ABSTRACT

Title of Thesis: EVALUATING FEEDBACKS BETWEEN VEGETATION AND SEDIMENT DYNAMICS IN SUBMERSED AQAUTIC VEGETATION (SAV) BEDS AND CREATED MARSHES OF LIVING SHORELINES IN CHESAPEAKE BAY

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Intertidal marshes and subtidal submersed aquatic vegetation (SAV) provide similar ecosystem services such as wave attenuation, provision of nursery habitat, water filtration, and sediment and nutrient retention. They are often found together in the coastal zone, especially when marshes have been created for shoreline protection in living shorelines. This study examines sediment dynamics within the created marshes of living shorelines and adjacent nearshore SAV habitat in mesohaline Chesapeake Bay, and within emergent, patchy SAV beds of the Susquehanna Flats. The naturally occurring radioisotopes ^{7}Be (half-life: 53.3 days) and ^{210}Pb (half-life: 22.3 years) were used to calculate seasonal- and decadal-scale sedimentation rates. Mud content, organic content, and nutrient concentrations were analyzed to describe sedimentary characteristics. Coastal habitats in the Chesapeake Bay exert significant influence on local sediment dynamics, further research on feedbacks between coastal vegetation and sediment dynamics can improve our understanding on how coastal ecosystems interact with Bay-wide shifts in sediment dynamics.

EVALUATING FEEDBACKS BETWEEN VEGETATION AND SEDIMENT DYNAMICS IN SUBMERSED AQAUTIC VEGETATION (SAV) BEDS AND CREATED MARSHES OF LIVING SHORELINES IN CHESAPEAKE BAY

by

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Evaluating feedbacks between vegetation and sediment dynamics in Submersed Aquatic Vegetation (SAV) beds and created marshes of living shorelines in Chesapeake Bay

Introduction

The ability of coastal habitats to support healthy vegetation communities in the Chesapeake Bay, as well as on a national/global scale, is threatened by a number of environmental and anthropogenic stressors, including but not limited to: sea-level rise (SLR), coastal erosion, shoreline hardening, and eutrophication (Orth et al, 2017; Fisher et al, 2006; Kemp et al, 2005). These stressors affect both intertidal (marsh) and subtidal (submersed aquatic vegetation; SAV) plant communities, which often occur in tandem and have linked ecosystem services. For example, coastal marsh erosion is exacerbated in areas without SAV to reduce wave energy (Palinkas et al, 2018); this erosion can lead to increased levels of suspended sediments in the water column, diminishing water clarity, and making sediments unsuitable for SAV by increasing mud content and accumulation rates (Orth et al, 2010; Bilkovic et al, 2016; Han et al, 2012). Coastal erosion leads to property loss and often a desire by property owners to stabilize their shorelines.

In the Chesapeake Bay, about 25% of the shorelines are armored with “hard” structures (ex: bulkheads, seawalls, rip rap), with about 70% of Maryland’s shorelines being categorized as eroding (Palinkas et al, 2017). Shoreline armoring has been linked to habitat loss, shifts in sediment characteristics and dynamics, declines in benthic diversity, and increased scouring and erosion in front of these structures (Kornis et al, 2017; Patrick et al, 2014). Hardened shorelines are also expensive to maintain, as they generally require consistent maintenance and repairs over time (Temmerman et al, 2013). Living shorelines are an alternative strategy to combat coastal erosion while providing the ecosystem services that accompany marsh habitat. These ecosystem services include wave/current attenuation, sediment/nutrient sequestration, provision of nursery habitat, and water filtration (Bilkovic et al, 2016; Chapman and Underwood, 2011; Davis et al, 2015). The created marshes of living shorelines have the ability to self-repair and accrete enough sediment to keep up with local sea-level rise (Gittman et al, 2014; Temmerman et al, 2013). Legislation has been passed in recent years to encourage the installation of living shorelines in Maryland, however environmental managers and researchers are trying to assess the influence of these structures on nearshore SAV populations (Palinkas et al, 2018).

SAV are sentinel species for Chesapeake Bay and are used as a primary indicator of water quality (Orth et al, 2017). SAV distributions in the Bay have a complex history, with massive losses beginning in the 1900s, due to a confluence of compounding factors such as wasting disease, eutrophication, coastal erosion, storm impacts, and overall declining water quality (Orth et al, 2010; Kemp et al, 2005; Orth and Moore, 1984). The greatest loss of SAV occurred following Tropical Storm (TS) Agnes in 1972, severe flooding and large sediment loads overwhelmed SAV beds in the Bay (Orth and Moore, 1984). SAV populations in the Chesapeake Bay began to recover in the 1980s as water clarity improved from a string of dry years (low precipitation) and improvements in wastewater management practices helped decrease nitrogen inputs and improve water clarity (Gurbisz and Kemp, 2014; Orth et al, 2010). Since 1984, as a result of management efforts, nitrogen concentrations have been reduced by 23% and phosphorus concentrations decreased by 8%; this reduction in nutrient inputs has allowed SAV area to expand by 170 km^2 between 1984-2015, the greatest area covered by SAV in the Chesapeake Bay in nearly 50 years (Lefcheck et al, 2018).

SAV beds offer a suite of ecosystem services to the surrounding environment such as wave/current attenuation, sediment stabilization, provision of nursery habitat, and sediment/nutrient sequestration (Orth et al, 2017; Heck et al, 2003). Sediment dynamics in SAV beds in the Chesapeake Bay and elsewhere have been the focus of several recent studies. These studies and others show that SAV presence can enhance deposition rates within established beds, protect sediment beds from erosion, and improve local nutrient cycling (Russ and Palinkas, 2018; Gurbisz and Kemp, 2014; Ganthy et al, 2013; Orth et al, 2010). Studies performed within the Susquehanna Flats in the main SAV bed show that the influence of SAV beds on sediment dynamics is seasonal, in line with SAV phenology (Russ and Palinkas, 2018). SAV abundance is greatest during the summer months, SAV abundance is lower during the spring as SAV is recovering from winter dieback (Russ and Palinkas, 2018). It remains unclear how smaller patches of SAV, which are more common in the Bay and elsewhere, influence local sediment dynamics.

This study evaluates sediment dynamics within these intertidal and subtidal vegetated environments, from the created marshes of living shorelines to SAV habitats adjacent to shorelines in mesohaline Chesapeake Bay and in patches near the main SAV bed of the Susquehanna Flats. We consider seasonal- and decadal-scale sedimentation within these habitats. Chapter 1 focuses on living shorelines, evaluating how their installation affects sedimentary characteristics in adjacent SAV habitat and whether these structures impact local SAV presence/absence. Chapter 2 evaluates seasonal sedimentation rates and characteristics inside and outside emergent, patchy SAV beds in the Susquehanna Flats to determine whether insights from past studies apply to smaller patches.

Chapter 1: Impact of Living Shorelines on Sediment Characteristics in Adjacent Nearshore Habitats in the Chesapeake Bay, U.S.A.

1.1 Introduction

Coastal resilience has become an increasingly prevalent topic when discussing the emergent challenges posed by climate change, such as sea-level rise (SLR), coastal inundation, and increasing storminess. In the U.S., more than 40 million people and over $3 trillion in combined assets are consolidated along the coastlines (Temmerman et al, 2013). Global mean SLR from 1970-2009, was calculated to be 0.98 ± 0.33 mm yr^{-1}, however mean SLR is 3-4 times the global average along the Northeast US Atlantic Coast (3.80 ± 1.06 mm yr^{-1}) (from Cape Hatteras to Boston; 1970-2009) (Sallenger Jr. et al, 2012; Boesch et al, 2013; Boon, 2020). According to Pew Charitable Trust, 14% of the U.S. shorelines are armored with manmade defenses and by 2100, NOAA estimates that 33% of the U.S will have hardened shorelines (Pace & Morgan, 2017). In the Chesapeake Bay specifically, 18% of the tidal shorelines have been hardened (Bilkovic et al, 2016).

Traditional shoreline armoring structures require consistent maintenance and disconnect terrestrial and aquatic systems (Patrick et al, 2014; Davenport et al, 2018). These structures can increase the amount of wave energy reflecting back into nearshore benthos, scouring the bottom and increasing the amount of suspended sediment in the water column (Palinkas et al, 2018; Patrick et al, 2014; Currin et al, 2010). This is harmful for benthic organisms like SAV because excessive erosion can cause plant rhizomes to be exposed and ultimately dislodged if erosion is severe (Palinkas & Koch, 2012). In recent years, living shorelines have been proposed as a more ecologically sound alternative to conventional shoreline armoring approaches such as sea walls, rip rap, bulkheads, etc. (Temmerman et al, 2013; Davis et al, 2006). Living shorelines (LS) are a coastal protection strategy that involves creating salt or freshwater marshes to buffer shorelines and reduce shoreline erosion (Gittman et al, 2016). Living shorelines require minimal maintenance after installation as they can accrete enough sediment to keep pace with SLR, reduce coastal erosion, improve water quality, and provide critical nursery habitat to a number of aquatic species (Pace & Morgan, 2017; Gedan et al, 2011; Currin et al, 2010; Bilkovic & Roggero, 2008). Living shorelines are resilient to storm impacts and have the ability to self-repair after enduring storm-related damages (Leonardi et al, 2018; Leonardi et al, 2016). The ability of living shorelines to self-repair by means of sediment accretion is dependent on regional sediment loads and transport and is by no means a given (Neubauer, 2008).

Living shorelines can be a more ecologically sound alternative to shoreline armoring when it comes to combatting shoreline abatement. Living shorelines, when successfully installed, improve nearshore conditions for a variety of fish, invertebrates, etc. while potentially restoring ecosystem services by up to 90% as compared to an armored or bare shoreline (Bilkovic et al, 2016; Rodríguez- Calederón, 2014). Living shorelines have been shown to help increase the diversity and abundance of nearshore benthic and infauna organisms. Shoreline vegetation from the living shorelines supports these communities by providing energy inputs to nearshore, detrital-based food webs, helping contribute to trophic transfer by means of secondary production (Davenport et al, 2018).

Although salt marshes and living shorelines have been credited with helping support local nearshore benthic communities and local coastal processes (nutrient/sediment sequestration), there have not been many studies looking specifically into the changes in SAV abundance and/or sedimentary environment following the installation of living shorelines (Davenport et al, 2018; Gittman et al, 2016; Kornis et al, 2017). This information could be important for environmental managers as SAV is sensitive to significant changes in their sedimentary environment and water clarity (Patrick et al, 2018; de Boer, 2007).

The purpose of this study, is to assess the potential impacts of living shoreline installation on sediment characteristics and/or SAV presence in adjacent nearshore habitat at several sites in mesohaline Chesapeake Bay. To address this goal, we (1) compare seasonal-scale sedimentation in nearshore environments adjacent to living shorelines and nearby reference (unaltered) shorelines, (2) establish sediment geochronologies from which potential changes in sediment characteristics before and after installation are evaluated, and (3) assess the presence of SAV before and after installation. We hypothesize that the installation of living shorelines will increase the mud and organic content in the nearshore sedimentary environment by enhancing deposition of fine-grained sediment in the subtidal environment. SAV generally thrives in sandier sediment where organic content is below 5%, whereas in muddier environments with high organic content (5%>) there is a higher chance of dislodgement (Wicks et al, 2009). This study informs management and permitting of living shorelines by environmental managers in the Chesapeake Bay and provide additional information on how the installation of living shorelines affect nearshore environments.

2.1 Methods

Study area and site selection

The living shorelines included in this study are located within the Choptank River basin and the Miles River basin, subtributaries of the Chesapeake Bay. Eight living shoreline sites were selected with the guidance of managers and restoration practitioners (Maryland Department of Natural Resources). Two distinct types of shorelines (8 sites each) were selected: living shorelines and adjacent unaltered shorelines. Four of the living shorelines sites that were selected had persistent SAV beds in the adjacent subtidal prior to the installation of the living shoreline (Queens Landing, Oppenheim, Ruesch, and Hatton Garden) while the other four living shoreline sites did not have SAV present before installation (Myrtle Grove, Maritime Museum, Environmental Concern, San Domingo).

A weighted overlay of SAV density data (aerial photographs) from 1978 (earliest available) to 2005 (year prior to installation of most of the study sites) was performed in ArcGIS to select 4 living shorelines with and 4 living shorelines without relatively dense SAV beds prior to installation. Nearby (typically within ~0.5 km) reference shorelines (unaltered shorelines) were selected for comparison with each site (see Fig. 1). The presence or absence of submersed aquatic vegetation (SAV) post-installation was determined using 2004-2018 SAV abundance data referenced from the VIMS Interactive SAV Map (https://www.vims.edu/research/units/programs/sav/access/maps/index.php).

Field Methods

Push-cores (~10 cm long) were taken in 2017 and 2018, both within the created marshes of the living shorelines, as well as the subtidal environment adjacent to the living shorelines and reference shorelines. Within the created marshes, 3 replicate cores were taken at Ruesch (RUL), Hatton Garden (HG), Environmental Concern (EC), and San Domingo (SDL) in 2017; cores were taken in the high and low marsh at Queens Landing (QL), Maritime Museum (CBMM), Oppenheim (OP), and Myrtle Grove (MG) in 2018. Results reported for mud and organic content reflect average values for both replicate push cores as well as push cores taken in the high and low marsh. Due to the relatively short half-life of ^{7}Be, sediment deposition rates were determined for only one core per site (see below) Vibracores (~3m long, ~8cm diameter) were collected in the subtidal adjacent to all living shorelines and all reference shorelines in 2018 to assess decadal-scale sedimentation.

Laboratory Methods

All cores were returned to the laboratory for further analysis. Push-cores were sectioned into 1-cm intervals; vibracores were sectioned into 1-cm (top 20 cm), 2-cm (20-80 cm), and 4-cm (80 cm end of core) increments; increment thicknesses varied down core to provide the highest resolution near the top of the core (youngest sediments). All samples were analyzed for grain size, bulk density, organic content, and the presence of natural occurring radioisotopes ^{7}Be (half-life 53.3 days) or ^{210}Pb (half-life: 22.3 years).

To analyze grain-size, wet sediment was disaggregated in an ultrasonic bath for 15 minutes with 100mL of 0.05% sodium metaphosphate solution. Sediment was then wet-sieved at

64 μm to separate the sand (>64 μm) and mud (silts and clays; <64 μm) fractions, after which the mud and sand fractions were dried at 60°C until sediment weight was constant. Bulk density was calculated from water loss after drying samples, assuming sediment density of 2.65 g cm^{-3} and that bulk density was a function of porosity. The sand component was dry sieved through a standard set of 13 sieves, from 500 to 64 μm (1/4 phi size intervals; phi=-$\log_2$ (particle diameter, mm)). Organic content was calculated via loss on ignition after combusting samples at 450°C for 4 hours.

Seasonal-scale deposition rates in the created marshes of the living shorelines, and in the subtidal adjacent to living and reference shorelines were calculated from ^{7}Be activities. ^{7}Be (53.3-day half-life) is a naturally occurring radioisotope produced in the atmosphere by the cosmic ray spallation of nitrogen and oxygen (Olsen et al, 1986). ^{7}Be is deposited onto the Earth's surface by particle dry deposition and/or by precipitation, where it attaches to sediment particles before being eroded and delivered to adjacent waters and settling on the bottom sediment surface (Olsen et al, 1986). Due to its relatively short half-life, ^{7}Be serves as a marker for recent sediment deposition and has been used in previous studies in dynamic estuarine environments to decipher spatial and temporal (seasonal) patterns (Dibbs and Rice, 1989; Neubauer et al, 2002; Russ and Palinkas, 2018).

To analyze ^{7}Be, bulk sediment from the 1-cm increments of the push-cores, was ground and put into 60-mL plastic jars, starting with the topmost 1 cm (0-1cm section of core). ^{7}Be activity (dpm g^{-1}; disintegrations per minute per gram) was measured through gamma spectroscopy of the 477.7 keV photopeak using a calibrated Canberra germanium detector and following the procedure of Palinkas et al. (2005). Gamma emissions were counted over a 24-hour period. For every core analyzed, gamma analysis continued for every 1-cm interval down the core until ^{7}Be activity was no longer detected. Measured ^{7}Be activities were decay-corrected to the time of core collection and used to calculate depth-integrated ^{7}Be inventories (I_{total}; dpm cm^{-2})

$$I_{total} = \sum A_i \times p_i \times h_i$$

where A_i is the activity (dpm g^{-1} in interval I), p_i is the bulk density (g cm^{-3}) in interval i, and h_i is the thickness (cm) of interval i. (Russ and Palinkas, 2018).

Decadal-scale accumulation rates were calculated from ^{210}Pb (half-life 22.3 years) activities, which were analyzed using alpha spectroscopy, following the procedures of Palinkas and Engelhardt (2016). ^{210}Pb is naturally derived from the decay of ^{238}U decay and as consequence, all sediments have "supported" ^{210}Pb activity from the decay of its effective parent ^{226}Ra. Particles that have been transported into aqueous environments during erosive processes can pick up "excess" ^{210}Pb from the water column, supplied from the atmosphere (gaseous ^{222}Rn escapes to the atmosphere, decays to ^{210}Pb and is scavenged by precipitation) and direct runoff (Szmytkiewicz & Zalewska, 2014). Excess ^{210}Pb activities were obtained by subtracting supported ^{210}Pb activity from the measured total ^{210}Pb activities. Supported ^{210}Pb activity was obtained from the bottom of the vibracore for each site. All activities were decay-corrected to the time of collection; excess activities were normalized to the corresponding measured mud fraction, because ^{210}Pb preferentially adsorbs to fine particles (Nittrouer et al. 1979; Goodbred and Kuehl, 1998).

Sedimentation rates were calculated using the Constant Initial Concentration (CIC) Age-Depth model (Appleby and Oldfield, 1978). The CIC model assumes that there is a constant supply of unsupported ^{210}Pb inventory while allowing for time-variable sedimentation rates. In the CIC model, sediment age at depth z (t_z) is calculated by

$$t_z = 1/\lambda_{210Pb} \ln(A_o/A(z))$$

where λ_{210Pb} is the ^{210}Pb decay constant (0.031 year $^{-1}$), A_o is the cumulative inventory of excess ^{210}Pb activity in the sediment column and the $A(z)$ is the cumulative inventory of excess ^{210}Pb activity below depth z. ^{210}Pb inventory for each interval is calculated by multiplying excess ^{210}Pb activity by the bulk density and thickness of that interval. The sedimentation rate is determined by dividing sediment depth by its calculated age.

Mass accumulation rates (g cm^{-2} y^{-1}) were used instead of accretion rates (cm y^{-1}) as sediment autocompaction could introduce large error margins into accretion rate calculations, especially over a ~100 yr period. Autocompaction refers to the process by which recently deposited sediment gets compacted over time, increasing in bulk density as additional sediment gets deposited over it (Bird et al, 2004).

Statistical Tests

We conducted statistical analyses on the data obtained from our push-core and vibracore samples we used R statistical software (RStudio 1.2.5042). For push-cores, t-tests were used to test for differences in means between related variables (mud content, organic content, ^{7}Be inventory) were used. T-tests were also used to test for differences in mud content, organic content, and mass accumulation rates before and after the living shorelines were installed. For this study, if $p<0.10$ then it was considered to be statistically significant. We decided to use a p-value threshold of 0.10 due to the inherent variability of geological data, which can make it more difficult to ascertain statistical significance.

1.3 Results and Discussion

Seasonal-Scale Sedimentation

Marsh Push-cores

In the created marshes of the living shorelines, mud content of surficial sediment (top 1 cm) averaged 41.6 ± 21.6%, with a range in values from 18.4% (Oppenheim) to 62.7% (Ruesch) (see Table 1a). There was high variability in mud content and sediment characteristics between different selected sites. Organic content averaged 14.5 ± 5.7% (see Table 1a), with a range in values from 1.7% (Maritime Museum) to 24.6% (Myrtle Grove) (see Fig. 3a). ^{7}Be inventory (dpm cm^{-2}) averaged 2.35 ± 1.69 dpm cm^{-2} and ranged from 1.02 dpm cm^{-2} (Maritime Museum) to 6.16 dpm cm^{-2} (Queens Landing) (see Table 1a). Corresponding deposition rates calculated from these inventories had a mean of 0.87 ± 0.67 g cm^{-2} y^{-1}; the deposition rate at Queens Landing was 2.43 g cm^{-2} y^{-1}, more than twice as high as the next closest value (Myrtle Grove: 1.16 g cm^{-2} y^{-1}) (see Table 1a).

Mud content in the subtidal adjacent to the living shorelines was highly variable, with a mean of 35.08 ± 30.35% (see Table 1b). Average organic content in front of the living shorelines was 4.9 ± 3.2%. Organic content was highest at Ruesch 11% and lowest at Oppenheim 1.84%.

^{7}Be inventory (dpm cm^{-2}) averaged 1.16 ± 1.49 dpm cm^{-2} in front of living shoreline sites, with an average deposition rate of 0.42 ± 0.48 g cm^{-2} y^{-1} (see Table 1b).

Mud content in front of reference shoreline sites had a mean of 52.33 ± 32.62%. Mud content in the adjacent subtidal is higher at reference shoreline sites than it was at living shoreline sites (see Table 1b). Organic content had a mean of 4.5 ± 5.2% at reference shoreline sites, reaching a maximum at Maritime Museum (16.0%). ^{7}Be inventory averaged 2.37 ± 2.93 dpm cm^{-2} with a mean deposition rate of 0.73 ± 1.03 g cm^{-2} y^{-1}at reference shoreline sites. Mud (p=0.29) and organic content (p=0.22) were similar between living shoreline and reference shoreline sites. However, deposition rates were higher at reference shorelines
There was no statistically significant difference between living shoreline and reference shoreline sites for mud content (p=0.29) and organic content (p=0.22). However, there was a statistically significant relationship found when comparing trends in deposition rates (p=0.017).

We also looked at statistical significance between marsh push-cores taken at living shorelines that had SAV pre-installation and those that did not to assess the impact of SAV presence on sediment characteristics in the living shorelines. We did not find any statistically significant differences for mud content (p=0.95), organic content (p=0.32), or deposition rates (p=0.41) between the sites that had SAV and those that did not.

Decadal-Scale Sedimentation

^{210}Pb Activity

Differences in down-core profiles of ^{210}Pb activities (normalized to mud content) varied among sites, indicating differences in decadal patterns of sedimentation. For example, the profile for Hatton Garden reflects steady-state accumulation, with excess activities generally decreasing logarithmically down-core and reaching a relatively constant, low ^{210}Pb activity reflecting supported activity. In contrast, the ^{210}Pb profile for Ruesch peaked in the middle of core (between 30-54 cm core-depth), with variable activities throughout the core indicating non steady-state accumulation. Activities were used in the CIC model (see Methods) to calculate sediment ages, and therefore also rates, for discrete depth horizons throughout the cores. This model was used to calculate the depth horizon corresponding to the installation year for each living shoreline site; core data were then separated into those prior (below this horizon) and after (above this horizon) installation.

In the subtidal adjacent to the living shorelines, mud content averaged 34.7 ± 12.8% at before installation, mud content averaged 43.5 ± 6.4% after installation (see Fig. 6a) ($p=0.59$) In the subtidal adjacent to the reference shorelines, mean mud content for the pre-installation period was 32.6 ± 12.1% mud content was 43.4 ± 8.3% post-installation (see Fig. 6b) (Table 2). Average organic content was calculated to be 2.74 ± 1.41% at living shoreline sites pre-installation and after installation organic content increased at living shoreline sites to 2.87 ± 0.58% (see Fig. 7a) ($p=0.87$). Pre-installation mean organic content was 2.46 ± 0.85% at reference shoreline sites, post-installation average organic content increased to 3.21 ± 1.12% (see

Fig. 7b) ($p=0.60$) (see Table 3). Mass accumulation rates (MAR) (g cm^{-2} y^{-1}) increased at living shoreline sites after installation from 0.89 ± 0.40 g cm^{-2} y^{-1} to 1.06 ± 0.55 g cm^{-2} y^{-1} (see Fig. 8a) ($p=0.64$) (see Table 4). At the reference shorelines, MAR increased from 0.79 ± 0.25 g cm^{-2} $y^{-1}$to 1.01 ± 0.63 g cm^{-2} y^{-1} following the installation of the corresponding nearby living shoreline (see Fig. 8b) ($p=0.66$) (see Table 4).

There were more pronounced differences in observed trends at individual sites, that are not readily apparent when looking at averages for the sites. Queens Landing, for instance, experienced a dramatic increase in mud content in the subtidal adjacent to the LS following installation, increasing from $19.1 \pm 11.2\%$ to $65.9 \pm 7.4\%$ ($p<0.001$). Another example is in the subtidal adjacent to the reference shoreline for Ruesch where mud content increased from $29.1 \pm 27.0\%$ to $91.4 \pm 1.5\%$ ($p=0.06$) (see Table 2).

At living shoreline sites, changes in mud content were statistically significant at every site except for San Domingo and Environmental Concern, which did not have enough post-installation data to perform statistically significant tests. At reference shoreline sites there were statistically significant relationships at 6 out of 8 sites. Differences in mud content at Queens Landing was not statistically significant, and we did not have vibracore data to run tests for the reference shoreline site at Environmental Concern.

Statistical differences in organic content before/after installation at living shoreline and reference shoreline sites led to an expectation of similar statistical before/after trends in mud content and organic content. However, this was not the case, for living shoreline sites statistically significant differences were observed at 3 out of the 8 sites and the same was true at reference shoreline sites.

Testing for statistical significance for trends in MAR at living shoreline and reference shoreline sites in the adjacent subtidal before/after installation, we did not find many statistically significant differences between sites. There were statistically significant differences observed at Ruesch ($p=0.02$), Hatton Garden ($p=0.08$), Myrtle Grove ($p=0.07$), and Maritime Museum ($p=0.005$) living shoreline sites but not at the other living shoreline sites. At reference shoreline sites, there were statistically significant relationships found at 3 out of the 8 sites; Oppenheim ($p=0.005$), Myrtle Grove ($p=0.05$), and Maritime Museum ($p=0.10$). Myrtle Grove and Maritime Museum were the only sites that had statistically significant relationships for MAR trends at both living shoreline and reference shoreline sites.

SAV Presence Before/After Installation

SAV presence varied throughout the years after installation at all sites (Table 5). "SAV present" after installation means that a site had SAV for more than half of the years post-installation. Sites that did not have SAV before installation also did not have SAV after installation. Oppenheim and Ruesch, had SAV before and after installation; however Hatton Garden and Queens Landing while they had SAV present prior to installation but only had SAV for 1 and 2 years, respectively, after installation and so these sites were considered as SAV absent.

To gauge the effect of SAV presence on sedimentary characteristics within the living shorelines, we also compared averages (mud content, organic content, and deposition rates) between sites that had SAV pre-installation and those that did not. At sites that had SAV, mean mud content was 41.2 ± 20.1%, mean mud content was 42.2 ± 26.9% at sites where SAV was absent (see Fig. 2a). Organic content averaged 11.2 ± 4.9% at SAV sites and 17.87 ± 10.9% at sites without SAV (see Fig 3a). Deposition rates had a mean of 1.1 ± 0.9 g cm^{-2} y^{-1} at SAV sites and 0.6 ± 0.3 g cm^{-2} y^{-1} (see Fig. 4a) at sites without SAV pre-installation. Average mud content was comparable between sites regardless of SAV presence/absence, whereas organic content was approximately 6% percent greater at sites without SAV. Deposition rates were noticeably higher at sites that had SAV pre-installation, deposition rates at SAV sites were nearly twice the deposition rates at sites without SAV.

Discussion

Seasonal-scale observations in the subtidal adjacent to living and reference shorelines show significant differences in mud content and deposition rates. Sediments in the subtidal adjacent to reference shorelines had a considerably higher mean mud content 52.3 ± 32.6% and deposition rates 0.73 ± 1.03 g cm^{-2} y^{-1} than the subtidal zone adjacent to living shoreline sites (35.1 ± 30.4%; 0.42 ± 0.48 g cm^{-2} y^{-1}) (see Table 1b). This could indicate that the installation of the living shorelines is helping improve sediment conditions for SAV in the adjacent subtidal by trapping fine sediment. Alternatively these differences could reflect differences in sedimentary characteristics that preceded living shoreline installation. SAV prefers sandy sediment, which can be a limiting factor if sand supply is constrained (Palinkas & Koch, 2012). Trapping of fine sediment and runoff filtration by the created marshes can also help local water clarity by reducing turbidity in the water column (Pace & Morgan, 2017; O'Donnell, 2016; Morgan et al, 2009).

Mean mud content was comparable between living shoreline and reference shorelines sites starting at an average of 34.7 ± 12.8% then increasing to 43.5 ± 6.45% ($p=0.59$) at living shoreline sites after installation. At reference shorelines mean mud content pre-installation was 32.6 ± 12.1% before increasing an average of 43.5 ± 6.45% post-installation ($p=0.52$) (see Table 2). Queens Landing was an outlier for mud content trends at living shoreline sites, mud content increased from 19.1 ± 11.2% to 65.9 ± 7.4% post installation ($p<0.001$). Analyzing differences in organic content before and after living shoreline installation between living shoreline and reference shoreline sites, there were distinct differences between sites. Reference shorelines and living shoreline sites both experienced net increases in organic content following installation; increasing from 2.4 ± 0.8% to 3.2 ± 1.1% ($p=0.60$) at reference shorelines post installation while increasing slightly from 2.7 ± 1.4% to 2.8 ± 0.5% ($p=0.87$) at living shoreline sites post installation (see Table 3). Referencing a number of studies, persistent SAV beds generally thrive when organic content is <5%, so we also looked at organic content trends specifically within the sites that had SAV pre-living shoreline installation (Wicks et al, 2009; Palinkas & Koch, 2012). While organic content rich sediment connotes nutrient rich sediment that aids the growth of SAV, sediments with high organic content and mud content posed a greater risk for SAV survival due to the increased chance of dislodgement as well as the potential for the accumulation of sulfides (Wicks et al, 2009). Queens Landing is the only site that had SAV present prior to installation but lost its subtidal SAV beds following living shoreline installation. Organic content increased 2%, 1.3 ± 0.4% to 3.3 ± 1.6% ($p=0.002$) and mud content increased by

46.8%, 19.1 ± 11.2% to 65.9 ± 7.4% ($p<0.001$) at Queens Landing after the living shoreline was installed, this sharp shift in sediment properties in the nearshore likely caused the SAV beds to be buried, dislodged, or subject to low levels of light availability due to potentially turbid conditions (Cabaço et al, 2008). This dramatic increase in mud and organic content at Queens Landing could be driven by nearby construction, as a new hotel was constructed in close proximity to the site and produced a sandbar (seen on VIMS SAV Map) near the outlet into the Bay which likely impacted sediment input. We do not know the exact cause of this shift in mud and organic content at Queens Landings.

In terms of observed subtidal mass accumulation rates (MAR) there were some interesting trends at living shoreline and reference shoreline sites. Living shoreline and reference shoreline sites had nearly identical trends in MAR increasing from 0.89 ± 0.40 g cm^{-2} y^{-1} to 1.06 ± 0.55 g cm^{-2} y^{-1} ($p=0.64$) and 0.79 ± 0.25 g cm^{-2} y^{-1} to 1.01 ± 0.63 g cm^{-2} y^{-1} ($p=0.66$), before/after living shoreline installation, respectively. Initially we hypothesized that sediment deposition would be higher at living shoreline sites when compared to reference shoreline sites because of the added sediment trapping capacity conferred by the living shorelines.

Across our study sites, MAR increased after the living shorelines were installed at both living shoreline and reference shoreline sites, contrary to our hypothesis. However, for environmental manager and restoration practitioners, the lack of significant differences in MAR between living shoreline and reference shoreline sites when averaged together suggests that installing the living shorelines will not drastically change subtidal sedimentary environments.

This is an important finding as it suggests that the installation of living shorelines may not change sedimentary characteristics in a way that impacts SAV presence/absence.

The effectiveness of living shorelines can be influenced by SAV abundance in nearshore habitat, as SAV beds have comparable ecosystem services (wave attenuation, nutrient/sediment sequestration, etc.), therefore historical trends of SAV abundance in the Chesapeake Bay should offer further insight into the trends observed in decadal scale sedimentation. Referencing SAV abundance studies at Virginia Institute of Marine Science (VIMS), our selected study sites are primarily in the lower Choptank region (Orth et al, 2010). From 1960-1983, SAV abundance throughout the Chesapeake Bay fluctuated due to declining water quality (Orth & Moore, 1984). SAV abundance in the Chesapeake Bay did not start increasing again until the 1990s following a series of improvements in wastewater management in the surrounding region to help decrease nutrient and sediment inputs to the Bay (Gurbisz et al, 2014). In the lower Choptank region, SAV abundance increased by nearly 20 km^2 peaking at nearly 28 km^2 in 1997, before experiencing a steep decrease and balancing out around 27 km^2 in 2002 (Orth et al, 2010). SAV coverage in the Chesapeake Bay overall ranges approximately from 3-10% of historical SAV coverage (Orth and Moore, 1984), as SAV beds continue to be adversely affected by salinity fluctuations, sediment loading, and sediment resuspension (Arnold et al, 2000). The dominant control on SAV abundance above all is the resuspension of sediment and nutrients as it negatively impacts water clarity and thus interferes with photosynthetic processes necessary for growth and survival (Orth et al, 2010; Cabaço et al, 2007; Boer et al, 2007; Golden et al, 2010).

While living shorelines remain the preferred feature for coastal protection, the presence of SAV in adjacent nearshore habitat could compliment the sediment trapping capacity of the created marshes to reduce local turbidity and improve water clarity (Bilkovic et al, 2016). At reference shoreline sites where SAV was present prior to the installation of a nearshore living shoreline, MAR remained high and SAV beds had a strong influence on sediment accretion rates. The presence of living shorelines can improve nearshore conditions for SAV by acting as a nutrient filter for terrestrial runoff as well, creating positive feedbacks between the SAV and the living shoreline (Gittman et al, 2016; Morgan et al, 2009).

More vibracore samples, over a longer study period, are needed to determine whether these observed differences might be related to living shoreline installation or whether they reflect historical differences in local sediment dynamics. Differences at specific sites are captured in vibracore data. When averaged together, there were no statistical differences pre- vs. post-installation in mud content, organic content, or accumulation rates adjacent to the living shorelines or the reference shorelines. Although, there were differences at individual sites. The majority (13 out of 16) of the study sites exhibited statistically significant relationships for mud content following living shoreline installation. For MAR and organic content, less than half of our study sites showed statistically significant differences for these variables.

1.4 Summary/Synthesis

Going into this study, we expected that there be distinct differences in the nearshore sedimentary environment of living shorelines and reference shorelines. Contrary to our hypotheses in regard to trends in mud content, organic content, and mass accumulation rates, trends for all the aforementioned variables were similar at both living shoreline and reference shoreline sites when taken on average. While mud content and mass accumulation rates both increased after the living shoreline was installed, they increased at nearly the same rate at both living shoreline and reference shoreline sites. This would suggest that either the living shoreline did not have a significant impact on the nearshore sedimentary environment or that the installation of the living shoreline affected trends at both living shoreline and reference shoreline sites. It could be possible that these increases in mud content and MAR following the installation are more reflective of local trends in sediment supply at individual sites than the influence of the living shorelines themselves. However, more vibracore samples are needed to determine whether these observed differences might be related to living shoreline installation or whether they reflect historical differences in local sediment dynamics. Differences at specific sites are captured in vibracore data, when averaged together for comparison, there were no statistical differences pre- vs. post-installation in mud content, organic content, or accumulation rates adjacent to the living shorelines or the reference shorelines.

The installation of the living shorelines did not affect SAV presence/absence, SAV remained present at 3 out of the 4 sites that had SAV prior to installation. However, the one site that lost SAV coverage, Queens Landing, likely lost its SAV beds due to reasons unrelated to the installation of the living shoreline.

In terms of broad implications for the Chesapeake Bay and coastal managers, our study did not find any evidence suggesting that the installation of living shorelines affects SAV presence/absence. Further research is needed to assess the impacts of living shorelines on adjacent subtidal habitat to determine if the shifts in sedimentary characteristics are more related to bay-wide shifts in sediment supply as opposed to the installed living shorelines. More research is also recommended to assess whether or not the presence of SAV is beneficial to living shorelines in terms of sediment and nutrient deposition and rates of accretion vs. rates of erosion.

1.5 Figures and Tables

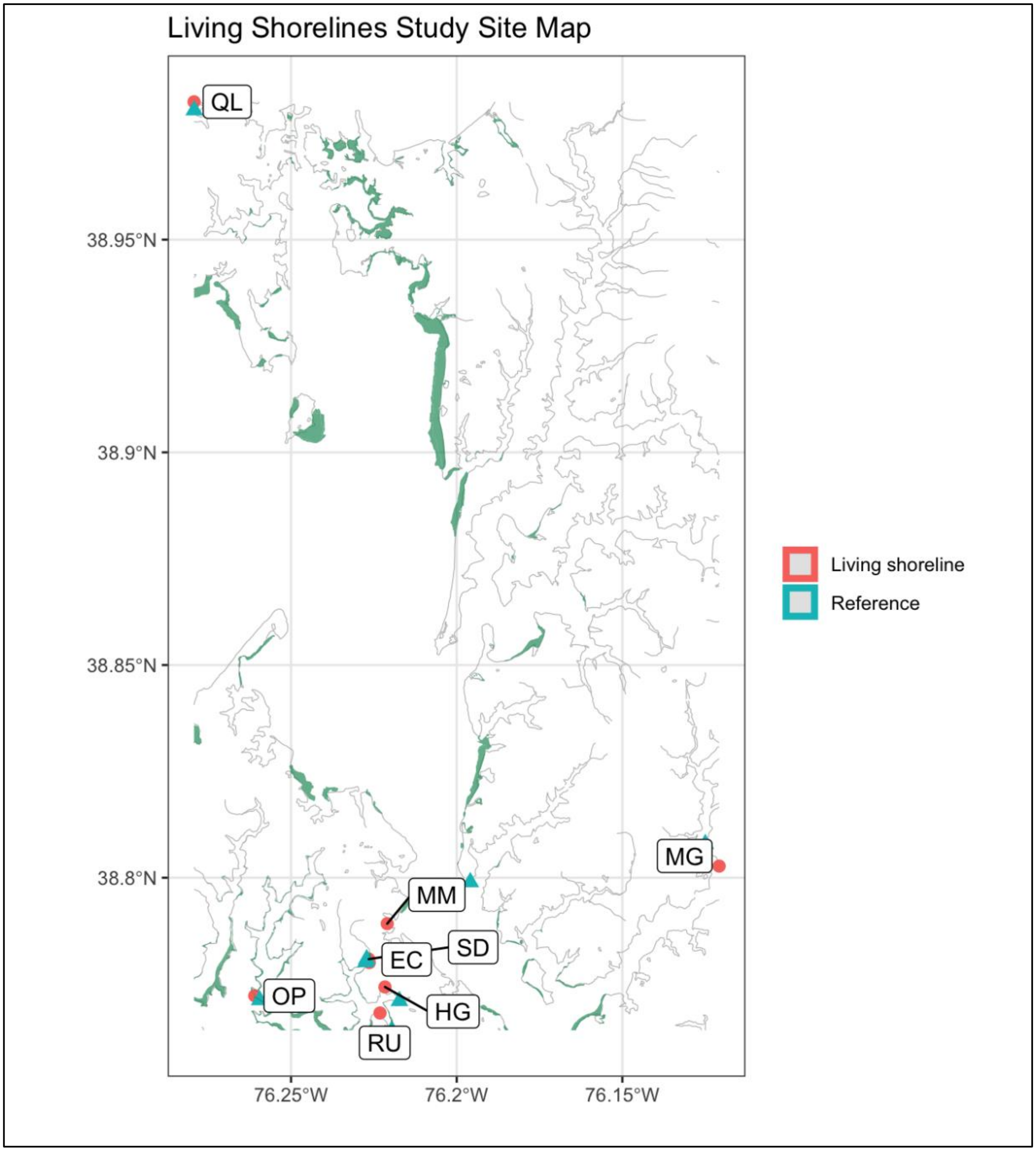
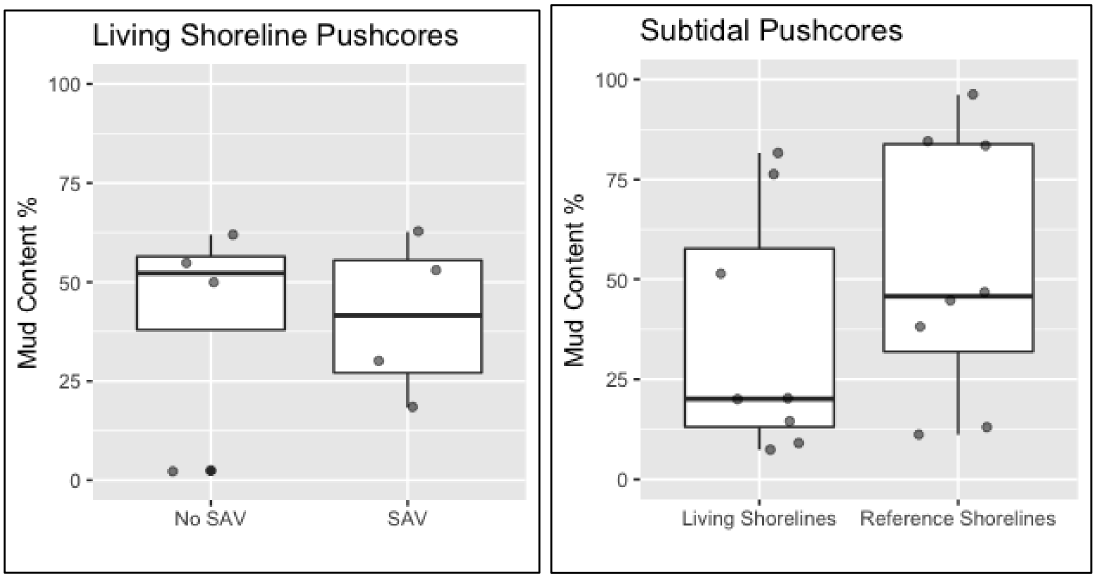
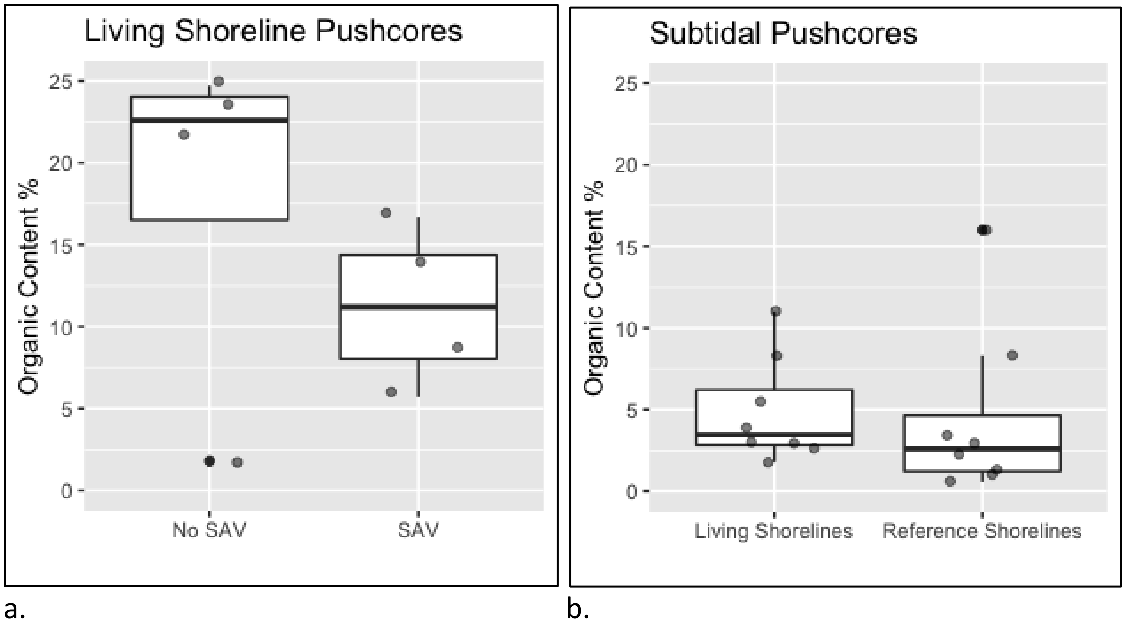


Figure 1: Study site map. The selected sites are located within the Choptank River basin and the Miles River basin in Chesapeake Bay, MD.

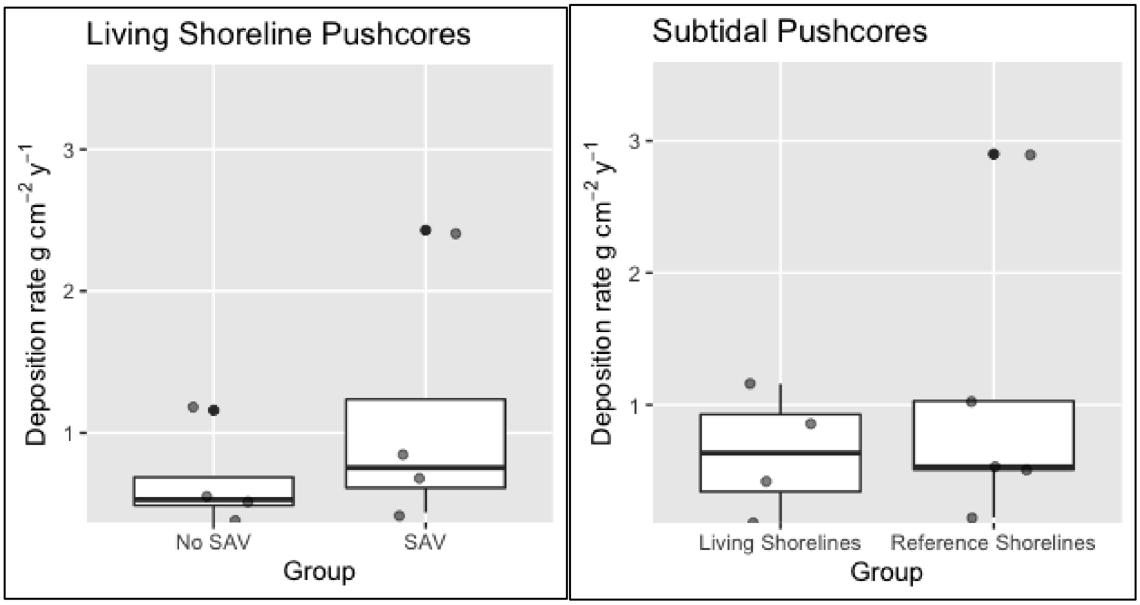


a. b.

Figures 2a-b: Average mud content (%) within the living shorelines (SAV/No SAV) and in the adjacent subtidal in front of living shorelines/reference shorelines.

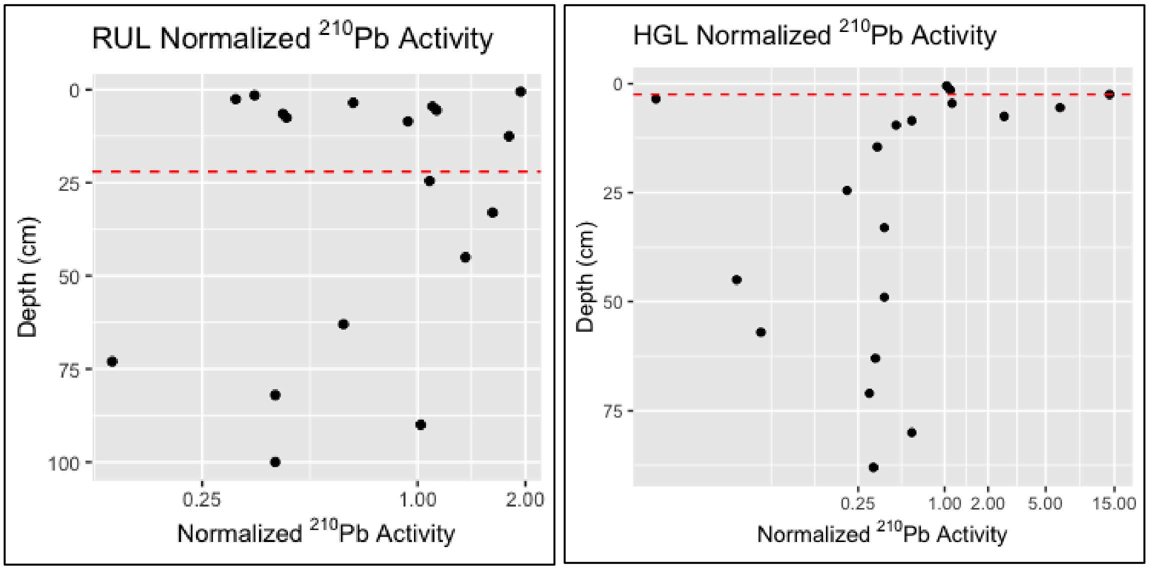


Figures 3a-b: Average organic content (%) within the living shorelines (SAV/No SAV) and in the adjacent subtidal in front of living shorelines/reference shorelines.

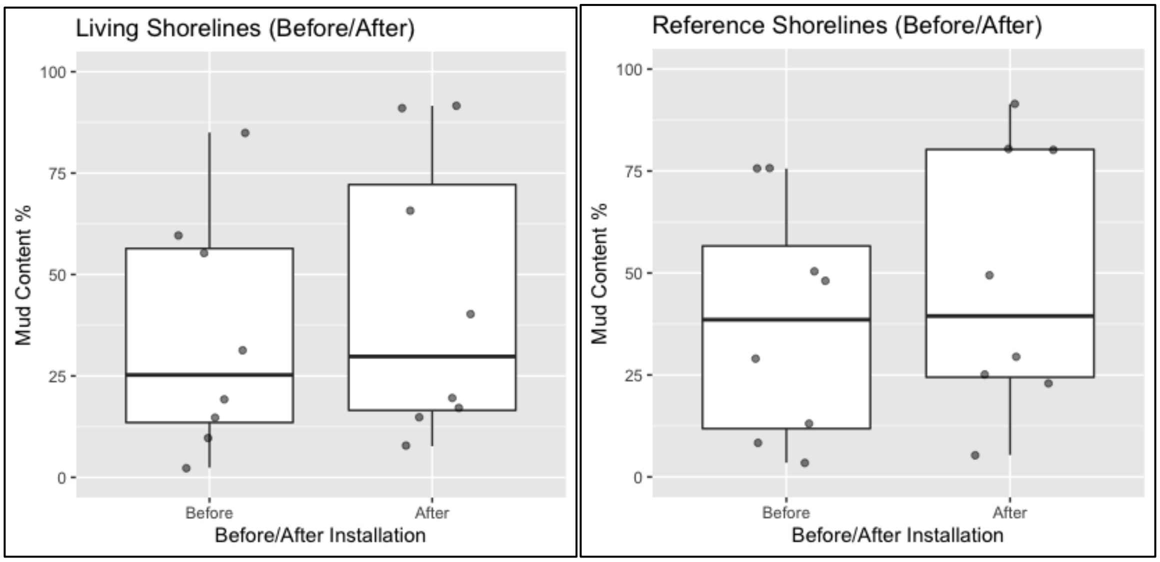


a. b.

Figures 4a-b: Average deposition rates (g cm^{-2} y^{-1}) within the living shorelines (SAV/No SAV) and in the adjacent subtidal (Living Shorelines/Reference Shorelines) (top 0-1 cm of pushcores).

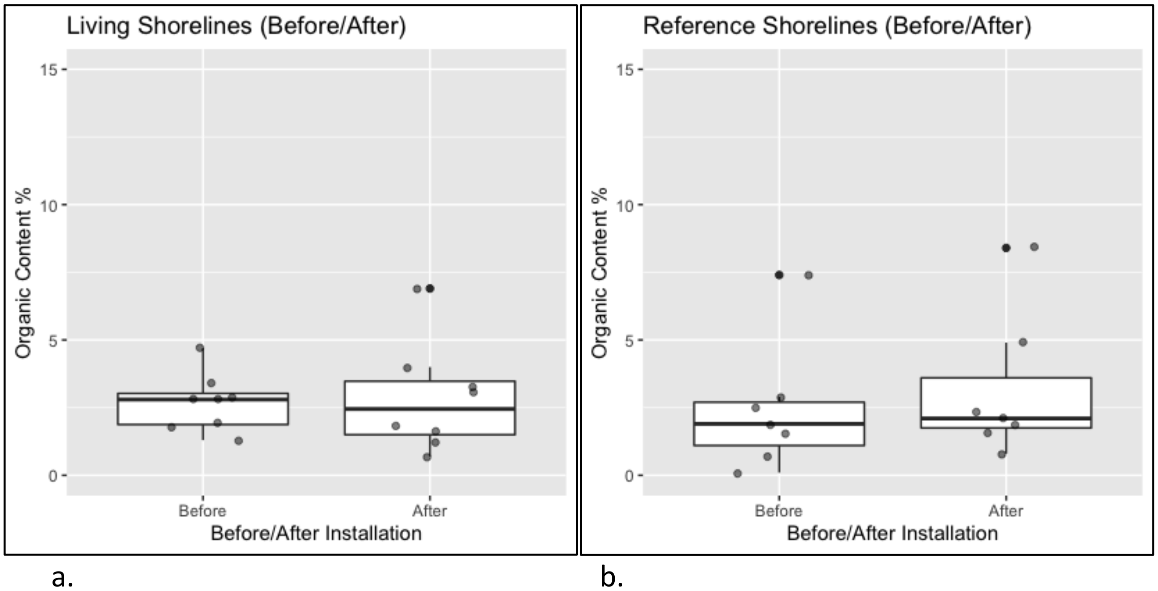


Figures 5a-b: HG (Hatton Garden) exhibits the traditional ^{210}Pb profile, with the excess activity concentrated in the upper portion of the vibracore, whereas RU's (Ruesch) ^{210}Pb profile shows excess activity down core. The dashed line indicates the approximate depth where the living shoreline was installed.

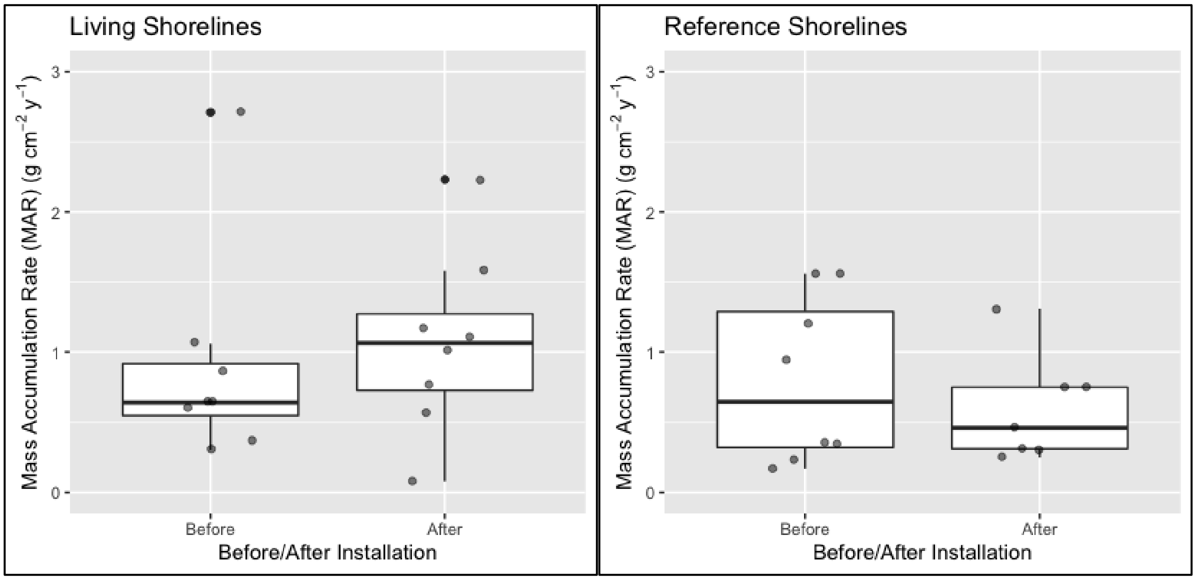


a. b.

Figures 6a-b: Box and whisker plots displaying average mud content at living shoreline and reference shoreline sites before and after the living shoreline was installed.



Figures 7a-b: Box and whisker plots displaying average organic content at living shoreline and reference shoreline sites before and after the living shoreline was installed.



Figures 8a-b: Box and whisker plots displaying average mass accumulation rates (g cm^{-2} y^{-1}) at living shoreline and reference shoreline sites before and after the living shoreline was installed.

Tables

Table 1a. Sediment characteristics for surficial sediments (topmost centimeter) of push-cores collected in the created marshes of living shorelines.

Site	Installation Year	Pre-SAV (Yes/No)	^{7}Be Inventory	Deposition Rate (g cm^{-2} y^{-1})	Mud %	Organic Content
Queens Landing (QL)	2005	Yes	6.16	2.43	30.0 ± 22.4	8.9 ± 1.4
Oppenheim (OP)	2006	Yes	1.85	0.67	18.4 ± 21.1	5.7 ± 3.3
Rusech (RUL)	2008	Yes	2.29	0.84	62.7 ± 25.2	13.6 ± 6.9
Hatton Garden (HG)	2007	Yes	1.79	0.44	53.1 ± 42.3	16.7 ± 12.8
Myrtle Grove (MG)	2004	No	3.17	1.16	49.8 ± 36.1	21.4 ± 7.7
Maritime Museum (CBMM)	2008	No	1.02	0.37	2.4 ± 1.8	1.7 ± 0.07
Environmental Concern (EC)	2005	No	1.07	0.53	61.9 ± 15.2	23.8 ± 9.5
San Domingo (SDL)	2007	No	1.45	0.53	54.6 ± 8.4	24.6 ± 5.8
Averages			2.35 ± 1.69	0.87 ± 0.67	41.6 ± 21.6	14.5 ± 1.4

Table 1b. Calculated, ^{7}Be Inventory, deposition rate, mud content and organic content for subtidal push-cores.
N/A-Not available
ND- Not detected, there may be trace amounts of ^{7}Be but it is below the detection limit.

Site	^{7}Be Inventory (dpm cm^{-2})	Deposition Rate (g cm^{-2} y^{-1})	Mud %	Organic Content %
Living Shorelines				
Queens Landing	3.18	1.16	14.6	3.9
Oppenheim	N/A	N/A	20.0	1.8
Rusech	1.14	0.42	20.2	11.0
Hatton Garden	ND	ND	51.6	5.5
Myrtle Grove	2.34	0.85	7.4	2.6
Maritime Museum	N/A	N/A	9.0	2.9
Environmental Concern	ND	ND	81.6	3.0
San Domingo	0.30	0.11	76.3	8.3
Averages	1.16 ± 1.49	0.42 ± 0.48	35.1 ± 30.4	4.9 ± 3.2
Reference Shorelines				
Queens Landing	8.02	2.92	46.8	2.3
Oppenheim	N/A	N/A	13.2	0.6
Rusech	ND	ND	38.2	8.3
Hatton Garden	0.41	0.15	96.2	3.4
Myrtle Grove	2.82	1.03	44.9	2.9
Maritime Museum	N/A	N/A	11.3	16
Environmental Concern	1.57	0.53	83.5	1.0
San Domingo	1.4	0.51	84.7	1.3
Averages	2.37 ± 2.93	0.73 ± 1.03	52.3 ± 32.6	4.5 ± 5.2

Table 2. Mud content in the adjacent subtidal at living shoreline and reference shorelines (control) sites, before/after living shoreline installation.
*assumed similar to San Domingo reference site due to the close proximity to the Environmental Concern reference shoreline.
**2004-2018 SAV abundance data was referenced from the VIMS Interactive SAV Map. (https://www.vims.edu/research/units/programs/sav/access/maps/index.php).

Site	Year of Installation	Pre-SAV (Post-SAV)**	Mud % Pre-Installation at Living Shorelines	Mud % Post-Installation at Living Shorelines	Mud % Pre-Installation at Reference Shorelines	Mud % Post-Installation at Reference Shorelines
Queens Landing	2005	Yes (No)	19.1 ± 11.2	65.9 ± 7.4	50.3 ± 22.5	49.4 ± 8.9
Oppenheim	2006	Yes (Yes)	9.7 ± 2.9	14.8 ± 1.0	13.0 ± 11.2	5.4 ± 0.8
Ruesch	2008	Yes (Yes)	31.4 ± 15.9	19.5 ± 5.8	29.1 ± 27.0	91.4 ± 1.5
Hatton Garden	2007	Yes (Yes)	59.5 ± 20.4	40.1 ± 1.6	48.0 ± 24.0	22.8 ± 0.6
Myrtle Grove	2004	No (No)	2.4 ± 0.5	17.1 ± 18.9	3.5 ± 2.1	25.0 ± 17.4
Maritime Museum	2008	No (No)	14.8 ± 11.6	7.7 ± 13.9	8.4 ± 5.6	29.5 ± 33.2
Environmental Concern	2005	No (No)	55.4 ± 30.9	91.6 ± 2.8	75.6 ± 4.2*	80.3 ± 4.2*
San Domingo	2007	No (No)	85.0 ± 8.7	91.0 ± 0.0	75.6 ± 4.2	80.3 ± 4.2
Averages			34.7 ± 12.8	43.5 ± 6.4	32.6 ± 12.1	43.4 ± 8.3

Table 3. Organic content at living shoreline and reference shorelines (control) sites, before/after living shoreline installation.
*2004-2018 SAV abundance data was referenced from the VIMS Interactive SAV Map. (https://www.vims.edu/research/units/programs/sav/access/maps/index.php).

Site	Pre-SAV (Post-SAV)*	Organic Content (%) Pre-Installation (Living Shorelines)	Organic Content (%) Post-Installation (Living Shorelines)	Organic Content (%) Pre-Installation (Reference Shoreline)	Organic Content (%) Post-Installation (Reference Shoreline)
Queens Landing	Yes (No)	1.3 ± 0.4	3.3 ± 1.6	2.9 ± 1.9	2.1 ± 0.5
Oppenheim	Yes (Yes)	2.8 ± 1.7	1.8 ± 0.1	0.7 ± 0.4	0.8 ± 0.9
Ruesch	Yes (Yes)	2.8 ± 1.1	1.6 ± 0.7	1.9 ± 1.8	4.9 ± 0.3
Hatton Garden	Yes (Yes)	2.9 ± 0.5	3.1 ± 0.1	2.5 ± 0.8	1.9 ± 0.05
Myrtle Grove	No (No)	1.8 ± 2.7	1.2 ± 1.4	0.1 ± 0.03	1.6 ± 1.1
Maritime Museum	No (No)	1.9 ± 1.2	0.7 ± 0.3	1.5 ± 0.4	2.3 ± 3.5
Environmental Concern	No (No)	3.4 ± 2.5	4.0 ± 0.1	N/A	N/A
San Domingo	No (No)	4.7 ± 0.9	6.9 ± 0.0	7.4 ± 0.3	8.4 ± 1.2
AVERAGES		2.7 ± 1.4	2.8 ± 0.5	2.4 ± 0.8	3.2 ± 1.1

Table 4. Decadal-scale sedimentation. Mass accumulation rates (MAR) (g cm^{-2} y^{-1}) at living shoreline and reference shoreline sites.
*2018 SAV abundance data was referenced from the VIMS Interactive SAV Map. (https://www.vims.edu/research/units/programs/sav/access/maps/index.php).

Site	Install Year	Pre-SAV/ Post-SAV	MAR (g cm^{-2} y^{-1}) Pre-Installation (1900-Install Year) at Living Shorelines	MAR (g cm^{-2} y^{-1}) Post-Installation (Install Year-2018) at Living Shorelines	MAR (g cm^{-2} y^{-1}) Post-Installation (1900-Install Year) at Reference Shorelines	MAR (g cm^{-2} y^{-1}) Post-Installation (Install Year-2018) at Reference Shorelines
Queens Landing	2005	Yes (No)	2.71 ± 0.93	1.17 ± 0.85	1.20 ± 0.31	1.31 ± 0.46
Oppenheim	2006	Yes (Yes)	0.64 ± 0.35	1.12 ± 0.74	1.56 ± 0.46	0.75 ± 0.19
Ruesch	2008	Yes (Yes)	1.06 ± 0.80	2.23 ± 0.80	0.17 ± 0.12	0.25 ± 0.16
Hatton Garden	2007	Yes (Yes)	0.36 ± 0.26	0.57 ± 0.10	1.56 ± 0.32	0.75 ± 0.36
Myrtle Grove	2004	No (No)	0.64 ± 0.11	1.58 ± 1.22	0.94 ± 0.53	3.97 ± 3.57
Maritime Museum	2008	No (No)	0.61 ± 0.08	1.01 ± 0.34	0.23 ± 0.04	0.46 ± 0.26
Environmental Concern	2005	No (No)	0.87 ± 0.56	0.78 ± 0.39	0.35 ± 0.10*	0.31 ± 0.03*
San Domingo	2007	No (No)	0.30 ± 0.18	0.08 ± 0	0.35 ± 0.10*	0.31 ± 0.03
Averages/ Standard Deviation			0.89 ± 0.40	1.06 ± 0.55	0.79 ± 0.25	1.01 ± 0.63

SAV Table (Presence/Absence)

Table 5. SAV presence/absence at living shoreline and reference shorelines sites. SAV presence data were referenced from the VIMS Interactive SAV Map. (https://www.vims.edu/research/units/programs/sav/access/maps/index.php).

Site	SAV Presence at Living Shoreline (years present/total years post-installation)	SAV Presence at Reference Shoreline (years present/total years post-installation)
Queens Landing	2 out of 14	2 out of 14
Oppenheim	8 out of 13	8 out of 13
Ruesch	7 out of 11	7 out of 11
Hatton Garden	1 out of 11	4 out of 11
Myrtle Grove	0 out of 15	0 out of 15
Maritime Museum	0 out of 15	0 out of 11
Environmental Concern	0 out of 14	0 out of 14
San Domingo	0 out of 12	0 out of 12

Chapter 2: Evaluating seasonal trends in sediment/nutrient retention in patchy Submersed Aquatic Vegetation (SAV) beds of the upper Chesapeake Bay

2.1 Introduction

Subtidal vegetated habitats provide critical ecosystem services to estuarine ecosystems such as wave attenuation, enhanced sediment and nutrient burial, and critical nursery habitat that promotes the growth, survival, and reproduction of diverse taxa such as crustaceans, mollusks, polychaetes, and juvenile fish (Gittman et al, 2016; Duarte et al, 2013; Waycott et al, 2009; Heck Jr. et al, 2003). By stabilizing the sediment, SAV beds dampen the hydrodynamic energy of incoming waves subsequently increasing sediment deposition and decreasing erosion in both subtidal SAV habitat and at adjacent coastline (Moki et al, 2020; Boer, 2007). The effectiveness of shoreline protection from SAV is seasonal as there is significant dieback during the winter due to the seasonal nature of the growth cycles of SAV (Russ and Palinkas, 2018; Ganthy et al, 2013) Patterns of sediment and associated particulate nutrient deposition also vary seasonally with the presence/absence of plants, generally being greater when plants are present but also varying with bed geometry and sediment transport pathways (Russ and Palinkas, 2018; Bos et al, 2004). Even during the dieback period, erosion rates can be lower within SAV as compared to unvegetated area due to the presence of rhizomes in these habitats (Ganthy et al, 2011).

SAV beds are sensitive to high levels of turbidity and nitrogen inputs, and negative correlations between both parameters and SAV growth have been well documented in previous studies on SAV growth trends (Lefcheck et al, 2018; Orth et al, 2010; de Boer, 2007). Eutrophication is the most well documented cause for SAV decline, promoting blooms of phytoplankton and macroalgae, as well as indirect feedback mechanisms, that increase turbidity and decrease light availability (McGlathery et al 2007; Cabaço et al, 2008; Kemp et al, 2005;). Indirect feedbacks include sediment resuspension following SAV loss, increased system respiration, sediment anoxia, and internal nutrient loading from enhanced nutrient fluxes from sediment to water column (Burkholder et al, 2007).

Much recent research on Chesapeake Bay SAV beds has focuses on the Susquehanna Flats, which is located at the head of the Chesapeake Bay, and is the shallow, oligohaline, subaqueous delta of the Susquehanna River (the Bay's main tributary). Historically, the Flats has supported extensive beds of SAV ($50km^2$). However, the extent and abundance of SAV in the region has fluctuated significantly since the 1930s (Gurbisz and Kemp, 2014). The greatest loss of SAV habitat occurred after the impact of Hurricane Agnes in 1972, with the Susquehanna Flats experiencing some of the most significant losses of SAV in the entire Chesapeake Bay (Gurbisz et al, 2014). The loss of SAV in the Chesapeake Bay resulted in increased rates of shoreline erosion, decreases in migratory waterfowl populations, the collapse of the bay scallop fishery, and higher rates of suspended nutrients in the water column (Orth et al, 2006; Ganthy et al, 2011).

In more recent decades, Chesapeake Bay nitrogen concentrations have been decreased by 23% since 1984, resulting in a $170km^2$ increase in total SAV bed area (Lefcheck et al, 2018). Resurgence of SAV on the Flats has occurred more recently in the early 2000s, following a period of reduced rainfall and nitrogen reductions that have allowed it to recolonize its historical coverage (Gurbisz et al, 2014). With the resurgence of SAV on the Flats, comes the return of key ecosystem services to the region. For example, the regrowth of SAV provides a buffer for sediment and nutrient inputs to the Bay via retention (Biddle et al, in review; Russ and Palinkas, 2018). The Susquehanna Flats receives 1,000,000 tons of sediment from the Susquehanna River annually which accounts for the majority of the sediment that is delivered to the Chesapeake Bay (Donoghue & Bricker, 1989), but generally only 37% of this sediment is transported beyond the Susquehanna Flats (Donoghue & Bricker, 1989).

The Susquehanna River controls the circulation of sediment and freshwater on the Flats, transporting sediment and freshwater down bay (Shubel and Pritchard, 1986). The Susquehanna River supplies 87% of the freshwater down Bay, with an annual river discharge of 1100 $m^3\ s^{-1}$ (Donoghue & Bricker, 1989). River discharge has surpassed 20,000 $m^3 s^{-1}$ twice, since the United States Geological Survey gauging station was installed in 1967 (https://waterdata.usgs.gov/usa/nwis/uv?01578310) ~15 km upstream of the Flats in – during Hurricane Agnes in 1972 (~ 32,000 $m^3\ s^{-1}$) and Tropical Storm (TS) Lee in 2011 (~ 22,000 $m^3\ s^{-1}$) (Russ and Palinkas, 2018; Gross et al, 1978). Precipitation from TS Agnes resulted in both the highest recorded river discharge and the highest recorded sediment loads (30 x 10^6 t) to the Chesapeake Bay (Gross et al. 1978). TS Lee had the second highest recorded river discharge (~22,000 $m^3\ s^{-1}$), with an estimated sediment load of 6.7 to 19 x 10^6 t of sediment (Cheng et al, 2013). During years with more average river discharge rates, 50-60% of the annual sediment load delivered by the Susquehanna River to the Chesapeake Bay is transported during the spring freshet, when river discharge is highest. River discharge is generally at its lowest during the summer months (Gross et al. 1978).

Historically the Susquehanna Flats hosted extensive beds of SAV (>50 km^2); however there were significant declines in SAV habitat starting in the 1930s due to a myriad of factors including poor water quality, overgrazing, nutrient pollution and storm impacts there were significant declines in SAV habitat starting in the 1960s (Gurbisz and Kemp, 2014). Nitrogen loading from the Susquehanna River to the Chesapeake Bay increased by 250% between 1945-1984, making environmental conditions unsuitable for SAV growth due to high turbidity and low water clarity (Orth et al. 2009). The Susquehanna Flats experienced its greatest losses in SAV area following Tropical Storm Agnes in 1972 (Orth et al. 1984). Water quality in the Susquehanna Flats and the Chesapeake Bay overall started to recover after nitrogen loads began to decrease in the 1980s due to improvements in wastewater management and the banning of phosphorus in detergents in 1980 (Orth et al. 2009). However, it was not until the early 2000s that SAV began to recolonize the Flats reaching its previous range 55 km^2 in 2008. Following TS Lee in 2011, SAV declined to 25 km^2 in 2016 (Gurbisz et al, 2016) but has gradually recovered and reached >35 km^2 in 2016 (Orth et al, 2016). The SAV beds of the Susquehanna Flats are comprised of up to 13 plant species but are primarily composed of *Vallisneria americana* (wild celery), *Myriophyllum spicatum* (Eurasian watermilfoil), *Hydrilla verticillata* (water thyme), and *Heteranthera dubia* (water stargrass) (Gurbisz et al, 2014). Small patches of SAV (1-2 km^2) lie adjacent to the main bed, with even smaller (<1 km^2) narrow patches fringing the upper Bay and Susquehanna River shorelines. To our knowledge, sediment and nutrient deposition have not been studied in these patches.

Recent research has focused on spatial and temporal sedimentation patterns within and along the edges of the main SAV bed (>25km^2 in 2020;

https://www.vims.edu/research/units/programs/sav/access/maps/index.php) in the Susquehanna Flats (Russ and Palinkas, 2018; Gurbisz et al, 2016), however there hasn't been much research within emerging, patchier beds of SAV (<1 km^2) in the Flats. Along the edge of the main SAV bed, vegetation and channels exert influences on water flow and sediment transport, generally diverting sediment-rich fluvial waters into the main bed while reducing the rate of sedimentation upstream (Russ and Palinkas, 2018). However, vegetation can also divert these waters around the main bed, especially during high flows (Biddle et al, in review). The timing of sediment input relative to vegetation presence/absence may ultimately drive short-term differences in sediment retention. For example, in the main bed of the Flats, sedimentation rates can be similar for spring and summer even though sediment loads are highest in spring, because SAV is present to trap sediment only in the summer (Russ and Palinkas, 2018). The main bed of the Flats is much larger than most SAV beds in the Chesapeake and elsewhere (>25km^2) (Russ and Palinkas, 2018). It is unclear whether sediment and vegetation interact in similar ways in smaller, patchier beds that are more common elsewhere in the Bay. Differences in sedimentation patterns inside and outside of the SAV beds between sampling seasons in Russ and Palinkas (2018) study were reflected in differences in median grain size, median grain size, which was lower when plant biomass was high because SAV, when abundant, attenuate waves/currents and promote the settling of finer particles (Russ and Palinkas, 2018).

The purpose of this study is to (1) compare deposition rates and sediment composition inside and outside of small patches of SAV beds in the Susquehanna Flats, and (2) determine how patchy SAV beds affect local sedimentation patterns. We hypothesize that sediment deposition rates, nutrient concentrations, and mud content will be higher within the SAV beds, compared to outside of the SAV beds, in the summer when SAV beds have fully recovered from winter dieback. The SAV beds have enhanced sediment trapping capacity when aboveground biomass is high . We also hypothesize that sedimentary characteristics will be highly influenced by river discharge from the Susquehanna River, where deposition rates and mud content will increase with higher average river discharge.

2.2 Methods

Study Site

The Susquehanna Flats, the subaqueous the delta of the Susquehanna River is located at the head of the Chesapeake Bay and is characterized by sandy sediments that host extensive seasonal beds of submersed aquatic vegetation (SAV) (growing season June-October; Bayley et al. 1978; Gurbisz and Kemp 2014). The upper Chesapeake Bay is freshwater-oligohaline and has a salinity range of 0.5-5 ppt, with an average water depth within the Flats at mean lower low water (MLLW) of 1m, and an average tidal range of 0.6 m (Bayley et al. 1978; Orth & Moore, 1984).

Field Methods

Sites in the Susquehanna Flats were selected after analyzing aerial imagery and associated SAV spatial shapefiles acquired through the Chesapeake Bay Program SAV aerial monitoring program (http://web.vims.edu/bio/sav/maps) in the year prior to the study (2017). SAV beds are relatively persistent in the upper Bay, so data from the most recent year was assumed to be representative of SAV presence for site selection. For this study, 11 patchy SAV beds outside of the main bed were selected. Within each patch, one push core (30 cm long, 5 cm diameter) was collected inside and one just outside the vegetated area in spring and summer 2018 (6 June, 21 June, 5 September, and 6 September) and 2019 (5 May, 8 August, 22 August) to evaluate seasonal-scale sedimentation (Fig. 1). Spring samples were intended to capture sediment conditions in the absence of SAV, while summer samples were intended to capture conditions near peak biomass.

Laboratory Methods

Intact cores were returned to the laboratory and sectioned into 1-cm increments. For grain-size analyses, sediment was placed in an ultrasonic bath for 15 minutes with 100mL of 0.05% sodium metaphosphate solution in a beaker to disaggregate the mud fraction of the sediment. Sediment was then wet-sieved at 64μm to separate the sand (>64 μm) and mud (silts and clays; <64 μm) fractions. The mud and sand fractions were dried and bulk density was calculated from water loss after drying samples, assuming sediment density of 2.65 g cm^{-3} and that bulk density was a function of porosity. The sand component was dry sieved through a standard set of 13 sieves, from 500 to 64 μm (1/4 phi size intervals; phi=-$\log_2$ (particle diameter, mm)). Organic content was calculated via loss on ignition after combusting samples at 450°C for 4 hours. Nutrient analyses for carbon and nitrogen were carried out by Analytical Services at Horn Point Laboratory.

To determine deposition rates, samples were prepared and analyzed for ^{7}Be (53.3-day half-life), which is a naturally occurring radioisotope produced in the atmosphere by the cosmic ray spallation of nitrogen and oxygen (Olsen et al, 1986). ^{7}Be is deposited onto the Earth's surface by precipitation and by particle dry deposition (the removal of pollutants by sedimentation under gravity, diffusion processes, or by turbulent transfer resulting in impaction and interception) where it attaches to sediment particles before being eroded and delivered to adjacent waters and settling on the bottom sediment surface (Wilson et al, 2003; Olsen et al, 1986). Due to its relatively short half-life, ^{7}Be serves as a marker for recent sediment deposition and has been used in previous studies in dynamic estuarine environments to decipher spatial and temporal (seasonal) patterns (Olsen et al, 1986; Dibbs and Rice, 1989).

To analyze sediments for ^{7}Be, bulk sediment from the topmost 1-cm increments of the push cores was ground and put into 60-mL plastic jars. ^{7}Be activity (dpm g^{-1}; disintegrations per minute per gram) was measured through gamma spectroscopy of the 477.7 keV photopeak using a Canberra germanium detector and following the procedure of Palinkas et al. (2005). Gamma emissions were counted over a 24-hour period. For every core analyzed, gamma analysis continued for every 1-cm interval down the core until ^{7}Be activity was no longer detected. Measured ^{7}Be activities were decay-corrected to the time of core collection and used to calculate depth-integrated ^{7}Be inventories (I_{total}; dpm cm^{-2}) where A_i is the activity (dpm g^{-1}) in interval *I*, p_i is the bulk density (g cm^{-3}) in interval *i*, and h_i is the thickness (cm) of interval *i.* (Russ and Palinkas, 2018).

$$I_{total} = \sum A_i \times p_i \times h_i$$

Deposition rates were calculated following Neubauer et al. (2002):

$$S = \lambda_{Be}\left(\frac{I_{total} - I_{atm}}{A_{catch}}\right)$$

S is the calculated rate of sedimentation/deposition, λ_{Be} is the ^{7}Be decay constant, I_{total} represents total ^{7}Be inventory (dpm/cm^2), I_{atm} is local atmospherically derived inventory (dpm/cm^2); and A_{catch} serves as mean ^{7}Be inventory for catchment derived sediments (Neubauer et al, 2002).

River discharge data was sourced from Conowingo Dam sampling station data from the USGS website (https://waterdata.usgs.gov/md/nwis/uv?site_no=01578310).

Statistical Tests

We performed statistical tests using R statistical software (RStudio 1.2.5042). T-tests were used to test for statistically significant differences between observations taken within and outside of the SAV beds for mud content, ^{7}Be inventory, and sediment/nutrient deposition rates. For this study, if $p<0.10$ then it was deemed to be statistically significant. A p-value of 0.05 is normally used to determine statistical significance, but due to small sample size we opted to use a p-value threshold of 0.10.

2.3 Results and Discussion

Results are organized by the season and year of collection, since observations were strongly influenced by seasonal conditions (e.g. average and peak river discharge) (see Table 2).

Spring 2018

Average river discharge at Conowingo Dam was calculated to be 1423.88 ± 624.44 m^3 s^{-1}, with a peak of 4813 m^3 s^{-1} in Spring 2018 (see Table 2). Mud content, both inside and outside of the observed SAV beds, was low in Spring 2018. Inside of the SAV beds, mean mud content was 3.60 ± 1.52%, and outside of the SAV beds mean mud content was 3.66 ± 2.06 % (p=0.94) (see Fig. 2). Average sediment deposition rates in Spring 2018 were 0.12 ± 0.08 g cm^{-2} y^{-1} inside of the SAV beds and 0.19 ± 0.11 g cm^{-2} y^{-1} outside of the SAV beds ($p=0.29$) (see Fig. 4). Without the presence of SAV during the Spring, the sediment trapping capacity of the SAV beds recovering from winter dieback was reduced and less fine-grained material was deposited.

Summer 2018

In Summer 2018 the Chesapeake Bay watershed experienced uncharacteristically heavy rains and run-off (http://www.water.usgs.gov), which resulted in high river discharge and abnormally large sediment loads moving into the Bay. Average river discharge was recorded at 1733.22 ± 1933.08 m^3 s^{-1}, with a peak of 9995 m^3 s^{-1}. Average mud content inside the SAV beds was 30.82 ± 33.97% and outside the SAV beds average mud content was 13.92 ± 26.19% ($p=0.20$) (see Fig. 2). Mud content was much higher within the SAV beds, due to the wave attenuation properties of the SAV beds, helping enhance the deposition of fine-grained particles. Organic content averaged 0.36 ± 0.48% within the SAV beds and had a mean of 0.17 ± 0.20% outside of the beds ($p=0.26$) (see Fig. 3). Sediment deposition rates (g cm^{-2} y^{-1}) were higher inside the SAV beds as well, 0.79 ± 0.78 g cm^{-2} y^{-1}, when compared to sediment deposition rates outside of the SAV beds, 0.28 ± 0.39 g cm^{-2} y^{-1} ($p=0.07$) (see Fig. 4). Carbon concentrations had a mean of 1.98 ± 0.85% (burial rate: 16.54 ± 16.32 mg/cm^2/y) inside of the SAV beds and outside of the beds carbon concentrations averaged 0.82 ± 0.69% (burial rate: 5.83 ± 8.27 mg cm^{-2} y^{-1}) (see Fig. 5). Nitrogen concentrations averaged 0.08 ± 0.11% (burial rate: 0.53 ± 0.56 mg cm^{-2} y^{-1}) within the SAV beds and had a mean of 0.11 ± 0.09% (burial rate: 0.15 ± 0.30 mg cm^{-2} y^{-1}) outside of the SAV beds (see Fig. 6).

Spring 2019

Spring 2019 had even higher average river discharge than Summer 2018 (2236.88 ± 848.18 m^{-3} s^{-1}) however the peak levels of river discharge were significantly lower (Spring 2019: 4474 m^3 s^{-1}; Summer 2018: 9995 m^3 s^{-1}) (see Table 2). A key distinction to make when comparing Spring and Summer data is the presence of SAV, in the Spring SAV is still in the process of growing back following winter dieback and the beds do not reach full biomass until late Summer. Average mud content within the SAV beds was 12.29 ± 12.06% and 9.29 ± 8.93% outside of the SAV beds in Spring 2019 (p=0.50). Mud content was significantly lower overall than it was in Summer 2018, despite average river discharge being higher on average (see Fig. 2). Average organic content within the SAV beds was 3.44 ± 2.86%, whereas organic content had an average of 2.18 ± 1.50% outside of the SAV beds (p=0.22). Organic content was highest in Spring 2019, both inside and outside of the SAV beds (see Fig. 3). With the lack of SAV beds, mass deposition rates inside and outside of the SAV did not show any significant differences in values. Sediment deposition rates averaged 0.48 ± 0.29 g cm^{-2} y^{-1} inside the SAV beds and 0.49 ± 0.30 g cm^{-2} y^{-1}outside the SAV beds (p=0.92) (see Fig. 4). Without the presence of SAV, there is less sediment deposition in the Flats as the sediment trapping capacity provided by SAV when present is absent. Despite experiencing the highest average river discharge in Spring 2019, amongst our sampling seasons, carbon and nitrogen concentrations were low (see Figs. 5 and 6). Carbon concentrations had a mean of 0.86 ± 0.79% (burial rate: 4.45 ± 5.49 mg cm^{-2} y^{-1}) inside the SAV beds and 0.50 ± 0.48 (burial rate: 2.78 ± 2.95 mg cm^{-2} y^{-1}) outside of the SAV beds. Nitrogen concentrations averaged 0.06 ± 0.05% (burial rate: 0.22 ± 0.21 mg cm^{-2} y^{-1}) within the SAV beds and had a mean of 0.11 ± 0.03 ± 0.02% (burial rate: 0.17 ± 0.15 mg cm^{-2} y^{-1}) outside of the SAV beds (see Fig. 6).

Summer 2019

Summer 2019 was the driest season out of the four sampling seasons, average river discharge was 804.38 ± 490.29 $m^3 s^{-1}$ with a peak of 3369 $m^3 s^{-1}$. As a result, sediment loads and run-off from the Susquehanna River were likely lower than expected summer loads, especially when compared to Summer 2018. There was a discernible difference in mud content when comparing between inside 29.42 ± 32.67% and outside 14.05 ± 26.14% of the SAV beds (see Fig. 2). The presence of SAV helps trap a higher proportion of sediment and fine-grained particles, resulting in higher sediment deposition rates and mud content. Mean organic content within the SAV beds was 1.03 ± 1.02% and averaged 0.41 ± 0.62% outside of the SAV beds (see Fig. 3). Average sediment deposition rates inside of the SAV beds was 0.51 ± 0.46 g cm^{-2} $y^{-1}$and outside of the SAV beds deposition rates averaged 0.48 ± 0.55 g cm^{-2} y^{-1} (see Fig. 4). Carbon and nitrogen concentrations were lower in Summer 2019 than they were in Summer 2018 due in most part to having significantly lower average river discharge. Carbon concentrations averaged 0.98 ± 0.67% (5.25 ± 6.87 mg cm^{-2} y^{-1}) inside of the SAV beds as compared to 0.65 ± 0.56% (3.27 ± 5.71 mg cm^{-2} y^{-1}) outside of the SAV beds (see Fig, 5). Nitrogen concentrations inside of the SAV beds averaged 0.06 ± 0.05% (0.33 ± 0.53 mg cm^{-2} y^{-1}) and outside of the beds average nitrogen concentrations were 0.03 ± 0.02 % (0.15 ± 0.23 mg cm^{-2} y^{-1}) (see Fig. 6).

Discussion

We hypothesized that small patches of SAV would influence sedimentary characteristics as expected from studies in larger SAV beds by increasing sediment and nutrient deposition rates, mud content, and organic content within the vegetation. By sampling in the spring and summer we were able to better ascertain the impacts of seasonal SAV growth cycles. During the spring sampling season, river discharge was generally higher but SAV was largely absent as the beds were still in the process of regrowth following winter dieback, whereas during the summer sampling season river discharge was lower but SAV abundance was much higher than in the spring. Looking at observations for mud content, sediment deposition rates, and nutrient concentrations, taken both inside and outside of the SAV beds there were distinct differences in results between the Spring and Summer sampling seasons. During the summer, there was higher mud content, sediment deposition, and nutrient concentrations within the SAV beds whereas during the spring there was not much difference between the inside and outside of the SAV beds. During winter dieback, SAV abundance is minimal to non-existent and SAV does not play as much of a role in wave and current attenuation when compared to the summer when SAV is abundant (Koch et al, 2010).

Summer 2018 had the highest mud content within the SAV beds (30.82 ± 33.97% inside; 13.92 ± 26.19% outside), likely because of river floods just prior to sampling, as sediment loads transported to the Flats are closely related to river discharge from the Susquehanna River (Russ and Palinkas, 2018). Summer 2019 had the lowest average and peak river discharge, however differences between mud content inside and outside of the SAV beds followed a similar pattern to Summer 2018 nonetheless (29.42 ± 32.67% inside; 14.05 ± 26.14% outside), suggesting that mud content within the beds is more dependent on SAV presence than river discharge (Work et al, 2020; Russ and Palinkas, 2018).

Organic content was highest in Spring 2019 (3.44 ± 2.87% inside; 2.18 ± 1.50% outside) when average river discharge was highest and SAV biomass was low. Looking back at our hypotheses, we expected organic content to be highest when river discharge was high and SAV biomass was high in the summer. These results suggest that organic content is more dependent on river discharge than SAV biomass. Comparing past studies on organic content and SAV, average organic content was consistently below the commonly cited 5%, suggesting the SAV in in these patches is likely not limited by organic content (Koch, 2001; Wicks et al, 2009). When organic content in the sediment is above 5%, there is a greater risk of rhizome dislodgement when exposed to high wave energy (Wicks et al, 2009). SAV in the Chesapeake Bay has a preference for sandy sediments with low organic content, SAV is generally absent in areas where sediment is predominantly fine grained and muddy (Noe et al, 2020; Palinkas and Koch, 2012)

Sediment deposition rates and nutrient content were highest in Summer 2018, coinciding with the highest peak river discharge and the 2nd highest average river discharge. Sediment deposition rates averaged 0.79 ± 0.78 g cm^{-2} y^{-1} inside of the SAV beds, outside of the SAV beds sediment deposition rates had a mean of 0.28 ± 0.39 g cm^{-2} y^{-1} (p=0.07) (see Fig 4). Differences in sediment deposition between inside/outside of the SAV beds in Summer 2018 had the only statistically significant differences when comparing statistical analyses across sampling seasons. Summer 2018 also saw peaks in nutrient concentrations; carbon concentrations averaged 1.98% (carbon burial rate: 16.54 ± 16.32 mg cm^{-2} y^{-1}) and nitrogen concentrations had a mean of 0.08%

(nitrogen burial rate: 0.53 ± 0.56 mg cm^{-2} y^{-1}). Although mean river discharge was highest in Spring 2019, due to low SAV abundance there was reduced sediment trapping capacity in the Flats, therefore sediment deposition rates and mud content were lower inside of the SAV beds than they were in Summer 2018.

Comparing averages for mud content between seasons mud content inside of the SAV beds was much higher during the summer when SAV biomass was high, however our statistical tests (paired t-tests) did not show any statistically significant differences across the 4 sampling efforts. Regardless, the presence of SAV clearly had an influence on mud content within the beds. During Summer 2018, 2019, there was nearly 20% difference in mud content between inside and outside of the SAV beds. We found statistically significant differences for sediment deposition rates (g cm^{-2} y^{-1}) between rates inside and outside of the SAV beds (2 out of 4 seasons: Spring 2018, Summer 2018).

In order to better understand how the re-emergence of SAV beds in the Flats affects local sedimentation patterns, it is essential to consistently evaluate changes in the sedimentary environment within the beds and the surrounding environment. From our results we were able to see distinct differences in sedimentation patterns between the inside and outside of the SAV beds. In summer 2018, when there was high river discharge and high plant biomass, mud content, sediment deposition rates, and nutrient concentrations were higher within the SAV beds. However, during the spring months when river discharge was high but SAV biomass was low, sedimentary properties were comparable between the inside and outside of the SAV beds. In past studies performed within the main SAV bed in the Susquehanna Flats, sedimentation rates were comparable between spring and summer months. River discharge was higher in the spring than the summer due to the spring freshet. However, since SAV biomass is low during the spring, higher river discharge does not necessarily translate to higher rates of deposition within the Flats (Russ and Palinkas, 2018). Comparing results, sedimentation rates were higher in the patchy beds in our study than they were within and along the fringes of the main bed, this could be due to the influence of channels alongside the main bed on sediment transport processes. In Russ and Palinkas, 2018; they propose that while the channels in the Flats direct the majority of sediment-laden fluvial waters into the interior of the main bed, these waters can "spill over" and deposit large sediment loads into shallow SAV beds adjacent to said channels (Russ and Palinkas, 2018). Further study and analysis of the sediment dynamics of the Susquehanna Flats is required to better understand the interconnected relationship between sediment dynamics, river discharge, and SAV abundance.

Summary/Synthesis

The presence of the SAV beds in the Susquehanna Flats during the summer months exerted a strong influence on sedimentary characteristics within the SAV beds. Sediment/nutrient deposition rates, mud, and organic content were all higher within the SAV beds during the summer months. During the spring season, when SAV beds are still recovering from winter dieback, there was not the same observed relationship between the inside and outside of the SAV beds in terms of seasonal-scale sedimentation trends, trends inside/outside of the beds were nearly identical. When SAV biomass is high in the summer, the SAV beds in the Susquehanna Flats increase sediment/nutrient deposition rates and modify local sediment dynamics.

Further research is recommended to study the impact of sedimentation on SAV during periods of high rainfall where river discharge and suspended sediment concentrations are high. Periods of excessive sedimentation can negatively impact SAV coverage and seedling emergence, while high suspended sediment concentrations can also decrease light availability for SAV growth (Noe et al, 2020). Sedimentation patterns within the SAV beds in the Susquehanna Flats are directly linked to river discharge from the Susquehanna River so it is essential to continue to study and monitor sediment deposition in relation to river discharge (Russ and Palinkas, 2018). Future studies should measure suspended sediment concentrations and light attenuation inside and outside of the SAV beds to gain a better understanding on how changes in river discharge and sediment loads affects conditions for SAV growth in the Susquehanna Flats.

2.5 Figures and Tables

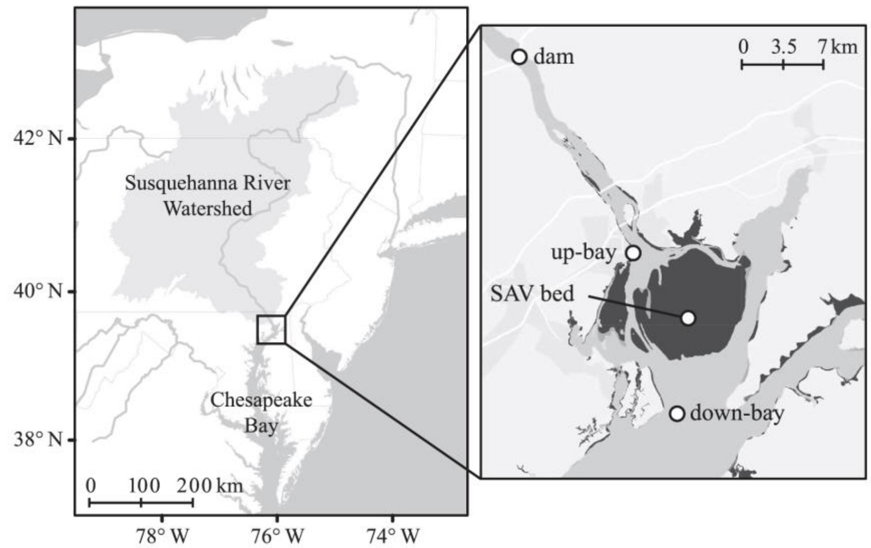


Fig. 1. Location of the Susquehanna River and water quality sampling sites. Dark shaded area in the figure on the right indicates the SAV bed aerial extent in 2010.

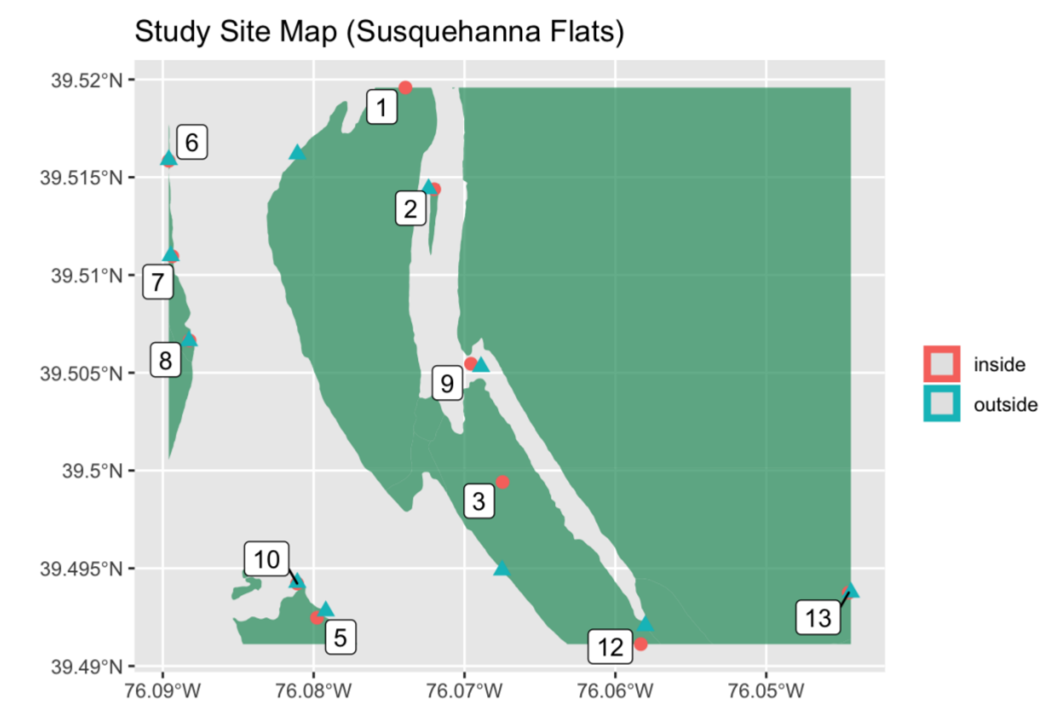


Figure 1. Map of the Chesapeake Bay and Susquehanna Flats (Gurbisz et al, 2014), with an accompanying R-generated map displaying SAV density (sourced from VIMS 2018 SAV abundance data) and the selected SAV beds in the Susquehanna Flats, Chesapeake Bay, MD, U.S.A.

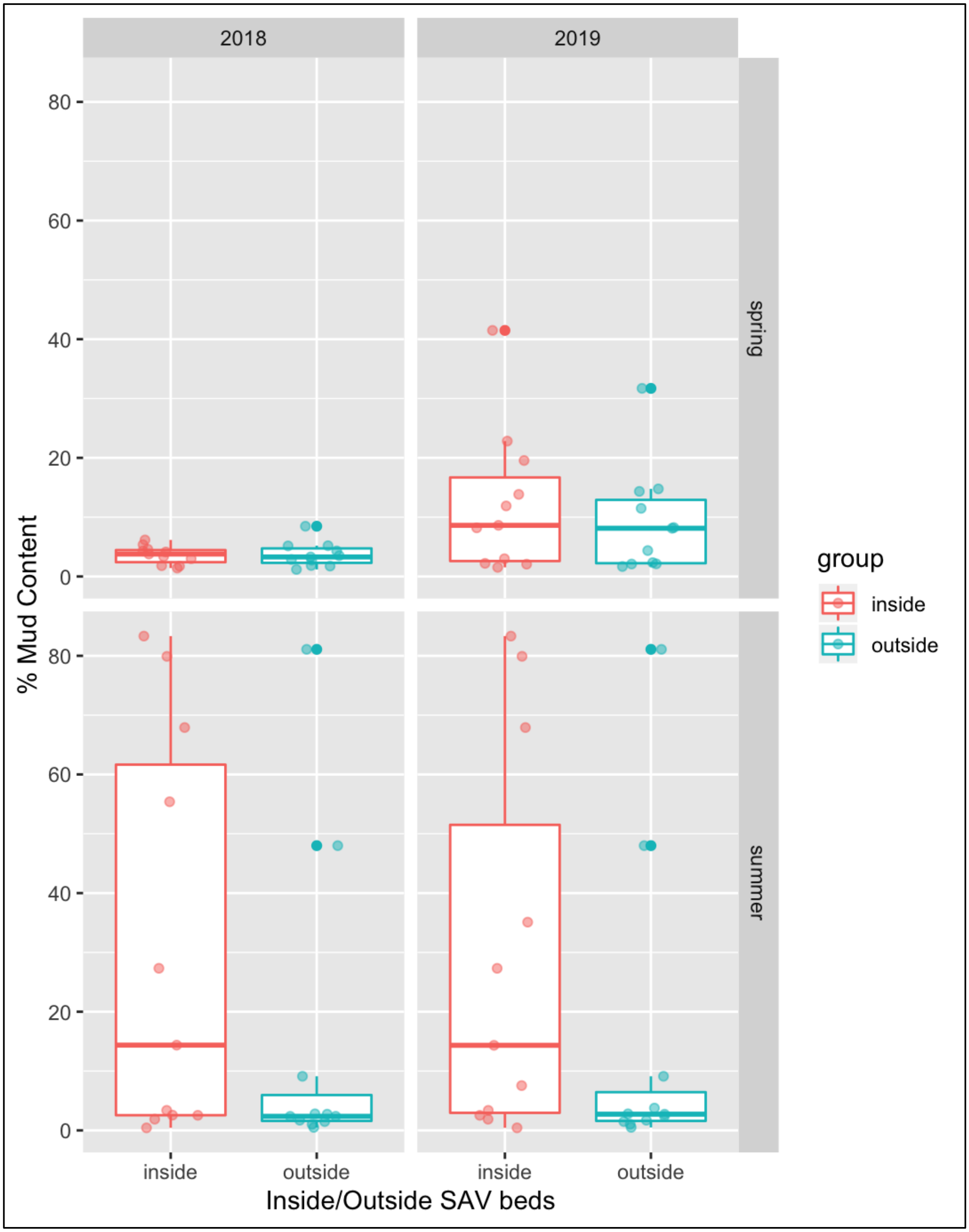


Figure 2. Box and whisker plots displaying average mud content as well as the range of values between the inside and outside of the SAV beds across sampling efforts.

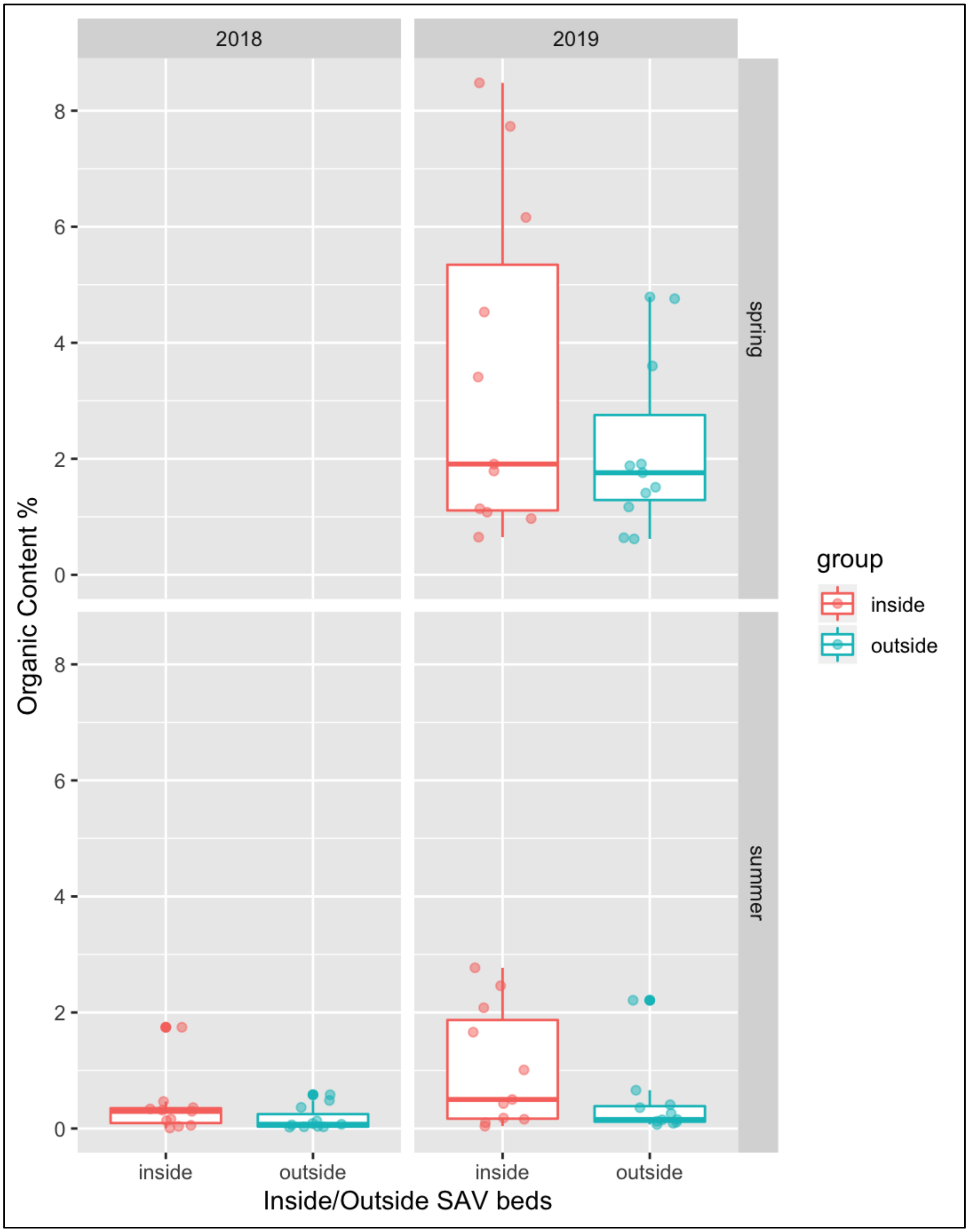


Figure 3. Box and whisker plots showing average organic content across sampling seasons inside and outside of the SAV beds. Organic content data was not available for spring 2018.

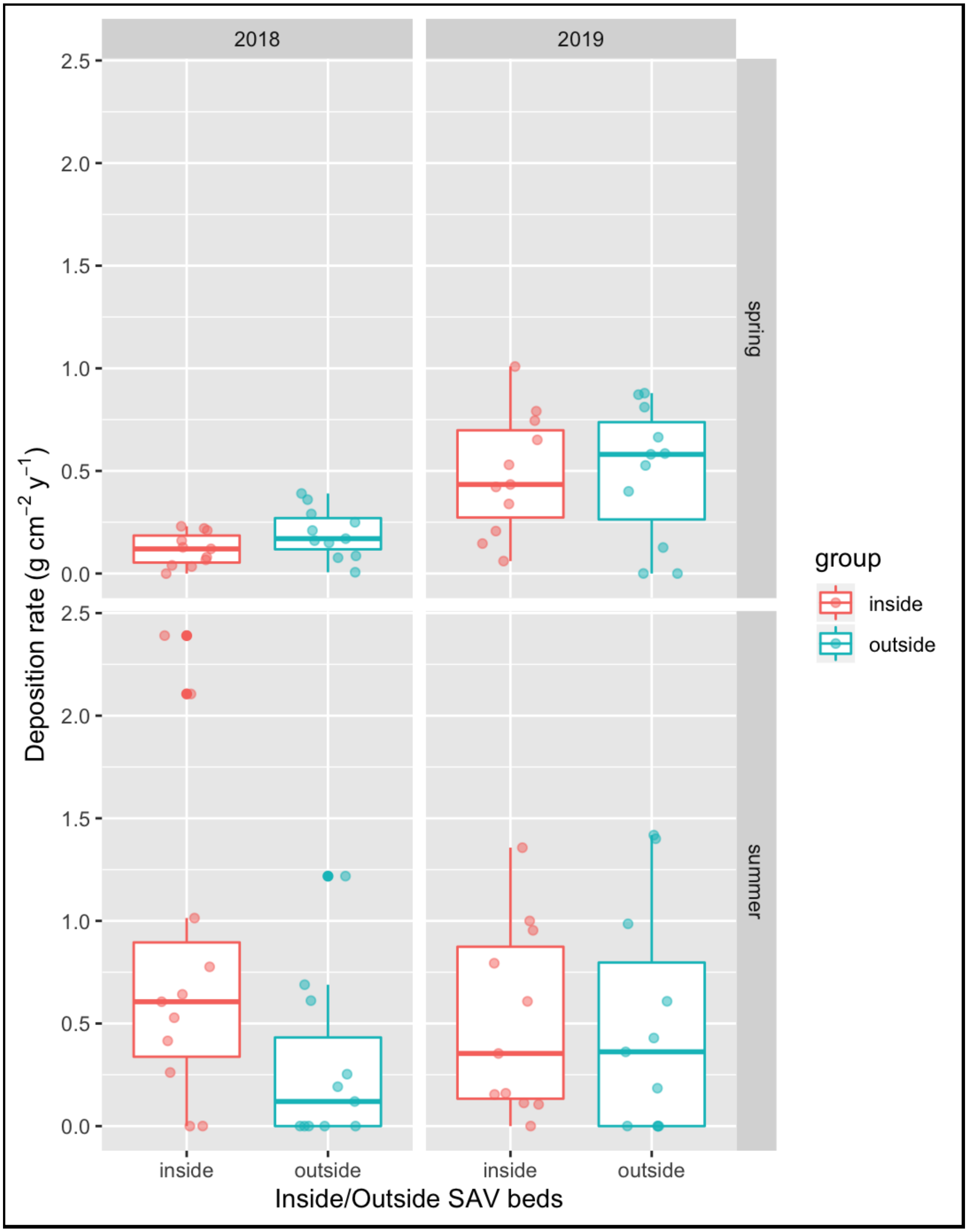


Figure 4. Box and whisker plots showing average deposition rates inside and outside of the SAV beds for each sampling season.

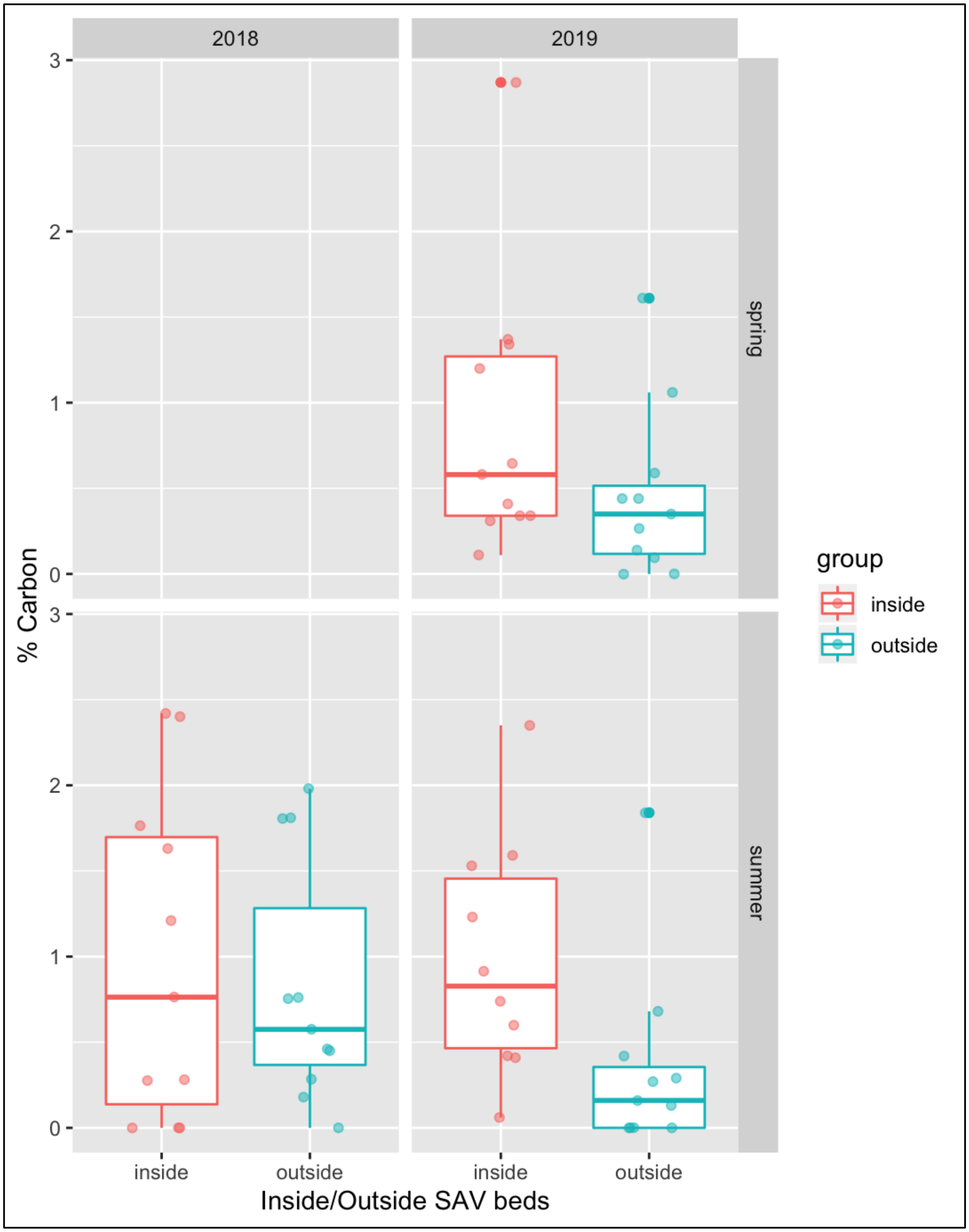


Figure 5. Box and whisker plots displaying average carbon concentrations inside and outside of the SAV beds. Carbon concentration data was not available for spring 2018.

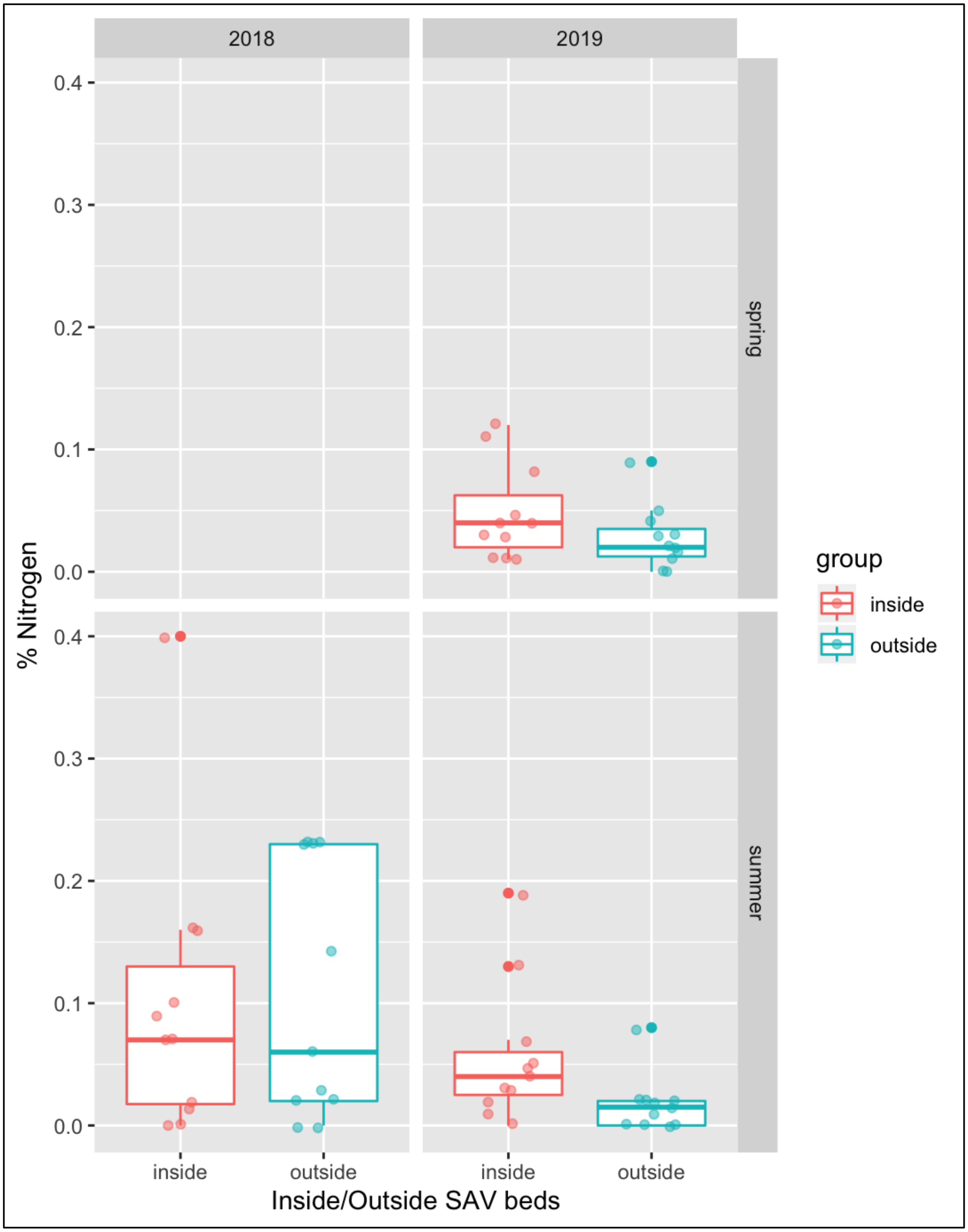


Figure 6. Box and whisker plots showing nitrogen concentrations inside and outside of the SAV beds. Nitrogen concentration data is not available for spring 2018.

Table 1. Table displays averages and standard deviations for mud content, organic content, deposition rates, and nutrient concentrations from each sampling season.

Season/Year	In/Out	Mud Content	Organic Content %	Deposition Rate ($g/cm^2/y$)	Carbon % /Burial Rate ($mg/cm^2/y$)	Nitrogen %/Burial Rate ($mg/cm^2/y$)
Spring 2018	Inside	3.60 ± 1.52		0.12 ± 0.08		
Summer 2018	Inside	30.82 ± 33.97	0.36 ± 0.48	0.79 ± 0.78	1.98 ± 0.85%/16.54 ± 16.32	0.08 ± 0.11%/0.53 ± 0.56
Spring 2019	Inside	12.29 ± 12.06	3.44 ± 2.86	0.48 ± 0.29	0.86 ± 0.79%/4.45 ± 5.49	0.05 ± 0.04%/0.22 ± 0.21
Summer 2019	Inside	29.42 ± 32.67	1.03 ± 1.02	0.51 ± 0.46	0.98 ± 0.67%/5.25 ± 6.87	0.06 ± 0.05%/0.36 ± 0.43
Average		**19.05 ± 20.05**	**1.61 ± 1.45**	**0.47 ± 0.40**	**1.27 ± 0.77%/8.74 ± 9.56**	**0.06 ± 0.07%/0.36 ± 0.43**
Spring 2018	Outside	3.66 ± 2.06		0.19 ± 0.11		
Summer 2018	Outside	13.92 ± 26.19	0.17 ± 0.20	0.28 ± 0.39	0.82 ± 0.69%/5.83 ± 8.27	0.11 ± 0.09%/0.15 ± 0.30
Spring 2019	Outside	9.29 ± 8.93	2.18 ± 1.50	0.49 ± 0.32	0.50 ± 0.48%/2.78 ± 2.95	0.03 ± 0.02%/0.17 ± 0.15
Summer 2019	Outside	14.05 ± 26.14	0.41 ± 0.62	0.48 ± 0.55	0.65 ± 0.56%/3.27 ± 5.71	0.03 ± 0.02%/0.15 ± 0.23
Average		**10.23 ± 15.83**	**0.92 ± 0.77**	**0.36 ± 0.34**	**0.65 ± 0.57 /3.96 ± 5.58**	**0.06 ± 0.04%/0.16 ± 0.22**

Table 2. Table displays averages and peaks for river discharge for each sampling season. Data is sourced from (https://waterdata.usgs.gov/md/nwis/uv?site_no=01578310).

Sampling Season	Average River Discharge (m^3/s)	Peak River Discharge (m^3/s)
Spring 2018	1423.88 ± 624.44	4813
Summer 2018	1733.22 ± 1933.08	9995
Spring 2019	2236.88 ± 848.18	4474
Summer 2019	804.38 ± 490.29	3369
Averages	1549.59 ± 973.99	5662.75 ± 2593.24

Summary/Synthesis

This study evaluated sediment dynamics in coastal habitats (created marshes of living shorelines, SAV), resulting in a better understanding of the role vegetations plays in trapping sediment.

In the case of the living shorelines (Chapter 1), the driving questions were concerned with how the installation of these structures impact subtidal sedimentation and SAV presence/absence in front of the living shoreline and nearby unaltered reference shorelines. We found from our study that living shorelines, on average, do not have a strong influence on shifts in sedimentary characteristics or SAV presence/absence in the adjacent nearshore. The observed parameters for mud content, organic content, and mass accumulation rate (MAR) exhibited similar trends following living shoreline installation at both living shoreline and reference shoreline sites. However, changes at individual sites can be significant such as at the living shoreline sites at Queens Landing or Environmental Concern where there were significant increases in mud content. SAV presence/absence at individual sites seemed to follow regional trends, likely driven by water quality. This suggests that either that living shorelines do not have a significant impact on local sedimentation patterns or the installation of the living shorelines affected trends at both living shoreline and reference shoreline sites.

Reviewing Chapter 2, we found that the emergent, patchy SAV beds in the Susquehanna Flats did assert influence on local sedimentary properties (mud content, deposition rates, and nutrient concentrations) when SAV abundance was high during the summer months. Local sedimentation inside and outside of the beds was dictated to a degree by river discharge from the Susquehanna River. River discharge is highest in the spring, aided by the spring freshet, however SAV abundance is lower in the spring as the SAV is still recovering from winter dieback thus rates of sedimentation were comparable inside and outside of the SAV beds. Assessing average organic content across spring and summer sampling, organic content remained below the 5% threshold, suggesting that the risk of potential dislodgement is minimal (Han et al, 2012).

This study helps provide supporting evidence for the use of living shorelines while building on recent studies done on the Susquehanna Flats to better understand the influences of these resurgent SAV beds on local sediment dynamics. Recent studies have shown that the presence of SAV can help increase the resilience of salt marshes and living shorelines against sea level rise by lowering flow velocity and sediment transport dynamics while also increasing the sediment budgets of shallow estuaries (Donatelli et al, 2018). Future studies should look into long term trends studying feedbacks between living shorelines and SAV while quantifying how these ecosystems impact local sediment budgets.

In contrast to past studies performed on the main SAV bed in the Flats (Russ and Palinkas, 2018) there were observable differences in seasonal-scale sedimentation rates between the inside and outside of the SAV beds. Our study in the Susquehanna Flats, shows that smaller, patchier SAV beds can also have influence on local sediment dynamics. Further study is needed to better understand the relationship between SAV abundance, river discharge and sedimentation patterns in the Flats. Excessive accumulation of fine-grained sediment with high organic content can create unsuitable conditions for SAV growth, as SAV prefer sandy sediment with low organic content (Noe et al, 2020; Wicks et al, 2009; Koch et al 2001). During periods of high precipitation and river discharge, SAV growth can be affected by high sediment loads, higher levels of suspended sediment, as well as decreased light availability (Noe et al, 2020; Work et al, 2020).

References

Appleby, P., & Oldfield, F. (1978). The calculation of lead-210 dates assuming a constant supply of unsupported ^{210}Pb to the sediment. *Catena, 5*, 1-8.

Bayley, S., Stotts, V. D., Springer, P. F., & Steenis, J. (1978). Changes in submersed aquatic macrophyte population at the head of the Chesapeake Bay, 1958-1975. *Estuaries, 3,* 73-84.

Bilkovic, D. M., & Roggero, M. M. (2008). Effects of coastal development on nearshore estuarine nekton communities. *Marine Ecology Progress Series, 358*, 27-38.

Bird, M. I., Fifield, L. K., Chua, S., & Goh, B. (2004). Calculating sediment compaction for radiocarbon dating of intertidal sediments. *Radiocarbon, 46*(Nr 1), 421-435.

Boesch, D. F., Atkinson, L. P., Boicourt, W. C., Boon, J. D., & Cahoon, D. R. (2013). Updating Maryland's sea-level rise projections. *CCPO Publications,* 1-24.

Boon, J. D. (2020). Evidence of sea level acceleration at U.S. and Canadian tide stations, Atlantic Coast, North America. *Journal of Coastal Research, 28*(6), 1437-1445.

Bos, A. R., Bouma, T. J., De Kort, G. L., & Van Katwijk, M. M. (2004). Ecosystem engineering by annual intertidal seagrass beds: Sediment accretion and modification. *Estuarine, Coastal, and Shelf Science, 74*, 344-348.

Cabaço, S., Santos, R., & Duarte, C. M. (2008). The impact of sediment burial and erosion on seagrasses: A review. *Estuarine, Coastal, and Shelf Science, 79*, 354-366.

Cheng, P., Li, M., & Li, Y. (2013). Generation of an estuarine sediment plume by a tropical storm. *JGR Oceans, 118*(2), 856-868.

Currin, C. A., Chappell, W. S., & Deaton, A. (2010). Developing alternative shoreline armoring strategies: The living shorelines approach in North Carolina. *Puget Sound Shorelines and Impacts of Armoring-Proceedings of a State of the Science Worksop,* 91-102.

Davenport, T. M., Seitz, R. D., Knick, K. E., & Jackson, N. (2018). Living shorelines support nearshore benthic communities. *Estuaries and Coasts, 41,* S197-S206.

Davis, J. L., Takacs, R. L., & Schnabel, R. (2006). Evaluating ecological impacts of living shorelines and shoreline habitat elements: An example from the upper western C. *Living Shoreline Summit. Conference Proceedings.,* 55-61.

De Boer, W. (2007). Seagrass-sediment interactions, positive feedbacks and critical thresholds for occurrence: A review. *Hydrobiologia, 591,* 5-24.

Dibbs, J. E. (1989). Atmospheric deposition of beryllium 7 in the Chesapeake Bay region. *Journal of Geophysical Research, 94*(D2), 2261-2265.

Donatelli, C., Ganju, N. K., Fagherazzi, S., & Leonardi, N. (2018). Seagrass impact on sediment exchange between tidal flats and salt marsh, and the sediment budget of shallow bays. *Geophysical Research Letters, 45*, 4933-4943.

Donoghue, J. F., Bricker, O. P., & Olsen, C. R. (1989). Particle-borne radionuclides as tracers for sediment in the Susquehanna River and Chesapeake Bay. *Estuarine, Coastal, and Shelf Science, 29*, 341-360.

Duarte, C. M., Losada, I. J., Hendriks, I. E., Mazarrasa, I., & Marbà, N. (2013). The role of coastal plant communities for climate change mitigation and adaptation. *Nature Climate Change, 3*, 961-967.

Fisher, T. R., Hagy, J. D., Boynton, W. R., & Williams, M. R. (2006). Cultural eutrophication in the Choptank and Patuxent estuaries of Chesapeake Bay. *Limnology and Oceanography, 51*(1), 435-447.

Ganthy, F., Sottolichio, A., & Verney, R. (2013). Seasonal modification of tidal flat sediment dynamics by seagrass meadows of Zostera noltii (Bassin d'Arcachon, France). *Journal of Marine Systems, 109-110*, S233-S240.

Gedan, K. B., Kirwan, M. L., & Wolanski, E. (2011). The present and future role of coastal wetland vegetation in protecting shorelines: Answering recent challenges to the paradigm. *Climatic Change*, *106*, 7-29.

Gittman, R. K., Peterson, C. H., Currin, C. A., Fodrie, F. J., Piehler, M. F., & Bruno, J. F. (2016). Living shorelines can enhance the nursery role of threatened estuarine habitats. *Ecological Applications*, 150511113453001.

Golden, R. R., Busch, K. E., Karth, L. P., Parham, T. A., Lewandowski, M. J., & Naylor, M. D. (2010). Large-Scale *Zostera marina* (eelgrass) restoration in Chesapeake Bay, Maryland, USA. Part 2: A comparison of restoration methods in the Patuxent and Potomac Rivers. *Restoration Ecology, 18*(4), 501-513.

Gross, M.G., M. Karweit, W.B. Cronin, and J.R. Schubel. 1978. Suspended sediment discharge of the Susquehanna River to northern Chesapeake Bay, 1966 to 1976. *Estuaries* 1 (2): 106.

Gurbisz, C., Kemp, W. M., Sanford, L. P., & Orth, R. J. (2016). Mechanisms of storm-related loss and resilience in a large submersed plant bed. *Estuaries and Coasts, 39*, 951-966.

Gurbisz, C., & Kemp, M. W. (2014). Unexpected resurgence of a large submersed plant bed in Chesapeake Bay: Analysis of time series data. *Limnology and Oceanography*, *59*(2), 482-494.

Han, Q., Bouma, T. J., Brun, F. G., Suykerbuyk, W., & Van Katwijk, M. M. (2012). Resilience of *Zostera noltii* to burial or erosion disturbances. *Marine Ecology Progress Series, 449*, 133-143.

Heck, K. L., Jr., Hays, G., & Orth, R. J. (2003). Critical evaluation of the nursery role hypothesis for seagrass meadows. *Marine Ecology Progress Series, 253,* 123-136.

Koch, E. W. (2001). Beyond light: Physical, geological, and geochemical parameters as possible submersed aquatic vegetation habitat requirements. *Estuaries, 24*, 1-17.

Koch, E. W., Ailstock, M. S., Booth, D. M., Shafer, D. J., & Magoun, A. D. (2010). The role of currents and waves in the dispersal of submersed angiosperm seeds and seedlings. *Restoration Ecology, 18*(4), 584-595.

Kornis, M. S., Breitburg, D., Balouskus, R., & Bilkovic, D. M. (2017). Linking the abundance of estuarine fish and crustaceans in nearshore waters to shoreline hardening and land cover. *Estuaries and Coasts, 40,* 1464-1486.

Lefcheck, J. S., Orth, R. J., Dennison, W. C., Wilcox, D. J., Murphy, R. R., Keisman, J., & Gurbisz, C. (2018). Long-term nutrient reductions lead to the unprecedented recovery of a temperate coastal region. *Proceedings of the National Academy of Sciences, 115*(14), 3658-3662.

Leonardi, N., Carnacina, I., Donatelli, C., Ganju, N. K., Plater, A. J., Schuerch, M., & Temmerman, S. (2018). Dynamic interactions between coastal storms and salt marshes: A review. *Geomorphology, 301*, 92-107.

Leonardi, N., Ganju, N. K., & Fagherazzi, S. (2016). A linear relationship between wave power and erosion determines salt-marsh resilience to violent storms and hurricanes. *PNAS, 113*(1), 64-68.

Moki, H., Taguchi, K., Nakagawa, Y., Montani, S., & Kuwae, T. (2020). Spatial and seasonal impacts of submerged aquatic vegetation (SAV) drag force on hydrodynamics in shallow waters. *Journal of Marine Systems, 209,* 1-12.

Morgan, P. A., Burdick, D. M., & Short, F. T. (2009). The functions and values of fringing salt marshes in Northern New England, USA. *Estuaries and Coasts, 32*, 483-495.

Naijar, R. G., Pyke, C. R., Adams, M., Breitburg, D., & Hershner, C. (2010). Potential climate-change impacts on the Chesapeake Bay. *Estuarine, Coastal, and Shelf Science*, *86*, 1-20.

Neubauer, S. C. (2008). Contributions of mineral and organic components to tidal freshwater marsh accretion. *Estuarine, Coastal, and Shelf Science, 78,* 78-88.

Neubauer, S. C., Anderson, I. C., Constantine, J. A., & Kuehl, S. A. (2002). Sediment deposition and accretion in a Mid-Atlantic (U.S.A.) tidal freshwater marsh. *Estuarine, Coastal, and Shelf Science, 54*, 713-727.

Nittrouer, C. A., Sternberg, R. W., Carpenter, R., & Bennett, J. T. (1979). The use of Pb-210 geochronology as a sedimentological tool: Application to the washington continental shelf. *Marine Geology, 31*, 297-316.

Noe, G. B., Cashman, M. J., Skalak, K., Gellis, A., & Hopkins, K. G. (2020). Sediment dynamics and implications for management: State of the science from long-term research in the Chesapeake Bay watershed, USA. *Wires Water, 7*(4), 1-28.

O'Donnell, J. E. (2016). Regulatory issues for implementing living shorelines. *National Wetlands Newsletter*, *38*(2), 19-24.

Olsen, C. R., Larsen, I. L., Lowry, P. D., Cutshall, N. H., & Nichols, M. M. (1986). Geochemistry and deposition of ^{7}Be in river-estuarine and coastal Waters. *Journal of Geophysical Research*, *91*(C1), 896-908.

Orth, R. J., Dennison, W. C., Lefcheck, J. S., Gurbisz, C., Hannam, M., Keisman, J., . . . Moore, K. (2017). Submersed aquatic vegetation in Chesapeake Bay: Sentinel species in a changing world. *Bioscience*, *67*(8), 698-712.

Orth, R. J., & Moore, K. A. (1984). Distribution and abundance of submersed aquatic vegetation in Chesapeake Bay: An historical perspective. *Estuaries*, *7*(4B), 531-540.

Orth, R. J., Williams, M. R., & Marion, S. R. (2010). Long-term trends in submersed aquatic vegetation (SAV) in Chesapeake Bay, USA, related to water quality. *Estuaries and Coasts, 33*, 1144-1163.

Pace, N. L., & Morgan, N. (2017). Living shorelines: Eroding regulatory barriers to coastal resilience. *Natural Resources & Environment*, *31*(3), 44-47.

Palinkas, C. M., & Engelhardt, K. A. (2016). Spatial and temporal patterns of modern (~100 yr) sedimentation in a tidal freshwater marsh: Implications for future sustainability. *Limnology and Oceanography, 61*, 132-148.

Palinkas, C. M., & Koch, E. W. (2012). Sediment accumulation rates and submersed aquatic vegetation (SAV) distributions in the mesohaline Chesapeake Bay, USA. *Estuaries and Coasts*, *35*, 1416-1431.

Palinkas, C. M., Halka, J. P., Li, M., Sanford, L. P., & Cheng, P. (2014). Sediment deposition from tropical storms in the upper Chesapeake Bay: Field observations and model simulations. *Continental Shelf Research, 86*, 6-16.

Palinkas, C. M., Sanford, L. P., & Koch, E. W. (2018). Influence of shoreline stabilization structures on the nearshore sedimentary environment in mesohaline Chesapeake Bay. *Estuaries and Coasts*, *41*, 952-965.

Palinkas, C. M., Nittrouer, C. A., Wheatcroft, R. A., & Langone, L. (2005). The use of 7Be to identify event and seasonal sedimentation near the Po River delta, Adriatic Sea. *Marine Geology, 222*, 95-112.

Patrick, C. J., Weller, D. E., Orth, R. J., & Wilcox, D. J. (2018). Land use and salinity drive changes in SAV abundance and community composition. *Estuaries and Coasts, 41*, S85-S100.

Patrick, C. J., Weller, D. E., Li, X., & Ryder, M. (2014). Effects of shoreline alteration and other stressors on submersed aquatic vegetation in subestuaries of Chesapeake Bay and the Mid-Atlantic coastal bays. *Estuaries and Coasts, 37*, 1516-1531.

Peterson, M. S., & Lowe, M. R. (2009). Implications of cumulative impacts to estuarine habitat quality for fish and invertebrate resources. *Reviews in Fisheries Science, 17*(4), 505-523.

Russ, E. R., & Palinkas, C. M. (2018). Seasonal-scale and decadal-scale sediment-vegetation interactions on the subaqueous Susquehanna River delta, Upper Chesapeake Bay. *Estuaries and Coasts*, *41*, 2092-2104.

Russ, E., & Palinkas, C. (2020). Evolving sediment dynamics due to anthropogenic processes in the Chesapeake Bay. *Estuarine, Coastal, and Shelf Science, 235*, 1-12.

Sallenger, A. H., Jr., Doran, K. S., & Howd, P. A. (2012). Hotspot of accelerated sea-level rise on the Atlantic coast of North America. *Nature Climate Change, 2*, 884-888.

Szmytkiewicz, A., & Zalewska, T. (2014). Sediment deposition and accumulation rates determined by sediment traps and ^{210}Pb isotope methods in the Outer Puck Bay (Baltic Sea). *OCEANOLOGIA, 56*(1), 85-106.

Temmerman, S., Meire, P., & Bouma, T. J. (2013). Ecosystem-based coastal defence in the face of global change. *Nature, 504*, 79-83.

Waycott, M., Duarte, C. M., Carruthers, T. J., Orth, R. J., Dennison, W. C., Olyarnik, S., & Calladine, A. (2009). Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *PNAS, 30*, 12377-12381.

Wicks, E. C., Koch, E. W., & O'Neil, J. M. (2009). Effects of sediment organic content and hydrodynamic conditions on the growth and distribution of *Zostera marina*. *Marine Ecology Progress Series, 378*, 71-80.

Work, P. A., Downing-Kunz, M., & Drexler, J. Z. (2020). Trapping of suspended sediment by submerged aquatic vegetation in a tidal freshwater region: Field observations and long-term trends. *Estuaries and Coasts*, 1-16.