COASTAL BLUE CARBON OPPORTUNITY ASSESSMENT FOR THE SNOHOMISH ESTUARY

THE CLIMATE BENEFITS OF ESTUARY RESTORATION

RESTORE AMERICA'S ESTUARIES

ESA

WESTERN WASHINGTON UNIVERSITY

EARTH CORPS

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Cover Photo Caption and Credit

Photo of Quilceda marsh, currently owned by The Tulalip Tribes, looking southwest down Steamboat Slough of the Snohomish River toward Port Gardner, Washington, USA. Photo by K. O'Connell (2013).

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EXECUTIVE SUMMARY

Background

Blue carbon is a term that recognizes the role of coastal wetlands in the global carbon cycle. Tidal marshes, tidal forested wetlands, and seagrasses sequester carbon dioxide from the atmosphere continuously over thousands of years, building stocks of carbon in organic-rich soils. When coastal wetlands are drained and converted to terrestrial land uses, carbon is rapidly released back to the atmosphere in the form of carbon dioxide. Restoring coastal wetlands stops the drainage-induced releases of carbon and reactivates carbon sequestration.

Coastal wetlands are one of the world's most rapidly disappearing ecosystems due to human impact (Pendleton et al., 2012). At current rates, within less than 100 years most of the world's coastal wetlands will be lost. Though wetlands are protected by law in the United States, losses continue. Restoration of coastal ecosystems brings benefits that support the livelihoods of local communities, improve water quality, reduce risk of flooding, facilitate future adaptation to climate change, and not only reduce greenhouse gas (GHG) emissions from converted wetlands but reverse emissions from that land.

The need for improved management of coastal wetlands for climate change mitigation benefits is recognized and advanced by a number of important recent actions:

- In November 2012 the **Verified Carbon Standard** recognized *Wetland Restoration and Conservation* as an eligible project activity for carbon finance[1]. In December 2013 the first global *Methodology for Tidal Wetlands and Seagrass Restoration* was submitted to the Verified Carbon Standard for review[2]. Once approved, there will be mechanisms for coastal wetlands restoration projects in the U.S. and internationally to apply for carbon financing.

- In October 2013, **the Intergovernmental Panel on Climate Change (IPCC)** adopted the *2013 Supplement to the 2006 Guidelines for National Greenhouse Gas Inventories: Wetlands (Wetlands Supplement)*[3]. This document fills a gap in the 2006 Agriculture, Forestry and other Land Use (AFOLU) GHG guidelines to cover wetlands and organic soils. In November 2013, at the Conference of Parties in Warsaw (COP 19), the Subsidiary Body for Scientific and Technological Advice (SBSTA) invited Parties (Nations) to apply the Wetlands Supplement in developing national GHG inventories and report back to the SBSTA in 2017 on their experiences in application[4].

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[1] http://www.v-c-s.org/wetlands_restoration_conservation

[2] https://www.estuaries.org/draft-greenhouse-gas-methodology-for-wetland-restoration.html

[3] http://www.ipcc-nggip.iges.or.jp/home/wetlands.html

[4] http://unfccc.int/resource/docs/2013/sbsta/eng/l29.pdf

- In the **United States**, federal agencies have established an interagency team to support blue carbon efforts. These include integrating blue carbon science and policy into the National Ocean Policy and activities to develop tools and methodologies for blue carbon management. *The National Assessment of Ecosystem Carbon Sequestration and Greenhouse Gas Fluxes* recognizes that national estimates of GHG fluxes are lacking and that filling this data gap is a priority[5].

Purpose of the Report

The purpose of this report is to support the above actions by providing information to: (1) inform policy makers of the scale of GHG emissions and removals associated with management of coastal lowlands under conditions of climate change; and (2) identify information needs for future scientific investigation to improve quantification of GHG fluxes with coastal wetlands management.

The Snohomish Estuary was selected as a system representative of the wider Puget Sound and Pacific Northwest Region in terms of geomorphology, land use, and management issues. The historic estuary, the second largest in Puget Sound, consisted of a suite of forested wetlands, scrub-shrub wetlands, and emergent tidal wetlands. Clearing and draining the wetlands resulted in subsidence of organic soils. Today the subsided lands include agriculture (lowered water table), anthropogenic Palustrine wetlands (high water table), and a small area of planted forest. Soils are a mix of organic and mineral materials. The estuary hosts remnant emergent and forested wetlands; an example of a large-scale regenerating wetland, North Ebey Island, breached in the 1960s; and drained wetland soils under various forms of management.

We hypothesize that because of geomorphic form, sediment delivery, and the composition of emergent wetland communities, estuaries in the Pacific Northwest offer locations where restoring wetlands would be of relatively high resilience to sea level rise and act as effective sinks for carbon sequestration. The Snohomish Estuary has benefited from a number of scientific and engineering investigations to support coastal management planning and restoration activities.

Approach and Methods

This study provides a first assessment of carbon fluxes over multiple decades for historic drained and future restoring wetlands.

Soil carbon stocks were determined from cores collected at 12 sites across the Snohomish Estuary, representing emergent tidal wetland, forested wetland, a regenerating emergent wetland, and drained wetlands. Soil and carbon accumulation rates were determined by ^{210}Pb (lead) radiometric dating at five natural and restoring wetlands sites. Changes in living forest biomass, a significant component of the historic landscape, were derived from a prior regional assessment. In the absence of field data on

[5] http://www.usgs.gov/climate_landuse/land_carbon/default.asp

methane (CH_4) fluxes from wetland and drained soils, IPCC Tier 1 default values from the Wetlands Supplement were used.

The historic carbon emissions of drained wetlands were calculated from soil carbon density values obtained from field samples, combined with the volume of soil subsidence resulting from wetland drainage (derived from publically available digital elevation data). The total carbon sequestration potential for restored estuarine wetlands was calculated from changes in carbon stock when wetland soils rebuild up to elevations at which vegetation can colonize. Rebuilding of wetlands to typical elevations at Mean Higher High Water was assumed. A future condition was based upon a 1 m sea level rise.

Projections of wetland restoration in the estuary were based upon state plans for recovery of emergent wetlands. An assessment of restoration potential was based upon geomorphic metrics that determine the potential for diked and drained wetlands to rebuild once the dike is breached. These metrics were supported by observed wetland rebuilding at North Ebey Island, a restoration site in the Snohomish Estuary that has been breached for decades.

Findings

Carbon measurements in natural wetland areas support the hypothesis that this estuary is representative of typical West Coast estuaries. The Snohomish Estuary is an excellent case study for restoration of tidal wetlands and estimates of carbon storage along the northwest coast of the U.S. and southwest coast of Canada. This study found that restoring wetland sites show good potential for high rates of accretion and high rates of carbon storage.

Historic land use change resulted in estimated emissions of 4.5 million tons of carbon (MtC), of which 2.8 MtC was a result of clearing forested wetland (loss of living biomass) and 1.7 MtC from draining soils. Of the 4,749 ha of converted and drained wetlands, 1,353 ha are currently in planning or construction for restoration. These projects are anticipated to rebuild soil carbon stocks of 0.32 MtC as wetlands recover to former tidal elevations, and an additional 0.38 MtC with sea level rise of 1 m. Full estuary restoration would rebuild soil carbon stocks of 1.2 MtC as marshes build to emergent wetland tidal elevations, and a further 1.2 MtC as they accrete with sea level rise of 1 m. Any recovery of forest biomass would be additional to projected accumulation of soil carbon.

Rates of soil carbon accumulation in natural emergent and forested tidal wetlands in the Snohomish estuary are in balance with the rate of current rate of sea level rise. Carbon sequestration at two restoring sites ranged from 0.9 t C ha^{-1} yr^{-1} to 3.52 tC ha^{-1} yr^{-1}. These two restoration sites represent different ages of restoration; North Ebey Island has been breached for over 40 years, whereas Spencer Island has only been open for about 20 years. Spencer Island is lower than North Ebey Island by 30 cm. This difference in elevation is large in tidal wetland development. Spencer Island is at the low-end of plant colonization elevation, while North Ebey Island is at an excellent elevation for emergent plant colonization. As emergent plants colonize restoring tidal wetlands, soil accretion rates may increase as sediment becomes trapped by dense vegetation and rootmats. We anticipate that as restoring sites age in the Snohomish estuary they are likely to span these two values, with increasing soil carbon accumulation rates over time. The rate observed at North Ebey is likely more than sufficient to offset estimated CH_4 emissions. For example, using the IPCC Tier 1 default value for CH_4 emissions from non-saline tidal wetlands, 1.8 tC ha^{-1} yr^{-1} equivalent, the measured carbon accumulation of 3.5 tC ha^{-1} yr^{-1} suggests that this recovering system is a net GHG sink of 1.7 tC ha^{-1} yr^{-1}, in soils alone. Future restoration at other tidal wetland sites in the Snohomish estuary is expected to achieve similar climate change benefits.

This study did not evaluate the rates of carbon emissions from drained soils, CH_4 emissions from drainage ditches, or nitrous oxide (N_2O) emissions from drained soil surfaces as a result of organic decomposition or fertilizer application. As a consequence, the calculated net reduction in carbon emissions associated with restoration of wetlands underestimates the net benefit of halting ongoing emissions from drained soils, and as such is conservative.

In addition to this carbon storage potential, measurements at preserved, natural, undiked sites in the Snohomish Estuary show that the hydrology of the estuary is healthy and soils are building up (accreting) at rates equal to or exceeding rates of current estimated sea level rise. This healthy accretion rate means that this estuary has built-in resilience to sea level rise. Future tidal wetland habitat will be able to keep up with rising waters over the 100-year timeframe of this study, rather than changing to an inundated shallow bay.

Recommended Next Steps

1) Establish a regional blue carbon working group to build local capacity to deliver coordinated scientific findings, improve land management, and inform policy. Coastal lowlands of the Pacific Northwest offer the potential for coastal wetlands restoration with natural resilience to sea level rise and carbon sequestration benefits. A coordinated action is required to further explore this opportunity and test it through demonstration projects.

2) With a carbon finance methodology for tidal wetlands restoration submitted to the Verified Carbon Standard for review, the next step to delivering a carbon finance project is selection of a potential project site and detailed feasibility assessment. This report suggests estuaries similar to the Snohomish offer potential to host successful projects. Such projects could include elements of upper estuary forest and floodplain restoration as buffers to sea level rise and components of climate change adaptation, as well as emergent tidal wetland restoration.

3) Expand the geographical extent of this study, regionally and nationally. The approach developed in this study is readily transferable to other coastal lowland settings. There is a need for regional quantification of GHG emissions and reductions associated with coastal land use practice in this and other regions. This study provides a first step.

4) Conduct higher resolution research to quantify carbon emissions and removals associated with wetlands management to inform best management practices for state and national reporting. In particular, further work is required to: (1) determine CO_2, CH_4, and N_2O emissions from drained soils;

and (2) develop refined regional (IPCC Tier 2) default values for state and national GHG inventory efforts.

5) Develop regional landscape-level management plans that incorporate both climate change adaption and mitigation. Restoring tidal wetlands sooner rather than later will enable marsh building before sea level rises above the elevation at which emergent vegetation will colonize. Creating buffer zones into which marshes can migrate with sea level rise will support both sea level rise mitigation and adaptation. Coordinating climate change adaption and mitigation planning will improve project outcomes.

GLOSSARY AND ABBREVIATIONS

C - carbon

CCAP – Coastal Change Analysis Program

CH_3 – methane

cm - centimeter

cm^3 – cubic centimeter

CO_2 – carbon dioxide

DEM – Digital Elevation Model

DFIRM – Digital Flood Insurance Rate Map

Estuary –"that part of a river, stream, or other body of water having unimpaired connection with the open sea, where the sea water is measurably diluted with freshwater derived from land usage" (Armantrout, 1998).

ft - feet

FEMA – Federal Emergency Management Agency

g - gram

ha – hectare

kg - kilogram

m – meter

m^2 – square meter

m^3 – cubic meter

LiDAR – a combination of the words light and laser, LiDAR is a high resolution remote sensing technology, commonly used for topographical analysis of vegetation and the earth's surface, among many other diverse applications.

MHW – Mean High Water

MHHW - Mean Higher High Water

MLW – Mean Low Water

MLLW – Mean Lower Low Water

MSL – Mean Sea Level

MtC – Million metric tons of carbon

Mm^3 - Million cubic meters

NAVD88 - North American Vertical Datum 1988

NOAA – National Oceanic and Atmospheric Administration

Pg – Petagrams

Restored – Final state of restoration, at equilibrium

Restoring – In transition from former or degraded wetland to functional wetland

Tidal Wetland Habitat Zones

> **Emergent tidal wetland zone –** tidal wetland habitat dominated by low and high salt marsh vegetation (Hayman, Beamer, & McClure, 1996).
>
> **Scrub-shrub tidal wetland transition zone –** tidal wetland habitat that is a combination of estuarine scrub-shrub wetland and tidally influenced emergent wetland, transitioning between the saltmarsh-dominated area and forest-dominated wetlands (Hayman, Beamer, & McClure, 1996).
>
> **Forested tidal wetland zone –** tidal wetland habitat dominated by forested wetlands that are tidally influenced (Hayman, Beamer, & McClure, 1996).
>
> **Palustrine wetland –** "all nontidal wetlands dominated by trees, shrubs, persistent emergents, emergent mosses or lichens, and all such wetlands that occur in tidal areas where salinity due to ocean-derived salts is below 0.5 %." (USGS, 2013)

tC – metric tons carbon

WDFW – Washington Department of Fish and Wildlife

yr - year

UNIT CONVERSIONS

> **1 hectare (ha) = 2.47 acres (ac)**
>
> **100 g m^{-2} = 1 t ha^{-1}**
>
> **1 meter (m) = 3.28 feet (ft)**

EQUATIONS

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1.0 INTRODUCTION

Coastal wetlands are under pressure from land use change and sea level rise. Yet these important ecosystems are recognized for their values in underpinning coastal health and maintaining economic resources, such as fisheries. Recently, there has been growing awareness that the loss of coastal wetlands is also contributing to global warming and that restoration of these wetlands may help to reduce or possibly reverse some of these impacts. In a global synthesis, Pendleton et al. (2012) estimate that converted and degraded coastal wetlands (including tidal wetlands, mangroves, and seagrass meadows) emit 450 million tons (Mt) of carbon dioxide (CO_2) (range 150 to 1,000 $MtCO_2$). Such emissions are equivalent to 3 to 19% of those from deforestation globally and result in economic damages of US $6 to 42 billion, annually.

A number of actions are ongoing to link wetlands management to climate change mitigation responses.

This year the Intergovernmental Panel on Climate Change (IPCC) produced guidance to nations on incorporating the human impacts to wetlands within accounting for national greenhouse gas (GHG) emissions and reductions (IPCC, 2013). Chapter 4 of that document provides guidance on accounting for emissions associated with conversion and drainage of coastal wetlands along with the GHG removal potential associated with restoration. At the Conference of Parties (Warsaw, November 2013) the Subsidiary Body for Scientific and Technological Advice (SBSTA) of the United Nations Framework Convention on Climate Change (UNFCCC) requested that nations apply the IPCC guidance and report back on their experiences by March 1, 2017.

The United States annually report the official national GHG Inventory, meeting the U.S. commitment under the UNFCCC. This is led by the Environmental Protection Agency, with data provided by federal agencies and states. Compilation of accurate data for GHG emissions and sequestration of coastal wetlands in the United States, and elsewhere, would be enhanced by quantification at the regional level.

In parallel, carbon market institutions are also exploring the potential to expand their range of activities to recognize wetland management, and in particular coastal wetland management. In 2011 the Verified Carbon Standard (VCS) recognized Wetland Restoration and Conservation activities eligible as potential carbon projects. This was followed by submission to the VCS in December 2013 of the first global methodology for *Greenhouse Gas Accounting Methods for Tidal Wetland and Sea Grass Restoration* (Emmer et al., 2013). Once approved this methodology will enable carbon project development for restoration of coastal marshes, mangroves, and seagrasses as well as the management of drained organic soils.

Submission of this methodology marks a major step in connecting carbon finance to wetlands management. This is consistent with the Action Plan released by Restore America's Estuaries' Blue Ribbon Panel, on *the Development of a Greenhouse as Offset Protocol for Tidal Wetlands* (Crooks et al., 2010). The plan calls for future quantification of GHG fluxes and regional demonstration and planning to integrate tidal wetlands into carbon project development.

In the United States, federal agencies have established an interagency team to support blue carbon efforts. These include integrating blue carbon science and policy into the National Ocean Policy and activities to develop tools and methodologies for blue carbon management. *The National Assessment of Ecosystem Carbon Sequestration and Greenhouse Gas Fluxes* (USGS, 2013) recognizes that national estimates of GHG fluxes are lacking and that filling this data gap is a priority.

Whether to support national climate change goals, e.g. under a carbon finance framework, or to encourage less formal good practice, there is a need for refined quantification of GHG emissions and removals due to wetlands management at the regional scale. Moreover, wetland climate change mitigation activities should be embedded within regional climate change adaptation strategies to avoid future conflicts in planning outcomes (Crooks et al., in prep).

The Snohomish Estuary represents a typical coastal system for the Pacific Northwest Region from which improved understanding can be extrapolated regionally. Over thousands of years, coastal wetlands in Puget Sound have built up with sea level rise, accumulating deep sequences of organic soils with sediment deposition at the mouth of large rivers, such as the Snohomish. Over the past 150 years, substantial emissions likely occurred when historic wetland soils were diked and drained. While existing wetlands continue to gradually accumulate carbon, remaining wetlands are vulnerable to drainage. Drained soils may also be continuing to release carbon dioxide. Restoration offers potential to rebuild some or all of the displaced carbon stocks, as well as restore other degraded but valued ecosystem services.

This project investigates the Snohomish River Estuary as a demonstration area to quantify changes in soil carbon stock associated with wetland drainage and restoration (Section 6.5). Quantification of emissions and removals is based upon (1) field sampling at undisturbed, drained, and restoring wetlands sites at representative locations within the estuary, and (2) spatial analyses within a context of land use change and rising sea level (Section 5.6). The results highlight historic and future changes (Section 6.6), while the final chapter in the report discusses the implications of ongoing emissions and potential carbon sequestration under different futures, and notes study limitations and uncertainty (Section 7.0).

1.1 Project Background

Coastal wetlands contain a significant amount of carbon as both biomass and soil. Coastal wetlands include mangroves, temperate forested tidal wetlands, emergent tidal marshes, and seagrass meadows. This study focuses on emergent tidal marshes and forested tidal wetlands. According to Pendleton et al. (2012), emergent tidal marshes comprise approximately 0.8 Petagrams (Pg), or 800,000,000 metric tons, of carbon (midrange). This project affords an excellent opportunity to demonstrate localized changes in tidal wetland carbon emissions with land use, and to inform local and national stakeholders in the role coastal wetland carbon cycling plays in local GHG inventories as well as the multifaceted benefits of restoration.

This project is located in an easily accessible and relatively intact estuary, affording opportunities to study the potential benefits of coastal wetland restoration as a carbon sink and actively observe changes in the carbon cycle and the landscape as restoration back to tidal wetland conditions occurs. Tidal wetlands in the Snohomish Estuary include emergent vegetation, scrub-shrub tidal wetland habitat, and forested tidal wetlands. Historically, this estuary was predominantly forested tidal wetland. This history (~1885) offers a compelling context for future tidal wetland restoration, carbon sequestration of rich tidal wetland soils, and the potential for large carbon sink in forest biomass.

1.2 Project Team

Restore America's Estuaries (RAE) is the only national organization that focuses exclusively on the restoration and protection of estuarine habitats. RAE is a national alliance of community-based organizations working to preserve the nation's network of estuaries by protecting and restoring critical habitats. RAE elevates the restoration of critical estuarine habitat to a national priority by adhering to a strategy of delivering on-the-ground restoration, convening and educating, and strategically addressing emerging issues. In 2010 RAE convened a Blue Ribbon Panel to develop an Action Plan outlining a path for developing a tidal wetland GHG offset protocol (methodology). RAE then led and supported the VCS Wetlands Technical Working Group, which resulted in the new VCS Wetland Restoration and Conservation requirements. In December 2013, RAE submitted the first global tidal wetland and seagrass restoration methodology to VCS for review and approval. Steve Emmett-Mattox is the Senior Director for Strategic Planning and Programs and the lead for wetland carbon activities.

Environmental Science Associates (ESA) brings multidisciplinary expertise in climate change, wetland restoration, energy, water, infrastructure planning, and sustainable community development, along with knowledge of current GHG policies and protocols. In addition to a 35+ year legacy of successfully meeting restoration challenges, ESA assists clients with climate change adaptation and mitigation. ESA works with clients to reduce vulnerability to climate change, plan for future change, assess and verify carbon footprints, give strategic guidance to minimize and reduce GHG emissions, or pursue economic growth and development in a socially equitable manner that is sensitive to the environment. Stephen Crooks is the Climate Change Program Manager for ESA, a member of IPCC working group that developed the 2013 Wetlands Supplement, a Methodology Expert for the Verified Carbon Standard, a Co-Chair of the International Blue Carbon Scientific Working Group, and chaired Restore America's Estuaries Blue Ribbon Panel on Delivering a Tidal Wetland Offset Protocol.

The Department of Environmental Science at Western Washington University (WWU) addresses today's environmental issues and prepares tomorrow's interdisciplinary problem solvers. Faculty at WWU work to accomplish this mission by integrating outstanding educational programs, faculty-student collaboration, applied research, and professional and community service. The Wetlands Research Laboratory led by Dr. John Rybczyk uses an integrated field and modeling approach to study the overall effects of climate change, and the particular effects of rising sea levels on coastal systems. His group is especially interested in modeling the non-linear feedback processes in coastal wetlands that allow these systems to maintain dynamic equilibrium with sea level rise. These models help to predict the resiliency of estuarine systems to rising water levels and to guide the course of restoration and mitigation efforts.

The mission of **EarthCorps** is to build a global community of leaders through local environmental service. EarthCorps assists community groups, governments, non-profit organizations, and private citizens in their efforts to survey, map, restore, and maintain ecological habitat in the Puget Sound region. Over the past eight years, EarthCorps ecologists have worked on more than 30 projects to provide detailed site assessments, restoration monitoring and mapping, management recommendations, GIS maps of project sites, and a wide variety of on-the-ground restoration services. The EarthCorps Science Team also offers local project management services throughout the region on projects like Coastal Blue Carbon.

1.3 Report Organization

This report is organized similarly to a peer-reviewed academic paper with the following sections:

Introduction (Sections 1.0 - 4.0)

Analytical Approach (Section 5.0)

Results (Section 6.0)

Discussion (Section 7.0)

2.0 STUDY APPROACH

The following steps are undertaken in this study:

1) Estimate historic CO_2 releases to the atmosphere from drained wetland soils associated with the diking and conversion of wetlands.
2) Evaluate the potential for ongoing emissions or removals of CO_2 to the atmosphere from existing drained soils.
3) Explore potential consequences for carbon storage, emissions, and removals with restoration activities and wetland response to sea level rise.

This analysis utilizes available science and mapping coupled with regional restoration planning. Additional field data collection was undertaken to quantify soil carbon stocks at representative natural, drained, and restoring sites within the Snohomish Estuary. A research team from Western Washington University, led by Dr. John Rybczyk, and EarthCorps conducted fieldwork in the spring and summer of 2013. This field effort included soil coring and rapid vegetation assessment. Twelve sites were sampled throughout the Lower Snohomish Estuary. Geomorphic analysis and landscape-level carbon budgets were provided by ESA.

The potential for restoration under both current conditions and future conditions with sea level rise is investigated using geomorphic principles and GIS analysis. Future scenarios for changes in carbon stocks include planned wetlands restoration projects and further restoration action. While direct measurements of non-CO_2 GHG emissions were beyond the scope of this study, we summarize what is known along with discussion of information gaps and future information needs.

3.0 CARBON MANAGEMENT IN COASTAL SETTINGS

3.1 Role of Coastal Wetlands in Carbon Cycling and Climate Change Regulation

Wetlands are important components of the global carbon cycle because of their capacity to sequester CO_2 directly from the atmosphere and accumulate carbon within biomass. Moreover, wetlands also have a capacity to build organic-bearing soils, transferring carbon from biomass to long-term storage in accumulating soils and sediments. This capacity is a result of wet, low-oxygen soil conditions which are ideal for preserving organic material in situ. Such a transfer leads to vast stores of carbon (and nitrogen) held within coastal wetland soils. Coastal wetlands exist in a dynamic equilibrium with respect to sea level and have, for at least the last 3,000 years, kept pace with moderate rates of sea level rise through the accumulation of organic and mineral sediments. These stores of soil carbon may be held for thousands of years unless disturbed by human activity.

Carbon sequestration and storage in coastal wetlands was enhanced by moderate sea level rise over the past 7,000 years. Vegetated wetlands were able to keep pace with rising sea levels by accumulating and burying organic matter beneath new layers of deposited sediment. Thus the extent of intact coastal wetlands and adjacent drained coastal wetlands reflects regions of long-term carbon sequestration. This long-term removal of atmospheric CO_2 acted as a negative feedback, reducing gradual warming since the post-glacial period.

In addition to carbon sequestration, intact wetlands also produce and emit methane (CH_4). Methane is a product of microbially-mediated organic decay in freshwater, low-oxygen, and low-salinity settings. In settings where salinity is greater than half that of sea water, CH_4 production in wetlands is negligible (Poffenbarger et al., 2011).

A third GHG of concern is nitrous oxide (N_2O). Nitrous oxide emissions are a function of anthropogenic nitrogen loading from catchments and atmospheric deposition. In natural conditions, emergent wetlands are not commonly a source of N_2O, which is typically emitted from soils subject to periodic wetting and drying.

In the context of long-term climate change (more than 100 years), carbon sequestration is the most important driver in influencing atmospheric warming. While CH_4 has a warming potential 25 times that of CO_2, it has a half-life in the atmosphere of only 12 years. However, in the context of immediate and near-term climate change mitigation, both CO_2 and CH_4 emissions and reductions associated with wetland management are important.

3.2 Greenhouse Implications of Wetland Conversion and Restoration

While much of the science in coastal wetlands has focused on the carbon sequestration capacity of vegetated coastal ecosystems, attention has turned to the climate implications of wetland conversion and destruction and the associated high rates of CO_2 emissions from vulnerable, vast long-term carbon stores in coastal wetland soils (Crooks et al., 2011; Pendleton et al., 2012; IPCC, 2013). Restoration of wetlands offers the potential to reduce or, in certain circumstances, reverse those emissions.

Globally, it is estimated that between 150 and 1,020 million tons of CO_2 are released annually from drained and converted coastal wetlands (Pendleton et al., 2012). Drainage of wetland organic soils by lowering the water table, which occurs when wetlands are converted to agricultural land, increases the oxygen content of the soil, resulting in increased microbial degradation of organic matter and emission of CO_2 from the soil to the atmosphere. These emissions may be protracted, occurring over many years depending upon water management and magnitude of carbon stock. Drainage can reduce CH_4 emissions, which are generally negligible in drained soils. However, methanogenesis may take place in drainage ditches or in drained soils where the water table has been raised to near soil surface elevations to create wet pasture, causing a significant source of CH_4. Nitrous oxide (N_2O), a potent GHG, may also be emitted from drained wetlands, particularly if the land is artificially enriched with fertilizer or holds a stock of cattle.

Restoration of coastal wetlands through dike breaching reintroduces natural tidal flooding, sediment delivery, and soil rewetting. It allows recolonization by tidal wetland plants and reactivation of carbon sequestration. Restoration may result in a state where net GHG emissions are greatly reduced or reversed and the wetland functions as a net remover of GHGs from the atmosphere.

3.3 Approaches to Greenhouse Gas Management

Approaches to managing GHG emissions and removals by coastal wetlands through conservation and restoration activities are illustrated in Figure 1. The intent is to reduce net GHG emissions with respect to the “business as usual” or baseline condition. Depending upon whether the drained soils are sources of ongoing emissions, the benefits of improved management practices may include both avoided GHG emissions from drained soils as well as GHG removals associated with rebuilding wetland carbon stocks. On terrestrial peatlands the primary benefits of wetland restoration come through lowering ongoing emissions, as the restored sequestration potential is relatively minor. For coastal wetlands both the avoided emissions from drained soils and the sequestration potential of restoring wetlands may be significant.

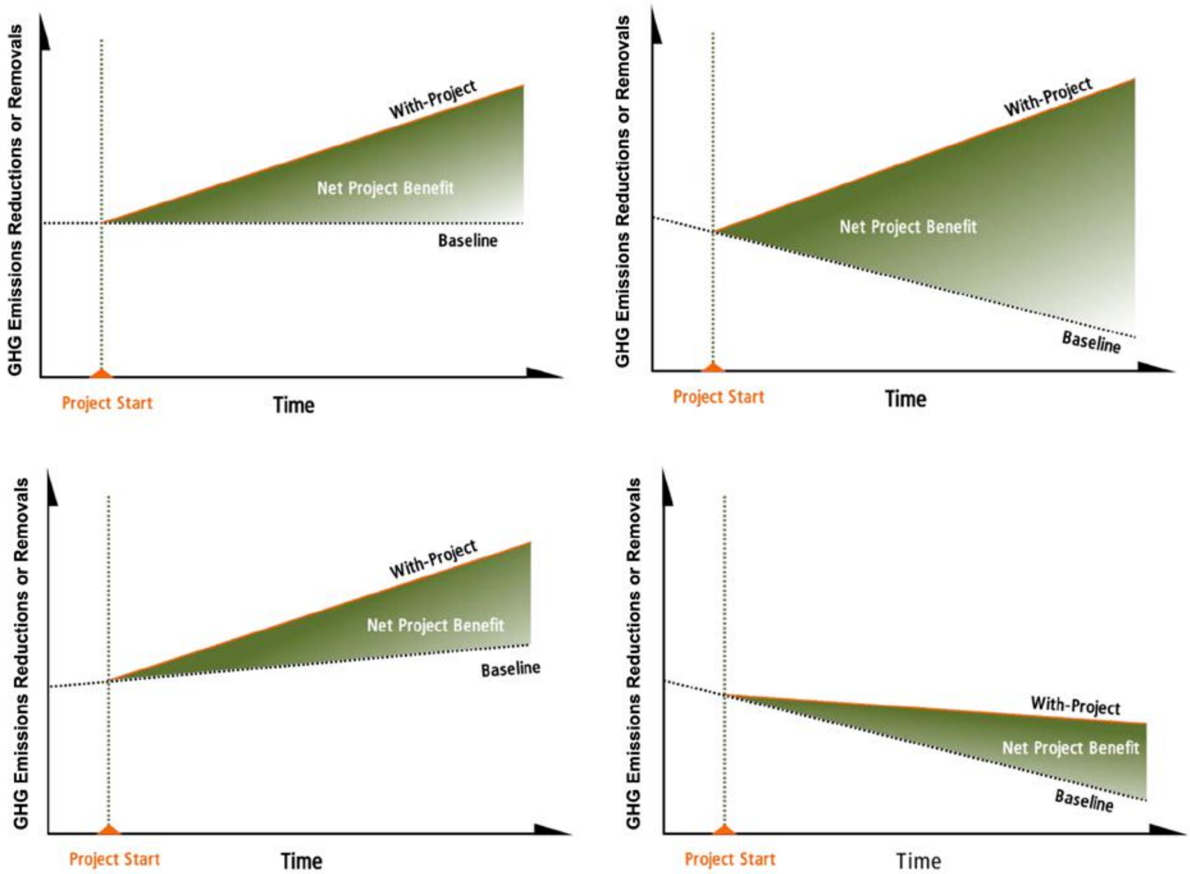


Figure 1 Hypothetical scenario illustrating the net impacts of restoration projects on carbon stock (Based on ForestTrends.org).

3.4 Geomorphic Considerations for Wetland Management

In the coastal setting, wetlands management activities for climate change mitigation should be taken in the wider context of landscape change with sea level rise. Given adequate space, wetlands migrate landwards with sea level rise; marine ecosystems replace coastal terrestrial ecosystems.

The capacity of natural wetlands to respond resiliently to sea level rise, and the capacity of restoring tidal wetlands to rebuild from subsided lands, depends upon geomorphic and ecological conditions. Mineral and organic sedimentation are both key to tidal wetland building. Mineral sediment accumulation is required to build mudflats to elevations at which emergent tidal marsh vegetation can colonize. The marsh vegetation contributes to the accumulation of tidal marsh soils. In the absence of an adequate supply of mineral sediment, restoring wetlands on drained sites may never attain, or be delayed in attaining, vegetated tidal wetland elevations (Orr et al., 2003; Stralberg et al., 2011). By contrast, restoration projects in coastal systems subject to high sediment loading may recover very quickly once tidally reconnected.

The three-dimensional shape of the estuary influences the capacity for restoration and resilience to sea level rise. Systems with relatively expansive shallow areas, where subsided lands lie within the elevation range that tidal wetland plants will colonize, offer high restoration potential compared to deeply subsided systems, if sediment supply to fill accommodation space is limited. Similarly, slope of lands above sea level also influence the potential extent of coastal wetlands area with sea level rise. Level areas provide greater capacity for wetland creation with sea level rise whereas steeper slopes do not provide this opportunity.

These geomorphic considerations are summarized in a series of simple metrics developed by Crooks et al. (2011) and will be considered later in reviewing the carbon sequestration potential of restoration opportunties.

4.0 STUDY AREA

4.1 Historic, Current, and Possible Future Conditions

Puget Sound in Washington State is a large tidal estuary connected to the Pacific Ocean. It consists of a complex of estuaries that interconnect through marine waterways and basins. The Sound's shoreline is 12,144 km long, encompassing seven major watersheds that drain the west Cascade and east Olympic mountain ranges. The watersheds range in size from 1,770 km^2 (Stillaguamish) to 7,800 km^2 (Skagit). Freshwater flow is seasonal; while the mean annual river discharge to the Sound is 1,161 m^3s^{-1}, this increases to 10,390 m^3s^{-1} during spring months.

Puget Sound's dense river-bottom forest has been almost entirely cleared and its wetlands drained. The mid-19th century mixed hardwood-conifer riverine forest was dominated by smaller hardwoods such as vine maple (*Acer circinatum*), willow (*Salix spp.*), and red alder (*Alnus rubra*). Less abundant conifers accounted for the majority of the biomass. Sitka spruce (*Picea sitchensis*) was a common large conifer found in tidal water areas (Collins et al., 2001). Now these areas exist as a mix of land uses.

The Snohomish Estuary is the second largest Puget Sound drainage at 4,807 km^2 (Haas & Collins, 2001). The Snohomish River is formed by the convergence of the Skykomish and Snoqualmie Rivers and flows 34 km from this confluence to Possession Sound at Port Gardner Bay, adjacent to the City of Everett. Like many of the Sound's estuaries, the Lower Snohomish River consists of an expansive alluvial floodplain. Historically, this was heavily forested but the trees have been systematically removed. Wetlands have been drained and converted to agriculture and other uses, and the river banks diked and hardened (Collins et al., 2001). As forested and emergent wetlands were drained, the land subsided relative to the tidal elevations and the remaining wetlands outside of diked areas.

Prior to European settlement in the mid-19th century, the Snohomish Estuary included tidal wetland, mudflats, and many blind channel and slough networks. While the channel margin, tidal wetland, and floodplain have been dramatically altered, the distributary sloughs and mainstem channels have changed little in area or location. Today, only 16% of the historic tidal wetland and 25% of the blind tidal sloughs remain (Haas and Collins, 2001). There are currently 71 km of dikes in the estuary that separate the river from former tidal wetland (Pentec Environmental Inc., 1998). This dike-protected land is largely used for agricultural purposes. Other uses include industrial purposes, municipal sewage treatment (Cities of Everett and Marysville), waste disposal (two landfills), and infrastructure (roads, railroads, etc.). All told, these more permanent, non-agricultural land uses account for 10% of the former tidal wetland area (Haas and Collins, 2001).

Large areas of the Snohomish Estuary are undergoing restoration. Snohomish County and The Tulalip Tribes, in collaboration with local cities, state agencies, and non-profit groups like EarthCorps, are working to restore former habitats, from emergent tidal wetland to forested riverine tidal wetland. Figure 2 shows both completed and planned restoration projects for the Snohomish Estuary. Please note that while the northwest portion of North Ebey has been restoring since the 1960s, a large southern portion has been funded by NOAA and Snohomish County for additional habitat enhancements such as expanding natural dike breeches.

The sites sampled for the fieldwork portion of this study are composed of natural tidal wetland (Figure 3), restoring tidal wetland (Figure 4), and areas that are still influenced by dikes and levees (Figure 5). The sample sites are described below.

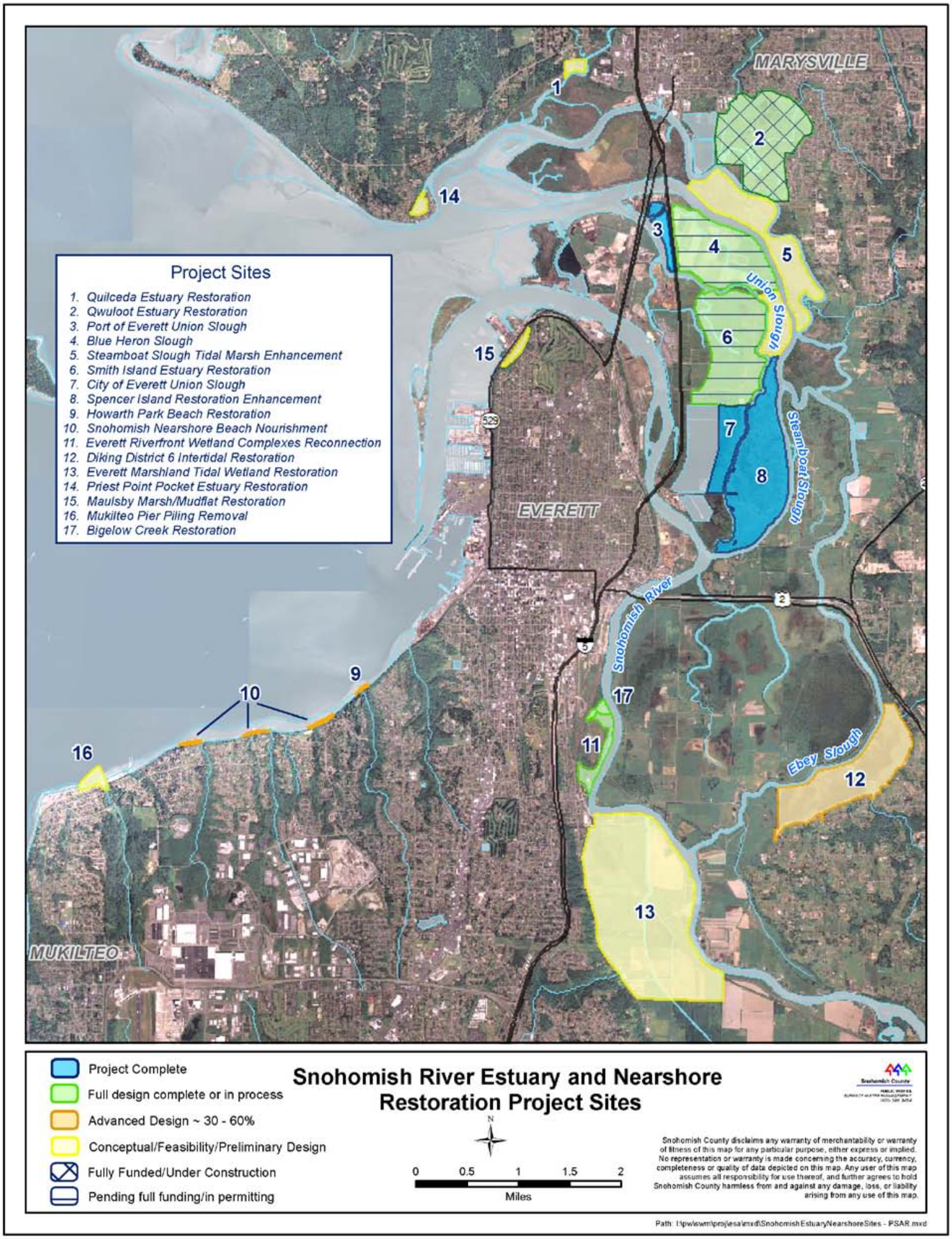


Figure 2 Snohomish Estuary nearshore restoration sites (Snohomish County, 2013).

4.1.1 Sites in their Historic Natural Condition

This study used three areas as points of reference for historic conditions (Figure 3):

- **Quilceda Marsh (QM)** represents an emergent herbaceous wetland at the mouth of the Snohomish Estuary. It is located on the north bank of Ebey Slough, across from North Ebey Island, and is bordered on the west by the outlet of Quilceda Creek. The City of Marysville forms the landward border. The emergent tidal marsh complexes in this area are among the most pristine in the study area and support rare plant species (black or Indian rice lily) and high-quality native plant communities and wetland types. This area is owned by The Tulalip Tribes.

- **Heron Point (HP)** is a forested wetland consisting of more than a dozen plant species. The authors of this report were unable to determine the historic land use, but the area is currently characterized as Palustrine forested wetland (NOAA CCAP, 2006). This wetland is a remnant 6.5 ha Sitka spruce forested tidal wetland behind a levee along Ebey Slough. This site is owned by Snohomish County.

- **Otter Island (OI)** is a mixture of both forested wetland and emergent herbaceous wetland habitats. This area is owned by Snohomish County. The majority of this 66.3 ha island has never been cleared or farmed and is assumed to represent the historic conditions within the Snohomish Estuary. This area provides very diverse habitat consisting of a Sitka spruce forested tidal wetland along the periphery and western half of the complex, and a cattail/bulrush tidal wetland in the central portion. Soil samples were focused in the Sitka spruce swamp area of this site.

These three areas represent typical ecosystems historically present in the Lower Snohomish Estuary (Table 1). This table is based on historic habitat research by Haas and Collins (2001) and the University of Washington River History Project (Geomorphological Research Group, Quaternary Research Center, 2005), as well as the *Snohomish Estuary Wetland Integration Plan* (City of Everett Department of Planning and Community Development, 1997).

Tidal Marsh Habitat	Vegetation Type	Vegetation Examples	Typical Vegetation Elevation (meters NAVD88)	Historic Area (Hectares)
Estuarine Emergent Marsh	Low and High Marsh Vegetation	*Deschampsia sp.* (grass), *Carex sp.* (sedge), *Scirpus sp.* (bulrush)	0.9 - 2.0	198
Emergent/ Forested Transition Wetland	Scrub-Shrub Wetland	*Typha latifolia* (Cattails), *Salix sp*. (willow), *Rosa sp*. (rose)	2.0 - 3.5	1,146
Forested Riverine Tidal Wetland	Palustrine and Riverine Forest	*Picea sp.* (Spruce), some Fir and Cedar	> 3.5	2,607
Palustrine Wetland/ Floodplain Forest	Palustrine and Riverine Forest			3,370
Total				7,320

Table 1 Historic landscape conditions (Haas and Collins, 2001; University of Washington River History Project, 2005).

NATURAL AREAS

Quilceda Marsh (QM)

Heron Point (HP)

Otter Island (OI)

Figure 3 Photos of natural areas where soil cores and vegetation plots were taken, June-July 2013. Photo Credit: D. Devier, with aerial support from LightHawk.

4.1.2 Sites Restoring from Diked Farmland to Tidal Wetland

This study also chose five areas that are in the process of restoring from diked farmland to tidal wetland conditions (Figure 4). These five areas include:

- **North Ebey (NE)**, located at the north end of Ebey Slough and sometimes called the "Steamboat Slough" site (Figure 2), was breached sometime between 1965 and 1970. This site has the largest cattail and bulrush tidal marsh (approximately 100 ha) in the Snohomish Estuary. The eastern and southern margins of these wetlands are dominated by scrub-shrub and forested habitat consisting of black twinberry (*Lonicera involucrata*), Nootka rose (*Rosa nutkana*), crabapple (*Malus fusca*), ninebark (*Physocarpus capitatus*), hardhack (*Spiraea douglasii*), Sitka spruce, and red alder. This area is owned by Snohomish County.
- **Spencer Island (SP)** was breached in 1994 (south) and 2005 (north). This is an 81 ha public property managed by Snohomish County. The levees and dikes around this property remain intact, and it is managed as a non-tidal wetland park and duck hunting reserve. A "hog-fuel" dike on the property has breached and small areas of the site are tidally influenced.
- **Marysville Mitigation site (MA)** was breached 1994. This is a 5 ha tidal wetland restoration site completed in 2011 by the City of Marysville for mitigation purposes. The site is owned by the City of Marysville.
- **Union Slough (US)** originally was breached in 2001, and an additional 1.9 ha were added through another dike breach in 2006. The 11 ha site is fully tidal and includes extensive mudflats with a mature sedge (*Carex*) tidal wetland fringe. This site is owned by the Port of Everett.
- **Smith Island-City (SS)** was breached in 2007. This is a 38 ha tidally influenced wetland restoration site owned by the City of Everett. Three 55 m dike breaches occurred in 2007 on the Union Slough side of the site.

These five sites range both in time since restoration, from 43 years to 6 years old, and historic tidal wetland habitat, from emergent to forested tidal wetland. See Section 4.0 for further description of these historic tidal wetland ecosystems.

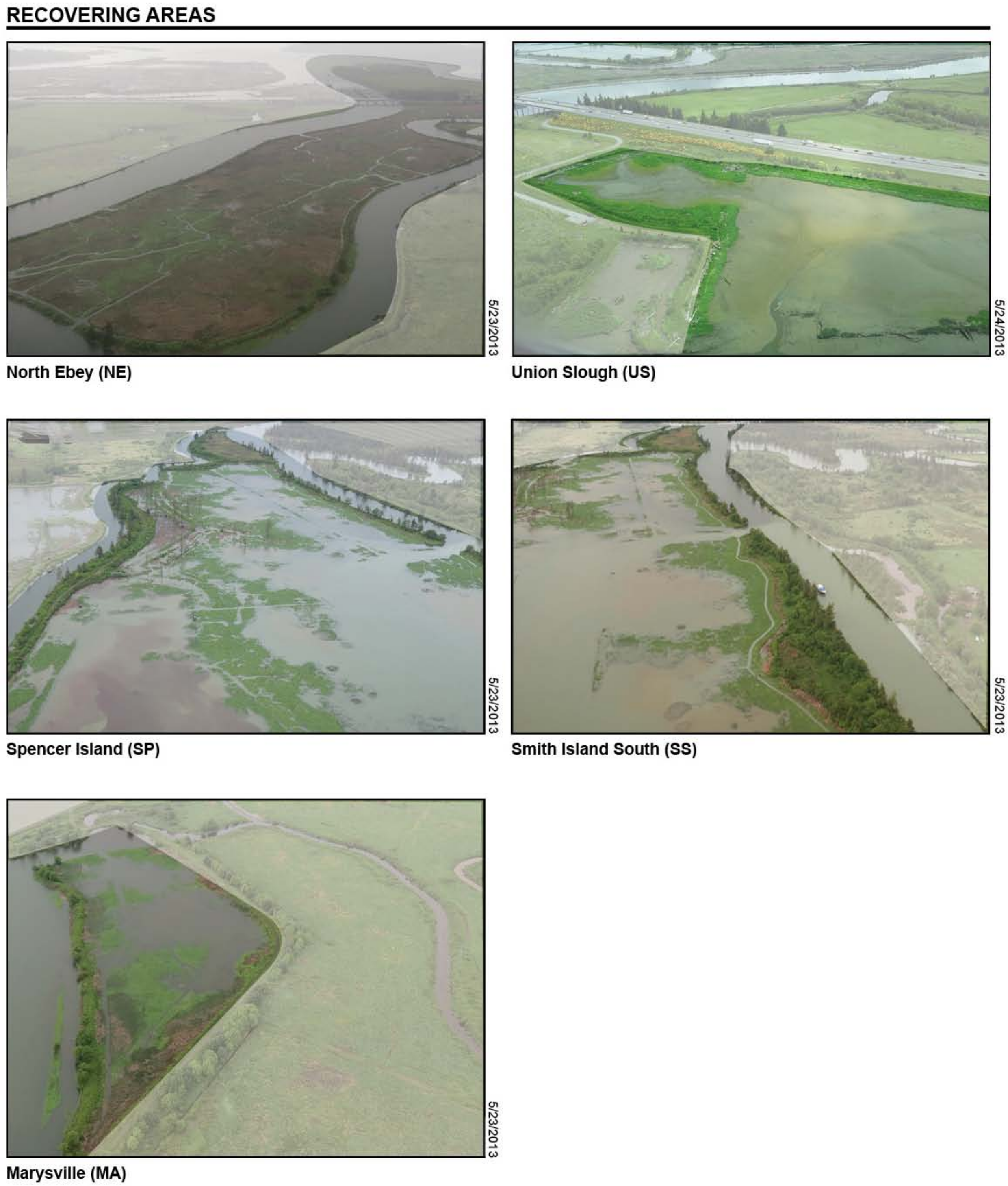


Figure 4 Photos of recovering areas where soil cores and vegetation plots were measured, June-July 2013. Photo Credit: D. Devier with aerial support from LightHawk.

4.1.3 Sites in Potential Future Restoration Areas

There are currently nine large restoration sites planned for the Snohomish River. Projects range from restoration of tidal wetlands and wetland complexes to increased connectivity to the Snohomish River mainstem.

This study investigated the following four areas that are planned for restoration in the short- to long-term:

- **Qwuloolt (QW)** levee breach, scheduled for 2 to 5 years from now (2013). This is an approximately 131 ha diked and drained former agricultural field at the mouth of Allen Creek. This parcel is now owned, almost in its entirety, by The Tulalip Tribes. Restoration of this site via a single, large levee breach is set for implementation in 2014.
- **Smith Island-County (SN)** levee breach, scheduled for 5 years or more from now (2013). This 113 ha diked and drained former agricultural site is owned by Snohomish County. The site has several large, remnant channels that the restoration designers anticipate reconnecting to tidal influence. Multiple smaller dike breaches are set for implementation in 2013.
- **Washington Department of Fish and Wildlife Diked and Drained Wetland (WW)** is a potential future restoration site located on Ebey Island, upstream of Spencer Island. The grassland portion of the site (332 ha) is vegetated largely by reed canarygrass (*Phalaris arundinaceae*) and Baltic rush (*Juncus balticus*). This dominant habitat type is divided by the forks of Deadwater Slough which span the property from north to south. Ebey Island was completely diked in the first half of the 1900s. This area is owned by Washington Department of Fish and Wildlife (WDFW).
- **WDFW Forest (WF)** is a potential future restoration site, located on Ebey Island. The forested portion of this site (169 ha) was logged in the 1890s and is regenerating naturally into one of the few remaining Sitka spruce swamps in the Snohomish Estuary. Field sampling revealed that this site is a restoring forested wetland, not yet to its apex of a historic Sitka spruce wetland. Few spruce were found; see results and discussion for details. This area is owned by WDFW.

The following five additional areas within the Lower Snohomish Estuary will likely be restored (see Figure 2):

- **Quilceda Estuary** - Restoration of 2 to 4 ha of historic estuarine emergent tidal wetland along Quilceda Creek. This project is north of the historic Quilceda Marsh sampled in this study. The project is currently in the feasibility phase of design and is owned by The Tulalip Tribes, who are also the project sponsor.
- **Blue Heron Slough** (former Biringer Farms) - A 138 ha site, also called the Steamboat Slough tidal wetland enhancement project, is hydrologically connected to the North Ebey site sampled in the fieldwork component of this project. Small sections of dikes around this area were naturally breached in the 1960s. The proposed enhancement project would strategically add or expand existing breaches to maximize tidal connection and salmon access to the existing habitat. The project is currently in the design phase. The property is owned by Snohomish County.

- **Diking District 6 property** - A 162 ha site within the tidally-influenced freshwater area of the estuary. This project is well into design to breach dikes on the site; however, staff capacity and funding have slowed the process in recent years. This site is owned and managed by Diking District 6. The project is sponsored by Snohomish County and City of Everett.
- **Bigelow Creek** - A small project that is part of a larger proposed restoration plan to reconnect the Everett riverfront freshwater wetland complexes to the Snohomish River. The Bigelow Creek portion of the project will reconnect 245 m of off-channel, tidally-influenced freshwater wetlands to the mainstem of the Snohomish River. The property is owned by City of Everett, which is also the project sponsor. This City has completed design and permitting and is seeking implementation funds.
- **Everett Marshland** – This project will restore tidal wetland within a large area that was diked and drained for agricultural purposes. There are currently proposals to convert 160 to 325 ha of this area back to tidally-influenced freshwater wetland and tidal channels.

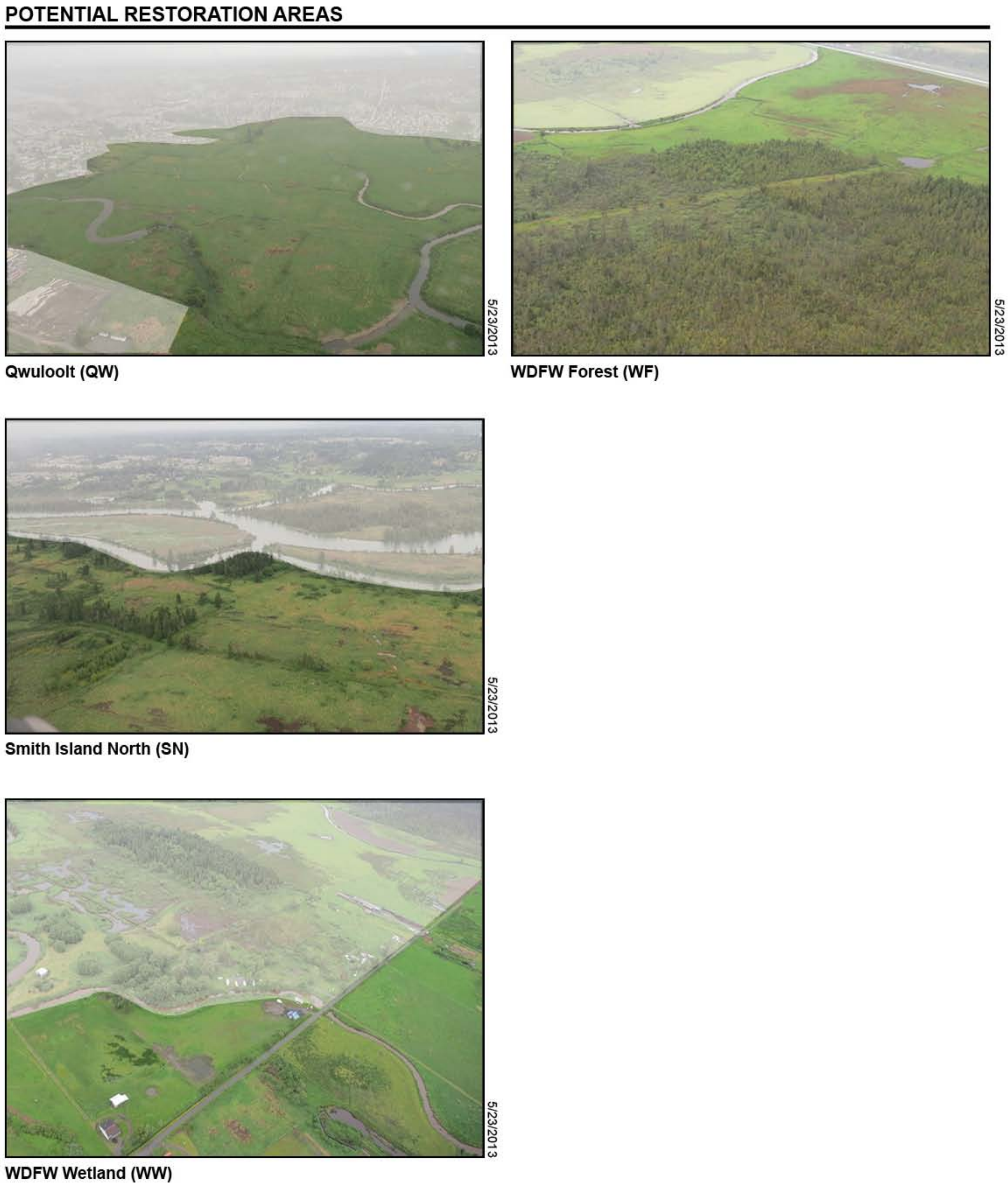


Figure 5 Photos of areas to be restored where soil cores and vegetation plots were taken, June-July 2013. Photo Credit: D. Devier, with aerial support from LightHawk.

5.0 ANALYTICAL APPROACH

5.1 Overview

This investigation quantifies changes in soil carbon stock resulting from historic conversion of wetlands to drained lands, possible ongoing GHG emissions under existing land uses, and potential rebuilding of stocks with wetlands restoration. Primary data of interest are *soil carbon density* (mass of carbon per unit volume of soil) and *soil volume change* under historic drainage and future tidal wetland building conditions. Soil carbon density is determined through direct field measurements (although for a rapid assessment it can be estimated from the science literature). Soil volume change is determined by GIS topographic analysis, quantifying the difference between historic or future tidal wetland elevations and the existing surface topography derived from a digital elevation model (Puget Sound LiDAR Consortium et al., 2009).

Dating the rates of soil accumulation adds valuable information on the capacity of wetlands and their soil carbon stock to recover once levees have been breached. Soil accretion rates are sensitive to mineral sediment supply and the capacity of vegetation to build organic soil.

Determination of landscape hypsometry also assists in assessing the feasibility of wetland restoration. Hypsometry is a measurement of land elevation with respect to sea level. Hypsometric variation is a function of the processes that shape the landscape, or geomorphic processes. A hypsometric analysis is used in this study to determine the potential restoration area and volume of the Snohomish Estuary and how this potential restoration area and volume change with sea level rise.

Field data were collected from 12 sites, including undisturbed, drained, and restoring wetland areas. Together these sites cover a reasonable range of environments found across the estuary. To supplement the soil carbon data, surface vegetation cover data were also logged. Although living biomass was not calculated, the data provide the basis to do so in future projects, and they support future interpretation of relationships between vegetation community composition and soil carbon sequestration.

5.2 Estuary Spatial Analysis

5.2.1 Site Boundary

The boundary for this project was derived from the potential elevation of mean higher high water (MHHW) in 100 years (NOAA Coastal Services Center, 2013). A site boundary was created at this elevation using 2009 bare earth digital elevation data (Puget Sound LiDAR Consortium, NOAA/NWFSC/Watershed Program, 2009).

5.2.2 LiDAR Digital Elevation Model and Tidal Datum

Land surface topography was derived from the validated LiDAR Digital Elevation Model (DEM) developed for the 2009 Snohomish River Estuary LiDAR project and made available by the

NOAA/NWFSC/Watershed Program and the Puget Sound LiDAR Consortium. Data are provided in Washington State Plane North Coordinate System FIPS 4601, in the NAD83/NAVD88 Geoid03 datum.

Reference vertical tidal data were derived from the Everett, WA, tide gauge (9447659). Together these data provided an absolute topographical surface and reference of that surface to tidal flooding elevations.

5.2.3 Historic Land Cover

The historic extent of tidal wetland for the Lower Snohomish Estuary was best described in Haas and Collins' *A Historical Analysis of Habitat Alterations in the Snohomish River Valley, Washington, Since the Mid-19th Century: Implications for Chinook and Coho Salmon* (2001). This study used government land office (GLO) records from 1884 – 1885, as well as other government records, to characterize the three different tidal wetland habitats: emergent tidal marsh, scrub-shrub tidal wetland, and forested tidal wetland. The work done by Haas and Collins is also supported by the University of Washington's Puget Sound River History Project which used the same topographical sheet (t1681.tif) from 1884 - 1885 to create polygons designating bluffs, buildings, channels, deciduous forest, fenced areas, mixed forest, salt marsh, forested salt marsh, and wooded marsh (Geomorphological Research Group, Quaternary Research Center, 2005). While the conditions in 1884 - 1885 do not represent an unmodified estuary, it is the oldest known spatial representation of this system and is the current baseline for the historic ecological condition.

5.2.4 Existing Land Cover

To represent existing land cover, NOAA's Coastal Change Analysis Program (CCAP) 2006 land cover dataset was used. This raster dataset uses remotely sensed imagery to identify about two dozen different land cover types including developed land, agricultural land, and wetland (NOAA Coastal Services Center, 2013).

5.2.5 Soils

In estimating historic carbon density and other characteristics of wetland soils, we relied on information from the USDA Natural Resources Conservation Service SSURGO data (Soil Survey Staff 2013), as well as field data from existing wetland soils (see field sampling and lab analyses below).

5.3 Field Sampling Methodology

5.3.1 Rapid Vegetation Assessment

Vegetation was assessed at each site at two scales. Prior to taking a soil core, the field team measured a square subplot of 0.25 m^2 placed directly over the location of each core sample. This square was used to estimate the cover class of all vegetation or other substrate such as litter and bare soil. This estimate did not include plant species that overhang the subplot such as tree canopy. In addition to the subplots, a second larger vicinity plot was taken at each site using a 5 m radius circular plot (19.63 m^2) originating from the soil core sample location. If the vegetation was relatively homogenous, a plot was taken nearby in order to expedite sampling. The vicinity plot was used to identify all vegetation species present by cover class. See Table 2 for cover class values and associated percent cover. In addition to species present and percent cover, data were recorded for any species overhanging the plot area, such as overstory tree species not rooted within the plot. Relative height of each species was also recorded; see below for height strata values. Lastly, trees greater than 12.5 cm diameter at breast height (DBH) rooted more than halfway within the 5 m radius plot were counted.

Vegetation Cover Class Values (%)	
1.	Trace
2.	0-5
3.	6-10
4.	11-25
5.	26-50
6.	51-75
7.	76-95
8.	96-100

Vegetation Height	Height Strata (meters)
Ground	0-1
Shrub	1-4
Tree	>4

Table 2 Vegetation cover as percent cover (%), left, and vegetation height in meters (m), right.

5.3.2 Collection of Sediment Cores

Two replicate sediment cores were collected from each site from an area that was, upon visual inspection, determined to be representative of vegetation. Cores were collected by driving PVC coring tubes (10 cm diameter) at a depth of 60 to 90 cm into the sediments. Cores were capped in the field and stored in a vertical position until they were returned to the laboratory. Cores were frozen to facilitate extraction and slicing. Extracted, frozen cores were sectioned into 2 cm increments and dried at 60°C for at least 96 hours to determine bulk density. Subsamples of each section were ground to a fine powder in a Wiley Mill for the determination of percent organic matter, percent carbon, and radioisotope ^{210}Pb (lead) activity.

5.4 Laboratory Analysis

5.4.1 Bulk Density, Organic Matter, and Mineral Matter

Bulk density was calculated as the ratio of the oven-dried weight of each 2 cm core section to the known wet volume of that section. Percent organic matter, by weight, of each 2 cm oven-dried section was determined by loss on ignition (Allen, 1974). Percent mineral matter, by weight, was calculated as the remainder.

5.4.2 Carbon Content

Sections from each soil core were analyzed for carbon content using a FlashEA 1112 nitrogen and carbon analyzer (Thermo Electron Corp.). Approximately 150 mg of each dried soil sample was packaged in a tin capsule, with some adjustment of sample amounts to allow the mass of carbon to fall within the range appropriate for the carbon calibration curve. A chemical standard and soil standard were analyzed for quality control at the start of each day. The quality control soil standard was also analyzed after every 10 samples. Approximately 5% of the samples were randomly selected and reanalyzed on a different day.

5.4.3 Accretion Rates

Long-term sediment accretion rates were determined for five sites. Assuming the depositional rate of excess ^{210}Pb does not change with time, a profile of excess ^{210}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}Pb. Excess ^{210}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). We analyzed approximately 15 g of 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}Pb and ^{214}Pb dropped below approximately 10%. Because the gamma spectrometer measures total ^{210}Pb activity which includes both excess and supported ^{210}Pb at 46 keV, excess ^{210}Pb activity must be isolated to determine sedimentation rates. Supported ^{210}Pb is measured by detecting ^{214}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}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 0.75 g pitchblende silica-ore standard (CRM 103-A, New Brunswick Laboratory, USDOE) to a previously analyzed 15 g sample. A linear regression of the natural log of excess ^{210}Pb activity versus depth was used to determine the accretion rate. The accretion rate is equal to $–\lambda/s$, where λ is the half life of ^{210}Pb (22.2 yr^{-1}) and s is the slope of the regression.

5.5 Calculations

5.5.1 Carbon Density

Carbon density was calculated as the product of percent carbon and bulk density (Eq. 1) for each core section with available percent carbon values.

$$\boldsymbol{Carbon\ density}\ \left[\frac{\boldsymbol{g}}{\boldsymbol{cm^3}}\right] = \frac{\boldsymbol{Carbon\ content}\ [\%]}{\boldsymbol{100}\%} \times \boldsymbol{Bulk\ density}\ \left[\frac{\boldsymbol{g}}{\boldsymbol{cm^3}}\right]$$

Eq. 1 Carbon Density

Similarly, mineral density was calculated as the product of percent mineral matter and bulk density, for sections with available mineral content values. Percent mineral matter was calculated as 100-Organic matter content (percent).

Interpolated values of carbon density and mineral density were included in calculations of average carbon and mineral density, and total carbon and mineral mass, in the top 30 cm for each core. Interpolated values were included to avoid over-representation of shallow core sections in these summary statistics, since fewer sections were measured for carbon, organic, and mineral content with depth. Interpolated values were fit to the trend of actual density values with depth at each site (typically a linear trend). Average carbon density in the top 30 cm was then calculated for each core including the interpolated carbon density values, and similarly for average mineral density. Average density values were then averaged across cores for each site to report one density value per site.

5.5.2 Carbon Mass

The total mass of carbon in the top 30 cm was calculated as the sum of the mass of carbon in each section from 0 to 30 cm depth. The mass of carbon in each section is the product of the carbon density in that section and the thickness of that section (Eq. 2). Total mineral mass was determined similarly using mineral density.

$$\boldsymbol{Carbon\ mass}\ \left[\frac{g}{m^2}\right] = \boldsymbol{Carbon\ density}\ \left[\frac{g}{cm^3}\right] \times \boldsymbol{Section\ thickness}\ [cm] \times \mathbf{10{,}000}\left[\frac{cm^2}{m^2}\right]$$

Eq. 2 Carbon Mass

5.5.3 Carbon Accumulation Rate

Carbon accumulation rates were calculated as the product of the sediment accretion rate and the average carbon density in the top 30 cm of the core (Eq. 3). Mineral accumulation rates were determined similarly using mineral density.

$$\boldsymbol{Carbon\ accum.}\left[\frac{g}{m^2 yr}\right] = \boldsymbol{Accretion}\ \left[\frac{cm}{yr}\right] \times\ \boldsymbol{Average\ carbon\ density}\ \left[\frac{g}{cm^3}\right] \times \mathbf{10{,}000}\left[\frac{cm^2}{m^2}\right]$$

Eq. 3 Carbon Accumulation

5.6 Carbon Emissions Modeling Approach

5.6.1 Carbon Stock and Estimating Carbon Emissions

To evaluate carbon dynamics within historic tidal wetlands of the Snohomish Estuary, it was necessary to evaluate pools and fluxes across the range of different tidal wetland habitats. The historic landscape of the estuary included a rich mosaic of habitat types, including estuarine tidal marsh, estuarine/forested transitional tidal wetlands, and forested tidal wetlands. In these systems, carbon was stored in above-ground biomass and soils rich in organic matter. The magnitude of the soil carbon pool within these systems was a function of (1) the extent of area covered by tidal wetland habitats, (2) the mass of carbon per unit volume in wetland soils (known as carbon density and determined by soil carbon content and soil density), and (3) the depth of accumulated wetland organic soils. Plant tissues, especially recalcitrant components of below-ground biomass, were the dominant carbon source to soil carbon pools (e.g. $\delta^{13}C$) (Malamud-Roam and Ingram, 2001). Prior to decomposition and incorporation into the soil, above- and below-ground components of vegetation represented significant, albeit short-term, standing stocks of carbon.

5.6.2 Volumetric Analysis

Using the 2009 DEM, we calculated two curves that characterize subsided area and volume:
(1) hypsometric curve, and (2) stage-volume curve. The hypsometric curve represents the cumulative distribution of elevation versus area. A point on the hypsometric curve represents the total area within the delta that lies below the specified elevation. The stage-volume curve tabulates elevation versus storage volume for a given delta. The storage volume is the volume required to fill the subsided area back up to the specified elevation. Together, these two curves provide detailed information about the distribution and depth of subsided areas within a particular delta system.

5.6.3 Carbon Storage Change

Estimated CO_2 emissions from drained wetland soils were derived from the following datasets:

- Acreage of converted wetlands (including current land use as defined by NOAA CCAP);
- Loss in elevation of soil of converted wetlands; and
- Current wetland soil characteristics, which can be used as a proxy for historic soil characteristics.

To calculate the mass of CO_2 released with soil drainage, the simplifying assumption is made that the amount of carbon emitted from impacted wetlands soils (kg) is equal to the amount of historic carbon throughout that section of the surface soil profile that has been lost (see Eq. 4). This first-order approximation could underestimate carbon loss by not accounting for additional loss from below the surface of remaining drained soils. It could overestimate carbon loss if organic carbon is eroded away with soil and remains deposited in other locations.

Carbon released (t C) =[Area of Drained Wetland (m^2)] *[Subsided depth (m)] * [Carbon Content of Soil (% C)] * [Historic Soil Bulk Density (t m^{-3})]

Eq. 4 Carbon Emissions

5.7 Future Restoration Scenarios

The way in which carbon pools within the Lower Snohomish Estuary will change over time is dependent on the type and extent of restoration activities that may occur in the future. Four general restoration scenarios, described below, were developed to bracket the potential range of future conditions within the estuary. The first scenario included near-term salmon restoration targets and restoration projects currently in design or implementation and likely within the next 10 years or so. The second scenario considered the broader restoration potential for the Lower Snohomish Estuary, allowing for estimation of the high end of carbon sequestration over the next century. These near-term restoration and broader, entire estuary restoration scenarios were analyzed under current and future sea levels (1 m of sea level rise), producing four restoration scenarios of varying scale and sea level inundation.

Future Restoration Scenario 1 (FS1) – Planned Restoration, Near-Term

The 2005 *Salmon Recovery Plan* for the Lower Snohomish Estuary established a target of 500 ha of restored tidal wetland within the lower estuary by 2015 (a 10-year period starting in 2005). As of 2011, approximately 150 ha of that target had been restored, leaving 350 ha. The Recovery Plan target was used as a general target for the near-term restoration potential. Many projects have been identified or proposed in the Snohomish Estuary (see for example Snohomish County and The Tulalip Tribes, 2001). Projects considered to be more conceptual or long-term as also included in FS1 scenario. The Puget Sound Nearshore Ecosystem Restoration Project (PSNERP), is a nearshore habitat restoration and preservation program with 36 conceptual design areas throughout the Puget Sound basin. Of these conceptual designs, six are located in the Snohomish Estuary. These areas include Spencer Island, Smith Island, Quilceda Estuary, Snohomish Estuary Mainstem Connectivity, Everett Riverfront, and Everett Marshland Tidal Wetland restoration. All together, the projects included in this scenario were the following:

1. Qwuloolt – 180 ha (includes Marysville Restoration)
2. Smith Island – 467 ha
3. Blue Heron – 181 ha
4. Marshlands –171 ha (out of 1,220 ha)
5. Spencer Island – 143 ha
6. North Ebey Island – 211 ha

Taken together, these projects would result in about 1,353 ha restoring from primarily agriculture to mudflat and tidal wetland habitat (Snohomish County, 2005; Anchor, 2008; PSNERP, 2011).

Future Restoration Scenario 2 (FS2) – FS1 Plus 1 m Sea Level Rise

The second scenario adds 1 m of sea level rise to FS1, the existing and planned restoration areas. This scenario uses the volume of accreted tidal wetland between the current MHHW, 2.76 m, and a future MHHW of 3.76 m. This increase in elevation not only increases the volume of accreted soil, it also expands the estuary area at or below MHHW to 1,594 ha.

Future Restoration Scenario 3 (FS3) – Large Scale, Near-Term

The third scenario is a hypothetical larger scale restoration scenario assuming present-day tidal elevations. This scenario determines the carbon sequestration effects of restoring the entire estuary. Like FS1 and FS2, this area does not include existing channels, water bodies, or urban areas. The potential restoration area is 4,393 ha, assuming restored tidal wetlands achieve an elevation of 2.76 m.

This calculation consists of the area in FS1 plus areas that are restorable but not yet included in an official restoration plan, such as active agricultural areas.

Future Restoration Scenario 4 (FS4) – FS3 Plus 1 m Sea Level Rise

This scenario applies a sea level rise of 1 m to FS3. Like FS2, the soil volume from 2.76 m to 3.76 m increases by a meter, and the tidal wetland area increases from 4,393 ha to 5,258 ha due to tidal inundation of lands currently above tidal elevations. In other words, sea level rise increases tidal wetland soil depth and breadth as soils accrete both up in elevation and broaden across the higher elevations.

6.0 RESULTS

6.1 Sea Level Rise, Historic MHHW, and Site Boundary

NOAA calculated a historic rate of sea level rise of 1.98 mm per year (NOAA Coastal Services Center, 2013). This rate of change was used to estimate historic change in tidal wetland elevation of 0.16 m from 1930 – 2009. Furthermore, the IPCC estimated that sea level may rise up to 1 m globally. This 1 m increase in sea level was used to estimate the future tidal wetland elevations. To create the study area boundary, an additional 0.5 m was added to the predicted 100-year MHHW elevation of 3.76 m (NAVD88) to create a generous site boundary (Figure 6) around the river valley wall at elevation 4.26 m using 2009 bare earth digital elevation data for the Snohomish Estuary, see Figure 7 (Puget Sound LiDAR Consortium, NOAA/NWFSC/Watershed Program, 2009). Tide elevations were also gathered online from NOAA, tidesandcurrents.noaa.gov, and are shown in Table 3.

Tidal Elevations based on NOS# 9447130	Elevation (m NAVD88)
MHHW	2.76
MHW	2.49
MTL	1.36
MSL	1.36
MLW	0.23
NAVD88	0.00
MLLW	-0.62

Table 3 Tidal datum from City of Everett tide gauge (47° 58.8' N, 122° 13.3' W).

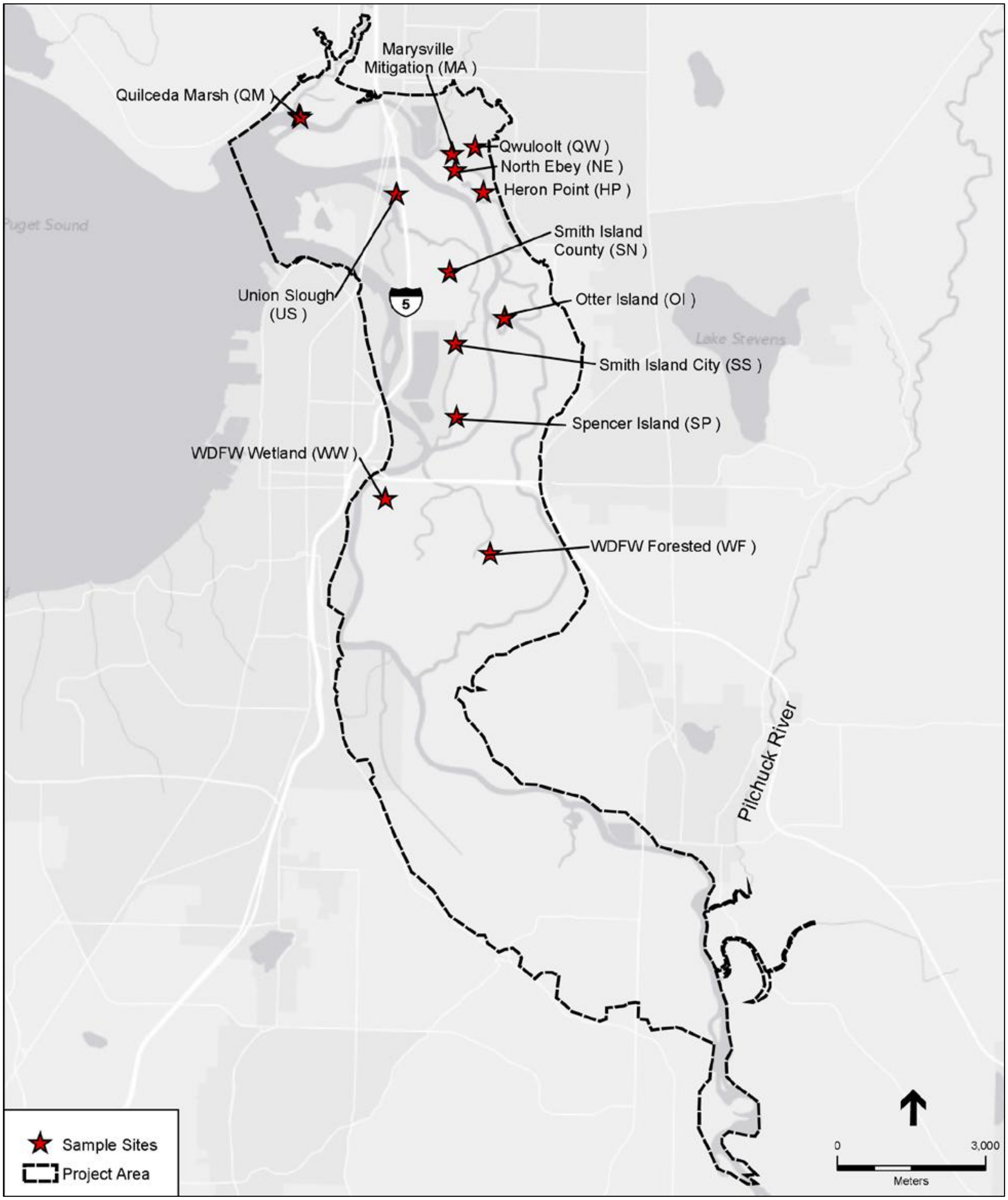


Figure 6 Study Area (dashed black line) and 2013 field sampling sites (red star).

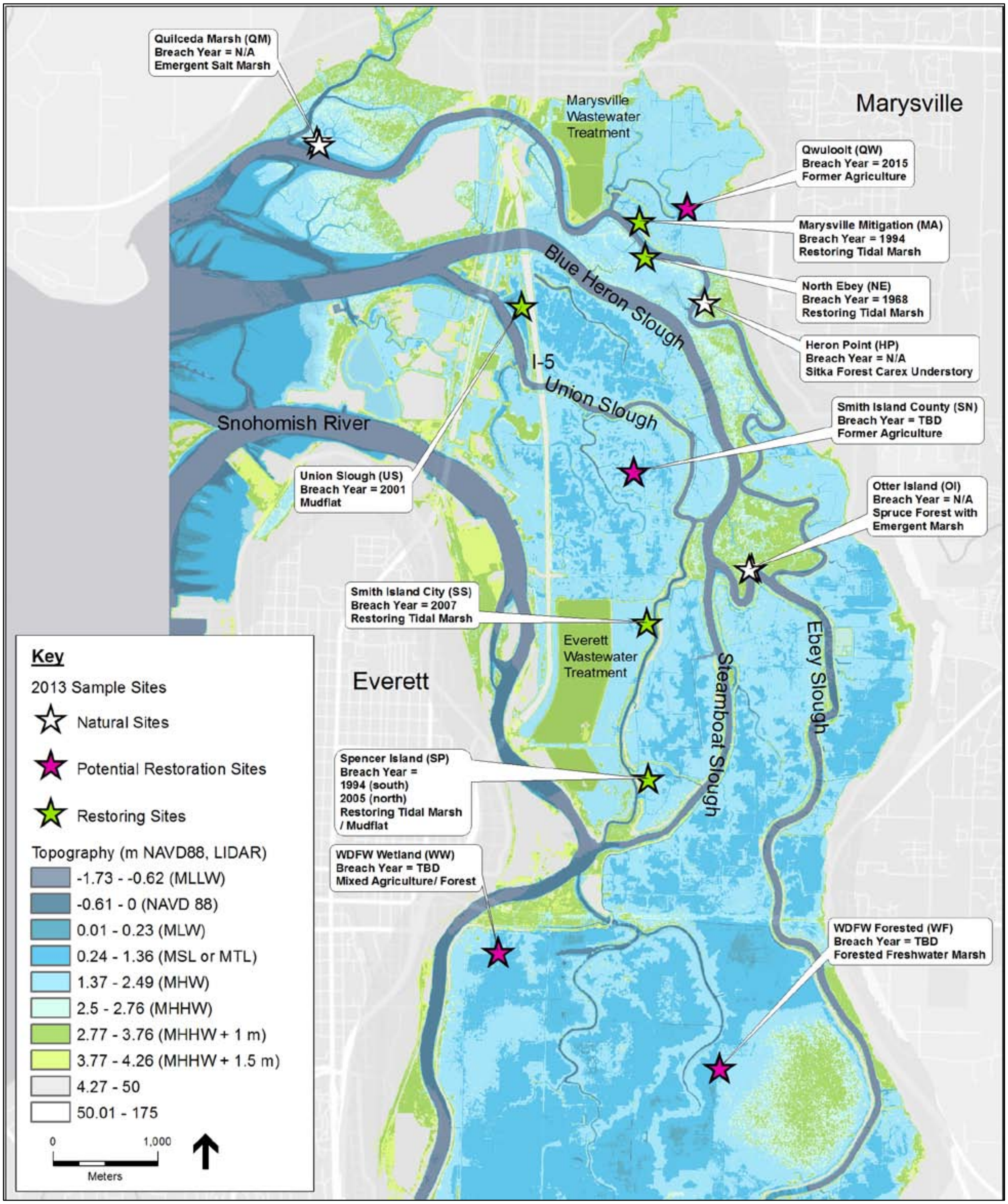


Figure 7 Topographic map (meters, NAVD88) with restoration status of 2013 field sites identified with a green star for breached and restoring, a magenta star for diked and drained areas, and a white star for sites in their natural state (Puget Sound Lidar Consortium et al., 2009).

6.2 Historic and Existing Landscape Conditions

The landscape of the Lower Snohomish Estuary around 1885 was largely wooded marsh (31%) and a variety of salt marsh habitats (63%) (Table 4, Figure 8). Mixed and deciduous forest made up the remainder of the landscape, with the exception of one building and fencing around some grassland and mixed forest. The historic land cover analysis conducted by the Geomorphological Research Group at the University of Washington (2005) used "No Symbol" to indicate areas having no symbology in the historical maps. "No Symbol fenced" denotes areas with no symbology that are enclosed by a fence (riverhistory.ess.washington.edu).

Historic Landscape - Lower Snohomish Estuary (~1885)	Historic Area in Hectares (ha)	Historic Area in Percent (%)	Current Landscape - Lower Estuary (NOAA CCAP, 2006)	2006 Landcover in Hectares (ha)	2006 Landcover in Percent (%)
Wooded marsh	1,254	31%	Palustrine Emergent Wetland	1,299	32%
Salt marsh/Mixed forest	998	24%	Agriculture	897	22%
Salt marsh	814	20%	Palustrine Scrub/Shrub Wetland	476	12%
Salt marsh/Pine	758	19%	Developed	375	9%
Mixed forest	154	4%	Palustrine Forested Wetland	348	9%
NO SYMBOL	22	1%	Open Water	235	6%
Grassland fenced	22	1%	Grassland	149	4%
Deciduous	19	0.5%	Forested	123	3%
Grassland	15	0.4%	Barren Land	87	2%
NO SYMBOL fenced	15	0.4%	Scrub Land	66	2%
Waterbody	1	0.0%	Estuarine Emergent Wetland	14	0%
Bluff	1	0.0%	Palustrine Aquatic Beds	5	0%
Building	1	0.0%	Estuarine Aquatic Beds	0	0%
Mixed forest fenced	1	0.0%			
Total	4,075	100%	**Total**	4,075	100%

Table 4 Historic and current estuary landscape conditions.

According to 2006 NOAA CCAP land use data, the lower portion of the Snohomish Estuary is currently dominated by Palustrine emergent wetland (32%), agriculture (22%), and Palustrine scrub-shrub wetland (12%) (Table 4, Figure 9). Developed land, Palustrine forested wetland, open water, grassland, forest, and barren and scrub land make up the bulk of the remaining land, about 35%. While at first glance it may appear that there is only 897 ha of restorable land, classified as agriculture, there are large areas of diked and drained land classified as Palustrine wetland that may be restored to forested wetland with restored connection to the river. Another landscape change to note is the increase in open water of 235 ha since the late 1800s, which may be attributed to the construction of two large wastewater treatment plants as well as several ponds.

It is difficult to compare historic and modern wetland conditions because the historic landscape is described in the cultural vernacular of the time in letters, journals, and maps; whereas, the 2006 data use a very specific scientific methodology for categorizing landscape conditions. Nevertheless, we see expansive, almost total, conversion of wetlands (forested, scrub-shrub, and emergent) to drained and subsided lands. Of these drained lands, approximately 50% are now classified as Palustrine wetlands. The occurrence of subsided lands classified as Palustrine suggests an abandonment of agricultural drainage resulting in near-surface or surface water table elevations that support wetlands vegetation. Water level management will influence (lower) the rates of soil CO_2 emissions, but increase CH_4 emissions. Table 5 summarizes broad classes of land use change in the Lower Snohomish Estuary. Almost 1,700 ha of wetland have been converted to agriculture, development, grassland, or forested scrub-shrub.

The presence of lands described as Palustrine wetlands should not be taken to suggest that these areas are not suitable for tidal wetlands restoration. In fact, these lands are below sea level and as such exist in locations vulnerable to levee failure, the risk of which increases with sea level rise.

Landscape Type (Lower Snohomish Estuary Only)	Historic (1885) Area (ha)	2006 Area (ha)	Change (ha)
Aquatic			
Open water	1	235	+234
Aquatic Bed	-	5	+5
Wetlands	3824	2137	-1687
Terrestrial			
Developed	1	375	+374
Agriculture	-	897	+897
Grassland	37	149	+112
Forested/Scrub Shrub	174	189	+15
Other	38	87	+49

Table 5 Summary of landscape changes from 1885 – 2006.

While we do not have the historic data for the upper estuary, Figure 9 shows that a similar change has occurred.

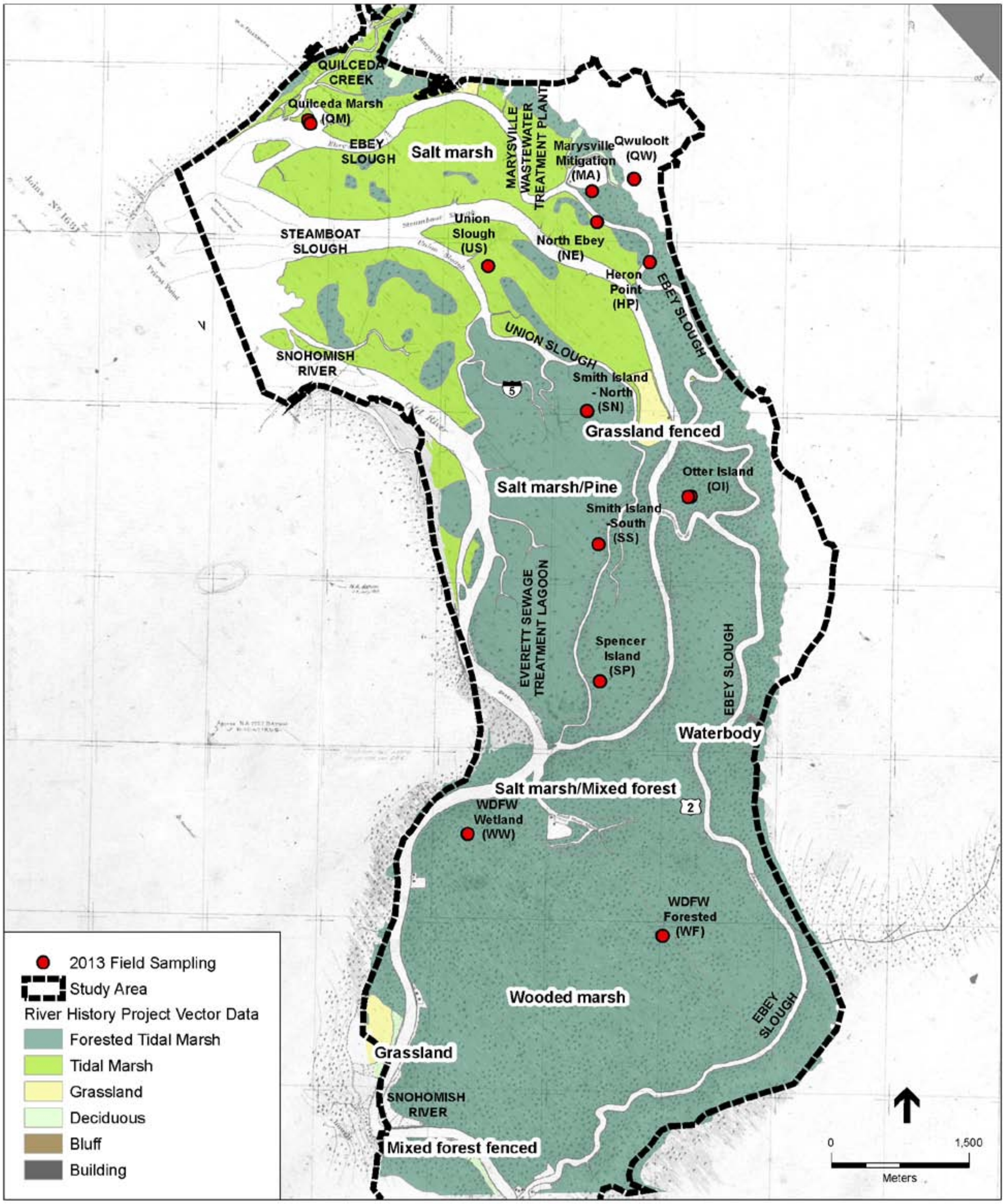


Figure 8 Historic habitats of the Lower Snohomish Estuary based on River History Project (Geomorphological Research Group, Quaternary Research Center, 2005) **and Haas and Collins (2001) and 2013 soil core and vegetation plot locations.**

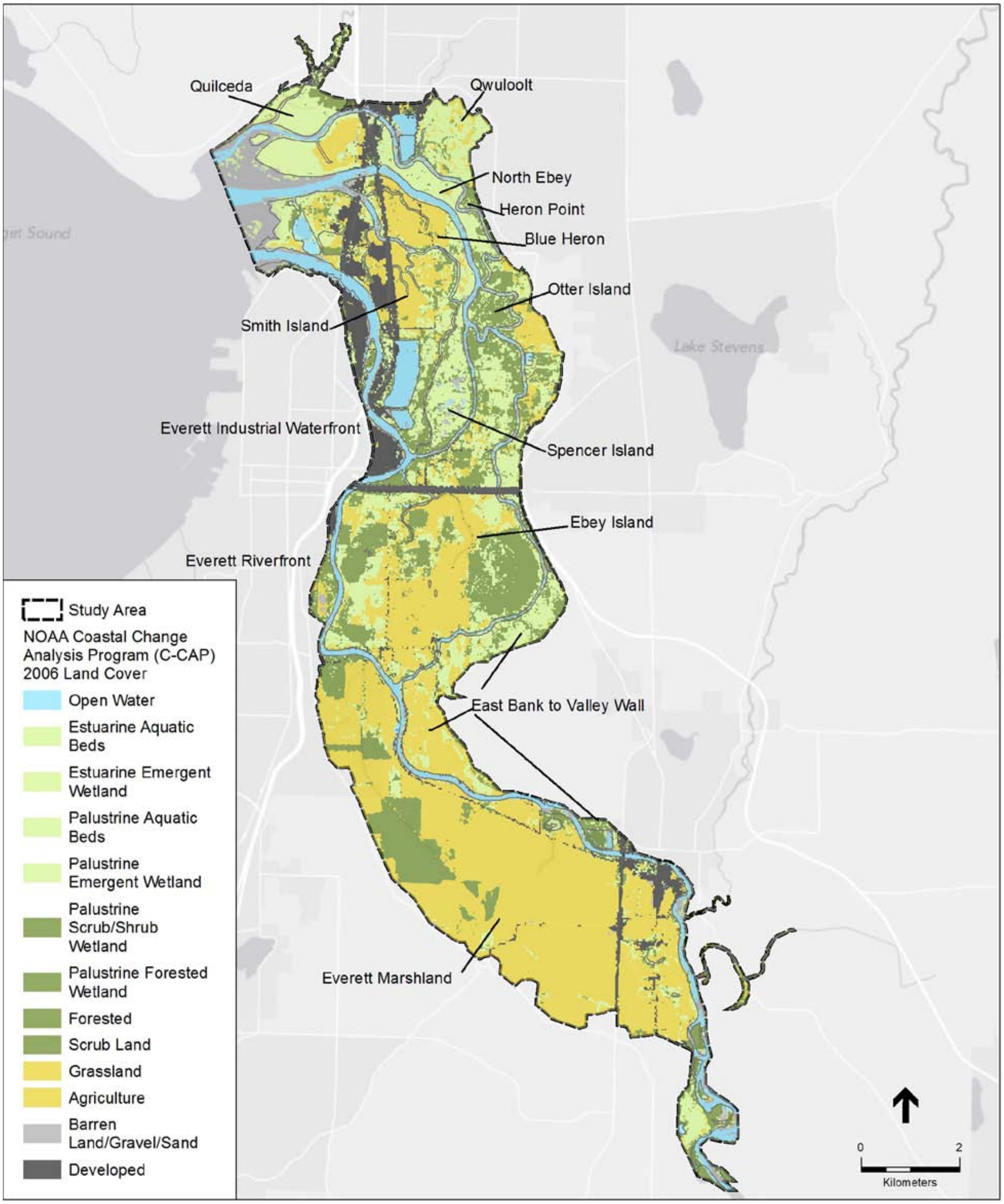


Figure 9 Existing land cover and land use (NOAA CCAP 2006).

6.3 Soils Description

Three dominant soil series are found in and adjacent to the Snohomish Estuary: Mukilteo, Puget, and Snohomish. All three of these soil series are classified as hydric soils and together cover more than 76% of the Snohomish Estuary. Several less common hydric soil series include Bellingham, Fluvaquents, Norma, Riverwash, Sumas, and Terric Medisaprists as well as several non-hydric soil series and modified "urban" soils (Figure 10, Table 6).

We focused the analysis on loss of carbon from wetland soils that have been altered in the past 150 years by drainage for agricultural activities including row crops, orchards, and grazing, as well as other human impacts including diking, creation of ponds, and urbanization.

Soil Series	Hydric	Hectares (ha)	Soil Order	% Area	Depth (cm)	Organic Matter (OM, %)
Snohomish	Yes	850	Entisol	15.9%	0-15	10-20.
Mukilteo	Yes	1,051	Histosol	19.7%	0-10	20-40
Puget	Yes	3,428	Entisol	64.3%	0-23	5-10.
Other	-	63	-	0.0%		
Total		5,392		100%		

Table 6 Soil series in the Snohomish Estuary.

This project focuses on the three dominant soils series, Mukilteo, Puget, and Snohomish (USDA Soil Conservation Service, 1983). These three soil series are all deep, often drained and highly organic and are representative of soil conditions since diking and draining of the estuary at a coarse large scale.

Mukilteo muck is found in depressional areas. The top three soil layers are made of organic material, to an average depth of 137 cm. These organic layers are followed by a layer of fine sandy loam, to a depth of 150 cm or more. Woody debris is often found buried in this soil, and the surface is typically vegetated by sedges and rushes.

Puget silty clay loam is found in depressional areas and on floodplains. In the published soil survey (1983), this soil is described as often artificially drained with native vegetation consisting primarily of hardwoods. It is composed of two layers of different colored silty gray loam, dark grayish brown on top and olive gray underneath, to a depth greater than 175 cm.

Snohomish silt loam is found on floodplains, "formed in alluvium underlain by peat and muck" (USDA Soil Conservation Service, 1983). Like Puget soils, it is often artificially drained and is vegetated with hardwoods. The surface layer of this soil is about 15 cm of silt loam, followed by 58 cm of silty clay loam. The depth to the organic layers is 43 to 81 cm.

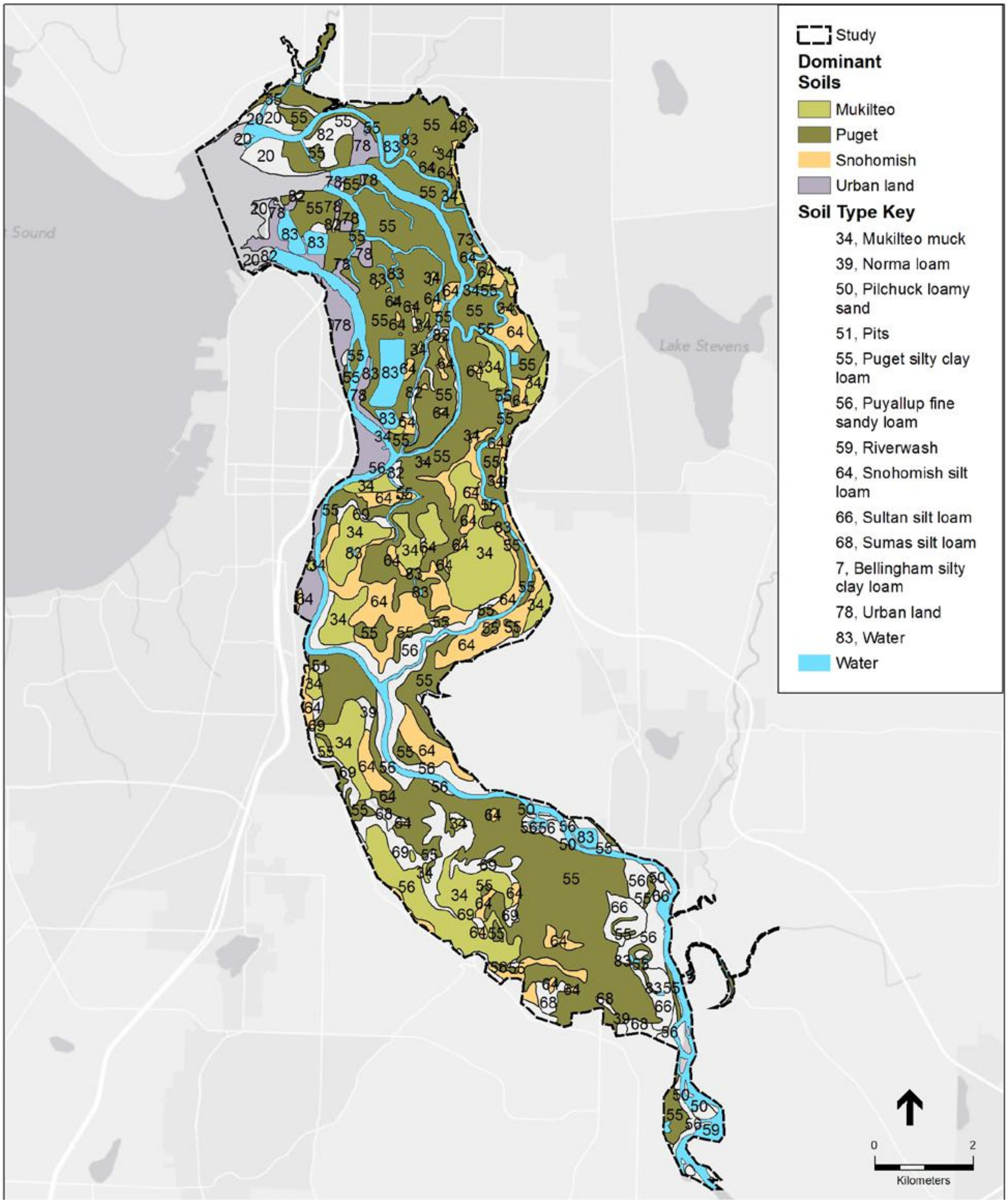


Figure 10 Soils map of the Lower Snohomish Estuary (USDA – NRCS soils data 2013).

6.4 Field Results

6.4.1 Rapid Vegetation Assessment – Natural Areas

The natural areas that were sampled during this project, Quilceda Marsh, Heron Point, and Otter Island, are located in two different tidal wetland zones. Quilceda Marsh is in the estuarine emergent zone, as reflected by the Lyngbye's sedge (*Carex lyngbyei)* dominated vegetation (Table 7). Conversely, Heron Point and Otter Island are in the forested riverine tidal zone with Sitka spruce (*Picea sitchensis*), clustered wildrose (*Rosa pisocarpa*), Nootka rose (*Rosa nutkana*), twinberry (*Lonicera involucrata*), water parsley (*Oenanthe sarmentosa*), sweet myrtle *(Myrica gale*), and many other species (Table 7). Several non-native species were also observed at each site. Photos of each site are shown in Figure 11.

Site	*Scientific Name*	Common Name	Native	Average Cover Class (%)
Quilceda (QM)	*Carex lyngbyei*	Lyngbye's sedge	Yes	68%
	Agrostis sp.	bentgrass	X	20%
	Triglochin maritima	seaside arrow-grass	Yes	11%
	Eleocharis palustris	spike rush	Yes	3%
	Potentilla anserina ssp. pacifica	Pacific silverweed	Yes	3%
	Schoenoplectus tabernaemontani	soft-stemmed bulrush	Yes	3%
	Spergularia canadensis	Canadian sandspurry	Yes	3%
	Chenopodium rubrum	red goosefoot	Yes	1%
	Juncus sp.	rush	X	1%
Heron Point (HP)	*Rosa pisocarpa*	clustered wildrose	Yes	18%
	Rosa nutkana	Nootka rose	Yes	18%
	Lonicera involucrata	twinberry	Yes	18%
	Rubus spectabilis	salmonberry	Yes	8%
	Lysichiton americanus	skunk cabbage	Yes	8%
	Vicia nigricans ssp. gigantea	black vetch	Yes	3%
	Solidago lepida	western Canada goldenrod	Yes	3%
	Ribes divaricatum	wild gooseberry	Yes	3%
	Phalaris arundinacea	reed canarygrass	No	3%
	Iris pseudacorus	yellow flag iris	No	3%
	Heracleum maximum	cow parsley	Yes	3%
	Gaultheria shallon	salal	Yes	3%
	Galium aparine	common bedstraw	Yes	3%
	Athyrium filix-femina	ladyfern	Yes	3%
	Rumex occidentalis	western dock	Yes	1%
	Fritillaria camschatcensis	indian rice-root, black lily	Yes	1%
	Callitriche sp.	water-starwort	X	1%
		Unknown herb sp.	X	1%

Table continues on following page.

Site	*Scientific Name*	Common Name	Native	Average Cover Class (%)
Otter Island (OI)	*Oenanthe sarmentosa*	water parsley	Yes	80%
	Myrica gale	sweet myrtle	Yes	38%
	Phalaris arundinacea	reed canarygrass	No	38%
	Picea sitchensis	Sitka spruce	Yes	38%
	Rosa pisocarpa	clustered wildrose	Yes	18%
	Spiraea douglasii	hardhack	Yes	18%
	Rubus spectabilis	salmonberry	Yes	17%
	Rubus ursinus	creeping blackberry	Yes	16%
	Lysichiton americanus	skunk cabbage	Yes	13%
	Rosa nutkana	Nootka rose	Yes	11%
	Typha latifolia	cattail	Yes	11%
	Athyrium filix-femina	ladyfern	Yes	8%
	Gaultheria shallon	salal	Yes	6%
	Lonicera involucrata	twinberry	Yes	6%
	Alnus rubra	red alder	Yes	3%
	Cornus sericea	red-osier dogwood	Yes	3%
	Frangula purshiana	cascara	Yes	3%
	Heracleum maximum	cow parsley	Yes	3%
	Lotus corniculatus	garden bird's-foot trefoil	No	3%
	Mahonia aquifolium	tall Oregon grape	Yes	3%
	Maianthemum dilatatum	false lily-of-the-valley	Yes	3%
	Polystichum munitum	sword fern	Yes	3%
	Ribes divaricatum	wild gooseberry	Yes	3%
	Taxus brevifolia	western yew	Yes	3%
	Amelanchier alnifolia	serviceberry	Yes	2%
	Carex stipata	sawbeak sedge	Yes	1%
	Galium trifidum	small bedstraw	Yes	1%
	Impatiens capensis	jewelweed	No	1%
	Symphoricarpos alba	snowberry	Yes	1%
	Vicia nigricans ssp. gigantea	black vetch	Yes	1%

Table 7 Natural sites vicinity rapid vegetation assessment results. Plants identified by species and percent cover (%). Native and non-native species indicated with a Yes or No, respectively. An “X” denotes identification to the genus level only.

NATURAL AREAS

Quilceda Marsh (QM)

Heron Point (HP)tt

Otter Island (OI)

Figure 11 Natural area site photos, courtesy of EarthCorps (2013).

6.4.2 Rapid Vegetation Assessment – Restoring Areas

The five restoring areas that were sampled varied historically from the estuarine emergent zone to the emergent/forested zone and up to the forested riverine/tidal zone. Today these areas are all classified as estuarine emergent, dominated by Lyngbye's sedge and soft-stemmed bulrush *(Schoenoplectus tabernaemontani*) (see Table 8). The exception was Union Slough which is currently a mudflat, slowly vegetating at the margins. Photos of each site are featured in Figure 12.

Site	*Scientific Name*	Common Name	Native	Average Cover Class (%)
Marysville Mitigation (MA)	*Carex lyngbyei*	Lyngbye's sedge	Yes	98%
	Callitriche sp.	water-starwort	X	3%
	Phalaris arundinacea	reed canarygrass	No	3%
	Schoenoplectus tabernaemontani	soft-stemmed bulrush	Yes	3%
North Ebey (NE)	*Schoenoplectus tabernaemontani*	soft-stemmed bulrush	Yes	63%
	Typha angustifolia	narrow-leaf cattail	No	38%
	Sium suave	hemlock water-parsnip	Yes	3%
	Deschampsia cespitosa	tufted hairgrass	Yes	3%
	Agrostis sp.	bentgrass	X	1%
	Carex lyngbyei	Lyngbye's sedge	Yes	1%
	Epilobium ciliatum	fringed willowherb	Yes	1%
	Potentilla anserina ssp. pacifica	Pacific silverweed	Yes	1%
	Grindelia integrifolia	Pacific gumweed	Yes	1%
	Sagina Sp.	pearlwort	X	1%
	Lilaeopsis occidentalis	western grasswort	Yes	1%
	Galium trifidum	small bedstraw	Yes	1%
Spencer Island (SP)	*Carex lyngbyei*	Lyngbye's sedge	Yes	98%
	Oenanthe sarmentosa	water parsley	Yes	8%
	Epilobium ciliatum	fringed willowherb	Yes	3%
	Phalaris arundinacea	reed canarygrass	No	3%
	Schoenoplectus tabernaemontani	soft-stemmed bulrush	Yes	3%
	Typha latifolia	cattail	Yes	3%
	Callitriche sp.	water-starwort	X	1%
Smith Island City - South (SS)	*Carex lyngbyei*	Lyngbye's sedge	Yes	86%
	Schoenoplectus tabernaemontani	soft-stemmed bulrush	Yes	18%
	Phalaris arundinacea	reed canarygrass	No	3%
	Sium suave	hemlock water-parsnip	Yes	3%
	Deschampsia cespitosa	tufted hairgrass	Yes	3%
	Callitriche sp.	water-starwort	X	1%
	Symphyotrichum subspicatum var. subspicatum	Douglas aster	Yes	1%
	Triglochin maritima	seaside arrow-grass	Yes	1%
	Rumex crispus	curly dock	No	1%
	Mimulus guttatus	seep monkey-flower	Yes	1%
Union Slough (US)		mud flat	X	98%

Table 8 Restoring sites vicinity rapid vegetation assessment results. Plants identified by species and percent cover (%). Native and non-native species are also indicated with a Yes or No, respectively. An "X" denotes identification to the genus level only.

RECOVERING AREAS

North Ebey (NE)

Union Slough (US)

Spencer Island (SP)

Smith Island South (SS)

Marysville (MA)

Figure 12 Restoring area site photos, courtesy of EarthCorps (2013).

6.4.3 Rapid Vegetation Assessment – Potential Restoration Areas

The four potential restoration sites are discussed in the *Snohomish Estuary Wetland Integration Plan* (City of Everett Department of Planning and Community Development, 1997) and other restoration plans (such as Puget Sound Nearshore Ecosystem Restoration Project, PSNERP). These sites vary in current land use from abandoned agricultural land to mixed agriculture land and forest. Non-native species such as reed canarygrass and bentgrass *(Agrostis sp.*) are dominant at two of the sites, Qwuloolt and the WDFW Wetland (Table 9). The two other sites, Smith Island County and WDFW Forested, are dominated by native Palustrine wetland species such as spike rush (*Eleocharis palustris*) and forested wetland species such as western redcedar *(Thuja plicata*), skunk cabbage (*Lysichiton americanus*), bitter cherry (*Prunus emarginata*), salmonberry (*Rubus spectabilis*), and hardhack. Photos of each of these sites are shown in Figure 13.

Site	*Scientific Name*	Common Name	Native	Average Cover Class (%)
Qwuloolt (QW)	*Phalaris arundinacea*	reed canarygrass	No	98%
Smith Island County (SN)	*Eleocharis palustris*	spike rush	Yes	63%
		litter	X	18%
	Phalaris arundinacea	reed canarygrass	No	3%
	Potentilla anserina ssp. pacifica	Pacific silverweed	Yes	3%
	Juncus effusus	soft rush	Yes	3%
WDFW Wetland (WW)	*Agrostis sp.*	bentgrass	X	98%
	Alopecurus geniculatus	water foxtail	Yes	18%
	Phalaris arundinacea	reed canarygrass	No	3%
	Atriplex sp.	saltbush	X	1%
WDFW Forested (WF)	*Athyrium filix-femina*	ladyfern	Yes	38%
	Lysichiton americanus	skunk cabbage	Yes	38%
	Thuja plicata	western red cedar	Yes	38%
	Prunus emarginata	bitter cherry	Yes	38%
	Rubus spectabilis	salmonberry	Yes	18%
	Spiraea douglasii	hardhack	Yes	18%
	Alnus rubra	red alder	Yes	8%
	Salix sitchensis	Sitka willow	Yes	8%
	Phalaris arundinacea	reed canarygrass	No	3%
	Cornus sericea	red-osier dogwood	Yes	3%
	Galium aparine	common bedstraw	Yes	1%
	Rubus ursinus	creeping blackberry	Yes	1%
	Scutellaria sp.	skullcap	Yes	1%
	Vaccinium parvifolium	red huckleberry	Yes	1%

Table 9 Potential restoration sites vicinity rapid vegetation assessment results. Plants identified by species and percent cover (%). Native and non-native species are also indicated with a Yes or No, respectively. An "X" denotes identification to the genus level only.

POTENTIAL RESTORATION AREAS

Qwuloolt (QW)

WDFW Forest (WF)

Smith Island North (SN)

WDFW Wetland (WW)

Figure 13 Potential restoration area site photos, courtesy of EarthCorps (2013).

6.5 Lab Results

Measured values for bulk density, organic matter content, and carbon content for each core section are reported in Appendix A (Tables 14 -25). Bulk density ranged from 0.08 g cm^{-3} at the WDFW Forested site (Figure 31) to 1.50 g cm^{-3} at the Marysville site (Figure 22). The lowest organic matter content (2.3%) was recorded at the Smith Island City site (Figure 28) and the highest, 85.1%, at the WDFW Forested site (Figure 31). Carbon content ranged from a low of 0.51% at the Smith Island City site (Figure 28) to a high of 45.78% at the WDFW Forested site (Figure 31). Carbon content values for all core sections were approximately 55% of their corresponding organic matter values (Figure 17).

6.5.1 Carbon and Mineral Mass

The total mass of carbon and mineral matter in the top 30 cm are reported for all sites in Table 10. Total carbon mass ranged from a low of 5.37 kg m^{-2} at Union Slough (an unvegetated mudflat) to a high of 23.36 kg m^{-2} at the WDFW Wetland site. Total mineral mass in the top 30 cm was lowest at the WDFW Forested site with 75.61 kg m^{-2}, and highest at the Smith Island City site with 331.08 kg m^{-2}.

6.5.2 Carbon and Mineral Density

The average carbon density in the top 30 cm ranged from 0.018 g C cm^{-3} at Union Slough (Figure 15) to 0.078 g C cm^{-3} at the WDFW Wetland site (Figure 16). Similar to total mineral mass, the average mineral density in the top 30 cm was lowest at the WDFW Forested site with 0.252 g cm^{-3} and highest at the Smith Island City site with 1.104 g cm^{-3}. Average carbon and mineral densities for all sites are reported in Table 10.

6.5.3 Accretion Rates

Sediment accretion rates ranged from 0.18 cm yr^{-1} to 1.61 cm yr^{-1} (Table 11), with the lowest rate at Heron Point (Figure 24), and the highest at North Ebey (Figure 23). The accretion rate at North Ebey appeared to decrease below 50 cm, corresponding to the time of restoration. Only the samples above 50 cm were included in the determination of the reported accretion rate at the North Ebey site, representing the post-restoration accretion rate.

6.5.4 Carbon and Mineral Accumulation Rates

Carbon accumulation rates ranged from a low of 58.0 g C m^{-2} yr^{-1} at Heron Point to a high of 352.1 g C m^{-2} yr^{-1} at North Ebey. Mineral accumulation rates ranged from 484 to 7,585 g m^{-2} yr^{-1}, with the lowest rate again at Heron Point and the highest rate at North Ebey (Table 11).

Site	Site Name	Total carbon mass in top 30 cm (kg C m^{-2})	Average carbon density in top 30 cm (g C cm^{-3})	Total mineral mass in top 30 cm (kg m^{-2})	Average mineral density in top 30 cm (g cm^{-3})
Natural sites					
QM	Quilceda Marsh	7.17 (0.67)	0.024 (0.00)	148.02	0.493
HP	Heron Point	9.85 (0.07)	0.033 (0.00)	82.71	0.276
OI	Otter Island	7.81 (1.72)	0.026 (0.01)	132.66	0.442
Transitional restored sites					
NE	North Ebey	6.48 (0.12)	0.022 (0.00)	141.38	0.471
SP	Spencer Island	8.29 (0.73)	0.028 (0.00)	182.69	0.609
MA	Marysville Site	9.74 (0.09)	0.032 (0.00)	246.63	0.822
SS	Smith Is. - City	6.11 (2.17)	0.020 (0.01)	331.08	1.104
US	Union Slough	5.37 (0.15)	0.018 (0.00)	236.98	0.790
Future restoration sites					
QW	Qwuloolt	11.31 (0.03)	0.038 (0.00)	188.45	0.725
SN	Smith Is. - County	18.52 (1.59)	0.062 (0.01)	163.15	0.544
WW	WDFW Wetland	23.36 (1.79)	0.078 (0.01)	87.34	0.291
WF	WDFW Forested	15.34 (0.29)	0.051 (0.00)	75.61	0.252

Table 10. Total carbon mass and average carbon density in the top 30 cm of cores, with averages (± standard deviation) reported for each site (n = 2). Mineral mass and mineral density were determined using one core from each site.

Site	Site Name	Sediment accretion rate (cm yr^{-1})	Carbon accumulation rate (g C m^{-2} yr^{-1})	Mineral accumulation rate (g m^{-2} yr^{-1})
QM	Quilceda Marsh	0.43	110.2	2134
HP	Heron Point	0.18	58.0	484
OI	Otter Island	0.58	173.1	2543
NE	North Ebey	1.61	352.1	7585
SP	Spencer Island	0.35	91.4	2148

Table 11. Rates of sediment accretion, carbon accumulation, and mineral accumulation for five sites. Accretion rates were determined from the distribution of excess ^{210}Pb activity with depth using one core from each site. Carbon and mineral accumulation rates were calculated from the accretion rates and the average carbon or mineral density within the top 30 cm.

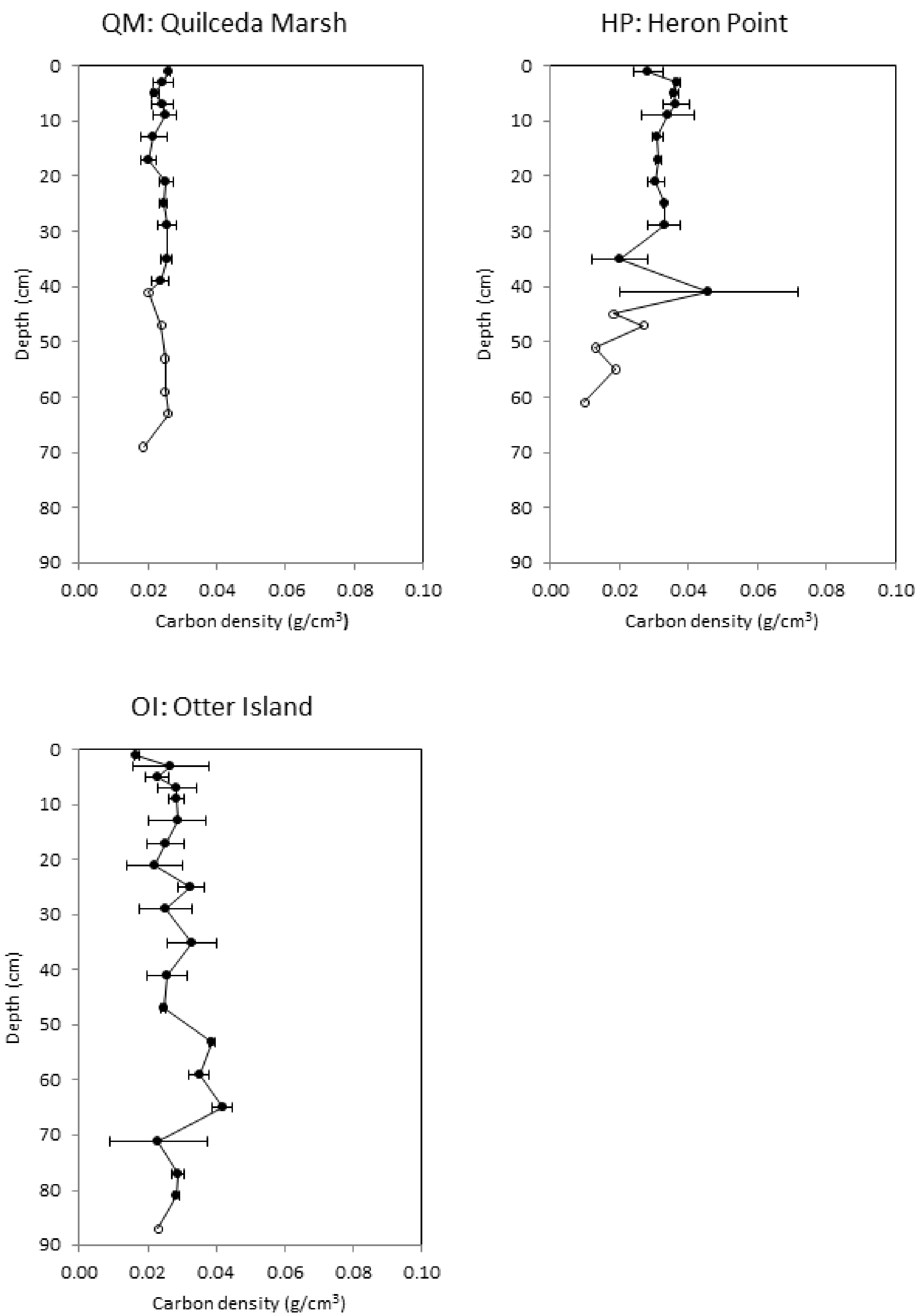
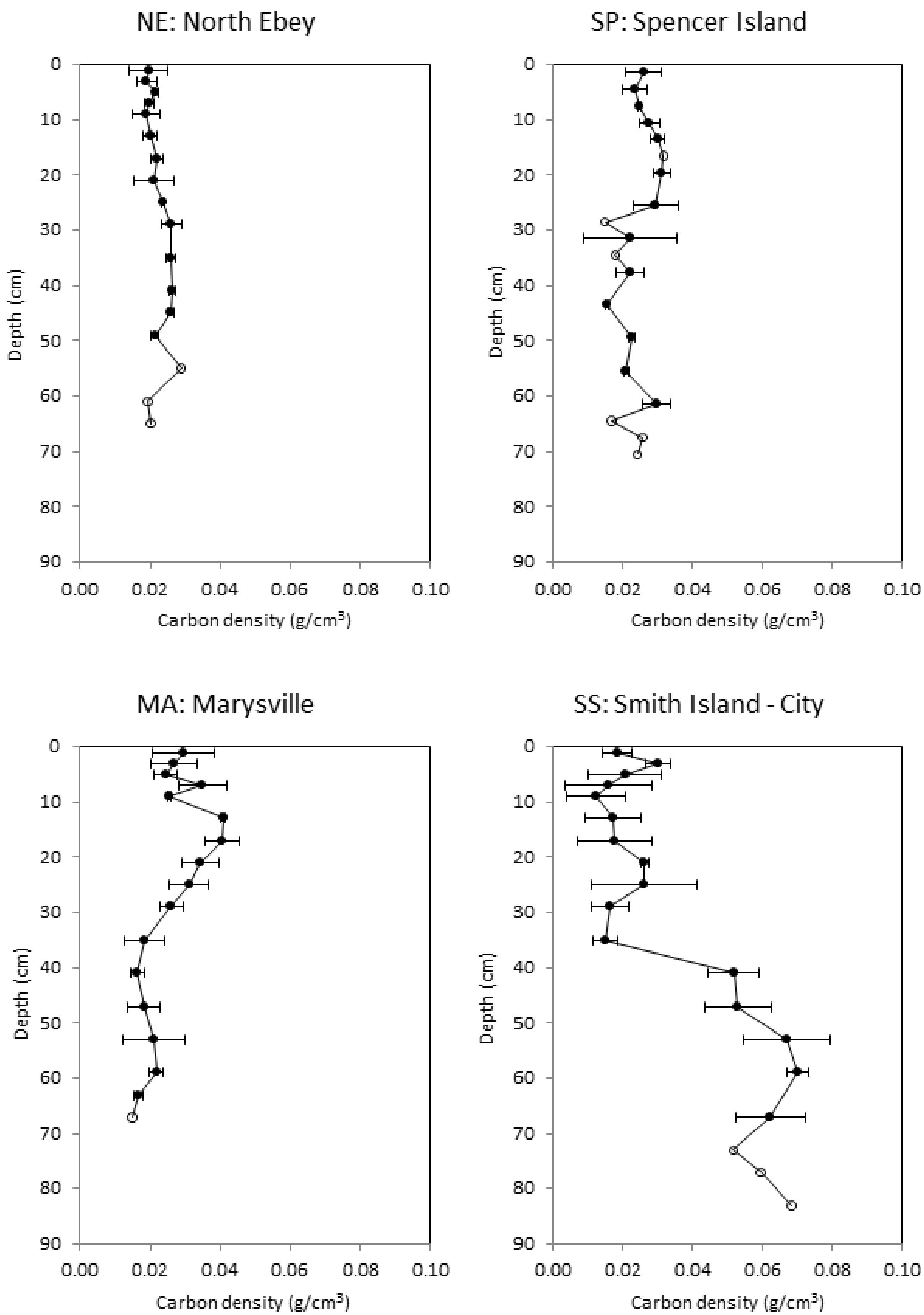


Figure 14. Carbon density profiles at sites with no diking or drainage alterations. Error bars represent ± 1 standard deviation (n = 2). Hollow points denote depths with no replication.



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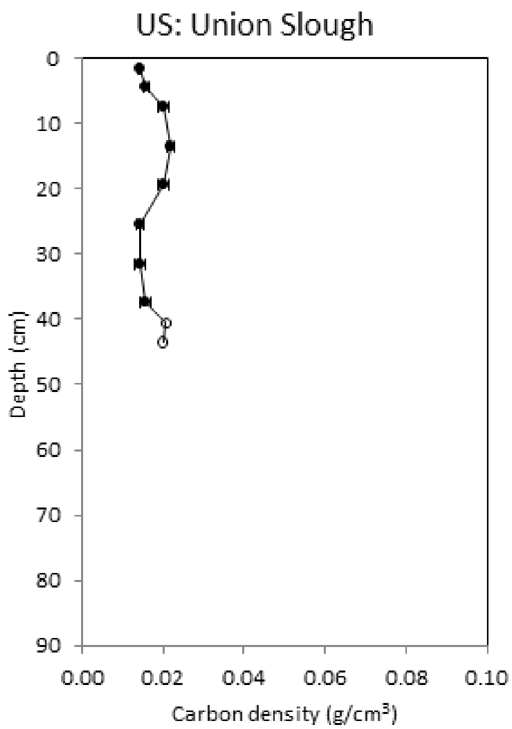

Figure 15. Carbon density profiles at sites restoring from diked farmland to restored tidal wetland. Error bars represent ± 1 standard deviation (n = 2). Hollow points denote depths with no replication.

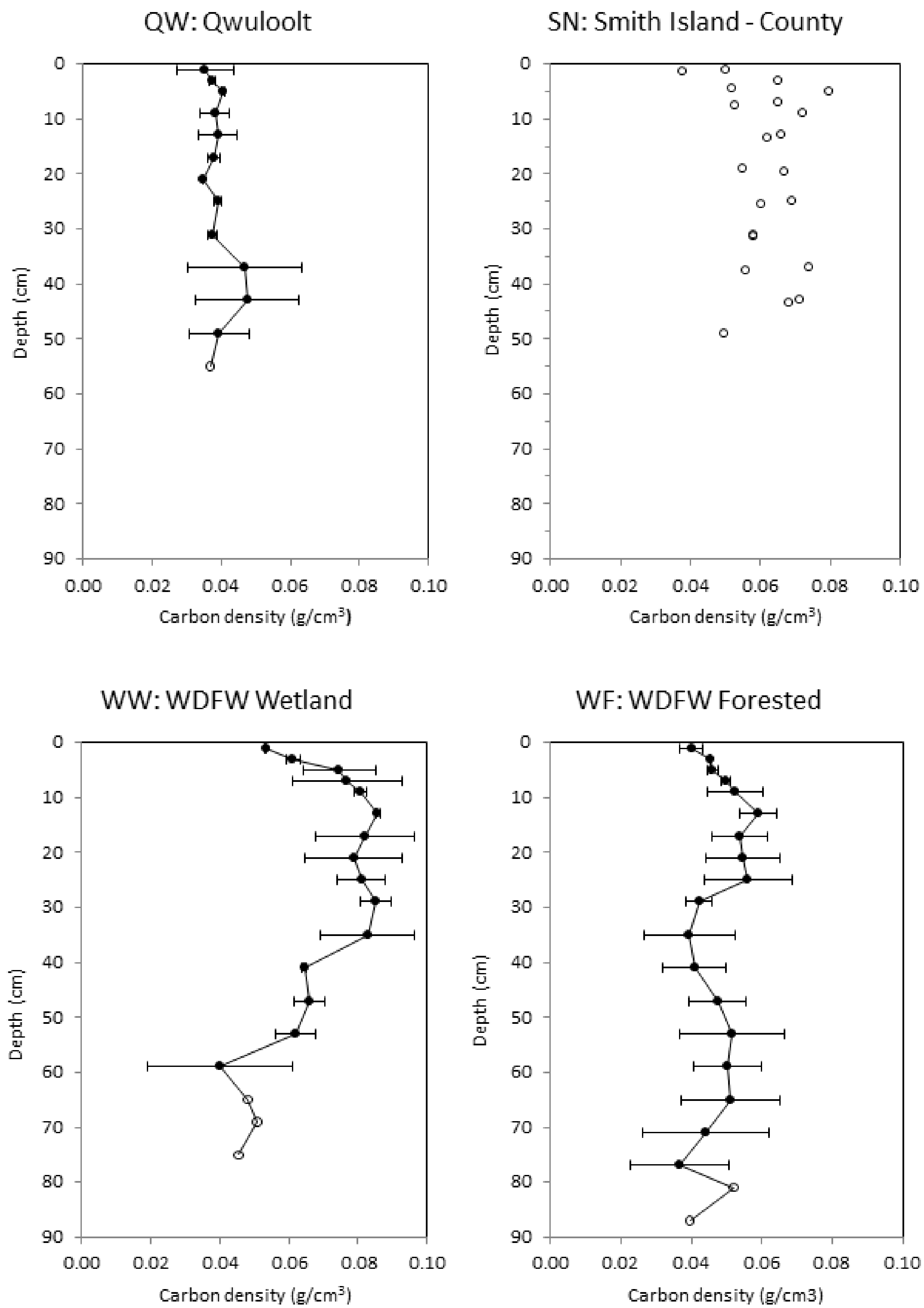

Figure 16. Carbon density profiles at sites intended for future restoration. Error bars represent ± 1 standard deviation (n = 2). Hollow points denote depths with no replication.

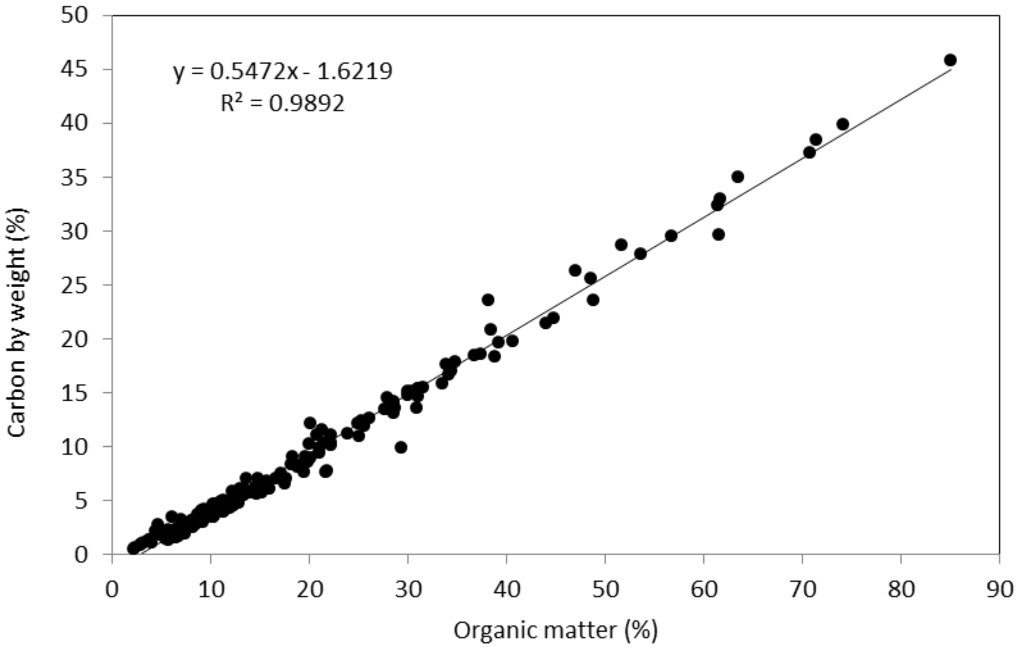

Figure 17. Relationship between organic matter and carbon content based on soil sections from one core at each site.

6.6 Geomorphic Change

6.6.1 Hypsometric Analysis

The geomorphic form of an estuary is readily described by a graph of the open surface area at defined elevations. This graph is called a hypsometric curve (Figure 18), against which we can assess the geomorphic changes that have occurred historically and could occur in the future through management activities. Estimates of historic subsidence volume can be calculated by referencing estimates of historic wetland surface elevation against the existing surface elevation. When the elevation at which tidal wetlands colonize intertidal sediments is known, then from this curve it is possible to calculate the volume of the estuary that would be restored to mudflat or emergent marsh, should restoration actions take place. By coupling the volumetric analysis with soil carbon density derived from field data, it is possible to estimate the quantity of carbon that has been released when wetlands have been drained, and to estimate the carbon sequestration that would result from rebuilding emergent marshes. By including topography above tidal elevations, we can also assess the potential area that would be converted to wetlands with sea level rise, should those lands be reconnected to tidal influence.

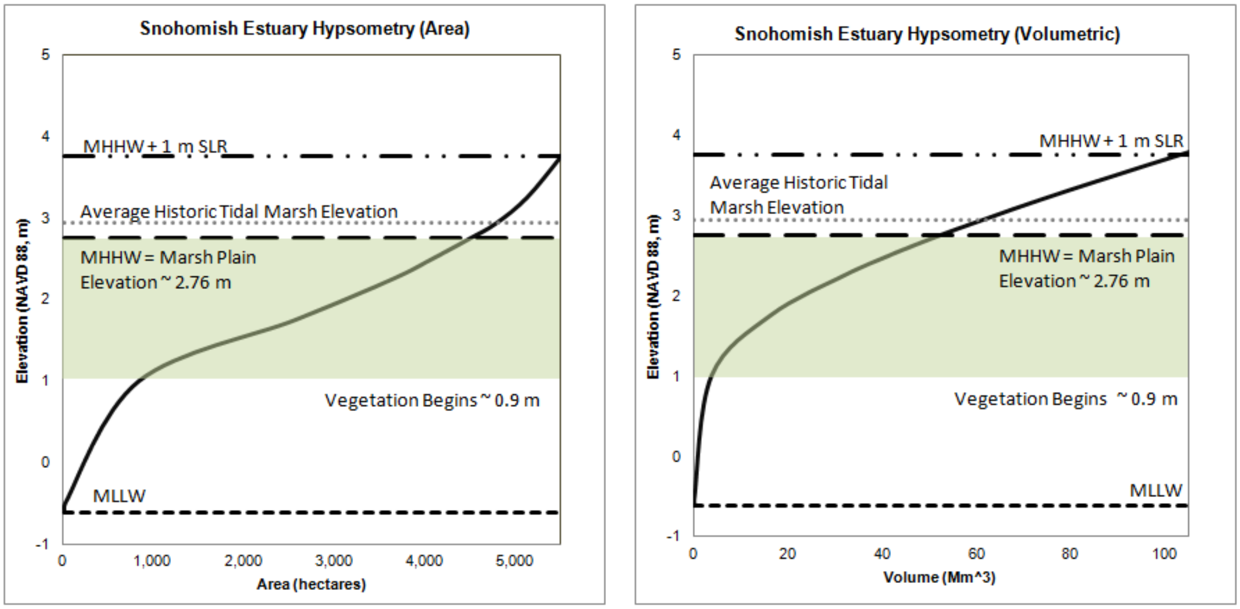

Figure 18 Hypsometric analysis of entire project area (ha).

The graphs shown above reflect the area, left, and volume, right, of space between a vertical plane and the 2006 landscape surface (digital elevation model, LiDAR). The current MLLW elevation is shown in a short dashed line at an elevation of -0.62 m. The MHHW line is in a long dash at an elevation of 2.76 m and the current MHHW elevation plus 1 m of sea level rise is shown at 3.76 m with a dash-dot-dot line. These graphs are useful for determining the empty area and volume at different elevations. The x-intercept at MHHW shows the approximate area (ha) and volume (million m^3) available for tidal wetland creation. In this case, the area at 2.76 m is about 4,500 ha and the volume is about 55 Mm^3. The graph on the left shows the potential restoration area, the curve above shows the volume between the restored elevation (MHHW) and the existing surface, about 52,517,533 m^3, or 52.5 Mm^3. The area and volume increases up and out with 1 m of sea level rise to 5,500 ha and 132 Mm^3, respectively. This volume is used to calculate the carbon storage potential for this project, which increases with sea level rise, if marshes can build at a rate equal or greater than sea level rise rates. The elevation at which carbon storage begins is the elevation at which plants begin to colonize mud flats, changing the mud flat into an emergent tidal wetland. This elevation is about 0.9 m in the Snohomish Estuary. As the tidal wetland soils build and increase in elevation, the carbon storage increases. For the purposes of this study, the wetlands are predicted to build up to MHHW elevation of 2.76 m up to 3.76 m with sea level rise. These assumptions and predictions are discussed in greater detail further in sections 7 and 8.

6.6.2 Elevation of Field Sampling Locations

Determining the elevation of field sites provides validation of the DEM used in the geomorphic assessment as well as confirming the elevations at which different tidal wetland plant communities establish in the Snohomish Estuary. When combined with the field data, the elevations inform the conceptual model of landscape change through time.

Of the 12 field sites sampled in the Lower Snohomish Estuary, three were in undisturbed locations (Quilceda Marsh, Heron Point, and Otter Island). Currently, Quliceda Marsh is a young low marsh, with a surface elevation of 2.0 m NAVD88. Wetlands at both Heron Point and Otter Island were sampled at 2.6 m NAVD88.

The remaining nine sampling locations were on drained lands, and all are subsided relative to natural tidal wetland elevations. In this estuary, the estuarine emergent marshplain was historically about 1.7 m (NAVD88), and forested riverine/tidal wetland zone was 2.3 m (NAVD88). Today, these wetland habitat types are formed at about 2.0 and 2.6 m (NAVD88), respectively, due to increases in tidal wetland elevation with sea level rise.

Figure 19 shows the current elevation of the 12 sites sampled during the fieldwork portion of this project (orange bars). The three natural areas are the highest in elevation, whereas the sites that have yet to be breached are some of the lowest. The green shaded area bounded by dashed green lines shows the elevation at which vegetation begins to colonize tidal wetland in the Snohomish Estuary, 0.9 m, up to the elevation at which tidal wetlands are "maintained" by the tidal and sediment fluxes of the estuary, at MHHW, in this case 2.76 m. This shaded band reflects the potential soil carbon sequestration depth.

Subsidence was calculated using the difference in current elevation from historical. As noted above, the historical tidal wetland elevations are predicted to have ranged from 1.7 m for emergent tidal wetland to 2.3 m or higher, for forested tidal wetland. Spencer Island, WDFW Forested, and WDFW Wetland show the greatest subsidence ranging from 1.4 to 1.6 m, while Smith Island City, restoring since 2007, shows the least subsidence at 0.7 m. Union Slough, while appearing rather low in the graph below, subsided only a meter or so. This area is at the mouth of the estuary and likely had a prior elevation of about 1.7 m.

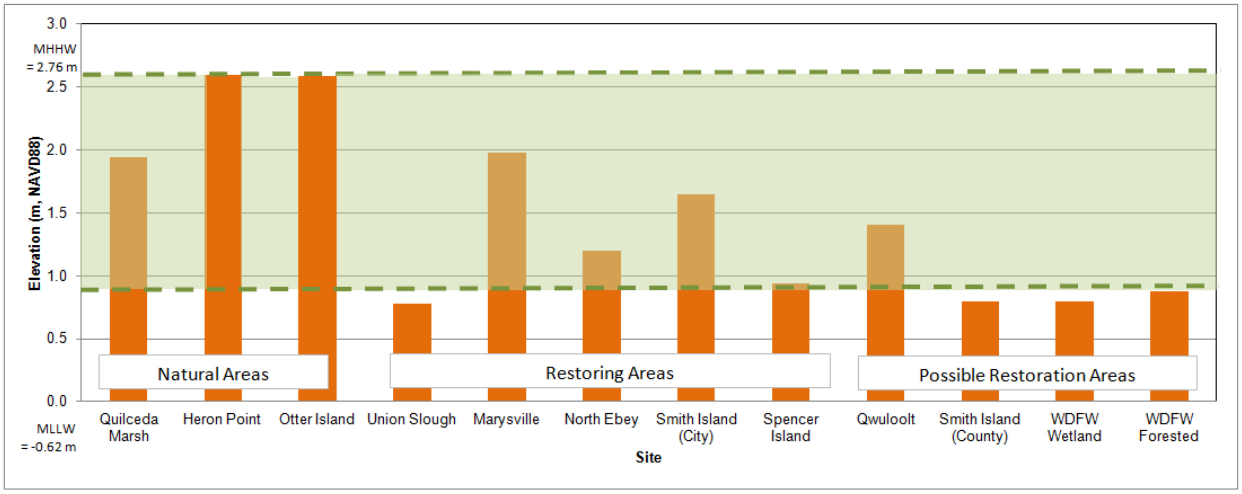

Figure 19 Existing and approximate targeted restoration elevations by site as of 2013. Units are in meters (m), NAVD88.

6.6.3 Calculation of Soil Carbon Emissions

Table 12 divides the estuary into regions and details the calculation of: (1) carbon emissions from drained wetlands; (2) the amount of carbon that would be sequestered were the areas restored to present day marshplain elevations; and (3) the amount of additional carbon sequestration that would result with 1 m of sea level rise. This analysis is also summarized in Table 13.

Estimates of carbon emissions are based upon hypsometric determination of a subsided volume below an assumed historic wetland surface elevation (for emergent and scrub-shrub wetlands, 2.76 m NAVD88; for forested wetlands, 3.5 m NAVD88) and an ascribed conservative soil carbon density value derived from the field analysis of 0.025 tC m^{-3}. With a calculated subsided volume of 67.7 Mm^3 we derive an estimate of 1.7 MtC released through historic drainage of wetland soils. This does not include loss of carbon pools from living biomass in forested areas.

In estimating carbon sequestration with marsh building, we account only for the carbon that accumulates within soils once the restoring areas have attained an elevation above the lower vegetation colonization elevation, and we assume a maximum attained elevation equal to that of MHHW. If all of the restorable land within the tidal reach of the estuary were restored (4,393 ha), the volume of soil that would include carbon derived from *in situ* plants is 4.9 Mm^3 resulting in 1.22 MtC sequestered as the marshes rebuild to present day marshplain elevations. The 1,353 ha of land already planned for restoration will sequester 0.32 MtC as wetlands rebuild to their marshplain elevations. Because we do not account for carbon that accumulates above MHHW as the wetlands transition from emergent wetland to scrub-shrub and forested wetlands, our estimates of carbon sequestration over time are conservative.

In calculating the ongoing sequestration with 1 m of sea level rise, we use the measured marsh building rate of up to 16 mm yr^{-1}, which suggests that restoring wetlands in the Snohomish Estuary potentially would be resilient to high rates of sea level rise. With 1 m of sea level rise, total wetland area on planned restoration sites increases only slightly to 1,594 ha, resulting in an additional 0.38 MtC stored in accumulating soils between present day MHHW and future MHHW. Total carbon sequestration with sea level rise from present day conditions would total 0.70 Mt.

Were the whole estuary to be restored under a future sea level rise scenario of 1 m, total tidal wetland area would increase to 5,258 ha, resulting in an additional 1.2 MtC accumulation of carbon and a total sequestration of soil carbon of 2.45 Mt from existing conditions.

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	Current Conditions (Note 1)	HS1 - Historic Carbon Emissions (Notes 2 - 4)									FS1 & FS2: Carbon Sequestration to Current MHHW (Note 5)				FS3 & FS4: Carbon Sequestration 1 m SLR (Note 5)		
Snohomish Estuary Zones	Current Condition	Historic Tidal Wetland Elevation (m NAVD88)	Area at Tidal Wetland Elevation (ha)	Subsided Soil Volume (M m^3)	Soil Accretion Rate (cm yr^{-1})	Accreted Soil Volume Since Breach (M m^3)	Total Subsided Soil Volume (M m^3)	Accreted Soil Volume Carbon Sequestration (T C)	Historic Soil Carbon Emissions (T C)	Historic Soil Carbon Emissions per hectare (T C ha^{-1})	Soil Volume Below Marsh Colonization (M m^3) See Note 5.	Soil Volume Above Marsh Colonization (M m^3)	Area At Marsh Colonization (2.76 m) (ha)	Future Soil Carbon Storage at Elevation 2.76 m (T C)	Additional Soil Volume with 1 m Sea Level Rise (M m^3)	Total Area with 1 m Sea Level Rise (ha)	Additional Soil Carbon Storage with 1 m Sea Level Rise (T C)
Otter Island	Natural State	-		-	-	-	-	unknown	-	-	-	-		-	0.4		-11,000
Quilceda Marsh	Natural State	-		-	-	-	-	unknown	-	-	-	-		-	1.3		-32,561
Heron Point	Natural State	-		-	-	-	-	unknown	-	-	-	-		-	0.1		-3,208
Everett Riverfront	Natural State	-		-	-	-	-	unknown	-	-	-	-		-	0.1		-2,720
Blue Heron	Restoring	2.6	177	1.93	1.61	0.55	2.48	-13,766	61,928	298	0.04	2.17	181	-54,219	1.9	192	-46,716
North Ebey Island	Restoring	2.6	168	0.84	1.61	2.24	3.08	-56,009	77,121	227	0.06	1.09	211	-27,310	2.5	260	-62,343
Smith Island	Restoring	2.6	448	3.96	0.35	0.29	1.19	-7,220	29,803	46	0.09	4.60	467	-115,098	5.1	535	-126,289
Spencer Island	Restoring	3.3	153	0.93	0.35	1.10	1.52	-27,446	37,880	223	0.05	1.56	143	-39,055	1.5	158	-38,085
Qwuloolt Marsh and Marysville Restoration Site	Partially Restored	3.3	209	2.55	1.61	0.01	2.56	-302	64,094	275	0.04	1.53	180	-38,203	2.0	218	-51,181
East River/Slough Bank to Valley Wall	Diked	3.3	629	9.40	N/A	N/A	9.40		234,892	266	0.12	6.26	573	-156,593	6.3	669	-156,543
Ebey Island	Diked	3.3	1481	26.63	N/A	N/A	26.63		665,727	420	1.42	17.99	1,389	-449,666	14.7	1,517	-368,110
Everett Marshland	Diked	3.3	1457	20.81	N/A	N/A	20.81		520,135	213	0.76	13.34	1,220	-333,470	14.5	1,651	-362,180
Everett Industrial Waterfront	Developed	2.6	27	0.65	N/A	N/A	0.65		16,196	107	0.24	0.45	29	-11,214	0.4	59	-10,591
Full Estuary Restoration Totals		**2.9**	**4749.0**	**67.7**		**4**	**68.3**	**-104,743**	**1,707,775**	**256**	**Full Restoration (FS2 & FS4):**	**49**	**4,393**	**-1,224,827**	**49**	**5,258**	**-1,222,037**
Partial Restoration Values											**Partial Restoration (FS1 & FS3):**	12.82	**1,353**	**-320,570**	15	**1,594**	**-375,319**
											Cumulative Full Restoration:		**4,393**	**-320,570**		**5,258**	**-2,446,864**
											Cumulative Partial Restoration:		**1,353**	**-320,570**		**1,594**	**-695,889**

Assumptions and Notes

Note 1. See report for detailed description of sites and zones.

Note 2. Assume historic elevation was that of the historic habitat, adjusted for SLR, with exception of Scrub-shrub tidal wetland areas = 2.6 m. This overestimates the historic elevation for this area. Forested tidal wetlands are underestimated at 3.3m.

Note 3. Subsided volumes in restoring areas augmented with accreted volumes to estimate total subsided volume. Accretion rates based on nearest 2013 sample site with similar current conditions.
Note 4. Used average carbon density of 0.025, based on 2013 natural tidal wetland carbon density.
Note 5: Tidal wetland plant colonize at 0.9 m. Soil volume above this elevation calcuated only to 2.76 m (MHHW),underestimating future carbon storage. Assuming full restoration of all areas.
Note 6: Carbon stock estimates based upon a conservative carbon density estimate of 0.025 gC cm^{-3}.

Table 12 Change in Soil Carbon Stock due to Historic Land Use Change and with Future Scenarios of Restoration.

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Scenario	Elevation (m NAVD88)	Area (ha)	Soil Carbon Emissions (t C)	Forest Biomass Carbon Emissions (t C)	Total Emissions (t C)
HS1: Historic Wetland Drainage	2.6-3.3	4,749	1,707,775	2,811,654	4,519,429
FS1: Planned and Existing Restoration, Restore to Current Tidal Wetland Elevation (2.76 m)	0.9-2.76	1,353	-320,570	-	-320,570
FS2: Planned and Existing Restoration, Restore to Future Tidal Wetland Elevation (3.76 m)	2.76-3.76	1,594	-375,319	-	-695,889
FS3: Restore Entire Estuary to Current Tidal Wetland Elevation (2.76 m)	0.9-2.76	4,393	-1,224,827	-	-1,224,827
FS4: Restore Entire Estuary to Future Tidal Wetland Elevation (3.76 m)	2.76-3.76	5,258	-1,222,037	-	-2,446,864
Notes: Conservative goal of restoration is to return estuary to emergent tidal wetland elevation. Emergent and scrub-shrub tidal wetland biomass was indeterminate. For these reasons, forest biomass carbon emissions were not calculated for future scenarios. Far right column shows cumulative emissions for different scenarios. Negative numbers reflect carbon sequestration, or net carbon uptake.					

Table 13 Summary of carbon emissions due to subsidence by site and state of restoration. The historic scenario (HS1) is the only scenario that includes forested tidal wetland biomass losses. Future restoration scenarios conservatively estimate carbon emissions with recovery of emergent tidal wetlands only.

6.6.4 Calculation of Total Carbon Emissions

Table 13 summarizes historic carbon stock emissions from wetland soils and forest biomass. Biomass for emergent wetlands was not included in this analysis because comparable data for drained lands were not known. Of the 4.52 Mt of carbon released from clearing and drainage of wetlands in the Lower Snohomish Estuary, 62% of that carbon was stored within forest biomass and 38% in soils.

Looking at soil carbon alone, about 1.71 Mt C lost, planned wetland restoration projects will restore 18% of historic soil carbon losses under present day sea level conditions, or 0.32 Mt C, and 41% with 1 m of sea level rise, 0.70 Mt C. Restoration of the full estuary will recover 71% of the historic soil carbon stock, 1.22 Mt C, and 147% of that lost soil carbon should the marshes build to 1 m above current marshplain elevations, 2.45 Mt C.

7.0 DISCUSSION

7.1 Sediment Profiles

In a sediment core collected from an undisturbed, vegetated estuarine wetland, undergoing a relatively constant rate of accretion, we expect to see the following sediment profile pattern:

- The highest percent organic matter, and associated carbon, at the top of the core, decreasing with depth, as a consequence of both the decomposition of organic matter over time, and of the nature of root growth (typically decreasing exponentially with depth).
- Bulk density increasing with depth, as a consequence of both compaction with depth, and the decomposition of relatively low bulk-density labile organic matter.
- The highest ^{210}Pb activity at the top of the core, decreasing exponentially with depth, as a consequence of its source (the atmosphere) and its known rate of decomposition.

Most of the 12 sites surveyed during this study generally exhibit this pattern, including the two undisturbed and undiked sites, Quilceda Marsh (Appendix, Figure 20), dominated by Lyngbye's sedge, and Otter Island (Appendix, Figure 25), a scrub-shrub forest. The other sites exhibiting this pattern include Marysville, Spencer Island, and Smith Island County.

Union Slough, an unvegetated mudflat, revealed no change in organic matter with depth (as would be expected) although bulk density did increase with depth, indicating some moderate primary compaction. North Ebey also exhibited a pattern (no change in organic matter with depth). However, in this case, this pattern is the consequence of a high rate of mineral sedimentation, which dilutes the organic matter pool.

Discontinuities in the sediment core pattern can be caused by numerous factors, both natural (e.g., large pulses of sediment from episodic flooding, or bioturbation) and anthropogenic (e.g., land use changes such as dams, diking, draining, and farming). The sediment profiles from four sites (Heron Point, Smith Island City, WDFW Wetland, and WDFW Forested) revealed similar discontinuous profiles. Specifically, the top 20 to 40 cm exhibited a typical profile sequence, followed by a transition to higher, or increasing, organic matter and carbon content in the lower core. This pattern is shown most markedly in the Smith Island City core (Appendix A, Figure 28) where the transition is abrupt at a depth of approximately 35 cm. Three of these sites were diked and farmed, starting in 1930; therefore this transition layer at 20 to 40 cm may reflect this landscape-level change.

However, three other sites (Marysville, Spencer Island, and Qwuloolt) that were diked and farmed in the 1930s do not show any evidence of this transition in the sediment profile. Perhaps the farming and associated tilling in these sites thoroughly mixed the soil, thus erasing the transition layer, or perhaps the transition layer is lower in the sediment profile and the core was not long enough to capture the layer.

7.2 Accretion and Accumulation of Organic Matter, Carbon, and Mineral Matter

Total carbon mass and densities in the top 30 cm are highest in the diked and unrestored sites, while the lowest values are found in the transitional, restored sites (Table 10). Natural sites are intermediate in terms of carbon mass and density. However, carbon mass and density should not be confused with *rates* of carbon storage, which are a function of carbon density and, crucially, rates of sediment accretion.

We measured rates of accretion at the three natural sites and two of the restored sites (Table 11). The highest rate of carbon accumulation was found at the North Ebey site (7,585 g C m^{-2} yr^{-1}), which is twice that of any other site (Table 10). However, at 0.22 g C cm^3, North Ebey had one of the lowest average carbon densities in the top 30 cm (Table 10). This is because the site has concomitantly high rates of sediment accretion and mineral accumulation (Table 11). The North Ebey site is illustrative because the high rates of mineral accumulation and vertical accretion indicate great potential for restoration (with regard to resilience to sea level rise) as well as carbon storage.

7.3 Changes in Carbon Storage with Historic Land Use Change

The historic landscape of the Snohomish Estuary composed an ecological gradient of coastal wetlands that transitioned down slope, from occasionally flooded, mixed coastal forested wetlands to scrub-shrub to emergent wetland at typical intertidal elevations. All of these wetlands accumulated and stored carbon within their soils, as well as in above-ground biomass.

Logging of the forests and drainage of the wetland soils resulted in a loss of carbon and a loss of ongoing sequestration processes. There was a release of carbon from cleared forest and drained wetland soils. Assuming that wetlands were in a state of equilibrium and building with sea level it, is reasonable to conservatively assume soil carbon sequestration rates across the system were 1.5 tC ha^{-1} yr^{-1} (Table 11). Consequently, land use change, peaking in the 1930s, has resulted in a loss of 70 years of soil carbon accumulation, or approximately 485,780 tC to date across 4,749 ha.

Carbon stock estimates of historic forests are not well known. Using estimates for living biomass carbon stocks of 80-year-old mixed coastal forest (grand fir, Douglas fir, and Sitka spruce) of 501 tC ha^{-1} (Dushku et al., 2007), we estimate that 2.8 MtC were released with forest clearing[1]. The estimate is conservative as it does not reflect carbon stock densities of fully mature trees, or recognize the carbon stock within dead wood, a significant component of Pacific Northwest lowland forests.

Conversion and drainage of wetlands resulted in subsidence from the former marshplain elevation to the present day surface. For the restoring wetlands (such as North Ebey Island), the full level of subsidence was greater than present day as some amount of wetland rebuilding has occurred. Derived from the field data, we use a soil carbon density for all wetland types of 0.025 g cm^{-3}, and a total subsided volume of 68.3 Mm^3 to calculate a total soil carbon emission of 1.7 MtC across the total area of drained wetlands since 1930 (Table 12). This estimate is sensitive to the assumption about initial elevation of habitat type. We have used existing MHHW, 2.76 m NAVD88, to represent the elevation of

[1] The fate of wood material has not been determined in this study. Some portion of felled wood will likely remain in conserved construction products. Over time this stock will be released as not replaced by new biomass.

emergent and scrub-shrub wetlands. We did not calculate carbon sequestration for wetlands that may eventually become forested. The lowest elevation at which these wetlands transition to forested wetlands is about 3.5 m NAVD88, and it is unknown if forested wetlands are an achievable habitat under future hydrologic and landscape conditions. The estimates of future carbon emissions and sequestration are therefore conservative.

Overall, we calculate that 4.5 MtC have been released from the Snohomish Estuary with wetland drainage. To some degree, the 485,780 tC of forgone carbon sequestration will have been reduced by carbon accumulation on abandoned drained wetlands as vegetation reestablished on rewetted soils. This value has not been determined within this study, but it is likely to be a small fraction of total forgone sequestration potential, because soil accretion rates would presumably be low in diked farmlands. In support of this, we recently measured the rate of long-term accretion in the diked WDFW Wetland area and found it to be the lowest of all sites measured (0.12 cm yr^{-1}).

7.4 Potential Ongoing Emissions

We do not have direct measurement of ongoing elevation change or carbon fluxes from the drained wetlands. While the soil survey data indicate organic soils remain in the lower estuary, we speculate that ongoing emissions have declined from maximum rates for the following reasons:

- Drained wetlands lose organic carbon quickly in the months and years following drainage. If soils are mineral, then emissions tend to decline toward zero over time. If organic emissions persist, it is until the carbon stock is fully depleted or soil water levels change.
- Distribution of Palustrine wetlands and presence of reed canarygrass suggest a high water table in subsided areas of the estuary, reducing rates of ongoing carbon loss.
- Measured subsidence levels are similar to those observed in mineral soil areas.

Further analysis is required to determine whether soils on subsided lands are continuing to release carbon. Halting those emissions as part of a restoration process would be a beneficial contribution to GHG management.

7.5 Potential Future Carbon Stock Changes with Restoration

Understanding how future carbon stocks will recover depends upon the prognosis for wetlands restoration activities under existing and future climate change conditions.

Historically, forested tidal wetlands changed to subsided drained lands. A large percentage, 58%, of these subsided drained lands currently lie within the elevation range suitable for colonization by emergent wetland plants, should they be reconnected to tidal waters. Examples include the potential restoration sites on Ebey Island, WDFW wetland and forested sites, Everett Marshland, and Qwuloolt. At such elevations wetland restoration can be supported by both a return of the mineral sediment supply from the river and the organic contributions from vegetation. While organic contributions from vegetation are not critical for reestablishment of wetlands in an estuary that receives sediment delivery from the catchment, it does offer an accelerated path to vegetation reestablishment and tidal wetlands rebuilding (Crooks et al., 2011).

Further, wetland rebuilding capacity in the Snohomish Estuary is greatly supported by the presence of the plants themselves, such as Lyngbye's sedge and bulrush species. Through rhizome production these plants offer high capacity to rebuild a soil fabric, as the wetland accretes from low intertidal elevations to a dynamic equilibrium at MHHW. Soil accretion rates are facilitated by the robust structure of tangled rhizome mat upon which the soil fabric is built. Sustained accretion rates measured at North Ebey Island of 1.6 cm per year are very high values compared to those measured elsewhere in the Snohomish Estuary. However, the other accretion rate sites were in scrub-shrub tidal wetland areas, or did not have the hydrological connectivity to facilitate high sediment load. Furthermore, similar rates of accretion have been observed in similar restoring bulrush (*Schoenoplectus)* marshes of San Francisco Bay (Orr et al., 2003). Moreover, these plants likely offer high capacity to build resiliently with high rates of sea level rise. The rates of wetland building match or exceed rates of sea level rise projected this century under higher IPCC (2013) scenarios. Once these tidal wetlands have attained equilibrium with tidal marshplain elevations, typically at MHHW, then the rate of tidal wetland building will decline to that of ambient rates of sea level rise.

An additional consideration is the slope of the land available for tidal wetlands to migrate into as sea level rises, should levees be removed. The Snohomish Estuary has areas that are currently just above MHHW and present an opportunity for additional tidal wetland restoration. Such areas enhance the resilience of restoring tidal wetlands and the potential to maintain carbon sequestration.

Table 12 summarizes changes in carbon stock under historic emissions and potential future conditions. The future scenarios do not specify time but rather the total carbon stock that would be stored when the wetland rebuilds to an emergent wetland elevation at MHHW. Potentially, given continued mineral sediment supply and low rates of sea level rise, these wetlands could continue to accrete to scrub-shrub or higher elevations. The eventual equilibrium habitat would also depend on salinity concentrations at a given site. This analysis assumes that with increasing rates of sea level rise, accreting wetland elevations do not attain an elevation higher than MHHW. As a consequence, the future scenario targets recovered and expanded areas of emergent and scrub-shrub tidal wetland but little recovery of forested wetlands, which historically dominated the Snohomish Estuary landscape. However, the likelihood of recovering forested tidal wetlands may be increased with further restoration in the upper watersheds.

Overall, we characterize the restored wetlands in the Snohomish Estuary to be potentially highly resilient to sea level rise for the following reasons: (1) the high capacity of emergent marsh vegetation in the Snohomish to rebuild marsh; (2) the high percentage of the estuary that is within the elevation range that would be colonized by emergent vegetation if sites were reconnected to tides; (3) the high mineral sediment availability from river and nearshore supplies; and (4) the gradual slope of the upper estuary.

7.5.1 Accounting for Methane and Nitrous Oxide Emissions

No studies were identified that directly measured the emissions of CH_4 or N_2O on undisturbed, drained or restoring wetlands in the Pacific Northwest, representing a significant data gap. Emissions of CH_4 and N_2O gases contribute to global warming, and changes in net flux of these gases should be accounted for in landscape assessments. According to the IPCC Fifth Assessment Report, CH_4 and N_2O each have a global warming potential of 34 and 298 times that of CO_2 over a 100-year timeframe, respectively. The global warming potential of CH_4 has been raised from a factor of 25 to a factor of 34 since the IPCC Fourth Assessment Report.

The approach of the IPCC in land use change assessments is to quantify only those emissions resulting from direct human impacts. Prior to anthropogenic disturbance, our atmosphere was largely in balance with the natural fluxes of gases from the planet. Humans have altered this balance.

Drainage of wetland organic soils is a human impact on GHG fluxes recognized by the IPCC 2013 Wetland Supplement. Drainage of organic wetland soils can release significant amounts of N_2O from nitrogen in organic matter or nitrogen added by fertilization. Standing water in drainage channels is also a significant source of CH_4 emissions. Rewetting organic soils decreases N_2O emissions to close to zero and increases CH_4 emissions compared to the drained state as oxygen levels in the soil drop and methanogenesis starts again.

Thus, drained wetlands are likely a source of N_2O which is not accounted for in this study, and restoring these areas to wetlands leads to a reduction in that emission. However, CH_4 emissions are a byproduct of organic matter decomposition by wetlands at salinities less than half that of seawater (Poffenbarger et al., 2011). The wetlands in the Snohomish Estuary fall within the low salinity range at which CH_4 emissions would result.

In the absence of direct measurements, and as a first approximation, it is to possible estimate the net balance in GHG emissions and removals for restoring wetlands by utilizing the IPCC Tier 1 emissions factor for CH_4 from restoring tidal wetlands (IPCC Wetland Supplement, 2013, Table 4.14). An emissions factor due to radiative forcing of 0.194 $tCH_4\ ha^{-1}\ yr^{-1}$ equates to 1.80 $tC\ ha^{-1}\ yr^{-1}$. This equivalent emission is roughly in balance with soil carbon sequestration rates within the forested wetlands at Otter Island (1.73 $tC\ ha^{-1}\ yr^{-1}$) and half that of the restoring tidal wetland at North Ebey Island (3.52 $tC\ ha^{-1}\ yr^{-1}$).

Therefore, restoring emergent tidal wetlands in the Snohomish Estuary appears to be a net remover of GHGs. The value of this removal is approximately 1.7 $tC\ ha^{-1}\ yr^{-1}$. With 1,353 ha of planned restoration, or 4,393 ha of full estuary restoration, this equates to estimated removals of 2,300 $tC\ yr^{-1}$ and 7,468 $tC\ yr^{-1}$, respectively, in the Snohomish Estuary.

These carbon sequestration values are considered conservative. On the emissions side, they do not include possible ongoing CO_2, CH_4 and N_20 emissions from the existing drained wetlands, ditches and wet soils, fertilizer, and cattle. In addition to this conservative emissions estimate, the carbon sequestration benefits of restored tidal wetland vegetation are also not taken into account.

Reforestation of former forested tidal wetlands in particular would greatly increase rates of carbon sequestration especially if included as part of an estuary wide ecosystem restoration.

7.5.2 Biomass

Biomass was not measured for this project. However, there are restored and recovering wetland forests within the study area. Forested tidal wetland dominated by Sitka spruce is the historic landscape condition. Old-growth forested wetlands are rare, even rarer than upland old-growth forests. While gathering background literature for this project, the research team found two studies conducted in the forested wetlands of the Snohomish Estuary. According to these two tree core studies, conducted in 1997 and 2005, the oldest Sitka spruce was about 450 years old (Painter, 2007). These studies concluded that these types of forests are slow-growing, and wetland forest trees are significantly smaller than similarly aged trees in a mature upland forest. This observation should be included in future carbon sequestration calculations for restored wetland forests, both in the time it takes to become forested and the amount of carbon sequestered within the biomass of the forest.

7.6 Implications for Management of Coastal Wetlands

This study finds that restoring tidal wetlands within the Snohomish Estuary, as an example of regional practice, would most likely contribute to state and national GHG reduction goals. Landscape-level planning that takes a forward-looking view, by restoring coastal wetlands and lowland forest, in balance with other land uses, would support both climate change adaptation (e.g. improved environmental conditions, reduced flood risk, etc.) and climate change mitigation.

Elements of a coastal lowland climate change mitigation strategy that would be supportive of adaption include the following:

1) Restore tidal wetlands sooner rather than later to:
    a. Reverse emissions, and
    b. Increase resilience of landscape capacity to rebuild prior to accelerated sea level rise.
2) Protect remaining wetlands from drainage or disturbance to conserve carbon stocks that have accumulated over centuries.
3) Quantify GHG emissions and removals on existing drained wetlands to aid GHG management practices.
4) Identify and protect coastal lowlands as buffer areas, providing space for wetlands to migrate with sea level rise. These areas may be planted with lowland forest species providing near-term carbon sequestration and ecological benefits, and if part of a connected floodplain, may provide flood risk reduction benefits.
5) Avoid irreversible, or costly to reverse, activities and land use changes on subsided lowlands that conflict with future adaptation and mitigation opportunities.

7.7 Conservative Approach to Analysis

This analysis applies conservative assumptions in its calculation of carbon fluxes to avoid overstating climate mitigation benefits of wetland restoration. Assumptions include the following:

1. Not accounting for dead wood carbon pools in historic assessment, nor dead wood and living forest biomass in future scenarios.
2. Assuming restoring wetlands build only to emergent marsh elevations and not to former supratidal wetland elevations.
3. Assuming that ongoing emissions from drained lands are zero, therefore underestimating any CO_2 emissions that are occurring, and not recognizing CH_4 and N_2O emissions. Quantifying these emissions in the baseline scenario will increase the relative value of the restoration alternative.

7.8 Study Limitations and Uncertainty

This study benefits from fieldwork conducted in 12 sites across the Snohomish Estuary, representing natural emergent wetlands (n=1), natural forested wetland (n=2), a mudflat (n=1), a restoring emergent wetland (n=1), and drained wetlands with shallow water tables (n=7).

The study was limited in the number of sites available to characterize emergent wetland and restoring wetland conditions. In particular, this study and a similar study in the San Francisco Estuary (Orr et al., 2003) suggest that brackish wetlands consisting of species that contribute to soil matter by rhizome production offer high restoration capacity, resilience to sea level rise, and carbon sequestration benefits. Future quantification is warranted to confirm this finding and build a regional/national database.

Emissions of CH_4 and N_2O currently can be derived only from direct field measurement. Such data are absent for the Pacific Northwest and West Coast in general. While the IPCC provides default values (Tier 1) for emissions of CH_4 at a global scale, for restoring marshes, no emissions were assumed for the drained wetlands. This scenario is likely incorrect and these estimates can be improved upon through direct measurement.

This assessment has not determined the ratio of autochthonous (site derived) versus allochthonous (off-site derived) carbon. Further analysis might reduce the total carbon sequestration by up to 20% (Brian Needleman, unpublished data).

In the Snohomish Estuary, there is some uncertainty as to when the various parts of the estuary were diked and drained. Improved historical knowledge about when particular areas were diked and drained would refine the estimates of ongoing sequestration as well as the historic carbon emissions.

7.9 Comparison with Other Regions

Because of geomorphology, ecology, climate, and the nature of human impacts, it is highly likely that the findings in this study are broadly applicable to estuaries from Northern California, Oregon, Washington, and British Columbia. Tidal wetlands along the Gulf Coast and Atlantic Coast are different in geomorphology, ecology, climate, and the nature of human impacts, and regionally specific studies are recommended. There is a small but growing body of evidence that bulrush (*Schoenoplectus)* species, because of their high productivity and rhizomal root mat, are particularly resilient contributors to tidal wetland and marsh building and carbon sequestration. These species are common in brackish settings along the West Coast, as well as being found in other regions.

Our data indicate that current rates of carbon sequestration, and potential sequestration in restored wetlands, are at least equivalent to sequestration rates measured elsewhere. Specifically, the rates of carbon sequestration measured here in the natural marshes (Quilceda Marsh and Otter Island, 110.2 and 173.1 g C m^{-2} yr^{-1} respectively) are within the range of values measured at other salt and brackish marshes in the United States (see Callaway et al., 2012). The 352.1 g C m^{-2} yr^{-1} measured at the restored North Ebey site greatly exceeds even the global mean of 220 g C m^{-2} yr^{-1} for tidal saline wetlands reported by Chmura et al. (2003), which was later revised to 115 C m^{-2} yr^{-1} (Morris et al., 2012).

8.0 SUMMARY OF MAIN FINDINGS

8.1 Restoration Potential and Sea Level Rise Resilience

Wetlands in the Snohomish Estuary demonstrate great potential for restoration and high resilience to sea level rise. Much of the topography of the drained wetlands lies within the elevation suitable for emergent marsh colonization should the wetlands become tidally reconnected. Slopes within the upper estuary are gradual, offering potential for wetland migration landwards with sea level rise, again subject to floodplain tidal reconnection.

High rates of tidal wetland building, such as that observed on the restoring emergent marsh at North Ebey Island (1.6 cm yr^{-1}), illustrate the capacity of *Schoenoplectus* species, or bulrush, to restore marshes on subsided lands. The capacity of *Schoenoplectus* species to build marsh rapidly has been observed in the San Francisco Estuary (Orr et al., 2003) and on other managed wetlands at rates of several centimeters per year (Miller et al., 2009).

8.2 Historic Carbon Emissions with Wetland Conversion

Historic emissions have been substantial for a relatively small estuary but are likely to be representative of estuaries across the Pacific Northwest. In the Snohomish Estuary, more than 4,749 ha of forested, scrub-shrub, and emergent wetlands have been cleared and drained, releasing 4.5 MtC. This equates to

0.95 MtC ha^{-1} released back to the atmosphere. Of these historic carbon emissions, 2.8 MtC (62%) originated from the cut forest and 1.7 MtC (38%) from wetland soils.

8.3 Ongoing Carbon Emissions and Removals

The restoring marsh of North Ebey Island was found to be accumulating carbon at a rate of 3.5 tC $ha^{-1}yr^{-1}$, three times that of mature high marshes at balance with sea level, or at MHHW. This accumulation rate reflects the capacity of vegetation in the Pacific Northwest to build marshes at rates greater than sea level rise.

Water salinities in the Snohomish Estuary are relatively fresh, below half that of sea water, and as such the wetlands are likely emitting CH_4. Measurement of CH_4 emissions on restoring wetlands was beyond the capacity of this study, but applying an IPCC Tier 1 default value for CH_4 emissions from restoring tidal wetlands suggests that the North Ebey Island site is a net remover of GHGs.

Further analysis is required to determine whether drained wetlands are continuing to emit carbon from soils, and to quantify the magnitude of CH_4 and N_2O emissions. Drained lands are likely to be a net source of GHGs, if still drained, through carbon dioxide and N_2O emissions. Additionally, where water tables are high, such as areas that are mapped as Palustrine wetlands, these areas will likely be emitting CH_4.

8.4 Future Carbon Sequestration

In the long-term, or beyond 20 years, carbon sequestration benefits and net GHG removals will result from restoring tidal marshes. This study found that existing planned wetland restoration activities will sequester 0.32 MtC within soils as they rebuild to mature marshes. With sea level rise of 1 m, a further 0.37 MtC or total of 0.7 MtC will be sequestered within these wetland soils as they accrete. Should all potential tidal wetland restoration in the estuary occur, it is estimated that 1.2 MtC will be sequestered in soils alone, as marshes rebuild, and a total of 2.4 MtC will be sequestered with sea level rise of 1 m. This calculation is conservative in that it does not account for the significant accumulation of tidal wetland vegetation biomass and the associated carbon accumulation, both above and below ground. If some of these areas return to forested tidal wetland, the biomass accumulation could easily match the soil carbon sequestration.

8.5 Recommended Next Steps

1. Establish a regional blue carbon working group to build local capacity to deliver coordinated scientific findings, improve land management, and inform policy. Coastal lowlands of the Pacific Northwest offer potential for coastal wetlands restoration with natural resilience to sea level rise and carbon sequestration benefits. A coordinated action is required to further explore this opportunity and testing through demonstration projects.

2. With a carbon finance methodology for tidal wetlands restoration submitted for review to the Verified Carbon Standard, the next step to delivering a carbon finance project is selection of a potential project site and a detailed feasibility assessment. This report suggests estuaries similar to the Snohomish offer potential to host successful projects. Such projects could include elements of upper estuary forest and floodplain restoration as buffers for sea level rise and climate change adaptation as well as emergent tidal wetland restoration.

3. Expand the geographical extent of this study, regionally to the Puget Sound, and nationally to other estuaries of the United States and the world. We also recommend including seagrass habitats, which were not studied here. The approach developed in this study is readily transferable to other coastal lowland settings. There is a need for regional quantification of GHG emissions and reductions associated with coastal land use practice in this and other regions. This study provides a first step.

4. Report back in March 2017 on the lessons learned in applying the IPCC 2013 Wetland Supplement guidance on GHG accounting for wetlands management to the Subsidiary Body for Scientific and Technological Advice (SBSTA), an organization with a mandate to advise the United Nations Framework on Climate Change (UNFCC). For example, there is an absence of data on CH_4 and N_2O fluxes for natural, drained, and restoring coastal wetlands in the Pacific Northwest. Gathering this data would assist in developing IPCC Tier 2 emissions factors for land use activities in the region, and support state and national GHG accounting. The application of the Wetlands Supplement should be tested in the Pacific Northwest and other regions of the country. Tier 2 level quantification of GHG fluxes with wetland management should be developed.

5. Develop landscape-level management plans that incorporate both climate change adaption and mitigation. Restoring tidal wetlands sooner rather than later will enable marsh building before sea level rises above the elevation at which emergent vegetation will colonize. Creating buffer zones into which marshes can migrate with sea level rise will support both sea level rise mitigation and adaptation. Coordinating climate change adaption and mitigation planning will improve project outcomes.

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APPENDIX SUPPLEMENTAL FIGURES AND TABLES

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QUILCEDA MARSH

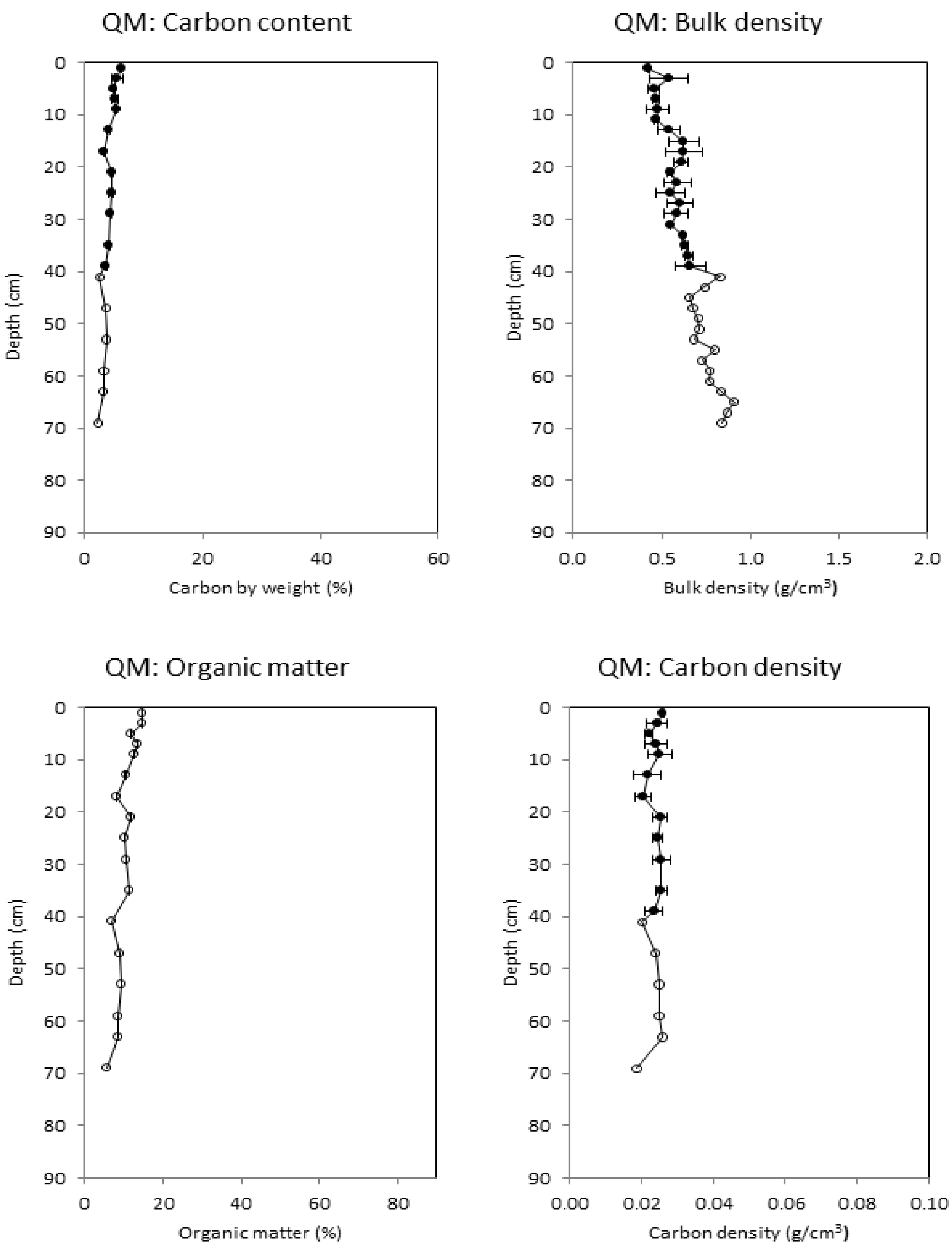

Figure 20. Quilceda Marsh soil profile with carbon content, bulk density, organic matter content, and carbon density. Error bars represent ± 1 standard deviation (n = 2). Hollow points denote depths with no replication.

QWULOOLT

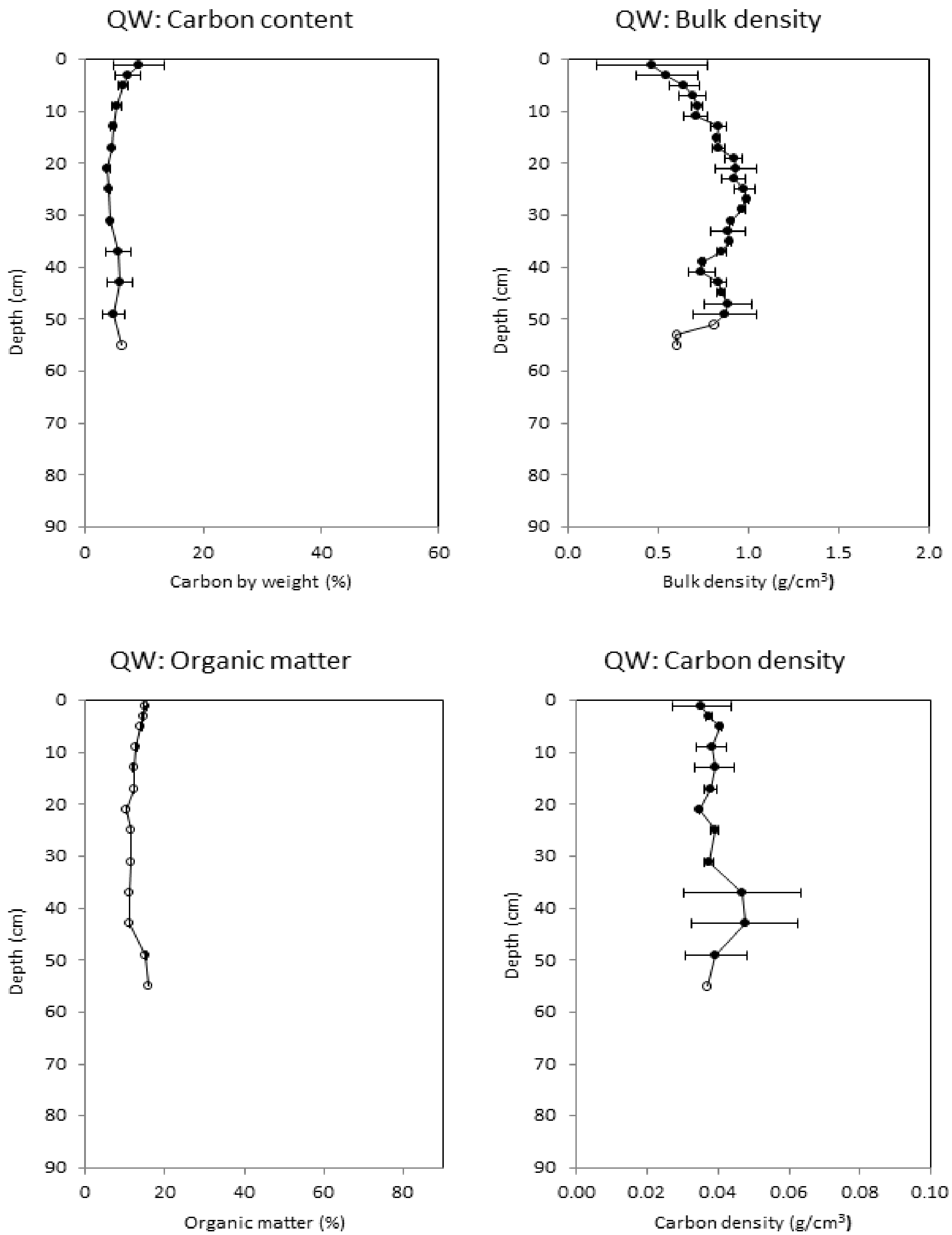

Figure 21. Qwuloolt soil profile with carbon content, bulk density, organic matter content, and carbon density. Error bars represent ± 1 standard deviation (n = 2). Hollow points denote depths with no replication.

MARYSVILLE

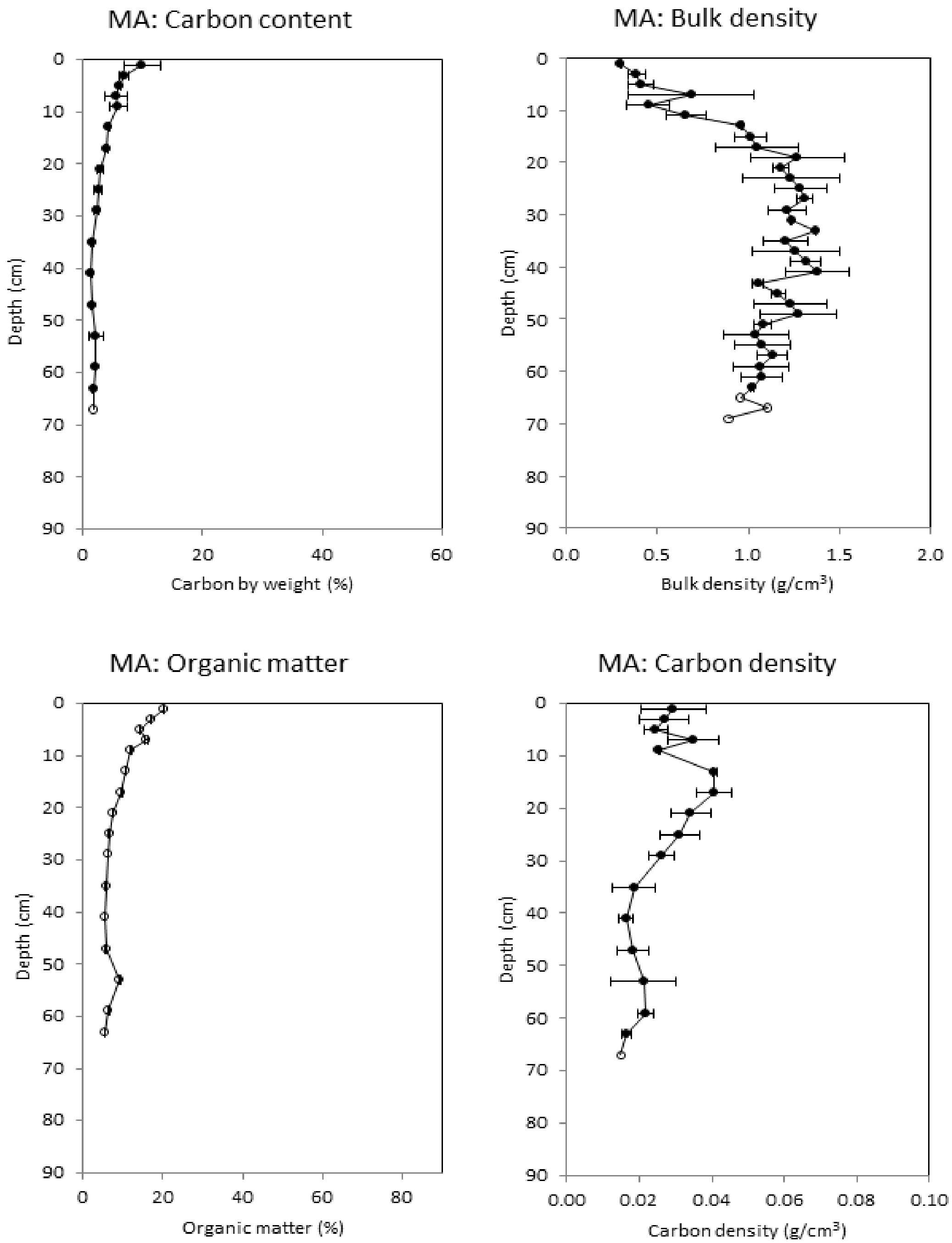

Figure 22. Marysville restoration site soil profile with carbon content, bulk density, organic matter content, and carbon density. Error bars represent ± 1 standard deviation (n = 2). Hollow points denote depths with no replication.

NORTH EBEY

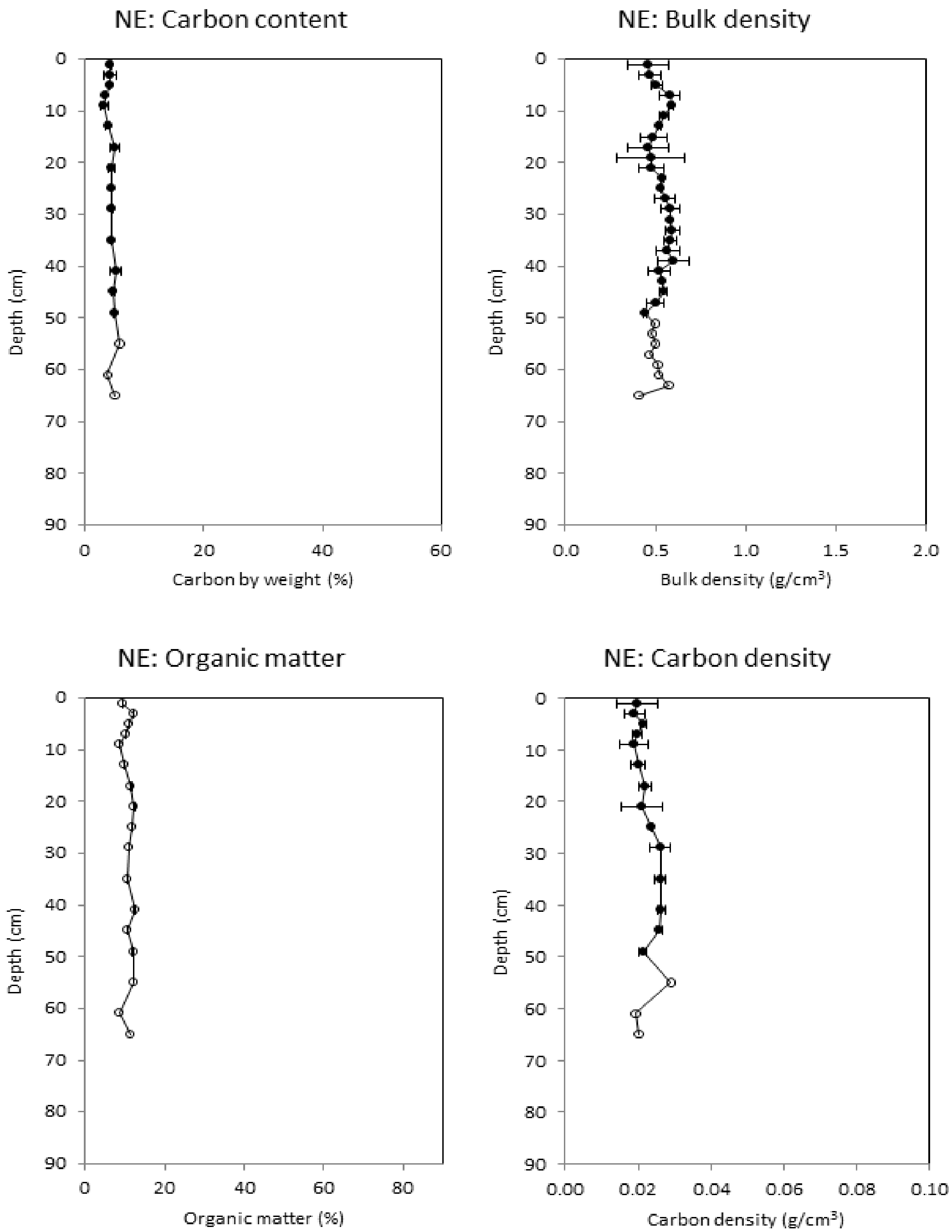

Figure 23. North Ebey soil profile with carbon content, bulk density, organic matter content, and carbon density. Error bars represent ± 1 standard deviation (n = 2). Hollow points denote depths with no replication.

HERON POINT

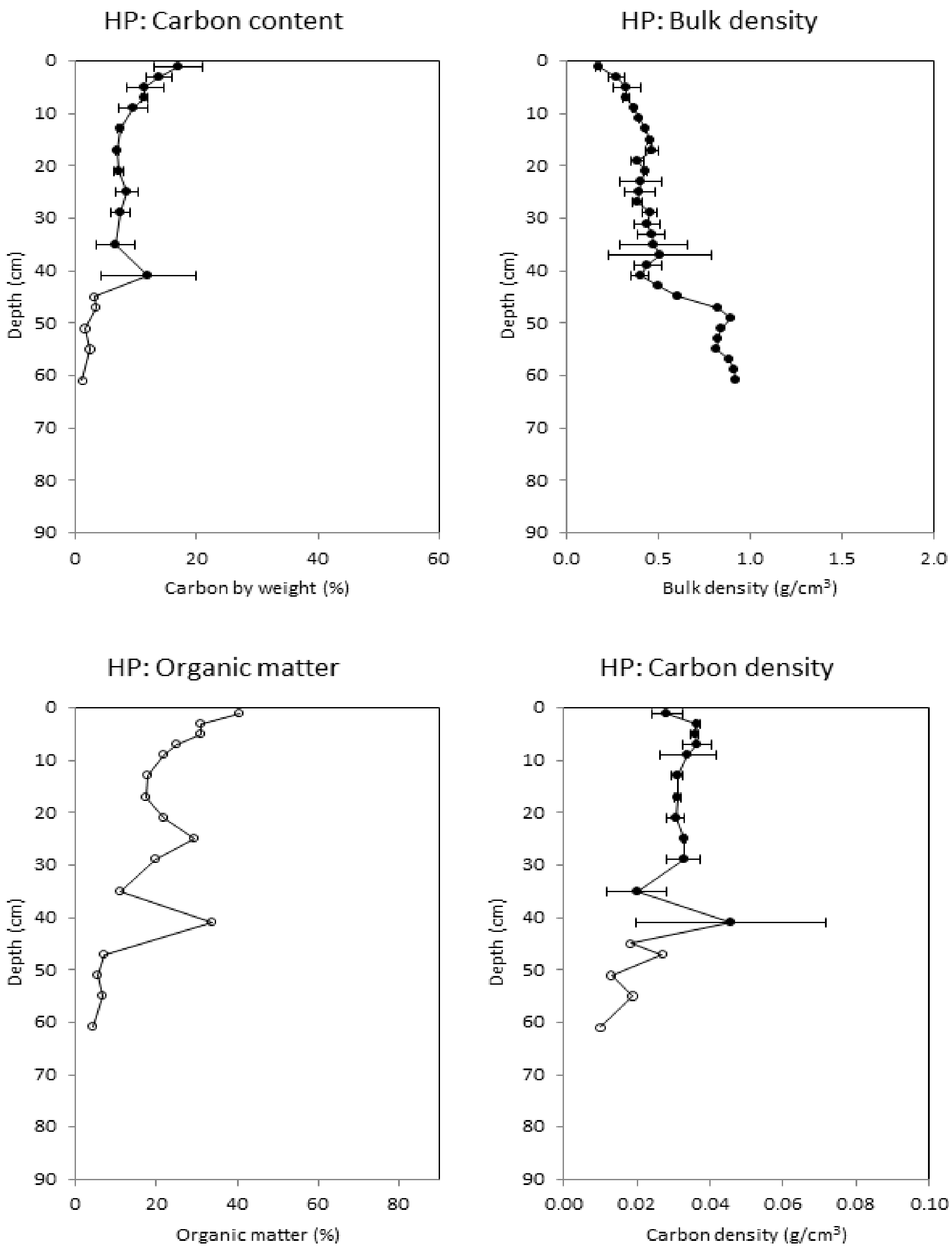

Figure 24. Heron Point soil profile with carbon content, bulk density, organic matter content, and carbon density. Error bars represent ± 1 standard deviation (n = 2). Hollow points denote depths with no replication.

OTTER ISLAND

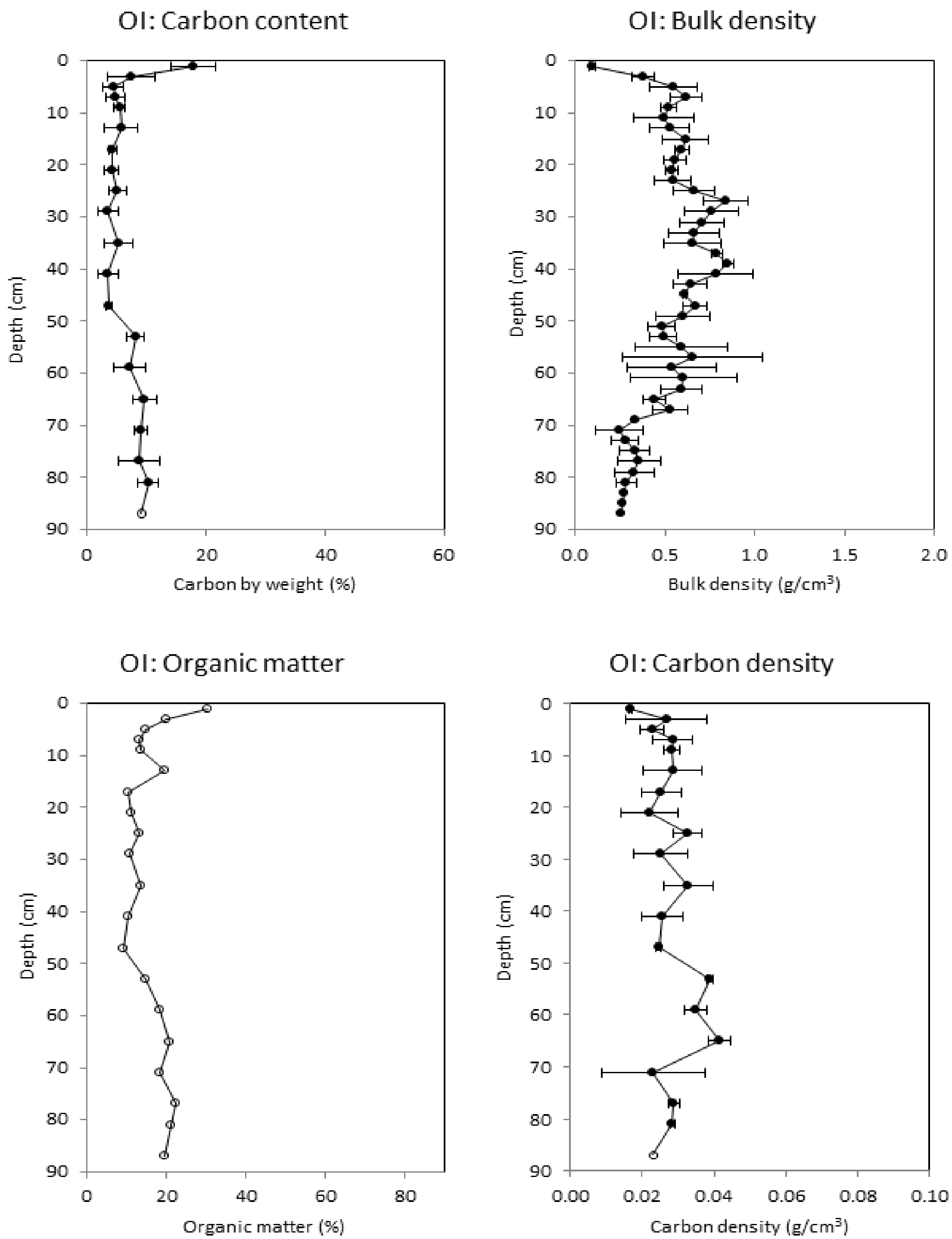

Figure 25. Otter Island soil profile with carbon content, bulk density, organic matter content, and carbon density. Error bars represent ± 1 standard deviation (n = 2). Hollow points denote depths with no replication.

SPENCER ISLAND

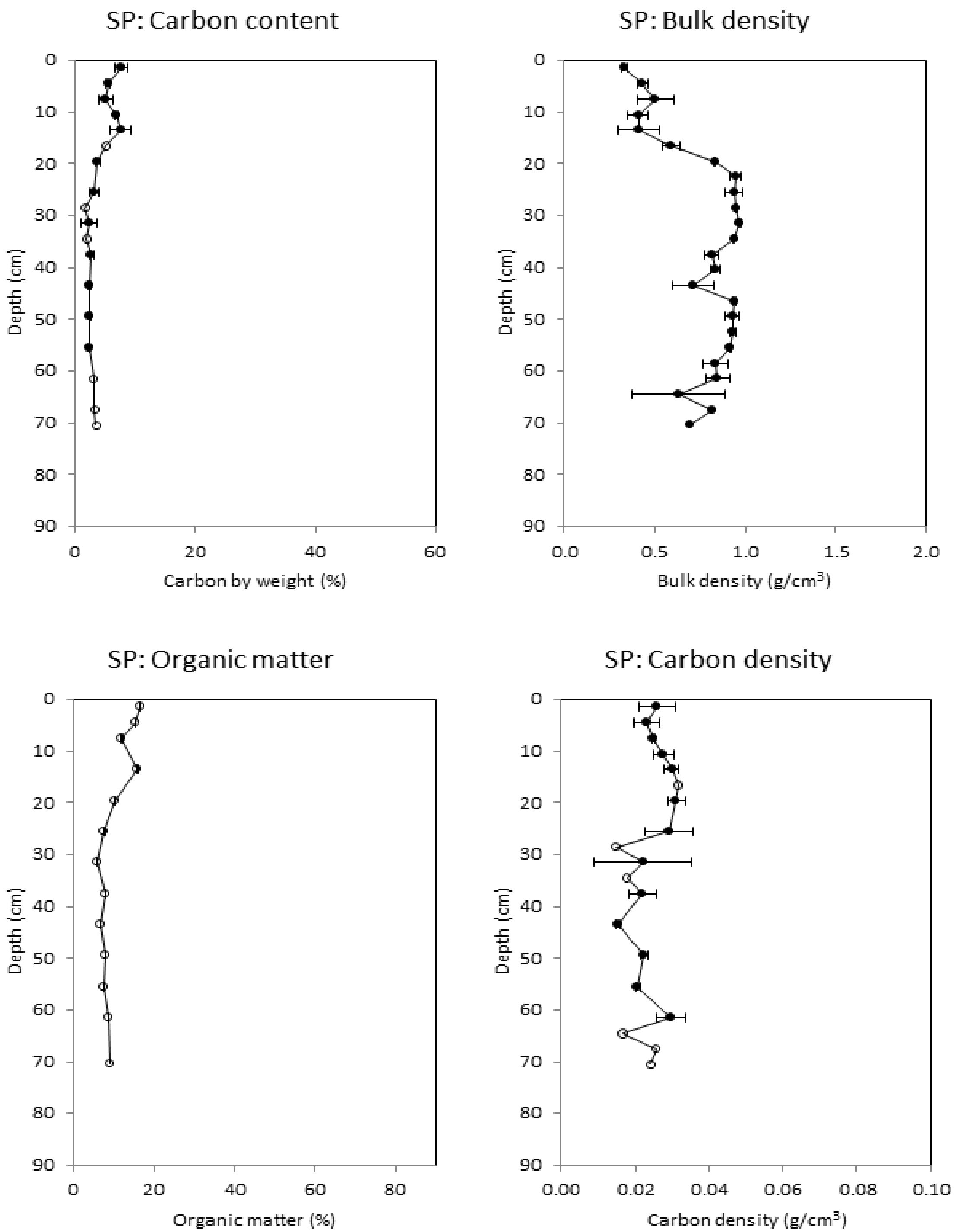

Figure 26. Spencer Island soil profile with carbon content, bulk density, organic matter content, and carbon density. Error bars represent ± 1 standard deviation (n = 2). Hollow points denote depths with no replication.

SMITH ISLAND – COUNTY

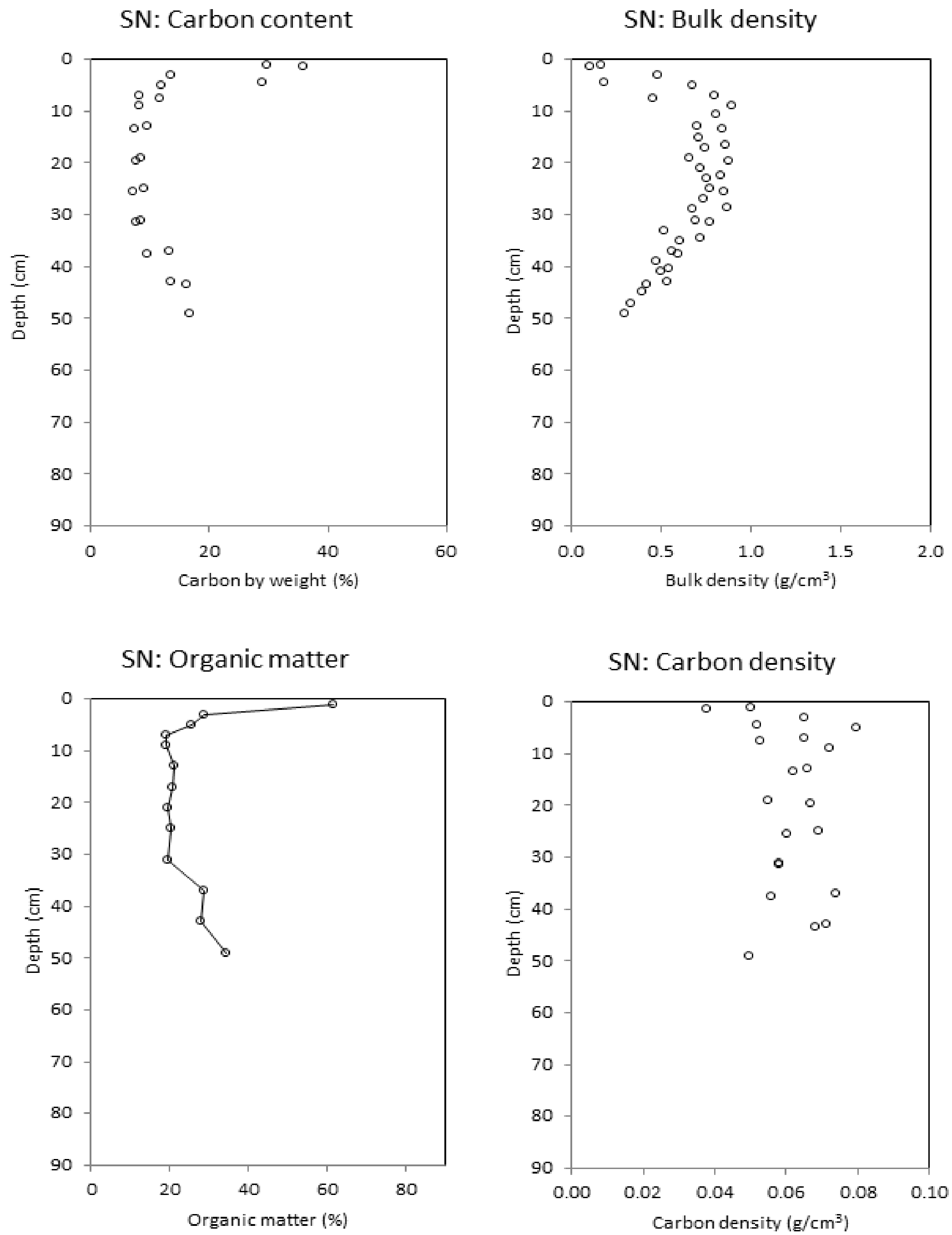

Figure 27. Smith Island County site soil profile with carbon content, bulk density, organic matter content, and carbon density. Error bars represent ± 1 standard deviation (n = 2). Hollow points denote depths with no replication.

SMITH ISLAND – CITY

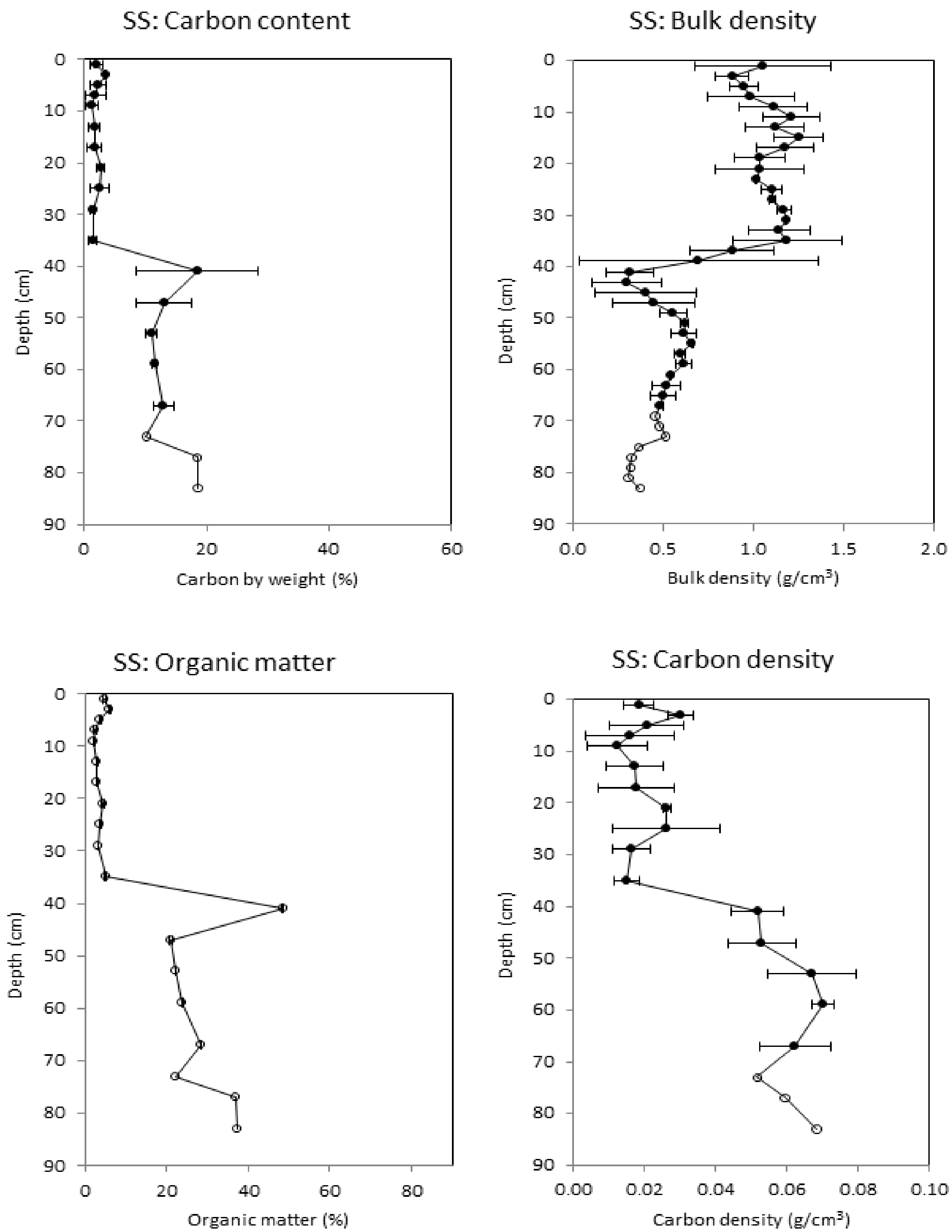

Figure 28. Smith Island City site soil profile with carbon content, bulk density, organic matter content, and carbon density. Error bars represent ± 1 standard deviation (n = 2). Hollow points denote depths with no replication.

UNION SLOUGH

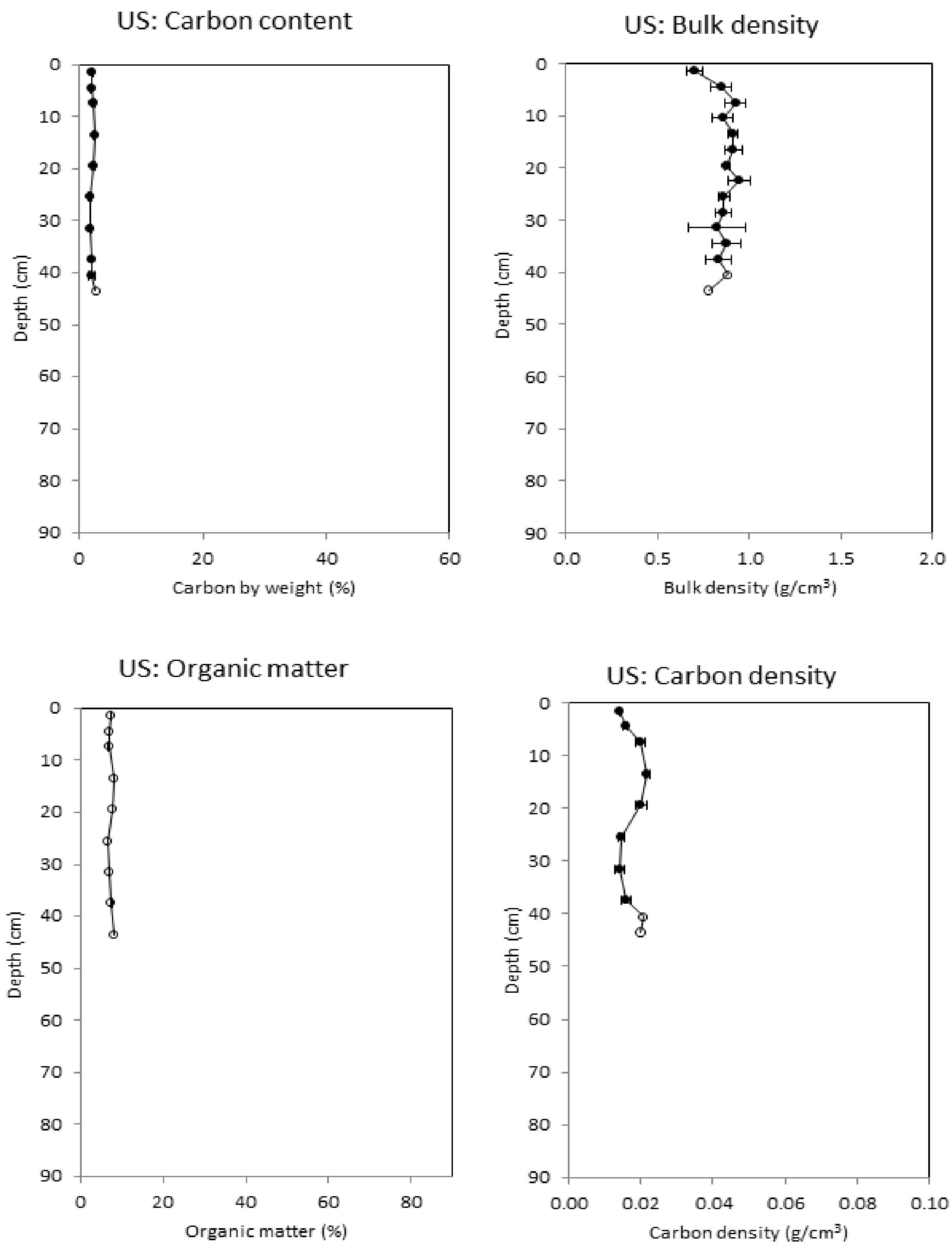

Figure 29. Union Slough soil profile with carbon content, bulk density, organic matter content, and carbon density. Error bars represent ± 1 standard deviation (n = 2). Hollow points denote depths with no replication.

WDFW WETLAND

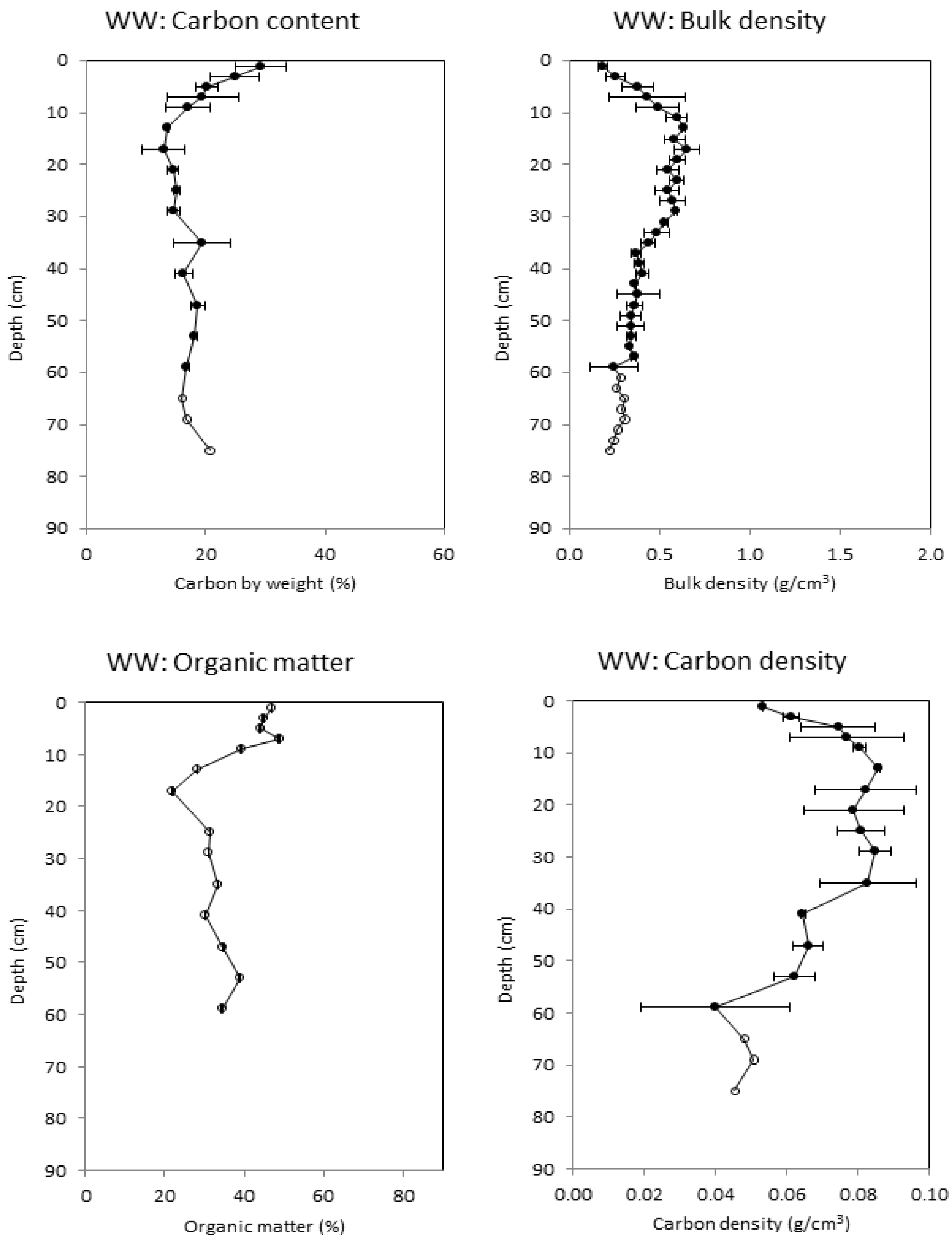

Figure 30. WDFW Wetland site soil profile with carbon content, bulk density, organic matter content, and carbon density. Error bars represent ± 1 standard deviation (n = 2). Hollow points denote depths with no replication.

WDFW FORESTED

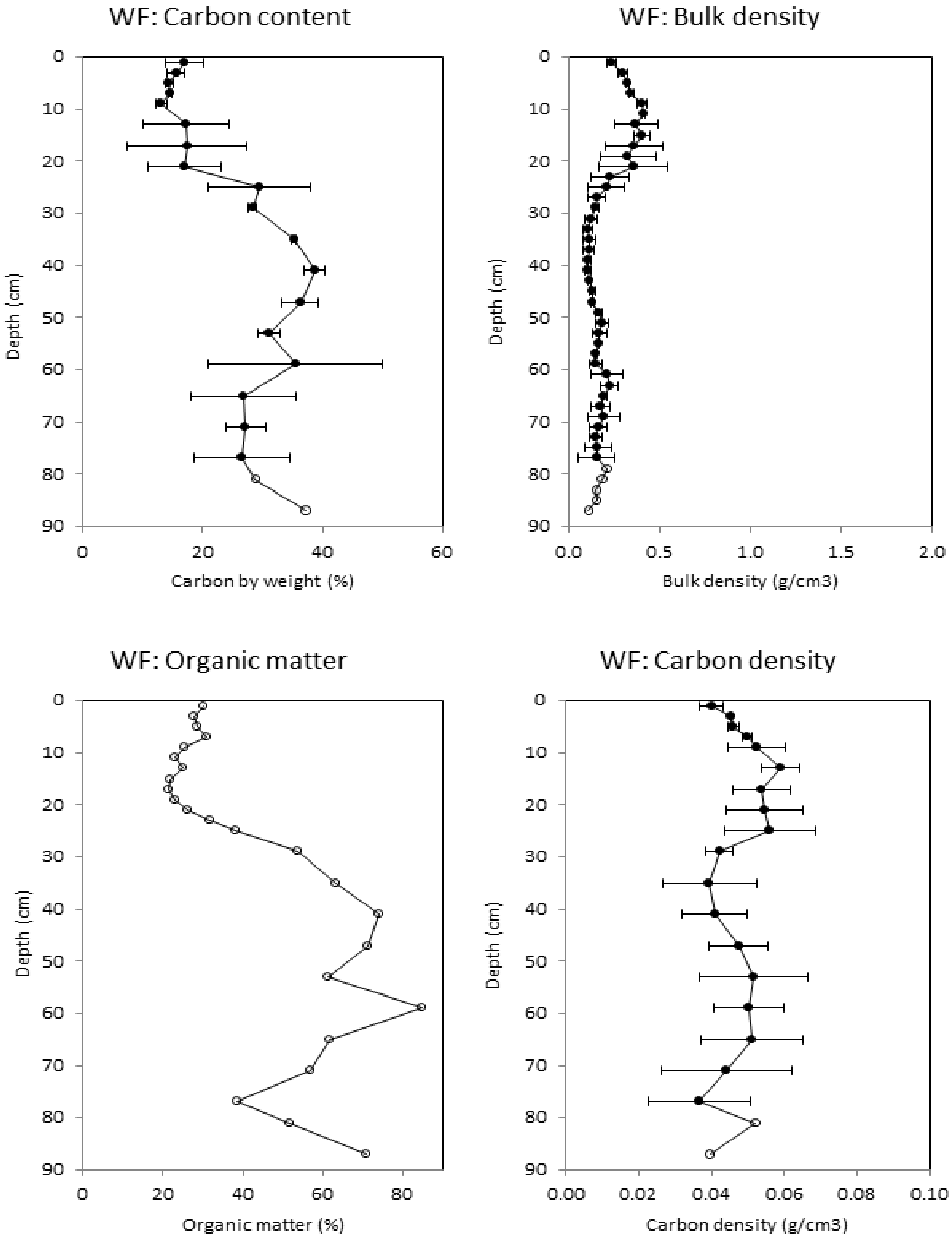

Figure 31. WDFW Forested site soil profile with carbon content, bulk density, organic matter content, and carbon density. Error bars represent ± 1 standard deviation (n = 2). Hollow points denote depths with no replication.

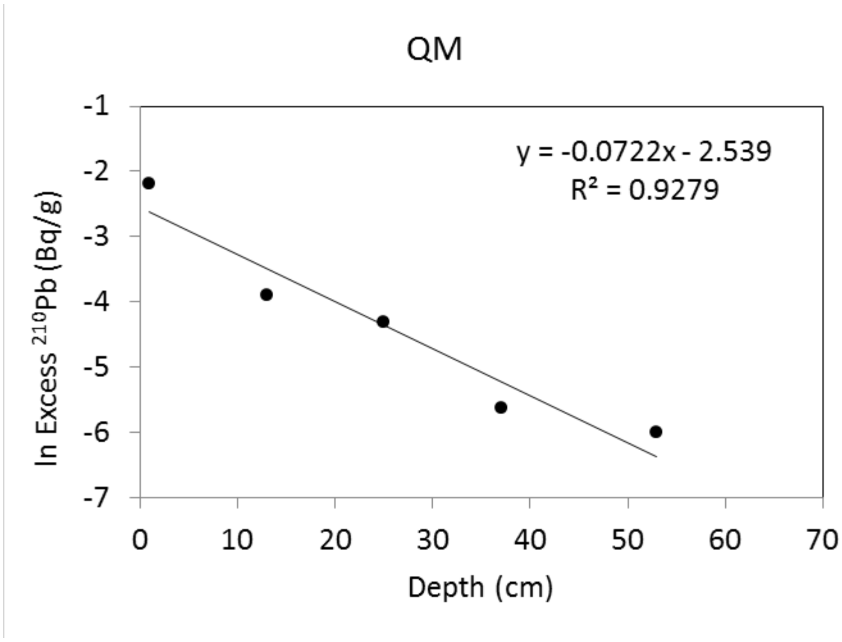

Figure 32. Downcore distribution of excess ^{210}Pb at Quilceda Marsh. The sediment accretion rate of 0.43 cm yr^{-1} was calculated from the slope of the regression of the natural log of excess ^{210}Pb activity vs. depth.

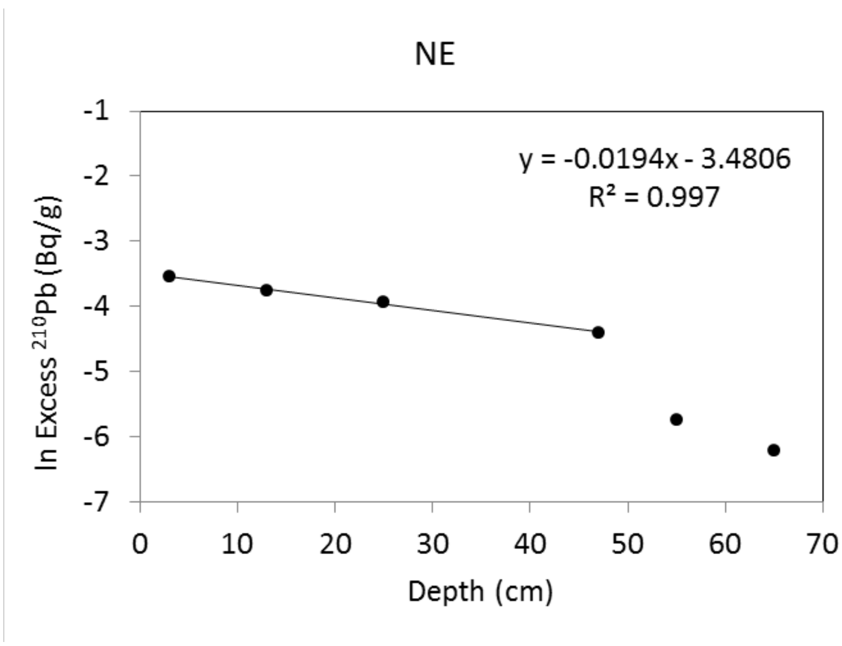

Figure 33. Downcore distribution of excess ^{210}Pb at North Ebey. The sediment accretion rate of 1.61 cm yr^{-1} was calculated from the slope of the regression of the natural log of excess ^{210}Pb activity vs. depth. Due to an apparent disjunct in rates below 50 cm, only the samples above 50 cm were included to represent the post-restoration accretion rate.

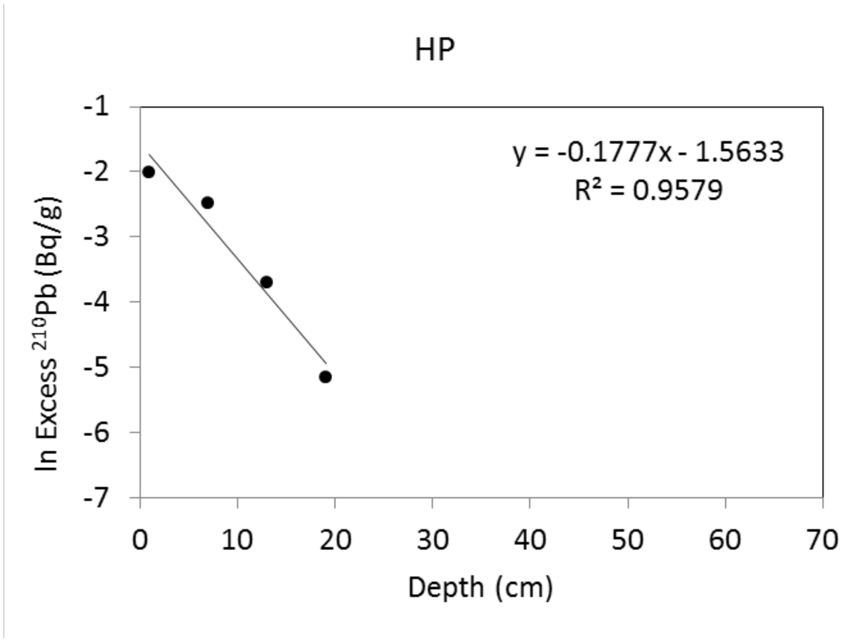

Figure 34. Downcore distribution of excess ^{210}Pb at Heron Point. The sediment accretion rate of 0.18 cm yr^{-1} was calculated from the slope of the regression of the natural log of excess ^{210}Pb activity vs. depth.

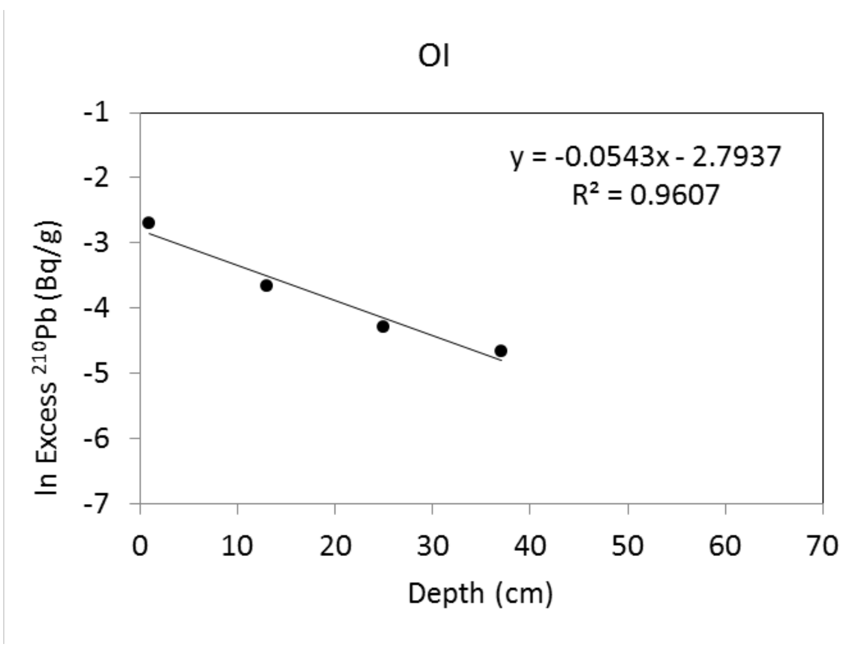

Figure 35. Downcore distribution of excess ^{210}Pb at Otter Island. The sediment accretion rate of 0.58 cm yr^{-1} was calculated from the slope of the regression of the natural log of excess ^{210}Pb activity vs. depth.

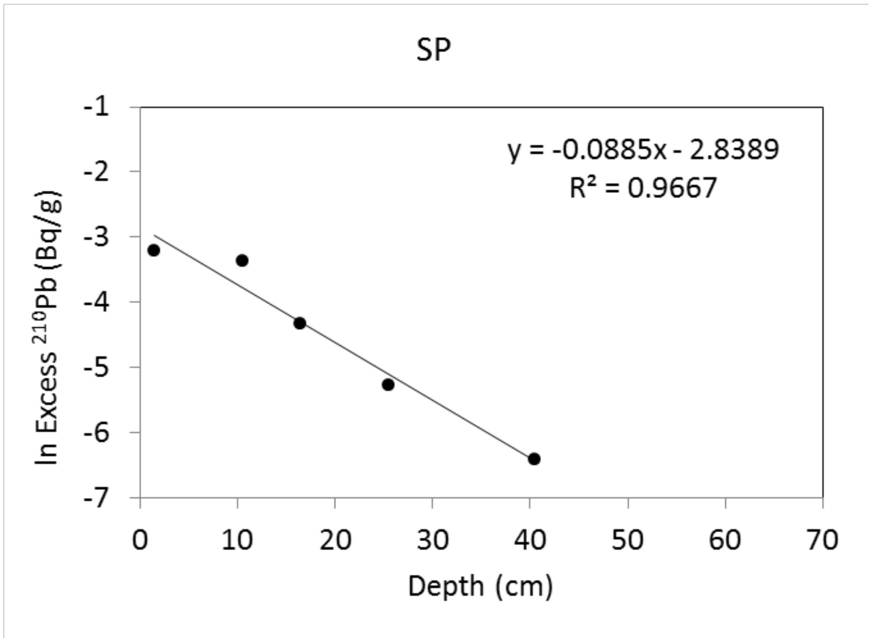

Figure 36. Downcore distribution of excess ^{210}Pb at Spencer Island. The sediment accretion rate of 0.35 cm yr^{-1} was calculated from the slope of the regression of the natural log of excess ^{210}Pb activity vs. depth.

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APPENDIX – SUPPLEMENTAL TABLES

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Table 14. Measured values of carbon content, organic matter content, and bulk density at Quilceda Marsh.

		QM: Quilceda Marsh						
		Carbon content (%)				Organic matter (%)	Bulk density (g cm^{-3})	
Section	Depth (cm)	Core A	Replicate	Core B	Replicate	Core A	Core A	Core B
0-2	1.0	6.14		6.15		14.53	0.42	0.43
2-4	3.0	6.12		4.81		14.61	0.43	0.46
4-6	5.0	4.75		4.93		11.74	0.48	0.43
6-8	7.0	5.46		4.80		13.36	0.48	0.45
8-10	9.0	5.26	5.33	5.23		12.74	0.52	0.43
10-12	11.0						0.47	0.47
12-14	13.0	4.14		3.81		10.36	0.59	0.50
14-16	15.0						0.68	0.57
16-18	17.0	3.14		3.40		8.04	0.70	0.55
18-20	19.0						0.64	0.58
20-22	21.0	4.83		4.43		11.68	0.55	0.54
22-24	23.0						0.64	0.53
24-26	25.0	4.19		4.80		10.07	0.60	0.49
26-28	27.0						0.65	0.55
28-30	29.0	4.34		4.47		10.60	0.63	0.53
30-32	31.0						0.55	0.55
32-34	33.0						0.62	0.62
34-36	35.0	4.12		3.94		11.54	0.65	0.62
36-38	37.0						0.67	0.63
38-40	39.0	3.49		3.64			0.72	0.60
40-42	41.0	2.42				6.90	0.83	
42-44	43.0						0.74	
44-46	45.0						0.66	
46-48	47.0	3.53				8.95	0.68	
48-50	49.0						0.71	
50-52	51.0						0.71	
52-54	53.0	3.65				9.35	0.68	
54-56	55.0						0.80	
56-58	57.0						0.73	
58-60	59.0	3.22				8.40	0.77	
60-62	61.0						0.77	
62-64	63.0	3.09				8.48	0.83	
64-66	65.0						0.91	
66-68	67.0						0.87	
68-70	69.0	2.23				5.80	0.84	

Table 15. Measured values of carbon content, organic matter content, and bulk density at Qwuloolt.

		QW: Qwuloolt						
		Carbon content (%)				Organic matter (%)	Bulk density (g cm^{-3})	
Section	Depth (cm)	Core A	Replicate	Core B	Replicate	Core B	Core A	Core B
4-6	1.0	12.10		6.02		15.02	0.24	0.68
6-8	3.0	8.64		5.75		14.55	0.42	0.66
8-10	5.0	6.91		5.78		14.04	0.58	0.70
10-12	7.0						0.63	0.74
12-14	9.0	5.93		4.77		12.83	0.69	0.74
14-16	11.0						0.75	0.66
16-18	13.0	4.98		4.38		12.09	0.86	0.80
18-20	15.0						0.81	0.84
20-22	17.0	4.55		4.56		12.40	0.80	0.86
22-24	19.0						0.95	0.88
24-26	21.0	4.10		3.43		10.30	0.84	1.01
26-28	23.0						0.96	0.87
28-30	25.0	4.10		3.93		11.33	0.93	1.02
30-32	27.0						0.98	0.99
32-34	29.0						0.98	0.95
34-36	31.0	4.29		4.01		11.39	0.89	0.90
36-38	33.0						0.95	0.82
38-40	35.0						0.88	0.89
40-42	37.0	7.10	7.00	4.06		10.93	0.83	0.87
42-44	39.0						0.74	0.75
44-46	41.0						0.68	0.79
46-48	43.0	7.27		4.28		11.15	0.80	0.86
48-50	45.0						0.86	0.83
50-52	47.0						0.97	0.79
52-54	49.0	3.37		6.16		15.17	0.99	0.74
54-56	51.0							0.81
56-58	53.0							0.60
58-60	55.0			6.12		16.03		0.60

Table 16. Measured values of carbon content, organic matter content, and bulk density at the Marysville site.

		MA: Marysville Site							
		Carbon content (%)					Organic matter (%)	Bulk density (g cm^{-3})	
Section	Depth (cm)	Core A	Replicate	Core B	Replicate	Replicate	Core A	Core A	Core B
0-2	1.0	12.12		7.66			20.12	0.29	0.30
2-4	3.0	7.49		6.30			17.14	0.42	0.35
4-6	5.0	5.80		6.13			14.34	0.46	0.36
6-8	7.0	6.79		4.32	4.25		15.69	0.44	0.93
8-10	9.0	4.85		6.07	8.31	6.19	11.99	0.53	0.37
10-12	11.0							0.73	0.58
12-14	13.0	4.32		4.17			10.46	0.96	0.97
14-16	15.0							1.07	0.95
16-18	17.0	3.66	3.64	4.18			9.53	1.21	0.89
18-20	19.0							1.45	1.09
20-22	21.0	2.52		3.31			7.60	1.21	1.15
22-24	23.0							1.42	1.05
24-26	25.0	1.96		2.95			6.57	1.39	1.18
26-28	27.0							1.34	1.28
28-30	29.0	1.84		2.51			6.26	1.29	1.14
30-32	31.0							1.24	1.23
32-34	33.0							1.37	NA
34-36	35.0	1.76		1.40			5.68	1.29	1.12
36-38	37.0							1.43	1.09
38-40	39.0							1.38	1.26
40-42	41.0	1.42		1.19			5.49	1.26	1.50
42-44	43.0							1.08	1.03
44-46	45.0							1.14	1.19
46-48	47.0	1.56		1.34			5.73	1.37	1.09
48-50	49.0							1.43	1.12
50-52	51.0							1.05	1.11
52-54	53.0	3.01	2.97	1.26			9.21	0.92	1.17
54-56	55.0							0.97	1.18
56-58	57.0							1.19	1.07
58-60	59.0	1.98		2.05			6.37	1.18	0.96
60-62	61.0							1.15	0.99
62-64	63.0	1.55		1.84			5.62	1.02	1.03
64-66	65.0								0.96
66-68	67.0			1.67					1.11
68-70	69.0								0.89

Table 17. Measured values of carbon content, organic matter content, and bulk density at North Ebey.

		NE: North Ebey						
		Carbon content (%)				Organic matter (%)	Bulk density (g cm^{-3})	
Section	Depth (cm)	Core A	Replicate	Core B	Replicate	Core A	Core A	Core B
0-2	1.0	4.15		4.39		9.42	0.38	0.54
2-4	3.0	4.98		3.33		12.27	0.42	0.51
4-6	5.0	4.27		4.21		11.00	0.49	0.53
6-8	7.0	3.48		3.38		10.26	0.54	0.62
8-10	9.0	2.69	2.85	3.62		8.53	0.58	0.60
10-12	11.0						0.53	0.56
12-14	13.0	3.52		4.13		9.64	0.53	0.52
14-16	15.0						0.54	0.44
16-18	17.0	4.33		5.46		11.52	0.54	0.38
18-20	19.0						0.61	0.34
20-22	21.0	4.78		3.94	3.98	12.34	0.53	0.43
22-24	23.0						0.55	0.53
24-26	25.0	4.53		4.44		11.63	0.53	0.53
26-28	27.0						0.59	0.51
28-30	29.0	4.56		4.41		10.91	0.62	0.55
30-32	31.0						0.58	0.58
32-34	33.0						0.62	0.57
34-36	35.0	4.45		4.53		10.50	0.61	0.56
36-38	37.0						0.62	0.52
38-40	39.0						0.54	0.66
40-42	41.0	5.68		4.53		12.43	0.48	0.57
42-44	43.0						0.54	0.54
44-46	45.0	4.51		5.00		10.44	0.56	0.53
46-48	47.0						0.53	0.47
48-50	49.0	4.96		4.74		11.99	0.45	0.44
50-52	51.0						0.50	
52-54	53.0						0.48	
54-56	55.0	5.80				12.21	0.50	
56-58	57.0						0.47	
58-60	59.0						0.51	
60-62	61.0	3.72				8.71	0.52	
62-64	63.0						0.57	
64-66	65.0	4.97				11.33	0.41	

Table 18. Measured values of carbon content, organic matter content, and bulk density at Heron Point.

		HP: Heron Point							
		Carbon content (%)					Organic matter (%)	Bulk density (g cm^{-3})	
Section	Depth (cm)	Core A	Replicate	Core B	Replicate	Replicate	Core A	Core A	Core B
0-2	1.0	19.74		14.22			40.68	0.16	0.18
2-4	3.0	15.38		12.36			31.08	0.24	0.30
4-6	5.0	13.59		9.31			30.89	0.27	0.38
6-8	7.0	10.90		11.68			25.14	0.31	0.34
8-10	9.0	7.74		11.08			21.79	0.37	0.36
10-12	11.0							0.39	0.39
12-14	13.0	7.06		7.54			17.64	0.43	0.43
14-16	15.0							0.44	0.46
16-18	17.0	6.57		7.02			17.52	0.49	0.43
18-20	19.0							0.36	0.41
20-22	21.0	7.60		6.61			21.69	0.42	0.44
22-24	23.0							0.32	0.48
24-26	25.0	9.86		7.19			29.41	0.34	0.45
26-28	27.0							0.37	0.40
28-30	29.0	8.54		6.17			19.95	0.42	0.48
30-32	31.0							0.48	0.39
32-34	33.0							0.51	0.40
34-36	35.0	4.20	4.36	8.29	9.03	9.16	10.88	0.60	0.34
36-38	37.0							0.70	0.31
38-40	39.0							0.50	0.38
40-42	41.0	17.61		6.41			33.95	0.36	0.43
42-44	43.0								0.50
44-46	45.0			3.02					0.60
46-48	47.0	3.27					7.04	0.82	
48-50	49.0							0.89	
50-52	51.0	1.54					5.43	0.84	
52-54	53.0							0.83	
54-56	55.0	2.31					6.54	0.82	
56-58	57.0							0.88	
58-60	59.0							0.91	
60-62	61.0	1.08					4.10	0.92	

Table 19. Measured values of carbon content, organic matter content, and bulk density at Otter Island.

		OI: Otter Island						
		Carbon content (%)				Organic matter (%)	Bulk density (g cm^{-3})	
Section	Depth (cm)	Core A	Replicate	Core B	Replicate	Core A	Core A	Core B
0-2	1.0	15.13		19.97	20.79	30.42	0.11	0.08
2-4	3.0	10.29		4.55	4.38	20.00	0.34	0.42
4-6	5.0	5.59		3.17		14.73	0.45	0.64
6-8	7.0	5.85		3.63		13.01	0.56	0.68
8-10	9.0	6.08		4.83		13.44	0.49	0.55
10-12	11.0						0.37	0.61
12-14	13.0	7.56	7.80	3.77		19.52	0.45	0.60
14-16	15.0						0.52	0.70
16-18	17.0	4.68		3.80		10.34	0.62	0.57
18-20	19.0						0.59	0.51
20-22	21.0	4.92		3.21		11.08	0.56	0.51
22-24	23.0						0.47	0.62
24-26	25.0	6.40	5.76	4.01		13.02	0.58	0.74
26-28	27.0						0.92	0.75
28-30	29.0	4.70		2.29		10.65	0.65	0.86
30-32	31.0						0.62	0.79
32-34	33.0						0.56	0.76
34-36	35.0	7.19	6.89	3.66		13.67	0.54	0.76
36-38	37.0						0.77	0.81
38-40	39.0						0.87	0.83
40-42	41.0	4.66		2.29		10.45	0.64	0.93
42-44	43.0						0.57	0.71
44-46	45.0						0.62	0.60
46-48	47.0	3.95	4.16	3.37		9.17	0.62	0.71
48-50	49.0						0.49	0.71
50-52	51.0						0.43	0.53
52-54	53.0	7.04		9.18	9.06	14.84	0.54	0.43
54-56	55.0						0.41	0.77
56-58	57.0						0.38	0.93
58-60	59.0	9.01		5.32	5.12	18.32	0.36	0.71
60-62	61.0						0.39	0.81
62-64	63.0						0.51	0.67
64-66	65.0	11.12		8.19		20.79	0.39	0.48
66-68	67.0						0.60	0.46
68-70	69.0							0.34
70-72	71.0	8.34		10.06	9.38	18.17	0.16	0.34
72-74	73.0						0.23	0.33
74-76	75.0						0.27	0.39
76-78	77.0	11.10		6.28		22.19	0.27	0.44
78-80	79.0						0.25	0.40
80-82	81.0	11.53		8.95		21.30	0.24	0.32
82-84	83.0						0.27	
84-86	85.0						0.26	
86-88	87.0	9.11				19.66	0.25	

Table 20. Measured values of carbon content, organic matter content, and bulk density at Spencer Island.

		SP: Spencer Island						
		Carbon content (%)				Organic matter (%)	Bulk density (g cm^{-3})	
Section	Depth (cm)	Core A	Replicate	Core B	Replicate	Core A	Core A	Core B
0-3	1.5	6.97	7.04	8.91	8.11	16.67	0.32	0.35
3-6	4.5	5.67		5.08		15.22	0.46	0.41
6-9	7.5	4.27		5.88		11.96	0.58	0.43
9-12	10.5	6.58		6.92			0.45	0.37
12-15	13.5	6.37		8.41	9.03	15.88	0.49	0.33
15-18	16.5	5.08					0.62	0.56
18-21	19.5	3.53		3.97	3.95	10.41	0.83	0.83
21-24	22.5						0.97	0.93
24-27	25.5	2.68	2.42	3.77		7.60	0.97	0.90
27-30	28.5	1.57					0.95	0.96
30-33	31.5	1.32		3.24		5.74	0.97	0.97
33-36	34.5	1.89					0.94	0.93
36-39	37.5	2.47		2.94		7.87	0.78	0.84
39-42	40.5						0.81	0.85
42-45	43.5	2.01		2.40		6.61	0.79	0.63
45-48	46.5						0.94	0.95
48-51	49.5	2.57		2.28	2.29	7.73	0.90	0.95
51-54	52.5						0.92	0.95
54-57	55.5	2.31		2.22		7.34	0.92	0.92
57-60	58.5						0.78	0.88
60-63	61.5	3.02				8.71	0.89	0.80
63-66	64.5						0.82	0.45
66-69	67.5	3.16					0.81	
69-72	70.5	3.48				9.11	0.69	

Table 21. Measured values of carbon content, organic matter content, and bulk density at the Smith Island County site.

SN: Smith Island - County									
Section			Carbon content (%)				Organic matter (%)	Bulk density (g cm^{-3})	
Core A	Core B	Depth (cm)	Core A	Replicate	Core B	Replicate	Core B	Core A	Core B
	0-2	1.0			29.66		61.62		0.17
0-3		1.5	35.34	36.45				0.11	
	2-4	3.0			13.58		28.74		0.48
3-6		4.5	28.71	28.92				0.18	
	4-6	5.0			11.86		25.66		0.67
	6-8	7.0			8.19		18.95		0.80
6-9		7.5	11.57	11.66				0.46	
	8-10	9.0			8.11		18.91		0.89
9-12		10.5						0.80	
	12-14	13.0			9.37		21.04		0.70
12-15		13.5	7.35	7.39				0.84	
	14-16	15.0							0.71
15-18		16.5						0.85	
	16-18	17.0					20.58		0.74
	18-20	19.0			8.44				0.65
18-21		19.5	7.44	7.91				0.87	
	20-22	21.0					19.61		0.71
21-24		22.5						0.83	
	22-24	23.0							0.76
	24-26	25.0			8.95		20.22		0.77
24-27		25.5	7.09					0.85	
	26-28	27.0							0.73
27-30		28.5						0.86	
	28-30	29.0							0.67
	30-32	31.0			8.47		19.62		0.69
30-33		31.5	7.55					0.77	
	32-34	33.0							0.52
33-36		34.5						0.72	
	34-36	35.0							0.60
	36-38	37.0			13.09	13.17	28.55		0.56
36-39		37.5	9.00	9.77				0.59	
	38-40	39.0							0.47
39-42		40.5						0.54	
	40-42	41.0							0.49
	42-44	43.0			13.45		27.74		0.53
42-45		43.5	16.19					0.42	
	44-46	45.0							0.39
	46-48	47.0							0.33
	48-50	49.0			16.69		34.17		0.30

Table 22. Measured values of carbon content, organic matter content, and bulk density at the Smith Island City site.

		SS: Smith Island - City						
		Carbon content (%)				Organic matter (%)	Bulk density (g cm^{-3})	
Section	Depth (cm)	Core A	Replicate	Core B	Replicate	Core B	Core A	Core B
0-2	1.0	1.06	1.26	2.75		4.70	1.32	0.78
2-4	3.0	3.40		3.46		6.09	0.82	0.95
4-6	5.0	3.15		1.32		3.77	0.89	1.00
6-8	7.0	3.05		0.62		2.41	0.81	1.15
8-10	9.0	1.86		0.51		2.26	0.98	1.24
10-12	11.0						1.09	1.32
12-14	13.0	2.28		0.94	0.92	2.98	1.00	1.23
14-16	15.0						1.15	1.34
16-18	17.0	2.39		0.77	0.81	2.95	1.06	1.28
18-20	19.0						0.93	1.13
20-22	21.0	3.14		2.13		4.43	0.86	1.20
22-24	23.0						1.01	1.01
24-26	25.0	3.49		1.35		3.81	1.06	1.14
26-28	27.0						1.09	1.11
28-30	29.0	1.69		1.09		3.31	1.20	1.14
30-32	31.0						1.18	1.17
32-34	33.0						1.26	1.02
34-36	35.0	0.90		1.80		4.99	1.40	0.97
36-38	37.0						1.05	0.72
38-40	39.0						1.16	0.23
40-42	41.0	11.38		25.54		48.57	0.41	0.22
42-44	43.0						0.16	0.43
44-46	45.0						0.20	0.60
46-48	47.0	16.15		9.86		21.05	0.29	0.61
48-50	49.0						0.50	0.60
50-52	51.0						0.63	0.60
52-54	53.0	11.56		10.34		22.22	0.66	0.56
54-56	55.0						0.66	0.66
56-58	57.0						0.57	0.61
58-60	59.0	11.79		11.21		23.96	0.58	0.65
60-62	61.0						0.54	0.54
62-64	63.0						0.57	0.46
64-66	65.0						0.54	0.45
66-68	67.0	11.72		14.12		28.63	0.47	0.49
68-70	69.0							0.45
70-72	71.0							0.48
72-74	73.0			10.10		22.26		0.51
74-76	75.0							0.36
76-78	77.0			18.44		36.84		0.32
78-80	79.0							0.32
80-82	81.0							0.31
82-84	83.0			18.58		37.43		0.37

Table 23. Measured values of carbon content, organic matter content, and bulk density at Union Slough.

		US: Union Slough						
		Carbon content (%)				Organic matter (%)	Bulk density (g cm^{-3})	
Section	Depth (cm)	Core A	Replicate	Core B	Replicate	Core B	Core A	Core B
0-3	1.5	1.92		2.13		7.20	0.73	0.67
3-6	4.5	1.73		2.02		6.94	0.88	0.81
6-9	7.5	1.97		2.37		6.72	0.97	0.88
9-12	10.5						0.89	0.81
12-15	13.5	2.36	2.23	2.52		7.91	0.93	0.89
15-18	16.5						0.88	0.95
18-21	19.5	2.14		2.43		7.59	0.89	0.87
21-24	22.5						0.98	0.90
24-27	25.5	1.80		1.57		6.55	0.84	0.88
27-30	28.5						0.83	0.89
30-33	31.5	1.89		1.63	1.63	6.85	0.71	0.93
33-36	34.5						0.82	0.93
36-39	37.5	1.91		1.90		7.41	0.78	0.88
39-42	40.5	1.65	1.65	2.35				0.88
42-45	43.5			2.58	2.56	8.22		0.78

Table 24. Measured values of carbon content, organic matter content, and bulk density at the WDFW Wetland site.

		WW: WDFW Wetland						
		Carbon content (%)				Organic matter (%)	Bulk density (g cm^{-3})	
Section	Depth (cm)	Core A	Replicate	Core B	Replicate	Core A	Core A	Core B
0-2	1.0	26.27		32.27		47.08	0.20	0.17
2-4	3.0	21.91		27.80		44.80	0.29	0.22
4-6	5.0	21.39		18.74		44.04	0.31	0.44
6-8	7.0	23.60		15.24		48.80	0.28	0.58
8-10	9.0	19.66		14.42		39.21	0.40	0.57
10-12	11.0						0.55	0.63
12-14	13.0	13.45		13.77		28.18	0.63	0.63
14-16	15.0						0.62	0.54
16-18	17.0	9.99	10.62	15.44		21.99	0.70	0.60
18-20	19.0						0.62	0.56
20-22	21.0	13.75		15.10			0.50	0.59
22-24	23.0						0.62	0.56
24-26	25.0	15.45		14.70		31.53	0.49	0.58
26-28	27.0						0.52	0.62
28-30	29.0	15.31		13.82		31.12	0.58	0.59
30-32	31.0						0.54	0.52
32-34	33.0						0.53	0.43
34-36	35.0	15.85		23.29	22.14	33.55	0.46	0.41
36-38	37.0						0.35	0.38
38-40	39.0						0.40	0.36
40-42	41.0	15.15		17.26		30.07	0.42	0.38
42-44	43.0						0.36	0.36
44-46	45.0						0.46	0.29
46-48	47.0	17.86		19.43		34.78	0.39	0.32
48-50	49.0						0.38	0.29
50-52	51.0						0.39	0.29
52-54	53.0	18.32		17.87		38.81	0.36	0.32
54-56	55.0						0.33	0.33
56-58	57.0						0.34	0.36
58-60	59.0	17.02		16.47		34.48	0.15	0.33
60-62	61.0							0.28
62-64	63.0							0.26
64-66	65.0			16.00				0.30
66-68	67.0							0.29
68-70	69.0			16.76				0.30
70-72	71.0							0.26
72-74	73.0							0.24
74-76	75.0			20.70				0.22

Table 25. Measured values of carbon content, organic matter content, and bulk density at the WDFW Forested site.

		WF: WDFW Forested						
		Carbon content (%)				Organic matter (%)	Bulk density (g cm^{-3})	
Section	Depth (cm)	Core A	Replicate	Core B	Replicate	Core A	Core A	Core B
0-2	1.0	14.81		19.31		30.09	0.25	0.22
2-4	3.0	14.49		16.54		27.94	0.31	0.28
4-6	5.0	13.84		14.87		28.66	0.33	0.32
6-8	7.0	14.70		14.46		31.13	0.33	0.35
8-10	9.0	12.34		13.80		25.37	0.38	0.42
10-12	11.0					22.93	0.41	0.40
12-14	13.0	12.20		22.20		24.94	0.45	0.28
14-16	15.0					21.63	0.43	0.37
16-18	17.0	10.30	10.34	24.41		21.60	0.47	0.24
18-20	19.0					22.93	0.43	0.22
20-22	21.0	12.67		21.26		26.07	0.49	0.22
22-24	23.0					31.79	0.30	0.15
24-26	25.0	23.55		35.56		38.22	0.28	0.13
26-28	27.0						0.18	0.12
28-30	29.0	27.86		28.37	29.15	53.62	0.16	0.14
30-32	31.0						0.15	0.10
32-34	33.0						0.12	0.08
34-36	35.0	35.01		35.52		63.47	0.14	0.08
36-38	37.0						0.13	0.09
38-40	39.0						0.12	0.10
40-42	41.0	39.89		37.50		74.12	0.12	0.09
42-44	43.0						0.12	0.10
44-46	45.0						0.14	0.11
46-48	47.0	38.47		34.18		71.39	0.14	0.12
48-50	49.0						0.18	0.15
50-52	51.0						0.21	0.15
52-54	53.0	32.32		29.77		61.49	0.19	0.14
54-56	55.0						0.17	0.16
56-58	57.0						0.14	0.15
58-60	59.0	45.87	45.69	25.17		85.09	0.12	0.17
60-62	61.0						0.14	0.27
62-64	63.0						0.19	0.26
64-66	65.0	32.92		20.54		61.71	0.19	0.20
66-68	67.0						0.21	0.14
68-70	69.0						0.25	0.13
70-72	71.0	29.49		24.73		56.81	0.19	0.13
72-74	73.0						0.17	0.12
74-76	75.0						0.21	0.10
76-78	77.0	20.83		31.96	32.25	38.48	0.22	0.08
78-80	79.0						0.21	
80-82	81.0	28.71				51.69	0.18	
82-84	83.0						0.15	
84-86	85.0						0.15	
86-88	87.0	37.20				70.85	0.11	