ABSTRACT

Title of Dissertation:	ECOSYSTEM DYNAMICS IN TIDAL MARSHES CONSTRUCTED WITH FINE GRAINED, NUTRIENT RICH DREDGED MATERIAL
	Lorie Winchell Staver, Doctor of Philosophy, 2015
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An ecological study was undertaken of the tidal marshes at Poplar Island, a restoration project utilizing fine grained dredged material from the shipping channels in upper Chesapeake Bay. The goals of the study were to examine the effect of a nutrient rich substrate on vegetation development, elevation change, and nutrient cycling in the constructed tidal marshes. Specifically, I examined macrophyte production, nitrogen (N), carbon (C) and silicon (Si) budgets, as well as the success of a silica amendment in enriching plant tissue concentrations. Establishment of *Spartina alterniflora* and *S. patens* on fine grained dredged material was rapid, and growth peaked in the second year. Thereafter *S. alterniflora* dieback occurred sporadically during the growing season, but the causes remain unclear. Elevation change averaged 7.9 ±0.8 mm y^{-1} in the dredged material marshes (low marsh only), compared to 7.4 ±1.4 mm y^{-1} in a low nutrient onsite reference marsh. Elevation change was significantly correlated with biomass production suggesting that inputs of organic matter from high rates of aboveground biomass production on nutrient rich dredged material offset the reduced contribution of belowground biomass to vertical accretion. However, dieback may have a detrimental effect on vertical accretion, which is essential for keeping up with apparent sea level rise (13.6 and 11.0 mm y^{-1} at Baltimore and Solomon's tide gauges, respectively) since elevation monitoring began at Poplar Island. The tidal flux study revealed that the marsh exports ~665 kg of N y^{-1}, including 100 kg NH_4^+-N y^{-1}, and 67,874 kg y^{-1} TSS, and imports 35 kg NO_3-N y^{-1}. Silicon is also exported on both a seasonal and annual basis, including 4,337 kg dissolved Si y^{-1} and 3,924 kg biogenic Si y^{-1}, with highest exports in July, an overlooked benefit of dredged material restoration projects. Soil Si amendments increased plant tissue concentrations significantly, but this study did not show increased resistance to N related stress effects on the vegetation. Overall, this study suggests that when considering trajectories of vegetation development, nutrient exchanges and elevation change in constructed marshes, it is essential to consider the initial nitrogen content of the substrate.

ECOSYSTEM DYNAMICS IN TIDAL MARSHES CONSTRUCTED WITH FINE GRAINED, NUTRIENT RICH DREDGED MATERIAL

by

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Dissertation submitted to the Faculty of the Graduate School of the
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Preface

In a comprehensive study of the rain forest at El Verde in Puerto Rico directed by H.T. Odum, he asserted that one way to learn about ecosystem response to stress is to look at the extremes – in his case a high diversity tropical cloud forest in Puerto Rico exposed to a high level radiation source. His student, Scott Nixon took a much more modest approach in a study of low diversity salt pans, aquatic ecosystems with salinities at extreme levels, upwards of 60 ppt. Bradshaw (1984) insisted that restoring an ecosystem from the ground up is "the acid test" of our understanding concerning critical system functions, and that getting the soil substrate right is a fundamental step in restoration. In my study, both perspectives were combined, using tidal marsh systems, "created" from the ground up using nutrient rich material dredged from the shipping channels in upper Chesapeake Bay, to address a variety of topics related to tidal marshes.

The marsh systems studied are part of a larger project, the Paul S. Sarbanes Ecosystem Restoration Project at Poplar Island. These marshes are extreme in several respects. They are completely engineered, beginning with bay bottom sediments that were once upland soils. Each of the marshes at Poplar Island has been filled with this material dredged from the shipping channels in upper Chesapeake Bay, graded and planted primarily with two species of marsh vegetation (*Spartina alterniflora* Loisel and *S. patens* (Aiton) Muhl.). The result is marshes that initially are quite homogeneous in terms of sediment and vegetation, which is unique compared to mature natural marshes at this salinity range which are more diverse floristically (and faunistically), with well-developed soils (i.e. sediment profiles). Furthermore, the plant available nitrogen content of the dredged material used as a substrate represents an extreme level of N fertilization, on par with the most productive agricultural rice ecosystems which are often triple cropped annually. The sediment N reservoir is enormous and the supply is continuous. In addition, the series of marshes have been created sequentially, using similar substrates, design and planting techniques and plant material, allowing consideration of trajectories, or successional patterns which have been little studied.

The project affords the opportunity, therefore, to quantify the success of the engineering design in producing marshes that are self-sustaining and resilient to sea level rise, a test of our overall understanding of the processes that lead to resiliency. The pace of ecosystem development toward equivalency with natural marshes in terms of function and complexity can be monitored, and compared with similar projects that utilized low nutrient material as a substrate such as those studied by Craft et al. (2002). This yields not only insights into the appropriateness of using this dredged material to build tidal marshes, but also provides critical information relevant to the response of natural marshes to long term nutrient enrichment. We can study the exchange of carbon and nutrients between the marshes and the estuary to learn how these engineered projects affect the local carbon and nitrogen balances. In addition, we can consider how constructed marshes conform to tidal marsh paradigms, including one raised by Childers et al. (2000) that young marshes with small nutrient pools import nutrients while older marshes in which the pools have built up tend to export, and another that tidal marshes act as sediment traps (Stumpf 1983).

An exhaustive study on the scale of the Poplar Island project is logistically daunting and involves numerous researchers. Through the efforts of a long term monitoring program there are now sufficient data to describe and begin to quantify how these tidal marsh ecosystems develop. Furthermore, using comparisons with natural tidal marshes, their long-term resiliency to sea level rise as well as how they interact with the adjacent estuary can be assessed. The portion of the monitoring program presented here includes primarily vegetation, elevation, and nutrient (including silica) monitoring. The detailed results of geochemical sediment monitoring and on initial plant establishment will be reported elsewhere, and the ecology of birds, fish, turtles and other fauna on Poplar Island will be hopefully published by other groups. All the data is currently available in annual reports submitted to the Maryland Environmental Service, United States Army Corps of Engineers and the Port of Baltimore.

In this dissertation, I report on the development of vegetation in the Poplar Island marshes with relevant data from sediment monitoring (Chapter 1), elevation change (Chapter 2), the role of silicon in fine grained dredged material marshes (Chapter 3), and the nitrogen and carbon budgets for one marsh (Chapter 4). Through this study of the ecology of the Poplar Island marshes, we put both Nixon's and Bradshaw's recommendations (as well as H.T. and E.P. Odum's concepts) to work in an attempt to elucidate some new information relevant to ecological restoration and to more recent debates surrounding nutrient enrichment (e.g. Deegan et al. 2012; Duarte et al. 2009), vertical accretion (e.g. Stevenson and Kearney 2009), and silicon (e.g. Struyf and Conley 2009) in tidal marshes.

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There are many individuals to acknowledge for their contributions to the work on the Poplar Island project that is the basis for this dissertation, and many who have supported my studies there. First among them is my adviser, J. Court Stevenson. Although my enthusiasm for marsh work was somewhat tepid at first, his interest in marshes was infectious, and his enthusiasm for field work inspiring. In addition, he allowed me the freedom to pursue my studies while balancing work and family life, for which I am grateful. Finally, he encouraged me to attend conferences and present the work, meet others working in marshes and marsh restoration, and put the Poplar Island project in a wider context. I also thank Court, as well as his wife Catherine, for their friendship over many years.

Another key individual, who has been supportive in so many ways, is Jeff Cornwell. Jeff and his collaborator Mike Owens are responsible for the sediment work on the Poplar Island project, and have shared that data with me in support of the vegetation work presented here. Jeff provided many helpful suggestions and ideas, and always provided positive reinforcement.

My other committee members, Walter Boynton, Pat Kangas, Eva Maria Koch and Mike Kearney were generous with ideas during committee meetings, always reminding me to keep an ecological perspective. Sadly, Eva passed away before I finished, after having encouraged me early on to pursue a PhD. She is missed.

Although not a committee member, Nick Nidzieko was a collaborator on the tidal flux portion of this work, and provided the water volume fluxes for calculating constituent fluxes. I am grateful for his efforts and for his patience in working with me.

Daniel Conley and Joanna Carey provided methods and standards that were needed to set up silicon analysis at Horn Point Laboratory for the work in Chapter 3, and Lawrence Datnoff provided some early sample analysis as well as advice on setting up the silicon field trial. Wade Elmer has also been a collaborator, especially on the silicon work, performing fungal analyses on plant samples.

Dozens of students and staff have helped with field and lab work on the Poplar Island project, and although I will not attempt to name all of them their help has been invaluable. A few key individuals must be named, however. Philippe Hensel, a fellow MEES graduate and currently at NOAA NGS, collaborated with us on SET installations, measurements, and data analysis. He always makes time to work with us despite his busy schedule, and his collaboration was essential in the elevation work described in Chapter 2. Debbie Hinkle from Horn Point Laboratory (HPL), has provided field and laboratory assistance throughout all phases of this study, and without her many tasks would not have been accomplished on time. Richie Long, also from HPL, has helped with equipment installation and biomass harvesting numerous times, and Daniel Stevenson assisted with field work on many occasions, and their help is much appreciated.

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The Maryland Environmental Service (MES) staff at Poplar Island has facilitated all of our field work, and although there have been personnel changes over the years several key individuals, Michelle Osborne, Claire Ewing and Alexa Boos, have gone above and beyond to make sure field work goes smoothly and to accommodate our needs on the island. Likewise, Carolyn Blakeney has been very supportive at MES headquarters.

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Chapter 1: Vegetation dynamics in nutrient rich tidal marshes constructed with fine grained dredged material

Introduction

Nitrogen in constructed marshes

The effects of nitrogen (N) enrichment on natural tidal marshes have been the focus of studies for over 40 years (e.g. Valiela et al. 1975). Responses to N fertilization of marshes include increased shoot production (Broome et al. 1975; Valiela et al.1975) and, frequently, decreased root production (Valiela et al. 1976; Smart and Barko 1978), leading to shifts in root:shoot ratios (Turner et al. 2004; Darby and Turner 2008a). Long term marsh fertilization can reduce soil integrity (Turner 2011), and increase erodibility (Deegan et al. 2012), potentially compromising the resiliency of the marsh to physical disturbance. Furthermore, the ability of N-enriched marshes to keep abreast of sea-level rise may also be compromised by reductions in belowground biomass, which can contribute significantly to elevation gain (Nyman et al. 2006; Langley et al. 2009), and to increased rates of decomposition (Morris and Bradley 1999; Wigand et al. 2009). Thus, N enrichment is currently considered one of the most important threats, along with sea level rise (SLR), to tidal marshes.

Many aspects of constructed marshes have been studied during the same period, but N enrichment has not been considered a threat to successful establishment or long term sustainability. In contrast to natural marshes, many constructed marshes have suffered from an N deficit due to the low nutrient content of the substrate (Langis et al. 1991). Nitrogen fertilization at planting has been recommended for successful establishment (Broome et al. 1975), and in some cases supplemental fertilization thereafter (Zedler and Callaway 1999). Concerns in these systems assume (and demonstrate) an initially lower level of function in comparison to natural marshes, with functionality rising toward levels of natural reference systems over time (Bradshaw 1984; Kentula et al. 1992; Zedler and Callaway 1999).

However, there is conflicting evidence for this type of trajectory toward equivalency in a number of functions, including biomass production. In most examples production has started out below reference marsh levels and increased slowly, over time scales of years to decades (Craft et al. 2002; Craft et al. 2003). In some equivalency has not been reached and trajectories toward equivalency have not emerged nearly a decade or more after establishment (Simenstad and Thom 1996; Zedler and Callaway 1999; Streever 2000). A nitrogen deficit has been implicated in the failure of these systems to develop as expected (Zedler and Callaway 1999) due to the use of low nutrient substrates dominated by sand (Costa-Pierce and Weinstein 2002). Few studies have addressed the reverse, marshes constructed with a substrate that contains a high percentage of silts and clays and significantly more N than most natural marshes. Streever (2000) finds some evidence of an alternate trajectory in dredged material marshes, with a flush of very high biomass production initially followed by a decline toward levels of natural marshes. Yet the initial N supply of the sediment is not considered as a determining factor of marsh trajectories. In their volume on wetland restoration, Kentula et al. (1992) devote only a page and a half to soil considerations, and dwell mainly on soil carbon. Given the importance of N in the ecology of natural wetlands and considering the initial sediment N supplies in constructed wetlands may help explain the lack of consistency in trajectories of a number of functions (Simenstad and Thom 1996; Streever 2000). Although replicating the energy signature is considered the most important factor in restoration ecology (Kangas 2003), especially tidal energy and sunlight in the case of tidal wetlands, soil N may be underappreciated as a driver of development trajectories.

In a large scale project in Chesapeake Bay, the Paul S. Sarbanes Ecosystem Restoration Project at Poplar Island (Poplar Island), tidal marshes are being constructed using fine grained, nutrient rich material dredged from the shipping channels approaching Baltimore Harbor, in upper Chesapeake Bay (Figure 1-1). This material has average in situ porewater and adsorbed ammonium (NH_4^+) concentrations of 5 mmol L^{-1} and 11 μmol g^{-1}, respectively (Cornwell and Owens 2011). While much of this N load is lost during dredging and placement activities leading up to marsh establishment, initial porewater NH_4^+ concentrations remain extremely high compared to many other reported marsh restoration projects, up to 2 mmol L^{-1} (Stevenson et al. 2007). Thus the Poplar Island project provides the opportunity to examine the effects of an N rich substrate on trajectories of tidal marsh development.

Fine grained dredged material that contains high concentrations of pyrite has the potential to acidify when the pyrite oxidizes during dewatering (Fanning and Fanning 1989; Demas et al. 2004; Cappuyns et al. 2006), which is part of the management process prior to grading and planting at Poplar Island. Furthermore, fine grained marsh soils with high clay contents (>8%) are associated with high porosity but low advective flow (Bradley and Morris 1990). Transport is primarily diffusive in these soils, potentially reducing soil oxygenation and allowing phytotoxins to accumulate. Thus, in addition to the N load, the upper Bay dredged material presents potential challenges to plant establishment and growth related to acidity, soil oxygenation and accumulation of phytotoxins.

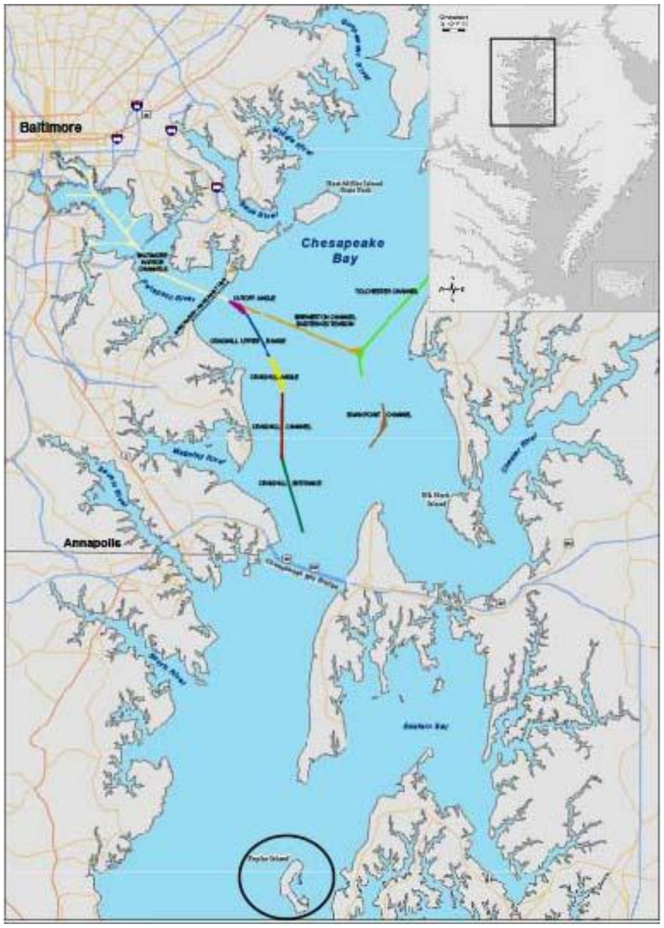

Figure 1-1. Location of Paul S. Sarbanes Ecosystem Restoration Project at Poplar Island. Shipping channels approaching Baltimore Harbor (shown in color) are the source of dredged material for the Poplar Island project. Tracey Saxby, Kate Boicourt, IAN Image Library.

Studies at Poplar Island began in 2003 in small test plots in Cell 4D, a pilot marsh constructed primarily with sand to develop optimal design and planting methods. The experimental plots contained a thin (30 cm) layer of dredged material overlying the sand (Mielcarek 2006). The initial work demonstrated that plant growth on upper-Bay dredged material was luxuriant in the first two years, similar to the trajectory described by Streever (2000).

This work also indicated that low pH was unlikely to pose an impediment to plant establishment or growth, probably because opening the marshes to tidal exchange several months before planting ameliorated sediment acidification (Cornwell et al. 2005). Observations suggested, however, that other problems could develop in the future dredged material marshes, including lodging (the collapse of weak-stemmed plants during mid-season), intense grazing by muskrats (*Ondatra zibethicus*), and *S. alterniflora* dieback in the low marsh.

Nevertheless, based on the preliminary work the subsequent dredged material marshes were expected to become established quickly, with initial macrophyte production exceeding natural marshes in the mid-Atlantic region due to the abundance of plant available N, but which would decline to the levels of natural marshes in response to a diminishing N supply over time. This trajectory would contrast with results in marshes constructed on low nutrient substrates, where it took a decade or more to approach production levels of natural marshes, if at all (Craft et al. 2002). We use monitoring data from these dredged material marshes to address the effect of N rich dredged material on vegetation establishment and growth in constructed marshes.

Hypotheses

Using a series of constructed marshes at Poplar Island, the following hypotheses were formulated: (a) fine-grained, nutrient-rich dredged material from Upper Chesapeake Bay can be used as a substrate to rapidly establish tidal marsh vegetation; and (b) macrophyte production in these marshes would initially be high compared to natural marshes and to marshes constructed on low N substrates, and would decline toward levels commensurate with natural marshes in response to a diminishing sediment N supply. Both may be critical to the long term sustainability of these marshes given the rapid historical rate of sea level rise (SLR) in Chesapeake Bay (Kearney and Stevenson 1991), and especially in view of recent projections of accelerating SLR (Rahmstorf et al. 2012). When vegetation dieback developed in the dredged material marshes, sediment porewater characteristics and transplant success in dieback compared to healthy areas was also examined, in order to assess the role of biogeochemical processes (e.g. sulfide toxicity) in dieback.

Research Approach

This work is part of a comprehensive monitoring program which is intended to provide feedback for an adaptive management plan and evaluation of success in meeting project goals. As such, sampling strategies changed over the years as new questions arose. Here we provide a summary of data that focuses on approximately ten years of vegetation development in Cell 3D, the first marsh constructed (in 2005) with upper Chesapeake Bay dredged material. These results are compared with Cell 4D, constructed primarily with a low nutrient sand substrate in 2003, excluding the experimental dredged material plots.

An initial sediment characterization in 2005 in Cell 3D was followed by annual monitoring of sediment characteristics including porewater nutrient concentrations and solid phase characteristics, and plant attributes including net annual aboveground (AG) and belowground (BG) biomass production (measured at the end of the growing season), plant coverage and height, and plant tissue nutrient contents. Results from Cell 3D are compared to annual vegetation monitoring and less frequent sediment monitoring data from Cell 4D which began later (2008), after vegetation dieback became apparent in the low marsh of Cell 3D and the value of monitoring Cell 4D as an on-site reference was recognized. Some supporting data from other dredged material marshes at Poplar Island planted after Cell 3D (Table 1-1), and a small natural marsh at Horn Point Laboratory (Cahoon and Stevenson 1986) are presented to compare a mature natural marsh with the emerging trajectories of constructed marshes of different ages.

Table 1-1. Tidal marsh containment cells at Poplar Island.

Cell	Year Planted	Primary Substrate	Area (ha)
4D	2003	sand	10.1
3D	2005	Upper Bay Dredged Material	12.5
1A	2009	Upper Bay Dredged Material	15.4
1C	2011	Upper Bay Dredged Material	17.8
1B	2012	Upper Bay Dredged Material	15.4

Methods

Site description

Poplar Island is located approximately 25 km SSE of Annapolis, Maryland (Figure1-1). It comprised approximately 463 ha of upland and wetland habitat in 1847, but by 1990 erosion had reduced the island to a few small remnants with a total area of about 1.6 ha. The Poplar Island project was conceived as a beneficial use project to replace the original island habitat lost to erosion, as well as to provide a placement site for dredged material resulting from maintenance dredging of the upper Chesapeake Bay shipping channels.

Currently a perimeter dike 3.0 m above mean low tide, armored with rock riprap, surrounds 461 ha, creating an island 4.3 km long and 0.8 km wide approximating the 1847 footprint of the original island (Figure 1-2). The interior has been subdivided into 17 containment cells where dredged material from upper Chesapeake Bay shipping channels can be deposited. The containment cells on the eastern side of the island are being developed as tidal marshes; those on the western side will eventually be upland habitat. To date approximately 71 ha of tidal marsh has been completed. Each marsh is comprised of 20% high marsh (elevation 0.47-0.68 m above mean lower low water, MLLW), planted with *Spartina patens* and 80% low marsh (elevation 0.32-0.47 m above MLLW), planted with *S. alterniflora*, primarily using 3.75 cm nursery grown transplant plugs. The marshes are connected to Chesapeake Bay via tidal inlet structures through the eastern perimeter dike. The inlet structure in Cells 4D and 3D consisted of corrugated steel pipes, while concrete box culverts (1.8 x 1.8 m) were used in all other wetland cells.

Dredged material placement is planned to accommodate de-watering and compaction prior to grading to achieve target elevations, but sand is sometimes added at the final grading to make up small deficits. Dewatering occurs over a period of 1 - 5 years (e.g. 1 year in Cell 3D), encouraged by a network of drainage ditches, until the dredged material is desiccated and can support heavy track vehicles used for grading, a process referred to as crust management. Inlet structures are opened to tidal exchange approximately six months prior to planting, exposing the marsh sediment to inundation with mesohaline Bay water (average salinity range 9 to 15 based on measurements at nearby DNR buoy CB4.2C). The overall management plan for the site requires annual vegetation and sediment monitoring for five years following planting (2005-2010 in Cell 3D) to evaluate success in achieving project goals (85% vegetation coverage in the tidal marshes. Less frequent and intensive monitoring occurred in Cell 3D thereafter (2011-2014).

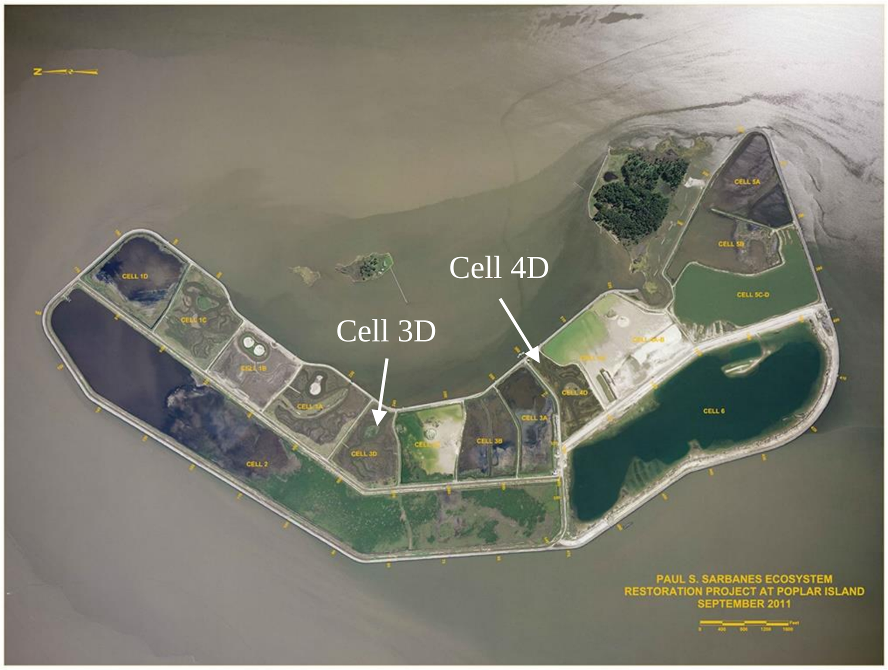

Figure 1-2. Aerial photograph of Poplar Island (September 2011), showing the location of developed marsh cells, highlighting Cells 3D and 4D, the focus of this study (USACE).

Vegetation

Stem height and Percent Cover

Permanent transects were established in each cell running from the upland dike to the creek bank, perpendicular to the high marsh-low marsh zonation. Transects were established in Cell 3D in 2005 following planting, and in Cell 4D in 2008, five years post planting. Three permanent points located along each transect, one in the high marsh, one in the low marsh and one at the creek edge, were monitored in autumn thereafter (usually October) for vegetative cover and stem height. Cover was estimated visually using a 0.25

m^2 PVC quadrat with a 10 x10 cm grid and counting the number of grid cells containing rooted plant material. Canopy height was calculated as the mean of the five tallest culms within 1 meter of the sample point.

Biomass

Aboveground (AG) biomass samples were collected using 0.25 m^2 PVC quadrats at the conclusion of the growing season (usually October) beginning in 2006 in Cell 3D, the year following planting, and in 2008 in Cell 4D (with one set in 2004). Samples were separated into dead and living biomass prior to being dried at 60°C in a forced draft oven (Grieve Model 343) and weighed to the nearest mg on a top-loading balance (O'haus 1500D). One belowground biomass (BG) sample was collected from each AG biomass quadrat using a specially fabricated stainless steel piston corer. Sediment cores were washed free of sediment over a 1 mm mesh sieve before being dried and weighed as described above. In 2010 the depth distribution of belowground biomass was evaluated by sectioning BG biomass cores into 2 cm increments and processing each section separately as described above.

Plant Spread

In order to assess optimal spacing for transplanting vegetation plugs, the vegetative spread of *S. alterniflora* during the first growing season was surveyed in Cell 1B. The distance from what appeared to be the most distant shoot originating from an individual transplant to the center of the clump was recorded for 100 clumps.

Transplant experiment

In 2008, lingering effects from the 2007 vegetation dieback in the northeast corner of Cell 3D were tested using three randomly selected 1.0 m^2 plots in the dieback area, by planting them with 3.75 cm nursery grown *S. alterniflora* plugs. Nine plugs per plot were planted on 0.5 m centers, and at the end of the growing season surviving plugs were counted in each plot.

Plant tissue nutrients

Plant samples used for tissue nutrient analysis consisted of leaves (2005) or whole culms (all subsequent years) or whole root/rhizome samples. Dried samples were ground in a Thomas-Wiley Intermediate Mill or a Wiley Model 4 Mill (20 mesh), and homogenized. One gram subsamples were sent to A&L Eastern Laboratories, Inc. (Richmond, VA) for analysis of: nitrogen (N), sulfur (S), phosphorus (P), potassium (K), magnesium (Mg), calcium (Ca), sodium (Na), iron (Fe), aluminum (Al), manganese (Mn), boron (B), copper (Cu), and zinc (Zn). Carbon analysis was carried out on an Exeter Analytical Inc. CE-440 Elemental Analyzer.

Sediment

Solid Phase Chemistry

The chemistry of sediment solids was measured at 104 sites in Cell 3D in 2005 by J. Cornwell and M. Owens (HPL Biogeochemistry Lab). From 2006-2010 five sites were intensively sampled, with 5 replicate samples at each site each year.

Iron monosulfides (FeS or AVS) and pyrite-sulfur (FeS_2) were measured using potentiometric detection of hydrogen sulfide after extraction with hydrochloric acid and acidic chromium (II), respectively (Cornwell and Morse 1987). Carbon-hydrogen-nitrogen (CHN) analysis (Exeter Analytical Inc. CE-440 Elemental Analyzer) was used to determine total sediment N and carbon (C) content. Total and inorganic P analysis followed (Aspila et al. 1976) and HCl-extractable Fe was analyzed after acid extraction using atomic absorption spectrophotometry for the final analysis (Leventhal and Taylor 1990). Loss on ignition was determined by loss of mass after combustion at 550°C. Sediment pH was measured using a hand held YSI pH meter.

Deep core sampling was carried out in 2010 (HPL Biogeochemistry Lab) to examine deep sediment concentrations of NH_4^+, particularly the adsorbed (KCl-extractable) NH_4^+ component (Mackin and Aller 1984, Cornwell and Owens 2011). Cores were obtained by vibracoring at four standard Cell 3D sampling sites and four at the newly-established Cell 1A. The longest core was approximately 1 m in length.

Sediment Pore Water

In 2005 sediment porewater was sampled at 32 stations (HPL Biogeochemistry Lab) using multi-chambered equilibrium dialysis samplers (Hesslein 1976). Porewater was sampled at a subset of the original 32 sites (12 each time) three times in 2006 and 2007 (May, July and October), and during autumn 2008, August 2009 and November 2010. In 2005 a pair of samplers was also deployed in the marsh near Horn Point Laboratory described in Cahoon and Stevenson (1986), one each in the high and low marsh. In 2008 porewater sampling in Cell 3D focused on assessing differences in nutrient and sulfide concentrations between an area which experienced vegetation dieback in 2007 in the NE corner of the cell and an unaffected area on the west side of the cell.

Porewater sampling in Cell 4D occurred in 2003 and 2004. Data reported by Cornwell et al. (2005) and Mielcarek (2006) are used here to characterize the nutrient status of the Cell 4D substrate, as well as the natural marsh at Horn Point Laboratory.

Porewater concentrations of NH_4^+, soluble reactive phosphorus (SRP), S (sulfate, SO_4^-, and hydrogen sulfide, H_2S), Fe, chloride (Cl^-) and pH were determined, using standard methods (Cornwell 1999; Cornwell and Owens 2011). Standard colorimetric analyses were used for the analysis of SRP, NH_4^+, H_2S (Parsons et al. 1984) and Fe (Gibbs 1979). Ion chromatography was used for the analysis of major anions (SO_4^- and Cl^-) and direct insertion of electrodes was used to measure pH. The sediments at the sampling sites were very compacted, so prior to insertion of the PVC pore water sampler a metal pilot was pounded into the substrate to avoid tearing membranes on the samplers.

Results

Vegetation

Transects

After planting in late spring 2005 vegetation establishment was rapid in Cell 3D, with average vegetation coverage in the high and low marsh reaching 100% and 80%, respectively, by 2006 (figure 1-4). In 2006, however, large areas of *S. alterniflora* in Cell 3D lodged in mid-summer (Figure 1-3), and small areas of vegetation dieback became apparent in the low marsh. By 2007 larger areas were affected, especially the northeast corner of the cell (Figure 1-3). Dieback was not observed in the same locations each year, but was observed to repeat in some locations after several years of recovery. It was initially observed as yellowing vegetation during the growing season, followed by mortality, and frequently little or no vegetation emergence the following spring (Figure 1-3). Neither lodging nor extensive dieback was observed in the sand substrate marshes (Cell 4D). The effect of vegetation dieback in the low marsh of Cell 3D is evident in the more variable mean annual cover and canopy height in the low marsh compared to the high marsh, and in Cell 3D compared to Cell 4D (Figure 1-4).

Figure 1-3. *Spartina alterniflora* lodging in Cell 3D, July 2006 (a); September 2007 aerial photograph of Cell 3D showing large area of dieback (b) and ground photo of dieback area in Cell 3D, September 2007 (c). Arrow in (b) highlights a large dieback area in the northeast corner of Cell 3D.

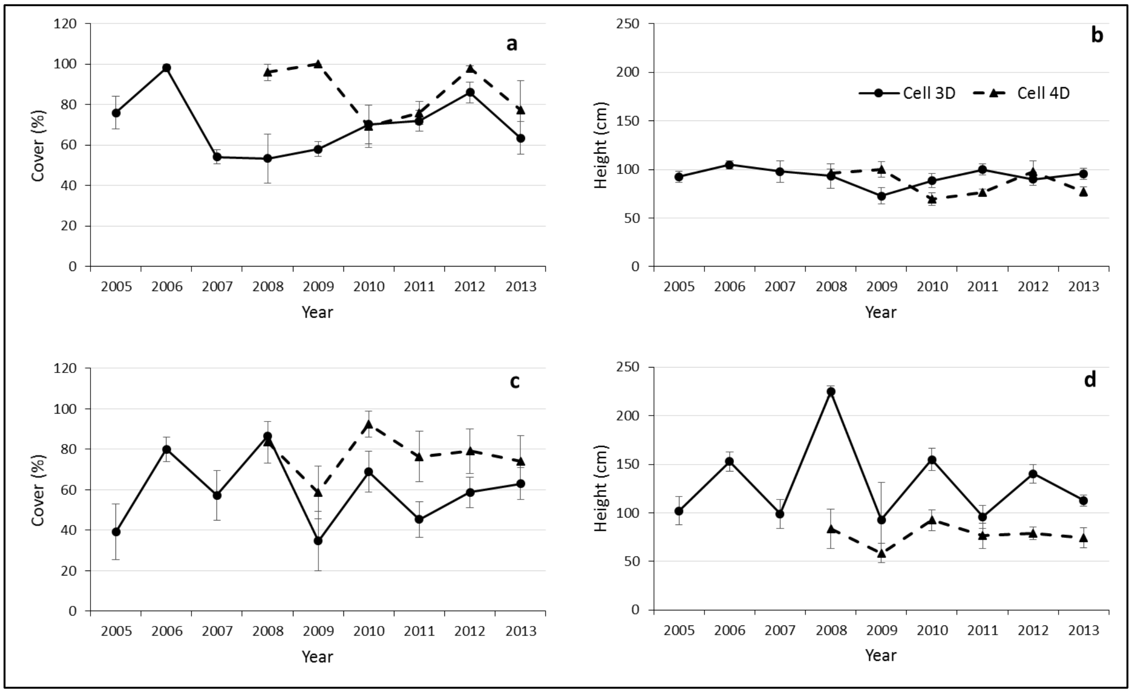

Figure 1-4. Time series of mean vegetation cover (a, c) and canopy height (b, d) in the high marsh (a, b) and low marsh (c, d) of Cells 3D (dredged material) and 4D (sand) from annual vegetation surveys. Error bars represent standard error.

Aboveground Biomass

Average AG biomass production in Cell 3D at Poplar Island peaked in 2006, the second year after planting, in both high (2761 ±233 g dry weight m^{-2}) and low marsh (1934 ±226 g dry weight m^{-2}). There was an overall downward trend thereafter, punctuated by *S. alterniflora* dieback events in 2007 (not at sampling sites), 2009 and 2011 (Figure 1-5). In Cell 4D, AG biomass production in the high marsh showed only a slight change (606 – 675 g m^{-2}) between the initial sampling in 2004 and the next sampling in 2008, and then an erratic but upward trend from 2008 – 2014 (Figure 1-5a). In the low marsh of Cell 4D AG biomass more than doubled (416 to 1164 g m^{-2}) from 2004 to 2008, and reached a peak in 2011 (Figure 1-5b). In 2012 there was about a 50% decline from a high in 2011 of 1463 ±135 g m^{-2} to 756 ±105 in 2012, and no recovery to previous levels in 2013 or 2014 (Figure 1-5b). High marsh AG biomass in Cell 3D exceeded Cell 4D until 2012, after which both cells were within the same range (Figure 1-5a). Low marsh AG biomass showed overlap between Cells 3D and 4D after 2007, largely due to the effect of dieback in Cell 3D (Figure 1-5b).

During the period 2008-2014, total biomass in the Cell 3D high marsh usually exceeded the low marsh. This resulted from an accumulation of dead biomass in the high marsh compared to the low marsh in most years (Appendix 1). In Cell 4D total high and low marsh biomass were similar in magnitude during the 2008-2014 period, and the difference in the proportion of live and dead material between high and low marsh was not as consistent or pronounced as in Cell 3D (Appendix 1).

In Cell 3D, an extensive low marsh dieback occurred in 2007 (which was observed but did not occur at the sampling sites), followed by a recovery period in 2008, and repeat diebacks in 2009 and 2011, followed by recovery in 2013-2014 (Figure 1-5). High marsh vegetation did not appear to be affected by dieback events. In Cell 4D both high and low marsh production tended to be more consistent, ranging between approximately 750 and 1250 g dry weight m^{-2} y^{-1}. There was an apparent shift to lower AG production in the low marsh in 2012, with production prior to 2012 ranging from 1200 to 1500 g dry weight m^{-2} y^{-1} and a reduction to 500-800 g dry weight m^{-2} y^{-1} from 2012 to 2014. Dieback was not observed in Cell 4D in association with the 2012 decline.

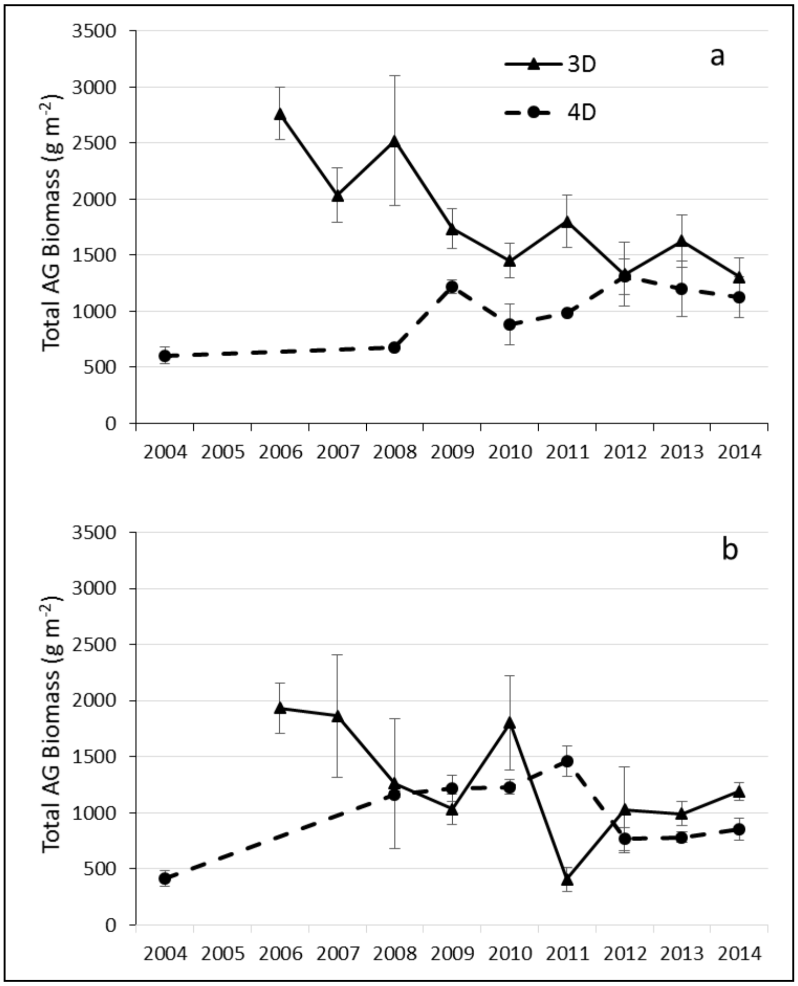

Figure 1-5. Time series of mean high marsh (a) and low marsh (b) total aboveground biomass (±SE) in Cell 3D (dredged material, planted 2005l) and Cell 4D (sand, planted 2003), 2004-2014.

Compared to biomass values reported for natural marshes in the mid-Atlantic region (New Jersey to Virginia), the average total aboveground biomass the second year after planting is on the low end of the range in Cell 4D, while the Cell 3D value is the highest reported (Table 1-2).

Table 1-2. End of season aboveground biomass from S. alterniflora marshes in the mid-Atlantic region. * denotes studies cited in (Turner 1976), ǂ denotes studies cited in Kirwan et al. (2013), PI Cell 4D and 3D denote Poplar Island low nutrient and dredged material.

Location	**Live AG Biomass (g dry wt. m^{-2})**	**Dead AG Biomass (g dry wt. m^{-2})**	**Total AG Biomass (g dry wt. m^{-2})**	**Reference**
VA			1140	Wass and Wright 1969*
VA	363	94	459	Mendelsohn 1973*
MD-VA	506	150-280	608-774	Keefe and Boynton 1973*
MD	471			Cahoon 1975*
DE	385-413	187	572-600	Morgan 1961*
NJ			300	Good 1965*
NJ	592-1592	91-110	682-1702	Squiers and Good 1974*
VA	350-375			Mendelsohn and Marcellus 1976[ǂ]
VA	398			Gross et al. 1991[ǂ]
VA	609			Reidenbaugh 1983[ǂ]
DE	356			Gross et al. 1991[ǂ]
NJ	167			Good 1965[ǂ]
MD			324	Stribling 1989
MD			359	Stribling and Cornwell 1997
MD			429 - 525	Smith et al. 1979
PI Cell 4D (2004)			416	this study
PI Cell 3D (2006)			1934	this study

Belowground Biomass

Total belowground biomass from 2008-2014 was consistently higher in the sandy substrate of Cell 4D than in the dredged material of Cell 3D in both high marsh (Figure 1-6a) and low marsh (Figure 1-6b). Belowground biomass in Cell 4D showed a higher than expected level of inter-annual variability. Production varied by a factor of 2.5 in the high marsh (Figure 1-6a) and 1.7 in the low marsh (Figure 1-6b). Root/rhizome distribution in both cells was concentrated in the top 15 cm of the substrate (Figure 1-7), although in most cores from both cells there were roots and/or rhizomes present throughout the entire depth profile to 24 cm.

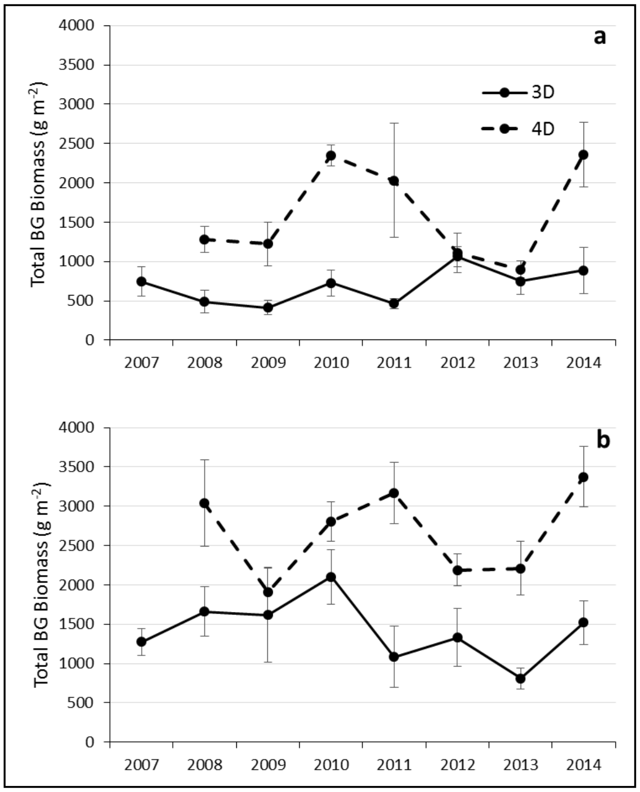

Figure 1-6. Time series of mean high marsh (a) and low marsh (b) total belowground biomass (± SE) in Cells 3D (dredged material, planted 2005) and 4D (sand, planted 2003), 2007-2014.

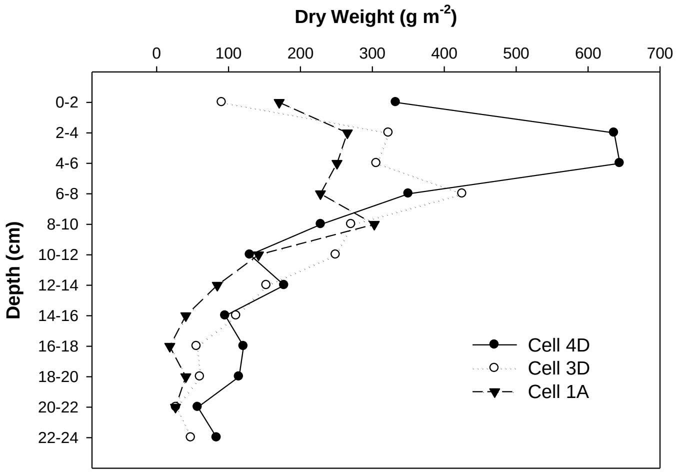

Figure 1-7. S. alterniflora mean (n=3) belowground biomass depth distribution in Cell 4D (sand) and Cells 3D and 1A (dredged material), in 2010. Cells 3D and 1A were 5 years and 1 year since planting, respectively.

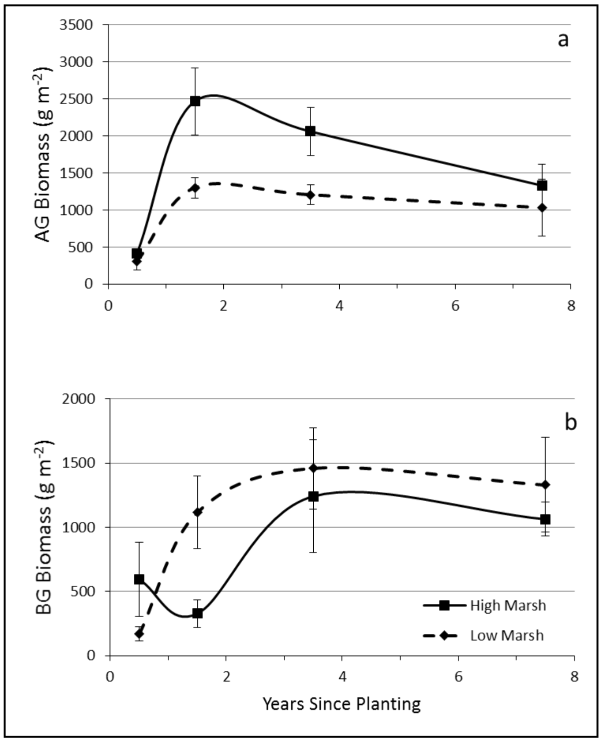

Figure 1-8. Performance curves of total aboveground (a) and belowground (b) biomass (±SE, n = 6) in the dredged material marshes at Poplar Island, 2012.

In the early years after planting before *S. alterniflora* dieback was widespread, the ratio of total belowground to total aboveground biomass, the root: shoot ratio (RSR), was very low in Cell 3D due to disproportionate shoot growth. In 2007, for example, in Cell 3D RSR was 0.68 versus 2.61 in Cell 4D. In subsequent years, dieback in the low marsh produced highly variable RSR as shoot vegetation died and decomposed but root tissue persisted in the anaerobic sediment. In non-dieback years RSR was 2-3 times higher in Cell 4D than 3D over the 2008-2014 period (Appendix 1).

Plant spread

In the December 2012 survey of the most distant shoot emerging from 100 *S. alterniflora* clumps in the low marsh of a newly planted dredged material marsh, Cell 1B, the average (±SE) distance to the center of the clump was 46 ± 0.59 cm. Nursery grown plugs were planted on 1 meter centers, so the youngest shoots from neighboring clumps were poised to completely fill in the bare sediment between transplants the following growing season. When *S. alterniflora* shoots emerged at the beginning of the second growing season (i.e. 2013, one year after planting) the individual clumps planted the previous year were barely discernable.

Transplant experiment

Spartina alterniflora plugs planted into 1 m^2 plots in the NE corner of Cell 3D in 2008 had a survival rate of 100%, indicating an absence of persistent phytotoxins in the sediment.

Plant Tissue Nutrients

Shoot tissue nutrient concentrations of N, P and K, the nutrients which frequently limit plant growth, declined throughout the growing season in *S. alterniflora* at Poplar Island, based on monthly sampling in Cells 3D and 4D in 2010 (Figure 1-9). Therefore, in years in which only end of season sampling was done, the concentrations reported are considered minimum concentrations for the season.

In September 2008, when both Cell 3D and 4D were sampled, shoot tissue concentrations of N, P and K were higher in the low marsh vegetation (*S. alterniflora*) of each cell than in high marsh vegetation (*S. patens*) of the respective cell (Figure 1-9). Overall, plants growing on dredged material (Cell 3D) had higher tissue concentrations of N, P and K than plants gwoing on sand (Cell 4D) in both high and low marsh zones. Other macronutrients showed similar patterns but with less difference in magnitude. Plant tissue concentrations of Fe and manganese (Mn) were elevated in vegetation from Cell 3D compared to Cell 4D, especially in the low marsh *S. alterniflora*.

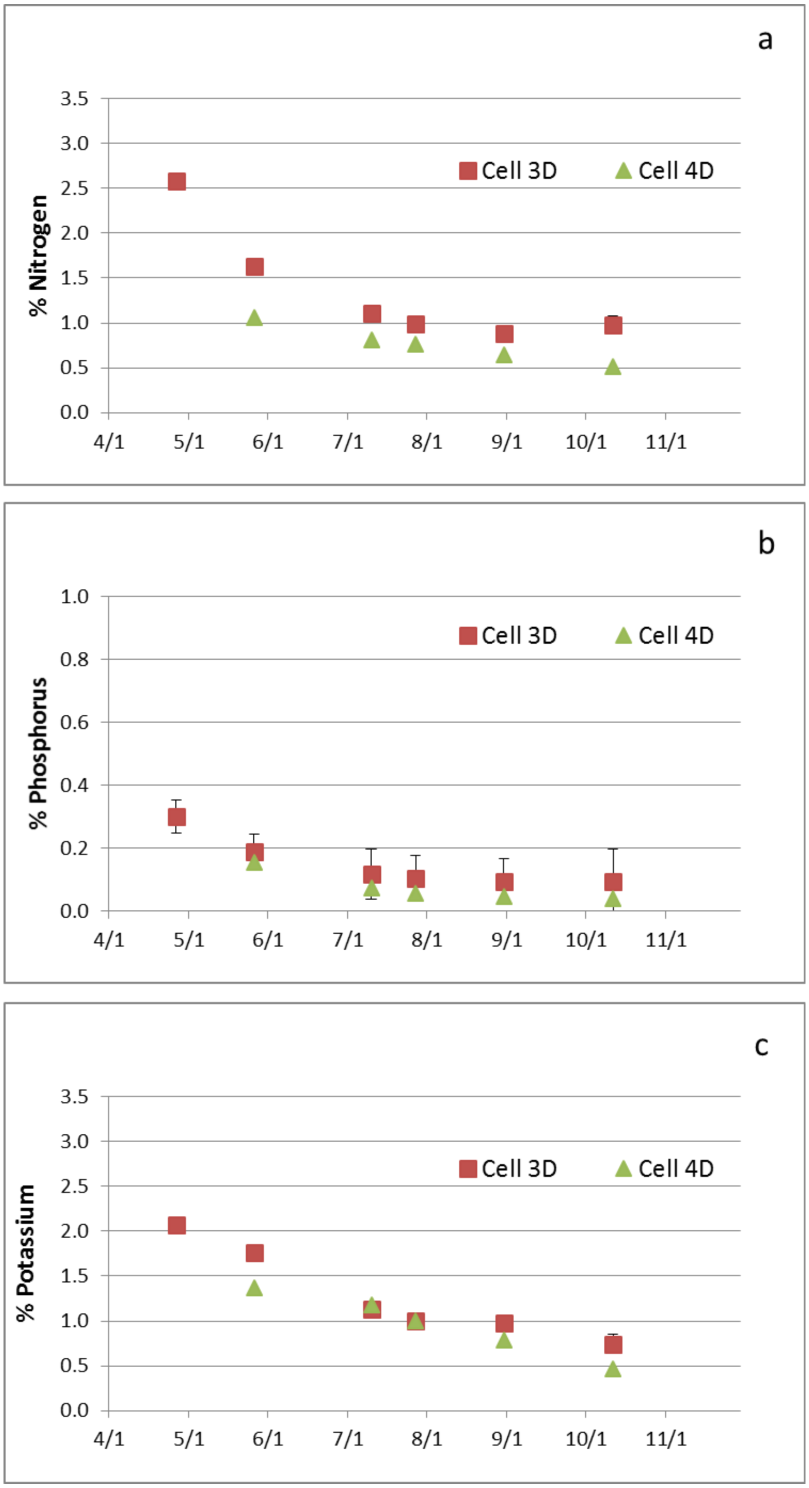

Figure 1-9. Time series of shoot tissue nitrogen concentrations in S. alterniflora during the 2010 growing season in the northeast corner of Cell 3D and Cell 4D (n=3, ± SE for Cell 3D; n=1 for Cell 4D).

Threshold values for most macro and micro nutrients for crop species (Epstein and Bloom 2005) and for *S. alterniflora,* N and P only, (Smart and Barko 1980) are shown in Table 1-3. These values represent adequate tissue concentrations for growth. Threshold tissue concentrations in crop species are generally derived from leaf tissue, which usually has a higher N content than whole shoots (Marschner 1995), while the concentrations for *S. alterniflora* were derived from whole shoots (Smart and Barko 1980). The marsh vegetation did not exceed the agronomic threshold when sampled in September 2008, but both *Spartina* species growing on dredged material in Cell 3D three years after the marsh was planted exceeded the *S. alterniflora* threshold reported by Barko and Smart (1980). This indicates a high level of N availability through the entire growing season. Overall, for tissue N and P concentrations Cell 3D low marsh > Cell 3D high marsh > Cell 4D low marsh > Cell 4D high marsh (Table 1-3).

Over time, end of season plant tissue N concentrations have remained above the threshold concentrations in Cell 3D, but remained below the threshold in Cell 4D. In 2010, five years post-planting, end of season tissue N concentrations were about 1% in Cell 3D compared to about 0.5% in Cell 4D and about 0.7% in *S. alterniflora* from the natural marsh at Horn Point Laboratory.

Table 1-3. End-of-season tissue nutrient concentrations for S. alterniflora and S. patens in 2008. LM denotes low marsh (S. alterniflora) and HM denotes high marsh (S. patens). The S. alterniflora thresholds from Barko and Smart (1980) and the crop thresholds from Epstein (2005) are the concentrations below which growth is limited.

	Cell 4D		**Cell 3D**		***S. alterniflora***	**Agronomic**
	HM	**LM**	**HM**	**LM**	**Threshold**	**Threshold**
Macronutrients (%)						
N	0.54	0.66	0.86	1.29	0.73	1.50
P	0.03	0.06	0.08	0.13	0.04	0.20
K	0.36	0.53	0.53	0.66		1.00
S	0.18	0.36	0.17	0.44		0.10
Ca	0.22	0.27	0.20	0.53		0.50
Mg	0.18	0.36	0.18	0.60		0.20
Na	0.27	1.80	0.56	4.06		
Micronutrients (ppm)						
Fe	321.0	1122.7	443.7	3003.3		100.0
Al	126.3	395.3	130.3	501.0		
Mn	43.0	120.0	279.3	985.7		50.0
Cu	6.0	5.0	5.3	5.3		6.0
Zn	17.0	14.3	11.3	25.0		20.0
B	7.0	15.0	5.3	13.7		20.0

Micronutrients tended to follow the same pattern in plant tissue concentrations, low marsh > high marsh and Cell 3D (dredged material) > Cell 4D (sand), with the exception of copper (Cu) which was similar in all samples. Iron (Fe) and manganese (Mn) were higher in the low marsh vegetation, and especially in plants growing on dredged material.

Sediment

Solid phase

The initial sediment characterization in Cell 3D (Table 1-4) showed that average sand, silt and clay content of the top 10 cm was 52.6 % ±26.2, 21.8 % ±18.6 and 25.6 % ±17.1, respectively (Stevenson et al. 2007). The carbon (C) and nitrogen (N) content in Cell 3D averaged 22 ±19 and 2.3 ± 1.3 mg g^{-1}, respectively, typical of Chesapeake Bay sediments (Hennessee et al. 1986), but low compared to most natural marsh sediments (Craft et al. 1999, 2003). Inorganic P was the dominant form of total P, and was significantly correlated with extractable iron (Fe), which binds inorganic P (Cornwell 1987). Initial pH values averaged 6.6 ± 0.4 and 6.2 ± 0.7 in the surficial and deeper sediments, respectively. After 2005 all measurements at 10 cm were >6 (Figure 1-10), although there was some inter-annual variation.

Table 1-4. Summary of available data on initial sediment characteristics in Cell 3D (2005) and Cell 4D (2003), with some comparative values from the natural reference marsh at Horn Point Laboratory. Data were compiled from Cornwell et al. (2005), Stevenson et al. (2007) and Mielcarek (2006). BD = below detection. Porewater units are μmol L^{-1}, except Cl- (mmol L^{-1}).

	Cell 3D		**Cell 4D**		**Horn Point Marsh**	
	Surface	**Deep**	**Surface**	**Deep**	**Surface**	**Deep**
Grain size:						
Sand	52.6					
Silt	21.8					
Clay	25.6					
Percent water	48 ±13	34 ±10				
pH	6.6	6.2				
Porewater NH_4^+	50 - 2500	200 - 2700	10 - 100	20 - 100	9 - 500	100 - 500
Porewater PO_4	BD - 75	BD - 85	BD	BD	2 - 80	12 - 42
Porewater Fe	BD - 220	BD - 580	BD - 8	BD - 10	BD	BD - 3
Porewater H_2S	BD	BD	BD	BD - 2	400 - 800	600 - 800
Porewater Cl^-	90 - 390	0 - 390			100 - 200	100 - 200
Solid phase Total N (mg g^{-1})	2.3 ±1.3					
Solid phase Total P (mg g^{-1})	0.44±0.23					
Solid phase Organic C	22 ±19					

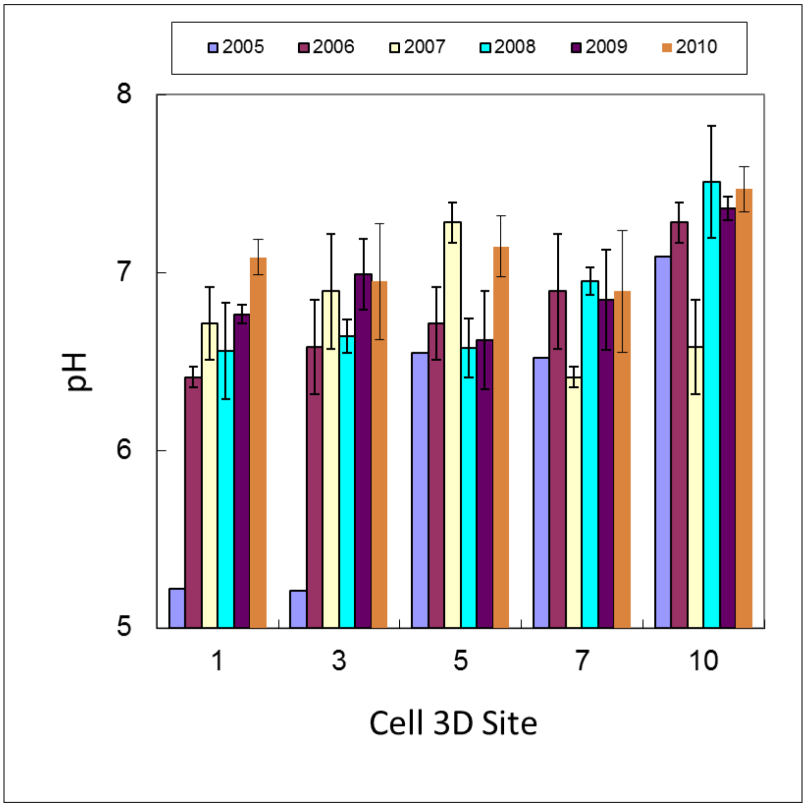

Figure 1-10. Long term average pH measurements (N = 5 for each site each year after 2005), for years 2005-2010 at 10 cm depth (Stevenson et al. 2013).

In the long cores from the dredged material cells, extractable NH_4^+ concentrations were variable but overall concentrations were high (Figure 1-11). Below 60 cm depth in the deepest core (1 m in length) extractable NH_4^+ concentrations were between 2-8 mmol dm^{-3} (3-16 µmol g^{-1}), indicating that deep sediment continued to provide a source of N more than 5 years after planting (Stevenson et al. 2013). The reasons for the variability in concentrations are not obvious, but could relate to the density of plant roots at the sampling sites, which could cause variability in concentrations due to plant uptake of NH_4^+.

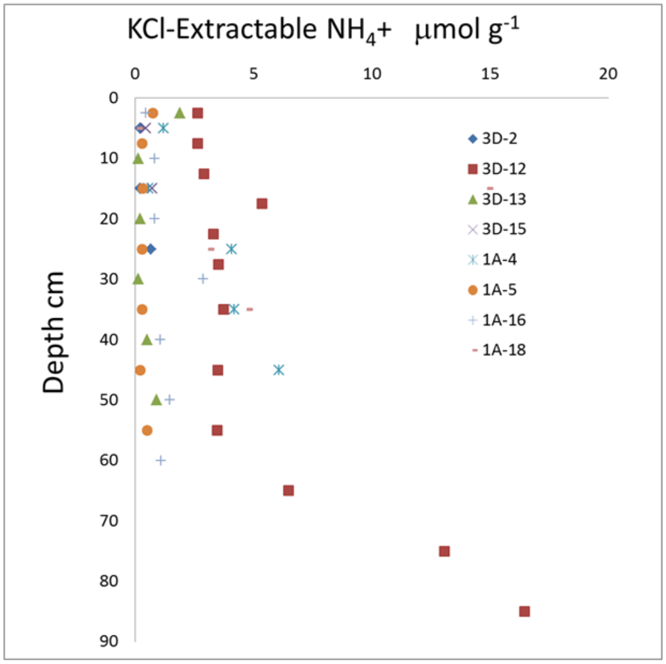

Figure 1-11. KCl-extractable NH4+ concentrations from 8 long cores collected in 2010 (Stevenson et al. 2013).

Porewater

In 2005, porewater ammonium concentrations in Cell 3D were variable but uniformly high, ranging from 50 - 2500 µmol L^{-1} in the root zone and 200 – 2700 µmol L^{-1} in the deeper sediment (Stevenson et al. 2007). Soluble reactive P, also variable, was low to moderate, less than 10 µmol L^{-1} at most sites in both shallow and deep sediment (Stevenson et al. 2007). Within one year average NH_4^+ concentrations in surface sediment declined to < 500 µmol L^{-1} at most sites, while deeper sediment concentrations at most sites remained above 700 µmol L^{-1} through 2009, but showed a steep decline to about 200 µmol L^{-1} by 2010 (Figure 1-12). A 2013 survey of the four Poplar Island dredged material marshes of different ages showed a strong trend of decreasing median porewater NH_4^+ concentrations below 20 cm depth (the root zone) suggesting that the pool of N in the deep sediment becomes depleted with increasing marsh age (Figure 1-12). Average porewater SRP concentrations generally increased in both surface and deep sediment from 2005 to 2009, but appeared to level off by 2010 (Figure 1-12).

Porewater H_2S concentrations were below detection at most sites in 2005, with only 2 samples out of 858 showing measurable levels (Stevenson et al. 2007). In contrast, H_2S in the Horn Point Laboratory marsh ranged from 400 - 800 µmol L^{-1} in 2003 (Stevenson et al. 2003).

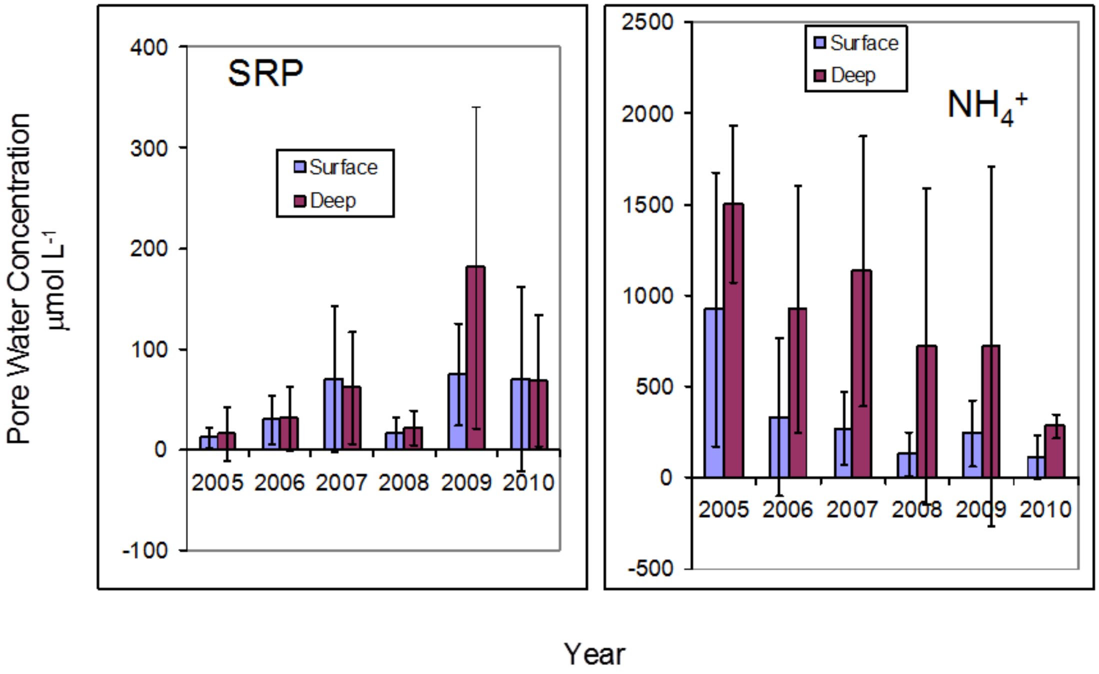

Figure 1-12. Time series of average porewater SRP and NH4+ concentrations in surficial (0-12 cm) and deep (20-50 cm) sediment sections from 12 sites in Cell 3D. Error bars are standard deviation (N = 12; Stevenson et al. 2013).

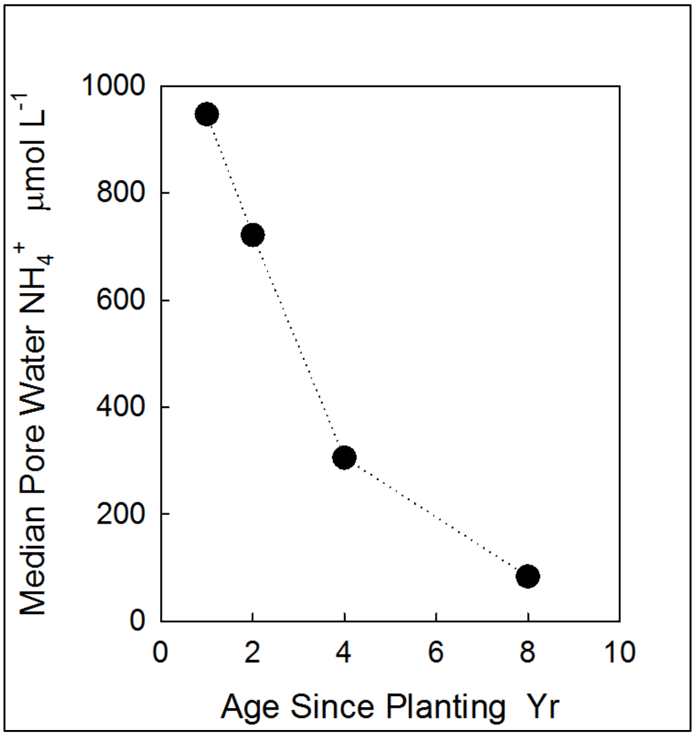

Figure 1-13. Performance curve of median pore water NH4+ concentrations below a sediment depth of 20 cm in a survey of four dredged material marshes in 2013 (Stevenson et al. 2014).

Dieback versus healthy marsh zones

In the 2008 survey of dieback vs. healthy zones of the Cell 3D low marsh, porewater NH_4^+ concentrations at the dieback sites were generally somewhat higher in deep sediment (100 – 250 µmol L^{-1}) than in root zone (9 – 190 µmol L^{-1}) and average concentrations were higher in both sediment sections in the dieback zone compared to the healthy zone (Stevenson 2009). Higher NH_4^+ concentrations in the dieback zone likely reflect the reduced N demand resulting from macrophyte losses due to dieback.

Porewater SRP concentrations were variable in the dieback zone but high, ranging from 90 – 200 µmol L^{-1} in the root zone and 150 – 220 µmol L^{-1} in the deep sediment. Healthy zone sites were only slightly lower (Stevenson 2009). Sulfide concentrations were similar in dieback and healthy sites, ranging from 10 – 450 µmol L^{-1} in the root zone and mostly below detection in the deep sediment (Stevenson 2009). Other than SRP enrichment at the dieback site, there was little difference in measured parameters between the dieback and healthy low marsh zones.

Discussion and Conclusions

Plant establishment

The fine grained, nutrient rich dredged material from upper Chesapeake Bay promoted rapid plant establishment in the Poplar Island marshes and luxuriant growth during the first two years. *Spartina alterniflora* transplants planted on 1 m centers spread at a rate of nearly 1 m y^{-1}, completely filling the space between transplants during the second growing season. Tissue N concentrations at the end of the growing season exceeded threshold concentrations, indicating that even after a full season of plant growth, N still was not limiting. Sediment pH was not excessively low, and sulfide concentrations were mostly below detection Thus, it appears that sediment pH, sulfide concentrations and nutrient content did not present a short-term impediment to plant establishment or growth in fine grained dredged material using transplant plugs. Furthermore, the abundance of plant available N and P eliminated the need for supplemental fertilizers which have been used in other marsh restoration projects (Broome et al. 1988; Zedler and Callaway 1999; Craft et al. 2002). The dredged material marshes at Poplar Island, therefore, represent a type of constructed marsh that is distinct from the low nutrient constructed marshes described by Craft (2002) and Zedler and Callaway (1999).

After year two, *S. alterniflora* growth was compromised by sporadic dieback, resulting in reduced AG biomass production and a high degree of inter-annual variability. The low nutrient substrate in Cell 4D produced less inter-annual variability and no dieback, although overall production was lower than Cell 3D in most years.

Inter-annual variability in macrophyte production on the scale observed in Cell 4D is common in both tidal and non-tidal marshes, although the drivers and mechanisms may be different. Non-tidal marsh production is often a function of river discharge, which delivers pulses of nutrients to riparian marshes in high flow years (Mitsch and Gosselink 2007). Production in some tidal marshes is also positively correlated with river discharge (Więski and Pennings 2014) or with rainfall (Morris and Haskin 1990; Kearney and Riter 2011), but sea level, flooding, surface water N concentrations and, most frequently, temperature are cited as the most important drivers (Visser et al. 2006; Kirwan et al. 2009; Kearney and Riter 2011). The result is pulsed macrophyte production in marshes, leading to inter-annual variability.

While these factors obviously play a role in the inter-annual variability of the constructed Poplar Island marshes, low marsh dieback in the dredged material cells amplifies the variability considerably. The result is highly pulsed production in the dredged material marshes, with years of high biomass production apparently driven by the abundant nutrient supplies, punctuated by vegetation dieback on varying scales. While small patches of dieback have a minimal effect on overall production, the dieback of 2013 affected >50% of two marshes (Cell 1A and 1C) and only slightly less in Cell 1B, while the oldest dredged material marsh (Cell 3D) remained largely unaffected (Stevenson et al. 2014). Dieback on this scale is of potentially greater consequence because recovery takes longer and because the loss of organic matter accumulation is likely essential for these marshes to keep up with sea level rise. Obvious questions are, what causes dieback in the dredged material marshes at Poplar Island, and why is it absent from Cell 4D? In addition, why was Cell 3D, the oldest dredged material marsh largely unaffected by the 2013 dieback while large scale dieback affected the younger dredged material marshes?

***Spartina alterniflora* dieback**

Sudden dieback of marsh vegetation, first observed in *S. townsendii* marshes in the UK in 1924 (Goodman et al. 1959), has been observed in *S. alterniflora* marshes with increasing frequency since the 1970s (Alber et al. 2008; Elmer et al. 2012). Despite the long history of observations, the causes of marsh vegetation dieback remain elusive. The primary driver has been attributed to drought, herbivory, changes in hydrology and freshwater flow, and sea level rise in different locations, but there is a general consensus that multiple stressors are involved in most incidents (Alber et al. 2008). Although transient soil conditions are thought to be important, there is little direct evidence of this because sampling generally has not been conducted prior to dieback and post-dieback sampling has not shown large differences with healthy marsh zones. This was also the case at Poplar Island, where large differences in porewater concentrations were not detected, and transplants into the dieback zone the following year had 100% survival. Mesocosm studies have also been inconclusive (Twilley et al. 2005; Elmer et al. 2012).

Soil monitoring has indicated that several factors that have been implicated in other diebacks are unlikely causes at Poplar Island. There was no indication from sediment monitoring in dieback areas that pH, salinity or sulfide were outside the tolerance limits of *S. alterniflora*. Sulfide in particular is known to be a phytotoxin which causes growth inhibition at concentrations of 2 mmol L^{-1} in *S. alterniflora,* and can cause mortality at higher concentrations (Koch et al. 1990). Dieback in Cell 3D, however, was first observed when sulfide concentrations in were still largely below detection. Furthermore, transplants of *S. alterniflora* into the worst dieback area of Cell 3D in 2008 (following the 2007 dieback) showed no inhibition of growth, indicating that there was not a persistent toxicity problem, consistent with the findings of others (Goodman et al. 1959 in Alber et al. 2008). Chloride measurements gave no indication that excessive salt concentration in the root zone had occurred, as it did in Louisiana marshes during the drought of 2000 (Stevenson et al. 2009).

The timing of sediment sampling may not have allowed detection of detrimental sediment conditions, since sampling occurred “after the fact” in most cases of dieback, but sampling prior to a dieback would be a matter of coincidence. Confirming the sediment conditions present immediately prior to dieback has not been possible at Poplar Island thus far, although comparing the distribution of dieback identified from aerial photography to sediment sampling sites may allow this to be done retroactively.

Observations during this study of the Poplar Island marshes indicate that dieback has followed a number of events, including plant lodging, muskrat damage from grazing and prolonged flooding early in the growing season which submerged small plants for an extended period. Based on these observations and the work thus far, we hypothesize that the rich supply of N in the dredged material predisposes *S. alterniflora* to a number of factors that may reduce photosynthesis or limit oxygen translocation to belowground portions of the plants, including lodging, grazing and disease.

In general, high soil N concentrations enhance shoot growth, increasing canopy height, and inhibit root elongation in graminoids (Levin et al. 1989; Marschner 1995). Increased shoot biomass and reduced root biomass compared to plants grown in low fertility conditions, lead to a downward shift in the root to shoot ratio. This shift in biomass allocation has been demonstrated in *S. alterniflora* fertilization studies under both controlled laboratory conditions (Smart and Barko 1980) and field studies (Valiela et al. 1976; Gusewell 2005; Darby and Turner 2008b; Deegan et al. 2012). In addition, these morphological changes lead to lodging - falling over due to weak stems (Mulder 1954), and N rich plant tissue is vulnerable to high rates of fungal and bacterial infection, and to increased rates of predation from vertebrate and invertebrate grazers (Marschner 1995). All of these plant characteristics have the potential to limit photosynthesis via shading (lodging), direct removal of (grazing) or damage to photosynthetic tissue (pathogens). They can also limit oxygen translocation by physically blocking the aerenchyma through which oxygen is conducted to the roots and rhizomes.

Low rates of oxygen production and transport to the BG plant tissue resulting from these factors, as well as inundation early in the growing season, may leave plants vulnerable to the reducing conditions in the fine grained dredged material. Toxic metabolites of anaerobic respiration can accumulate in the root zone due to low hydraulic conductivity (even when sulfides are not present in the sediment). In addition the plants may develop an energy deficit after prolonged anaerobic respiration, which is less efficient than aerobic respiration (Koch et al. 1990). These may be a key factors explaining the incidence of dieback in the Poplar Island marshes. Cell 4D, the low nutrient marsh, and Cell 3D, the oldest dredged material marsh, may be resilient to dieback due to higher

RSR, producing a greater capacity for sediment oxygenation. In addition, an increase in sediment organic matter content in older dredged material marshes may increase hydraulic conductivity and improve flushing of phytotoxins during tidal flooding. These hypotheses could be tested in future work.

Elucidating the mechanistic causes of dieback at Poplar Island was beyond the scope of this project, but may be critical to the long term sustainability of the marshes. Ultimately, if N is the driver of dieback in the dredged material marshes at Poplar Island, dieback should diminish as the available N supply from deep sediments is depleted over time. The timing may be critical, however, depending on the larger effects of dieback on elevation change in the dredged material marshes. An adverse effect on vertical accretion could reduce the ability of the marshes to keep abreast of sea level rise, which is critical to their long term sustainability.

Vegetation trajectories

The trajectory of low marsh vegetation development at Poplar Island contrasts sharply with the trajectory described for tidal marsh restoration projects in North Carolina (NC), where low nutrient, coarse grained dredge material or exposed upland subsoil produced marshes with low productivity compared to nearby natural marshes (Craft et al. 1991, Craft and Sacco 2003). In these created NC marshes aboveground biomass, belowground biomass and stem height took 10-15 years to increase to levels equivalent to paired natural marshes. At Poplar Island, very high peak biomass is attained by the second growing season, followed by a slow downward trend toward levels reported for natural marshes in the mid-Atlantic region, punctuated by vegetation dieback in the low marsh.

A similar trajectory, but without dieback, was described in a review of the literature on marshes established on dredged material by Streever (2000) as being generally accepted, although the data did not seem to support the contention.

The Streever (2000) analysis concluded that monitoring individual pairs of marshes over long periods would be the most appropriate method for judging marsh trajectories due to inter-site variation and random temporal differences between sites. Perhaps events such as dieback, which are not mentioned in reviews by Streever (2000) or Simenstad (1996), would be detected more readily in annual surveys rather than single surveys of different aged marshes. The Poplar Island data suggest that indeed, following individual marshes over time is important given the degree of interannual variability in marsh production and other transient phenomena such as dieback.

Streever (2000) further asserted that general conclusions about trajectories of created marshes toward functional equivalency should not include different “types” of created marshes, for example upland sites where soil was removed to achieve appropriate elevation for marshes and dredged material deposition sites, due to different soil characteristics (e.g. Sacco et al. 1994). If the “type” of marsh is interpreted as the N content of the substrate used in construction, which may be the key factor determining the trajectory of vegetation development, rather than the process (e.g. downgrading versus “upgrading”), improved relationships might be found.

There are a number of differences between the substrate used in the NC marshes studied by Craft (2002) and the dredged material used at Poplar Island, but the soil N content at planting is the most notable. Differing by over two orders of magnitude, it is likely the driver of the very different vegetation development trajectories. As N availability in the

Poplar Island dredged material declined over time, a decline in aboveground biomass production resulting in an increase in the root:shoot ratio was expected In contrast, in Cell 4D, the low nutrient marsh, an increase in biomass production similar to that of the NC marshes (Craft et al. 2003) was expected. There was some evidence of both in the Cell 3D time series biomass data (Figure 1-5), and in the biomass data from all dredged material cells in 2012, a year when there was little dieback observed in any of the Poplar Island marshes and biomass was compared across the age spectrum (Figure 1-8). It appears that at 10 years, biomass production in Cell 3D is approaching the levels in Cell 4D, comparable to natural marshes in the Mid-Atlantic region.

Based on the trajectory of the nutrient rich dredged material marshes at Poplar Island compared to the low nutrient NC created marshes, it seems that pooling dredged material marshes with vastly different N contents would not produce consistent predictions of the type of trajectory or the time to functional equivalency, as N availability is a primary driver of plant growth and allometry (Chapin et al. 1987; Marschner 2012). This study suggests that categorizing restored marshes by the initial N content of the substrate may improve our understanding of trajectories toward functional equivalency and estimates of the time required to reach to functional equivalency with natural marshes in biomass production.

Pulsing

The response of macrophyte production to pulses of nutrients can be described by a subsidy stress model (Odum 1995). In wetlands, when these nutrient pulses occur on the lower end of the subsidy spectrum an increase in production is a direct result. At the higher end, however, reductions in primary production can occur, likely as an indirect result of the nutrient input. The nutrient rich dredged material at Poplar Island provides N at a level that likely pushes the system into the stress region of the model. Heightened vulnerability to secondary factors likely result in dieback, varying in scale on depending upon the severity of the secondary factors (e.g. degree of tidal flooding, or grazing pressure). As the intermittent secondary stress diminishes, production returns to higher levels resulting in the oscillations seen in the Poplar Island marshes.

Although pulsing is a normal part of wetland ecosystem function, vegetation dieback is an area of concern at Poplar Island for several reasons. The pulsed production caused by dieback in the dredged material marshes may not be consistent with a steady accumulation of autochthonous organic matter, or allochthonous material trapped by standing vegetation (Kirwan and Megonigal 2013).

Deegan et al. (2012) found that chronic eutrophication of coastal marshes can drive a conversion of marsh to mudflat even in the absence of widespread vegetation dieback. By altering the rate of plant production, biomass allocation, and sediment microbial decomposition, N drives a number of sediment changes that undermine the structural integrity of the substrate, leading to creek bank erosion and increasing the area of unvegetated mudflat along tidal channels. The main concern at Poplar Island is the ability of the marshes to accrete at rates equivalent to local SLR. The same N driven changes that allowed creek bank collapse in the marshes studied by Deegan et al. (2012) may negatively affect accretion rates, especially in areas of low tidal range and suspended sediment loads (Stevenson et al. 1986; Stevenson et al. 1988; Morris et al. 2002; Stevenson 2002). The implication for natural marshes of the Poplar Island study is that under current N loads marshes may be undergoing subtle changes that reduce marsh resiliency, but further N enrichment may lead to more acute problems, such as vegetation dieback. This can be expected to exacerbate problems leading to erosion and accretion deficits, and may have impacts on nutrient cycling and exchange with coastal areas.

As an indicator of problems that may develop in natural marshes under extreme levels of eutrophication, the Poplar Island marshes are not a perfect model. The grain size and low organic content of the upper Chesapeake Bay dredged material are markedly different from the highly organic substrate typical of natural marshes. The dieback observed in the Poplar Island marshes may not have a direct analogue in natural marshes, but the instability produced by this high level of N at Poplar Island is consistent with field studies showing increasing vulnerability to stress in natural marshes (Alber et al. 2008; Deegan et al. 2012). The effects of dieback on elevation change and nutrient dynamics are examined in subsequent chapters.

Conclusions

The Poplar Island tidal marsh restoration shows that fine-grained, nutrient-rich dredged material can be used to quickly establish marsh vegetation, but will have a destabilizing effect on the marshes within two years by promoting *S. alterniflora* dieback. Low marsh vegetation dieback may compromise the ability of the marshes to keep up with SLR, especially if projections of accelerated SLR are realized.

Conflicting conclusions in recent reviews of vegetation development in constructed marshes including, but not limited to dredged material marshes, may be driven by differences in the initial N content of the material used as a substrate. These high N constructed marshes initially have higher aboveground biomass production than natural marshes, and become highly pulsed systems, rather than proceeding monotonically toward a lower level of production as the sediment nutrient supplies decline. The rich N supply, likely in combination with problems associated with the silt/clay dominated sediments, is presumed to be the ultimate driver of these oscillations.

Although the exact mechanism which causes dieback is not clear, it is suspected that a reduction in the oxygen supply (due to lodging, grazing, fungal infection, submergence, stem borers and other factors) to belowground portions of the plants is important. Eutrophication has been implicated in dieback in natural marshes, but it is unclear whether the mechanisms at work in natural systems are similar to those which produce dieback in the Poplar Island constructed marshes. Determining what individual or combined factors result in dieback, and the mechanism that leads to plant mortality may help improve marsh design to minimize dieback in future projects. Comparing rates of elevation change in areas with differing dieback impacts will help determine the overall importance of dieback to the long term sustainability of these marshes.

Chapter 2: Elevation change in tidal marshes constructed with fine grained dredged material at Poplar Island

Introduction

Background

In spite of a federal “no net loss” policy for wetlands in effect in the U. S. since 1989, losses have outpaced gains from mitigation and restoration, leading to a continued decline in wetland acreage (Dahl 2011). Tidal marsh losses have been especially high due to a combination of human induced and natural processes, and interactions between the two (Day et al. 2008). Replacing natural tidal marshes lost through natural or anthropogenic forces with marshes constructed using dredged material from maintenance dredging of shipping channels has become a widespread practice (Streever 2000). These constructed marshes are vulnerable to many of the same stresses as natural marshes, including sea level rise (SLR).

Positioned in the intertidal zone, tidal marshes are sensitive to small changes in sea level, making SLR one of the most important threats to their sustainability. Projections of accelerating SLR resulting from climate change have led to predictions of increased tidal marsh losses, along with their important ecosystem services (Craft et al. 2009). In this study, we examine elevation change in a tidal marsh constructed with fine grained, nutrient rich dredged material in relation to SLR and factors which may contribute to accretion to a greater or lesser degree in all tidal marsh systems.

There has been an increase in the global SLR rate from 1.2 mm y^{-1} over the last century to 3.3 mm y^{-1} beginning in the early 1990s, based on NASA altimetry (Figure 2-1b), but this is much lower than the worst case scenarios projected (Rahmstorf 2007, Rahmstorf et al. 2012). Rahmstorf et al. (2012) find evidence that the rate of SLR over the last few decades exceeds projections in the 2007 International Panel on Climate Change (IPCC) report, suggesting that these future projections may also be conservative. There could be a global rise of 14 mm y^{-1} by the end of the century if carbon emissions remain in the IPCC intensive range (Church et al. 2013).

Global SLR is not uniform, however, and the middle Atlantic coastline has been described as a global hotspot for SLR resulting from a combination of temperature, salinity and ocean circulation effects, and larger scale dynamic processes, with SLR rate increases 3 – 4 times the global average since the middle of the twentieth century (Kearney and Stevenson 1991; Sallenger et al. 2012). Relative SLR, the combination of eustatic SLR and local isostatic changes, is also especially high in the Chesapeake Bay region where local subsidence resulting from glacial fore bulge collapse and groundwater withdrawals have produced a 100 year rate of apparent SLR of 3 - 4 mm y^{-1} (Figure 2-1a).

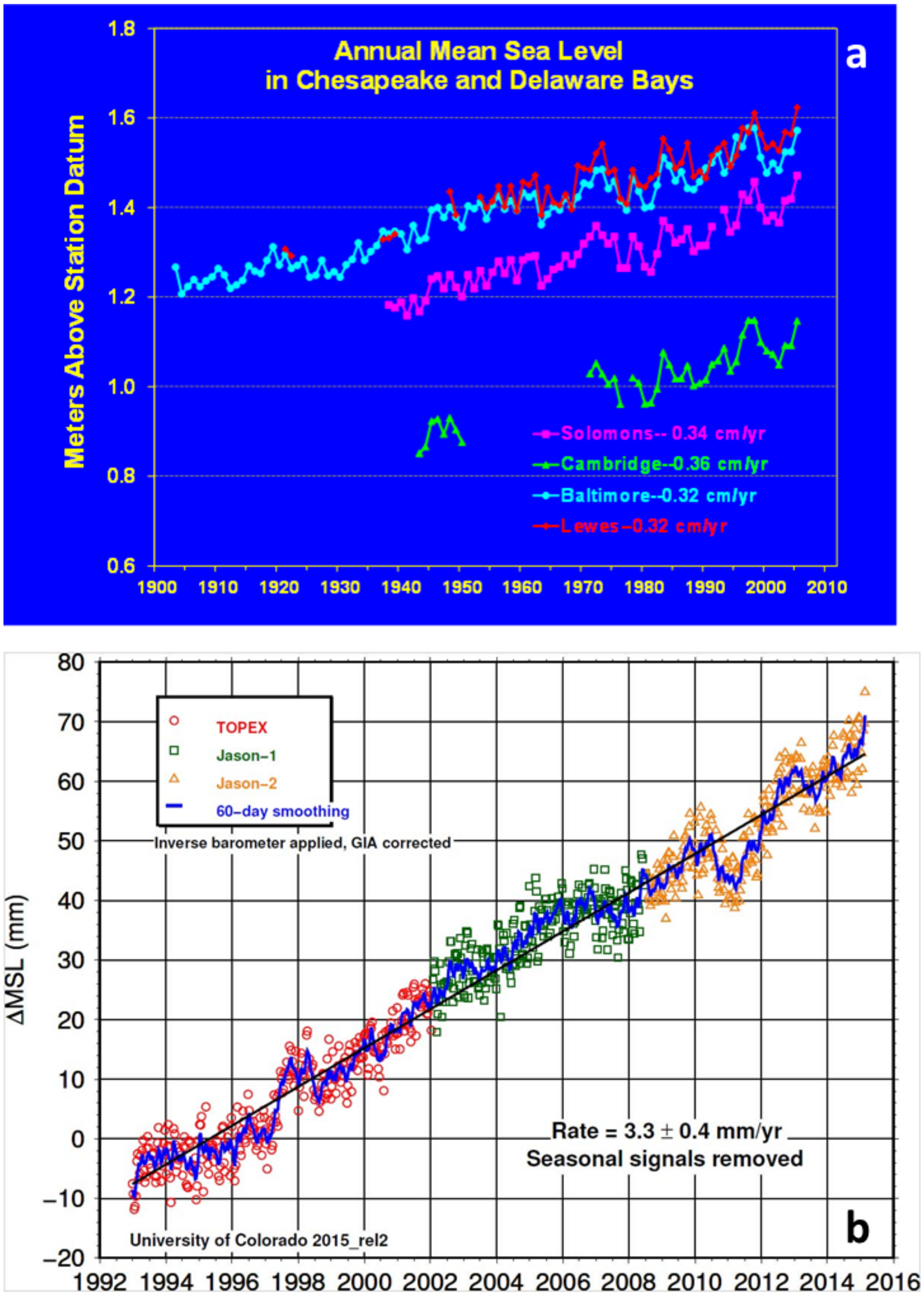

Figure 2-1. Mean historical annual sea level at regional NOAA tide gauges (a); and recent average global sea level estimated via altimetry by NASA satellites, 1993-2013 (b, University of Colorado).

The combined effects of the accelerated rate of global SLR, the regional enhancement of global SLR in the mid-Atlantic region, and local isostatic changes in Chesapeake Bay have resulted in recent rates of apparent SLR in mid-Chesapeake Bay of 3.9 and 5.6 mm y^{-1} (1993 to 2013) at the NOAA tide gauges at Baltimore and Solomons, Maryland, respectively (Figure 2-2a). Over the last five years that data are available (2008 – 2013), the Baltimore and Solomons rates are even higher, 13.6 and 11.0 mm y^{-1}, respectively (Figure 2-2b), in the same range as the maximum threshold of 12 mm y^{-1} projected for survival of marshes along the southeast coast of the U.S. (Morris et al. 2002).

Given the variability in mean sea level on time scales of seasons to decades, the reliance on these shorter term records of local sea level for projecting long term rates would not be entirely appropriate. Using them, however, as indicators of the water levels experienced by the marshes during an elevation monitoring period may be instructive, because *S. alterniflora* responds to stress such as inundation on time scales of hours to days (Mendelssohn et al. 1981; Bradley and Morris 1990; Koch et al. 1990; Maricle and Lee 2002).

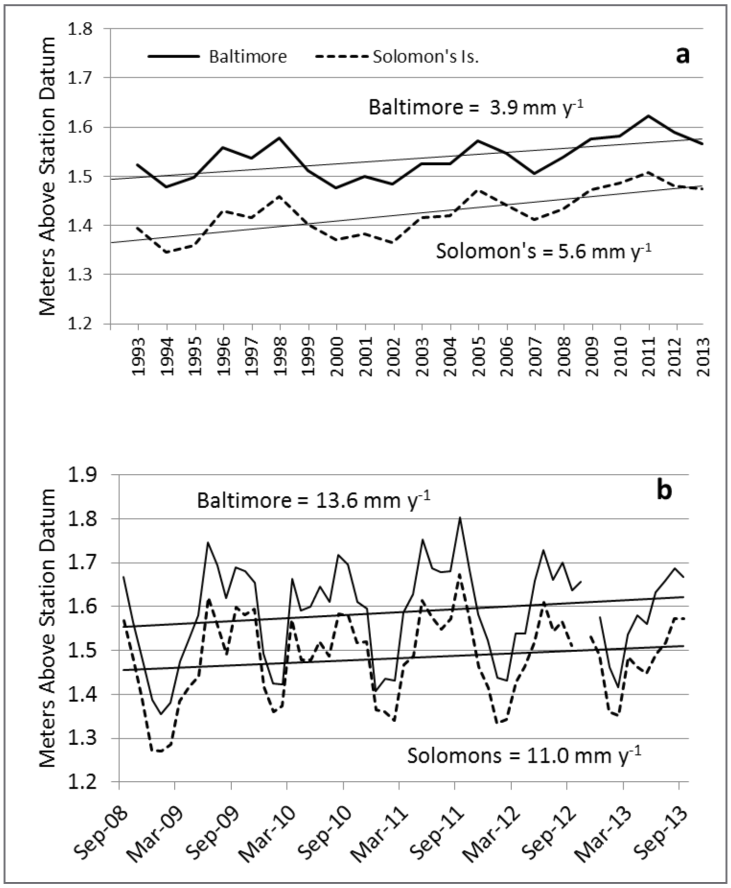

Figure 2-2. Mean annual (a) and mean monthly (b) sea level at Baltimore (NOAA Station 8577330) and Solomons (NOAA station 8574680) tide gauges, 1993-2013 and 2008- 2013, respectively, expressed as meters above station datum. Rates of sea level rise over respective periods are shown next to each curve. For the twenty year period (a) $\overline{x}$ = 4.9 mm y^{-1}; for the five year period (b) $\overline{x}$ = 12.3 mm y^{-1}.

Spartina alterniflora can withstand short term inundation through a variety of physiological and morphological responses, including switching to anaerobic respiration (Mendelssohn et al. 1981) and the formation of new aerenchyma during extended flooding events (Arenovski and Howes 1992). Resilience to SLR, however, is accomplished by vertical accretion of the marsh surface to maintain positive elevation with respect to sea level, and transgression (landward migration) where possible. Where transgression is constrained by coastal development or natural landforms, vertical accretion is essential to sustainability during extended periods of SLR.

Vertical accretion occurs through the accumulation of organic matter produced in situ (Stumpf 1983; DeLaune et al. 2003; Nyman et al. 2006), and allochthonous organic and inorganic matter transported by tidal flooding or wind and trapped by the vegetation and organic soil matrix (Redfield 1972; McCaffrey and Thompson 1980; Stevenson et al. 1986). Studies of natural tidal marshes such as the Barnstable Marsh (Redfield 1972) indicate their persistence through millennia at SLR rates of 1-2 mm y^{-1}.

Feedback mechanisms allow tidal marshes to adjust to changing SLR (Morris et al. 2002; Day et al.2011), but the limits of vertical accretion to compensate for SLR depend upon a number of factors, including rates of macrophyte production, the local supply of inorganic sediment, tidal range and the magnitude of SLR (Stevenson et al. 1988; Morris et al. 2002; Temmerman et al. 2003; Mudd et al. 2009; Kirwan et al. 2010). There are a range of potential effects of climate change on the sustainability of tidal marshes (i.e. increases in temperature, CO_2, salinity and water levels), but there is agreement that climate change induced SLR could have substantial detrimental impacts on these systems depending on magnitude and rates of change (Morris et al. 2002; Stevenson and Kearney 2009; Kirwan and Megonigal 2013; Weston 2014).

Extensive losses of tidal wetlands in Chesapeake Bay are attributed at least in part to the historically high rate of apparent SLR, including losses associated with island disappearance (Kearney et al. 1988; Kearney and Stevenson 1991). Projections of accelerating SLR over the next century (Rahmstorf et al. 2012; Church et al. 2013) place both remaining natural and newly constructed tidal marshes in the region at risk, including the recently constructed *Spartina alterniflora*/*S. patens* marshes at the Paul S. Sarbanes Ecological Restoration Project at Poplar Island, Maryland (Poplar Island).

The Poplar Island marshes are part of an island restoration project utilizing fine grained dredged material from the upper Chesapeake Bay, replacing habitat lost when the original island eroded almost entirely during the last 150 years (Chapter 1). Importantly, the project also provides a placement site for dredged material produced by maintenance dredging of the approach channels for the Port of Baltimore, Maryland., a significant contributor to the Maryland economy. The project is expected to remain active until approximately 2030, resulting in nearly 300 hectares of constructed wetland habitat in mid-Chesapeake Bay. Tidal marsh establishment, a key project goal, has been successful (Chapter 1), but the long term sustainability of the Poplar Island marshes will depend primarily on vertical accretion, as transgression is severely constrained by adjacent upland habitat with an elevation approximately 10 m above mean sea level.

Historical estimates of vertical accretion in natural marshes are based on Pb^{210} and Cs^{137} dating of cores and marker horizons (Stevenson et al. 1986), and more recently, rod-surface elevation tables (Cahoon et al. 2002). Few estimates exist, however, for constructed marshes where historical sediment profiles are absent. In this study, we examine vertical accretion using rod-surface elevation tables (rSETs) for three marshes at Poplar Island during a period of about six years. Two of the marshes were constructed using fine grained dredged material from the upper Chesapeake Bay (Cells 3D and 1A), and one was constructed largely with locally obtained sand (Cell 4D).

The upper Bay dredged material is rich in nutrients (Cornwell and Owens 2011, Stevenson et al. 2014), particularly nitrogen (N). The resulting marshes initially have exceptionally high aboveground (AG) biomass and low root:shoot ratios (RSR; Chapter 1), typical effects of N enrichment for *S. alterniflora* (Morris and Bradley 1999; Darby and Turner 2008; Deegan et al. 2012).

There are conflicting evidence for the influence of N enrichment on elevation change in natural tidal marshes (Morris et al. 2002; Reed et al. 2009; Turner et al. 2009; Anisfeld and Hill 2012). The shift in the RSR may result in lower contributions to vertical accretion by belowground biomass deposition (Deegan et al. 2012), but the increased aboveground macrophyte production and sediment trapping ability may enhance surface accretion, offsetting the losses in belowground biomass. Here we use the Poplar Island marsh with a sand substrate (Cell 4D), containing a much smaller pool of N (Mielcarek 2006) as a low nutrient control to examine the impact of nutrient supply on elevation change. Cell 4D has lower AG biomass and a higher RSR than the dredged material marshes, more typical of natural marshes in the mid-Atlantic (Chapter 1),

Recurring *S. alterniflora* dieback in the dredged material marshes at Poplar Island, sometimes affecting many hectares of the low marsh (Chapter 1), may also influence sediment elevation change. Dieback has also been observed in natural *Spartina* marshes (Linthurst and Seneca 1980; McKee et al. 2004; Alber et al. 2008; Elmer et al. 2012), but the impact of dieback on elevation change has also not been well studied (Baustian and Turner 2006). At Poplar Island, some areas that have experienced repeated vegetation dieback have converted to non-vegetated mudflat. We used these marshes to address how vegetation dieback may affect elevation change in constructed tidal marshes.

We also examined differences in rates of elevation change in high marsh and low marsh zones of one dredged material marsh (Cell 1A), which are graded to different elevations prior to planting. The Poplar Island marshes are designed to be 80% low marsh and 20% high marsh, and similar rates of accretion would maintain these proportions while different rates could result in the expansion of one at the expense of the other.

Extensive ground surveys of marsh surface elevation in these constructed marshes were completed prior to planting, and a follow-up survey was conducted in 2014 by the United States Army Corps of Engineers (USACE). The rates of elevation change estimated with rSET measurements in one dredged material marsh, Cell 1A, were compared to rates estimated from these ground surveys in order to apply rates derived from rSET measurements to larger scales.

Finally, the sources of accretionary material in the dredged material marshes were considered. Marker horizons were used to evaluate surface versus sub-surface contributions to vertical accretion. Tidal flux measurements made in 2013 and 2014 allowed estimates of total suspended solids (TSS) exchange with the adjacent estuary, and a decomposition experiment helped estimate how much macrophyte shoot biomass may be available for burial and accretion.

Hypotheses

This study focused on the ability of tidal marshes constructed with fine grained, nutrient rich upper Chesapeake Bay dredged material at Poplar Island, Maryland to keep abreast of local sea level rise. Hypotheses tested were that (1) nutrient rich dredged material marshes will maintain equilibrium with the local apparent rate of SLR; (2) elevation change in the dredged material marshes and the onsite low nutrient reference marsh (Cell 4D) would not be equivalent; (3) vegetation dieback in the low marsh of the dredged material cells would have a negative effect on elevation change, and (4) high marsh rates of elevation change would be lower than low marsh rates. We also examine the potential contributions of surface versus subsurface, and organic versus inorganic material contributions to elevation change in the dredged material marshes using data from rSETs, marker horizons, sediment monitoring, a decomposition experiment and a tidal flux study. Finally, we examined elevation change in the Poplar Island marshes in the context of reported estimates of elevation change based on rSET measurements in regional tidal marshes.

Methods

Site description

A full description of the Poplar Island project site is provided Chapter 1. Three tidal marshes on the island were the focus of the current study, tidal marsh containment Cells 3D (planted 2005) and 1A (planted 2009), constructed with upper Bay dredged material as a substrate, and Cell 4D (planted 2003), constructed with local sand as a substrate (Figure 2-3). Cell 1B (planted 2012), also a dredged material marsh, was the site of a tidal flux study used to estimate net sediment exchange in tidal waters. A perimeter dike approximately 3 m in height surrounds all the marsh cells, with a tidal opening consisting of a series of corrugated metal pipes (Cells 3D and 4D), or concrete box culverts (Cells 1A and 1B). Though there were three rSETs installed in the high marsh zone of Cell 1A, the primary focus of this study was the low marsh zone, covering approximately 80% of marsh area at Poplar Island.

Figure 2-3. Aerial photograph of Poplar Island (September 2014), showing containment cells (Cells 1B, 1A, 3D, 4D) being developed as tidal marsh habitat on the east side of the island (a). Locations of rSETs used in this study are indicated by white dots in images (b) and (c).

Biomass

Aboveground biomass samples were collected using 0.25 m^2 PVC quadrats at the conclusion of the growing season (usually October) beginning in 2006 in Cell 3D, the year following planting, and in 2008 in Cell 4D (with one set in 2004). Samples were separated into dead and living biomass prior to being dried at 60°C in a forced draft oven (Grieve Model 343) and weighed to the nearest mg on a top-loading balance (O'haus 1500D). Belowground biomass samples were collected from each of the aboveground biomass quadrats using a stainless steel piston corer. Sediment cores were washed free of sediment over a 1 mm mesh sieve before being dried and weighed as described above. In 2010 the depth distribution of belowground biomass was evaluated by sectioning BG biomass cores into 2 cm increments and processing each section separately as described above.

Surface Elevation Tables

Rod-surface elevation tables (rSETs) were used to monitor elevation change (Cahoon et al. 2002). Concrete monuments were established, secured over underlying stainless steel rods driven to refusal with a Cobra gasoline driven concrete hammer (provided by NOAA National Geodetic Survey in Silver Spring, Maryland). Thirty six measurements of surface elevation were made at each rSET annually, usually in late winter or early spring prior to the emergence of vegetation.

Rod-SETs were installed in Cell 3D in 2008, three years after planting, to examine the effect of dieback on elevation change (Table 2-1). Two groups of three rSETs were located in the low marsh, one in an area where dieback was observed to affect > 1 ha in 2007 and occurred repeatedly in subsequent years, leaving the marsh sparsely vegetated. At the other site dieback was not observed in 2007, and only on a smaller scale and less frequently in subsequent years.

In 2009, the year Cell 1A was planted, twelve rSETs were installed in groups of three. The experimental design in this marsh was intended to compare high and low marsh accretion rates, as well as rates adjacent to creek banks where higher deposition rates were expected. Creek banks in the Chesapeake often have levees, so increased sediment accumulation was anticipated in these areas compared to interior low marsh sites. Consequently, one group of three rSETs was located in the high marsh, two groups were located in interior low marsh areas, and each rSET in the remaining group was located adjacent to a creek bank. Although the original design did not include a comparison of accretion rates in areas differentially affected by dieback, the two interior low marsh groups provided that opportunity when one group was affected more by dieback in 2011 and 2013 than the other, based on visual observations and annual aerial photography. Three rSETs were installed in 2011 in the low marsh of Cell 4D to compare rates in the dredged material marshes to the low nutrient sand marsh.

Rates of elevation change relative to the first measurement at each rSET were compared with SLR rates derived from the Baltimore and Solomons NOAA tide gauge stations over the full tide gauge record, the previous ~20 years, and since 2008 when the first rSETs were installed at Poplar Island. Comparisons of elevation change rates were made between the low nutrient marsh (Cell 4D) and the high nutrient dredged material marshes (Cells 3D and 1A) and between high marsh and interior low marsh (Cell 1A).

To examine the potential influence of incoming allochthonous material on accretion rates comparisons were made between rates of change at individual rSETs and the direct distance from the rSET to the tidal inlet,. The average group rates were compared to aboveground, belowground and total biomass at sampling sites located within the same area. In Cells 3D and 4D there were three sampling stations located near the group of rSETs, while in Cell 1A there was only one biomass station located near each group.

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Table 2-1. Installation and monitoring dates of rSETs at Poplar Island.

Cell	**Number Installed**	**Installation Date**	**Measurement Date – Month/Year**							
			3/09	**7/09**	**12/09**	**5/10**	**5-7/11**	**2/12**	**3/13**	**4/14**
3D	6	November 2008	√		√	√	√	√	√	√
1A	9	April 2009 (#1-9)		√		√	√	√	√	√
	3	July 2009 (#10-12)			√ (#10, 11)	√	√	√	√	√
4D	3	April 2011					√	√	√	√

SET Ground Survey Comparison

There are often concerns about how applicable rSET measurements from a limited number of observations are to the elevation changes in a large area of a given marsh over time. In order to assess overall errors, estimates of surface elevation change in Cell 1A based on the rSETs were compared with estimates based on ground surveys conducted by the United States Army Corps of Engineers (USACE). For the USACE survey Cell 1A was divided into sections, and elevation change for each section was determined from the average as-built (2009) and 2014 elevations for the section (Stevenson et al. 2014). Rates of elevation change in the sections were compared to the mean rate estimated from the rSETS located within each section.

Marker Horizons

Twenty marker horizons were established in spring 2006 (HPL Biogeochemistry Lab), and about 20 more were added in November 2006, by applying a ~2 mm layer of feldspar clay over the marsh surface to cover an area of ~0.18 m^2 (Stevenson et al. 2008). Fourteen of these sites were revisited in April 2007 for a short-term assessment and again in winter 2009, 2010 and 2011 (Stevenson et al. 2008). Depth was determined by measuring the layer above the marker horizon in a small (~2 cm) core removed from each plot with a ruler. Sediment analyses on the newly accreted material included loss on ignition (LOI), determined after combustion in an oven at 400ºC; HCl-extractable iron (Fe) using atomic absorption spectronomy (Leventhal and Taylor 1990); total and inorganic phosphorus (P) following Aspila et al. (1976). New marker horizons were installed on March 14, 2014 in Cells 4D, 1A and 1C and April 10 in Cell 3D. They were evaluated on May 15, 2015 to assess recent accretion.

Tidal Flux Study

A tidal flux study was conducted seasonally from November 2012 through November 2014 in Cell 1B, adjacent to Cell 1A. Although it is not one of the marshes where vertical accretion is being measured with rSETs, it is similar in design, size and aspect to the other dredged material cells, and is considered representative of them for estimating TSS fluxes. Due to initial problems with velocity measurements, only data from July 20, 2013 to November 2014 are reported.

For each season, two consecutive tidal cycles were monitored. Velocity measurements were made with a Nortek Aquadopp Acoustic Doppler Profiler (ADP) mounted on a weighted PVC plate and placed in the center of one of the two primary tidal inlets, 1.83 x 1.83 m concrete culverts, equidistant from the open ends and sidewalls. Net water flow between Cell 1B and adjacent cells through two additional inlets was previously determined to be negligible (Stevenson et al. 2013). Because the water depth in the culvert is very shallow, often only a single velocity measurement was possible (as opposed to obtaining a vertical profile from multiple measurements). Velocities were uncorrected for any vertical or lateral shear in the flow and it was assumed that the velocity measurement of the first bin, which was nominally located 45 cm above the bed, was representative of the mean velocity. Velocities were recorded every ten minutes by averaging two minutes of observations.

Discrete one liter water samples were collected hourly using an ISCO model 2700 automatic water sampler positioned above the culvert with approximately 3.6 m head. The sampling intake was located approximately 0.3-0.4 cm above the bottom of the culvert. The samples were chilled with ice and returned to the lab at the conclusion of the sample period and refrigerated until processing the following day. Subsamples were vacuum filtered using and 4.7 mm and 2.5 mm Whatman GFF filters (nominal pore size 0.7µ), analyzed for total suspended solids (TSS), and particulate carbon (C) and N, respectively. Total suspended solids analysis followed USEPA Method 160.2 (USEPA 1979); CHN analysis was conducted according to standard protocols of the Horn Point Laboratory Analytical Lab, by combustion in pure O_2 using an Exeter Analytical, Inc. CE-440 Elemental Analyzer (Lane et al. 2000).

For computing fluxes using concentrations from the ISCO samplers, the 10 minute velocity records were used, and hourly water quality data were interpolated to pair with velocity data points. Fluxes can be calculated as (Stevenson et al. 1988):

$$q_i = c_i u_i A_i \ .$$

Where q_i = instantaneous flux at time i, c = concentration, u = velocity and A = cross sectional area of the culvert, with subscript i referring to an instantaneous, discrete measurement. Due to variations in the diurnal tides at this site, which can be exaggerated by wind forcing, large differences in water volume between the beginning and end of the sampling period (Figure 2-4) can lead to large errors in calculations of net fluxes (Boon 1975; Nixon 1980). Therefore an alternative method was used here. Volume weighted concentrations were multiplied by the average tidal prism for the 24 hour monitoring period to obtain the flux (q) for each flood or ebb period. Average water volumes used to calculate fluxes for each period were: July 2013, 18583 m^3; November 2013, 8217 m^3; February 2014, 6028 m^3; May 2014, 8820 m^3; July 2014, 9886 m^3; November 2014, 10873 m^3.

The cross-sectional area of the culvert was calculated by multiplying the observed depth by the width of the culvert (1.83 m). Depth was calculated from hydrostatic pressure measurements obtained by a pressure sensor located on the Aquadopp, and confirmed with data from the Poplar Island tide gauge (USACE).

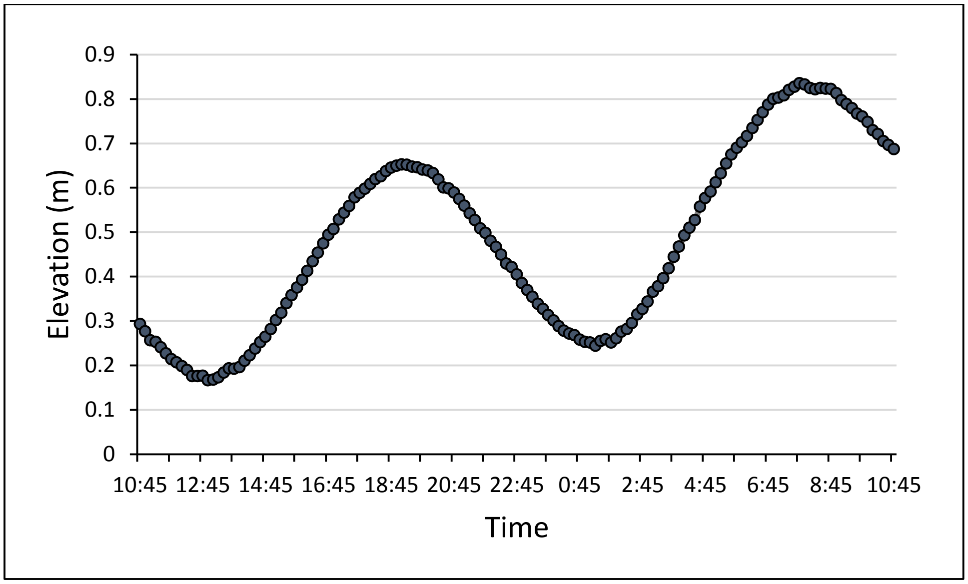

Figure 2-4. Water level at the tidal inlet to Cell 1B, July 26 - 26, 2013.

Decomposition

Intact stems of *S. alterniflora* were collected from each marsh on January 5, 2012, rinsed with deionized water and dried at 60°C to constant weight in a forced draft oven (Grieve model 343). Approximately 40 g of dried plant material was placed in 20 x 30 cm nylon mesh bags (mesh size 2 mm; Collins Cottage Industries). Eighteen bags were deployed in each marsh on February 15, 2012, containing plant litter from the respective marsh. Three bags were retrieved from each marsh at approximately three month intervals for twelve months and the final three were collected at 22 months. Bags were rinsed with deionized water in a shallow basin, and the detrital material was removed from the bag, dried in a forced draft oven at 60°C (GCA Corp. model 28) and weighed to the nearest 0.01 g on a top loading balance (O'Haus model 1500D). Detrital material was analyzed for C and N content by combustion in pure O_2 using an Exeter Analytical, Inc. CE-440 Elemental Analyzer (Lane et al. 2000). Soil temperature was monitored during the period with a Hobo Water Temp Pro v2 data logger (Onset Corp.) buried ~2 cm beneath the soil surface in Cell 1C.

Statistics

Regression analyses were carried out to detect significant elevation changes in Cells 1A, 3D and 4D. Analysis of variance (ANOVA) was used to detect significant differences between mean elevation rates in the high marsh and low marsh zones of Cell 1A, and between low marsh rSETs in the dredged material cells (Cells 3D and 1A) and the low nutrient reference marsh (Cell 4D). Correlation analyses were carried out to detect significant relationships between rates of elevation change at individual rSETs and (1) 2014 aboveground biomass production, and (2) direct distance from the rSET to the tidal inlet. Statistical analyses were performed using R statistical software by Philippe Hensel, National Oceanographic and Atmospheric Administration (NOAA) National Geodetic Survey (NGS).

Results

Elevation change

Rod-surface elevation tables

Generally, rates of elevation change for all rSET groups exceeded the average 1993 – 2013 rate of SLR at Baltimore and Solomons ($\overline{x}$ = 4.9 mm y^{-1}), with the exception of the creek bank group, where severe bank erosion at one rSET resulted in a low group mean

(Table 2-2). Compared to the average 2008-2013 SLR rate ($\bar{x}$ = 12.3 mm y^{-1}), however, only one low marsh interior group in Cell 1A (rSETs 7, 8, 9) was close to this rate of SLR. Elevation rates among groups ranked: dredged material, low intensity dieback > sand > dredged material, high intensity dieback > high marsh > creek bank (Table 2-2).

Table 2-2. Characteristics and mean rates of elevation change in groups of rSETs in Cells 3D and 1A (n = 3), and individual rSETs in Cell 4D at Poplar Island. Substrates are dredged material (DM) or sand. Dieback intensity estimated visually from aerial photography.

Cell	**Substrate**	**rSET #**	**Zone**	**Dominant Species**	**Dieback Intensity**	**Mean Distance from Inlet (m)**	**Mean 2015 Rate of Elevation Change (mm y^{-1})**
Cell 3D	DM	1-3	LM	*S. alterniflora*	L	390 ± 7.9	9.5 ± 0.97
	DM	4-6	LM	*S. alterniflora*	H	98 ± 7.6	5.6 ± 1.25
Cell 1A	DM	1-3	LM	*S. alterniflora*	H	323 ± 27.6	6.8 ± 0.77
	DM	4-6	HM	*S. patens*	-	360 ± 33.8	5.4 ± 1.20
	DM	7-9	LM	*S. alterniflora*	L	97 ± 20.5	11.9 ± 0.55
	DM	10-12	Creek Bank	*S. alterniflora*	H	312 ± 7.1	0.2 ± 6.06
Cell 4D	Sand	1-3	LM	*S. alterniflora*	-	209 ± 43.5	7.4 ± 1.36

The mean rate of elevation change in the Cell 1A high marsh group (5.4 mm y^{-1}) was significantly lower than the mean rate for the interior low marsh (P = 0.026). There was not a significant difference (P = 0.83) between the low marsh rSETS located in the dredged material marshes (Cells 1A, excluding creek bank sites, and 3D) and the sandy reference marsh (Cell 4D). This is not surprising considering the widely varying rates among the different low marsh rSET groups in the dredged material marshes. There was not a significant correlation (P = 0.51) between the rate of elevation change at individual rSETs and the straight line distance to the tidal inlet (Figure 2-5a). There was a highly significant correlation (P = 0.001) between the rate of elevation change at individual rSETs and the 2014 end of season total aboveground biomass standing stock at the nearest biomass sampling station (Figure 2-5b).

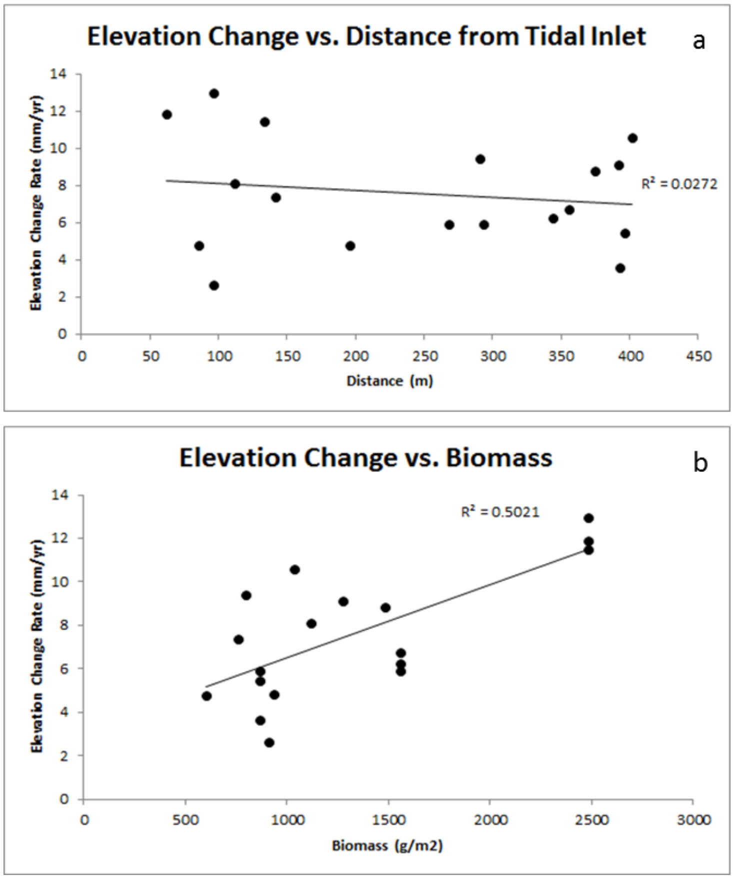

Figure 2-5. Rate of elevation change at individual rSETs versus distance from tidal inlet (a), and versus 2014 end of season total aboveground biomass standing stock (b).

Overall, the rate of elevation change in the low marsh of all three cells appears to be slow after first 2 – 3 years of monitoring (Figure 2-6b, d, f), and variance increases within each cell as the group rates diverge (Figure 2-6a, c, e).

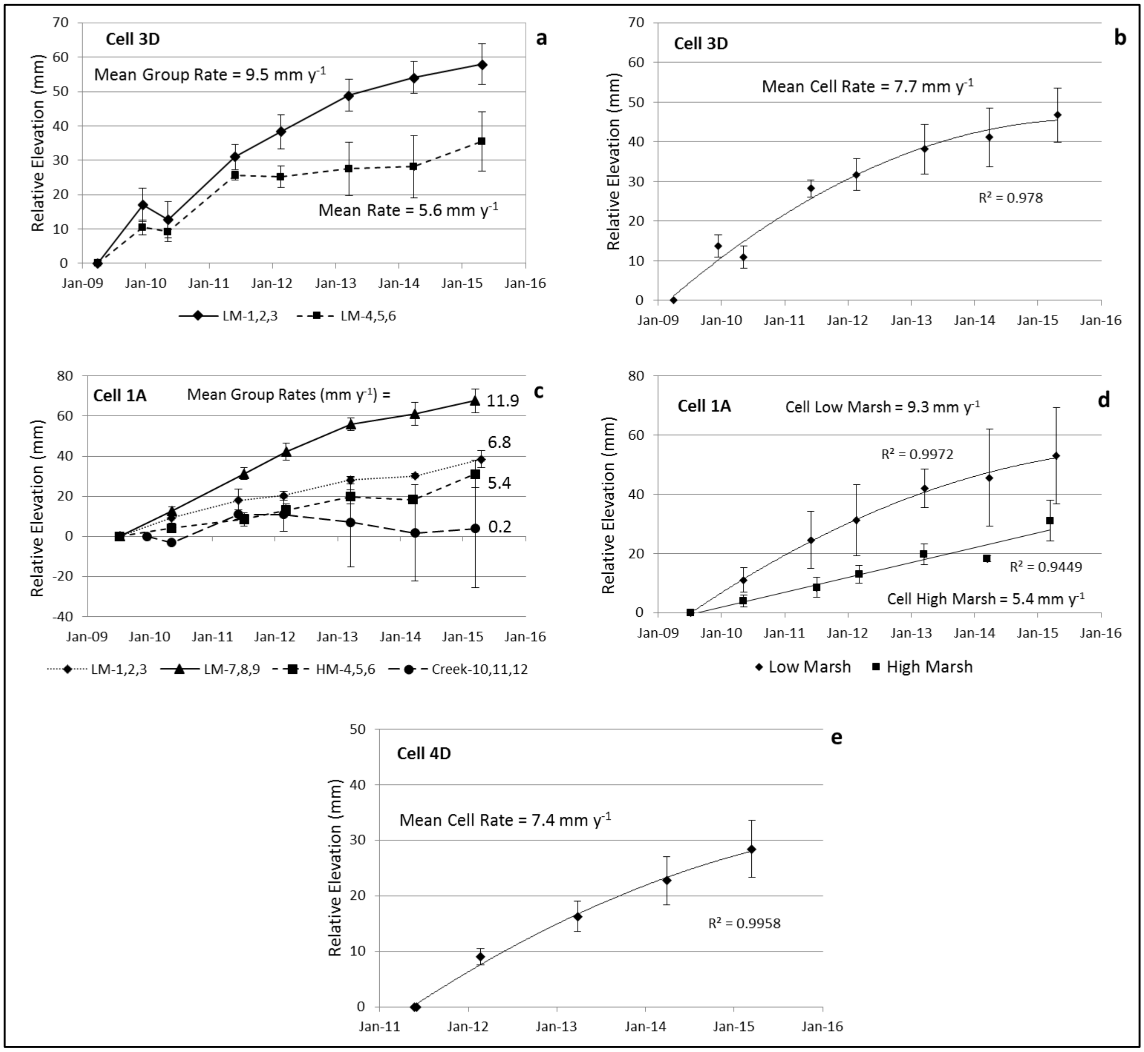

Figure 2-6. Elevation change in Cell 3D (a, b), Cell 1A (c, d) and Cell 4D (e). Rates of change calculated from 2015 measurements. Group means (n = 3) plotted for Cells 3D (a), 1A (c) and 4D (e). Mean rates of change for all rSET within a cell are plotted with a best fit regression line for Cell 3D (b, n = 6) and 4D (e, n = 3), and for Cell 1A (d) low marsh (n = 6, excluding creek bank rSETs) and high marsh (n = 3). Bars represent standard error.

SET Ground Survey Comparison

Overall there is agreement between estimates of elevation change in Cell 1A based on the USACE ground surveys and rSETs (Figure 2-7). Outliers include the creek bank rSETs, which are not representative of the entire section, and one interior low marsh rSET. The latter was located in a section with a high number of ground survey points that were likely affected by erosion near the creek banks, likely from the activity of *Ondatra zibethicus* (muskrat), but which did not affect the rSET location. A regression analysis excluding these four points shows a significant (P = <0.001) relationship, with r^2 = 0.974.

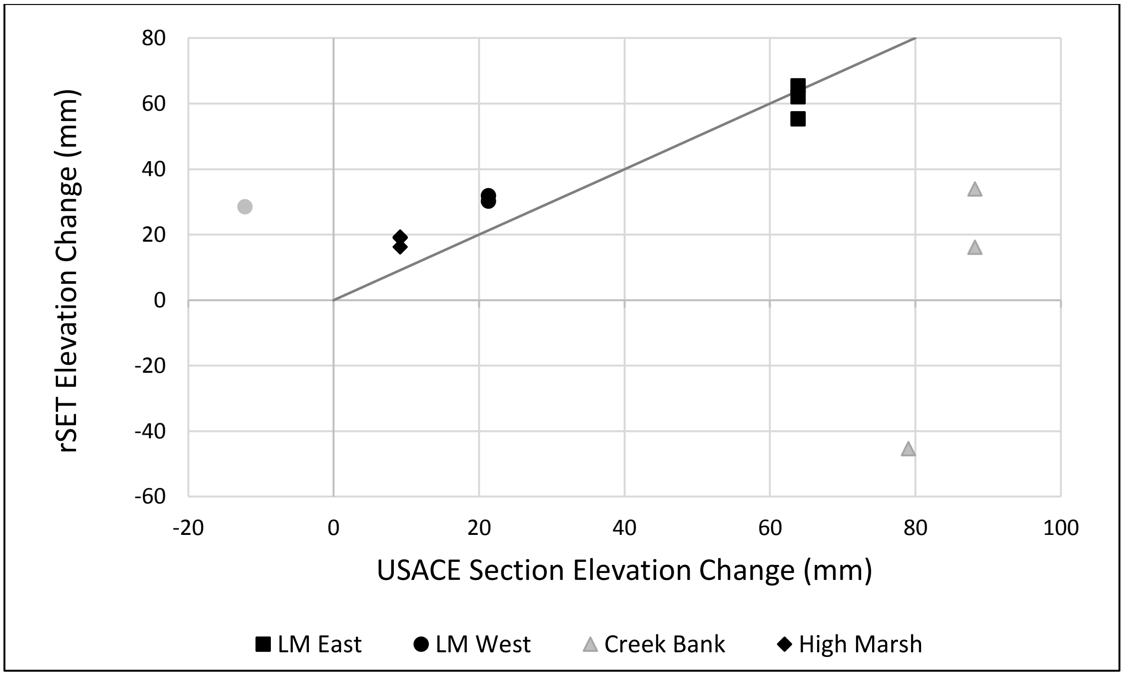

Figure 2-7. Elevation change at individual SETs compared with mean elevation change in the corresponding USACE section. LM = interior low marsh; solid line represents 1:1. Light gray points were outliers.

Marker horizons

In April 2007, one year after deployment, accretion at 14 marker horizon sites in Cell 3D ranged from 0 to 11.5 mm y^{-1}, with an average of 6 ±1.2 mm y^{-1} (Stevenson et al. 2009)). The composition of the accreted material above the marker horizon was similar to the 0 - 10 cm averages from the original solid phase sediment characterization of the marsh (Stevenson et al. 2008). Loss on ignition ranged from about 3–7 %, similar to the 0-10 cm average at most of the sites, with one site (8) having LOI > 25, indicating an accumulation of organic plant material (Stevenson et al. 2008).

In February 2011, when 10 of the same marker horizons were sampled, the mean surface accretion rate was 6.2 ±0.77 mm y^{-1}. Loss on ignition was variable, with 5 sites showing a substantial increase over previous years, and the remainder showing little change (Stevenson et al. 2013).

New marker sites were installed on March 14, 2014 (Cells 1A, 1C and 4D) and April 10, 2014 (Cell 3D). After approximately one year, the new sites were sampled on May 15, 2015 and averaged 4.0 mm in Cell 4D, 6.0 in Cell 3D and 2.0 in Cell 1A. The longer term rates of vertical accretion estimated by rSETs are 7.4, 7.7 and 9.3 mm y^{-1} in Cells 4D, 3D and 1A, respectively.

Tidal Flux

Tidal exchange monitoring in Cell 1B indicated a net export of TSS during four of six periods of ranging from 75-170 kg day^{-1}, and net import during two periods ranging from 25 to 44 kg d^{-1} (Table 2-3). Monitoring occurred during July and November during both 2013 and 2014.

Table 2-3. Total suspended solids (TSS) concentrations and tidal exchange during six monitoring periods. Positive values represent import and negative values represent export from the marsh system.

Date	Range ($mg L^{-1}$)	Mean ($mg L^{-1}$)	C:N	Net Flux ($kg d^{-1}$)	Net Flux ($g m^{-2} d^{-1}$)
July 2013	20.42 – 68.39	36.94	6.04	-420	-3.74
November 2013	7.39 – 22.74	12.51	7.17	-32	-0.28
February 2014	3.01 – 42.18	11.19	6.48	-243	-2.16
May 2014	8.09 – 29.52	16.12	6.18	-165	-1.47
July 2014	21.31 – 43.37	30.60	6.74	-6	-0.05
November 2014	4.31 – 18.80	8.65	7.02	-30	-0.27

The daily net flux estimates were used to calculate quarterly flux rates (averaging the two July and November fluxes, respectively). Using the quarterly rates, a net export of TSS of 67,874 kg y^{-1}, or 0.60 kg m^{-2} y^{-1} was found (Table 2-4). Using a bulk density of 1.1 g cm^{-3}, this amounts to about 0.6 mm y^{-1} eroded from the marsh and creek surface of Cell 1B (11.22 ha) annually. The mean volume averaged concentration of TSS over the study was 19.3 and the mean C:N ratio of TSS was 6.6.

Table 2-4. Estimate of annual net flux of TSS using daily flux estimates to calculate quarterly estimates from Cell 1B tidal flux study. Positive values represent import; negative values represent export.

Month	Quarter (months)	Daily Rate (kg d^{-1})	Quarterly Rate (kg 3 $month^{-1}$)
February	1st (J-F-M)	-78.69	-13,691
May	2nd (A-M-J)	-78.24	-13,763
July	3rd (J-A-S)	-211.5	-37,618
November	4th (O-N-D)	-15.76	-2,801
Annual (kg y^{-1})			**-67,874**
Annual (kg m^{-2} y^{-1})			**-0.60**

Decomposition

The rate of *S. alterniflora* decomposition in all marshes was highest at 200 - 300 days and 400 - 500 days, coinciding with peaks in soil temperature (Figure 2-8a, b). There was approximately 20-40% of the original plant mass remaining at the end of 2012, and approximately 15-30% remaining at the end of the study in December 2013 (Figure 2-8a). Decomposition occurred most rapidly during the first year in Cell 1A, where the plant tissue contained a higher initial percentage of N (1.33%) compared to Cells 3D, 1C and 4D (0.97%, 0.84%, and 0.63%, respectively). The molar C:N ratio of the plant material remaining at the end of the study was approximately 20.

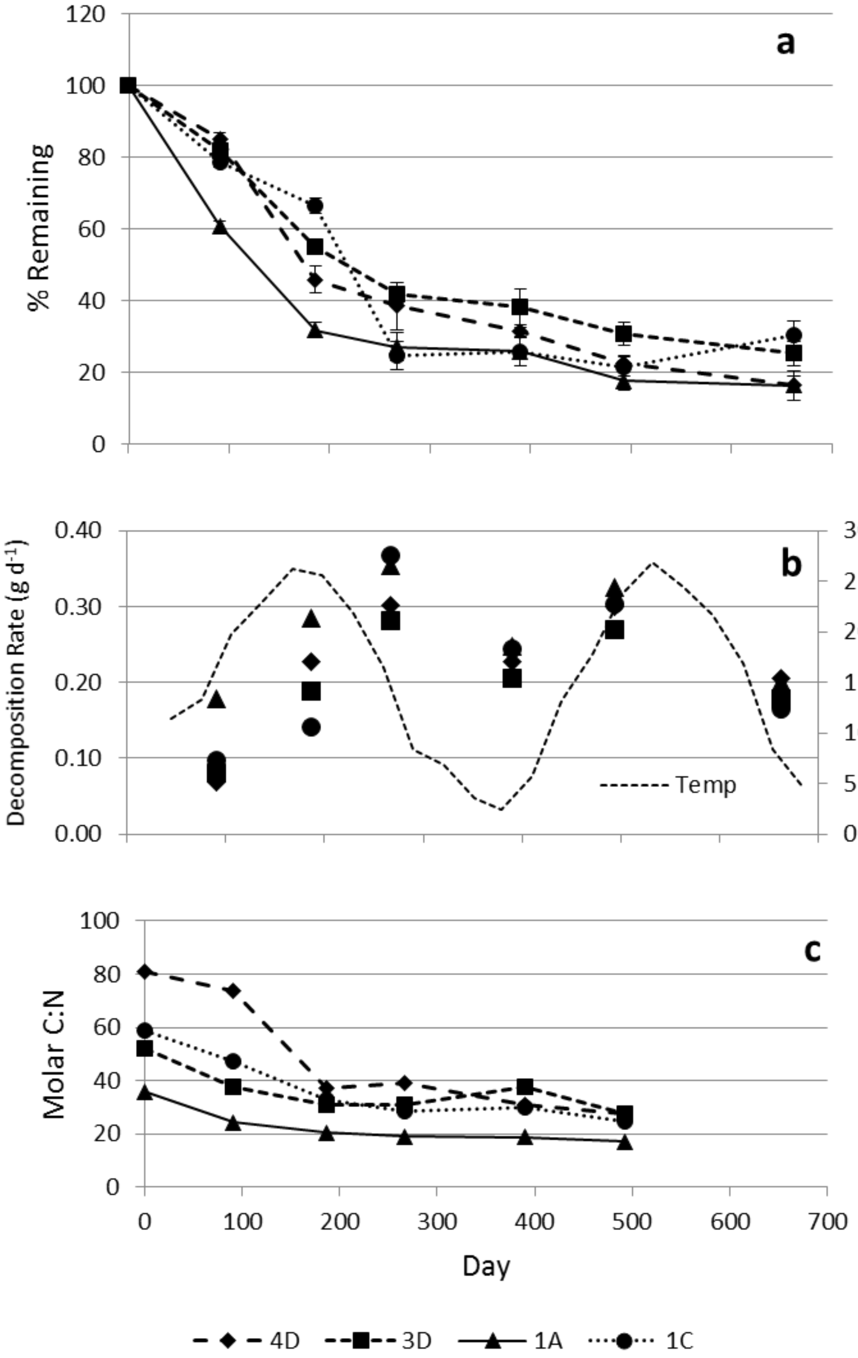

Figure 2-8. Percent biomass (± SE) remaining in litter bags (n=3) (a); decomposition rate since previous sample date (symbols) and average monthly soil temperature (dotted line) (b); and molar C:N ratio of remaining litter (c). Time 0 = February 17, 2012.

Discussion and Conclusions

Elevation trends

The key to the survival of tidal marshes in the face of SLR, including the constructed marshes at Poplar Island, is their ability to maintain a vertical accretion rate equivalent to relative SLR. Using the appropriate rate of SLR to compare with rates of elevation change is essential, but complicated due to large differences in estimates related to the length of the tide gauge records used. The long term historical rate of SLR (e.g. Baltimore) is not appropriate for comparisons with rates of elevation change during short time spans, especially when it seems clear that the rate of SLR has been increasing during the last few decades. Using the SLR rate during the previous 20 years likely makes a better comparison, but tidal marshes respond to much shorter term changes in water level. For example, *Spartina patens* has been observed to initiate adventitious roots when exposed to inundation for just three weeks (Nyman et al. 2006).

Over all, the rates of vertical accretion in both the dredged material marshes and the sandy reference marsh at Poplar Island are clearly high enough to keep abreast of the 20 year rates of SLR at Baltimore and Solomon’s. They are in the higher end of the range reported for natural marshes in the region. Only one group of low marsh rSETs, however, indicate accretion rates high enough to keep up with the mid-Chesapeake SLR trends since 2008, when the first rSETs were installed. More importantly, although the 5 year rate of SLR has been especially high, there has been a slowing of the rate of elevation gain in all three Poplar Island marshes, based on best fit regression models (Figure 2-6b, d, f). Thus, it appears that in most areas of the Poplar Island marshes, vertical accretion is at best marginally keeping up with SLR. This conclusion seems to be supported by field observations of *S. alterniflora* colonizing the high marsh zone in some areas of all three marshes, and by the increasing conversion of low marsh area to mudflat, evident in aerial photography (Figure 2-9b).

The mean rate of accretion in Cell 4D (Figure 2-6f), the low nutrient marsh, was not significantly different from the mean of interior low marsh rSETs in the dredged material cells due to the variance among the dredged material rSETs. However, when the dredged material rSETs are separated into replicate groups, the Cell 4D rate is comparable to the low marsh groups more affected by dieback. Annual aboveground biomass production is lower in Cell 4D than the dredged material marshes, on average, but belowground biomass production is higher (Chapter 1). If belowground biomass production were the dominant process in vertical accretion, as has been suggested for some marshes (Turner et al. 2004), rates of vertical accretion would be expected to be higher in Cell 4D marsh where BG biomass is higher. Furthermore, lower rates of organic matter decomposition under lower nutrient conditions could enhance accretion rates (Marinucci et al. 1983), although in this study decomposition rates were similar in the dredged material and sand marshes except for Cell 1A, which had a higher initial tissue N concentration. In these marshes, belowground biomass production does not appear to produce accretion rates equivalent to SLR, suggesting that enhanced *Spartina* spp. AG biomass production resulting from nutrient availability in the upper Bay dredged material is beneficial in terms of accretion where dieback has not been too intense.

Table 2-5. Reported estimates of elevation change in tidal marshes along the Atlantic coast based on SET measurements.

Location	**SLR (mm y^{-1})**	**Elevation Change (mm y^{-1})**	**Dominant Vegetation**	**Reference**
Nauset Marsh, MA	2.6	2.7	*S. alterniflora*	Erwin et al. 2006
Little Beach, NJ	4.1	1.7	*S. alterniflora*	Erwin et al. 2006
Wachapreague, VA	3.9	0.7-1.4	*S. alterniflora*	Erwin et al. 2006
Mockhorn, VA	3.9	1.4	*S. alterniflora*	Erwin et al. 2006
Jug Bay, MD	2 - 4	4.1 – 7.1	*Tidal Fresh*	Delgado et al. 2012
Rhode River, MD	2 - 4	1.3 – 11.9	*S. patens, Schoenoplectus americanus*	Childers et al. 1993
Patuxent R., MD	2 - 4	-16.2 – 20.7	*S. patens*	Childers et al. 1993
North Inlet, SC	2 - 4	-7.9 – 6.4*	*S. alterniflora*	Childers et al. 1993
Blackwater, MD	3.5	3.8-5.0	*S. patens, Schoenoplectus americanus*	Cahoon et al.
Poplar Island, MD	12.3	5.4	*S. patens*	This study
Poplar Island, MD	12.3	7.4 – 11.9	*S. alterniflora*	This study

*Except 11.9 mm y^{-1} where wrack was deposited and contributed extra organic matter.

Increased AG biomass may not only contribute directly to vertical accretion through organic matter deposition, but also improves the ability to trap TSS during tidal flooding (Baustian et al. 2012). In the microtidal system where the Poplar Island marshes are located, and where TSS concentrations are relatively low, macrophyte biomass production is likely more important that sediment trapping. The strong association between annual AG biomass production and elevation gain and lack of correlation between accretion rates and distance to tidal inlets support this conclusion.

The design of the Poplar Island marshes may contribute to the importance of organic matter retention. Tidal exchange is restricted to the inlets by the three meter dikes surrounding the marshes, so ice scour, king tides and storm surges cannot remove large amounts of material from the marshes. Most of the AG biomass produced in the marshes is likely retained on the marsh surface. In this situation, dieback has both direct and indirect impacts on the marsh via reductions of organic matter production and, secondarily, the trapping ability of the macrophyte detritus.

The average accretion rate at the high marsh rSETs in Cell 1A (5.4 mm y^{-1}) is just over half the average for all interior low marsh rSETS (Figure 2-6c). The high marsh areas may benefit from the transport of material off the dike slopes surrounding each marsh, in addition to deposition during tidal flooding. Presently there is little evidence that they are receiving enough material from either source to prevent conversion to low marsh, a process which already appears to be occurring, indicated by colonization of the some original high marsh zones by the low marsh species, *S. alterniflora*.

The difference between the high and low marsh zones is likely attributable to several factors. Tidal flooding is one, but biomass production is likely at least as important. *Spartina patens* produces less belowground biomass than *S. alterniflora* at Poplar Island (Chapter 1), and although *S. patens* total aboveground biomass is often higher than *S. alterniflora*, a large proportion is usually dead material from previous years (Chapter 1). With higher elevation, the high marsh zones at Poplar Island are designed to be flooded less frequently than the low marsh, and consequently appear to be drier. Thus slower decomposition is expected, with less opportunity for deposition and accumulation of organic or inorganic material in flood water.

Contrary to expectations, a relationship was not found between elevation change and distance from the tidal inlet, suggesting that flood water entering the marsh was not an important source of material for surface accretion at any of the rSET locations. Mean volume-weighted TSS concentrations during the tidal flux study varied throughout the year, peaking in July at ~30 mg L^{-1}, concentrations not likely to support vertical accretion in a microtidal system like Chesapeake Bay. Marker horizon data showed, however, that early in the development of the Poplar Island marshes the material accreting on the marsh surface had characteristics similar to the dredged material, suggesting that erosion and re-deposition within the marsh provides a small source of material that is transported on to the marsh platform during tidal flooding. Creek bank erosion, evident in the creek bank rSET measurements and visible in aerial photography (Figure 2-9a), appears to be the source of this material. The net annual export of TSS indicated by the tidal flux study also suggest that there is erosion occurring in the marsh, but it is localized rather than uniform, indicated by deposition over the marker horizons.

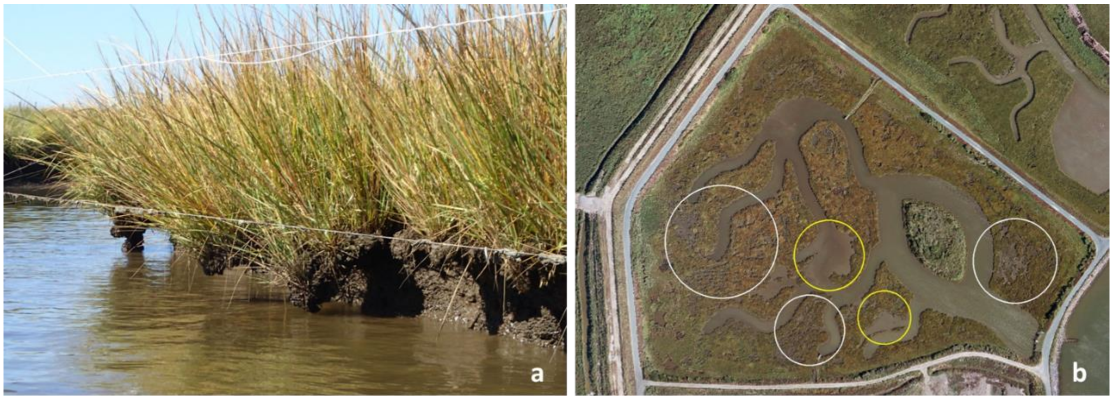

Figure 2-9. Creek bank erosion in Cell 1A; white lines in foreground mark the original edged of the marsh (a). Aerial photo (2014) of Cell 3D with areas which have converted to mudflat circled in yellow, and areas with thinning vegetation circled in white (b).

The C:N ratio of TSS (6 – 7) suggest that it is dominated by inorganic material and phytoplankton most of the year, with enrichment of macrophyte material after senescence in late fall. In the decomposition study here, initial *S. alterniflora* C:N ratios were ~40 – 80, declining to ~20 after nearly two years of decomposition in the field. Thus the November peak in C:N ratio may represent an export of organic matter from the marsh surface.

Slowing rates of accretion over time in natural marshes have been considered part of the maturation process over long time scales (Hazelden and Boorman 1999). In the Poplar Island marshes, however, the slowing accretion rates over such a short time scale (≤7 years) raises questions about marsh sustainability, especially with the anticipated escalation of SLR. Several mechanisms may contribute to elevated rates of accretion early in the development of constructed marshes, including rehydration of dredged materials in the first few years following opening to tidal exchange, rearrangement of sediment eroded within the marshes, especially off the creek banks and very high macrophyte production in the first year or two, prior to the onset of dieback. The decline after the first few years may be due to increasing levels of dieback, sporadic insect infestations or fungal infections which lag behind plant introductions, or to declining nutrient supplies. The latter should increase belowground biomass production, but may not be sufficient to offset losses in aboveground production.

A similar pattern of slowing accretion is reported for restored marshes in former polder areas of the Dutch Wadden Sea (Muller et al. 2013). The changing accretions rates are attributed to a maturation process where low elevation in young marshes promotes surface accretion through trapping, leading to high burial rates, because they are flood dominated. Older marshes are thought to be in equilibrium with sea level and have shorter and less frequent inundation, and therefore trap less, become ebb dominated, and tend toward export of nutrients (Boorman, 1999; Hazeldon and Boorman 1999).

Expansion due to rehydration of dredged material in the Poplar Island marshes is a potential source of vertical accretion. Most rehydration occurs occurrs within the first year following opening of the tidal inlets (M. Snyder, USACE, pers. com) and could contribute to the higher initial accretion rates. However, effects related to rehydration would presumably not be an important factor in the sandy substrate of Cell 4D, where there is also an apparent slowing in the rate of vertical accretion.

Creek bank erosion has been observed in Cell 4D as well as the dredged material marshes and declining erosion could be a factor in the slowing accretion rates as the creeks reach equilibrium configurations (Fagherazzi et al. 2012). Although the time frame at Poplar

Island is uncertain, it seems likely that erosion is most intense in the first few years after planting, and the increasing organic content of the sediment above the marker horizons between 2008 and 2011 may indicate that this is occurring. Erosion may continue, however, as a result of destabilizing factors such as vegetation dieback and muskrat herbivory which both continue at Poplar Island. Sediment nutrient levels are declining in the root zone, but do not appear to have reached limiting levels yet. Vegetation dieback appears to have a negative influence on accretion rates, but the slowing accretion rates in all three marshes suggests that it isn’t the single factor driving the decline. It may be that the marshes are approaching a threshold, already reached in the worst dieback areas such as the one monitored in Cell 3D, where accretion rates are subject to negative feedback resulting from increasing water levels and declining macrophyte biomass.

Without intervention, it appears that some areas of the dredged material marshes at Poplar Island where dieback has been intense may soon convert to mudflat. Efforts to determine the causes of and to reduce dieback are warranted. Improved monitoring of the different components of elevation change through paired marker horizons with rSETs, cryogenic coring of marker horizons and paired vegetation monitoring can help refine our understanding of surface and subsurface processes at Poplar Island and elsewhere in microtidal environments. A better understanding of accretionary processes will help improve designs to make future projects utilizing fine grained dredged material more resilient to SLR. This may be achieved by something as simple as layering sand over the dredged material to allow access to nutrients, which enhances production, while improving drainage, which would reduce the build-up of sulfide and toxic metabolites of anaerobic respiration, thereby reducing the risk of dieback. Connecting upland sites to the marshes to increase the supply of inorganic material may also improve resilience by augmenting vertical accretion in the marshes after the redistribution of creek bank material has diminished.

Conclusions

Based on the data presented here, it appears that the stability of the Poplar Island marshes under current and future rates of SLR is uncertain despite the fact that Poplar Island has some of the highest sediment accretion rates along the Atlantic Coast. While overall rates of accretion at Poplar Island exceed historical rates of SLR for the region, most are below more recent rates estimated for the period since the SETs were installed, which are some three times the historical rates. If this is a temporary situation the marshes may remain stable, but if they are part of a long-term increase in SLR, it appears the Poplar Island marshes will succumb.

The healthier low marsh areas of dredged material marshes have accretion rates that exceed the low nutrient marsh, but periodic diebacks in the dredged material marshes appear to result in lower accretion rates, similar to the low nutrient marsh. Efforts to keep the rate of vertical accretion positive with respect to SLR may include efforts to control vegetation dieback in the low marsh as well as connecting the upland areas to the marshes to provide a terrestrial source of accretionary material.

The difference between the dredged material and low nutrient marshes appears to be the higher AG macrophyte production in the dredged material marshes. Overall, organic matter production is likely more important than inorganic matter deposition to vertical accretion in the all the constructed marshes once the creek banks reach a stable configuration, because the mid-Chesapeake Bay region does not provide a large source of inorganic material to the marshes. Aboveground production likely dominates BG production in the dredged material marshes due to the N induced low RSR, and the retention of autochthonously produced organic material resulting from the limited connection with the estuary.

The lower rates of accretion in dieback areas of the dredged material marshes may indicate that they are nearing a threshold from which they cannot recover. If this is the case, much of the created marshes at Poplar Island will be vulnerable if dieback persists. Migration of the low marsh vegetation (*S. alterniflora*) into the original high marsh zones suggest this process is under way. In dieback areas where vertical accretion has stalled and biomass production is low, directly augmenting them with new material through thin layer application (Ford et al.1999) may be the only way of keeping them from converting to open water.

Chapter 3: Silicon in a eutrophic tidal marsh restoration: An estimate of pools, fluxes, and the potential benefits of a soil amendment

Introduction

Background

Silicon (Si) is essential for diatom production, which forms the base of the food web in estuaries and coastal areas and supports commercially valuable fisheries. Terrestrial vegetation and weathering of terrigenous rock provide Si to these systems primarily via riverine transport (Tréguer et al. 1995; Conley 1997), but also as groundwater inputs (Ragueneau et al. 2006). As the link between terrestrial sources of Si and coastal ecosystems which require it, there is speculation that tidal marshes play a key role in regulating primary productivity not just through their influence on nitrogen (N) and phosphorus (P) processes and organic matter production, but also on Si cycling and availability (Hackney et al. 2000).

Coastal eutrophication has disrupted the historically diatom based food web in many areas, including Chesapeake Bay (Officer and Ryther 1980; Conley and Malone 1992; Cooper and Brush 1998). Nitrogen enrichment can lead to Si depletion early in the growing season (Conley et al. 1993), promoting the growth of non-diatomaceous species of phytoplankton, especially dinoflagellates, and is hypothesized to be behind the proliferation of harmful algal blooms in some coastal areas (Ragueneau et al. 2006). Thus, although Si is an abundant element globally, seasonal depletion in Chesapeake Bay and other estuaries has led to decreased diatomaceous productivity and allowed other algal species to proliferate in summer (Cooper and Brush 1991).

Recent reviews have pointed out the need for more studies which examine the potential impact of marshes on Si cycling, and particularly of tidal marshes which border estuarine and coastal waters where seasonal Si depletion occurs (Struyf and Conley 2009; Schoelynck et al. 2014). In drier terrestrial ecosystems Si deposited in plant tissue (phytoliths) can persist in the soil indefinitely, making them useful to paleobotanists who can often reconstruct ancient plant communities based in part on phytolith presence (Piperno 2009). Conley et al. (1993) report that lakes, which promote burial with relatively quiescent waters and long residence times, may sequester up to 90% of biogenic silica (BSi). However, because phytolith solubility is several orders of magnitude greater than mineral silicates (Van Cappellen 2003) groundwater flow and tidal flooding in marshes promotes dissolution of buried phytoliths (Hackney et al. 2000), leading to leached soils in more mature (indicated by elevation) marshes (Müller et al. 2013). Exposure to seawater increases the rate of phytolith dissolution due to higher pH and the catalytic effect of cations (e.g. Mg^{+2} and Na^{+}) which are abundant in sea water (Loucaides et al. 2008). Thus, regular flooding in tidal salt marshes further promotes Si cycling, resulting in greater fluxes of DSi from tidal salt marshes than from tidal freshwater marshes (Struyf and Conley 2009).

In Chesapeake Bay, tidal marshes are being constructed at Poplar Island, Maryland with dredged material from upper-Bay shipping channels. The fine grained material is largely of terrigenous origin (Cornwell and Owens 2011), and is therefore a potentially rich source of Si. Yet it is also a rich source of N, and has resulted in *S. alterniflora* that shows distinctive N induced stress responses, including low root:shoot ratios, lodging, high rates of fungal infection and intense grazing pressure, all of which may contribute to the frequent vegetation diebacks observed in these marshes (Chapter I). The site differs from many tidal marsh restoration projects conducted during the last 30 years in the use of this fine-grained, nutrient rich dredged material, which contrasts with coarse-grained substrates, containing low to moderate nutrient levels used in many projects, and has produced very different trajectories of plant growth and soil development (Craft et al. 1999). Currently there are similar restoration projects utilizing dredged material under way in other nutrient rich environments (e.g. the Gulf Coast of Louisiana; Costa-Pierce and Weinstein 2002).

Few natural marshes would be exposed to the levels of N found in agricultural systems, but marsh restoration projects utilizing dredged material from eutrophic estuaries may be subject to a very high N supply over many years, and in a form (NH_4^+) preferred by *S. alterniflora* (Mendelssohn 1979). Thus far, however, Si cycling in these types of restored marshes has not been examined.

The effects of N on tidal salt marshes are becoming clearer, but evidence of the impact on Si cycling in those marshes is contradictory. Silicon imports and exports have been estimated for salt marshes from only a few locations (Norris and Hackney 1999; Struyf et al. 2006b; Vieillard et al. 2011; Carey and Fulweiler 2013; Müller et al. 2013; Carey and Fulweiler 2014), making it hard to evaluate the influence over coastal and estuarine Si budgets. Furthermore, eutrophication, climate change and rising sea level are affecting salt marshes worldwide (Stevenson and Kearney 2009) and the impact of each and in combination is unclear. There are no estimates of Si pools or fluxes from created marshes, which may be expected to be very different depending on the source of the substrate, and therefore comparisons of Si budgets in natural and created marshes have not yet been made. The potential benefits to *S. alterniflora* of adequate Si in extremely high nutrient environments has also received little attention (Querne et al. 2012; Carey and Fulweiler 2013), and studies of Si amendments in marsh restoration have not been reported. The creation of tidal marshes at Poplar Island provides an opportunity to address the ramifications of Si dynamics in constructed tidal marshes and fluxes to nearby waters, as well as the potential for Si amendments to improve *S. alterniflora* resistance to stress imposed by exposure to extremely high N levels.

Silicon as a beneficial nutrient

Silicon (Si) is the second most abundant element in the earth's crust, after oxygen, and occurs mainly as alumino-silicate minerals in soils. It also occurs in solution as silicic acid (H_4SiO_4) in soil porewater following mineral weathering, a process mediated in part by plant roots (Brezonik and Arnold 2011), in concentrations ranging from 0.1-0.6 mM (Epstein 1994).

Although Si has not historically been considered essential for growth of most terrestrial plants, it is now known that Si promotes survival of many taxa, including the Bryophyta (bryophytes), the Lycopsida and Equisetopsida of Pteridophyta (ferns), and among the angiosperms the Cyperaceae and Gramineae (Hodson et al. 2005). Even in plants for which Si is not considered essential, the range of plant tissue concentrations (0.1-10% Si) is equivalent to those elements considered macronutrients (e.g. N), making it a major mineral constituent of many plant groups (Epstein 1991, 1994).

Plant uptake occurs passively in the transpiration stream, actively by transporters in the roots, or Si can be selectively rejected (Ma et al. 2001). Within the plant it is transported as silicic acid via the xylem and deposited throughout the plant as amorphous hydrated silica ($SiO_2 \cdot nH_2O$), referred to as plant opal, silica gel or biogenic silica (BSi). In higher plants Si is deposited as phytoliths, morphologically distinct structures which can be taxonomically characteristic (Sangster et al. 2001). The transformation to amorphous Si is irreversible during the life of the plant, so remobilization does not occur and accumulation continues until senescence (Epstein 1994; Norris and Hackney 1999).

Agronomic research began to focus on Si around the turn of the 20th century in Japan, where organic (and later inorganic) nitrogen fertilizers were being used to increase the yield of paddy-grown rice, *Oryza sativa* (Ma and Takahashi 2002). Nitrogen fertilization improves crop yields by stimulating shoot growth, which can lead to growth induced deficiencies of other nutrients, including Si, and a number of undesirable changes in plant morphology and physiology which can ultimately reduce yields. Morphological changes such as increased height and internode length leads to stem lodging, where tall thin stems are blown over by wind; wider and thinner leaves droop, reducing light interception efficiency; and exudation of amino acids on leaf surfaces increases susceptibility to pathogens and predators (Marschner 1995). As a consequence, heavily fertilized crop plants are often treated with pesticides, fungicides and plant hormones in order to maintain the higher yields that are the goal of N fertilization. The application of Si amendments to rice paddy soils, which often become leached from continuous flooding, increases Si availability, raises plant tissue Si concentrations and improves resistance to both biotic and abiotic stresses (Ma and Takahashi 2002). Several other crop species, including sugarcane (*Sacharinum sacharinum*), turfgrass (*Poa spp., Agrostis spp., Zoysia spp.*), and banana (*Musa* spp.) are now also being treated with Si soil amendments to counteract the negative effects of N fertilization while minimizing the use of harmful chemicals.

The beneficial effects of Si are most apparent under biotic or abiotic stress, where Si appears to work in several ways to improve plant fitness. Deposition in stems and leaves imparts rigidity which can improve the upright stature of leaves (improving light interception) and resistance to lodging; deposition beneath the cuticle helps armor plant leaves from bacterial and fungal infection and predation by grazers, all of which increase under high N fertility. Silicon can also help ameliorate the effects of metal toxicities, drought and salinity stress, can moderate the uptake of other nutrients and plays a role in the enzyme activities involved in plant defense (Epstein 2001).

Lanning and Eleuterius (1981, 1983) speculated that coastal eutrophication could impose stress on marsh grasses such as *S. alterniflora*, the dominant species in North American tidal salt marshes that could be related to Si deficiencies similar to heavily fertilized crops. In a survey to provide a baseline of Si plant tissue concentrations in coastal areas that were still considered pristine, *S. alterniflora* was one of the higher Si accumulating species (Lanning and Eleuterius 1981). Plant taxa are categorized by their ability to take up and accumulate Si as “accumulators,” with plant tissue concentrations >1%, “intermediate,” with tissue concentrations of 0.5-1%, or “non-accumulators,” with tissue concentrations < 0.5% (Ma et al. 2001). *Spartina alterniflora* is generally considered intermediate, with tissue concentrations ranging from 0.52 - 2.28% SiO_2 (Table 3-1).

Silicon in marshes

An extensive literature exists with respect to nitrogen and phosphorus effects on *Spartina alterniflora,* from the scale of individual plants (e.g. uptake kinetics) to the ecosystem scale, where recent work suggests that long-term exposure to high levels of N and P undermine the structural integrity of salt marshes, leading to peat collapse and increasing vulnerability to sea level rise (Turner 2011; Deegan et al. 2012). There have been few similar studies of Si in marshes.

Agronomic research has focused on the ability of Si soil amendments to overcome stress imposed by high rates of N fertilization and on yield improvement in some crops. The tidal marsh research that has been has been conducted is more concerned with the potential ecological impacts and the sustainability of coastal marshes increasingly affected by coastal eutrophication (Nixon 1995). Early work focused more on the plant level, establishing baseline tissue Si concentrations of marsh plants, but with an awareness that as eutrophication intensified marsh vegetation could suffer from high N loadings and N induced Si depletion (Lanning and Eleuterius 1981, 1983).

In one study the effects of Si amendments on tissue concentrations of marsh vegetation including *S. alterniflora* were examined in situ (de Bakker et al. 1999), but little enrichment was observed. The level of N enrichment in this study was not clear, but may have been below the level at which the plants would benefit from Si amendments, or the amendment application rate may not have been high enough to discern a difference from background levels. Querné et al. (2012) found that tissue Si concentrations in *S. alterniflora* were not related to the level of exposure to abiotic stress (wave action). However these two studies have added to the inventory of *S. alterniflora* tissue Si concentrations (Table 3-1). In the last decade studies have begun to focus more on the Si pools in tidal marshes and exchanges of dissolved Si (DSi) and BSi with adjacent waters, primarily from natural marshes (Struyf et al. 2006a; Struyf and Conley 2009; Vieillard et al. 2011; Querne et al. 2012; Carey and Fulweiler 2013; Carey and Fulweiler 2014).

Table 3-1. Concentration of silicon in the tissue of *Spartina alterniflora* shoots.

Author	Location	Percent SiO_2	Method
Lanning and Eleuterius (1981)	Davis Bayou, MS, USA	0.83	Gravimetric
Lanning and Eleuterius (1983)	Graveline Bayou, MS, USA	2.28	Gravimetric
Norris and Hackney (1999)	Cape Fear River, NC, USA	0.52	Wet alkaline digestion w/ $Na_2CO_3^b$
Carey and Fulweiler (2013)	Narragansett Bay, RI, USA	0.98-1.81	Wet alkaline digestion w/ $Na_2CO_3^b$
Querne et al. (2012)	Bay of Brest, France	0.33-0.68	Wet alkaline digestion w/ $Na_2CO_3^b$

Poplar Island

Nitrogen availability in the marshes at Poplar Island is considerably higher than reported for most natural marshes (Chapter 1), making this an ideal site to examine Si cycling. Key questions that can be addressed in the restored marshes at Poplar Island are related to both the role of restored tidal marshes in Si processing and the potential for Si to alleviate the stress of N enrichment and promote stability of restored marshes in eutrophic systems. In addition, the symptoms of stress imposed by excess N (lodging, fungal infection, grazing pressure, dieback) are widespread and persistent, and the potential benefits of Si amendments for *S. alterniflora* are expected to be most apparent under these conditions.

Fine grained dredged material which accumulates in the bottom of shipping channels from riverine sources and local shoreline erosion may represent a significant pool of sequestered Si. It is probable that Si in the root zone can be remobilized by the restored marsh and released into Chesapeake Bay, where seasonal Si depletion occurs (Conley and Malone 1992). Supplying Si to local waters to sustain diatom growth and ultimately contribute to secondary production, including commercially important fisheries, may more be important in a eutrophic system like Chesapeake Bay than in a more pristine coastal system, although the benefits may be outweighed by the export of N from the marshes. This potentially overlooked ecosystem service is addressed here by characterizing the pools of Si in the Poplar Island dredged material marshes and exchanges with the adjacent bay water to provide insight into how this type of restored marsh affects the estuarine Si cycle.

Tidal exchange in the Poplar marshes occurs through concrete box culverts, restricting transport of vegetation wrack and simplifying flow estimates, making them well suited to a tidal flux study. Estimates of the pools of Si in marsh vegetation, sediment and porewater were made from data collected in the marsh in 2012 and 2013. Recycling of Si assimilated by macrophyte vegetation was addressed in a decomposition experiment in 2012 - 2013, and exchange of DSi and BSi are reported for a series of flux studies carried out seasonally from 2013 - 2014. To place restored marshes in context, a Si budget for the Poplar Island marshes is compared to natural marshes receiving high and low nutrient inputs (Vieillard et al. 2011; Carey and Fulweiler 2013).

The potential for Si soil amendments to improve *S. alterniflora* plant fitness in an N rich environment was addressed in this study with a mesocosm experiment conducted at Horn Point Laboratory in 2012, followed by a two year field experiment at Poplar Island. The experiments were designed to address the ability to increase *S. alterniflora* tissue Si concentrations and changes in incidence of lodging, grazing, fungal infection rates, and ultimately dieback, to evaluate how long these benefits may persist.

Methods

Site description

This research was conducted at the Paul S. Sarbanes Environmental Restoration Project at Poplar Island (Poplar Island), located in mid-Chesapeake Bay (Figure 1-1) and described in detail in Chapter 1. The silicon study was carried out from 2012 to 2014 in Cell 1B, a 12.41 hectare tidal marsh planted in 2012 (Figure 3-1). A perimeter dike 3 meters in height surrounds the marsh, limiting direct tidal exchange with the estuary to two concrete box culverts (each 1.83 m x 1.83 m) armored with stone. Although this cell was connected via channels to adjacent marsh cells for fish passage, water transport was determined to be minimal during a several month current meter deployment at each channel (Nidzieko, pers. com. 2012).

`Silicon budget

Potential inputs were identified as atmospheric deposition, tidal exchange, and diffusion from the sediment below the root zone. Outputs are tidal exchange and burial. The pool of Si in marsh vegetation was estimated using *S. alterniflora* shoot and root tissue concentrations from the control subplots in the Si amendment experiment described below. These values were also applied to the high marsh for *S. patens*, which is reported to have tissue SiO_2 concentrations similar to *S. alterniflora* (Hodson et al. 2005). Annual shoot and root production in Cell 1B were estimated by harvesting biomass in the fall (September-October) in 2013 and 2014 (see Chapter 1 for complete description of methods). The tissue SiO_2 concentration of standing dead material was determined as part of a decomposition experiment described below. High marsh and low marsh pools were calculated separately and summed.

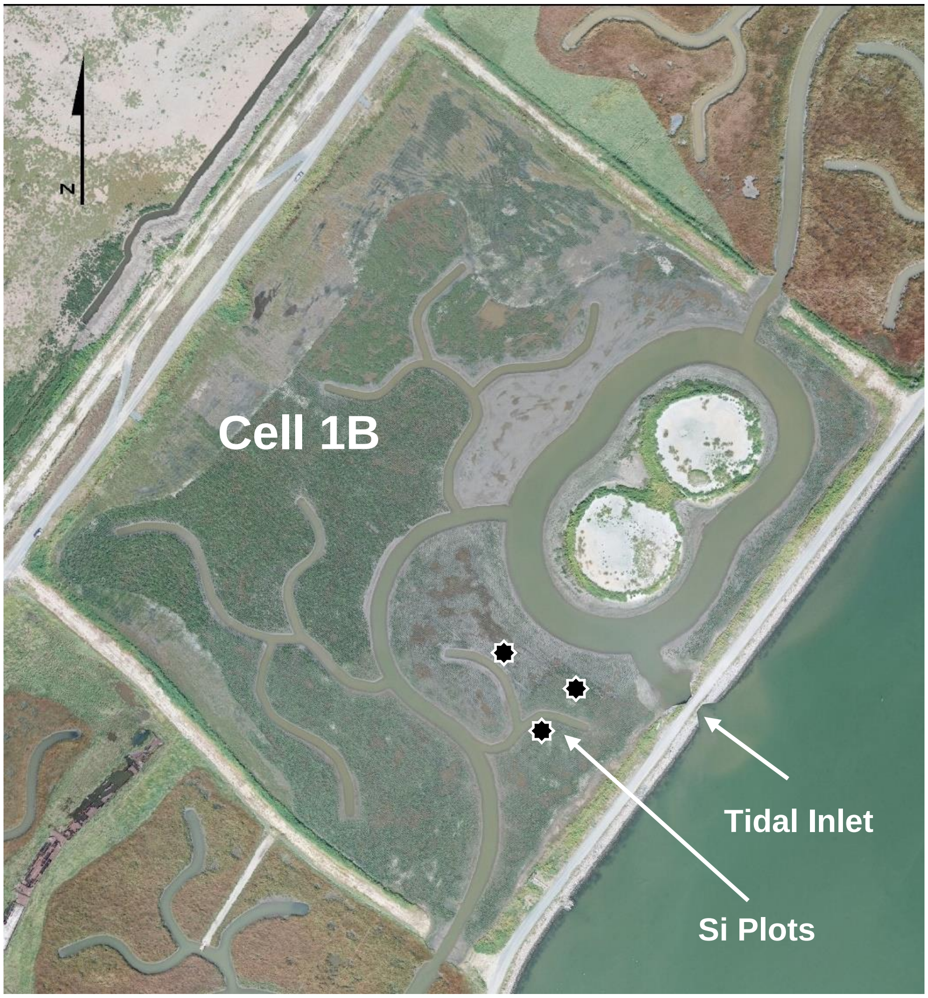

Figure 3-1. Map of Cell 1B at Poplar Island.

The sediment porewater DSi pool was estimated using concentrations measured in the control subplots of the Si amendment experiment in 2012 (HPL Biogeochemistry Lab) described below and porewater monitoring elsewhere in the cell in 2013 and 2014.

Porewater DSi ranged from approximately 200 to 500 µmol L^{-1} in the root zone (0 – 15 cm depth; Figure 3-2). The minimum value, 200 µmol L^{-1}, was used to calculate a conservative estimate of the root zone DSi pool. Sediment BSi measurements were made on the 18 solid phase samples collected in 2011, prior to planting, using the base digestion technique (Demaster 1981) modified by Saccone et al. (2006) followed by silicate analysis using the molybdate blue methodology (Technicon Industrial Method 186-72W-Modified) with ascorbic acid as the reductant on a Technicon AutoAnalyzer II system. The range among those samples was 2.34 – 3.45 %, and the mean value, 2.96 % (±0.07 se), was used to calculate the root zone BSi pool.

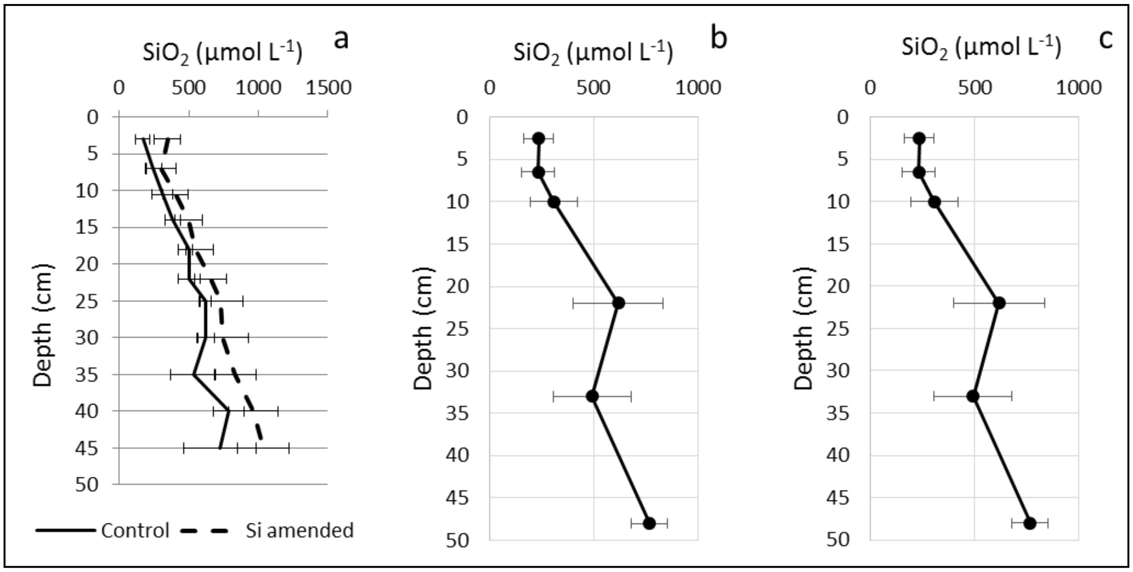

Figure 3-2. Porewater DSi concentrations measured in 2012 in the experimental Si amendment plots (a), and in 2013 (b) and 2014 (c) elsewhere in Cell 1B.

Fluxes of DSi, BSi, TSS and Chl *a* (among other constituents) were estimated during a larger tidal flux study from 2012-2014 in Cell 1B (Chapter 2) to gain a better understanding of the net exchange of dissolved and particulate Si between the restored marshes and the estuary (Stevenson et al. 2015). Biogenic and dissolved Si concentrations were measured seasonally in the tidal inlet during 2014. Water velocity measurements were made with Nortek Aquadopp acoustic Doppler profilers (ADP), mounted on weighted PVC plates and placed in the center of one culvert, equidistant from the open ends and sidewalls.

Concurrent with the velocity measurements, an ISCO automatic water sampler was deployed for a 24 hour period (two full tidal cycles) in the same culvert at the mouth of Cell 1B. The ISCO collected discreet water samples at hourly intervals from 30-40 cm off the bottom of the culvert. The samples were chilled with ice in the ISCO sampler, and were returned to the lab at the conclusion of the sample period and kept refrigerated during processing. Samples were subsampled for DSi within 24 hours, usually immediately upon return from the field (i.e. within 2 hours). For the February 2014 flux study, 30 ml subsamples were filtered through 0.45 μm polycarbonate filters, which were analyzed for BSi as for sediments (described above). Full descriptions of velocity measurements and flux calculations are given in Chapter 2.

Additional water samples from three long term water quality monitoring sites (the inlets of Cell 4D and Cell 1C, WQR1 – Poplar Narrows, approximately 2 km east of Poplar Island) were analyzed for DSi throughout 2014. Concentrations at these long term monitoring sites were compared to concentrations measured in Cell 1B during the tidal flux study periods to examine how representative the Cell 1B concentrations were of other dredged material marshes at Poplar Island, and how they compared to the reference marsh (Cell 4D) and the adjacent estuary.

An average value for the tidal creek pool was estimated by using the mean annual ebb concentrations of DSi and BSi in the tidal exchange study, combined with the tidal creek area (1.79 ha) and an assumed average depth of 1 m. It is assumed in this study that the Si pool in edaphic algae is small compared to other pools and recycles rapidly (Teal 1962). It was not quantified here, although it is recognized that benthic diatoms are likely important in Si cycling within the marsh seasonally (Ragueneau et al. 2006). Remineralization was determined in a litter bag study in three other dredged material marshes at Poplar Island (Cells 3D, 1A and 1C) and a low nutrient marsh on the island (Cell 4D). A complete description of methods can be found in Chapter 4.

Phytotron Experiment

In 2012 a soil enrichment experiment was conducted at Horn Point Laboratory to evaluate the potential for a soil Si amendment to enrich the tissue Si concentrations of *S. alterniflora*. Dredged material was collected in September 2010 from Cell 1B, which had undergone crust management and was at the time being graded, but had not yet been opened to tidal exchange. The material was relatively dry and crumbly, having been worked by heavy equipment. It was transported to Horn Point Laboratory where it was sieved (1 cm mesh) to remove plant debris and large chunks, and stored dry (average moisture = 10.9 %) in large covered plastic containers.

Dredged material (2100 g) or sand (2865 g) was placed in 30 cm sections of 10 cm ID PVC pipe (pots) with one end covered with 1 mm mesh screen. Sand was used to simulate the substrate in Cell 4D at Poplar Island, which acts as a control for many aspects of monitoring. An attempt was made to flush residual acidity from the oxidation of pyrite in the dredged material by submerging the pots in ambient filtered (2 μm) Choptank River water (salinity 6.8, SiO_2 5 μmol L^{-1}) for approximately 24 hours, draining them, and repeating twice. A calcium meta-silicate soil amendment (Cal-Sil Corporation, Tennessee) was then mixed into the top 10 cm of the substrate at rates of 0 (control), 1, 2, 4, and 8 metric tons ha^{-1}, with no amendment added to the pots containing sand. Porewater DSi concentrations were monitored using 10 cm porewater samplers (Rhizosphere Research Products, model 19.21.21) inserted into the top of the soil in 5 of the 10 replicates of each treatment. One nursery grown *S. alterniflora* plug grown from wild collected seed, obtained from Environmental Concern (St. Michaels, Maryland), was placed in each pot after rinsing the nursery substrate from the root mass. Each pot was placed in a 19 liter bucket, filled to 15 cm depth with the same ambient Choptank River water described above. The buckets containing the plants were placed in a walk-in controlled environment chamber (Environmental Growth Chambers, Chagrin Falls, Ohio) with a 15 hour light, 26 °C, 60 % humidity/9 hour dark, 22 °C, 80% humidity cycle for 93 days.

Temperature and light were recorded continuously using a HOBO Water Temp Pro V2 datalogger (Onset) and a Licor UW sensor and LI 1000 datalogger, respectively. Soil pH was monitored approximately every 2 - 3 weeks with a Corning model 315 pH meter equipped with a Sensoret model S175CD spear tip electrode. At the conclusion of the experiment, measurements of plant height were made and flowering was noted. Aboveground portions of plants in each pot were removed, rinsed in deionized water and dried to constant weight in a forced draft oven (Grieve model 343). Roots and rhizomes were rinsed free of sediment over a 1mm mesh sieve and similarly dried, weighed and ground in a Wiley Mill equipped with a 1 mm mesh screen. Shoot and root samples were analyzed for BSi using the modified wet alkaline digestion technique described above.

Field experiment

In 2012 three experimental plots were established using a split plot design in the low marsh of Cell 1B to test the effects of a soil Si amendment on the growth and survival of *S. alterniflora*. A granular form of calcium meta-silicate (Calcium Silicate Corporation, Columbia, Tennessee) was broadcast on the sediment surface of an amended subplot at a rate of 6 kg ha^{-1} prior to planting and roto-tilled into the top 5 cm using a small rototiller attachment on a Stihl Model 115R Kombi system. Each plot included an un-amended tillage control subplot that was roto-tilled but received no amendment, and a control subplot which received no amendment or roto-tilling. Plots were 15 x 5 m (16.4' x 16.4'), divided into three 5x5 m adjacent subplots. Plant tissue Si was monitored throughout the growing season by collecting whole shoots monthly. Samples were washed clean in deionized water and dried at 60 °C to constant weight in a forced draft oven (Grieve Model 343). Silicon content was determined using the wet alkaline extraction technique described above.

Porewater DSi concentration profiles were determined using 46 cm sediment equilibrators (Hesslein 1976), which sampled at depths of up to 40 cm below the sediment surface. On July 24, 2012, pore water analyses were carried out in the three Si amendment and control subplots in Cell 1B. Because the marsh sediments in the restored marshes are very compacted, a metal pilot was pounded into the substrate and removed prior to insertion of the polyvinyl chloride (PVC) pore water sampler. DSi concentrations were determined as silicate using the molybdate blue methodology as described above. Solid phase analysis of the top 10 cm of sediment was carried out at eighteen sites in Cell 1B in fall 2011.

The experimental plots were checked monthly throughout the growing season (April to September) for lodging and muskrat grazing, and sampled once toward the end of the growing season for fungal infection, which can be higher when tissue Si concentrations are low. Fungal infection rates (Elmer et al. 2007) were determined for 150 stem sections per treatment (50 per plot) by Wade Elmer, USDA, Connecticut Agricultural Experiment Station, New Haven, Connecticut.

Statistical analysis of plant tissue Si concentrations in the soil amendment study were carried out in R software. A Box-Cox transform was used, followed by analysis of variance (ANOVA) with treatment and time as variables. Statistical analyses were performed by Slava Lyubchich, University of Maryland Center for Environmental Science (UMCES) Environmental Statistics Collaborative.

Results

Silicon budget

Silicon Pools

The major pools of Si within the Cell 1B marsh are given in Table 3-2. The macrophyte value derived from the 2013 and 2014 biomass data represents the maximum annual Si pool contained in vegetation. The value is likely an underestimate due to the extensive marsh dieback observed during the 2013 growing season. The estimate includes the standing dead vegetation, which has a slightly lower SiO_2 content than live shoot material.

The biogenic Si in the sediment contains the largest pool in the system. The sediment DSi pool is comparatively small, based on 2012 porewater data, with surveys in 2013 and 2014 having similar concentrations (Figure 3-2). The water column value represents the standing stock of combined DSi and BSi in the tidal creeks, using the average of volume weighted ebb concentrations for each quarter in the tidal exchange study.

Table 3-2. Major pools of Si in the Cell 1B tidal marsh, Poplar Island, Maryland including macrophyte vegetation, the sediment root zone (top 15 cm), and tidal creek water column.

Component	**SiO_2 (kg)**
Macrophyte vegetation	3,486
Sediment - DSi	85
Sediment - BSi	460,561
Water column	73

Atmospheric deposition

In a review of global atmospheric Si transport, Tegen and Kohfeld (2006) estimated that the mid-Atlantic coast receives between 0.1 – 1.0 g total Si m^{-2} y^{-1}. Within this range, Anderson and Downing (2006) estimated that in Iowa, an agricultural area similar to the Maryland Eastern Shore where this study was conducted, atmospheric deposition averages 0.61 g total Si m^{-2} y^{-1}, with highest deposition rates occurring during the (agricultural) planting season. This value was used here, and given that biogenic Si is likely a small proportion of total Si, and the low solubility of mineral Si, this represents a very minimal input of plant available Si on short time scales.

Remineralization and burial

The change in SiO_2 content in decomposing *S. alterniflora* (Figure 3-4) varied widely with the initial SiO_2 content. The two marshes with the highest starting plant tissue concentrations (Cells 1A and 1C) had the most rapid Si losses during the first three to six months, but there was not an apparent trend in the subsequent year (Figure 3-3). The plant material from Cells 3D and 4D had much lower starting tissue concentrations and showed no discernable trend in tissue SiO_2 over the duration of the experiment. The range of BSi change over the first six months is 0.015 – 0.34 % per day for the dredged material marshes, and a gain of 0.015% per day for Cell 4D, the sand marsh. With such a wide range in the rates of Si loss, it is difficult to determine an appropriate value remineralization for Cell 1B, but for calculating a burial rate, a concentration of 0.5% SiO_2 was used.

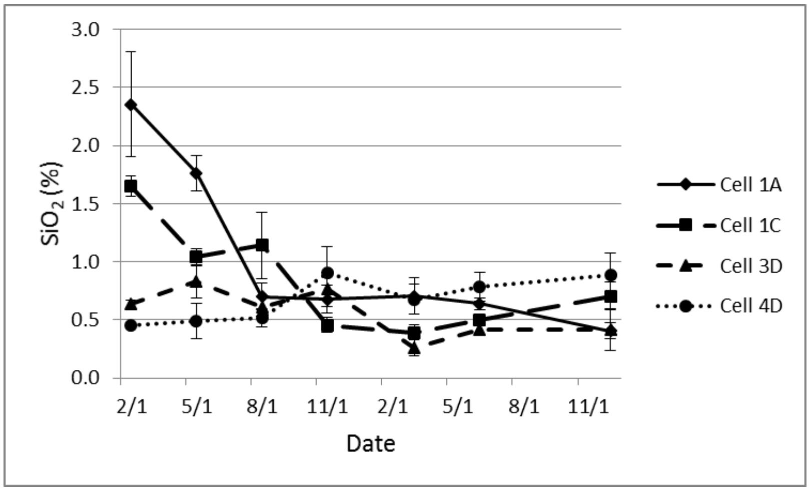

Figure 3-3. Mean (n = 3) biogenic silica composition of decomposing *S. alterniflora* vegetation in four marshes at Poplar Island. Cells 3D, 1A and 1C are dredged material marshes; Cell 4D has a sand substrate.

Tidal exchange

Average volume weighted ebb concentrations of DSi, BSi, total suspended solids (TSS) and chlorophyll a (Chl *a*) exceeded flood concentrations during all sampling periods (Table 3-3). There was a strong relationship between BSi and TSS ($r^2 = 0.78$), and between BSi and Chl a ($r^2 = 0.68$) Figure 3-4). The relationship between DSi and Chl *a* was much weaker ($r^2 = 0.19$, Figure 3-5).

Table 3-3. Flood and ebb tide concentrations of dissolved (DSi), particulate (BSi) silica, total suspended solids (TSS) and chlorophyll a (Chl *a*) measured during the tidal exchange study in Cell 1B during 2014.

Date	**Tide Stage**	**TSS (mg L^{-1})**	**Chl a (µg L^{-1})**	**Silicate (µmol Si L^{-1})**	**Bsi (µmol Si L^{-1})**
Feb-14	Flood	5.09	6.80	18.57	7.26
	Ebb	18.15	7.39	22.64	17.09
May-14	Flood	11.50	13.23	16.96	19.40
	Ebb	20.37	12.82	20.44	26.01
Jul-14	Flood	30.48	10.36	48.87	28.79
	Ebb	30.80	15.12	92.82	52.76
Nov-14	Flood	7.77	3.11	29.78	8.11
	Ebb	9.39	3.99	31.85	9.50

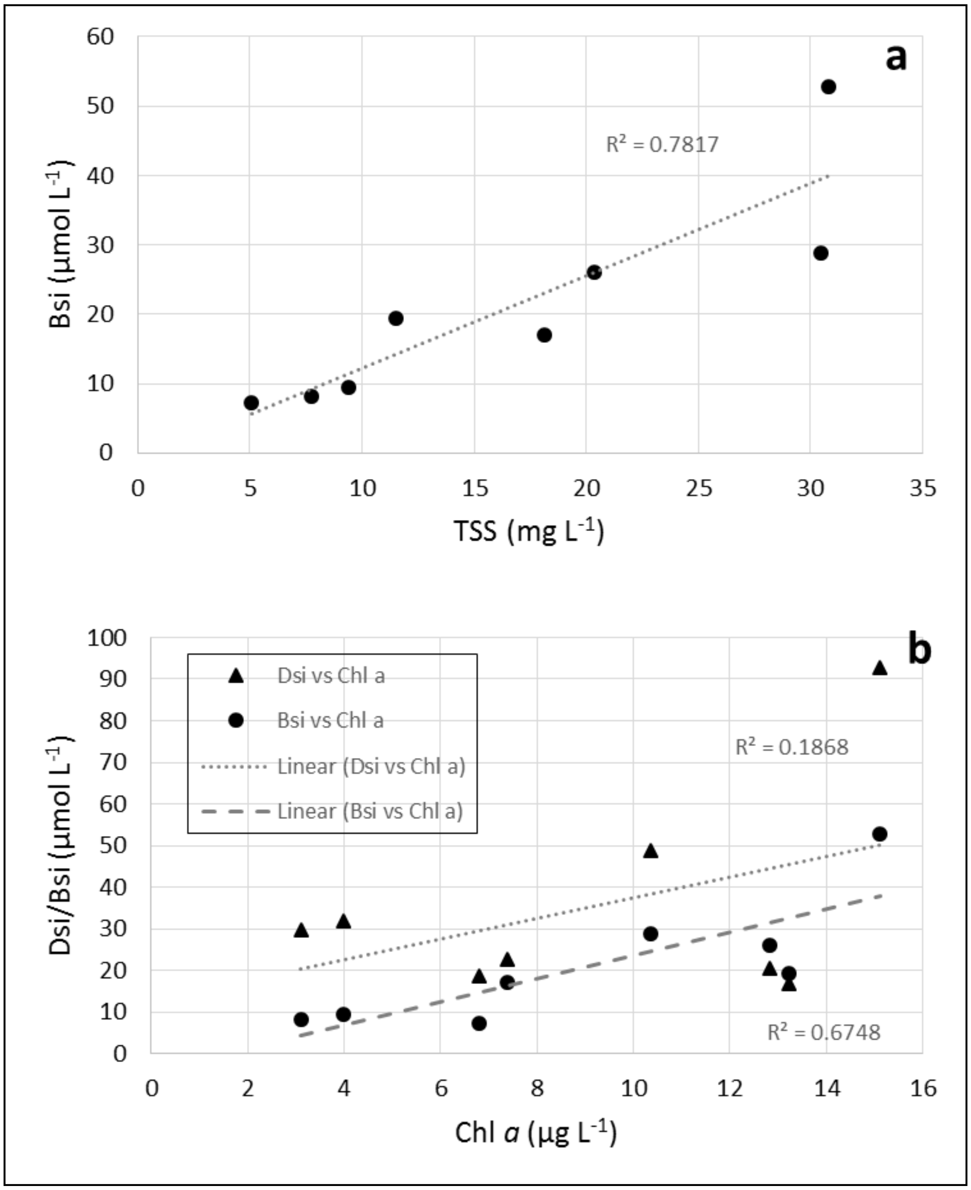

Figure 3-4. Linear regressions of BSi with TSS (a) and both DSi and BSi with Chl *a* (b).

Net fluxes were exports for both DSi and BSi during all monitoring periods in 2014, with the largest net export of both constituents in July (Table 3-4). The net exchange on a quarterly basis was export of both DSi and BSi, with the largest next export in the third quarter, and a combined (DSi + BSi) net annual export of 3924 kg y^{-1} (Table 3-5).

Biogenic Si dominated exports during the first two quarters of the year, while DSi dominated exports during the latter half of the year.

Table 3-4. Mean (n = 2) flux of dissolved (DSi) and biogenic (BSi) silica during flood and ebb tides, and mean net flux for entire tidal cycle. Negative values represent export.

Deployment	Tide	DSi (μmol Si L^{-1})	Flux/tide (kg)	BSi (μmol Si L^{-1})	Flux/tide (kg)
Feb-14	Flood	18.57	6.71	7.26	2.63
	Ebb	22.64	-8.19	17.09	-6.18
	Δ		**-1.47**		**-3.56**
May-14	Flood	16.96	8.97	19.40	10.27
	Ebb	20.44	-10.82	26.01	-13.77
	Δ		**-1.85**		**-3.50**
Jul-14	Flood	48.87	28.99	28.79	17.08
	Ebb	92.82	-55.06	52.76	-31.29
	Δ		**-26.07**		**-14.21**
Nov-14	Flood	29.78	19.43	8.11	5.29
	Ebb	31.85	-20.78	9.50	-6.20
	Δ		**-1.35**		**-0.91**

Table 3-5. Estimates of net quarterly and annual DSi and BSi exports in 2014. Negative values represent export.

Quarter	DSi ($kg\ SiO_2$)	BSi ($kg\ SiO_2$)
1st	-256	-619
2nd	-325	-616
3rd	-3332	-2528
4th	-424	-162
Annual ($kg\ y^{-1}$)	**-4337**	**-3924**
($kg\ ha^{-1}\ y^{-1}$)	**-387**	**-350**

Dissolved SiO_2 concentrations in samples collected by MES at the inlets of Cells 4D and 1C, and in Poplar Narrows (Figure 3-5) varied by a factor of four throughout 2014. Concentrations were consistently highest at the inlet to Cell 1C, the dredged material marsh, where they were at least double the other stations on all but one sampling date. Concentrations at the inlet to Cell 4D, the sand marsh, were close to ambient Poplar Harbor concentrations and showed a similar pattern. Minimum concentrations were in April at all stations, when diatom uptake is high in Chesapeake Bay (Conley and Malone 1992). They tended to increase through the warmer summer months, and declined after September at all stations, consistent with patterns reported for Chesapeake Bay (Conley and Malone 1992).

The major pools and exchanges of Si in the Cell 1B marsh are summarized in Figure 3-6.

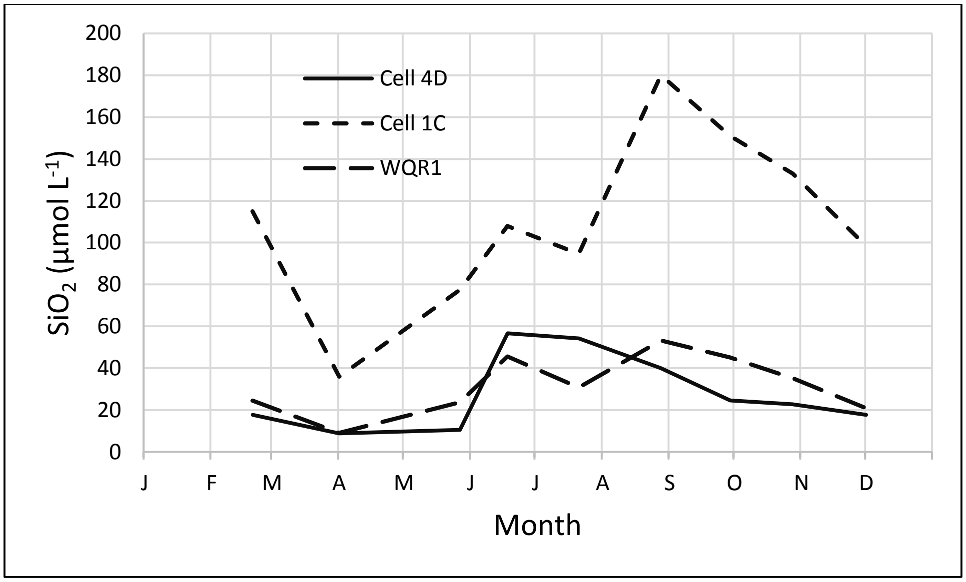

Figure 3-5. Ambient 2014 DSi concentrations at the inlets to Cell 4D (sand), Cell 1C (dredged material) and site WQR1 (Poplar Narrows).

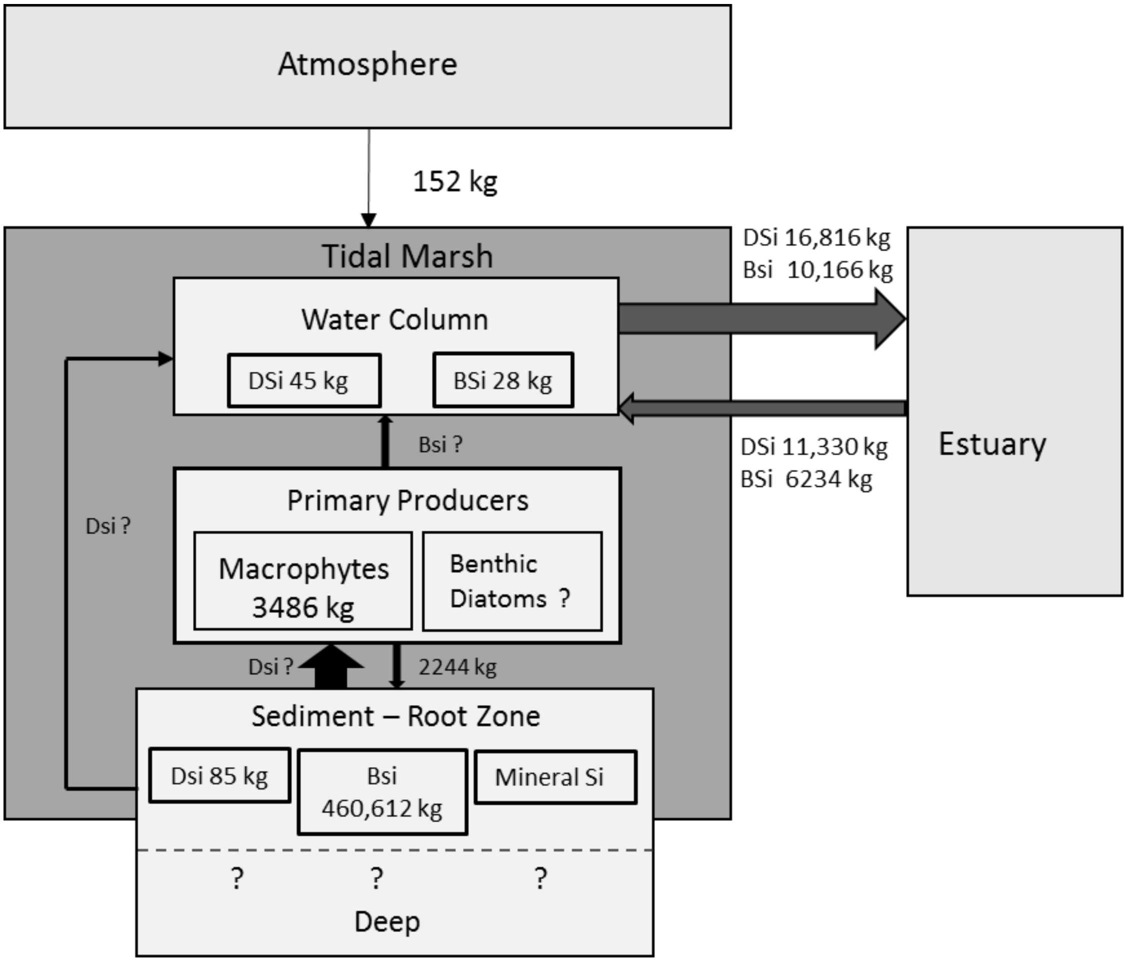

Figure 3-6. Conceptual model of annual Si budget for the Cell 1B tidal marsh at Poplar Island. Values presented above represent kg SiO_2 y^{-1}.

Silicon amendment experiments

Phytotron experiment. In the 2012 soil Si enrichment experiment conducted in the controlled environmental chamber at Horn Point Laboratory, porewater DSi concentrations were similar in at all levels of Si amendment tested (Figure 3-7), but lower on the beginning and ending sampling dates in the sand treatment (no amendment). Soil pH reflected the application rate of the Si amendment, and the differences persisted throughout the duration of the experiment (Figure 3-8a).

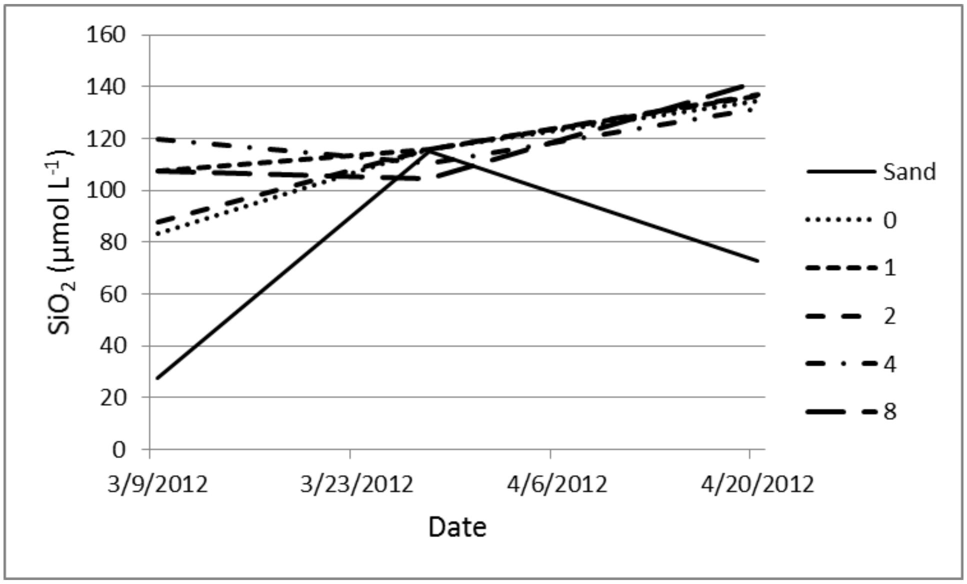

Figure 3-7. Average porewater concentrations in each treatment in the Si amendment experiment. Silicon treatments on dredged material were applied at rates equivalent to 0, 1, 2, 4, and 8 tons Si ha-1. Plants grown on a sand substrate received no Si amendment.

Shoot biomass at the termination of the experiment was similar across all dredged material treatments, but the sand treatment contained much less biomass (Figure 3-8b). Root biomass at termination across the dredged material treatments was variable, but again the sand treatment contained much less biomass (Figure 3-8c). Plant tissue SiO_2 concentrations at the termination of the experiment were similar across all treatments (Figure 3-9).

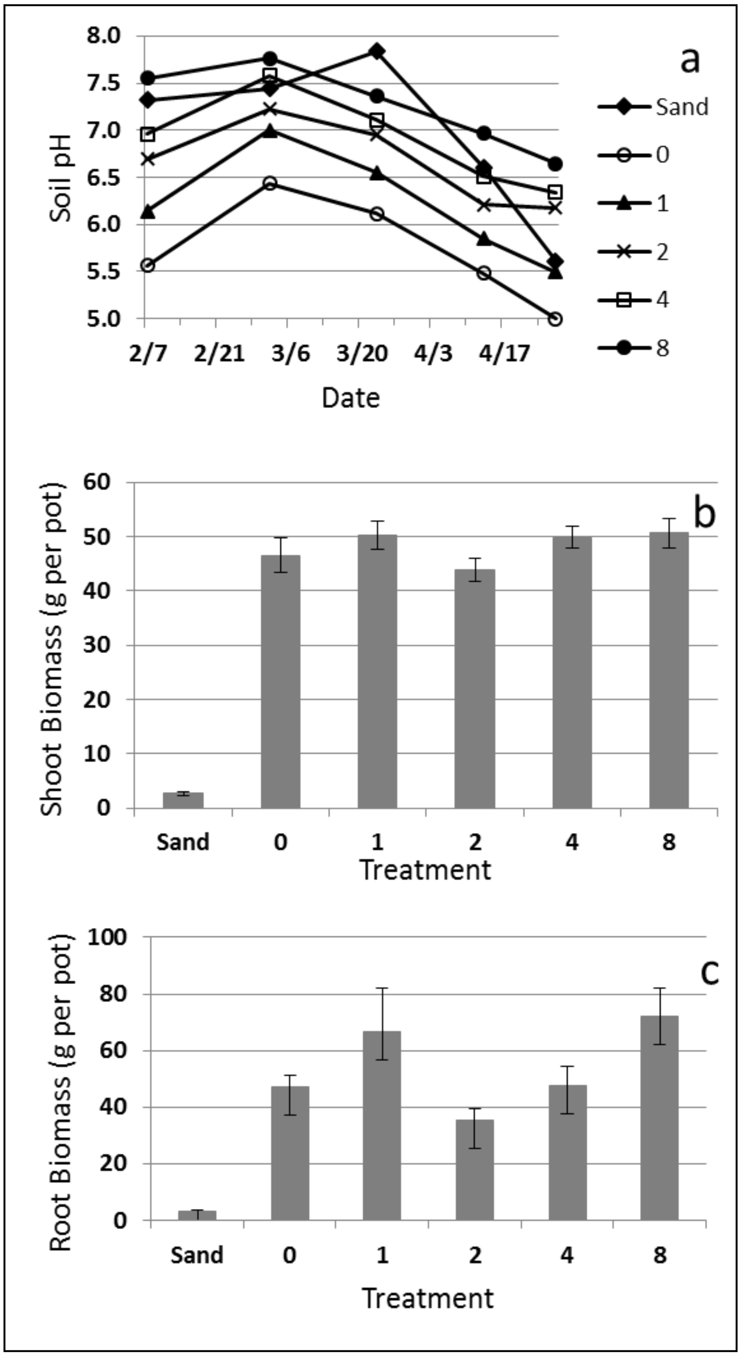

Figure 3-8. Soil pH throughout duration (a), and shoot biomass (b) and root biomass (c) at termination of 2012 phytotron Si enrichment experiment after approximately 12 weeks. Treatment levels are application rates in metric tons per hectare. Error bars are standard error.

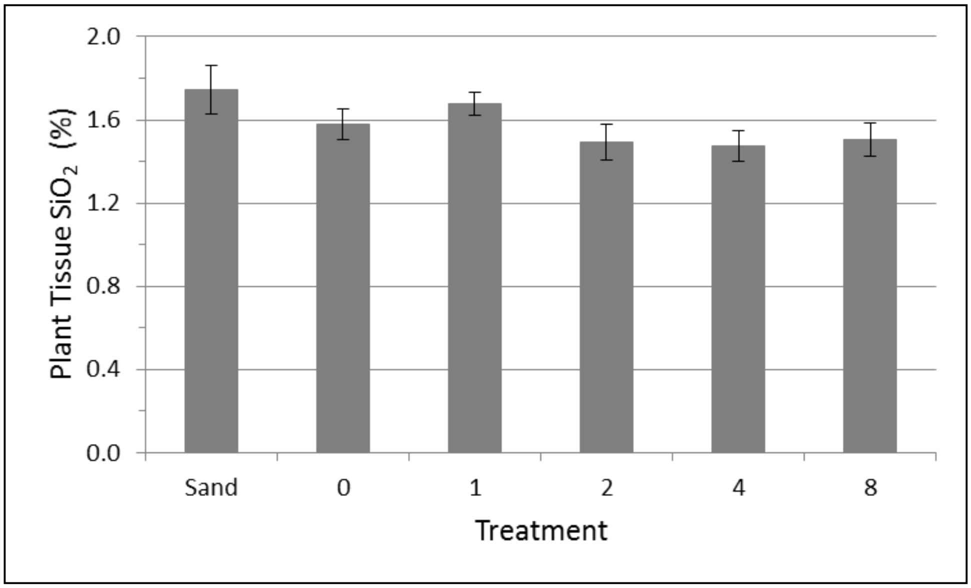

Figure 3-9. Shoot tissue SiO2 concentrations at the termination of the phytotron Si enrichment experiment. Treatments are Si application rates in metric tons per hectare. Error bars are standard error.

Field experiment. Background soil BSi concentrations in Cell 1B at the start of the Si amendment field trial ranged from 2.34 – 3.45 % SiO_2. In November 2012, the first year of the silicon amendment field trial, mean soil pore water silicate concentrations ranged from 169-788 μmol L^{-1} and 302-1035 μmol L^{-1}, in the un-amended control and Si amended plots, respectively (Figure 3-3a). Porewater measurements were not made in the tillage control plots in either year.

Plant tissue concentrations in the experimental Si amendment field plots at Poplar Island were similar at the beginning of the 2012 growing season, but diverged as the season progressed (Figure 3-10a). In 2013 the Si amended plots again produced plants with tissue SiO_2 concentrations that were elevated relative to control plots on the September and November sampling dates (Figure 3-10b). End of season sampling in 2014 of the Control and Si amended plots showed that tissue SiO2 concentrations in the amended plots were still slightly elevated (2.41 ±0.14) compared to the control plots (2.21 ±0.12).

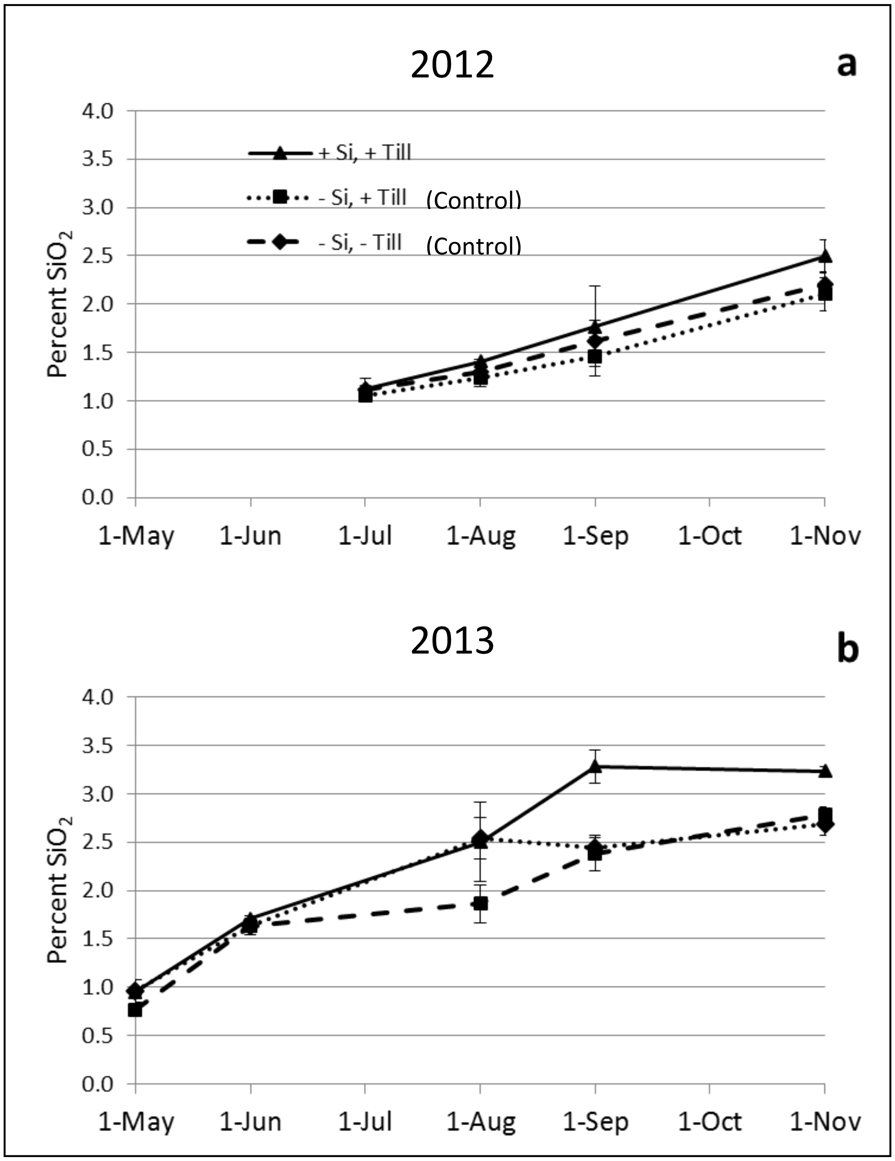

Figure 3-10. Mean (± SE) *S. alterniflora* tissue SiO2 concentrations in the Si amended and control plots through 2012 (a) and 2013 (b), the first and second growing seasons following planting and soil Si amendment application.

Fungal analysis of *S. alterniflora* plant tissue from the experimental plots in 2012 revealed a higher incidence of *Fusarium* infection in control plots than in Si amended plots (Table 3-6), suggesting that higher tissue Si concentrations may confer resistance to fungal infection in *S. alterniflora*. In addition, there were four different *Fusarium* species found in the samples from control plots (*F. proliferatum, F. solani, F. incarnatum* and *F. oxysporum*), while only two were found in samples from the Si amended plots (*F. oxysporum* and *F. incarnatum*).

Table 3-6. Incidence of fungal infection in plants from the experimental Si amended and control plots in September, 2012.

	Treatment	
Plot	+ Si	Control
1	0	6
2	5	5
3	0	2

In 2013 mean percent incidence of fungal infection in stem sections tested (Table 3-7) was higher in Si amended plots (66.7 ±13.5) than in control plots (84.0 ±8.3). Of note however, was that different *Fusarium* species were dominant in the treated and control plots. *Fusarium palustre*, a species unique to *S. alterniflora* (Elmer and Marra 2009), was dominant in 2 of the 3 control plots but was not found in any amended plots, while *F. proliferatum* dominated amended plots. Both species have been reported from natural *S. alterniflora* stands on the east coast of the U. S. and are considered pathogenic only when plants are stressed by other factors, e.g. drought (Elmer and Marra 2011).

Table 3-7. Incidence of fungal infection and frequency of fungal species in plants form the experimental Si amended and control plots in September, 2013.

Treatment	Plot	Incidence	Incidence	Species				
		(%)	x̄ ± sd	*F. palustre*	*F. incarnatum*	*F. proliferatum*	*Sporotrichiodes*-like	*Chlamydosporum spp.*
Control	1	46	66.7 ±13.5	15	0	0	6	4
	2	62		21	0	0	2	2
	3	92		6	1	11	3	4
Silicon	1	72	84.0 ±8.3	0	0	16	5	4
	2	100		0	0	25	0	0
	3	80		0	0	23	1	1

Grazing by muskrats was not observed in either 2012 - 2014 in any of the experimental plots. Lodging was also not observed in 2012 - 2014 in any of the plots. An infestation of stem boring insect larvae in 2013 appeared to kill the meristem of a high percentage of plants, limiting plant height and reducing the propensity for lodging. Stem borers persisted through 2014 based on a summer survey and included several species, including *Languria taedata*, a beetle found on *S. alterniflora* along the Atlantic and Gulf coasts (Gimmel et al. 2012).

Discussion

Silicon budget

The Si budget in the Cell 1B marsh is notable for the large net annual export of combined Si (DSi + BSi) in this study. The export of DSi is consistent with Si flux studies of other marshes (Struyf et al. 2006a; Jacobs et al. 2008; Vieillard et al. 2011). Unlike other marshes, which import BSi with TSS, especially in summer (Struyf et al. 2007), there was a constant export of BSi from Cell 1B at Poplar Island. It is apparent that for Si, as for N, the dredged material in the Cell 1B marsh provides a rich source to macrophyte vegetation, as well as benthic and pelagic diatoms, and for net export. Sediment porewater concentrations indicate that while there is some depletion in the root zone (≤ 15 cm depth), deep sediment concentrations are among the highest reported (Muller et al. 2013). The range in Cell 1B at Poplar Island was 200 – 1000 µmol L^{-1} in 2012, higher that values reported for a natural marsh in Rhode Island, near 0 to 300 µmol L^{-1}, (Vieillard et al. 2011; Carey et al. 2013), and a Delaware marsh ~100 to 900 μmol L^{-1} (Scudlark and Church 1989). As the dredged material ages, Si concentrations will likely decline, but there was little change during this three year study (Figure 3-3) suggesting the large volume of dredged material beneath the marsh will provide a long term source.

The peak export of DSi and BSi in July coincides with the highest soil temperatures, when sediment flux rates are highest (Scudlark and Church 1989) and the decomposition rate reaches a maximum (Chapter 4). The loss of BSi from *S. alterniflora* detritus appears to be greatest during the spring leading up to peak DSi and BSi export (Figure 3-4), and may be recycled and assimilated by benthic diatoms, only to be released in summer as diatoms are replaced by other algal species.

In Danish marshes the summer export of DSi is considered a valuable source of Si to coastal waters when ambient concentrations are at a minimum (Jacobs et al.2008; Struyf et al. 2006), but at Poplar Island monitoring suggests that this peak does not coincide with the ambient minimum concentration (Figure 3-6). Rather, it coincides with a summer peak in ambient DSi resulting from remineralization of diatom BSi following the spring diatom bloom (Conley and Malone 1992). The positive relationships between BSi and TSS, and BSi and Chl *a* (Figure 3-5) suggest that much of the BSi exported from the marsh is in pelagic diatoms. If this is the case, it indicates that diatoms remain abundant in the marsh creeks at Poplar Island despite the high inorganic N concentrations (Chapter 4) due to the availability of DSi. This would support the hypothesis that DSi depletion in eutrophic estuaries is responsible for the shift in species composition toward non-diatomaceous species, including harmful algal blooms (Conley and Malone 1992). The Si:N ratio in diatoms is ~1 (Sterner and Elser 2002), while the molar Si:N ratio of exports from Cell 1B is approximately 6 (6.6 for dissolved forms of Si and N), indicating that overall the marsh is exporting a surplus of Si relative to N.

The retention of macrophyte detritus within Cell 1B due to the limited tidal exchange likely promotes vertical accretion (Chapter 2), although it is not clear what effect it may have on Si burial rates. Vertical accretion is not currently measured in Cell 1B, but the rate in other dredged material marshes at Poplar Island is approximately 1 cm y^{-1} (Chapter 2). Struyf et al. (2007) report that young Dutch marshes have higher deposition and burial rates than mature marshes, with rates of both declining after the marshes reach an elevation in equilibrium with sea level. The resulting recycling efficiency ranges from 15% in young marshes (85% buried) to 60% in mature marshes (40% buried). The young marshes at Poplar Island also have higher rates of vertical accretion in the first few years (Chapter 2), with rates slowing after about 5 years. Upward diffusion of N and DSi contained within the deep sediment, however, likely drives high remineralization rates in the root zone and at the sediment surface, so the recycling rate should be much higher and the burial rate much lower than in the Dutch marshes.

Although the marshes studied by Stuyf et al. (2007) were formed on former agricultural land, they do not consider the effects of soil N on Si recycling or burial. This was examined in a study of two Rhode Island marshes exposed to different levels of N (Carey and Fulweiler 2013). They found N enrichment was associated with higher sediment BSi and DSi concentrations, as well as higher plant tissue concentrations. Increased Si availability due to high TSS deposition and high rates of recycling induced by N enrichment were thought by Carey and Fulweiler (2013) to be responsible for the elevated sediment and plant Si concentrations, although there was likely also a difference in Si inputs. Thus the effect of N enrichment on Si recycling and burial rates remains unclear, and should be explored in future studies at Poplar Island.

Silicon amendments

Since Eleuterius' (1981, 1983) speculation that *S. alterniflora* could develop Si deficiency in response to N enrichment, several studies have examined the interactions of marsh macrophytes, stress (biotic and abiotic) and Si. Results have been mixed, with some marsh species showing enhanced tissue concentrations in response to Si enrichment (Schaller et al. 2012; Mateos-Naranjo et al. 2013), while others did not (de Bakker et al. 1999). Stress due to excess N, thought to be an important factor at Poplar Island (Chapter 1), was not a variable in any of these studies.

The N availability in the dredged material substrate at Poplar Island, combined with the numerous indications of stress resulting from excess N in the *S. alterniflora*, make this an ideal site to test Eleuterius (1983) hypothesis that a growth induced Si deficiency decreases fitness and increases vulnerability to a variety of stresses, and the hypothesis that Si amendments can help improve resistance to disease, lodging, grazing and perhaps dieback.

The results of our small scale experiment growing *S. alterniflora* in dredged material enriched with various levels of calcium meta-silicate were inconclusive. While Si amendments had a pronounced effect on soil pH, there was no difference in plant tissue Si concentrations among the dredged material treatments. Plants grown in sand contained similar tissue Si concentrations (Figure 3-8). The experiment was only 12 weeks long, and may not have reflected the results that would be obtained in a full growing season.

In the field experiment tissue concentrations early in the growing season were similar in treated and control plots, as they had been in the lab experiment, but during the latter half of the growing season the amendment increased tissue concentrations. The result was an increase in the separation between control and amended tissue concentrations as the season progressed, indicating that the soil in the root zone becomes depleted as the plants grow and uptake exceeds the rate of resupply. This highly significant ($P = <0.0001$) effect persisted through year two following soil amendment. Thus, tissue Si concentrations in *S. alterniflora* responded to Si availability, in this N rich environment even though Si is abundant in the soil.

Also noteworthy in the field experiment, tissue Si concentrations of plants from Cell 4D, which has a low nutrient substrate, were substantially lower than plants from the dredged material plots, while plants from the Horn Point marsh, with higher soil porewater NH_4^+ concentrations (unpublished) were closer to the dredged material marsh controls. These results concur with the results of Carey and Fulweiler (2013) that plants exposed to higher N availability contain higher tissue Si concentrations. It is not clear whether this is due to differences in N availability and plant growth, or differences in Si availability or uptake. Nitrogen enrichment delays senescence, and *S. alterniflora* stays greener late into the season than natural marshes nearby. Passive Si uptake persists as long as the plants are transpiring, so a prolonged growing period for N enriched plants could result in higher tissue Si concentrations at the end of the growing season. Unfortunately, although plant tissue concentrations in the experimental plots and reference marshes were monitored in this study, leaf chlorophyll *a* concentrations were not and soil Si concentrations were only monitored in the plots. Future work in this area could readily resolve this question.

The question of whether increased tissue concentrations confer resistance to stress in *S. alterniflora* has not been resolved by these studies. Lodging and *Ondatra zibethicus* grazing were not observed in or near the experimental plots during the course of the field experiment. The fungal analysis indicated that there is likely a protective mechanism conveyed by Si enrichment, at least with respect to the more pathogenic strains found in *S. alterniflora*, but further experiments would be required to support this contention. A survey of stem strength in the treated and control plots (Harris et al., unpublished) did not find higher stem strength in treated plots, but the results may have been compromised by an invasion of stem boring insects. However, there was no obvious visual evidence that plants in the treated plots were more resistant to stem borers than those in control plots.

Although there are some intriguing questions raised by this study, it does not suggest that *S. alterniflora*, like rice (*Oriza sativa*), sugarcane (*Saccharum spp.*) and other crop species, shows marked improvement in fitness when supplemented with Si, even under extreme N enrichment. Although they are in the same family, *Poacea*, a family known to have higher tissue concentrations that most other taxa, *S. alterniflora* has recently been designated an intermediate Si user (Querne et al. 2012), whereas *O. sativa* is toward the furthest end of the spectrum in terms of Si assimilation, showing both passive and active uptake. The reasons for this difference are unclear, but may be related to their evolution in brackish to saltwater environments versus freshwater environments, respectively. With the terrestrial sources of Si closer to freshwater marshes, *O. sativa* may have evolved under Si replete conditions, whereas *S. alterniflora* evolved in a coastal environment where the Si availability is lower and seasonally depleted by diatoms.

Conclusions

The tidal exchange portion of this study has demonstrated that fine grained dredged material marshes can provide a source of both DSi and BSi to local estuarine waters in stoichiometric excess of exported N. There is likely rapid recycling of Si within the marsh, and the tidal creeks appear to be important in the transformation and export of Si. *Spartina alterniflora* shows slightly enhanced tissue concentrations in response to a soil amendment, but improved plant fitness was not detected in this study.

Overall, it appears from these studies that tidal marshes are important in Si cycling, and fined grained dredged material marshes can provide a source of Si to the local estuarine environment, an overlooked ecosystem service. The concerns of Lanning and Eleuterius (1981, 1983) relative to stress induced by excess N in this one species, *S. alterniflora*, is likely not related to Si deficiency.

Chapter 4: Carbon and nitrogen balances in a tidal marsh created with fine grained, nutrient rich dredged material

Introduction

Background

Wetland creation and restoration is frequently undertaken to replace the highly valued ecosystem services associated with natural wetlands (de Groot et al. 2012; Costanza et al. 2014) that have been lost or damaged through natural or anthropogenic processes (Chmura et al. 2012). The restoration of these ecosystem services depends upon the ability of restored wetlands to function as natural wetlands, sequestering carbon (C) as organic matter (Zedler and Kercher 2005), taking up inorganic nitrogen (N) and phosphorus (P) and exporting organic matter that is the basis of coastal food webs (Odum 2000). Increasingly, fine grained sediment dredged during shipping channel maintenance is finding a beneficial use as a substrate in constructed tidal marshes. A key factor determining the trajectories of created marshes toward functional equivalency appears to be the substrate (Chapter 1).

Fine grained dredged material in eutrophic estuaries is particularly rich in inorganic N, primarily as exchangeable ammonium (NH_4^+), due to the low redox in the sediment (Cornwell and Owens 2012). Since the substrate in constructed marshes is often at least several meters deep, this represents a large pool of available N. The prevailing paradigm is that young marshes typically import N on an annual basis because the sediment N pool is small, while in older marshes where organic N has accumulated, uptake and remineralization allow higher rates of export (Childers et al. 2000). The effect of an initially large sediment pool of N in a young constructed marsh on cycling and on the net flux of N between the marsh and adjacent estuary has not been well studied, but may force these young marshes to function more like mature marshes in terms of N retention. Large N losses from the sediment pool to the adjacent estuary would represent mobilization of N that had formerly been sequestered in the bay bottom, an undesirable outcome in a eutrophic system. Alternatively, the supply of NH_4^+ may indirectly fuel high rates of denitrification in near-surface sediments, preventing major N export to the estuary.

With respect to C cycling, N fertilization results in high macrophyte production (Valiela et al. 1975; Callaway et al. 1995; Darby and Turner 2008). The availability of N is also positively correlated with macrophyte decomposition rates in tidal marshes (Morris and Bradley 1999), and the balance of these two processes determines the amount of organic matter available for export and burial. Although C sequestration in natural marshes is described as a potential offset to anthropogenic atmospheric CO_2 enrichment, and an anticipated benefit of tidal marsh creation (Chmura et al. 2012), the net effect of the N rich dredged material on C cycling and fluxes in constructed tidal marshes is not well understood. In addition, the design of constructed marshes may influence the fate of C in these systems, potentially favoring burial of the abundant organic matter over export where containment dikes constructed to keep the dredged material in place limit tidal exchange to a narrow inlet.

To examine the influence of fine grained dredged material on N and C fluxes between a constructed tidal marsh and adjacent estuary, mass balances for N and C were developed for a tidal marsh at the Paul S. Sarbanes Ecosystem Restoration Project at Poplar Island, Maryland (Poplar Island), constructed using fine grained dredged material from the shipping channels in upper Chesapeake Bay (Chapter 1). The rich supply of plant available N in the substrate used at Poplar Island produces tidal marshes that initially have very high rates of primary production, but are prone to *Spartina alterniflora* dieback, leading to highly pulsed production (Chapter 1). The two years covered in this study include one year of extensive low marsh dieback (2013) and one year of much less dieback (2014).

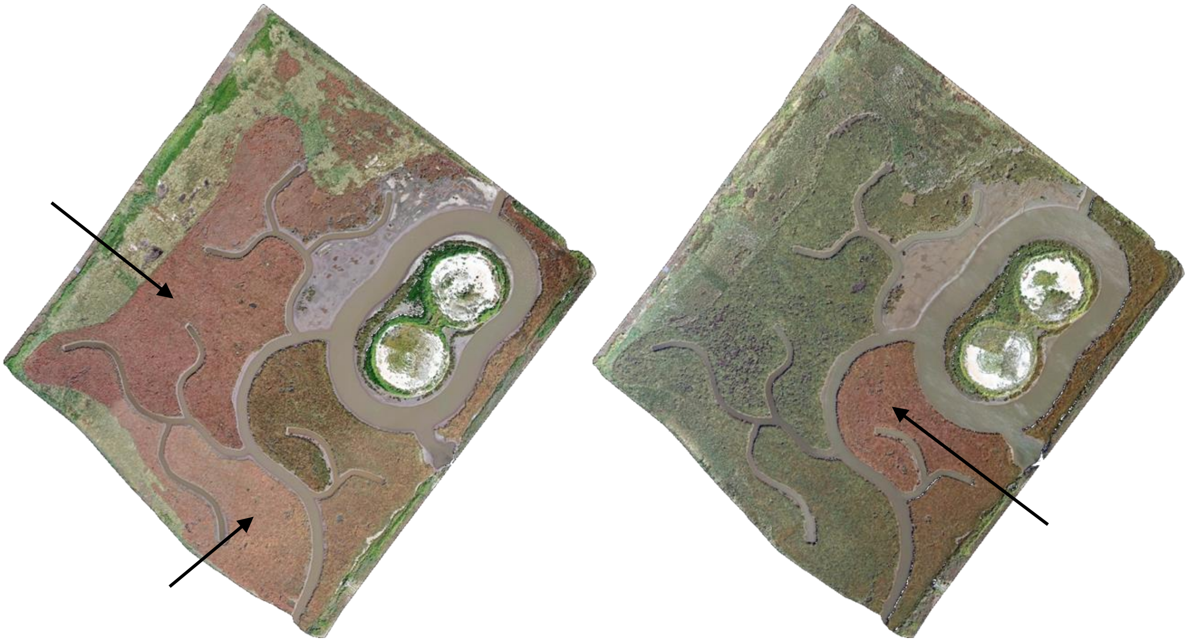

Figure 4-1. Aerial imagery of Cell 1B in September 2013 (left) and September 2014 (right), with arrows indicating areas of dieback, brown areas of prematurely senescent *S. alterniflora*.

Using data from sediment and vegetation monitoring, a tidal flux study, local atmospheric deposition data from the National Atmospheric Deposition Program (NADP) site MD13, approximately 27 km from Poplar Island, and selected literature values the main sources and sinks of N and C in this engineered marsh system were identified, and net annual exchange of N and C with the adjacent estuary estimated. We examine whether high production rates lead to C export, or whether the marsh configuration leads to burial and C sequestration; how dieback may affect the N and C exchange with the estuary; the magnitude of N delivery from the deep sediment to the root zone, and whether this young, N rich, engineered marsh is a net source or sink for N and C on seasonal and annual bases.

Methods

Site description

The Poplar Island project, located in mid-Chesapeake Bay, is a large scale island restoration project utilizing dredged material from the approach channels to Baltimore Harbor in the upper Bay (Chapter 1). The focus of this study is a 12.41 ha tidal marsh, Cell 1B (Figure 3-1), which has been monitored since planting in 2012. Cell 1B experienced a large scale *S. alterniflora* dieback in 2013, and vegetation recovery in 2014. A perimeter dike 3 m in height surrounds the marsh, limiting direct tidal exchange with the estuary to two concrete box culverts (each 1.83 m x 1.83 m) armored with stone. Although this marsh was opened up via channels to adjacent marshes for fish passage, minimal water transport was detected during a several month current meter deployment at each channel (Nidzieko, pers. com. 2012). Groundwater inputs, which can be an important source of N in natural marshes (Valiela et al. 1978), are assumed to be minimal in this system due to the limited upland area adjoining the marsh and because movement through the fine grained substrate is likely mostly diffusive (Stevenson et al. 2007). The marsh consists of a high marsh zone, 2.51 ha, low marsh zone 6.92 ha, tidal creeks 1.79 ha and an island with an approximate elevation of 2 m, intended to provide habitat for colonial ground nesting birds, 1.19 ha, determined by hand digitization of September 2014 aerial imagery in ArcMap 9.1. The majority of the island is above tidal flooding and is largely devoid of vegetation, covered instead with a variety of coarse materials. Neither the sediment nor the vegetation of the island was characterized in this study, and the island area is excluded from calculations of marsh standing stocks. It is included in calculations of net fluxes on an areal basis in the final summary table.

Conceptual Models

Conceptual models were developed for the Cell 1B marsh to help define the N and C budget terms. For the N model (Figure 4-2) the inputs are tidal flood water, atmospheric deposition, and upward transport from the deep sediment, below the root zone. Previous work (Chapter 1) established that ≥75% of marsh vegetation roots are confined to the top 15 cm of the sediment. For this analysis the sediment concentrations from the top 10 cm and ≥20 cm were used to characterize the root zone and deep sediment, respectively. Groundwater inputs are considered negligible. Nitrogen fixation can represent a significant contribution to the N budget of a marsh (e.g. ~9% of Great Sippewissett Marsh, or ~6.8 g N m^{-2} annually, Valiela and Teal 1979), but was not measured in this study. The large pool of NH_4^+ in the dredged material likely suppresses fixation in the Poplar Island marshes, however, making it a small percentage of the overall budget. An N_2 fixation rate measured in the marsh at Horn Point Laboratory (Lipschultz 1978), 11.5 mg NH_3-N m^{-2} d^{-1} was used here. At the time the measurements were made, this Horn Point marsh was receiving large amounts of N in agricultural runoff from surrounding fields (Stevenson et al. 1976). Potential outputs are denitrification, tidal exchange and burial. There is undoubtedly some exchange of N via biota (e.g. fish, turtles, birds). There are presently no quantitative estimates for this system, and because it is young net exchange is considered small. In Great Sippewissett Marsh (Valiela and Teal 1979) inputs and outputs from birds and shellfish were balanced.

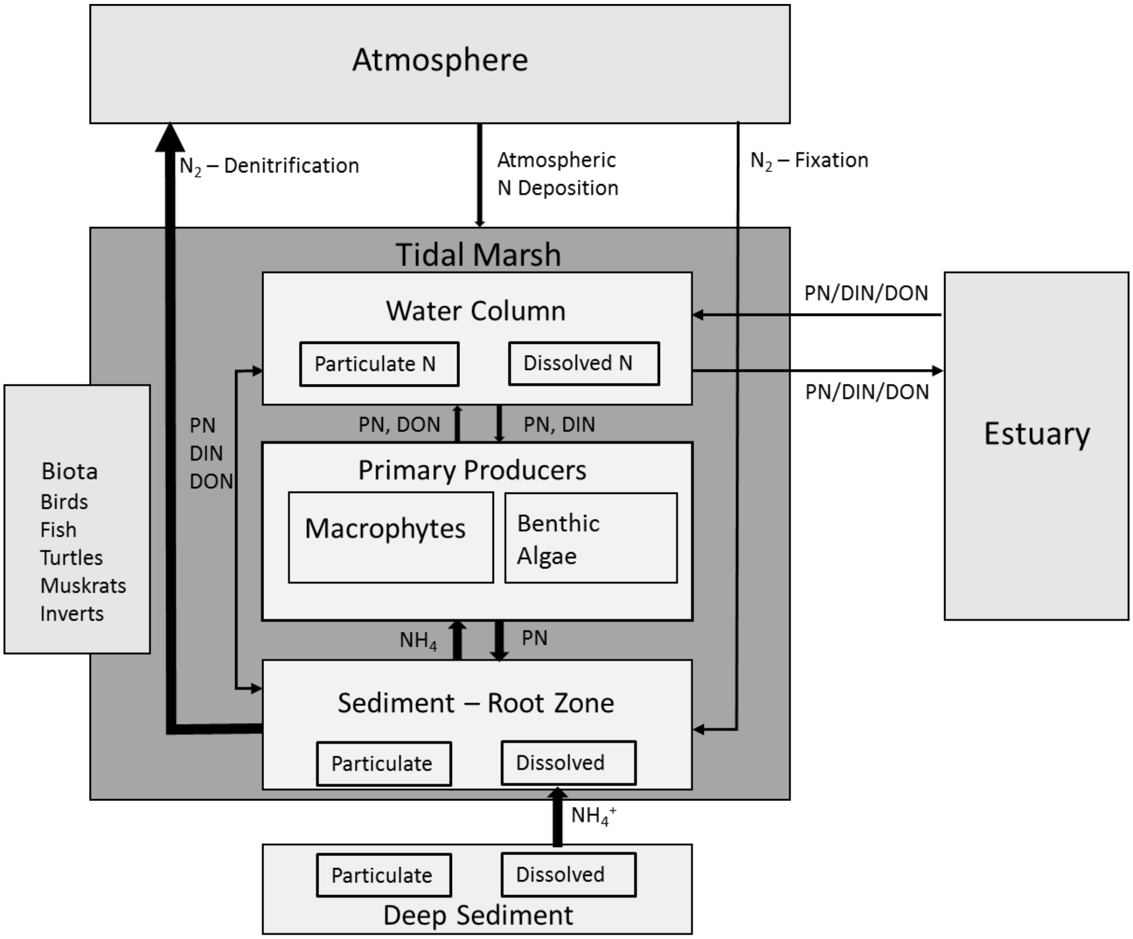

Figure 4-2. Conceptual model of N cycling in the Cell 1B marsh at Poplar Island, 2013-2014.

For the C budget (Figure 4-3) inputs include C fixation by macrophytes and benthic algae and deposition from tidal inputs. Outputs include respiration, burial and tidal exports. Again, exchange via biota was not quantified, but is considered to be small. Carbon stored in the deep sediment is considered constant, but C content in the top 10-20 cm increases over time with organic matter deposition on the surface and belowground biomass production (discussed below) as these marshes age (Stevenson et al. 2013).

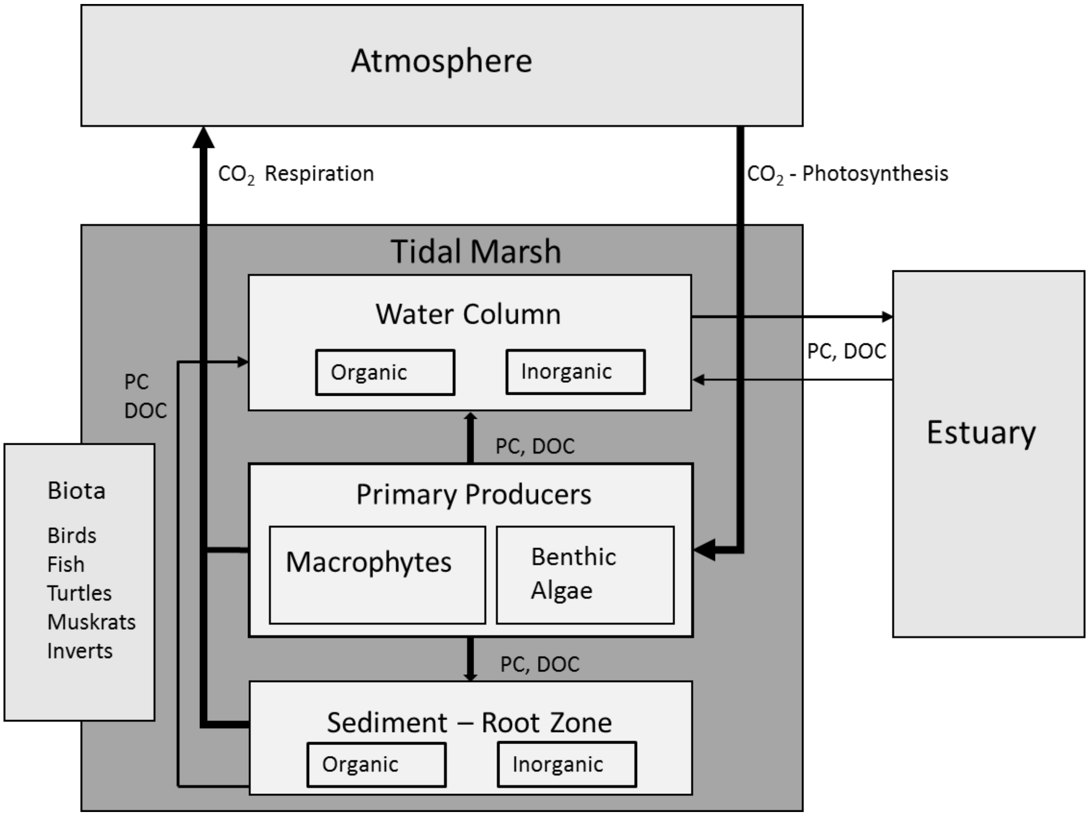

Figure 4-3. Conceptual model of C cycling in the Cell 1B marsh at Poplar Island.

Macrophyte production

Annual macrophyte production was estimated in Cell 1B by harvesting aboveground and belowground biomass samples at the conclusion of the growing season (October) in 2013 and 2014. Six high marsh and six low marsh sites distributed throughout the marsh were sampled each year, and flags were placed at the sampling locations to avoid repeat sampling in the exact locations. Aboveground biomass samples were collected using 0.25 m^2 PVC quadrats. Samples were separated into dead and living biomass prior to being dried at 60°C in a forced draft oven (Grieve Model 343) and weighed to the nearest mg on a top-loading balance (O'haus 1500D). Belowground biomass samples consisted of a single sediment core collected from each of the aboveground biomass quadrats using a specially fabricated 7.5 cm diameter stainless steel piston corer. Sediment cores were washed free of sediment over a 1 mm mesh sieve before being dried and weighed as described above. After weighing, all samples were ground in a Wiley Mill (1 mm mesh size) and analyzed for C and N content by combustion in pure O_2 using an Exeter Analytical, Inc. CE-440 Elemental Analyzer according to standard protocols of the Horn Point Laboratory Analytical Services (Lane et al. 2000). Standing stock computations were made using biomass estimates and tissue C and N contents for belowground biomass (total) and aboveground biomass (live and dead), in the high and low marshes separately.

Benthic algal production

Net carbon fixation by benthic microalgae was estimated from measurements of oxygen production in sediment cores from the low marsh and tidal creeks in May and September, 2012 by J. Cornwell and M. Owens (HPL Biogeochemistry Lab), described at length in

Stevenson et al.(2013). Computation of algal C fixation assumed a 1:1 ratio with O_2 production (Stevenson et al. 2014). Annual algal marsh C fixation was calculated by converting average hourly light and dark rates to day rates using the day/night lengths on the sampling date. The May and September 2012 marsh and creek rates were applied to the areal coverages (9.43 and 1.79 ha, respectively). Averages of the May and September measurements for the marsh platform and tidal creeks were multiplied by the respective areas to calculate C fixation for the Cell. High marsh rates were not measured, so the low marsh rate was applied to the high marsh area, and the habitat island was excluded from the calculation.

Water column standing stocks

The water column standing stocks of N and C were estimated by using the areal coverage of the tidal creeks in the 2014 aerial imagery, an average water depth of 1 m, and average annual total nitrogen (TN), concentrations and the sum of average annual non-purgeable organic carbon (NPOC) and particulate carbon (PC) obtained from the tidal exchange study described below.

Sediment standing stocks

For calculating the N and C standing stocks in sediment, the root zone was defined as the top 15 cm based on root distribution at Poplar Island (Chapter 1), which is consistent with other studies (e.g. Craft et al. 1988). To calculate the N pool in the sediment below the root zone, the volume occupied by the root zone was subtracted from the total volume of dredged material contained in the cell (from USACE), and a soil density of 1.1 g cm^{-3}. The tidal creek bottom area (1.79 ha) was included in the root zone calculation because it is exposed to the water column and therefore open to exchange, but the soil C and N concentrations have not been determined separately. The C and N concentrations used to calculate standing stocks were means (± standard error) of 18 sediment samples (0 – 10 cm depth) collected in 2015, 34.52 (±1.79) mg C g^{-1} and 2.84 (0.13) mg N g^{-1}, respectively (HPL Biogeochemistry Lab). These are within the ranges reported for a variety of salt marshes in a review by Nixon (1980).

Decomposition

In January 2012 a litter bag study was initiated to estimate decomposition rates of *S. alterniflora* shoots in the dredged material marshes and the low nutrient sand marsh at Poplar Island. The Cell 1B marsh was not planted until May 2012 and, thus, was not included in this experiment. Data from this experiment are used here, however, to estimate rates of C sequestration and N mineralization in Cell 1B.

Intact shoots of *S. alterniflora* were collected from three dredged material marshes (Cells 3D, 1A and 1C) and the sand marsh (Cell 4D) at Poplar Island on January 5, 2012. The plant material was rinsed with deionized water and dried at 60°C to constant weight in a forced draft oven (Grieve model 343). Approximately 40 g of dried stems and leaves were placed in 20 x 30 cm nylon mesh bags (mesh size 2 mm; Collins Cottage Industries). Eighteen bags were deployed on February 15, 2012. Three bags were retrieved at approximately three month intervals for eighteen months and the final three were collected at 22 months. Bags were rinsed with deionized water in a shallow basin, and the detrital material was removed from the bag, dried in a forced draft oven at 60°C (GCA Corp. model 28) and weighed to the nearest 0.01 g on a top loading balance

(O'Haus model 1500D). Detrital material was analyzed for C and N content as described above. Burial rates were estimated using the C and N concentrations and % dry mass remaining after 22 months.

Solid Phase Chemistry

Solid phase analysis of the top 10 cm of sediment was carried out (HPL Biogeochemistry Lab) at 18 sites in Cell 1B in autumn of 2011 (Stevenson et al. 2012). Organic matter was determined by loss on ignition (LOI) in Cell 1B, measured by combustion at 400°C. Total N and C concentrations were not measured in the initial solid phase survey in Cell 1B, so average root zone (0 – 10 cm) values of N (2.17 mg g^{-1}) and C (23.5 mg g^{-1}) determined by CHN analysis (as above) from an adjacent marsh, Cell 1A, were used for calculating the root zone N and C pools (Stevenson et al.2011). Cell 1B had a slightly higher average LOI (~ 8%) than Cell 1A (6.9 %) in the respective initial solid phase surveys. The initial Cell 1A root zone C concentration was close to the deep concentration determined for a 1 meter long core obtained from Cell 3D in 2010 (20 – 30 mg g^{-1}), and so was also used for calculating the deep sediment C pool in Cell 1B. From the same core, a total phosphorus (TP) concentration of ~1.25 mg g^{-1} and a molar N:P ratio of ~4 (< 20 cm) reported in Stevenson et al. (2013) were used to determine a TN concentration of ~5 mg g^{-1}, which was used for calculating the deep sediment TN pool in Cell 1B. For calculating sediment N and C pools, the total area of the marsh (12.41 ha) minus the habitat island (1.19 ha) was used, for a total of 11.22 ha.

Sediment-water exchange

Sediment-water N fluxes reported in Stevenson et al. (2014) were used to estimate annual sediment N fluxes for the marsh. In brief, seasonal measurements of NH_4^+, NO_x (nitrate plus nitrite), and N_2 gas were made in 2013 and 2014 in Cell 1B (HPL Biogeochemistry Lab). Cores were collected during each sampling period. For 2013 and 2014, sampling was seasonal, with cores collected in June, September and November 2013 and April and May 2014. Methods are described at length in Stevenson et al. (2014). In brief, sediment cores were collected in acrylic tubes with an inside diameter of 7 cm and a length of 30 cm. Marsh cores were collected by hand insertion into the soil and creek bottom cores were collected using a pole corer.

Cores were transported to Horn Point Laboratory and placed in a water bath for about 4 hours in the dark followed by a two–three hour incubation with gentle stirring under light conditions in most cases. Overlying water was sampled at seven time points, three in the dark, one at the light/dark transition, and three in the light. This provided a four point regression for both light and dark periods. Samples were analyzed for dissolved oxygen, di-nitrogen (N_2), NH_4^+, and nitrate plus nitrite (NO_x).

Annual estimates for the N budget were calculated using the representative seasonal estimates, taking into account average light and dark periods for the quarter. Although the cores were flooded during measurements, the period during which the marsh was flooded was not taken into account. Therefore, estimates for the creeks may be a more accurate reflection of actual rates than estimates for the marsh surface, which is not continuously flooded.

Nitrogen fixation was not measured in this study, so an average value from the *S. patens* zone of the tidal marsh at Horn Point Laboratory (11.5 mg NH_3-N m^{-2} d^{-1}) determined by Lipschultz (1978) was used here. This rate is within the range of salt marsh estimates of N_2 fixation reported for the USA and European salt marshes by Rozema et al. (2000), but may be somewhat high for the Poplar Island marshes given their high NH_4^+ concentrations.

Tidal exchange

A study of tidal exchange of N, C and other constituents was conducted seasonally from November 2012 through November 2014 in the tidal inlet to Cell 1B. Due to initial problems with velocity measurements, only data from July 20, 2013 to November 2014 are reported here. Methods are described at length in Stevenson et al. (2014). Briefly, two tidal cycles over a 25 hour period were monitored each season. Velocity measurements were made with a Nortek Aquadopp Acoustic Doppler Profiler (ADP) mounted on a weighted PVC plate and placed in the center of one of the 1.83 x 1.83 m concrete culverts, equidistant from the open ends and sidewalls. Velocities are reported uncorrected for any vertical or lateral shear in the flow and it is assumed that the velocity measurement of the first bin, which was nominally located 45 cm above the bed, is representative of the mean velocity. Velocities were recorded every ten minutes by averaging two minutes of observations.

Discrete one liter water samples were collected hourly using an ISCO model 2700 automatic water sampler positioned above the culvert with approximately 3.6 m head. The sampling intake was located approximately 0.3-0.4 cm above the bottom of the culvert.

The samples were chilled with ice during sampling and returned to the lab at the conclusion of the sample period and refrigerated until processing the following day. Subsamples were vacuum filtered using and 4.7 mm and 2.5 mm Whatman GFF filters (nominal pore size 0.7µ), analyzed for total suspended solids (TSS) and particulate C and N as well as dissolved fractions. Analytical methods are summarized in Table 4-1.

For computing fluxes using concentrations from the ISCO samplers, the 10 minute velocity records were used, and hourly water quality data were interpolated to pair with velocity data points. Fluxes can be calculated as (Stevenson et al. 1988):

$$q_i = c_i u_i A_i \,.$$

Where q_i = instantaneous flux at time i, c = concentration, u = velocity and A = cross sectional area of the culvert, with subscript i referring to an instantaneous, discrete measurement. Due to variations in the diurnal tides at this site, which can be exaggerated by wind forcing, large differences in water volume between the beginning and end of the sampling period (Figure 4-4) can lead to large errors in calculations of net fluxes (Boon 1975; Nixon 1980). Therefore an alternative method was used here. Volume weighted concentrations were multiplied by the average tidal prism for the 24 hour monitoring period to obtain the flux (q) for each flood or ebb period. Average water volumes used to calculate fluxes for each period were: July 2013, 18583 m^3; November 2013, 8217 m^3; February 2014, 6028 m^3; May 2014, 8820 m^3; July 2014, 9886 m^3; November 2014, 10873 m^3.

The cross-sectional area of the culvert was calculated by multiplying the observed depth by the width of the culvert (1.83 m). Depth was calculated from hydrostatic pressure measurements obtained by a pressure sensor located on the Aquadopp, and confirmed with data from the Poplar Island tide gauge (USACE).

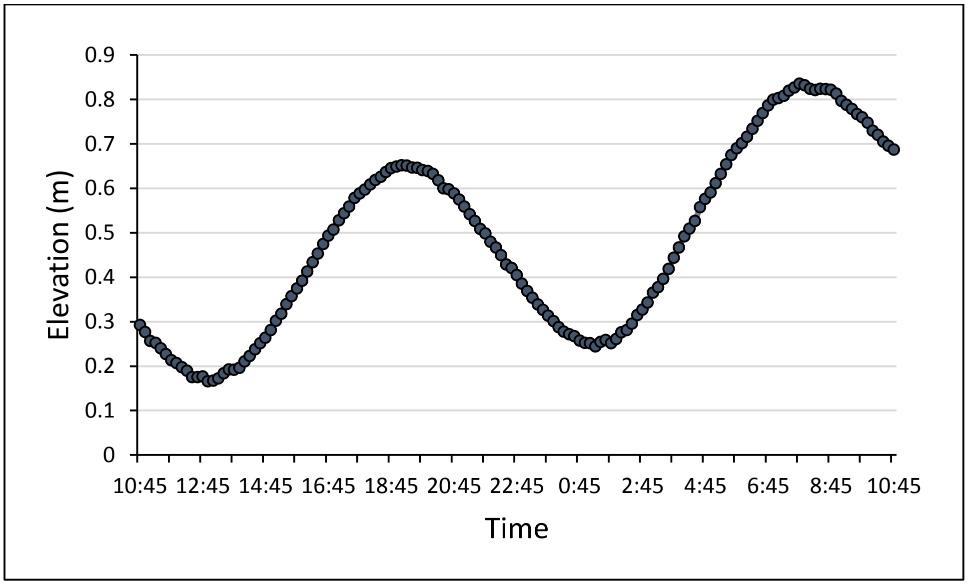

Figure 4-4. Water level at the tidal inlet to Cell 1B, July 26 - 26, 2013.

Concentrations of constituents that are not the focus of this study (phosphorus, total suspended solids, silica) are reported here as supporting data. Analytical methods are presented in Table 4-1.

Table 4-1. Summary of analytical methods used on water samples in the tidal exchange study.

Constituent	Method	Reference	Notes
NO_2+NO_3	USEPA Method 353.2	USEPA (1979)	Technicon Autoanalyzer II
NO_2	USEPA Method 353.1	USEPA (1979)	Technicon Autoanalyzer II
NH_4^+	USEPA Method 350.1	USEPA (1979)	Technicon Autoanalyzer II
PO_4	USEPA Method 365.1	USEPA (1979)	Technicon Autoanalyzer II
Total N & P	Digestion: Persulfate oxidation Colorimetric analysis: USEPA 353.2 (TN) and 365.1 (TP)	Valderama (1981); USEPA (1979)	Technicon Autoanalyzer II
Dissolved organic C	High temperature catalytic oxidation	Sugimura and Suzuki (1988)	Shimadzu TOC-5000A
Chlorophyll a	Acetone extraction, fluorescence	Greenberg et al. (1992)	Whatman 2.5 cm GFF glass fiber filter; Turner Designs Model 10-AU fluorometer
TSS	USEPA Method 160.2	USEPA (1979)	Whatman 4.7 cm GFF glass fiber filter
Particulate N & C	Combustion in pure O_2	Lane et al. (2000)	Exeter Analytical, Inc. (EAI) CE-440 Elemental Analyzer

Atmospheric deposition

Wet deposition of NH_4^+ and NO_3 at the National Atmospheric Deposition Program (NADP) Site MD13 (Wye Mills, MD) was used to estimate annual atmospheric N deposition on the marsh. (Boynton et al. 2008) use an estimate that wet deposition comprises approximately 67% of total N deposition. Average annual wet deposition for the 6 most recent years of available data at MD13, 2009 – 2014, were adjusted accordingly to estimate total annual N deposition.

Deep sediment N input

The input of N from the deep sediment is calculated as the difference between the sum of outputs (tidal, burial, denitrification) and the sum of the other inputs (atmospheric deposition and N-fixation).

Results

Standing stocks

Table 4-2 summarizes estimates of N and C standing stocks in the Cell 1B marsh. Ignoring the N pool in the biota, for which no estimate has been attempted but assuming it is negligible in proportion to the total N pool, the sediment below the root zone contains >90% of total N in the marsh (441,880 kg). The macrophyte vegetation at the end of the growing season represents about 0.5 % of the total N pool, or about 6.6 % of the root zone pool. Overall for N, the deep sediment ≫ root zone sediment > vegetation > water column.

The deep sediment C standing stock is not considered as part of the total C pool because it is considered sequestered, but the sediment root zone C reservoir is the largest part of the total C pool. The macrophyte vegetation represents about 14.4 % of the active C pool (above the root zone-deep sediment boundary). An estimate of the C reservoir in the biota is not given here, but is included in the table because it may be added as data become available on birds, fish, muskrats, turtles and other biota.

Table 4-2. Estimates of N and C standing stocks in Cell 1B.

	Standing Stocks (kg)	
Component	**TN**	**TC**
Macrophytes	2,829	73,129
Sediment –		
root zone <15 cm	40,173	435,056
deep >15 cm	441,880	---------
Water column	11	83
Biota	?	?
Total	497,120	711,910

Estimates of macrophytic vegetation biomass, N and C standing stocks in Table 4-2 were derived from the 2013-2014 biomass data in Table 4-3. The average biomass standing stock for the entire marsh in was ~ 2000 g dry weight m^{-2}, or ~189 metric tons. Using percent C and N values of 30 % and 1.5 %, respectively, the average standing stocks of C and N in biomass were 73,129 and 2829 kg, respectively.

The C content of the low marsh vegetation (*S. alterniflora*) was about 25% lower than expected, at least partially due to salts exuded on stems and leaves by the plants, which were not rinsed with fresh water prior to drying and analysis. The C concentration in *S. alterniflora* plants used in the decomposition experiment, which were rinsed with deionized water prior to analysis, was ~40% (Figure 4-5). The large sample size and number of replicates made rinsing biomass samples impracticable, so for the C standing stock estimate, the 30% C value was used with the biomass estimates.

The C:N ratio of the sediment, 10.8, is lower than five pairs of natural and constructed marshes in North Carolina surveyed by Craft et al. (1988) and marshes reviewed by

Nixon (1980). This reflects the high N content of the upper Chesapeake Bay dredged material, as well as the low organic matter content compared to natural marshes.

Table 4-3. Summary of mean (n = 6) macrophyte biomass and C and N standing stocks in Cell 1B, 2013 and 2014.

Year	**Component**	**Biomass (g m^{-2})**	**% C**	**C Mass (g m^{-2})**	**% N**	**N Mass (g m^{-2})**	**Area (ha)**	**C Standing Stock (kg)**	**N Standing Stock (kg)**
2013	LM - AG Live	17	30.74	5.1	2.07	0.3	6.92	355	24
	LM - AG Dead	894	34.88	311.8	1.64	14.6	6.92	21577	1012
	LM - BG	522	33.70	176.0	1.20	6.3	6.92	12177	434
	LM Total	**1433**		**492.9**		**21.2**		**34109**	**1470**
	HM - AG Live	1713	43.91	752.0	1.29	22.1	2.51	18875	555
	HM - AG Dead	292	42.15	123.2	1.38	4.0	2.51	3092	101
	HM - BG	458	38.15	174.7	1.21	5.5	2.51	4386	139
	HM Total	**2463**		**1049.9**		**31.7**		**26353**	**795**
	$\bar{x}$ Total Biomass	**1948**				**2013 Marsh Total**		**60462**	**2266**
2014	LM - AG Live	1174	39.90	468.6	1.63	19.2	6.92	32425	1326
	LM - AG Dead	176	27.37	48.0	1.73	3.0	6.92	3324	210
	LM - BG	1400	33.59	470.2	1.00	14.0	6.92	32536	967
	LM Total	**2750**		**986.8**		**36.2**		**68285**	**2502**
	HM - AG Live	930	45.43	422.3	1.27	11.8	2.51	10600	296
	HM - AG Dead	434	40.04	173.7	1.61	7.0	2.51	4359	175
	HM - BG	255	39.90	101.7	1.05	2.7	2.51	2552	67
	HM Total	**1618**		**697.7**		**21.5**		**17511**	**539**
	$\bar{x}$ Total Biomass	**2184**				**2014 Marsh Total**		**85796**	**3041**
	2013-2014 $\bar{x}$	**2066**						**73129**	**2653**

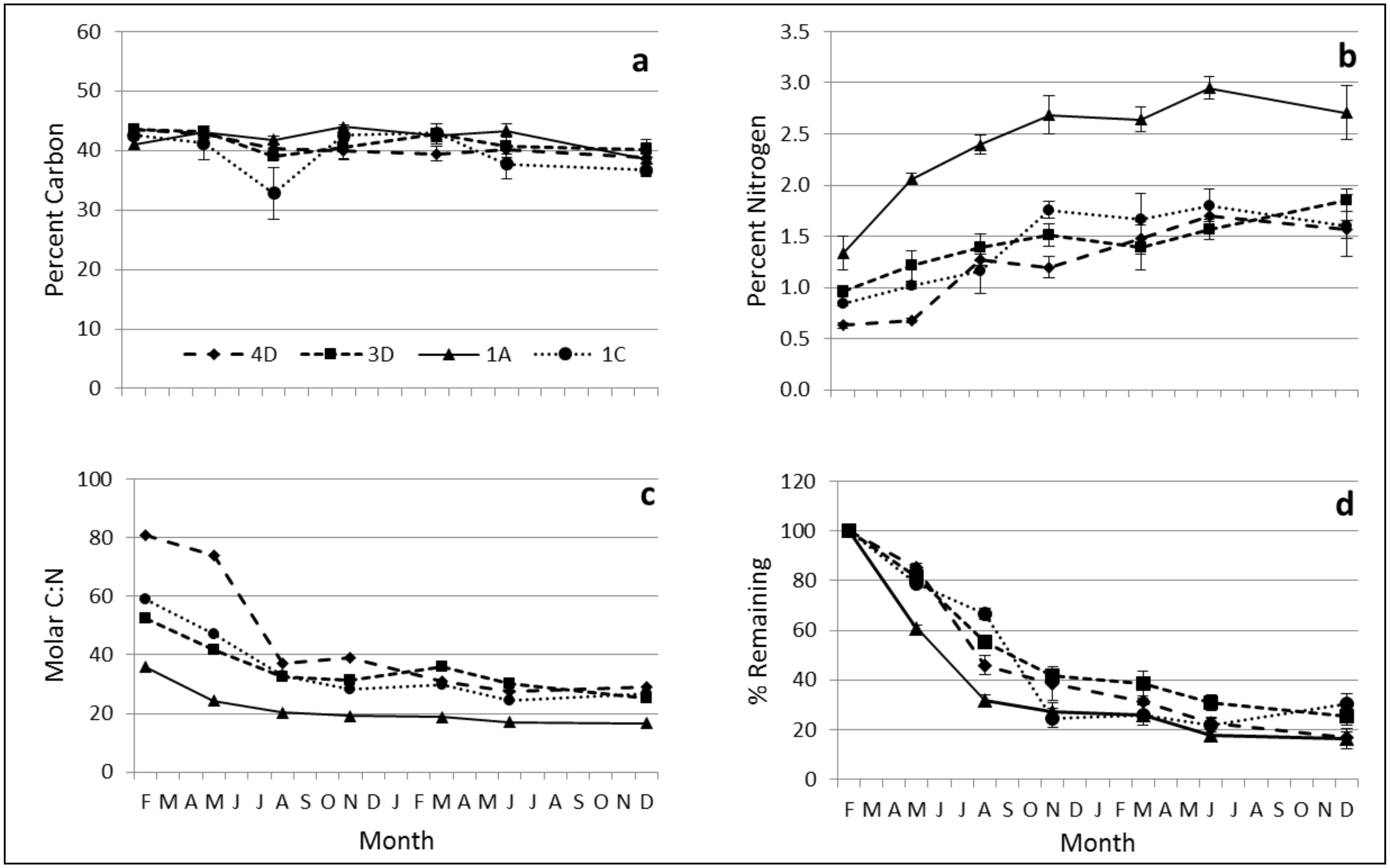

Figure 4-5. Results of the 2012-2013 litter bag study in four different marsh cells at Poplar Island. Cells 3D, 1A and 1C are dredged material marshes; Cell 4D has a sand substrate. Data points represent mean (n=3) ±standard error.

Rates

Important rates of N and C processes are summarized in Table 4-4. The N_2-fixation rate used here (11.5 mg m^{-2} d^{-1}) is about half the denitrification rate measured in Cell 1B. Burial, mineralization (N process) and decomposition (C process) are determined using an average decomposition rate from the litter bag study (Figure 4-5) and mean 2013-2014 biomass production (Table 4-3). The rate for all marshes in the litter bag study at the end of 9 months was similar, but it is clear from Figure 4-5d that the over the first six months the rate was higher in Cell 1A, where the initial tissue N concentration was about 25 % higher than in the other dredged material cells. Using ending tissue concentrations and percent biomass remaining, a burial rate of 16.44 mg N m-2 d-1 and 329 mg C m-2 d-1 were calculated (Table 4-4).

Table 4-4. Rates of processes used in calculating the N and C budgets for the Cell 1B marsh.

Constituent	Process	Rate (mg m^{-2} d^{-1})
N	Sediment NH_4^+ flux	-25.05
	Sediment NOx flux	8.36
	Sediment denitrification	-30.75
	Sediment N_2-fixation	11.50
	Assimilation (macrophyte)	82.19
	Mineralization	65.75
	Burial	-16.44
C	C-fix - macro	1644
	C-fix - algae - marsh	634
	C-fix - algae - creek	259
	Decomposition (macrophyte)	1315
	Burial	329

Annual wet deposition of N at NADP site MD13 near Wye Mills, MD averaged 4.53 kg N ha^{-1} y^{-1} from 2009 to 2014 Figure 4-6). Using the value cited by Boynton et al. (2008) for conversion of wet deposition to total atmospheric N deposition (67%), the annual N deposition on the 12.41 ha marsh is 83.9 kg N y^{-1}, a value consistent with the dominant agricultural land use in the region, (Anderson and Downing 2006).

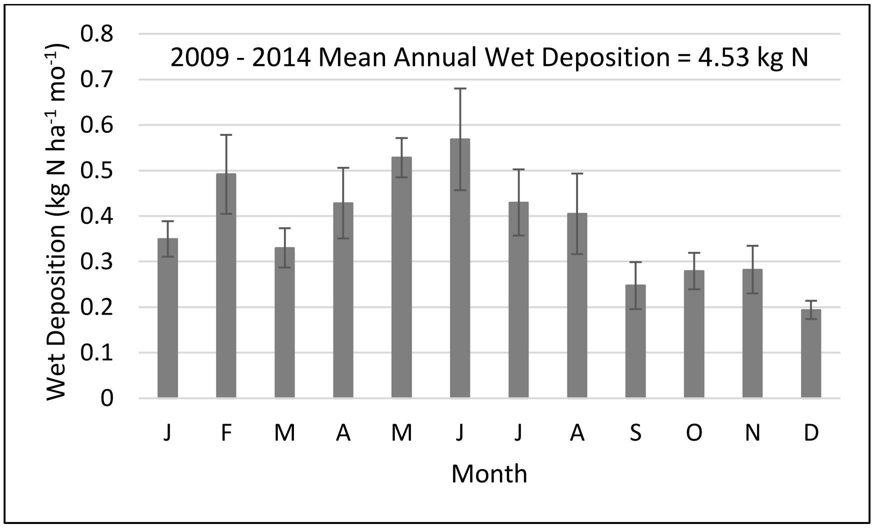

Figure 4-6. Monthly wet deposition of NH4+-N and NOx-N at NADP site MD13, Wye Mills MD, 2009-2014.

Tidal exchange

Average volume weighted flood and ebb concentrations of all constituents measured in the tidal exchange study are shown in Table 4-5. In general, ebb concentrations exceeded flood concentrations for all constituents except NO_3. On a seasonal basis, flood concentrations of inorganic N exceeded ebb concentrations during the growing season (May and July, shaded values in Table 4-5), but the reverse was true during the remainder of the year (November and February). During May 2013 the ambient NO_3 concentration at the edge of Eastern Bay (Buoy CB4.1C located just north of Poplar Island) was >25 µmol L^{-1} (Figure 4-7), leading to elevated flood NO_3 concentrations during that monitoring period. Particulate N (PN) concentrations were higher on the ebb tide throughout the study except for November 2013, and was extraordinarily high in July 2013 during the large scale vegetation dieback. Total N concentrations were always higher on ebb than flood tides.

With TN concentrations higher on all ebb tides, there was a net export of N for all monitoring periods (Table 4-6), leading to a net annual export of 665 kg N (Table 4-7). There was a net import of dissolved inorganic N, primarily as NO_3, during the second and third quarters, coinciding with high ambient concentrations and the growing season. Particulate N was exported throughout the year, and was especially high (-4.31 kg N) on ebb tides in July 2013, during vegetation dieback. There was a net export of total dissolved N (TDN) in all seasons except May 2014, when the high ambient NO_3 concentrations resulted in a net import due to NO_3 import. Total dissolved N export was in part due to NH_4^+ export, except in the third quarter during the peak of the growing season. On an annual basis, the NH_4^+ export is about 15% of the TN export (Table 4-7). The difference between TDN and the sum of NOx + NH_4^+ (329 - 133 = 196 kg) represents dissolved organic N (DON), indicating that a large proportion of net TDN export is organic N.

Carbon concentrations (particulate carbon, PC, and non-purgeable organic carbon, NPOC) were generally all higher on the ebb tides than on the flood, with the exception of November 2013, following the vegetation dieback. Particulate C concentrations were especially high in July 2013 at the height of the dieback. There was, overall, a net export of PC both seasonally and annually, with the largest export in the third quarter (Table 4-7). Net export of NPOC occurred in the first three quarters and on an annual basis (Table 4-7), but there was a net import in November 2013 following dieback.

Total suspended solids (TSS) concentrations were consistently higher on ebb tide than flood tide (Table 4-5). The highest concentrations of TSS occurred on ebb tide in July 2013 during the vegetation dieback, but were also high in July 2014. They coincide with high Chlorophyll *a* (Chl *a*) concentrations, which were 2x and 1.5x on the ebb versus flood tides in July 2013 and 2014, respectively. Overall, there is a moderately strong relationship between TSS and Chl *a* concentrations, and a strong relationship between Chl *a* and PC concentrations (Figure 4-8), suggesting that TSS concentrations were driven largely by phytoplankton during the periods monitored. The molar C:N ratio of particulate material for each deployment ranged from 6.92 to 8.06, with an average of 7.56, slightly higher than the Redfield (1958) ratio for phytoplankton.

Table 4-5. Volume weighted average concentrations of water column constituents on flood and ebb tides. Shaded values highlight where flood concentrations exceeded ebb concentrations in the same tidal cycle.

Date	Tide	TN	PN	TDN	NH$_4^+$	NOx	NO2	NO3	TP	SRP	TDP	PC	NPOC	TSS	Chl a	Si
		(μmol L^{-1})	(mg L^{-1})	(μmol N L^{-1})					(μmol P L^{-1})			(mg C L^{-1})		(mg L^{-1})	(ug L^{-1})	(umol Si L^{-1})
Jul-13	Flood	37.48	0.35	19.18	1.62	0.87	0.18	0.69	2.38	0.17	0.81	2.13	2.79	25.63	14.08	43.61
	Ebb	51.48	0.59	24.02	1.54	0.63	0.16	0.47	4.10	0.22	1.08	3.54	3.01	48.23	28.16	53.84
Nov-13	Flood	23.57	0.13	16.14	1.21	2.21	0.17	2.04	0.91	0.08	0.40	0.89	3.28	11.76	6.97	30.11
	Ebb	28.37	0.11	22.75	4.57	6.08	0.25	5.83	1.03	0.08	0.40	0.78	2.78	13.45	4.52	37.03
Feb-14	Flood	38.58	0.10	29.85	0.98	17.59	0.30	17.29	0.67	0.08	0.21	0.64	2.56	5.09	6.80	18.57
	Ebb	47.17	0.18	35.73	3.30	18.52	0.35	18.17	1.17	0.11	0.28	1.12	2.97	18.15	7.39	22.64
May-14	Flood	67.08	0.26	52.22	1.30	37.98	0.69	37.29	1.29	0.10	0.28	1.72	2.53	11.50	13.23	16.96
	Ebb	70.35	0.30	52.00	2.90	33.57	0.74	32.83	1.86	0.09	0.34	1.78	2.96	20.37	12.82	20.44
Jul-14	Flood	35.92	0.24	22.45	1.96	1.33	0.15	1.18	1.67	0.09	0.71	1.62	2.82	30.48	10.36	48.87
	Ebb	37.51	0.32	24.47	1.16	1.18	0.15	1.04	2.34	0.15	0.94	2.15	3.03	30.80	15.12	92.82
Nov-14	Flood	22.12	0.09	20.08	1.71	1.35	0.17	1.18	0.92	0.11	0.56	0.59	2.77	7.77	3.11	29.78
	Ebb	27.65	0.11	23.28	2.31	2.63	0.17	2.46	1.36	0.17	0.68	0.76	3.02	9.39	3.99	31.85

Table 4-6. Fluxes of water column constituents (kg per flood or ebb tide) for individual tidal exchange deployments and difference between flood and ebb fluxes (Δ). Flood and ebb fluxes are expressed as positive and negative values, respectively. Shaded values highlight flood concentrations that exceeded ebb concentrations in the same tidal cycle.

Deployment	Tide	TN	PN	TDN	NH_4^+	NOx	NO2	NO3	TP	SRP	TDP	PC	NPOC	TSS	Chl a	DSi
Jul-13	Flood	9.75	6.57	4.99	0.42	0.23	0.05	0.18	1.37	0.10	0.46	39.63	51.83	476.33	0.26	22.77
	Ebb	-13.39	-10.88	-6.25	-0.40	-0.16	-0.04	-0.12	-2.36	-0.12	-0.62	-65.78	-55.87	-896.17	-0.52	-28.12
	Δ	**-3.64**	**-4.31**	**-1.26**	**0.02**	**0.06**	**0.00**	**0.06**	**-0.99**	**-0.03**	**-0.16**	**-26.14**	**-4.04**	**-419.84**	**-0.26**	**-5.34**
Nov-13	Flood	2.71	1.05	1.86	0.14	0.25	0.02	0.23	0.23	0.02	0.10	7.33	26.94	96.62	0.06	6.95
	Ebb	-3.26	-0.88	-2.62	-0.53	-0.70	-0.03	-0.67	-0.26	-0.02	-0.10	-6.44	-22.84	-110.56	-0.04	-8.55
	Δ	**-0.55**	**0.18**	**-0.76**	**-0.39**	**-0.45**	**-0.01**	**-0.44**	**-0.03**	**0.00**	**0.00**	**0.89**	**4.10**	**-13.94**	**0.02**	**-1.60**
Feb-14	Flood	3.26	0.60	2.52	0.08	1.48	0.03	1.46	0.13	0.02	0.04	3.88	15.44	30.69	0.04	3.14
	Ebb	-3.98	-1.09	-3.02	-0.28	-1.56	-0.03	-1.53	-0.22	-0.02	-0.05	-6.77	-17.90	-109.38	-0.04	-3.84
	Δ	**-0.72**	**-0.49**	**-0.50**	**-0.20**	**-0.08**	**0.00**	**-0.07**	**-0.09**	**0.00**	**-0.01**	**-2.89**	**-2.46**	**-78.69**	**0.00**	**-0.69**
May-14	Flood	8.28	2.33	6.45	0.16	4.69	0.09	4.60	0.35	0.03	0.08	15.13	22.32	101.47	0.12	4.20
	Ebb	-8.69	-2.64	-6.42	-0.36	-4.15	-0.09	-4.05	-0.51	-0.03	-0.09	-15.72	-26.08	-179.70	-0.11	-5.07
	Δ	**-0.40**	**-0.31**	**0.03**	**-0.20**	**0.54**	**-0.01**	**0.55**	**-0.16**	**0.00**	**-0.02**	**-0.59**	**-3.76**	**-78.24**	**0.00**	**-0.86**
Jul-14	Flood	4.97	2.33	3.11	0.27	0.18	0.02	0.16	0.51	0.03	0.22	16.02	27.88	301.34	0.10	13.58
	Ebb	-5.19	-3.17	-3.39	-0.16	-0.16	-0.02	-0.14	-0.72	-0.05	-0.29	-21.25	-29.99	-304.54	-0.15	-25.79
	Δ	**-0.22**	**-0.84**	**-0.28**	**0.11**	**0.02**	**0.00**	**0.02**	**-0.20**	**-0.02**	**-0.07**	**-5.23**	**-2.11**	**-3.19**	**-0.05**	**-12.21**
Nov-14	Flood	3.37	0.95	3.06	0.26	0.21	0.03	0.18	0.31	0.04	0.19	6.46	30.16	84.50	0.03	9.10
	Ebb	-4.21	-1.15	-3.54	-0.35	-0.40	-0.03	-0.37	-0.46	-0.06	-0.23	-8.27	-32.89	-102.07	-0.04	-9.73
	Δ	**-0.84**	**-0.20**	**-0.49**	**-0.09**	**-0.19**	**0.00**	**-0.20**	**-0.15**	**-0.02**	**-0.04**	**-1.80**	**-2.73**	**-17.57**	**-0.01**	**-0.63**

Table 4-7. Net seasonal and annual fluxes of all constituents measured during the 2013 - 2014 tidal exchange study.

Quarter	TN	PN	TDN	NH_4^+	NOx	NO2	NO3	TP	SRP	TDP	PC	NPOC	TSS	Chl a	SiO_2
							Net Flux (kg y^{-1})								
1st	-126	-85	-86	-34	-14	-0.7	-13	-16	-0.8	-2.1	-503	-428	-13691	-1.9	-120
2nd	-71	-55	5	-35	96	-1.1	97	-27	0.2	-3.0	-104	-662	-13763	1.3	-152
3rd	-344	-458	-137	12	7	0.5	7	-106	-4.2	-20.4	-2790	-547	-37618	-30.9	-1561
4th	-124	-2	-111	-43	-57	-0.8	-56	-16	-2.0	-3.8	-81	121	-2801	2.6	-198
Annual (kg y^{-1})	**-665**	**-600**	**-329**	**-100**	**33**	**-2.1**	**35**	**-166**	**-6.8**	**-29.2**	**-3478**	**-1515**	**-67874**	**-28.9**	**-2031**
(kg ha^{-1} y-1)	**-59**	**-54**	**-29**	**-9**	**3**	**0**	**3**	**-15**	**-1**	**-3**	**-310**	**-135**	**-6049**	**-3**	**-181**

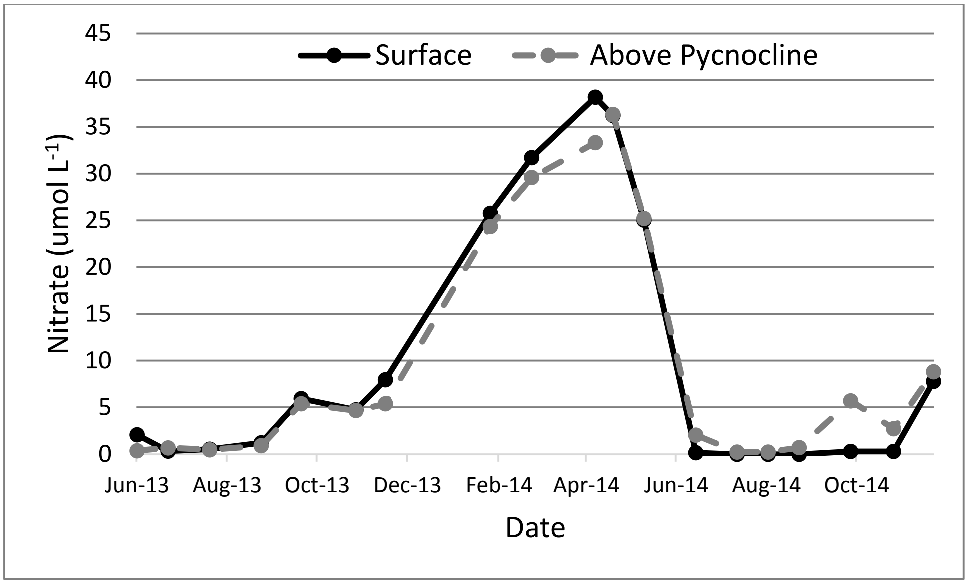

Figure 4-7. Nitrate concentrations at Chesapeake Bay monitoring buoy CPB4.1C, located approximately 5 km north of Poplar Island, June 2013 to October 2014, showing the winter/spring spike in ambient NO3 concentrations.

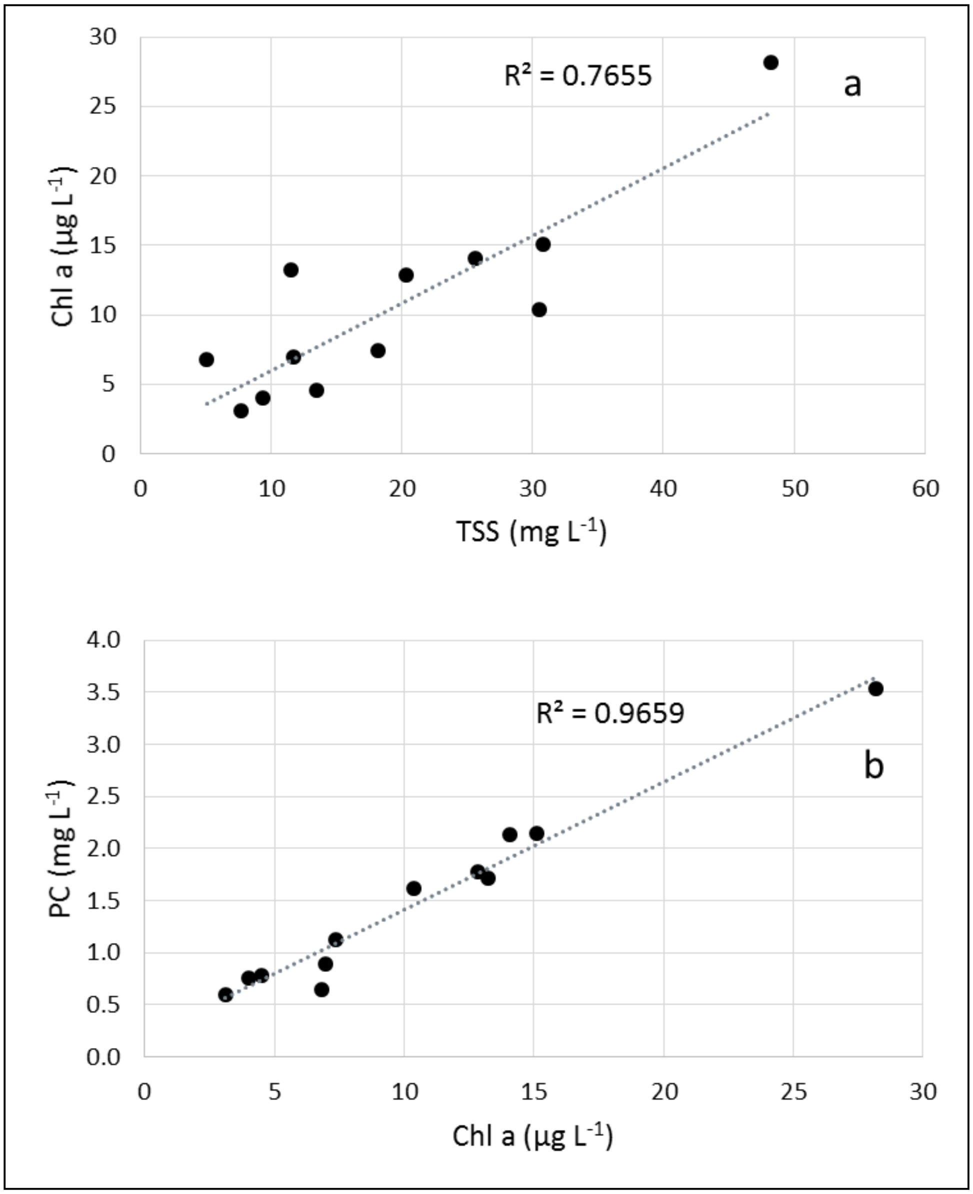

Figure 4-8. Relationship between Chl *a* and TSS concentrations (a) and PC and Chl a concentrations (B) over all deployments.

Overall, on an annual basis the Cell 1B marsh is exporting TN Table 4-7. Inorganic N is imported during the growing season as NO_3, which is imported on an annual basis, and

NH_4^+, which is exported on an annual basis. On an annual basis C is exported, as PC throughout the year, and also as NPOC most of the time.

Nitrogen Budget

The N exchanges of the Cell 1B tidal marsh determined in this study are summarized in Table 4-8. Total inputs from atmospheric deposition (84 kg y^{-1}) and N fixation (471 kg y^{-1}), excluding the deep sediment, are 555 kg y^{-1}. Total outputs are 2489 kg y^{-1}. The difference, 1934 kg y^{-1} is assigned to transfer from the deep sediment below the root zone, > 15 cm below the surface. This represents 78% of N inputs. Denitrification represents the largest output, about 50% of total outputs. No attempt is made to quantify N exchange associated with the biota, and the net flux expressed on an areal basis does not include the 1.19 ha habitat island.

Table 4-8. Nitrogen budget for the tidal marsh, Cell 1B, at Poplar Island.

Process	**Input**	**Output**	**Net Flux**	**Net Flux**
		(kg y^{-1})		**(g m^{-2} y^{-1})**
Atmospheric Deposition	**84**		**84**	**0.68**
N_2-fixation	**471**		**471**	**3.80**
Denitrification		**-1258**	**-1258**	**-10.13**
Sediment - NOx	342			**2.76**
Sediment - NH_4^+		-1025		**-8.26**
Tidal Exchange - TN		**-665**	**-665**	**-5.36**
Tidal Exchange - TDN		-329		**-2.65**
Tidal exchange - NH_4^+		-100		**-0.80**
Tidal exchange - NO3	35			**0.28**
Tidal exchange - PN		-600		**-4.84**
Burial		**-566**	**-566**	**-4.56**
Biota	**?**	**?**	**?**	**?**
Subtotal	**555**	**-2489**	**-1934**	
Deep sediment (by difference)	**1934**		**1934**	**15.58**

Carbon budget

Carbon exchanges for the Cell 1B tidal marsh are summarized in Table 4-9. Total inputs from *Spartina* spp. and algal photosynthesis are estimated at 80,082 kg y^{-1}. Carbon fixation by edaphic algae is about one third of macrophyte C fixation. Outputs are estimated to be 61,579 kg y^{-1}. Carbon loss associated with macrophyte decomposition is the single largest C loss, ~75% of fixed C. The difference between inputs and outputs, 18,509 kg y^{-1}, is close to the estimate for C fixed by edaphic algae, 23,502 kg $y^{-1,}$ or ~79%.

Table 4-9. Carbon budget for the tidal marsh, Cell 1B, at Poplar Island.

Process	**Input**	**Output** (kg y^{-1})	**Net Flux**	**Net Flux** (g m^{-2} y^{-1})
C - fixation - macrophyte	56,580		**56,580**	**504.3**
C - fixation - benthic algae	23,502		**23,502**	**209.5**
Decomposition - macrophyte		-45,262	**-45,262**	**-403.4**
Tidal exchange - PC		-3478	**-3,478**	**-31.0**
Tidal exchange - NPOC		-1,515	**-1,515**	**-13.5**
Burial - macrophyte		-11,324	**-11,324**	**-100.9**
Biota	?	?	**?**	**?**
Totals			**18,509**	**164.9**

Discussion and Conclusions

Nitrogen

Although the Poplar Island marsh is completely man-made, using an N rich substrate which is comprised of a high proportion of silts and clays and initially had a much lower organic content than most natural marshes, the imports and exports are largely within the ranges reported for natural marshes. For instance, denitrification is proportionately the largest output of N in the budget both in this study and in the study of Great Sippewisset Marsh (Valiela and Teal 1979), representing about 50% of total outputs in each study. Denitrification rates in this marsh, 10.1 g N m^{-2} y^{-1} were in the mid-range of those reported in Nixon's (1980) review, 3 – 22 g N m^{-2} y^{-1}; and Boynton et al. (2008) for Patuxent River marshes, 9.8 - 13.5 g N m^{-2} y^{-1}. In contrast, Rozema et al. (2000) reported a much lower rate of 0.8 g N m^{-1} y^{-1} for denitrification in the marshes of the Wadden Sea, but this estuarine system is much further north and experiences lower temperatures. Total N export via tidal exchange in Cell 1B, 5.36 g N m^{-2} y^{-1}, compares well with 11 g N m^{-2} y^{-1} for Great Sippewisset Marsh (Valiela and Teal 1979), and 1.2 – 24.6 g N m^{-2} y^{-1} for marshes reviewed by Nixon (1980). In spite of the enormous pool of N contained in the deep sediment, imports and exports appear to be within the ranges for natural marshes when normalized to marsh unit area.

Variability in precipitation, tidal prism (Boon 1975), macrophyte production (Kearney and Riter 2011) and other factors such as episodic grazing by herbivores can bias estimates of nutrient fluxes in a budget upward or downward, and scaling point measurements to the entire marsh is risky considering the heterogeneity in many processes (Bradley and Morris 1990). Yet in this study, although the marsh is very different from natural marshes in terms of the substrate and production levels, estimates of many of the rates and exchanges are remarkably similar.

The marsh also appears to be transforming N during the growing season, as natural marshes do (Heinle and Flemer 1976; Stevenson et al. 1976; Valiela and Teal 1979; Nixon 1980), by taking up inorganic N and exporting N in organic forms. There was net import of NO_3 and NH_4^+ during the growing season, indicated by net import in May 2014 and July 2013 and 2014. There was net export of TN during all sampling periods. The import is relatively small, 0.28 g m^{-2} y^{-1}, comprising less than 1% of macrophyte uptake. Outside the growing season, however, there is net DIN export, dominated by a net export of 8 kg NH_4^+ ha^{-1} y^{-1}, about half the amount reported by Valiela and Teal (1979) for Great Sippewissett Marsh. The difference between annual exports of TDN (329 kg) and DIN (102 kg) indicate that about two thirds of dissolved N exported is organic N. Thus, despite the deep sediment reservoir, the availability of inorganic N to the marsh is surprisingly limited, allowing the marsh to function as other marshes in taking up ambient inorganic N during the growing season to meet demand, and exporting it as organic N.

While inorganic N assimilation can't be partitioned here between macrophytes, edaphic algae and the microbial community, it appears that inorganic N assimilation exceeds the rates of remineralization and external inorganic N inputs during the growing season, assuming the N-fixation rate measured at HPL by Lipschultz (1978), is reasonable for this marsh. Seasonal porewater monitoring showing depletion of porewater NH_4^+ concentrations in the root zone through the growing season (Stevenson et al. 2007) also indicate that demand exceeds supply while macrophytes are actively growing and temperatures are high.

The difference between inputs and outputs is balanced here by transfer from the deep sediment. Although this rate has not been measured, it is key to understanding how long it may take production in these dredged material marshes to decline to levels near natural marshes, and for the problems presumed to be associated with an over-supply of N (e.g. dieback) to abate. It is interesting that the rate of the N input attributed here to the deep sediment, 15 g m^{-2} y^{-1}, is roughly equal to the groundwater input reported by Valiela and Teal (1979), 14 g m^{-2} y^{-1}. This is about half the annual standing stock of N in the macrophytes at the end of the growing season. *S. alterniflora* has been shown to translocate about 40 – 50% of the peak N standing stock in plant tissue (Morris 1980; Hopkinson and Scubauer 1984; White and Howes 1994b), although the amount re-translocated into new shoot biomass the next spring is questionable (White and Howes 1994b).

With a sediment pool of >400,000 kg N beneath the marsh in Cell 1B at Poplar Island, the estimated diffusion of NH_4^+ from the deep sediment into the root zone represents a depletion rate of only 0.2 % y^{-1} if all the sediment N were in an available form. Extractable NH_4^+ in dredged material, however, is only about 10 % of total N (Cornwell, pers. comm.; Stevenson et al.2010), but this still represents many years to depletion with steady diffusion upward. Diffusion of NH_4^+ through saturated soil is correlated with the clay content of the soil, and soils approaching 50% clay have a diffusion rate of <2 mm d^{-1} (Trostle et al. 2011). Consequently, diffusion in these marsh soils, with clay contents of ~20 – 50 % is likely to be on the order of 1-4 mm d^{-1}. What is not clear in these dredged material marshes is whether there are barriers to diffusion, such as dry layers, which may limit the amount of N potentially available for upward transport to the root zone. This area could be a focus of future research that would be critical in the design of dredged material marshes.

Burial was calculated in this study based on the N content and mass of macrophyte detritus remaining after two years. For this calculation it was assumed that the rate of decomposition of buried material slows due to anaerobic conditions and an increasing percentage of recalcitrant C compounds (White and Howes 1994a), which was indicated in our litter bag study. This is somewhat simplistic and does not include inorganic sediment inputs, which appear to be negligible based on a consistent net export of TSS. There is growing evidence of sediment erosion of along the creek banks which has been responsible for negative elevation changes in Cell 1A. This eroded material could supplement the marsh platform, but TSS concentrations in flood water are generally low, ≤ 20 mg L^{-1}, which is considered a threshold concentrations for a significant inorganic contribution to marsh vertical accretion (Kirwan et al. 2010) except in mid-summer. Thus the assumption here that N burial is primarily a function of organic matter deposition and decomposition, during which there is some inorganic N uptake (White and Howes 1994a), represent the bulk of buried N.

The tidal exchange study suggests that in tidal marshes constructed using fine grained dredged material, the chronological age of the marsh is not a good indicator of whether the marsh will import or export N. Childers et al. (2000) suggest that younger marshes import inorganic N on an annual basis until they build up large nutrient and C pools in the sediment, while older marshes reach an equilibrium between production and respiration and tend to show little net nutrient flux. However, it is not clear whether the marshes surveyed by Childers et al. (2000) were ebb or flood dominant, a primary consideration in whether marshes import or export sediment and dissolved materials on an annual basis. There is evidence from European studies that new marshes tend to be flood dominant and import TSS and nutrients, while older marshes become ebb dominant and export more C and nutrients (Boorman 1999; Hazelden and Boorman 1999). Theories of community succession (Bertness and Pennings 2000) and ecosystem development (Odum 1969) may be behind these changes, or they may be driven more by physical processes. It has not been established that these paradigms developed in systems with larger tidal ranges apply to microtidal systems such as Chesapeake Bay or in constructed marshes with initially high sediment N content.

In this study, there is a large export of TN, and only a small import of inorganic N (and a net export of P), suggesting that nutrient rich dredged material marshes do not follow this pattern described for young natural marshes over the first few years of development. How long they will take to reach an equilibrium, both in terms of production and respiration, and in terms of nutrient and sediment balances is an open question. A follow-up tidal exchange study in this marsh every few years would provide some insight into this process.

The tidal exchange study also provides an indication of the effect of vegetation dieback on N dynamics in the marsh. In July 2013, at the height of the large scale *S. alterniflora* dieback (Figure 4-1), there was a large export of TN, PN, PC, TP and TSS compared to July 2014, when dieback was confined to a much smaller area of the marsh and the formerly affected area was recovering. Chlorophyll *a* concentrations on the ebb tide also peaked that month, suggesting a large release of nutrients into the creeks. A large tidal prism that month likely inflated the numbers somewhat, but volume weighted ebb tide concentrations of those constituents were exceptionally high. Nonetheless, it is clear that there was a large release of nutrients in both organic and inorganic forms during the dieback. Edaphic algae was obviously abundant on the marsh surface at that time, and likely buffered the nutrient release to some extent.

Based on this study there appears to be a large amount of N cycling taking place within the marsh. NH_4^+ release rates from the sediment are estimated at 1025 kg y^{-1}, but tidal export of NH_4^+ is estimated at only 100 kg y-1. This suggests that most of the inorganic N released in the marsh is assimilated within the marsh system, either by benthic algae in the marsh or tidal creeks, by phytoplankton in the creek water, or by nitrifiers in the surface sediment. The high Chl *a* concentrations indicate that in the summer the creeks are acting as incubators, with shallow waters warming more than the estuary, and with abundant inorganic (and organic) nutrients from the marsh. Low dissolved oxygen near the heads of the creeks during the summer months (Stevenson et al. 2013) and measured summer oxygen demand in creek sediments (Stevenson et al. 2015) also indicate that these areas become heterotrophic in summer, and are probably hotspots of remineralization, especially when large scale dieback occurs during the growing season.

Carbon

Carbon cycling in marsh systems is of interest primarily for several reasons: it is a potential source of organic matter for secondary production in coastal systems via “outwelling” (Odum 2000), and buried C in the form of organic matter is the dominant component of vertical accretion in many, if not most, marshes (Turner et al. 2000; Nyman et al. 2006). On a global scale, marshes are considered a C sink and tidal marshes sequester more C per unit area than northern peatlands (Chmurra et al. 2003; Bridgam et al. 2006) while emitting negligible amounts of gaseous C that freshwater wetlands can release (Badiou et al. 2011).

The main question regarding C cycling in the marshes at Poplar Island is if the high rate of production in these nutrient rich dredged material marshes translates into high rates of C sequestration, export or CO_2 losses? Morris and Bradley (1999) found that long term fertilization of a tidal marsh increased the soil respiration rate more than the loss of soil organic matter. They attribute the difference to an increase in primary production resulting from the nutrient subsidy. The balance was an increased rate of sediment C turnover, but also a net C loss from the fertilized marsh.

The C budget for Cell 1B shows that tidal export (44.5 g C m^{-2} y^{-1}) is relatively low in this marsh compared to other marshes. Tidal C export in the marshes reviewed by Nixon (1980) ranged from 53 to 165 g C m^{-2} y^{-1}. This low export value is likely related to a number of factors, both process oriented and physical. The N supply in the dredged material produces elevated levels of primary production, but increased rates of decomposition likely result in higher losses of CO_2 through respiration. Although respiration wasn’t measured directly, an indication of this is the net oxygen consumption found in light/dark sediment core incubations in the 2013-2014 sediment–water exchange work (Stevenson et al. 2014).

In addition to high rates of internal processing, the microtidal environment does not promote large export from the marsh, and the limited size of the inlets of the Poplar Island marshes minimize losses of organic material via storm tides and ice scour (and also prevents deposition of inorganic matter on the marsh surface), which may remove large amount of organic matter from natural marshes. The result is large areas of *S. alterniflora* wrack deposited near the low marsh – high marsh boundary, especially when there is a large dieback event. This also can lead to extra CO_2 losses through the loss of production and increased respiration associated with patches of wrack (Macreadie et al. 2013).

Macrophytes are the dominant primary producers in the Poplar Island marshes, but edaphic algae can be responsible for a significant portion of fixed C, estimated to be about 30 % in this study. The algal production rates used here are likely on the high end of potential rates because they are derived from measurements made in the Cell 1B marsh in 2013 and 2014, during a large scale dieback event and recovery (Stevenson et al. 2015). Observations indicated that edaphic algae responded rapidly to the increased light reaching the surface of the marsh and the abundant nutrients during the *S. alterniflora* dieback by forming a thick mat on the sediment surface. Undoubtedly this compensated somewhat for lost macrophyte C fixation. Algal production in an earlier study comparing dieback areas to healthy marsh areas in another marsh cell at Poplar Island indicated that is indeed the case, with production rates about 25 % lower in healthy areas (Stevenson et al. 2013). Similar trends have been reported from other marshes (Zedler 1980, Sullivan and Currin 2000). Nonetheless, this community is responsible for a significant part of the carbon inputs in these marshes.

The C fixed by the benthic algal community is also important because it has a lower lignin content that macrophyte vegetation, and stable C and S isotope ratios indicate that organic matter of algal origin readily enters the aquatic food web (Sullivan and Moncreiff 1990; Stribling and Cornwell 1997). Thus the C fixed by benthic algae in the Poplar Island marshes, although probably insignificant in contributing to vertical accretion, is important in nutrient cycling as part of the base of the food web in the marsh systems.

The overall surplus of C in the budget is of similar magnitude to the C fixed by benthic algae, and likely represents the decomposition and uptake of that organic matter within the marsh, and perhaps export in biota, which was not quantified during the tidal flux measurements. This can be a significant fraction of primary production in marshes (Nixon 1980).

Carbon sequestration occurs through burial of organic matter, and especially in marshes like the Poplar Island marshes which receive little inorganic sediment subsidies, it is the most important factor in maintaining vertical accretion over the long term. Measurements of vertical accretion in the Poplar Island marshes indicated rates of 5 - 10 mm y^{-1} in the low marsh (Chapter 2). While some of this material appears to be material eroded off the marsh surface or tidal creek banks (Stevenson et al.2010), lower accretion rates in dieback areas (Chapter 2) suggest that organic matter accumulation is, indeed, the most important factor in vertical accretion.

Thus, it would appear that the high N status of the dredged material in the Poplar Island marshes may lead to high rates of internal C cycling (respiration) which, combined with high production rates, lead to high rates of vertical accretion. It does not, however, appear to lead to high rates of C export. It is unclear, however, how much may be transported out of the marsh in juvenile or predatory fish, birds, turtles or other biota.

Conclusions

By attempting to quantify the inputs and outputs of N and C in one dredged material marsh at Poplar Island, some insights have been gained into the processes at work and how these marshes function with respect to natural marshes. It appears that most rates, imports and exports are within ranges reported for many natural marshes (on an areal basis) along the Atlantic and Gulf coasts. There is not an extraordinary export of N considering the large N pool contained in the dredged material, so the construction design has been effective at retaining N. The N export is probably larger than would be expected from such a young marsh, however, so chronological age is likely not a good indicator of N export rates in constructed marshes, which may have higher or lower export rates than similarly aged natural marshes.

High rates of primary production appear to lead to rates of organic matter burial that are adequate to maintain vertical accretion, albeit at rates below the current rate of SLR in Chesapeake Bay (Chapter 2). However, *S. alterniflora* dieback had a dramatic effect on the marsh, promoting N and C losses during the summer when rates of remineralization and denitrification are high, and likely changing the balance between macrophyte and benthic algal primary production. This has implications for C and N sequestration, as well as for vertical accretion, and warrants efforts to elucidate the causes and undertake design revisions to reduce dieback in created marshes. The transfer of NH_4^+ from the deep sediment appears to be substantial, probably similar to the rate of N export via denitrification, to support the observed rates of primary production. Finding a way to slow this transfer may be beneficial in the short term.

Overall, the dredged material marshes at Poplar Island demonstrate that although they may differ from natural marshes in some fundamental ways, and that the time to full functional equivalency with natural marshes may be long, in many respects they are already functioning quite normally. Their long term sustainability is still in question, however, and future studies to help resolve questions about deep sediment transfer, N-fixation rates and C sequestration will improve our understanding of nutrient cycling and long term trends.

Chapter 5: Synthesis

If restoration is the acid test of our knowledge of tidal marshes, the Poplar Island studies indicate that our understanding of how these ecosystems function is growing, but still incomplete. We cannot fully anticipate their response to nitrogen (N) or sea level rise (SLR), or to predict whether they will be importers or exporters of nutrients and materials, in part because we are looking through the lens of previous studies. For example, the initial plans for the marshes at Poplar Island included the use of fertilizers at planting to promote rapid plant establishment and growth, because the design and expectations for functional trajectories were based on experiences where substrates containing low levels of nutrients were used to construct new marshes (e.g. Craft et al. 2002). As we found, a nutrient rich substrate like the upper Chesapeake Bay dredged material requires no fertilization and produces trajectories quite different from low nutrient constructed marshes, approaching functional equivalency with natural marshes from a surplus rather than a deficit. More forethought might have avoided that particular misconception, but based on our current level of understanding of marsh response to N and of vegetation dieback, the scope and frequency of dieback in these marshes is not likely to have been anticipated. A firm link has yet to be made between the two, but our studies strongly suggest such a connection.

Likewise, there has been much debate surrounding the combined effects of N on marshes, due to coastal eutrophication, and SLR. Some studies have suggested that reductions in belowground biomass production under high N loads will result in loss of marsh integrity and erosion with rising sea level (Turner 2011; Deegan et al. 2012). Others seem to suggest that the increased shoot production may trap more sediment and contribute more organic matter to the surface, offsetting belowground losses and allowing vertical accretion to keep up with SLR (Kirwan et al. 2010; Morris et al. 2013). In both cases, experience in one type of marsh (e. g. macrotidal, mesotidal or microtidal; high or low nutrient) may have limited the conclusions. Our studies suggest that accretion rates are correlated with biomass production, especially aboveground biomass, but that implies that dieback will have a detrimental effect on accretion. If dieback is linked to the N supply, there may be a threshold N supply level beyond which the asset becomes a liability, as in the subsidy stress model (Odum et al. 1995). This appears to be the case in our study, with oscillations resulting not so much from pulses of growth as from pulses of dieback. This may have important implications for the sustainability of marshes being constructed with nutrient rich dredged material, as well as for natural marshes in areas where coastal eutrophication is intensifying, such as Asia.

Marshes have also become a focus for silicon (Si) research, due to their location at the interface between terrestrial sources, primarily weathered mineral Si, and coastal ecosystems where Si is essential for the growth of diatoms, the basis of coastal food webs including commercially important fisheries. A few studies have suggested that marshes may provide a source of Si in summer (Struyf et al. 2006; Vieillard et al. 2011), when it has been depleted in many aquatic systems. This study demonstrates that constructed marshes can be a source of Si throughout the year if they are constructed with fine grained material. Although sediment Si fluxes were not measured directly in this study, the net annual export of both dissolved and biogenic Si imply that large amounts of biogenic Si contained in the sediment are supplying macrophytic vegetation and benthic diatoms, leading to large scale export to the local estuary, a potential benefit of dredged material marshes that has been overlooked.

This study also challenges some ideas about marsh maturity, the import-export balance of nutrients and materials, and accretion rates, that have developed primarily as a result of studies of restored marshes in Europe and the U.S. Some of these studies have implied that younger, expanding marshes in mesotidal to macrotidal systems tend to be flood dominated, have higher accretion rates and tend to import nutrients and total suspended solids (TSS) on an annual basis (Boorman 1999; Childers et al. 2000). Furthermore, they import large amounts of inorganic nutrients and export organic forms. At Poplar Island, in a microtidal system, there are higher accretion rates early in the development of the dredged material marshes with a slowing as they age, similar to the trends described for the European marshes. However, this appears to be related to high rates of in situ organic matter production rather than flood domination and sediment trapping. We found a net export of TSS from the dredged material marsh studied, and although there is a net import of nitrate on an annual basis, there is a much larger export of ammonium producing an overall export of inorganic N. Thus, the definition of marsh maturity based on soil nutrient content, import-export balances or flood-ebb domination is not valid in the context of these nutrient rich dredged material marshes.

Thus it seems that Bradshaw's (1994) statement about restoration and soils was well informed. The Poplar Island marsh studies demonstrate that we can create marshes that in many ways behave like natural marshes, but not entirely, and understanding the soil should be a top priority in marsh restoration projects. The approach of Odum et al. (1995) and Nixon (1969), too, to study extreme environments to learn about how ecosystems function, may be well heeded. The Poplar Island studies may have produced some valuable insights into the responses of natural marshes to N and SLR, and to the causes and consequences of marsh vegetation dieback, and leave much to pursue in future studies.

The use of dredged material to build habitat suitable for wildlife has become a focus of wetland restoration efforts over the last 30 years (Costa-Pierce and Weinstein 2002) . As a result, large dredged material wetlands are being constructed within bays and estuaries where the dredged material can be put to a beneficial use replacing lost habitat. The cost of these projects can run into the hundreds of millions of dollars, so insuring that expectations are realistic, designs are appropriate, the systems are self-sustaining and they are ecologically similar to their natural analogs is important. Studying these marshes is essential not only to insure that tax payer money is well spent, but to learn about the natural systems they are intended to model, and eventually to become.

Appendix

Table A-1. Mean and standard error of annual aboveground (AG) and belowground (BG) biomass production in Cell 3D and Cell 4D.

Year	Cell 3D									Cell 4D								
HM	**AG Live mean**	**se**	**AG Dead mean**	**se**	**AG Total mean**	**se**	**BG Total mean**	**se**	**n**	**AG Live mean**	**se**	**AG Dead mean**	**se**	**AG Total mean**	**se**	**BG Total mean**	**se**	**n**
2004														606	71.9			3
2005																		
2006					2761	233			3							308	62	3
2007	532	168	1504	153	2036	244	745	191	3									
2008	1205	245	713	228	2521	576	488	142	6	470	38	204	34	675	16	1279	165	3
2009	803	242	931	219	1734	178	412	90	6	728	21	491	52	1219	56	1222	274	3
2010	807	20.3	644	109	1451	156	725	163	3	693	2828	189	70	882	183	2351	132	3
2011	1430	162	369	75	1799	234	467	64	3	713	114	274	90	986	25	2028	727	3
2012					1329	287	1063	131	6					1308	155	1109	248	6
2013	1030	144	598	224	1629	233	750	168	6	535	65	666	268	1201	248	900	105	6
2014	929	121	376	95	1305	171	889	294	6	619	94	510	146	1129	182	2357	414	6
LM																		
2004														416	69			4
2005																		
2006					1934	226			3							1219	253	3
2007	1402	603	461	167	1863	549	1277	171	3									
2008	1197	551	63	36	1261	576	1662	316	6	950	92	215	119	1164	28	3035	550	3
2009	462	146	570	112	1032	137	1616	602	6	683	90	53.2	44	1215	116	1913	313	3
2010	1598	133	205	165	1804	421	2103	345	3	974	10	257	26	1231	64	2802	250	3
2011	300	78	106	34	406	106	1085	391	12	1263	155	200	24	1463	135	3168	392	3
2012					1029	384	1331	370	6					765	105	2190	205	6
2013	732	126	261	74	993	104	811	135	6	422	54	360	38	781	46	2211	342	6
2014	1061	90	131	31	1192	77	1520	281	9	653	67	199	39	852	101	3373	384	6

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