

# A blue carbon assessment for the Stillaguamish River estuary: Quantifying the climate benefits of tidal marsh restoration

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## Project Summary Report

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## Key Findings

- This is one of the first studies anywhere, and particularly in the PNW, to report sediment carbon stock and carbon sequestration rates for a restored salt marsh and adjacent natural salt marshes.
- The mean rate of carbon sequestration in a recently restored marsh in the Stillaguamish River Delta/Estuary ( $230.49 \text{ g C m}^{-2} \text{ yr}^{-1}$ ) is currently twice that of carbon sequestration rates in adjacent natural marshes.
- The mean rate of elevation change in the same restored marsh ( $+ 2.74 \text{ cm yr}^{-1}$ ) is nearly three times the rate of elevation change measured in adjacent natural marshes.
- Salt marshes immediately adjacent to the active delta distributary, including the restored marsh, are all accreting at rates that exceed the rate of sea level rise indicating some potential for resiliency.
- Carbon stocks and carbon sequestration rates are driven by different processes in the Stillaguamish River estuary. Carbon stocks appear driven primarily by elevation (as a proxy for tidal inundation frequency), whereas sequestration rates appear driven primarily by sediment accretion rates.
- The 60-hectare restoration site is expected to accumulate approximately 4,500 to 9,000 Tonnes of carbon before it reaches equilibrium with adjacent marshes, which is equivalent to removing approximately 3,500 to 7,000 cars from the roads for one year. That amount of carbon is worth approximately \$65,000 to \$165,000 using a national average carbon offset price of \$4 to \$5 per Tonne  $\text{CO}_2$ .

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## Introduction

Coastal wetlands are among the most valuable ecosystems in terms of the ecosystem services they provide (Batker et al. 2008). Recently, coastal wetlands have been recognized for their role in blue carbon sequestration and climate change mitigation (Callaway et al. 2012). “Blue carbon” refers to the fraction of atmospheric carbon captured by salt marsh plants during photosynthesis and incorporated into plant organic matter that ends up stored, more or less permanently, in salt marsh soils. Draining and converting coastal wetlands to other land uses often results in the release of that stored carbon back into the atmosphere. Restoring these same wetlands can potentially reverse this loss.

In the Pacific Northwest United States (PNW), neither land managers nor policy makers have had the region- or site-specific quantitative carbon data needed to incorporate the value of carbon sequestration into improved coastal resource management plans or to support coastal resource management actions through carbon finance mechanisms (e.g., wetland restoration financed by regulatory carbon markets). For example, a recent review of North American blue carbon (CEC 2013) found no published studies relating to blue carbon stocks or emissions in the coastal wetlands of the PNW, and the Pacific Northwest Coastal Blue Carbon Working Group identified the absence of comprehensive site-scale data that quantify PNW coastal wetland carbon stocks and net carbon sequestration rates as a key information gap. Since then, a few PNW blue carbon studies have been completed (e.g., Brophy et al. 2017, 2018, Crooks et al. 2014, Diefenderfer et al. 2018, Poppe and Rybczyk 2018), but much work remains to document the full extent of variability in blue carbon potential across the region. This study addresses that need by providing estimates of the carbon stocks and accumulation rates for the existing and recently restored tidal marshes of the Stillaguamish River estuary in Port Susan Bay, WA.

## Methods

### Study Area

The Stillaguamish River discharges into the Port Susan Bay Preserve, owned by The Nature Conservancy (TNC). The Preserve contains a 60-hectare restoration site that was reintroduced to tide and river pulses, via levee removal in 2012. Prior to the restoration, the 60-hectare site had been managed as diked and drained farmland for many decades. Since 2011, the wetlands research group at Western Washington University (WWU) has partnered with TNC to conduct pre- and post-restoration monitoring within the restoration site and throughout the estuary (Figure 1). This included the establishment of 21 permanent Surface Elevation Tables (SETs) to monitor long-term elevation change and sediment dynamics throughout the estuary. As an extension, in this study we measure carbon stocks and carbon sequestration rates in these marshes and quantify the climate/carbon mitigation benefits of both maintaining and restoring healthy tidal marshes.

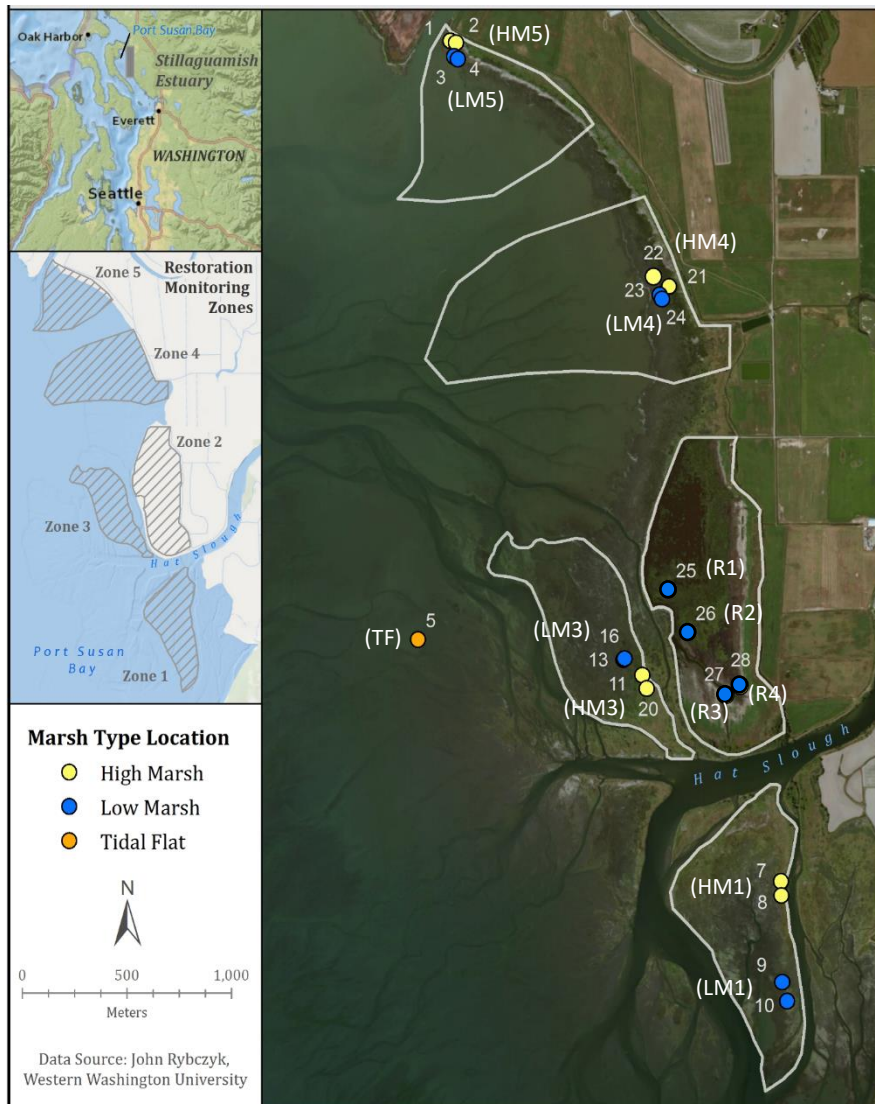


Figure 1. The Stillaguamish Estuary and the permanent sediment elevation table (SET) benchmarks, indicated by the colored dots. Site names are shown in parentheses next to their associated SET(s). Zone 2 indicates the 60-hectare restoration zone accomplished by dike removal and the subsequent re-establishment of tidal and riverine pulsing.

Study sites were located in five marsh zones (Figure 1). These zones represent a range of conditions, from a healthy, undisturbed marsh (Zone 1), to a regenerating, rapidly accreting emergent marsh in the restoration zone (Zone 2). The study zones also vary hydrologically, with some located directly adjacent to the Stillaguamish River mouth (shown as Hat Slough) and others more distant from the river and its distributaries.

With the exception of the restoration zone, each of the zones consists of two sites (high and low marsh) and each site has two SETs. The restoration zone consists of four low marsh sites each containing one SET (there is currently no high marsh there due to subsidence prior to restoration) (Figure 1). An additional site is located in the lower elevation tidal flats for a total of 13 sites. At each site, total ecosystem carbon stocks were partitioned into the three accepted Intergovernmental Panel on Climate

Change (IPCC) components that apply to non-forested coastal wetlands: aboveground biomass pools, belowground biomass pools, and soil carbon pools (IPCC 2014).

### **Field Sampling**

Field sampling was conducted in the spring and summer of 2016. At each of the 13 sites, two replicate cores were collected to determine soil carbon stocks and sediment accretion rates. PVC coring tubes (10.2 cm internal diameter) were driven up to 1 meter into the sediment with a sledgehammer. Cores were kept upright during transport to the laboratory and were frozen for later processing.

Aboveground and belowground plant biomass samples were collected at the end of the growing season (late August) from three replicate plots (0.15 m<sup>2</sup>) distributed haphazardly at each site. All aboveground biomass within each plot was clipped to the sediment surface. Belowground biomass samples were collected by pushing in PVC coring tubes (10-cm internal diameter) to a depth of 50 cm in the center of each replicate plot to capture the entire root zone. Belowground biomass cores were extracted in the field and sectioned into 10-cm depth sections. Biomass samples were stored in plastic bags for transport to the laboratory and temporary storage at 2°C.

We measured elevation change rates at 21 previously established SETs in the spring of 2016, 2017, and 2018 to continue building a time series of elevation change that began in 2011. Field measurement methodology remained consistent with our previous SET monitoring, following methods described by Cahoon and Lynch (2003). Elevation change rates were determined using an ordinary least squares linear regression, with time as the independent variable, elevation relative to initial as the dependent variable, and the slope of the regression indicating the rate of surface elevation change.

Sediment porewater salinity was measured in late summer of 2016 at the time of plant biomass collection. At each site, one of the pits remaining from the belowground biomass cores was left open to allow porewater to collect. Porewater salinity was measured in the field with a portable refractometer. A few sites were completely dry during sampling, preventing any salinity measurements at those sites.

### **Laboratory Analyses**

Plant aboveground biomass samples were washed with a 0.5-mm sieve and separated by species. Belowground biomass samples were washed and separated into live and dead root fractions, with the dead fraction then discarded. All samples were oven dried at 60 °C for at least 96 hours and weighed. We used a 1:0.38 ratio to convert biomass dry weight to biomass carbon stock (Westlake 1963).

In the laboratory, frozen sediment cores were extracted, sliced into 2-cm sections, and dried at 60 °C for at least 96 hours to determine bulk density. A subsample of each section was pulverized using a Wiley Mill with 0.425-mm mesh screen. The organic matter (OM) content of each section was determined by loss on ignition (LOI). Subsamples were burned at 500 °C for approximately 8 hours, and weighed before and after burning (Craft et al. 1991).

We measured organic carbon ( $C_{org}$ ) content for a subset of samples ( $n = 124$ ) with a FlashEA 1112 CN analyzer (Thermo Fisher Scientific, Waltham, MA). We first measured total (organic and inorganic) carbon content, then measured inorganic carbon by analyzing the ashed subsamples that remained after LOI, and calculated the  $C_{org}$  content by subtraction. We developed a OM- $C_{org}$  conversion from these 124 samples that was then applied to all sediment samples to produce the  $C_{org}$  contents reported here ( $C_{org} = 0.0035 \cdot OM^2 + 0.4135 \cdot OM - 0.4496$ ;  $R^2 = 0.97$ ;  $P < 0.001$ ) (Appendix A, Figure 7). Carbon density was also calculated for each core section as the product of  $C_{org}$  content and bulk density. Each sediment

characteristic parameter was averaged across the top 30 cm of each core for reporting purposes. Sediment carbon stocks were also calculated to a depth of 30 cm. However, a thick layer of organic material (presumably from a depositional event or burial of pre-restoration biomass) was observed in the restoration site cores between 10 cm and 20 cm in depth, which biased the results high when averaged across the top 30 cm. Therefore we used only the top 10 cm of the restoration site profiles and extrapolated to 30 cm to best represent post-restoration conditions without the influence of this depositional layer.

We analyzed sediment grain size composition by separating fine from coarse sediments, to evaluate the relationship of grain size with sediment carbon stocks. A subset of six samples distributed in even depth intervals were selected for analysis from each core. Each 30 mL subsample was redried, weighed, soaked, and rinsed through a 63- $\mu\text{m}$  sieve to remove the fine sediment (silt and clay) portion. The remaining coarse-grained (sand) portion was dried and weighed.

Long-term sediment accretion rates were determined from the downcore distribution of excess  $^{210}\text{Pb}$  activity at one core per site. Our analysis was limited to one core per site due to time constraints (the analysis requires approximately one month per core). We used the Canberra Germanium Detector (model GL2820R, Mirion Technologies (Canberra) Inc., Meriden, CT), with gamma emissions at 46 keV and 351 keV recorded by Genie 2000 software (Canberra 2002). Excess (unsupported)  $^{210}\text{Pb}$  was calculated as the difference between total  $^{210}\text{Pb}$  (at 46 keV) and supported  $^{210}\text{Pb}$  (at 351 keV) to distinguish between excess  $^{210}\text{Pb}$  deposited at the sediment surface and supported  $^{210}\text{Pb}$  that has decayed from radium in the sediment. Cores were analyzed in 2-cm sections from the surface to the depth at which excess  $^{210}\text{Pb}$  declined to zero. With the constant initial concentration (CIC) model (Robbins et al. 1978), a linear regression of the natural log of excess  $^{210}\text{Pb}$  activity versus depth was used to determine the sediment accretion rate, which is equal to  $-\lambda/s$ , where  $\lambda$  is the half-life of  $^{210}\text{Pb}$  (22.2  $\text{yr}^{-1}$ ) and  $s$  is the slope of the regression. Because the four restored site  $^{210}\text{Pb}$  profiles showed evidence of disturbance, preventing us from assigning them a long-term accretion rate that could be compared with rates from the undisturbed sites, we used the mean ratio of  $^{210}\text{Pb}$ -based rate to SET-based rate from the natural marsh sites to estimate the  $^{210}\text{Pb}$ -based accretion rate for the restoration sites.

Carbon accumulation rates were calculated in two ways: 1) as the product of sediment carbon density and the  $^{210}\text{Pb}$ -based accretion rate, and 2) as the product of carbon density and the SET-based elevation change rate. These two rates are reported as the  $^{210}\text{Pb}$ -based carbon accumulation rate and the SET-based carbon accumulation rate, respectively. Although SET-based carbon accumulation rates are not as commonly reported in blue carbon studies, we hypothesized that, in concept, these rates should be comparable to  $^{210}\text{Pb}$ -based accumulation rates. We include them to 1) provide a validation for the  $^{210}\text{Pb}$ -based rates, 2) test the feasibility of using SET and  $^{210}\text{Pb}$  accretion rates interchangeably, and 3) provide a potential alternative to the  $^{210}\text{Pb}$ -based rates in the event that any cores were unable to be dated with  $^{210}\text{Pb}$  due to sediment mixing or other disturbance.

Statistical analyses were conducted in the R programming environment using a critical value ( $\alpha$ ) of 0.05 (R Core Team 2018). We used simple and multiple linear regressions to determine the primary environmental drivers of plant biomass, sediment carbon stocks, and carbon accumulation rates. We used a one-way ANOVA to compare high marsh and low marsh biomass.



## Results

### Plant biomass and biomass C stocks

Aboveground biomass ranged from 0 g dw m<sup>-2</sup> (at Site TF) to 978.64 g dw m<sup>-2</sup> (HM5) (Figure 2). Aboveground biomass at low marsh sites in particular averaged 279.09 g dw m<sup>-2</sup>, compared to 879.40 g dw m<sup>-2</sup> at high marsh sites, with 353.37 g dw m<sup>-2</sup> across the four restoration area sites. Belowground biomass across all sites was an average of 2.4 times greater than aboveground biomass (median of 1.6), ranging from 0 g dw m<sup>-2</sup> (TF) to 2,243.98 g dw m<sup>-2</sup> (HM5). Belowground biomass at low marsh sites averaged 397.83 g dw m<sup>-2</sup>, compared to 1,686.54 g dw m<sup>-2</sup> at high marsh sites, with 329.11 g dw m<sup>-2</sup> at restoration area sites.

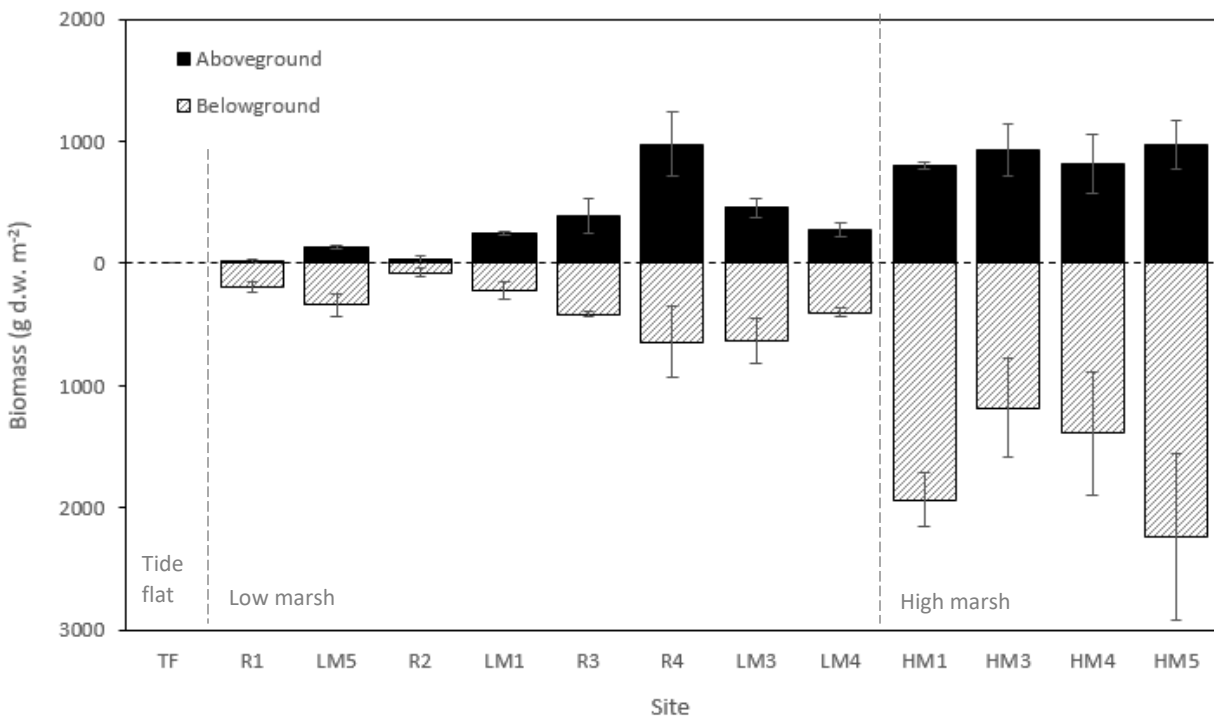


Figure 2. Aboveground and belowground biomass by site, with sites arranged in order of increasing elevation. The restored site (R1, R2, R3, and R4) elevations were within the range of other low marsh sites. Error bars represent standard error.

### Sediment characteristics

The mean sediment bulk density observed at both the natural (undisturbed) and restored marsh sites was identical, at 0.92 g cm<sup>-3</sup>, lower than the tidal flat site bulk density of 1.48 g cm<sup>-3</sup> (Table 1). However, patterns in organic matter content, organic carbon content, and carbon density differed across the three site categories, with restored site sediment characteristics being intermediate between those from natural marsh and tidal flat sites. Organic matter content averaged 6.45% at natural marsh sites, 4.93% at restored sites, and 2.70% at the tidal flat site. Carbon content averaged 2.38% at natural marsh sites,

1.74% at restoration sites, and 0.70% at the tidal flat site. Carbon density averaged  $0.020 \text{ g C cm}^{-3}$ ,  $0.015 \text{ g C cm}^{-3}$ , and  $0.010 \text{ g C cm}^{-3}$ , respectively.

The percentage of fine grained sediments, as averaged over the top 30 cm, was lowest (23.46%) at the tidal flat site (TF), and highest (98.60%) at high marsh site HM4. Although the percentage of fines was more variable across the natural marsh sites, the mean from both natural marsh sites and restored sites was approximately equal, with 76.02% and 76.66% fines, respectively (Table 1). As expected, sediment grain size was a significant predictor of organic content ( $R^2 = 0.45$ ,  $p < 0.001$ ,  $n = 154$ ). However, the percentage of fine grained sediments best predicted the minimum organic content rather than the full range of observed values at any given site (Appendix A, Figure 8).

Table 1. Site elevation, marsh type, porewater salinity, and sediment characteristics. Sediment characteristics (bulk density, organic content, carbon content, carbon density, and grain size) are reported as the average value across the top 30 cm.

| Site                  | Marsh type  | Elevation (m MLLW) | Porewater salinity (ppt) | Bulk density ( $\text{g cm}^{-3}$ ) | Organic content (% by weight) | Carbon content (% by weight) | Carbon density ( $\text{g C cm}^{-3}$ ) | Grain size (% fines) |
|-----------------------|-------------|--------------------|--------------------------|-------------------------------------|-------------------------------|------------------------------|---|----------------------|
| <i>Natural marsh</i>  |             |                    |                          |                                     |                               |                              |   |                      |
| HM5                   | High        | 3.34               | –                        | 0.92                                | 7.60                          | 2.90                         | 0.026                                   | 85.77                |
| LM5                   | Low         | 2.48               | 20                       | 1.19                                | 4.39                          | 1.44                         | 0.016                                   | 66.65                |
| HM1                   | High        | 3.04               | 6                        | 0.82                                | 7.18                          | 2.73                         | 0.020                                   | 68.50                |
| LM1                   | Low         | 2.65               | 12                       | 1.36                                | 3.26                          | 0.94                         | 0.012                                   | 42.88                |
| HM3                   | High        | 3.24               | 20                       | 0.84                                | 6.70                          | 2.49                         | 0.020                                   | 81.62                |
| LM3                   | Low         | 2.86               | 20                       | 0.76                                | 7.11                          | 2.67                         | 0.020                                   | 67.10                |
| HM4                   | High        | 3.27               | –                        | 0.76                                | 7.86                          | 3.02                         | 0.023                                   | 98.60                |
| LM4                   | Low         | 2.86               | 15                       | 0.72                                | 7.49                          | 2.85                         | 0.020                                   | 97.06                |
|                       | <b>Mean</b> | <b>2.97</b>        | <b>15.5</b>              | <b>0.92</b>                         | <b>6.45</b>                   | <b>2.38</b>                  | <b>0.020</b>                            | <b>76.02</b>         |
|                       | SE          | 0.11               | 2.3                      | 0.08                                | 0.60                          | 0.27                         | 0.00                                    | 6.55                 |
| <i>Unveg tideflat</i> |             |                    |                          |                                     |                               |                              |   |                      |
| TF                    |             | <b>2.15</b>        | –                        | <b>1.48</b>                         | <b>2.70</b>                   | <b>0.70</b>                  | <b>0.010</b>                            | <b>23.46</b>         |
| <i>Restored marsh</i> |             |                    |                          |                                     |                               |                              |   |                      |
| R1                    | Low         | 2.40               | 17                       | 1.10                                | 3.99                          | 1.26                         | 0.013                                   | 78.22                |
| R2                    | Low         | 2.50               | 19                       | 0.83                                | 5.56                          | 1.96                         | 0.016                                   | 83.90                |
| R3                    | Low         | 2.76               | 15                       | 0.82                                | 5.57                          | 2.20                         | 0.016                                   | 73.44                |
| R4                    | Low         | 2.85               | 11                       | 0.92                                | 4.60                          | 1.53                         | 0.014                                   | 71.08                |
|                       | <b>Mean</b> | <b>2.62</b>        | <b>15.5</b>              | <b>0.92</b>                         | <b>4.93</b>                   | <b>1.74</b>                  | <b>0.015</b>                            | <b>76.66</b>         |
|                       | SE          | 0.11               | 1.7                      | 0.06                                | 0.39                          | 0.21                         | 0.001                                   | 2.83                 |

## Elevation change rates

Rates of surface elevation change ranged from  $-0.03 \text{ cm yr}^{-1}$  (TF) to  $3.79 \text{ cm yr}^{-1}$  (R2) (Table 2). The mean rate across the eight natural marsh sites was  $1.03 \text{ cm yr}^{-1}$  (Figure 3), while the mean rate across the four restored sites was nearly three times higher at  $2.74 \text{ cm yr}^{-1}$ .

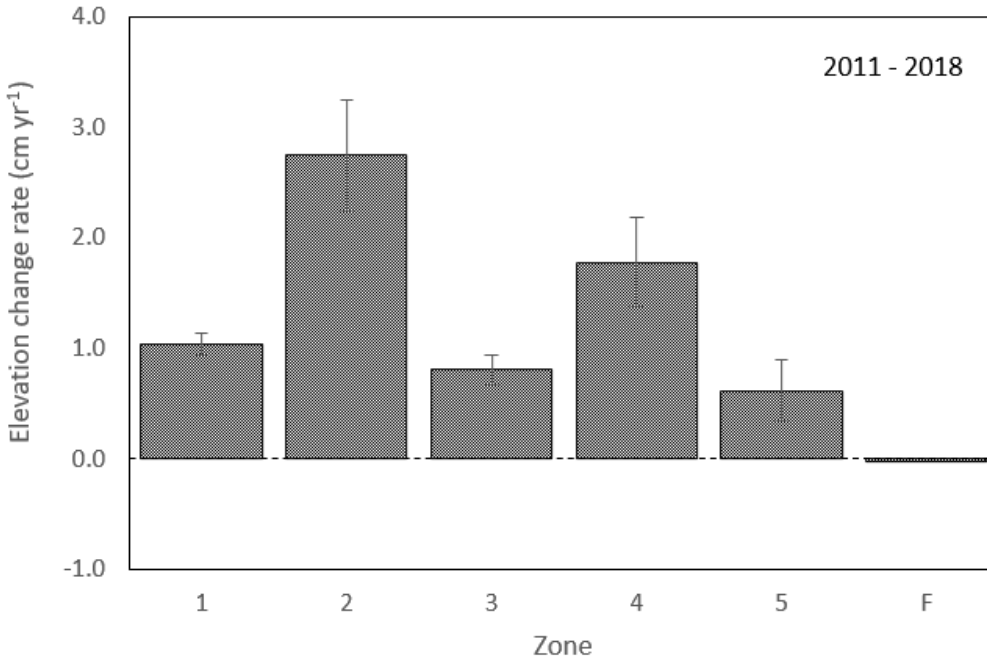


Figure 3. Mean annual elevation change as measured by SETs, by zone, for the entire period of record (2011 – 2018) with standard error bars. Zone 2 is the restored area, Zone F is the tidal flat site, and the remaining Zones 1, 3, 4, and 5 are natural marsh zones (see Figure 1 for a map showing the location of all zones).

Although some sites showed elevation loss during certain years within the seven-year period of record, the overall pattern was that of elevation gain in all but the tidal flat zone (Figure 4).

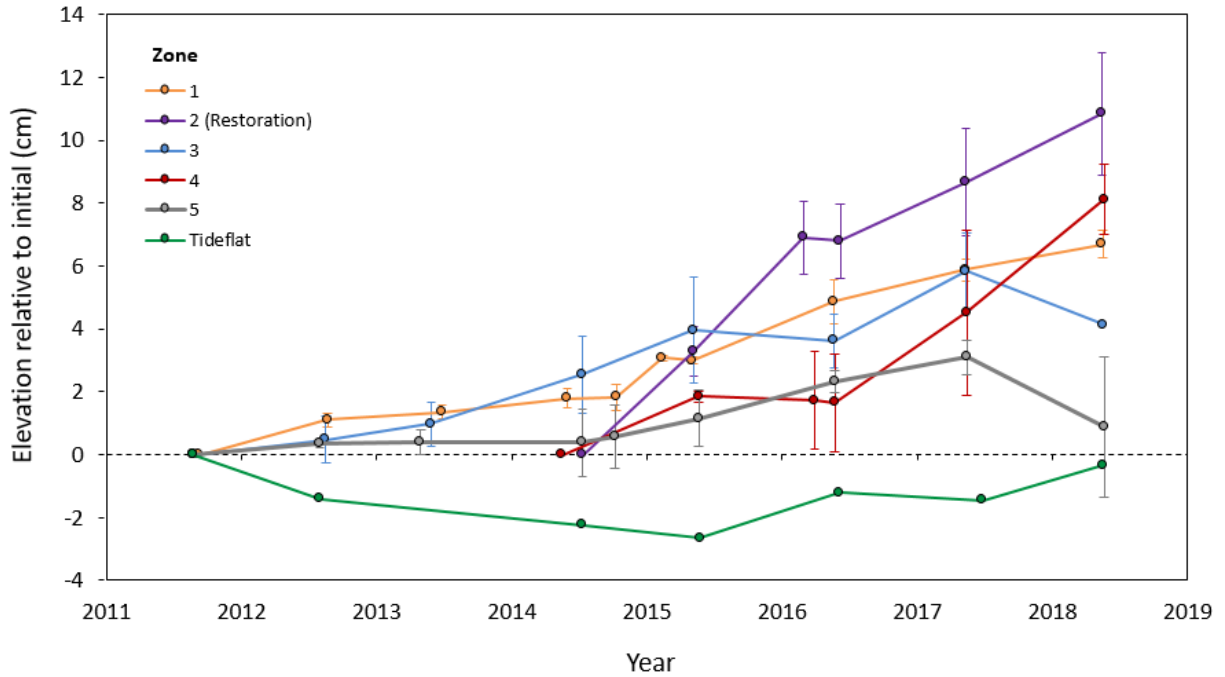


Figure 4. Time series of mean elevation change by zone, as measured by SETs. Most zones contain four SETs, with the exception of the tidal flat, which contains one SET. Error bars represent standard error.

### Long-term accretion rates

Long-term accretion rates determined from radioisotope dating ranged from 0 cm yr<sup>-1</sup> (TF) to 2.17 cm yr<sup>-1</sup> (R2) (Table 1). The mean accretion rate at natural marsh sites was 0.60 cm yr<sup>-1</sup>, while the mean rate at restored sites was 1.57 cm yr<sup>-1</sup>. The restoration site <sup>210</sup>Pb profiles appeared disturbed, with discontinuous layers and a mixed surface layer corresponding to the most recent post-restoration period (Appendix B), therefore the mean <sup>210</sup>Pb:SET ratio of 0.57 was applied to SET-based rates at these four sites to estimate the <sup>210</sup>Pb accretion rate.

Table 2. Rates of elevation change, accretion, and carbon accumulation, and carbon stocks across the three carbon pools.

| Site                  | SET elevation change rate (cm yr <sup>-1</sup> ) | <sup>210</sup> Pb accretion rate (cm yr <sup>-1</sup> ) | SET-based C accumulation rate (g C m <sup>-2</sup> yr <sup>-1</sup> ) | <sup>210</sup> Pb-based C accumulation rate (g C m <sup>-2</sup> yr <sup>-1</sup> ) | Sediment C stock in top 30 cm (kg C m <sup>-2</sup> ) | Aboveground biomass C stock (kg C m <sup>-2</sup> ) | Belowground biomass C stock (kg C m <sup>-2</sup> ) |
|-----------------------|--|---|---|---|---|---|---|
| <i>Natural marsh</i>  |  |   |   |   |   |   |   |
| HM5                   | 0.67   | 0.34  | 177.53  | 90.09   | 7.95  | 0.37  | 0.85  |
| LM5                   | 0.38   | 0.24  | 61.63   | 39.45   | 4.93  | 0.05  | 0.13  |
| HM1                   | 0.88   | 0.28  | 176.81  | 56.26   | 6.03  | 0.31  | 0.74  |
| LM1                   | 1.20   | 0.30  | 148.63  | 37.31   | 3.73  | 0.09  | 0.08  |
| HM3                   | 0.73   | 0.60  | 147.88  | 121.55  | 6.08  | 0.35  | 0.45  |
| LM3                   | 0.84   | 0.82  | 166.79  | 162.82  | 5.96  | 0.17  | 0.24  |
| HM4                   | 2.46   | 1.24  | 561.02  | 282.79  | 6.84  | 0.31  | 0.53  |
| LM4                   | 1.09   | 0.95  | 222.28  | 193.73  | 6.12  | 0.11  | 0.15  |
| <b>Mean</b>           | <b>1.03</b>                                      | <b>0.60</b>   | <b>207.82</b>   | <b>123.00</b>   | <b>5.95</b>   | <b>0.22</b>   | <b>0.40</b>   |
| SE                    | 0.22   | 0.13  | 52.95   | 30.47   | 0.44  | 0.05  | 0.10  |
| <i>Unveg tideflat</i> |  |   |   |   |   |   |   |
| TF                    | -0.03  | 0.00  | -2.95   | 0.00  | 2.95  | 0.00  | 0.00  |
| <i>Restored marsh</i> |  |   |   |   |   |   |   |
| R1                    | 2.98   | 1.71*   | 394.24  | 226.22  | 3.97  | 0.01  | 0.07  |
| R2                    | 3.79   | 2.17*   | 611.69  | 350.23  | 4.84  | 0.01  | 0.03  |
| R3                    | 1.37   | 0.78*   | 217.85  | 124.03  | 4.77  | 0.15  | 0.16  |
| R4                    | 2.82   | 1.61*   | 387.95  | 221.49  | 4.13  | 0.37  | 0.24  |
| <b>Mean</b>           | <b>2.74</b>                                      | <b>1.57</b>   | <b>402.93</b>   | <b>230.49</b>   | <b>4.43</b>   | <b>0.13</b>   | <b>0.13</b>   |
| SE                    | 0.50   | 0.29  | 80.69   | 46.34   | 0.22  | 0.09  | 0.05  |

\* <sup>210</sup>Pb accretion rates at restored marsh sites were estimated with a <sup>210</sup>Pb:SET rate conversion (see Methods section for further description).

### Carbon stocks and sequestration rates

Sediment carbon stocks in the top 30 cm ranged from 2.95 kg C m<sup>-2</sup> (TF) to 7.95 kg C m<sup>-2</sup> (HM5) (Table 1). As with other sediment characteristics, the restored site carbon stocks were intermediate between the tidal flat site and the natural marsh sites, averaging 4.43 kg C m<sup>-2</sup>, 2.95 kg C m<sup>-2</sup>, and 5.95 kg C m<sup>-2</sup>, respectively. Sediment carbon stocks were approximately 30 times greater on average than aboveground and belowground carbon stocks (Figure 5, Table 1). Aboveground biomass carbon stocks ranged from 0 kg C m<sup>-2</sup> (TF) to 0.37 kg C m<sup>-2</sup> (HM5 and R4), averaging 0.22 kg C m<sup>-2</sup> at the natural marsh sites, and 0.13 kg C m<sup>-2</sup> at the restored sites. Belowground biomass carbon stocks ranged from 0 kg C m<sup>-2</sup> (TF) to 0.85 kg C m<sup>-2</sup> (HM5), averaging 0.40 kg C m<sup>-2</sup> at natural marsh sites and 0.13 kg C m<sup>-2</sup> at restored sites.

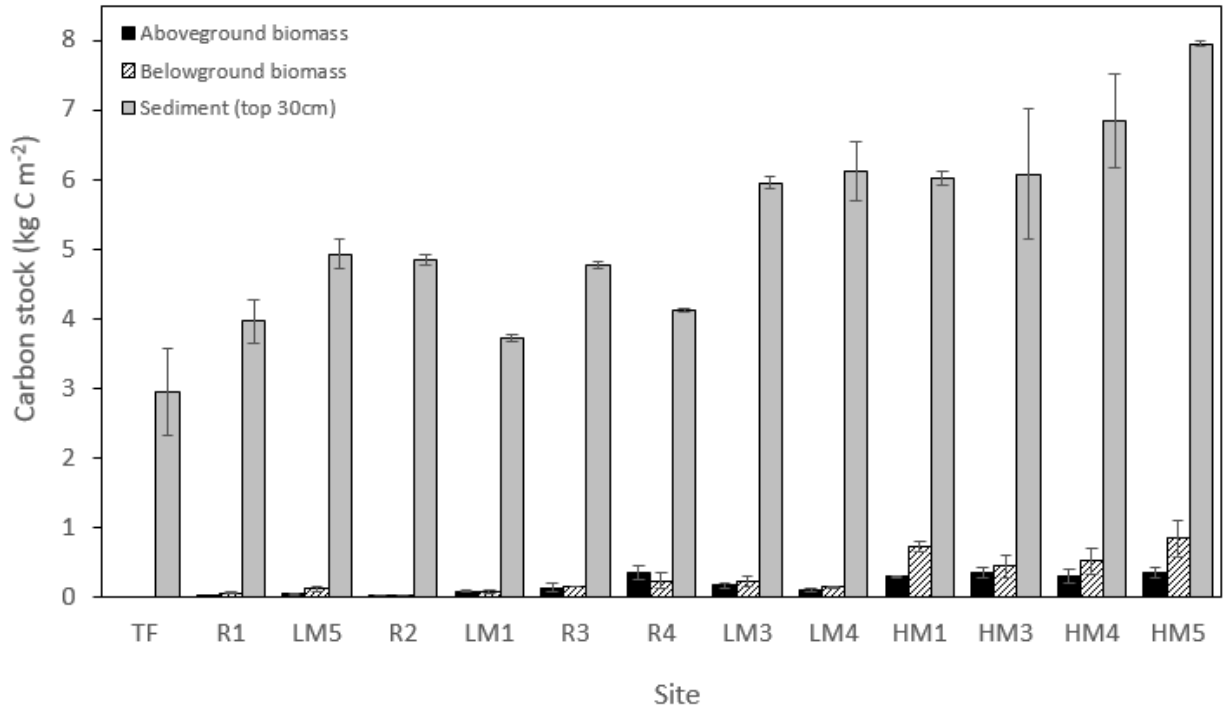


Figure 5. Carbon stocks in aboveground biomass, belowground biomass, and the top 30 cm of sediment at each site, with standard error bars. Sites are arranged in order of increasing elevation.

Carbon accumulation rates calculated with <sup>210</sup>Pb-based accretion rates ranged from 0 g C m<sup>-2</sup> yr<sup>-1</sup> (TF) to 350.23 g C m<sup>-2</sup> yr<sup>-1</sup> (R2), averaging 123.00 g C m<sup>-2</sup> yr<sup>-1</sup> at natural marsh sites and 230.49 g C m<sup>-2</sup> yr<sup>-1</sup> at restored sites. Carbon accumulation rates calculated with SET-based elevation change rates ranged from -2.95 g C m<sup>-2</sup> yr<sup>-1</sup> (TF) to 611.69 g C m<sup>-2</sup> yr<sup>-1</sup> (R2), with an average of 207.82 g C m<sup>-2</sup> yr<sup>-1</sup> at natural marsh sites and 402.93 g C m<sup>-2</sup> yr<sup>-1</sup> at restored sites (Figure 6, Table 1).

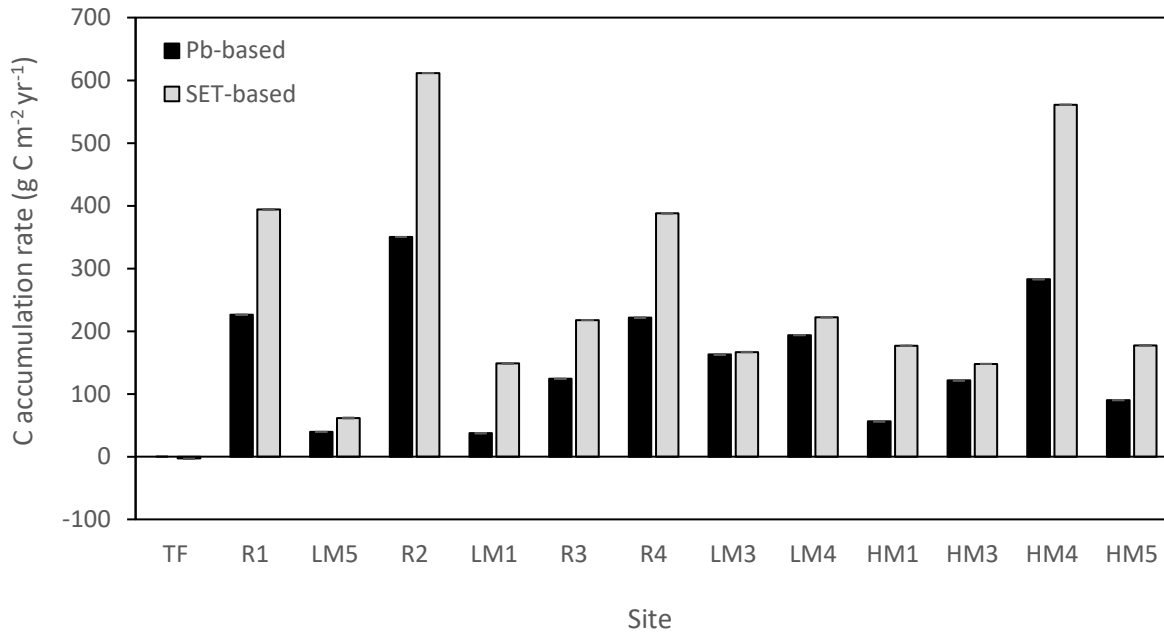


Figure 6. Carbon accumulation rates at each site, with both <sup>210</sup>Pb-based and SET-based rates shown. Sites are arranged in order of increasing elevation.

### Environmental characteristics

Porewater salinity, as measured at the end of the growing season in mid-August, ranged from 6 ppt (at HM1 adjacent to Hat Slough) to 20 ppt (at several sites), and averaged 15.5 ppt within both natural marsh sites and restored sites (Table 1).

Site elevations ranged from 2.15 m MLLW (TF) to 3.34 m MLLW (HM5). Undisturbed low marsh sites ranged from 2.48 to 2.71 m MLLW, while high marsh sites were approximately 0.5 m higher, ranging from 3.04 to 3.34 m MLLW. Elevations within the restoration area fell within the range of undisturbed low marsh elevations, ranging from 2.40 to 2.85 m MLLW (Table 1).

### Drivers of plant biomass, carbon stocks, and sequestration rates

Plant biomass appears driven more by elevation than salinity. A multiple linear regression with both elevation and salinity revealed that only elevation significantly predicted aboveground biomass (elevation:  $p = 0.003$ ; salinity:  $p = 0.30$ ). Among the natural marsh sites, a one-way ANOVA determined that both aboveground and belowground biomass were significantly greater at high marsh sites compared to low marsh sites (Aboveground:  $F_{1,6} = 56.6$ ,  $p < 0.001$ ; Belowground:  $F_{1,6} = 24.6$ ,  $p = 0.003$ ).

Sediment carbon stocks could be significantly predicted by elevation, aboveground biomass, and belowground biomass, but elevation was the strongest predictor ( $R^2 = 0.76$ ,  $p < 0.001$ ,  $n = 13$ ). Carbon accumulation rates, however, could only be predicted by variation in accretion rates ( $R^2 = 90$ ,  $p < 0.001$ ,  $n = 13$ ). No significant relationship was detected between carbon accumulation rates (both SET-based and <sup>210</sup>Pb-based) and elevation, biomass, or even sediment carbon stocks.

## Discussion

### Plant biomass

The restored sites varied substantially in terms of both aboveground and belowground biomass for reasons discussed below. Excluding the restored sites, aboveground biomass at the high marsh sites was approximately three times greater than at low marsh sites. Belowground biomass showed even more of a contrast, with over four times greater biomass at high marsh sites. This difference in low and high marsh biomass can be attributed to higher density plant communities in the high marsh, but also (and perhaps more importantly) flooding stress which limits root development in the low marsh.

Restored site R4 had a higher aboveground biomass than other low marsh sites. This was the southernmost site in the restoration area where vegetation was dominated by cattails (*Typha latifolia* and *T. angustifolia*) and maritime bulrush (*Bolboschoenus maritimus*), both relatively productive species. Cattails were not found at any other sites across the estuary. Their prevalence in the restoration area is likely a result of their occurrence across the site's freshwater wetlands prior to restoration, and their ability to persist after restoration despite suboptimal conditions (Fuller 2017). Restored sites R1 and R2 contained very little aboveground biomass. These two sites were located near the northern dike breach where a developing sand splay appeared to be reducing the area's capacity for permanent vegetation establishment.

### Sediment characteristics

Sediment profiles at all four restored sites showed a spike in organic material between 15 and 20 cm in depth. The cause of this spike is uncertain but it is hypothesized to be from either pre-restoration standing plant material that was quickly buried after the dike breach (or trampled as part of the levee removal process), or a layer of material originating outside the site that was deposited during a storm event. The depth of this layer reasonably corresponds to the depth of sediment we expect to have accumulated after the dike breach, based on SET elevation change rates from the last four years.

All other undisturbed sites (beyond the restoration area) generally exhibited typical sediment profile patterns with highest percent organic matter and carbon at the surface, and a decrease with depth due to organic material decomposition and diminishing root growth. Low marsh and tidal flat sites tended to have lower levels of carbon at the surface, but they still showed a decrease with depth. Bulk density was generally lowest at the surface, as expected, and increased with depth due to compaction and the decomposition of less dense organic matter.

The restoration area sites, although low in elevation, had a mean sediment bulk density comparable to that of the high marsh sites. Despite the restoration and natural marsh sites having a similar mean bulk density and percentage of fine-grained sediments, organic content, carbon content, and carbon density were all lower in the restoration area. Because the restoration area has experienced very rapid sediment deposition since the dike breaches in 2012, plant material has had little opportunity to develop and influence the sediment profile. However, even the sites with little or no plant material (TF, R1, and R2) still contained a small amount of sediment carbon, indicating that some carbon is likely allochthonous, presumably deposited with the incoming sediment independent of the *in situ* vegetation. Similarly, Wollenberg et al. (2018) measured a 2.3% carbon content in newly deposited sediment in the Bay of



Fundy, concluding that short-term soil C density was driven by the inherent C content of incoming sediment.

Sediment organic matter content, carbon content, and carbon density across the Stillaguamish River estuary were slightly lower than those reported for the Snohomish River estuary near Everett, WA (Crooks et al. 2014). For example, carbon density at six Snohomish tidal marsh sites ranged from 0.018 g C cm<sup>-3</sup> to 0.32 g C cm<sup>-3</sup>, whereas Stillaguamish marsh carbon density ranged from 0.12 g C cm<sup>-3</sup> to 0.26 g C cm<sup>-3</sup>. Bulk density was generally higher across the Stillaguamish River estuary compared to the Snohomish River estuary. Although these two neighboring estuaries are only separated by approximately 20 miles, the most likely explanation for the difference in sediment characteristics is their differing hydrodynamics. The Stillaguamish estuary marshes are located at the northern end of Port Susan Bay with full exposure to waves from the south. In contrast, the Snohomish estuary marshes are situated further inland, where they are more protected from storms, allowing finer sediments to settle. In addition, the Snohomish estuary sites are situated more directly adjacent to the river, providing an accessible supply of fine riverine sediments and organic material. Having blue carbon data from both estuaries provides a window into the variability we might expect to see across the Salish Sea, from more river-dominated estuaries to more ocean-dominated estuaries.

### **Accretion and elevation change rates**

We originally predicted that SET-based elevation change rates and <sup>210</sup>Pb-based accretion rates would be comparable because they measure similar processes of sediment deposition and shallow subsidence. However, we found the <sup>210</sup>Pb accretion rates to be consistently lower than SET rates, with an average <sup>210</sup>Pb:SET ratio of 0.57 across the eight natural marsh sites. One possible explanation for this difference is that the <sup>210</sup>Pb method does not measure the most recent depositional layer with as much precision as the SET method, providing only one data point over the top 2 cm. This may result in the most recently deposited sediment not being fully accounted for, which would reduce the reported rate. Additionally, this surface layer may be compacted somewhat during the coring process. Another possible reason for the discrepancy is that SETs have a shorter length of record. Very few other studies have used both methods at the same sites to allow for a comparison, but Breithaupt et al. (2018) also found that SETs recorded a higher rate of accretion than that measured by <sup>210</sup>Pb. Our results indicate that the two methods cannot be used interchangeably in the Stillaguamish estuary, although one rate may potentially be predicted from the other with this site-specific ratio.

SETs in Zones 1 and 3 have seen a fairly steady increase in elevation over their 7-year period of record. These zones are nearest to the Stillaguamish River mouth and riverine sediment supply. Previously at Zones 4 and 5 (farthest from the river), sediment starvation and wave-induced erosion were believed to be deteriorating the low marsh and causing the high marsh to retreat. However, these areas have seen very dynamic conditions in the most recent years, with pools and channels encroaching in one year, and an influx of large woody debris and sediment deposits in the next. Although these dynamic conditions lead to more intermittent deposition rather than steady and predictable accretion, they appear to be slowing erosion in these areas.

The restored site SETs have shown 2.74 cm yr<sup>-1</sup> of elevation change on average, over their four-year period of record. This rate is nearly three times that of the other undisturbed marsh sites. While it was expected and desired that the return of tidal and riverine pulsing to this formerly hydrologically isolated

and subsided farmland would result in enhanced sedimentation, the rates observed here are remarkably high. However, it is expected that as infill continues and the elevation inside the restoration zone approaches that of the adjacent undisturbed marshes, rates of elevation change will gradually decline. These initial high rates demonstrate that the restored marsh has the capacity to restore its elevation without additional human intervention, and this is likely to apply to other Puget Sound estuaries as well.

All sites, with the exception of the tidal flat site, are accumulating sediments and gaining elevation at a rate that exceeds current and even some predicted future rates of sea level rise, indicating that the estuary is not currently sediment-limited.

### **Carbon stocks and sequestration rates**

It appears that carbon stocks and sequestration rates are driven by different processes in the Stillaguamish River estuary. In other words, a high carbon stock does not necessarily imply a high carbon sequestration rate. Carbon stocks were highest at high marsh sites, whereas sequestration rates at those same sites were moderate. Stocks appear driven primarily by elevation (as a proxy for tidal inundation frequency), whereas sequestration rates appear driven primarily by sediment accretion rates (similar to that observed by Wollenberg et al. 2018), which themselves are highly variable and dependent on more complex sediment dynamics, at least here in the Stillaguamish River estuary.

A few studies have reported C accumulation rates calculated with SET elevation change rates (e.g., Howe et al. 2009, Rogers et al. 2013, Lovelock et al. 2014) but most use radiometric ( $^{210}\text{Pb}$  or  $^{137}\text{Cs}$ ) accretion rates. We have reported both rates here to allow for comparisons with both types of studies. SET-based C accumulation rates here were all higher, and on average were nearly twice the  $^{210}\text{Pb}$ -based C accumulation rates, so it is important to be deliberate about which type of rate is used for further analyses. But considering that most studies use  $^{210}\text{Pb}$ -based C accumulation rates, the rest of our discussion emphasizes those rates.

Global average rates of salt marsh carbon accumulation are generally higher than those reported here for Stillaguamish natural marshes. For example, Mcleod et al. (2011) reported a global average rate of  $218 \text{ g C m}^{-2} \text{ yr}^{-1}$ . A more recent review by Ouyang and Lee (2014) reported a global average of  $244.7 \text{ g C m}^{-2} \text{ yr}^{-1}$ , but a lower NE Pacific regional average (based on eight California sites) of  $173.6 \text{ g C m}^{-2} \text{ yr}^{-1}$ . However, some of the studies included in those global reviews used shorter-term accretion methods, which overestimate long-term accretion (Callaway et al. 2012). Other studies along the west coast of North America report natural marsh rates more comparable to ours (all using  $^{210}\text{Pb}$  dating). Callaway et al. (2012) estimated  $79 \text{ g C m}^{-2} \text{ yr}^{-1}$  from San Francisco Bay marshes. Brophy et al. reported a  $85 \text{ g C m}^{-2} \text{ yr}^{-1}$  average from the northern Oregon coast (Brophy et al. 2018), and  $79 \text{ g C m}^{-2} \text{ yr}^{-1}$  in the lower Columbia River estuary (Brophy et al. 2017). In British Columbia, Chastain et al. (2018) reported an average of  $146 \text{ g C m}^{-2} \text{ yr}^{-1}$ . Finally, Crooks et al. (2014) reported  $110 \text{ g C m}^{-2} \text{ yr}^{-1}$  from a Snohomish estuary natural marsh site, similar to the Stillaguamish average of  $123 \text{ g C m}^{-2} \text{ yr}^{-1}$ .

Few studies report carbon sequestration rates following marsh restoration. Burden et al. (2019) reported a similar pattern to ours in eastern England using a chronosequence of marsh ages since restoration, finding a high initial rate which declined at older sites. Their rates, however, were lower than those in the Stillaguamish River estuary, with a high of  $104 \text{ g C m}^{-2} \text{ yr}^{-1}$  16 years after restoration, and a low of  $65 \text{ g C m}^{-2} \text{ yr}^{-1}$  at a site that was 114 years post-restoration. In contrast, Wollenberg et al. (2018) studied a restored marsh in the Bay of Fundy, and found much higher initial C accumulation

rates, averaging  $1329 \text{ g C m}^{-2} \text{ yr}^{-1}$  six years post-restoration, which was more than five times higher than their  $259 \text{ g C m}^{-2} \text{ yr}^{-1}$  reported from a nearby mature marsh. Their elevation change measurement methods were similar to our SET methods, but their average rate of  $1329 \text{ g C m}^{-2} \text{ yr}^{-1}$  is still three times higher than our average SET-based rate of  $402.93 \text{ g C m}^{-2} \text{ yr}^{-1}$  from the Stillaguamish estuary restoration area. In Oregon, Brophy et al. reported  $80 \text{ g C m}^{-2} \text{ yr}^{-1}$  in Columbia River estuary restored sites (Brophy et al. 2017), and  $150 \text{ g C m}^{-2} \text{ yr}^{-1}$  in Tillamook Bay restored sites (Brophy et al. 2018), both lower than Stillaguamish rates. Snohomish estuary rates, however, were similar to Stillaguamish rates, with up to  $352 \text{ g C m}^{-2} \text{ yr}^{-1}$  based on  $1.61 \text{ cm yr}^{-1}$  of accretion (Crooks et al. 2014).

### **Implications for carbon finance and restoration**

Since the Stillaguamish River estuary restoration site is 0.5 to 1.0 meters below the elevation of adjacent natural marshes, it is expected to continue accumulating sediments until it reaches that elevation. Assuming a constant sediment carbon density (which is likely conservative), the entire 60-hectare restoration site is expected to accumulate approximately 4,500 to 9,000 Tonnes of carbon, which is equivalent to removing approximately 3,500 to 7,000 cars from the roads for one year. That amount of carbon is worth approximately \$65,000 to \$165,000 using a national average carbon offset price of \$4 to \$5 per Tonne  $\text{CO}_2$ . Once its elevation reaches an equilibrium with the surrounding marsh, the site is expected to continue accumulating carbon at a rate similar to that of the surrounding marsh, estimated at  $1.23 \text{ Tonnes C ha}^{-1} \text{ yr}^{-1}$ , or approximately  $75 \text{ Tonnes C yr}^{-1}$  within the entire restoration site. This C accumulation rate is valued at approximately  $\$20 \text{ ha}^{-1} \text{ yr}^{-1}$ , or  $\$1,200 \text{ yr}^{-1}$  within the entire restoration site.

Unfortunately the national carbon offset value is currently quite low compared to the estimated social cost of carbon of \$41 per Tonne (Pendleton et al. 2012). There is some indication that the carbon offset value awarded to blue carbon projects may increase, in light of the many other co-benefits that marsh restoration provides, such as coastal storm protection, habitat restoration, and fisheries enhancement. But until then, carbon credits alone are unlikely to fund an entire restoration project. However, although the monetary value of carbon sequestration is currently low, there are other potential benefits to recognizing and quantifying the climate benefits of marsh restoration:

- The climate benefits of restoration may garner interest from additional funders seeking to support a project with a climate change mitigation component.
- Annual carbon offset payments may help fund monitoring efforts, which are typically difficult to fund.
- Blue carbon adds to a long list of other marsh restoration benefits, increasing justification for restoration.

Tidal wetland restoration projects tend to be relatively small, losing cost-effectiveness once the carbon quantification, feasibility, design, and monitoring are accounted for. Fortunately, it is possible to lump smaller carbon finance projects together within a larger project area to increase cost-effectiveness. Larger project areas could be at the scale of an estuary or even a region, and the individual restorations need not be implemented at the same time. This possibility should be an important consideration for tidal wetland restoration projects.

## Recommended next steps

This study substantially expands the amount of PNW blue carbon data available to scientists, managers, and policymakers. No other blue carbon study in the Pacific Northwest has included carbon sequestration rates from both  $^{210}\text{Pb}$  and SET measurements, in addition to carbon stocks within all three marsh carbon pools (i.e., sediment, aboveground biomass, and belowground biomass). The extensive dataset generated by this study can contribute to a feasibility analysis that determines if the site would be appropriate for the carbon market, using the Port Susan Bay Preserve restoration site as an example for other potential projects within the region. This study will also allow for comparisons of carbon stocks and sequestration rates across estuaries, which until only recently were not possible due to a lack of data within the region.

Some further research is recommended that was outside the scope of this study, to allow for a full feasibility analysis of the estuary's carbon finance potential:

- Greenhouse gas ( $\text{CO}_2$ ,  $\text{CH}_4$ , and  $\text{N}_2\text{O}$ ) emissions must be quantified for a complete blue carbon assessment. Even with substantial carbon sequestration, the estuary's brackish marshes could simultaneously be emitting greenhouse gases, and it is yet unknown whether the estuary is a net greenhouse gas sink or source. It is possible to use published emission data from other estuaries, but published methane ( $\text{CH}_4$ ) emissions can be quite variable, particularly from brackish marshes such as those in the Stillaguamish estuary, therefore site-specific data are preferred.
- Carbon offset credits are awarded based on net autochthonous carbon accumulation (carbon produced *in situ*). Allochthonous carbon (carbon produced outside the system) must be subtracted from the carbon balance for crediting purposes. Further analysis using stable isotopes would allow for a differentiation between autochthonous and allochthonous carbon to provide this information.

We recommend continuing to monitor elevation change rates and soil carbon within the Stillaguamish estuary restoration site, to determine the length of time needed for it to match the elevation and the soil profile of the adjacent undisturbed marshes. Current rates of accretion are unlikely to remain constant, and there are a paucity of other studies that have tracked this progression.

Both the Stillaguamish and Snohomish estuaries have very high rates of accretion (and consequently carbon accumulation) immediately following restoration, higher than rates reported from Oregon estuaries (Brophy et al. 2017, 2018). It would be helpful to expand this assessment to restored marshes in other PNW estuaries to determine whether this high capacity for accretion is estuary-specific or region-specific.

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## Appendix – Supplemental Figures and Tables

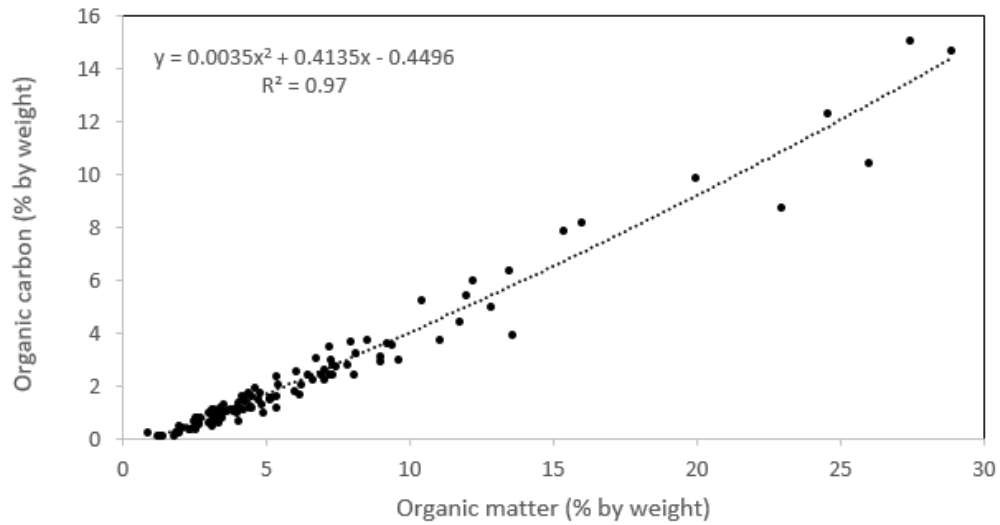


Figure 7. Relationship between organic matter and organic carbon content, based on a subset of 124 sediment core sections.

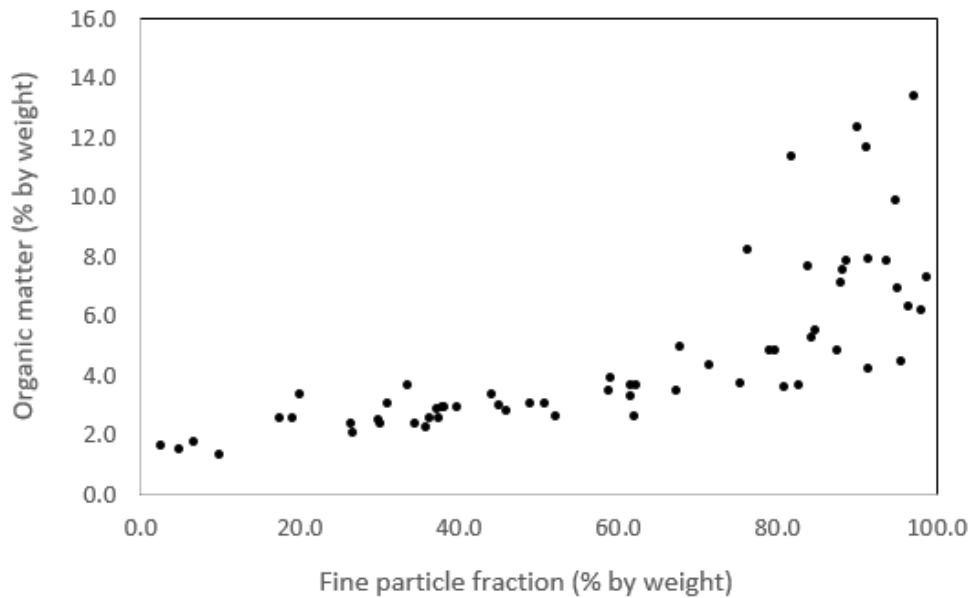


Figure 8. Relationship between the percentage of fine-grained particles and sediment organic content, based on a subset of 154 sediment core sections.