VARIATIONS IN SURFACE SOIL ORGANIC CARBON AT THE DUCKABUSH RIVER DELTA, WASHINGTON

by

Daniel M. Masello

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Daniel M. Masello

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by

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Dr. Erin Ellis
Member of the Faculty

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Date

ABSTRACT

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Intertidal wetland soils are estimated to sequester carbon at a rate that greatly exceeds those of other terrestrial ecosystems (Pidgeon, 2009). Despite this, data regarding intertidal wetland area and soil carbon content and density are scarce. Puget Sound intertidal wetlands have been greatly diminished from their historical extent, and sea level rise threatens those that remain. Further, there are currently no estimates of carbon content, density, or stocks of Puget Sound intertidal wetland soils. This study examined the organic carbon content and density of the top 30 cm of soil at the Duckabush River Delta, located on the Hood Canal in the Puget Sound, Washington. Carbon content (percent organic carbon), density (grams of carbon per cm^3), and soil texture (percent sand, silt and clay in sediment) were measured at varying salinities and elevations gradients. The study area was divided into three zones by elevation (1: low, 2: middle, 3: high), which was based on apparent groupings of the non-linear distribution of soil bulk density plotted against elevation. Carbon content was significantly different in all zones, with 2 > 3 > 1. Carbon density was not significantly different between zones 2 and 3, but both were significantly higher than zone 1. The increase in carbon density with elevation between the zones was driven by significantly greater soil bulk density in 3 than 2, as well as the silt-dominated soil of zones 2 and 3, which was positively correlated with carbon content in this study (r^2=0.57). Estimates of surface soil organic carbon stocks (Mg/ha) were 52.9, 93.9, and 106.3 for zone 1, 2, and 3, respectively. These stocks are similar to estimates of soil organic carbon stocks in other terrestrial systems (e.g. 96.2 in Temperate Forests, 127. 4 in Tundra (Pidgeon, 2009)). However, taking into account the extremely high sequestration rate previously reported for intertidal wetland soils, it is logical to conclude that surface soils at the Duckabush River Delta and other intertidal wetlands are high-value ecosystems in the effort to mitigate climate change. Long-term research examining sequestration rates at the Duckabush Delta (as well as soil organic carbon content, density, and sequestration rates at other Puget Sound intertidal wetlands) would be useful supporting or dismissing this claim.

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Introduction

As global temperatures rise and consensus builds that anthropogenic climate change is the culprit, it becomes more and more valuable to understand how greenhouse gasses are allocated globally in the terrestrial, oceanic, and atmospheric reservoirs. The terrestrial carbon reservoir is perhaps the most complex of the three carbon reservoirs, due to the diversity of environments and ecosystems that fall under the umbrella of the greater terrestrial system (Crooks et al., 2010; Chmura, 2009).

Coastal ecosystems, at the interface of the terrestrial and oceanic world, account for a small fraction of the earth's area, but may account for a disproportionately large portion of organic carbon stored in their soils (Pidgeon, 2009; Hussein et al., 2004). They account for 1% or less of global terrestrial ecosystems, but constant accretion of sediment in these systems over millennia results in huge stores of buried carbon (Pidgeon, 2009) (Table 1.1). Not only do coastal ecosystems, including intertidal wetlands, store large amounts of carbon, but they emit negligible methane (CH_4), a potent greenhouse gas (Crooks et al., 2010; Chmura et al., 2003; Saarnio et al., 2009; Bartlett et al., 1987). There is debate in the literature, but many sources assert that intertidal wetlands also emit negligible nitrous oxide (N_2O), another potent greenhouse gas (Chmura et al., 2003). Unfortunately, the global extent of these systems, including intertidal wetlands, is currently unknown, due to their ever-shifting geomorphology and a legacy of land-use change along coasts all over the world.

As these coastal ecosystems are developed, their ability to continuously sequester carbon is also lost, thus contributing to atmospheric carbon dioxide (CO_2) levels. Sea level rise, which has been accelerated by climate change, will result in coastal ecosystems becoming trapped between rising seas and inland development, further diminishing carbon accumulation and thus becoming another positive feedback to anthropogenic CO_2 emissions (Hopkinson et al., 2012; Hussein et al., 2004).

The value of these ecosystems in the face of climate change cannot be overemphasized. This study seeks to examine whether claims made in previous research from other parts of the world regarding the high capacity of intertidal wetland soils to accumulate large carbon stocks, and subsequently mitigate a fraction of anthropogenic CO_2, holds true in an unstudied location by measuring carbon content and density, and estimating surface soil organic carbon stocks of an intertidal wetland located in the Puget Sound, Washington State. Data regarding carbon in intertidal wetland soil in this region are absent from the literature, so this study will be this first to look at soil carbon density, content and stocks in this unique region. The Puget Sound is unique because it is historically rich in intertidal wetland area and diversity, relative to the global extent of intertidal wetlands. The historical extent of intertidal wetlands has been greatly diminished, and opportunities for restoration and conservation abound in the region (Collins & Sheikh, 2005; Correa, 2003). Furthermore, Puget Sound is more vulnerable to climate-change-accelerated sea level rise than other coastal regions of Washington, including the northern coast of the Olympic Peninsula along the Strait of San Juan de Fuca and the west coast of the state along the Pacific Ocean, which are experiencing tectonic uplift, largely negating the effects of local sea level rise (Mote et al., 2008).

Most similar research has taken place on the Atlantic and Gulf Coasts of North America, and few studies have been done at all on the entire Pacific Coast, with none being conducted in Washington State. Puget Sound shorelines and intertidal wetlands are already greatly diminished from their historic extent, due mostly to land-use change, and the combination of shoreline armoring and sea level rise. Those who support the conservation and restoration of Puget Sound intertidal wetlands often focus on the value for fish and wildlife, as well as public recreation. However, with baseline data of the carbon content and density of an intertidal wetland, the value as greenhouse gas sinks could further support the argument for conservation and restoration of these ecosystems. For example, if it is demonstrated that high amounts of carbon are stored in these ecosystems relative to other upland ecosystems in Washington State, one could conclude that emphasis should be placed on preserving these ecosystems. Furthermore, because sea level rise and and coastal development threaten the continued existence of coastal ecosystems and intertidal wetlands around the world, this study examines carbon content and density along an elevation gradient. In addition, other factors that can influence carbon content, such as soil texture and salinity, were concurrently measured. Soil organic carbon content is also known to correlate to physical soil texture. Correlating the content of different soil particle size classes to elevation and carbon (content and density) may provide insight into understanding the distribution of carbon in an intertidal wetland using these physical characteristics as a proxy. As such, the goal of this study is to provide an estimate of surface soil carbon stocks in a unique intertidal wetland system in an understudied region, and to contribute to the larger goal of understanding the mechanisms controlling carbon storage in the terrestrial carbon reservoir.

1. **Literature Review**

1.1. ***Intertidal wetlands and the global carbon reservoir***

In May of 2013, the atmospheric concentration of CO_2 exceeded 400 ppm for the first time in more than 2.5 million years (NOAA, 2013). Only two-and-a-half centuries ago, the pre-industrial atmospheric concentration of CO_2 was about 280 ppm (Lal, 2004; NOAA, 2013). The consensus in the scientific community is that the anthropogenic addition to atmospheric CO_2 is a central driver behind climate change (Denman et al., 2007; Chmura, 2009). Anthropogenic CO_2 is released through fossil fuel combustion and through land-use change (Crooks et al., 2010; Lal, 2004). It is believed that anthropogenic CO_2 can be partly mitigated by carbon sequestration in terrestrial ecosystems, both in vegetation and in soils (Hopkinson et al., 2012; Hussein et al., 2004; Phachomphon, 2008). Therefore, assessing the content, density, and stocks of carbon stored in different types of soil is the first step in evaluating the potential of soils to sequester carbon.

The amounts of carbon in oceanic and atmospheric reservoirs are better understood relative to the amount of carbon stored in terrestrial reservoirs (Lal, 2004). Studies of carbon stocks in terrestrial ecosystems have been most extensive in peatland, freshwater wetlands, and large upland ecosystems (including temperate and tropical systems) (Chmura et al., 2003). Coastal wetlands have received less attention relative to these other terrestrial systems, despite the agreement in the literature that various types of coastal ecosystems sequester carbon at a rate far exceeding their other terrestrial counterparts (Crooks et al., 2010; Chmura, 2009; Li et al., 2010). The low attention paid to coastal ecosystems may stem from the fact that they account for only a small fraction, 1% or less of the greater terrestrial system (Pidgeon, 2009). Estimates regarding the areal extent and carbon stocks of coastal wetlands cover a wide range, due mostly to limited data and differences in methodology between studies (Hopkinson et al., 2012). Furthermore, estimating the area of coastal wetlands is complicated by sea level rise, and the constantly-changing geomorphology of coastal wetlands (Pidgeon, 2009). Estimates of the area of global coastal systems (including mangroves, intertidal marshes, and seagrass beds) lie between 5.15x105 and 1.2x106 km^2, with intertidal marshes accounting for between 2x105 and 4x105 km^2 (Hopkinson et al., 2012), or approximately 0.13-0.26% of Earth's total land surface area. In the IUCN's report, The Management of Natural Coastal Carbon Sinks, it is stated that the total global area of tidal salt marshes is simply "unknown," with 0.22x1012 m^2 currently reported (Pidgeon, 2009).

Synthesis studies of global carbon sequestration have come up with varied estimates, but all agree that relatively huge amounts of carbon are sequestered per unit area in coastal ecosystems and intertidal wetlands. Because intertidal wetlands must continuously accrete vertically to stay above sea level, they are constantly sequestering more and more carbon. Other terrestrial systems, on the other hand, have been shown to approach a point of equilibrium in which carbon is no longer sequestered at a rate exceeding respiration, because accretion does not occur at a comparable pace in these other terrestrial ecosystems (Ellis, 2003; Crooks et al., 2010). Chmura et al. (2003) estimated the average carbon sequestration rate of coastal wetlands at 210 grams of carbon per square meter per year (g C m^2/yr). Hopkinson et al. (2012) estimate the sequestration rate at 57 ± 6 to 218 ± 24 g C m^2/yr, based on sequestration rates estimated in other studies of specific intertidal wetlands. These estimates clearly stand out when compared to carbon sequestration rates of other, well-studied terrestrial systems, where rates range from 0.2-20 g C m^2/yr (Table 1.1). The variability in the rate of carbon sequestration further supports the need to more broadly assess carbon stocks distributed in different coastal wetland systems around the world.

Ecosystem Type	**Standing carbon (g/m^2) Plants**	**Standing carbon (g/m^2) Soil**	**Total global area (*$10^{12}m^2$)**	**Global Carbon Stocks (*10^{15}g) Plants**	**Global Carbon Stocks (*10^{15}g) Soil**	**Annual rate of carbon accumulation in sediment (g/m^2)**
Tropical forests	12,045	12,273	17.6	212	216	2.3-2.5
Temperate forests	5,673	9,615	10.4	59	100	1.4-12.0
Boreal forests	6,423	34,380	13.7	88	471	0.8-2.2
Tropical savannas and grasslands	2,933	11,733	22.5	66	264	
Temperate grasslands and shrublands	720	23,600	12.5	9	295	2.2
Deserts and semi-deserts	176	4,198	45.5	8	191	0.8
Tundra	632	12,737	9.5	6	121	0.2-5.7
Croplands	188	8,000	16	3	128	
Wetlands	4,286	72,857	3.5	15	225	20
Tidal salt marshes			Unknown (0.22 reported)			210
Mangroves	7,990		0.152	1.2		139
Seagrass meadows	184	7,000	0.3	0.06	2.1	83
Kelp forests	120-720	na	0.02-0.4	0.009-0.02	na	na

Table 1.1 Carbon stocks and long-term accumulation of carbon in various ecosystems (Pidgeon, 2009).

Coastal wetlands span a broad range of ecosystem types, from mangrove to marsh, and carbon sequestration dynamics are unique to each (Crooks et al,. 2010). Understanding amount of carbon stored in each system is necessary to better assess the distribution of carbon in the terrestrial reservoir. The Puget Sound, Washington, presents an important opportunity to study the carbon content and density of a particular type of coastal ecosystem, the intertidal wetland, composed of salt marsh and bare flats (Dethier, 1990). Studies of carbon in soils in intertidal wetlands are scarce, as most of these studies have taken place on the North American Atlantic and Gulf coasts, European coasts, and Chinese coasts (Zhou, et al., 2007; Bouchard & Lefeuvre, 2000; Andrews et al., 2008). For a sense of perspective, a synthesis report of global coastal ecosystem carbon density and sequestration rates, only six studies accounted for the entire west coast of North America, while the same report included a total of 84 studies from the Gulf and Atlantic coasts of North America (Chmura et al., 2003). Studies of soil carbon content in the Puget Sound (and Pacific Northwest coast in general) are absent from the literature.

Historically, the Puget Sound was estimated to have 29,500 hectares (ha) of intertidal wetland (Collins & Sheikh, 2005), accounting for approximately 1.3% of the Puget Sound lowland ecoregion (DellaSalla et al.,, 2013). Relative to the estimate of the area of intertidal wetlands globally, the Puget Sound was historically a region rich in intertidal wetland ecosystems, but due to land-use change and development, intertidal wetlands now only account for approximately 0.23% of the Puget Sound lowland ecoregion. By the late nineteenth century, approximately 38% of Puget Sound wetlands had been converted to agricultural and urban land uses (Essington et al., 2011). Currently, Puget Sound wetlands occupy only 17-19% of their historic extent, and the median size of wetlands has decreased from approximately 0.93 hectares to 0.57 hectares (Collins & Sheikh, 2005). This loss is the result of land-use change such as diking for agriculture as well as shoreline modification and armoring for industry and development. The lost intertidal wetland represents lost potential to sequester carbon (Lal, 2004; Hopkinson et al., 2012). By better understanding the carbon content and density of an existent Puget Sound intertidal wetland, natural resource managers and those working to mitigate CO_2 will be able to estimate potential additional carbon stored in the soil as a result of restoring degraded and lost intertidal wetlands (Andrews et al., 2008; Chmura, 2009).

To better understand the objectives and results of this study, it helps to be familiar with the biogeochemical properties of intertidal wetlands and past work done on this topic. The following sections of this literature review are meant to provide background in the biological, geomorphological, and chemical processes of intertidal wetlands.

1.2. *The Development of Intertidal Wetlands*

The development of intertidal wetlands is dependent on rates of sea level rise, sediment supply, and the ability to accrete vertically and and move laterally (Hopkinson et al., 2012; Hussein et al., 2004). Intertidal wetlands began to develop with the slowing of sea level rise to a rate of about 5 mm/yr, about four to five thousand years ago following the last glacial period (Hopkinson et al., 2012). When sea level rise slowed to about 3.5 mm/yr, rapid expansion of intertidal wetlands occurred (Hopkinson et al., 2012). Intertidal wetlands first colonized the tidal fringes of estuarine ecosystems and then transgressed inland as sea level rise continued to flood higher into terrestrial ecosystems (Hopkinson et al., 2012). As higher elevation terrestrial ecosystems experienced tidal submergence, forest species died and eventually halophytes (salt-tolerant plants) came to dominate (Spohn & Giani, 2012; Hussein et al., 2004).

As intertidal wetlands transgressed inland, they also accreted vertically on pace with the rate of sea level rise. Intertidal wetlands accrete vertically through sediment accumulation of organic and inorganic material from a variety of sources. Upland sediment is transported to intertidal wetlands by rivers and streams. Some of this sediment remains in the intertidal wetland, some of the sediment is transported seaward in the water beyond the intertidal wetland, and some upland sediment is washed back into an intertidal wetland upon tidal inundation (Zhou et al., 2007). When vertical sediment accretion surpasses rates of sea level rise, intertidal wetlands move seaward (laterally), in a process known as progradation (Hopkinson et al., 2012). Above- and below-ground halophyte biomass in intertidal wetlands leads to the addition of organic matter. Furthermore, these marsh plants trap sediment and organic matter as tides inundate and subside (Hussein et al., 2004). The soils of intertidal wetlands have continued to accumulate for the last five thousand years as a result of constant sediment accretion (in the absence of human intervention) (Crooks et al., 2010).

1.3. *Productivity*

Favorable nutrient and water supply position intertidal wetlands as some of the most productive ecosystems on earth (Spohn & Gianni, 2012; Chmura et al., 2003). Tidal mixing and fluvially transported upland sediments provide important nutrients for intertidal wetland plants, particularly high levels of nitrogen. (Hussein et al., 2004; Hopkinson et al., 2012). Although intertidal wetland plants can tolerate high levels of pore water salinity, the saline soils still cause physiological stress, which causes a greater nitrogen demand, and in turn drives greater root production to obtain the limiting nutrient resulting in high levels of biomass above and below ground (Chmura, 2009). These high levels of subtidal, intertidal, and emergent primary producer organisms, result in exceptionally high levels of primary production (Hopkinson et al., 2012). Duarte et al., (2005) estimate rates of gross primary production to range between 100 and 4,000 grams of g C m^2/yr. The high net primary productivity of these systems, coupled with the contribution of tidal inundation, leads to high inputs of both autochthonous (on-site input) and allochthonous (tidal input) organic matter (Spohn & Gianni, 2012; Chmura et al., 2003). Fluvially transported sediments are also important sources of organic matter. The relative importance of fluvial versus tidal sediment varies from one intertidal wetland to another, depending on the quantity of these sediments relative to one another and the relative abundance of organic matter between the two groups of sediments (Zedler & Callaway, 2001).

Multiple factors contribute to the ability of intertidal wetlands and other coastal wetlands to sequester such high levels of carbon. Tidal inundation events not only contribute sediments and nutrients, but saturate the soil, creating anoxic conditions and inhibiting aerobic decomposition (Chmura, 2009; Hussein et al., 2004; Zedler & Callaway, 2001). In intertidal wetlands, anaerobic sediments store carbon by slowing the decomposition of the *in situ* primary production, especially below-ground primary production, which results in carbon-rich peat deposits (Zedler and Callaway, 2001). Anoxic conditions impede decomposition in intertidal wetlands because heterotrophic bacteria have a reduced energy yield per unit of substrate consumed under anoxic conditions (Bastviken et al., 2004). Furthermore, oxygenase reactions which are required to break down certain compounds, cannot occur in the absence of oxygen (Schink, 2005; Bastviken et al., 2004).

1.4. *Soil texture and carbon*

These systems are not only productive because of the reasons discussed above. Physical soil properties, particularly composition of different sized soil particles in the sediment, influences the vegetation of an intertidal wetland, as well as the carbon content and density. The very presence of above-ground vegetation diminishes the scouring effect of tidal currents, which allows finer soil particles to accumulate, and subsequently improves nutrient retention (Zedler & Callaway, 2001).

It is generally accepted that there is a positive correlation between smaller soil particles, particularly clay content, and soil organic carbon preservation in soils (Krull et al., 2001; Ladd et al., 1985). The physical protection of soil organic carbon is a function of soil texture, specific surface area (SSA), and soil mineralogy (Krull et al., 2001). The SSA of soil particles increases from large to small particles. Sand particles range in size from 0.062 mm to 2.0 mm and have the lowest SSA. Silt particles range from 0.004 mm to 0.062 mm and have greater SSA than sand. Clay particles are everything under 0.004 mm and have the highest SSA (USGS, 2013). Ransom et al. (1998) showed the impact that small amounts of high SSA material can have on the total SSA of mineral particle textures. In the study conducted by Ransom et al., the presence of 1% weight of high-SSA clay (SSA of 100 m^2/g) in 1.0mm diameter sand grains (SSA of 0.001 m^2/g) increases the total SSA of the particle mixture by three orders of magnitude (Ransom et al., 1998; Krull et al., 2001). Therefore, it is clear that clay particles, because of the extremely high SSA relative to larger soil particles, have the most significant surface area to adsorb organic carbon (Krull et al., 2001.)

In addition to the fact that soil organic carbon content is positively correlated with SSA, almost all organic carbon is found within pores between mineral grains in the form of discrete particles, as molecules sorb onto the surface of minerals (Krull et al., 2001.) Kilbertus (1980) and Van der Linden et al. (1989) demonstrated that the micro-organisms responsible for the decomposition of organic matter are excluded from entering pores below a certain size. As clay content increases, the proportion of total porosity in small pores increases, resulting in the exclusion of biological decomposers, thus protecting stores of organic carbon (Krull et al., 2001).

The properties of soil texture and carbon retention described above were consistent with studies of carbon content and clay content in intertidal wetlands. Li et al. (2010) studied microbial activity, carbon content, and soil texture in the Yangtze River Estuary, China. Soil organic carbon content in sandy soil was significantly lower than clay soil (Li et al., 2010). This is consistent with other studies that have demonstrated a correlation between soil composition and particle size distribution and carbon sequestration (Zhou et al., 2007). Specifically, soil with a higher clay content has a greater ability to sequester more carbon (Ellis et al., 2003).

1.5. *Carbon and elevation*

The trend in previous studies from around the world suggests that concentration of soil carbon generally increases with elevation within the intertidal wetland studied (Spohn & Giani, 2012; Zhou et al., 2007;Li et al., 2010; Bouchard & Lefeuvre, 2000). Bouchard and Lefeuvre (2000) found that net annual primary production is significantly lower in low marsh than high and middle marsh. This is probably due to less frequent inundation at the higher levels of the marsh, allowing for higher primary production. Since primary productivity tends to positively correlate with elevation, there are greater inputs of autochthonous organic matter at higher elevations within the intertidal wetland. However, higher sites within the intertidal wetland must, at least occasionally, be submerged by tidal inundation. This keeps the anaerobic conditions in place that decrease the respiration of soil organic carbon (Li et al., 2010). Although these higher elevation areas are less frequently inundated, they still receive and accumulate tidal detritus, and the decreased tidal energy at these elevations, combined with vegetation, allows organic matter to settle at these elevations instead of being scoured by tides. However, once the conditions that define an intertidal wetland cease, such as primary productivity of halophytes and tidal inundation leading to anaerobic conditions, upland soil organic carbon stocks decline as increasing aerobic respiration releases more carbon (Mudd et al., 2009).

1.6. *Intertidal wetlands and methane*

It is important to note that CO_2 is not the only greenhouse gas of concern in the discussion of climate change mitigation. While the atmospheric concentration of CO_2 has been climbing steadily over our recent past, so too has the atmospheric concentration of methane (CH_4), from 700 ppb in 1750 to 1745 ppb in 1999 (Lal, 2004). Although other terrestrial ecosystems, such as peatlands and freshwater wetlands, have demonstrated their ability to sequester substantial amounts of carbon, there is a great deal of debate as to whether these systems in fact act as positive inputs to the global warming cycle because of their high levels of methane emissions (Table 1.2) (Sha et al., 2011; Saarnio et al., 2009). Intertidal wetland soils are inundated by tides by definition, and this frequent saturation of saline sea-water results in sulfate-rich soils. The sulfate-rich soils of coastal wetland systems inhibit the microbial activity which produces methane (Chmura, 2009).

Source	Total Global Methane emmisions Estimate (Tg CH_4/year)	Methane emission range (Tg CH_4/year)
Northern wetlands/bogs	42.7	24-72
Tropical wetlands/swamps	127.6	81-206
Oceans, estuaries, and rivers (Including intertidal wetlands)	9.1	2.3-15.6
Lakes	30	10-50
Wild animals	8	2-15

Table 1.2 Current methane emmisions from natural sources (EPA, 2013).

Bartlett et al. (1987) studied methane efflux in an intertidal wetland in the Chesapeake Bay, VA. Their results demonstrated a strong negative correlation between methane efflux (g CH_4 m^2/yr) and soil salinity (ppt). According to their work, as sulfates increase in concentration, so do sulfate-reducing bacteria and archaea. Some of these sulfate-reducing micro-organisms can consume methane, and can therefore be a controlling variable determining differences in methane concentrations along a salinity gradient (Bartlett et al., 1987). A more recent study conducted by Saarnio et al. (2009) compared methane efflux from different types of European wetlands, including various freshwater peatlands, bogs, and marshes, and salt marsh. The results showed a marked contrast in methane efflux from saltwater marsh versus freshwater systems. Freshwater marsh effluxed 0.48 teragrams of methane per square kilometer per year (Tg CH_4 km^2/yr), while saltwater marsh effluxed a meager 0.01 Tg CH_4 km^2/yr (Saarnio et al., 2009). These studies demonstrate that intertidal wetlands are negligible sources of CH_4, especially relative to freshwater wetlands. Other studies of greenhouse gas fluxes also suggest that nitrous oxide efflux from intertidal wetlands and coastal wetlands are also diminished by the presence of sulfates in the soil, although this is debated in the literature (Chmura et al., 2003). The negligible efflux of CH_4 adds to the capacity of intertidal wetlands and coastal wetland systems to act as more powerful greenhouse gas sinks than their freshwater counterparts, which in many cases act as sources (Chmura, 2009; Crooks et al., 2010).

When considering questions of climate change mitigation, it is important to take into account that CH_4 has a much higher capacity to retain heat than CO_2, making it a more potent warming agent despite its low relative atmospheric concentration compared to CO_2 (Crooks et al., 2010). Pound for pound, CH_4 is over twenty times more efficient at trapping radiant heat than CO_2 (EPA, 2013). Therefore, when climate change mitigation opportunities through restoration and conservation are prioritized, the question whether a site is a sink or source of all greenhouse gasses arises. Obviously, if climate change mitigation is one of the objectives of a conservation or restoration scenario, it is important that the site be a sink, not a source of greenhouse gasses. Since intertidal wetlands sequester carbon with negligible output of methane, these are excellent areas to protect and restore to mitigate climate change (Crooks et al., 2010).

1.7. *Soil sampling*

A common inconsistency between studies of soil carbon sequestration in coastal systems and intertidal wetlands is the depth to which soil samples are gathered. As soil depth increases, so does bulk density and the portion of minerals, and the amount of carbon stored in deeper soils diminishes (Spohn & Giani, 2012). For this reason, a significantly higher proportion of carbon is stored higher in the soil, and subsequently sampling has stayed in shallower soil depths. Generally, these studies measure soil carbon by depth increments as opposed to soil horizons. The trend in previous studies has demonstrated that concentrations of carbon decrease with depth, but the depth to which samples have been taken range from the top 20 cm (Santín et al., 2007) to 210 cm deep (Hussein et al., 2004). Most studies have ranged between 30 and 60 cm sample depths (Chmura et al., 2003; MacClellan, 2011).

1.8. *Impacts of anthropogenic change on intertidal wetlands*

So far the discussion has focused on the ability of coastal wetland ecosystems and intertidal wetlands to store carbon. The ability of these systems to store carbon may indeed mitigate a portion of anthropogenic atmospheric CO_2 (Lal, 2004; Hussein et al., 2004; Crooks et al., 2010; Hopkinson et al., 2012). However, these systems are highly vulnerable to climate change (Hopkinson et al., 2012; Hussein et al., 2004; Chmura, 2009). Recall from earlier that intertidal wetlands must accrete vertically in sync with sea level rise. Simultaneously with vertical accretion, intertidal wetlands must be allowed to transgress inland (when sediment supplies are insufficient for progradation), gaining elevation relative to sea level rise. Different intertidal wetlands have different rates of sediment accretion, depending on tidal currents, wave energy, and suspended sediment concentrations.

To minimize intertidal wetland loss, researchers suggest conserving adjacent uplands for inland transgression (Stralberg et al., 2011). As intertidal wetlands accrete ahead of sea level, carbon will continue to accumulate in intertidal wetland soils. However, when the rate of sea level rise catches up with, and surpasses the rate of sediment accretion, intertidal wetlands will essentially drown and represent unrealized potential carbon storage (Mudd et al., 2009).

Estimates of the rate sea level rise for the last 100 years are 1-2 mm/yr, however rates over the last decades have accelerated and are projected not to slow (Mudd et al., 2009; Denman et al., 2007). The average global rate of sea level rise from 1961-2003 was 1.8 ± 0.5 mm/yr. Data from the Poseidon and Jason satellites suggested that rates of sea level rise have accelerated, with a rate of 3.1 ± 0.7 mm/yr from 1993-2003 (Denman et al., 2007). Based on the current science, sea level rise in Puget Sound is likely to match global projections of sea level rise (Mote et al., 2008). The northwest coast of the Olympic Peninsula will show little apparent sea level rise due to high rates of local tectonic uplift, which exceed current rates of sea level rise (Mote et al., 2008). Data for the central and southern Washington coast are scarce, but available data suggests that tectonic uplifting is occurring in this region as well (Mote et al., 2008). Low-probability, high-impact estimates of sea level rise in the Puget Sound are 55 cm by 2050 and 128 cm by 2100. Low-probability, high-impact estimates of sea level rise are less for the central/southern coast and Northwest Olympic Peninsula, at 45 cm and 35 cm by 2050, and 108cm and 88 cm by 2100, respectively (Mote et al., 2008). Regardless of whether these worst-case estimates are realized, Puget Sound is the marine area of Washington where the effects of sea level rise will be most apparent. Because of shoreline modification and the historic loss of intertidal wetlands in the Puget Sound, remaining intertidal wetlands are in danger of being lost as sea levels continue to rise.

To add another dimension to this scenario, a substantial quantity of sediment is delivered from upland sources to intertidal wetlands through fluvial transportation (Zedler and Callaway, 2001). The seaward journey of sediment, however, has been widely interrupted in systems throughout the world. This is certainly the case in the watersheds that flow into Puget Sound. The culprit is land-use change, mostly in the form of dams built for hydroelectric power (Hopkinson et al., 2012; Mudd et al., 2009). The dams halt the seaward journey of upland sediment, allowing the sediment to descend through the water column as the current velocity diminishes. Instead of being transported to intertidal wetlands and beyond, the sediment settles on the floor of the reservoir behind the dam. For example, in the case of Alder Lake, a hydroelectric reservoir on the upper Nisqually River, sediment withheld by the dam has accumulated to 48 m deep (South Sound Science Symposium, 2012). Withheld sediment diminishes an intertidal wetland's ability to accrete vertically and maintain the pace with sea level rise necessary to ensure its continued existence (Mudd et al., 2009).

As sea levels rise, and vertical accretion is unable to keep pace because of diminished sediment supply and climate change-induced acceleration of sea level rise, intertidal wetlands will transgress inland to higher elevations. As in the case of dams, sediment, and vertical accretion, anthropogenic drivers pose a major problem for the inland transgression of intertidal wetlands as well. Shoreline armoring, agricultural dikes, and other infrastructure designed to support and protect coastal industry and development stand in between intertidal wetlands and the higher ground these systems must retreat to in order to maintain themselves (Chmura, 2009; Andrews et al., 2008). Where shoreline modifications exist prohibiting transgression, intertidal wetlands quite literally have their back up against a wall, a phenomenon known as "coastal squeeze" (Chmura, 2009).

For the reasons outlined above, it is important to continue studying these systems around the world (Chmura et al., 2009). Puget Sound intertidal wetlands soil carbon has not been thoroughly studied, and represents an opportunity to further our understanding of the allocation of carbon in the terrestrial reservoir. This work will build on previous studies by measuring soil carbon content and density in the top 30 cm of different elevations within a Puget Sound intertidal wetland. Data from a Puget Sound intertidal wetland is valuable because it contributes data from another unique location, and as noted in previous studies, carbon content and density can vary widely between locations. Furthermore, Puget Sound intertidal wetlands are under threat from a combination of land-use change and sea level rise, so demonstrating the value of intertidal wetlands to offset anthropogenic CO_2 may strengthen the argument to restore and conserve Puget Sound intertidal wetlands. The study will seek to further the body of knowledge on relationships between soil carbon and intertidal wetland elevation, salinity, tidal inundation frequency, and physical soil attributes such as grain size and presence of clay. Results of this work support the theory that coastal wetland soil contains large stores of carbon per unit area, and that carbon content and density levels will positively correlate with higher elevation zones within the intertidal wetland.

The ability of intertidal wetlands and other coastal wetlands to sequester carbon, combined with negligible methane emissions, support the theory that coastal wetlands are valuable greenhouse gas sinks and make up a small but significant portion of the terrestrial carbon reservoir (Crooks et al., 2010; Chmura et al., 2003; Hussein et al., 2004). Although these systems may mitigate a portion of anthropogenic CO_2, climate change and subsequent sea level rise threaten their continued existence (Mudd et al., 2009). In planning for restoration and conservation activities, natural resource managers should consider the benefit these systems play as greenhouse gas sinks combined with the fact that they are under threat. Losing coastal systems and intertidal wetlands would represent a loss for the terrestrial carbon reservoir. It is therefore important to understand these systems and their role in the greenhouse gas cycle, and to build upon the case for their preservation where they still exist, and their restoration to where they once belonged.

As we continue the plunge into the realm of prehistoric levels of atmospheric CO_2, increasing our understanding the global carbon cycle and all of its many components becomes more crucial every day.

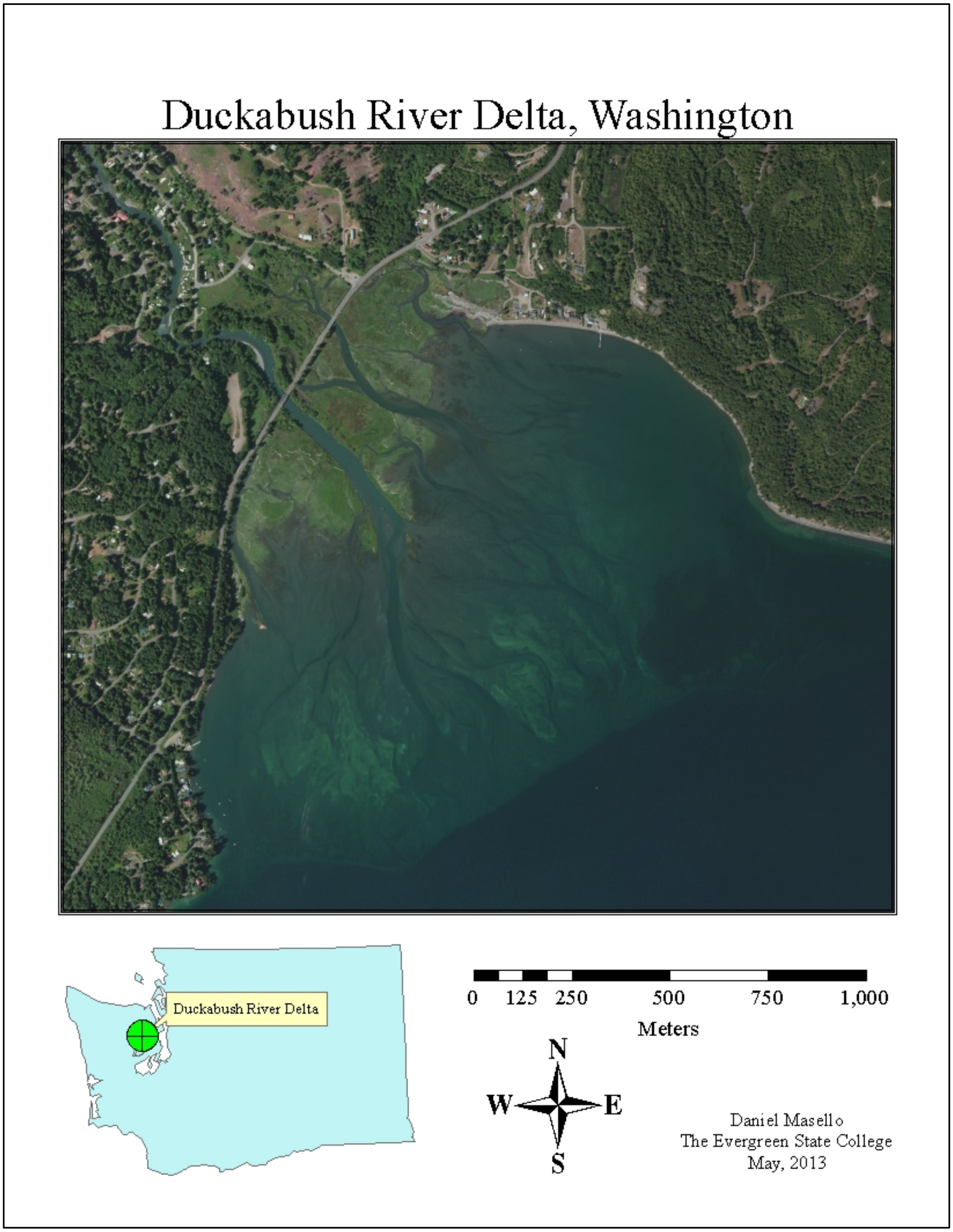
2. Manuscript

2.1. *Site description*

2.11. *Physical setting*

The Duckabush Watershed is one of four sub-watersheds (the Dosewallips, Duckabush, Hamma Hamma and Skokomish) that drain the Dosewallips-Skokomish Watershed of the eastern Olympic Peninsula, Washington state (Correa, 2003). The Duckabush River originates in the vicinity of Mount Duckabush and Mount Steele in the Olympic Mountains and flows generally eastward, draining approximately 20,207 ha (Correa, 2003). The Olympic National Park contains 11,685 ha of the watershed, and an additional 6,345 ha lie within the Olympic National Forest, together accounting for 89% of the watershed (Hood Canal Coordinating Council, 2000). The remaining portion of the watershed is mostly privately-held forest land, along with residential and park land (Hood Canal Coordinating Council, 2000; Correa, 2003). There is no commercial or industrial-zoned land in the entire Duckabush Watershed, and there are no dams along the river (Hood Canal Coordinating Council, 2000). The mainstem of the Duckabush River is approximately 39.43 km, with over fifty tributary streams for a total of 191.19 stream kilometers in the watershed (Correa, 2003). The average annual discharge is average of 11.64 cubic meters per second, with bimodal hydrology resulting in winter and spring peaks (Hood Canal Coordinating Council, 2000).

The Duckabush River terminates at its delta on the northwestern western shore of the Hood Canal, the westernmost sub-basin of the Puget Sound. The delta is located at approximately N 47°38’56”, W 122°56’05 (Map 2.1) (National Geodetic Survey, 2013).



Map 2.1 Aerial view of the Duckabush River Delta and it location within Washington state.

The Duckabush delta is approximately 60 ha. Tides at the delta range from extreme highs of 4.11 m to extreme lows of -1.19 m (NOAA, 2013). All sample points in this study were located in the eulittoral zone, between the mean high water (MHW) and mean low water (MLW) of the intertidal zone (Dethier, 1990). MHW and MLW at the Duckabush Delta are approximately 3.12 m and 0.82 m, respectively (Mojfeld, et al., 2002; NOAA, 2013).

The delta is owned and maintained by the Washington Department of Fish and Wildlife. It is open to the public for recreation, particularly shellfish gathering (WDFW, 2013).

2.12. *Pre-European contact*

Around 12,000 BP, after 2,000 years of glacial retreat and subsequent temperature and climatic changes, conditions in the region became suitable for human habitation (Mather et al., 2006). The post-glacial landscape was sparsely-vegetated, but within approximately two millennia of glacial retreat, dense forests of *Pseudotsuga menziesii* (Douglas Fir), *Thuja plicata* (Western Red Cedar), and *Tsuga heterophylla* (Western Hemlock) came to dominate the landscape. Early human habitation in the region, 9,000-5,000 BP, was characterized by upland site occupation, along river terraces, and in temporary hunting camps. Evidence of task-specific, year-round, broad-based activities date from about 4,200 BP. Permanent villages, which served as bases for other seasonal activities, were established at this same time. Salmon was the primary food source for the people of the region, supplemented by steelhead and cutthroat trout, shellfish, deer, elk, roots, bulbs, and berries (Elmandorf & Kroeber, 1992).

The Duckabush River Delta is located within the traditional territory of the Twana People, who occupied the entire Hood Canal Drainage and spoke a Salishan language unintelligible to neighboring groups (Mather et al., 2006). The Twana peoples' first contact with Europeans came in 1792 when Captain George Vancouver explored the area (Mather et al., 2006). The banks of the Duckabush River, close to the delta, was the site of a Twana-speaking winter fishing village. The name "Duckabush" is the Anglicized version of the village name, *duxwyabu's*, which means, "place of crooked-jaw salmon," (Elmandorf & Kroeber, 1992).

2.13. *Post-European contact*

Washington Territory was established in 1853, the same year Euro-American settlement began along the shores of the Hood Canal at the Duckabush and Dosewallips River Deltas (Mather et al., 2006). In 1855, the Treaty of Point-No-Point compelled many of the native people in the region to move south to the Skokomish reservation, located on the lower Skokomish River in the southwest corner of Hood Canal (Elmandorf & Kroeber, 1992).

Elwell Brinnon is considered to be the first Euro-American to settle in the region permanently. Mr. Brinnon settled at the mouth of the Duckabush River and married a Clallam woman named Kate, the sister of a local chief (Mather et al., 2006). In the 1860's, Mr. Brinnon sold his claim to Thomas Pierce and moved to a new claim at the mouth of the Dosewallips River, about 5 km to the north. The town located between the Duckabush and Dosewallips Rivers still bears the name "Brinnon". In 1859, Mr. Pierce began logging the area by hand for the Washington Mill Company in Seabeck, on the other side of Hood Canal (Mather et al., 2006).

The passage of the Homestead Act of 1862 spurred settlement in the Brinnon area during the mid-1860's. In the absence of roads, railroads, or even a dock, Brinnon remained fairly isolated, 40 miles north or south to the nearest towns through rugged terrain and dense woods. A road connecting Brinnon to Quilcene was finally built in 1896. Following the construction of the road, logging continued to be the central driver of the local economy, progressing from hand-logging to ox teams, to horse teams, to railroads, and finally to logging trucks. In 1938, the Olympic National Park was established and logging ceased within its borders (Mather et al., 2006). Logging practices in areas outside the park have continued throughout the region.

During the last century, the Duckabush River Delta has been intersected by State Route 101, truncating approximately 5.26 ha of existing intertidal wetland habitat (Correa, 2003) (Map 2.1). This truncation by SR 101 has resulted in intertidal wetland habitat prograding eastward and seaward to the north of the mainstem of the Duckabush River beyond historic boundaries. The mainstem of the Duckabush River is armored with concrete bulkheads and riprap within the intertidal zone of the wetland. Armoring along the mainstem, SR 101, and other residential armoring and development has disrupted backshore sediment recruitment (Correa, 2003). Historic intertidal wetland habitat exists to the south of the main stem of the Duckabush, but its extent has been reduced from approximately 2.43 ha to 1.46 ha. Correa (2003) estimates (conservatively) that approximately 9% of the shoreline at the Duckabush River Delta is armored.

Samples for this study were taken from within the historic habitat to the south of the Duckabush River.

2.2. *Methods*

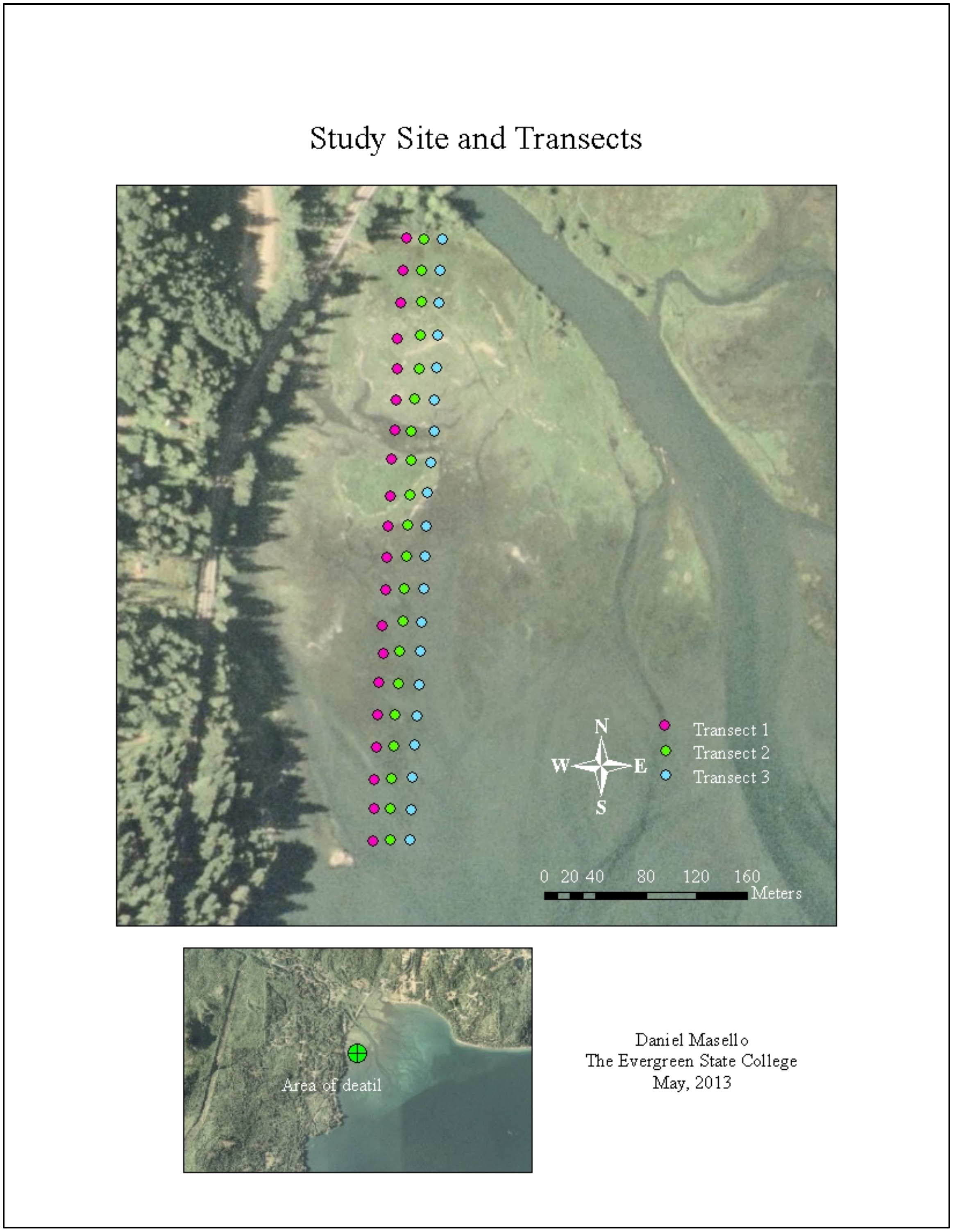
Field

2.21. *Transect Design*

Sample points were established with a grid sampling transect (Pennock, et al., 2008). The transects were oriented north to south, because the elevation gradient is most varied along this bearing, providing about a 2 m range of elevation. Other orientations would require significantly longer transects to include a similar elevation gradient. Also, deep channels in other areas of the delta presented logistical complications that could not be overcome without watercraft. This grid transect captures the eulittoral zone, with salinities ranging from mesohaline (5-18 ppt) to euhaline (30-40 ppt) in intertidal wetland soils (Dethier, 1990). The grid was composed of three parallel transects, spaced 15 m apart. Twenty samples were taken every 25 m on each of the three transects, for a total of 60 samples, and total transect length of 475 m (Map 2.2)

2.22. *Elevation*

Elevation was measured at each sample point with a *TopCon G-7* auto level and stadia rod. Base elevation was established from the National Geodetic Survey marker PID – SY1194 (47°38’56” N, 122°56’05” W) with subsequent measurements taken from the marker to the first sample point on the delta. The elevation of the marker is 6.70 m above sea level (National Geodetic Survey, 2013). The auto level, mounted on a tripod,



Map 2.2 Detail of the study site and sample locations.

was placed over the marker. From the marker, elevation measurements were taken proceeding towards the northernmost sample point of the west transect. Starting with the known elevation at the first sample point, subsequent elevation measurements were taken at each sample point. Before positioning the stadia rod, vegetation and debris was cleared at each point so that elevation would be measured from the soil surface. The stadia rod was then placed on the cleared soil surface, gently to avoid compaction, and held vertically.

2.23. *Salinity*

Approximately 70-100 cm^3 of soil from the sample point were placed in a paper coffee filter. The soil for salinity tests was taken from approximately 30 cm depth. The soil in the coffee filter was gently squeezed so as not to tear the filter. Water then dripped onto an *A366ATC* salinity refractometer, which was held under the soil in the coffee filter. Salinity readings were taken in parts per thousand (ppt).

2.24. *Soil cores*

Soil cores were extracted with a PVC electrical conduit pipe cut to 30 cm, which was beveled for cutting on one end. Samples were taken to 30 cm because this was consistent with the range of depths from previous studies (Ellis & Aherton, 2003; MacClellan, 2011; Chmura et al., 2003); the top 30 cm captures the area of the soil closest to the interface between atmospheric and terrestrial carbon reservoirs. Furthermore, sampling to greater depths rapidly becomes more and more logistically difficult. The inside diameter of the pipe was 5.2 cm, which resulted in cores with a volume of 637 cm^3. The pipe was placed on the bare soil, after it had been cleared of vegetation and debris, and driven into the ground. The pipe containing the soil core was excavated from the ground, and any soil protruding from the bottom of the pipe was cut off with a knife. The core was removed, sealed in polyethylene bag, and refrigerated at a temperature of 4-5 °C until analysis.

Laboratory

2.25. *Soil bulk density*

To obtain bulk density, samples were dried at 65 °C until a constant weight was reached (Blake and Hartge, 1986). This took anywhere from 12-36 hours depending on the water content of the sample. Once samples were oven-dry, the soil was passed through a 2.0 mm sieve (#10 sieve) to remove any coarse organic matter, coarse sand, gravel, and cobbles. The remaining soil was composed only of soil particles less than or equal to 2.0 mm. The bulk density was calculated by the oven-dry mass of soil less than or equal to 2.0 mm (M_S) divided by the sample volume (V_S), 637.37 cm^3 (Blake and Hartge, 1986):

$$\textbf{(2.1)}\quad \text{Soil bulk density } (g/cm^3) = M_S\,(g) / V_S\,(cm^3)$$

Bulk density units are grams of soil ≤2.0 mm per cubic centimeter (g/cm^3). All subsequent analyses were conducted with soil that was ≤2.0 mm in size.

2.26. *Soil texture composition*

The hydrometer method was used to obtain soil texture measurements (Kroetch and Wang, 2008; Gee and Bauder, 1986). Essentially, the hydrometer method creates a liquid suspension of soil and, upon settling, the depths of different-sized particles are measured. The hydrometer method procedure and calculations used in this study are described by Gee and Bauder (1986).

Briefly, a dispersing solution was prepared containing 50 g of sodium hexametaphosphate per liter of deionized water. Fifty to 100 g of oven-dried, sieved soil (S_{od}) was added to a liter mason jar, to which 100 mL of the dispersing solution and 300 mL of deionized water were added. The jar was then shaken vigorously for one minute and allowed to sit overnight for at least twelve hours, allowing the dispersing solution to completely soak the soil and disperse the soil particles. The soil solutions were then mixed with an electric mixer for five minutes, and then poured into 1,000 mL graduated cylinders. Deionized water was added to the soil solution until a total volume of 1,000 mL was reached in the cylinders. Soil solutions were allowed to equilibrate to a temperature of 22-23 °C. A plunger was then inserted into the cylinder and the solution was vigorously mixed to create a suspension. As soon as the plunger was removed, a timer was started and the hydrometer was gently lowered into the suspension. A reading was taken at 40 seconds ($R_{40\ s}$), and again at 7 hours ($R_{7\ h}$). The procedure was repeated for each soil sample. The hydrometer was calibrated by adding 100 mL of the dispersing solution to a graduated cylinder. To this, 900 mL of deionized water was added for a total volume of 1,000 mL. The hydrometer was lowered into the calibration solution and the scale reading was taken from the hydrometer (R_L). Content of sand, silt, and clay were calculated as follows (Gee and Bauder, 1986):

$$\textbf{(2.2)}\ \text{Sand \%} = 100 - [R_{40\,s} - R_L] \times [100 \div S_{od}]$$

$$\textbf{(2.3)}\ \text{Clay \%} = [R_{7\,h} - R_L] \times [100 \div S_{od}]$$

$$\textbf{(2.4)}\ \text{Silt \%} = 100 - [\text{sand \%} + \text{clay \%}]$$

2.27. *Organic carbon content and density*

To calculate organic carbon content of the soil samples, the loss-on-ignition (LOI) method was used (Skjemstad and Baldock, 2008: Wright et al., 2007). LOI combusts any organic matter in the soil, and the carbon content is calculated as a percent of the organic matter present in the sample. LOI only combusts organic carbon, because inorganic carbon requires a higher temperature to combust, closer to 825 °C (Wright et al., 2007). Ceramic crucibles were precombusted by baking them for five hours at 550 °C. Once cooled, the each crucible was placed on a scale and the mass was recorded. While each crucible was on the scale, 100 mg of soil from a sample was added, and the total mass of the crucible plus soil was recorded. The crucibles plus soil were covered with aluminum foil and placed in the muffle furnace for five hours at 550 °C. The crucibles were removed from the muffle furnace and placed in a desiccator for 20 minutes, and then reweighed. The organic matter content was measured as the difference between the oven-dry soil mass (S_A) and the soil mass after combustion (S_B), divided by the oven-dry soil mass (Wright et al., 2007):

$$\textbf{(2.5)}\ \text{OM \%} = 100 - [\{S_A - S_B\} \div S_A]$$

Two conversion factors were used to estimate organic carbon content of the soil. It was first calculated as 58% of the organic matter. This calculation was based on the recommended conversion factor from the EPA (Schumacher, 2002), and used by Lal (2004) in his study of terrestrial soil carbon stocks. The second conversion factor was 68%, based on the conversion factor used by MacClellan (2011) in a similar study of Oregon estuaries. These two conversion factors provide a useful range for estimates of soil carbon content. This study reports the average of this range, organic carbon at 63% of organic matter:

$$\textbf{(2.6)}\ \text{OC \%} = \text{OM \%} \times 0.63$$

Carbon density is measured as grams of organic carbon per cubic centimeter (g/cm^3). Carbon density was calculated by multiplying carbon content by soil bulk density (BD):

$$\textbf{(2.7)}\ \text{OC density } (g/cm^3) = \text{OC \%} \times \text{BD } (g/cm^3)$$

2.28. *Estimating carbon stocks*

Carbon stocks are the mass of carbon for a given surface area, unlike carbon density which measures carbon mass for a given volume. There are a variety of metrics used to convey carbon stocks, and for the purposes of consistency with carbon stock estimates from other systems, this study estimates carbon stocks in megagrams per hectare (Mg/ha). Since this study only sampled soil to 30 cm depth, this estimate only represents carbon stocks in the top 30 cm of soil, excluding any organic matter >2.0mm. To arrive at this estimate, organic carbon density (g/cm^3) we multiplied by 30 cm. This number represents the mass of carbon in one square centimeter (g/cm^2), to a depth of 30 cm. To arrive at megagrams (Mg) of carbon per hectare, carbon g/cm^2 were simply multiplied by 100, since 1 g/cm^2 = 100 Mg/ha.

This study used a GIS digital elevation model (DEM) of the Duckabush River Delta to divide the delta into zones 1, 2, and 3, and to calculate the area of the different zones (see "Statistical Analyses" section for zone designation). The area of each zone, in hectares, was then multiplied by the mean carbon stock of each zone (Mg/ha), and all three zones were added together for an estimate of organic carbon stocks in the top 30 cm of the intertidal zone of the Duckabush River Delta.

Estimates of carbon stocks at the Duckabush River Delta do not include carbon stored belowground in the form of coarse woody debris or root biomass. Because these sources of carbon are not accounted for, and because carbon density was only estimated to 30 cm depth, estimates of carbon stocks at this site should be considered conservative.

2.29. *Statistical analyses*

Bootstrap analysis was used to determine the distribution of salinity, elevation, organic matter content, organic carbon content, organic carbon density, and soil texture composition. The Shapiro-Wilkes Goodness-of-Fit test was applied to each distribution to see if data was from the normal distribution and met the assumptions of regression and analysis of variance tests. The relationships between elevation and salinity, organic matter content, organic carbon content, organic carbon density, and soil texture compositions were tested with regression analyses. Correlation analysis was used to examine the relationship between salinity and organic matter content, organic carbon content, organic carbon density, and soil texture compositions. The study site was also divided into three zones. The three zones are based on observations during laboratory analysis, in which three types of groups emerged: (1) a low-elevation, high- bulk density zone (0.92-1.59 m), (2) a mid-elevation, low-bulk density zone (1.6-2.54 m), and (3) a high-elevation, high-bulk density zone (2.55-2.81 m). These zone corresponded to field observations, with regard to apparent soil bulk density, and differences in vegetation. Most notably, zone 1 was sparsely vegetated with *Fucus* spp. (rockweed) or bare. Zones 2 and 3 were densely vegetated mostly with *Distichlis spicata* (seashore saltgrass). This field work was conducted in February, so vegetation was mostly dormant and not flowering, making thorough plant identification difficult. The different zones acted as categorical variables by which to analyze significant differences of elevation, salinity, organic matter content, organic carbon content, organic carbon density, and soil texture composition through analysis of variance (ANOVA). Significant differences between zones were assessed with Tukey's Honest Significant Difference test.

2.3. *Results*

An analysis of the results indicated that surface soil organic carbon content is driven largely by the increased proportion of smaller particles, particularly silt, in the sediment. Organic carbon density was driven by both the carbon content of the sediment as well as the soil bulk density of the sediment. Bulk density did not correlate linearly with elevation; two distinct bulk densities among three distinct elevation ranges appeared in the results. Samples with high bulk density and high silt content had the highest carbon density. This is discussed in more detail below.

2.31. *Soil bulk density*

Soil bulk density values ranged from 0.19 to 0.95 g/cm^3. The mean and median bulk density was 0.57 and 0.56 g/cm^3, respectively. The relationship between bulk density and elevation was not linear (Figure 2.1); mean soil bulk density was significantly lower in zone 2 at 0.39 g/cm^3 than zones 1 or 3, which had virtually identical bulk density values at 0.69 g/cm^3 ($F[2,57]=21.25$, $p<0.01$) (Figure 2.2), despite nearly a meter of elevation separating them. Organic matter and carbon content were significantly correlated with soil bulk density ($r^2=0.56$, $p<0.\ 01$) (Figure 2.3), but soil bulk density and carbon density were not significantly correlated.

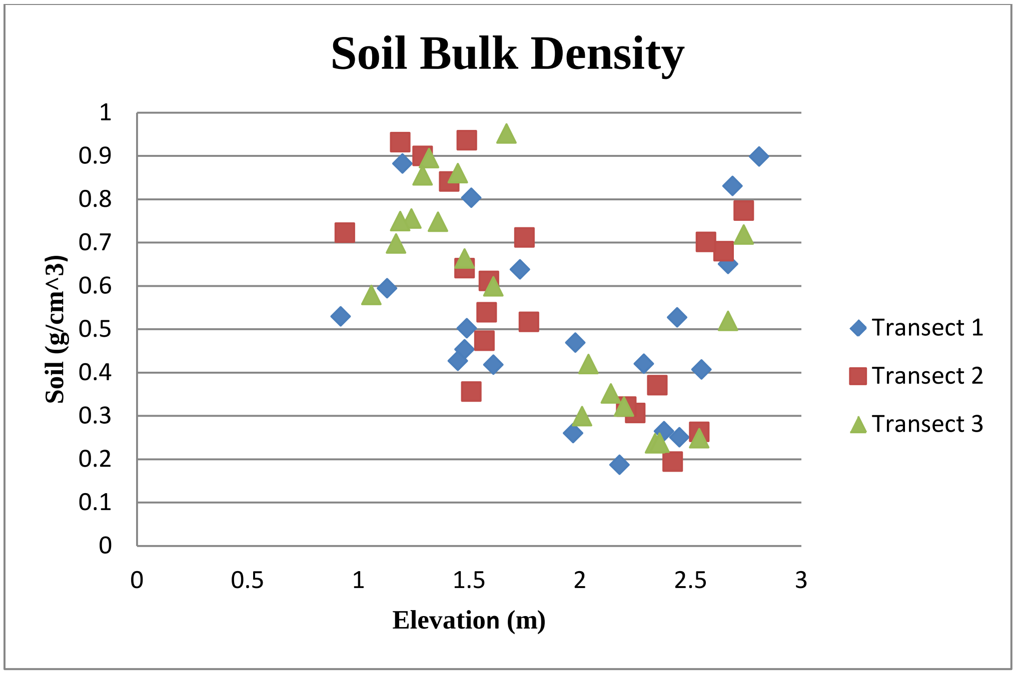


Figure 2.1 Soil bulk density against elevation.

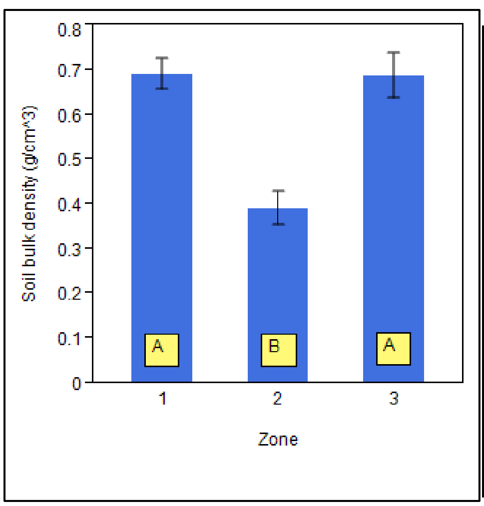


Figure 2.2 Analysis of variance of soil bulkdensity between different zones. Different letters indicate significant differences between zones. Error bars represent one standard error from the mean.

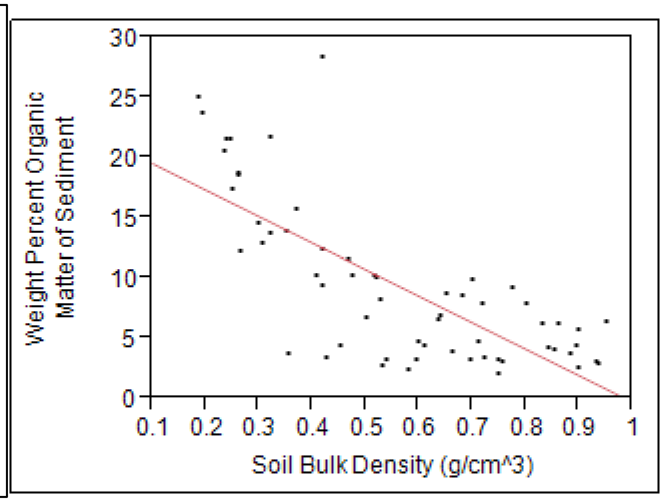


Figure 2.3 Organic matter content against bulk density.

2.32. *Elevation and salinity gradients*

A strong negative relationship between salinity and elevation (r^2=0.80) was observed (Figure 2.4). Salinity values ranged from 12 ppt to 33 ppt, and elevation ranged from 0.92 to 2.81 m. The mean salinity was 22.8 ppt. The mean elevation value was 1.87m and the median was 1.74m.

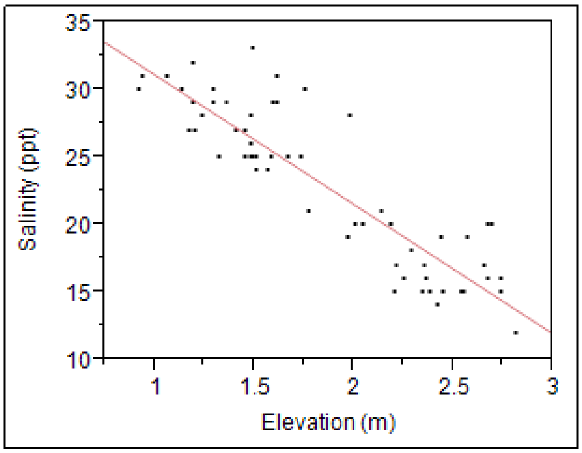


Figure 2.4 Salinity in parts per thousand against elevation in meters.

2.33. *Organic carbon content and density*

The organic matter content (and subsequently, organic carbon content) varied widely with a maximum value of 28.27% at 2.04m elevation, and a minimum value of 2.04% at 1.19m. Organic matter and therefore carbon content were positively correlated with elevation (r^2=0.37, $p<0.01$) and negatively correlated with salinity (r^2=0.46, $p<0.01$) (Figures 2.5, 2.6). Organic carbon content ranged from 1.29% to 17.81%. This positive correlation, however, does not capture the drop-off in organic matter content at some of the high-elevation, high- bulk density sites of zone 3. Therefore, it is useful to look at for differences between the three zones. Organic matter and organic carbon content varied significantly between all three zones, with zone 1 < zone 3 < zone 2 ($F[2, 57]$=36.83, $p<0.01$) (Figure 2.7, 2.8).

Estimates of organic carbon content (percent of sediment that is organic carbon) were multiplied by soil bulk density (mass of soil in a given volume) for an estimate of organic carbon density (g/cm^3). Carbon density values ranged from 0.008 g/cm^3 to 0.075 g/cm^3. Regression analysis displayed a modest positive correlation between elevation and organic carbon density (r^2=0.38, p<0.01) (Figure 2.9) and negative correlation between organic carbon density and salinity (r^2=0.32, p<0.01). Mean organic carbon density of zone 3 was greater than that of zone 2, 0.035 g/cm^3 and 0.031 g/cm^3, but the difference was not statistically significant. Zones 2 and 3 both showed significantly greater carbon density than zone 1, with a value of 0.018 g/cm^3 ($F[2,57]$=20.35, p<0.01) (Figure 2.10).

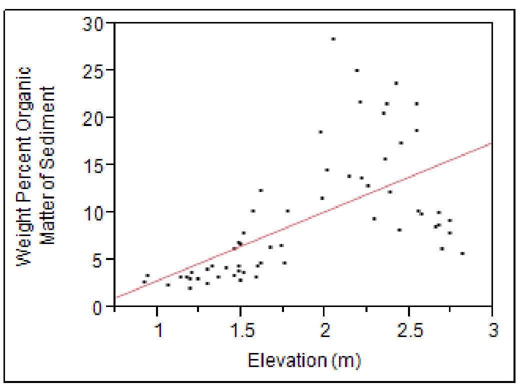


Figure 2.5 Organic matter content against elevation.

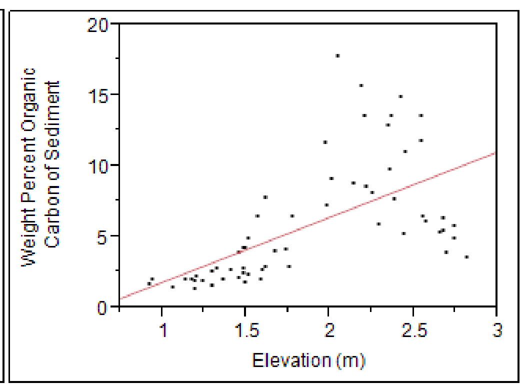


Figure 2.6 Organic carbon content against elevation.

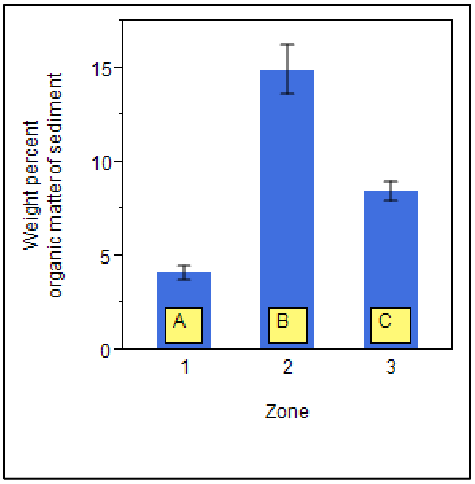


Figure 2.7 Analysis of variance of organic matter content between different zones. Different letters indicate significant differences between zones. Error bars represent one standard error from the mean.

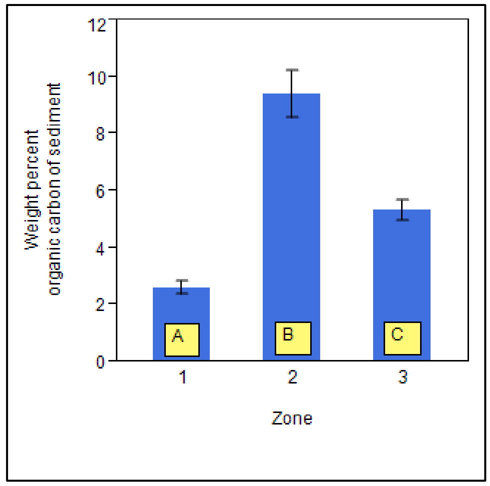


Figure 2.8 Analysis of variance of organic carbon content between different zones. Different letters indicate significant differences between zones. Error bars represent on standard error from the mean.

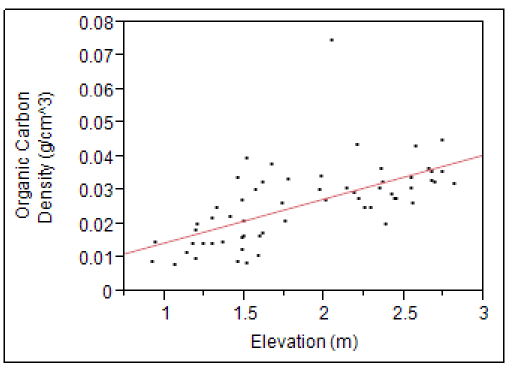


Figure 2.9 Organic carbon density against elevation.

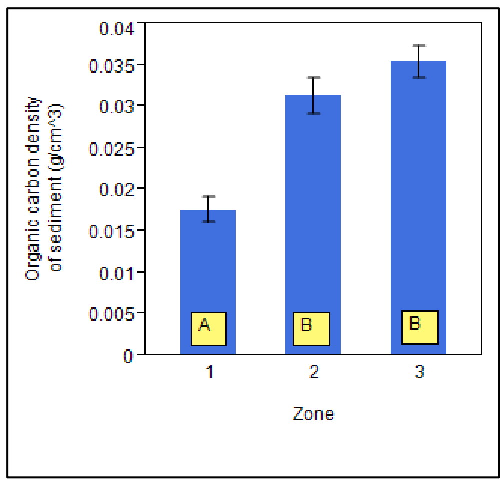


Figure 2.10 Analysis of variance of organic carbon density between different zones. Different letters indicate significant differences between zones. Error bars represent one standard error from the mean.

2.34. *Soil texture composition*

Hydrometer analysis sheds light on relationships between soil texture composition and other variables. Generally, sand and silt content displayed an inverse relationship, with sandier soil at lower elevations and siltier soil at higher elevations. Sand content ranged from 22% to 85%, and silt content ranged from 10% to 66%, with means of 49%

and 42%, respectively. The clay content was the least variant, with a mean of 9% and a range between 5% and 16%.

Sand content displayed a significant negative correlation with elevation (r^2=0.63, p<0.01) (Figure 2.11). The mean sand content was significantly higher in zone 1 (67.62%), than zone 2 (35.61%) or zone 3 (35.04%). Although the mean sand content was greater in the zone 2 than the zone 3, the difference was not significant (F[2,57]=44.57, p<0.01) (Figure 2.12).

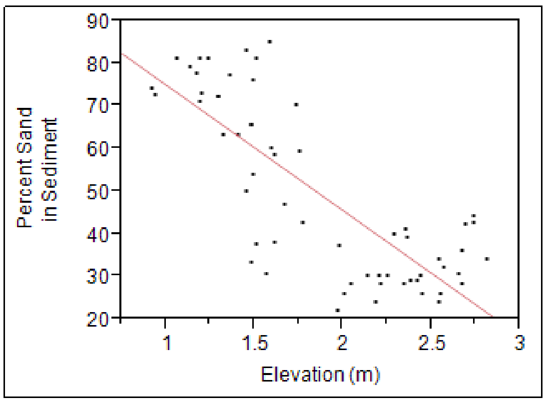


Figure 2.11 Sand content against elevation.

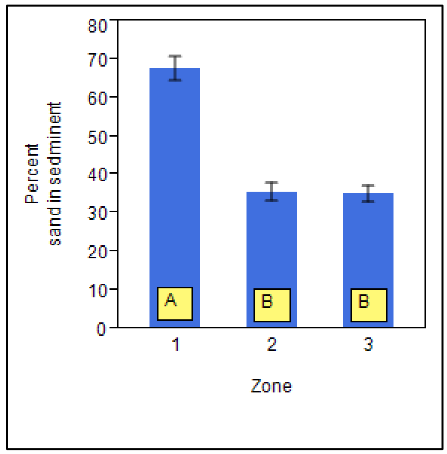


Figure 2.12 Analysis of variance of sand content between different zones. Different letters indicate significant differences between zones. Error bars represent one standard error from mean.

The silt content displayed a significant positive correlation with elevation (r^2=0.64, p<0.01) (Figure 2.13). Mean silt content in zones 1, 2 and 3 were 24.38%, 54,71%, and 55.19%, respectively. As with sand content, silt content did not vary significantly between zones 2 and 3, but both were significantly greater than zone 1 (F[2,57]=49.76, p<0.01)(Figure 2.14).

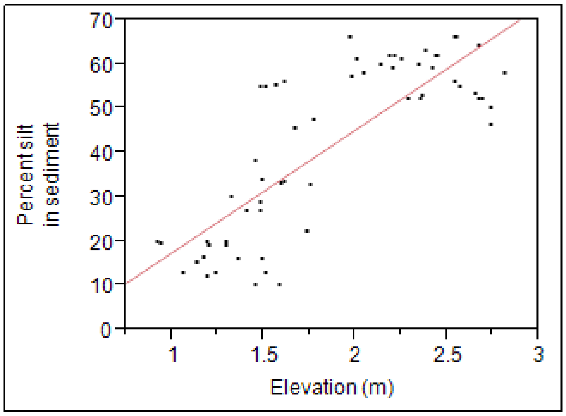


Figure 2.13 Silt content against elevation.

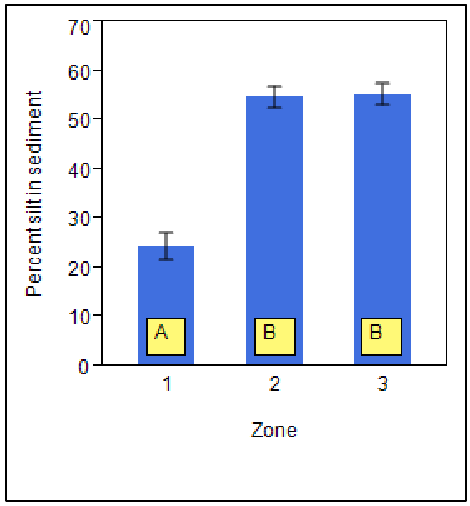


Figure 2.14 Analysis of variance of silt content between different zones. Different letters indicate significant differences between zones. Error bars represent one standard error from the mean.

Correlation analysis between clay and elevation showed a slight positive relationship, however the correlation coefficient was quite low (r^2=0.14, p<0.01) (Figure 2.15). Clay did not vary significantly between any of the zones, with means of 8.00%, 9.68%, and 9.78% in zones 1, 2, and 3, respectively (F[2,57]=3.45, p=0.04) (Figure 2.16).

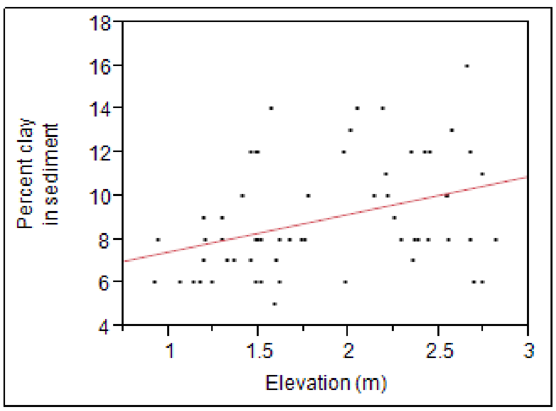


Figure 2.15 Clay content against elevation.

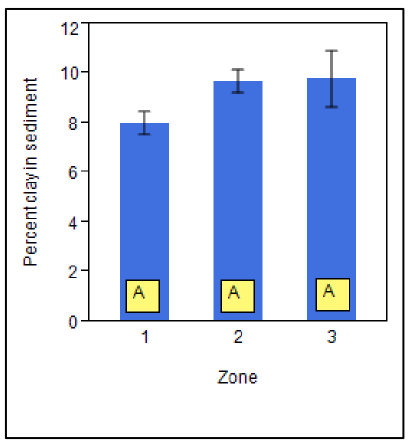
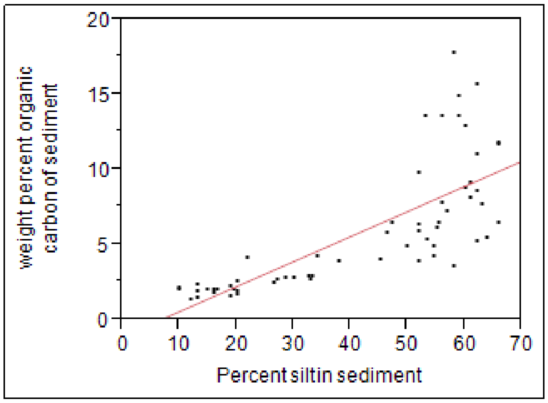


Figure 2.16 Analysis of variance of clay content between different zones. Different letters indicate significant differences between zones. Error bars represent one standard error from the mean.

Sand, silt, and clay all displayed a significant relationship with organic matter content and subsequently with organic carbon content. The relationship was strongest between silt, positively correlated (r^2=0.57, p<0.01), and sand, negatively correlated

(r^2=0.60, p<0.01) and organic carbon content, and somewhat more modest between clay, positively correlated (r^2=0.32, p<0.01) and organic carbon content (Figure 2.17-2.19).

Organic carbon density also displayed significant relationships with all three soil texture classes. As with organic matter and carbon content, sand was negatively correlated with organic carbon density (r^2=0.53, P,0.01), while silt and clay were both positively correlated with carbon density (r^2=0.51, p<0.01; r^2=0.30, p<0.01) (Figure 2.20-2.22).



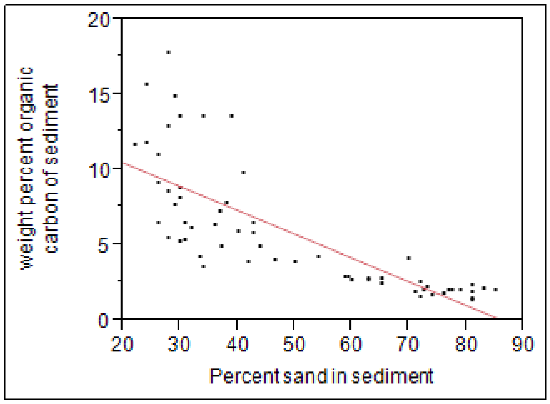
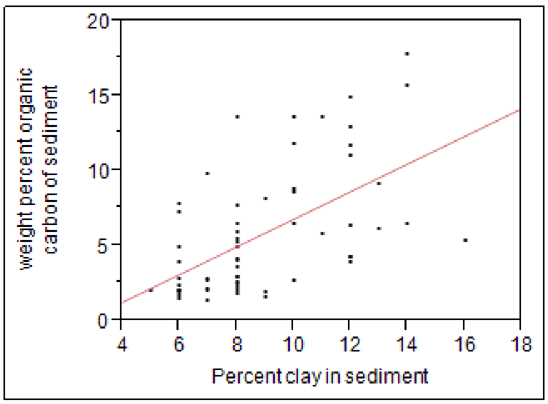


Figure 2.17 Organic carbon content against silt content. **Figure 2.18** Organic carbon content against sand content.



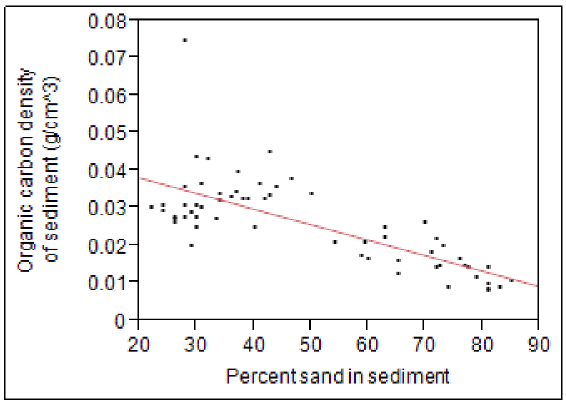


Figure 2.19 Organic carbon content against clay content. **Figure 2.20** Organic carbon density against sand content.

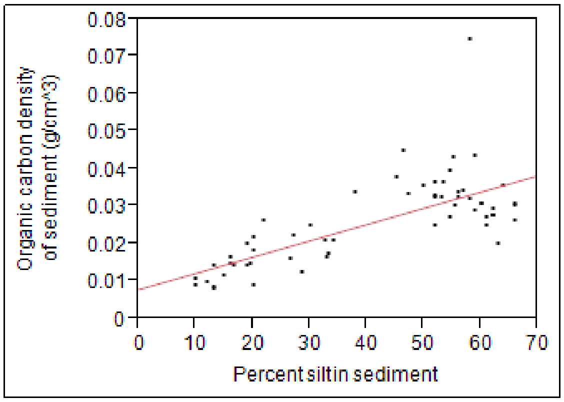


Figure 2.21 Organic carbon density against silt content.

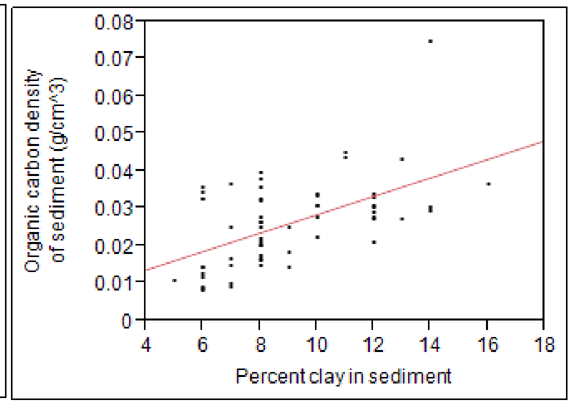


Figure 2.22 Organic carbon density against clay content.

2.4. *Discussion*

2.41. *Relationships between organic carbon, soil texture, and soil bulk density*

The correlation results of this study support the hypothesis that as elevation increases, so does the organic carbon content and density. The patterns from this study are consistent with other studies of carbon distribution in intertidal wetlands. Spohn & Gianni (2012) measured organic carbon stocks against an inundation frequency gradient in intertidal wetlands of the north German coast. Because inundation frequency is a proxy for elevation, their data is useful for comparison (Zedler & Callaway, 2001). The results of their study showed that sites that were least frequently inundated had the highest organic carbon density, while the most frequently inundated sites had the least organic carbon density (Spohn & Gianni, 2012).

Analysis of variance between zones of this study show that other factors besides elevation contribute to carbon content and density, most importantly the soil bulk density and the silt and sand content of the sample. Zone 3 displayed the highest mean carbon density value, 0.035 g/cm^3, while zone 2's mean value was only slightly lower, with a density of 0.031 g/cm^3, although these differences are not statistically significant (Figure

2.10). The soil of zone 2 exhibits significantly lower bulk density values relative to zone 3, while the organic carbon content of zone 2 is greater than zone 3. In the case of this study, the significantly higher mean soil bulk density of zone 3 compensated for the lower organic carbon content in this zone, allowing zone 3 to have the greatest mean carbon density, despite having lower carbon content than zone 2.

This raises another question: if samples from zone 1 and zone 3 have about the same mean soil bulk density (Figure 2.2), why does zone 3 have a significantly higher value for carbon content and density than zone 1? In this case, soil texture composition explains a large part of why zone 3's carbon density is higher than zone 1's. As mentioned earlier, silt content was positively correlated with carbon content and density (Figures 2.17, 2.21) Because silt has greater SSA than sand, and SSA has been previously shown to positively correlate to soil carbon content, it follows that zones 1 and 3, with nearly identical soil bulk density, have significantly different mean carbon content. Furthermore, zone 3 is vegetated salt marsh, while zone 1 contains soil sparsely vegetated with Fucus spp., or bare sediment. Therefore, a logical conclusion is that greater above and below-ground primary productivity at zone 3 drives the higher carbon content of the soil. The very presence of above-ground biomass contributes to higher carbon content in the following two ways. First, tidal organic matter input, in the form of marine floral and faunal detritus, as well as coarse woody debris, is transported to higher-elevation areas with the tide, and when the tide recedes, the dense mat of vegetation snags and catches a great deal of this tidal organic matter input, keeping it in the higher elevation zones, and (Zedler & Callaway, 1998). Further, tidal current energy could be dissipated by dense vegetation, decreasing the scouring action and allowing silt to remain in place and accumulate; in areas without vegetation, tidal energy scours silt and clay and leaves mostly heavier, coarser soil particles, which retain much less organic carbon (Zedler & Callaway, 1998). This immediate organic matter input, coupled with the carbon retaining capacity of high-SSA soil results in zone 3 having the greatest mean carbon density value of all three zones. Although zone 3 has the highest mean value of carbon density, it is not significantly different than zone 2, which had a much greater carbon content but much lower soil bulk density.

Li et al. (2010) looked at soil organic carbon content and density by sampling along two transects at the Chongming Island in the Yangtze River Delta, China. Li et al. divided the transects into four zone (high tidal, mid-tidal, low tidal, and bare flat). These zones also serve as a proxy for elevation. Soil organic carbon density was significantly lower in their "bare flat" zone than low, mid-, or high tidal zones. These results are consistent with the results of this study, in which the low zone had significantly lower organic carbon density. Furthermore, both the Li et al. study and this study showed no significant difference of organic carbon density between zones above the lowest zone. This does not come as a surprise; high inputs of autochthonous organic matter from on-site primary production of vascular plants, and allochthonous organic matter from tidal and upland detritus that is deposited in higher tidal zones result in greater organic matter content than in low tidal zones (Spohn & Gianni, 2012; Chmura et al., 2003).

J. Zhou et al. (2007) also conducted a study in the Yangtze estuary in China, examining spatial variations in carbon. Their results for organic carbon content were considerably lower than organic carbon content from the Duckabush River Delta, however the clear trend in their data showed that as elevation decreased, so did organic carbon content (Zhou et al., 2007).

Consistent with other studies (Zhou et al., 2007; Li et al., 2010), carbon was correlated with soil texture. Clay content and organic carbon content in the Yangtze River Estuary displayed a strong significant positive relationship (r^2=0.77, $p<0.01$) (Zhou et al., 2007). At the Duckabush River Delta, the relationship between clay content and organic carbon content was also significant, but less strong (r^2=0.32, $p<0.01$) (Figure2.19). The positive correlation between silt content and organic carbon content, and the negative correlation between sand and organic carbon content, accounted for more variation in carbon content (silt: r^2=0.57 and sand: r^2=0.60, p<0. 01) (Figures 2.17, 2.18). Likewise, organic carbon density was most strongly negatively correlated with sand content at the Duckabush River Delta (r^2=0.53, $p<0.01$), followed by a positive correlation with silt content (r^2=0.51, $p<0.01$), and least strongly correlated with clay (r^2=0.30, $p<0.01$) (Figures 2.20-2.22). Although the relationship between soil texture and carbon density and content is not as strongly correlated as other studies, higher percentage smaller particle sizes, clay and silt, were both positively correlated with carbon content and density, while higher percentages of sand particles were negatively correlated with both carbon content and carbon density. This is supported by the literature, which agrees that organic carbon is positively correlated with specific surface area, and specific surface area is negatively correlated with soil texture size (Krull et al., 2001).

Organic carbon density at the Duckabush was driven largely by soil texture composition as discussed above, but variations in soil bulk density also drove the organic carbon density. Bearing in mind that soil texture composition is correlated to carbon storage, it is important to think about how the different zones of the study site came to be. The difference in soil bulk density between zones 2 and 3 is particularly interesting. Both zones are composed of vary similar proportions of sand, silt and clay, but the soil of zone 3 is so much more dense than zone 2. One factor that may influence bulk density of the surface soil is below-ground biomass. During the process of drying the soil samples and passing them through a sieve, zone 2 consistently contained more of below-ground biomass (plant roots) than zone 3. The removal of this biomass represented a larger portion of the original mass of samples from zone 2 than zone 3. Zones 2 and 3 were vegetated mostly by *Distichlis spicata*. Zone 3 did support some, which was absent from lower elevation sites in zone 2. Zone 2 supported *Salicornia virginica* (Pickleweed), while zone 3 did not. Because zones 2 and 3 were both dominated by *Distichlis spicata*, it is difficult to say that plant differences in plant communities account for the difference in carbon content. An in-depth study of plant communities and their relationship with soil organic carbon would be useful to understanding these differences. Because zones 2 and 3 have approximately the same silt content, the soil bulk density difference between these sites appears to be a stronger driver of carbon density, which appears to be driven by below-ground biomass. However, given that there isn't a strong compositional difference between plant communities, it is difficult to say why the below-ground biomass is appears greater in zone 2. Unfortunately, the below-ground biomass was not measured in this study, because it was too difficult to separate that biomass from the soil samples through the 2.0 mm sieve.

Another possible explanation for this difference in soil bulk density is elevation, and subsequently tidal inundation. Because zone 2 is lower in elevation than zone 3, it spends more time submerged, increasing the average water content of the soil relative to zone 3. Since zone 3 spends less time saturated than zone 2, and its soil could be more consolidated without the dispersing influence of water and tidal energy. A third possible explanation for the increased bulk density may be compaction by elk. At the study site, in zone 3, there was evidence of elk moving from the wooded hills down to the river in the form of tracks, scat, and evidence of browsing. The pressure of a herd of elk may be enough to cause significant compaction of the surface soil, resulting in a much higher bulk density than zone 2. Dethier's (1990) A Marine and Estuarine Habitat Classification Systems for Washington State notes that eulittoral marsh, specifically including the Duckabush River Delta, is frequently used by deer and elk, supporting the notion that elk may indeed contribute to soil compaction.

This study has shown that at the Duckabush River Delta, carbon content and density is positively correlated with silt content. Zones 2 and 3 had about the same silt content, while zone 1 had only about half the silt content. Because zone 1 is the lowest elevation zone, it contains the most saline water, spends the most time tidally submerged, and subsequently supports the least vegetation. In the relative absence of vegetation, zone 1 does not have the on-site input of organic matter that zones 2 and 3 have. Without above ground biomass, zone 1is totally exposed to the full force of tidal energy, waves and freshwater flows (Dethier, 1990), subsequently, organic matter and silt that settle in zone 1 is likely to be scoured and moved by incoming tides into zones 2 and 3, or removed by outgoing tides. Because of this scouring, a greater proportion of heavier soil particles, as well as gravel and cobble are left. This results in zone 1 having high bulk density but low organic carbon content and density.

2.42. *Organic matter, organic carbon, and carbon stock density*

The range of organic matter and carbon content in this study was consistent with the range in MacClellan's 2011 study of carbon content in Oregon tidal wetland soils. The maximum organic matter content in MacClellan's was 27.75% with a corresponding organic carbon content of 18.87% (they assumed that 68% of the organic matter was comprised of carbon). By comparison, this study produced a maximum organic matter content of 28.27% and organic carbon content of 19.22% (calculated at 68% of organic matter for the basis of comparison). Minimum organic matter and carbon content varied more widely between the two studies, probably because the minimum carbon content of this study came from a zone 1 sample with little or no vegetation and a very high sand content. The MacClellan study minimum content was 9.28% organic matter and 6.31% organic carbon, while the Duckabush minimum content was 2.04% organic matter and 1.39% organic carbon.

Chmura et al. (2003) estimated average soil carbon density globally in tidal, saline wetland soils. They found the average soil organic carbon density of all sites to be 0.043 ± 0.002 g/cm^3, but this included mangroves as well as salt marshes. The global intertidal wetland average carbon density was 0.039 ± 0.003 g/cm^3. This estimate is somewhat higher than the average carbon density at the Duckabush River Delta, which is 0.026 ± 0.002 g/cm^3. However, average carbons density at the Duckabush River Delta excluding the zone 1 was 0.033 g/cm^3, which is more consistent with the finding of Chmura et al.

Chmura et al. also note that carbon stocks may increase at lower latitudes due to higher productivity in warmer climates, and the Duckabush River Delta is at a higher latitude than many of the sites included in the Chmura et al. study. Sites at similar latitudes to the Duckabush River Delta included in the Chmura et al. study generally exhibited similar density of carbon stocks, but there is noticeable variation in carbon stocks in different regions of the world (Table 2.1).

Region	Number of Sites	Latitude Range	Longitude Range	Mean carbon g/cm^3
Gulf of Mexico	27	28.4-30.4 °N	84.2-96.8 °W	0.051
Northeastern Atlantic	12	51.5-55.5 °N	0.7-8.4 °E	0.033
Mediterranean	1	43.3 °N	4.6 °E	0.073
Northeastern Pacific*	6	32.5-48.9 °N	117.1-125.5 °W	0.019
Northwestern Atlantic	57	35.0-47.4 °N	63.2-76.4 °W	0.036

Table 2.1 Mean carbon stock density in various regions of the world (Chmura et al., 2003). *Northeastern Pacific does not include results of Duckabush River delta study.

The six Northeastern Pacific sites included in the Chmura et al. study had a mean carbon density of 0.019 g/cm^3, which is less than half the density of the global mean (Chmura et al., 2003). The carbon density found at the Duckabush River Delta is well within the range of other sites studies in the Northeastern Pacific (Table 2.2).

Site	Latitude (°N)	Longitude (°W)	Carbon g/cm^3
Tijuana Slough 1, CA	32.5	117.1	0.018
Tijuana Slough 2, CA	32.6	117.1	0.017
Tijuana Slough 3, CA	32.6	117.1	0.040
Alviso, San Francisco Bay, CA	37.5	122.0	0.009
Bird Island, San Francisco Bay, CA	37.6	122.2	0.014
Duckabush River Delta, WA	47.6	122.9	0.026
Uculet, BC	48.9	125.5	0.017

Table 2.2 Comparison carbon density at salt marsh sites in the NE Pacific (Chmura et al., 2003).

2.43. *Variations in organic carbon and estimating carbon stocks at the Duckabush River Delta*

To estimate carbon stocks at the Duckabush River Delta, the area of each zone throughout the entire delta was calculated using DEM and categorizing the zones by their corresponding elevations. The total area of each zone was multiplied by megagrams of organic carbon per hectare as described in the methods (Table 2.3). This study estimates of total carbon stored in zones 1, 2 and 3 at the Duckabush River Delta to be

Zone	**OC %**	**OC density (g/cm^3)**	**OC stock (Mg/ha)**	**Area (ha)**	**Total C (Mg)**	**% of total C found in Duckabush River Delta**	**% of total area of Duckabush River Delta**
1	2.60	0.018	52.875	20.79	1099.27	26.72	40.45
2	9.39	0.031	93.875	19.19	1801.46	43.78	37.33
3	5.32	0.035	106.292	11.42	1213.86	29.50	22.22
Total				50.75	4049.96		

Table 2.3 Estimates of total carbon stocks, carbon per hectare, and carbon densities of different zones throughout the entire Duckabush River Delta. All values represent carbon found within the top 30 cm of soil and exclude coarse organic matter great than 2.0 mm. This estimate should be considered conservative.

approximately 4,050 Mg of carbon in the top 30 cm of soil, excluding coarse organic matter.

Depth increments from other studies vary widely, from 20-210 cm, but the highest concentrations of carbon was consistently found in the top 50 cm of soil. Unfortunately, gathering soil samples from a depth greater than 30 cm would have become exponentially more difficult to extract, due to logistical constraints of this study.

Furthermore, it is unclear in other studies how soil bulk density is measured (what size of soil particles are excluded by sieving). The depth increment and soil bulk density measurement variations between studies may represent a source of error in comparing estimates. In higher elevation zones at the Duckabush River Delta, organic-rich soil may be several meters deep as a result of years of accretion. Assuming this is the case, the reported carbon stocks per unit area are very low.

For the sake of comparison, consider other terrestrial system carbon stocks (Table 2.4). The conservative estimates of carbon stocks (Mg/ha) at the Duckabush River Delta may at first glance appear comparable to ecosystems such as temperate forests and tundra, and perhaps may seem weak compared to freshwater wetlands. However, estimates from the Duckabush River Delta only include the top 30 cm of soil and exclude any coarse organic matter. Because intertidal wetland soils, including at the Duckabush Delta, are constantly accreting and sequestering carbon at a rate far exceeding other terrestrial systems (Table 2.4), significantly more carbon is stored at the Duckabush Delta than these informal estimates indicate. Furthermore, the estimates of carbon stocks in other terrestrial systems listed in Table 2.4 include the entire soil column, not just the top 30 cm.

Biome	**C density (Mg/ha)**	**C sequestration (g/m^2y^1)**
Tundra	105	0.2-5.7
Boreal/Taiga	343	0.8-2.2
Temperate	96	1.4-12.0
Tropical	123	2.3-2.5
Wetlands	723	20.0
Duckabush River Delta*	53-106	Unknown

Table 2.4 Estimated carbon density and sequestration rates comparing the Duckabush river Delta to other terrestrial systems. *The C density range for the Duckabush river delta only accounts for the top 30cm of soil and does not include buried organic material >2.0 mm diameter. (Data sourced from: Lal, 2005; Pidgeon,2009).

2.44. *Climate change, sea level rise, and "coastal squeeze"*

In addition to the ability of intertidal wetlands to sequester carbon, intertidal wetlands produce negligible methane. Gail Chmura (2009) summarizes the significance of intertidal wetland soil organic carbon thus:

> When one considers feedbacks to climate, each molecule of carbon dioxide sequestered in soils of tidal salt marshes and their tropical equivalents, mangrove swamps, probably has greater value than that stored in any other natural ecosystem, due to lack of production of other greenhouse gases.

It is important to convey this information to people in decision-making positions where the opportunity to conserve or restore intertidal wetlands is a real possibility. The general public may easily grasp the value that intertidal wetlands and coastal systems have for fish and wildlife habitat, recreation, etc., but communicating the ability of intertidal wetlands and coastal wetlands to mitigate anthropogenic CO_2 emissions is the real challenge for the scientific community to convey to the public.

One of the main challenges for intertidal wetlands and coastal wetlands to mitigate climate change is the history of land-use change and reclamation associated with intertidal wetlands. As mentioned earlier, anthropogenic CO_2 is a key driver of global warming and climate change (IPCC, 2007). Fossil fuel combustion is most frequently associated with anthropogenic CO_2, but land-use change has made significant contributions to the atmospheric reservoir of anthropogenic CO_2 (Crooks et al., 2010; Hopkinson et al., 2012). In the Puget Sound, where the Duckabush River Delta is located, opportunities to restore intertidal wetlands abound, because less than 20% of historical intertidal wetlands are left. Since it is well-documented that intertidal wetland soils sequester carbon at a high rate (Chmura et al., 2003; Pidgeon, 2009; Hopkinson et al., 2012) restoring the areal extent of regional intertidal wetlands could cause more carbon to be locally sequestered. However, it will be important to first study the carbon stocks of degraded intertidal wetlands in Puget Sound before estimating how much additional carbon could be locally sequestered due to restoration. In addition, carbon content and density should be studied in sites that are undergoing restoration, such as the Nisqually River Delta or Stillaguamish River Delta. This would provide valuable insight into degraded and restored wetland carbon dynamics relative to reference conditions, such as at the Duckabush River Delta. Furthermore, in degraded intertidal wetlands where saline water has been excluded due to diking and shoreline armoring, it is likely that the presence of sulfate-reducing bacteria that consume methane is diminished or absent (Bartlett et al., 1987; Zedler & Callaway, 2001). Returning saline influence to these soils may subsequently inhibit local methane flux, further mitigating a fraction of local greenhouse gas emissions (Crooks et al., 2010). If Puget Sound intertidal wetlands are restored, they may indeed mitigate past land-use change that led to anthropogenic inputs of greenhouse gasses, but further studies regarding the points made above are needed to support this claim.

Unfortunately, in the Puget Sound, as well as other coastal areas, shorelines have been developed and armored, prohibiting the inland transgression of intertidal wetlands, which is necessary to ensure the survival of intertidal wetlands in the face of sea level rise. Although intertidal wetlands grow by vertically accreting sediment, if the rate of sediment accretion is insufficient to keep pace with sea level rise, intertidal wetlands will move vertically by transgressing inland and upland, rather than by accreting sediment. However, in areas where shorelines have been armored and developed, intertidal wetlands have nowhere to transgress to, and are trapped between and armored shoreline and rising seas. Eventually, rising sea levels will outpace vertical accretion, an intertidal wetlands will drown (Figure 2.23). This phenomenon is called “coastal squeeze” (Chmura, 2009). As these systems drown carbon sequestration will slow and eventually halt in the absences of primary production and sedimentation.

Opportunities to remove shoreline armoring and other shoreline modifications should be seriously considered; this represents an opportunity to increase local carbon storage and offset anthropogenic CO_2. Cost-benefit analyses of restoring and protecting intertidal wetlands and other coastal systems should be undertaken to compare the costs and benefits of greenhouse gas mitigation by restoring these systems versus other strategies.

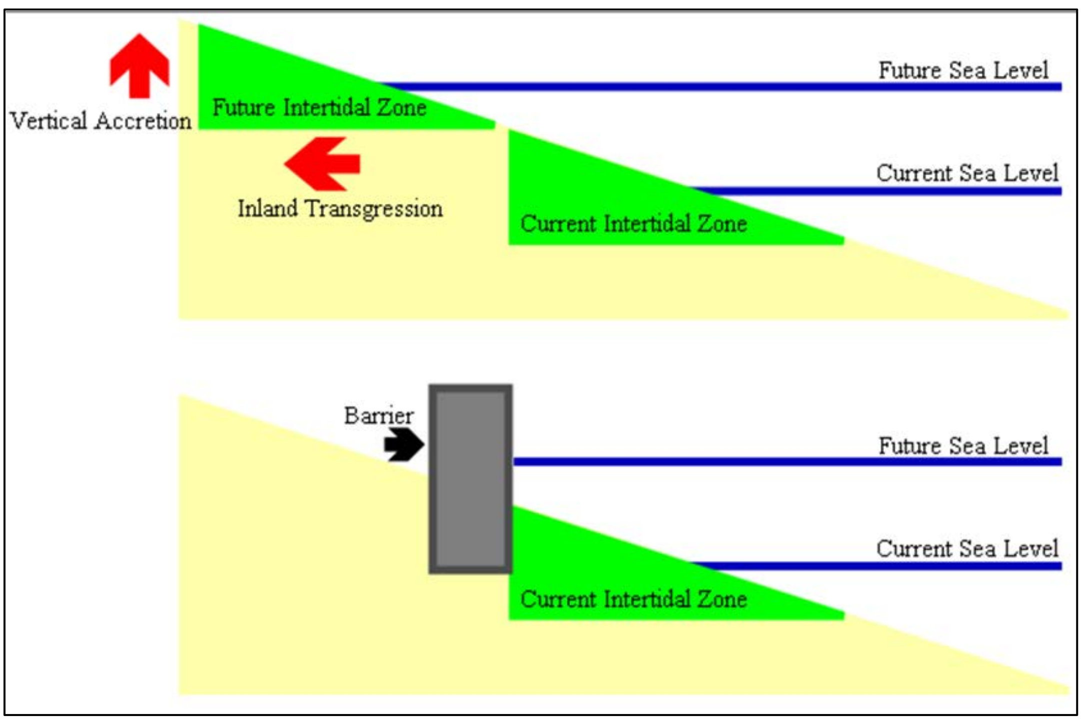


Figure 2.23 “Coastal Squeeze” (Chmura, 2009).

3. Conserving and restoring Puget Sound intertidal wetlands

This study has demonstrated that soil organic carbon density is positively correlated with elevation, and negatively correlated with increasing soil texture size. Soil carbon density at the Duckabush River Delta is slightly higher than the mean carbon density of sites in the Northeastern Pacific, and slightly lower than the global mean carbon density of salt marshes. More studies in the region would help shed light on whether or not the carbon content and density estimated for this site is consistent with that of intertidal wetlands in the Puget Sound. The next step with this information is to apply it to a broader range of environmental issues, and further our understanding the role of intertidal wetlands in the global carbon reservoir and greenhouse gas cycle.

3.1. *Why it matters*

The global extent of intertidal wetlands and coastal ecosystems is diminishing, and is substantially reduced in Puget Sound (Hopkinson et al., 2010, Collins & Sheikh, 2005). As this study has discussed, conserving and restoring intertidal wetlands may have benefits regarding carbon storage and their ability to act as greenhouse gas sinks. In addition to greenhouse gas benefits, intertidal wetlands offer many other important ecosystem services. In the Puget Sound, intertidal wetlands provide complex habitat and abundant food sources, which in turn supports a complex and diverse array of terrestrial and marine organisms (Dethier, 1990). Insects emerging from intertidal wetlands are consumed by fish at high tides, and tidal detritus is consumed and filtered by benthic invertebrates, including mollusks, crustaceans, and annelids. These fish and invertebrates are subsequently consumed by larger fish and invertebrates, mammals, waterfowl, shorebirds, and a huge array of other birds (Dethier, 1990).

In the Puget Sound, the plight of the salmon receives more attention than the problems faced by any other fauna in the region. All salmon species spend at least part of their lives in intertidal wetlands. Intertidal wetlands are particularly important in the life history of *Oncorhynchus tshawytscha* and *Oncorhynchus keta* (Chinook and Chum salmon). Many genetically-distinct runs of Puget Sound Chinook and Chum salmon are endangered, and the disappearance of intertidal habitat, critical for juveniles of these species, has played a role in the shrinking salmon populations.

Conserving and restoring intertidal wetlands is critical for so many environmental reasons, and they are all intertwined. Even the least-disturbed intertidal wetlands in Puget Sound, such as the Duckabush River Delta, have been undergone anthropogenic alteration, presenting opportunities for restoration and conservation.

3.2. ***Restoring the Duckabush River Delta***

Ginna Correa's 2003 report, which assessed habitat for salmon and steelhead in the Dosewallips-Skokomish Watershed, made several important restoration recommendations. The recommendations in the report are chiefly aimed at salmon and steelhead habitat improvement, but the benefits of restoring salmon and steelhead habitat also benefit the rest of the intertidal wetland community, as well as the biogeochemical processes that take place at the delta, including the storage of carbon and other greenhouse gasses. At the Duckabush River Delta, approximately 9% (conservatively) of the shoreline is armored (Correa, 2003). Most of the armoring at the delta is along SR

101, as well as a WDFW parking lot that was formerly part of the intertidal zone. Not only does shoreline armoring prohibit natural processes and diminish the extent critical fish and wildlife habitat, but it provides a location for invasive species to take hold and spread. Armoring at the Duckabush River Delta is dominated by the highly-invasive *Cytisus scoparius* (Scotch Broom). Correa recommends removing armoring and associated invasive species wherever possible, and replanting the area with native species.

The Puget Sound Nearshore Ecosystem Restoration Project (PNSERP) has designed a restoration plan at the Duckabush River Delta. The key elements of the plan include the removal of 640 m of the present SR 101 and its associated armoring. SR 101 would be reconstructed further upstream from where it currently exists, and would include a 335 m elevated roadway, which would allow for restored tidal connection and restoration of backshore sediment recruitment (PNSERP, 2013). Another 21 m section of armored roadway in the northwest part of the delta would be replaced with a bridge, restoring the intertidal zone to its historic extent. The PNSERP estimates that this design would add 15 ha of restored intertidal wetland.

This would have obvious benefits for fish and wildlife, including juvenile Chinook and Chum Salmon (both of which are endangered in the Duckabush River), but would have the potential to sequester additional carbon as well. If the project proposed by the PSNERP restores 15 ha of intertidal wetland, a “back-of-the-envelope” calculation based on the carbon stocks of zones 2 and 3 (which would correspond to the area to be restored) could account for an addition of approximately 1,485 Mg of carbon stored at the Duckabush River Delta. Again, this is an informal estimate, and many other factors may alter the rate of carbon sequestration and subsequently carbon stocks in the restored area (MacClellan, 2011; Santín et al., 2007).

3.3. *Restoring the areal extent of Puget Sound intertidal wetlands*

Only 17-19% of the historical extent of Puget Sound intertidal wetlands still exist (Collins & Sheikh, 2005). Intertidal wetlands at the deltas of Olympic Peninsula rivers, including the Duckabush, historically accounted for only 1% of Puget Sound Intertidal wetlands, but now account for 5% due to drastic losses of intertidal wetlands in other, more developed areas of Puget Sound (Collins & Sheikh, 2005). This historic extent of some of the largest intertidal wetland complexes on the Skagit, Stillaguamish, and Samish rivers has been greatly diminished, and intertidal wetlands have been almost totally eliminated in some rivers including the Green, Lummi, Puyallup, and Duwamish (Collins & Sheikh, 2005).

It is unlikely that the historic extent of Puget Sound intertidal wetlands will ever be restored, mostly due to the massive infrastructure, development and human population in many parts of the Puget Sound. Therefore, conserving and protecting the remaining intertidal wetlands of the Puget Sound is very important. The challenge in restoring many intertidal wetlands is simply that they no longer exist, having been diked, armored, and developed to a point that restoration is not an option. Some of the best opportunities to restore the historic extent of Puget Sound intertidal wetlands lie in degraded, but still existent, intertidal wetlands (PSNERP, 2013).

The PSNERP has outlined a set of general restoration objectives for Puget Sound nearshore ecosystems. The objectives are: (1) restore the size and quality of large river deltas and the nearshore processes the deltas support; (2) restore the number and quality of coastal embayments; (3) restore the size and quality of beaches and bluffs; (4) increase understanding of natural process restoration in order to improve effectiveness of restoration actions.

Achieving the last objective will be the most important in guiding effective restoration for intertidal wetlands and other nearshore ecosystems of the Puget Sound. Communicating new and better understandings of ecosystem services and functions to the public should be an additional objective in restoring Puget Sound intertidal wetlands and nearshore ecosystems. Broader public understanding will strengthen public support, funding, and restoration and conservation efforts. For a task as great as restoring the intertidal wetlands of the Puget Sound, it is absolutely necessary to work with the public as much as possible, because ultimately, the people of the Puget Sound region will enjoy the benefits of restoration first-hand.

Conclusion

As the results of this study demonstrated, carbon content and density are positively correlated with elevation within the intertidal wetland, driven largely by variations in soil texture composition and soil bulk density. Further research in the Puget Sound Region should consider accretion (or erosion) rates against rates of sea level rise to estimate local sequestration rates. Future research should compare the Duckabush River Delta to other intertidal systems in the Puget Sound to examine variation between systems locally. Other studies have investigated differences between carbon stored in natural intertidal wetlands, anthropogenically altered and degraded intertidal wetlands, and restored intertidal wetlands. The Puget Sound is home to intertidal wetlands that run the gamut of anthropogenic alteration, and comparisons between these different wetlands would be useful in understanding the potential to store carbon through restoration, and the amount of carbon that has been lost due to land-use change.

Much of the world's coastal systems, and certainly much of the Puget Sound's intertidal areas have already been lost, and much of what remains is trapped between development and rising sea levels. As sea levels rise, higher elevation areas will be more frequently inundated by tides, diminishing the ability to store carbon. As the this study shows, combined with previous studies' estimates of sequestration rates, intertidal wetlands, including the Duckabush River Delta, store huge amounts of carbon (relative to their size) and thus offset anthropogenic CO_2. Therefore, in the face of climate change, it is important to protect these systems where they exist, and restore them wherever it is feasible.

References

Andrews, J.E., G. Samways, and G.B. Shimmield, "Historical storage budgets of organic carbon, nutrient and contaminant elements in saltmarsh sediments: Biogeochemical context for managed realignment, Humber Estuary, UK," *Science of the Total Environment*. 405 (2008) 1-13.

Armamento, T.V., "Drainage of Organic Soils as a Factor in the World Carbon Cycle," *BioScience*. 30 (1980): 825-830.

Bartlett, K.B., D.S. Bartlett, R.C. Harris, and D.I. Sebacher, "Methane Emissions along a Salt Marsh Salinity Gradient," *Biogeochemistry*. 4 (1987): 183-202.

Bastviken, D., L. Persson, G. Odham, and L. Tranvik, "Degredation of Dissolved Organic Matter in Oxic and Anoxic Lake Water," *Limnology and Oceanography*. 49 (2004): 109-116.

Blake, G.R. and K.H. Hartge, "Bulk Density," in *Methods of Soil Analysis: Part 1-Physical and Minerological Properties.* Arnold Klute, ed. American Society of Agronomy, Inc., Soil Science Society of America, Inc., Madison, WI, (1986): 363-375.

Bouchard, V. and J.C. Lefeuvre, "Primary production and macro-detritus dynamics in a European salt marsh: carbon and nitrogen budgets," *Aquatic Botany*. 67 (2000): 23-42.

Chmura, G.L., "Tidal Salt Marshes," in: D. Laffoley and G. Grimsditch, Eds. *The Management of Natural Coastal Carbon Sinks*. IUCN. Gland, Switzerland. (2009): 5-11.

Chmura, G.L., S.C. Anisfeld, D.R. Cahoon, and J.C. Lynch, "Global carbon sequestration in tidal, saline wetland soils," *Global Biogeochemical Cycles*. 17 (2003): 1111-1133.

Collins, B.D, and A.J. Sheikh, *Historical reconstruction, classification and change analysis of Puget Sound tidal marshes*. University of Washington. Washington Department of Natural Resources. (2005): 120 pp.

Cresser, M.S., R.L. Gonzalez, A. Leon, “Evaluation of the use of soil depth and parent material data when predicting soil organic carbon concentration from LOI values,” *Geoderma*. 140 (2007) 132-139.

Crooks, S., M. Hatziolos, and D. Herr, *Capturing and conserving natural coastal carbon: Building mitigation, advancing adaptation*. The World Bank, IUCN, ESA PWA. (2010): 8 pp.

DellaSala, D., G. Orians, K. Kavanagh, and M. Sims, “Puget lowland forests,” *Ecoregions.* World Wildlife Fund. (2013): worldwildlife.org/ecoregions/na0524

Denman, K.L., G. Brasseur, A. Chidthaisong, P. Ciais, P.M. Cox, R.E. Dickinson, D. Hauglustaine, C. Heinze, E. Holland, D. Jacob, U. Lohmann, S Ramachandran, P.L. da Silva Dias, S.C. Wofsy and X. Zhang, “Couplings Between Changes in the Climate System and Biogeochemistry,” *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change.* Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M.Tignor and H.L. Miller (eds.). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA. (2007).

Dethier, M.N. *A Marine and Estuarine Habitat Classification System for Washington State*. Washington Natural Heritage Program. Washington Department of Natural Resources. (1990): 56 pp.

Duarte, C.M., J.J. Middleburg, & N. Caraco, “Major role of marine vegetation in the oceanic carbon cycle,” *Biogeosciences*. 2 (2005): 1-8.

Ellert, B.H., H.H. Janzen, A.J.VandenBygaart, and E. Bremer, “Measuring Change in Soil Organic Carbon Storage,” in *Soil Sampling and Methods of Analysis, Second Edition*. M.R. Carter and E.G. Gregorich, Eds. CRC Press, Boca Raton, FL, (2008): 25-38.

Ellis, S., and J.K. Atherton, “Properties and developments of soils on reclaimed alluvial sediments of the Humber Estuary, eastern England,” *Catena*. 52 (2003): 129 147.

Elmandorf, W.W., and A.L. Kroeber, *The Structure of Twana Culture, With Comparative Notes on the Structure of Yurok Culture*. Washington State University Press, Pullman, WA. (1992): 576 pp.

Essington, T., T. Klinger, T. Conway-Cranos, J. Buchanan, A. James, J. Kershner, I. Logan, and J. West, "Biophysical Condition of Puget Sound," *Puget Sound Science Update*. (2011): www.psp.wa.gov

Gee, G.W. and J.W. Bauder, "Particle-size Analysis," in *Methods of Soil Analysis: Part 1- Physical and Minerological Properties*. Arnold Klute, ed. American Society of Agronomy, Inc., Soil Science Society of America, Inc., Madison, WI, (1986): 383-411.

Gruneberg, E., I. Schoning, E.K.V. Kalko, and W.W. Weiser, "Regional organic carbon stock variability: A comparison between depth increments and soil horizons," *Geoderma*. 155 (2010): 426-433.

Hood Canal Coordinating Council. (2013): hccc.wa.gov

Hopkinson, C.S., W. Cai, and X. Hu, "Carbon sequestration in wetland dominated coastal systems – a global sink of rapidly diminishing magnitude," *Current Opinions in Environmental Sustainability*. 4 (2012): 186-194.

Hussein, A.H., M.C. Rabenhorst, and M.L. Turner, "Modeling of Carbon Sequestration in Coastal Marsh Soils," *Soil Science Society of America Journal*. 68 (2004): 1786-1795.

Kilbertus, G., "Microhabitats in soil aggregates. Their relationship with bacterial biomass and size of procaryotes present," *Revue d'Ecologie et de Biologie du Sol*. 17 (1980): 543-557.

Kroestch, D. and C. Wang, "Particle and Size Distribution," in *Soil Sampling and Methods of Analysis, Second Edition*. M.R. Carter and E.G. Gregorich, Eds. CRC Press, Boca Raton, FL, (2008): 713-725.

Krull, E., J. Baldock, and J. Skjemstad, "Soil Texture Effects on Decomposition and Soil Carbon Storage," in *Net Ecosystem Exchange: Workshop Proceedings*. M.U.F. Kirschbaum and R. Mueller, eds. Cooperative Research Centre for Greenhouse Accounting (2001): 103-110.

Ladd, J.N., M. Amato, and J.M. Oades, “Decomposition of plant materials in Australian soils. III Residual organic and microbial biomass C and N from isotope-labelled legume materials and soil organic matter decomposing under field conditions,” *Australian Journal of Soil Research*. 23 (1985): 603-611.

Laffoley, D., “Introduction,” in: D. Laffoley and G. Grimsditch (eds). The Management of Natural Coastal Carbon Sinks. IUCN. Gland, Switzerland. (2009): 53 pp.

Lal, R., “Soil carbon sequestration to mitigate climate change,” *Geoderma.* 123 (2004): 1-22.

Lal, R., “Forests and soil carbon sequestration,” *Forest Ecology and Management*. 220 (2005): 242-258.

Li, Y., L. Wang, W. Zhang, S. Zhang, H. Wang, X. Fu, and Y. Le, “Variability of soil carbon sequestration capability and microbial activity of different types of salt marsh soils at Chongming Dongtan,” *Ecological Engineering*. 36 (2010): 1754 1760.

MacClellen, M.A. *Carbon content in Oregon tidal wetland soils*. Marine Resource Management Program, College of Oceanic and Atmoshpheric Sciences, Oregon State, Corvallis, OR. (2011): 14 pp.

Mather, C.A., J. Chambers, J. Schumacher, and G.D. Hartmann, *Cultural resource Assessment for the Proposed Pleasant Harbor Marina and Golf Resort, Jefferson County, Washington*. Western Shore Heritage Services, Inc. (2006): 39 pp.

Mote, P., A. Peterson, S. Reeder, H. Shipman, & L.W. Binder, *Sea Level Rise in the Coastal Waters of Washington State*. University of Washington Climate Impacts Group. Washington Department of Ecology. (2008): 11 pp.

Mudd, S.M., S.M. Howell, and J.T. Morris, “Impact of dynamic feedbacks between sedimentation, sea-level rise, and biomass production on near-surface marsh stratigraphy and carbon accumulation,” *Estuarine, Coastal and Shelf Science*. 82 (2009): 377-389.

National Geodetic Survey, "Brinnon (1985). PID: SY1194." NGS DATASHEET PIDS FORM. (2013): www.ngs.gov

National Oceanic and Atmospheric Administration, United States Department of Commerce. (2013): www.noaa.gov.

Nicholls, R.J., P.P. Wong, V.R. Burkett, J.O. Codignotto, J.E. Hay, R.F. McLean, S. Ragoonaden and C.D. Woodroffe, “Coastal systems and low-lying areas,” *Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. M.L. Parry, O.F. Canziani, J.P. Palutikof, P.J. van der Linden and C.E. Hanson, Eds., Cambridge University Press, Cambridge, UK, (2007): 315-356.

Pennock, D., T. Yates, and J. Braidek, “Soil Sampling Designs,” in *Soil Sampling and Methods of Analysis, Second Edition*. M.R. Carter and E.G. Gregorich, Eds. CRC Press, Boca Raton, FL, (2008): 1-14.

Pidgeon, E., “Carbon Sequestration by Coastal Marine Habitats: Important Missing Sinks,” in: D. Laffoley and G. Grimsditch, Eds. *The Management of Natural Coastal Carbon Sinks*. IUCN. Gland, Switzerland. (2009): 47-51.

Phachomphon, K., P. Dlamini, and V. Chaplot, “Estimating carbon stocks at a regional level using soil information and easiliy accessible auxiliary variables,” *Geoderma*. 155 (2010): 372-380.

Puget Sound Nearshore Ecosystem Restoration Project. (2013): www.pugetsoundnearshore.org/technical_papers/historical_shoreline_dnr.pdf

Ransom, B., D. Kim, M. Kastner, and S. Wainwright, “Organic matter preservation on continental slopes: Importance of mineralogy and surface area,” *Geochimca et Cosmochimica*. (1998): 1329-1345.

Saarnio, S., W. Winiwarter, and J. Leitão, “Methane release from wetlands and watercourses in Europe,” *Atmospheric Environment*. 43 (2009): 1421-1429.

Santín, C., X.L. Otero, S. Fernandez, M. Gonzalez-Perez, and M. A. Alvarez, “Variations of organic carbon stock in reclaimed estuarine soils (Villaviciosa estuary, NW Spain),” *Science of the Total Environment*. 378 (2007): 138-142.

Schink, B., “Principles and limits of anaerobic degradation: Environmental and technological aspects,” in *Biology of Anaerobic Microorganisms*, B. Zehnder, ed. (1988): 771-846.

Schumacher, B.A., Methods for the Determination of Total Organic Carbon (TOC) in *Soils and Sediments*. United States Environmental Protection Agency, Environmental Sciences Division, National Exposure Research Laboratory. Las Vegas, NV. (2002): 23pp.

Sha, C., W.J. Mitsch, Ü. Mander, J. Lu, J. Batson, L. Zhang, and W. He, "Methane emmisions from freshwater riverine wetlands," *Ecological Engineering*. 37 (2011): 16-24.

Setia, R., P. Gottschalk, P. Smith, P. Marschner, J. Baldock, D. Setia, and J. Smith, "Soil salinity decreases global organic carbon stocks," *Science of the Total Environment*. Article in press. (2012).

Skjemstad, J.O. and J.A. Baldock, "Total and Organic Carbon," in *Soil Sampling and Methods of Analysis, Second Edition*. M.R. Carter and E.G. Gregorich, Eds. CRC Press, Boca Raton, FL, (2008): 225-237.

Spohn, M. and L. Giani, "Carbohydrates, carbon and nitrogen in soils of a marine and a brackish marsh as influenced by inundation frequency," *Estuarine, Coastal and Shelf Science*. 107 (2012): 89-96.

Stralberg, D., M. Brennan, J.C. Callaway, J.K. Wood, L.M. Schile, D. Jongsomjit, M. Kelly, V.T. Parker, and S. Crooks., "Evaluating Tidal Marsh Sustainability in the Face of Sea-Level Rise: A Hybrid Modeling Approach Applied to San Francisco Bay," *PLoS ONE.* 6 (2011): www.plosone.org/article/info%3Adoi%2F10.1371%2Fjournal.pone.0027388

Tuxen, K., L. Schile, D. Stralberg, S. Siegel, T. Parker, M. Vasey, J. Callaway, and M. Kelly, "Mapping changes in tidal wetland vegetation composition and pattern across a salinity gradient using high spatial resolution imagery," *Wetlands Ecology and Management*. 19 (2011): 141-157.

United States Environmental Protection Agency, *Methane and Nitrous Oxide Emmisions from Natural Sources* (2013): www.epa.gov/outreach/pdfs/Methane-and Nitrous-Oxide-Emissions-From-Natural-Sources.pdf

United States Geological Survey, *Wentworth Grain Size Chart.* (2013): www.usgs.gov van der Linden, A.M.A., L.J.J. Jeurisson, J.A. Van Veen, and G. Schippers, "Turnover of soil microbial biomass as influenced by soil compaction," in *Nitrogen in Organic Wastes Applied to Soil*, J. Attansen and K. Henriksen, eds. Academic Press, London, UK. (1989): 25-36.

Washington Department of Fish and Wildlife. (2013): wdfw.wa.gov/fishing/shellfish/beaches/270286/

Zedler, J.B., and J.C. Callaway, "Tidal Wetland Functioning," *Journal of Coastal Research*. Special Issue No. 27. *Tidal Wetland Restoration: Physical and Ecological Processes.* (2001): 38-64.

Zhang, J., Z. Li, T. Ning, and S. Gu, "Methane uptake in salt-affected soils shows low sensitivity to salt addition," *Soil Biology & Biochemistry*. 43 (2011): 1434 1439.

Zhou, J., Y. Wu, Q. Kang, and J. Zhang, "Spatial variations of carbon, nitrogen, phosphorus, and sulphur in the salt marsh sediments of the Yangtze Estuary in China," *Estuarine, Coastal and Shelf Science*. 71 (2007): 47-59.