An Ecogeomorphic Model to Assess the Response of Padilla Bay's Eelgrass Habitat to Sea Level Rise

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AN ECOGEOMORPHIC MODEL TO ASSESS THE RESPONSE OF
PADILLA BAY’S EELGRASS HABITAT TO SEA LEVEL RISE

By

Katrina L. Poppe

Accepted in Partial Completion
of the Requirements for the Degree
Master of Science

Kathleen L. Kitto, Dean of the Graduate School

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MASTER’S THESIS

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Katrina Poppe

December 14, 2015
AN ECOGEO MORPHIC MODEL TO ASSESS THE RESPONSE OF
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A Thesis
Presented to
The Faculty of
Western Washington University

In Partial Fulfillment
of the Requirements for the Degree
Master of Science

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December 2015
ABSTRACT

Estuaries worldwide are facing the possibility of conversion to open water if accretion cannot keep pace with increasing rates of sea level rise. Recent research into sediment elevation dynamics in Padilla Bay, a National Estuarine Research Reserve in Puget Sound, has revealed a mean bay-wide elevation deficit of -0.37 cm yr\(^{-1}\) since 2002. However, a more mechanistic prediction of the estuary’s response to future sea level rise should also incorporate non-linear feedback mechanisms between water depth, plant growth, and sediment deposition. Therefore, I used measurements of sediment accretion rates, suspended sediment concentrations, eelgrass stem density, and above- and belowground eelgrass biomass to build and calibrate a marsh equilibrium model (MEM), developed elsewhere but applied here for the first time to this eelgrass-dominated intertidal habitat. I then coupled the MEM with a relative elevation model (REM), which has previously been applied here, to create a hybrid that combines each model’s strengths in mechanistically simulating above-and belowground processes, respectively. The model predicts elevation change under various scenarios of sea level rise and suspended sediment concentrations. I used a 12-year elevation change dataset obtained from an extensive surface elevation table (SET) network in Padilla Bay for model validation.

Field measurements indicated sediment accretion rates to be primarily determined by eelgrass stem density instead of biomass or relative elevation. I modified the hybrid model to reflect this relationship, which differentiates it from its predecessors. The model validation exercise revealed the need for an erosion parameter, without which projected relative elevation gain was substantially overestimated. Model projections without erosion showed an increase in relative elevation over much of the bay’s elevation gradient over a 100-year
timeframe, reaching an equilibrium at an elevation where *Zostera japonica* stem density is maximized. These scenarios would involve an increase in *Z. japonica* cover in Padilla Bay, and a decrease in *Z. marina* cover. In contrast, model projections with erosion revealed a loss in relative elevation along the entire elevation gradient for all but the most conservative sea level rise scenario. The magnitude of loss was predicted to be greater at higher elevations. The suspended sediment concentrations required for the bay to maintain a stable relative elevation were higher than the current concentration of 3.93 mg L$^{-1}$ for all sea level rise scenarios, with up to 15 mg L$^{-1}$ being required for the most extreme scenario.
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1 INTRODUCTION

1.1 Estuarine habitat sustainability

1.1.1 Eustatic sea level rise

It is becoming increasingly apparent that climate change is causing sea level to rise at a faster rate than it has historically, and concern is mounting over the sustainability of coastal wetlands (Craft et al. 2009, Reed 1995). Climate change is causing increased eustatic sea level rise (ESLR), which is a change in global ocean volume due to melting land ice and thermal expansion of the ocean (NRC 2012). The current global average rate of ESLR is estimated at 3.2 mm yr\(^{-1}\), nearly twice the historical rate of 1.7 mm yr\(^{-1}\) from 1900-2010 (Church et al. 2013). The most recent Intergovernmental Panel on Climate Change (IPCC) report predicts a sea level rise of 0.40 to 0.78 m above 1986-2005 levels by the end of the 21\(^{st}\) century (Church et al. 2013), and although daunting, this is still considered by many to be conservative. Rahmstorf (2007), for instance, reports a less conservative ESLR prediction of 0.5 to 1.4 m above the 1990 level by 2100. The National Research Council’s Committee on Sea Level Rise in California, Oregon, and Washington is also planning for this less conservative projection (NRC 2012).

Thermal expansion of the ocean was thought to be the greatest contributor to ESLR until an ocean temperature bias discovered in the IPCC’s Fourth Assessment Report (Solomon et al. 2007) was corrected (NRC 2012). Melting of land ice became the largest component of global SLR in a more recent estimate, contributing 68% of ESLR for the 1993-2008 period (Church et al. 2011). Complete melting of the entire Greenland and Antarctic ice sheets could add at least 65 m to global sea level (NRC 2012).
Although sea level rise has become a household term due to more than a decade of media coverage related to climate change, it is not a recent phenomenon; global sea levels have been rising since the close of the Pleistocene Epoch over 10,000 years ago (Day et al. 1989). What has recently become apparent, however, is an unprecedented accelerating rate of ESLR. Estuaries have historically been able to keep pace with sea level rise through system feedbacks involving plant productivity, water depth, and sedimentation rates (Morris et al. 2002). Many coastal ecosystems have shown an ability to accumulate sediment and keep pace with historic or current sea level changes (Redfield 1965, 1972; Friedrichs and Perry 2001; Rybczyk and Cahoon 2002). However, their ability to keep pace with accelerated future rates of ESLR is questionable.

1.1.2 \textit{Relative sea level rise}

Relative sea level rise (RSLR) is the effective rate of sea level rise characteristic of a particular location, determined through a combination of ESLR and vertical land movement. Vertical land movement can be attributed to plate tectonics, isostatic rebound from glacial retreat, or deep subsidence due to sediment compaction or fluid withdrawal. These values can vary significantly on a fairly small spatial scale. For example, at Anacortes in Puget Sound, WA, the current rate of vertical land movement is reported as 0.10 ± 0.15 cm yr$^{-1}$ (NRC 2012), reducing the local rate of RSLR. The uplift here is primarily caused by plate tectonics associated with the Cascadia Subduction Zone (NRC 2012). On the outer coast at Neah Bay, WA, plate movement is causing the land to rise at a rate exceeding ESLR, resulting in a negative rate of RSLR (NRC 2012).

In addition to ESLR and vertical land movement, a few other factors are involved in determining estuarine surface elevation relative to sea level on a local scale. Relative
elevation is increased by accretion, which is the accumulation of sediment and organic material minus erosion. Relative elevation is decreased by shallow subsidence, which includes both decomposition and compaction. In total, the processes influencing relative elevation at a particular site can be described as:

Relative Surface Elevation = Accretion – ESLR – Shallow Subsidence
\[\pm \text{Vertical Land Movement}\]  

While the current rate of ESLR cannot be controlled on a short-term and local scale, we do have more immediate control over local sediment supply. According to D’Alpaos et al. (2011), the two dominant factors causing the drowning of intertidal habitats worldwide are rising sea levels and a paucity of available sediment. In fact, anthropogenic alteration of sediment inputs may be the greater threat to coastal wetland sustainability; there is no documented evidence of coastal wetland loss due to ESLR alone (Kirwan and Megonigal 2013, and references therein). For example, Kirwan and Megonigal (2013) state: “Regions of the world with rapid wetland conversion to open water (for example, the Gulf of Mexico, Venice Lagoon and along tributaries of the Chesapeake Bay) are all located in sediment-deficient areas.” The widespread reduction in estuarine sediment inputs is caused in many cases by streams and coastlines “hardened” by dams, dikes, and levees, which reduce natural downstream sediment transport and sediment deposition in estuaries.

In Padilla Bay, a 4,200 ha estuary in northern Puget Sound, a cutoff from its primary historical sediment supply via Skagit River distributaries has caused it to become a net erosional system as it loses more sediment than it gains every year (Kuhlman 2011). In addition to the bay’s historical sediment inputs via several Skagit River distributaries now filled and converted to farmland, the former network of wetlands across the Skagit floodplain
would have also delivered sediment in overland flow during river flood events (Collins 2000). Today, the major source of freshwater to Padilla Bay is the Swinomish Channel which is regularly dredged (Thom 1990), supplemented by other small freshwater inputs draining agricultural land including Joe Leary and Indian sloughs. This loss of sediment inputs to the Padilla Bay estuary is just one example of a worldwide trend threatening estuarine habitat sustainability.

While some coastal wetlands may be able to persist despite rising sea levels by migrating shoreward to follow their optimum depth, other wetlands in more developed areas may be completely inundated where shorelines have been modified such that upslope migration is prevented, a process termed “coastal squeeze” (Scavia et al. 2002, Orth et al. 2006). An estimated 27% of Puget Sound’s 2,500 miles of shoreline are armored, and although the Puget Sound Partnership now has a goal to reduce the net amount of armored shoreline in Puget Sound, it is unclear whether this goal will be realized over time (Puget Sound Partnership 2012). Much of the Padilla Bay shoreline is modified with dikes that prevent tidal action from reaching upland farms. As ESLR is predicted to increase, putting the Padilla Bay eelgrass meadows at risk of inundation and disappearance, mitigation options include increasing sediment supply by importing and dumping sediment or reconnecting the bay to its historical Skagit River sediment source, and removing dikes to allow the eelgrass meadows to migrate inland.

1.1.3 Plant-sediment dynamics

Changes in water depth affect intertidal plant characteristics, which in turn influence sediment dynamics that feedback to alter water depth. When ESLR causes water depth to increase, sediment accumulation tends to increase in part due to the increased duration of
tidal inundation, providing more suspended sediment that could potentially settle. Greater depths also tend to provide calmer conditions, allowing for more sediment deposition and less erosion. Plant density also shows an increasing trend as water depth increases from the plant’s upper limit to the species-specific optimal depth, then decreasing to the plant’s lower limit. The upper limit is dictated by tolerance to desiccation (Boese et al. 2005) or wave energy (Koch 2001), while the lower limit is determined by light attenuation (Duarte 1991). In addition to the influence of increasing water depth on sediment accumulation, an increase in plant density also increases sedimentation through the reduction of water velocity or the trapping of sediment on leaves (Fonseca and Kenworthy 1987, Mudd et al. 2010, Temmerman et al. 2012). Other studies have emphasized the ability of seagrass meadows to reduce sediment resuspension (Gacia and Duarte 2001) which is significant considering an estimated 70-95% of deposited sediment is resuspended (Dauby et al. 1995, Gacia and Duarte 2001). Thus, sedimentation can be influenced by water depth both directly and indirectly through plant density. Finally, an increase in plant density also increases surface elevation via organic matter deposition. It is these ecogeomorphic feedbacks that have allowed intertidal wetlands to persist in their current location despite sea level changes for thousands of years (Kirwan et al. 2008). However, there is likely a threshold rate of RSLR at which the estuary can no longer accumulate sediment fast enough to maintain equilibrium, the plants can no longer thrive, and the habitat is converted to open ocean (Morris et al. 2002).

The goal of my research was to use a combined field and modeling approach that addresses the feedback mechanisms between water depth, plant growth, and sedimentation, using Padilla Bay, WA as my study area. The model is used to predict long-term relative
elevation change given current and predicted conditions, and also to determine the suspended sediment concentrations necessary to maintain equilibrium. Collecting site-specific field data for selected parameters not only provides some real-time indication of coastal elevation dynamics, but also provides data for model initialization, calibration, and validation. Modeling long-term relative elevation change can improve our understanding of future ecosystem sustainability, since current rates and relationships will not necessarily be maintained as constants over time due to non-linear feedbacks within the system. Model projections can then be used to guide restoration efforts by indicating what the system would require to be sustained.

1.1.4 Predictive models

Several different predictive surface elevation models have been used in recent years to estimate future ESLR impacts on coastal wetlands. They vary in terms of the factors included, which is determined in part by their spatial scale. Model spatial scales range from zero-dimensional models, which represent processes at a single point on the landscape, to two-dimensional models that are typically less mechanistic but more spatially-explicit with processes that vary across the landscape. A model is considered to be “ecogeomorphic” when it includes feedbacks between vegetation and physical processes (Rybczyk and Callaway 2009).

Zero-dimensional surface elevation models have typically fallen into one of two categories, focusing on either mineral sedimentation or organic sediment processes (Rybczyk and Callaway 2009). Mineral sedimentation models emphasize the role of sedimentation in changing elevation, and although their relative simplicity can make them more approachable, they have an inherent tendency to overestimate elevation increases because they do not

Models focusing on organic sediment processes incorporate the effect of shallow subsidence into predicted elevation change, typically using a sediment cohort approach that tracks sediment packets from the surface down through the soil column. These provide an additional option for calibration or validation, where simulated profiles of soil characteristics are compared to actual soil cores. These include models by Morris and Bowden (1986), Chmura et al. (1992), Callaway et al. (1996), and Rybczyk et al. (1998).

These models have typically included sediment deposition as a linear function of depth because sediment availability increases with the duration of tidal inundation. More recently, some models have also included the effect of vegetation on accretion rates (Morris et al. 2002, Swanson et al. 2013), including an accretion component modeled as a function of plant biomass or productivity. Several studies have examined this plant-accretion relationship and the mechanisms behind it, finding that vegetation plays a very important role in facilitating sediment accretion (Fonseca et al. 1982, Gacia et al. 1999, Gacia and Duarte 2001, Baustian et al. 2012). The plant-accretion relationship may thus be a valuable addition to elevation change models.

1.1.5 Study area

Padilla Bay (48° 30' N; 122° 30' W) was designated a National Estuarine Research Reserve in 1980, to protect its relatively large eelgrass meadows and to provide opportunities for scientific research and public education. Eelgrasses (Zostera spp.) are intertidal flowering plants that provide habitat for many ecologically and commercially valuable species.
including macroalgae, oysters, clams, crabs, salmon, herring, migratory waterfowl, and shorebirds (Bulthuis 1995). An ecosystem services assessment in Puget Sound estimated the value of eelgrass beds at up to $15,000 acre\(^{-1}\) yr\(^{-1}\), ranking among the top 25% of Puget Sound Basin ecosystems evaluated (Batker et al. 2008). These services include biodiversity, food provisioning, shoreline stabilization, storm protection, human disease control, waste processing, and carbon sequestration. Unfortunately, anthropogenic alteration of the shoreline throughout the plant’s range has confined total eelgrass habitat to a small fraction of its historical area. However, this is not just a regional concern; on a global scale, an estimated 29% of total seagrass area has been lost (Waycott et al. 2009), due to both loss of habitat and declining water quality (Orth et al. 2006, Lotze et al. 2006). Padilla Bay currently contains over 3,800 ha of eelgrass (approximately 9,390 acres), making it the largest eelgrass meadow in the greater Puget Sound region (Berry et al. 2003) and one of the largest contiguous eelgrass stands along the North American Pacific coast (Bulthuis and Shull 2006). This fact makes the sustainability of Padilla Bay’s eelgrass habitat all the more important for the ecosystem services it provides and the species that depend on it.

There are two species of eelgrass in Padilla Bay, including one native species \((Zostera marina)\) and one non-native species \((Z. japonica)\). The native \(Z. marina\) occupies approximately 3,131 ha (82% of total eelgrass coverage in Padilla Bay), while \(Z. japonica\) occupies approximately 669 ha (18% of total) as of 2004 (Bulthuis and Shull 2006). Although there is some overlap between 0.3 and 0.6 m above MLLW (Thom 1990), the two species generally occupy different depth ranges, with \(Z. marina\) occurring lower in the intertidal zone than \(Z. japonica\). In Padilla Bay, \(Z. marina\) typically grows from 0.3 m above MLLW to subtidal depths (to \(~3.0\) m below MLLW), while \(Z. japonica\) ranges from 0.3 to
0.8 m above MLLW (Thom 1990). The non-native species was likely first introduced to neighboring Samish Bay in the early 1900s as packing material for Japanese oyster transportation to Washington for commercial production, and later spread to Padilla Bay when oysters were planted there in the 1930s (Dinnel 2000). Since then, the Z. japonica population has spread as far south as Humboldt, California, and north to Campbell River, British Columbia. The long-term ecological implications of this introduction are still not entirely understood (Mach et al. 2010).

1.2 Project history

This project was built on research performed by several previous graduate students in the research group led by Dr. John Rybczyk at WWU, focused on evaluating surface elevation dynamics in Padilla Bay’s eelgrass meadows. Surface elevation tables (SETs) were installed to monitor elevation change beginning in 2002 by Maxwell (2004). Maxwell measured sediment accumulation at the SET sites, concluding that accretion and subsidence rates were roughly equivalent, resulting in no net elevation gain. Ball (2004) used these SETs to assess the impact of Spartina eradication on elevation change rates. Gwozdz (2006) looked at sediment deposition rates and eelgrass biomass and density to develop a Z. marina production model for Padilla Bay. Kairis (2008) installed additional SETs, and developed a spatially-explicit version of the Relative Elevation Model from Rybczyk et al. (1998) to predict long-term relative elevation change in Padilla Bay’s Z. marina meadows under various ESLR scenarios. Most recently, Kuhlman (2011) analyzed relationships between eelgrass biomass, relative elevation, and surface elevation change rates, using an 8-year SET dataset.
The SET elevation change dataset compiled by these researchers - now extended to a 12-year dataset - has been a crucial element of this study. The work of Kairis and Rybczyk (2010) to develop and apply an elevation change model to Padilla Bay has been particularly useful. Several components of their model have been integrated into the hybrid model presented here.

1.3 Study objectives

The purpose of this study was to develop a hybrid model based on two existing models of relative elevation change, and apply the model to Padilla Bay’s eelgrass habitat to predict long-term rates of relative elevation change under various RSLR scenarios. I used several field measurements to allow model parameters to be as site-specific as possible. The hybrid model incorporates both above- and belowground processes that are important components of surface elevation dynamics.

To accomplish this goal, I addressed the following objectives, with each objective being the subject of chapters 2, 3, and 4, respectively:

1. Obtain sediment characteristics and quantify relationships among sediment accretion, plant characteristics, and relative elevation for model use.
   a. Determine sediment accretion rates along an elevation gradient with wire mesh screens, feldspar markers, and \(^{210}\text{Pb}\) analysis.
   b. Evaluate relationships between sediment accretion rates, relative elevation, and plant characteristics (biomass and stem density).
   c. Describe sediment characteristics (bulk density, organic content, pore space) from sediment cores for model initialization and calibration.
2. Analyze elevation change rates across Padilla Bay.
   a. Determine rates of elevation change at SET sites for model validation.
   b. Examine spatial and temporal trends in elevation change rates.
   c. Compare elevation change rates with sediment accretion rates.
3. Develop a hybrid model of relative elevation change and apply the model to Padilla Bay’s eelgrass habitat.
   a. Construct a hybrid model based on two existing models of relative elevation change, modified to apply to Padilla Bay’s eelgrass habitat based on site-specific field data.
   b. Predict long-term relative elevation change under a variety of RSLR scenarios.
   c. Determine the suspended sediment concentration necessary to maintain a relative elevation equilibrium, for each RSLR scenario.
2 SEDIMENT ACCRETION RATES AND PLANT AND SEDIMENT CHARACTERISTICS ALONG AN ELEVATION GRADIENT

2.1 Introduction

The hybrid surface elevation model includes a variety of parameters that are best estimated using site-specific field data. Decomposition and compaction rates for model initialization are based on sediment profiles of bulk density, organic content, and porosity. Accretion measurements as a function of both relative elevation and plant characteristics are used to estimate sediment settling and trapping coefficients for model calibration. Previous studies in Padilla Bay have reported accretion rates, elevation, and biomass from various sites throughout the bay. However it is best to maximize site-specificity when relating multiple parameters to each other. I chose to obtain my own accretion measurements and sediment profiles from the Padilla Bay National Estuarine Research Reserve (PBNERR) biomonitoring sites to correspond with the plant characteristics and elevation data collected from the same sites by PBNERR.

It is widely accepted that the surface elevation of a coastal wetland, with respect to sea level, has a substantial effect on sediment accretion (Friedrichs and Perry 2001, and references therein). As such, this relationship is typically incorporated into surface elevation models (Rybczyk and Callaway 2009). However, the additional effect of plant characteristics, such as biomass and stem density, are less often considered. Although largely qualitative, a few studies have shown that plants can have a significant impact on sediment dynamics, describing these plants as “ecosystem engineers” in the coastal wetland context (Jones et al. 1997, Bouma et al. 2005, Bos et al. 2007). Seagrass study results have shown a reduction in erosion within seagrass beds due to the plant canopy’s reduction of water velocity (Gambi et al. 1990, Bouma et al. 2005) and the prevention of resuspension by roots
and rhizomes (Orth 1977, Dauby et al. 1995). For example, Gacia and Duarte (2001) used direct measurements of primary and resuspended sediment flux and found the Mediterranean seagrass, *Posidonia oceanica*, to cause a three-fold reduction in sediment resuspension relative to unvegetated areas. Although they found vegetation to affect resuspension, they found little significant effect on primary deposition, contrary to popular opinion (e.g., Orth 1977, Walker et al. 1996). However, several others have actually documented an increase in sediment deposition within seagrass beds, for both subtidal perennial and intertidal annual populations (e.g., Fonseca and Bell 1998, Gacia et al. 1999, Koch 2001, Bos et al. 2007).

The objectives of this field study were to 1) obtain short- and long-term accretion rates along an elevation gradient, 2) quantify the effects of eelgrass stem density, aboveground biomass, and relative elevation on accretion rates, and 3) obtain sediment profiles of bulk density, organic content, and pore space for model initialization and calibration. Field data obtained by PBNERR are also reported here as they are integral to this analysis.

### 2.2 Methods

I established seven accretion monitoring sites in June 2013 at a sub-set of sites along a previously established permanent PBNERR biomonitering transect (Figure 1). The PBNERR transect runs approximately 4 km west from the shore, in the northern region of the bay. This transect is separated into six different zones, determined by the dominant vegetative cover. (1= Mudflat; 2 = *Zostera japonica*; 3 = Mixed *Z. japonica* and *Z. marina*, with *Z. japonica* dominant; 4 = Mixed *Z. japonica* and *Z. marina*, with *Z. marina* dominant; 5 = Intertidal *Z. marina*; and 6 = Subtidal *Z. marina*). These zones differ in length, with Zone 5 (Intertidal *Z. marina*) being the longest. I installed a total of seven accretion monitoring
plots along the transect, with one station in each of Zones 1-4 and three stations in Zone 5. I did not measure accretion in the subtidal zone. The farthest accretion monitoring site was 3.29 km from shore, with a total distance of 3.12 km between the first and last site.

![Figure 1](image-url)  
Figure 1. Location of accretion monitoring sites along the northernmost PBNERR biomonitoring transect (modified from image courtesy of D. Bulthuis). Inset shows transect location within Padilla Bay.

Each of my accretion monitoring sites was installed approximately 3 m west of a corresponding PBNERR eelgrass monitoring plot, in which above- and belowground eelgrass biomass and stem density data were collected monthly for one year by PBNERR staff from February 2011 to January 2012. Each of these plots continues to be monitored annually in June/July using nondestructive methods to obtain stem density and percent cover data. My
accretion monitoring sites were named for their corresponding PBNERR plot name, with a number corresponding to the transect zone (1-6) and a letter corresponding to the plot within that zone (A-K).

2.2.1 Transect elevation

North American Vertical Datum of 1988 (NAVD88) elevations along the permanent eelgrass monitoring transects were surveyed by PBNERR staff between July 12 and 14, 2011. They used a Trimble R8 GNSS Base Receiver and Rover to collect Real-Time Kinematic (RTK) measurements using GEOID03. ForeSight DXM software was used to download the data from the TDS field unit. The xyz field data were obtained from the center of each transect plot, using a 2-m fixed height pole with a bubble level. The latitudes, longitudes, and orthometric elevations were exported to a geodatabase and excel spreadsheet. Elevations were determined in meters relative to NAVD88 with an estimated accuracy of 1-5 cm. Extremely low HDOP and PDOP (dilution of precision) values at the time of acquisition indicate a relatively high precision.

I converted NAVD88 elevations to tidal elevations using a conversion factor provided by Suzanne Shull at PBNERR (Figure 2). The conversion from NAVD88 to Mean Lower Low Water (MLLW) using the NOAA VDATUM model, the most common conversion method used by others, was determined to be inaccurate for Padilla Bay because the VDATUM model is based on La Conner survey data with a very different tidal regime. Shull determined a preferred conversion for Padilla Bay based on correspondence with Robert Burrows and Steven Gill at NOAA CO-Ops, using historic survey data at the North End Swinomish Slough station (9448682), with an estimated accuracy of ± 0.08 m (95% CI). Tidal datums at this site were last computed and benchmark elevations were established in
the 1960’s. The datums were determined relative to the 1941-59 National Tidal Datum Epoch (NTDE). NOAA CO-OPS estimated updated tidal datums at this location using the change in relative mean sea level observed at Seattle between the 1941-59 NDTE and the current (1983-2001) NDTE. Mean sea level rose approximately 1.0 ft (0.31 m) between NDTE periods. Tidal range was assumed to remain the same. Based on this estimation, MLLW is 0.12 m below 0 m NAVD88 (Figure 2), whereas the conversion based on VDATUM indicates MLLW 0.461 m above NAVD88. I converted NAVD88 elevations along the transect to relative elevations (cm above MLLW) by adding 12 cm to the NAVD88 elevation.

Figure 2. Relationship between NAVD88 elevations and tidal elevations for Padilla Bay (S. Shull, pers. comm., 2013).
2.2.2  *Plant biomass and stem density*

PBNERR provided me with measurements of stem density and above- and belowground biomass for both species of eelgrass occurring in Padilla Bay. This section describes PBNERR sampling methods as well as my methods for analyzing these data. Three of the 11 permanent PBNERR plots within each of five zones along the northern transect (Transect 1) were surveyed monthly by PBNERR staff from February 2011 to January 2012. Three replicate 0.0625 m$^2$ samples were taken from each plot, with the plot average ± SE ($n = 3$) reported for each month. Three plots were also surveyed in a deeper sixth zone from December 2011 to January 2012, but these samples were not included in my analysis because they did not provide annual mean values of biomass and stem density.

Some mean biomass values reported by PBNERR were negative as a result of the dried weight of very small samples being less than the initial filter weight, perhaps caused by differences in moisture adsorption onto the filters. I adjusted any negative plot mean biomass values to 0.0001 g m$^{-2}$. It would be preferable to change all the negative measurements instead of merely the negative plot means, but I only had access to the plot means. However, this adjustment had minimal impact on the biomass means, raising the mean annual aboveground biomass value by only 0.001%.

PBNERR reported aboveground biomass separately for *Z. marina* and *Z. japonica*, after shoots and leaves were separated in their laboratory. A portion of leaves were not confidently identifiable to species, although they were most likely *Z. japonica* leaves. PBNERR reported two separate *Z. japonica* biomass values, one with only confidently identified leaves and the other with the addition of the remaining leaves. I elected to use the latter values in my analysis, so the sum of both species’ biomass values would produce the
actual total biomass value. Belowground biomass was reported with both species combined. I thus determined one root-to-shoot biomass ratio to apply to both species using the ratio of belowground to total aboveground biomass. Stem densities were surveyed for each species separately in each plot, allowing me to assess the relationship between aboveground biomass and stem density for each species.

I chose to use the annual mean biomass and stem density values for analysis instead of the annual peak, primarily because the two eelgrass species differ in the timing of their density peaks; whereas biomass for both species appears to peak in mid-summer in Padilla Bay, stem density peaks in summer for *Z. japonica* but mid-winter for *Z. marina* (Bulthuis et al. 2013). Incorporating winter measurements could be important due to the effect of vegetation on erosion during winter storms, which will affect the annual accretion rate. Furthermore, a previous study in Padilla Bay also used annual means (Kairis and Rybczyk 2010), and I chose to maintain this consistency with other studies.

2.2.3  *Sediment characteristics*

I collected one sediment core from each of the seven accretion monitoring sites along the PBNERR transect. The cores were taken approximately 1 m outside of the marked northwest corner of each accretion plot. The PVC corers had an internal diameter of 10.0 cm and they were driven into the sediment to a depth of 36 – 44 cm. Cores were kept upright during transport to the lab where they were frozen to aid removal of the sediment from the PVC. Once removed, each frozen core was sliced at 2 cm increments using a sawzall, then measured for wet volume, and oven-dried at 105 °C for at least 72 hours. The mass of each dried section was recorded and bulk density was calculated as the ratio of dry weight to wet
These sections were pulverized using a Thomas Wiley Mini-Mill with #40 screen (0.425 mm mesh), and stored in plastic bags.

The organic and mineral content of each section was determined through loss on ignition. A 15 g subsample of each section was re-dried for at least 5 hours prior to weighing for loss on ignition, to ensure that any moisture gained during storage was eliminated. These subsamples were then placed in a pre-weighed ceramic crucible and burned at 500 °C for 24 hours. Percent organic matter by weight was calculated as the ratio of the mass lost to initial mass. The percentage of mineral matter was calculated by subtracting organic content from 100. After calculating the volume of organic and mineral matter using the product of the percent by weight and a known organic or mineral particle density of 1.14 and 2.62 g cm⁻³, respectively (Callaway et al. 1996), percent pore space by volume was calculated by subtracting both organic and mineral volumes from 100.

I obtained mean profiles of bulk density, organic matter, and pore space by averaging each 2-cm section across all cores. These mean profiles were then used for model calibration. The two highest sites (1G and 2J) were excluded from the means due to the presence of atypical layers. Cores from all sites extended to at least 32 cm below the surface, so the mean profiles included each 2-cm section from 0-32 cm depth range. Some of the deepest core sections were also excluded from the means when these sections differed substantially from the sections above them based on a visual assessment of the profiles. These differences in the lowest sections may have been caused by surface organic material being pushed to the bottom when pushing the corer into the sediment.
2.2.4 Short-term accretion

I measured short-term accretion in two ways, employing both feldspar marker horizons and wire mesh screens at each accretion monitoring site. Feldspar marker horizons are commonly used to measure accretion on a seasonal or annual basis (Nolte et al. 2013). The wire mesh screens are a slightly modified version of the plastic grid method, developed particularly for use in unvegetated environments that experience more mixing and erosion than vegetated sites (Turner et al. 2012). The screens allow for repeated measurements with the same temporal resolution as the feldspar marker horizons. Both were installed in June of 2013.

2.2.4.1 Feldspar marker horizons

Each accretion monitoring site contained four 0.5 m by 0.5 m feldspar horizon markers, each approximately 2 m apart to form the corners of a square. The four corners of this 4 m² plot were marked with a 0.5 inch internal diameter PVC pipe, located 0.7 m from the outside corner of feldspar. The PVC stakes were placed at this distance from the feldspar to prevent any algae or eelgrass detritus that settled around the stakes from disturbing the feldspar marker. Once the PVC stakes were pounded into the sediment, they were cut such that no more than 30 cm emerged above the water level at installation, to reduce the buildup of detritus.

I took sediment cores from the feldspar markers to measure sediment accumulation in June 2014, one year after their installation. I used a clear plexiglass core with a 5 cm internal diameter and a sharp, beveled bottom edge. I pounded the core into the sediment at least 15 cm and capped it with a rubber stopper to remove the core. The clear plexiglass allowed any
feldspar in the sediment to be observed immediately after extraction and the distance from the feldspar to the sediment surface was be measured with a ruler.

2.2.4.2 Wire mesh screens

Each accretion monitoring site included two 0.25 m by 0.25 m screens placed approximately 1 – 2 m west of the feldspar marker horizons. These screens were cut from galvanized steel hardware cloth with a 0.5 inch mesh size. They were secured on the sediment surface with 1-ft wire stakes at the corners and the center. In order to allow complete contact with the sediment surface, I trimmed any plant leaves under the screens to approximately 1 cm above the sediment surface, with the assumption that this short-term reduction in aboveground biomass within a 0.0625 m² area would not significantly affect sediment dynamics. One advantage of using these screens over feldspar markers is that they allow for measurements of erosion. The screen is held by the stakes at the same vertical location regardless of whether sediment accumulates or erodes. Any eroding sites can thus be included quantitatively in the analysis, instead of being qualitatively included or even eliminated from any accretion analysis as has previously been the case.

I visited these sites to measure the height of sediment accumulation atop the screens in August and December of 2013, and April, June, July, and November of 2014. To measure sediment accretion on the screens, I pushed a ruler lightly through the sediment until it touched the screen, and recorded the depth of the screen below the surface. To obtain one accretion value for each site, I averaged the four measurements from each screen, and then averaged again the screens (n = 2) for each site.

Annual accretion rates were calculated using the 2014 measurements obtained approximately one year after installation. The June 2014 measurements were used for this
purpose at all sites except 1G for which July 2014 measurements were used. Because screens were installed on different days, these June-July 2014 accretion measurements were standardized to one year to compare rates among sites.

2.2.5 Long-term accretion

One core from each of the seven accretion monitoring sites was analyzed for long-term sediment accretion using the downcore distribution of excess $^{210}\text{Pb}$. I obtained these long-term accretion rates to compare with short-term rates, and to compare with long-term rates reported by other researchers for Padilla Bay. Assuming the depositional rate of excess $^{210}\text{Pb}$ does not change with time, and the profile is not mixed due to bioturbation, a profile of excess $^{210}\text{Pb}$ activity in the sediment column provides an indication of long-term sediment accretion rates. The sediment accretion rate for an unmixed sediment column is determined from the exponential decrease in activity with depth and the known decay rate of $^{210}\text{Pb}$.

Excess $^{210}\text{Pb}$ activity was analyzed using a Canberra Germanium Detector (model GL2820R), and gamma emissions at 46 keV and 351 keV were recorded by Genie 2000 software (Canberra 2002). I analyzed approximately 30 g of dried and ground sediment from sections at various depths throughout the sediment column. Each sample was analyzed for 48 to 72 hours, until the counting error rates for $^{210}\text{Pb}$ and $^{214}\text{Pb}$ dropped below approximately 10%. Because the gamma spectrometer measures total $^{210}\text{Pb}$ activity which includes both excess and supported $^{210}\text{Pb}$ at 46 keV, excess $^{210}\text{Pb}$ activity must be isolated to determine sedimentation rates. Supported $^{210}\text{Pb}$ was measured by detecting $^{214}\text{Pb}$ activity at 351 keV. The difference between the activity (disintegrations s$^{-1}$ or Bq) measured at 46 keV and 351 keV represents the excess $^{210}\text{Pb}$ activity in the sample. To account for different spectrometer counting efficiencies at different energy levels, a calibration standard was analyzed for each
core. The standard was created by adding approximately 0.75 g pitchblende silica-ore standard (CRM 103-A, New Brunswick Laboratory, USDOE) to a previously analyzed 30 g sample. A linear regression of the natural log of excess $^{210}\text{Pb}$ activity versus depth was used to determine the accretion rate. The accretion rate is equal to $-\lambda/s$, where $\lambda$ is the half-life of $^{210}\text{Pb}$ (22.2 yr$^{-1}$) and $s$ is the slope of this regression. Error was calculated for excess $^{210}\text{Pb}$ by propagating error from supported $^{210}\text{Pb}$ and $^{214}\text{Pb}$ counts quadratically, following methods described by Gwozdz (2006).

Downcore profiles of $^{210}\text{Pb}$ exhibiting a trend other than exponential decay were assumed to be mixed by polychaetes, shrimp, or other bioturbators, or influenced by sediment deposition events. However, it is important to note that an exponential curve does not indicate the absence of bioturbation. Bioturbation is likely to have occurred at all sites, and it is not necessarily apparent in the $^{210}\text{Pb}$ profile.

Previous studies in Padilla Bay have encountered mixed $^{210}\text{Pb}$ profiles more often than not. These layers can extend at least 25 cm in depth below the surface (Gwozdz 2006, Kairis 2008), which is generally deeper than the useful portion of the $^{210}\text{Pb}$ profile given the isotope’s decay rate. When accretion rates are low, excess $^{210}\text{Pb}$ values can decline to zero within the top 10 cm of the sediment surface. It is thus very difficult to obtain an acceptable accretion rate from sediment cores in Padilla Bay, due to the prevalence of mixing and the small portion of the core providing positive $^{210}\text{Pb}$ values. Because distinct mixed layers were difficult to reliably identify, I did not exclude any core sections from the regressions for this reason. I did, however, exclude sections with negative excess $^{210}\text{Pb}$ values from the regressions, because these values could not be log transformed.
Considering the likelihood of bioturbation at all core sites in Padilla Bay, the resulting long-term accretion rates should be considered maximum possible rates. To assess the relative effect of bioturbation, I calculated the excess $^{210}$Pb inventories for each core to obtain the $^{210}$Pb flux to the sediment in Padilla Bay, to then compare to the expected flux from the atmosphere for the region (Nevissi 1985). An observed flux lower than expected implies that the site is not depositional, and the apparent long-term accretion rates would be due to bioturbation instead of sediment deposition. An observed flux that is similar to the expected rate implies that the site is depositional, and the apparent long-term accretion rate is due to both accretion and bioturbation.

2.2.6 Relationships between accretion rates and site characteristics

Because eelgrass biomass and stem density were monitored at plot 1F and not at the adjacent plot 1G where my accretion monitoring plot was set up, I made the assumption that eelgrass biomass and stem density at 1G were identical to 1F for the purpose of examining relationships between plant characteristics and accretion rates.

I examined the relationship between accretion rates and site characteristics including annual mean eelgrass aboveground biomass, annual mean stem density, and relative elevation, using data from the accretion monitoring sites. I used accretion rates obtained from the wire mesh screens for this analysis. I used an ordinary least squares linear regression analysis to test the ability of each of these site characteristics to predict accretion rates. All four variables were tested for normality with the Shapiro-Wilk test, while auto-correlation was tested with the Durbin-Watson test. Variance heterogeneity was assessed qualitatively. All statistical tests were performed in the R programming environment using a critical value ($\alpha$) of 0.05 (R Core Team 2014).
2.3 Results

2.3.1 Transect elevation

The elevation of the entire PBNERR transect 1 relative to Mean Lower Low Water (MLLW) ranges from a shoreward high of 122.4 cm to a low of -262.2 cm at the last subtidal plot in zone 6. Elevation at my seven accretion monitoring sites ranges from 80.8 cm in zone 1 to -66.5 cm near the end of zone 5 (Table 1). The location of MLLW is at the transition between zones 4 and 5, coinciding with the lower limit of Z. japonica along this transect in Padilla Bay.

Table 1. Location and elevation of accretion monitoring sites. Site locations are shown in Figure 1.

<table>
<thead>
<tr>
<th>Plot</th>
<th>Latitude (N)</th>
<th>Longitude (W)</th>
<th>Distance from shore (m)</th>
<th>Elevation (cm NAVD88)</th>
<th>Elevation (cm above MLLW)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1G</td>
<td>48°32.910'</td>
<td>122°30.083'</td>
<td>169.2</td>
<td>68.8</td>
<td>80.8</td>
</tr>
<tr>
<td>2J</td>
<td>48°32.922'</td>
<td>122°30.525'</td>
<td>712.6</td>
<td>37.0</td>
<td>49.0</td>
</tr>
<tr>
<td>3F</td>
<td>48°32.932'</td>
<td>122°30.864'</td>
<td>1131.0</td>
<td>17.5</td>
<td>29.5</td>
</tr>
<tr>
<td>4F</td>
<td>48°32.947'</td>
<td>122°31.356'</td>
<td>1736.7</td>
<td>-2.9</td>
<td>9.1</td>
</tr>
<tr>
<td>5B</td>
<td>48°32.956'</td>
<td>122°31.719'</td>
<td>2183.9</td>
<td>-13.0</td>
<td>-1.0</td>
</tr>
<tr>
<td>5F</td>
<td>48°32.970'</td>
<td>122°32.167'</td>
<td>2736.4</td>
<td>-34.6</td>
<td>-22.6</td>
</tr>
<tr>
<td>5J</td>
<td>48°32.982'</td>
<td>122°32.616'</td>
<td>3288.3</td>
<td>-78.5</td>
<td>-66.5</td>
</tr>
</tbody>
</table>

2.3.2 Plant biomass and stem density

Annual mean aboveground biomass totals including both eelgrass species at the seven accretion monitoring sites ranged from 0.02 g m$^{-2}$ at site 1G to 46.38 g m$^{-2}$ at 5J (Table 2). Z. marina biomass was greatest at site 5J with 44.82 g m$^{-2}$, while Z. japonica biomass was greatest at site 3F with 33.52 g m$^{-2}$ (Figure 3). Annual mean stem density totals ranged from 7.8 shoots m$^{-2}$ at site 1G to 3519.0 shoots m$^{-2}$ at 2J (Table 2). Z. marina stem density was
greatest at site 4F with 671.8 shoots m\(^{-2}\) while \textit{Z. japonica} stem density was greatest at site 2J with 3515.2 shoots m\(^{-2}\) (Figure 4).

Table 2. Annual mean aboveground eelgrass biomass (± SE) and stem density (± SE) at seven accretion monitoring sites. ZM = \textit{Z. marina}; ZJ = \textit{Z. japonica}; Total = sum of both species.

<table>
<thead>
<tr>
<th>Site</th>
<th>Aboveground biomass (g DW m(^{-2}))</th>
<th>Stem density (# shoots m(^{-2}))</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>ZM</td>
<td>ZJ</td>
</tr>
<tr>
<td>1G</td>
<td>0.0 (0.0)</td>
<td>0.0 (0.0)</td>
</tr>
<tr>
<td>2J</td>
<td>0.1 (0.0)</td>
<td>26.8 (6.8)</td>
</tr>
<tr>
<td>3F</td>
<td>4.6 (1.4)</td>
<td>33.5 (8.6)</td>
</tr>
<tr>
<td>4F</td>
<td>17.0 (5.0)</td>
<td>19.4 (5.7)</td>
</tr>
<tr>
<td>5B</td>
<td>24.0 (4.4)</td>
<td>2.7 (0.6)</td>
</tr>
<tr>
<td>5F</td>
<td>44.7 (8.2)</td>
<td>1.3 (0.8)</td>
</tr>
<tr>
<td>5J</td>
<td>44.8 (9.8)</td>
<td>1.6 (1.4)</td>
</tr>
</tbody>
</table>

The two eelgrass species appeared to have different optimal elevations inferred from annual mean aboveground biomass with a zone of overlapping distribution (Figure 3). The \textit{Z. japonica} elevation range is almost entirely above MLLW here, while \textit{Z. marina} biomass continues increasing below MLLW. The highest \textit{Z. japonica} annual mean biomass observed along the transect occurred at site 3F with a relative elevation of 29.5 cm MLLW, while the highest \textit{Z. marina} biomass was observed at site 5J at -66.5 cm MLLW.
Figure 3. Annual mean aboveground eelgrass biomass for each species (± SE) along an elevation gradient.

The two species also appeared to differ in their elevations for maximum annual mean stem density (Figure 4). In addition, the elevations for maximum stem density and biomass differed for each species. The highest *Z. japonica* annual mean stem density observed along the transect occurred at site 2J at 49.0 cm MLLW, while the highest *Z. marina* stem density was observed at site 4F at 9.1 cm MLLW.
Figure 4. Annual mean eelgrass stem density for each species (± SE) along an elevation gradient.

2.3.3 Sediment characteristics

The mean bulk density profile showed a small increase in bulk density with depth in the sediment column (Figure 5A). Surface bulk density was 1.14 g cm\(^{-3}\) in the top 2-cm section, increasing gradually with depth until stabilizing after reaching 1.52 g cm\(^{-3}\) at 15 cm below the surface (Figure 5A). Organic content was highest at the surface with 2.4% organic matter by weight, declining with depth and stabilizing after reaching 1.3% organic matter at 13 cm depth (Figure 5B). Pore space showed a similar trend to organic content, with 55.1% pore space in the surface section, decreasing gradually then stabilizing after reaching 41.0% at 15 cm depth (Figure 5C).
2.3.4  Short-term accretion

2.3.4.1 Feldspar marker horizons

I was not able to obtain accretion measurements from the feldspar markers because they did not survive the entire year, likely due to a combination of erosion and sediment mixing. Sediment cores extracted with a transparent tube from the feldspar markers one year after installation showed no evidence of any detectable feldspar layer.

2.3.4.2 Wire mesh screens

Annual sediment accretion rates measured at wire mesh screens ranged from 0.12 cm yr$^{-1}$ at site 5J, to 2.29 cm yr$^{-1}$ at site 3F (Table 3). Erosion below the screens was observed at a few sites, but the vast majority of measurements were positive resulting in positive site
Any observed erosion covered less than the area of the wire mesh screen. This typically occurred in areas with a lumpy sediment surface. Seasonal accretion measurements are reported in Table 3 and shown in Figure 6.

Table 3. Seasonal accretion measurements (± SE) and annual accretion rates (± SE) at accretion monitoring sites (n = 2 at each site). Sites 5F and 5J were inaccessible in April 2014. The annual accretion rate was obtained from June 2013-June 2014 measurements standardized to one year to account for different site visit dates (with the exception of 1G from July 2013-July 2014).

<table>
<thead>
<tr>
<th>Site</th>
<th>Aug-13</th>
<th>Dec-13</th>
<th>Apr-14</th>
<th>Jun-14</th>
<th>Jul-14</th>
<th>Nov-14</th>
<th>Annual accretion rate (cm yr⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1G</td>
<td>0.31 (0.11)</td>
<td>0.40 (0.13)</td>
<td>0.03 (0.10)</td>
<td>0.54 (0.34)</td>
<td>0.71 (0.11)</td>
<td>0.64 (0.24)</td>
<td>0.73 (0.11)</td>
</tr>
<tr>
<td>2J</td>
<td>0.64 (0.04)</td>
<td>1.80 (0.20)</td>
<td>1.25 (0.03)</td>
<td>1.84 (0.01)</td>
<td>2.03 (0.13)</td>
<td>2.98 (0.28)</td>
<td>1.91 (0.01)</td>
</tr>
<tr>
<td>3F</td>
<td>0.76 (0.21)</td>
<td>1.96 (0.04)</td>
<td>1.38 (0.20)</td>
<td>2.20 (0.33)</td>
<td>-</td>
<td>2.84 (0.06)</td>
<td>2.29 (0.34)</td>
</tr>
<tr>
<td>4F</td>
<td>0.13 (0.00)</td>
<td>0.85 (0.10)</td>
<td>0.33 (0.15)</td>
<td>0.73 (0.45)</td>
<td>-</td>
<td>1.98 (0.53)</td>
<td>0.75 (0.47)</td>
</tr>
<tr>
<td>5B</td>
<td>0.23 (0.10)</td>
<td>0.51 (0.01)</td>
<td>0.10 (0.00)</td>
<td>0.26 (0.06)</td>
<td>-</td>
<td>1.48 (0.53)</td>
<td>0.27 (0.06)</td>
</tr>
<tr>
<td>5F</td>
<td>0.25 (0.05)</td>
<td>0.91 (0.06)</td>
<td>-</td>
<td>0.73 (0.05)</td>
<td>-</td>
<td>0.78 (0.08)</td>
<td>0.75 (0.05)</td>
</tr>
<tr>
<td>5J</td>
<td>0.34 (0.01)</td>
<td>0.28 (0.40)</td>
<td>-</td>
<td>0.11 (0.11)</td>
<td>-</td>
<td>0.03 (0.00)</td>
<td>0.12 (0.12)</td>
</tr>
</tbody>
</table>
The position of sites relative to one another in relation to accretion rates was generally maintained for the duration of the study. Sediment continued to accumulate on the screens from June 2013 to December 2013. All sites experienced some erosion of previously accumulated sediment between December 2013 and April 2014, followed by rapid accumulation from April to June 2014 and continued slower accumulation from June 2014 to November 2014 (Figure 6).

![Figure 6. Mean field measurements of sediment accretion at each accretion monitoring site from August 2013 to November 2014. Sites 5F and 5J were inaccessible in April 2014.](image-url)
The trend in aboveground eelgrass biomass with elevation did not correspond with accretion rates (Figure 7). Accretion rates followed biomass trends at the three higher elevation sites, but then decreased dramatically as biomass continued to increase at lower elevations.

Figure 7. Total annual mean eelgrass biomass (± SE) and accretion rate (± SE) at each accretion monitoring site. Total biomass is the sum of both eelgrass species at each site, averaged across 12 months.
In contrast, accretion rates did generally follow the trend in eelgrass stem density along the elevation gradient (Figure 8). The peak in accretion rates approximated the peak in stem density, both followed by a sharp decline and a leveling off at lower elevations.

![Graph showing stem density and accretion rate with error bars over a relative elevation range.]

Figure 8. Total annual mean stem density (± SE) and accretion rate (± SE) at each accretion monitoring site. Total stem density is the sum of both eelgrass species at each site, averaged across 12 months.

2.3.5 Long-term accretion

Results from the $^{210}$Pb analysis are presented here but I did not use them to analyze relationships between accretion and other site characteristics because I could not exclude the possibility of bioturbation or sediment deposition events at any of the sites, making the resulting rates unreliable. The rates presented here must be considered maximum possible rates, because bioturbation increases the apparent accretion rate. I did not analyze the core from site 1G because I considered it unlikely to yield a useful $^{210}$Pb profile based on its profiles of sediment characteristics. The Site 4F profile most closely resembled exponential decay with depth, followed by 3F and 5B (Figure 9). Sites 2J, 5F, and 5J produced $^{210}$Pb
profiles with less of an exponential decay pattern. Excess $^{210}$Pb values declined to zero within 10 cm depth at sites 3F, 4F, and 5B, suggesting fairly low accretion rates.

Figure 9. Excess $^{210}$Pb activity versus depth in the sediment column at six sites. Error bars represent ± 1 standard deviation. Hollow points represent excess $^{210}$Pb values less than 0 Bq g$^{-1}$, which were not included in the natural log regression used to calculate the accretion rates.
If it is assumed that these profiles are not influenced by bioturbation, then long-term accretion rates can be estimated. These rates are included in Figure 10. The lowest rate was 0.079 cm yr\(^{-1}\) from site 4F. Sites 2J, 3F, 5B, and 5F had similar accretion rates ranging from 0.18 cm yr\(^{-1}\) to 0.24 cm yr\(^{-1}\). Site 5J appeared to have the highest accretion rate of 0.49 cm yr\(^{-1}\) (Figure 10). The mean value (± SE) for long-term accretion from these six cores was 0.23 ± 0.06 cm yr\(^{-1}\).
Figure 10. Linear regressions of excess $^{210}$Pb against depth below the sediment surface, used to calculate long-term accretion rates at each site.
The $^{210}$Pb inventory exercise revealed that all six core sites have a lower $^{210}$Pb flux to the sediment than expected based on a published rate of 0.44 dpm cm$^{-2}$ yr$^{-1}$ (Nevissi 1985) from the atmosphere. The calculated $^{210}$Pb flux ranged from 0.15 to 0.43 dpm cm$^{-2}$ yr$^{-1}$, with a mean flux of $0.19 \pm 0.04$ dpm cm$^{-2}$ yr$^{-1}$. These low rates indicate that the core sites are net erosional, and the apparent long-term accretion rates are overestimates, being more an effect of bioturbation instead of sediment deposition.

2.3.6 Relationships between short-term accretion and site characteristics

All variables met the regression assumption of normality. The linear regression analysis revealed no effect of total aboveground eelgrass biomass on short-term sediment accretion rates ($R^2 = 0.0003$, $p = 0.97$, $n = 7$) (Figure 11A). I did observe a significant effect of stem density on sediment accretion rates ($R^2 = 0.68$, $p = 0.02$, $n = 7$) (Figure 11B). Finally, I observed no linear effect of relative elevation on sediment accretion rates ($R^2 = 0.28$, $p = 0.22$, $n = 7$) (Figure 11C). I also assessed a potential nonlinear relationship between elevation and accretion with a nonlinear multiple regression, including stem density and a quadratic term for elevation to predict accretion rates. When compared to the simple linear stem density regression, this multiple regression resulted in a minimal improvement to model fit, but an insignificant relationship that is only a minor improvement over the simple linear relative elevation regression ($R^2 = 0.77$, $p = 0.18$, $n = 7$). I therefore elected to use the simpler linear relationship between stem density and accretion for modeling purposes.
Figure 11. Effect of (A) mean annual aboveground eelgrass biomass, (B) mean annual stem density, and (C) relative elevation, on sediment accretion rates (mean ± SE) at each accretion monitoring site.
2.4 Discussion

2.4.1 Comparison of methods

This is not the first study to have difficulty obtaining consistent accretion rates from Padilla Bay using short- and long-term accretion methods. This typically inundated, erosional environment combined with a low sediment supply and large amount of bioturbation reduces the likelihood of success with most common accretion methods. Feldspar marker horizons in Padilla Bay tend to be either resuspended and washed away or mixed into deeper layers by bioturbation, as seen during this study as well as that by Maxwell (2004). Others have had limited success with feldspar marker horizons in other coastal ecosystems as well, such as mangroves (Woodroffe 1992) and tidal freshwater forests (Ensign et al. 2014), especially at relatively low elevation sites. Filters and sediment traps used to measure sediment deposition rates in Padilla Bay have also been washed away during the winter (Gwozdz 2006) and could cause localized scour (Nolte et al. 2013). Moreover, collection of sediment from traps can be difficult in an inundated environment leading to sediment loss and underestimation of accretion (Nolte et al. 2013), as Gwozdz (2006) observed. Long-term $^{210}$Pb accretion rates from sediment cores have been largely unreliable due to the prevalence of mixed sediment layers, either from bioturbation or physical reworking (Gwozdz 2006, Kairis 2008, this study). Sediment pins are another common method that would not likely provide reliable results in Padilla Bay. The pins would likely cause scour around the pin, particularly in this erosive environment. The wire mesh screens I employed in this study appear to have shown relative success in allowing for at least one year of vertical accretion measurements. The screens were superior to the feldspar markers because they persisted throughout the entire year and were easy to relocate. They allowed for measurements even in erosive conditions,
reducing the chance of overestimating accretion rates, which is often the result of using only the subset of feldspar marker horizons that are not eroded (Cahoon and Lynch 1997). The screens had no obvious impact on localized sediment dynamics because they were flat, and the mesh maintained the natural sediment surface texture while allowing eelgrass growth through the screen. The screens were also not affected by any potential sediment mixing occurring below the surface. Finally, of all the short-term accretion methods reviewed by Nolte et al. (2013), sediment plates (functionally similar to wire mesh screens) have the highest estimated accuracy of 1.5 mm compared to the 5-10 mm accuracy of other methods.

It is possible that the presence of the wire mesh screens on the sediment surface interfered with natural sediment dynamics, especially immediately after installation. The wire screen and stakes could potentially increase sedimentation or erosion, or reduce resuspension. Any effect of the equipment, however small, could have a significant impact on the measured accretion rates when the accretion rates themselves are so low. However, I did not observe any discontinuity in the sediment surface at the screen locations relative to the surrounding surface, thus I believe the screens had no significant impact on the measured accretion rates. To maximize the success of the screens in future studies, I would recommend installing them at the beginning of the growing season to allow the screens to become buried as soon as possible.

2.4.2 Temporal trends in accretion rates

Short-term accretion rates followed a common seasonal pattern with rapid accretion in the spring and summer and some erosion occurring during the winter (Pasternack and Brush 1998, 2001; Bos et al. 2007; Delgado et al. 2013). This is the same seasonal pattern that has been observed with elevation change at the SET sites in Padilla Bay (Kuhlman
Accretion rates are typically highest during the growing season when plant stem density is highest and water is calmer, allowing suspended sediment to be trapped and deposited on the surface. Erosion tends to occur in the winter when high-energy waves are more frequent and plant density is lower, allowing surface sediments to be resuspended and relocated (Orson et al. 1992). The accretionary period was observed starting again the second spring, and if this pattern continues at these rates, these sites will continue to gain elevation over time. However, additional years of data from these sites would make these accretion rates more trustworthy. Longer monitoring time periods have a better chance of integrating episodic accretionary or erosional events into the trend, instead of allowing a small number of these events to completely determine the accretion rates (Day et al. 1995, Neubauer et al. 2002, Delgado et al. 2013). The 2013-2014 year could have been a year with a relatively large amount spring/summer accretion, or relatively little winter erosion. Another study looking at seasonal accretion rates saw intertidal annual eelgrass beds lose all their accreted sediment during the winter (Bos et al. 2007). This is in contrast with their lower-density perennial eelgrass beds, which accreted more on an annual basis because although they gained less during the growing season, they eroded less during the winter. My sites did not lose all the accreted sediment in the winter that they had gained during the spring/summer. It can also take some time (several years) before this kind of system returns to equilibrium after the disturbance of equipment installation. For example, some SETs in Padilla Bay took up to three years before beginning to reveal more consistent rates of elevation change (see Chapter 3). Short-term accretion at several screens along the biomonitoring transect appears to be starting to level off as of 1.5 years following installation. In any case, the seasonal accretion trend observed at these sites so far appears to be reasonable.
2.4.3  Comparison of accretion rates across methods and studies

Short-term accretion rates obtained from wire mesh screens for this study were roughly an order of magnitude higher than the long-term rates I obtained from the same sites using excess $^{210}$Pb profiles. Although this is quite a substantial difference, it is not unexpected. Longer-term accretion measurements are typically lower because sediment becomes denser due to decomposition over time and compaction with depth (Callaway et al. 1996, Neubauer et al. 2002).

Previous studies in Padilla Bay have reported short-term accretion rates somewhat similar to mine, if not slightly higher. Maxwell (2004) used feldspar marker horizons at the SET sites throughout Padilla Bay to obtain accretion rates ranging from 1.33 cm yr$^{-1}$ to 4.10 cm yr$^{-1}$, with a mean rate of 2.51 cm yr$^{-1}$. These rates are somewhat higher than my short-term rates, which ranged from 0.12 cm yr$^{-1}$ to 2.29 cm yr$^{-1}$. Maxwell was unable to recover feldspar from four of the 18 sites she surveyed, attributing the loss of feldspar to bioturbation. Using a sediment trap method to obtain even shorter-term accretion rates over a bi-weekly timeframe instead of a yearly timeframe, Gwozdz (2006) reported an average of 55 g m$^{-2}$ day$^{-1}$ at a Z. marina site, and approximately 25 g m$^{-2}$ day$^{-1}$ at a Z. japonica site. These rates convert to roughly 0.29 cm yr$^{-1}$ and 0.13 cm yr$^{-1}$ respectively, assuming a surface bulk density of 1 g cm$^{-3}$. These rates are nearer the lower end of the range of my short-term accretion rates.

Other studies of long-term accretion rates in Padilla Bay have reported a range of rates, some very similar to the rates I obtained, and some much higher. Carpenter et al. (1985) reported $^{210}$Pb accretion rates from 27 sites throughout Puget Sound in the 1970s. They sampled two sites in Padilla Bay and determined those rates to be 0.270 g cm$^{-2}$ yr$^{-1}$
(approximately 0.24 cm yr\(^{-1}\)) in northern Padilla Bay, and 0.910 g cm\(^{-2}\) yr\(^{-1}\) (approximately 1.12 cm yr\(^{-1}\)) in southern Padilla Bay. The three cores that they sampled from Padilla Bay (one from the northern half of the bay and two from the southern half) had mixed layer depths of 15, 6.7, and 13 cm. Their average rate of 0.24 cm yr\(^{-1}\) from northern Padilla Bay is similar to rates I obtained from three cores along the transect also in northern Padilla Bay. Gwozdz (2006) obtained long-term accretion rates from two of three cores that he analyzed from sites dominated by \(Z.\ marin\a\). Eliminating portions of the \(^{210}\)Pb profiles that did not indicate exponential decay, Gwozdz reported accretion rates of 0.948 cm yr\(^{-1}\) and 1.024 cm yr\(^{-1}\), roughly two to ten times the rates I obtained. Using the same methods, Kairis (2008) reported rates from four SET sites ranging from 0.109 cm yr\(^{-1}\) to 0.488 cm yr\(^{-1}\) with a mean of 0.255 ± 0.088 cm yr\(^{-1}\), which more closely resembles the rates determined from this study.

Accretion studies in seagrass meadows from other parts of the world are few. Bos et al. (2007) reported an accretion rate of 0.5 cm to 0.7 cm during the growing season in an intertidal annual \(Z.\ marin\a\) bed in the Wadden Sea, using a Stanley Compulevel for short-term rates. This amount may have increased had they used experimental planting units larger than 1.8 m in diameter. Garcia and Duarte (2001) reported an accretion rate of 0.2 cm yr\(^{-1}\) within a subtidal, perennial \(P.\ oceanica\) meadow in the Mediterranean Sea. They used sediment traps for one year to obtain this rate, which corresponds closely with the majority of my Padilla Bay long-term accretion rates.

Flow velocities, and perhaps accretion, are also affected by proximity to channels or open water (Christiansen et al. 2000). I did not examine the effect of channel proximity on accretion rates, although this is known to be an important factor (DeLaune et al. 1989, French and Spencer 1993, Esselink et al. 1998, Neubauer et al. 2002, Chmura and Hung 2004,
Rates of accretion and elevation change are known to be higher nearer channels also because there is a higher chance of sediment deposition near the freshwater source. However, this correlation has not been reported from seagrass meadow studies. It is possible that channel proximity has a much smaller effect on accretion in Padilla Bay’s eelgrass meadows because the entire meadow is virtually always inundated. Channel proximity could potentially explain why site 3F had such a high accretion rate because a small channel developed approximately 10 m from it, although the accretion rate here could also be reasonably explained by the stem density. Furthermore, site 2J also showed a similarly high accretion rate and it was not located near a channel. Ultimately, although a more thorough examination of factors affecting accretion rates could include channel proximity, I did not include it in this study because my primary objective was to obtain field data to inform a model that is not spatially explicit.

The short-term accretion rates reported here should not be used to draw conclusions about the long-term sustainability of Padilla Bay’s eelgrass meadows in the face of sea level rise. Although one can calculate the accretion deficit based the rates of accretion and sea level rise, this deficit does not incorporate other important processes operating at different time scales, such as decomposition and compaction (Boumans and Day 1993). It is better to assess wetland sustainability using the elevation deficit, which is based on sea level rise and elevation change as measured by a surface elevation table (Cahoon and Lynch 1997). My accretion rates were obtained for the purpose of determining the relative effects of elevation and plant characteristics on accretion rates, and for calibrating the short-term sediment inputs in the hybrid surface elevation model (see Chapter 4). Surface elevation change rates better suited for an analysis of long-term sustainability are documented in Chapter 3.
2.4.4 Relationships between short-term accretion, plant characteristics, and elevation

I expected accretion rates to be dependent on a combination of aboveground biomass and relative elevation, as measured and modeled by Morris et al. (2002). In fact, neither of these factors appeared to affect short-term accretion rates at the sites I monitored in Padilla Bay. Stem density was the only factor significantly affecting accretion rates along the entire transect. Although this was unexpected, it is not unreasonable. Other studies have shown increased stem density to increase accretion rates by reducing water velocity, allowing for increased deposition and/or reduced resuspension (Gambi et al. 1990, Leonard and Luther 1995, Neumeier and Ciavola 2004, Tsai et al. 2010, Baustian et al. 2012). Stem density has also been shown to affect accretion rates by increasing particle collisions (Hendriks et al. 2008). Agawin and Duarte (2002) report that the tropical seagrass, *Thalassia hemprichii*, can trap up to 70% of suspended particles within the canopy in less than one hour, and the dominant trapping mechanism may be active trapping (sediment ingestion by epibionts) rather than passive trapping (particle collisions on leaves). That said, it is possible that biomass has a greater effect on accretion in other types of coastal wetlands under different conditions, and it is possible that aboveground biomass could be proportional to stem density in a system with only one dominant plant species, or multiple plant species with similar density-biomass relationships (e.g., Morris 2006). In Padilla Bay however, the two eelgrass species occurring there demonstrated vastly different stem density-biomass relationships, thus one cannot be used as a proxy for the other. Stem density and aboveground biomass were not linearly related, particularly with *Z. marina*. Where *Z. marina* biomass continues to increase at lower elevations as its leaves lengthen toward its light source at the surface, stem density peaks (at its “maximum sustainable density”) then declines at lower elevations due to
self-shading (Zimmerman 2006). In contrast, *Z. japonica* shows a more linear stem density-biomass trend. This species may never reach the height or the elevation that results in self-shading. It is also possible that leaf litter, which could be estimated as a function of aboveground biomass, forms a greater component of accreting material in other systems. In Padilla Bay’s frequently inundated and highly erosive environment, it is probable that a large fraction of sloughed leaves are swept away and deposited elsewhere, instead of being deposited in situ to contribute to the short-term accretion I measured. The low surface layer sediment organic content in Padilla Bay (this study, Gwozdz 2005) relative to what would be expected based on measured productivity rates (Thom 1990), suggests that only a small fraction of leaf production remains in situ. In this case, the majority of accreted material appears to be allogenic mineral sediment trapped by plants, which is primarily driven by stem density.

Recent research suggests that CO$_2$ enrichment of coastal waters caused by ocean acidification will likely increase eelgrass stem densities because eelgrass is typically C-limited (Palacios and Zimmerman 2007, Alexandre et al. 2012). This could create a positive feedback for eelgrass populations, wherein increased stem densities lead to increased accretion rates, enhancing light availability and allowing the population to expand at the lower end of its elevation range.

Rich Gwozdz analyzed the effect of eelgrass biomass on short- and long-term accretion rates at three sites in Padilla Bay (Gwozdz 2006). Contrary to my results, Gwozdz found short-term accretion rates in a *Z. japonica* meadow site to be less than half that observed in the *Z. marina* site, and no significant difference between the *Z. japonica* site and the mixed species site. These results could be due to different measurement time periods,
different short-term accretion methods, or the fact that Gwozdz’s results are based on fewer sites, with one site from each of the three habitat types. It is also possible that accretion is driven by different factors in different locations within the bay. For example, the southern half of the bay has larger and more numerous channels with more exposure to the Swinomish Channel, hence sediment may be transported differently there than in the northern half of the bay. Kairis (2008) found significant differences between the northern and southern bay regions in bulk density, organic content, and porosity, suggesting regional hydrodynamic differences.

Although my Padilla Bay field measurements showed relative elevation to have no significant effect on accretion rates, elevation is frequently used as a determinant of accretion rates in intertidal surface elevation models because it is proportional to the fraction of time that the wetland is tidally inundated (Krone 1987, Callaway et al. 1996, Rybczyk et al. 1998, Morris et al. 2002, Kairis and Rybczyk 2010, Rogers et al. 2012, Swanson et al. 2013). Mineral sediment deposition can only occur during a period of inundation. The relationship between relative elevation and accretion may be relatively weak in Padilla Bay because the eelgrass meadows occur much lower in the intertidal zone than the salt marshes typically studied and modeled for similar purposes. Much of the eelgrass elevation range in Padilla Bay is below MLLW, below which the sediment surface is rarely exposed. Even during very low spring tides, the ability of the eelgrass meadow to retain water during the ebbing tide (Powell and Schaffner 1991, Tsai et al. 2010) causes the eelgrass meadow to be inundated longer than would be expected based on tidal elevation alone. Therefore, the minimal difference in the effective inundation period at different transect elevations could explain the lack of any significant effect of elevation on sediment accretion rates observed in this study.
Cahoon and Lynch (1997) similarly found no effect of relative elevation on accretion rates in a Florida mangrove forest, though this was attributed to reasons differing from those mentioned above. One of their sites was a basin receiving most of its sediment as organic litter from within the basin. Periods of tidal inundation at their other more exposed sites experienced more erosion than deposition, due to the occurrence of high wave energy and plants with low sediment trapping abilities. Not only did I fail to detect a negative linear trend in accretion with elevation, I actually observed a non-significant positive linear relationship between accretion and relative elevation (Figure 11). There does, however, appear to be a nonlinear relationship between elevation and accretion, with an accretion peak at approximately 30 cm above MLLW, but this is most likely a reflection of variation in stem density instead of elevation itself. Perhaps Padilla Bay’s lower elevations experience similar conditions to the outer mangrove forests reported by Cahoon and Lynch (1997), with tidal inundation causing more erosion than accretion. Further studies here with additional sites could allow for a more robust analysis of the relationship between elevation, stem density, and accretion. However, this study indicates that stem density is simply more important than elevation as a determinant of accretion rates.

Accretion has also been shown by other studies to be affected by plant stiffness. Bouma et al. (2005) compared accretion rates in patches exposed to similar currents and waves but dominated by either Spartina anglica or Zostera noltii, finding accretion rates to be much higher in the patches with the stiffer Spartina plants. Although this stiffness effect makes intuitive sense, I did not include it in my analysis under the assumption that the difference in stiffness between the two Zostera species is negligible. It could be a valuable
addition to other accretion studies comparing plant species with more drastic differences in their physical structure.

My analysis of the relationships between accretion and plant characteristics could be flawed because I did not use accretion and plant data from the same time period. The available biomass and stem density data were from 2011-2012 measurements, while accretion data were from 2013-2014 measurements. Although this is only a two-year difference, eelgrass patch distribution has been observed varying interannually (Bulthuis and Shull 2002). It is possible that eelgrass biomass and stem density have changed during that two-year lag, causing my analysis to be inaccurate, although no notable interannual patch differences were observed during the 2013-2014 period. Although PNBERR is no longer monitoring eelgrass biomass and stem density on a monthly basis, this potential temporal variability in plant characteristics could be assessed in the future with a comparison of annual peak values, given the annual surveys continue to be performed at approximately the same time of year.

2.4.5 Carbon accumulation rates

My field measurements of bulk density, organic content, and long-term accretion rates allowed me to estimate site-specific rates of carbon accumulation. The ability of coastal wetlands to sequester carbon in their sediment has become a topic of interest within the research and restoration communities in recent years. Coastal wetland ecosystems, including seagrass meadows, tidal marshes, and tidal forests, are now known to be efficient carbon sinks, collectively accumulating carbon 35 times faster than tropical rainforests due to their high rates of productivity, low rates of decomposition, and carbon burial by mineral sediments (Macleod et al. 2011). This carbon sequestration capacity is a newly recognized
coastal wetland ecosystem service, and these ecosystems mitigate climate change by removing carbon from the atmosphere and burying it long-term in the sediment. This service allows these wetlands to be entered into the voluntary carbon market, once an estimate of the actual carbon sequestration rate is determined. Of the three coastal wetland types, carbon sequestration by seagrasses is the least understood (Grimsditch 2013). The relatively high variability in seagrass carbon sequestration and storage could be due to variability among species and habitats (Grimsditch 2013), or possibly due to the relative difficulty in obtaining reliable long-term accretion rates in these environments as I experienced.

Based on a review of seven available studies summarizing a total of 123 sites, Macleod et al. (2011) reported carbon accumulation rates in seagrass meadows to range from 45 to 190 g C m\(^{-2}\) yr\(^{-1}\), with a mean rate of 138 ± 38 g C m\(^{-2}\) yr\(^{-1}\). Methods have not been consistent across all these studies, but they do all use some measure of long-term accretion rather than short-term measures. Research methodologies have been refined and standardized to develop a greenhouse gas offset protocol, allowing for more robust comparisons of rates across sites and ecosystem types (Emmer et al. 2015). I followed methods described by Crooks et al. (2014) and Callaway et al. (2012) to calculate carbon accumulation rates at the accretion monitoring sites in Padilla Bay, as the product of the sediment carbon density and the long-term sediment accretion rate. I used long-term rather than short-term accretion rates because short-term rates do not incorporate the long-term processes of compaction and decomposition (Callaway et al. 1996, Neubauer et al. 2002), which would result in a gross overestimate of long-term carbon accumulation rates (Callaway et al. 2012). Since I did not directly measure sediment carbon content, I converted organic matter content to organic carbon content using an equation from a similar Snohomish River estuary study (Poppe 2013).
unpublished data). This regression equation was based on 191 core sections indicating carbon content (% by weight) to be roughly half of organic matter content (Carbon = 0.0017*OM content\(^2\) + 0.4147*OM content). This equation was very similar to equations from other published studies (Callaway et al. 2012, Craft et al. 1991). I determined carbon density based on bulk density and carbon content for each core section, then found a representative carbon density value for each core by averaging carbon density from the surface to the depth at which excess \(^{210}\)Pb declines to zero. The carbon density and accretion rate used to calculate the carbon accumulation rate for each core thus approximated 100-year average values.

Carbon accumulation rates from my Padilla Bay transect sites ranged from 8.2 to 40.2 g C m\(^{-2}\) yr\(^{-1}\), with the lowest rate at site 4F and the highest rate at site 5J. The mean (± SE) rate at these six sites was 20.1 ± 4.5 g C m\(^{-2}\) yr\(^{-1}\). Rates from sites 2J, 5F, 5B, and 3F were similar to each other with 13.9, 15.8, 20.9, and 21.6 g C m\(^{-2}\) yr\(^{-1}\), respectively. All carbon accumulation rates followed the same trend as long-term accretion rates among sites because accretion rates vary more than carbon content among sites. These rates are all overestimates due to the likelihood of particle mixing affecting the \(^{210}\)Pb-based accretion rates used in their calculation. Yet, despite this unreliability, this range of carbon accumulation estimates is valuable in demonstrating a low carbon sequestration capacity of Padilla Bay’s eelgrass meadow, relative to seagrass meadows at other locations as well as other types of coastal wetland ecosystems. The mean carbon accumulation rate from Padilla Bay is lower than any other currently published rates from similar studies. Even the highest carbon accumulation rate reported here is less than the lowest rate seen in tidal marshes along the U.S. west coast. For example, rates from 58.0 to 352.1 g C m\(^{-2}\) yr\(^{-1}\) were observed in the Snohomish River estuary in Puget Sound (Crooks et al. 2014), and rates from salt marsh sites in San Francisco...
Bay ranged from 46 to 117 g C m\(^{-2}\) yr\(^{-1}\) (Callaway et al. 2012). Padilla Bay rates are also lower than the lowest reported rate of 45 g C m\(^{-2}\) yr\(^{-1}\) from a number of other seagrass studies (Macleod et al. 2011). Greiner et al. (2013) reported a carbon accumulation rate of 36.68 g C m\(^{-2}\) yr\(^{-1}\) for a 10-year restored Z. marina meadow in the coastal bays of Virginia. This rate is closest but still higher than the majority of Padilla Bay rates.

The relatively low carbon sequestration rates from Padilla Bay are a result of both low long-term accretion rates and low sediment organic content. If short-term accretion rates were used in the calculation instead of long-term rates, the resulting carbon accumulation rates would range from 9.9 to 252.6 g C m\(^{-2}\) yr\(^{-1}\) with a mean (± SE) of 94.0 ± 37.1 g C m\(^{-2}\) yr\(^{-1}\). However, the use of short-term accretion rates for this purpose is discouraged because much of this accretion is eventually lost to shallow subsidence, resulting in an overestimate of long-term carbon accumulation rates as stated above (Callaway et al. 2012). Most studies now use long-term accretion rates for this reason, and I chose to use these same methods to allow for cross-study comparisons of results. In contrast, my reported carbon accumulation rates would be lower (or even negative) if accretion rates were replaced with elevation change rates in the calculations. A negative elevation change rate measured at a surface elevation table implies that sediment is being lost along with the carbon stored within it, and eelgrass roots and rhizomes are not being buried. These findings are contrary to the prevalent assumption that seagrass meadows must have a substantial capacity to sequester carbon, because they have high rates of net primary production (Zieman and Wetzel 1980) and decay-resistant roots and rhizomes (Fourqurean and Schrlau 2003). A high carbon sequestration rate under these conditions is contingent on the leaf litter being buried in situ instead of being transported elsewhere, and also on roots and rhizomes being buried by new
sediments, which this study suggests is not happening very quickly in Padilla Bay. If the eelgrass leaf litter is exported beyond Padilla Bay, the system could still be considered a carbon sink if the litter is buried at its destination. However, our current lack of information about the fate of the exported eelgrass limits this analysis to on-site carbon dynamics.

Future carbon sequestration rates may actually be enhanced as ocean acidification causes higher eelgrass stem densities and a concomitant increase in accretion (Palacios and Zimmerman 2007). Garrard and Beamont (2014) estimated an 86% increase in seagrass carbon sequestration rates by 2100, based on an expected 80% increase in stem density and a 125% increase in belowground biomass. This may be the single positive outcome of climate change in Padilla Bay’s eelgrass meadow, though it remains to be seen whether the benefit of CO₂ enrichment can outweigh the negative effects of sea level rise.

It seems the vast array of crucial ecosystem services provided by seagrass meadows is growing daily (Batker et al. 2008), and it is tempting to add carbon sequestration to the list, especially when this ecosystem service is an emerging conservation tool (Crooks et al. 2010). Unfortunately, carbon sequestration may not be the service to highlight when applied to seagrasses, particularly sediment-starved eelgrass meadows with very low accretion rates, unless exported material is tracked and eventually buried elsewhere. I would therefore suggest that further research be focused on obtaining carbon sequestration rates from other seagrass meadows varying in sediment supply, dominant species, and litter fate. And if the need persists to champion the carbon sequestration capacity of a seagrass meadow such as Padilla Bay, the sequestration rate could be increased by increasing the sediment supply with freshwater-derived sediment, such as with a Skagit flood diversion channel. Although the carbon sequestration rates that I calculated from Padilla Bay accretion rates are relatively
low, it is possible they are still greater than zero, which would qualify this habitat as a carbon sink. However, better measures of long-term sediment accretion are necessary to confirm this. Given a total eelgrass extent of 3,800 ha and an average carbon accumulation rate of 20.1 g C m$^{-2}$ yr$^{-1}$, the bay’s eelgrass meadows could be accumulating up to (though likely less than) 764 t C yr$^{-1}$. If the eelgrass meadows in Padilla Bay were replaced with macroalgae, as is occurring in many places worldwide due to human-caused eutrophication (Vitousek et al. 1997), the carbon sequestration capacity of the bay could decline to nearly zero due to the highly labile qualities of macroalgal tissue (Macreadie et al. 2012). Roughly 0.4% of total macroalgae NPP is stored in the soil in contrast to 15.9% of seagrass NPP (Duarte and Cebrian 1996). Finally, although carbon sequestration rates in seagrass beds may be low relative to tidal marshes and mangroves per unit area, the greater global coverage of seagrasses suggests that the carbon they sequester may be comparable on a global scale (Garrard and Beaumont 2014).

### 2.4.6 Suggestions for future research

It might be beneficial for future studies to measure leaf area to allow for estimates of leaf area index (LAI). This parameter combines leaf surface area and stem density and may be a better predictor of sediment accretion than stem density alone. Gacia et al. (1999) found LAI to be significantly correlated with sediment trapping in a *Posidonia oceanica* meadow up to an LAI threshold above which other processes complicate the relationship. Fonseca et al. (1982) found seagrass LAI to significantly affect current flow.

Spatial variation in suspended sediment concentration (SSC) was not considered in this study, but it is probable that SSC differs among sites along the transect, and this may be a confounding factor affecting accretion rates. Total suspended solids (TSS) measurements
taken throughout an entire year at each sediment accretion monitoring site would reveal whether SSC varies across sites, and if so, this spatial SSC variation could be a valuable addition to this accretion analysis.

Channel proximity may be another confounding factor not considered in this study. The simplest way to capture the effect of this variable may be to measure the distance from each accretion monitoring site to the nearest channel. Of course, this does not consider complexities such as channel size or flow direction and velocity, but it may still capture the effect of channel sediment delivery on accretion rates.
3 SURFACE ELEVATION CHANGE

3.1 Introduction

A surface elevation table (SET) is an instrument used in tidal wetlands to measure surface elevation change over time. Once established, SETs can remain in place indefinitely, thus providing an opportunity for monitoring long-term surface elevation change. SET measurements incorporate accretion or erosion at the surface as well as shallow subsidence due to decomposition or compaction. SETs allow for relatively precise (< 1.5 mm) (Boumans and Day 1993) measurements of surface elevation change that can then be compared with short-term accretion measurements using wire screens or feldspar marker horizons, and the difference between the accretion and elevation change measurements can be attributed to shallow subsidence (Figure 12).

Figure 12. A diagram of a surface elevation table (SET) (also known as a sedimentation erosion table) and marker horizon, illustrating the sediment processes that they measure (Figure from Cahoon and Lynch 2003).
Relative elevation change can then be determined from the SET-derived elevation change rates by incorporating rates of sea level rise and vertical land movement:

\[
\text{Relative elevation change (cm yr}^{-1}\text{)} = \text{Surface elevation change} - \text{RSLR} \quad (2)
\]

I used these values of relative elevation change from SETs in Padilla Bay for model validation, as described in Chapter 4. This relative elevation change dataset relied heavily on prior work done by Maxwell (2004), Kairis (2008), and Kuhlman (2011). I sampled the SETs in 2012, 2013, and 2014 to add to this dataset.

### 3.2 Methods

#### 3.2.1 SET installation

18 SETs were installed in 2002 by Maxwell (2004), followed by the installation of five more SETs in 2004 and 2005 by Kairis (2008), and one more in 2010 by Kuhlman (2011). Sites were selected to cover a broad range of locations throughout the bay and an elevation range of almost 2 m within the intertidal zone (Figure 13). Lower (subtidal) elevations are under-represented, due to SET installation and measurement being limited to intertidal elevations. Four of the original 18 SETs were disturbed and removed from analysis due to their proximity to channels with periodic dredging and frequent boat traffic. Six additional sites were eliminated from ongoing monitoring in 2010 due to extensive scour around the SET cement collar.
Each SET was installed by pounding a stainless steel rod into the sediment to the point of refusal (typically 4-5 m). The rod is considered a permanent benchmark from the bottom of the zone of shallow subsidence. Cement was poured around the rod to form a stabilizing collar approximately 10-20 cm deep, 16 cm in diameter, and flush with the sediment surface. A metal coupler was attached to the top of the rod remaining approximately 30 cm above the surface. Four of the eight possible arm positions opposite the designated approach side at each SET were selected for measurements after installation. During each site visit, a portable SET device consisting of a metal arm with levels was
attached to the rod coupler. Nine fiberglass pins fit through the SET arm, and the pins were lowered to rest on the sediment surface. The distance from the SET arm to the top of each pin was measured at each designated arm position.

Additionally, elevation at each site was surveyed between May 2010 and January 2011 using rapid static GPS surveying, with the GPS receiver (Javad Maxor GGDT) securely fastened to the top of the SET benchmark using a custom-made 3/8” – 5/8” threaded adapter. Elevation was adjusted in reference to the height of each SET cement collar, which was flush with the surface at installation. Thus, the 2010-2011 elevation measurements represent installation elevations. NAVD88 elevations were converted to tidal elevations (MLLW) using the same 0.12 m conversion factor described in Chapter 2 (Table 4).
### Table 4. Relative elevation and location of all SET sites in Padilla Bay.

<table>
<thead>
<tr>
<th>Site</th>
<th>Elevation (m MLLW)</th>
<th>Latitude</th>
<th>Longitude</th>
<th>Date of last measurement</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>0.47</td>
<td>48.5623</td>
<td>-122.5175</td>
<td>1/3/2011</td>
</tr>
<tr>
<td>1N</td>
<td>0.47</td>
<td>48.5623</td>
<td>-122.5175</td>
<td>6/16/2014</td>
</tr>
<tr>
<td>2</td>
<td>0.32</td>
<td>48.5593</td>
<td>-122.5242</td>
<td>6/15/2014</td>
</tr>
<tr>
<td>3</td>
<td>0.75</td>
<td>48.5540</td>
<td>-122.5093</td>
<td>1/2/2011</td>
</tr>
<tr>
<td>4</td>
<td>0.29</td>
<td>48.5526</td>
<td>-122.5160</td>
<td>6/15/2014</td>
</tr>
<tr>
<td>5</td>
<td>0.66</td>
<td>48.5433</td>
<td>-122.5047</td>
<td>8/5/2010</td>
</tr>
<tr>
<td>6</td>
<td>0.21</td>
<td>48.5402</td>
<td>-122.5171</td>
<td>6/15/2014</td>
</tr>
<tr>
<td>7</td>
<td>0.58</td>
<td>48.5186</td>
<td>-122.4923</td>
<td>6/14/2014</td>
</tr>
<tr>
<td>8</td>
<td>0.26</td>
<td>48.5166</td>
<td>-122.5034</td>
<td>6/14/2014</td>
</tr>
<tr>
<td>9</td>
<td>0.73</td>
<td>48.5090</td>
<td>-122.4915</td>
<td>5/17/2010</td>
</tr>
<tr>
<td>10</td>
<td>0.07</td>
<td>48.5096</td>
<td>-122.5014</td>
<td>6/13/2014</td>
</tr>
<tr>
<td>11</td>
<td>0.57</td>
<td>48.5025</td>
<td>-122.4900</td>
<td>6/13/2014</td>
</tr>
<tr>
<td>12</td>
<td>0.04</td>
<td>48.5031</td>
<td>-122.5001</td>
<td>6/23/2013</td>
</tr>
<tr>
<td>13</td>
<td>0.26</td>
<td>48.4935</td>
<td>-122.4865</td>
<td>6/16/2014</td>
</tr>
<tr>
<td>14</td>
<td>0.20</td>
<td>48.4929</td>
<td>-122.4895</td>
<td>6/16/2014</td>
</tr>
<tr>
<td>15</td>
<td>0.97</td>
<td>48.4765</td>
<td>-122.4748</td>
<td>6/15/2010</td>
</tr>
<tr>
<td>16</td>
<td>1.02</td>
<td>48.4740</td>
<td>-122.4768</td>
<td>6/15/2010</td>
</tr>
<tr>
<td>17†</td>
<td>0.78</td>
<td>48.4760</td>
<td>-122.5246</td>
<td>8/9/2010</td>
</tr>
<tr>
<td>18†</td>
<td>0.41</td>
<td>48.4797</td>
<td>-122.5277</td>
<td>8/9/2010</td>
</tr>
<tr>
<td>4B†</td>
<td>-0.38</td>
<td>48.5441</td>
<td>-122.5391</td>
<td>6/14/2010</td>
</tr>
<tr>
<td>5B</td>
<td>-0.92</td>
<td>48.5337</td>
<td>-122.5380</td>
<td>7/12/2014</td>
</tr>
<tr>
<td>12B</td>
<td>-0.41</td>
<td>48.5044</td>
<td>-122.5263</td>
<td>7/13/2014</td>
</tr>
<tr>
<td>14B</td>
<td>0.01</td>
<td>48.5441</td>
<td>-122.5390</td>
<td>5/29/2013</td>
</tr>
<tr>
<td>16B†</td>
<td>0.71</td>
<td>48.5337</td>
<td>-122.5380</td>
<td>5/14/2010</td>
</tr>
</tbody>
</table>

Mean: 0.34 ± 0.45 (sd)
Min: -0.92
Max: 1.02

† Sites were eliminated from analysis, but location data are included here.

### 3.2.2 Elevation change measurements

Beginning in 2002, monitoring was conducted biannually for the first four years of the study, with less frequent sampling every one to two years in subsequent years. I visited two of these SET sites in the summer of 2012, 13 sites in 2013, and 12 sites in 2014 to
continue measuring surface elevation change. 10 of the existing 23 SETs were excluded from my SET field survey due to scour reported by Kuhlman (2011). My SET measurements followed methods described by Kairis and Rybczyk (2010), consistent with methods used by all the previous SET studies in Padilla Bay. At each designated arm position, nine pins on the SET arm were adjusted until they settled on the sediment surface. Where the surface sediment was very soft, the pins were placed such that they rested on the more consolidated surface below the floc layer. The length of each pin above the arm was then recorded.

### 3.2.3 Data analysis

#### 3.2.3.1 Surface elevation change rate

SET measurements were used to determine elevation change rates following methods used by Kairis and Rybczyk (2010) and Kuhlman (2011). SET pin heights above the leveling arm were averaged for all four SET arm positions to obtain one value for each SET (n = 36) for each sampling event. Elevation change rates were determined for each SET individually using an ordinary least squares linear regression, with time as the independent variable and elevation relative to initial as the dependent variable. The slope of the regression indicated the rate of surface elevation change. All statistical analyses were executed in the R programming environment using a critical value (α) of 0.05 (R Core Team 2014).

#### 3.2.3.2 Scour correction

Several of the SET elevation change rates had been adjusted for scour by Kuhlman (2011) at which time those sites were eliminated from subsequent surveying. The scour observed at these sites was evidenced by a small depression surrounding the SET rod, caused by localized erosion due to the presence of the rod. Kuhlman performed a systematic analysis of SET measurements, and I followed these methods for my measurements from 2012, 2013,
and 2014 to potentially identify any new scour sites. Each pin’s height was averaged across the four arm positions. The average pin height was regressed against the pin position along the SET arm from nearest to farthest from the SET rod. If the slope of this regression exceeded 0.19 cm pin$^{-1}$, that SET was determined to be scoured. Any measurements at that SET were removed from the analysis beginning at the first scour detection, and including any subsequent sampling periods. I used the scour-adjusted elevation change rates reported by Kuhlman (2011) for these sites in my analysis, and only continued surveying SETs that did not meet Kuhlman’s scour criterion.

3.2.3.3 Relative elevation change rate

To determine relative elevation change rates (relative to sea level), I used parameter values that differed from those used by Kairis and Rybczyk (2010) and Kuhlman (2011). Whereas Kairis and Rybczyk (2010) used a global average ESLR rate of 0.33 ± 0.04 cm yr$^{-1}$ and a vertical land movement rate of 0.02 ± 0.08 cm yr$^{-1}$, I used the sea level rise rate of 0.113 ± 0.033 cm yr$^{-1}$ reported for the nearest tide gauge station in Friday Harbor, WA for the period 1934-2006 (NOAA Tides and Currents) with no additional vertical land movement. This rate represents relative sea level rise (RSLR) because tide gauge measurements incorporate vertical land movement. The current rate of relative elevation change was calculated using Equation 2 (above).

3.2.3.4 Spatial and temporal trends in elevation change rates

I analyzed the relationship between surface elevation and elevation change rates to compare current relationships against those reported by Kuhlman (2011). I used Pearson’s correlation test to maintain consistency with Kuhlman’s analysis.
I then sought to examine how elevation change rates had changed between 2010 and 2014 at different sites. I subtracted the elevation change rates as of 2010 from the rates as of 2014, and performed a Pearson correlation test on these rate differences against elevation.

### 3.3 Results

#### 3.3.1 Elevation change rates

Rates of surface elevation change determined from 20 SET sites in Padilla Bay ranged from $-0.80 \text{ cm yr}^{-1}$ at site 1 to $0.10 \text{ cm yr}^{-1}$ at site 14B (Table 5). The mean rate of elevation change was $-0.26 \pm 0.23 \text{ cm yr}^{-1}$. Three of these 20 SETs exhibited positive rates of elevation change.

Once RSLR was accounted for, all SET sites exhibited a negative rate of relative elevation change (Table 5). These rates ranged from $-0.91 \text{ cm yr}^{-1}$ to $-0.01 \text{ cm yr}^{-1}$, with a mean of $-0.37 \pm 0.23 \text{ cm yr}^{-1}$ across all 20 sites.
Table 5. Rates of elevation change at SET sites in Padilla Bay. Relative elevation change rates incorporate the current RSLR rate of 0.113 cm yr\(^{-1}\).

<table>
<thead>
<tr>
<th>Site</th>
<th>Elevation change (cm yr(^{-1}))</th>
<th>R(^2)</th>
<th>Relative elevation change (cm yr(^{-1}))</th>
<th>Years</th>
</tr>
</thead>
<tbody>
<tr>
<td>1'</td>
<td>-0.80</td>
<td>0.75*</td>
<td>-0.91</td>
<td>2002-2005</td>
</tr>
<tr>
<td>1N</td>
<td>-0.54</td>
<td>0.70</td>
<td>-0.66</td>
<td>2010-2014</td>
</tr>
<tr>
<td>2</td>
<td>-0.24</td>
<td>0.90*</td>
<td>-0.35</td>
<td>2002-2014</td>
</tr>
<tr>
<td>3'</td>
<td>-0.15</td>
<td>0.11</td>
<td>-0.26</td>
<td>2002-2005</td>
</tr>
<tr>
<td>4</td>
<td>-0.46</td>
<td>0.96*</td>
<td>-0.58</td>
<td>2002-2014</td>
</tr>
<tr>
<td>5'</td>
<td>-0.01</td>
<td>0.00</td>
<td>-0.12</td>
<td>2002-2004</td>
</tr>
<tr>
<td>6</td>
<td>-0.21</td>
<td>0.89*</td>
<td>-0.32</td>
<td>2002-2014</td>
</tr>
<tr>
<td>7</td>
<td>-0.43</td>
<td>0.94*</td>
<td>-0.54</td>
<td>2002-2014</td>
</tr>
<tr>
<td>8</td>
<td>0.04</td>
<td>0.05</td>
<td>-0.08</td>
<td>2002-2014</td>
</tr>
<tr>
<td>9'</td>
<td>-0.12</td>
<td>0.16</td>
<td>-0.23</td>
<td>2002-2005</td>
</tr>
<tr>
<td>10</td>
<td>-0.29</td>
<td>0.84*</td>
<td>-0.41</td>
<td>2002-2014</td>
</tr>
<tr>
<td>11</td>
<td>-0.26</td>
<td>0.74*</td>
<td>-0.38</td>
<td>2002-2014</td>
</tr>
<tr>
<td>12</td>
<td>-0.28</td>
<td>0.72*</td>
<td>-0.39</td>
<td>2002-2013</td>
</tr>
<tr>
<td>13</td>
<td>-0.43</td>
<td>0.93*</td>
<td>-0.54</td>
<td>2002-2014</td>
</tr>
<tr>
<td>14</td>
<td>-0.14</td>
<td>0.70*</td>
<td>-0.25</td>
<td>2002-2014</td>
</tr>
<tr>
<td>15'</td>
<td>-0.21</td>
<td>na</td>
<td>-0.32</td>
<td>2002-2003</td>
</tr>
<tr>
<td>16'</td>
<td>-0.57</td>
<td>na</td>
<td>-0.68</td>
<td>2002-2003</td>
</tr>
<tr>
<td>5B</td>
<td>-0.12</td>
<td>0.44</td>
<td>-0.24</td>
<td>2005-2014</td>
</tr>
<tr>
<td>12B</td>
<td>0.02</td>
<td>0.01</td>
<td>-0.09</td>
<td>2004-2014</td>
</tr>
<tr>
<td>14B</td>
<td>0.10</td>
<td>0.12</td>
<td>-0.01</td>
<td>2004-2013</td>
</tr>
</tbody>
</table>

Mean \(-0.26 \pm 0.23\) (sd) \(-0.37 \pm 0.23\) (sd)

\(^{1}\) Data were scour corrected by Kuhlman (2011)

\(^{*}\) Significant regression at \(p \leq 0.05\)

Only three SET sites have shown any substantial elevation gain for the majority of the 12-year monitoring period, and only two of the 20 sites are currently higher than their initial elevation (Figure 14). Several sites revealed overall elevation loss even within the first year following SET installation, despite some elevation gain during the growing season.
3.3.2 Scour correction

I observed some evidence of scour during my 2012-2014 site visits at sites 1N, 2, 4, 6, and 14. However, none of these sites met Kuhlman’s quantitative criterion for scour, thus no new sites were eliminated from analysis or subsequent surveying. Site 1N appeared to be scoured only on one side of the SET rod; two of the four arm positions would qualify as scoured on their own, but with all four arm positions averaged, the pin height slope of 0.18 cm pin\(^{-1}\) was just under the qualifying threshold of 0.19 cm pin\(^{-1}\). All other sites with some scour observed in the field had pin height slopes ranging from 0.07 cm pin\(^{-1}\) to 0.14 cm pin\(^{-1}\).
3.3.3 Spatial and temporal trends in elevation change rates

I observed a negative yet non-significant correlation between elevation and the rate of elevation change across 20 SET sites (i.e., landward sites lost more elevation than seaward sites) (Pearson’s $r = -0.34$, $p = 0.15$, $n = 20$) (Figure 15). The three sites demonstrating positive rates of elevation change were located near the middle of the surveyed elevation range, near 0 m MLLW, although they were lower than the median SET elevation.

Figure 15. Correlation between elevation and rate of elevation change in Padilla Bay ($n = 20$).
An analysis of the difference in elevation change rates from the previous 2010 measurements to 2014-inclusive rates revealed that the majority of sites surveyed during this study have seen decreased rates of elevation change during the most recent four years of monitoring, relative to the previous eight years (Figure 16). This loss includes the lower elevation sites, which weakened the relationship between elevation and elevation change. However, rates at the lower elevation sites have not been declining significantly more than the higher elevation sites (Pearson’s $r = 0.41$, $p = 0.17$).

![Figure 16. Difference in elevation change rates between 2010 and 2014 datasets versus elevation ($n = 13$). A positive change indicates an increase in the elevation change rate in recent years, whereas a negative change indicates a decrease in the elevation change rate in recent years.](image)
3.4 **Discussion**

3.4.1 *Elevation change rates*

Although a few SET sites indicated positive rates of surface elevation change, the entirely negative rates of relative elevation change indicate that none of the sites are keeping pace with sea level rise. SET sites in Padilla Bay on average have become over 4.4 cm deeper relative to sea level during the 12-year period since the beginning of SET installation. If the current local rate of relative elevation change continues, the SETs will become increasingly difficult to access and measure, considering that many of them are accessible only a few days each year and even then, many are still inundated.

Surface elevation change is responsible for approximately 70% of the average relative elevation loss in Padilla Bay. The local rate of RSLR is a relatively small contributor to the current rate of relative elevation loss. Padilla Bay has the benefit of positive vertical land motion due to isostatic rebound and plate tectonics (NRC 2012). The apparent lack of acceleration in the regional rate of RSLR since 1980 could also be partially explained by wind stress patterns that suppress sea level rise (Bromirski et al. 2011). Unfortunately this age of local RSLR stabilization may soon end, as recent wind stress patterns indicate an imminent Pacific Decadal Oscillation regime shift to pre-1980 patterns (Bromirski et al. 2011). This shift could cause regional RSLR to match or exceed global ESLR rates, which would cause RSLR to then exceed the rate of surface elevation loss in Padilla Bay. One positive interpretation of the current relative elevation loss dominance by surface elevation loss considers our ability to manipulate sediment supply to affect surface elevation change on a local and short-term scale. We do not have the same ability to manipulate RSLR at the
same small scale, due to the larger- and longer-term climatic and geological processes affecting it.

The mean rate of surface elevation loss reported here may be an underestimate because many of the SETs eliminated from continuing measurements due to scour also showed higher rates of elevation loss. The higher rates of elevation loss due to erosion are in fact responsible for the observed scour. In order to continue acquiring data at these sites, we would need to either reinstall the SETs with more deeply buried cement collars, or use a longer SET arm attachment that allow the pins to settle outside the scour zone.

3.4.2 Scour correction

Although no new SETs were eliminated from analyses or further surveying due to scour, a few sites came close to meeting Kuhlman’s scour criterion. Future measurements from these sites (particularly 1N and 14) should be watched closely as this scour could soon exceed the threshold. Although I chose to continue using Kuhlman’s scour detection criterion, I believe this method is quite conservative, allowing continued use of sites that do not qualify as scoured but showed clear evidence of scour in the field. Elevation change rates obtained from these potentially scoured sites should thus be used with caution, as they likely overestimate elevation loss.

3.4.3 Spatial and temporal trends in elevation change rates

In a previous analysis of elevation change in Padilla Bay, Kuhlman (2011) also observed a negative yet non-significant correlation between elevation of the SET sites and their corresponding elevation change rates (with Pearson’s r = -0.39, p = 0.10). Kuhlman’s correlation was close to significant based on a critical value (α) of 0.05, leading me to expect to see a significant correlation with my additional four-year period of elevation change data,
especially with several drastic elevation losses at the higher elevation sites since the 2010 measurements (Figure 15). Surprisingly, my updated correlation test resulted in a weaker correlation between elevation and elevation change rates. There are no obvious explanations for the accelerating elevation loss occurring at the deeper sites. One of these sites (site 8) had become dominated by macroalgae in 2014, whereas it was dominated by *Z. marina* in previous years. This site lost enough elevation in 2014 to decrease its elevation change rate from 0.10 cm yr\(^{-1}\) to 0.04 cm yr\(^{-1}\) in one year. It is difficult to draw conclusions about the recent changes in elevation change rates, however, when the magnitude of change is actually quite small and there are relatively few sites at these lower elevations.

3.4.4  *Elevation change rates relative to accretion rates*

SET elevation change results demonstrate the importance of combining SET monitoring with short-term accretion monitoring rather than assuming that short-term accretion rates can approximate elevation change. SET measurements incorporate shallow subsidence and erosion in addition to accretion. Without the SET data, we would conclude from the short-term accretion measurements alone (which range from 0.12 cm yr\(^{-1}\) to 2.29 cm yr\(^{-1}\)) that surface elevation change in Padilla Bay is keeping pace with sea level rise. However, the SET data reveal that not a single site within the study area is keeping pace with sea level rise (Figure 17).
Considering the positive rates of accretion I observed (see Chapter 2) in contrast to the negative rates of elevation change from the SETs, Padilla Bay must be experiencing a considerable amount of erosion or shallow subsidence, or a combination of these. It is not possible to conclusively determine which of these two is responsible, without the combination of SETs and screens at the same sites. In a previous Padilla Bay study, Maxwell (2004) measured accretion at the SET sites and observed accretion rates exceeding 2 cm yr\(^{-1}\) despite no significant changes in elevation change. These rates are similar to my rates, despite being measured a decade apart, suggesting that both of our accretion and elevation change rates are valid. Maxwell concluded that shallow subsidence must be responsible for the lack of significant elevation change. The mean short-term accretion rate of 0.97 cm yr\(^{-1}\) determined from my study along with a mean elevation change rate of -0.26 cm yr\(^{-1}\) point to...
a shallow subsidence rate of 1.23 cm yr$^{-1}$. Alternatively, replacing this short-term accretion rate with the mean long-term accretion rate of 0.23 cm yr$^{-1}$ in the calculation indicates a subsidence rate of 0.49 cm yr$^{-1}$. Given that the long-term accretion rate is almost certainly an overestimate due to bioturbation (as discussed in Chapter 2), the subsidence rate is likely less than 0.49 cm yr$^{-1}$. Sediment profiles of bulk density, organic matter, and pore space (described in Chapter 2) show only the slightest changes in these three variables with increasing depth in the soil column, similar to the soil profiles observed by Kairis and Rybczyk (2010). It is possible that the surface sediment layer is quickly decomposed and compacted once buried, which would lead to the observed homogeneous soil profile with the exception of the surface layer.

It is also possible that a substantial amount of subsidence is occurring below the 40-cm layer sampled by the cores, due to continued primary consolidation or secondary compression. This explanation would be corroborated by the absence of any evidence of erosion on the wire mesh screens. SET data represent cumulative accretion and subsidence over an approximately 5-m depth range (to the bottom of the SET rod), which spans an additional 4.5 m of potentially compacting sediment below my 40-cm core data. The 40-cm cores themselves are comprised mostly of sandy sediment, which is known to be less compressible than other sediment types except when under high stress or with softer sands (Lambe and Whitman 1969). However, it is quite possible that finer alluvial sediments lie below the sandy surface, deposited in Padilla Bay before the bay was disconnected from Skagit River distributaries. Without deeper cores, I cannot conclude on the likelihood of this consolidation explanation relative to the erosion explanation.
Bos et al. (2007) observed more spring/summer accretion but also more winter erosion in intertidal annual eelgrass beds compared to subtidal perennial eelgrass beds. Although I did not observe significantly more erosion in the shallower areas dominated by *Z. japonica*, which has a more annual habit than *Z. marina* (Harrison 1982), if additional years of field measurements tend toward this pattern, this could explain the SET trend of greater elevation loss at shallower sites.
4 SURFACE ELEVATION MODEL

4.1 Introduction

Although rates of sediment accretion and elevation change are often used to assess the sustainability of coastal wetlands in the face of sea level rise, these rates are not reliable indicators of future wetland conditions. Changes in relative elevation can feed back to alter many of the processes involved in elevation change. Models can incorporate these non-linear feedbacks, and more accurately predict wetland sustainability for a range of relative sea level rise (RSLR) scenarios. As discussed in Chapter 1, many models of relative elevation change have been developed for this very purpose (Rybczyk and Callaway 2009, and references therein). They have been applied to every coastal wetland type, from mangroves to marshes to seagrass meadows. However, seagrass meadow elevation change has been relatively under-modeled. In fact, to my knowledge, Padilla Bay is the only seagrass meadow that has been modeled for long-term relative elevation change (Kairis and Rybczyk 2010).

The hybrid model presented here is a mechanistic and non-linear, zero-dimensional model based on two earlier elevation change models: the Marsh Equilibrium Model (MEM) from Morris et al. (2012), and the Relative Elevation Model (REM) from Kairis and Rybczyk (2010). The MEM is an analytical model originally developed by Morris et al. (2002) for a South Carolina Spartina salt marsh. It simulates the effects of aboveground plant biomass on accretion, which most other models do not consider. The MEM was later modified by Morris (2006) to include multiple plant species, then again by Morris et al. (2012) to include the growth and decomposition of belowground plant biomass. The REM is a mechanistic, numerical model developed by Rybczyk et al. (1998) for a Louisiana hardwood swamp. This model includes the simulation of belowground processes such as decomposition and
compaction, tracking these processes within individual sediment cohorts with depth and time. The REM is an extension of earlier cohort models by Morris and Bowden (1986), Chmura et al. (1992), and Callaway et al. (1996). It has since been applied to Italy’s Venice Lagoon (Day et al. 1999), the Bay Islands in Honduras (Cahoon et al. 2003), mangrove forests in Micronesia (Fullagar 2006), and most recently converted to a spatially explicit version for Padilla Bay’s seagrass meadows (Kairis and Rybczyk 2010).

The hybrid model was developed using select components from both the MEM and REM. I included the sediment deposition algorithm from the MEM, which replaced the mineral input algorithm from the REM. The MEM (and hence the hybrid model) simulates sediment deposition based on aboveground plant characteristics, relative elevation, tidal range, and suspended sediment concentration (SSC), whereas the REM simulates sediment deposition based on relative elevation alone. The MEM uses a sediment input in the form of total suspended solids (TSS), whereas the REM requires a sediment deposition rate, which is not as widely available. Linking the MEM sediment deposition algorithm to the REM sediment dynamics sub-model then allowed for the elimination of the MEM simulation of belowground plant growth and decomposition, because these processes were already incorporated into the REM within the primary productivity and sediment dynamics sub-models. The REM relative elevation sub-model translates the simulated soil column height into a relative elevation, which then feeds back to the MEM sediment deposition sub-model. The addition of the MEM to the REM thus adds another layer to the feedbacks governing relative elevation. Rather than increasing sediment deposition rates linearly with a decrease in relative elevation, the modified MEM component adds an increase in sediment deposition with eelgrass stem density, which varies non-linearly with elevation.
Field measurements obtained from Padilla Bay, as part of this research as well as other studies, were used for model initialization, calibration, and validation. Wire mesh screens were used to estimate short-term accretion rates to calibrate the MEM sub-model. Measurements of plant productivity, above- and belowground biomass, and stem density as a function of relative elevation were used to initialize the MEM and REM primary productivity sub-models. Sediment core analyses and decomposition studies were used to calibrate the REM sediment dynamics sub-model. Finally, measurements of elevation change were used for model validation.

There are several advantages to applying this hybrid model to Padilla Bay over the Spatial REM (SREM) used by Kairis and Rybczyk (2010). The hybrid model includes the effect of both vegetation and relative elevation on sediment deposition, instead of just elevation. Vegetation was shown to be the most important determinant of accretion rates in Padilla Bay, as discussed in Chapter 2. The hybrid model also allows for the inclusion of multiple plant species. Whereas Kairis and Rybczyk (2010) included just Z. marina in the SREM, Padilla Bay’s eelgrass meadow includes two eelgrass species that may affect elevation change rates differently. Chapter 2 illustrated these two species’ differential effects on accretion field measurements. Additionally, the hybrid model includes a SSC forcing function as part of the sediment deposition algorithm that is not included in the REM. This addition allows the user to input a suspended sediment concentration (typically a TSS measurement), which tends to be more widely available and faster to obtain than an annual deposition rate. The SSC can also be manipulated in model exercises to determine the necessary conditions for optimal wetland sustainability, and it can point directly to management targets for sediment supply. Finally, the hybrid model provides a larger
calibration tool set than either the MEM or REM as used separately. The accretion rate generated by the MEM sub-model provides a calibration point as do the soil profiles generated by the REM sediment dynamics sub-model.

One disadvantage of using the hybrid model is that it is not spatially explicit as is the SREM. The hybrid model simulates elevation change at one point in the landscape, although it can be run separately for multiple points. However, this zero-dimensional nature allows the hybrid model to be more versatile for use by other modelers on other study systems because it is less computer-intensive, with fewer extensive spatial data set requirements.

Although the hybrid model presented here was initialized, calibrated, and validated for Padilla Bay, it was developed with the intention to maximize its applicability to other coastal wetlands. The model was designed to assess the long-term sustainability of any intertidal wetland under a variety of suspended sediment and RSLR scenarios, with the provision of the site-specific parameters described in this chapter.

The objectives of this modeling study were to: 1) develop this hybrid model by linking two existing models, modifying in accordance with site-specific field data to apply to Padilla Bay’s eelgrass habitat, 2) simulate long-term relative elevation change under a variety of RSLR scenarios, and 3) determine the suspended sediment concentration necessary to maintain a relative elevation equilibrium under each RSLR scenario.

4.2 Methods

4.2.1 Linking two models

All three REM sub-models (primary productivity, sediment dynamics, and relative elevation) used in the SREM application to Padilla Bay by Kairis and Rybczyk (2010) were retained in the hybrid model. The structure of the primary productivity and relative elevation
sub-models were largely unmodified. The sediment dynamics sub-model was modified primarily with the omission of the mineral input algorithm. The mineral deposition algorithm was instead included in the MEM component, then used as an input to the sediment dynamics sub-model (Figure 18).

Figure 18. Conceptual diagram of the hybrid model including the modified MEM and REM components. The REM diagram component is modified from Rybczyk et al. (1998).

The MEM component was more extensively modified to allow it to link to the REM component and apply to Padilla Bay’s intertidal eelgrass habitat. The contribution of belowground plant biomass to elevation change (represented in the MEM by belowground plant production, refractory fraction, and root turnover rate) was eliminated because this is simulated in more detail in the REM sediment dynamics sub-model. And instead of converting the mass-based accretion rate to a vertical accretion rate within the MEM
component using a simulated bulk density, I eliminated this conversion and used the mass-based accretion rate (g cm\(^{-2}\) yr\(^{-1}\)) as input to the sediment dynamics REM sub-model as described above.

The hybrid model was programmed using STELLA iconographic modeling software (High Performance Systems, Inc. 2002), similar to the original REM. STELLA provides a graphical interface for building a qualitative model. After assigning values to the inputs and functions to the relationships, the software determines the differential equations used for simulations at each timestep using an Euler numerical method. I ran the hybrid model with a timestep of one week and dt of 0.25 (i.e., parameters are set with a weekly time unit, but calculations are made four times per week).

4.2.2 Marsh Equilibrium Model component

The original MEM component simulates changes in plant biomass as a function of water depth, and surface elevation change is then modeled as the sum of sedimentation trapped by plants, sedimentation by settling out of the water column, and refractory belowground plant production:

\[
\frac{dS}{dt} = \left( q + k_s B_s \right) m D^2 + k_r B_r
\]

where:

- \( S \) = sediment deposition (g cm\(^{-2}\))
- \( q \) = settling rate of suspended sediments (yr\(^{-1}\))
- \( k_s \) = sediment trapping coefficient specific to plant species (m\(^2\) g\(^{-1}\) yr\(^{-1}\))
- \( B_s \) = above-ground end-of-season (EOS) plant biomass (g m\(^{-2}\))
- \( m \) = suspended sediment concentration (g cm\(^{-3}\))
- \( D \) = depth of the sediment surface below MHW (cm)
- \( T \) = mean tidal range from MHW-MLW (cm)
- \( k_r \) = refractory fraction of below-ground biomass (g g\(^{-1}\))
- \( B_r \) = below-ground production (g m\(^{-2}\) yr\(^{-1}\))
In the original MEM, sedimentation from settling is modeled as a function of suspended sediment concentration, water depth, tidal range, and a settling coefficient. Sedimentation from plant trapping is modeled as a function of the same variables as settling, but with the addition of aboveground biomass and the replacement of the settling coefficient with a trapping coefficient.

I modified the MEM to replace the aboveground plant biomass variable with stem density, based on analyses of relationships between field measurements of sediment accretion, biomass, stem density, and relative elevation (see Chapter 2). I removed the belowground contribution of plant production \((k_r B_r)\) because plant production is accounted for separately in the primary productivity sub-model. I modified the tidal range \((T)\) parameter to represent the range from MLLW to MHHW instead of MLW to MHW, which encompasses a larger portion of Padilla Bay’s eelgrass elevation range. I added an elevation threshold at MLLW to prevent the sedimentation rates from increasing below MLLW due to depth alone, while allowing stem density and suspended sediment concentrations to affect sedimentation rates at all elevations. Finally, I replaced the water depth \((D)\) input with relative elevation \((h_r)\) and adjusted the equation accordingly, to maintain consistency with the REM. With the above modifications and a few unit conversions, the modified MEM component of the hybrid model is as follows:

\[
\text{If } h_r > 0, \quad \frac{dS}{dt} = \frac{(q + k_s P)m(T - h_r)^2}{T}, \quad \text{else } \quad \frac{dS}{dt} = (q + kP)mT
\]  

(4)

where:

- \(S\) = sediment deposition (g cm\(^{-2}\))
- \(q\) = settling rate of suspended sediments (week\(^{-1}\))
- \(k\) = sediment trapping coefficient (cm\(^2\) shoot\(^{-1}\) week\(^{-1}\))
\[ P = \text{stem density (\# shoots cm}^{-2}\text{)}\]
\[ m = \text{suspended sediment concentration (g cm}^{-3}\text{)}\]
\[ h_r = \text{elevation of the sediment surface above MLLW (cm)}\]
\[ T = \text{maximum tidal range from MLLW-MHHW (cm)}\]

I developed elevation switching functions for stem density (\( P \)) separately for both eelgrass species based on annual mean stem density field data at the accretion monitoring sites (PBNERR unpublished data) (Table 6).

Table 6. Elevation-dependent stem density (\( P \)) equations determined from field-based annual mean stem density values.

<table>
<thead>
<tr>
<th>Elevation range (cm above MLLW)</th>
<th>( Z. \text{marina} ) stem density (# shoots m(^{-2}))</th>
<th>( Z. \text{japonica} ) stem density (# shoots m(^{-2}))</th>
</tr>
</thead>
<tbody>
<tr>
<td>elevation &gt; 80.8</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>80.8 ≥ elevation &gt; 49.0</td>
<td>0</td>
<td>-110.22 * elevation + 8915.50</td>
</tr>
<tr>
<td>49.0 ≥ elevation &gt; 29.5</td>
<td>-9.82 * elevation + 485.14</td>
<td>40.46 * elevation + 1531.50</td>
</tr>
<tr>
<td>29.5 ≥ elevation &gt; 9.1</td>
<td>-23.28 * elevation + 882.66</td>
<td>52.49 * elevation + 1176.70</td>
</tr>
<tr>
<td>9.1 ≥ elevation &gt; -1.0</td>
<td>20.10 * elevation + 489.59</td>
<td>140.92 * elevation + 375.49</td>
</tr>
<tr>
<td>-1.0 ≥ elevation &gt; -22.6</td>
<td>0.41 * elevation + 469.75</td>
<td>10.06 * elevation + 243.62</td>
</tr>
<tr>
<td>-22.6 ≥ elevation &gt; -66.5</td>
<td>5.97 * elevation + 595.10</td>
<td>-2.75 * elevation + 45.56</td>
</tr>
<tr>
<td>-66.5 ≥ elevation &gt; -300</td>
<td>0.850 * elevation + 255.01</td>
<td>0</td>
</tr>
<tr>
<td>-300 ≥ elevation</td>
<td>0</td>
<td>0</td>
</tr>
</tbody>
</table>

Annual mean stem density was measured down to -66.5 cm MLLW. I added a point to the lowest end of the \( Z. \text{marina} \) curve to allow it to decline to zero at -300 cm MLLW, based on aerial photo observations of \( Z. \text{marina} \) extending down to this lower elevation in Padilla Bay (Bulthuis 1995). \( Z. \text{japonica} \) stem density was assumed to be zero below its lowest measured elevation at -66.5 cm MLLW (Figure 19).
Suspended sediment concentration

I determined the suspended sediment concentration \( (m) \) forcing function value from bimonthly measurements of TSS at Padilla Bay National Estuarine Research Reserve (PBNERR) at four sites during the period 2009-2013. Although PBNERR began sampling TSS in 2002, I chose to include only the most recent five year period in my average following the suggestion of Dr. Douglas Bulthuis (D. Bulthuis, pers. comm., 2013). Water samples were analyzed in the PBNERR laboratory beginning in 2009, producing less variable results within and among years.

PBNERR collected water samples bimonthly from 2009-2011 and monthly beginning September 2011 from up to six sites within the bay. Two to three samples were collected from each site during each sample event. Replicate samples were collected from 0.5 m above the sediment surface at Bayview and Ploeg Channel sites. At the Gong site, surface samples were taken at a depth of 0.5 m, and deep samples were taken approximately 20 m above the
bottom. Beginning in 2010, the Joe Leary Slough site was abandoned. The other sample sites included Bayview Channel, Ploeg Channel, Gong Surface, Gong Deep, and Joe Leary Estuary. I excluded the Joe Leary Slough and Joe Leary Estuary sites from my analysis due to their proximity to shore. The locations of the remaining four sites were determined to be more representative of Padilla Bay eelgrass habitat. The water samples were processed by PBNERR following the methods of Greenberg et al. (1995).

To obtain a mean annual TSS value, I first averaged TSS values within each sample site and sample event to account for varying numbers of samples at different sites. I then averaged across sites to produce the mean per sample event. The annual mean was determined from these sample event values (n = 24 for 2009-2011; n = 12 for 2012-2013). Finally, the average of these five years produced the value of 3.93 ± 0.69 mg L\(^{-1}\) (± 1 SD) used in the model.

*Sediment trapping and settling coefficients*

The sediment settling and trapping coefficients (q and k, respectively) were first determined by fitting the MEM sub-model to the observed accretion rates (Eq. 2). The simulated height of the surface sediment cohort was used to calibrate the settling and trapping coefficients. This surface cohort simulates the accumulation of both inorganic and organic material for one year at a time, which best represents the accretion measured with wire screens for a one-year period. Coefficient values for q and k were initialized at 1 week\(^{-1}\) and 117 cm\(^2\) shoot\(^{-1}\) week\(^{-1}\), respectively.
4.2.3 **Relative Elevation Model component**

*Primary productivity sub-model*

Net primary productivity (NPP) is used in this model to estimate leaf and root litter rates, which are the predominant sources of organic material to the soil surface and the soil column, respectively. NPP was modified in this model to be a function of aboveground biomass, instead of a function of relative elevation as Kairis and Rybczyk (2010) used. I chose to make this modification because the NPP-elevation relationship utilizing field data from Thom (1990) did not pair well with the stem density and biomass-elevation relationships using PBNERR data. The field measurements from these two studies were taken from different locations, leading to different relationships, resulting in certain elevations with positive eelgrass density and biomass but no NPP when NPP was initially programmed as a function of elevation. Gwozdz (2006) also noted a discrepancy in elevation-productivity relationships between studies done by Thom (1990) and Gwozdz (2004) in Padilla Bay, suggesting an incongruity in elevation measurements. I avoided using previous elevation data altogether by deriving a linear relationship between NPP and aboveground biomass from Thom’s (1990) field data, with separate functions for each eelgrass species (Figure 20), similar to the approach taken by Morris (2007). These relationships were significant with $p < 0.001$ for both species.
It would be preferable to use field measurements of NPP from the same sites as stem density and biomass to allow for a NPP-elevation function, but these measurements would have required an additional year. For this reason, aboveground biomass was included as a stepwise elevation function (Table 7) to estimate NPP in the REM primary productivity sub-model.
Table 7. Elevation-dependent biomass equations determined from field-based annual mean aboveground biomass values.

<table>
<thead>
<tr>
<th>Elevation range (cm above MLLW)</th>
<th>Z. marina biomass (g m(^{-2}))</th>
<th>Z. japonica biomass (g m(^{-2}))</th>
</tr>
</thead>
<tbody>
<tr>
<td>elevation &gt; 80.8</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>80.8 ≥ elevation &gt; 49.0</td>
<td>0</td>
<td>-0.84 * elevation + 68.02</td>
</tr>
<tr>
<td>49.0 ≥ elevation &gt; 29.5</td>
<td>-0.24 * elevation + 11.59</td>
<td>-0.35 * elevation + 43.74</td>
</tr>
<tr>
<td>29.5 ≥ elevation &gt; 9.1</td>
<td>-0.60 * elevation + 22.41</td>
<td>0.69 * elevation + 13.13</td>
</tr>
<tr>
<td>9.1 ≥ elevation &gt; -1.0</td>
<td>-0.70 * elevation + 23.29</td>
<td>1.66 * elevation + 4.34</td>
</tr>
<tr>
<td>-1.0 ≥ elevation &gt; -22.6</td>
<td>-0.96 * elevation + 23.03</td>
<td>0.06 * elevation + 2.73</td>
</tr>
<tr>
<td>-22.6 ≥ elevation &gt; -66.5</td>
<td>0.00 * elevation + 44.68</td>
<td>-0.01 * elevation + 1.20</td>
</tr>
<tr>
<td>-66.5 ≥ elevation &gt; -300</td>
<td>-0.40 * elevation + 18.26</td>
<td>0</td>
</tr>
<tr>
<td>-300 ≥ elevation</td>
<td>0</td>
<td>0</td>
</tr>
</tbody>
</table>

Although *Z. marina* has been observed extending to -300 cm MLLW in Padilla Bay (Bulthuis 1995), the PBNERR monthly biomass monitoring survey did not extend below -66.5 cm MLLW, with the exception of December 2011 and July 2012 monitoring events which extended to -262.2 cm MLLW. Unlike *Z. marina* stem density, which appears to peak at 9.1 cm MLLW then decrease toward zero as the species approaches its lower limit, aboveground *Z. marina* biomass appears to continue increasing because the *Z. marina* leaves grow longer at greater depths. I estimated an annual mean aboveground biomass value at -300 cm MLLW, based on the relationship between summer biomass measurements at -66.5 cm MLLW in 2011 and -262.2 cm MLLW in 2012. Biomass increased from the -66.5 cm elevation to a maximum approximately 2.75 times higher at the lowest surveyed site at -262.2 cm in the summer of 2012. Biomass was assumed to increase linearly with a decrease in elevation. Applying this relationship to the annual mean biomass of 44.82 g m\(^{-2}\) at -66.5 cm MLLW, I estimated the annual mean biomass at -262.2 cm MLLW to be 123 g m\(^{-2}\). This value was used to determine the elevation-dependent biomass function for the -66.5 to -300
cm elevation range (Table 7), which translates to an estimated maximum biomass of 138 g m$^{-2}$ at -300 cm MLLW (Figure 21).

Figure 21. Mean annual aboveground biomass with additional estimated value at -300 cm MLLW.

Belowground productivity was calculated as a constant fraction of aboveground productivity. The root-to-shoot productivity ratio ($r_m$) was assumed to be equal to the annual mean root-to-shoot biomass ratio of 1.34 ± 0.34 (± SD, n = 6). This root-to-shoot biomass ratio was calculated as the average of monthly root-to-shoot biomass ratios across the six vegetated accretion monitoring sites along the PBNERR transect. This value is within the range of 1:1 (in summer) to 2:1 (in winter) reported by Phillips (1972) for eelgrass in Puget Sound, although a much wider range of values has been reported from other locations (Phillips 1984). The seventh accretion monitoring site (1G) lacked an associated root-to-shoot ratio because it was unvegetated. Because differentiation between $Z. marina$ and $Z. japonica$ roots was not possible in the lab, one ratio is assumed to apply to both eelgrass
species. This ratio is higher than the ratio of 0.503 used by Kairis and Rybczyk (2010) which was based on literature values from other locations.

Since the MEM was developed, others have determined the root-to-shoot ratio to be elevation-dependent (Mudd et al. 2009). In Padilla Bay, the annual mean root-to-shoot biomass ratio shows a slight decline with decreasing relative elevation, but this relationship is not significant ($R^2 = 0.11$, $p = 0.31$, df = 9) (Figure 22). For this reason, I chose to use a constant root-to-shoot ratio that does not vary with elevation.

![Graph showing root:shoot ratio vs. relative elevation](image)

Figure 22. Root:shoot biomass ratio of both eelgrass species at each PBNERR biomonitorting site for which annual mean biomass ratios could be obtained from one year of monthly samples ($n = 11$).
Sediment dynamics sub-model

The sediment dynamics sub-model simulates longer-term processes within the sediment column such as decomposition and compaction, using mineral and leaf litter inputs at the sediment surface and root inputs within the column. Discrete sediment packets are transferred from the surface cohort to the bottom through 18 cohort layers. Each cohort is programmed to have unique decomposition rates and transfer rates based on its relative depth within the sediment column, while the actual height of each cohort changes over time depending on its content. Stock variables of mineral matter, labile organic matter, and refractory organic matter are maintained for each cohort.

I retained the cohort transfer rates from Kairis and Rybczyk (2010). From the surface to the bottom cohort, sediment packets were transferred to the subsequent cohort at a time sequence of 1, 5, 5, 5, 10, 10, 10, 10, 10, 10, 20, 20, and 20+ years. I ran the model for 400 simulated years to build a minimum 40-cm sediment column to then use as the starting state for calibration and simulation runs. Simulated bulk density, organic and mineral content, and porosity of each cohort are used to calibrate this sub-model. The algorithms for mineral input, root distribution, and sediment compaction are described below.

Mineral input

As stated above, the original REM mineral input algorithm was removed to allow mineral inputs to be simulated by the MEM component as a function of both relative elevation and stem density (Eq. 4). Regardless of plant stem density, mineral inputs increase as relative elevation decreases due to a longer duration of tidal inundation. Below MLLW however, inundation frequency no longer increases and mineral inputs are programmed to be unaffected by relative elevation beyond MLLW. The mineral input \( (S) \) is delivered at each
timestep to the surface sediment cohort (cohort 1) in the REM sediment dynamics sub-model where it is then transferred to subsequent cohorts at a rate determined by the equation developed by Day et al. (1999):

If \( n = 1 \), \( \frac{dM(n)}{dt} = S - (Z_m(n) \cdot M(n)) \), else \( \frac{dM(n)}{dt} = (Z_m(n-1) \cdot M(n-1)) - (Z_m(n) \cdot M(n)) \)  

(5)

where:

\( M(n) \) = mineral matter in cohort \( n \) (g cm\(^{-2}\))
\( S \) = mineral input at the sediment surface (g cm\(^{-2}\) week\(^{-1}\))
\( Z_m(n-1) \) = transfer rate of mineral matter from overlying cohort (g cm\(^{-2}\) week\(^{-1}\))
\( M(n-1) \) = mineral matter in overlying cohort (g cm\(^{-2}\))
\( Z_m(n) \) = transfer rate of mineral matter to underlying cohort (g cm\(^{-2}\) week\(^{-1}\))

Root distribution

The root distribution equation uses root productivity simulated in the primary productivity sub-model and distributes this root production as an exponential decline with depth in the soil column. The root distribution algorithm is based on an equation originally developed by Morris and Bowden (1986) to simulate exponential decline in root biomass at depth \( h \):

\[ R(h) = se^{(-rh)} \]  

(6)

where:

\( R(h) \) = root biomass at depth \( h \) (g DW cm\(^{-2}\))
\( s \) = root biomass at the sediment surface (g DW cm\(^{-2}\))
\( r \) = root distribution constant (cm\(^{-1}\))
\( h \) = depth within the sediment column (cm)
Kairis and Rybczyk (2010) determined a root distribution constant of 0.307 ± 0.080 cm$^{-1}$ by fitting this curve to data reported by Vichkovitten and Holmer (2005) from a study of belowground Z. marina biomass within 2 cm sections of cores down to 10 cm depth in sandy sediments. Lacking any more site-specific field data, I used the same root distribution constant value. Although I changed units of biomass from grams carbon to grams dry weight, the value of this constant remained the same.

Root biomass within each cohort was calculated using this constant value with the following equation from the original REM model (Rybczyk et al. 1998):

$$r_i(n) = \frac{s(e^{-rh_b} - e^{-rh_a})}{-r}$$

where:

- $r_i(n)$ = root input to sediment cohort $n$ (g DW cm$^{-2}$)
- $s$ = root biomass at the sediment surface (g DW cm$^{-2}$)
- $r$ = root distribution constant (cm$^{-1}$)
- $h_b$ = depth to the bottom of cohort $n$ (cm)
- $h_a$ = depth to the top of cohort $n$ (cm)

**Organic matter decomposition**

Leaf and root litter from the primary productivity sub-model were each divided into labile and refractory components. Leaf litter was exported to the surface cohort of the sediment dynamics sub-model. Leaf labile and refractory fractions were determined by Kairis and Rybczyk (2010) based on results from a Z. marina decomposition study at Shannon Point Marine Center (Rybczyk 2005 unpublished data) (Figure 23).
Figure 23. *Z. marina* leaf litter decomposition study results, used by Kairis and Rybczyk (2010) to determine labile and refractory fractions of leaf litter, and decomposition rates. Figure from Kairis (2008).

Results from this study were also used to determine four decomposition rates: surface labile and refractory matter, and deep labile and refractory matter. Kairis and Rybczyk (2010) used the following equation describing exponential decay with depth in the soil column, to determine the values of these constants:

\[
  k_1 = \frac{\ln \left( \frac{N_t}{N_0} \right)}{t}
\]

(8)

where:

- \( k_1 \) = decomposition constant (day\(^{-1}\))
- \( N_t \) = mass remaining at time \( t \) (g)
- \( N_0 \) = mass at start (g)
- \( t \) = time since start (days)

Root litter was also divided into labile and refractory components by Kairis and Rybczyk (2010) using results from a *Z. marina* root decomposition study in North Carolina (Kenworthy and Thayer 1984).
The model uses the following equation from Day et al. (1999) with each timestep to simulate labile organic matter decomposition and transfer through the sediment column:

\[
\begin{align*}
\text{If } n &= 1, \quad \frac{dQ(n)}{dt} = (a_1 \cdot f_2) + (r_1(n) \cdot f_3) - (Q(n) \cdot k_x) - (Z_L(n) \cdot Q(n)), \\
\text{else } \quad \frac{dQ(n)}{dt} &= (r_1(n) \cdot f_3) + (Z_L(n-1) \cdot Q(n-1)) - (Q(n) \cdot k_x) - (Z_L(n) \cdot Q(n))
\end{align*}
\]

(9)

where:

- \(Q(n)\) = labile organic matter in cohort \(n\) (g DW cm\(^{-2}\))
- \(a_1\) = leaf litter inputs to surface cohort (g DW cm\(^{-2}\) week\(^{-1}\))
- \(f_2\) = labile fraction of leaf litter (unitless)
- \(r_1(n)\) = root litter inputs to cohort \(n\) (g DW cm\(^{-2}\) week\(^{-1}\))
- \(f_3\) = labile fraction of root litter (unitless)
- \(Z_L(n-1)\) = transfer rate of labile organic matter from overlying cohort (g DW cm\(^{-2}\) week\(^{-1}\))
- \(Q(n-1)\) = labile organic matter in overlying cohort (g DW cm\(^{-2}\))
- \(k_x\) = \(k_2\) or \(k_3\) depending on cohort position: decomposition rate of labile organic matter (week\(^{-1}\)); see \(k_2\) and \(k_3\) in Table 8
- \(Z_L(n)\) = transfer rate of labile organic matter to underlying cohort (g DW cm\(^{-2}\) week\(^{-1}\))

The surface decomposition constant for labile organic matter \((k_2)\) was used in Eq. 9 for cohort 1, after which the deep decomposition constants \((k_3)\) was used for cohorts 2-18.

The decomposition of refractory organic matter was modeled similarly, with the following equation (Day et al. 1999):

\[
\begin{align*}
\text{If } n &= 1, \quad \frac{dB(n)}{dt} = (a_1 \cdot (1 - f_2)) + (r_1(n) \cdot (1 - f_3)) - (B(n) \cdot k_i) - (Z_R(n) \cdot B(n)), \\
\text{else } \quad \frac{dB(n)}{dt} &= (r_1(n) \cdot (1 - f_3)) + (Z_R(n-1) \cdot B(n-1)) - (B(n) \cdot k_i) - (Z_R(n) \cdot B(n))
\end{align*}
\]

(10)

where:

- \(B(n)\) = refractory organic matter in cohort \(n\) (g DW cm\(^{-2}\))
\[ Z_R(n-1) \] = transfer rate of refractory organic matter from overlying cohort (g DW cm\(^{-2}\) week\(^{-1}\))

\[ B(n-1) \] = refractory organic matter in overlying cohort (g DW cm\(^{-2}\))

\[ k_i \] = \( k_4 \) or \( k_5 \) depending on cohort position: decomposition rate of refractory organic matter (week\(^{-1}\)); see \( k_4 \) and \( k_5 \) in Table 8

\[ Z_R(n) \] = transfer rate of refractory organic matter to underlying cohort (g DW cm\(^{-2}\) week\(^{-1}\))

The surface decomposition constant for refractory organic matter (\( k_4 \)) was used in Eq. 10 for cohorts 1-10, while the deep decomposition constant (\( k_5 \)) was used for cohorts 11-18.

*Sediment compaction*

Compaction of the sediment column results from both decomposition and reduction of pore space (primary consolidation). The reduction of pore space was modeled as an asymptotic decrease with depth. Kairis and Rybczyk (2010) simulated pore space within each cohort as a function of the mass of material above it:

\[
p_s(n) = p_m + (p_x - p_m) \cdot C_{\text{func}}(n)
\]

where:

\[ p_s(n) \] = pore space in cohort \( n \) (%)

\[ p_m \] = minimum pore space for the entire sediment column (%)

\[ p_x \] = maximum pore space for the entire sediment column (%)

\[
C_{\text{func}}(n) = 1 - \frac{g(n)}{p_k + g(n)}
\]

where:

\[ C_{\text{func}}(n) \] = Michaelis-Menten reduction in pore space (unitless)

\[ g(n) \] = total mass of sediment overlying cohort \( n \) (g cm\(^{-2}\))

\[ p_k \] = half saturation compaction constant (g cm\(^{-2}\))
I determined the minimum pore space \( (p_m) \), maximum pore space \( (p_x) \), and compaction constant \( (p_k) \) from a representative pore space profile from 5 cores collected along the permanent eelgrass monitoring transect. Each 2-cm core depth section was averaged across the 5 cores. To determine the compaction constant for this representative pore space profile, I used a non-linear regression within the R programming environment (R Core Team 2014) to fit an asymptotic curve used by Kairis and Rybczyk (2010) to a plot of pore space as a function of the mass overlying each core section:

\[
poreSpace = A + B \left( \frac{1}{tMass + C} \right)
\]

where:

- \( poreSpace \) = pore space in core section (\%)
- \( tMass \) = total mass above core section (g cm\(^{-2}\))
- \( A, B, C \) = fitting parameters

The fitting parameters were initialized at 0.001. The half saturation compaction constant \( (p_k) \) was determined as the total sediment mass above the depth where pore space was halfway between the minimum and maximum values on the fitted curve.

The half saturation compaction constant \( (p_k) \) was determined to be 5.82 g cm\(^{-2}\) based on the non-linear curve fitted to the representative downcore distribution of pore space with depth (Figure 24). The five sites along the transect with porosity profiles demonstrating exponential decay were included in this mean porosity profile. Sites 1G and 2J were excluded from this mean due to atypical surface layers of dense sand. The curve fitting parameters A, B, and C were estimated at 39.72, 74.27, and 4.64, respectively, with a convergence tolerance of \( 9.655 \times 10^{-6} \). All three parameters were significantly different from
zero at p < 0.01. The minimum and maximum pore space values ($p_m$ and $p_x$) from this mean porosity profile were 40.50 % and 55.13 %, respectively.

Figure 24. Downcore distribution of pore space (± SE) used to determine the half-saturation compaction constant ($p_k$). Sections of the core were averaged across the five sites demonstrating exponential decay of pore space with depth.

**Relative elevation sub-model**

The relative elevation sub-model determines relative surface elevation based on the total height of the sediment column exported from the sediment dynamics sub-model, in addition to changes in water depth using local rates of RSLR and erosion as external forcing functions. The total height of the sediment column incorporates both accretion and shallow subsidence. Accretion consists of mineral matter deposition simulated by the MEM.
component, and leaf and root litter simulated by the primary productivity sub-model. Shallow subsidence consists of organic matter decomposition and compaction, both simulated by the sediment dynamics sub-model. RSLR consists of both ESLR and vertical land movement (deep subsidence). Using an equation modified from Kairis and Rybczyk (2010), replacing ESLR and deep subsidence with just RSLR, and adding an erosion forcing function, relative elevation is modeled as:

$$ h_r = h_t - R - N + I_r $$

(14)

where:

- $h_r$ = sediment surface elevation relative to MLLW (cm)
- $h_t$ = total height of sediment column (cm)
- $R$ = cumulative relative sea level rise (cm)
- $N$ = cumulative erosion (cm)
- $I_r$ = initial elevation relative to MLLW (cm)

Changes to local mean sea level can be estimated with tide gauge records. Tide gauges measure changes to sea level relative to a fixed point on land, incorporating both ESLR and vertical land movement. I used a RSLR rate of $0.113 \pm 0.033$ cm yr$^{-1}$ from the closest tide gauge at Friday Harbor, WA (Station 9449880) (Figure 25).
I added the erosion term as a forcing function based on field observations of lost marker horizons, scour at several SET sites, and deep mixed layers in sediment cores used for radioisotope analysis (Carpenter et al. 1985, Ball 2004, Maxwell 2004, Gwozdz 2006) (see discussion in Chapter 2). Erosion was set to zero for calibration and validation runs and all RSLR scenarios. These same RSLR scenarios were later run with erosion as well.

4.2.4 Hybrid model initialization and calibration

I built the sediment column by running the model for 400 simulated years with a timestep of 1 week. The resulting stocks of mineral matter, labile organic matter, and refractory organic matter in each sediment cohort were then used as starting values for calibration runs. The initial relative elevation was set to 9.1 cm MLLW as the median of the seven accretion monitoring sites. I used the local relative sea level rise rate of 0.113 cm yr\(^{-1}\) (NOAA Tides and Currents) to represent the rate for the last century as well as the current rate. This rate was determined from Friday Harbor tide gauge measurements from 1934-
2006, and the rate has not changed significantly in more recent years (Figure 25). The rate of 0.113 cm yr$^{-1}$ was applied to pre-simulation, calibration, and validation runs. I used the maximum tidal range (MLLW-MHHW) of 257 cm for the tidal range parameter value (Shull, pers. comm., 2013; see Figure 2 in Chapter 2).

I calibrated the model with a 100-year simulation run, adjusting the initial relative elevation to 9.1 cm MLLW again to account for the starting height of the sediment column. The sediment trapping and settling coefficients in the MEM component were adjusted first until the simulated height of the year-old surface cohort best approximated field-based accretion rates. The sediment dynamics sub-model was then calibrated by comparing simulated bulk density, organic matter, and pore space profiles to the mean measured profiles for these same parameters. Decomposition, compaction, and root distribution constants were adjusted until the simulated profiles matched mean measured profiles within one standard deviation when possible. Initialization parameters are reported in Table 8.
Table 8. Initialization parameters for the hybrid relative elevation model after calibration.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Description</th>
<th>Value</th>
<th>Parameter Type</th>
</tr>
</thead>
<tbody>
<tr>
<td>m</td>
<td>Suspended sediment concentration</td>
<td>3.93 mg L$^{-1}$</td>
<td>Forcing function</td>
</tr>
<tr>
<td>T</td>
<td>Tidal range</td>
<td>257 cm</td>
<td>Forcing function</td>
</tr>
<tr>
<td>q</td>
<td>Sediment settling coefficient</td>
<td>1 week$^{-1}$</td>
<td>Rate/Constant</td>
</tr>
<tr>
<td>k</td>
<td>Sediment trapping coefficient</td>
<td>117 cm$^{-2}$ shoot$^{-1}$ week$^{-1}$</td>
<td>Rate/Constant</td>
</tr>
<tr>
<td>$r_m$</td>
<td>Root:shoot production ratio</td>
<td>1.34 (unitless)</td>
<td>Rate/Constant</td>
</tr>
<tr>
<td>$f_2$</td>
<td>Labile fraction of leaf litter</td>
<td>0.45 (unitless)</td>
<td>Rate/Constant</td>
</tr>
<tr>
<td>$f_3$</td>
<td>Labile fraction of root material</td>
<td>0.15 (unitless)</td>
<td>Rate/Constant</td>
</tr>
<tr>
<td>$k_2$</td>
<td>Decomp. rate of surface labile o.m.</td>
<td>0.2 week$^{-1}$</td>
<td>Rate/Constant</td>
</tr>
<tr>
<td>$k_3$</td>
<td>Decomp. rate of deep labile o.m.</td>
<td>0.09397 week$^{-1}$</td>
<td>Rate/Constant</td>
</tr>
<tr>
<td>$k_4$</td>
<td>Decomp. rate of surface refractory o.m.</td>
<td>0.006 week$^{-1}$</td>
<td>Rate/Constant</td>
</tr>
<tr>
<td>$k_5$</td>
<td>Decomp. rate of deep refractory o.m.</td>
<td>0.0001 week$^{-1}$</td>
<td>Rate/Constant</td>
</tr>
<tr>
<td>$p_m$</td>
<td>Minimum fraction of pore space</td>
<td>0.4050 (unitless)</td>
<td>Rate/Constant</td>
</tr>
<tr>
<td>$p_x$</td>
<td>Maximum fraction of pore space</td>
<td>0.5513 (unitless)</td>
<td>Rate/Constant</td>
</tr>
<tr>
<td>$p_k$</td>
<td>Soil compaction constant</td>
<td>5.82 g cm$^{-2}$</td>
<td>Rate/Constant</td>
</tr>
<tr>
<td>$r$</td>
<td>Root distribution constant</td>
<td>0.307 cm$^{-1}$</td>
<td>Rate/Constant</td>
</tr>
</tbody>
</table>

**Sediment Dynamics**

**Relative Elevation**

$R$ Relative sea level rise rate See Table 9 Forcing function

$I_r$ Initial relative elevation See Table 1 State variable

$N$ Erosion rate 0 cm wk$^{-1}$ Forcing function 0.0385 cm wk$^{-1}$

4.2.5 Validation

The model was tested by running it for 12 simulated years and comparing the simulated relative elevation change rate to the actual rate obtained from surface elevation tables (SETs) in Padilla Bay from 2002-2014. Simulated elevation change rates were calculated by dividing the simulated change in relative elevation by the number of simulation years. This rate was equivalent to the relative elevation change rates determined from SET measurements (see Chapter 3). I performed separate validation model runs altering only the
initial relative elevation. Each validation run’s initial elevation corresponded to the 2002
elevation of one of 20 SET sites (see Chapter 3 for SET elevations).

I calculated the modeling efficiency (EF) to quantify the results of the validation
exercise. EF is a dimensionless statistic that can be used to compare simulated to observed
values across a range of sites. EF is defined by Loague and Green (1991) as:

\[
EF = 1 - \frac{\sum (y_i - y_o)^2}{\sum (y_i - y_m)^2}
\]  \hspace{1cm} (15)

where:

\( y_i \) = observed value
\( y_o \) = simulated value
\( y_m \) = mean of observed values

EF has an upper limit of 1 and a lower limit of negative infinity. An EF value of 1
indicates a perfect match between simulated and observed values.

4.2.6 Sensitivity analyses

I evaluated the model’s sensitivity to several parameters. I adjusted the value for each
of these parameters by ±5% and ±50% to observe its effect on simulated relative elevation
change for both short- and long-term predictions of 10 and 100 years, respectively. Adjusted
parameter values were all determined to be within the realm of physical and biological
possibility, either currently or in the future. However, adjusted parameters with small initial
values did not necessarily represent the range of possible values (e.g., SSC and RSLR). Once
the sensitive parameters are identified, they could be further examined across any range of
interest. I conducted the sensitivity analyses using an initial relative elevation of 9.1 cm
MLLW with no erosion. For both 10- and 100-year simulations I calculated the sensitivity range, defined by Rybczyk and Cahoon (2002) as:

\[
\text{Sensitivity Range} = \left( h_{(1+x)} \right) - \left( h_{(1-x)} \right)
\]

where:

\[
h_{(1+x)} = \text{final relative elevation with initial parameter value multiplied by } (1+x) \text{ (cm)}
\]

\[
h_{(1-x)} = \text{final relative elevation with initial parameter value multiplied by } (1-x) \text{ (cm)}
\]

where:

\[x = \text{fraction of parameter value variation, either 0.05 or 0.50 (unitless)}\]

### 4.2.7 Sea level rise scenarios

I ran the model for 100 simulated years starting in model year 2002, using six different RSLR scenarios (Table 9). The most conservative scenario (“current”) uses the current rate of RSLR based on local tide gauge measurements, and assumes no increase in RSLR. Three scenarios are based on predictions from the IPCC Fifth Assessment Report (Church et al. 2013). Since the IPCC values represent the global mean ESLR rate, I converted them to RSLR rates by subtracting the local vertical land movement rate of +0.10 cm yr\(^{-1}\) based on a continuous GPS record at Anacortes, WA, reported as the current and projected rate from 2010-2100 (NRC 2012). The positive land movement in this region indicates uplift dominated by plate tectonics with the Cascadia Subduction Zone (NRC 2012). Finally, I included two RSLR projections from Mote et al. (2008) and NRC (2012). I determined these rates by dividing the reported projected increase in sea level height for the 2000-2100 period by the number of projected years.
Table 9. Model scenarios of rates of sea level rise. IPCC scenarios reported as ESLR were converted to RSLR by subtracting +0.10 cm yr\(^{-1}\) from vertical land movement. Other scenarios were reported as RSLR rates, requiring no conversion.

<table>
<thead>
<tr>
<th>Scenario Name</th>
<th>Description</th>
<th>Source</th>
<th>ESLR rate (cm yr(^{-1}))</th>
<th>RSLR rate (cm yr(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td>Current</td>
<td>Based on tide gauge records at Friday Harbor for the period 1934-2006</td>
<td>NOAA</td>
<td>NA</td>
<td>0.113</td>
</tr>
<tr>
<td>IPCC-Low</td>
<td>Based on the lower limit of all IPCC AR5 scenarios (from scenario RCP26)</td>
<td>Church et al. 2013</td>
<td>0.274</td>
<td>0.174</td>
</tr>
<tr>
<td>Mote</td>
<td>Based on middle projection for 2100 relative to 2000 in Puget Sound</td>
<td>Mote et al. 2008</td>
<td>NA</td>
<td>0.340</td>
</tr>
<tr>
<td>IPCC-Mid</td>
<td>Average of all IPCC AR5 scenario medians</td>
<td>Church et al. 2013</td>
<td>0.513</td>
<td>0.413</td>
</tr>
<tr>
<td>NRC</td>
<td>Based on projected 2100 sea level relative to 2000 in Seattle region</td>
<td>NRC 2012</td>
<td>NA</td>
<td>0.618</td>
</tr>
<tr>
<td>IPCC-High</td>
<td>Based on upper limit of all IPCC AR5 scenarios (from scenario RCP8.5)</td>
<td>Church et al. 2013</td>
<td>0.853</td>
<td>0.753</td>
</tr>
</tbody>
</table>

4.2.8 Erosion scenario

I included an erosional scenario for each RSLR scenario, with a constant height removed from the relative elevation at each timestep for 26 weeks per year during the winter. The erosion rate was set to 0.0385 cm week\(^{-1}\), which approximates 1 cm yr\(^{-1}\) over the 26-week period. This rate was based on field observations of sediment loss at the majority of higher elevation SET sites in Padilla Bay. Erosion was set to zero at elevations below MLLW. Results for each RSLR scenario are presented both with and without erosion.

4.2.9 Suspended sediment concentration requirements

I ran experimental model simulations to determine the minimum SSC required to maintain relative elevation equilibrium over a 100-year timeframe. I performed these experimental simulations for the seven initial relative elevations corresponding to the transect accretion monitoring sites. For a given initial relative elevation, I determined this equilibrium SSC separately for each RSLR scenario with erosion.
4.3 Results

4.3.1 Calibration

Simulated bulk density and pore space values were within one standard deviation of the mean at all depths within the mean measured profiles (Figure 26). Simulated percent organic matter was consistently lower than measured values by approximately 1.5% after adjusting decomposition rates. However, a 1.5% discrepancy is quite small relative to the range of values typically seen in coastal wetlands, and no further adjustment of parameter values substantially improved the fit of the simulated to measured organic content.

Figure 26. Actual and simulated sediment profiles of bulk density, organic content, and pore space. Actual values are the means (± 1 SD) from five cores along the PBNERR transect.
4.3.2 Validation

Simulated rates of relative elevation change for the 12-year period from 2002-2014 were predominantly positive, whereas observed rates were mostly negative (Figure 27). This contradiction indicated the need for a scenario with an erosion parameter. A scenario including seasonal erosion at intertidal elevations improved model performance, with an EF of -2 instead of -17.

Figure 27. Simulated versus observed rates of relative elevation change at all SET sites (n = 20) for the period 2002-2014. Model performance was improved with the addition of an erosion forcing function.

For the validation simulations without erosion, the mean (± SD) simulated rate of relative elevation change at the SET sites was 0.48 ± 0.38 cm yr\(^{-1}\). The observed mean rate of relative elevation change at these sites was -0.37 ± 0.23 cm yr\(^{-1}\). For the validation simulations with erosion, the mean simulated rate of relative elevation change was -0.33 ± 0.37 cm yr\(^{-1}\).
4.3.3 Sensitivity analyses

The model was most sensitive to the minimum and maximum pore space, followed by tidal range, SSC, sediment trapping coefficient, and RSLR. The remaining parameters had little to no effect on the final relative elevation (Table 10). The 100-year sensitivity ranges were actually smaller than the 10-year values for several parameters. This resulted from simulations during which relative elevation stabilized at some point, for one or both of the plus and minus parameter values. This stabilization was more likely to happen with longer simulations. Stabilization typically occurred around 75 cm MLLW, just above the location of maximum stem density. This is similar to the equilibrium point reported by Morris et al. (2002) just above the maximum plant biomass.
Table 10. Sensitivity of relative elevation to changes in model parameters for the non-erosional scenario. Sensitivity range was calculated as the difference in final relative elevation between simulations with parameter values varied by plus and minus 5%, or plus and minus 50%.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Description</th>
<th>Sensitivity range (cm)</th>
<th>10 yr</th>
<th>100 yr</th>
<th>10 yr</th>
<th>100 yr</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>± 5%</td>
<td></td>
<td></td>
<td>± 50%</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>10 yr</td>
<td>100 yr</td>
<td>10 yr</td>
<td>100 yr</td>
<td></td>
</tr>
<tr>
<td>m</td>
<td>Suspended sediment concentration</td>
<td>0.99</td>
<td>2.16</td>
<td>9.66</td>
<td>30.88</td>
<td></td>
</tr>
<tr>
<td>k</td>
<td>Sediment trapping coefficient</td>
<td>0.93</td>
<td>2.04</td>
<td>9.34</td>
<td>29.26</td>
<td></td>
</tr>
<tr>
<td>q</td>
<td>Sediment settling coefficient</td>
<td>0.03</td>
<td>0.08</td>
<td>0.33</td>
<td>0.78</td>
<td></td>
</tr>
<tr>
<td>T</td>
<td>Tidal range</td>
<td>1.07</td>
<td>3.13</td>
<td>10.72</td>
<td>41.75</td>
<td></td>
</tr>
<tr>
<td>R</td>
<td>Relative sea level rise rate</td>
<td>0.11</td>
<td>0.42</td>
<td>1.15</td>
<td>4.22</td>
<td></td>
</tr>
<tr>
<td>r_m</td>
<td>Root:shoot production ratio</td>
<td>0.01</td>
<td>0</td>
<td>0.11</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>r</td>
<td>Root distribution constant</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>p_k</td>
<td>Soil compaction constant</td>
<td>0.31</td>
<td>0.10</td>
<td>3.23</td>
<td>0.94</td>
<td></td>
</tr>
<tr>
<td>f_2</td>
<td>Labile fraction of leaf litter</td>
<td>0.01</td>
<td>0</td>
<td>0.05</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>f_3</td>
<td>Labile fraction of root material</td>
<td>0</td>
<td>0</td>
<td>0.02</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>k_2</td>
<td>Decomp. rate of surface labile o.m.</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>k_3</td>
<td>Decomp. rate of deep labile o.m.</td>
<td>0</td>
<td>0</td>
<td>0.01</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>k_4</td>
<td>Decomp. rate of surface refractory o.m.</td>
<td>0.01</td>
<td>0</td>
<td>0.16</td>
<td>0.01</td>
<td></td>
</tr>
<tr>
<td>k_5</td>
<td>Decomp. rate of deep refractory o.m.</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>p_m*</td>
<td>Minimum fraction of pore space</td>
<td>11.58</td>
<td>3.86</td>
<td>NA</td>
<td>NA</td>
<td></td>
</tr>
<tr>
<td>p_x*</td>
<td>Maximum fraction of pore space</td>
<td>2.01</td>
<td>0.43</td>
<td>NA</td>
<td>NA</td>
<td></td>
</tr>
</tbody>
</table>

* Results for ±50% exclude +50% p_m and -50% p_x scenarios because they resulted in logic errors.

4.3.4 Sea level rise scenario projections

Projection results for both non-erosional and erosional scenarios are presented here for comparison. All RSLR scenario projections without erosion resulted in a gain in relative elevation along at least a portion of the transect sites (Figure 28). Simulations using the two most conservative RSLR scenarios (“Current” and “IPCC-low”) revealed no elevation loss, with substantial elevation gain at middle-elevation sites corresponding with the range of Z. japonica distribution. The other RSLR scenarios resulted in relative elevation loss at the deeper sites (below MLLW), and at the highest site. Relative elevation never exceeded 77 cm MLLW, and this equilibrium elevation decreased with increasing rates of RSLR.
Figure 28. Projected relative elevation under six RSLR scenarios without erosion. Initial elevations were varied under each RSLR scenario, representing the seven accretion monitoring sites along the transect. Projected elevations are shown at 20-year intervals, beginning in model year 2002 and ending in 2102.

All RSLR scenario projections with erosion showed a loss in relative elevation at over half of the transect sites, while three of the RSLR scenarios predicted relative elevation loss along the entire transect (Figure 29). Simulations using the two most conservative RSLR scenarios revealed some elevation gain at the lower elevations, with the threshold between gain and loss at approximately 0 cm MLLW. All except the two most pessimistic RSLR scenarios (“NRC” and “IPCC-high) resulted in stabilization of the surface at MLLW for the
middle portion of the transect, which is an artifact of the model’s elevation threshold for erosion. The RSLR rates for those two scenarios appeared to be too high for the system to reach an equilibrium at any site along the transect.

Figure 29. Projected relative elevation under six RSLR scenarios with erosion. Initial elevations were varied under each RSLR scenario, representing the seven accretion monitoring sites along the transect. Projected elevations are shown at 20-year intervals, beginning in model year 2002 and ending in 2102.

4.3.5 Suspended sediment concentration requirements

The current SSC in Padilla Bay of 3.93 mg L\(^{-1}\) is projected to be sufficient for one site to maintain elevation equilibrium over a 100-year timeframe, under three of the six RSLR
scenarios with erosion (Figure 30). Two other sites are projected to be stable under the two most conservative RSLR scenarios. The remaining sites appear to require more than the current suspended sediment load under all RSLR scenarios. The site with the highest relative elevation required substantially more sediment than the other elevations because this site was located above the limit of eelgrass growth (Figure 30). With the exception of this highest site, the majority of the transect may remain stable if provided with a suspended sediment load of approximately 15 mg L\(^{-1}\).

Figure 30. Minimum suspended sediment concentrations required to maintain relative elevation equilibrium over a 100-year timeframe. Results are shown for all RSLR scenarios, with initial elevations representing the seven accretion monitoring sites along the transect. The y-axis is expanded in the second plot to reveal more detail at lower SSC values. The dashed line represents the current SSC in Padilla Bay.
4.4 Discussion

4.4.1 Calibration

Simulated organic content was underestimated by approximately 1.5% throughout the core. Adjustment of model parameters such as decomposition rates, and leaf or root labile fractions, could not reduce this discrepancy. Other studies using the REM reported the same issue (Rybczyk et al. 1998, Kairis and Rybczyk 2010), and they suggested an inaccurate decomposition rate as the most likely cause. However, it is also possible that the highly refractory organic matter remaining constant in the sediment column beyond approximately 10 cm in depth is not completely sourced from eelgrass organic matter, but also from refractory organic matter, or “black carbon” sourced from suspended organic particles that settle out of the water column. Another possibility is that the eelgrass-derived organic matter no longer decomposes below approximately 10 cm in depth, either due to conditions below that depth being unsuitable for decomposition, or due to the inherent resistance of the organic matter to further decomposition. Bioturbation could also explain the model’s underestimate of organic matter below the surface. Deposit feeders transport sediment downward from the surface, increasing the organic content below the surface (Shull 2001).

4.4.2 Validation

The model underestimated simulated elevation loss relative to measured elevation loss over the 12-year validation period. Kairis and Rybczyk (2010) reported the same discrepancy, suggesting the cause to be erosion that is otherwise unaccounted for, or the use of a relatively short, and therefore unreliable, 4-year elevation change dataset. Unfortunately, the average elevation change rate has not increased with a longer time series. Now with a 12-year elevation change dataset, the measured rate of elevation loss is actually slightly higher,
therefore SET data reliability does not provide a likely explanation for the underestimated simulation results. The model’s underestimate of elevation loss is therefore most likely due to an underestimate of either erosion or shallow subsidence (or both). I determined erosion to be the most likely explanation, because I did not observe any significant discrepancies between actual and simulated sediment core profiles that would be expected if subsidence were at fault.

The use of long-term elevation change rates to validate a model based on short-term accretion rates may not be ideal in this situation, because *Z. japonica* distribution, which appears to significantly affect accretion, has been expanding in the last 15 years (Bulthuis and Shull 2002, 2006). It may take several more years for the SET dataset to more accurately represent current elevation change rates at the SET locations, after the *Z. japonica* distribution at these sites has stabilized. Plant characteristics play a role in elevation dynamics long after the plants themselves have died, due to the longer-term decomposition process. It would be preferable to use longer-term accretion measurements, from $^{210}$Pb or $^{137}$Cs analyses, but the prevalence of bioturbators and erosive conditions throughout Padilla Bay is not conducive to reliable estimates of long-term accretion rates from cores at each accretion monitoring site (see Chapter 2).

4.4.3 Erosion

The hybrid model’s predecessors did not include an erosion term, because they were developed for net depositional systems. However, the validation exercise made it clear that the hybrid model needed an additional erosion term to apply to Padilla Bay’s frequently inundated environment. The erosion added to the model is a forcing function that corrects relative elevation at each timestep, rather than changing sediment inputs and sediment
characteristics (although these are eventually altered as an indirect effect of relative elevation loss). Erosion was simulated for only half of the year, to allow the system to accrete during the growing season as observed in the field.

4.4.4 *Impact of sea level rise on relative elevation*

Results for the non-erosional scenario simulations indicate that much of the transect will equilibrate near the elevation with the maximum *Z. japonica* stem density. This elevation is close to the boundary between the *Z. japonica* and mudflat zones. This predicted equilibration would involve an expansion of the *Z. japonica* total cover and reduction of the *Z. marina* cover in Padilla Bay. Using the “Current” RSLR scenario, *Z. marina* would remain at only the two lowest elevation transect sites. At the other extreme, using the “IPCC-high” RSLR scenario, *Z. marina* would remain at the three lowest transect sites, which still is not much relative to its current coverage. The lower sites (below 0 cm MLLW) are predicted to be fairly stable over time under all but the “NRC” and “IPCC-high” RSLR scenarios, under which they are predicted to lose elevation substantially.

Results for the erosional scenario simulations indicate that at least some of the transect sites will experience a loss of relative elevation, for all RSLR scenarios. In this case, the higher elevation sites were predicted to lose the most, resulting in an increased cover of *Z. marina* throughout the bay and a reduced cover of *Z. japonica*. The “NRC” and “IPCC-high” scenarios resulted in a nearly even decline in relative elevation across the transect, indicating the inability of any ecogeomorphic feedbacks to keep up with those rates of RSLR. These more pessimistic RSLR scenarios may in fact be the most likely, because recent global mean ESLR increases have followed the upper end of previous predictions (Rahmstorf 2007).
4.4.5  Effect of suspended sediment concentration on model projections

The current suspended sediment load in Padilla Bay is unsustainable for almost all of the RSLR scenarios. Under current SSC conditions, only the few sites below MLLW were projected to keep up with the current rate of RSLR because these sites were not experiencing erosion. Most RSLR scenario projections required a higher SSC than is currently available, although in general, the requirements do not exceed 15 mg L\(^{-1}\). This SSC is not outside the realm of possibility. For context, the mean annual SSC value of 3.93 mg L\(^{-1}\) in Padilla Bay is approximately 20% of the “low” suspended sediment scenario used by Morris et al. (2002) for the MEM, and approximately 5% of their “high” suspended sediment scenario (20 and 80 mg L\(^{-1}\), respectively). Padilla Bay’s SSC is much lower than a typical estuary, in part because the bay lacks a direct connection to a major river, but also because most of its historic indirect river connections have been removed. The restoration of some of these Skagit River distributaries may be enough to make the bay’s eelgrass meadows sustainable, at least within a 100-year timeframe.

Before facilitating any increase in sediment load to Padilla Bay, further research is necessary to determine the maximum SSC that the eelgrass here can tolerate. An increase in SSC will decrease the amount of light available to eelgrass for photosynthesis. \textit{Z. marina} needs at least 10-20\% of surface irradiance to survive (Duarte 1991, Short and Burdick 1995), and as much as 50\% of surface irradiance to thrive (Ochieng et al. 2010). It is currently unknown what level of SSC would allow Padilla Bay’s eelgrass to meet its light requirements, given local conditions for phytoplankton and epiphytes.
4.4.6 Opportunities for model improvement

The calibration of the sediment trapping and settling parameters \((k\ and\ q)\) would be improved with stem density manipulations to isolate the effects of depth and density on sediment accretion rates, rather than relying on natural variation in these variables. I opted not to manipulate stem densities in the field because it was unknown how large an area would have to be altered to ensure that the observed accretion rate was not affected by the surrounding patch. This might require a large area, and I chose not to remove large amounts of eelgrass without any assurance of its effectiveness.

Erosion could be simulated more realistically by removing sediment from the top sediment cohort(s) to more accurately simulate changes to the height of the sediment column. This is difficult to do mechanistically when erosion removes more sediment than is contained in the surface cohort, and it removes both mineral and organic components. The model would be improved with a more mechanistic treatment of erosion, and this would also allow the model to simulate carbon accumulation rates. It would also be helpful to determine from field measurements what factors affect the magnitude of erosion. Erosion is currently programmed as a function of elevation, but there are certainly other factors involved. A similar model of elevation change developed by Kirwan and Murray (2007) included an erosion algorithm simulating erosion as a function of bed shear stress. Their approach to incorporating erosion was similar to the hybrid model, reducing the final relative elevation at each model timestep.

Accretion could be allowed to vary seasonally by incorporating seasonal variation in variables such as stem density and SSC. SSC could also be varied spatially, given the appropriate field measurements. SSC most likely varies as a function of distance to shore and nearby channels, and the incorporation of this variation could be a valuable improvement to
the model. Some other models have incorporated the distance to the nearest channel as an important factor in determining SSC (D’Alpaos et al. 2007, Kirwan and Murray 2007, Bartholdy et al. 2010, Rogers et al. 2012).

Light attenuation affects plant productivity and distribution (Mateo et al. 2006), and this could be a valuable addition to the model. There is a threshold to the SSC that eelgrass can tolerate, beyond which it cannot meet its light requirements and productivity decreases. It may be important to know where this SSC threshold exists, particularly if the suspended sediment load were to be altered to improve ecosystem sustainability.

Leaf litter inputs to the sediment surface are likely overestimated. The model does not consider loss of leaf litter due to herbivory and export. The fraction of seagrass leaf production lost to export is likely much larger than the fraction lost due to herbivory (Mateo et al. 2006). Unfortunately, few studies have quantified the export of seagrass production outside the seagrass bed because measurement can be difficult (Mateo et al. 2006). The few existing studies have suggested that seagrass leaf litter export can vary from 0 to 100% of leaf production (Bach et al. 1986, Hemminga and Nieuwenhuize 1990, Mateo and Romero 1997, Ochieng and Erftemeijer 1999, Hemminga and Duarte 2000). This variability is due to differences among eelgrass meadows in hydrodynamic conditions and bed exposure. A few studies have reported up to 90% of *Posidonia oceanica* litter being exported both landward and seaward (Romero et al. 1992, Cebrian and Duarte 2001), and there is no reason to think that export would not be this substantial in Padilla Bay. A review of carbon budgets for a variety of coastal wetland types by Duarte and Cebrian (1996) indicated that seagrass ecosystems export 24.3% of their total NPP on average, with 18.6% lost to herbivory, 50.3% to decomposition, and only 15.9% stored in the sediment. It is clear that there is some litter
transport occurring in Padilla Bay because I have observed large masses of leaf detritus concentrated in a few locations during the winter; we can thus assume that more than 0% of leaf litter is exported from the site of production. It is possible that Z. japonica has a lower export rate than Z. marina if the Z. japonica decomposition rate is higher (Hahn 2003). When litter remains on the surface for a longer period of time before decomposition or burial, it has a greater chance of being moved (Mateo and Romero 1997, Pérez et al. 2001). On the other hand, the two species could also have different leaf buoyancy properties, which may also affect their export rates. Zieman et al. (1979) compared the buoyancy of two seagrass species and found that the broad-leaved species exported only 1% of its production whereas the more buoyant, thin-leaved species exported 75% of its production. This would imply a higher export rate for Z. japonica, whereas decomposition rates suggest a higher export rate for Z. marina. Litter export rates for these two species could be estimated in the field by comparing leaf litter mass inside and outside enclosures. Litter export could be added to the model by reducing the amount of leaf litter by a specified amount before it enters the sediment column.

Just as some leaf litter is likely exported, other organic material may also be imported. A study by Garcia et al. (2002) estimated that only half of seagrass production was deposited in situ, in a Mediterranean Posidonia oceanica bed. This seagrass detritus was determined to contribute only 28% to the total carbon deposited, and 43% to total sediment carbon. The rest of the total carbon was found to be derived from sestonic particles. Similarly, a study of Thalassodendron ciliatum beds in Kenya reported that only 30% of sediment organic matter was derived from seagrass material (Hemminga et al. 1994). The possibility of other unconsidered organic contributions may help explain why the organic content simulated by the hybrid model was consistently lower than the actual organic content.
of Padilla Bay sediment cores. The model does not include organic inputs other than seagrass material, though studies suggest that inputs from seston could be the dominant source of organic material in the sediment.

The hybrid model does not consider the effect of bioturbation on sediment processes, though it is almost certainly affecting sediment characteristics across the bay. Deposit feeding benthos are abundant in Padilla Bay (Ray 1997), and their feeding results in the transport of surface sediment to the base of the $^{210}$Pb layer (e.g., Shull 2001). This affects the sediment profiles by increasing the organic content below the surface over what would be expected without bioturbation. The absence of any bioturbation treatment in the current hybrid model may explain why the model consistently overestimated organic content throughout the sediment column. Bioturbation could be incorporated into the model by modifying sediment cohort transfer rates, based on site-specific estimates of bioturbation rates. This would be a valuable addition to future model revisions, especially for model applications to lower-intertidal habitats such as Padilla Bay.

In contrast to the fate of aboveground production, belowground production is rarely exported, and most of it remains in situ to decompose or persist as refractory material (Mateo et al. 2006). Therefore, my estimate of root and rhizome production input in the soil column is likely more accurate than aboveground input.

Different plant species have different decomposition rates, and it may be worthwhile to modify the model to decompose organic matter using different rates. Decomposition rates for $Z. \text{japonica}$ were not measured as part of this study, thus they were assumed to be equivalent to previously measured $Z. \text{marina}$ rates. However, a previous study by Hahn (2003) did measure decomposition rates for both species, and reported $Z. \text{japonica}$ to
decompose roughly 20% faster than Z. marina. Hahn’s rates could be used in future modeling exercises. A faster Z. japonica decomposition may or may not affect long-term elevation change. However, the species could also differ in the fraction of labile versus refractory organic matter they contain, and this could affect rates of elevation change. For example, if Z. japonica has a higher labile organic matter fraction than Z. marina, then shallow subsidence may be greater at the Z. japonica sites than predicted based on Z. marina values.

Finally, the hybrid model does not include the shorter-term effects of the El Niño Southern Oscillation (ENSO) on eelgrass growth and density, which has been shown to affect these plant characteristics (Thom et al. 2014). Studies of the Z. marina meadow in Willapa Bay, Washington have shown an effect of temperature on stem density and aboveground biomass (Thom et al. 2003). Stem density was observed to be lowest at both extremes of the Oceanic Niño Index (ONI). With a predicted increase in frequency of “super” El Niño years due to anthropogenic climate change (Cai et al. 2014), this would raise the average water temperature and sea level and likely reduce eelgrass stem density, and push Padilla Bay’s eelgrass range further shoreward, beyond what the model predicts without the incorporation of this ONI effect.
5 CONCLUSIONS

I used a combined field and modeling approach to assess the long-term sustainability of a sediment-starved eelgrass meadow in the face of sea level rise. Under certain conditions, coastal wetlands can maintain a stable elevation with respect to a changing sea level due to feedbacks between water depth, plant growth, and sediment accretion. The purpose of this study was to predict long-term surface elevation change under current conditions in Padilla Bay, and then determine the conditions necessary for a stable eelgrass meadow under various sea level rise scenarios.

Field measurements of plant characteristics (biomass and stem density), relative elevation, and sediment accretion allowed for a quantification of the relationships among these factors. Results from this study indicated that short-term accretion rates could be predicted by stem density alone, while surprisingly, aboveground plant biomass and relative elevation had no significant effect on accretion rates. The non-native Zostera japonica grows in much higher densities than the native Z. marina. Although the expansion of Z. japonica is of ecological concern due its replacement of mudflat habitat, it may be beneficial in providing a valuable sediment trapping service in this sediment-starved estuary.

This study contributed to the ongoing monitoring of an extensive surface elevation table (SET) network in Padilla Bay, now providing a 12-year dataset of surface elevation change to use for model validation. Only three of the 20 SET sites exhibited positive rates of surface elevation change, although all sites are losing elevation with respect to sea level. This surface elevation loss at the SETs, in combination with the sediment accretion observed along an elevation gradient, indicate the occurrence of substantial shallow subsidence or erosion (or both). The surface elevation change rates observed at the SET sites did not show
the expected negative correlation with relative elevation. Several SET sites showed early signs of sediment scour in addition to previously identified scoured sites, and these should be closely monitored in future years to prevent them from creating a bias in the dataset.

I created a hybrid surface elevation model by combining two existing models (the Marsh Equilibrium Model (MEM) and Relative Elevation Model (REM)), and modified it as necessary to represent site-specific field measurements of plant characteristics and sediment dynamics across an elevation gradient. The hybrid model presented here has improved upon some aspects of the spatially explicit REM (SREM) previously applied to Padilla Bay. Although the hybrid model is not spatially explicit, it has been initialized and calibrated to reflect the empirical relationship between aboveground plant characteristics and sediment accretion. A suspended sediment concentration (SSC) parameter has been added for model initialization. The hybrid model also includes both eelgrass species that occur in Padilla Bay instead of just Z. marina. The inclusion of Z. japonica may be very important, given its relatively high stem densities, and the significant effect of stem density on sediment accretion rates, as indicated by this study. Several studies have determined stem density to be an important driver of sediment accretion rates in a variety of coastal wetlands, yet the hybrid model is the first model of its kind to incorporate density as a driving factor.

The model validation exercise revealed that the hybrid model overestimated relative elevation change, indicating the need for an erosion parameter to better approximate sediment dynamics in Padilla Bay’s frequently inundated, erosive environment. A sensitivity analysis indicated that the hybrid model was most sensitive to pore space, tidal range, SSC, the sediment trapping coefficient, and the rate of relative sea level rise (RSLR). Obtaining
accurate field measurements for these parameters should be a focus for future model applications.

Long-term model projections for all RSLR scenarios with no erosion predicted the development of a stable platform at an elevation that maximizes eelgrass stem density, and is therefore dominated by *Z. japonica*. These scenarios would result in a reduction in the total *Z. marina* cover in the bay. In contrast, simulations using RSLR scenarios with the included erosion term projected a substantial relative elevation loss, particularly at higher elevations. For the more conservative RSLR scenarios, projections revealed a reduction in the elevation gradient, leveling off near the MLLW tide level. However, simulations using the more extreme (and perhaps more likely) RSLR scenarios resulted in a consistent relative elevation loss across the entire elevation gradient, given current SSC conditions. These more extreme scenarios were predicted to require an SSC up to 15 mg L\(^{-1}\) to remain stable with respect to sea level. Although this SSC value is nearly four times the current annual mean SSC value in Padilla Bay, it is still lower than typical values seen elsewhere. The required SSC conditions may be attainable with the restoration of just a small number of Skagit River distributaries. Without the provision of an increased sediment supply, Padilla Bay’s eelgrass habitat could be retained longer with dike setbacks that allow for upland transgression, though this would only be a temporary solution.
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