SOIL ATTRIBUTE VARIATIONS FOLLOWING SHORELINE RESTORATION IN SHELTON, WASHINGTON

by

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ABSTRACT

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Shoreline ecosystems provide a variety of valuable services for human communities and marine wildlife, including significant carbon storage functions and critical habitat. Despite their abundant benefits, 74% of Puget Sound salt marshes and estuaries have been developed, degraded, or destroyed (The Seattle District Corps of Engineers, 2016). Furthermore, climate change and predicted sea level rise threaten the well-being and functionality of these ecosystems. To remedy this problem, coastal ecosystems are being restored and preserved around the Puget Sound region. One such area is the Bayshore Preserve in Shelton, Washington. This property was purchased by the Capitol Land Trust in 2014 and, as of that year, is being actively converted to a nature preserve. The preserve contains 4,000 feet of Oakland Bay shoreline, of which, in 1947, 1,400 feet had been armored with a tidal dike. In 2014, the dike was removed and several tidal channels and basins were excavated to create new saltmarsh habitat. All subsequent rehabilitation processes in the new Bayshore basins have been natural, facilitated by tidal exchange. This study examines how sediment, vegetation, and carbon variables interact as a function of elevation in these newly created tidal basins to approximate the progress and viability of the habitat. Soils were analyzed for relationships between salinity, percent carbon, texture, and bulk density. The results indicated that the unrestricted tidal influence appears to have indeed facilitated a healthy effect on the newly created saltmarsh habitat. Elevation appeared to be the common factor playing a significant role in the variations of most of the variables. Vegetation exhibited a relatively strong trend with salinity and texture. Furthermore, higher percentages of fine clay and silt sediments in the basin soil coincided with higher levels of soil carbon content. This monitoring data could allow land managers to retain and observe a robust evolving model of the health of a restored site. Continued documentation on how coastal ecosystems function while facing restoration challenges can be vital to influencing perceptions on the value of community investment in urbanized estuaries.

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1. Introduction

Shoreline habitats are important sites for a diverse abundance of flora and fauna and they also play a vital role in the Earth's interconnected climate system (Callaway and Crooks, 2015). Human communities are drawn to these coastal areas, as they provide a variety of ecosystem services and resources that are very valuable to our societies. However, in order to develop these resources and to economically progress, significant alterations are made upon the nearshore environments that disrupt the natural cycles and relationships that comprise their entire ecosystem (The Seattle District Corps of Engineers, 2014). The Puget Sound region is an excellent example of this form of altered habitat (Morley et al., 2012; Seattle District Corps of Engineers, 2014).

The Puget Sound is a large estuary in northwestern Washington State in the southern part of the Salish Sea, it is the second largest estuary in the US, and serves as an inlet of the Pacific Ocean (Morley et al., 2012; Seattle District Corps of Engineers, 2014). Puget Sound is made up of an elaborate network of rivers and watersheds and provides habitat to a diverse array of migratory birds, fish, benthic organisms, and marine mammals. Puget Sound is also home to thirteen fish and mammal species listed as threatened or endangered under the Endangered Species Act (Seattle District Corps of Engineers, 2014). Its shorelines not only serve as habitat for abundant plant and animal species, but within the Puget Sound region, these shorelines also play a significant part in complex trophic dynamics, biogeochemical cycling, regulating flood control, and water filtration (Callaway and Crooks, 2015).

Together with the ecosystem and habitat services that shoreline wetlands provide, globally, many studies have shown that they are powerful carbon sequestration sites

(Hopkinson et al., 2012; Weston et al., 2014). This is an important feature, in the sense that excessive anthropogenic carbon emissions appear to be the dominant driver for climate change (IPCC, 2014). One approach to stemming this outflow of greenhouse gases into the atmosphere could be to rehabilitate and conserve coastal ecosystems. Healthy saltmarshes continually bury carbon through tidal sediment inputs in a process called accretion (Hopkinson et al., 2012; Hansen and Nestlerode, 2013; Lovelock et al., 2013). In this way, they can play a formidable role in absorbing carbon and decelerating the progression of anthropogenic climate change (Hansen and Nestlerode, 2013).

Although these sites are essential for important natural functions, Euro-American colonization and land use change has led to profound modifications in their performance and geography (Jolivette et al., 2014). In fact, despite their abundant benefits, 74.2% of Puget Sound salt marshes and estuaries have been developed, degraded, or destroyed (The Seattle District Corps of Engineers, 2016). In order to protect private properties and coastal businesses, shoreline armoring in the forms of dikes, seawalls, and bulkheads have been erected along 27% of Washington's coastlines as of 2011 (Seattle District Corps of Engineers, 2016). Due to this rapid urbanization, modern-day shoreline ecosystems in the Puget Sound are being lost or degraded at a rapid rate, and those that are intact face various approaching threats, including development, pollution, and climate change. As of 2011, 56% of historic tidal wetlands in the 16 largest Puget Sound river deltas have been lost, and 17% of these existing wetlands are predicted to be lost by 2060 (Fresh et al., 2011; Seattle District Corps of Engineers, 2016). To decelerate this habitat loss pattern, it is vital to conserve and monitor the remaining shoreline ecosystems, as well as to restore and rehabilitate as many disturbed sites as possible.

This thesis will examine the progress of the Bayshore Preserve shoreline restoration site in Shelton, Washington, located in the Southern Puget Sound. The Bayshore Preserve is a small peninsula that juts out along the northwestern shoreline of Oakland Bay (Figure 3.1). This peninsula has undergone considerable land use changes following the area's Euro-American colonization in the mid-1800s (Jolivette et al., 2014). After being utilized as a sawmill-and-railway logging operation in the late 1800s, the land was repurposed and converted into a golf course resort in the 1930s (Jolivette et al., 2014). In 1947, a large 1,400-foot long dike was constructed along the shoreline to protect the golf course from heavy tides and flotsam, and substantial modifications were made upon the soil composition and plant communities of the peninsula. In 2015, the golf course land was purchased by the Capitol Land Trust (CLT) in Olympia, Washington. The CLT then initiated an extensive rehabilitation program on the site to return the peninsula to a more natural and ecologically functional state. Some of the most notable modifications include: the removal of the tidal dike, the excavation of new tidal basins for saltmarsh habitat, and the planting of native vegetation along the riparian zones.

Fine sediment inputs in saltmarsh environments are important for healthy soil, vegetation communities, wildlife habitat, and nutrient fluxes. The research question for this thesis is, "Are newly created tidal basins in Bayshore Preserve effective for collecting fine sediment material?" A follow-up question is, "How does variation in elevation, soil salinity, and vegetation cover affect the distribution of these parameters? To answer this question, specific soil attributes in the saltmarsh basins were measured to determine whether their construction indeed serves as a viable habitat. Healthy soil salinity levels are essential for a functional saltmarsh vegetation community, which is why this was a considered a valuable variable to evaluate (Howard et al., 2008; Janousek & Folger, 2014). Soil carbon content was chosen to be measured as well, as many scientists already agree that per acre, coastal wetlands sequester significantly more carbon than inland ecosystems (Hopkinson et al., 2012; Hansen & Nestlerode, 2013; Crooks et al, 2014). By observing carbon content in the soils, one could apply this information as a lens through which to observe saltmarsh restoration progress. Carbon content is typically controlled by soil texture, necessitating the measurements of the distribution of sand, silt and clay (Krull et al., 2001). Bulk density was also measured because it can also be related to carbon content and fine sediment concentration, and could also assist with determining soil surface carbon concentration (Beecher et al., 2001).

The creation of artificial tidal basins for saltmarsh habitat is a relatively novel approach to saltmarsh restoration. The findings of this thesis exhibit relationships between the soil surface carbon content, soil salinity levels, and fine sediment content variables. Moreover, the findings demonstrate that this rehabilitation approach does appear to be successful in accumulating carbon and fine sediments as well as accommodating saltmarsh vegetation. As such, this success can imply further incentive for continued support of intertidal wetland conservation and rehabilitation. Furthermore, this study could also expand the body of knowledge concerning the carbon and salinity relationships of estuarine restoration projects in the Puget Sound region for land managers and restoration ecologists.

This thesis is organized as follows: Chapter 2 is a literature review that examines scientific studies surrounding central themes from restoration ecology and the significance of nearshore ecosystems. The first section, 2.1, introduces the concepts of the complex network of abiotic and biotic mechanisms that comprise shoreline habitats and their valuable ecosystem services, details the Puget Sound region and its shoreline geography, and expands on the myriad of threats facing these ecosystems, particularly those provoked by human development and climate change. Section 2.2 explains the essentials of saltmarsh soil composition, vegetation associations, biotic communities, and ecosystem services. Next, Section 2.3 reviews the successes and pitfalls of local restoration ecology, restorative approaches, and monitoring strategies. Section 2.4 introduces the Bayshore Preserve in Shelton, Washington, the restoration site serving as the focus of the thesis. An evaluation of the literature in these topics will make clear the importance of maintaining and rehabilitating nearshore ecosystems in the Puget Sound. The third chapter reviews the research, goals, sampling methods, and statistical analysis. The fourth chapter covers the results, and the fifth chapter provides a discussion on the significance of the research and suggestions for future Puget Sound shoreline restoration and rehabilitation projects.

2. Literature Review

2.1 NEARSHORE ECOSYSTEMS AND CLIMATE CHANGE

Climate change is putting the functionality of the biosphere of planet Earth at risk (Smith, et al., 2000; IPCC, 2014; Mauger et al, 2015). Since the beginning of the Industrial Revolution, humankind has been continuously emitting carbon emissions and other greenhouse gases (GHG) into the planet's atmosphere through the burning and extraction of fossil fuels for transportation, manufacturing, agriculture, and deforestation (Smith et al, 2000, IPCC, 2014). The 2014 IPCC report indicates that anthropogenic greenhouse gas (GHG) emissions have increased from 27 gigatons of CO_2 –equivalents per year ($GtCO_2$ –eq/yr) in 1970 to 49 $GtCO_2$ – eq/yr in 2010 (IPCC, 2014). As these greenhouse gases are released into the atmosphere, they contribute to anthropogenic climate change, affecting biogeochemical cycles, including the carbon cycle (IPCC, 2014).

Intertidal Wetlands and Carbon Storage Services

All ecosystems, including forests, aquatic vegetation communities, and the ocean, play a vital function in the global carbon cycle. As humanity continues to emit greenhouse gases and the planet's climate continues to change, it becomes increasingly important to understand the current performance and condition of the Earth's carbon sinks (Seong Rhee, et al., 2009; Hansen & Nestlerode, 2014; Callaway & Crooks, 2015; Sheng et al., 2015). This entails continual monitoring and assessment to ensure that carbon reservoirs are functioning properly in changing environmental regimes. The science and land management communities have been discussing the utility of the nearshore wetlands' carbon sequestration properties as an economic incentive for restoration and preservation investment (Chmura et al, 2003; Crooks and Maddox, 2014; Weston et al. 2014).

Estuaries and coastal wetland ecosystems, if in proper functioning condition, serve as effective carbon sequestration sites (Chmura et al., 2003; Crooks and Maddox, 2014; Weston et al., 2014). Examples of these include salt marshes, mangroves, swamps, and seagrass meadows (Smith et al., 2000). Though they make up only a small proportion of the planet's landmass, these coastal ecosystems absorb and sequester significantly more carbon than most inland and aquatic sites (Hopkinson, 2012; Hansen & Nestlerode, 2014; Callaway & Crooks, 2015). Globally, nearshore saltmarshes and mangroves sequester at least 44.6 Tg C/yr (Chmura et al., 2003). High levels of carbon, nutrients, and sediment accumulate in their soils and vegetation due to deposition, continuous accretion, low decomposition rates, and tidal influence (Lovelock et al., 2013; Callaway & Crooks, 2015). Decomposition of organic matter is inhibited both by soil and water salinity, as well as the anaerobic conditions created by tidal inundation (Lovelock et al., 2013; Masello, 2013; Franklin et al., 2014). Furthermore, the nearshore wetlands vertically accrete sediments, continually burying carbon and organic matter (Hopkinson, 2012; Callaway & Crooks, 2015). This mechanism allows for intertidal marshes to keep up with sea level rise and to contribute to the carbon sequestration properties of the coastal estuaries (Hopkinson, 2012; Callaway & Crooks, 2015).

Callaway and Crooks (2015), who researched carbon sequestration in San Francisco Baylands, assert that the accumulation of organic matter and carbon in tidal wetland soils is positively correlated with vertical accretion. In addition, they explain that saltmarsh soils can contain a range of organic matter by weight, depending on their salinity levels, with the greatest amounts of carbon in less saline saltmarshes. This indicates higher soil carbon content and density in freshwater or brackish wetlands rather than in the very saline saltmarshes. A brackish marsh can contain up to 60% organic matter in their soils, whereas a more saline marsh could contain about 20% organic matter (Callaway, et al., 2012).

Nearshore Ecosystem Services and Threats

Aside from being formidable carbon sinks, the shoreline ecosystems provide other indispensable services, including inland flood protection, erosion control, pollution attenuation, nutrient and sediment filtration, and vital habitat for benthic organisms, bivalves, crab, shrimps, forage fish, juvenile and mature salmon, and migratory birds (Belleveau, 2012; Smith et al., 2000; Janousek & Folger, 2014). The shoreline ecosystems are critical transition zones between freshwater, terrestrial, and marine environments (The Seattle District Corps of Engineers, 2016). Their ecosystem services are vital for all coastal regions' ecological, economic, and community well-being.

Despite the coasts' abundance of benefits, these ecosystems continue to be degraded and destroyed at an alarming rate due to human development and climate change threats (Dean et al., 2001). Like many other organisms, human populations have a tendency to live in, near, and around coastal estuarine areas. Fifteen of the world's nineteen megacities are located on estuaries, which can be attributed to the fact that estuaries in urban areas serve as important transportation corridors, recreation sites, and aquaculture/agriculture areas (Morley et al., 2012). This appeal has encouraged seemingly incessant construction of commercial and residential projects along the coastlines. To protect coastal urban developments from heavy tides or sea level rise, many shorelines are armored, diked, and/or modified through various practices such as dredging or filling (Simenstad, et al., 2005). These practices, however, have led to a succession of disconcerting complications in shoreline ecosystems and waterways worldwide (Morley et al., 2012).

Puget Sound Shoreline Modification Impacts: Erosion and Benthic Communities

The Puget Sound region in western Washington state is no exception to this pattern of urban development, as its population has doubled over the last three decades and is primarily concentrated around the coasts (Morley et al., 2012). The Puget Sound is a large inlet of the Pacific Ocean and the Strait of Juan de Fuca on the northwest portion of the state. It is also the second largest estuary in the United States (Washington State Department of Ecology, 2015). The region consists of a complex estuarine system of smaller inlets, rivers and water bodies, and its watershed drainage area, referred to as the Puget Sound Basin, is made up of nineteen river basins and covers over 16,988 square miles (The Seattle District Corps of Engineers, 2016; Mauger et al., 2015; Washington State Department of Ecology, 2015). Over the past decades, due to population growth and development, one third of the Puget Sound shoreline has been armored with structures like seawalls, dikes, or bulkheads (Morley et al., 2012), and as of 2014, 74.2% of Puget Sound saltmarshes and estuarine wetlands had been developed, destroyed, or degraded (The Seattle District Corps of Engineers, 2016). These impacts have resulted in the loss or impairment of an array of shoreline ecosystem services, habitats, and sharply diminished carbon storage properties (Hansen & Nestleroad, 2013).

Shoreline armoring can veritably produce more erosion complications than it solves. Shorelines and beaches throughout the Puget Sound are dependent on natural erosion and sediment transfer processes to develop and function properly (The Seattle District Corps of Engineers, 2016). These environments are fed by nutrients and sediment transported from neighboring shorelines via wave action and tidal regimes throughout the region. While seawalls or bulkheads may protect the immediate shore and uplands of one site, their influence can potentially increase erosion on neighboring beaches by shifting and altering natural tidal flows (WA Department of Fish and Wildlife, 2016). Furthermore, structures placed along shorelines isolate these areas from sediment deposition and/or natural erosion processes, leading to changes in the physical attributes of the shoreline, including the coarsening of beach sediment composition, and changes in beach slope and width (The Seattle District Corps of Engineers, 2016; Patrick et al., 2015).

Other concerning effects of shoreline armoring are the limitations imposed on the complex habitats of benthic organisms, burrowing aquatic macrofauna that are often used as indicators of marine health (Morley et al., 2012; WA Dept. of Ecology, 2013). The benthic community holds important roles in the shoreline ecosystem, including sediment stabilization, nutrient cycling, and prey to a diverse assortment of marine organisms (Smith, et al., 2000). Coastal armoring and the removal of the vegetation communities profoundly disturb crucial benthic habitats as well as the vital transition zones between terrestrial and freshwater environments (Morley et al., 2012). In addition, unnatural changes in sediment composition due to nearshore modifications and developments alter these habitats, which causes cascading impacts throughout the marine ecosystem and food web. Morley et al. (2012) examined ecological effects of shoreline armoring in the Duwamish River Estuary and found that armored sites contained only a fraction of benthic organisms compared to unarmored sites (Morley et al., 2013). Their research evaluated chum salmon eating patterns and found that salmon from unarmored sites consumed significantly more benthic organisms than those captured at armored sites, demonstrating the impact that armoring can have on the shoreline trophic cycle and local fisheries (Morley et al., 2012).

Benthic organism communities also have an impact on the global carbon cycle. Healthy rates and fluxes of carbon and other nutrients in shoreline systems rely upon the metabolism, feeding patterns, and burrowing habits of these tiny benthic organisms (Smith et al., 2000; Lebrato, et al., 2010). Hohn et al. (2008) and Lebrato et al. (2010) explain that when benthic filter-feeding organisms consume other marine organisms or die and decompose, they release carbonate, nitrogen, and other nutrients directly into the benthic environment. In their research on the global contribution of benthic organisms on the carbon cycle, Lebrato et al. (2015) thoroughly stress that the benthic carbonate-producing metabolic processes hold a crucial role in the global carbon budget, particularly due to their carbonate shells. The impact on the benthic community alone demonstrates how the disruption of even one small component of the complex marine ecosystem can critically impact sediment stability, the marine trophic web, and the global carbon cycle (Smith et al., 2000; Hohn et al., 2008; Lebrato et al., 2010; Lovelock et al., 2013; Masello, 2013).

The cumulative impacts of shoreline armoring, upland roads and/or marine area development can greatly overwhelm nearshore ecosystems, particularly the embayment shorelines, wetlands, and landforms (The Seattle District Corps of Engineers, 2016). Ecologically-conscious development strategies and restoration projects should be implemented in order to salvage intertidal shoreline services, but land managers must also contend with climate change. The following section will discuss predicted climate change and sea level rise (SLR) concerns in the Puget Sound.

SLR and Climate Change Threats In the Puget Sound

In addition to shoreline armoring, Puget Sound estuaries face another imminent threat. Due to climate change, predicted sea level rise (SLR) during the 21^{st} century is forecasted to significantly endanger many coastal ecosystems (IPCC, 2014; Mauger et al., 2015). SLR, if left unchecked and unmitigated, could lead to the accelerated destruction of coastal estuaries, wetlands, and saltmarshes (Smith et al., 2000, Mauger et al., 2015). The global average sea level has risen at a mean rate of 1.7 mm/yr, or about 8 inches in the past 100 years, a greater rate than that of the past 2,000 years (IPCC, 2014; Mauger et al., 2015). Ordinarily, an intertidal saltmarsh could adapt to SLR through continual vertical accretion carried out by tidal influence (Callaway & Crooks, 2015). However, the predictions for SLR suggest that urban and industrial landowners will continue to armor their shorelines to protect their properties, prolonging the cycle of estuarine degradation (Morley et al., 2012). This armoring practice will only be exacerbated by the growth of human populations and increases in coastal real estate prices (Smith et al., 2000).

Nevertheless, not all coastal areas will be impacted by SLR. The Puget Sound is prone to varying tectonic movements, creating slow elevation changes along the Cascadia Subduction Zone (Mauger et al., 2015). This means that some areas in the Puget Sound will be slowly rising, while others will be slowly sinking. Those areas that are rising in elevation might not experience the full effects of the projected SLR, but those that are sinking will experience its effects much more. For example, a tidal gauge in Seattle has recorded a sea level rise of 8.6 inches since 1900, while a gauge in Neah Bay, Washington has recorded a drop in sea level of 5.6 inches since 1934 (Mauger et al., 2015). These tectonic trends and the progress of SLR suggest that the sea levels will rise 14-54 inches in the Puget Sound region by the year 2100, depending on the location in relation to vertical tectonic movement (Mauger et al., 2015).

As sea levels rise and/or fall, coastal wetlands will very likely continue to degrade, inevitably leading to disastrous outcomes if this positive feedback loop progresses (Mauger et al., 2015). Coastal river flooding regimes could be permanently affected, increasing the frequency, likelihood, extent, and duration of flood events and their associated hazards (Mauger et al., 2015). These changes would lead to significantly higher vulnerability of coastal habitats and higher intensity of eroding effects on unprotected beaches and bluffs (Mauger et al., 2015).

As greenhouse gas emissions continue to enter the planet's atmosphere, climate change could continue to impact the culture and economy of Puget Sound communities through changing temperature, precipitation variations, sea level rise, eutrophication, ocean salinity and acidification, and flooding (Mauger et al., 2015). These changes are predicted to influence unusual patterns in snowpack, timing of biological events, weather patterns, species distributions, agriculture, aquaculture, and ecosystems, in turn affecting the patterns of freshwater discharge into the shoreline habitats (Weston et al., 2014; Mauger et al., 2015). Due to these climate issues, as well as rapid nearshore development and degradation, it is imperative to rehabilitate and conserve these ecosystems wherever possible.

2.2 SIGNIFICANCE OF VEGETATION COMMUNITIES, SOIL SALINITY, AND CARBON RELATIONSHIPS FOR NEARSHORE RESTORATION TARGETS

Carbon concentration, soil salinity, tidal inundation regimes, and vegetation growth patterns are all interconnected elements in coastal ecosystems. A delicate balance is necessary for the proper functioning of the estuarine habitat system (Sheng et al., 2015). However, human development and sea-level rise have been influencing the many integrated elements of all natural systems (Weston et al., 2014). In the case of shoreline ecosystems, future climate changes are predicted to disrupt factors like flooding patterns, salinity levels, and carbon fluxes, which are then likely to lead to unsustainable changes in vegetation and microbial communities (Weston et al., 2014). Butzeck et al. (2015) assert that it could be instrumental to develop multivariate models to assess temporal and spatial variation as variables to anticipate soil deposition rates and the other complex variables in nearshore marshes. The following sections will cover the unique interactions and restoration implications between vegetation, inundation regimes, and carbon and salinity fluxes in saltmarsh soils.

Saltmarsh Vegetation and Soil Associations

Healthy vegetation communities fulfill a vital role in the entire coastal habitat. If functioning properly, a productive plant community should perform a range of interrelated services that contribute to the structure, habitat, and chemical fluxes of the ecosystem. Carbon cycling through vegetation growth, decay, and sequestration has a considerable influence on the entire saltmarsh habitat (Callaway & Crooks, 2015). Understanding the vegetation growth patterns along environmental gradients is a key component of working in nearshore restoration and conservation. This section describes the unique group of plants that make up a saltmarsh habitat, their role in the structure and development of saltmarshes, and their role in the carbon flux patterns.

Tidal saltmarsh plant communities are populated with halophytes, plants that have evolved mechanisms to be tolerant of and thrive in salty environments (Crain et al., 2004; Silvestri et al., 2004). The physiology of halophytes makes these plants quite extraordinary. For non-halophyte vegetation, too much salt can interfere with the plant's ability to uptake nutrients, in turn limiting growth and gains in biomass, and ultimately leading to cell death (Belleveau, 2012; Joshi et al., 2015). In fact, Joshi et al. (2015) go so far as to posit that out of all toxic substances, salt is the one that most restricts plant growth. However, halophyte capacity to withstand high salinity environments allows for them to hold important roles in the saltmarsh. For instance, their root systems stabilize the soil, which boosts erosion control and allows for finer tidal sediments, specifically clay and silt particles, to settle on the wetland floor (Butzeck et al., 2015). The plants themselves also reduce the energy of the tidal flows and deter the remobilization of captured sediment. Regarding carbon sequestration, the halophytes absorb CO_2 during photosynthesis and it is then stored in matted plant matter, broken down, buried, and preserved in the saline soils (Hussein et al., 2004). This vegetation-soil relationship illustrates how healthy halophyte communities in a saltmarsh habitat are fundamental to the success of effective carbon sequestration properties and erosion control.

Studies have shown that clay and silt content tends to be positively correlated with carbon content, as individual fine grains provide greater specific surface area for which carbon to adhere (Krull et al., 2001). Moreover, the presence of varying surface minerals (particularly Ca, Fe, and Al multivalent cations) on the clay particles is an important mechanism for determining how organic carbon bonds onto the clay particles (Krull et al., 2001). Furthermore, the smaller silt and clay sediments are less susceptible to disturbance than larger particles, such as sand or gravel (Crooks et al., 2002; Masello, 2012; Mao et al., 2014).

Janousek & Folger (2014), in their studies of Pacific Northwest tidelands in Oregon, determined that certain abiotic variables are positively related to the distribution pattern of saltmarsh plant species. Their research suggested that the abiotic characteristics of salinity, elevation, tidal inundation, and soil composition have a dominant role in determining the saltmarsh species composition. The configuration of these characteristics leads to spatial gradients in the percent cover and species composition throughout a tidal wetland site (Janousek & Folger, 2014). For instance, salinity was the most important variable for five of the twelve most recurrent species. Elevation was the most important for the *Hordeum* grass species and the *Atriplex* saltbush. Clay content did not appear to have a strong relationship with most plant occurrence, except for a *Deschampsia* grass and *Carex lyngbyei* sedge (Janousek & Folger, 2014). However, both soil salinity and clay content together were positively correlated with species richness (Janousek & Folger, 2014). The following sections will go into further detail about these soil variables, as they perform essential duties in the distribution of vegetation communities and shoreline ecosystem services.

Soil Salinity

As explained in the prior section, macrophyte distributions and growth patterns are strongly related to salinity levels in soils. Healthy salinity levels are essential for a functional saltmarsh vegetation community (Howard et al., 2008; Janousek & Folger, 2014). In fact, Howard et al. (2008) explain that soil salinity levels are more important to phenotypic variation of saltmarsh halophytes than soil organic matter content. However, depending on species, halophyte fitness can be inhibited and can develop reduced plant root activity, photosynthesis rates, and biomass if salinity levels are too high (Sheng et al., 2015). Crain et al. (2004) explain that the lack of understanding this salinity gradient/range is a major gap in coastal wetland knowledge. Therefore, investigating the processes of vegetation distributions along salinity gradients is critical for expanding our knowledge pool and for performing effective restoration practices (Crain et al., 2004). This information could help researchers and land managers to accurately anticipate ecosystemic reactions in estuarine habitats resulting from land use change, sea level rise, and coastal development.

Soil salinity levels throughout tidal marshes can fluctuate depending on variables such as their proximity to freshwater sources, elevation, and tidal influence (Crain et al., 2004). Elevation has negative correlation with soil salinity. When the tidewater flows into a saltmarsh, it seeps into the soils, bringing with it an influx of salt. The tidal flows cannot reach the higher elevation areas, which in turn establishes a salinity gradient along the elevation slope, leading to a negative correlation between the two (Veenklaas et al., 2015). The relationship between soil salinity and elevation has a significant influence on the zonation patterns of the plant communities of saltmarsh environments (Silvestre et al., 2005; Veenklaas et al., 2015).

The effect of salinity on the carbon cycle is subject to some uncertainty, since it can have varying effects on the estuarine carbon cycle. Qiang Sheng et al. (2015; pg. 543) explains that carbon cycling patterns in wetlands can be complex due to "…the interactions between salinity, plant type, productivity, and the water table." They found that nearshore wetlands emit negligible amounts of methane (CH_4) into the atmosphere, and that the methane and CO_2 emissions are significantly diminished by increased salinity (Sheng et al., 2015). They suggest that this likely due to inhibited microbial activity in higher salinity concentrations (>10%) (Sheng et al., 2015). The Sheng et al. (2015) findings demonstrated that carbon emissions from a restoration site were highest if there was low salinity and no vegetation. However, low salinity sites *with* vegetation had greater carbon absorption capacity than the other experimental sites (Sheng et al., 2015). Essentially, their findings boiled down to the assertion that the saltmarsh vegetation productivity is important for balancing wetland salinity and carbon fluxes, and that there is a positive correlation between macrophyte productivity and carbon absorption (Sheng et al., 2015).

Inundation Fluctuation and Elevation

Soil salinity levels, carbon concentration, sediment deposition, and vegetation communities are all elements of intertidal saltmarshes that are contingent on tidal fluxes and an elevation gradient. Butzeck et al. (2015) explain how sediment deposition rates vary between marsh types (e.g. brackish, salt, or freshwater), low to high elevations, and the annual cycle of the seasons. In their study of accretion rates along flooding gradients in the Elbe Estuary in Germany, they found that the suspended sediment concentrations of the tidal flows were higher by three to four times during the autumn and winter seasons than those of the spring and summer (Butzeck et al., 2015). They also found that the sediment deposition rates (SDR) more than doubled during the autumn and winter seasons (Butzeck et al., 2015). They explain that this increased SDR is likely due to the stormy season bringing increased precipitation events, higher tidal inundations, and concurrently lower sediment stability from increased wave erosion from neighboring shorelines and feeder bluffs (Butzeck et al., 2015; Seattle District Corps of Engineers, 2016). This same elevation gradient also accounts for the nearshore salinity gradient, as it influences the salinity recharge length of saltwater or brackish tidal inundations on the varying elevations of the saltmarsh sediments (Butzeck et al., 2015; Weston et al., 2014).

The elevation gradient on intertidal marshes serves as the driver for inundation frequencies, which in turn dictates vegetation zonation, sediment deposition rates, and soil carbon sequestration levels (Crooks et al., 2002). Butzeck et al. (2015) found that lower elevation marsh zones, unsurprisingly, have significantly longer inundation times than the higher zones. Suspended sediment concentration (SSC) of the tidal flows also shared a similar pattern, with higher SSC in the lower marsh compared to the higher

(Butzeck et al., 2015), likely due to sediments accumulating in the lower elevation marsh vegetation. These patterns provide a reliable explanation for the significantly higher SDR rates found in the lower marshes than those of the higher zones, due to the continual settling of silt and clay particles during "upmarsh" tidal advancement (Butzeck et al., 2015). Moreover, Butzeck et al. explain that the vegetation biomass has a large influence on soil deposition, as is captures the smaller sediments in its roots and plant matter.

One other factor that Butzeck et al. (2015) included in their analysis was the range of soil bulk density in their elevation gradient. Slightly higher bulk density was observed in the lower zone of the saltmarsh as opposed to the higher zone (Butzeck et al., 2015) (1.17 g/m^3 verses 1.03 g/m^3, respectively) (Butzeck et al., 2015). Their number also demonstrated an increase in organic carbon with a decrease of bulk density (Butzeck et al., 2015).

These elements are important to consider when planning for productive saltmarsh vegetation communities. Crooks et al. (2002) explain that the development and restoration of a successful saltmarsh habitat after a shoreline armoring breach is dependent on the elevation of the newly inundated marshland. As halophyte assemblages begin to develop, their biomass will facilitate further sediment deposition, carbon sequestration rates, and healthy salinity levels (Butzeck et al., 2015; Weston et al., 2014). Understanding these complex interactions is crucial for effective restoration work, as ineffectual changes in elevation and inundation rates can directly diminish carbon storage processes and habitat services, and lead to unsuccessful results.

2.3 RESTORATION OF PUGET SOUND SHORELINES

Restoration Ecology is the scientific discipline that concentrates on determining the most effective approaches towards rehabilitating disturbed ecosystems. The Society for Ecological Restoration describes restoration as “an intentional activity that initiates or accelerates the recovery of a degraded ecosystem with respect to its health, integrity, and sustainability” (SER, 2002). Due to the increased understanding of the high value of the ecosystem mechanisms, services, and resources of Puget Sound nearshore areas, recovery projects have become more attractive to land managers and to Puget Sound communities (Simenstad, 2005). However, opportunities for restoration work are often limited, as many of these urbanized estuaries are specially zoned public properties or on valuable private or commercial parcels (Simenstad, 2005). Feasibility for a successful rehabilitation is further complicated by persistent stressors, such as urban chemical pollutants, invasive plant and animal species, and light and noise pollution (Simenstad, 2005). These complications can lead to prohibitive costs and setbacks in the restoration process, making it difficult to assure prospective investors and to obtain resilient restoration sites.

Government agencies, non-profit organizations, scientists, and citizen volunteers work vigilantly to create, restore, preserve, and manage nearshore ecosystems along the Washington State coast to prevent coastal wetland degradation (Dean et al., 2001; Belleveau, 2012; Masello, 2013; Capitol Land Trust, 2015). Federal, state, and local legislation has been authorized to instigate and encourage nearshore restoration work (Seattle District Corps of Engineers, 2016). The following section will outline and discuss some of the legal parameters concerning Puget Sound nearshore restoration, the advantages and pitfalls of various shoreline restoration approaches, and the diversity of management practices used to assess and monitor the progress of rehabilitation projects.

Puget Sound Coastal Restoration Legislation and Stakeholders

Since Euro-American colonization, Washington's ecosystems have been facing environmental difficulties that threaten the state's diverse coastal resources. These dilemmas act as drivers for the state's nearshore restoration projects. The most prominent of these struggles include dwindling salmon populations, loss of shoreline ecosystem services and critical habitat, degraded recreational sites, and polluted waterways. Laws have been enacted at the federal, local, and tribal level to encourage restoration efforts and provide funding for natural resource agencies, ecologically oriented nonprofit organizations, and resource conservation partnerships.

The following legislations are perhaps of the most notable in regards to shoreline restoration. At the federal level, the Clean Water Act (CWA) was the first major law enacted to address water pollution issues. It mandates that the Environmental Protection Agency (EPA) implement water pollution control programs in every state. Through the CWA, the EPA awards funding to state and municipal governments to comply with their established regulations. These regulations make it illegal to dump pollutants into waterways unless the stakeholder has obtained a proper permit. The EPA must also comply with the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), also known as Superfund. This CERCLA program identifies areas of toxic contamination, identifies the responsible parties, and ensures that they clean their toxic messes. The Endangered Species Act (ESA) is another law which ensures that resource agencies throughout the country maintain and manage endangered species populations and their habitats. The ESA is administered by the National Oceanic and Atmospheric Administration (NOAA) and by the US Fish and Wildlife Service (USFWS).

The NOAA National Marine Fisheries Service (NMFS) also plays a large role in Washington restoration efforts. NOAA NMFS manages the Pacific Coastal Salmon Recovery Fund, which provides grants to state and tribal agencies for habitat restoration efforts and to remain in compliance with ESA regulations.

One important state agency is the Washington State Recreation and Conservation Office. This agency retains the Salmon Recovery Funding Board, which provides grants to agencies and organizations involved with a variety of salmon recovery programs across the state. These grants are awarded to restoration, planning, monitoring, hatchery programs, and land acquisition projects (WA State Recreation and Conservation Office, 2016).

These federal and state entities are major contributors to state and local restoration efforts. In most cases, restoration projects require a number of partnerships. The federal agencies provide significant funding, logistical support, and professional assistance to state agencies, non-governmental organizations and community members to realize the goal of successful resource management and restoration.

Restoration Targets and Practices

Every restoration site has a unique ecosystem, history, and geography. These intricacies can make it complicated for land managers to determine suitable targets for restoration. For many land managers, a return to pre-colonization conditions is the standard target for these sites (Thorpe & Stanley, 2011). This entails the establishment of native plant communities, based on inferences derived from remnant populations. However, this approach can lead to complications considering the uncertainty of historic conditions and that the ecosystem and land use have changed dramatically since historic times (Thorpe & Stanley, 2011).

For the Puget Sound, many direct changes to its shoreline ecosystems have been established since Euro-American settlement. The most evident of these include tidal barriers, altered sediment and nutrient cycling patterns, the degradation of river deltas, and an alarming loss of nearshore wetlands (Seattle District Corps of Engineers, 2016). To address the increasing habitat loss, citizen scientists and government agencies have come together to research the problems and identify strategies for restoration and management (Seattle District Corps of Engineers, 2016). Procedures essential for beginning the restoration process include: extensive planning, organizing, determining funding sources, and studying the laws and regulations.

For Puget Sound nearshore environments, one of the leading restoration goals is to recover critical habitat for avian and marine species, particularly salmon. Once restoration opportunities and objectives are identified and the partners are organized, the process of restoration can begin. For degraded shoreline ecosystems, these practices can include the establishment of native vegetation and a riparian buffer, the breaching or removal of tidal barriers, stabilization of soil, and treatment of invasive plant species. Understanding the chemical fluxes and environmental gradients of a healthy site should inform land managers of the appropriate geography conducive to a productive ecosystem.

A relevant example of this comprehensive approach to shoreline restoration is the massive Duwamish Estuary restoration project in Seattle (Simenstad et al., 2005). In 1986, only 2% of this estuary's historic wetlands remained due to coastal development (Simenstad et al., 2005). Roughly 75% of the estuary's historic flow had been diverted for various development projects (Simenstad et al., 2005). The estuary was contaminated with a medley of toxic wastes, and the area was not suitable for any marine wildlife. Small recovery projects began in 1988 (Simenstad et al., 2005). Over time, through pressure from the community, assistance from government programs, and many partnerships, funding was provided and efforts were made to restore the site to a relatively functional state. Freshwater flows were returned to the estuary, tidal barriers were removed, marshes were restored, and native vegetation was planted throughout (Simenstad et al., 2005). This is not to say that the entire estuary was restored to historic condition, as a true restoration is not possible in highly disturbed areas. However, a continual rehabilitation is sometimes all that can be done, given the complex network of constraints. The Duwamish remains a Superfund (Comprehensive Environmental Response, Compensation, and Liability Act, or CERCLA) site today; a title given to the most heavily polluted sites in the U.S. Nevertheless, even a partial restoration and/or rehabilitation is valuable for improved ecosystem services and wildlife habitat.

Monitoring, Assessment, and Education

To ensure that a restored site is functioning properly, a scientific monitoring protocol should be established. Depending on the site, monitoring programs can be administered by various agencies, consultants, or even citizen scientists. Variables can be assessed to develop a comprehensive understanding of the fitness of a restored shoreline. These can include vegetation cover and diversity, presence of invasive species or of certain indicator organisms, forage fish surveys, and soil and water quality testing. Keeping a record of these variables can help land managers track of the progress of the project, ensure they are within compliance of legal regulations, and make informed predictions and management decisions.

The Duwamish Estuary restoration began as a small action with a single authority initiation (Simenstad et al., 2005). Eventually, the project grew into a wide assembly of local and regional partnerships. As community members became more informed and aware of the significance and ramifications of shoreline preservation/rehabilitation, the restoration movement expanded (Simenstad et al., 2005). This process demonstrates how awareness campaigns can be implemented around the community to keep people updated and enthusiastic about new developments. Regular habitat monitoring, both scientific and citizen volunteer monitoring, contributes to the information pool, allowing for interested parties to grasp tangible data about their local restoration endeavors. Developing the concept that an urban area can coexist with a healthy estuary could potentially inspire other communities to conduct similar restoration ventures and partnerships (Simenstad et al., 2005).

2.4 SITE: BAYSHORE PRESERVE

An excellent example of shoreline restoration is the Bayshore Preserve in Shelton, Washington, owned and managed by Capitol Land Trust (CLT). CLT purchases and protects natural areas throughout Washington State (Capitol Land Trust, 2015). The

Bayshore Preserve (Figure 2.1) was added to their portfolio of conserved lands in 2014. This area had been a 74-acre golf course since 1930, was purchased by the CLT in 2014, and converted into a nature preserve (Capitol Land Trust, 2015). The preserve is on a peninsula on the western coast of Oakland Bay. The Bayshore Preserve contains roughly 4,000 feet of shoreline, 27 acres of pristine saltmarsh habitat, and 47 acres of newly established native habitat that once constituted the golf course (Capitol Land Trust, 2015).

In order to restore tidal influence to the shoreline, CLT removed a decades-old 1,400-foot tidal dike from the shoreline, providing tidal access to an area that was previously blocked from intertidal flows (Jolivette, 2014). This dike was originally created to protect the golf course from tidal influence, but it also provoked the degradation of the saltmarsh habitat. The CLT and Mason Conservation District worked together to create a series of tidal channels and basins on the western shoreline to create additional tidelands and increase the incursion of bay waters into the peninsula (Jolivette et al., 2014; Capitol Land Trust, 2015). Basins were excavated in the slightly more upland area of the shoreline, and thinner channels were created to connect them to the tide flows. They were excavated using a hydraulic excavator, and once completed, had only a sand/gravel bed. These channels and basins were not deliberately seeded, and in the time since their creation, a layer of fine sediment and organic matter had been deposited with the tide flows, allowing for vegetation communities to establish themselves in these areas. A riparian buffer of native plants was planted around the perimeters of the saltmarsh basins to create a defense between upland disturbances and the fragile saltmarsh vegetation communities. The buffer plants include Oregon Grape, *Rosa spp.*, snowberry, cottonwood, and willows.

In their 2016-2020 strategic plan, Capitol Land Trust's long term vision for their projects are to conserve marine shorelines and estuaries, maintain water quality, and improve salmon habitat (CLT, 2016). They plan to engage communities and stakeholders by providing public access to the Bayshore property and by carrying out stewardship projects and educational programs for any interested parties (CLT, 2016). They have partnered with local schools, resource companies and agencies, and conservation groups. Volunteers and both public and private funds have allowed CLT to conduct some monitoring throughout the site.

The Bayshore Preserve serves as a prime example of a rewarding restoration project. By allowing tidal access to return to the shoreline and by creating new tidal basins, the Capitol Land Trust and their partners have taken a large stride towards rehabilitating a healthy habitat for marine, avian, and plant species (Capitol Land Trust, 2015). Due to continual tidal influence and sediment deposition, this site could also serve as an effective carbon storage site, whilst playing a small, yet powerful, role in the global carbon cycle. To maintain a record of the progress of the new saltmarsh habitat, it could benefit the CLT to regularly monitor some of the saltmarsh attributes. Some post dike-removal programs have already been implemented by both CLT and Mason Conservation District, including annual monitoring of vegetation changes post-dike removal, monitoring of basins for effective tide fluxes, monitoring for large woody debris, and protection of basin side slopes. Some other potential variables to record could include sediment accretion rates and carbon concentration levels. Furthermore, recording soil salinity levels and vegetation characteristics directly in the saltmarsh could prove useful for their monitoring assessments. Shoreline restoration is an important investment that benefits entire communities through ecosystem services, recreation opportunities, and natural resources. If a restored nearshore saltmarsh appears to have strong carbon storage potential, its success can reinforce the case for further shoreline conservation, rehabilitation, and investment.

This study examines the relationship between soils attributes (including texture and carbon content), elevation and percent vegetation cover in the newly created tidal basins. The research focuses mostly on revealing whether the newly created tidal basins in Bayshore Preserve are effective for collecting fine sediment material, and how this variable relates to elevation and/or soil salinity. To uncover this information, soil salinity, carbon levels, soil texture, and percent vegetation cover were measured in the saltmarsh basins to determine the status of these characteristics and their relationships. Collecting numeric data about these parameters may provide some tangible insight concerning the progress and health of the saltmarsh basin habitat, relative to similar established saltmarshes.

Figure 2.1 *Bayshore Preserve, Shelton, WA*

3. Methods

3.1 Methods

Transect and Plot Design

Seven basins of varying sizes and their respective tidal channels were excavated on the west side of the removed tidal dike into the upland area of the Bayshore peninsula. Each basin is connected to Oakland Bay through excavated channels that allow for tidal waters to flow into them. These basins were created to establish extra estuary and saltmarsh habitat in the shoreline area of the site post-dike removal and to increase tidal exchange. Of the seven new basins, the four longest were selected to examine elevation and vegetation cover, and to extract soil samples to test for salinity, carbon content, texture, and bulk density. These are Basin 1, Basin 6.1, Basin 6.2, and Basin 7 (Fig. 3.1). These basins were selected because their large size made them more likely to be representative of the whole group, and their length allows for establishment of long transects on each. By being physically longer, the transects cover more area than if placed in smaller basins, allowing for more heterogeneity to sample along a tidal range and elevation gradient (near shore to further inland). Basins ranged from between approximately 76-94 meters long and 12-22 meters wide.

One transect was established along the bottoms of each of these four basins, parallel to the basin side slopes. Each transect was four feet offset from the basin center. The Basins 1 and 6.1 transects were created in November 2016, whereas the transects in Basins 6.2 and 7 were placed in March 2017. The soil samples and the percent vegetation cover data were collected in April 2017.

Plots were established on all four transects, and were placed 15.25 meters apart from each other to yield a total of 4 or 5 plots per basin. Transect 1 has five plots, Transect 2 has four plots, Transect 3 has four plots, and Transect 4 has five plots making a total of 18 plots. Each 1 x 1 m plot was marked with a labeled 2" PVC pipe hammered into the ground to a depth of a ~30 cm. The first plot of each transect is randomly located within the first two meters of the transect closest to the shoreline and the rest of the plots were plotted 15.25 meters (50 feet) apart from each other.

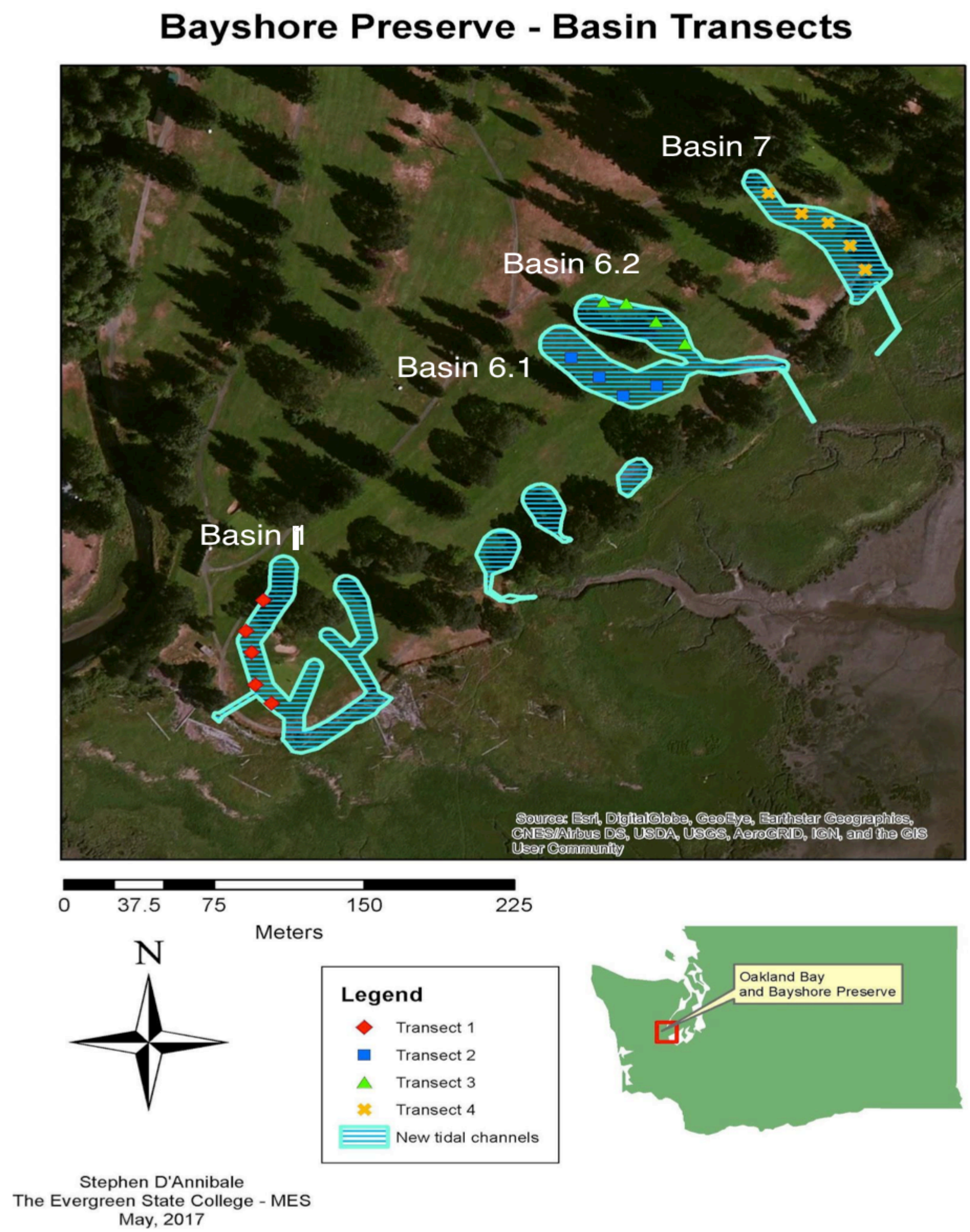

Figure 3.1 *Bayshore with Transects*

Elevation

A TopCon level scope was used to collect the elevation data. The base elevation measurements come from elevation survey tack markers left by Mason Conservation District land surveyors. Beginning with the known elevation from this point, an individual can collect accurate elevation numbers of each plot in feet (later converted to meters) with the instrument.

The procedure is as follows. First, the instrument and tripod was located near the plots and near a marker of known elevation. The surveyor with the stadia rod stood at the marker and the surveyor at the leveled scope read the height from the rod. This number added to the known elevation of the survey tack provides the elevation of the scope. From there, the surveyor handling the rod moved to the individual plots. Any debris from the plots was gently cleared so as not to disturb vegetation and/or the soil surface. This was to ensure an accurate reading with the stadia rod and scope. The survey rod is placed on the plot center. The surveyor at the scope reads the measurement on the rod. This number was then subtracted from the height of the instrument to obtain the precise elevation of each plot. Once the elevation of one plot was known, the stadia rod was moved to the next plot and the process repeated. This data was recorded on the datasheet.

Soil Collection

Three sets of soil samples were collected from the basins. The first set of samples was for soil bulk density. The second set was specifically for salinity, carbon, and particle size analysis. The third set was as base/foundation samples to obtain a rough approximation of the basins' soil attributes before tidal influence. The methods for this third set are elaborated upon in the following paragraphs.

For the first two sample sets, four sub-samples of soil were excavated with a hand trowel from small pits and collected from each outer border side or corner of the plot down to a depth of 15 cm. Three of these sub-samples were combined and used to measure soil salinity, carbon concentration, and texture (Janousek & Folger, 2014). This method was used in a similar study conducted by Janousek & Folger (2014), where the plot core samples were homogenized and used to extract salinity and nutrient measurements. The fourth sub-sample and pit from the research plot was used to determine bulk density (described below).

Ideally, one would use a core sampler to extract bulk density samples, particularly to obtain an accurate measure of volume for bulk density. However, the soil substrate is coarse and gravelly, rendering the core soil sampler ineffective. Instead, to obtain volume data for a sample, a different method called the volume excavation technique was implemented (Lichter & Costello, 1994). This method is described in a later section.

The literature reveals some variations in the depths and diameters of soil sampling pits, depending on the study objectives and characteristics of the sites. A number of wetland and bulk density studies extracted soil samples from a depth range between 10 – 30 cm (Lichter & Costello, 1994; Franklin et al., 2013; Hansen and Nestlerode, 2013; Masello, 2013; Sheng et al, 2015). Tom Terry, volunteer soil scientist for Capitol Land Trust, has extracted previous soil samples at Bayshore Preserve at a 15 cm depth (personal communication). For the sake of consistency, this study used soil-sampling pits at a 15 cm depth and diameter. This depth is appropriate on this site for three reasons: 1) the soil at the bottom of the basin was just recently exposed by the excavation to build the channel, thus there is little to no O or A horizons; 2) the rock content makes sampling at a deeper depth very difficult; and 3) any changes in soil organic matter the first year are most likely to occur within this depth.

The three subplot salinity/carbon samples were dug out using a trowel, were broken up and combined in a bucket with a trowel to get a uniform sample for each plot. Coarse plant matter, detritus and rocks were removed with a coarse sieve (2 mm) in the field and discarded. The combined sample was subdivided into Ziploc bags used for salinity measurements, carbon content, and texture. These were labeled with transect and plot identification numbers and placed in a chilled cooler for transport (Franklin et al., 2014). The leftover combined soil material in the bucket was returned to the sample pits. The bucket and trowel were rinsed between sampling plots. This method was applied to each plot of the four transects.

Once back in laboratory, the samples were placed in a refrigerator in The Evergreen State College laboratory for storage. Soon after, the samples were oven dried at 60 °C for 12-24 hours and stored in new, clean Ziploc bags, labeled, and stored in a cool, dry place until taken out for laboratory analyses (Lichter & Costello, 1994; Pan et al., 2015; Masello, 2013). Each dried sample was then subdivided into three separate subsamples and stored in 150 mL Nalgene bottles. One sub-sample was used to measure organic carbon, one was used for salinity, and one was for soil texture analysis.

*Soil Bulk Densi*ty

An accurate measure of soil bulk density is necessary to procure a reliable estimate of carbon stocks in the saltmarsh tidal basins. Bulk density is the dry weight of a soil sample divided by its volume, and is measured in grams per cubic centimeter (g/cm^3) (Lichter & Costello, 1994). In order to obtain an accurate bulk density measurement, a precise measure of volume is needed. Studies often use the soil core method to obtain an accurate volume measurement (Lichter & Costello, 1994; Masello, 2013; Pan, et al, 2014; Beaven, 2015). This method is measured as the volume of the coring tool minus the volume of 2 mm diameter rock fragment. However, the soil substrate of the Bayshore saltmarsh basins is quite gravelly and has significant amounts of coarse rock. This obstructs the penetration of the corer.

To circumvent this issue, the volume excavation technique was used. Though it is a more labor-intensive process than the core method, it does offer the benefits of being a low-cost alternative as well as retaining higher accuracy and flexibility in regards to sample volume (Pan et al., 2014; Lichter & Costello, 1994). At each plot, either one of the outer sides or one of the corners of the quadrat was randomly selected to collect the soil bulk density sample measurement. A small pit was dug at the center point of the selected outer side or corner. The pits were 15 cm in diameter and 15 cm deep. All of the soil and rocks from this pit were carefully placed in a plastic bag, labeled, and stored in a container until transported to the lab for oven drying. Next, the pit was lined with a plastic liner. The lined pit was then filled flush to the surface with sand from a graduated cylinder. The volume of the sand used to fill the pit was used to approximate the volume of the soil pit, which was recorded in a field notebook or datasheet (Lichter & Costello, 1994).

At the laboratory, the samples were dried at 105°C for 24 hours (Lichter &

Costello, 1994; Pan et al., 2015). The soil sample was then sieved through a 2 mm sieve. The rock/gravel extracted through the sieving processes was set aside and later measured for volume. The dry weight of the soil was recorded and the sample set aside and stored.

The total volume of the sample soil pit must be corrected for the rock and gravel fragments. To determine the volume of the rocks from the sample pit, the rocks were placed in a graduated beaker of water. The amount of water that is displaced by the rocks is their volume. This number was recorded and then subtracted from the original pit volume measurement. The dry weight of the sample was then divided by volume of the sample pit in order to obtain the g/cm^3 bulk density measurement unit (Lichter & Costello, 1994).

The following equation is used to calculate soil bulk density, where M_s is the oven dry-mass of soil minus rocks (in grams), and V_s is the sample volume of the pit (cm^3) minus the volume of the rock in the sample, respectively. The unit for bulk density is grams of soil ≤ 2 mm divided by cm^3 (g/cm^3) of soil (minus the rocks) (Pan et al., 2015).

$$Soil\ bulk\ density\ (g/cm^3) = \frac{(M_s)}{V_S}$$

Vegetation Percent Cover

To determine a relationship between vegetation cover and abiotic factors, such as elevation, soil salinity and soil carbon, data for percent cover of total vegetation was gathered. This collection was conducted in mid-April as the saltmarsh vegetation is beginning to grow. At each plot, a 1 m x 1 m quadrat is placed. For sampling consistency, the quadrat was placed facing the cardinal directions. Percent cover of the present vegetation in the quadrat was visually estimated and recorded the datasheet.

Soil Salinity

Soil salinity was measured with a Hanna HI 9813-6 electro-conductivity meter (in units of millisiemens/cm). Electrical conductivity is generally the measurement used to determine salinity levels. A 1:5 soil:deionized water (weight:volume) solution was made with the soil samples and deionized water at room temperature. This solution was then stirred every five minutes for a period of thirty minutes. After letting the sediments settle in the container (after one hour), the solution is then poured through medium filter paper.

The Hanna meter was properly calibrated for solution temperature and conductivity of a known salt concentration following the Hanna protocols and using appropriate cleaning and calibration solutions (Hanna HI70031 1413 μS/cm calibration solution for salinity). Salinity readings can fluctuate with temperature changes, so it is necessary to calibrate it to obtain accurate measurement. The probe was then placed into the filtered soil solution and the measurements in mS/cm (millisiemens) were recorded. mS/cm was then converted to ppt using the following equation: 1 mS/Cm = 640 ppm.

Carbon Content

Carbon content was measured using the PerkinElmer Carbon Hydrogen, Nitrogen (CHN) Analyzer at The Evergreen State College laboratory. This instrument combusts the samples and analyzes and quantifies the combusted elements using a thermal conductivity detector (PerkinElmer, 2011).

The soil samples to be analyzed were oven-dried at 105 °F for 24 hours, sieved with a 2 mm sieve, and the coarse material and roots were set aside. The soil was ground and homogenized using a mortar and pestle (Beecher et al., 2001). 3-5 milligrams of each sample were placed in small aluminum cups designed especially for use with the CHN analyzer. These were weighed with an autobalance, folded, and stored until ready to run the instrument. Several replicates were prepared, as well as several blanks, K-factors, and standards to ensure the accuracy of the measurements. A blank is a "sample" that does not actually hold any of the sample: in this case, an empty aluminum cup. The CHN analyzer uses the K-factors, an Acetanilide standard, to maintain accuracy with its analyses.

Once the carbon content per sample is quantified, the data is recorded as weight percent carbon per mass of sample (5% C means 5 % C by weight).

Carbon density of the top 15 cm of the saltmarsh basin bottoms is calculated as percent carbon per sample soil multiplied by bulk density (Masello, 2013).

$$C\ Density\ (g/cm^3) = C\%\ x\ Bulk\ Density\ (g/cm^3)$$

Soil Texture

In order to determine soil texture, Bouyoucos hydrometer method was used as explained by Lesikar et al. in their Soil Particle Analysis Procedure (Lesikar et al., 2005). This method analyzes grams of suspended particles per liter of water/soil solution. This method was used instead of the hand texturing method for several reasons. First, it is a simple and relatively fast approach to collecting accurate readings of clay and silt content of a sample (Lesikar et al., 2005). Also, high salinity levels in the soil can lead to problems when hand-texturing, as salts can reduce stickiness of clay and contribute to underestimations of clay content (Lesikar et al., 2005). Also, since the soils are likely to be mostly sand, it could be difficult to determine percent clay by feel alone.

The hydrometer method measures the suspended solids of a soil and deionized (DI) water solution after heavier particles sink to the bottom of the container. For the analysis, a soil sample of 100 grams was mixed with a 100 mL 5% sodium hexametaphosphate DI water solution in a dispersion cup. The sodium hexametaphosphate is a dispersant agent, which is added to the sample in the container. This is used so that the clay and/or silt particles do not bind to each other and settle to the bottom when shaken (Lesikar et al., 2005). This solution soaks into the soil for about 15 minutes. Next, the solution is mixed in an electric malt mixer for five minutes.

The solution was then added to a 1000 mL sedimentation cylinder, which was then filled to the 1000 mL line with DI water. Care must be taken to ensure that all particles from the dispersion cup make it into cylinder and that there is no loss (Lesikar et al., 2005). The sedimentation cylinder is then capped with a rubber cork and turned end over end once a second for one minute, and set down (Lesikar et al., 2005). Next, the hydrometer is placed in the solution and recorded the reading at 40 seconds (R_{40}) after the cylinder is placed upright (Gee and Bauder, 1986). A thermometer is used to collect the 40-second solution temperature and is then recorded. Next, the cylinder is set aside and upright in a safe place for 7 hours. This allows silt and sand particles to settle. Only clay particles should be left suspended (Lesikar et al., 2005). The temperature is to remain at about 20 °C. After 7 hours, another hydrometer and temperature reading is collected and recorded (Sheldrick and Wang, 1993). Also, the temperature and the hydrometer reading of the blank (R_L) must be recorded at the beginning of each 40-second and 7-hour measurement. Once the numbers are recorded, the following formulas are used to obtain percent sand, silt, and clay. R_L is the hydrometer reading of the blank.

Formulas for Sand, Silt and Clay
(Oven dry weight of soil = M_S)

$$\%\,Clay = \frac{(R_{7h} - R_L)\times 100}{M_S}$$

$$\%\,Sand = \frac{100 - (R_{40} - R_L) \times 100}{M_S}$$

$$\%\,Silt = 100 - (\%\,sand + \%\,clay)$$

3.3 DATA SUMMARY AND STATISTICAL ANALYSIS

For the statistical analyses, the correlation analyses from the JMP 12.1 statistics software was used to evaluate the relationships between the soil variables of bulk density, percent carbon, soil salinity, soil texture, and elevation. The initial intent was to plot trends in soil salinity, carbon concentration, and percent vegetation cover as a function of elevation in the created saltmarsh tidal basins. Non-normal distributions were either transformed with arcsine square root, or a nonparametric Spearman's rank correlation analysis was conducted between non-normally distributed sample sets.

4. Results

Soil Salinity

The distribution of soil salinity revealed a mean of 0.7 ppt (parts per thousand), a median of 0.7 ppt, a standard deviation of 0.3 ppt, and a range between 0.4 ppt and 1.5 ppt. Elevation had a small range between 2.76 m and 3.32 m, with a mean of 2.96 m, a median of 2.9 m, and a standard deviation of 0.14 m.

When measured against elevation, the higher reaches of the basins exhibited lower salinity levels, and the lower elevation areas exhibited higher salinity levels. This trend was significant and possessed an r^2 value of 0.46 (adjusted r^2=0.43; $p>0.002$) (Figure 4.1).

The variable with the strongest relationship with salinity was the percent of the soil that was composed of sand[1], which had an r^2 value of 0.82 (adjusted r^2=0.819; $p<0.001$) (Figure 4.1). This relationship was significant and negatively correlated, indicating higher salinity levels in areas of lower percentage of sand.

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[1] % Sand was arcsine square-root transformed to obtain a normal distribution.

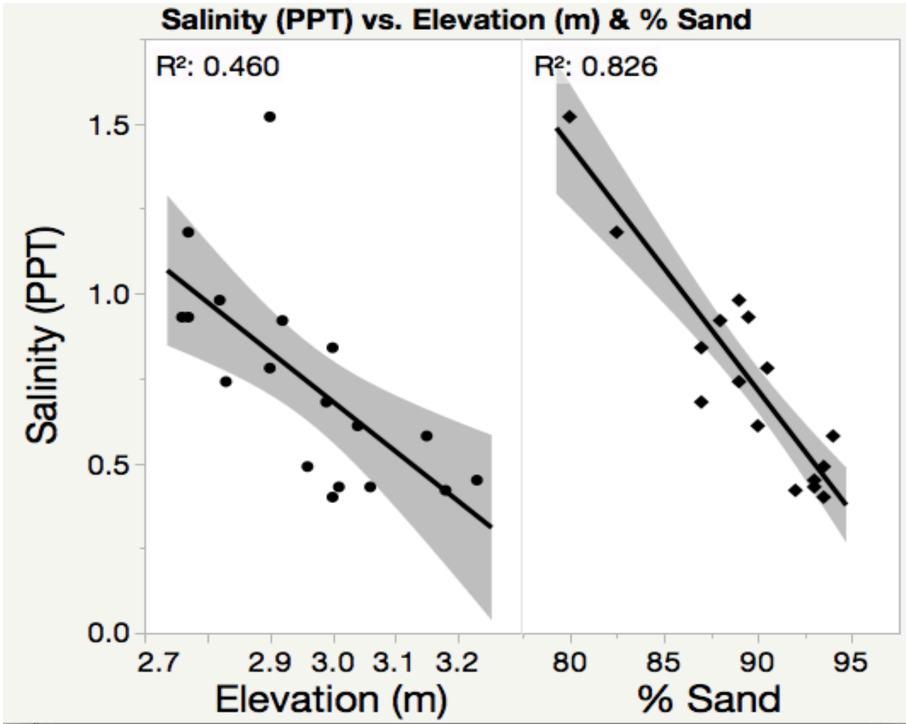

Fig. 4.1 *– Salinity against Elevation (r^2=0.46); Salinity against Percent Sand (r^2=0.826)*

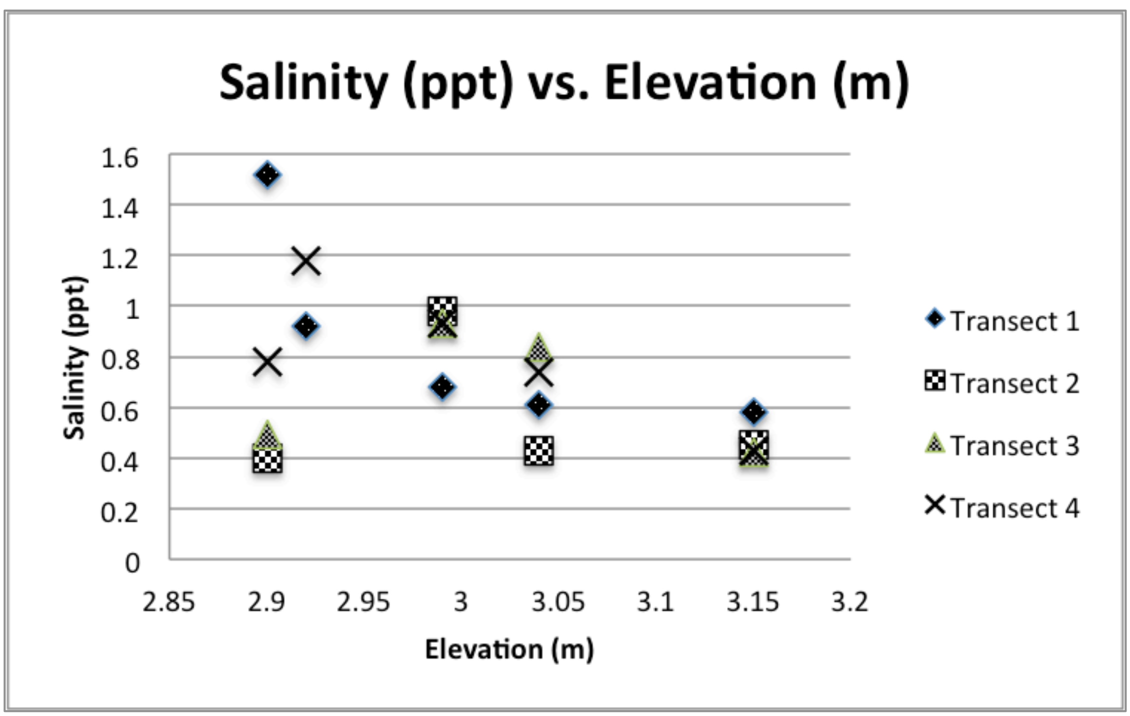

Fig. 4.2 *– Salinity against Elevation per transect*

Soil Bulk Density

Between the three transects, the soil bulk density measurements ranged between 0.66 g/cm^3 and 2.61 g/cm^3. The mean soil bulk density for all plot values was 1.56 g/cm^3, the median was 1.53 g/cm^3, and the distribution had a standard deviation of 0.5. The disparities between the soil bulk density samples are likely due to variations in rock fragment volume per sample. Against elevation, soil bulk density exhibited a positive relationship with higher measurements in the higher elevation plots of the tidal basins. This trend possessed an r^2 value of 0.14 (p=0.12; adjusted r^2 = 0.08) (Figure 4.3).

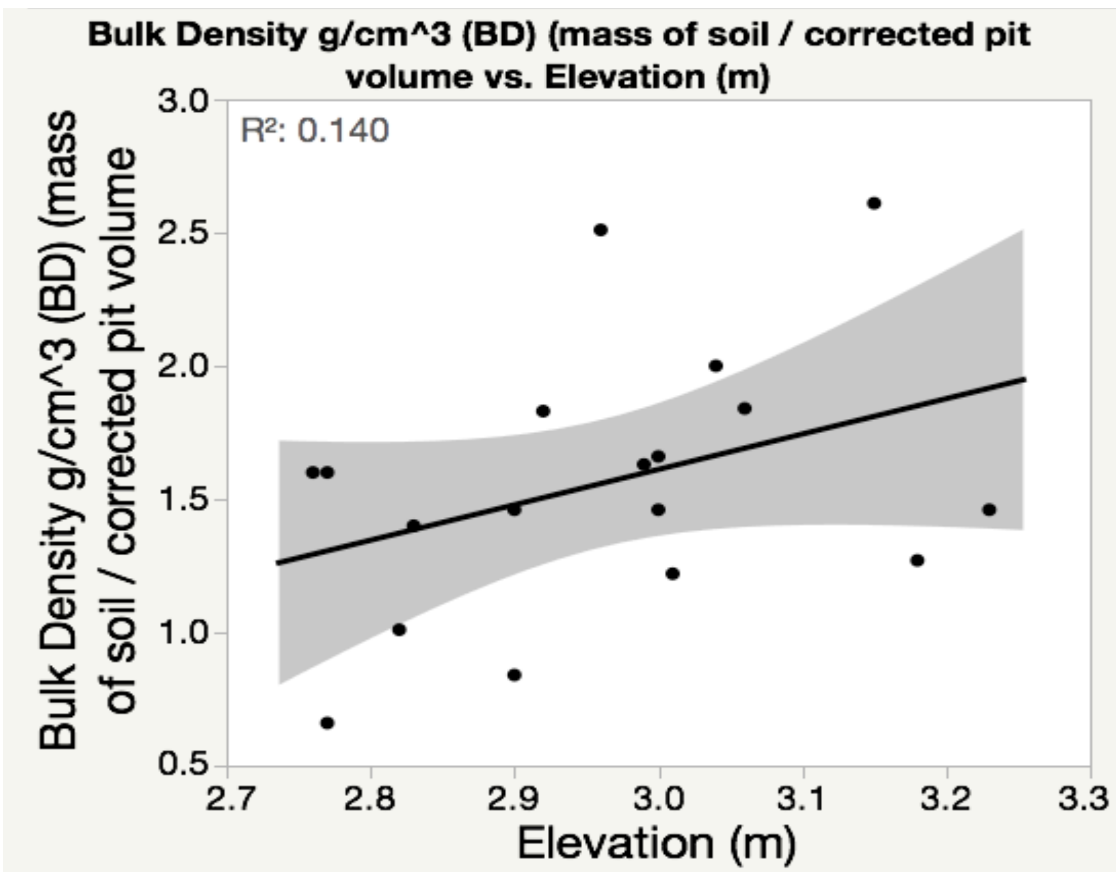

Fig. 4.3 *Soil Bulk Density against Elevation – r^2=0.14*

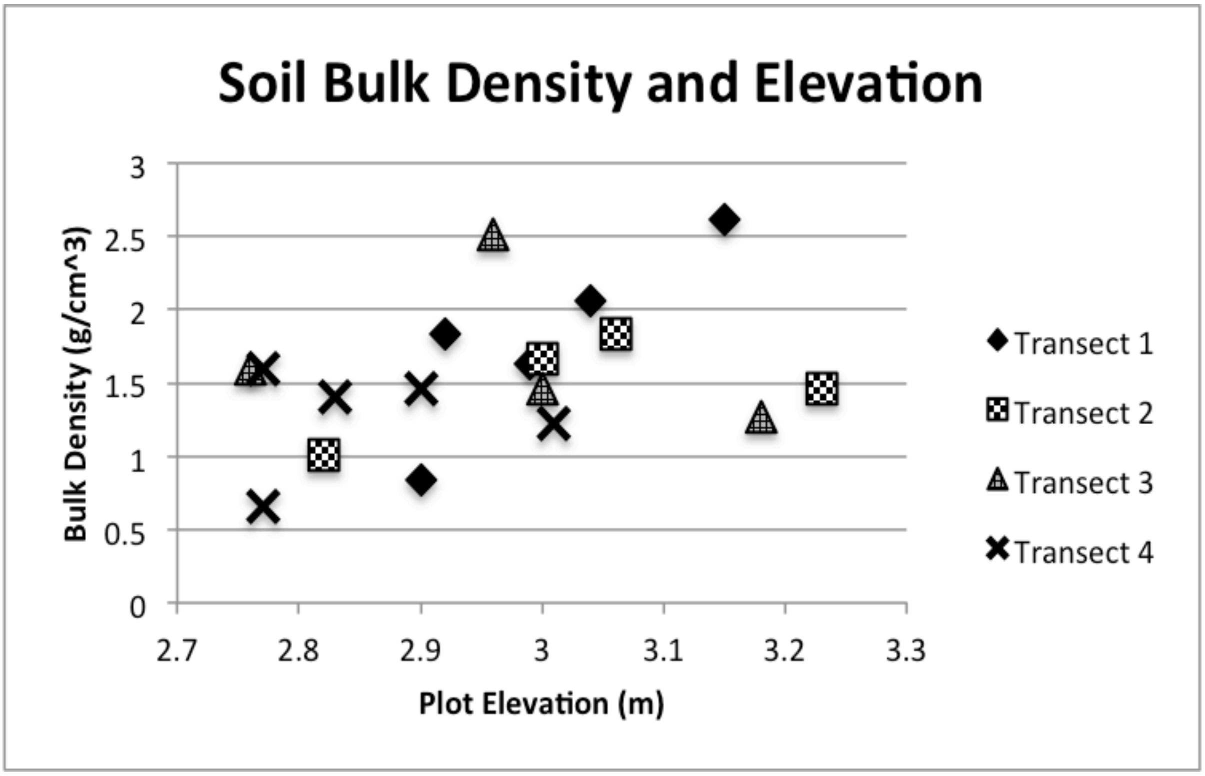

Fig. 4.4 *Soil Bulk Density and Plot Elevation by transect*

Soil Texture

Using the Bouyoucos hydrometer method, it was determined that the top 15 cm of soil from the saltmarsh basin bottom plots was composed predominantly of sand, with a mean percent sand of 89.7% and a range from 80% to 94%. The mean for percent clay was 2.4% per sample and the samples had a small range from between 1.5% and 4%. Silt content exhibited a mean percentage of 7.9% per sample with a range of 4.2%-14.5%. Percent sand, silt, and clay shared some varying correlations with elevation and percent carbon.

Percent clay by elevation displayed a very minimal and insignificant linear correlation, with a small r^2 value of 0.08 (adjusted r^2=0.02; p=0.25) (Figure 4.5). Percent silt was negatively correlated with elevation (an r^2 of 0.33; adjusted r^2= 0.29; p<0.01) (Figure 4.5), indicating higher silt content in the lower elevations of the tidal basin bottoms. Percent sand against elevation indicated higher percentages of sand in the higher elevations, with an r^2 value of 0.287 (adjusted r^2= 0.24; p<0.02) (Figure 4.5). Because Percent Sand was not normally distributed, it was arc-sin transported. There was significant positive correlation between the percent distribution of sand and elevation, with an r^2 value of 0.31 (adjusted r^2=0.27; p>0.01).

There were no notable trends between texture and bulk density.

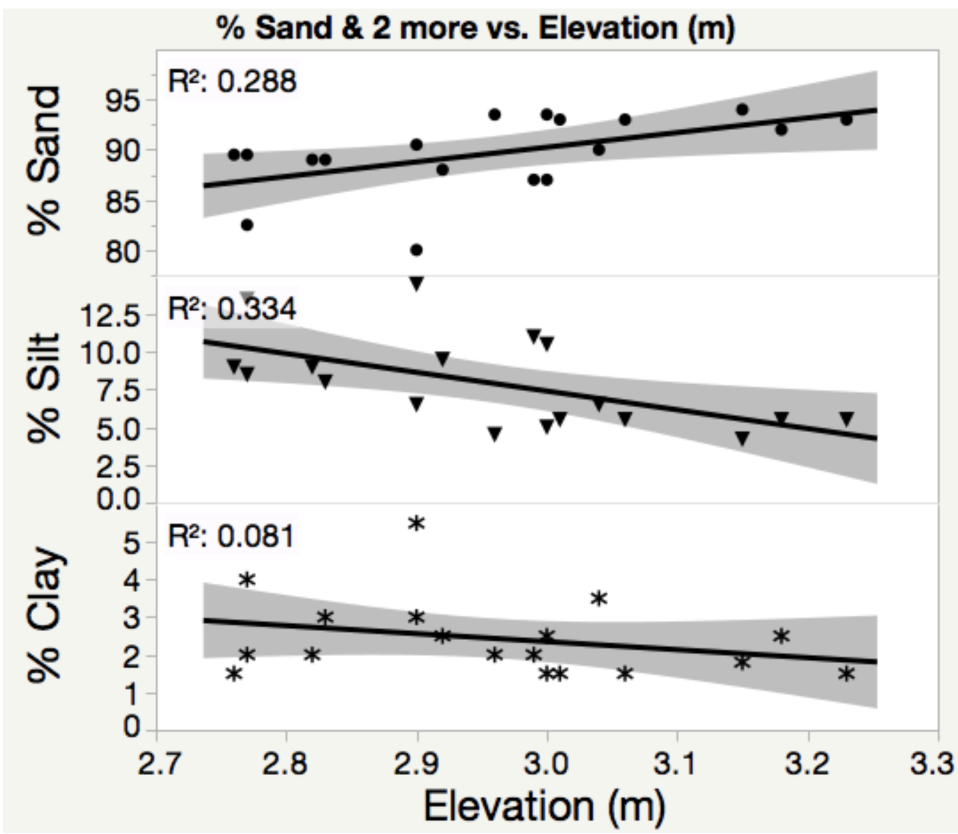

Fig. 4.5 *Percent Clay against Elevation of soil samples****;*** *Percent Sand by Elevation of soil samples; Percent Silt against Elevation of soil samples*

Vegetation Percent Cover

Percent cover of vegetation was visually estimated, and the comparisons with the other variables did not indicate any markedly substantial associations. A wide range of cover percentages was observed, from 3% to 90%, with a mean of 35% and median of 25%. Percent cover against salinity displayed the strongest relationship out of all the other variables, with a positive trend and a correlation coefficient r^2 value of 0.29 (adjusted r^2=0.25; p<0.02) (Figure 4.6), indicating higher vegetation cover in the areas of higher salinities, nearer to the shoreline.

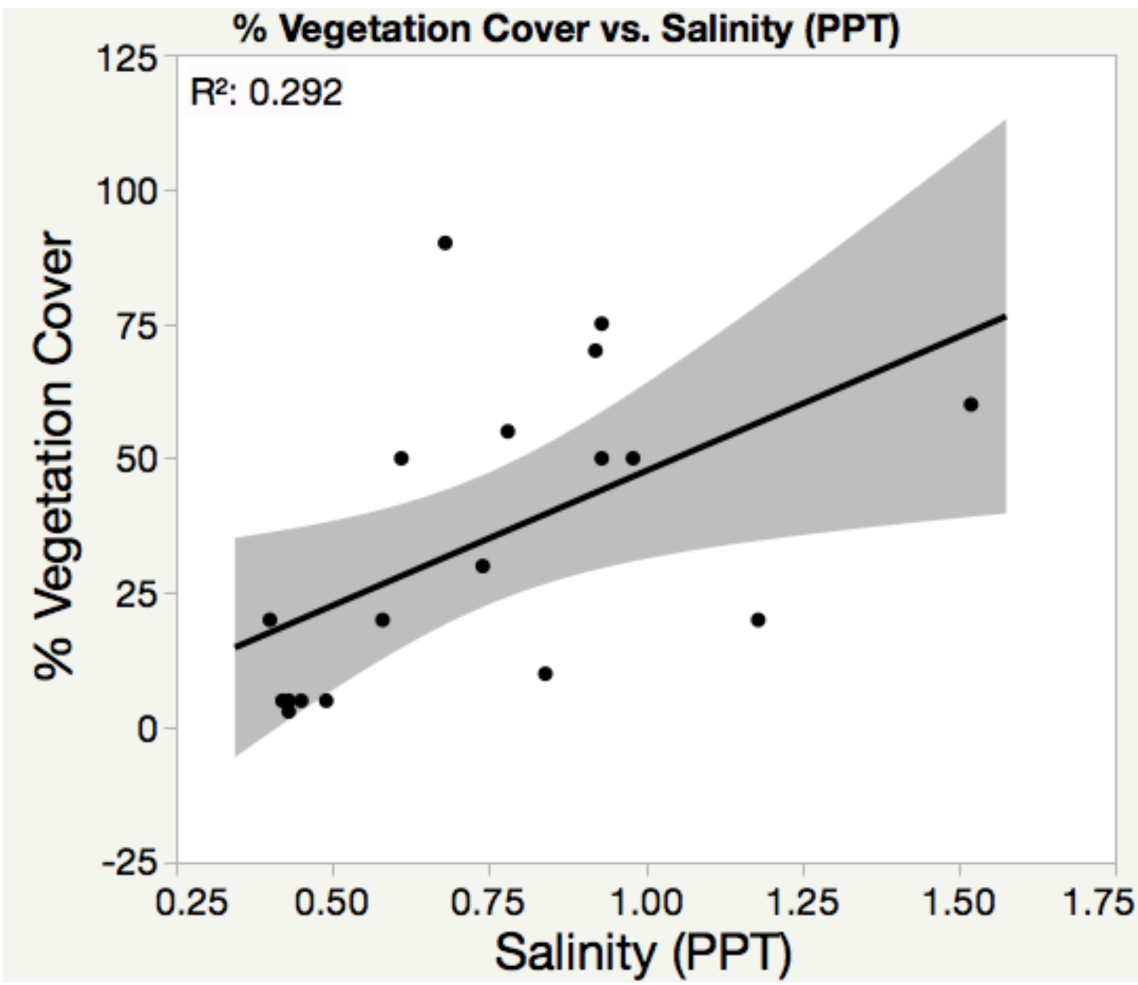

***Fig. 4.6** – Salinity against Percent Vegetation Cover*

The relationship between vegetation cover and silt content had the strongest positive correlation with an r^2 value of 0.26 (adjusted r^2=0.21; p<0.03) (Figure 4.7). The third strongest vegetation relationship was with elevation, exhibiting downward trend with an r^2 value of 0.256 (adjusted r^2=0.21; p<0.03) (Figure 4.8). These measurements suggest that there is a higher percent of vegetation cover in those areas with higher soil salinity levels, higher fine particulate content, lower elevations, and lengthier tidal inundation times (in lower elevations).

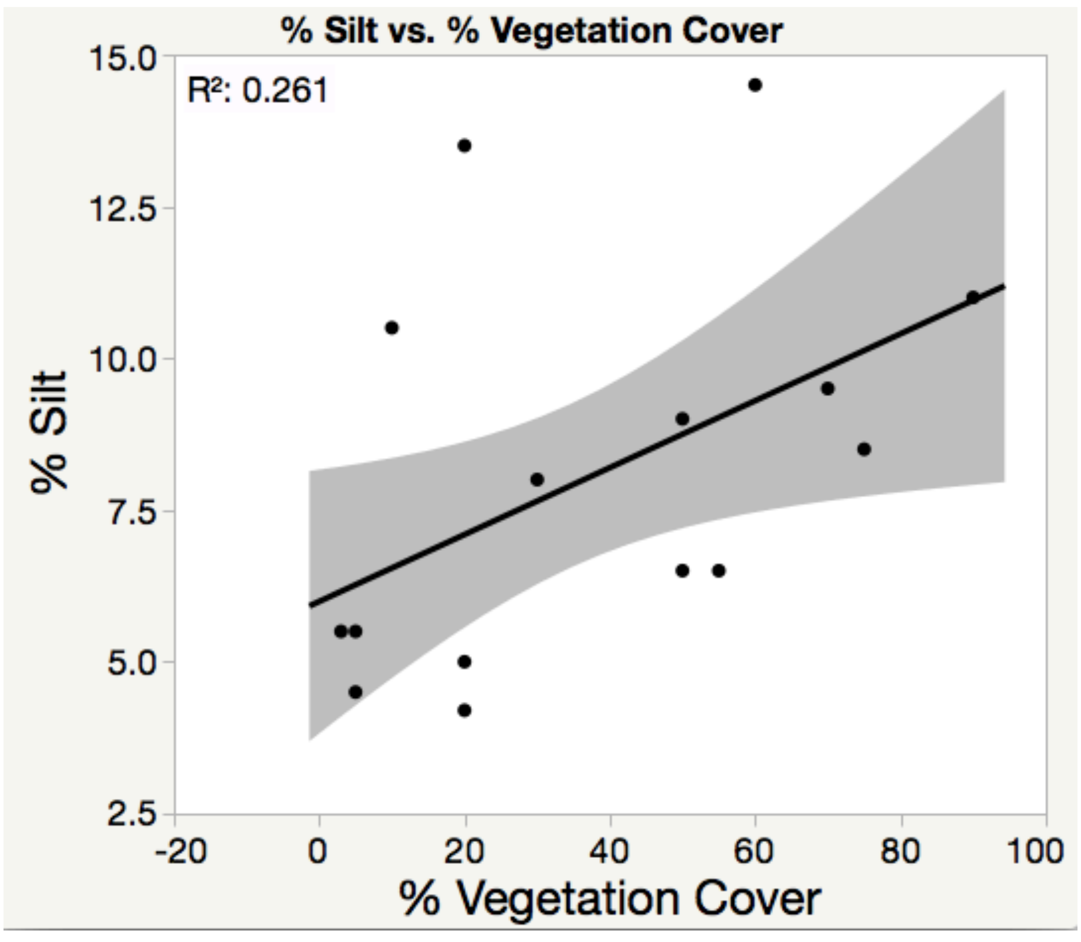

Fig. 4.7 *– Percent Silt against Percent Vegetation Cover*

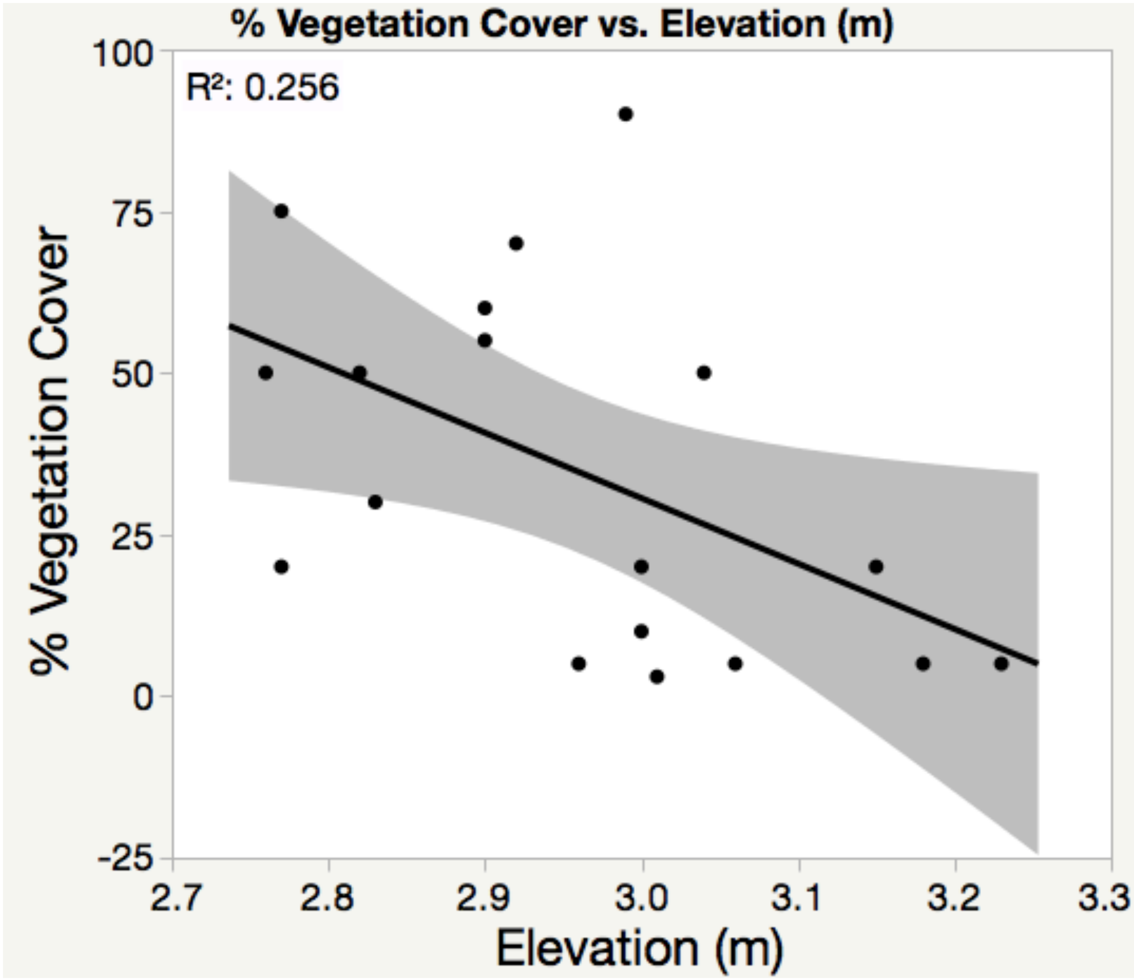

Fig. 4.8 *–Percent Vegetation Cover vs. Elevation*

Relationship Between Soil Carbon, Elevation and Salinity

The percent soil carbon from the top 15 cm of the basin bottom plots possessed a mean of 1.3%, a standard deviation of 0.54%, and the values ranged from 0.43% to 2.4 %. Percent carbon was negatively correlated witt elevation (r^2=0.29, p<0.02, adjusted r^2=0.24) (Figure 4.9). This indicates that there are higher soil carbon concentrations in the lower elevation sections of the basins, which receive more prolonged tidal influence than the higher elevation areas.

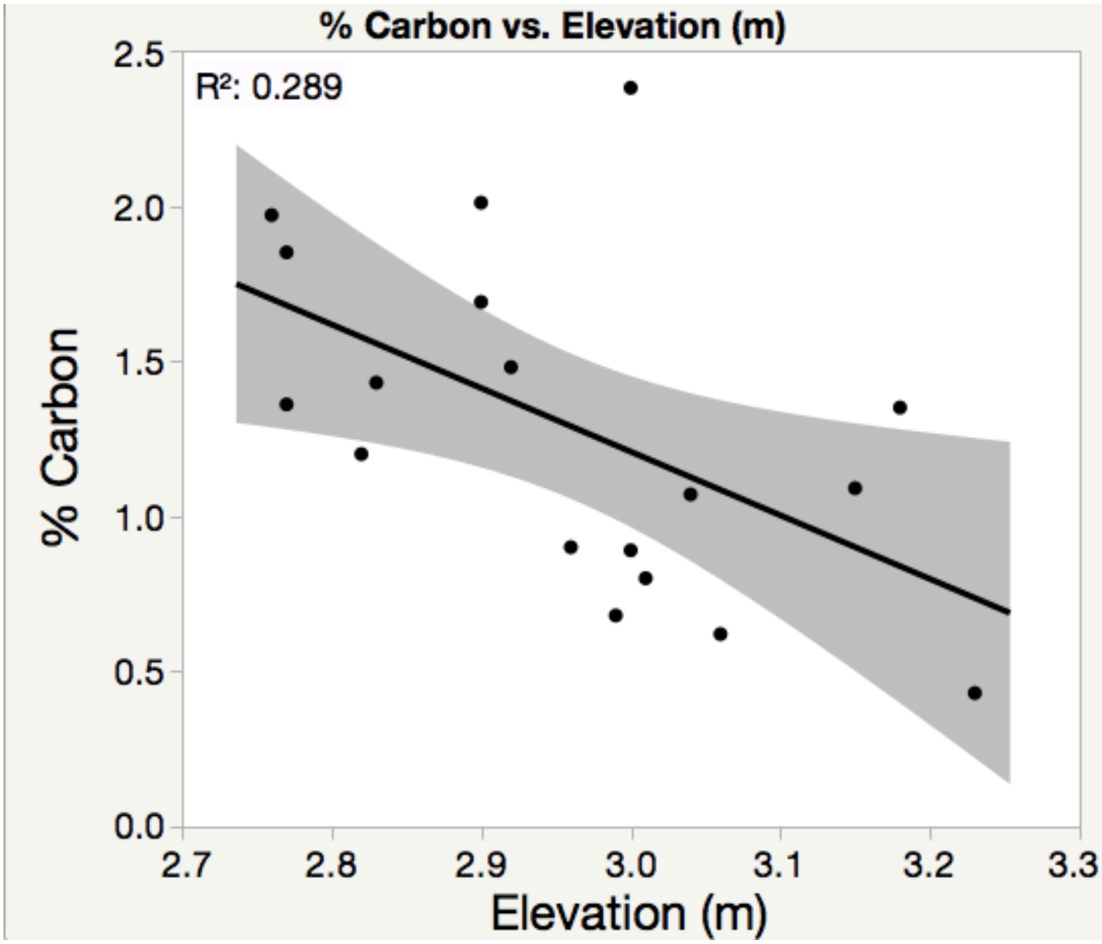

Fig. 4.9 *- Percent Carbon against Elevation*

Carbon content showed a strong positive relationship with soil salinity levels, revealing an upward trend with r^2=0.52 (adjusted r^2=0.49; $p<0.0007$) (Figure 4.10). This indicates higher carbon levels in the higher salinity areas. This agrees with the measurements of higher salinity levels in the lower elevations of the basins. Vegetation shows no discernable trend with carbon content.

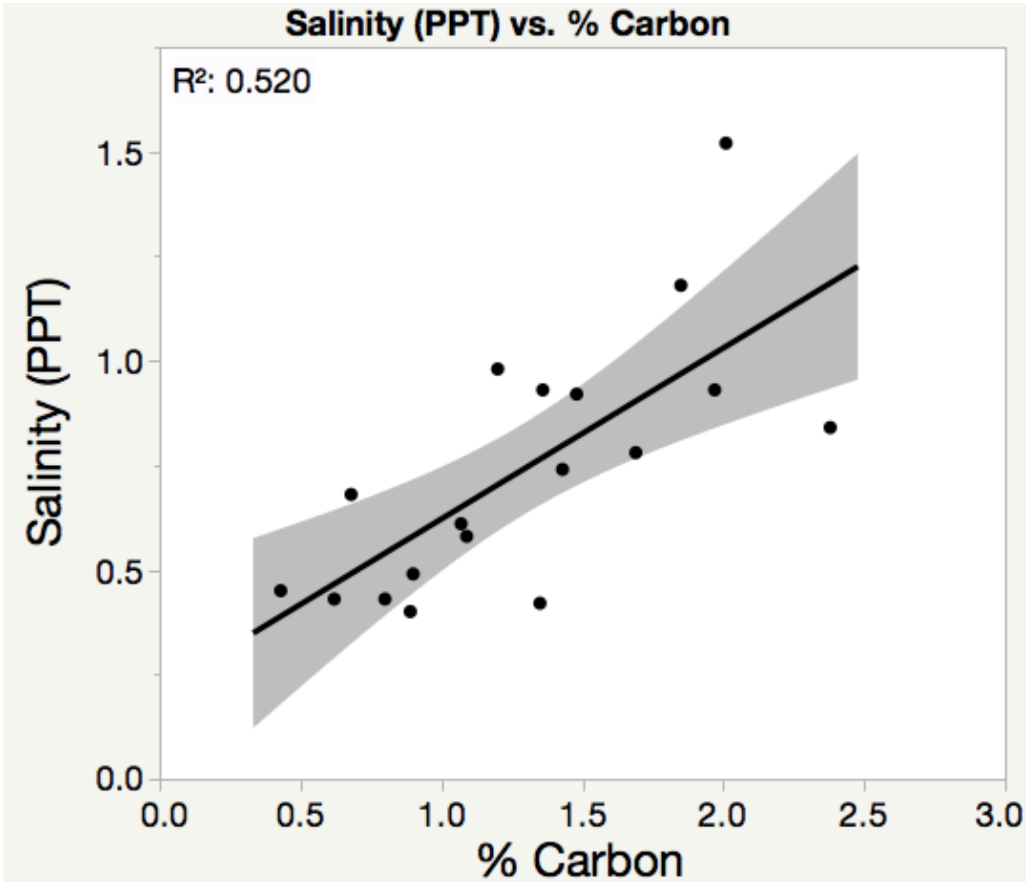

Fig. 4.10 *- Percent Carbon against Salinity (ppt)*

Bulk density and percent carbon were not notably related either, with a very low r^2 of 0.08 (adjusted r^2=0.02; $p<0.24$) (Figure 4.11).

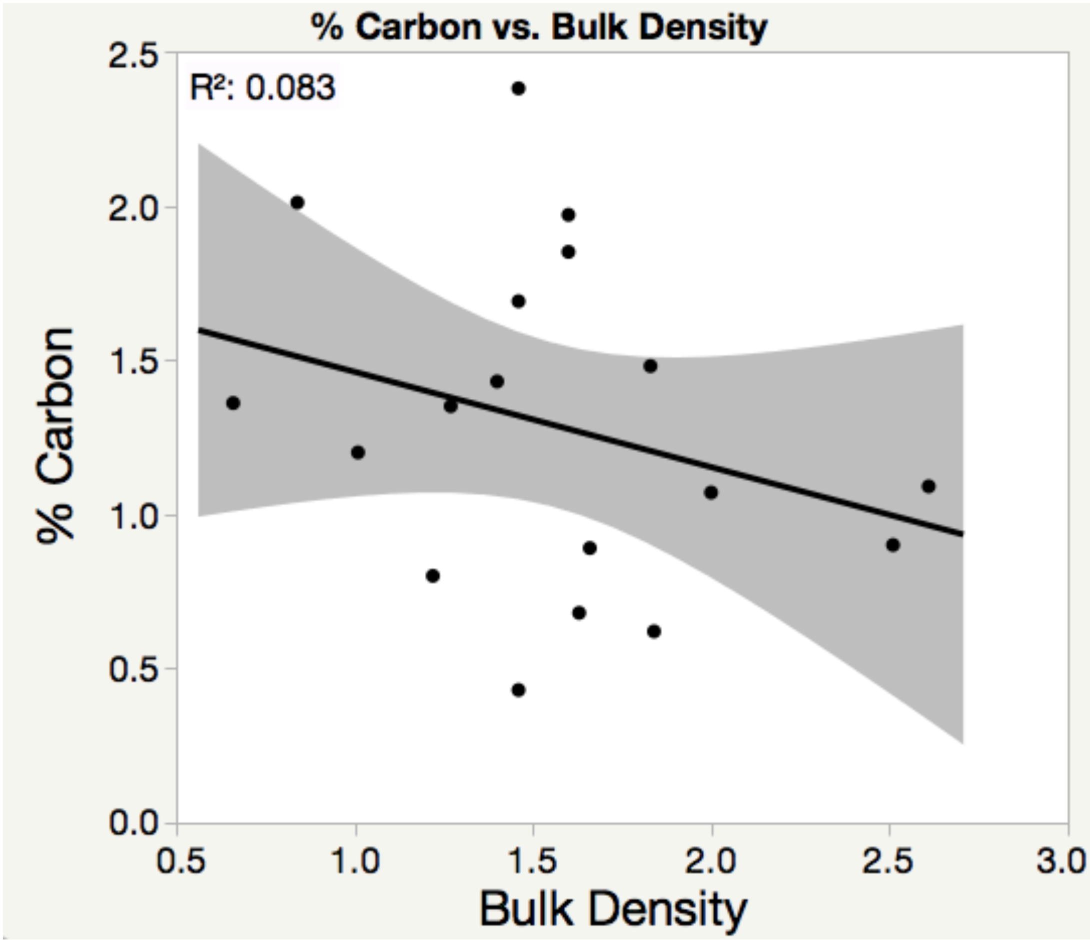

Fig. 4.11 *Soil Bulk Density against Percent Carbon*

Texture and Carbon

Percent silt also displayed a stronger relationship with percent carbon, displaying an r^2 value of 0.39 (adjusted r^2=0.36; p<0.005) (Figure 4.12). Percent clay also indicated an upward trend against carbon with an r^2 of 0.31 (adjusted r^2=0.27; p<0.01) (Figure 4.12). These trends appeared to be consistent with the percent sand against carbon content measurements, which revealed a strong downward trend, with an r^2 value of 0.427 (adjusted r^2=0.39 p<0.0032) (Figure 4.12).

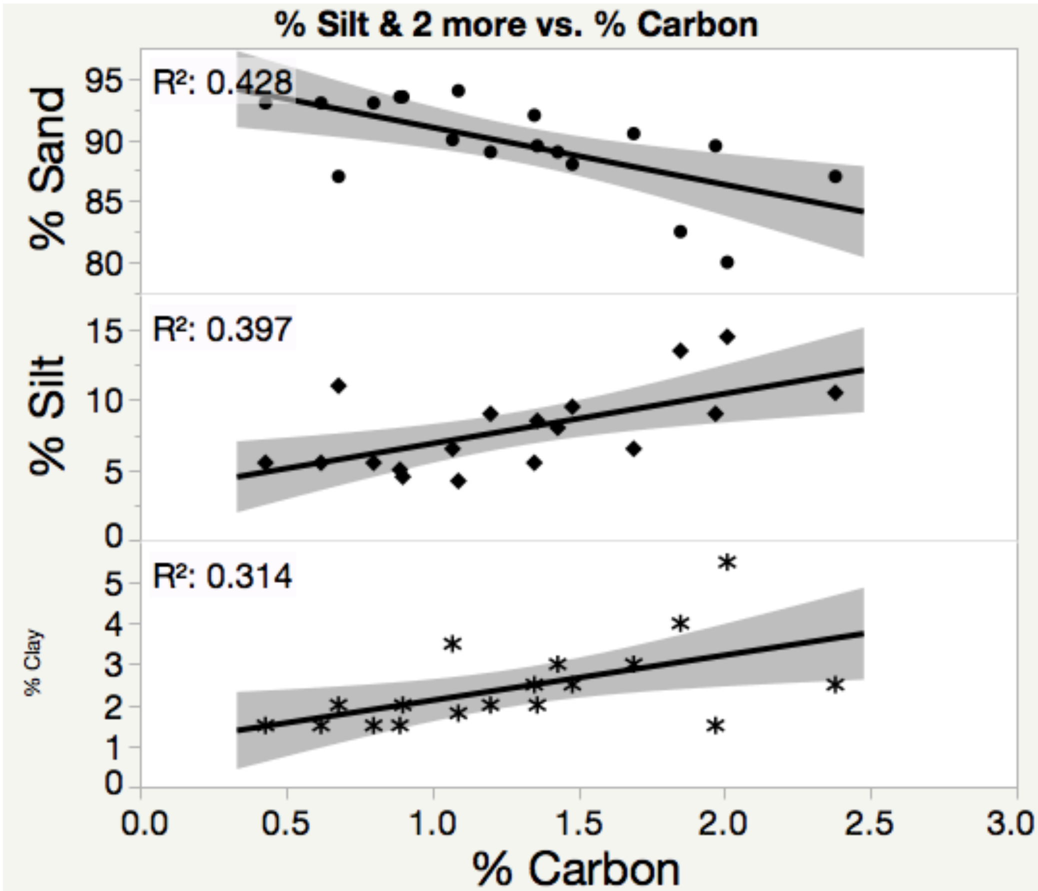

Fig. 4.12 Percent *Silt against Percent Carbon in soil samples; Percent Clay against Percent Carbon in soil samples; Percent Sand by Percent Carbon in soil samples*

Carbon Density

The concentration of carbon in sediment samples was obtained by multiplying the percent carbon content by bulk density. Measurements ranged from 6 mg C/cm^3 to 35 mg C/cm^3, with a median of 18.5 mg C/cm^3 and a mean of 19.2 mg C/cm^3. The results did not display trends against any of the variables. The carbon density values appeared to be scattered randomly and to not have any discernable distribution pattern.

5. DISCUSSION

The results indicated that the variables of elevation, fine sediment composition, salinity, and carbon content all shared some connections in the Bayshore tidal basins. The following discussion section explains how these results compare to other similar studies, questions and inferences that can be drawn from the results, suggestions for future monitoring and/or management, and some critiques on methodology.

Relationships between the Variables and Comparisons with Other Studies

The unrestricted tidal influence appears to have indeed facilitated a healthy effect on the newly created saltmarsh habitat in the tidal basins. My findings also displayed some strong trends between salinity and soil particle size proportions, elevation, and percent carbon (Figures 4.1 and 4.10). Soil grain size distribution against carbon content also revealed a strong relationship, with higher carbon percentages in those soils with a higher proportion of fine sediments (silt and clay) (Figure 4.12). This corresponds with the fact that finer sediments possess a higher surface area, allowing for soils with higher fine sediment proportions to retain more carbon. It appears that soil texture may be controlling the carbon dynamics of the site.

Vegetation displayed a relatively strong trend with salinity and texture. The halophytic plant communities are dependent on moderate soil salinity levels. The measurements exhibit slightly less vegetation in those areas with lower soil salinity levels (near shore). There was also more vegetation in areas more frequently inundated (lower elevations), possibly due to frequent access to tidal salinity and fine sediment inputs. It is very likely that the higher vegetation cover is due to the higher salinity levels more conducive to halophytic vegetation, and higher vegetation areas trap fine sediments in their plant and root structures, creating a finer sediment constitution in the lower elevations.

Relationships with soil bulk density were minimal. The results showed only a slight trend with elevation (r^2=0.14; p<0.12) (Figures 4.3 and 4.4), with greater bulk density in the higher elevations of the channels. The trend coincided with the slight trend of bulk density against % Sand (r^2=0.19; p<0.06), with higher bulk density measurements alongside the higher sand percentages. The USDA NRCS explain that sandier soils tend to have higher bulk density levels as the pore space in sand is less than the soil made up of silt and clay sediments (USDA, 2008). However, this trend seems to go against the pattern found in Butzeck et al. (2015), which indicated higher bulk densities in the lower elevations of a saltmarsh. When fit against carbon, bulk density showed almost no trend, but did display a very slight inclination (r^2=0.08; p<0.24) to possess less carbon content in the higher bulk density regions (Figure 4.11).

Some additional discrepancies were observed between my results and those of other studies. My bulk density results for the new Bayshore basins were between 0.66 g/cm^3 and 2.61 g/cm^3. Dan Masello's bulk density results from the nearby Duckabush River Delta were between 0.19 and 0.95 g/cm^3, indicating that he likely had less sandy soils. The Hansen et al. (2014) study of a Los Angeles saltmarsh indicated bulk density rates of 0.40 ± 0.26 g/cm^3 (Hansen and Nestlerode, 2014). The saltmarshes in the Beecher et al (2001) Bay of Fundy saltmarsh study displayed a soil bulk density range of 0.18-1.3 g/cm^3, albeit their measurements were from only the top 2 cm of the saltmarsh surface. If grain particle size proportions are indeed a predictor of soil bulk density, this may explain my seemingly high bulk density measurements, as the Bayshore tidal basins have not yet had the opportunity to accumulate more fine sediments, in comparison to more established saltmarshes.

Another interesting discrepancy between my results and those from other studies are the Percent Carbon and carbon density measurements. The Bayshore data indicated lower carbon content in the higher elevation areas with less tidal/inundation influence. Dan Masello's thesis (2013) displayed the opposite trend, with higher carbon content in the infrequently inundated higher elevations. Masello's carbon density measurement followed a similar trend against elevation. In contrast, my carbon density results did not indicate any trends.

Callaway and Crooks (2015) explain that most tidal wetlands contain a carbon density range between 0.02 and 0.04 g/cm^3 (Callaway and Crooks, 2015). In their study about carbon sequestration in saline wetland soils, Chmura et al. (2003) revealed that their carbon density results from all their saltmarsh study sites from around the world possessed an average of 0.039 ± 0.003 g/cm^{-3} (Chmura et al., 2003). The Hansen and Nestlerode (2014) study from a Los Angeles saltmarsh displayed a carbon density range of 0.035 ± 0.023 g/cm^3. Dan Masello's average soil carbon density was roughly 0.028 g/cm^3 (Masello, 2013). The mean from my carbon density results was 0.0192 g/cm^3, and the range was between 0.006 g/cm^3 and 0.035 g/cm^3. A likely explanation for the discrepancies between these C density measurements and those of Bayshore is that the soils from the new Bayshore tidal basins may not yet have experienced sufficient inundation time and sediment deposition to amass a substantial carbon bank and vegetation community since their creation. Additionally, carbon is likely concentrated on the top two or three centimeters of the soil surface. My samples were 15 centimeters deep, which possibly diluted the carbon readings of the surface soil. For this reason, I did not attempt to determine/estimate the carbon stocks of the basins as my carbon density results were scattered.

Inferences and Future Considerations

Another informative saltmarsh attribute to observe in the Bayshore basins would be their accretion rates. Because saltmarsh soil substrates are consistently accreting vertically, it would be informative to obtain a long-term dataset of the Bayshore basins accretion patterns. The continual accretion of fine sediments in coastal wetlands is a natural process essential for a robust fitness level in a saltmarsh, particularly for its carbon burying properties. Monitoring this data can be valuable for additional information about the coastal carbon stocks. There is a range of long-term methodologies used for measuring accretion in the literature. Common methods include measuring sediment deposition on plates or traps for short-term rates, and marker horizons and isotopic profiles for longer-term measurements (Butzeck et al., 2014; Hopkinson et al., 2012). Accretion is a necessary protection for the saltmarsh ecosystem to withstand sea level rise. By tracking the progression of vertical accretion, land managers can obtain another valuable parameter to evaluate shoreline habitats and their resilience. Moreover, when combined with carbon density measurements, land managers could collect valuable data about the carbon pool on the saltmarsh surface as well as carbon accumulation rates (Hansen & Nestlerode, 2013).

Maintaining a sound monitoring record of a restored wetland's vegetation and soil attributes over the wetland's lifetime is a proactive approach for effectively visualizing and communicating its progress. Doing this is valuable for examining the advancement of the constructed habitat and for understanding which restoration methods are most successful in the rehabilitation process. The information can be revealing to investors and other stakeholders, who may be interested in the land for community green space, recreation opportunities, and for local ecosystem service recovery. By having a continual statistical and visual representation of the fitness of their restoration venture, stakeholders may become more inclined to continue their support. Additionally, the data helps community members recognize that land managers are proactively inspecting their work to ensure success.

Sustaining this support and positive attitude about coastal restoration is imperative for successful conservation and restoration projects. If communities desire to expand and conserve salmon habitat, along with enjoying all the ecosystem and recreational services that these biomes provide, land managers need to communicate and demonstrate that they are determined to see these projects through.

CONCLUSION

My research question inquires if the newly created tidal basins in Bayshore preserve serve as effective sites for fine sediment deposition. The follow-up question is, “How does variation in elevation, soil salinity, and vegetation cover affect the distribution of these parameters?”

To address the questions, I analyzed several soil properties: salinity, carbon content, bulk density, grain size proportion, and percent vegetation cover. I compared these variables to their elevations and to each other. Though my sample size was small, I observed that elevation seems to be the common factor playing a significant role in the variations of most of the variables. This is because a saltmarsh area can receive more or less tidal influence, contingent on the elevation, which in turn influences access to salt and sediment inputs from the tidewaters. Soil salinity levels appeared to be related vegetation patterns, and also displayed a strong downward trend with percent sand. I conclude that by regularly assessing these variables in relation to each other, land managers could retain and observe a robust evolving model of the of health of a restored site.

The Puget Sound Nearshore Ecosystem Restoration Study (2014) explains that the marine natural resources in the Pacific Northwest perform an important part in the local economy and regional lifestyle. To accomplish the preservation of these resources, researchers expound the benefits of developing adaptive management strategies as well as the “increased understanding of natural process restoration to improve effectiveness of project actions” (The Seattle District Corps of Engineers, 2016). By evaluating the soil attributes in the Bayshore basins, land managers can obtain and utilize valuable information about attribute relationships and how a rehabilitated habitat can potentially progress overtime. Bayshore's basins provide a unique opportunity to observe how sediments and carbon accumulate over time in a created and/or restored saltmarsh. By continuing to evaluate the physiographic attributes in the Bayshore Preserve, stakeholders can confidently predict and communicate success and advancement in existing and future restoration actions. Puget Sound coastal ecosystems are progressively diminishing, and though it may seem of small significance, monitoring the restored nearshore wetlands is an important facet of coastal restoration.

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Appendices

{References or Notes}