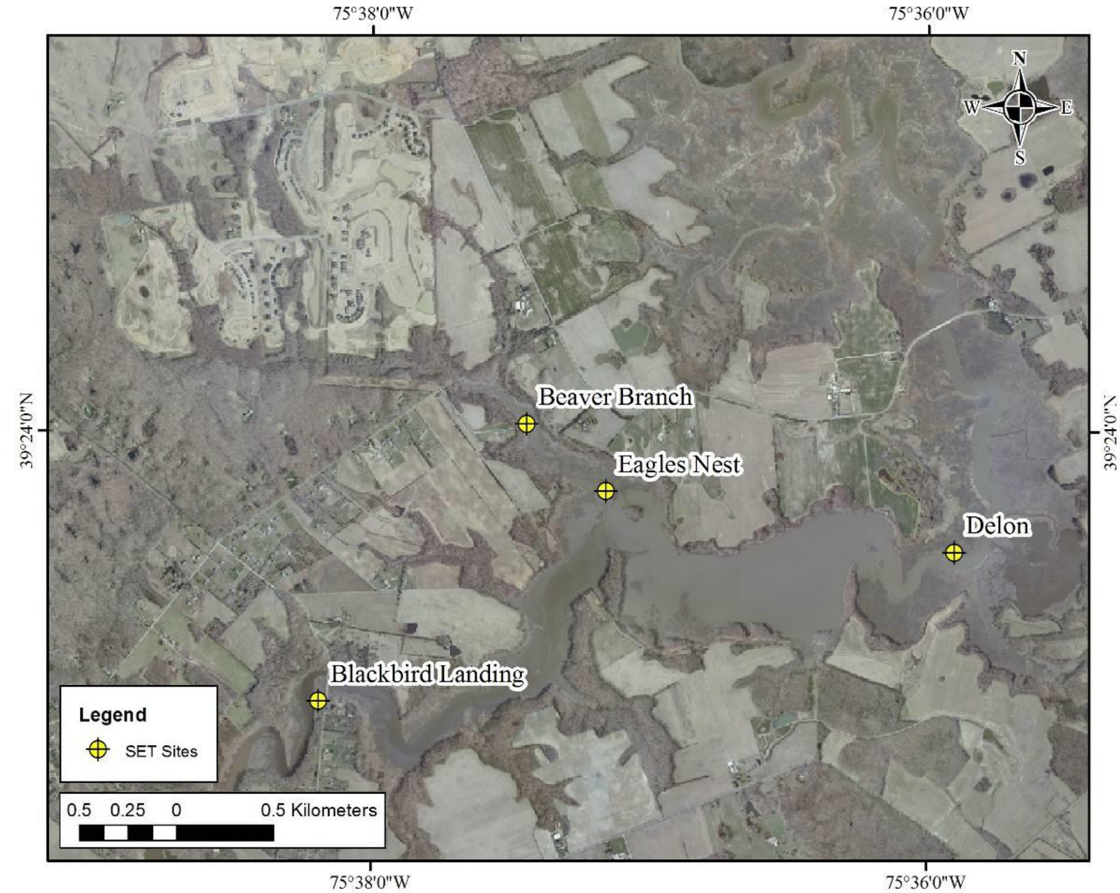


Figure 4.3 Map showing the locations of the four SET monitoring sites within the Blackbird Creek sub-estuary.

subsurface datum (Cahoon et al., 2002 a and b). They are typically sampled at low tide (at the same relative period in the lunar tidal cycle) every three to twelve months to ensure consistency of data collection. Spot sampling before and after major storm events may also occur. Thus, they can be used to determine both the influence of a single meteorological event on, and the long-term trends (i.e. decades) of, sediment surface elevation. Accurate measurements of surface elevation in wetlands over time are necessary to determine rates of change. SET measurements can be used to constrain changes in marsh surface elevations with respect to sea-level rise.

This study focuses on utilizing pre-established SETs on the St. Jones River (four sites) and on the Blackbird Creek (four sites; Figures 4.2 and 4.3). These SETs have recorded data spanning between 4.25 and 7.35 years. The data include deep (total) surface elevation change, shallow surface elevation change, accretion, shallow subsidence, and deep subsidence. At each of the SET sites, deep benchmarks were installed by driving 15 mm (9/16 inch) diameter rods into the marsh sediments to a point of refusal (7.0 to 25.0 m; Figure 4.5A and B). Shallow benchmarks were also installed consisting of four legged platforms with legs extending 35 to 50 cm into the marsh surface (Figure 4.6A and B). The marsh surface elevations (relative to the deep and shallow benchmarks) were recorded at four set angles (90º intervals) on the collar (Figure 4.7).

To measure accretion, G200 feldspar marker horizons were laid down in three plots (areas of 0.50 x 0.50 m) around each benchmark (Figure 4.8). New clay markers to

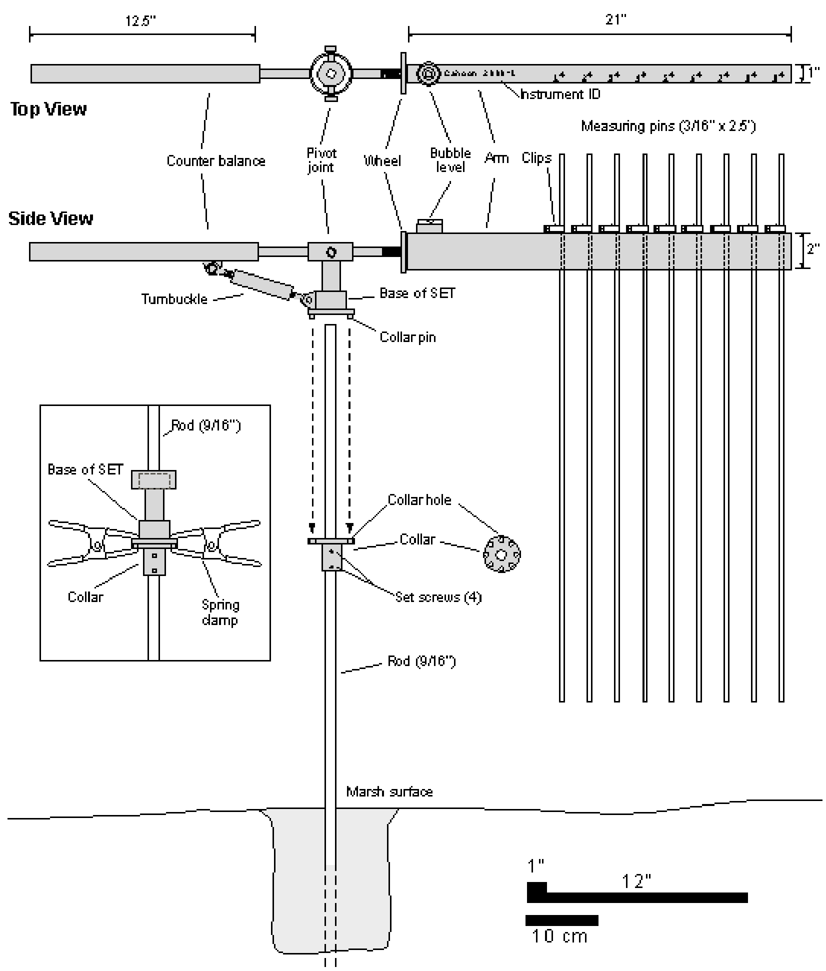
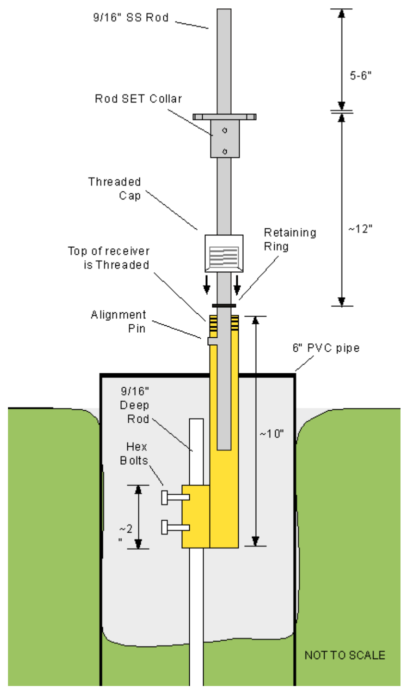
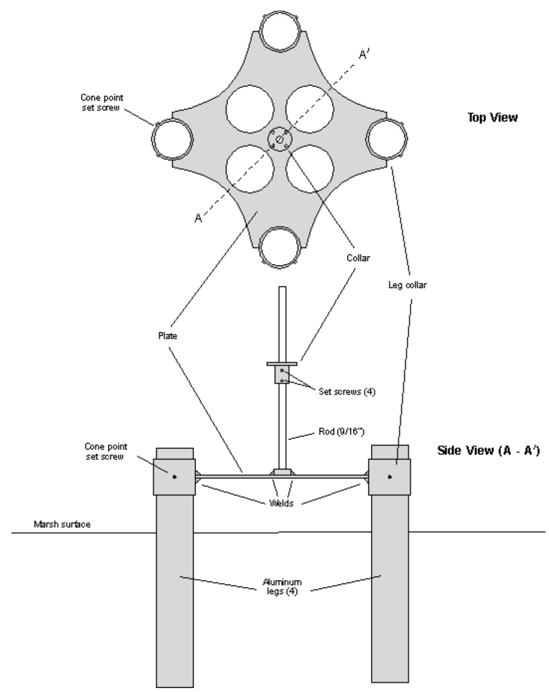


Figure 4.4 Schematic of a SET measurement arm and coupling design (figure courtesy of USGS SET website; Cahoon et al., 2002a and b).

A. 

B. 

Figure 4.5 A) Schematic of an installed survey rod coupled with a deep SET collar (figure courtesy of USGS SET website; Cahoon et al., 2002a and b). B) Photo of upstream St. Jones deep SET coupling and collar monument cemented in place with benchmark.



A.

B.

Figure 4.6 A) Shallow SET platform schematic (figure courtesy of USGS SET website; Cahoon et al., 2002a and b). B) Photo of installed Eagles Nest (Blackbird Creek) shallow SET platform.

A.

B.

Figure 4.7 A) Photo of deep SET with measurement arm connected and pins placed on marsh surface. Pin distance above arm mirrors the brackish marsh's hummocky surface (Eagles Nest deep SET site, Blackbird Creek). B) Photo of pin placement on marsh surface. Pins are clipped into place after adjustment.

replace ones that were diffused due to extensive bioturbation or that became too deep to be sampled effectively were re-deployed as necessary. Sediment plugs were randomly cut from each of the plots and the amount of sediment deposited on top of the marker horizon was measured (Figure 4.8). The sediment deposited was used to calculate the sedimentation rate (rate of vertical accretion) for that localized area, and the mean rate was calculated for each sampling station (benchmark area).

When used simultaneously, SET and marker horizon techniques provide information on below-ground processes that influence elevation change. Differences between rates of vertical accretion and elevation change in the shallow benchmark SET measurements may be attributed to processes, such as dewatering, enhanced root growth, decomposition and root growth collapse, occurring below the feldspar layer and above the bottom of the shallow SET pipe.

4.4 Results

Elevation changes measured by a SET are influenced by both surface and subsurface processes occurring within the soil profile (Figure 4.9). Separating the influences of the biological (root growth), geological (soil compaction) and hydrological processes (groundwater storage) can be very difficult. The shallow benchmark measures elevation changes due to factors occurring in the root zone (from the surface to ~30 to 50 cm below the sediment surface), such as enhanced growth due to nutrient fluxes, decomposition, shallow compaction, dewatering, and pore water fluxes. The root zone extends from the marsh surface to the bottom of the active root growth. The zone of

Figure 4.8 G200 Feldspar clay plot marker horizons are laid in 50 cm x 50 cm squares, marked by stakes on either side. A clay plug is taken by cryogenic corer (shown to right). Plug shows the total sediment that has been deposited after the clay layer was laid.

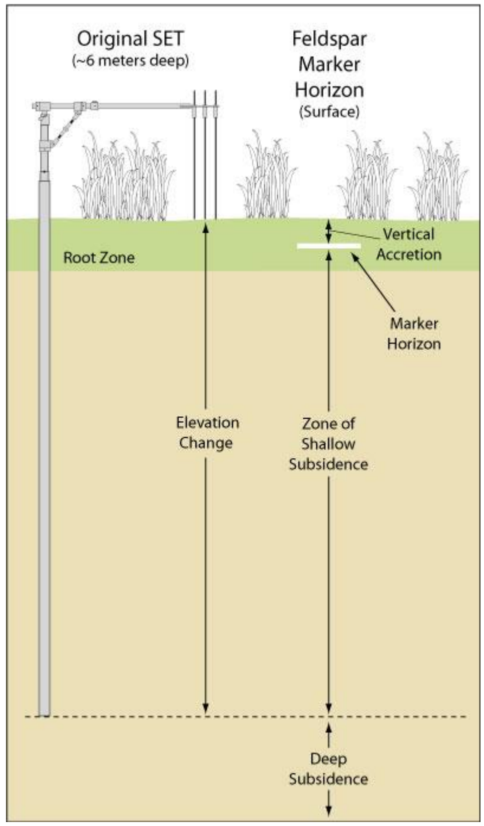


Figure 4.9 Schematic illustrating the areas of sub-surface processes that are studied by a shallow and a deep SET's zone of vertical coverage (figure courtesy of USGS SET website; Cahoon et al., 2002a and b). In this study, the zone of shallow subsidence is equal to the length of the shallow SET's leg (~70 cm). The deep SET's zone of deep subsidence was equal to the variable depth of the surveying rods penetration (~7.0 to 25.0 m).

shallow subsidence extends from the surface of the marsh to the bottom of the shallow benchmark. Measurements from the deep benchmark document the effect of subsidence and compaction (e.g., the consolidation and dewatering of the fine grained material deep in the marsh sediments) on the marsh surface elevation. The zone of deep subsidence extends from the bottom of the zone of shallow subsidence to the contact with the underlying estuarine deposits.

4.4.1 Surface Elevation Change – Deep Benchmark Measurements

All four SETs along the St. Jones River experienced an overall loss in marsh surface elevation as measured by the deep benchmarks (Table 4.1 and Figure 4.10). The elevation decreases range from -47.5 mm at the Impoundment SET to -23.7 mm at the Wildcat SET (Table 4.1 and Figure 4.10). The rate of overall elevation loss ranges from -11.1 to -4.4 mm/yr at the Impoundment and Boardwalk SETs, respectively (Table 4.1).

Over the time period sampled, the elevation changes measured by the SETs along the St. Jones River follow a similar general pattern. Initially, surface elevations are relatively constant, if not slightly increasing (Figure 4.10). Then, after October, 2008 at the Impoundment and Issac's Branch SETs and April, 2009 at the Boardwalk and Wildcat SETs, elevations begin to rapidly decrease, followed after October-November, 2009 at the Boardwalk, Impoundment, and Issac's Branch SETs, and after April, 2010 at the Wildcat SET, by a more gradual decrease in surface elevation (Figure 4.10).

Table 4.1 Ending deep and shallow elevations, deep and shallow elevation change, accretion rate, thickness of the Holocene sediments, deep or total subsidence rate and shallow subsidence rate for all eight SET sites.

	Ending Deep Surface Elevation (mm ± s.d.)	Ending Shallow Surface Elevation (mm ± s.d.)	Mean rate of Deep Elevation Change (mm/yr ± s.d.)	Mean Rate of Shallow Elevation Change (mm/yr ± s.d.)	Accretion Rate (mm/yr ± s.d.)	Thickness of Holocene Sediment (m)	Deep or Total Subsidence Rate (mm/yr)	Shallow Sub-sidence Rate (mm/yr)
St. Jones River								
Impoundment	-47.54 (17.11)	-88.00 (17.84)	-11.12 (4.0)	-28.56 (5.79)	37.27 (15.25)	24.30	48.38	65.83
Boardwalk	-31.64 (10.10)	-33.10 (6.81)	-4.38 (1.4)	-5.69 (1.17)	5.26 (1.81)	7.40	9.64	10.95
Wildcat	-23.68 (42.90)	-24.72 (23.74)	-5.58 (10.10)	-8.71 (8.37)	19.99 (15.67)	8.70	25.57	28.70
Isaac's Branch	-40.17 (20.24)	-43.42 (11.10)	-9.47 (4.77)	-15.62 (3.99)	20.44 (10.9)	8.50	29.91	29.34
Blackbird Creek								
Delon	-45.80 (37.8)	-49.40 (24.02)	-6.23 (5.14)	-11.89 (5.78)	22.26 (8.11)	7.30	28.49	34.15
Eagles Nest	14.93 (45.48)	21.15 (41.89)	2.14 (6.5)	4.81 (9.52)	16.59 (2.61)	12.20	14.45	11.78
Beaver Branch	8.63 (46.7)	59.46 (71.44)	1.27 (6.85)	14.87 (17.86)	16.85 (10.47)	11.20	15.58	1.98
Blackbird Landing	-72.83 (67.34)	-0.11 (12.53)	-10.42 (9.63)	-0.03 (2.87)	11.80 (3.47)	9.80	22.22	11.83

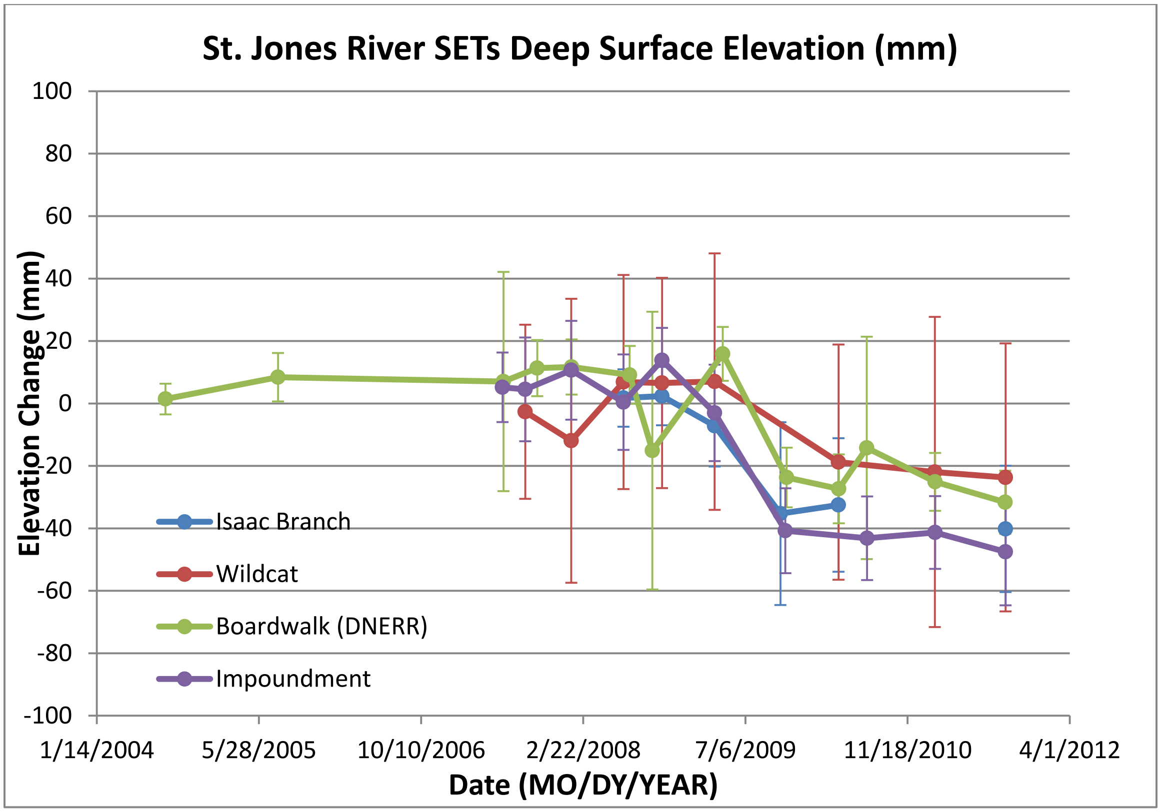


Figure 4.10 St. Jones River SETs surface elevation change as measured by deep benchmarks. Standard deviations of the mean elevation change for each site's readings are shown in the corresponding color.

Only two of the four SETs on Blackbird Creek record an overall decrease in marsh surface elevation as measured by the deep benchmarks (Table 4.1 and Figure 4.11). At the Blackbird Landing SET, a large overall decrease of -72.4 mm in elevation is observed; a decrease of -45.8 mm is measured at the Delon SET (Table 4.1 and Figure 4.11). The rates of elevation loss at these SETs (-10.4 and -6.2 mm/yr, respectively) are of similar magnitude to the St. Jones River SETs (Table 4.1). In contrast, at the Eagles Nest and Beaver Branch SETs overall increases in elevation of 14.9 and 8.6 mm, respectively are measured (Table 4.1 and Figure 4.11). The rates of surface elevation increase are low, 2.14 and 1.27 mm/yr, respectively, for the Eagles Nest and Beaver Branch SETs when compared to the rates of elevation change for the SETs that measure elevation decreases (Table 4.1).

The Blackbird Creek SETs did not have a common pattern of elevation change over the sampling period (Figure 4.11). The Eagles Nest and Blackbird Landing SETs until October-November, 2008 are characterized by overall small increases in elevation (9.9 and 5.8 mm, respectively), similar to that at the St. Jones River SETs (Figures 4.10 and 4.11). However, the Delon SET until September, 2008 has a much larger overall increase in elevation (33.5 mm), and the Beaver Branch SET until April, 2009 is characterized by a large elevation decrease (-103.0 mm; Figure 4.11). After September-November, 2008, elevations begin to decrease at the Eagles Nest, Blackbird Landing, and Delon SETs (Figure 4.11). Elevation decreases continue at the Blackbird Landing and Delon SETs. However, the Eagles Nest and Beaver Branch SETs between April and

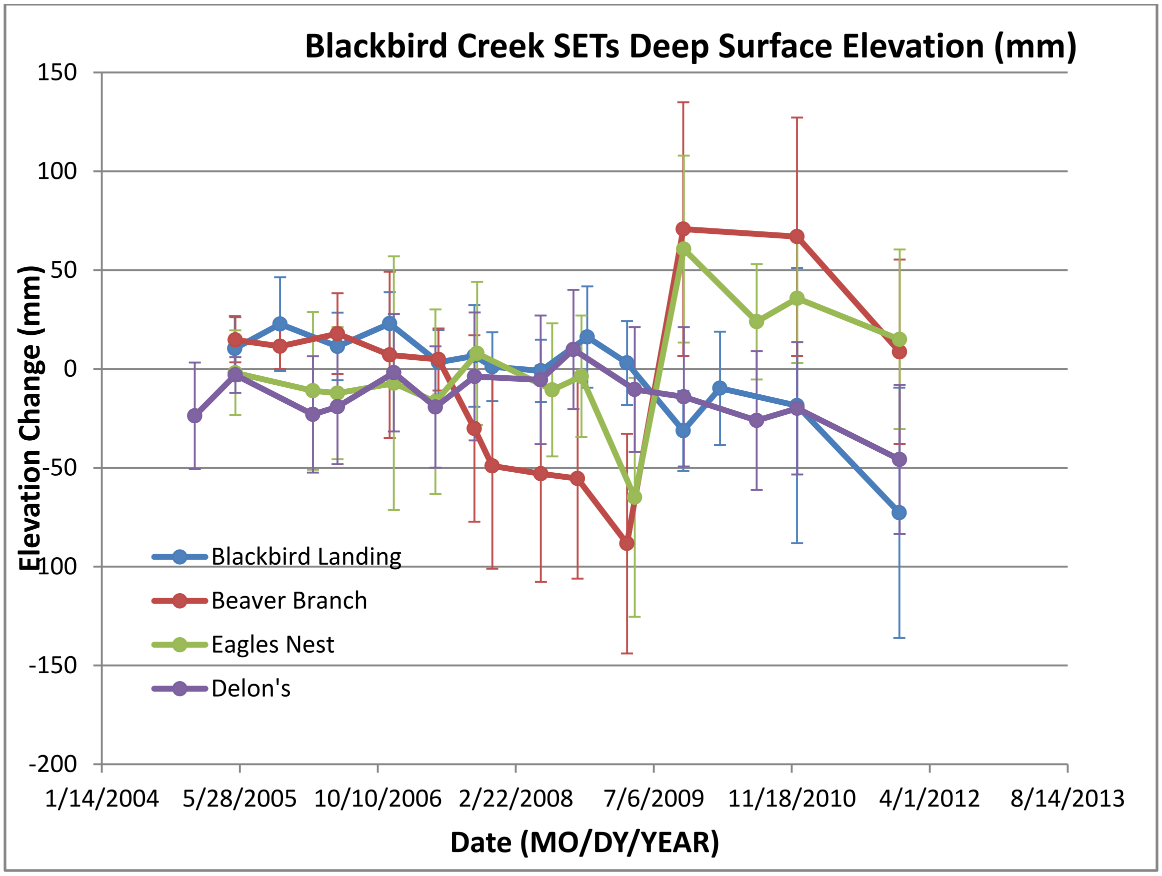


Figure 4.11 Blackbird Creek SETs surface elevation change as measured by deep benchmarks. Standard deviations of the mean elevation change for each site's readings are shown in the corresponding color.

October, 2009 experience large increases in surface elevation (125.6 and 159.1 mm, respectively), followed by lesser overall decreases in elevation (Figure 4.11).

As shown by the relatively large standard deviations for the elevation changes in both the deep and shallow benchmark measurements (Figures 4.10 – 4.13), there is a level of uncertainty in the data that reduces the extent to which changes in marsh elevation can be evaluated. The variability in the data at individual SET sites can in part be attributed to extensive bioturbation at the sites generated by burrowing and mound creation by fiddler crabs. Fiddler crabs were ubiquitous at most of the sampling sites, and produced local changes in elevation that were spatially distributed at a scale that corresponded to the spacing between the sampling pins of the SET instruments.

4.4.2 Surface Elevation Change - Shallow Benchmark Measurement

The length of time over which the shallow elevation change is measured is shorter than the deep SETs, as the shallow SET platforms were re-leveled on the St. Jones River in February 2011 and in December 2010 on Blackbird Creek. Initial baseline readings were re-started on the subsequent readings in 2011.

Along the St. Jones River, and similar to that observed for the deep benchmarks, the shallow surface elevation measurements are characterized by an initial gradual elevation increase, followed by a relatively large and rapid decrease (Figure 4.12). Overall initial elevation increases range between 20.9 mm at the Impoundment SET and 7.8 mm at the Wildcat SET (Figure 4.12). The rates at which elevations increase between sampling periods varies between 4.2 mm/yr at the Boardwalk SET and 18.1 mm/yr at the Impoundment SET. Starting after October, 2008 at the Impoundment, Wildcat, and Issac's Branch SETs, and after April, 2009 at the Boardwalk SET, shallow elevations begin to decrease at a rapid rate. Decreases ranging from -50.0 mm at the Issac's Branch SET to -129.2 mm at the Impoundment SET are measured (Figure 4.12). The rates at which surface elevations decrease between sampling periods varies between -18.9 mm/yr at the Wildcat SET and -74.6 mm/yr at the Impoundment SET. At the Boardwalk SET after November, 2009, the rate slows to -7.5 mm/yr.

Along the Blackbird Creek, only the Delon SET has a similar general pattern of an initial gradual elevation increase followed by a large and rapid decrease as observed at the St. Jones River (Figure 4.13). At Delon, the overall elevation increase is 15.7 mm at a rate of 8.8 mm/yr. The decrease in elevation starts after September, 2008, with a rapid rate of decrease (-153.5 mm/yr) occurring after April, 2009 (Figure 4.13). Similar to the St. Jones River Boardwalk SET, the rate of decrease slows after October, 2009 to -3.9 mm/yr.

At the Eagles Nest and Beaver Branch SETs, an opposite pattern to that at the Delon and the St. Jones River SETs is observed. Elevations have initial decreases reaching -40.6 mm and -63.4 mm in April, 2009, followed by rapid increases in elevation (71.0 mm and 128 mm) between April and October, 2009 (Figure 4.13). At the Eagles Nest SET, a subsequent decrease in elevation of -12.1 mm (rate of -16.7 mm/yr) is measured in July, 2010 (Figure 4.13). The Beaver Branch SET was not sampled for

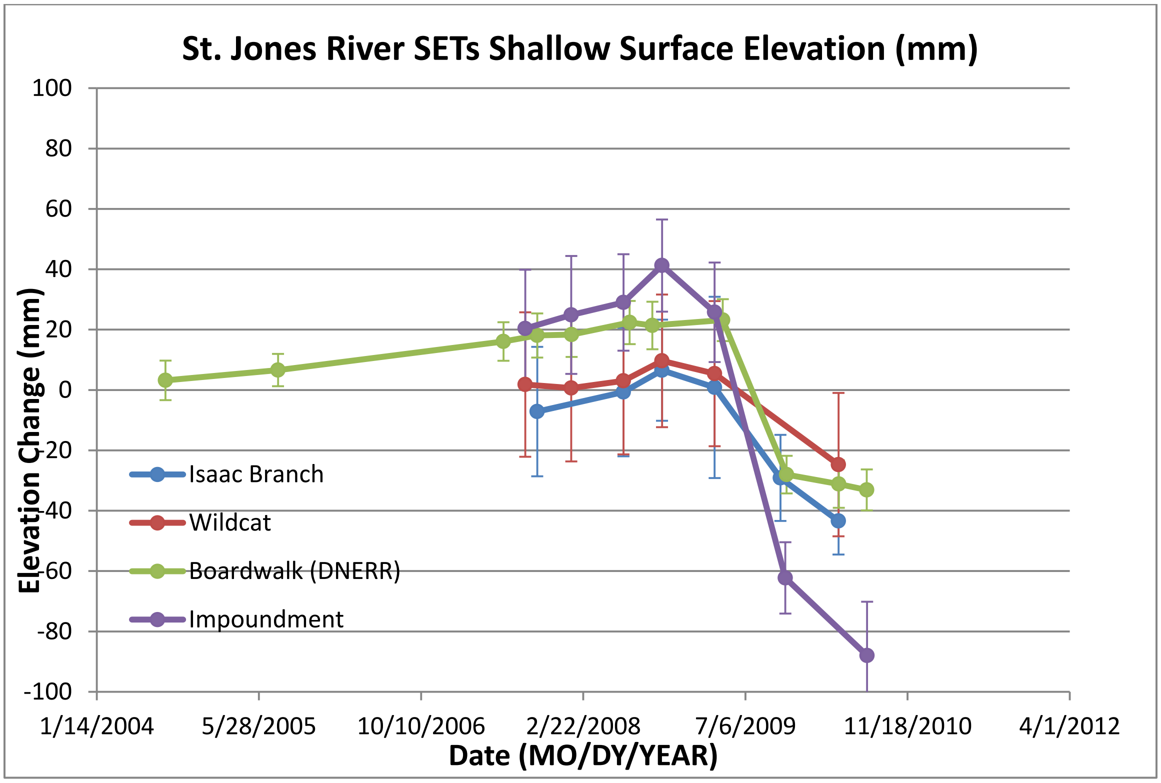


Figure 4.12 St. Jones River SETs surface elevation change as measured by shallow benchmarks. Standard deviations of the mean shallow elevation change for each site's readings are shown in the corresponding color.

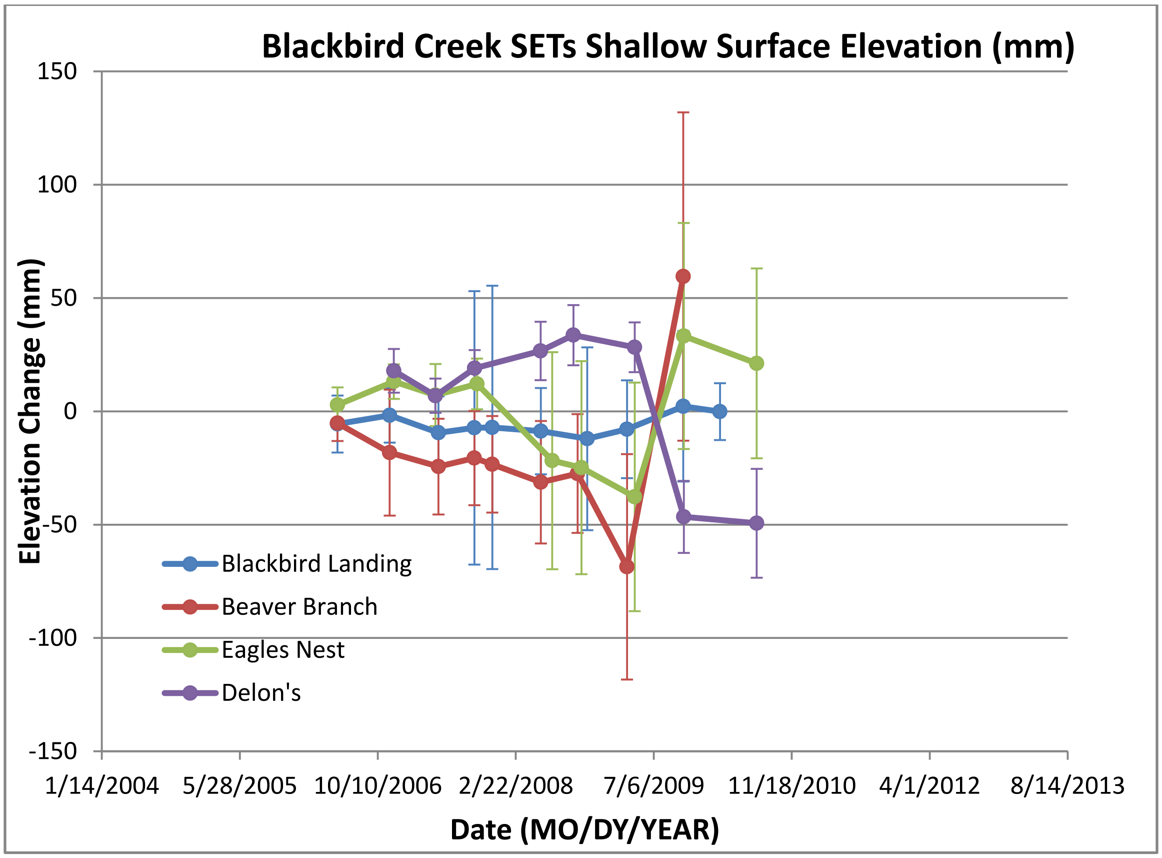


Figure 4.13 Blackbird Creek SETs surface elevation change as measured by shallow benchmarks. Standard deviations of the mean shallow elevation change for each site's readings are shown in the corresponding color.

shallow elevation change after October, 2009. The Blackbird Landing SET recorded little change in elevation between sampling periods. The average rate of absolute elevation change is only 8.5 mm/yr. An overall increase in elevation of only 5.5 mm is measured at this SET (Figure 4.13).

4.4.3 Accretion Rates

In a comparison of the data, the St. Jones River SETs have the greatest differences in accretion rates both in magnitude and degree of variability (Figures 4.14 and 4.15). The St. Jones River Boardwalk SET, with an average rate of 5.3 +/- 1.8 mm/yr and a minimum rate of 2.49 mm/yr, have the lowest and least variable accretion rates. Accretion rates at the St. Jones River Impoundment SET are the highest measured with an average rate of 37.3 +/- 15.2 mm/yr, and a maximum rate of 66.4 mm/yr. The St. Jones River Wildcat and Issac's Branch SETs have similar accretion rate patterns with initially high rates of accretion (51.6 mm/yr (August, 2007) and 43.8 mm/yr (October, 2007), respectively), that rapidly decrease (to 12.2 mm/yr (June, 2008) and 19.0 mm/yr (October, 2008), respectively), then remain relatively low over the remainder of the sampling interval (Figure 4.14). The average accretion rates at the Wildcat and Issac's Branch SETs are 20.0 +/- 15.6 mm/yr and 22.9 +/- 11.5 mm/yr, respectively.

Accretion rates at the Blackbird Creek SETs are much more consistent in terms of magnitude and variability than that along the St. Jones River. With the exception of only two sampling periods, one at the Delon SET and one at the Beaver Branch SET, accretion rates are between 26.7 and 7.5 mm/yr (Figure 4.15). The Delon SET has the highest

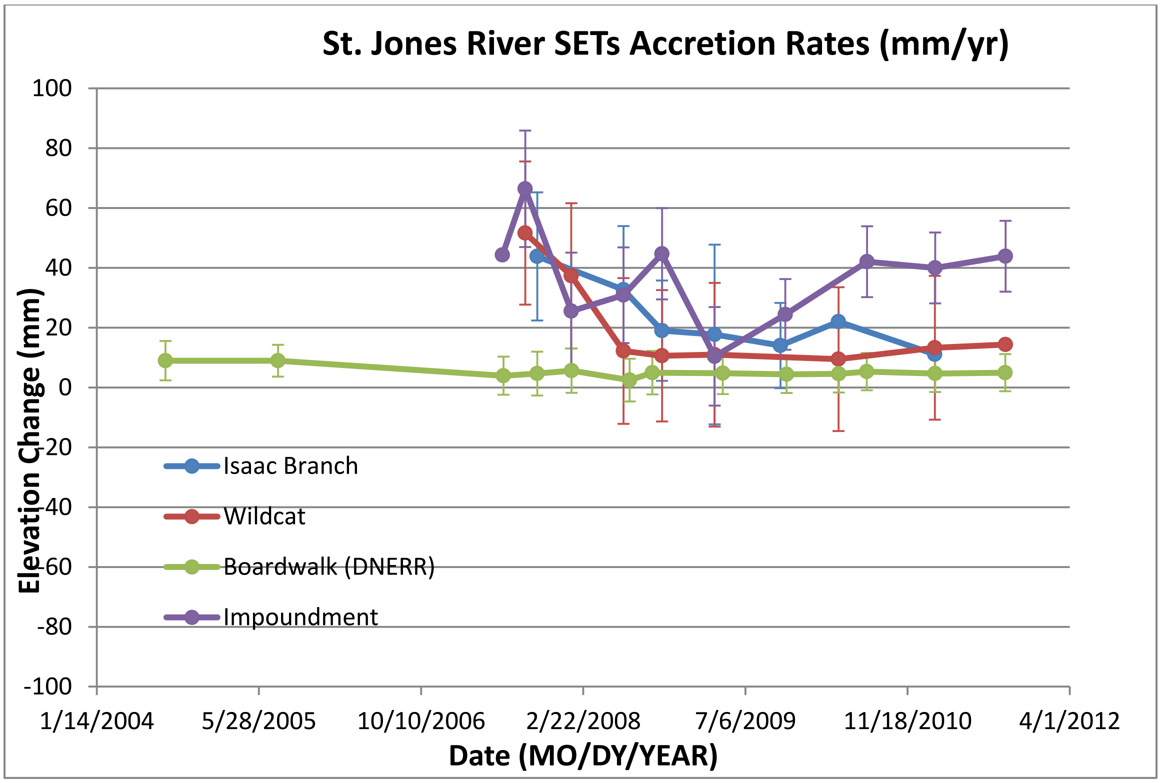


Figure 4.14 St. Jones River SETs accretion rates. Standard deviations of the mean accretion rate for each site's readings are shown in the corresponding color.

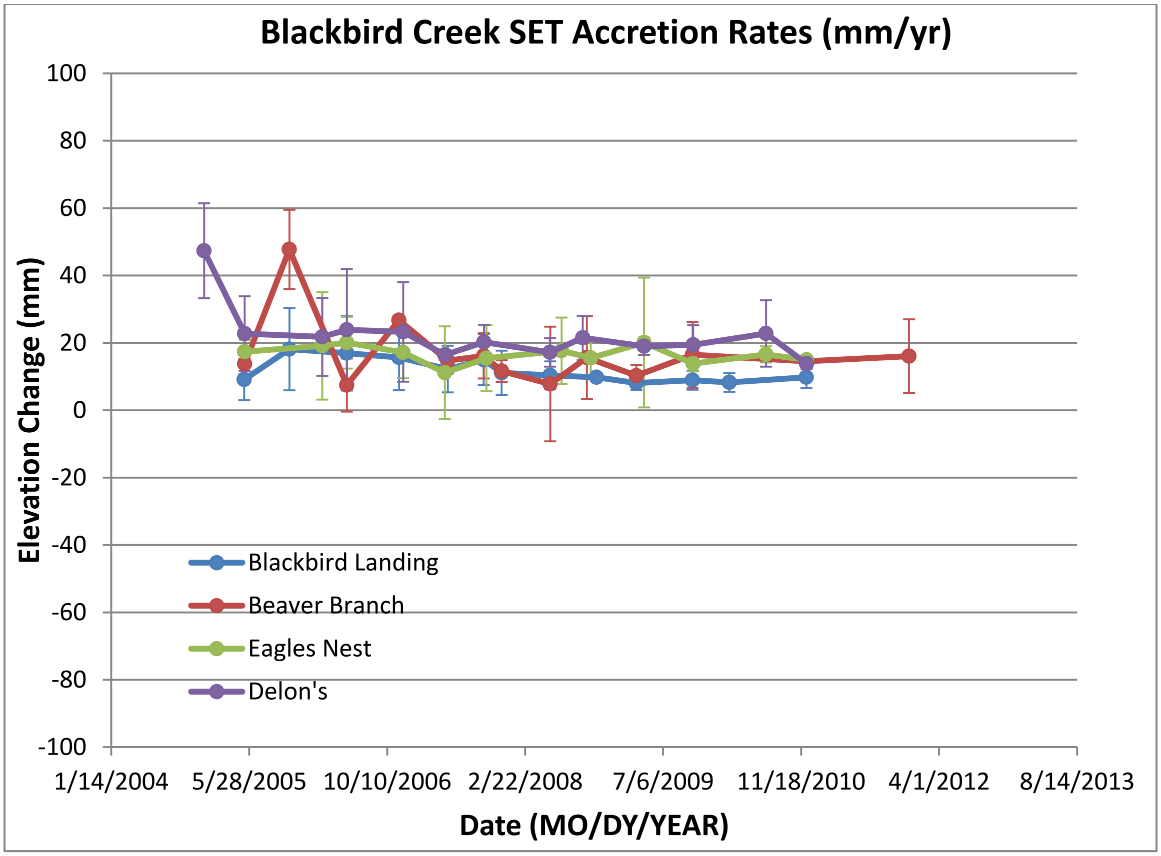


Figure 4.15 Blackbird Creek SETs accretion rates. Standard deviations of the mean accretion rate for each site's readings are shown in the corresponding color.

average accretion rate of 22.3 +/- 8.1 mm/yr, and the Blackbird Landing has the lowest average rate of 11.8 +/- 3.5 mm/yr (Table 4.1). The Eagles Nest and Beaver Branch SETs have accretion rates that are comparable (16.6 +/-2.6 mm/yr and 16.8 +/-10.4 mm/yr, respectively).

Accretion rates decrease upstream along both the St. Jones River and Blackbird Creek. The two SETs closest to the Delaware Bay, the Impoundment SET (37.3 +/- 15.2 mm/yr) and the Delon SET (22.3 +/- 8.1 mm/yr) have the highest average accretion rates for the St. Jones River and Blackbird Creek, respectively. Lower average rates of accretion are measured at the two SETs, Wildcat (20.0 +/- 15.6 mm/yr) and Isaac's Branch (22.9 +/- 11.5 mm/yr), that are the furthest upstream along the St. Jones River. Similarly, the lowest average rate of accretion (11.8 +/- 3.5 mm/yr) occur at the Blackbird Landing SET, furthest upstream along Blackbird Creek. A low average rate of accretion is measured at the Boardwalk SET (5.3 +/- 1.8 mm/yr) which is the nearest SET to the main stem of the St. Jones River (Table 4.1).

Lower accretion rates upstream and near the main stem of the St. Jones River are expected within these tidal wetlands (Reed, 1995; Christiansen, 1998; Chmura and Hung, 2004; Li, 2006). Sites closer to the bay would be exposed to higher concentrations of suspended sediments due to closer proximity to the sediment-rich Delaware Bay waters, and the re-cannibalization of marsh (bay and channel) edges during storm events. Sites further upstream would experience lower rates of accretion due to reduced sediment concentrations in the tidal waters as a result, in part, due to the longer flow path up the main stem and any tributaries.

Trends that were expected, but not clearly observed, are seasonal variations in accretion rates. A seasonal pattern of deposition due to higher stem and leaf densities during the summer, and lower stem densities and above-ground biomass during winter months (due to vegetation senescence), should result in a pattern of higher rates of accretion during the summer and lower rates during winter. Of the eight accretion records, only the Impoundment Site shows distinctly higher rates of accretion (during or right after summer months) in 2007, 2008, and 2010 (Figure 4.14). Two of the Blackbird Creek sites (Eagles Nest and Beaver Branch), show slightly higher rates during the summer, but the rates are not significantly different, and the trends are not consistent over the time period that data were collected (Figure 4.15). The lack of a clear seasonal accretion rate trend in the data is likely the result of allowing too much time to pass between sampling intervals, such that the effects of vegetation vigor on sediments trapping could not be constrained, and the effects of episodic coastal storms could not be excluded. If the sites had been sampled bi-monthly or even quarterly, then the seasonal effects could have been better constrained. Sampling before and after large coastal storms with significant storm surges (i.e. Nor'easters) would also be necessary to account or exclude the effects of episodic accretion events. It also cannot be discounted that the placement of a SET within the estuary or marsh may have an effect on the magnitude of the accretion. For example, if positioned along the fringe of the marsh (i.e. Boardwalk and Blackbird SET locations) where there are lower suspended sediment rates, the overall accretion rates would be lower and the degree of seasonal variability would be more difficult to measure. In contrast, sites (i.e. Impoundment SET) located closer to Delaware

Bay (the sediment source for these estuaries), and/or closer to the main stem of the estuaries, would experience higher suspended sediment concentrations, and with a larger signal would have an easier to detect seasonal trend.

The effect of re-suspension and re-deposition of mud by fiddler crab burrowing activities could be a factor in the rates of accretion that were measured. Areas of highest burrow intensity would most likely be associated with higher measured accretion rates, than those determined for non-altered sites. However, it was beyond the scope of this study to quantify the intensity of burrowing activity and volume of sediments brought to the surface by the activity.

4.4.4 Deep and Shallow Subsidence Rates

Deep and shallow rates of subsidence at a SET can be calculated by subtracting the accretion rate from the rate of elevation change. Deep subsidence rates are equivalent to the rate of elevation change as determined from the deep SET measurements minus the accretion rate; the shallow subsidence rate is equivalent to the rate of elevation change as determined from the shallow SET measurements minus the accretion rate. Deep and shallow subsidence rates for the St. Jones River and Blackbird Creek SETs are listed in Table 4.1.

Deep subsidence rates for the St. Jones River SETs vary between a maximum of 48.4 and a minimum of 9.6 mm/yr for the Impoundment and Boardwalk SETs, respectively. The mean deep subsidence rate for the St. Jones River SETs is 28.4 +/- 15.9 mm/yr. The deep subsidence rates for the Blackbird Creek SETs are less variable and have a lower mean rate when compared to the St. Jones River data. Deep subsidence rates vary between a maximum of 28.5 and a minimum of 14.5 mm/yr at the Delon and Eagles Nest SETs, respectively. The mean deep subsidence rate for the Blackbird Creek SETs is 20.2 +/- 6.5 mm/yr.

Shallow subsidence rates are also less variable and have a lower mean rate for the Blackbird Creek SETs when compared to the St. Jones River SETs. Shallow subsidence rates at the Blackbird Creek SETs vary between a maximum of 34.2 and a minimum of 2.0 mm/yr at the Delon and Beaver Branch SETs, respectively, with a mean shallow subsidence rate of 15.0 +/- 13.6 mm/yr. At the St. Jones River SETs, maximum and minimum shallow subsidence rates are 65.8 and 11.0 mm/yr at the Impoundment and Boardwalk SETs, respectively. The mean rate of shallow subsidence at the St. Jones River SETs is 35.4 +/- 22.9 mm/yr.

The rate of shallow subsidence can either amplify or ameliorate the processes that are resulting in deep subsidence. Shallow subsidence rates are a function of the consolidation or de-watering occurring within the root zone (within ~40 cm of the surface where the benchmark is located). Sites with a healthy vegetative community will have a high volume of biomass in the shallow zone, and will have a rate of subsidence that is either very low, at zero, or even negative (i.e. an elevation increase). This can be seen at the Blackbird Creek Beaver Branch SET with a shallow subsidence rate of only 2.0 mm/yr (Table 4.1). Sites with stressed plant communities and potentially higher rates of decomposition will experience rates of shallow elevation decrease that are large, and amplify the effects of deep subsidence processes. This is observed at the St. Jones River Impoundment SET with a high shallow subsidence rate of 65.8 mm/yr (Table 4.1).

Deep subsidence rates are a function of the degree of compaction, decomposition, and consolidation (de-watering) of the sediments, including those in the shallow zone, that underlie a SET. Thus, the rate of deep subsidence should differ between SETs due to variations in the composition (as it impacts compaction, decomposition, and consolidation properties) and the thickness of the Holocene sediments that underlie the SETs.

The Holocene sediments that underlie the SETs, in both the St. Jones River and Blackbird Creek, are mainly composed of fine grained silts and clays with varying quantities of organic content (Marx, 1981; Oertel and Kraft, 1994; Wilson, 2004). They can range in thickness from centimeters (at the fringes of the estuary) to greater than 20 m (at the center of the basins, toward the bay; Marx, 1981; Oertel and Kraft, 1994; Wilson, 2004). These sediments are easily compacted due to the weight of the overburden; which causes pore fluids to be squeezed out, clay and silt particles to realign and compress, and organic deposits to compress. Compaction results in a loss of vertical volume, which lowers the ground surface above these deposits. The top ~2 m of the vegetated marsh deposits, are composed of higher quantities of organic content, and therefore experience high rates of shallow compaction (due to de-watering, compression, and decomposition; Pizzuto and Schwendt, 1997). Decomposition (aerobic and anaerobic) plays a huge role in organic loss in the upper deposits, while below ~2 m the rate of decomposition is reduced due to the lack of available oxygen (Pizzuto and Schwendt, 1997; Kirwan et al., 2012 and 2013).

Cores to determine the composition of the underlying material at each SET were not collected as part of this study. Holocene sediment thicknesses at each of the SETs were determined by the depth of penetration of the surveying rod, in conjunction with the sediment type (Pre-Holocene sand and gravel deposits) sampled at the point of refusal. The Impoundment SET, with a large Holocene sediment thickness (24.3 m), is an outlier (Table 4.1). The other SETs have thicknesses that range between 12.2 and 7.3 m (Table 4.1). Past research confirms the thicknesses of the Holocene sediments in the vicinity of the SET locations (Marx, 1981; Oertel and Kraft, 1994; Wilson, 2004).

It is expected that the greater the Holocene sediment thickness, the larger the degree of compaction and consolidation (de-watering), and thus the higher the rate of deep subsidence (Chrzastowski and Kraft, 1985; Wilson, 2004). However, this relationship is not supported by the SET data collected in this study. A plot of the rate of deep subsidence as a function of the thickness of Holocene sediments does not show a strong correlation (Figure 4.16). A best-fit line to the data has an R^2 value of only 0.4884. Clearly shown on this plot is the outlier nature of the Impoundment SET 24 m Holocene sediment thickness data point. What the data are lacking is deep subsidence rate values for sediment thicknesses between 12 and 24 m (Figure 4.16). To better constrain the relationship between thickness of underlying marsh sediments and the rate of subsidence, additional SETs would need to be located in the study areas at sites corresponding to Holocene sediment thicknesses between 12 and 24 m.

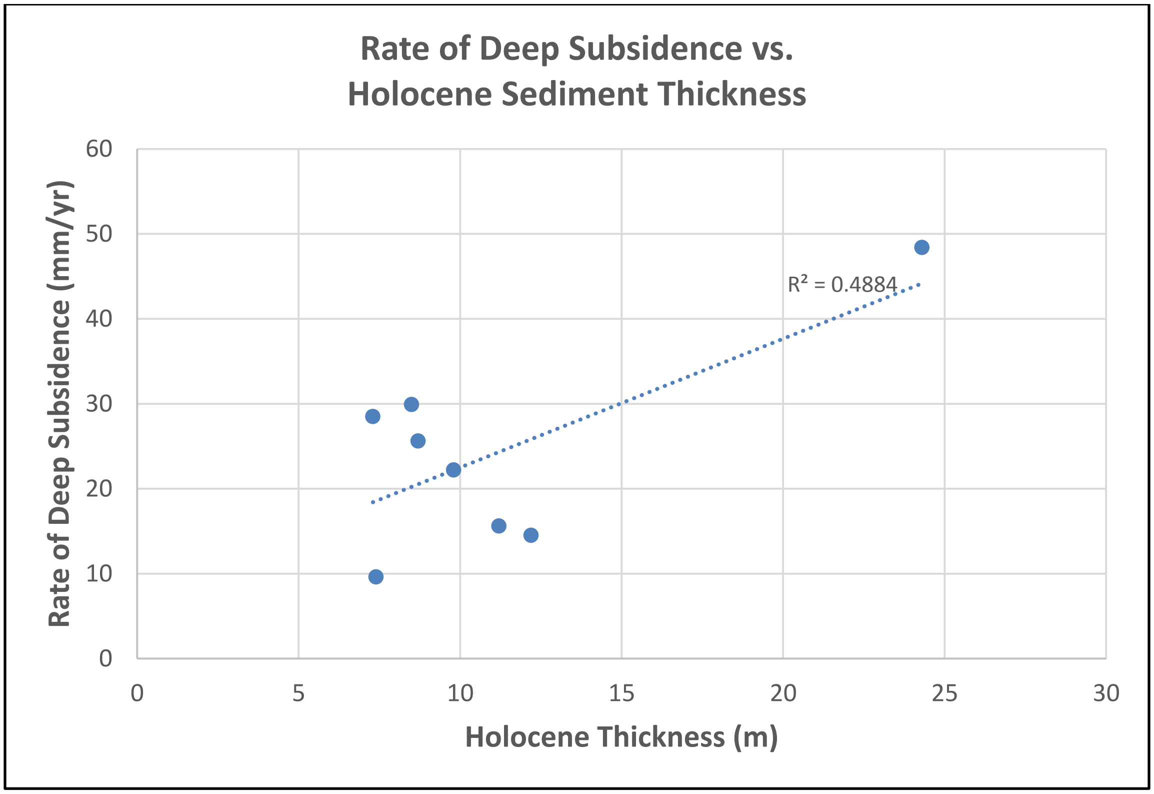


Figure 4.16 Relationship between Holocene sediment thickness and rate of deep subsidence for the eight SETs. Notice lack of data between sediment thicknesses of 12 and 24 m.

4.5 Discussion

The relationship between marsh surface elevation and tidal levels is one of the ultimate controls in dictating wetland flooding frequency, length of inundation, suspended sediment concentrations, and type and density of vegetation cover (Morris et al., 2002). A marsh's ability to pace sea-level rise is directly correlated with the rate of elevation change along the marsh surface, in relation to changes in tidal levels. Temporal variations in the rate of sea-level rise are one of the main controls on marsh development or destruction, as it pertains to wetland evolution outside of the effects of anthropogenic disturbances (i.e. ditching, channelizing, water level manipulations, etc.). Longer-term surface elevation and accretion records derived through fixed monitoring sites, such as SETs, are a means of monitoring the response of a marsh to seasonal, annual, and decadal changes in tidal levels. As with any marsh surface elevation study, it is important to note that marsh accretion rates may not necessarily always reflect changes in the elevation of the marsh surface. Subsurface processes such as decomposition and compaction can lower the elevation of the marsh surface, independent of the accretion rates, which may exceed the rate of sea-level rise (Cahoon and Reed, 1995).

Determining the factors that contribute to changes in marsh surface elevation is not easy. The eight SET sites in this study show a variety of trends that are consistent with both inter- and intra-site changes. Six SET sites (Impoundment, Boardwalk, Wildcat, Issac's Branch, Delon, and Blackbird Landing) show a precipitous decrease across two entirely different estuaries in their shallow and deep elevation records. A cause for this loss cannot be easily determined. Although changes within the vegetative community could be a factor, the vegetation in the six sites is not consistent. Two of the sites in the St. Jones (Impoundment and Boardwalk) are dominated by short-form *S. alterniflora*, while the other four sites (two in the St. Jones [Wildcat and Isaac's Branch], two in Blackbird Creek [Delon and Blackbird Landing]) are a mix of many species. These four sites show greater similarity and contain hummocks, with deep troughs transecting highly vegetative clumps, which are dominated by long-form *S. alterniflora, Peltandra virginica, Scurpus americanus and P. australis.*

Changes in the shallow elevation and vegetation can have several effects on the overall marsh surface elevation, resulting in decreased below-ground biomass production which can lower the marsh surface elevation. Variations in the rate of decomposition (both organic and inorganic) can also affect marsh surface elevation changes (Kirwan et al., 2012). Variations in decomposition rates likely result from changes in water levels as increased tidal levels shift decomposition toward more anaerobic conditions (Kirwan et al., 2012). Regular tidal inundation creates anaerobic conditions during flooding and then more aerobic conditions at the surface during periods of low tide. Increased tidal levels, through local sea-level rise, could result in a higher likelihood of un-interrupted anaerobic conditions.

The equilibrium between natural sedimentation (organic and inorganic) and sea level determines how a marsh will react to rapid and/or long-term changes in the rate of sea-level rise/fall (Reed, 1995). The primary productivity of salt marsh vegetation is regulated by changes in sea level (eustatic and/or land subsidence), sediment supply, and tidal range; the vegetation, in turn, constantly modifies the elevation of its habitat toward equilibrium with sea level (Stevenson et al., 1986; Reed, 1995). (Mendelssohn and Morris, 2000). Morris et al. (2002) suggest that every marsh has a theoretical optimum rate of relative sea-level rise (RSLR), an optimum depth at which the marsh community is most productive, and an equilibrium depth that can be greater or less than the optimum.

As sea-level rises, the long-term sustainability of a salt marsh is dependent upon the vegetation to maintain the elevation of the marsh platform within the intertidal zone (Morris et al., 2002). *S. alterniflora* is considered an indicator and foundation species because of its modification of the physical environment to optimize growth within tidal marshes (Pennings and Bertness, 2001). *S. alterniflora*, primary production (both above- and below-ground biomass) varies throughout the marsh itself and within the tidal range and is found to be highest at the lower elevations of its vertical growth range (Pomeroy et al., 1981).

At lower surface elevations (and thus greater extent of inundation), growth of *S. alterniflora* is likely limited and the marsh plant community is replaced by un-vegetated tidal mudflats (Morris et al., 2002). A marsh platform positioned above its ideal elevation for biomass production is more sustainable because, in the future, it will endure a higher RSLR (Morris et al., 2002). There is an ideal marsh platform elevation for tidal wetland vegetation productivity, though it can differ by study area as a function of tidal range and other factors (McKee and Patrick, 1988). Past research shows that MSL can vary seasonally by up to ~30 cm (Kjerfve and McKellar, 1980). Marsh surface elevations cannot respond with changes of this magnitude in such a short time period. Variability in summer MSL, as great as ± 20 cm, can have an effect on primary production (Morris et al., 2002). The variability of primary production is in part due to the time lag between changes in MSL and adjustment in elevation of the marsh surface. Within the Delaware Estuary as a result of steric effects on water temperature, seasonal variation in MSL can vary on the order of 16 cm (Figure 4.17). Can these seasonal changes in part explain the elevation losses recorded in the six SETs within the St. Jones River and Blackbird Creek? If the seasonal variations are the cause, then the resultant losses in elevation should be recorded on an annual basis. However, the observed data show noticeable changes in surface elevation only during the 2009 sampling season.

Data from the Lewes tide gauge indicate an increase in sea level in the Delaware Estuary over the past ~100 years. The rate of sea-level rise over this time period is 3.20 ± 0.28 mm/yr (Figure 4.18). However, superimposed along this overall increasing trend are shorter time period negative (sea level decreasing) and positive (sea level increasing) signals. For example, starting about half-way through 2008 and continuing until early in 2009, sea level decreases. This is followed by a large increase that continues until the end of 2009, when sea level again began to decrease (Figure 4.19). Similar sea-level trends are observed in the Reedy Point, Delaware tidal records (Figure 4.20). These inter-annual sea-level fluctuations would affect all of the SET sites in this study.

It is the 2009 large sea-level increase that correlates to the surface elevation decreases at six of the SET sites between the April and November, 2009 sampling intervals. The increase in sea level results in a period of higher tidal water levels

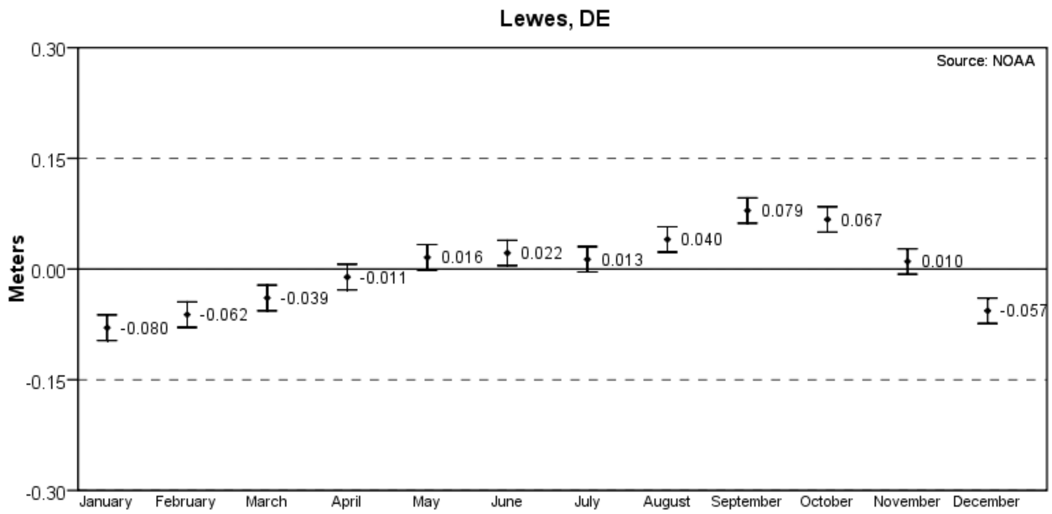


Figure 4.17 The average seasonal cycle of mean sea level, caused by regular fluctuations in coastal temperatures, salinities, winds, atmospheric pressures, and ocean currents, is shown along with each month's 95% confidence interval (NOAA, 2014).

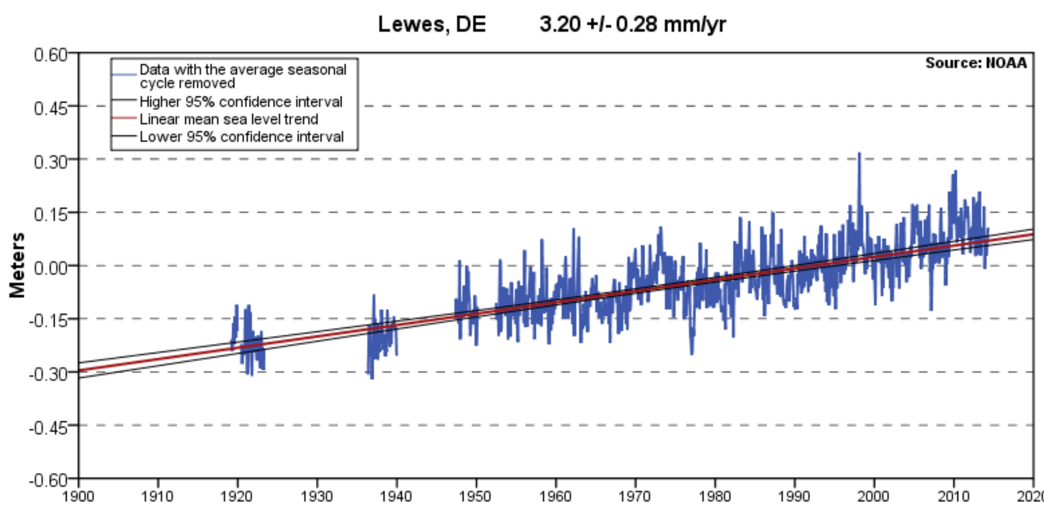


Figure 4.18 Mean sea-level trend for Lewes, Delaware. The plot shows the monthly mean sea level without the regular seasonal fluctuations due to coastal ocean temperatures, salinities, winds, atmospheric pressures, and ocean currents. The long-term linear trend is also shown, including its 95% confidence interval (NOAA, 2014).

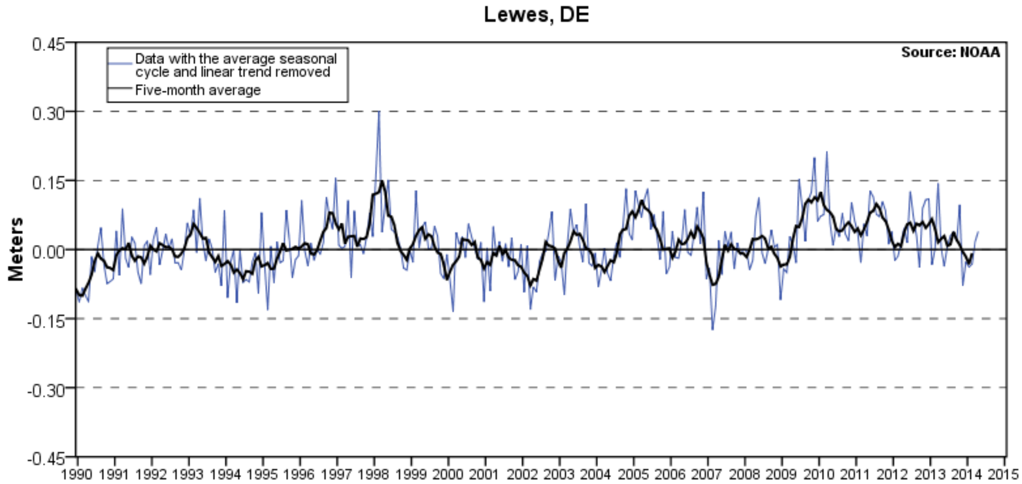


Figure 4.19 Interannual variation, since 1990, for the Lewes, Delaware tide station. The plot shows the interannual variation of monthly mean sea level and the 5-month running average. The average seasonal cycle and linear sea-level trend have been removed. Interannual variation is caused by irregular fluctuations in coastal ocean temperatures, salinities, winds, atmospheric pressures, and ocean currents (NOAA, 2014).

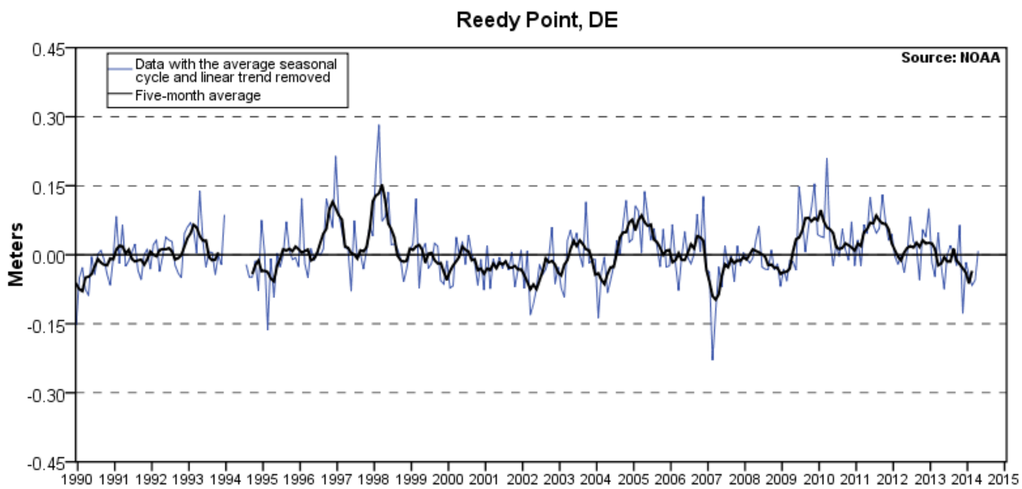


Figure 4.20 Interannual variation, since 1990, for the Reedy Point, Delaware tide station. The plot shows the interannual variation of monthly mean sea level and the 5-month running average. The average seasonal cycle and linear sea-level trend have been removed. Interannual variation is caused by irregular fluctuations in coastal ocean temperatures, salinities, winds, atmospheric pressures, and ocean currents. (NOAA, 2014).

superimposed on the already existing long-term increasing sea-level trend and seasonal sea-level variations. During 2009 through 2010, a moderate El Nino period contributes to higher seasonal sea-level anomalies (Figure 4.21; Sweet and Zervas, 2011). As a result of this moderate El Nino, the mean northeasterly wind component along the Atlantic Coast region (especially New England and the Mid-Atlantic) is at a 1960–2010 high during the 2009/2010 cooler than normal winter season (Sweet and Zervas, 2011). Northerly–northeasterly wind forcing is parallel to most of the Mid-Atlantic and New England coastline (Han, 2007; Lentz, 2008). The observed coastal sea-level rise patterns could stem from both an Ekman-related on-shore transport, an enhanced along-shelf pressure gradient, and a related southwestward geostrophic transport over the Mid-Atlantic Bight and Scotian continental shelves (Han, 2007; Lentz, 2008; Sweet and Zervas, 2011).Higher water levels due to seasonal cycles, the 2009 interannual rising sea-level signal (incursion), the 2009-2010 El Nino effect, and the long-term increase in sea level, would result in increased flooding in the studied marshes causing a longer duration of inundation on the marsh platform, shorter periods of low tide marsh platform exposure, increasing salinity levels on the marsh platform, and a shift of the salt line further up the estuary. These factors would reduce the health of the wetland vegetation and result in a reduction in below-ground biomass production.

The higher rates of subsidence through 2009 and 2010 could also be influenced by the increased water on the marsh during the higher water level period, due to greater overburden (weight of overlying water) on the marsh surface. This could result in

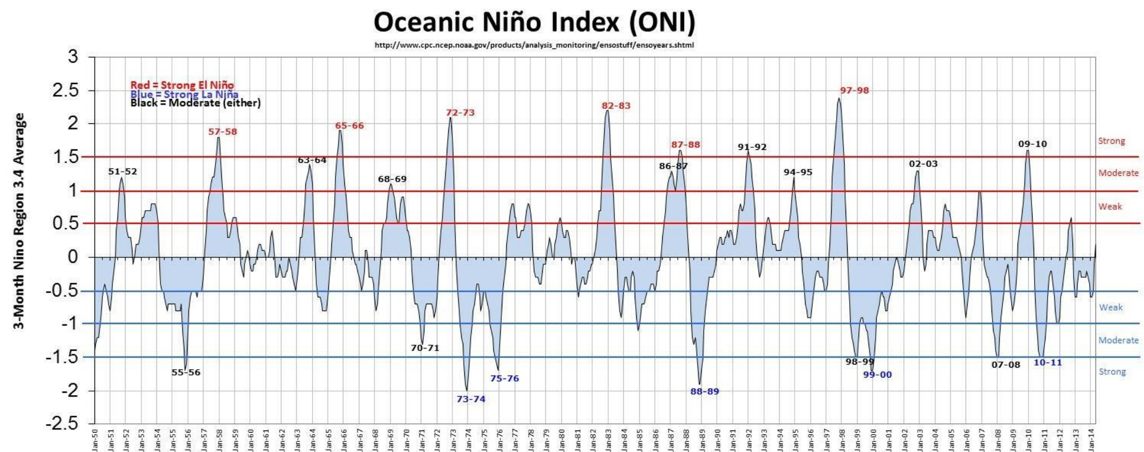


Figure 4.21 El Niño and La Niña Years and Intensities Based on Oceanic Niño Index (ONI). The Oceanic Nino Index (ONI) has become the de-facto standard that NOAA uses for identifying El Niño (warm) and La Niña (cool) events in the tropical Pacific. It is the running 3-month mean SST anomaly for the Niño 3.4 region (i.e., 5^{o}N-5^{o}S, 120^{o}-170^{o}W; (Null, 2014).

decreases in marsh surface elevation. Short-term increases in subsidence could account for a portion of the elevation losses in the shallow and deep records. Sites with large losses in shallow elevation (with apparently healthy vegetation) are examples of this process. The Impoundment, Boardwalk, Isaac's Branch, and Delon sites all have apparent "healthy" vegetative communities, and all have been exposed to extensive flooding due to storm events and lunar tides. Of these four sites, the Impoundment, Delon, and Isaac's Branch SETS are characterized by high shallow subsidence rates of 65.8, 34.2, and 29.3 mm/yr, respectively.

4.6 Conclusions

The use of SETs is important in identifying and monitoring changes within tidal wetlands in response to sea-level rise. The SETs were used to make precise measurements, over the course of many years, of shallow and deep elevation change of the marsh surface, surface accretion, and subsidence at each monitoring site. The long-term monitoring revealed that six of the eight sites showed a loss in marsh surface elevation (both shallow and deep), that has not recovered. Deep and shallow elevation changes all occurred independent of accretion. It was also shown that two (Eagles Nest and Beaver Branch) of the eight monitoring sites experienced rapid gains in elevation (and losses not experienced at the other six sites).

The results from the marsh elevation data for the SET sites within this study appear to be a result of both localized and larger-scale factors including: 1) localized effects on wetlands (as represented in the shallow and deep surface elevation records, 2) the effects of coast-wide sea-level anomalies, due to El Niño, on marsh surface elevation trends, and 3) the effects of localized watershed scale land-use changes.

The exact cause for the large scale wetland elevation losses experienced in 2009 is hard to determine. However, the platform elevation loss at six of the eight sites does coincide with a moderate El Niño and high sea-level anomalies experienced along the East Coast that led to high water levels. The lack of recovery of the 2009 elevation losses may denote the overall stress that the current rate of sea-level rise has upon these tidal wetlands. Under these stressed conditions, the wetland vegetation cannot rebound in a manner that some marshes have demonstrated under natural conditions (Reed, 1995).

It is also not clear as to the exact mechanism(s) that is/are responsible for the rapid gains in elevation that were recorded at the Eagles Nest and Beaver Branch sites. The gains could be linked to land-use changes (i.e. converting forest to suburban and agricultural land-uses) that resulted in uplift of the marsh surface, although a definitive determination cannot be made based upon the data available in this study. More detailed studies are needed to provide better constraints on the mechanisms or feedbacks that are responsible for both the dramatic 2009 wetland elevation losses and the subsequent lack of elevation recovery, and the rapid elevation gains at the Eagles Nest and Beaver Branch sites.

With only four SET locations within each watershed, the collected data can most appropriately be used only to make general assessments of the trends in marsh surface elevation and accretion for each watershed. Replicate SET locations would be necessary to make more detailed assessments. The small area of study for each SET location allows for high precision assessments, but also limits the ability of the data to be correlated between sites when the number of sample sites is small (as is the case for this study). As standalone data, the results from the SET measurements obtained in this study are not sufficient to warrant publication in a peer reviewed journal. As additional SET data from the Delaware Estuary becomes available, the measurements obtained in this SET study from the St. Jones River and Blackbird Creek should be included in larger-scale assessments of tidal wetland elevation trends.

Chapter 5

CONCLUSION

In this dissertation, three separate wetland monitoring and research studies in the vicinity of the Delaware River and Bay coastline within the Delaware Estuary are described with the goal of better understanding the short-term and historic evolution of natural and impacted tidal and impounded wetlands. The specific studies and their major results include:

- The determination of an optimal (or "Goldilocks") marsh platform elevation over which biomass production of *S. alterniflora* is maximized. This study examines the vertical growth range and above- and below-ground biomass production of *S. alterniflora* with respect to MLW and MHW tidal datums. Within the watersheds and sub-estuaries studied, *S. alterniflora* is found to have an optimal ("Goldilocks") growth range between -0.07 and 0.18 m relative to MHW, and between 1.25 and 1.72 m relative to MLW.

These results can be used to assess a marsh's ability to combat changing conditions associated with sea-level rise, by determining whether or not the marsh platform is in the optimal elevation of -0.07 to 0.18 m relative to MHW and 1.25 to 1.72 m relative to MLW, to optimize the growth of *S. alterniflora*. This assessment method, and the derived relationships to MHW and MLW, assist in developing a better means for determining whether a tidal wetland area is in need of restoration. Thin-layer application of material is a typical restoration technique for such circumstances and the findings in this study help determine how much material must be added to raise the marsh platform to allow for longer-term sustainability, through the optimization of below-ground biomass production.

- The evaluation of water level management actions on the rate of accretion and wetland platform elevation of impoundment marshes in comparison to neighboring un-impounded tidal wetlands. Nine impoundments (three in northern Delaware (in the vicinity of New Castle, Delaware), four in Central Delaware Bay, and two in lower Delaware Bay) and four reference tidal wetlands were sampled to determine their rates of accretion over the past 60 years and their wetland platform elevations relative to local tidal datums. This study finds that only two (Broad Dike and Gambacorta) of the nine impoundments have statistically significant differences in mineral mass accumulation (at the $p < 0.05$ level). Of the highest significance is the statistical difference in organic mass accumulation in seven of the nine impoundments (at the $p < 0.05$ and/or $p < 0.01$ levels).

It is the water level management effect on a wetland vegetation's ability to produce below-ground biomass and to also retain organic material (through lower rates of decomposition) that is likely the driving force in the observed differences in the accretion and elevation of impounded and natural or reference wetlands. The below-ground biomass can be the main mechanism for in-place vertical marsh accretion, but as the water level manipulations short-circuit these natural processes, the disparity in organic matter drives production and preservation of the impounded wetlands to have overall lower accretion rates and marsh platform elevation.

It is imperative to manage water levels for the species of concern (i.e. waterfowl, shore birds, mosquito control, etc.), but also to manage in a way that below-ground biomass is optimized and decomposition rates are not enhanced through long-term marsh platform exposures. The overall lower marsh platform elevation with respect to mean high tide makes the ability to manage water levels harder for the desired outcome. If catastrophic levee breaches occur, as witnessed at Prime Hook Unit II in 2008, the lowered platform will be permanently flooded (due to the disequilibrium with the tidal prism), and large-scale loss of the wetland would result. In 2008/2009, storms caused breaches in Prime Hook Unit II and now the entire impoundment is a lagoon rather than a tidally-flooded back-barrier wetland. This loss of managed impoundment wetland serves as a lesson to the effect of water level manipulations and the importance of managing the wetlands for multiple goals.

- The utilization of surface elevation table (SET) monitoring at sites within two Delaware watersheds, Blackbird Creek (relatively un-impacted watershed) and St. Jones River (heavily impacted mixed urban- suburban, and agricultural watershed), to evaluate recent trends in tidal marsh surface elevation and short-term vertical accretion. The elevation and accretion records provide a means of monitoring the response of a marsh to seasonal, annual, and decadal changes in tidal levels. SET and marker horizon data also provide information on surficial and below-ground processes. The long-term monitoring revealed that six of the eight sites showed a loss in marsh surface elevation (both shallow and deep), that has not recovered. Deep and shallow elevation changes all occurred independent of accretion. It was also shown that two (Eagles Nest and Beaver Branch) of the eight monitoring sites experienced rapid gains in elevation (and losses not experienced at the other six sites).

The SET monitoring sites, within the study, show both a reflection of localized factors or effects on wetlands (as represented in the shallow and elevation records) but, more importantly, these sites are influenced by the effects of coast-wide sea-level anomalies, due to El Niño on marsh surface elevation trends and the effects of localized watershed scale land-use changes. The loss of elevation at six of the monitoring sites is of concern for the overall health and sustainability of these watersheds. Although, the sampling sites are limited in their scope of coverage, they show that these sites are not keeping pace with sea-level rise and that other effects could be witnessed in the near future (i.e. lower vegetation productivity and/or loss of vegetation). These results need to be merged with additional datasets to develop a large more representative and complete assessment of these wetlands status in relation to sea-level rise. The use of fixed station monitoring (i.e. SETs) is important in identifying and monitoring changes within tidal wetlands in response to sea-level rise. At the heart of this monitoring is the limitation in the area of coverage and what those small-scale changes mean to the bigger picture within a wetland.

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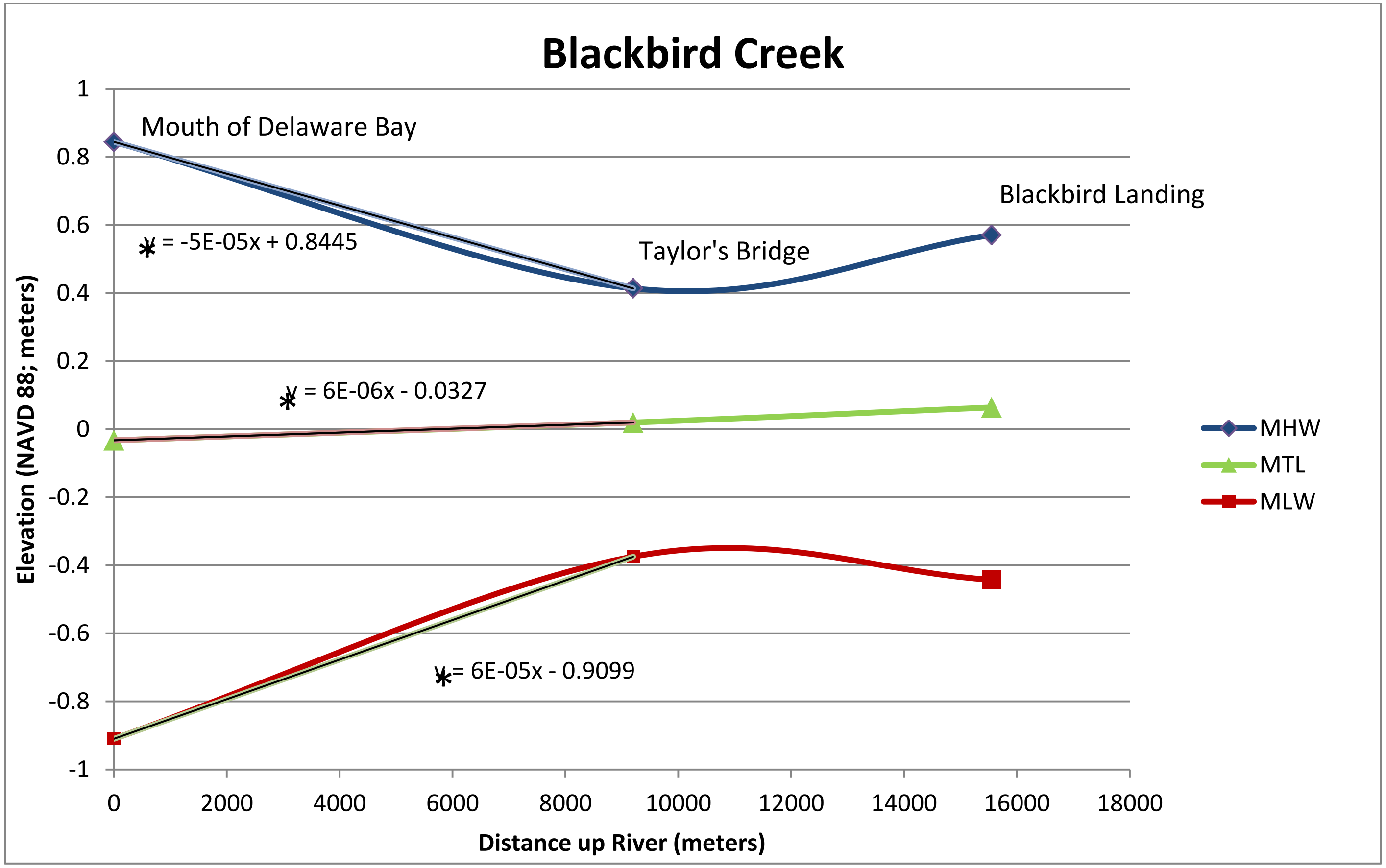
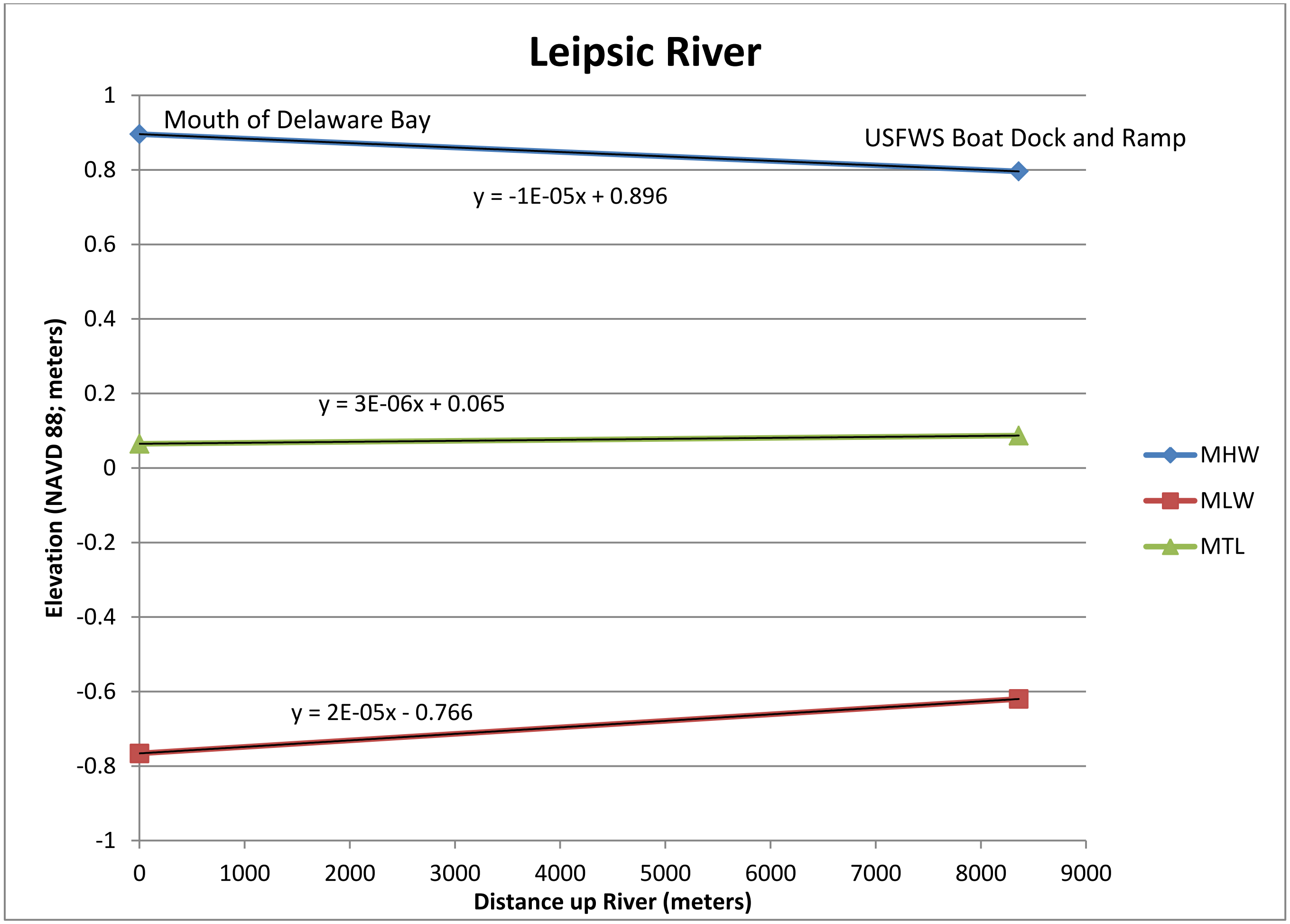
Appendix A

CONSTRUCTED TIDAL CURVES FOR *S. ALTERNIFLORA* STUDY

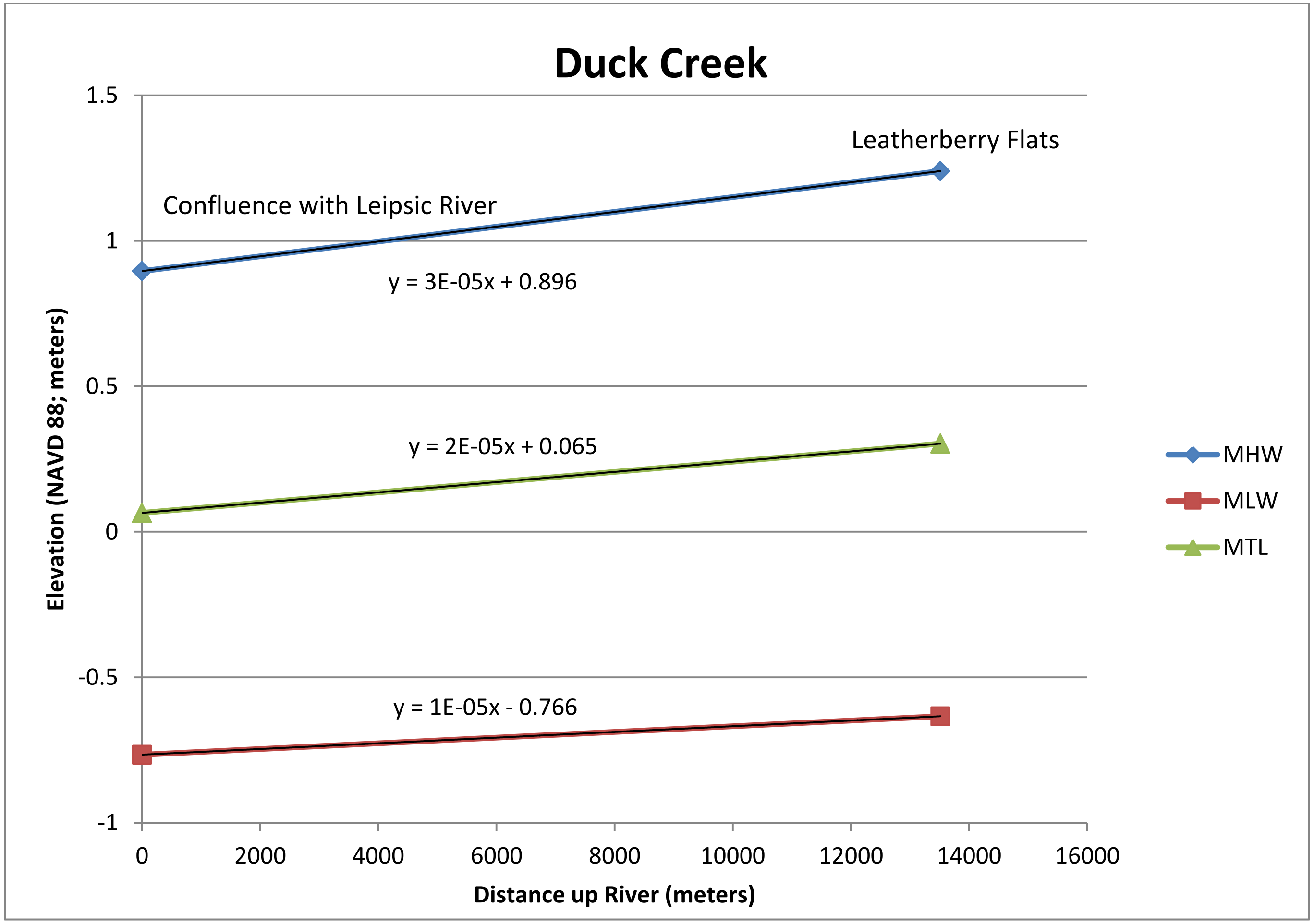


Blackbird Creek Tidal Elevation Curve

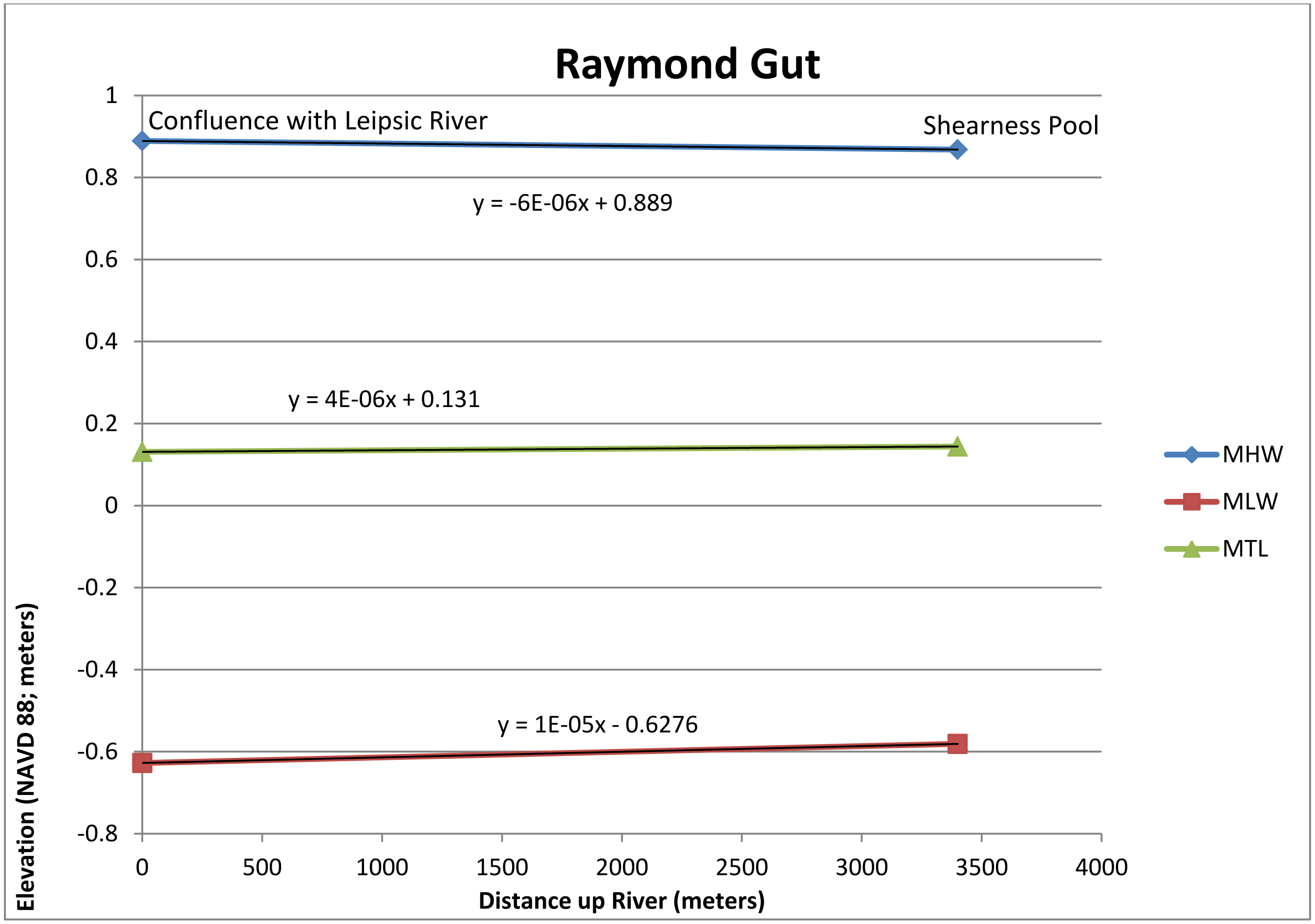
(* Fitted trendlines and equations, for each tidal component, were calculated only to Taylor's Bridge, as all short-form *S. alterniflora* occurred bayward of this tidal datum collection point.)



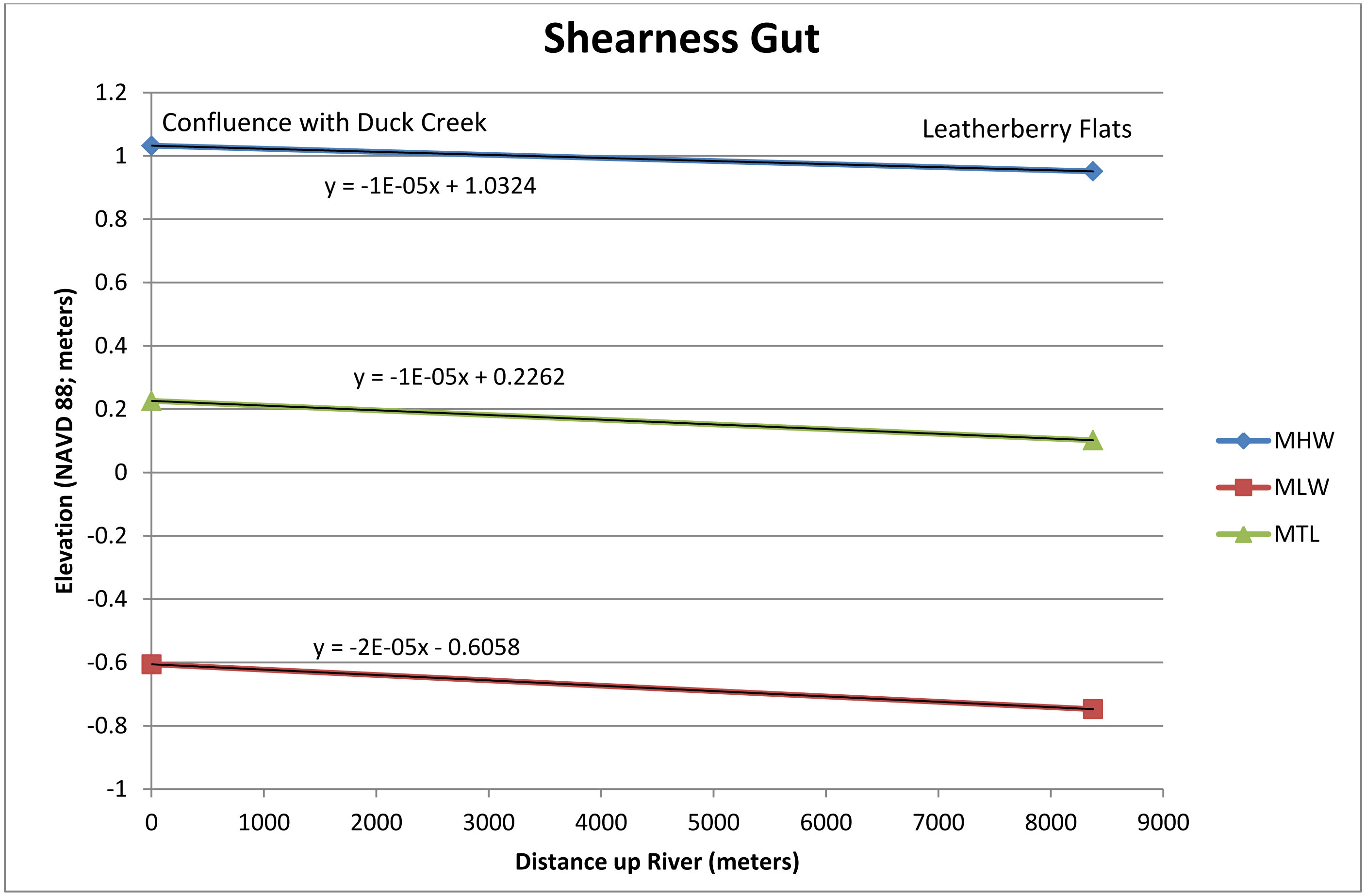
Leipsic River Tidal Elevation Curve



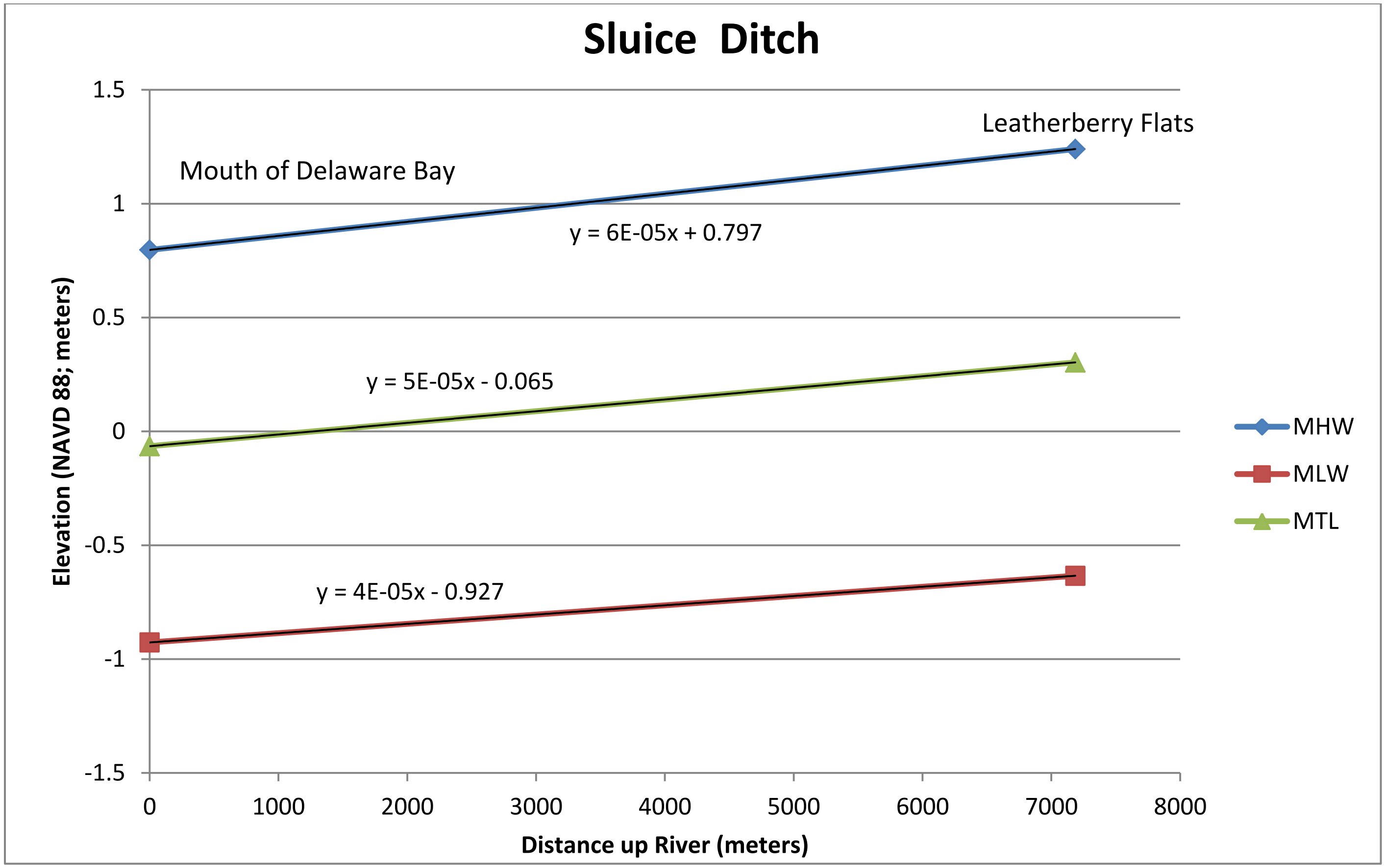
Duck Creek Tidal Elevation Curve



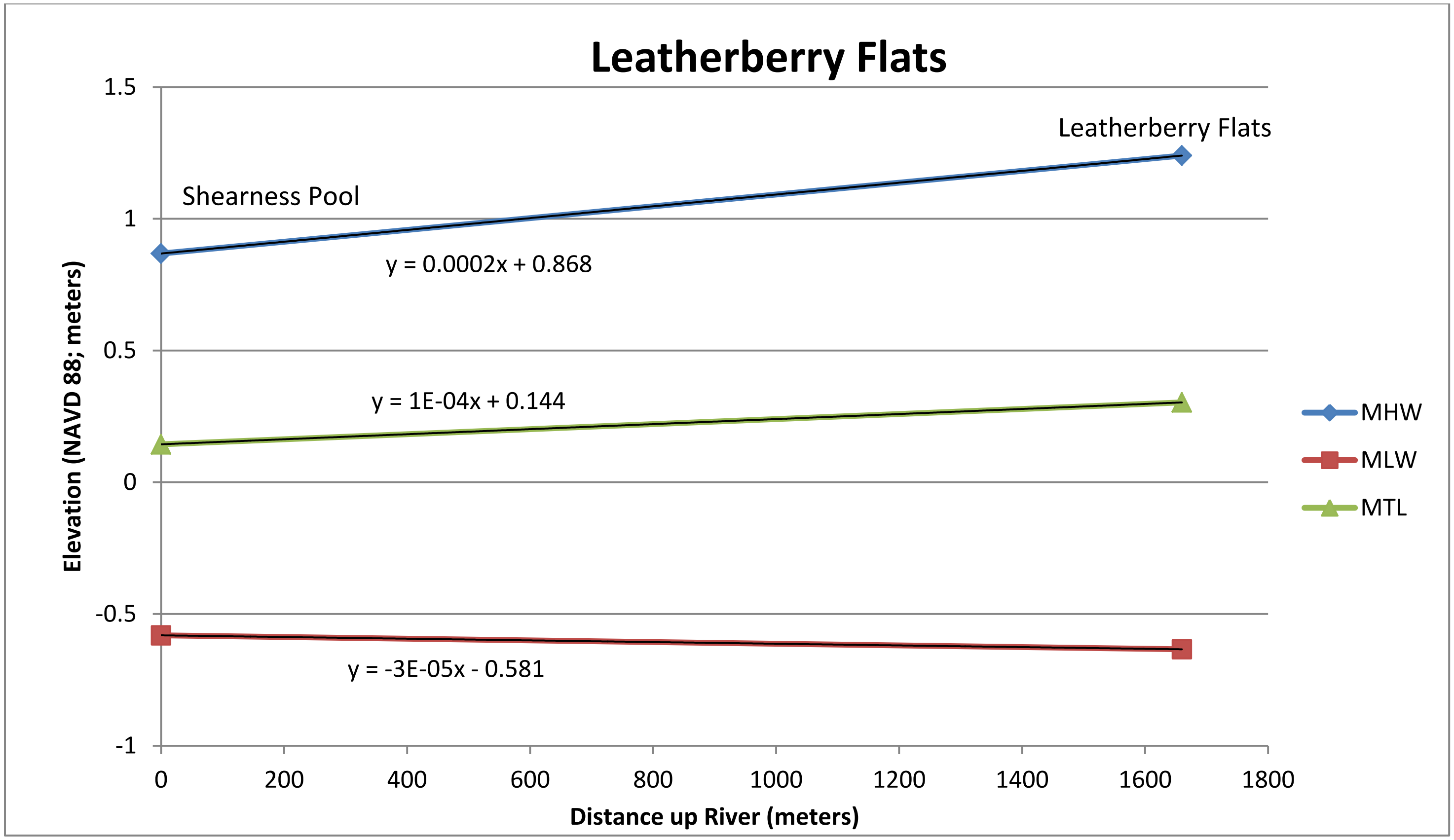
Raymond Gut Tidal Elevation Curve



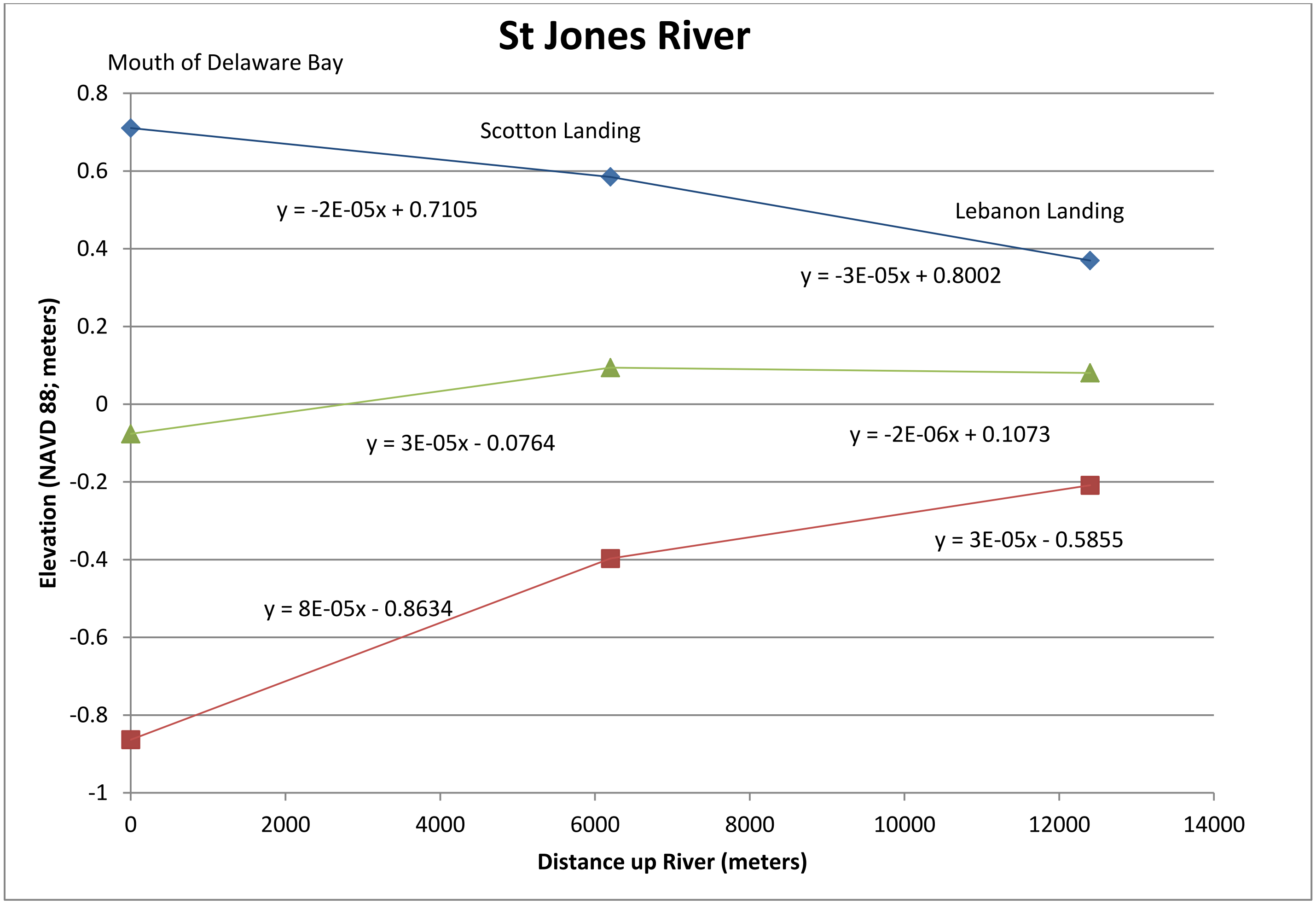
Shearness Gut Tidal Elevation Curve



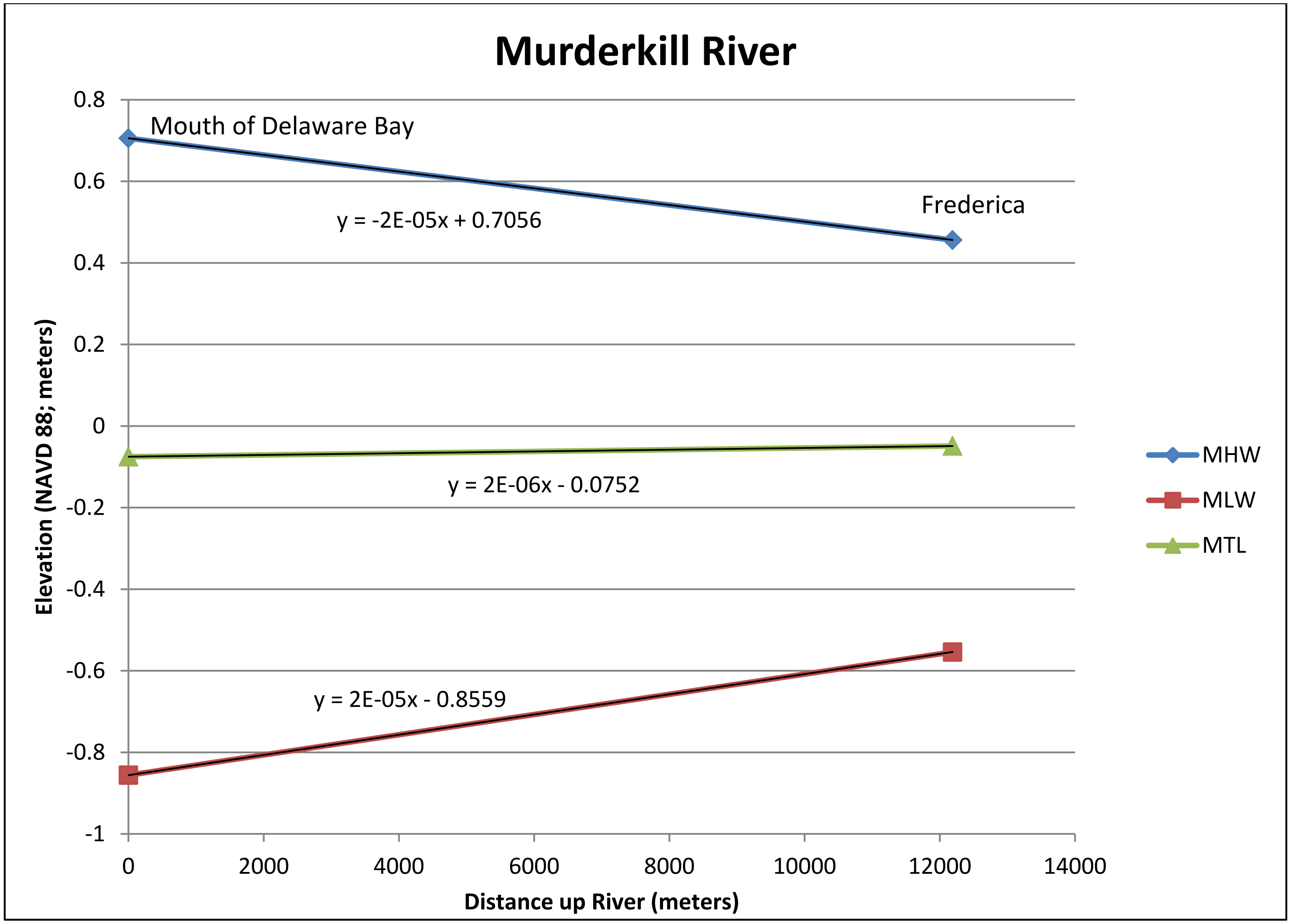
Sluice Ditch Tidal Elevation Curve



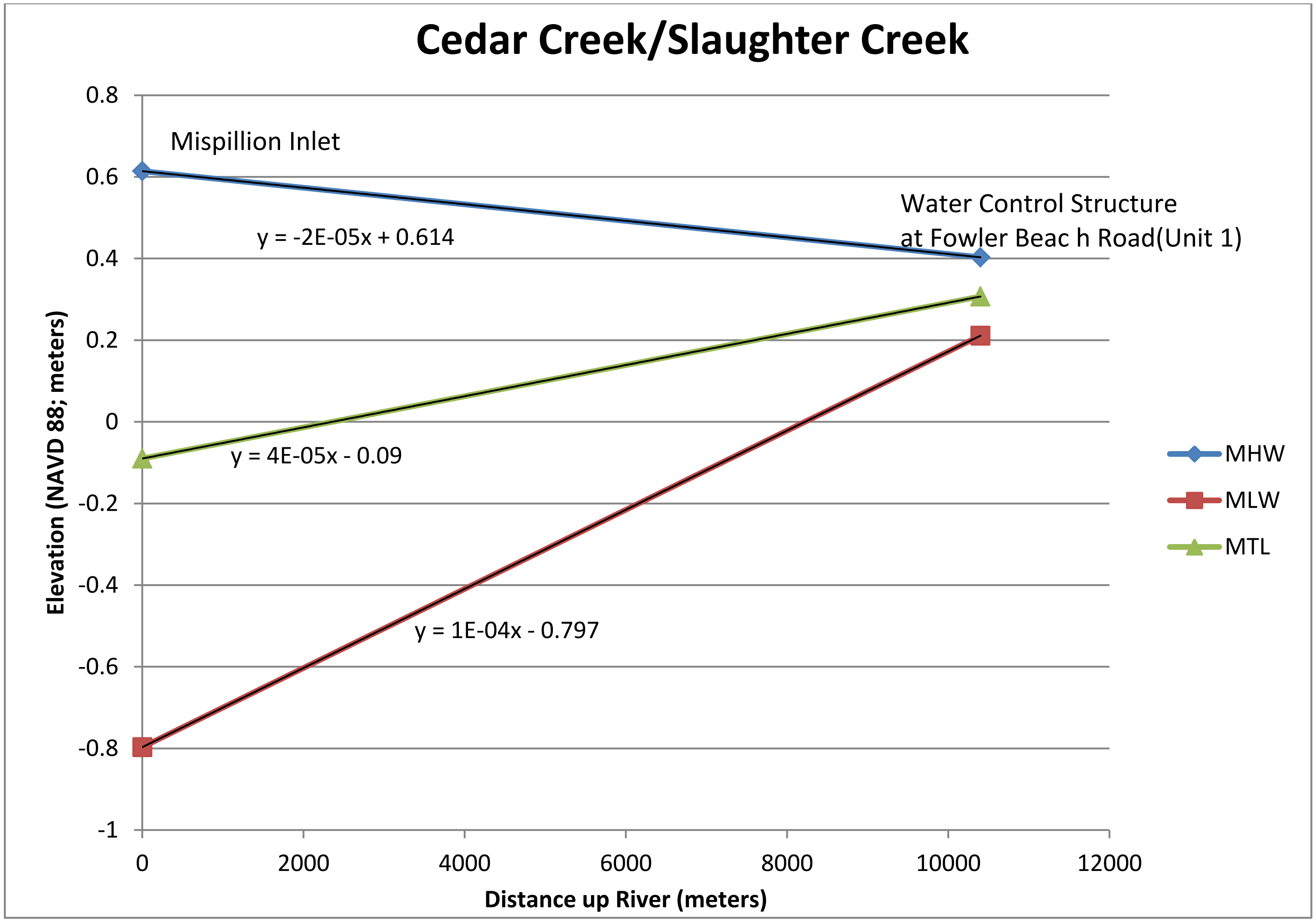
Leatherberry Flats Tidal Elevation Curve



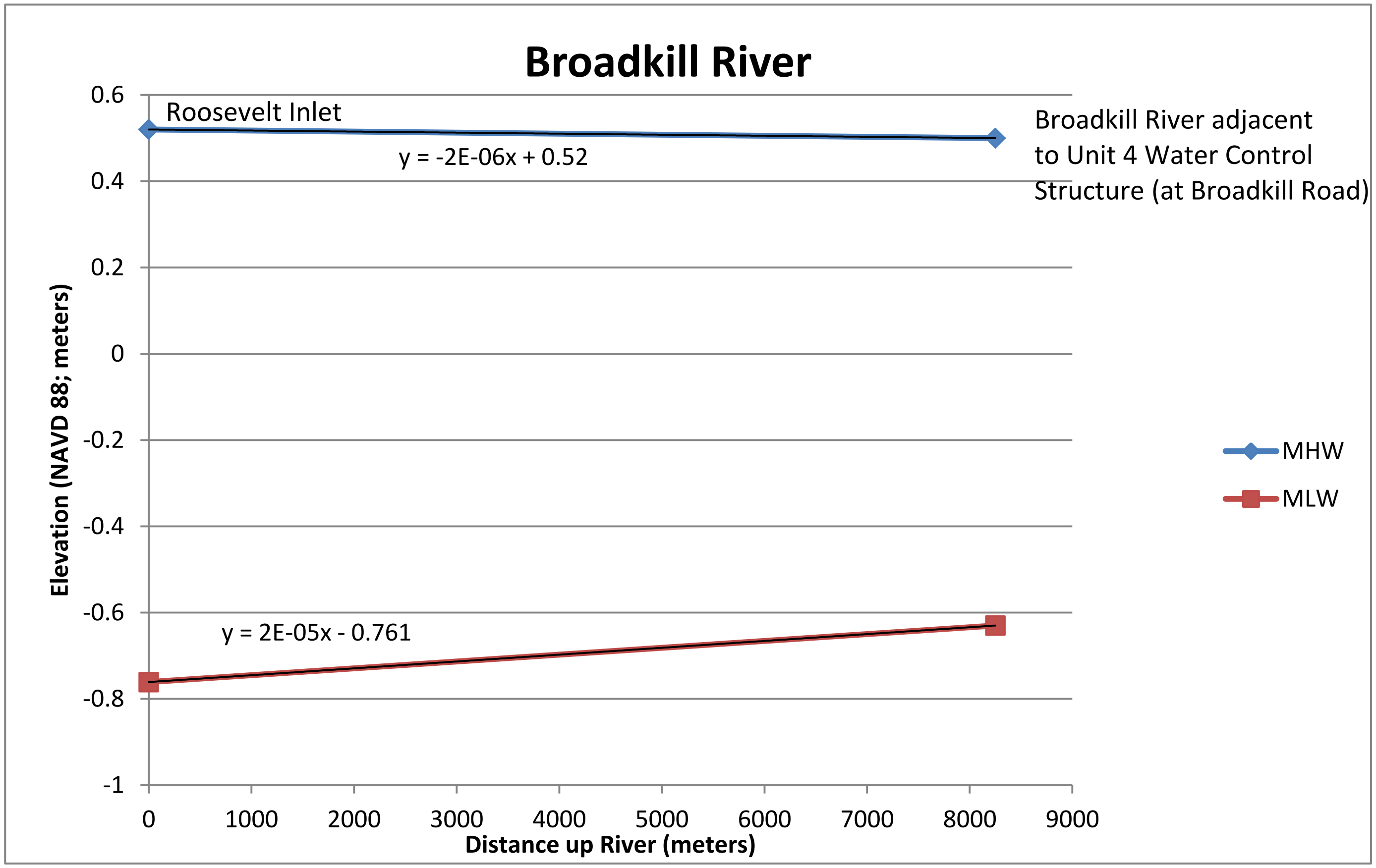
St. Jones River Tidal Elevation Curve



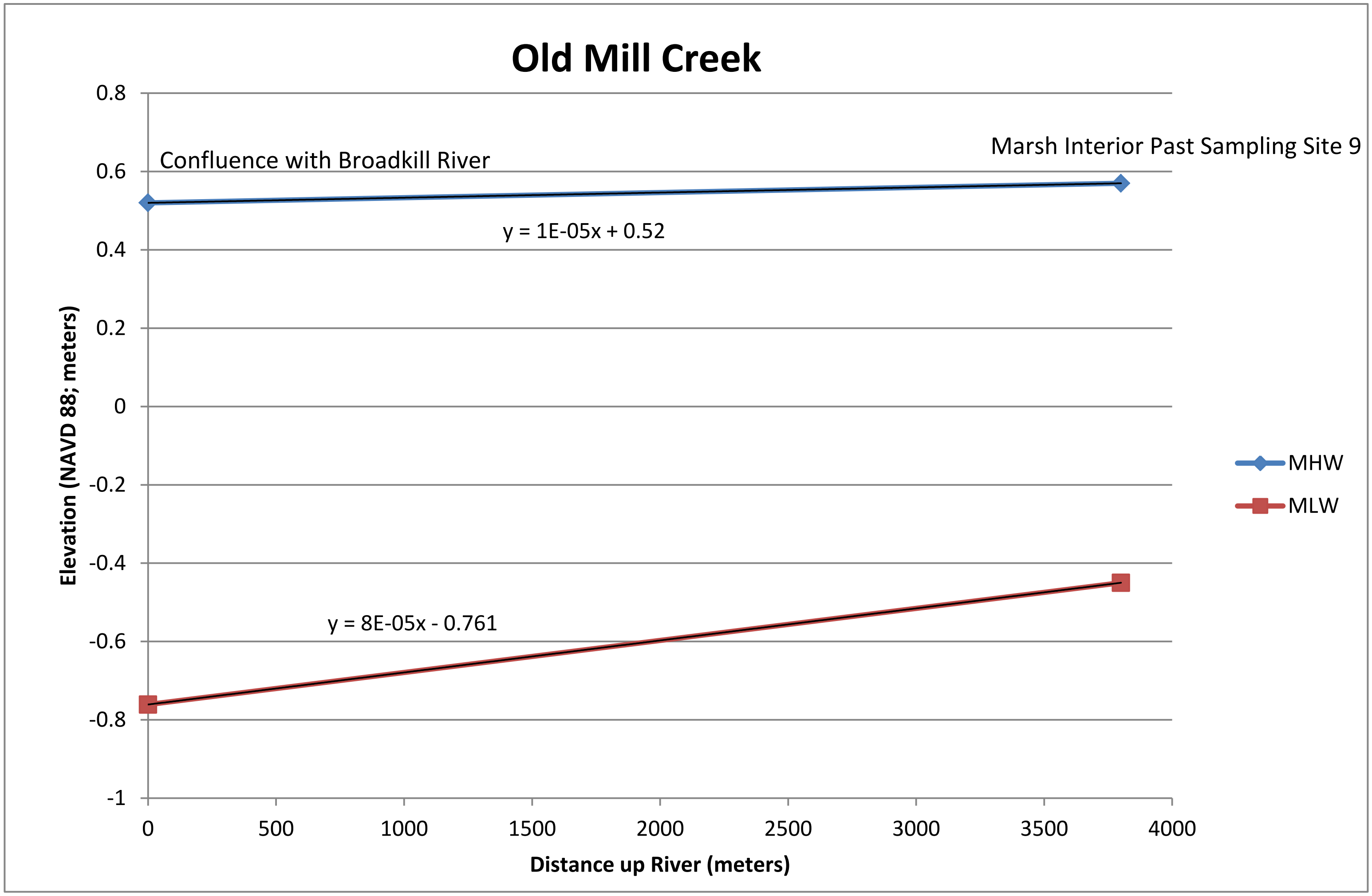
Murderkill River Tidal Elevation Curve



Cedar Creek through Slaughter Creek Tidal Elevation Curve



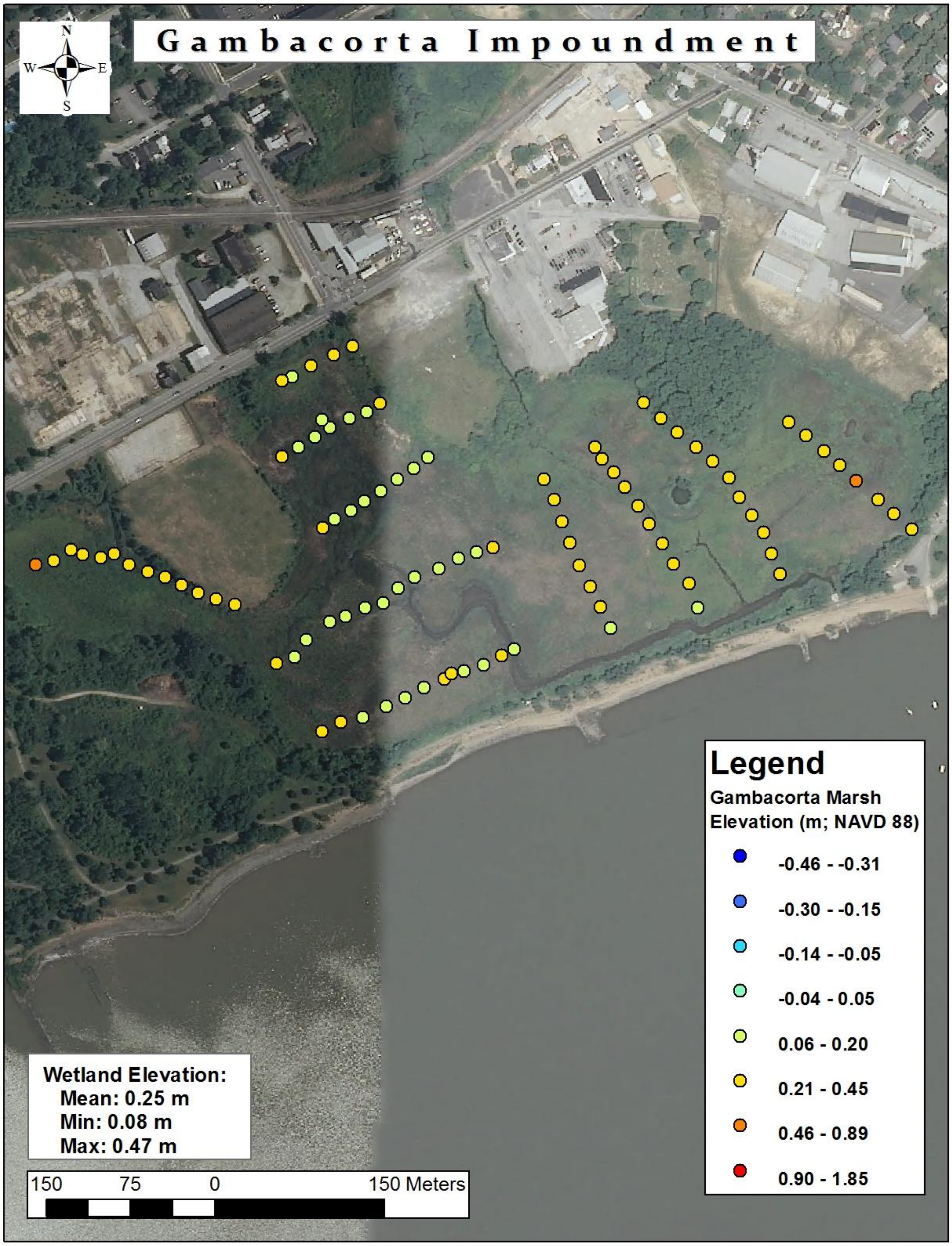
Broadkill River Tidal Elevation Curve

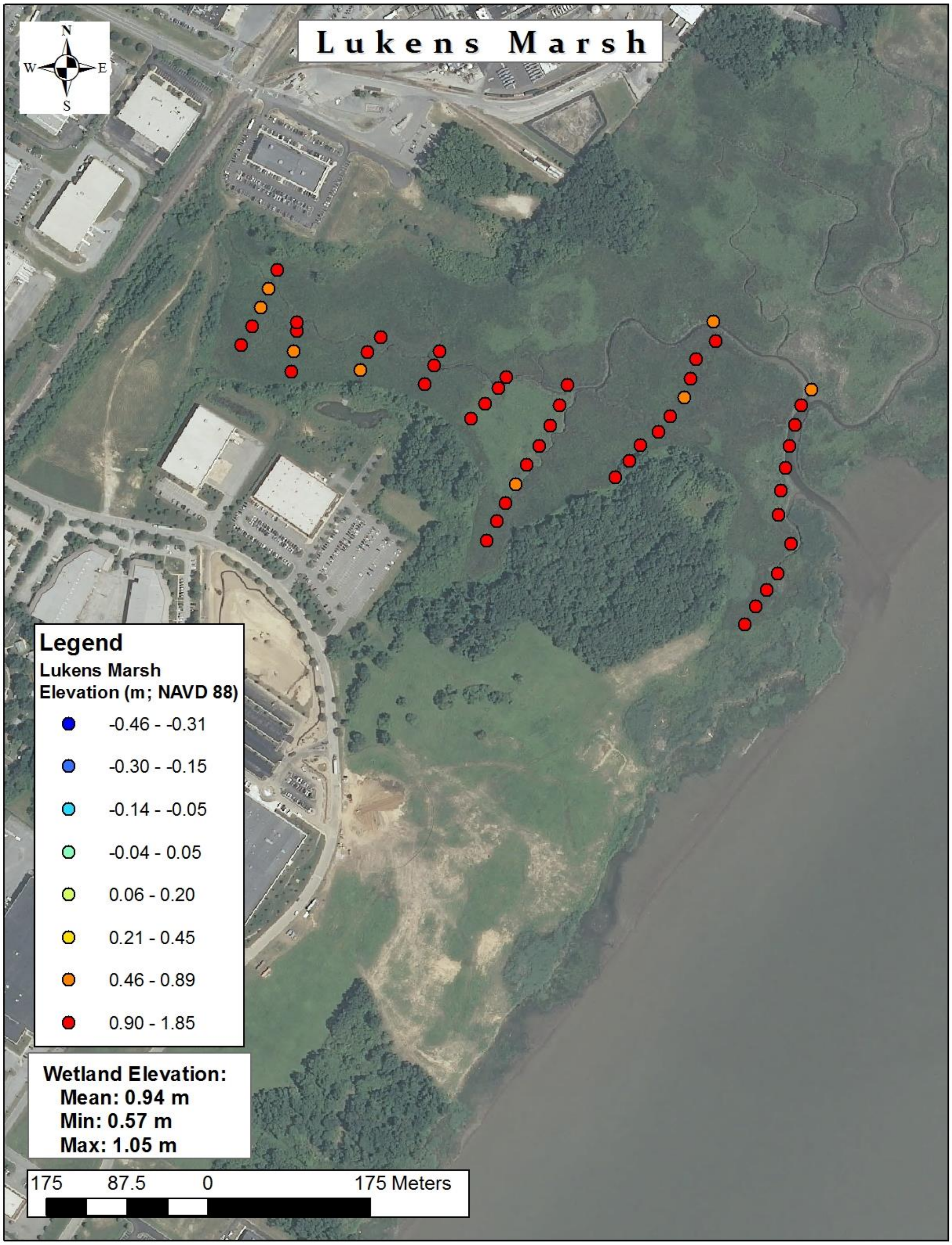


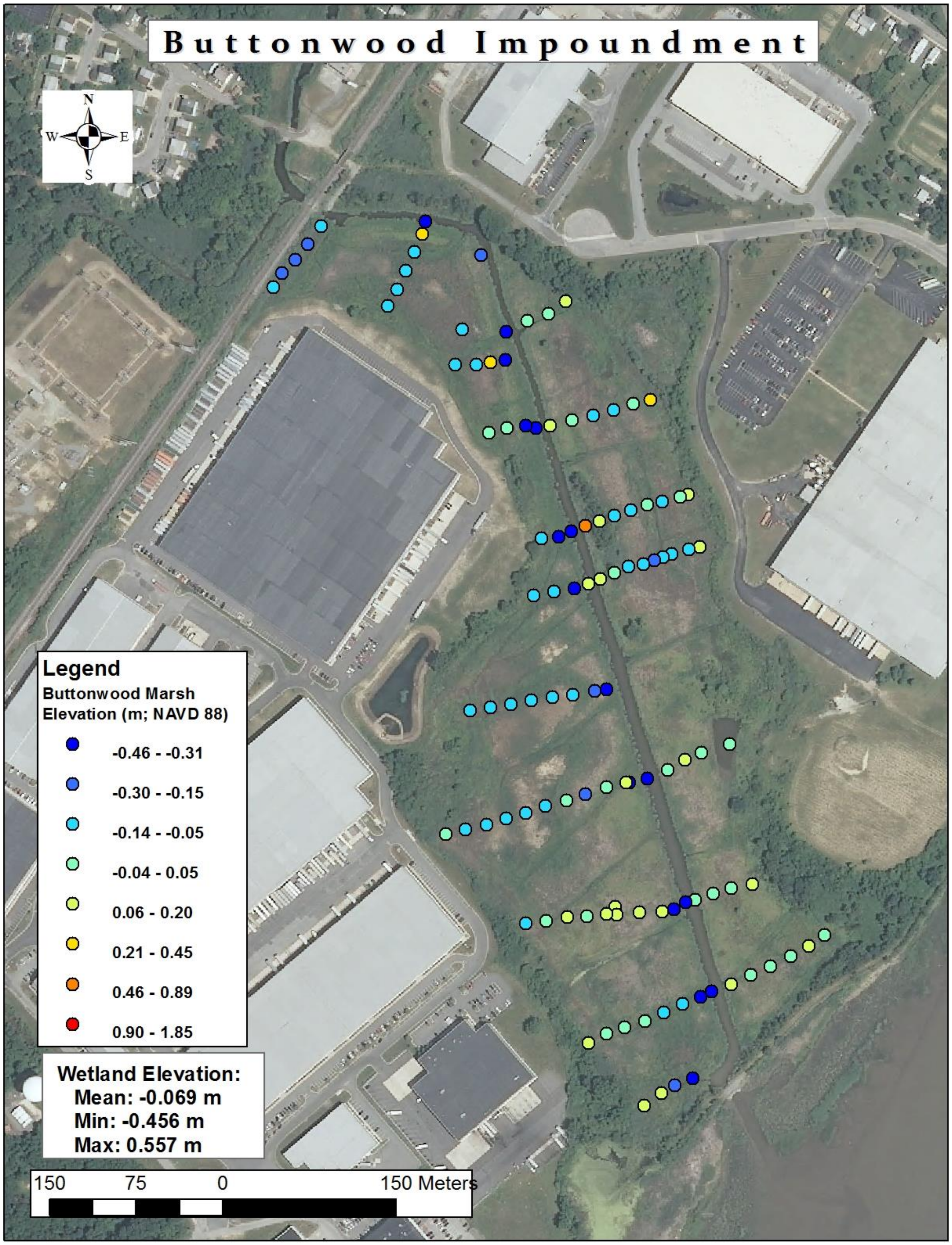
Old Mill Creek Tidal Elevation Curve

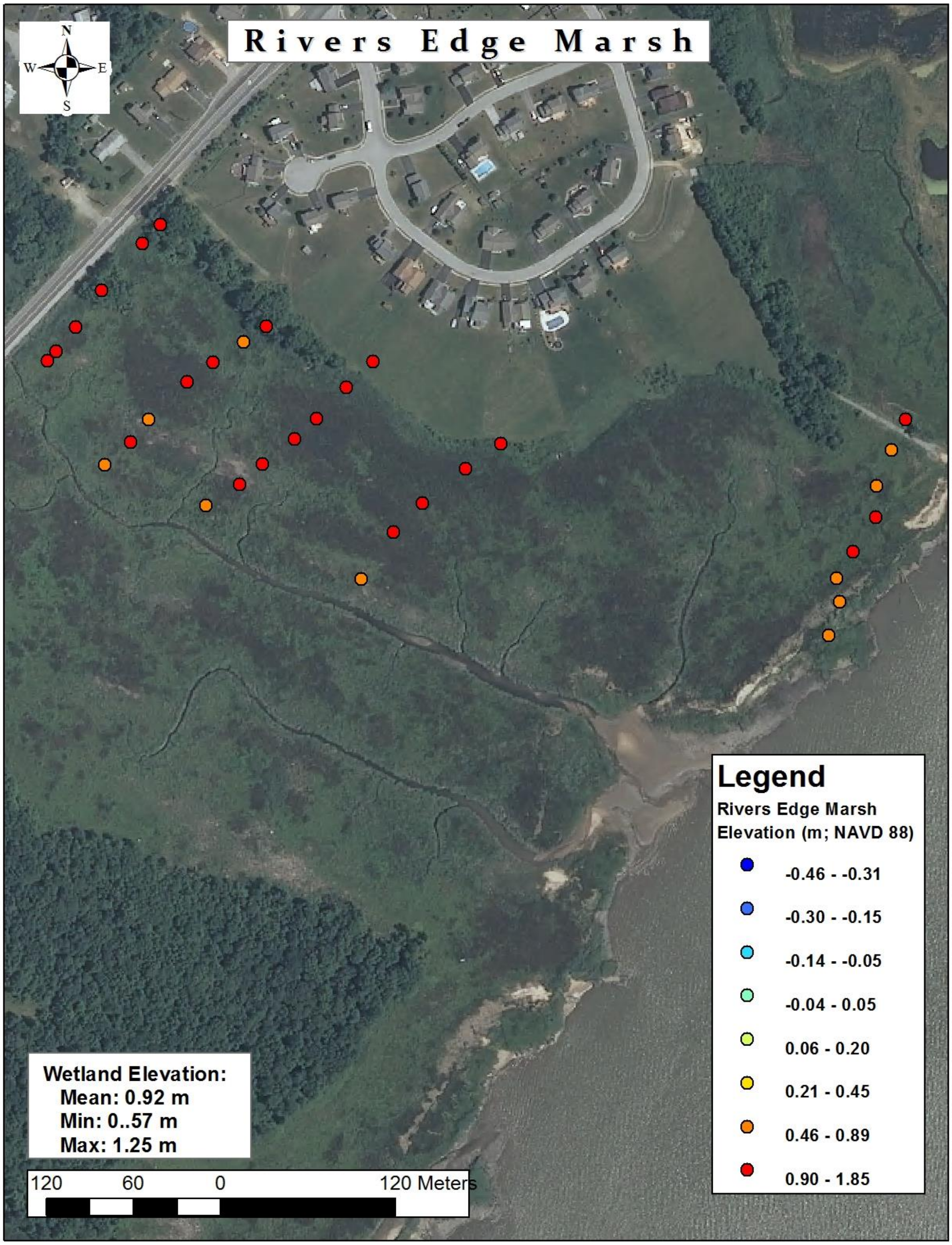
Appendix B

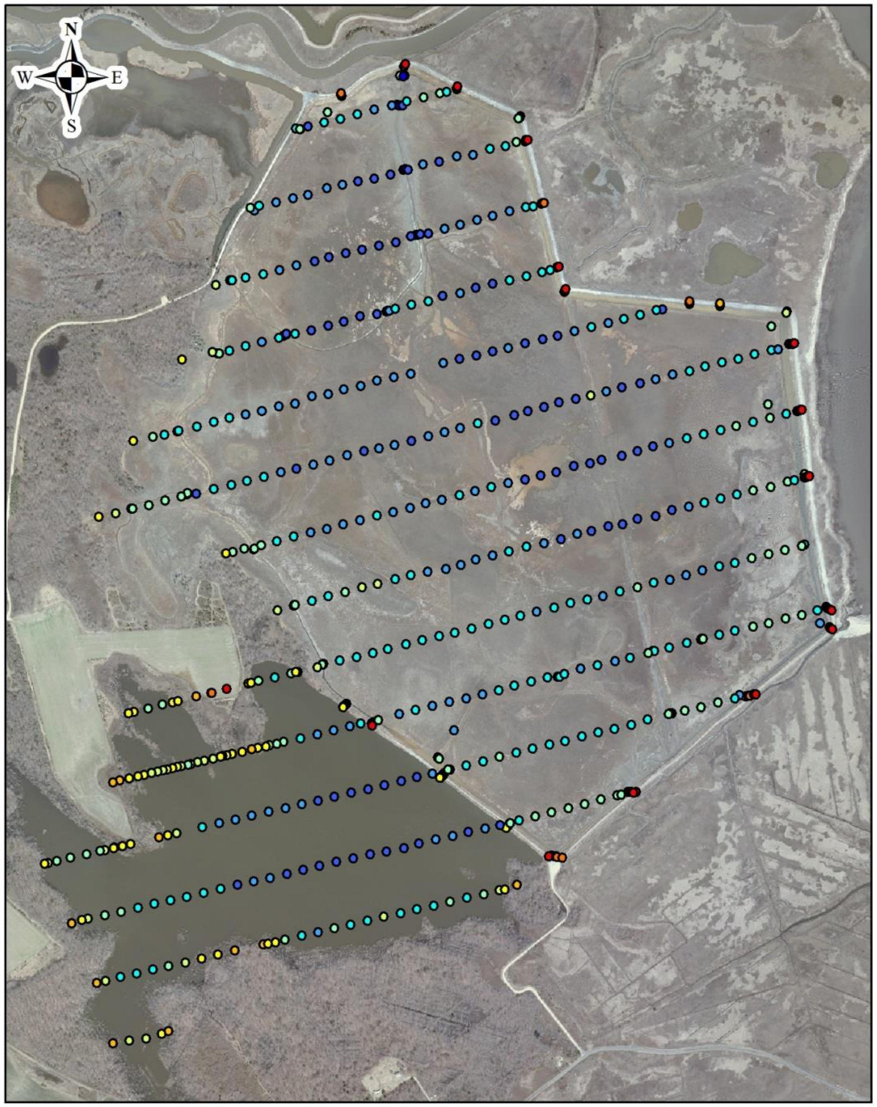
PLANE VIEWS OF IMPOUNDMENT RTK SURVEYS AND INTERPOLATED MARSH ELEVATION SURFACES


Gambacorta Impoundment
N
S
W
E
Legend
Gambacorta Marsh
Elevation (m; NAVD 88)
-0.46 - -0.31
-0.30 - -0.15
-0.14 - -0.05
-0.04 - 0.05
0.06 - 0.20
0.21 - 0.45
0.46 - 0.89
0.90 - 1.85
Wetland Elevation:
Mean: 0.25 m
Min: 0.08 m
Max: 0.47 m
150 75 0 150 Meters


Lukens Marsh
N
S
E
W
Legend
Lukens Marsh
Elevation (m; NAVD 88)
-0.46 - -0.31
-0.30 - -0.15
-0.14 - -0.05
-0.04 - 0.05
0.06 - 0.20
0.21 - 0.45
0.46 - 0.89
0.90 - 1.85
Wetland Elevation:
Mean: 0.94 m
Min: 0.57 m
Max: 1.05 m
175 87.5 0 175 Meters


Buttonwood Impoundment
N
W
E
S
Legend
Buttonwood Marsh
Elevation (m; NAVD 88)
-0.46 - -0.31
-0.30 - -0.15
-0.14 - -0.05
-0.04 - 0.05
0.06 - 0.20
0.21 - 0.45
0.46 - 0.89
0.90 - 1.85
Wetland Elevation:
Mean: -0.069 m
Min: -0.456 m
Max: 0.557 m
150 75 0 150 Meters


Rivers Edge Marsh
N
W
E
S
Legend
Rivers Edge Marsh
Elevation (m; NAVD 88)
-0.46 - -0.31
-0.30 - -0.15
-0.14 - -0.05
-0.04 - 0.05
0.06 - 0.20
0.21 - 0.45
0.46 - 0.89
0.90 - 1.85
Wetland Elevation:
Mean: 0.92 m
Min: 0..57 m
Max: 1.25 m
120 60 0 120 Meters



Little Creek Impoundment

Wetland Elevation RTK Survey

Legend

Little Creek RTK Survey

Height (m; NAVD 88)

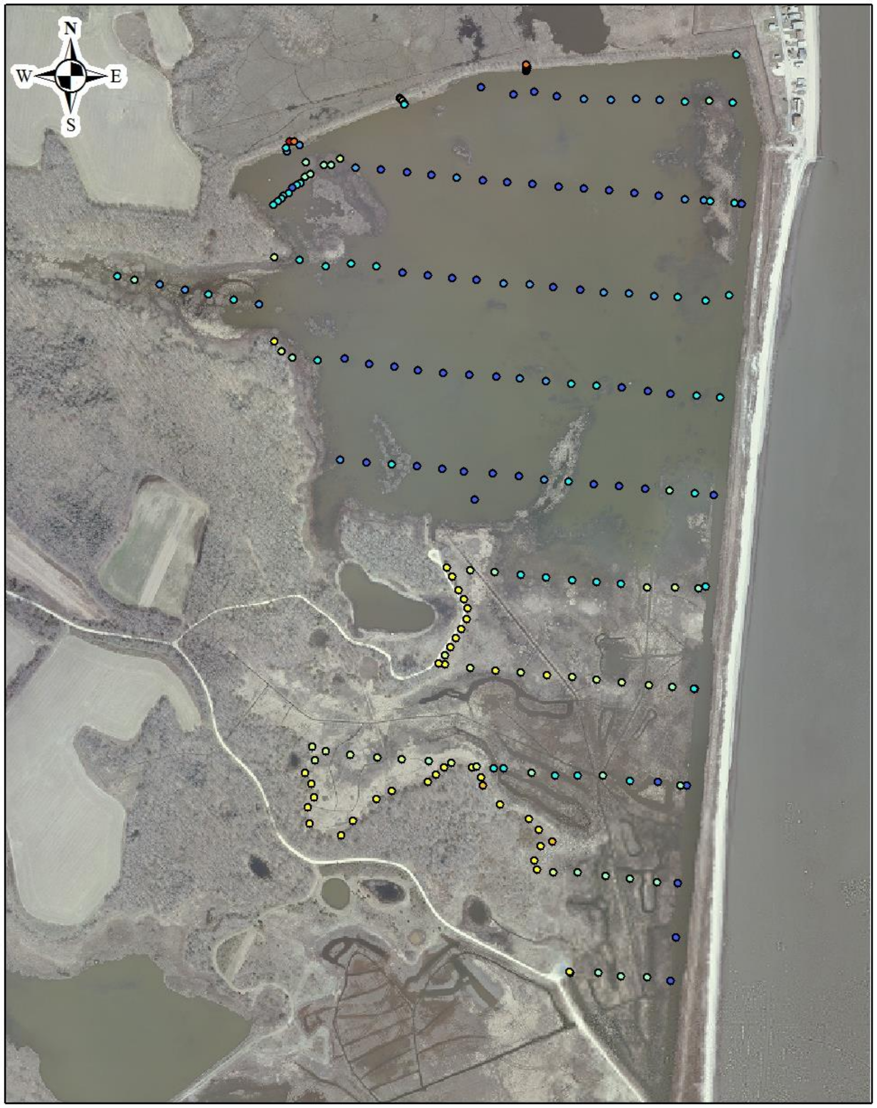
- -0.57 - 0.00
- 0.01 - 0.39
- 0.40 - 0.50
- 0.51 - 0.60
- 0.61 - 0.73
- 0.74 - 0.90
- 0.91 - 1.15
- 1.16 - 1.55
- 1.56 - 2.14
- 2.15 - 3.07

Wetland Elevation:	Ditch Elevation:
Mean: 0.60	Mean:0.43
Minimum:0.26	Minimum: -0.57
Maximum:2.34	Maximum:0.78

Kent County Impoundments

400 200 0 400 Meters

0.3 0.15 0 0.3 Miles



Logan Lane North Impoundment

Wetland Elevation RTK Survey

Legend

Logan Lane North RTK Survey

EleHeight (m; NAVD 88)vation

- -0.57 - 0.00
- 0.01 - 0.39
- 0.40 - 0.50
- 0.51 - 0.60
- 0.61 - 0.73
- 0.74 - 0.90
- 0.91 - 1.15
- 1.16 - 1.55
- 1.56 - 2.14
- 2.15 - 3.07

Wetland Elevation:	Un-vegitated Elevation:
Mean: 0.74	Mean:0.43
Minimum:0.20	Minimum: 0.32
Maximum:1.176	Maximum: 2.43

Kent County Impoundments

300 150 0 300 Meters

0.2 0.1 0 0.2 Miles

Logan Lane South Impoundment

Wetland Elevation RTK Survey

Legend

Logan Lane South RTK Survey

Height (m; NAVD 88)

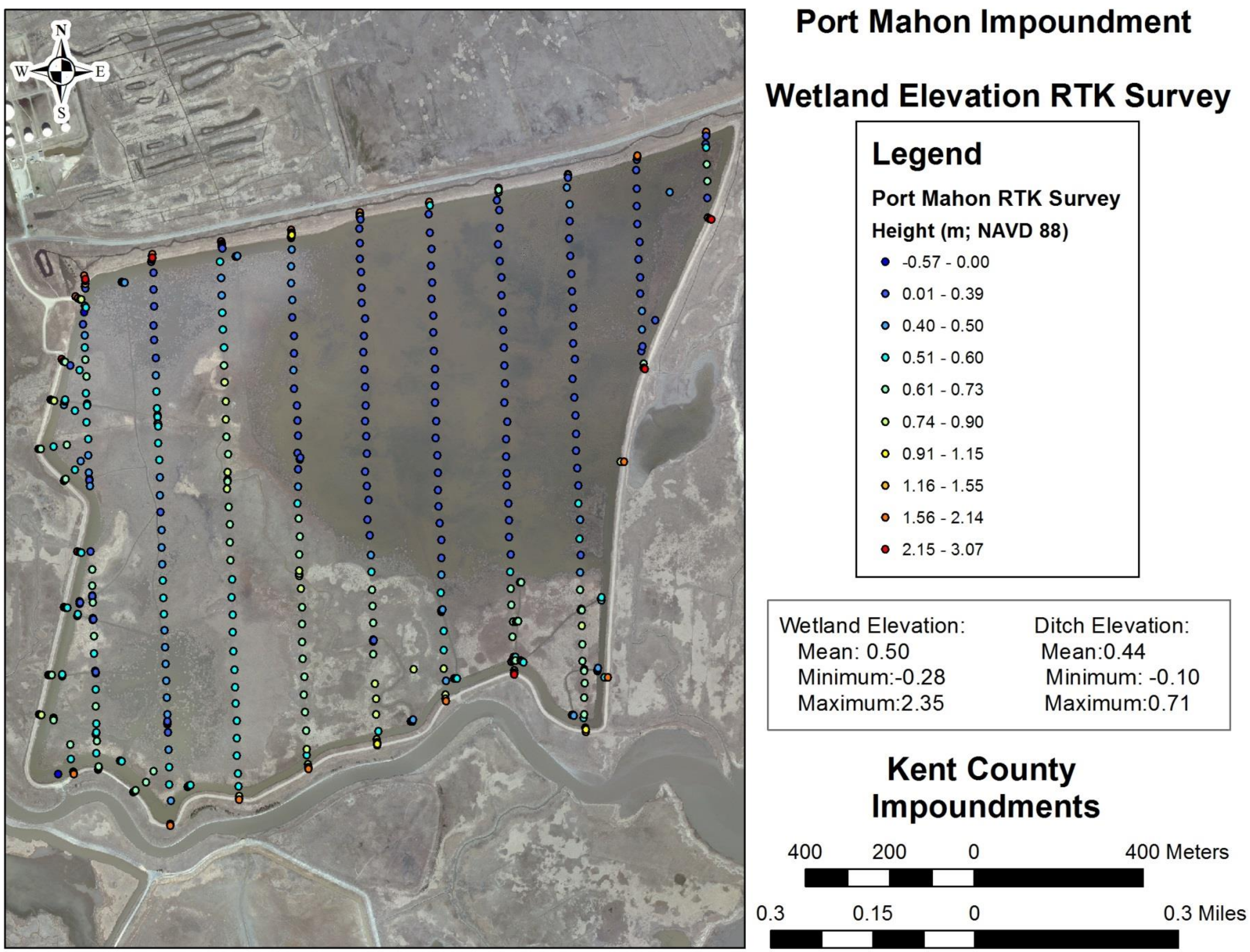
- -0.57 - 0.00
- 0.01 - 0.39
- 0.40 - 0.50
- 0.51 - 0.60
- 0.61 - 0.73
- 0.74 - 0.90
- 0.91 - 1.15
- 1.16 - 1.55
- 1.56 - 2.14
- 2.15 - 3.07

Wetland Elevation:	Ditch Elevation:
Mean: 0.62	Mean:0.52
Minimum:0.16	Minimum: 0.05
Maximum:1.61	Maximum: 0.85

Kent County Impoundments

540 270 0 540 Meters

0.4 0.2 0 0.4 Miles


Port Mahon Impoundment
Wetland Elevation RTK Survey
N
W
E
S
Legend
Port Mahon RTK Survey
Height (m; NAVD 88)
-0.57 - 0.00
0.01 - 0.39
0.40 - 0.50
0.51 - 0.60
0.61 - 0.73
0.74 - 0.90
0.91 - 1.15
1.16 - 1.55
1.56 - 2.14
2.15 - 3.07
Wetland Elevation:
Mean: 0.50
Minimum:-0.28
Maximum:2.35
Ditch Elevation:
Mean:0.44
Minimum: -0.10
Maximum:0.71
Kent County Impoundments
400 200 0 400 Meters
0.3 0.15 0 0.3 Miles

Pickering Beach Marsh

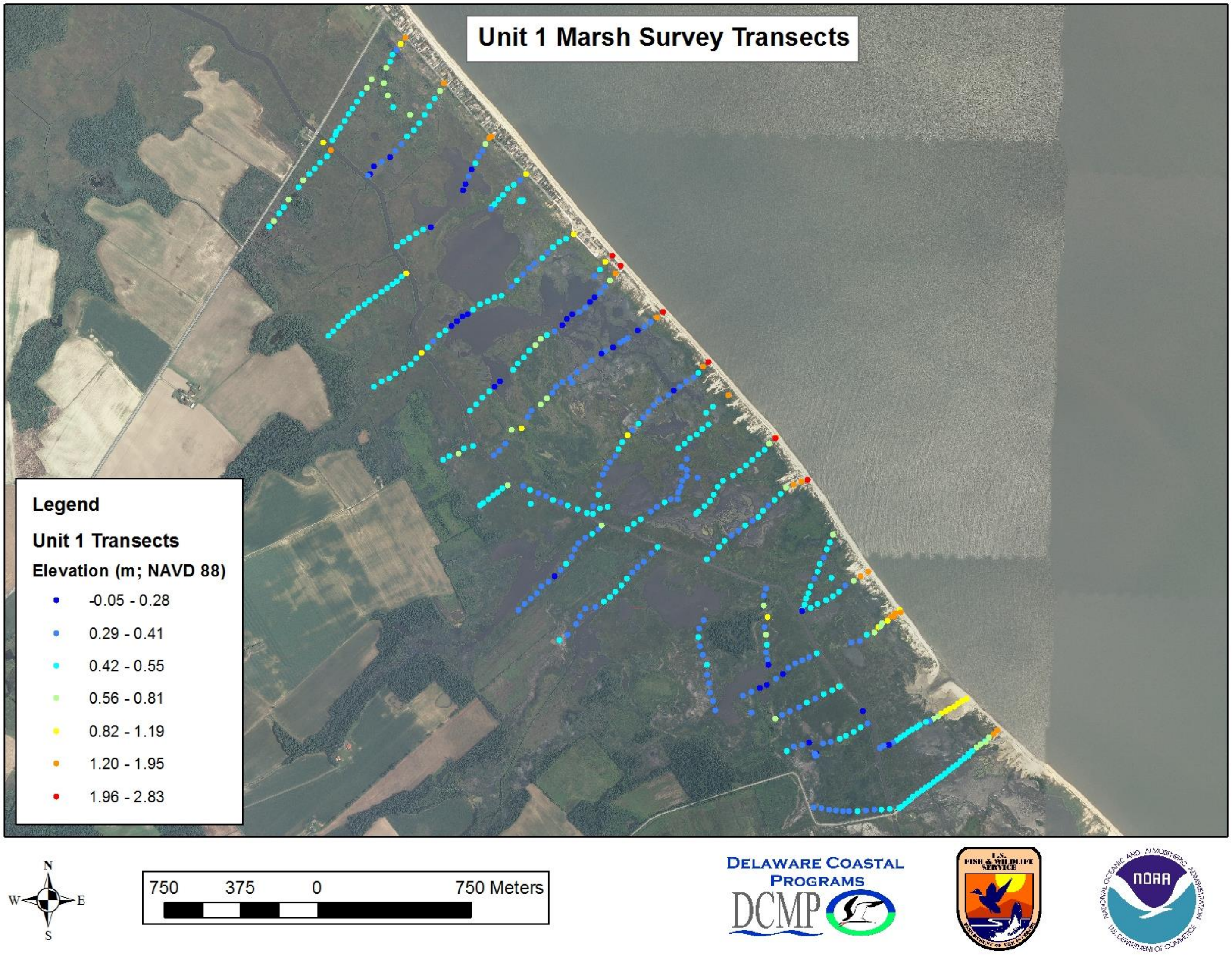
Legend
Pickering Transects
Elevation

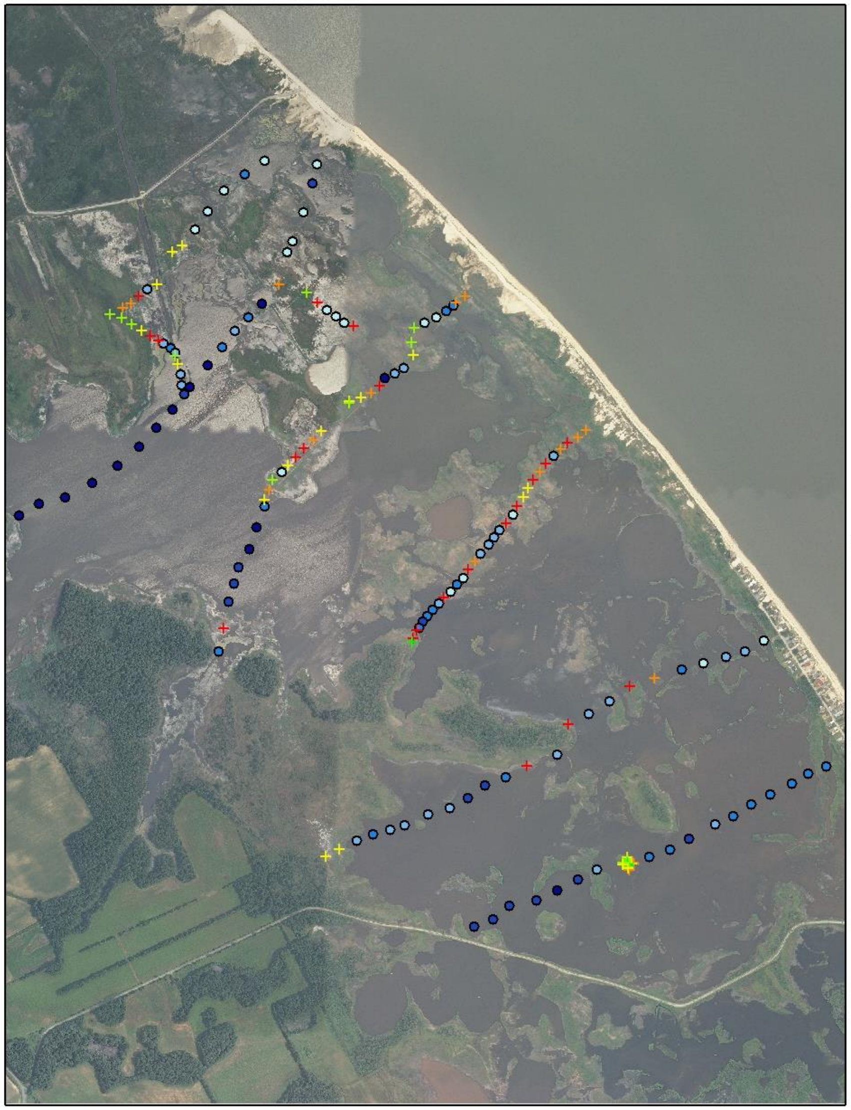
- -0.57 - 0.00
- 0.01 - 0.39
- 0.40 - 0.50
- 0.51 - 0.60
- 0.61 - 0.73
- 0.74 - 0.90
- 0.91 - 1.15
- 1.16 - 1.55
- 1.56 - 2.14
- 2.15 - 3.07

Wetland Elevation:
Mean: 1.07 m
Min: 0.44 m
Max: 1.38 m

0.2 0.1 0 0.2 Miles

DELAWARE COASTAL PROGRAMS
DCMP


Unit 1 Marsh Survey Transects
Legend
Unit 1 Transects
Elevation (m; NAVD 88)
-0.05 - 0.28
0.29 - 0.41
0.42 - 0.55
0.56 - 0.81
0.82 - 1.19
1.20 - 1.95
1.96 - 2.83
N
W
E
S
750 375 0 750 Meters
Delaware Coastal Programs
DCMP
NOAA



Unit 2 Elevations

Primehook National Wildlife Refuge

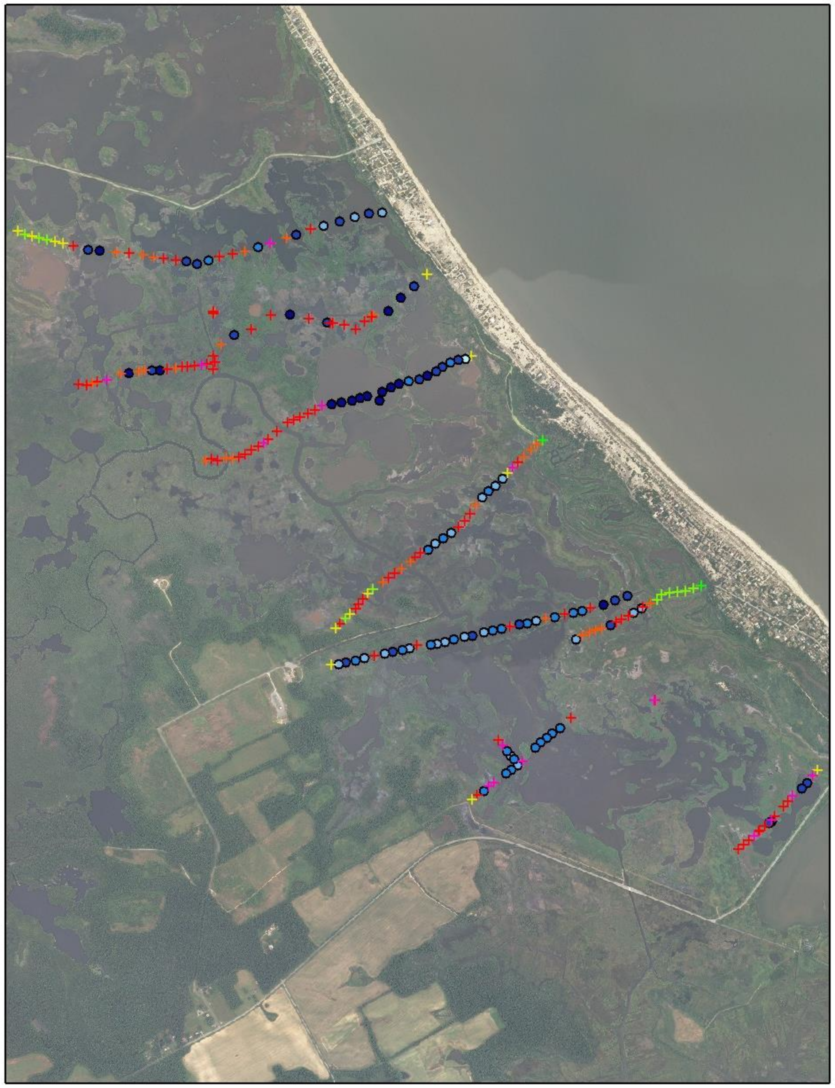
Legend

Unit 2 Wetland Elevation (m NAVD 88)	Unit 2 Bottom Elevation (m NAVD 88)
0.250 - 0.349	-0.901 - -0.065
0.350 - 0.392	-0.064 - 0.089
0.393 - 0.441	0.090 - 0.176
0.442 - 0.503	0.177 - 0.277
0.504 - 0.556	0.278 - 0.418

Wetland Elevation:	Bottom Elevation:
Mean: 0.407 m	Mean: 0.140 m
Min: 0.250 m	Min: -0.901 m
Max: 0.556 m	Max: 0.418 m

N
W E
S

410 205 0 410 Meters



Unit 3 Elevations

Primehook National Wildlife Refuge

Legend

Unit 3 Wetlands Elevation (m NAVD 88)	Unit 3 Bottom Elevation (m NAVD 88)
+ -0.074 - 0.250	● -0.901 - -0.065
+ 0.251 - 0.349	● -0.064 - 0.089
+ 0.350 - 0.392	● 0.090 - 0.176
+ 0.393 - 0.441	● 0.177 - 0.277
+ 0.442 - 0.503	● 0.278 - 0.418
+ 0.504 - 0.555	

Wetland Elevation:	Bottom Elevation:
Mean: 0.332 m	Mean: 0.102 m
Min: 0.155 m	Min: -0.307 m
Max: 0.555 m	Max: 0.257 m

N
W E
S

640 320 0 640 Meters

Delaware Coastal Programs
DCMP