SOIL ORGANIC CARBON CONTENT OF COMPENSATORY WETLAND MITIGATION PROJECTS IN AUBURN, WASHINGTON

by

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ABSTRACT

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Christina L. Stalnaker

Wetlands provide many essential ecosystem services, such as water quality improvement, flood prevention, and critical species habitat. 6% of global land cover is categorized as wetland, yet wetlands are estimated to account for 20-71% of earth's terrestrial carbon storage (Dahl, 2011, Reddy & DeLaune, 2008). Natural wetlands are often permitted to be developed, and replacement wetlands are subsequently either constructed or restored in their place to fulfill federal regulation. Laws dictate that no net loss of ecosystem function may result due to permitting activity, therefore it is obligatory to engineer wetlands functionally equivalent to those lost. However, the ability of wetlands to sequester carbon is often ignored during the evaluation and monitoring of natural and replacement mitigation wetlands. This study compares two ecosystem functions of constructed and restored wetland mitigation sites in Auburn, Washington: (1) soil organic carbon content and (2) species richness, and investigates if physical parameters such as size, age, or adjacent land use affect these functions. There was no correlation between size, age, and adjacent land use with species richness or carbon content. It was observed that the weight percent (%) soil organic matter content (SOM) of constructed wetlands was half that of restored (7.8 ±4.0%, 15.3 ±12.1%, respectively) ($\chi^2(1)=9.4$, $p=0.002$). These values are drastically lower than % SOM in similar natural wetlands in the area (45.5 ±34.2) (Horner, Cooke, Reinelt, Ludwa, & Chin, 2001). Soil bulk density was a much better predictor ($R^2=0.584$) of % SOM than soil texture ($R^2=0.020$). The wetlands that were excavated using heavy equipment and layered with top soil had the lowest % SOM values, indicating that this activity is compacting soil and limiting the soil development of these mitigation sites and capacity to store soil organic carbon. Future mitigation projects should choose soil with low bulk density and high soil organic matter content and avoid soil compaction during construction.

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1. Introduction

Wetlands are vastly complex ecosystems which deliver services that humans, fauna, and flora rely upon for survival. They deliver an array of benefits ranging from improving water quality, providing a habitat for a wide range of species, and preventing floods through water storage in porous soils. Wetlands are also extremely important global carbon sinks. While they represent less than 6% of global land cover, estimates of its share of terrestrial carbon storage range from as low as 20 and as high as 71% (Dahl, 2011, Hossler & Bouchard, 2010, Reddy, et al., 2008). Due to the immense contribution of their precious ecosystem services, wetlands are vital to the health of our watersheds and our planet. Despite their necessity, natural wetlands are destroyed through development and then compensated through artificially constructed or restored wetlands, known as compensatory wetland mitigation.

Prior to European settlement, the conterminous United States had approximately 215 million acres of wetlands; today less than half of that natural span remains intact (Kusler & Kentula, 1990). More recently, from 2004 to 2009, an estimated 551,870 acres of natural wetland were lost while 489,620 acres were gained through mitigation activity (Dahl, 2011). Because wetlands are an essential part of the landscape, the Clean Water Act mandates no net loss of ecosystem function caused by this permitting action. As human manipulation of wetlands persists, mitigation efforts depend chiefly upon these permits, and it is therefore crucial to understand the impacts of these trade-offs on wetland ecosystem function.

While the directive to mitigate these impacts originates in federal regulatory agencies, local governments bear the brunt of permit issuance and monitoring compliance and ecological success. City planners in Auburn, Washington are interested in incorporating knowledge about successful wetland mitigation strategies to ensure they satisfy no net loss of ecosystem function rules by assessing whether wetland mitigation projects are meeting key ecological standards. Auburn is located in the White River Valley of the Puget Sound Lowlands, nestled between the urban sprawl of Seattle, Bellevue, and Tacoma and the less-disturbed foothills of the Cascade Range. Auburn's topography includes many natural wetlands, but its proximity to a growing metropolis makes these wetlands subject to rapid development and intense permitting activity. In 2012 Auburn conducted a series of rapid assessments of wetland functions of 24 wetland mitigation sites in the area with support from the Environmental Protection Agency as part of their Wetland Mitigation Assessment Project (WMAP). From these assessments, Auburn's Environmental Services Department discovered that wetland mitigation efforts achieved mixed levels of ecological success.

To supplement the findings in WMAP, this paper evaluates an additional ecosystem service not currently considered in most wetland performance reviews: the soil organic carbon content of restored and constructed wetlands. Global climate change caused by rising atmospheric carbon dioxide and other greenhouse gas concentrations can be partially mitigated by terrestrial carbon sequestration. Therefore, it is ever more important to understand the ability of constructed and restored compensatory mitigation wetlands to store carbon.

This paper presents an investigation of the differences between these two mitigation project types and explores whether physical parameters that can be controlled, such as size, age, project type (construction or restoration), and adjacent land use, can impact ecological functioning of wetlands. More specifically, we test the following hypotheses: (1) there is a difference in ecosystem function, as characterized by species richness, between constructed and restored compensatory mitigation wetland project types in Auburn, Washington; (2) there is a difference in soil organic matter content between constructed and restored wetland mitigation projects in Auburn, Washington; (3) differences in soil organic content and species richness can be explained by size, age, and adjacent land use.

It was found that size, age, and adjacent land use were not good predictors of soil organic carbon or species richness. While there are no differences between constructed and restored wetlands in terms of species richness, there were significant differences between them regarding soil organic carbon content with restored wetland soils containing twice as much as constructed. Soil bulk density appeared to be a contributing to these variations as high soil bulk density values correlated with lower soil organic matter, and vice versa. This may be a result of heavy equipment use during excavation of project sites, and the use of topsoil with lower organic matter content as fill. These disparities illuminate the need to further study the mechanisms causing these differences, and incorporate findings into regulations at all levels of government.

This thesis is organized in the following way. Chapter 2 (Literature Review) reviews the legal requirements of wetland mitigation and the current systems permitting authorities use to evaluate their success or failure. It presents a synopsis of previous research illuminating the differences in ecosystem services provided by natural and constructed wetlands. Measures of areal and functional loss of freshwater wetlands are described, as well as the scientific principles that govern the ecosystem functions of freshwater wetlands.

Chapter 3 (Methodology) details the methods used to test the hypotheses. Study area, research design, and statistical analyses are described. Chapter 4 (Results) portrays the results of this analysis. Chapter 5 (Discussion) interprets the results in a framework relative to the original research question and hypotheses. This section discusses the implications of study results for present day mitigation efforts. It offers recommendations to permitting authorities and policy makers in regards to freshwater wetland mitigation with an emphasis on future research possibilities.

2. Literature Review

This literature review begins with a summary of basic wetland functions and their ecosystem services followed by a description of federal, state, and local regulations which govern wetlands and wetland mitigation projects. Next, wetland construction and restoration techniques are described with the criteria used to evaluate these mitigation strategies. Then, estimated functional and areal loss of wetlands attributed to mitigation permitting activity is discussed, followed by a synopsis of case studies using biological and abiotic factors to compare natural versus mitigation wetlands. The final section explores the relationship between the role of wetlands in global climate change and carbon sequestration through storage of organic matter in wetland soils.

2.1 Wetland Functions

Wetlands are terrestrial ecosystems that are defined by the presence of three unique characteristics. They are periodically or permanently inundated with water, home to hydrophytic plants, and contain hydric soils (Cowardin, Carter, Golet, & LaRoe, 1979). Hydric soils are able to store large volumes of water which produce anaerobic conditions (Reddy, et al, 2008). The Cowardin classification system of wetlands further divide wetlands into classes or subsystems according to their hydroperiod and predominant vegetative cover. In this study, freshwater non-tidal palustrine systems that are predominantly covered with emergent and scrub-shrub vegetation are considered (Cowardin, et al., 1979). The unique conditions in which wetlands exist account for their ability to provide a wide variety of essential ecosystem services.

Programs use different techniques to evaluate ecosystem services and functional success of wetlands and wetland mitigation sites. In order to describe functional success, it is important to understand functional qualities of natural wetlands. Appendix A displays Puget Sound Water Quality Wetland Preservation Program's list of 11 wetland function/value indicators. This program uses these indicators as criteria to select viable wetlands for preservation to compensate for development. Functional descriptions include: wildlife habitat support and biodiversity; floodwater, sedimentation and erosion control; nutrient/pollutant entrapment & assimilation; water flow; and several cultural values, such as recreational and educational opportunities (Washington State Department of Ecology, 1988).

Though it is quite difficult to put an absolute monetary value on ecosystem services, economists have estimated how much they could be worth by comparing them to man-made systems. For example, to determine the value of waste treatment one could compare the capital and operating cost of a local waste water treatment plant to process the same volume of water (Flores, Batker, Milliren, Harrison-Cox, 2012). Table 2.1 shows the high and low estimates in dollars per acre of a recent study of the value of wetland ecosystems in Thurston County, Washington.

Table 2.1 *Estimated monetary value of wetland ecosystem services in Thurston County, Washington. Recreated from Flores et al., 2012.*

Ecosystem Service	**Low Value ($ per acre)**	**High Value ($ per acre)**
Aesthetic and Recreational	$1.67	$4,641.41
Disturbance Regulation	$18.35	$8,578.76
Food Provision	$63.40	$9,372.90
Gas and Climate Regulation	$1.79	$774.40
Habitat Refuge and Nursery	$99.76	$13,560.51
Raw Materials	$2,816.44	$2,816.44
Waste Treatment	$76.39	$19,116.50
Water Regulation	$148.48	$17,351
Water Supply	$10.01	$33,969.02
Total	**$3,236.29**	**$110,180**

2.2 Compensatory Wetland Mitigation

2.2.1. Local, State, and Federal Regulations

When a land owner wishes to dredge, fill, or otherwise adversely impact an existing wetland on their property, in the United States, they are required by federal law to receive a permit from the US Army Corps of Engineers. First, applicants must show why they cannot avoid impacting the wetland by modifying their construction plans or using an alternative location. Then, if the impacts are proven to be unavoidable within reason, they must submit a compensatory wetland mitigation plan (Environmental Protection Agency, 1972). In the last 5 years, over 56,400 written authorization were issued by the Corps, with more than 5,600 of them requiring compensatory mitigation in the United States (Institute for Water Resources, 2015).

Prior to a ruling on wetland mitigation in 2008, compensation was established by using mitigation ratios calculated from areal extent. For example, given a 2:1 ratio, if 1 acre of freshwater marsh was filled, 2 acres of wetland would have to be created or restored in an alternate location, preferably on site. However, the 2008 Federal Mitigation Rule, emphasizes that the goal of mitigation was to achieve no net loss of ecosystem function, abandoning mitigation ratio requirements in favor of using a watershed perspective and equivalencies to the ecosystem functions (Title 33, 2008). Mirroring this requirement in Washington State, permits must indicate compensation in the form of a replacement wetland (restored or constructed) that delivers the same ecosystem function in accordance with RCW Title 90 Chapter 90.84 (Washington State Legislature, 1998).

The City of Auburn, Washington's Municipal Code also incorporates compensatory wetland mitigation in its regulations. They define wetlands as a critical area which performs important ecosystem functions, yet is environmentally sensitive. Auburn Municipal Code 16.10.010 states (2015):

> The primary goals of wetland regulation are to avoid adverse effects to wetlands; to achieve no net loss of wetland function and value – acreage may also be considered in achieving the overall goal; to provide levels of protection that reflect the sensitivity of individual wetlands and the intensity of proposed land uses; and to restore and/or enhance existing wetlands, where possible.

The Auburn Environmental Services Department is responsible for enforcing, monitoring, and validating wetland mitigation action within city limits.

2.2.2. Mitigation Practices

As previously mentioned, there are two types of wetland mitigation actions that can be taken: construction or restoration. Permit applicants indicate in their mitigation plan whether they will be conducting a construction or restoration project. Wetland construction establishes a new wetland ecosystem where none previously existed. This is accomplished through influencing hydrology by grading, digging, and excavation and then establishing vegetative communities of hydrophytes and other native wetland species through planting seedlings, cuttings, and natural recruitment. Wetland restoration either re-establishes or enhances existing degraded wetlands to improve specific ecosystem functions. This work can entail invasive species removal, native species planting, and habitat enhancements.

An interagency publication by Washington State Department of Ecology, U.S. Army Corps of Engineers, and U.S. Environmental Protection Agency (EPA) Region 10 provides guidelines for developing mitigation plans in Washington to follow from when it is decided a wetland will be impacted to the completion of the mitigation project (2006). The affected wetland should first be delineated by establishing the location and boundaries and the impacted physical, chemical, and biological functions determined by a qualified wetland professional. During site selection they recommend extensive consideration of the source of water, soil conditions (including organic matter content and compaction), prior and adjacent land use, wildlife species and corridors, and vegetation. Five environmental factors for project design are outlined: water, soil, vegetation, invasive species, and target functions.

While the guidelines recommend many soil functions to consider, such as improving water quality and nutrient availability for plants, no mention of soil organic carbon storage or carbon sequestration is mentioned. However, the authors do recommend salvaging topsoil from the impacted site- a practice which was not done in any of the 24 mitigation projects studied. They also strongly advocate consideration of using organic amendments when this is not possible, or when invasive species dominate the source, noting the importance of organic matter for vegetative establishment and nutrient cycling. Guidance for vegetation and species diversity include examination of nearby seed banks in soils as seeds from adjacent lands often colonize mitigation sites. Additionally, planting plans should incorporate a variety of appropriate species to support biodiversity and dynamic wildlife habitats (Washington State Department of Ecology, et al, 2006).

2. 3 Wetland Mitigation Project Evaluation

Zedler (1996) offers a complex view of constructing wetlands. She asserts that oversimplified evaluations of ecosystem function paired with little to no enforcement of stated performance standards does not properly address the complex nature of how ecosystems operate. She criticizes wetland mitigation strategies for failing to adequately address basic ecological principles of succession, habitat connectivity and distribution of wetlands within entire watersheds, and the effects of hydrogeological and climate changes on biological assemblages which are still poorly understood. Illustrating how monumental a task of creating an entirely new ecosystem is, she makes a salient analogy to issues faced reintroducing a single species to habitat, "Recent attempts to reintroduce a single species of rare plants to their historic habitats (Falk et al., *in press*) show how difficult it is to return even a single species to an ecosystem- the plant's environmental requirements may no longer be present; its pollinators may be absent; the small-scale disturbances required for recruitment may be lacking; and exotic species may invade the transplantation site and resist eradication efforts. Replacing an entire ecosystem multiplies the difficulties (pg. 34)."

Given this caveat, administrators are applying performance measures to evaluate whether mitigation efforts are successful. Guidelines written by Washington State Department of Ecology provide target roles for water, soil, and vegetation development in wetland mitigation plans. In order to feature tangible objectives, each performance standard should describe the specific mitigation goal in terms of qualitative indicators, quantitative attributes, specific actions accomplished, time-oriented benchmarks, and geographic location of monitored indicator (Washington State Department of Ecology, U.S. Army Corps of Engineers Seattle District, & U.S. Environmental Protection Agency Region 10, 2006).

In order for functional data and analysis obtained through mitigation research to be useful, land managers must be able to incorporate it into management decisions. The EPA Wetland Research Program (WRP) published *An Approach to Improving Decision Making in Wetland Restoration and Creation* in 1992 (Kentula, et al) in which they designed an approach that uses a series of systematic feedback loops to incorporate experience and lessons learned from previously constructed wetland mitigation sites and monitoring data to better inform future wetland management decisions. Figure 2.1 depicts WRP's model to continuously reintegrate new monitoring data and evaluations from wetland mitigation sites back into decision making processes (pg. 4). Additionally, WRP's approach uses two evaluation forms for monitoring and assessing

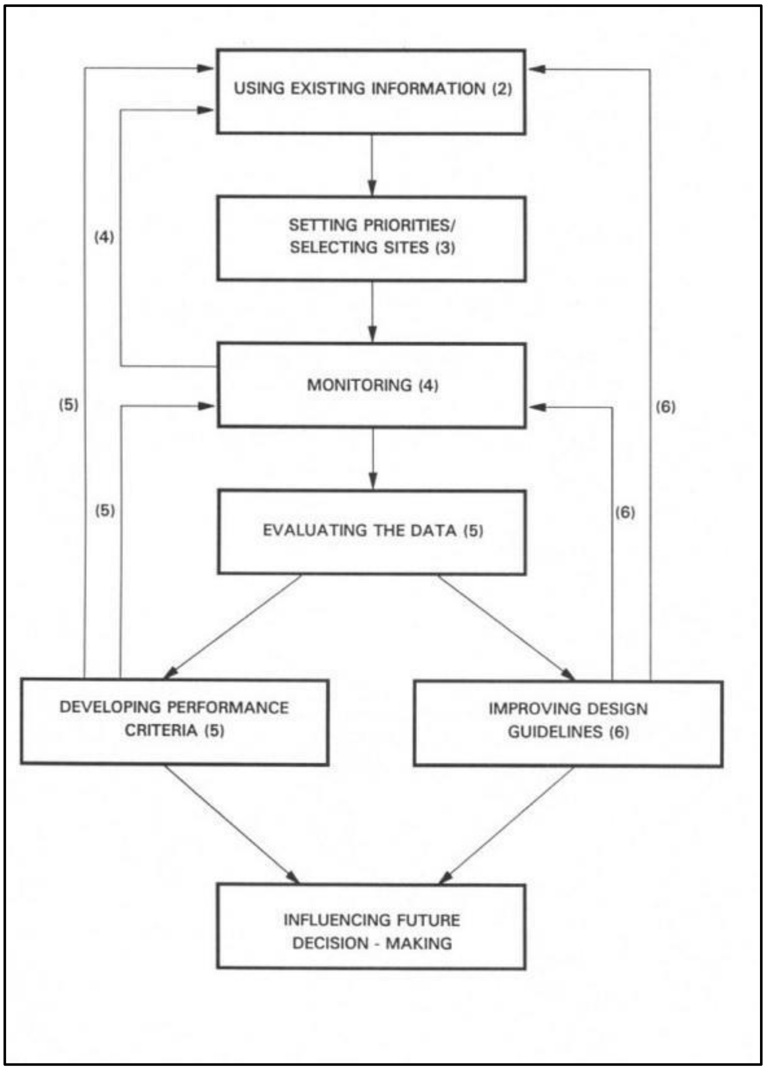

Figure 2.1 *Steps of the WRP Approach for using quantitative information to support decision making (Kentula, et al, 1992, pg 4).*

constructed wetlands. The first form is used to conduct an initial assessment of conditions immediately after wetland construction and the second form is intended for continuous monitoring of mitigation sites. Standard use of these, or other locally adopted evaluation tools, allows for consistent evaluation of wetland mitigation throughout time and makes it possible for policymakers and researchers alike to make broad comparisons across many different areas of concern.

2.4 Areal and Functional Loss

Many studies over the course of the last few decades support Zedler's suggestion that wetland creation is much more complex than Section 404 permitting requires. Ecosystem complexity, coupled with lax compliance monitoring for wetland mitigation permit requirements has resulted in a loss of wetland ecosystems in terms of both areal extent and function (Turner, et al., 2001). These mitigation programs have shown high rates of failure, falling short of no net loss goals (Robb, 2002, Brown & Veneman, 2001, Turner, et al., 2001).

Moreno-Mateos Power, Comín, & Yockteng completed a thorough examination of ecosystem recovery (2012). They compiled data from 124 articles assessing biological and biogeochemical recovery of restored and created wetlands of all types from around the globe. These wetlands were compared to nearby natural reference sites and captured data for 14 constructed sites as old as 100 years. Biological recovery in terms of species richness and abundance averaged 77% after 50-100 years, whereas the biogeochemical functions of nitrogen, carbon, and phosphorus cycling and storage reached only 74% and consistently showed a time lag behind biological recovery. Their study also found that inland depressional wetlands exhibited slower recovery trajectories than riverine or tidal wetlands. 50 years after construction these wetlands have not reached reference conditions, and the authors conclude that they may never achieve those biological or biogeochemical goals.

Turner, Redmond, & Zeller's meta-analysis examined eight studies of wetland mitigation projects. They calculated a mere 21 percent of compensatory wetlands delivered ecosystem services equivalent to those lost. This meta-analysis included studies from WA in 1994, 1998 and 2000 which revealed a range of permit percent compliance (21-53%) with stated mitigation goals in WA. Overall, permitting activity resulted in a major net loss of 80% wetland ecosystems in direct conflict with local, state, and federal no net loss mandates. These authors attribute a majority of failure to poor administration and recommend incorporating deadlines, ecological criteria, compliance monitoring, and mitigation programs implemented at watershed scales to improve ecological viability (2001).

In 2001, Brown and Veneman conducted a systematic review of compliance with stated mitigation goals for 319 permitted projects spanning 44 towns in the Commonwealth of Massachusetts. Over half of the mitigation sites did not comply with regulatory standards with 21.9% of failed mitigation projects never moving past the planning stage. Of completed projects, none of the plant community structures compared to natural, reference wetland sites, with lower biodiversity in the mitigation sites. Given the lack of proper hydrophitic and native vegetation, the mitigation sites also fell short in providing adequate wildlife habitat for amphibians, mammals, and birds. Similar to Robb's (2002) study of wetland mitigation in Indiana, which calculated a 71% failure rate for palustrine forested wetlands, Brown, et al. discovered that none of the forested wetland projects were successfully constructed. This is particularly concerning since just over one-fourth of wetland development permits in MA were on forested wetlands.

2.5 Soil Organic Carbon

Wetlands receive carbon from three sources: 1) dissolved inorganic carbon 2) organic carbon inputs from terrestrial sediment and particles and 3) from the carbon dioxide found in the atmosphere. As these compounds move through the system and are coupled with inorganic nutrients, wetland vegetation undergoes gross primary production. Biomass produced may then be consumed and respired by animals and microbes, outgassed into the atmosphere, or buried in the wetland's sediments (Figure 2.2) Carbon sequestration in wetlands occurs when this carbon is buried for long-term storage in the wetland sediments (Hopkinson, Cai, & Hu, 2012). Because of the unique hyrdric soils in wetlands, these ecosystems store much higher amounts of soil organic carbon compared to other terrestrial ecosystems (Table 2.2).

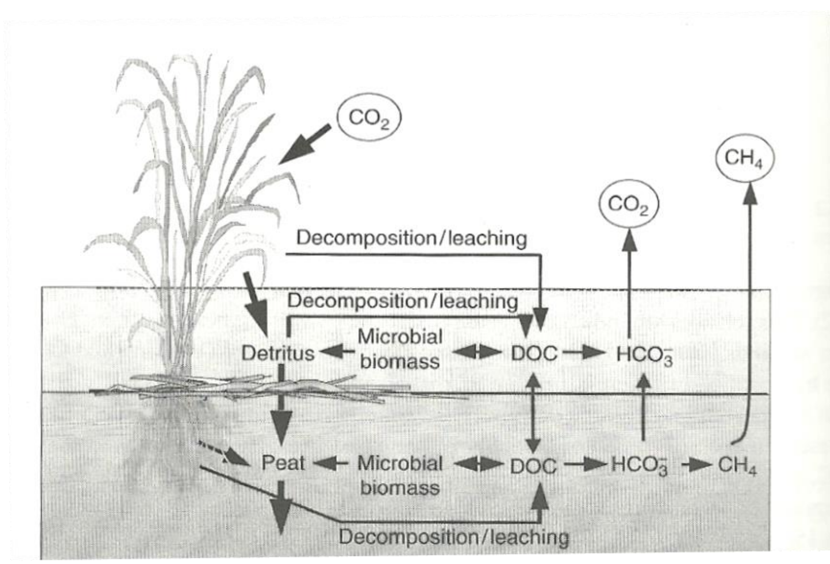

Figure 2.2 *Carbon cycling in wetlands. (Reddy, et al., 2008).*

Table 2.2 *Net Primary Productivity of different types of terrestrial ecosystems.*

Ecosystem	**Net Primary Productivity (g C/m^2 year)**
Desert	80
Boreal Forest	430
Tropical Forest	620-800
Temperate Forest	65
Wetland	**1,300**
Cultivated land	760
Tundra	130

Hopkinson and authors (2012) conducted a study of carbon sequestration in coastal wetland systems. They analyzed different measurements of this potential throughout the globe in order to quantify the rate at which this global sink is diminishing. In their meta-analysis they found that there is very large range of estimates, some as high as an order of magnitude in difference. This is due to the variation of carbon burial between different systems and the difficulty in estimating the areal extent of these wetlands. In order to quantify the rate of carbon sequestration of wetlands, they use the average of individual reports. They then measured the rate of carbon over a given area. According to their estimates (Table 2.3) mangroves bury 31.0-45.2 Tg C yr^{-1}, intertidal marshes bury 11.4-87.0 Tg C yr^{-1}, and seagrass beds bury 24.4-82.8 Tg C yr^{-1} in sediments (Hopkinson, et al., 2012). This study provides a clear picture of the potential of coastal systems to sequester carbon, but offers no insight into the capacity of freshwater systems to do so.

Table 2.3 *Estimates of global carbon burial in coastal vegetated ecosystems. Recreated from Hopkinson, et al., 2012.*

System	**Global area (km^2)**	**Carbon burial rate (g C m^{-2} yr^{-1})**	**Global carbon burial (Tg C yr^{-1})**
Mangroves	138,000–200,000	226 ±39	31.0–45.2
Intertidal marshes	200,000–400,000	57 ±6 – 218 ±24	11.4–87.0
Seagrass beds	177,000–600,000	138 ±38	24.4–82.8

Global carbon sequestration estimates for temperate freshwater wetland communities are even more difficult to obtain. There are several more subcategories of freshwater wetlands than coastal wetlands. These various wetland types have not been studied in as much detail as their coastal counterparts. A 2012 study by Bernal and Mitsch highlighted differences in carbon sequestration potential by community type in Ohio. Bernal, et al. found that, "the depressional wetland communities sequestered 317 ±93 g C m^{-2} yr^{-1}, more than the riverine communities that sequestered 140 ±16 g C m^{-2} yr^{-1} ... These differences in sequestration suggest the importance of addressing wetland types and communities in more detail when assessing the role of wetlands as carbon sequestering systems in global carbon budgets (2012)."

Bernal and authors' conclusion reemphasizes the need to conduct thorough investigation and comparison of carbon sequestration in different wetlands around the globe in order to understand their overall impact on the global carbon cycle. One such detailed analysis is that of carbon sequestration in the salt marsh Doñana Wetlands of southern Spain. This study incorporated estimations and direct measurements of primary production by vegetation, burial of organic carbon in sediments, and outgassing of carbon dioxide in order to measure the potential of the Doñana Wetlands as a carbon sink.

Through their detailed analysis, they were able to determine that although the water bodies were a net annual source of carbon dioxide, outgassing of C was still six times lower than the net primary production of the system which would indicate that the wetlands act as a carbon sink assuming no other losses (like lateral export) (Morris, Flecha, Figuerola, Costas, Navarro, Ruiz, Rodiguez, & Huertas, 2013).

While it is key to understand the dynamics of soil organic carbon storage and carbon sequestration of natural systems, it is also important to acknowledge the differences between constructed and restored systems compared to natural ones. In China researchers investigated the carbon sequestration potential of wetlands on a national scale in order to inform protection and restoration measurements aimed at preserving or increasing this capability of wetlands (Xiaonan, Xiaoke, Lu, and Zhiyun, 2008). They estimated carbon sequestration potential by using sedimentation rates and total organic carbon content of soil by a given distribution area. In their study they found a significant loss of soil carbon, 2,769.7 Gg C, due to the reclamation of lakes and swamps, but through wetland restoration they found carbon sequestration potential to be as high as 6.57 Gg C a^{-1} from 2006-2010 (Xiaonan, et al., 2008). Contrary to this study's optimism, in 2012 Moreno-Mateos, et al. also evaluated the structural and functional loss in restored wetlands and found that on average, after 50 to 100 years, restored wetlands recovered only 74% of their biogeochemical functioning in terms of nitrogen and carbon cycling potential.

It has been shown that wetlands of all types throughout the globe perform the crucial job of sequestering carbon dioxide from the atmosphere. However, we do not completely understand the differences in overall ability for specific wetland types, particularly in differentiating between the potential of separate freshwater wetland types. Further, it is evident that artificially constructed and restored wetlands do not perform as well as their natural counterparts. Given these gaps in knowledge and importance of carbon sequestration, it is essential to closely examine the soil dynamics of freshwater wetland mitigation projects.

3. Methodology

3.1 Site Selection

The City of Auburn's 2012 Wetland Mitigation Assessment Project (WMAP) evaluated ecological success and regulatory compliance of selected mitigation wetlands within city jurisdiction and Duwamish/Green Watershed Resources Inventory Area (WRIA) #9. To select sites for WMAP, city staff reviewed mitigation project files of all known projects in WRIA #9 that provided compensation for wetland impacts occurring in WRIA #9. For comparison, only projects involving the construction, restoration, or enhancement of freshwater, emergent depressional wetlands located on the Green River Valley Floor were considered (Appendix B). Projects lacking mitigation plan documents, planting plans, construction plans, performance standards, as-built reports and/or monitoring reports were excluded from WMAP, resulting in a final list of 24 mitigation sites (Soundview Consultants LLC, 2012).

Details reported from WMAP on these 26 wetlands were examined for this study, including data describing mitigation type, site age, site area, species richness, dominant species, vegetative coverage, and overall ecological success statistics. Wetland specialists visited each wetland site and recorded all vegetative species present and noted the dominant species, categorized by vegetation type: aquatic, herbaceous, shrub, or tree. Ecological success was determined by best professional judgement of the wetland specialist and labeled on a scale of 1, 3, or 5 with 1 being the lowest performing wetland and 5 being the best performing.

A subset of constructed and restored wetlands was chosen to collect supplemental data on soil bulk density, texture, and organic matter content. A list of 5 constructed and 9 restored sites of interest was provided to City staff with a request to access each site and collect soil samples. Some mitigation sites are located on privately owned land, and the ability to access the sites for continued sampling depended upon whether conservation easements which allow city employees access to the site for continued monitoring were still in place at the time of this study. The list was narrowed down to 4 constructed and 4 restored sites were chosen for which there are established conservation easements. However, only 3 of the 4 constructed wetland sites were accessible; the 4^{th} site had a chain-linked fence surrounding it with and a gate welded shut, preventing any access to the site.

3.2 Soil Samples

Depending on size, 5 to 10 soil cores were collected every 50 meters along a transect parallel to the wetland topographic contour (U.S. EPA, 2008). Some transects intersected areas of dense vegetation, which were cautiously navigated and/or avoided to prevent any adverse impact. The majority of soil organic matter accumulates in the root zone which occurs at 0-30 cm depth (Reddy, Clark, DeLaune, & Kongchum, 2013). To sample within this zone, a nickel-plated steel soil corer was used to extract intact vertical core samples to 30 cm depth and 2 cm in diameter. Coordinates were logged using a Garmin eTrex Vista. Soil compaction frequently occurred due to heavily saturated soil condition and the small diameter of soil corer. Each soil core was measured with a standard metric ruler to obtain core length (l). To calculate compaction (c) the following equation was used:

$$\textbf{(Equation 3.1) } c = \frac{30\ cm - l\ cm}{30\ cm} * 100\%$$

Soil cores were collected at each point until a sample with <50% compaction was obtained and recorded (overall soil compaction for all soil samples averaged 28%)(Ellert, Janzen, VandenBygaart, & Bremer, 2008). Each sample was then wrapped with low-density polyethylene and placed in a polyethylene zip sealed bag (Ellert, et al., 2008). Samples were stored in a cooler for transportation to the laboratory and then kept at 4ºC until ready for analysis, which occurred within a time frame of 6 to 14 weeks (Ellert, et al., 2008).

3.3 Soil Bulk Density

Soil cores were divided into 2 segments: 1)0-15 cm and 2)15-30 cm depths to perform laboratory analysis on soil samples (Badiou, McDougal, Pennock, & Clark, 2011). The first property analyzed was soil bulk density, or the ratio of solid, dry mass to total soil volume (Reddy, et al., 2013). Volume of soil cores was calculated as follows:

$$\textbf{(Equation 3.2) } volume = \pi(1\ cm^2)30\ cm$$

After oven-drying samples at 70ºC for 72 hours, each sample was weighed and soil bulk density was calculated using the following equation (Reddy, et al., 2013):

$$\textbf{(Equation 3.3) } bulk\ density = \frac{mass\ dry\ weight\ (g)}{volume\ (cm^3)}$$

3.4 Soil Organic Matter

Soil organic matter (SOM) was estimated by weight loss-on-ignition (LOI) methodology (Dean, 1974, Heiri, Lotter, & Lemcke, 2001, Skjemstad & Baldock, 2008, Wright, Wang, Reddy, 2008, Massello, 2013). Oven-dried samples were milled using a mortar and pestle, sifted through a 2 mm sieve, and then 5 +/- 1 g of each sieved sample (< 2 mm) were oven-dried overnight at 70°C (Skjemstad, et al., 2008). 15 mL crucibles with lids were combusted in a high temperature muffle furnace at 550°C for 5 hours to remove any contaminants (Massello, 2013). Each crucible was allowed to cool in a desiccator, and then weighed, with the final weight referred to as W_C. After drying soil and crucibles, samples were placed in a crucible and the dry weight (W_{60}) recorded. Crucibles containing soils were placed in the muffle furnace and heated at 550°C for 2 hours (Wright, et al., 2008). Samples were allowed to cool in the furnace, and then each was individually removed and placed on balance to record weight (W_{550}). % Soil Organic Matter (% OM) was calculated as follows, using the equation from Skjemstad, et al., 2008:

(Equation 3.4) $$\% OM = \frac{W_{550} - W_C}{W_{60} - W_C} x 100$$

3.5 Soil Organic Carbon

Soil organic carbon comprises only a portion of soil organic matter content. There is a wide range of estimated values to calculate this amount, however, as not to overestimate the amount of carbon contained in the samples, the most conservative ratio of 1:2 was used as recommended by Pribyl (2010). Therefore, soil organic carbon was estimated using the following calculation:

(Equation 3.5) $$\% Soil\ Organic\ Carbon = \frac{\% OM}{2}$$

3.6 Soil Particle Analysis

The Bouyoucos hydrometer method was used to determine soil texture (% silt, % sand, and % clay). The remaining oven-dried soil was ground with a mortar and pestle and sieved through a 2mm sieve. A 50 g sample (S_g) for each wetland and depth was prepared for analysis by soaking it in 100 mL of 1 M sodium hexametaphosphate dispersing solution, which was mixed vigorously with 250 mL deionized water. The samples were then placed in an electric mixer for five minutes. The resultant solution was poured into a 1,000 mL graduated cylinder which was then filled to 1,000 mL with deionized water. Measurements were also recorded for a blank cylinder with 100 mL 1M dispersant solution and 900 mL deionized water (R_B) (Bouyoucos, 1962, Massello, 2013).

Using a wooden plunger, the samples were further dispersed. A timer was started as soon as the plunger was removed, and hydrometer gently lowered into solution. The hydrometer has a scale read from the numbered mark which intersects the meniscus of the solution at specific time intervals. Hydrometer readings and temperature were measured at 40 seconds (R_{40S}) and 2 hours (R_{2H}). A blank reading (R_B) was taken to calibrate the hydrometer used in measurement. Soil particles were analyzed by calculating the following, where % sand is the portion of sand in the sample, % clay is the portion of clay, and % silt is the portion of silt (Bouyoucos, 1962, Massello, 2013):

(Equation 3.6) $$\%\ sand = 100 - (R_{40S} - R_B) * \left(\frac{100}{S_g}\right)$$

(Equation 3.7) $$\%\ clay = (R_{2H} - R_B) * \left(\frac{100}{S_g}\right)$$

(Equation 3.8) $\% \, silt = 100 - (\% \, sand + \% \, clay)$

3.7 Spatial Analysis

Geospatial analysis was completed using software and World Imagery base map provided by Esri ArcGIS10.3 (Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community, 2015). Adjacent Land Use for each mitigation site was determined by visual analysis from imagery and city parcel data obtained from Auburn's GIS database (City of Auburn, 2015). Each adjacent land use type along the border of a wetland mitigation project site location was measured. The type with the highest percentage was assigned as the primary adjacent land use.

3.8 Statistical Analyses

All statistical analyses was conducted using JMP Pro 11.2.0 statistical software. Both species richness and soil organic matter data were evaluated using the Shapiro-Wilk test for normality and were found to be non-normal, however, they met the assumptions to compare means using a Kruskal-Wallis test by ranks. The mean % OM of restored and constructed wetlands, mean % OM based on ecological score, species richness by project type and adjacent land use, and mean soil bulk density by project type were compared using this method. Simple linear regression was used to determine predictability of % OM and species richness by size and age, and % OM by soil texture. Cubic linear regressions were calculated to predict % OM by soil bulk density in restored and constructed wetlands.

4. Results

4.1 Physical Properties

Physical properties of the wetlands sampled for soil analysis, including information on project type, age, area, and adjacent land use are summarized in Table 4.1. Ages of wetlands ranged from 5-18 years and project site areas ranged from 14,374 to 1,698,840 sq. ft. with respective means of 12.8 years and 487,882 sq. ft. It is worth noting that the rage of ages and sizes of the restored wetlands varied considerably more than that of the constructed wetlands. Land uses adjacent to all mitigation sites were classified as vacant (8), industrial (8), road (5), or mitigation site (3). Also, industrial sites were found near some of the constructed wetlands, whereas none of the restored wetlands had industrial as an adjacent land use category. While these wetlands are not paired replicates due to their differences in physical properties, and therefore do not provide exact comparisons, however, they are representative of mitigation projects designed to construct or restore palustrine wetlands within the same municipality.

Table 4.1 *Physical properties of wetland mitigation sites examined in this study.*

Project #	Wetland	Project Type	Age (years)	Area (sq ft)	Ecological Score	Adjacent Land Use
92-0055A	A	Restoration	18	1,158,260	5	Vacant
92-0055B	B	Restoration	18	1,373,447	3	Vacant
97-0013	C	Restoration	6	429,066	5	Road
07-0001	D	Restoration	5	115,434	5	Mitigation Site
00-0038E	E	Construction	9	-	5	Road
04-0013	F	Construction	8	178,596	3	Industrial
97-0063B	G	Construction	13	200,376	3	Industrial

4.2 % Organic Matter

4.2.1 % Organic Matter by Mitigation Project Type

% OM values ranged from 1.4 to 49.2% in restored wetlands and 3.5 to 21.1% in constructed wetlands (Table 4.2). In all cases, samples at greater depth (15-30 cm) had lower means than those sampled at 0-15 cm (Figure 4.1). The mean of % OM in constructed wetlands (7.8 ±4.0) is nearly half the mean in restored wetlands (15.3 ±12.1) (Figure 4.2). Kruskal-Wallis test showed that there was a statistically significant difference in % OM between constructed and restored wetland mitigation sites ($\chi^2(1)=9.4$, $p=0.002$) with a mean rank score of 42.1 for constructed wetlands and 62.0 for restored wetlands.

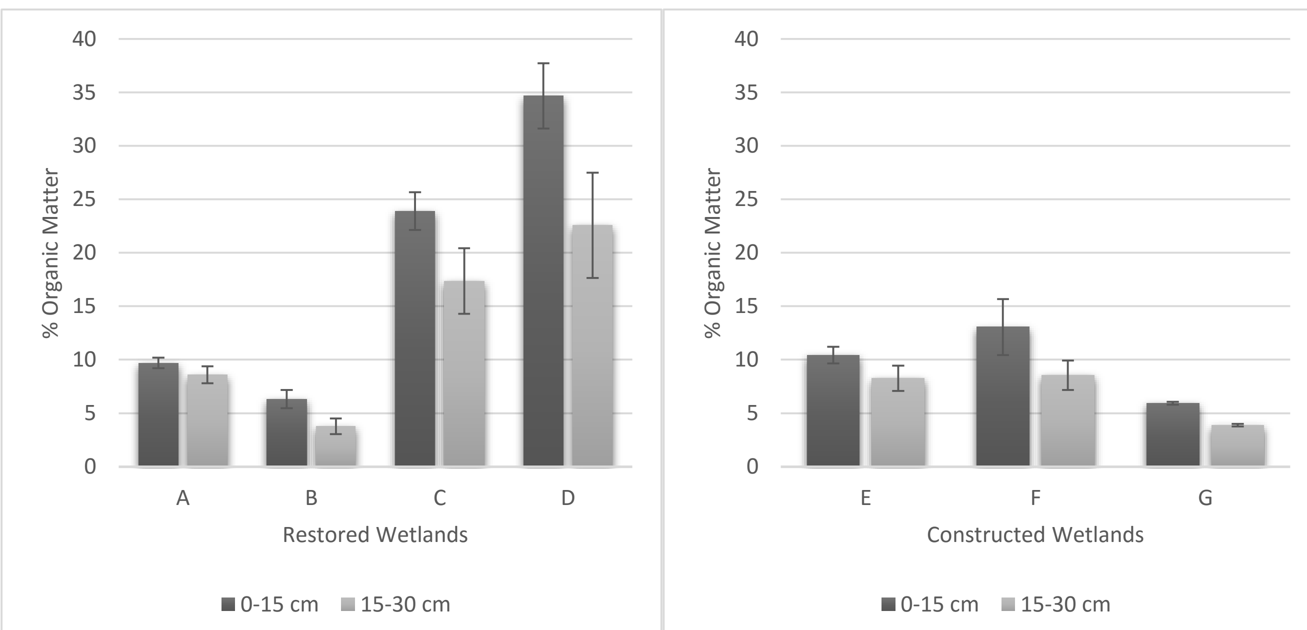

Figure 4.1 *Mean % Organic Matter of constructed and restored wetland mitigation sites with error bars constructed 1 Standard Error from the mean.*

Table 4.2 *% Organic Matter and % Organic Carbon in sampled wetland mitigation sites.*

Wetland	Project #	Project Type	*n* (soil cores)	Depth (cm)	$\bar{x} \pm \sigma$ (% OM)	Min (% OM)	Max (% OM)	% Organic Carbon
A	92-0055A	Restoration	10	0-15	9.7 ±1.6	8.0	13.3	4.9
				15-30	8.6 ±2.5	6.2	13.5	4.3
B	92-0055B	Restoration	10	0-15	6.3 ±2.7	3.5	11.0	3.2
				15-30	3.8 ±2.3	1.4	7.7	1.9
C	97-0013	Restoration	8	0-15	23.9 ±5.0	16.5	30.4	12.0
				15-30	17.4 ±8.7	7.9	31.2	8.7
D	07-0001	Restoration	9	0-15	34.7 ±9.2	20.2	46.7	17.4
				15-30	22.6 ±14.8	4.4	49.2	11.3
E	00-0038E	Construction	5	0-15	10.4 ±1.8	8.5	12.9	5.2
				15-30	8.3 ±2.7	3.7	10.4	4.2
F	04-0013	Construction	5	0-15	13.1 ±5.8	7.5	21.1	6.6
				15-30	8.5 ±3.1	5.8	13.4	4.3
G	97-0063B	Construction	8	0-15	5.9 ±0.41	5.5	6.5	3.0
				15-30	3.9 ±0.3	3.5	4.3	2.0

4.2.2 % Organic Matter by Age, Area, and Ecological Score

Separate simple linear regressions were calculated to predict % OM by age or area, and no significant regression equation was found. Similarly, no relationship was established between Ecological score of all 24 wetland mitigation projects and age or area. However, in the Kruskal-Wallis test found that those wetlands with a high ecological score of 5 did have statistically higher % OM means (15.3 ±12.1) than those with a moderate ecological score of 3 (7.8 ±3.9) with a mean rank score of 31.0 for wetlands with a score of 3 and 73.1 for wetlands with a score of 5 ($\chi^2(1)$=46.7, *p*=0.0001) (no wetlands with a low score of 1 were sampled) (Figure 4.2).

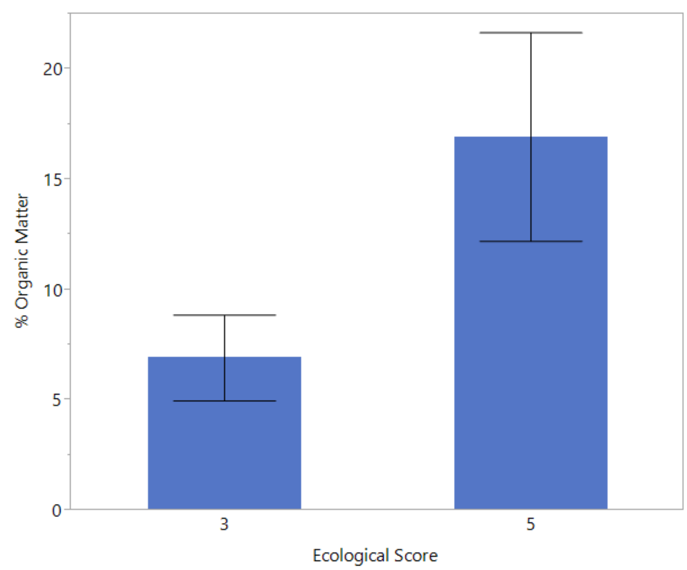

Figure 4.2 *Mean % Organic Matter for wetlands by ecological score shown with error bars constructed 1 Standard Error from the mean.*

4.3 Soil Bulk Density

Soil bulk density of restored wetlands ranged from 0.07 to 0.59 g/cm^3 with a mean of .30 ±.13. Values for soil bulk density of constructed wetlands ranged from 0.09 to 0.65 with a mean of .32 ±.13 (Table 4.3). A strong negative correlation between % OM and soil bulk density was observed (Figure 4.4). A cubic regression was calculated to predict % OM based on soil bulk density (g/cm^3). A significant regression equation was found with an R^2 of 0.584 (F=49.6, DF=3, p<0.0001) where x = soil bulk density:

(Equation 4.1) $\% \text{ OM} = 54.66 - 300.6*x + 644.7*x^2 - 475.48*x^3$

No significant difference was found when comparing the mean soil bulk density of constructed and restored wetlands. However, when separate cubic regressions for constructed and restored wetlands were calculated to predict % OM by soil bulk density, the regression line for restored wetlands predicted %OM far more accurately (R^2=0.727, F=61.99, DF=3, p<0.0001) (Figure 4.3, Equation 4.2) than for constructed wetlands (R^2=0.298, F=4.53, DF=3, p<0.0003) (Figure 4.4).

(Equation 4.2) $\% \text{ OM} = 59.87 - 302.4*x + 594.5*x^2 - 412.9*x^3$

Table 4.3 *Soil Bulk Density in g/cm^3 of sampled wetland mitigation sites.*

Wetland	Project #	Project Type	n (soil cores)	Depth (cm)	$\bar{x} \pm \sigma$ (g/cm^3)	Min (g/cm^3)	Max (g/cm^3)
A	92-0055A	Restoration	10	0-15	0.33 ±0.11	0.11	0.48
				15-30	0.39 ±0.12	0.13	0.52
B	92-0055B	Restoration	10	0-15	0.36 ±0.10	0.20	0.49
				15-30	0.43 ±0.09	0.32	0.59
C	97-0013	Restoration	8	0-15	0.23 ±0.07	0.12	0.32
				15-30	0.30 ±0.11	0.13	0.44
D	07-0001	Restoration	9	0-15	0.14 ±0.06	0.07	0.22
				15-30	0.20 ±0.11	0.07	0.37
E	00-0038E	Construction	5	0-15	0.28 ±0.10	0.11	0.35
				15-30	0.33 ±0.10	0.16	0.41
F	04-0013	Construction	5	0-15	0.21 ±0.12	0.09	0.40
				15-30	0.26 ±0.12	0.16	0.39
G	97-0063B	Construction	8	0-15	0.33 ±0.11	0.21	0.51
				15-30	0.42 ±0.12	0.27	0.65

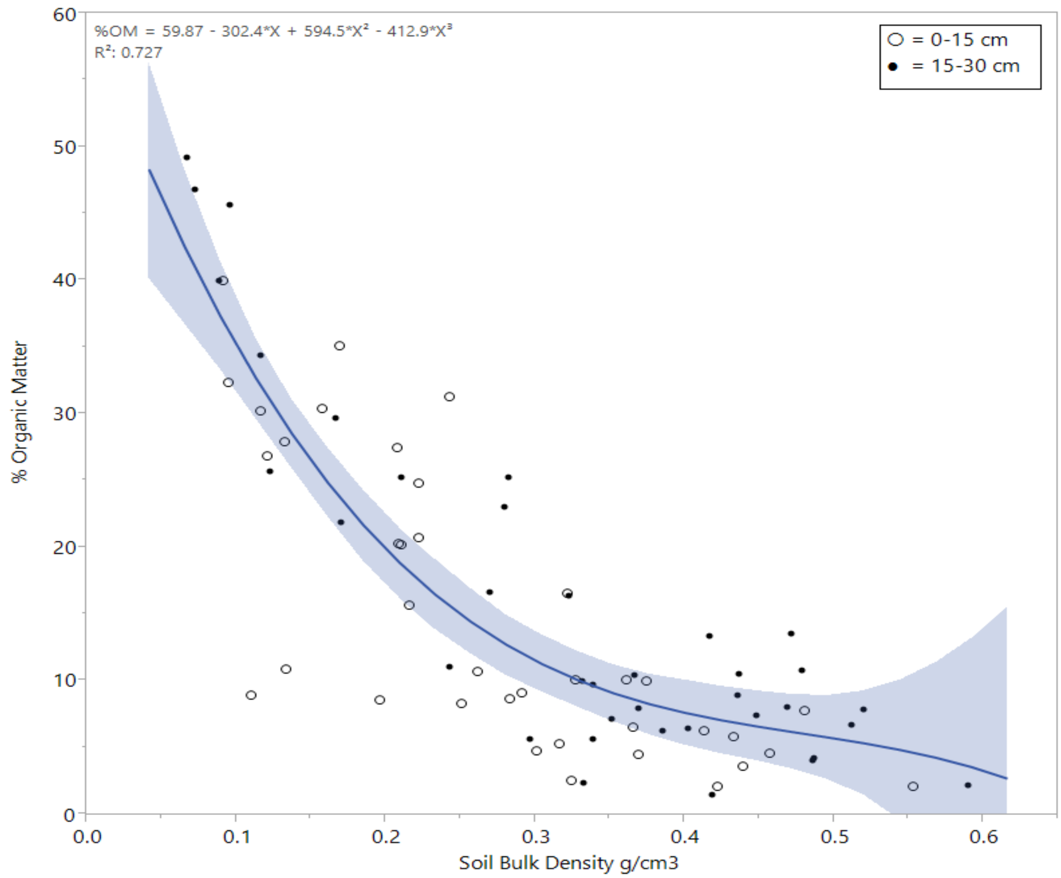

Figure 4.3 *Cubic regression line of fit for the prediction of % Organic Matter based on soil bulk density in g/cm^3 in restored wetland mitigation sites.*

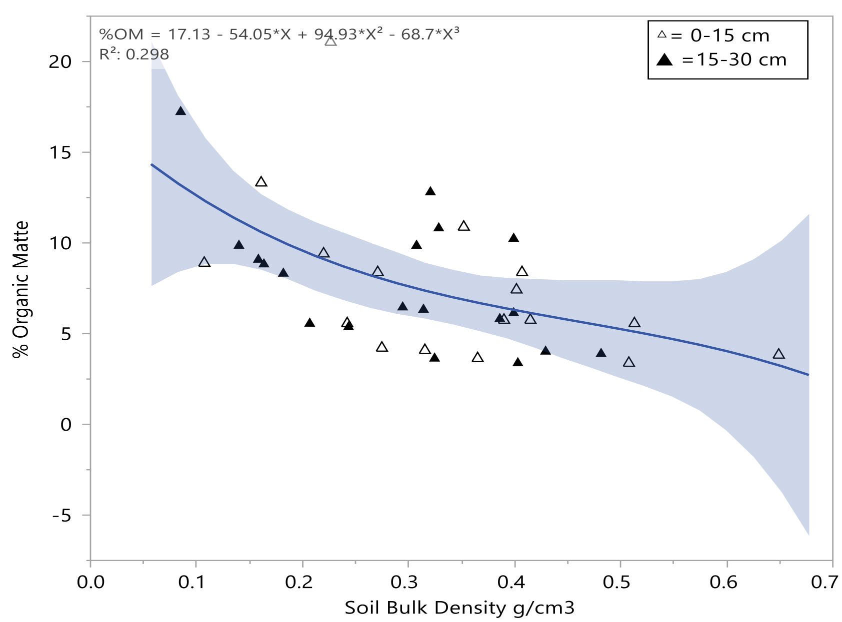

Figure 4.4 *Cubic regression line of fit for the prediction of % Organic Matter based on soil bulk density in g/cm^3 in constructed wetland mitigation sites.*

4.4 Soil Texture

Soil texture of restored wetlands varied widely and was estimated to range from a minimum 3 to a maximum of 82% sand ($\bar{x}$ =53, σ =29), 3 to 28% clay ($\bar{x}$ =11, σ =9), and 14 to 69% silt ($\bar{x}$ =36, σ =21). Soil texture of constructed wetlands was also highly variable and estimated to range from a minimum 48 to a maximum of 67% sand ($\bar{x}$ =56, σ =7), 6% to 15% clay ($\bar{x}$ =11, σ =4), and 25% to 46% silt ($\bar{x}$ =36, σ =21) (Figure 4.5). Simple linear regression was calculated, and failed to predict % OM by texture. % Sand, % clay, and % silt did not exhibit any correlation to % OM with respective R^2 values of 0.011 (Figure 4.6), 0.020 (Figure 4.7), and 0.006 (Figure 4.8).

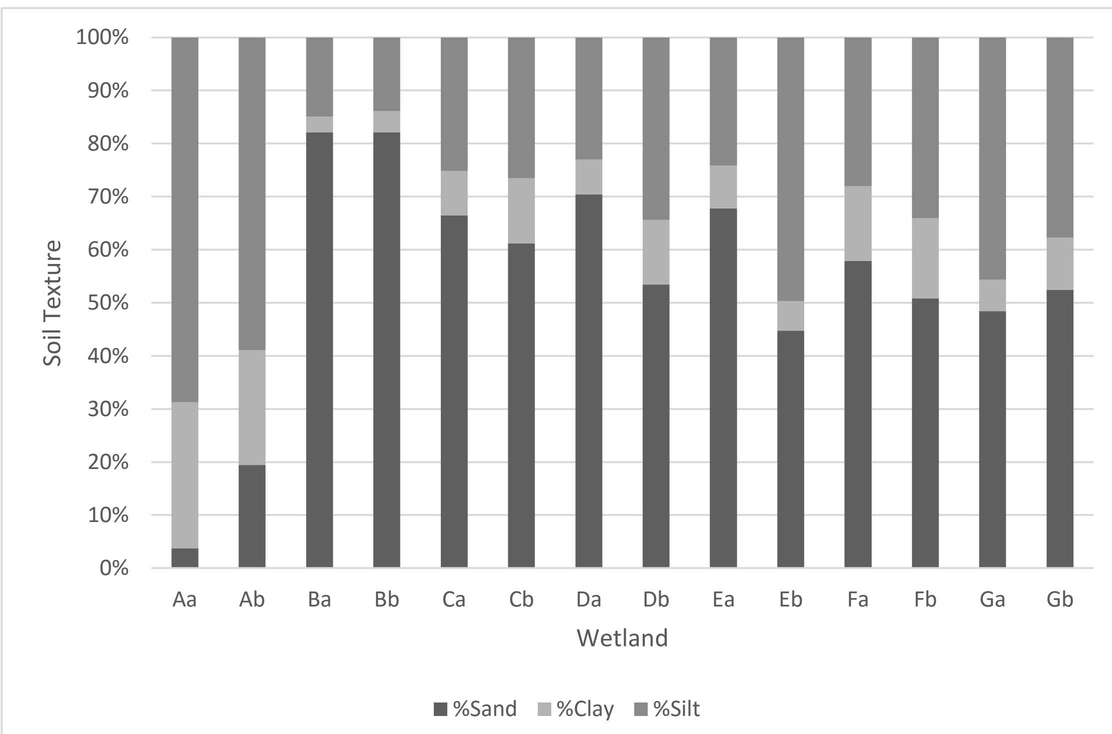

Figure 4.5 *Estimated soil texture in sampled wetland mitigation sites shown in % sand, % clay, and % silt. Wetlands A-D are restored and E-G are constructed;* a *indicates values at 0-15 cm depth and* b *for 15-30 cm.*

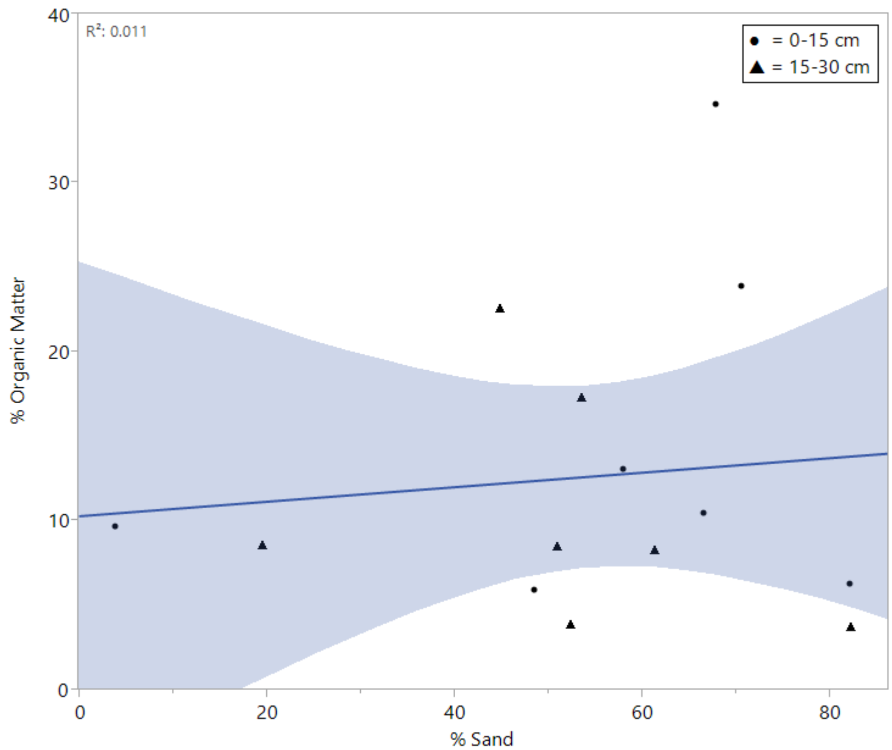

Figure 4.6 *Simple linear regression of % Organic Matter by % Sand.*

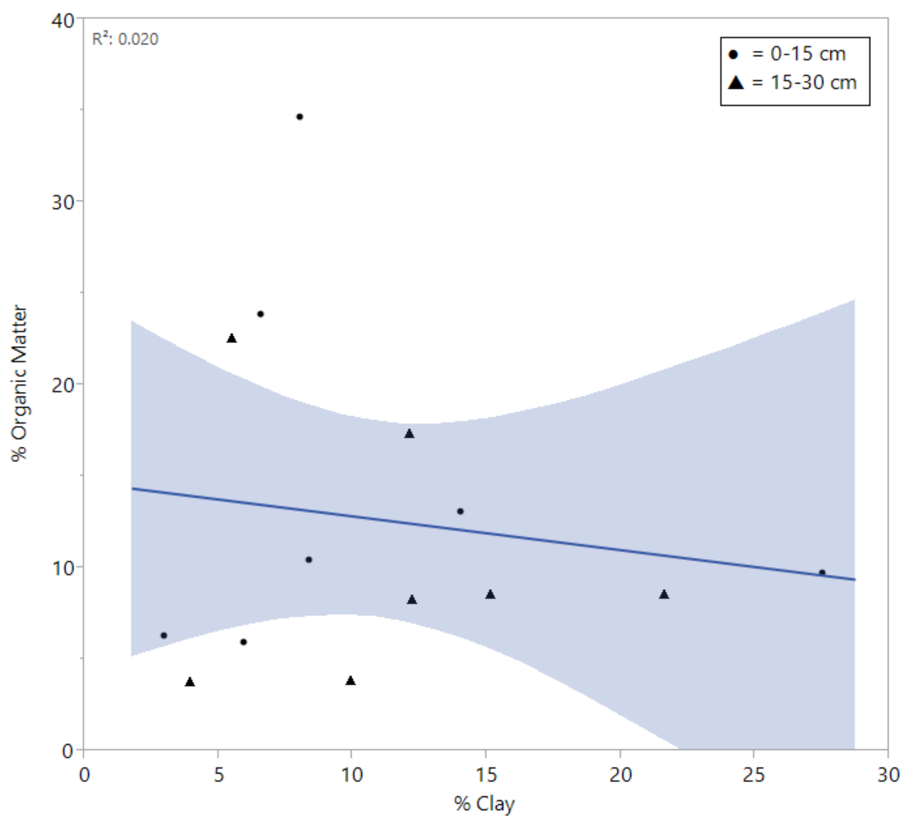

Figure 4.7 *Simple linear regression of % Organic Matter by % Clay.*

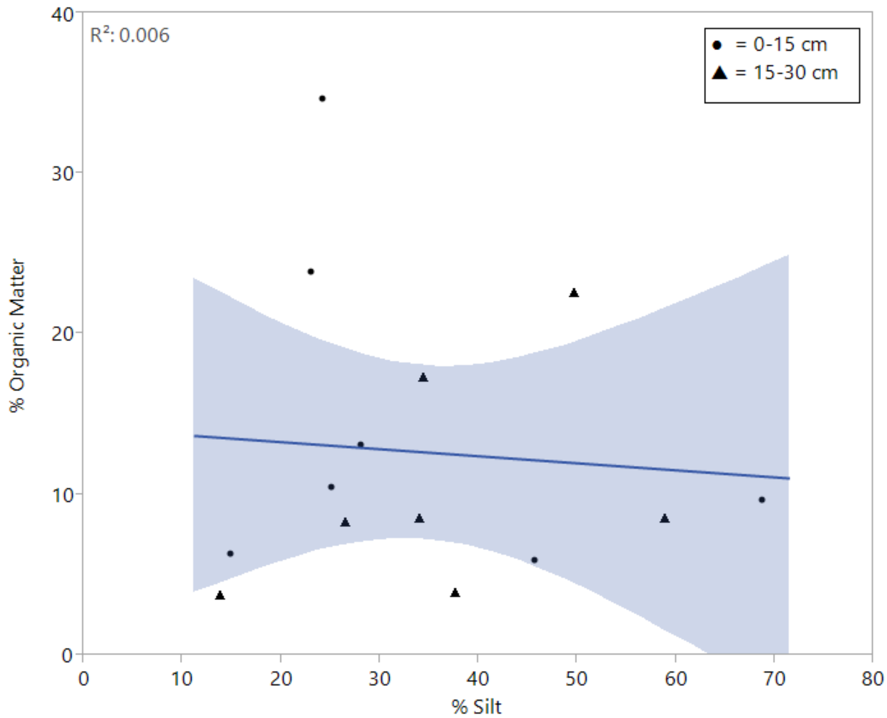

Figure 4.8 *Simple linear regression of % Organic Matter by % Silt.*

4.5 Species Richness

Species richness was evaluated for all 24 wetland mitigation sites in the original Auburn study. Total vegetative species richness of restored wetlands that were sampled ranged from 18-48 species for restored wetlands with a mean of 35.5 ±14.6, and 11-17 for constructed with a mean of 13.3 ±3.2. A summary categorized by type of vegetation with dominant species for each type is illustrated in Table 4.5; a graph and table of species richness for all 24 wetlands in the Auburn study is located in Appendix C. There is no significant difference in mean species richness of different mitigation project types. Additionally, species richness is not a good predictor of % OM for these wetlands.

Table 4.4 *Vegetative species richness and dominant species of sampled wetland mitigation sites.*

Wetland	Project Type	Aquatic	Dominant Species	Tree	Dominant Species	Shrub	Dominant Species	Herbaceous	Dominant Species
A	Restoration	13	*Typha latifolia, Nuphar polysepalum*	4	*Populus balsamifera*	13	*Salix sp.*	18	*Typha latifolia* and *Nuphar polysepalum*
B	Restoration	2	*Lemna minor, Phalaris arundinacea*	5	*Populus balsamifera*	19	*Populus balsamifera*	21	*Phalaris arundinacea* and *Ranunculus repens*
C	Restoration	3	*Polygonum persicaria*	2	*Salix sp., Thuja plicata, Picea sitchensis*	6	*Salix sp., Thuja plicata, Picea sitchensis*	7	*Phalaris arundinacea*
D	Restoration	12	*Glyceria occidentalis* and *Deschampsia cespitosa*	3	*Populus basamifera*	4	*Salix sp.*	10	*Glyceria occidentalis* and *Deschampsia cespitosa*
E	Construction	2	*Typha latifolia, Lemna minor*	2	*Salix sitchensis* and *Populus basamifera*	3	*Salix sitchensis*	5	*Typha latifolia* and *Lemna minor*
F	Construction	0		0		7	*Salix sp.*	10	*Phalaris arundinacea*
G	Construction	1	*Malus fusca*	3	*Salix sp.*	3	*Salix sp.*	4	*Carex obnupta*

Adjacent land use for mitigation projects include industrial, mitigation (wetlands adjacent to other mitigation projects), road, and vacant land use types. Species richness for all 24 wetlands ranged from 5 to 27 species present in wetlands with industrial adjacent land use with a mean of 11.6 ±7.2, 11 to 29 species for wetlands adjacent to mitigation sites with a mean of 22.3 ±9.9, 12 to 30 for wetlands adjacent to roads with a mean of 19±6.9, and 4 to 48 for vacant adjacent land use with a mean of 21.4±17.2 (Figure 4.9). These means were not statistically different.

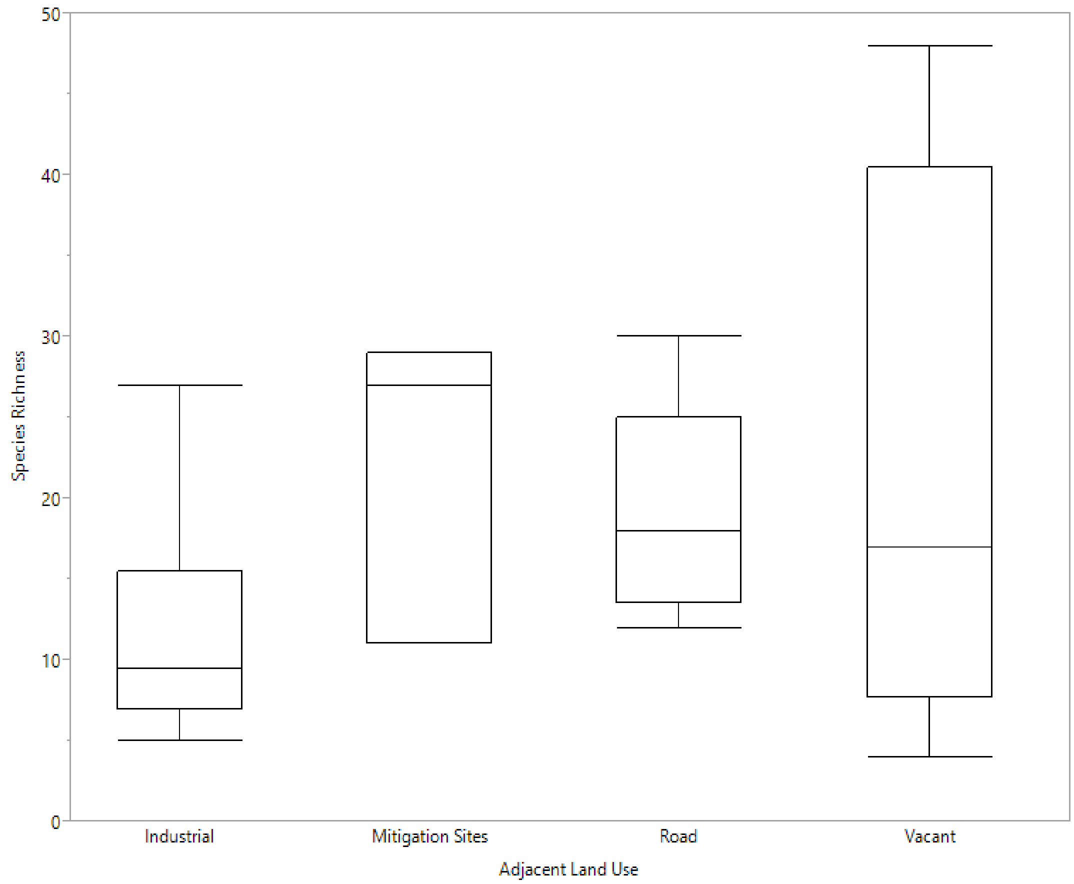

Figure 4.9 *Species Richness of compensatory wetland mitigation sites by adjacent land use.*

5. Discussion

Restored wetland mitigation project sites in Auburn, WA have higher soil organic matter content than constructed wetlands. This disparaity necessitates an examination of differences between the two project types, which could contribute to variations in ecosystem function with respect to soil carbon and development. To date, the published literature only draws comparisons between constructed wetlands to natural wetlands, restored wetlands to natural wetlands, or groups constructed and restored wetlands into one category. This strategy does not provide insight into functional differences between construction and restoration of wetlands. In constrast to the results presented here, a study in North Carolina compared natural, restored, and created wetland soil organic matter content, but the authors did not find a significant difference between constructed and restored wetlands, and consequently reported the remainder of their results with the two wetland types under a single category: constructed/restored (Bruland & Richardson, 2005).

5.1 Soil Organic Matter

As far as particle size distrubution and its effect on soil organic matter, the results do not fit with conventional wisdom of soil biogeochemistry. It is widely accepted that the capacity of soil to contain soil organic matter can be attributed to the relative compositon of and, silt and clay (Hassink, Whitmore, & Kubát, 1997). For example, clay particles increase surface area available in soil for organic carbon adsorption, and are positively associated with soil organic matter (Krull, Baldock, & Skjemstad, 2001). However, for the wetlands sampled in this study, there was no association between soil texture and soil organic matter content(Figure 4.6, 4.7, 4.8). These wetlands have been physically altered in the restoration or construction process, and some have been excavated to influence hydrologic regimes. Because the wetland soils have not developed naturally over time, but still undergo primary production activity and subsequent carbon burial in the soils, the relationship between soil texture and organic matter composition may not follow the theoretical relationship outlined above. In contrast, it could be an artifact of the creation process.

Differences in soil organic matter were not explained by age, which indicates that this function does not substantialy develop further with the timescales examined in this study. These findings are consistent with a previous study in Pennsylvania that compared constructed wetland mitigation projects from two age groups (<10 and >10 years old) to natural reference wetlands and found that the mean % organic matter of both wetland age groups did not vary and wetlands >10 years old contained just over 50% the amount of organic matter than their natural counterparts (Campbell, Cole, & Brooks, 2002). Moreno-Mateos, et al. established that biogeochemical functions of constructed and restored wetlands achieve 74% capacity of natural wetlands, even after as much as 100 years (2012). In terms of carbon storage, they contended that they only achieved 62% of natural wetlands after 30 years, supporting the supposition that these projects' wetland soils will not mature with age (Moreno-Mateos, et al., 2012). Furthermore, Hossler & Bouchard developed a model which projects a constructed wetland would need 300 years before it would be able to store carbon equivalent to the levels contained in a natural wetlands, and argue for a minimum mitigation ratio of 5.1:1 in order to reduce this loss of ecosystem function with the caution that this modeled trajectory has yet to be verified by observation (Hossler & Bouchard, 2010).

Soil organic matter values are much lower than other reported values for natural wetlands in the Puget Sound Lowlands in King County, Washington (The City of Auburn and all wetland mitigation project sites are located in southeast King County, Washington). Natural, palustrine wetlands were found to have a mean percent organic matter content of 45.5 ±34.2, which is much higher than the values measured in both constructed ($\bar{x}$ =7.8 ±4.0) and restored ($\bar{x}$ =15.3 ±12.1) wetland mitigation projects (Horner, et al., 2001). Previous studies have also exhibited this disparity (Shaffer & Ernst, 1999; Stolt, Genthner, Daniels, Groover, Nagle, & Haering, 2000; Campbell et al., 2002). For example, in Pennsylvania, Campbell et al. estimated that constructed wetlands store more than half the amount of organic matter ($\bar{x}$ =4.8) than natural, reference wetlands ($\bar{x}$ =11.5), noting that soil bulk density is at least twice as high in constructed wetlands (likely caused by compaction from heavy equipment during excavation), but do not make a connection between compacted soils and a low percentage of soil organic matter (2002).

Soil bulk density is another frequently used metric to explain soil organic matter content. Generally, high soil bulk density indicates low soil organic matter and low soil bulk density is related to high soil organic matter content (Ekwue, 1990; Aşkin & Özdemir, 2003). Addtionally, there is an inverse relationship between soil bulk density and porosity; less dense soil is more porous. These pores allow water to saturate the soil and generate the anearobic conditons characteristic of wetland hydric soils, which slow decompositon rates and are therefore able to retain soil organic matter for longer periods of time (Krull, et al., 2001; Reddy & DeLaune, 2008). This study’s findings of a strong negative correlation between soil bulk density and soil organic matter in wetland mitigation projects (Figure 4.5, 4.6, Equation 4.1, 4.2) coincide with this concept. (Figure 4.1). During the implementation of mitigation projects, heavy equipment is used to excavate, grade, and fill wetlands, and the weight of this equipment can compact soils, resulting in higher bulk densities (Shaffer et al., 1999; Stolt et al., 2000; Campbell et al., 2002). In fact, the two wetlands that contained soils with the highest organic matter content and lowest soil bulk densities, C & D, had mitigation plans that required the least amount of disturbance (Table 4.2, 4.3). Wetland C did not require grading actions and Wetland D was graded around existing mounds, whereas all other wetlands required excavations and all had significantly lower soil organic matter. This could suggest that the use of heavy equipment better accounts for differences in soil development than mitigation project type. Further, excavation and grading action removed native soils of the 3 constructed wetlands, F, G, & E, and were then covered in top soil which could account for the higher soil bulk density, and the lower soil OM content, found in all three of the constructed wetlands examined in this study.

When comparing soil organic matter content by ecological score, the highest ecological score had higher average soil organic matter with a difference of 7.5%. Although the ecological score assigned was subjective, left to the best professional judgement of the wetland scientists conducting the wetland rapid assessment, the value would seem to reflect this aspect of soil development. Although it would not be prudent to use this measure as a replacement to determining soil organic matter development, it is advantageous to have an alternative to estimating soil health, since obtaining accurate measurements of soil organic matter while in the monitoring phase of mitigation projects would be limited by time, and may be impractical.

5.2 Species Richness

Species richness of wetland mitigation projects in Auburn, WA were highly variable, but none of the parmameters measured in this study could account for these variations. There was no difference in species richness based on wetland mitigation project type. Although survivability of planted species is not guranteed, the quantity, type, and location of vegetation planted is planned during the intial project design, and should be evaluated during the monitoring period and adjusted, as needed.

While the planting regime can be controlled in developing a mitigation site, this does not mean that these sites achieve the same heights of biodiversity of natural wetlands. In King County, palustrine wetlands ranged from 35-109 vegetative species per wetland (Cooke & Azous, 2001). The minimum species richness of these wetlands (35) is higher than 92% of the wetland mitigation sites considered in the full 2012 Auburn Wetland Mitigation Assessment Project. This inequality is cause for concern over the ability of these mitigation projects to effectively replace in-kind the ecosystem function of natural wetlands, which appear to be a habitat for more biodiverse vegetative plant communities. Vegetative biodiversity impels the continued survival of native species and provides valuable habitat for fauna, complementing quality goals the Washington Natural Heritage Plan (Washington State Department of Ecology, 1988).

Although the means of species richness by adjacent land use were not statistically different, it is valuable to highlight that the maximum species richness for sites that were adjacent to industrial sites (27) was much lower than those adjacent to vacant parcels (48). In an examination of species richness in urbanizing areas of King County, WA, researchers determined that urbanization did cause a decrease over time, possibly due to increasing runoff events, increasing mean water level fluctuation, and changing hydrologic regimes, which all can inhibit the survival of species intolerant to these changes and may indicate that the adjacent land use and land use changes do have a direct impact on vegetative communities (Azous & Cook, 2001).

5.3 Conclusion

Wetland mitigation projects in Auburn, Washington differ in their soil development, which may be affecting their capacity to mitigate the loss of carbon storage ecosystem functionality. The methods of project construction of both constructed and restored wetlands may affect this function, as the use of heavy equipment to grade and fill wetlands and the replacement of native topsoil may be compacting soils, and thus limiting their ability to store soil organic carbon. Further, projects may add topsoil with a lower percent organic matter (<10%), which further diminishes carbon supplies. There were no patterns found indicating a difference in species richness in mitigation projects based on age, project type, or adjacent land use. It is possible to affect biodiversity by selecting appropriate species to introduce by planting, monitoring their survival at least 10-15 years, and replanting as necessary in order to provide adequate habitat and primary production to replace those functions lost due to 404(c) permitting activity.

5.3.1 Implications

The impending negative consequences of anthropogenic climate change forces us to examine any and all opportunities to mitigate these looming disasters. These strategies incorporate the utilization of systems, both natural and engineered, to capture and store greenhouses gases such as carbon dioxide. Wetlands are an essential system that delivers this indispensable ecosystem service. Although they are a source of methane emissions, they are a substantial carbon sink due to their ability to store carbon in their porous soils. Because wetland mitigation projects are not storing as much soil carbon as their natural counterparts, it is imperative to understand how this trade-off could be limiting earth's natural systems for counteracting rises in atmospheric carbon dioxide levels. Just as global action includes prevention of further deforestation, policymakers must also incorporate wetland conservation and mitigation plans into carbon sequestration planning scenarios (Badiou, et al., 2011).

This study demonstrated that constructed wetlands do not perform as well as restored wetlands in terms of soil organic carbon storage. For these wetland mitigation sties to be functional equivalents to natural systems and result in no net loss of ecosystem function, they must be able to provide the critical ecosystem function of soil carbon storage. As permitting activity endures, land managers should differentiate between constructed and restored wetlands and create mitigation plans that reflect their functional differences. While restored wetlands may seem more viable in terms of ecosystem function development, it is not possible to reverse the overwhelming loss of wetlands through restoration alone. A large number of constructed wetlands must also be established to achieve this goal, but should be done while acknowledging their limited ability to mimic natural wetland ecosystes.

Regulators need to address wetland mitigation projects' limited ability to function as their natural counterparts, and find ways to explicitly hold permitees responsible for these standards. Although the 2008 federal rule moves land managers away from using wetland mitigation ratios in favor of a simple policy of no net loss of ecosystem function, in some aspects, ratios can aide in determining a project's potential in achieving functional equivalencies such as carbon storage capacities. Hossler & Bouchard recommend using a conservative 5.1:1 minimum mitigation ratio to reflect a constructed wetlands ability to store carbon (2010).

Given the drastic inability of wetland mitigation projects to sequester carbon, these stark differences must be considered when using global climate models and terrestrial carbon storage projections. It is imperative that wetlands are classified by their status as constructed, restored, or natural and their respective carbon sequestration abilities factored in to these analyses at all levels. Mitigation projects are not functional equivalents to natural wetlands, and should not be treated as such. Further, every effort should be made to preserve natural wetland ecosystems and Section 404 permitting should be limit to absolutely essential development projects in order to preserve one of earth's most important landscapes in naturally balancing atmospheric greenhouse gas levels.

5.3.2 Recommendations for Further Research

Construction techniques may be hindering wetland mitgation projects' ability to store soil organic carbon. Remaining reliant on heavy equipment to create these landscapes could be detrimental to soil development and functionality. This may be difficult to avoid when attempting to manipulate hydrology, but limiting its use and developing a viable alternative to prevent soil compaction should be explored. Appropriate soil ammendment should be carefully considered for the next stage once topography formation is complete. Due to excavation, organic rich native top soils are removed, and must be replaced with a soil that would develop into a functional equivalent to wetland soils. Where possible, the feasibility of relocating top soil directly from the impacted wetland to the new mitigation site should be studied. Appropriate storage of the soil would be essential, as aeration would alter the physiochemical and biological soil properties (Zedler & Kercher, 2005). Alternatively, future studies could test the capacity of different organic ammendments to mature into hydric soils characteristic of naturally occuring wetlands.

This study established that in Auburn, WA there are differences in soil organic content between constructed and restored wetlands, which are frequently lumped into one category when researching wetland mitigation. Future study of mitigation sites should differentiate between these systems and report results in this manner. It would be useful to continue to compare and contrast a larger sample size of these systems to test whether these results hold true at a larger scale, and pair constructed and restored wetlands with similar physical parameters to one another. In order to understand the capacity of mitigation project sites to store carbon long term, sedimentation rates should be assessed. Further research should then estimate the carbon sequestration rate of different project types, and propose methods to incorporate findings into global climate models used to better understand carbon cycling and project the extent of future climate change.

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Appendix A

Wetland Function/Value Indicators used by Puget Sound Water Quality Wetland Preservation Program to evaluate whether a wetland should be preserved (Recreated from Washington State Department of Ecology, 1988)

Function/Value Indicators

I. Resident and Migratory Species Support

A. The site supports important fish and wildlife use such as nesting rookeries, nursery sites, migratory feeding routes, feeding areas, and spawning areas for resident and/or migratory animal and fish species.
B. The site contains a significant number of habitat features important for fish and/or wildlife support.

II. Species of Special Concern

A. The site is feeding, breeding, or wintering habitat for animal species on WA Dept. of Wildlife (WDFW) adopted or proposed lists of endangered, threatened, sensitive or monitor species.
B. The site is spawning or feeding habitat receiving special mention under the WA Dept. of Fisheries Hydraulic Project Approval WAC's Chapter 220-110 7/20/87 (salmon, herring, and surf smelt).
C. The site is habitat for plant species which are listed in the Department of Natural Resources (DNR), Natural Heritage Program list of *Endangered, Threatened, and Sensitive Vascular Plants of WA, 1987*.
D. The site is habitat for uncommon plant species listed by a local Native American tribe or academic ethnobotanist as important to native people for food, medicinal, or spiritual purposes.

III. Native Plant Communities

The site contains a high quality example of a native wetland listed in the Terrestrial and/or Aquatic Ecosystem elements of the current WA Natural Heritage Plan that is presently identified as such (documented in DNR records) or is determined to be of Heritage quality by DNR.

IV. Diversity

A. The site supports a high diversity of native plant and animal species.
B. The site contains high habitat and structural diversity.

V. Floodwater Detention

The site moderates high flows experienced downstream by intercepting, slowing, and storing storm water runoff.

VI. Sedimentation & Erosion Control

The site intercepts sediment-laden runoff and provides for settling of sediments, thereby reducing sediment deposition in downstream areas.

VII. Nutrient/Pollutant Entrapment & Assimilation

The site intercepts, stores, assimilates, or provides for the biological conversion of nutrients or other pollutants (such as coliform bacteria, oil & grease, etc.) in a highly efficient manner. (Note: emphasis on lower level, non-toxic pollutants… not waste dump areas.)

VIII. Groundwater & Surface Water Exchange

The site provides or contributes to base flow in streams that support sensitive downstream habitat areas. (Sensitive downstream habitat areas such as connecting waters for fish habitat, estuarine wetlands, or other habitat areas dependent upon base flow.)

IX. Recreation

A. The site is important for recreational opportunities that are appropriate in wetland settings and are consistent with the needs identified in the WA Wetlands Priority Plan and the *Statewide Comprehensive Outdoor Recreation Plan* (SCORP) or in local open space and/or park and recreation plans. (An appropriate opportunity is defined as that which is dependent on the setting, doesn't harm the wetland, or whose impacts, if any, can be mitigated through temporal r spatial distribution of the activity.)

X. Open Space & Aesthetics

A. The site contribute significant visual natural landscape characteristics or linkages as an open space in a surrounding urban area.

B. The site provides aesthetic amenity values and contributes aesthetic functions to the adjacent landscapes.

XI. Education and Research

A. The site offers a diverse environment and is readily accessible for instructional use by education facilities and the general public.

B. The site has significant archaeological or historic cultural value as identified per listing on the National Register of Historic Places.

C. The site provides an important wetland research opportunity.

Appendix B

Map of wetland mitigation project sites evaluated for WMAP.

LABEL	PROJECT ID
1	00-0038A
2	00-0038B
3**	0-0038E
4	01-0004
5	01-0027
6	02-0001
7	02-0018
8	02-0020
9	03-0018
10**	04-0013
11	04-0018
12	04-0037
13	04-0043
14	06-0032
15*	07-0001
16	90-0090
17*	92-0055B
18*	92-0055A
19	94-0021
20	95-0015
21*	97-0013
22	97-0063A
23**	97-0063B
24	99-0046B

* SAMPLED RESTORATION SITE

** ** SAMPLED CONSTRUCTION SITE**

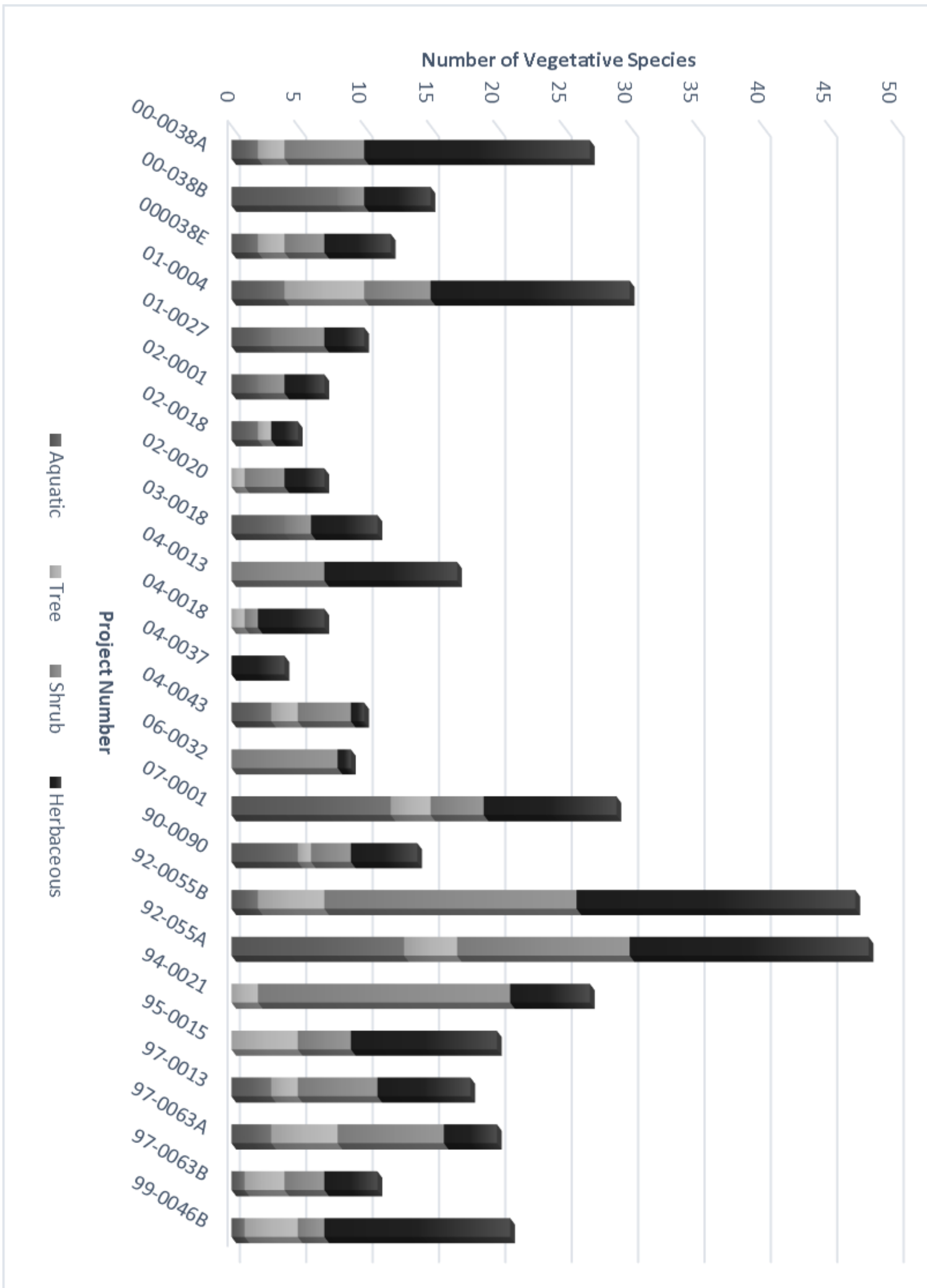
N

Christina Stalnaker
The Evergreen State College
2015

0 0.5 1 2 Kilometers

Appendix C

Species richness data for each compensatory wetland mitigation project showing number of aquatic, herbaceous, shrub and tree species by project number designation in the bar chart and including dominant species illustrated within table.



Project #	Project Type	Aquatic	Dominant Species	Tree	Dominant Species	Shrub	Dominant Species	Herbaceous	Dominant Species
00-0038A	Restoration	2	*Typha latifolia*	2	*Populus balsamifera*	6	*Salix sitchensis*	17	*Alopecurus pratense* and *Phalaris arundinacea*
00-038B	Both	8	*Nuphar polysepalum*	0		2	*Cornus sericea* and *Salix sp.*	5	*Juncus effusus*
00-0038E	Creation	2	*Typha latifolia, Lemna minor*	2	*Salix sitchensis* and *Populus basamifera*	3	*Salix sitchensis*	5	*Typha latifolia* and *Lemna minor*
01-0004	Restoration	4	*Nuphar polysepalum Typha, Elodea canadensis Lemnar*	6		5	*Salix sp.*	15	*Juncus effusus* and *Typha latifolia*
01-0027	Both	3	*Scirpis micracarpus, Typha latifolia*	0	*Populus balsamifera*	4	*Salix sp.*	3	*Typha latifolia*
02-0001	Restoration	2	*Typha latifolia, Phalaris arundinacea*	0		2	*Salix sp.*	3	*Phalaris arundinacea*
02-0018	Restoration	2	*Typha latifolia*	1	*Fraxinus latifolia*	0		2	*Typha latifolia*
02-0020	Creation	0	*Festuca sp*	1	*Salix lucida*	3	*Cornus sericea, Physiocarpus capitatus, Rosa pisac*	3	*Festuca rubra* and *Lotus corniculatus*
03-0018	Both	4	*Typha latifolia*	0		2	*Salix sp.*	5	*Scirpus microcarpus* and *Juncus effuses*
04-0013	Creation	0		0		7	*Salix sp.*	10	*Phalaris arundinacea*
04-0018	Both	0		1	*Populus balsamifera*	1	*Salix sp.*	5	*Phalaris arundinacea*
04-0037	Restoration	0		0	*Thuja plicata*	0		4	*Phalaris, Ranunculus repens, Alopecurus pretense*
04-0043	Both	3	*Typha latifolia*	2	*Populus balsamifera* and *Thuja plicata*	4	*Cornus sericea Physiocarpus capitatus Rosa Lonicer*	1	*Phalaris arundinacea*
06-0032	Both	0		0		8	*Salix sp*	1	*Phalaris arundinacea*
07-0001	Restoration	12	*Glyceria occidentalis* and *Deschampsia cespitosa*	3	*Populus basamifera*	4	*Salix sp.*	10	*Glyceria occidentalis* and *Deschampsia cespitosa*
90-0090	Restoration	5	*Iris pseudocorus, Salix sp on fringe*	1	*Populus balsamifera* and *Salix sp.*	3	*Salix sp.*	5	*Phalaris arundinacea*
92-0055B	Restoration	2	*Lemna minor, Phalaris arundinacea*	5	*Populus balsamifera*	19	*Populus balsamifera*	21	*Phalaris arundinacea* and *Ranunculus repens*
92-0055A	Restoration	13	*Typha latifolia, Nuphar polysepalum*	4	*Populus balsamifera*	13	*Salix sp.*	18	*Typha latifolia* and *Nuphar polysepalum*
94-0021	Restoration	0		2	*Populus balsamifera*	19	*Salix sp.*	6	*Phalaris arundinacea*
95-0015	Restoration	0		5	*Salix sp., Pseudotsuga menziesii*	4	*Rosa nutkana*	11	*Ranunculus repens*
97-0013	Restoration	3	*Polygonum persicaria*	2	*Salix sp., Thuja plicata, Picea sitchensis*	6	*Salix sp., Thuja plicata, Picea sitchensis*	7	*Phalaris arundinacea*
97-0063A	Restoration	3	*Polygonum amphibium*	5	*Populus balsamifera*	8	*Salix sp.*	4	*Carex obnupta*
97-0063B	Creation	1	*Malus fusca*	3	*Salix sp.*	3	*Salix sp.*	4	*Carex obnupta*
99-0046B	Creation	1	*Typha latifolia*	4	*Salix lucida*	2	*Picea sitchensis*	14	*Typha, Carex obnpta, Equisetum arvense*