LAND USE EFFECTS ON CARBON CYCLING IN OREGON COASTAL WETLANDS

by

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THESIS ABSTRACT

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Pacific Northwest coastal wetland extent has been significantly reduced due to development. To understand the effects of land use change on carbon cycling in coastal wetlands, we compared soil carbon dynamics in restored, disturbed (by diking or draining), and reference wetlands in both freshwater and saline conditions in Coos Bay, Oregon. We quantified soil carbon pools, measured *in situ* fluxes of methane (CH_4) and carbon dioxide (CO_2), and estimated sediment deposition and carbon sequestration rates. We found that land use change influences carbon cycling and storage in coastal wetlands. The disturbed marshes have likely lost all their organic material after draining or diking, except for a shallow A horizon. The restored marsh *in situ* CH_4 and CO_2 fluxes were intermediate between the disturbed and reference marshes. Generally, restored marshes showed a partial return of carbon storage functions, or an indication that reference level functions may be achieved over time.

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CHAPTER I

INTRODUCTION

Coastal Wetland Ecosystems

Wetlands make up an estimated 6% of Earth's land surface (Mitsch and Gosselink 2007). Coastal wetlands are considered to be among the most valuable ecosystem types per unit area based on the ecosystem functions they provide, including carbon and nutrient cycling (Costanza et al. 1997). These systems provide important habitat in the forms of shelter, spawning grounds, and food for waterfowl, birds, small mammals, amphibians and aquatic species including some that are threatened or endangered (Thayer et al., 2003). Additionally, coastal marshes support human ecosystem services including protection against shoreline erosion, flood control infrastructure, recreational activities, supporting fisheries, tourism, and tribal sustenance (Frenkel and Morlan, 1991; Thayer et al., 2003).

Coastal wetlands encompass a diverse assortment of wetland types from salt marshes and mangroves to freshwater marshes and shrub depressions (Dahl, 2013). They are hydrologically connected to both upland watersheds and the ocean (Dahl, 2013), and thus they occur along a salinity gradient from freshwater to saline tidal (Poffenbarger et al., 2011; Bartlett et al., 1987). This salinity gradient in turn produces unique compositions of adapted flora and fauna communities in both salt marshes and freshwater marshes *(*Barendregt et al. 2009*)*.

Salt marshes develop in areas with oceanic influence that also have low wave energy and shallow depths (Pennings, 2014*)*. There are characterized by the presence of emergent vegetation, salt adapted flora and fauna, and aerobic-anaerobic fluctuations

(Thayer et al., 2003). Plant communities colonize different elevations in the tidal frame based on the pattern of tidal inundation *(*Bertness, 1991; Wigand and Roman, 2012*)*. Incoming tides bring sediments and nutrients to marshes, which promote high rates of primary productivity, whereas falling tides deliver organic materials back into lower portions of the estuary, supporting food webs (Pennings, 2014). In contrast, freshwater marshes may be or may not be tidal but occur along the river and stream floodplains within estuaries beyond the influence of marine-derived salinity. Riverine inputs to freshwater marshes transfer organic material and nutrients from upland sources to lower reaches of the marsh (Batzer et al., 2014).

Wetland soils, i.e. hydric soils, have unique biochemical and physical processes, dictated by the hydrological regime, that support unique ecosystem functions (Batzer et al., 2014). Marsh soils vary in their proportions of organic to mineral matter based on allochthonous (derived from outside of the system) versus autochthonous (derived from within the system) inputs, which has major implications for rates of soil decomposition and the production of methane (CH_4) and carbon dioxide (CO_2) (Batzer et al., 2014; Bridgham et al., 2008; Bridgham et al., 2006).

Coastal Wetlands and Carbon Cycling

Wetlands are important ecosystems for the global carbon cycle because they contain large soil carbon pools and have high rates of soil carbon sequestration (Bridgham et al., 2006). In addition to the many ecosystem services provided by coastal wetlands, the high soil carbon sequestration potential is believed to provide a buffer to global climate change effects of greenhouse gas emissions by converting atmospheric carbon into biomass and soil carbon (Callaway et al., 2012; Kirwan and Mudd, 2012).

Estuarine wetlands sequester carbon at a higher rate than any other wetland ecosystem because of rapid sediment burial and low decomposition rates (Bridgham et al., 2006; Magenheimer et al., 1996; Bartlett et al., 1987). Roughly half of marine carbon burial takes place in wetlands and similar shallow water ecosystems (Kirwan and Blum, 2011). High sediment accretion allows for the burial and accumulation of soil carbon in sediments, while anoxia and low sediment hydraulic conductivity promote slow microbial decomposition (Greiner et al., 2013). Overtime, high accretion rates and low decomposition rates in wetlands can build substantial pools of soil carbon.

Coastal wetlands have high net primary productivity due to high sediment and nutrient deliveries from riverine and tidal flow into the system (Bauer et al., 2013; Duarte et al., 2012). The conversion of organic matter into inorganic matter via mineralization result in the production of greenhouse gases including CO_2, CH_4, and nitrous oxides (N_20) (Raich et al., 1992). Carbon dioxide is produced from both aerobic and anaerobic respiration, while CH_4 is produced via anaerobic respiration (Whiting and Chanton, 2001). The relative proportion of CO_2 and CH_4 is critical to climate change simulations because CH_4 has forty-five times the sustained flux greenhouse gas warming potential of CO_2 and is responsible for 18% of anthropogenic warming (Neubauer and Megonigal, 2015). Sulfate-reducing bacteria impede the production of CH_4 by competing with methanogens and oxidizing CH_4, resulting in low fluxes from high salinity estuarine wetlands (Bartlett, 1987, Poffenbarger et al., 2011). By contrast, larger CH_4 fluxes tend to occur in lower salinity portions of the estuary with lower concentration of sulfate. While both fresh and salt marshes tend to be net carbon sinks, the potential of freshwater marshes to attenuate climate forcing effects may be offset by the production of CH_4 (Bauer et al., 2013).

Variations in sediment supply to estuaries result from both natural and anthropogenic factors. Natural factors include stochastic weather and climate events and a background rate of erosion (Wheatcroft, 2013). Anthropogenic factors include changes in land use which often contribute to erosion and sediment delivery to the estuary. In the Pacific Northwest, active tectonics, prodigious rainfall and land use practices, particularly timber harvest, influence sedimentation and carbon burial in coastal estuaries (Mathabane, 2015). In addition, sediment accumulation rates in estuaries depend on ocean forcing through wave energy as well as riverine contributions (Wheatcroft, 2013). Previous studies indicate estuary sediment accumulation rates in the Pacific Northwest range from 0.004-0.81 cm/yr (Mathabane, 2015). The rate of sediment deposition relative to sea level rise may dictate estuary dynamics such that if estuaries do not accumulate sediment faster than sea-level rise, estuaries may migrate inland (Kirwan et al., 2016; Kirwan and Blum, 2011; Reed, 1995). Conversely, if sediment deposition outpaces sea level rise, estuaries may contract. In the Pacific Northwest, sea level rise varies regionally from -1.8 mm/yr in Neah Bay to 4.26 mm/yr in Humboldt Bay, based on variable uplift associated with the Cascadia Subduction Zone (Komar et al., 2011). Quantifying local sediment accumulation rates for estuarine systems is essential for determining carbon accumulation rates, coastal estuary potential for blue carbon storage, and the fate of estuaries with future sea-level rise.

The carbon sequestration potential of estuarine wetlands has attracted attention as a potential buffer to global climate change (Kirwan and Mudd, 2012; Chmura et al., 2003). There is a growing interest to commodify wetland carbon sequestration through carbon credits as a greenhouse gas mitigation policy (Wylie et al., 2016; Murdiyarso et al., 2014; Pidgeon et al., 2011; Chmura et al., 2003). Carbon credit potential, in addition to the many ecosystem services of estuarine wetlands, has encouraged restoration of degraded and historically diked wetlands. Improved knowledge of carbon dynamics in reference and altered wetlands can improve regional carbon balance inventories and carbon credit policies (Wylie et al., 2016).

Coastal Wetlands & Degradation

Prior to large scale European settlement, there was an estimated 221 million acres of wetlands in the lower 48 states of the U.S. (Thayer et al., 2003). Many North American coastal wetlands were lost to the development for agriculture and population density pressures (Dahl et al., 2013; Thayer et al., 2003; Frenkel and Moran, 1991). Coastal marshes have been developed and altered by ditches, culverts and dikes for wetland land conversion into agriculture and urban lands (Dahl et al., 2013; Thayer et al., 2003). Globally, two-thirds of human populations are settled on coasts and noncoastal wetlands and rivers (Duarte et al., 2012). This dense human settlement leads to major ecological disturbance in estuarine and riverine systems (Duarte et al., 2012). Ecological degradation of marshes owing to human development include eutrophication, storm water pulses, agricultural runoff, and enhanced nutrient delivery (Dahl et al., 2013; Gleason et al. 2011). The degradation of wetlands leads to the loss of essential wetland ecosystem services and functions.

Coastal wetlands in the Pacific Northwest of the United States have experienced substantial losses. Land use alterations including dams, levees and shoreline development have altered sediment delivery rates in the region (Gleason et al., 2011). Major threats to West Coast estuaries include altered tidal exchange, altered nutrient dynamics and water quality, altered freshwater inputs, altered sediment regime, direct habitat loss, and climate change (Gleason et al., 2011). In the state of Oregon, excluding the Columbia River estuary, it is estimated that 80% of pre-European settlement salt marshes have been degraded and lost through land use changes (Frenkel et al., 1981). In the Coos Bay estuary, the focal area of my research, 90% of original salt marshes have been estimated to have been developed, primarily for agricultural use (Frenkel et al., 1981).

Pacific Northwest coastal wetlands are unlike Atlantic and Gulf coast wetlands which have developed largely on shallow, expansive coastal plains (Dahl et al., 2013). Pacific Northwest coastal wetlands occur in small, discontinuous segments because of the steep topography of the surrounding landscape and the predominance of a high energy and rocky coastline (Dahl et al, 2013; Komar et al., 2011). Thus, the loss of the small number of coastal marsh habitat has large implications for ecosystem function loss.

Wetland Policy

The *No Net Loss* national policy was originally established by the National Wetlands Policy Forum of 1988 which recommended a national policy to “achieve no overall net loss of the nation’s remaining wetland base, as defined by acreage and function, and to restore and create wetlands, where feasible” (Bendor, 2008). However, the No Net Loss policy is more of an ecological goal, as there is no regulatory power associated with it (Bendor, 2008). This policy encouraged national wetland restoration and mitigation to curb wetland loss and preferably, slowly increase wetland acreage in the face of development (Frenkel and Morlan, 1991). It also encourages compensatory mitigation of wetland degradation to restore or preserve wetlands acreage and ecological function (Bendor, 2008).

Wetland Restoration

Elements of wetland restoration include the reestablishment of natural ecological processes, creation of self-sustaining ecosystems, inclusion of ecological function performance standards, and integration into the surrounding landscape (Thayer et al., 2003). Ideally restoration projects should target a return to historic, least-disturbed conditions (Thayer et al., 2003). The restoration of disturbed marshes and the conservation of remaining marshes could mitigate and/or curb marsh habitat loss.

Both passive and active forms of wetland restoration have been done in the Pacific Northwest. Passive restoration of coastal wetlands typically refers to the removal of dikes and tidal gates which allow for the return of natural tidal hydrology to historically tidal wetlands (Cornu and Sadro, 2002). However, due to the aerobic environment created from diking and draining, wetlands tend to subside with peat and organic matter becoming decomposed and consolidated (Cornu and Sadro, 2002; Frenkel and Morlan 1991*)*. Active restoration of coastal wetlands typically restores historic vegetation by removing invasive species and planting native marsh species (Cornu and Sadro, 2002).

Research Context

My thesis research quantifies and compares soil carbon dynamics including carbon pools and fluxes along a natural salinity gradient in Coos Bay, OR. My research compares soil carbon function in disturbed, restored and reference marshes. Quantifying these carbon functions has implications for understanding coastal or “blue” carbon storage in the Pacific Northwest. The results of this research will contribute to regional knowledge of how Oregon estuarine wetlands meet goals of the No Net Loss policy by evaluating the restoration success of essential carbon functions relative to reference and disturbed wetlands.

CHAPTER II

LAND USE EFFECTS ON CARBON STORAGE & FLUXES IN OREGON COASTAL WETLANDS

Introduction

Wetland ecosystems are globally important stores of carbon (Bridgham, et al., 2006). Coastal wetlands occur within estuaries along a salinity gradient from freshwater marshes to salt marshes (Bartlett et al., 1987; Poffenbarger et al., 2011). In the conterminous U.S., 15.7% of coastal wetlands are saline and 85.3% are freshwater (Dahl & Stedman, 2013). Coastal wetlands have among the highest net primary productivity of any ecosystem in the world due to high sediment and nutrient deliveries from riverine and tidal flow into the system (Bauer et al. 2013; Kirwan & Mudd, 2012). Coastal wetlands are also the source of roughly 50% of all oceanic carbon burial (Kirwan & Blum, 2011).

Carbon cycling dynamics, including sedimentation and decomposition, differ along this salinity gradient due to the biogeochemical attributes associated with the gradient (Marin-Muniz, 2014). Because soils are often saturated, these environments are often anaerobic with slow rates of decomposition (Bartlett et al., 1987; Magenheimer et al., 1996). Despite these anaerobic soil conditions, saline, tidal-influenced wetlands typically have low methane fluxes. This phenomenon is due to sulfate-rich, marine waters causing high sulfate-reduction rates, and sulfate-reducing bacteria who outcompete methanogens for energy substrates (Bartlett et al., 1987; Poffenbarger et al., 2011).

In contrast, freshwater coastal wetlands can emit substantial amounts of methane (Chu et al., 2014), a potent greenhouse gas with a sustained-flux global warming potential forty-five times that of carbon dioxide over a 100-year period (Neubauer et al., 2015). Understanding carbon accumulation and fluxes are important for understanding the "blue carbon" storage potential of coastal wetlands. Bridgham et al. (2006) report that there are major gaps in wetland literature on trace gas fluxes and soil carbon sequestration in both estuarine and freshwater mineral soil wetlands.

The important carbon ecosystem functions that coastal wetlands provide are threatened by human disturbances, including anthropogenic climate change-induced sea level rise, nutrient and sediment loads, and coastal development (Culbertson et al., 2009). Globally, two-thirds of human populations are settled on coasts and noncoastal wetlands and rivers (Duarte et al., 2012) and about half of North America wetlands have been lost or degraded (Dahl & Stedman, 2013). About one-third of all wetlands in Oregon and Washington have been lost (Dahl & Johnson, 1991). Estuarine wetlands in the Pacific Northwest have experienced even greater losses due to drainage, diking, ditching and/or filling primarily for development as agricultural land (Frenkel et al, 1981; Weilhoefer et al., 2012). In Oregon, excluding the Columbia River estuary, it is estimated that 80% of pre-European settlement salt marshes have been lost (Frenkel et al., 1981). In the Coos Bay estuary, the focal area of the following research, it is estimated that 90% of original salt marshes have been lost (Frenkel et al., 1981). Net loss of coastal wetlands continues in the Pacific Northwest, albeit at a slower rate than historically. From 2004 to 2009, 2112 hectares of coastal wetland were lost (-0.4% of the total), although almost all of this loss was from freshwater wetlands (Dahl & Stedman, 2013).

To overcome the damage caused by wetland land use change, wetland restoration has been a major focus of both governmental and non-governmental groups (Kassakian et al., 2017; Whigham, 1999). The federal (USA) "No Net Loss" policy promotes the maintenance of wetland acreage and function despite on-going wetland development through mitigation and restoration (Whigham, 1999). The implementation success of the No Net Loss policy has been contested because wetland restoration and mitigation often results in the change from one wetland type to another, and potentially a loss of ecosystem function (Heimlich, 1997). Previous studies have concluded that wetland restoration and assessment should be based on reference wetland characteristics, including biogeochemical functions, hydrology, and biodiversity to better achieve the goal of "No Net Loss" (Whigham, 1999).

Given the desire to maintain the essential biogeochemical carbon-storage functions coastal wetlands provide, this study seeks to address the following: Along the salt to fresh marsh continuum, how do rates of carbon accumulation and carbon pools and fluxes vary by wetland land use treatment? More specifically, do restored wetlands meet reference level conditions in terms of the storage and cycling of soil carbon? If not, do they appear to be regaining some of their past function? To what degree have disturbed (ditched, diked, drained) wetlands lost essential carbon functions as compared to reference wetlands? This research will contribute to a growing regional database of blue carbon pools and fluxes in the Pacific Northwest.

Methods

Site Description

This study took place in the Coos Bay watershed, located on the Central Oregon Coast. The Coos Bay estuary is the largest estuary within Oregon (SSNERR Management Plan, 2006). Coos Bay has a Mediterranean climate with warm, dry summers and cool, wet winters (SSNERR Management Plan, 2006). The Coos Bay area typically receives less than 101mm of rain between May and September and roughly 1371 mm of rain from October through April (SSNERR Management Plan, 2006). Because of the coastal influence, average monthly air temperatures in the region are relatively stable from 7.5 °C to 15 °C throughout the year (Intellicast, 2017). The estuary has a mixed semi-diurnal tidal pattern with two high tides and two low tides of unequal height daily (SSNERR Management Plan, 2006). The mean tidal range is 2.3 m, with the highest tides 3.3 m above mean low low water (MLLW) and the lowest tides -0.9 m below MLLW (SSNERR Management Plan, 2006). The freshwater flow into the system is largely from the Coos and Millicoma Rivers (SSNERR Management Plan, 2006).

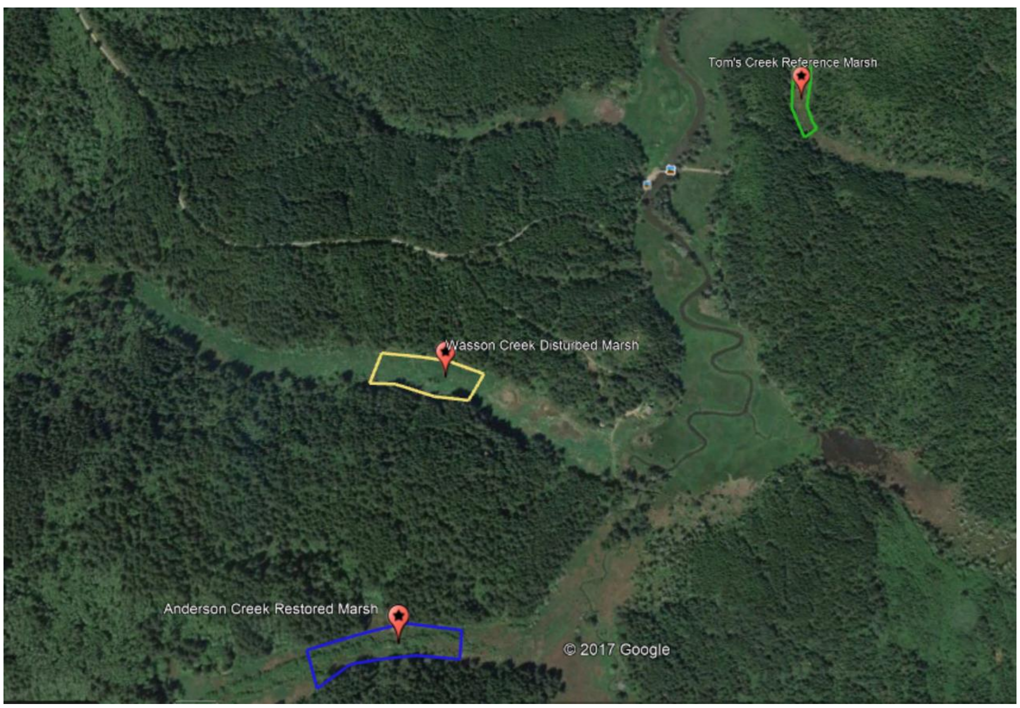


Figure 1. Google Earth image showing marsh land-use treatments within the fresh marsh complex in the South Slough National Estuarine Research Reserve. The reference marsh (Tom's Creek) is outlined in green, the disturbed marsh (Wasson Creek) is outlined in yellow, and restored marsh outlined in blue (Anderson Creek)

Figure 2. Google Earth image showing marsh land uses within the salt marsh complex. The restored marsh (Mangan Restored Marsh) is outlined in blue, the disturbed marsh (Mangan Disturbed Pasture) is outlined in yellow, and the reference marsh (Mangan Reference Marsh) is outlined in green.

This experiment focused on two different marsh systems at the endpoints of the salinity gradient. The freshwater marsh complex located in the South Slough Estuarine Research Reserve has three marsh land-use treatments-- reference, restored and disturbed marshes. The reference marsh, Tom's Creek marsh (43°16'44.97"N, 124°19'6.41"W), has little apparent anthropogenic disturbance. The lower part of the marsh has direct tidal exposure with a salinity of 5 ppt. The high marsh, the area used in this research, has a small tidal influence of several cm's with a salinity of 0-1 ppt. The other two freshwater marshes have no tidal effects in the study areas. There is a short transition area in the site where the vegetation dramatically shifts from halophytic graminoids to more diverse fresh vegetation trees, shrubs and sedges including *Alnus rubra*, *Lysichiton americanus*, and *Carex obnupta*. The marsh is wet throughout the year, but there tends to be more inundation in the winter because of greater precipitation.

The disturbed site, Wasson Creek (43°16'17.07"N, 124°19'22.47"W), was ditched, drained and converted to agricultural land during the early 1900s (Turnbull & Bridgham, 2015). It was used as agricultural land until the 1970s when it was abandoned (Turnbull & Bridgham, 2015). Currently, the ditches continue to divert water away from the site, and a constructed road causes the area to flood in the lower end (Turnbull & Bridgham, 2015). We sampled in the upper part of the marsh where the water table remained below 60 cm even during the winter. We chose to sample the drier end of the marsh because this better reflects the state of drained freshwater marshes. The site is now dominated by graminoids including *Phalaris arundinacea* and *Scirpus microcarpus* (Turnbull & Bridgham, 2015). The soil at this site has a thin, within the top 15 cm of soil, A horizon overlying a clayey B horizon.

Anderson marsh (43°16'3.78"N, 124°19'26.32"W) was actively restored in 2002 to represent the original vegetative community and hydrology with an ecological focus on recreating salmonid habitat (Cornu, 2005). Prior to restoration, this site had a similar disturbance history to Wasson Creek, which is only 400 m to the northeast in a parallel drainage. During restoration in Anderson Creek, the marsh was re-graded, channels created, and native species were planted to mimic the structure of the nearby Tom's Creek reference marsh (Cornu, 2005). The top 20 to 60 cm of topsoil was removed to fill the ditch draining the wetland, with 8-12 cm of silty sediment that has accreted since restoration lying on top of a clayey B horizon. Similar to Tom's Creek, the Anderson Creek site is wet throughout the year but experiences increased inundation during the winter months when rainfall is highest. Anderson Creek is populated by hydrophytic species including, *Carex obnupta, Scirpus microcarpus*, *Myrica californica, and Vaccinium ovatum* (Cornu, 2005).

The Mangan salt marsh complex is situated directly off the northern branch of Coos Bay in Haynes inlet. It was originally one contiguous salt marsh until 1934 when most of the marsh was diked and converted to agriculture (Mangan, L., landowner, personal communication, 2015). The disturbed land-use treatment (43°27'18.91"N, 124°12'21.02"W) remains diked and is currently used as pasture for cattle. Because of the dike, the site is fresh with a salinity of 0-1 ppt in the soil porewater. Faults in the tidal gates allow some salt water to flow in along the drainage ditches, but this water does not overflow into the pasture area. Winter rains inundate the site with standing water, around 8 cm above the soil surface. During the summer months, the site is dry with a water table nearly 1 m below the soil surface. Similar to the disturbed freshwater site, this diked area has a thin A horizon, within the top 15 cm of soil, over a clayey B horizon.

The restored site (43°27'21.99"N, 124°12'17.67"W) was similar to the disturbed site before passive restoration in 2003 during which the dike was removed and tidal flow returned. The site was restored as a mitigation project for the nearby Coos Bay airport (Mangan, L., landowner, personal communication, 2015). The restored site is a mudflat without vascular plants, but with some algae and the seagrass *Zostera japonica* sparsely located throughout the marsh. Similar to the freshwater restored site, 9-13 cm of silty sediment lies on top of a clayey B horizon.

The reference site (43°27'15.68"N, 124°12'25.96"W) is a small tidal high marsh fringing the dike for the disturbed salt marsh site. The reference marsh is populated mostly with *Salicornia sp.*, (pickleweed). The salinity in both the reference and restored salt marshes are around 33 ppt, representing very little freshwater influence.

Soil core sample collection

We extracted five shallow and five deep push soil cores from each site with PVC tubes (5 cm diameter, 15 cm length) during the summer 2015, winter 2016, and spring 2016 seasons. Shallow cores were removed from the depth of the soil surface to 15 cm, or from the surface to the initial clay layer in the restored sites, which was readily apparent above the loosely compacted recent sediments. This depth approximated the A horizon in the two disturbed sites. Deep cores went from the bottom of the shallow core to a 30 cm depth. Cores were plugged with compression caps on each end to create an airtight seal and transported from the field to the laboratory on ice.

To further estimate carbon storage in the reference sites, we used a Livingston corer to collect two 80 cm and 87.5 cm soil cores from the salt and fresh reference marshes, respectively. At each site, we first removed a surface layer of soil with a shovel, which we included in our subsequent analyses, from 0-10 cm and 0-19 cm depths in the fresh and reference marshes, respectively. We removed the surface layer to reduce the effects of compaction in driving the piston-corer through the dense root layers at both marsh complexes. In the field, the intact soil core was removed from the corer, wrapped in plastic wrap, and placed in PVC tubes for storage.

Percentage Carbon

Roots were removed from both the shallow and deep cores and the entire soil core homogenized, and then a subsample was oven dried at 60 °C for a minimum of 48 hours to calculate bulk density. Samples were then ground with pestle and mortar to determine total percent carbon on a Costech Elemental Combustion System Analyzer (Valencia, CA) using acetanilide-based standard curves. Known concentrations of San Joaquin NIST soils were used as standard checks.

The reference deep cores were used both to quantify carbon stocks and for radioisotope dating. A subset of soil from cores were dried and homogenized in 10 cm intervals for percentage carbon as described above.

Radioisotope dating analysis

For the radioisotope dating, both short and long cores were cut in 2 cm intervals and weighed. Both soil and plant material were retained in samples. The samples were oven dried at 60 °C for a minimum of 48 hours. The dried samples were ground into a fine powder with pestle and mortar, placed in polystyrene counting jars, and read for a minimum of 24 hours on a Canberra low energy germanium detector (Meriden, CT) to derive ^{210}Pb and ^{137}Cs radioisotope activity. The photopeaks were read at 46.5 keV for ^{210}Pb and 661.7 keV for ^{137}Cs. The downcore distribution of ^{210}Pb and ^{137}Cs radioisotope activity at depth was used to quantify sedimentation rates.

^{210}Pb is a naturally occurring radioisotope that is deposited on the Earth's surface through atmospheric deposition where it can absorb to the fine-grained components of soil and sediments which makes it useful as a geologic tracer or chronometer. ^{210}Pb is also produced naturally in soils and sediments through the decay of ^{226}Ra. This "supported" component must be separated from the "unsupported" component described above in order for sediment deposition rates to be estimated. After distinguishing the unsupported ^{210}Pb activity from the background supported ^{210}Pb activity, we can derive a sedimentation rate based on the known ^{210}Pb half-life of 22.3 years. We used the constant supply rate model, based on a constant atmospheric deposition of unsupported ^{210}Pb (Lubis, 2006). A linear regression with supported ^{210}Pb activity as a function of depth was used to quantify the accumulation rate using the following formula (Wheatcroft et al., 2013):

$$\ln A_z = \ln A_0 - \lambda / S(z)$$

where A_0 is ^{210}Pb activity at surface depth, A_z is ^{210}Pb activity at depth of z. λ is the ^{210}Pb decay constant, z is depth and S is sedimentation rate.

The ^{137}Cs-derived sediment accumulation rate was calculated from the activity associated with known global atmospheric nuclear fallout which peaked in 1963. Nuclear residual deposits first appeared in 1950s, peaked in 1963, and reached undetectable amounts after 1986 (Callaway et al., 2012). From the corresponding peak of ^{137}Cs at depth in the core, we divided that depth by the years since the 1963 peak fallout to determine sedimentation rate.

Soil carbon pools, accretion rates, and historical loss

Percent carbon and bulk density were used to calculate soil carbon content. We calculated carbon accumulation rates in the reference sites by multiplying the average carbon content in the appropriate depth interval by the sediment accumulation rate. We quantified both ^{137}Cs and ^{210}Pb carbon accumulation rates for both the fresh and salt reference marshes.

The depth of the recently deposited sediment and time since restoration was used to estimate accretion rates in the two restored sites. We assumed that the disturbed and restored sites had organic layers similar to their respective reference sites prior to disturbance and that this organic layer had completely oxidized in the 80+ years since they were drained and/or diked. From this information, we estimated a historical loss of soil carbon after correcting for the presence of the A horizon in the disturbed sites and the newly accreted layer in the restored sites. This assumption is well supported in the Mangan marsh complex since it was originally a single contiguous wetland. The freshwater complex of wetlands were non-contiguous but they did occupy very similar hydrogeomorphic positions in the watershed, which we believe justifies this assumption.

Gas flux sampling

To estimate methane (CH_4) and carbon dioxide (CO_2) fluxes, gas samples were collected using the static chamber method. One-year prior to sampling, 2.4 m x 0.6 m boardwalks were constructed to reduce erroneous gas fluxes associated with sample collection. Three boardwalks were placed at each site, with two chambers placed on either side of each boardwalk. Chambers consisted of a permanent chamber base, upon which a chamber top with a vent tube was placed only during sample collection. The fresh marsh complex chamber tops were made of PVC (39.5 cm diameter, 34 cm height). Chamber bases (39.5 cm in diameter, 14 cm in height) were driven down 7.5 cm into the soil. At the salt marsh sites, an alternative set of chamber tops constructed from retrofitted, airtight, screw-top buckets (27 cm in height, 29 cm radius) were used. The salt marsh chamber bases (28.5 cm radius) were constructed from the same buckets by cutting a hole through the bottom and pushing them 7.5 cm into the soil.

We collected one gas flux measurement per site during the winter wet season of 2016, within a three-week period during November and December. Using a needle and 30 mL syringe, eight samples were extracted from the chambers over the course of roughly 1.3 hours. Gas samples volumes of 14 mL or 17 mL were over pressurized into 9 mL or 12 mL (respectively), N_2 flushed, pre-evacuated serum bottles. The samples were stored upside-down on ice for transport back to the laboratory, and processed within a week. Gas flux samples were run on an SRI 8610 Gas Chromatograph (Torrance, CA) with flame ionization detector and methanizer to convert carbon dioxide to methane. Gas fluxes were calculated from the linear increase in methane and carbon dioxide over time.

Statistical Analyses

We used the combination of R 3.3.2 statistical package and IBM SPSS Statistics Version 24 for the statistical analysis. For the statistical analysis, we averaged the five replicate shallow and deep cores to derive mean values of bulk density, carbon stocks and percent soil carbon. Using the powerTransform function in the R "car" package to derive the best transformation to normalize the data. We transformed bulk density, percent carbon and carbon content using log and coefficient-based multipliers based on the output from the "powerTransform" function. We ran a two-way factorial ANOVA for each marsh complex, with land use and depth as fixed factors. If there were significant interactions between land use or depth on bulk density we ran one-way ANOVAs within each depth and within each treatment. Sedimentation and carbon accumulation rates in the restored fresh and salt marshes were compared with a t-test. We used p-values <0.05 as the threshold for significance.

Results

Carbon Stocks and Bulk Density

A. Freshwater Wetlands

In the fresh marsh complex the effect of land use depended upon depth ($p=0.008$, Figure 3). Across depths, the reference site had the lowest bulk density and was 25% of the bulk density of the disturbed site. In the shallow depth, the restored marsh had a bulk density intermediate of the reference and disturbed marshes (Figure 3). At the deeper depth, the restored and disturbed sites had similar soil bulk densities.

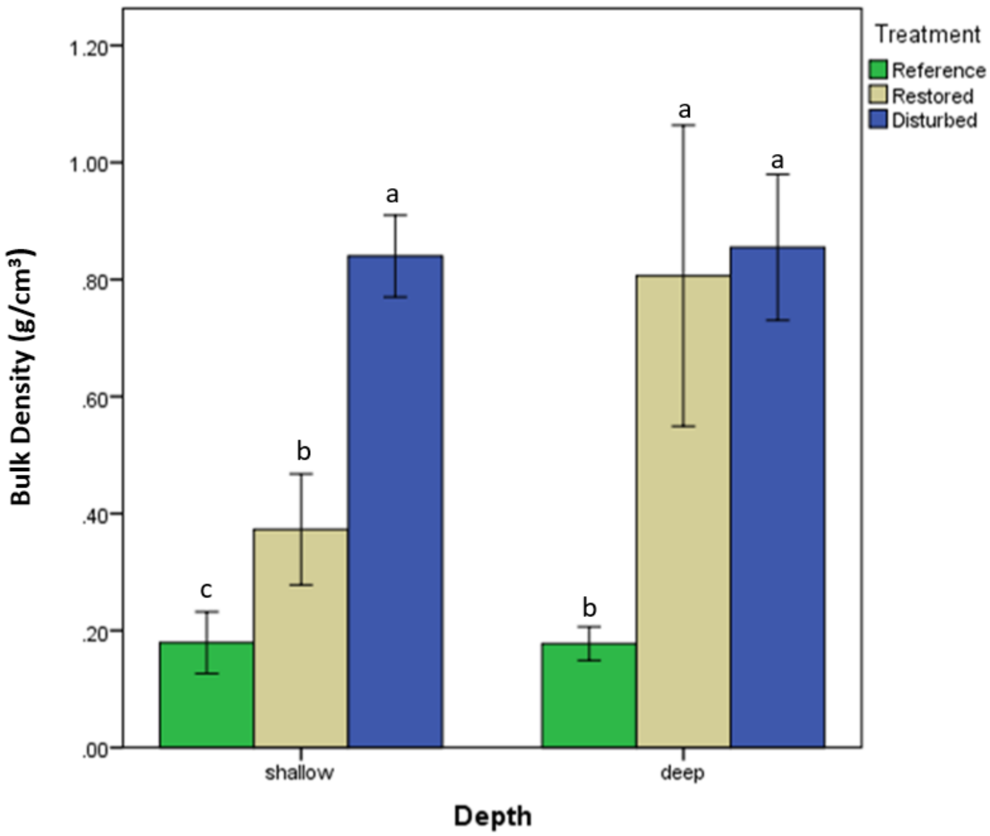


Figure 3. Soil bulk density for the fresh marsh complex. The shallow depth represents the first 0-15 cm or 0- to-clay layer. The deep depth represents soil cores from 15-30 cm. Letters indicate statistically significant differences among treatments within a depth. Error bars are based on a 95% confidence interval.

In the fresh marsh complex, the effect of land use on soil carbon concentration also depended upon depth ($p < 0.001$, Figure 4). Across depth, the reference site had between a three and four-fold higher soil carbon concentration than the disturbed site. In the shallow depth, the soil carbon concentration in the restored site was intermediate between the reference and disturbed sites (Figure 4). In the deeper depth, the restored and disturbed sites had similar soil carbon concentrations (Figure 4). Soil carbon concentrations of the restored and disturbed sites in the shallow depth were greater than the respective soil concentrations in the deep depth (Figure 4).

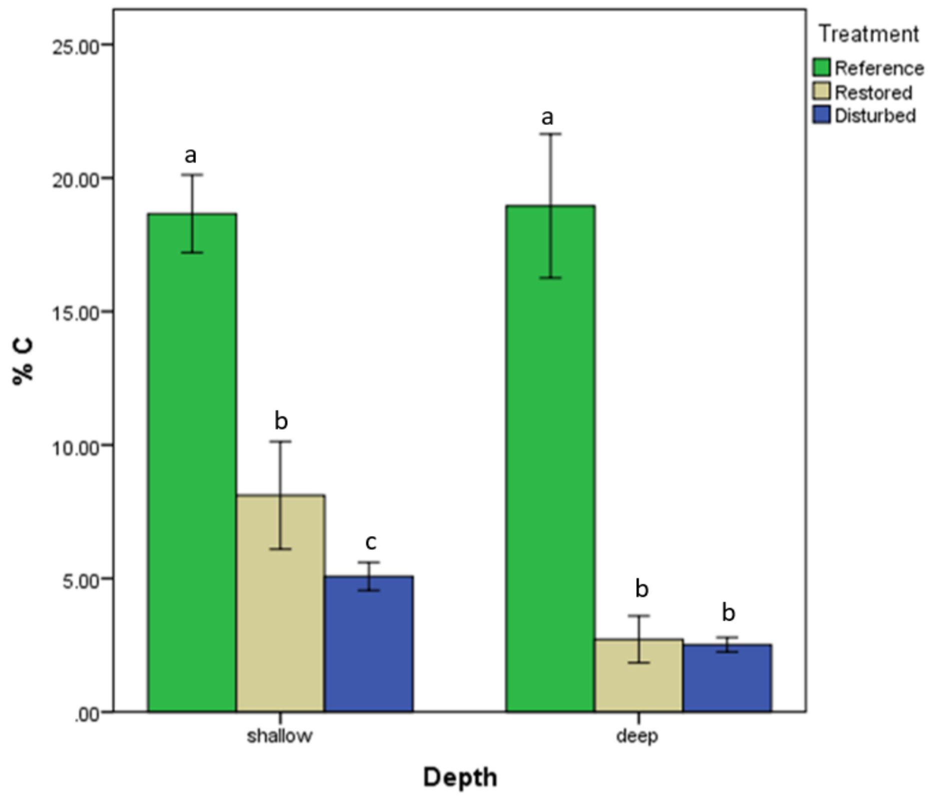


Figure 4. Percent soil carbon for the fresh marsh complex. The shallow depth represents the first 0-15 cm or 0- to-clay layer. The deep depth represents soil cores from 15-30 cm. Letters indicate statistically significant differences among treatments within a depth. Error bars are based on a 95% confidence interval.

The effect of soil carbon pools in the fresh marsh complex was also dependent on depth ($p<0.001$; Figure 5). In the shallow depth, the soil carbon pool was highest in the disturbed site and lowest in the restored site (Figure 5). In the deeper depth, the reference site had the highest carbon pool, with the restored and disturbed sites having similar pools. For all sites, carbon pools decreased from shallow to deeper soil depths.

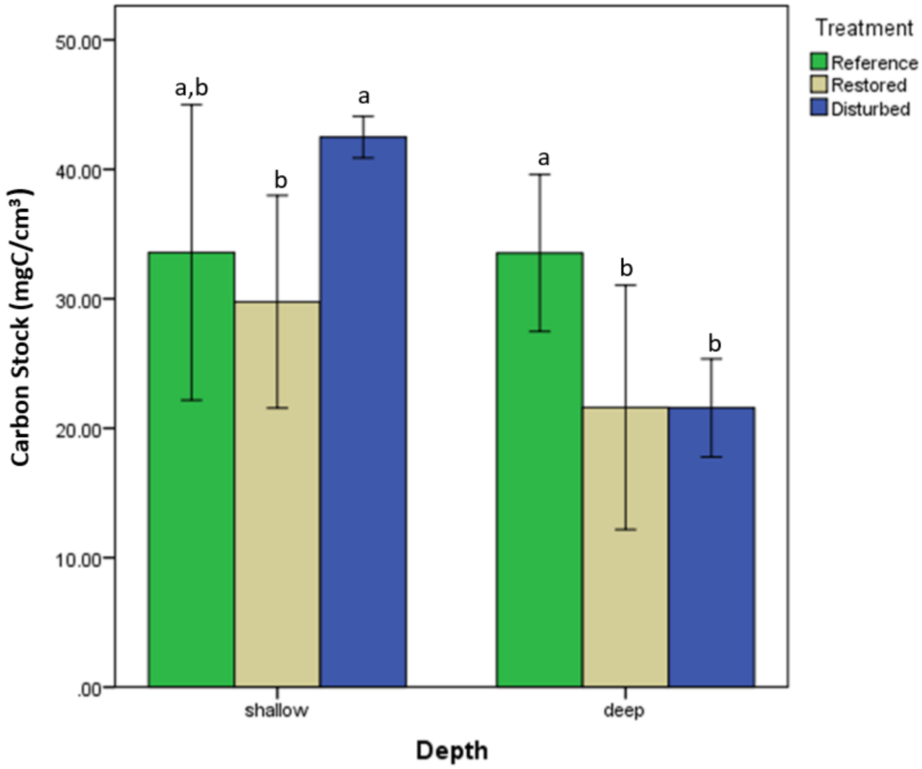


Figure 5. Soil carbon pools for the fresh marsh complex. The shallow depth represents the first 0-15 cm or 0- to-clay layer. The deep cluster represents soil cores from 15-30 cm. Letters indicate statistically significant differences among treatments within a depth. Error bars are based on a 95% confidence interval.

The reference wetland deep core had organic peat content from the surface down to 70 cm, below which there was a sharp transition to more mineral material (Figure 6, Figure 7), suggesting the initiation of the soil carbon accumulation in the wetland above this depth. The overall carbon pool in the fresh reference marsh was 33.86 kg C/m^2, including both peat and mineral matter, with 31.47 kg C/m^2 in the organic horizon (Figure 19). Overall the freshwater reference marsh has greater carbon storage, per area, than the reference salt marsh (Figure 6, Figure 11).

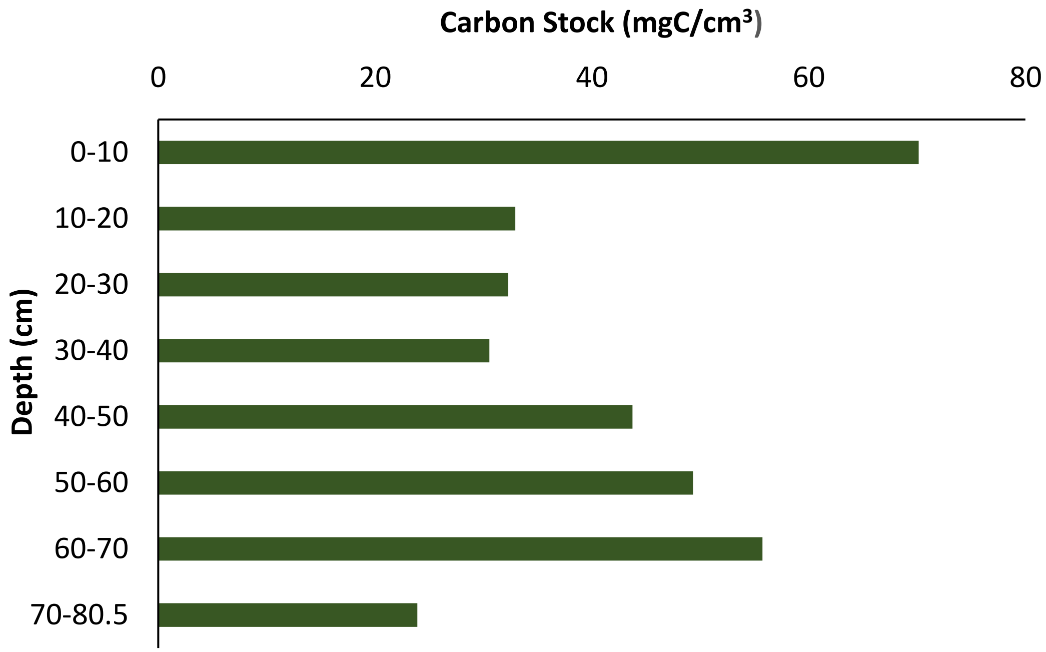


Figure 6. Soil carbon pools values for the fresh reference marsh, as a function of depth in the soil profile.

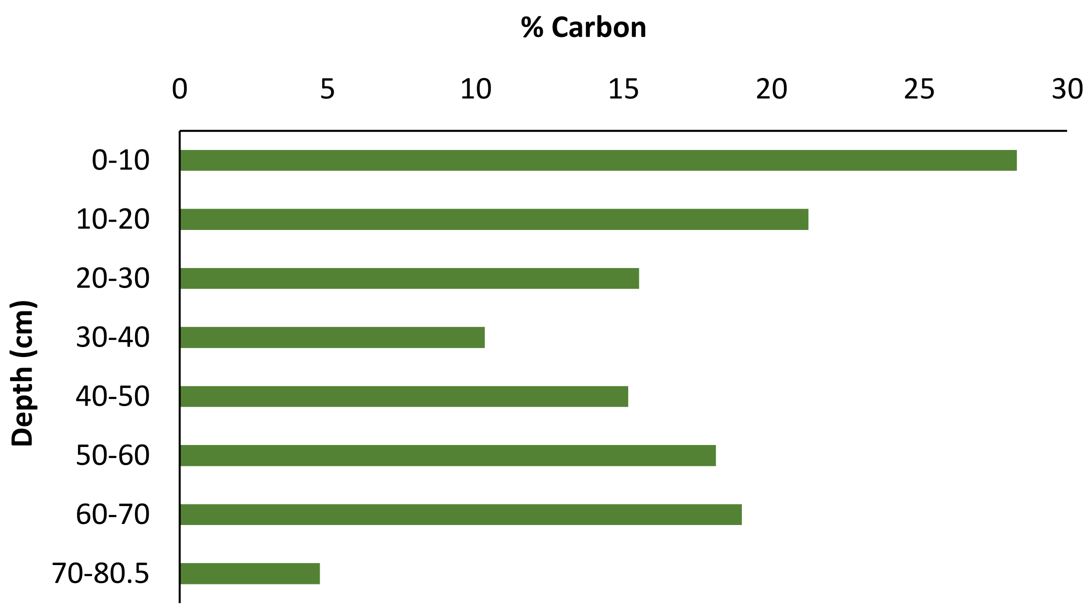


Figure 7. Percent carbon concentrations for fresh reference marsh with depth

A. Saline Wetlands

The effect of land use depended upon depth for soil bulk density in the salt marsh complex ($p<0.001$, Figure 8). The disturbed site had the highest bulk density in the shallow (0-15 cm) depth. In the deeper depth (~15-30 cm), bulk density was similar in the disturbed and restored sites and lowest in the reference site (Figure 8). Bulk density increased with depth except in the reference site.

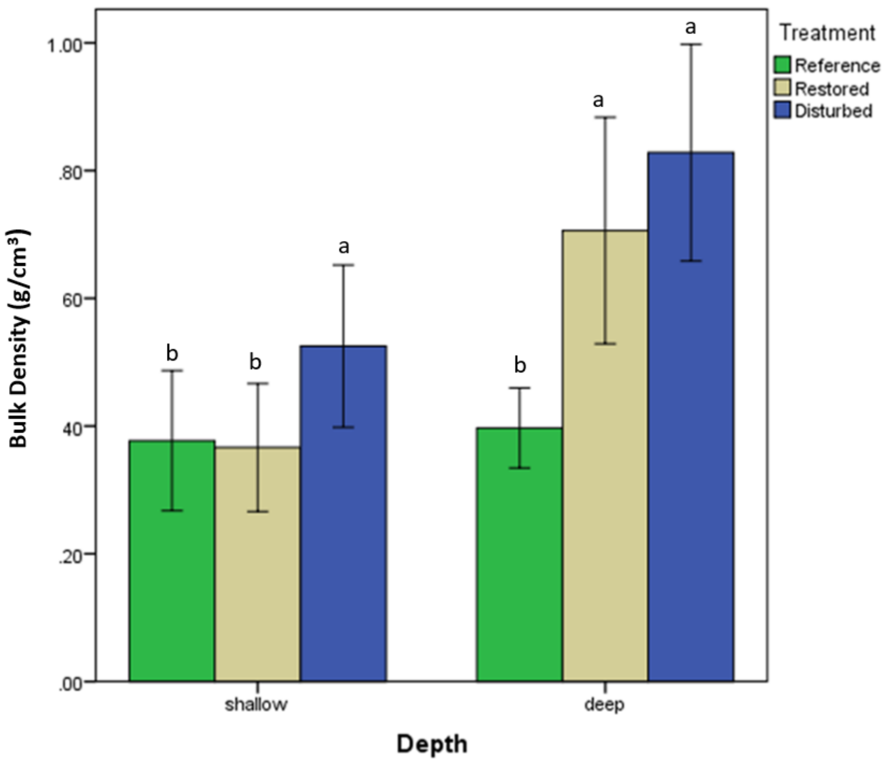


Figure 8. Soil bulk density for the salt marsh complex. The shallow depth represents the first 0-15 cm or 0- to-clay layer. The deep depth represents soil cores from 15-30 cm. Letters indicate statistically significant differences among treatments within a depth. Error bars are based on a 95% confidence interval.

The effect of land use on soil carbon concentration in the salt marsh complex again depended upon depth ($p<0.001$, Figure 9). In the shallow depth, percent carbon was lowest in the restored wetland and similar in the disturbed and reference sites. In the deeper depth, percent soil carbon in the disturbed site had half that of the reference site, with the restored site having an intermediate concentration (Figure 9). Percent soil carbon decreased with depth, expect for in the restored site which had comparable rates across depth.

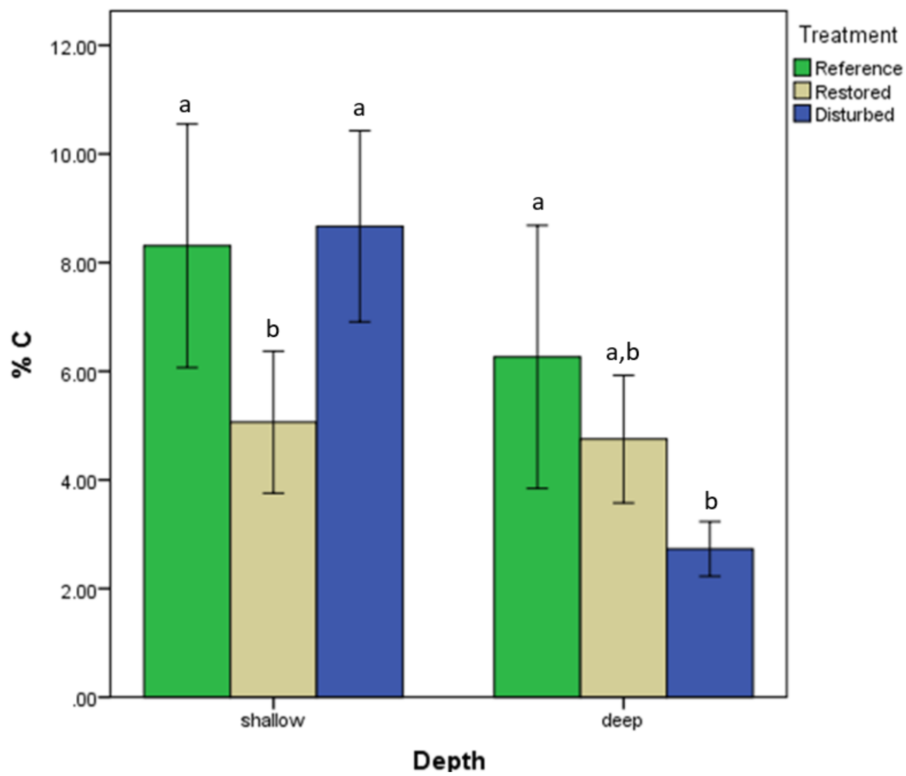


Figure 9. Percent soil carbon for the salt marsh complex. The shallow depth represents the first 0-15 cm or 0- to-clay layer. The deep depth represents soil cores from 15-30 cm. Letters indicate statistically significant differences among treatments within a depth. Error bars are based on a 95% confidence interval.

The effect of land use on soil carbon pools also varied with depth (p=0.003**,** Figure 10). In the shallow depth, the disturbed site had the highest carbon pool. The restored wetland had half the carbon of the disturbed wetland, with the reference wetland intermediate. In the deeper depth, the carbon pool was highest in the restored site and similar in both the reference and disturbed sites (Figure 10).

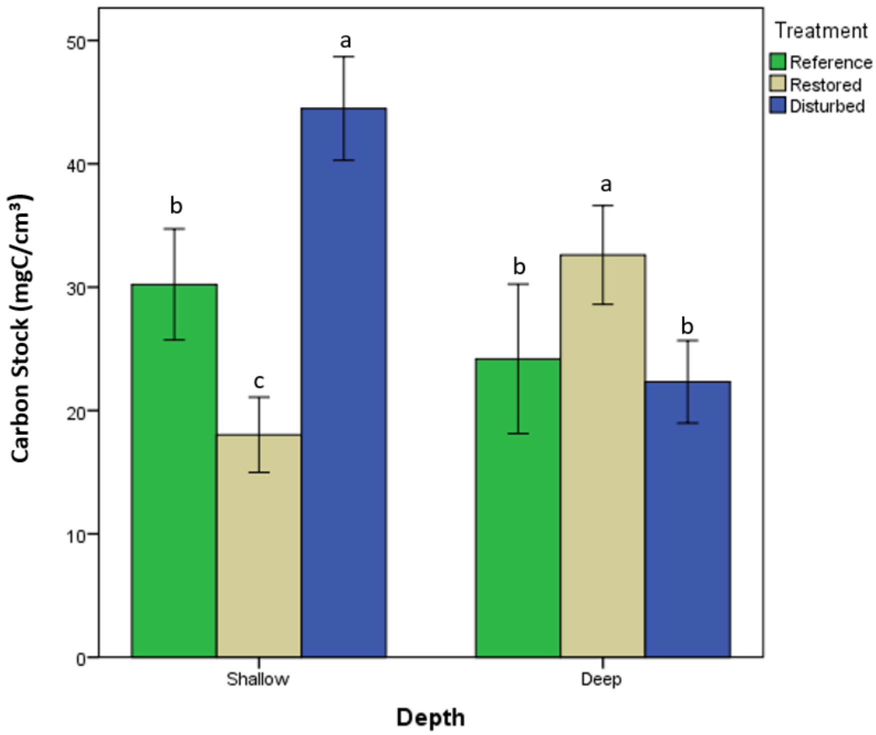


Figure 10. Soil carbon pools for the salt marsh complex. The shallow depth represents the first 0-15 cm or 0- to-clay layer. The deep depth represents soil cores from 15-30 cm. Letters indicate statistically significant differences among treatments within a depth. Error bars are based on a 95% confidence interval.

The reference marsh deep core contained an organic-rich sand from the surface down 40 cm. Below 40 cm, the soil was almost completely composed of sand (Figure 11, Figure 12). This transition from organic carbon to sand, also observed visually, suggests the formation of the salt marsh on the existing sand flat. The organic carbon pool in the reference salt marsh was 13.98 kg C/m^2, while the carbon content of the total marsh profile, including the sand mineral material was 18.31 kg C/m^2 (Figure 20).

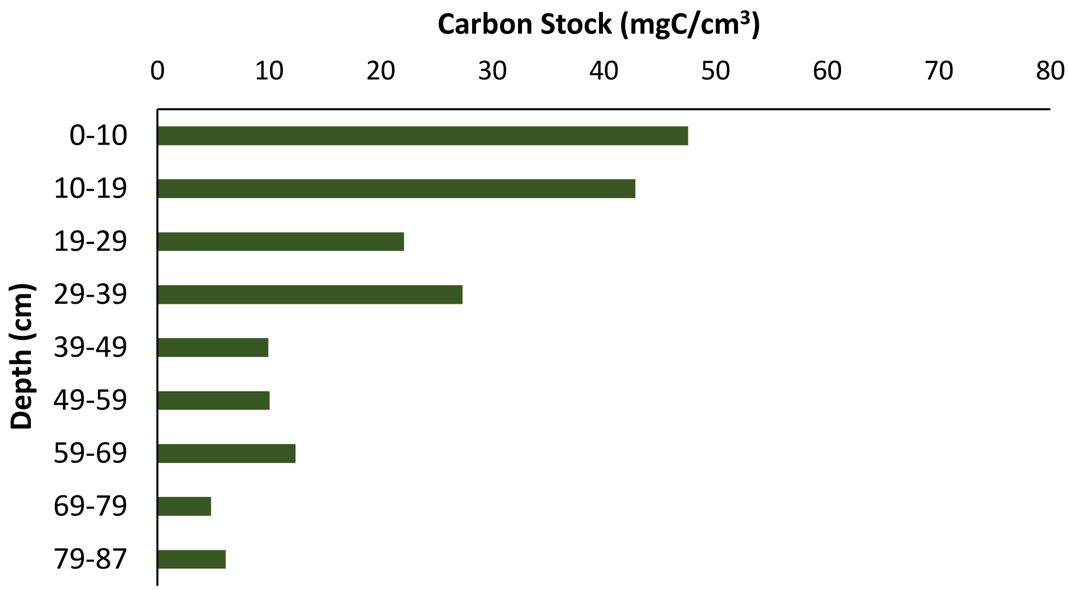


Figure 11. Soil carbon pool values for the reference salt marsh, as a function of depth in the soil profile.

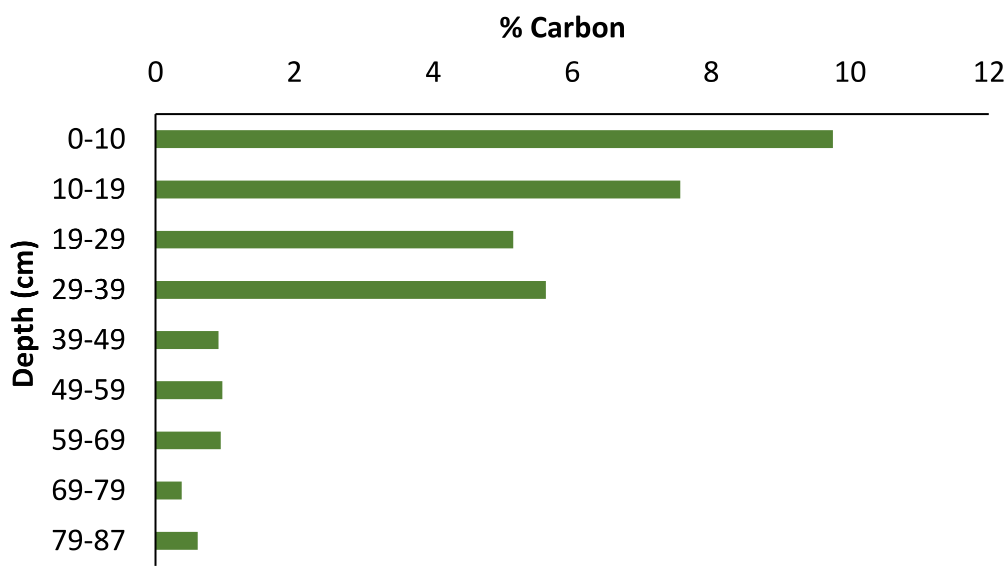
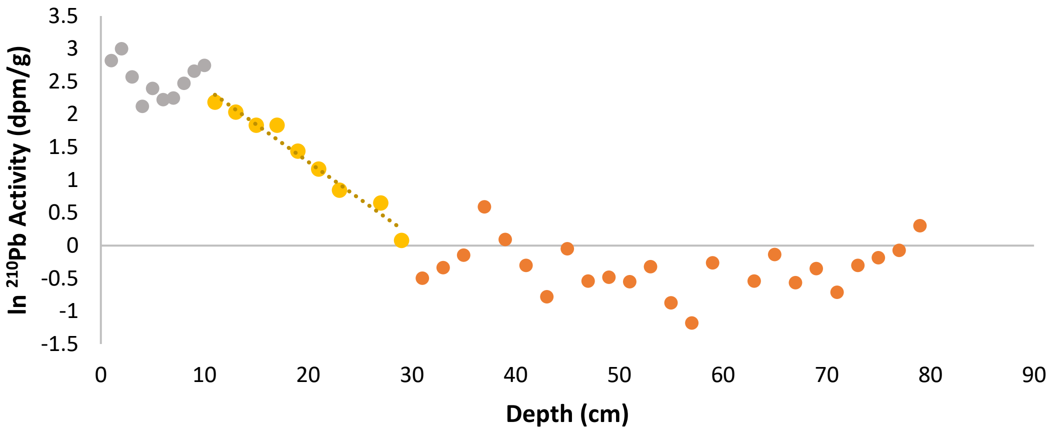


Figure 12. Percent carbon concentrations for the saline reference marsh with depth

Sedimentation & Carbon Accumulation Rates

The ^{210}Pb sedimentation rate was identified by isolating the interval in the core that excluded the bioturbated surface mixing layer, and background or “supported” activity (Figure 13) using the same procedure as Wheatcroft et al. (2013). We derived a sediment deposition rate of 0.28 ± 0.14 cm/year for the fresh marsh and 0.70 ± 0.17 cm/year for the salt marsh, suggesting that the reference salt marsh accumulates sediment roughly twice as fast as the freshwater reference marsh.

A.



B.

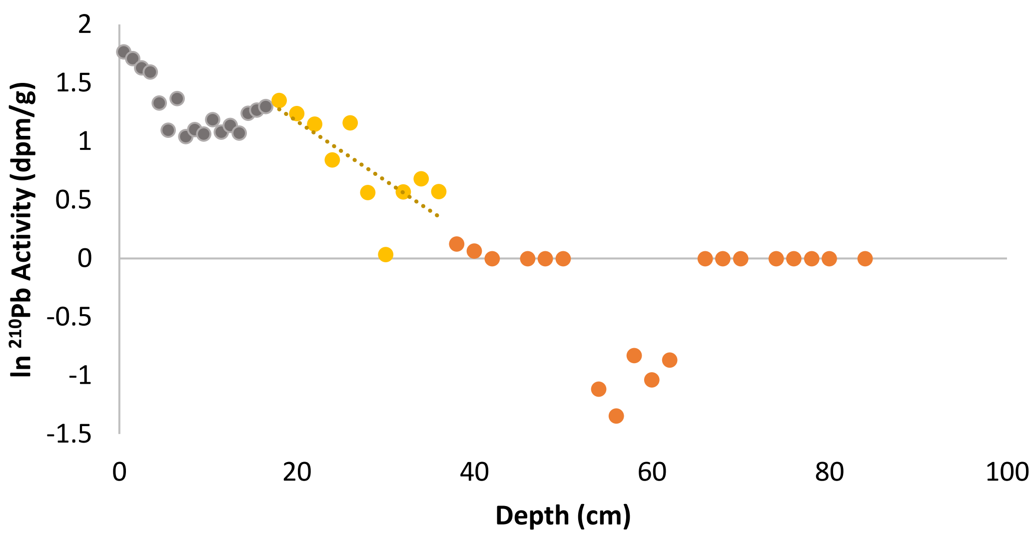
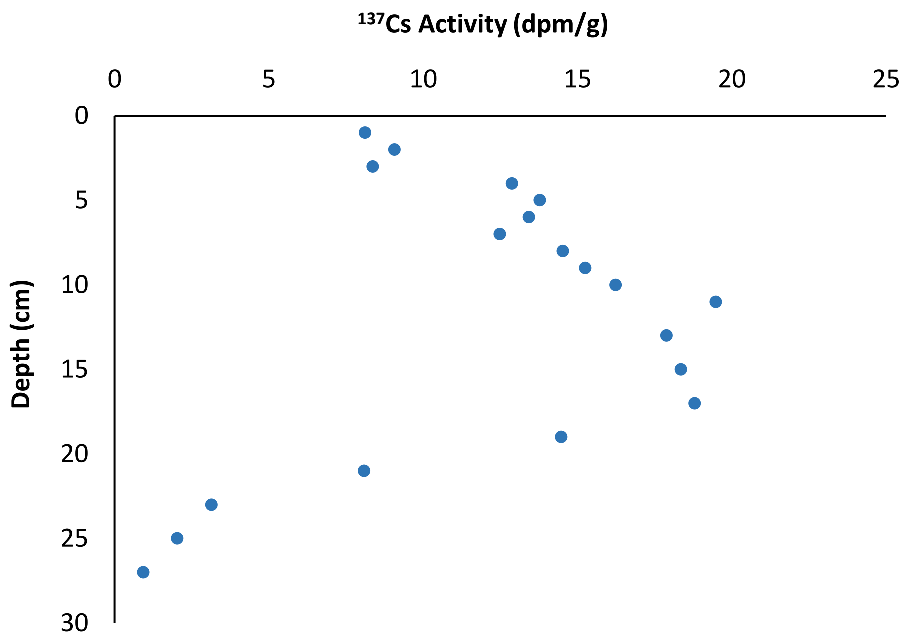


Figure 13. Natural log of ^{210}Pb as a function of depth in the reference fresh marsh (A) and the reference salt marsh (B). The gray points represent upper bioturbated soil layer. The orange points represent the background activity levels of supported ^{210}Pb from which. The yellow points represent the stable interval from which sedimentation rates were derived.

The 1963 nuclear ^{137}Cs fallout peak corresponded to a depth of between 11- 17 cm in the fresh reference marsh and 22- 26 cm in the salt marsh (Figure 14), giving a sedimentation range of 0.20- 0.32 cm/year in the fresh reference marsh and 0.41- 0.48 cm/year in the reference salt marsh. The ^{137}Cs rates between the fresh and salt marsh were more comparable than the ^{210}Pb rates. Importantly, both dating techniques reveal that the reference salt marsh accretes sediment faster than the reference fresh marsh (Table 1).

A.



B.

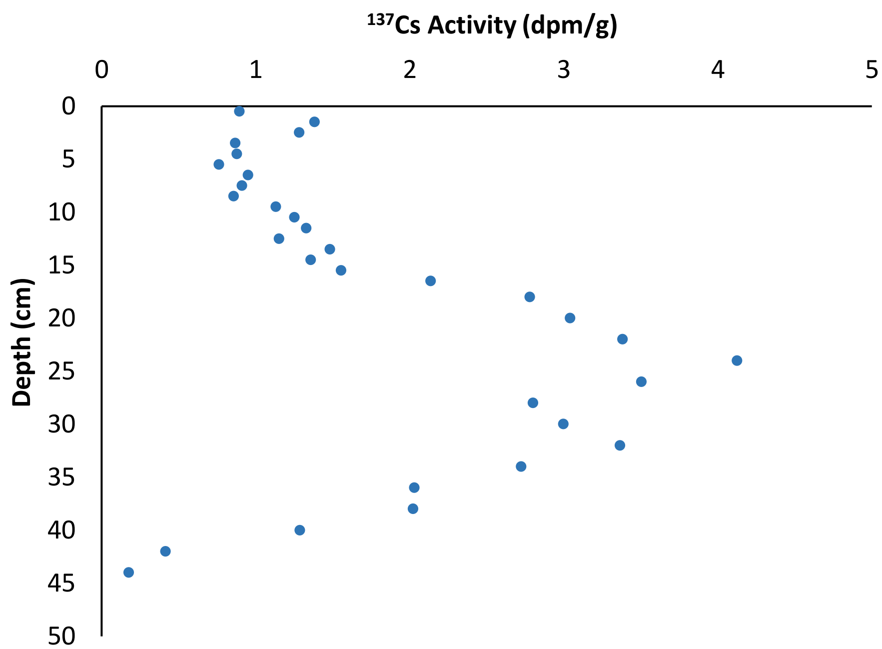


Figure 14. ^{137}Cs radioactivity as a function of depth in the fresh reference marsh (A) and the reference salt marsh (B). The peak in ^{137}Cs activity with depth represents the peak nuclear fallout year of 1963.

We calculated sedimentation rates of 0.70 cm/yr and 0.77 cm/yr for the fresh and salt marsh restored sites respectively, which were not statistically different (t-test; p=0.33). The sediment accretion rate in the restored salt marsh is comparable to the ^{210}Pb derived median sediment accretion rate in the reference salt marsh of 0.70 cm/year but lower than the ^{137}Cs rate of 0.44 cm/year (Table 1). However, the fresh restored marsh is accreting sediment approximately twice as rapidly as the reference fresh marsh (Table 1).

Table 1. Sediment accumulation rates in the reference and restored marshes derived from ^{210}Pb and ^{137}Cs radioisotope dating. Restored sedimentation rates derived from depth of accretion divided by years since restoration. Standard errors of ^{210}Pb are based on a 95% confidence interval.

	Sedimentation rate technique (cm/year)		
Site	**^{210}Pb**	**^{137}Cs**	**Manually measured accretion**
Fresh Reference Marsh	0.28 ± 0.14	0.20- 0.31	
Fresh Restored Marsh			0.70 ± 0.05
Salt Reference Marsh	0.70 ± 0.17	0.41- 0.48	
Salt Restored Marsh			0.77 ± 0.04

Carbon accumulation in the restored and reference wetlands show similar patterns to the sediment accretion rates. The ^{210}Pb derived carbon accumulation rates ranged from 126.3 ± 62.8 g C/m^2*yr in the fresh reference marsh, and 242.2 ± 57.7 g C/m^2*yr in the saline reference marsh. The ^{137}Cs derived rates ranged from 93.5- 144.5 g C/m^2*yr in the fresh reference marsh, and 140.9- 166.6 g C/m^2*yr in the saline reference marsh. Similar to the sedimentation rates, the ^{210}Pb and ^{137}Cs carbon accumulation rates are quite different in the saline reference marsh (Table 2). In contrast to the sedimentation rates between fresh and salt restored marshes, the restored fresh marsh is accreting carbon faster than the restored salt marsh (t-test, p=0.03).

Table 2. Carbon accumulation rates for the reference and restored marshes.

	Carbon accumulation rate (g C/m^2*yr)		
Site	**^{210}Pb**	**^{137}Cs**	**Manually measured accretion**
Fresh Reference Marsh	126.3 ± 62.8	93.5- 144.5	
Fresh Restored Marsh			209.1 ± 20.5
Salt Reference Marsh	242.2 ± 57.7	140.9- 166.6	
Salt Restored Marsh			143.1 ± 8.5

In situ Gas Flux

Land use determined CO_2 flux in both the fresh ($p=0.005$) and salt ($p = 0.001$) marsh complexes (Figure 15, Figure 16). The disturbed site had the highest rate of CO_2 flux in the fresh marsh complex and it was similar to the reference site in the salt marsh complex. In the fresh marsh complex, the reference and restored sites had similar rates of CO_2 flux. The restored salt marsh had a negligible CO_2 flux compared to the reference and disturbed sites. Overall the CO_2 fluxes were higher in the fresh marsh complex than the saline complex, except the reference fresh marsh which had a lower mean flux than the reference salt marsh (Figure 15, Figure 16).

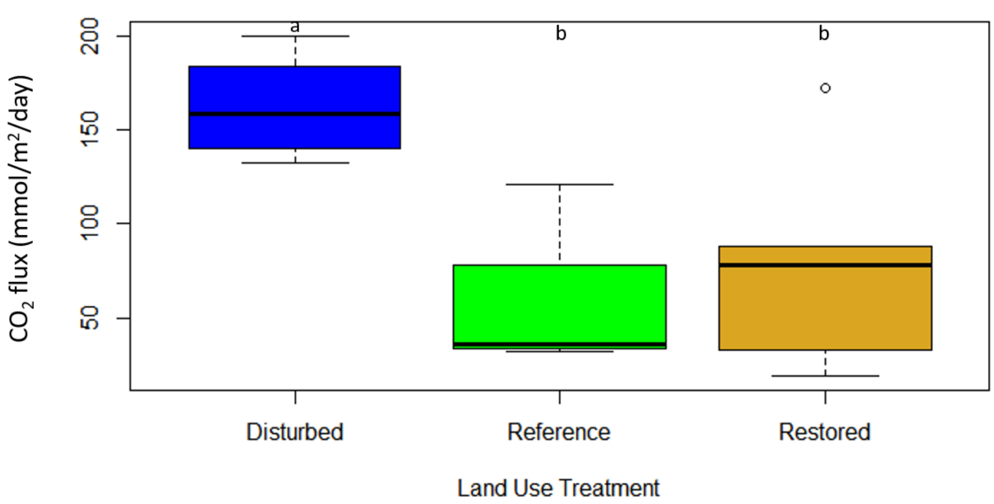


Figure 15. Carbon dioxide flux for each land use treatment within the fresh marsh complex. Letters indicate statistically significant differences across land use treatments (95% confidence interval).

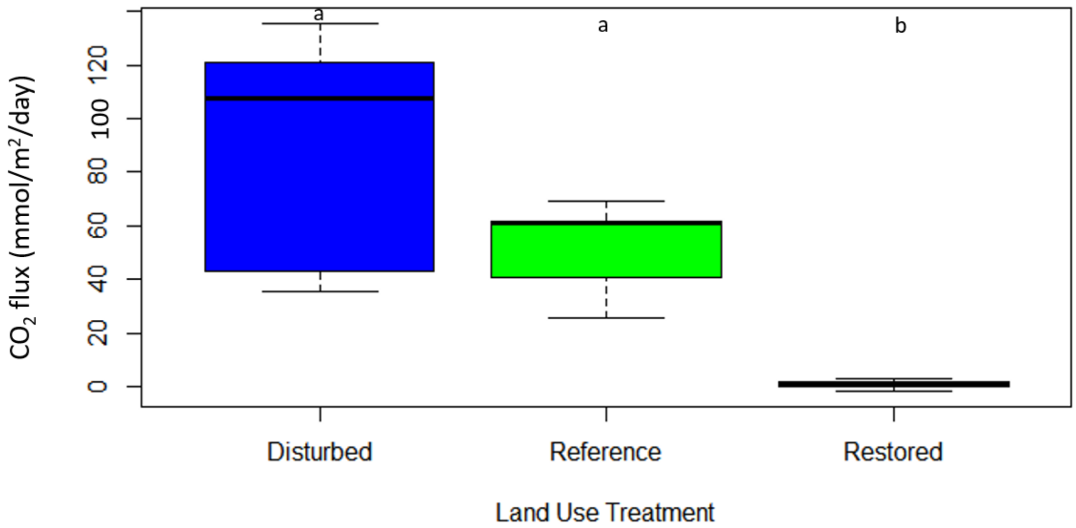


Figure 16. Carbon dioxide flux for each land use treatment within the salt marsh complex. Letters show statistically significant differences across land use treatments (95% confidence interval).

Methane fluxes from the fresh marsh complex were about four times higher than those in the salt marsh complex, except for the disturbed fresh marsh. Methane fluxes in the fresh marsh complex were similarly high in the reference and restored wetlands and essentially zero in the disturbed site ($p=0.003$, Figure 17). However, in the salt marsh complex, CH_4 emissions were essentially zero in all three sites and did not differ (Figure 18, $p=0.562$).

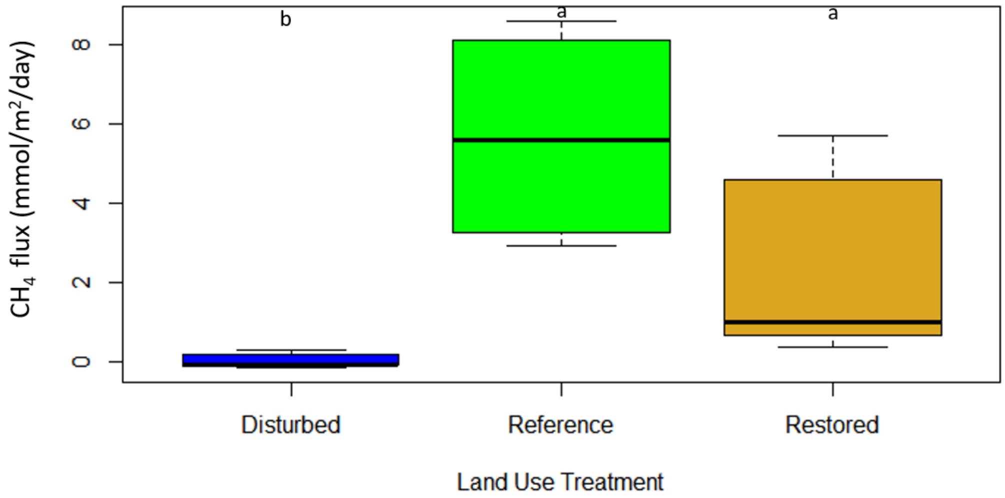


Figure 17. *In situ* methane flux for each land use treatment within the fresh marsh complex. Letters indicate statistically significant differences across land use treatments (95% confidence interval).

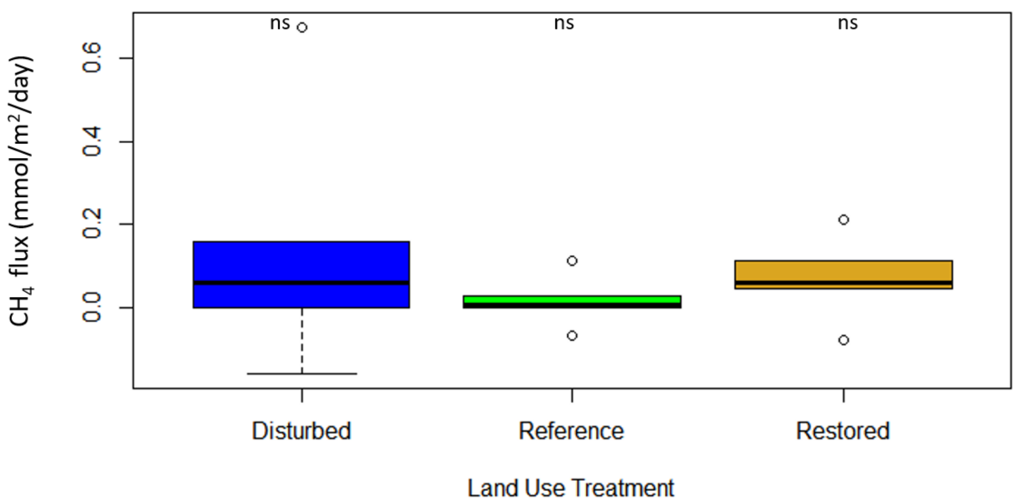


Figure 18. *In situ* methane flux each land use treatment within the salt marsh complex. Letters indicate statistically significant differences across land use treatments (95% confidence interval).

Discussion

Land use treatment influenced carbon fluxes and storage in both fresh and salt marshes along the endpoints of a salinity gradient. We saw significant effects of land use on soil properties including bulk density, percent soil carbon and carbon pools. The fresh reference marsh had roughly double the amount of total soil carbon with 33.7 kg C/m^2 than the salt reference marsh with 18.3 kg C/m^2 (Supplemental Table 1). Knowing the basal depth of organic material and sedimentation rate, we estimated that the fresh reference marsh has been accreting carbon for roughly 250 years. Using the same method, we estimated the reference salt marsh has been accreting carbon for roughly 50 years. Although these marshes have different tidal and hydrological influences, it does suggest an unexpected effect of land use on the sedimentation rate. The much younger age of accretion in the salt marsh suggests the reference marsh may be a result of dike construction in the mid-20th century or heavy sedimentation involved with logging in the watershed. However, the reference salt marsh is still a high marsh and can be used as a useful comparison in understanding how carbon cycling differs from restored and disturbed marshes.

Both fresh and saline disturbed and restored marshes are believed to have oxidized all the original organic peat after conversion from fresh marsh to diked and drained agricultural land in the early twentieth century (Figure 19, Figure 20). The resulting lower water table and hydrologic change due likely to aerated soil may have increased decomposition of the peat. At the salt marsh complex, which was originally one contiguous marsh, subsidence likely occurred with the biochemical consumption of historical carbon pools. Both the disturbed sites function at least part of the year as upland grasslands, with a relatively organic A-horizon of approximately 15 cm depth (Figure 5, Figure 10), which slightly offsets the loss of the organic matter in the former wetlands. We estimate that the disturbed marshes lost organic matter roughly equivalent to current reference pools (Figure 19, Figure 20). The disturbed fresh marsh likely lost an estimated net 25 kg C m^{-2} given the current reference pool of 31.5 kg C m^{-2}, and the current A horizon pool of 6.4 kg C m^{-2} (Figure 19). Similarly, the disturbed salt marsh likely lost roughly 7 kg C m^{-2} given the current reference pool of 13.98 kg C m^{-2} and the current A horizon pool of 6.7 kg C m^{-2} (Figure 20). Given the thin A-horizon and the advanced age of these disturbed sites, it seems likely they are currently approximately static in terms of soil gains or loss.

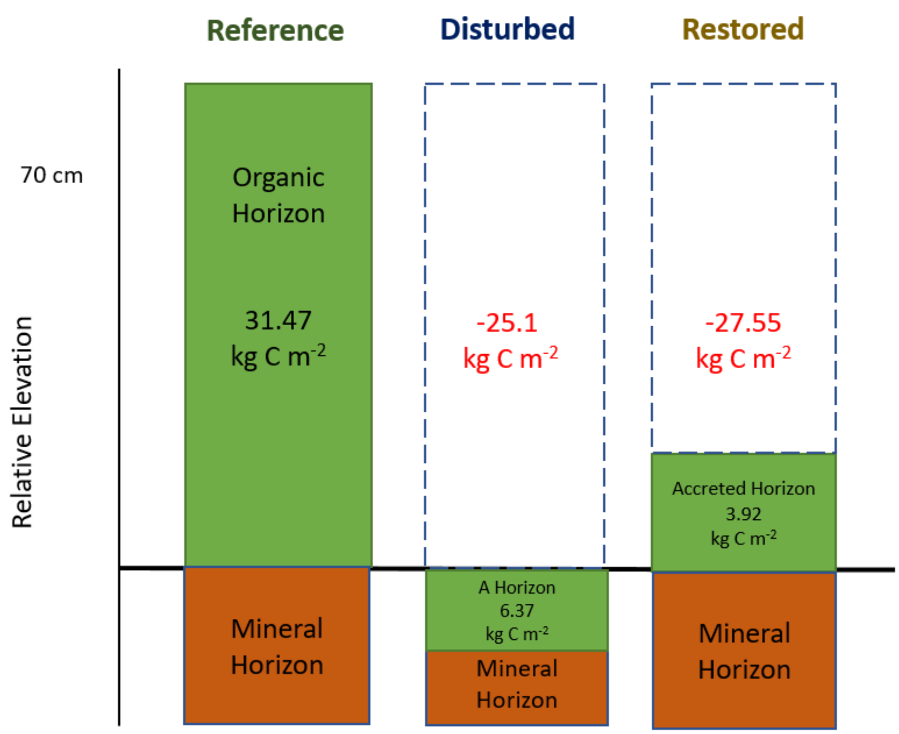


Figure 19. Current soil carbon stocks and estimated historical loss in the fresh marsh complex across wetland land use treatment. The green boxes represent accreted carbon, or for the disturbed marsh, carbon in the remaining A horizon. The dashed lines represent potential subsidence. Values in red are carbon pool loss estimates.

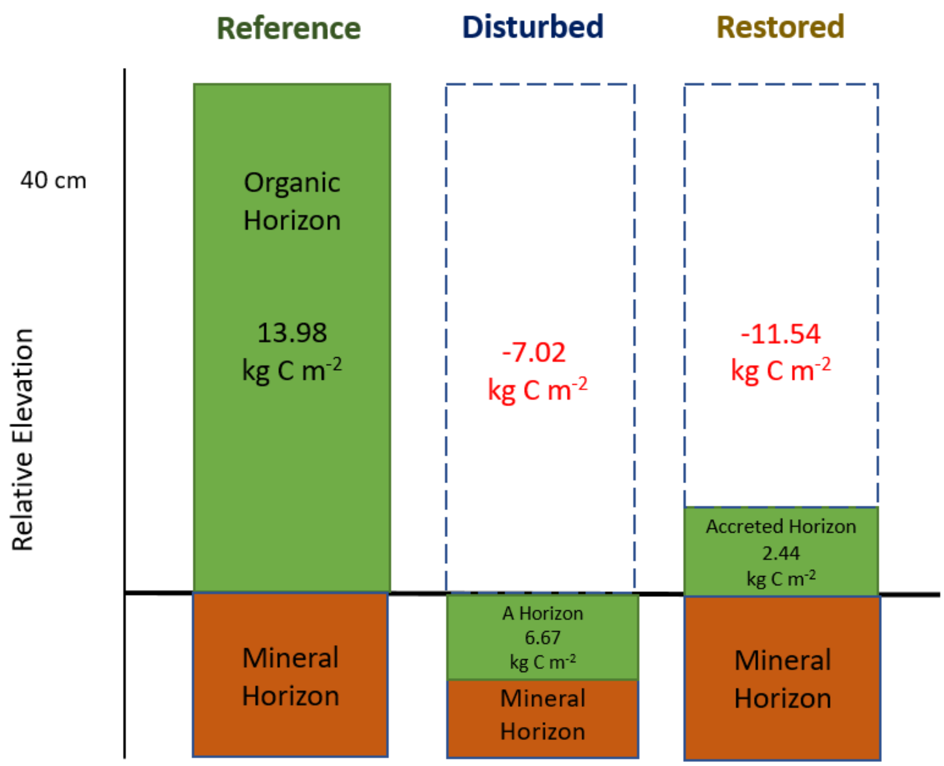


Figure 20. Current soil carbon stocks and estimated historical loss in the salt marsh complex across wetland land use treatment. The green boxes represent accreted carbon, or for the disturbed marsh, carbon in the remaining A horizon. The dashed lines represent potential subsidence. Values in red are carbon pool loss estimates.

Like the disturbed sites, both the fresh and saline restored sites also likely lost all the original peat in the marsh profile when the original marshes were converted to agriculture and the accreted organic material was oxidized. However, since restoration these marshes have been actively accreting carbon, as is readily evident from the uncompacted sediment setting on top of a clayey B-horizon. The saline and fresh restored marshes were restored 14 to 15 years ago, respectively. The restored fresh marsh contains 3.92 kg C m^{-2} and is accreting carbon at a rate of 209.1 ± 20.5 g C/m^2*yr in the accreted sediments since restoration, while the restored salt marsh contains a slightly smaller pool of 2.44 kg C m^{-2} and has a carbon accretion rate of 143.1 ± 8.5 g C/m^2*yr (Figure 19,

Figure 20, Table 2). These surface organic pools are smaller than the A horizons of the disturbed sites, which was scraped off of Anderson Creek and may have eroded away or become part of the fresh sediment layer in the salt marsh restored site. However, both of these restored sites are actively accreting organic matter at a rate that is similar to or higher than the respective reference sites, in juxtaposition to the probable static state of carbon accumulation in the disturbed sites. Thus, the restored sites are on a trajectory to recovered carbon storage ecosystem function. Still, the original transformation of these sites from marsh to agricultural land represents a major loss of soil carbon (Figure 19, Figure 20).

Methane flux was negligible in our reference and restored salt marsh complex study sites with active tidal exchange of high salinity water (Figure 18). This is consistent with our prediction of low CH_4 flux in the salt marsh sites due to the inhibitory characteristics of sulfate on methanogenesis (Poffenbarger et al., 2011; Bartlett et al., 1987). There was a small flux of CH_4 in the disturbed site of the salt marsh complex. However, we expected high CH_4 fluxes in the disturbed site because it was inundated with freshwater during the winter season when sampling occurred. Methane fluxes need to be further sampled in the salt marsh complex to verify this finding, because if the disturbed site did have high methane fluxes, then restoration of the site would result in a large greenhouse benefit by reducing those emissions.

In contrast, there was a significant land use effect on CH_4 production in the fresh marsh complex (Figure 17). High CH_4 fluxes in the reference and restored fresh marshes indicate a return of this function in the restored marsh, while lower CH_4 flux from the disturbed fresh marsh indicate a loss of methanogenic function (Figure 17). However, it is important to consider that high emissions of the potent greenhouse gas CH_4 is typically considered to have a negative societal value.

In the fresh marsh complex, the disturbed sites produced higher dark fluxes of CO_2 (Figure 15). The restored fresh marsh CO_2 flux rate was intermediate between the reference and disturbed sites, suggesting a partial return of function. This return of function was not realized in the restored salt marsh where CO_2 flux was very low, likely because of the low carbon content of its sediments and very limited vascular plant biomass.

More seasonal gas flux data are necessary to support our findings. These measurements represent a single winter sampling time point. However, the relatively stable average monthly temperatures might lead to similar seasonal fluxes, except for summer months when the disturbed fresh site dries up and potentially consume CH_4. Overall land use influenced *in situ* gas flux and in some cases restoration indicated a partial return of wetland function.

The role of coastal wetlands in mitigating the effects of climate change-induced sea level rise, and carbon storage through sequestration has been championed by scientists and policy makers. Recent studies highlight the uncertainties associated with coastal wetland resilience in the face of sea level rise. Some studies suggest that the feedback processes associated with sea level rise might lead to a net gain of coastal wetlands (Kirwan & Mudd, 2016; Kirwan et al., 2016). Other studies highlight that nuances associated with sea level rise and marsh extent showing a gain in some marsh areas and a reduction in others (Kassakian et al., 2017). However, these claims are typically based on marsh ability to migrate inland on the more expansive Atlantic and

Gulf Coasts marshes (Kassakian et al., 2017; Kirwan et al., 2016). In the Pacific Northwest, where salt marshes are segmented, marsh migration may be more limiting (Gleason et al. 2011). A previous study quantified the annual average rate of sea level rise in Coos Bay as 0.11 cm/year (Komar et al., 2011). Our rates of sediment accumulation in the reference and restored salt marshes revealed that these areas are currently outpacing sea level rise with an average sedimentation range of 0.44 to 0.77 cm/year.

This research will add to blue carbon inventories. The land use context may prove useful for any developing blue carbon mitigation markets in the region. Because of the small proportion of original coastal wetland ecosystems in the region, blue carbon dynamics were historically understudied. However, the small total area of coastal ecosystems and high proportion of land use change represent a disproportionate loss of ecosystem functions essential to the storage and transformation of carbon. Our research shows that with time, restored marshes may have the ability to regain these carbon cycling functions. Policymakers, private landowners and NGOs in the region should consider a concerted conservation and restoration plan to increase potential of blue carbon storage in the region, which may increase the regional ability to buffer the effects of climate change.

CHAPTER III

CONCLUSION

We found that different land uses influence carbon cycling and storage in fresh and salt marshes on the Oregon coast. We observed significant effects of land use on soil properties including bulk density, percent soil carbon and carbon pools. Soil carbon pools were greater in the fresh marsh complex than the salt marsh complex. We concluded based on field observations that most of the original organic soil carbon in both restored and disturbed wetlands was lost after draining and diking. However, wetland restoration presents an opportunity to regain carbon storage functions with the return of natural hydrology.

Land use also influenced overall carbon fluxes in both the fresh and salt marsh complexes. Carbon dioxide *in situ* fluxes were highest in the disturbed wetlands. We measured very low *in situ* CH_4 flux in all salt marsh land uses. Methane fluxes were similar between fresh reference and fresh restored marshes, but low in the disturbed marsh.

Although ^{210}Pb and ^{137}Cs carbon accumulation rates varied, overall the reference salt marsh appears to accumulate carbon faster than the reference fresh marsh. The fresh restored marsh is accumulating carbon faster than the restored salt marsh. Generally, restored wetlands showed a partial return of carbon storage functions. If not, at minimum restored wetlands are advancing in the direction of the reference marshes and return of carbon function may be achieved over time.

It is important to consider historic land use when assessing carbon pools in coastal wetlands. The disturbed wetlands were active or abandoned pastures with an A horizon within a 15 cm depth over a low carbon-density B horizon. The subsidence upon oxidation of the original deeper organic layer was visually evident in the salt marsh, and an even deeper organic layer was oxidized upon drainage in the freshwater disturbed marsh if it was originally similar to the reference marsh. We observed that with restoration, both the freshwater and salt marshes began rapidly accreting carbon. Although not measured in this study, and in addition to the loss of blue carbon, drained and diked salt marshes also represent a loss of other beneficial ecosystem services and functions associated with coastal wetlands, including shoreline protection, floodwater retention, recreation, wildlife habitat, etc. (Dahl et al., 2013; Gleason et al., 2011).

If we analyze the return of ecosystem carbon functions of restored marshes based on the federal No Net Loss policy, our findings suggest that meeting the collective goal of maintained or enhanced wetland function may depend on time frame. Some studies have found that major carbon cycling features can be returned in a restored wetland within five years of restoration (Cornell et al. 2007). However, Oregon coastal wetlands appear to show only a partial return of carbon storage functions 14-15 years post-restoration. Because of the scarcity of coastal wetlands in the Oregon, land-use changes represent a disproportionate loss of ecosystem function. More than 80% of coastal marshes have been lost in Oregon. Restoration of Pacific Northwest coastal wetlands may become increasingly important with regional interests in establishing blue carbon markets.

APPENDIX

SUPPLEMENTARY FIGURES AND TABLES

Supplementary Table 1. Carbon pool data for deep core highlights percent carbon, bulk density in reference sites for saline and fresh marsh complexes

Site	Depth Interval (cm)	Bulk Density (g/cm^3)	%C	C Stock (mg C/cm^3)	Overall C (kg C/m^2)	Sum of Overall C in complete core (kg C/m^2)	Sum of Overall C to organic horizon (kg C/m^2)
Fresh	0-10 Ped	0.25	28.3	70.2	7.0		
	10-20	0.16	21.3	32.9	3.3		
	20-30	0.21	15.5	32.3	3.2		
	30-40	0.30	10.3	30.5	3.2		
	40-50	0.29	15.2	43.7	4.4		
	50-60	0.27	18.1	49.3	4.9		
	60-70	0.29	19.0	55.7	5.6		
	70-80.5	0.50	4.7	23.9	2.4		
						33.7	31.5
Saline	0-10 Ped	0.49	9.8	47.6	4.8		
	10-19 Ped	0.57	7.6	42.8	4.3		
	19-29	0.43	5.2	22.1	2.2		
	29-39	0.49	5.6	27.3	2.7		
	39-49	1.1	0.9	9.9	1.0		
	49-59	1.0	1.0	10.0	1.0		
	59-69	1.3	0.9	12.4	1.2		
	59-79	1.3	0.4	4.8	0.5		
	79-87	1.0	0.6	6.1	0.6		
						18.3	14.0

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