5-13-2011

Comparison of geochemical and biological characteristics of natural and constructed wetlands in Western NY

Katrina Scheiner

Comparison of Geochemical and Biological Characteristics of Natural and Constructed Wetlands in Western NY

by:

Katrina R. Scheiner

Rochester Institute of Technology
College of Science: School of Biological and Medical Sciences
Program of Environmental Science

A thesis submitted in partial fulfillment
of the requirements for the degree
of Master of Science

Approved
May 13 2011
by:

_____
Anna Christina Tyler, Ph.D.
Chair of Committee

_____
Karl F. Korfmacher, Ph.D.

_____
Elizabeth N. Hane, Ph.D

_____
John M. Waud, Ph.D

TABLE OF CONTENTS

ABSTRACT

To protect wetlands and the critical ecosystem functions and services they provide, federal law mandates creation of new wetlands for wetlands that are destroyed. However, we don't yet fully understand if created wetlands are functionally equivalent to their natural counterparts. In this study of two natural (N1, N2) and two created (C1, C2) wetlands in western New York state, interdependent biological and geochemical characteristics were measured to assess equivalency of ecosystem function. With the exception of *Typha* sp, all wetlands contained unique vegetation zones. We sampled within these vegetation zones during the growing season of 2010. Overall cover of plant invasive species increased over the growing season, while native plants decreased, indicating a similar susceptibility to invasion. Invertebrate densities were very low and without within or between marsh trends. Differences in soil properties among wetlands did not fall out as a factor of wetland age, as N1 and C2 were similar and N2 and C1 were similar in terms of soil organic matter (OM) and phosphorus and there was a significant correlation between soil organic phosphorus and OM across all wetlands. *Typha* zones in the created wetlands tended to have low OM, but a significant relationship between vegetation type and OM was observed only at C1. When exposed to both ambient and high pulses of water column nitrate and phosphate, all wetlands showed an initial surge uptake of phosphate, followed by a more sustained flux. All wetlands were a phosphate sink, but only N1 was a consistent sink for both nitrate and phosphate. The significant differences that existed among the four wetlands suggest that the measured variables potentially have the greatest impact on overall ecosystem function. Overall, the created wetlands fell within the range of the natural wetlands for all tested parameters, suggesting similar structure and function in spite of the differences in age.

INTRODUCTION

Overview

The ecosystem functions and services associated with wetlands are well known. Of all their functions, the four that are most substantial are: supporting biodiversity, abating flood waters, improving water quality, and sequestering carbon (Turner *et al.* 2000, Whiting and Chanton 2001, Zedler and Kercher 2005). These functions lead to the innate value of wetlands and stem from the unique hydrological, biogeochemical and ecological structure of these ecosystems. Wetland values are attributes that have an economical benefit to human society, including fishery support and ecotourism (EPA 2001). Unfortunately, in spite of these critical values, wetlands have been and continue to be destroyed and degraded by human activities (Mitsch and Bouchard 1998, Mitsch and Wang 2000, Zedler and Kercher 2005). Two solutions to this problem are the restoration and creation of wetlands in order to prevent further loss of net wetland acreage (Zedler 2004). Given the significant impact that wetlands have on both the environment and human society, it is critical to know if created and restored wetlands are able to provide functions and values equivalent to their natural counterparts.

Wetland Degradation and Conservation

Until recently, wetlands were not under government protection and have suffered extensive losses. Between the 1780s and 1980s in the lower 48 United States, over 25 ha of wetlands were lost every hour (Dahl 1990). Thus, the past 200 years have seen a loss of more than 50% of the wetlands in the United States (Dahl 1990). The Laurentian Great Lakes in particular have seen drastic losses throughout this same time period, with an overall loss of about 66% and Lake Erie's western basin having lost an estimated 95% of its wetlands (Mitsch 1998).

Nontidal coastal wetlands are particularly vulnerable as they attract human development, but are not protected by inundating tide waters periodically throughout the day. This leaves them open to both drying during times of low lake levels and increased human encroachment (Mitsch 1998).

In 1977 an amendment to the Clean Water Act was passed, and wetlands came under federal protection (Clean Water Act 1977). The amendment introduced the practice of "zero net loss", monitored by the Army Corps of Engineers. With no net loss, compromised wetlands are restored or new wetlands are constructed to mitigate for wetlands destroyed by new human activity, resulting in, theoretically, zero net loss of wetland acreage or overall function. It is, however, unclear whether created wetlands fulfill native wetland functions. Given the importance of these ecosystems, it is critical to know how the important functions of wetlands, including nutrient uptake capacity, habitat quality, and biodiversity, vary between natural and constructed wetlands.

Vegetation and Biotic Factors

Wetlands are areas of high productivity and high diversity. The excellent habitat available and high levels of primary productivity in turn attract many animals and birds (Zedler and Kercher 2005). Some of these species are wholly dependent on the existence of wetlands while others, both aquatic and terrestrial, rely on wetlands for a portion of their life cycle. Biodiversity promotes stable ecosystems (Hooper *et al.* 2005, Smith 1995, Tilman 1996), and is thus critical for ecosystem health. Low-diversity systems, such as monocultures, may be at risk for total ecosystem collapse (Hooper *et al.* 2005) because they are not variable enough to sustain predation, invasive species, disease, or drought events. Such a collapse directly impacts plants

and animals in the immediate vicinity, and indirectly impacts any organisms that rely on that system during any point in their life cycle.

The base of ecosystem diversity is provided by its primary producers, vegetation. The success or failure of created wetland project is often determined simply by the presence of hydrophytic vegetation (Atkinson and Cairns 2001), but this does not necessarily indicate that the wetland provides the same functions and to the same degree as its naturally occurring counterpart. Numerous components interact to determine vegetation community structure, which in turn may determine the functionality, and thus "success" of a created wetland.

Due to a number of varying factors, vegetation communities may differ between natural and restored wetlands. Balcombe *et al* (2005) found that there were noticeable differences in species composition between natural and created wetlands, with a greater number of invasive species in the created wetlands. A greater diversity of plant species was found in the created wetlands, but there were also more invasive species present. Invasive species are often aggressive colonizers, and quickly spread in disturbed areas (Stevens and Hoag 2011), making created wetlands more susceptible to invasion. As natural wetlands are not subjected to this type of disturbance, they are considered to be les vulnerable to same colonization of invasive species. Seabloom (2003) found that while the invasive species distribution did not differ significantly between created and natural wetlands, overall vegetation cover was lower in the restored wetlands. Species richness was also lower in the restored wetlands. As the only factor addressed in this study was the vegetation composition, his final conclusion with regards to the difference in species richness and cover is limited by natural dispersal of the plants. Many other factors contribute to the suitability of wetlands for plant species, and could greatly impact the

3

community composition, so this conclusion may in fact only be a partial reason for the differences in the plant communities.

Benthic macroinvertebrates play a role in soil structure, organic and nutrient content of the soil, and thus vegetation composition as well. Macroinvertebrates play a critical role in nutrient cycling as they consume organic matter and other invertebrates (Stewart 2008). They can also impact plant litter decomposition rates (Atkinson and Cairns 2001) and nutrient cycling pathways and rates (Brinson 1981). They are food for larger vertebrates and can thus impact food webs and utilization of the wetlands by other species (Stanczak 2004). Invertebrates can be used as indicators of water quality as they are limited by factors that can indicate poor water, such as dissolved oxygen, nutrient concentrations, salinity, etc (Spieles and Mitsch 2000). Invertebrate community structure may differ depending upon the vegetation community structure (Atkinson and Cairns 2001), but the vegetation present can also be impacted by the invertebrates. As mentioned in Verhoeven (1996), lower nutrient concentrations can encourage a more diverse vegetation community. If the invertebrates present rapidly break down organic matter, greater amounts of inorganic and biologically available nutrients will be present, and thus the plant species diversity may subsequently be altered. It is evident that many components and players in wetland ecosystems are interconnected and impact one another synergistically. All of these components add together to provide the wetland's final functionality. Due to all of these inter-connected factors, the invertebrate community structure in the study wetlands can be used to assess water and ecological quality, and thus the potential functionality of the wetlands (Metcalfe 1989).

Hydrogeomorphology and Soil Structure

Of the potential determining factors, wetland hydrogeomorphology is typically the primary determining factor in vegetation community composition (Lougheed *et al* 2001, Bailey *et al* 2007). Organic content of the soil is another determining factor, though it is one that can vary with time and is subject to external inputs. Vegetation growth is in part limited by sediment quality (Lougheed *et al*. 2001), which is then in turn partially dependent on soil organic and nutrient content, density, pH, and soil particle structure. Of these, soil organic carbon is one of the most consequential attributes in determining soil quality (Shukla *et al*. 2006). Soil organic content is a result of partially and non-decayed plant matter blended with the mineral soil components. The source of the plant matter may come from the wetland vegetation that has died, or may be the result of flood and runoff events that carry in plant matter from outside sources. Local hydrology therefore can influence the amount and type of organic matter present in wetland soils. The decomposition of organic matter in the soil results in inorganic forms of nutrients that plants can utilize.

Bruland and Richardson (2006) addressed soil organic matter (SOM) in natural, created, and restored wetlands. They found that created and restored wetlands had significantly lower SOM than the natural wetlands in their study. They cite time as being a critical factor in the development of SOM, as well as previous land use. The method for actually creating wetlands may also play a significant role in organic matter, as excavation techniques can result in the removal of the organic matter –rich topsoil (Bruland and Richardson 2006).

Nitrogen and phosphorus are two of the primary limiting nutrients (Verhoeven 1996) for plants, and thus often determine growth rate and biomass, as well as diversity of species. High levels of nutrients result in rapid growth of aquatic vegetation, but may also actually result in

lower plant diversity. In the presence of excess nutrients, plants do not need to compete as strongly for these nutrients and instead are limited only by their own growth rates. High nutrient availability may also increase the susceptibility of the wetland to invasion. Lower concentrations of either or both of the nutrients limit the community to plants that have adapted traits to compensate for lower nutrient levels. Species often have varying techniques for fixing required nutrients, utilizing different forms of the chemicals. These unique traits allow for a more diverse community (Verhoeven 1996).

Run-off water, particularly in agricultural and residential regions, often contains high quantities of nitrogen and phosphorus. An issue of immediate concern in the Great Lakes region is nutrient runoff from agricultural fields and other non point source pollutants. These can greatly impact lake ecosystems, potentially leading to eutrophication and anoxic "dead zones".

The amount of nutrients in wetland soil and the soil's ability to process these nutrients has significant implications for areas farther down stream. Anaerobic bacteria can remove nitrogen from the water by way of denitrification. The process of denitrification converts nitrate, (NO_3^-) to a gaseous form of nitrogen and releases it to the atmosphere (Blackwell *et al*. 2002). Denitrification frequently occurs in wetlands as it requires anaerobic conditions, and the aquatic nature of wetlands leads to frequent periods of anaerobic conditions. Phosphorus adsorption also frequently occurs in wetlands as phosphorus has a tendency to adhere to soil particles (Smil 2000).

The nutrient sorption capacity of wetlands in regions where agricultural fields and nutrient-rich runoff abound is of even greater significance as higher nutrient inputs can result in greater concentrations being washed farther downstream. Thus, it is critical to know how other

important functions of wetlands, including nutrient uptake capacity, habitat quality, and biodiversity, vary between natural and constructed wetlands.

Just as differences in the soil quality and structure impact the vegetation community structure, so too does the vegetation community itself impact the soil structure. As previously mentioned, vegetation contributes organic matter to the soils. Submerged and emergent vegetation slows the rate of water flow and encourages sedimentation (Anderson 2006). Particulates, both mineral and organic, from outside sources or from elsewhere in the wetland then settle and contribute to the soil structure, and in turn contribute to potential nutrient content. Alternately, vegetation that is too dense may actually impede sedimentation as it prevents sediment-laden water from reaching areas containing emergent vegetation (Anderson 2006). Reed canary grass and narrowleaf cattail can both form dense mats of roots, stems, and leaves. If these areas become too densely packed, sediment is deposited where the plants are growing but does not continue downstream.

The geologic structure of wetlands allows them to assist in flood water abatement. Wetlands often form on low, relatively level land as such characteristics allow for water to remain still or with a slow flow. These areas of land are able to store large amounts of water (Hey and Phillipi 1995), preventing both ecologic and economic damage downstream. The physical slowing of water due to shallow, level ground surface and emergent vegetation allows soil particles to settle out. As phosphorous bonds to soil particles, the sedimentation process in wetlands also removes excess phosphorous from the water (Smil 2000). In particular, non-tidal coastal wetlands have a significant impact on the water bodies they border, promoting healthy aquatic ecosystems (Mitsch and Bouchard 1998).

Wetlands contain a great deal of carbon in varying forms in their soils and in their vegetation. Wetlands are capable of sequestering carbon, but they also release it in the form of CO_2 when plant matter decays. The rate and amount of CO_2 depends upon various environmental conditions, such as temperature and amount and duration of inundation. Wetlands are capable of being large carbon sinks since decomposition rates are typically low, due to anaerobic conditions in their soils (Mitra 2005). The ability to sequester or release carbon is particularly pertinent with the current concerns about green house gases and global climate change. Zedler and Kercher (2005) suggest that proper management and conservation of natural wetlands may assist in further sequestration of carbon, while the capacity of created wetlands to do the same is still unknown.

Like all ecosystems, the biotic and abiotic components of wetlands are interconnected, with feedbacks among the components leading to the alteration of functions to varying degrees. Differing plant types and density of vegetation influence the amount of accumulated litter and subsequently organic content of the soil (Atkinson and Cairns 2001). Decomposition rates in turn influence the amount of available nutrients in the soil and water, which may subsequently influence the growth rate, success, and types of plants present. These factors also influence the invertebrate assemblage, which has additional feedbacks to decomposition rates and soil organic content (Atkinson and Cairns 2001). All of these processes and characteristics influence other wetland functions, such as providing food web support for higher trophic levels, wildlife habitat, hydrological modification, or enhancing water quality (Atkinson and Cairns 2001).

Interconnectivity and Significance

Given the interconnections among the hydrology, biogeochemistry and biology within a wetland, it is important to understand how these factors, individually and as a whole, vary among created wetlands and in comparison to natural wetlands. Despite the importance of wetlands, we still have only a rudimentary understanding of the ability of created wetlands to replace natural wetland functions. While several analyses have been conducted to determine areas that may be most suitable for wetland restoration and techniques with which to restore the wetlands (Mitsch and Wilson 1996, Wilcox and Whillans 1999, Mitsch and Wang 2000, Gutrich and Hitzhusen 2004) few have assessed the functional equivalence of wetlands created on such sites. Created wetlands vary in age and are much "younger" than natural wetlands. It is likely that these young ecosystems are not functionally equivalent to mature, reference wetlands, but that these differences diminish over time (Atkinson and Cairns 2001, Campbell *et al* 2002). Based upon slow nutrient cycling rates, Atkinson and Cairns (2001) determined, however, that the 20 year-old created wetlands in their study still qualified as immature and developing ecosystems, differing greatly from both the two year old created wetlands also in the study and literature values for natural wetlands with similar hydrologic regimes. Such findings indicate that while created wetlands may be equivalent in structure to natural wetlands, it may take much longer for them to achieve the same degree of functionality.

Combined with the interconnected nature of wetland ecosystems, is it paramount to consider the relationship of different ecological and geochemical factors in determining whether or not natural and created wetlands are functionally equivalent.

Objectives & Hypotheses

The overall objective of this study was to **assess the functional equivalence of natural and created wetland**s. The approach was aimed at characterization of both physio-chemical and biological characteristics and is unique with respect to the breadth of variables measured. Due to length of time ecosystems can take to recover from disturbance and develop into a mature ecosystem, we anticipate that created wetlands will exhibit characteristics of younger ecosystems. Specifically, we hypothesize that:

1. the vegetation community structure will be less diverse in the younger ecosystems, with greater representation by invasive species;

2. the created wetlands will have a lower soil organic matter content and lower phosphorus concentrations because insufficient time has passed for accumulation of the substantial organic matter and nutrient reserves characteristic of mature wetlands;

3. the soils in the younger wetlands will release significantly more nutrients into overlying water due to their history as agricultural lands, though the sorption capacity of soils in nutrient loading conditions will be significantly greater in the younger wetlands, because the soil is not yet saturated with respect to nutrients and the organic matter will help to fuel microbial uptake of nutrients from the overlying water; and

4. the macroinvertebrate communities will be less diverse in the created wetlands because insufficient time has passed for colonization and the habitat heterogeneity is lower.

MATERIALS AND METHODS

Field Surveys

Site Description

Historically, the site of the Rochester Institute of Technology campus consisted of numerous wetlands, as evidenced by the presence of hydric soils throughout the lower elevations. Red Creek, which flows through campus prior to entering the Erie Canal and subsequently the Genesee River, flows through the campus (Figure 1). In the early-1900s, the area was drained for agriculture, with apple orchards on the higher elevations and row crops on the lower elevations. A few stands of old growth wood lots remained. In 1964, the campus was established (History of RIT 2010) and much of the prior farmland was converted either to campus buildings or reverted to wetlands. The southern part of the campus remained in row crops. The existing natural wetlands, and ones that developed after the property became the campus, are now primarily wooded, shrub/scrub, or emergent vegetation. In 2002 and 2007, two new mitigation wetlands were constructed on the campus (C2 and C1, respectively) to compensate for wetlands losses elsewhere on the campus. These wetlands were targeted as emergent marsh and wet meadows.

C1 and C2 were constructed adjacent to one another and within the same wetland complex. The soil is primarily Niagara silt loam, a shallow sloping and somewhat poorly drained soil (Soil Survey Staff 2010). The sites were chosen due to their proximity to existing water ways and wetlands, for their ability to diversify the existing wetlands, and to slow and filter runoff water from nearby agricultural fields (Terrestrial Environmental Specialists Inc 2002). Both wetlands were constructed by removing and saving the organic topsoil, excavating to the

planned depths, and then returning the top soil. Prior to replacement, the topsoil was amended

with shredded woody matter to increase the organic content to 7%-8% (D. Harris, personal

communication, April 6 2010). A variety of depths were created in each wetland to provide

heterogeneity of habitat types and appropriate vegetation was planted in each location. The

excavation for a portion of C2 was originally too deep and resulted in a pond rather than the

intended wetland flora. To remedy this, the top soil was once again removed, the site partially

filled with soil from the excavation of the newer created wetland in 2007 (McMullen 2007), and

the top soil replaced.

The total created and restored area of the younger constructed wetland, C1, was 29 ha (D.

Harris, personal communications, April 6 2010), consisting of a mix of persistent emergent and

herbaceous emergent, though it also contained a number of ponds, wet meadows, and wooded

areas (Cowardin *et al.* 1979). A specific study area of 1 ha consisting of wet meadow,

herbaceous emergent, and pond habitat types was selected (Figure 2). Portions of the eastern

two-thirds of wetland were being replanted at the time of this study. C2 was approximately 5.6

ha and comprised of three ponds with persistent emergent areas and wet meadows and shrubby

and wooded areas around the perimeter (Terrestrial Environmental Specialists Inc 2002) (Figure

2).The perimeter of the two northern ponds, an area of about 1.1 ha total, was used as the study

site. The deeper areas of the ponds were not used in the study.

All four wetlands had similar hydrologic regimes and soil types. C1, C2, and N2 are

primarily Niagara silt loam, a shallow sloping and somewhat poorly drained soil. Niagara silt

loam abuts N1, but the site is comprised primarily of Canandaigua and Odessa silt loams, which

are somewhat to very poorly drained (Soil Survey Staff 2010).

The two natural wetlands used for this study were identified based on a 2001 wetland delineation report of the RIT campus (Terrestrial Environmental Specialists Inc. 2001). Both sites are adjacent to Red Creek and consist of emergent herbaceous vegetation and wet meadows, but are somewhat smaller than the constructed sites. Shrub/scrub and wooded habitats within these wetlands were avoided. A 0.6 ha study area of the 1.2 ha total area of N1 was selected. It was comprised of mostly persistent emergent and wet meadow areas, with an herbaceous emergent zone between the two (Figure 3) The fringes of the wetland were invaded with *Typha angustifolia* (narrowleaf cattail). The N2 study area was approximately 0.3 ha, with an entire wetland area of approximately 1 ha and consisted primarily of a wet meadow with persistent emergent areas around the perimeter and along Red Creek (Figure 4).

The study areas were delineated by walking their perimeters and marking waypoints with a Garmin Etrex Venture HC GPS unit and GoogleEarth and used to create a perimeter polygon of each study site. A 10 m x 10 m sampling grid of Universal Transverse Mercator (UTM) coordinates was established at each site. Each point was groundtruthed and points falling within wooded, shrubby, or ponded areas were excluded, retaining only points with emergent vegetation as described above. Due to their varying sizes, the wetlands had differing numbers of survey plots (Table 1).

Ecological Community Structure

Vegetation Surveys

Cover of emergent vegetation was measured at all sampling points in May-June 2010. Percent cover for all dominant plant species in each wetland was determined using a 1 m^2

sampling quadrat partitioned into 16 equal 0.0625 m^2 sections. The dominant plant species (> 50% cover) was visually determined in each of the 16 grids. When no single species was dominant, a "mixed" designation was used. "Bare ground" was also used a designation when a section was > 50% bare ground or water.

Based upon the vegetation data collected in spring 2010, a map of vegetation zones was drawn for each wetland and the distinct vegetation zones were visually delineated based upon the dominant plant species (Figure 5a, b, c, and d). Most zones were comprised of only one species, such as *Typha sp.*, and so were named according to their dominant species. The zone types were: *Aulacomnium palustr*e (bog moss), *Typha latifolia* (broadleaf cattail), *Typha angustifolia* (narrowleaf cattail), *Festuca rubra* (red fescue), *Phalaris arundinacea* (reed canary grass), *Juncus effusus* (soft rush), *Eleocharis palustris* (spike rush), *Agrostis stolonifer* (spreading bent grass), and *Scirpus cyperinus* (wool grass). Three permanent plots were selected randomly within each vegetation zone in each wetland for all future sampling and partial vegetation surveys that were conducted in July and September 2010. Both N1 and N2 each contained nine permanent plots, C1 contained twelve, and C2 contained six based upon the number of habitats present in each wetland. Semi-permanent wooden platforms were constructed at each permanent sampling plot to prevent damage to and disturbance of the delicate habitat. Soil samples were collected immediately next to the permanent plots rather than within the square meter to avoid disturbing the vegetation.

Vegetation was again sampled in summer and fall at the three permanent sampling plots within each habitat type and an additional 1-3 plots representing that habitat type selected randomly from the original 10 m x 10m grid to approximately double the number of points being sampled. At times, the number of plots that fell within a vegetation zone was fewer than six, and

thus the number of plots could not be fully doubled. A total of 16 plots were used in N1, 17 in N2, 24 in C1, and nine plots in C2.

<u>Macroinvertebrate Sampling</u>

Adjacent to each permanent site, soil samples were taken for macroinvertebrate zone composition at the same time as the spring, summer and fall vegetation surveys. One core of the top 10 cm of soil was extracted using a 7.62 cm diameter metal auger. The sample was stored in plastic zip-top bags on ice and transported to the lab where it was sieved (1 mm mesh) and invertebrates were immediately preserved in ethanol (Stanczak and Keiper 2004). The invertebrates were later identified under a dissection microscope to lowest practical taxonomic level.

Geochemical properties

Two soil cores for organic matter (OM) and extractable phosphorus were extracted from the top 10 cm of soil at each plot May 24-June 4, July 26-August 6, and September 20- October 1, 2010 with a 7.62 cm diameter metal auger. The cores were frozen in plastic zip-top bags at -20°C until analysis. Soil was dried in a 60°C oven for at least 48 hours and then ground to homogeneity by hand using a mortar and pestle. Percent OM was determined gravimetrically by loss on combustion at 500°C for at least four hours on two 15 g subsamples from each core.

Soil extractable phosphorus content was determined on two 0.1 g subsamples each for total phosphorus (TP) and inorganic phosphorus (IP). Samples were placed into 20 mL glass scintillation vials. Samples for TP only were mixed with 0.5 mL of $Mg(NO_3)_2$ and ashed for two hours at 550°C. Ten milliliters of 1N hydrochloric acid was added to all samples and the vials

were placed on a horizontal shaker for 16 hours after which the samples were allowed to settle for 24 hours (adapted from Aspila *et al*. 1976). Once settled, the samples were analyzed spectrophotometrically using the ammonium molybdate method and a Shimadzu 2100 spectrophotometer.

Nutrient sorption capacity was measured on two additional cores collected during midsummer (July 25 – August 13 2010). The top 5cm of soil was extracted using a 9.5 cm inner diameter polycarbonate core tube. The core was inserted into the sediment, filled with overlying water, stoppered at the top, extracted and stoppered on the bottom. Following transport to the lab, the top stopper was removed and the cores were mostly submerged in a tank of water where they were allowed to acclimate for 24 hours. The top 2 cm of the core tube was above the surrounding water to maintain separate water columns. The headspace was gently aerated with a standard aquarium air stone and air pump to maintain water mixing and aerobic conditions. The water in the holding tank maintained at 74^oF (23.3^oC) and was exposed to a 14 hr light / 10 hr dark cycle using full spectrum fluorescent lamps. After the acclimation period, the overlying water was carefully siphoned out and replaced with water collected from nearby Red Creek.

Half of the cores (one per site) were spiked with potassium phosphate (KH_2PO_4) and sodium nitrate ($NaNO_3$) to a final concentration of 16.25 mg/L KH_2PO_4 and 481 mg/L $NaNO_3$. These concentrations were chosen to simulate nutrient loading after a fertilizer runoff event (Eghball and Gilley 1999). The phosphate concentration was comparable to literature values and nitrate was ten times literature values (Eghball and Gilley 1999). A 60 mL water sample was taken from each core at 0, 6, 24, 48, and 96 hours. Each sample was immediately filtered through a Supor ® 0.45 μm filter and frozen in Whirl-pak ® bags at -20^oC for total phosphorus and nitrogen (TP and TN, respectively) and nitrate and orthophosphate analysis. The samples

were analyzed using a Lachat Quickchem 8500 and methods 31-107-04-1-C, 31-115-01-1-J for nitrate and phosphate, respectively. TP and TN data were not complete and are not presented here.

Data Analysis

Biological factors

A principle component analysis (PCA) was run on the spring vegetation data using PRIMER version 6 (Clark and Warwick 2001) to determine vegetation zone distribution. Shannon-Weiner diversity indices, relative dominance, vegetation richness and evenness were calculated for all wetlands and all three seasons. Two separate one- way analysis of variance (ANOVA) tests in SPSS 11.0 were used to determine the effects of wetland type and then season on the invertebrate distribution. Shannon-Weiner diversity indices, relative dominance, species richness and evenness were calculated for all wetlands and all three seasons.

Geochemical Factors

All data were tested for normality using the Kolmogorov-Smirnov test and homogeneity of variance using Levene's test and SPSS 11.0 software. Some data did not have equal variance and no reasonable transformations produced equal variance, so nontransformed data were used for all analyses. Tamhane T2 post-hoc analyses were used on data that violated the assumption of equal variance. Tukey HSD post-hoc analyses were used on all other data to determine the differences among means and groups.

A two-way ANOVA was used to determine effects of season and wetland on soil organic matter content among all wetlands, and a one-way ANOVA was used to determine differences in

soil organic matter content among vegetation zones within each wetland for the pooled spring and summer data. Tukey HSD and Tamhane T2 post-hoc analyses were used to determine significant differences ($p < 0.05$) between factor groups. Spring and summer values were averaged as there was no significant difference between seasons and the remainder of the factor analyses were performed on the mean values. The organic content within dominant vegetation zones was analyzed separately to elucidate within-wetland differences associated with the different plant zones. Because narrow leaf and broad leaf cattail were found in the majority of the wetlands, these species were also analyzed for differences across wetlands where they were present.

A paried t-test was used to determine any significant differences between spring and summer soil phosphorus concentrations ($p < 0.05$) at each sampling plot. A two-way ANOVA was used to determine effects of season and wetland site on inorganic and organic phosphorus concentrations. A one-way ANOVA was run on each spring and summer data sets for inorganic and organic phosphorus to determine differences among vegetation zones within each wetland. A linear regression was used to determine any relationship between mean organic matter and soil phosphorus concentrations in each wetland.

All nitrate and phosphate flux rates were computed for each time point and compared visually to determine whether there were multiple phases of uptake. There was a clear distinction between the initial surge uptake between 0 and 6 hr and the more consistent sustained uptake rate between 6 and 96 hr, so the initial and sustained rates were compared separately across the wetland types using one-way ANOVA with wetland as the fixed factor.

Vegetation

A total of 30 plant species were recorded in all four study sites over the course of spring, summer and fall. Eight species were present only during summer and fall. Each wetland was dominated by two to four species, forming relatively distinct vegetation zones within each site. While several species were found in multiple wetlands, only *Typha latifolia* and *T. angustifolia* formed distinct zones in more than one wetland (Table 2).

The Shannon-Weiner diversity indices and species richness varied among wetlands and over time. Of all the wetlands, C1 consistently had the highest species richness and diversity throughout all three seasons, and the lowest overall dominance of invasive species (Table 2). Both created wetlands were most diverse in the spring, while N1 was most diverse in the fall and N2 in the summer. There was less fluctuation in diversity in the natural wetlands, however (Table 2). All wetlands except for N2 had greatest species richness in spring; N2 had 9 species in the spring and 10 in both summer and fall, though the species themselves varied (Table 2). While the natural wetlands did not have the highest richness, they maintained a more stable number of species over the course of the growing season (Table 2). In C2, *P. arundinacea* became increasingly dominant over time, and in both C1 and C2 native species such as *J. effusus*, *Scirpus atrovirens*, and *Carex vulpinoidea* became less prevalent. Species evenness varied among wetlands and seasons, with no apparent trend (Table 2).

Invertebrates

Only seven invertebrate species were found across spring, summer and fall in all four wetlands (Table 3). The small freshwater clam, *Pisidium compressum* was by far the most numerous species; additional species included lunged snails and *Lumbricus terrestris*, the

common earthworm. There were no significant differences among wetlands or season in terms of individual species. A two-way ANOVA also showed no significant effect of season on wetland in terms of species present.

Soil OM

A two-way ANOVA using season and wetland as fixed factors determined that while there were no significant differences between seasons (P = 0.608, F = 0.266), there were significant differences among wetlands (P < 0.001, F = 12.831). As the spring and summer OM values were statistically equivalent (P > 0.05), the pooled data for both seasons were used to compare across wetlands. Percent OM varied significantly between each wetland, ranging from 4.88 ± 0.31% in N1 to 13.39 ± 1.62 % in N2. There were no significant differences in %OM related to wetland age, as N1 and C2, with the lowest OM content, were statistically equivalent (P = 0.267). N1 was significantly lower than both N2 and C1 (P = 0.004 and 0.014, respectively), but C2 was only significantly lower than N2 (P = 0.022). N2 and C1 were also statistically equivalent to one another (P = 0.958) (Figure 6).

The % OM in N1 ranged from 4.56 ± 0.26% in *T. angustifolia* plots to 5.37 ± 0.85% in *A. palustr*e plots. The OM content in all three species plots were statistically equivalent (P = 0.325, F = 1.257) (Figure 7a). All vegetation species plots in N2 also had statistically equivalent OM content (P = 0.152, F = 2.618), and ranged from 10.72 ± 2.65% in *F. rubra* plots up to 17.80 ± 2.59% in broadleaf cattail plots (Figure 7b). Within N2, *F. rubra* plots varied the most with one plot averaging 6.26%, and another 15.44%. The *S. cyperinus*, and *T. latifolia* plots were also quite variable, though to a lesser degree.

In C1, *A. stolonifera* and *P. arundinacea*, plots were statistically similar (P = 0.950). *A. stolonifera* plots contained significantly more OM than both *J. effusus* and *T. latifolia* plots (P =

0.013 and 0.044, respectively) as did *P. arundinacea* plots (P = 0.006 and 0.21, respectively). *J. effuses*, and *T. latifolia* plots contained statistically equivalent amounts of OM (P = 0.808) (Figure 7c). Both *Typha* plots in C2 had statistically equivalent (P = 0.325) OM content (Figure 7d).

Out of all vegetation species, *P. arundinacea* plots had the highest % OM, with a spring and summer mean of 16.95 ± 3.06%, and *Eleocharis sp.* the lowest with a mean of 4.72 ± 0.45%. Even when a species was present in multiple study sites, the % OM varied significantly among wetlands. The OM content of *T. latifolia* plots in N2 was significantly higher than that found in either C1 (P = 0.003) or C2 (P = 0.008). However, the OM content of *T. angustifolia* sites in N1 and C2 were statistically equivalent (P = 0.378, F = 0.981) (Figure 8).

Soil Phosphorus

Inorganic phosphorus (IP) was higher in summer at all sites, but only significantly so at N2 (P = 0.034, F = 5.384) and C1 (P = 0.026, F = 5.684). Both seasons were statistically similar in N1 (P = 0.365, F = 0.869) and in C2 (P = 0.139, F = 2.578) (Figure 9a). There were no significant seasonal differences in the organic phosphorus (OP) concentrations (P = 0.822, F = 0.051) but compared to spring concentrations, the mean summer concentrations were lower in both natural wetlands and higher in both created wetlands (Figure 9b). In spring, IP was similar at all sites, ranging from 12.39 ± 0.82 mmol/kg in C2 to 16.12 ± 1.89 mmol/kg in N2. In summer, the only significant difference that arose was between N2 and C2 (P = 0.016), while all other sites remained statistically similar (between N2 and C1 P = 0.051, all other wetlands P > 0.450).

In spring, OP values ranged from 6.25 ± 0.99 mmol/kg in N1 up to 16.41 ± 1.98 mmol/kg in N2. N1 and C2 OP concentrations were statistically similar (P = 0.923) and

significantly lower than N2 (P < 0.001 and 0.007 for N1 and C2, respectively) and C1 (P = 0.001 and 0.024 for N1 and C2, respectively), which were similar to one another (P = 0.871). The relative concentrations of P in summer were similar, with N1 again significantly lower than N2 (P = 0.026) and C1 (P < 0.001). However, N2 was only significantly higher than N1, while C1 was still significantly higher than both N1 and C2 (P = 0.014) (Figure 9a and b).

There were no significant differences (Table 4) in either IP or OP among vegetation types in either spring or summer within N1, C1 or C2 (Figure 10 a, c, and d, respectively). In N2, *T. latifolia* plots had the highest IP and OP concentrations. IP values were significantly higher than both *F. rubra* (P = 0.004 and 0.001 in spring and summer, respectively) and *S. cyperinus* (P = 0.008 and 0.002 in spring and summer, respectively) plots (Figure 10b). Tukey HSD post-hoc was used for spring data, and Tamhane T2 post-hoc was used for summer data due to unequal variances. A linear regression between mean OP and % OM showed a significant (R^2 = 0.58, P = 0.000) correlation, indicating that OM and OP are directly impacted by one another (Figure 11).

Nitrate and phosphate fluxes

In all wetlands and all treatments there was an initial phase of "surge" uptake or release during the 0-6 hour time period. After this initial time interval, flux rates slowed and were consistent for the remainder of the experiment (Figures 12 and 13, a and b). The nitrate flux rate in the unspiked C1 cores did differ slightly from this trend, and its fastest flux rate occurred between T6 and T24. Due to the variability of the data, however, this difference was not statistically significant from the rates between 0-6 hr (P = 0.902). With the exception of nitrate in the unspiked cores, there was a decrease in overlying water nitrate and phosphate concentration

over time, indicating overall uptake of nutrients by the soil. Phosphate uptake in unspiked cores was higher in the natural wetlands, but only significantly so for N1 at T0-6 (P = 0.024), which was higher than all others (Figure 12a). N1 had the lowest phosphate uptake rate during T0-6 in the spiked cores, but the rate was not significantly lower than any others (P = 0.555, 0.920, and 0.555 between N1 and N2, C1, C2, respectively). The phosphate flux rates during T6-96 in both spiked (P = 0.498, F = 0.798) and unspiked cores (P = 0.970, F = 0.082) were statistically similar (Figure 12a).

Nitrate flux rates were much more variable among time points and wetlands than the phosphate flux rates (Figure 13 a and b). N1 still had nitrate uptake capacity in the unspiked cores, while samples from all other wetlands released nitrate into the water column. Variability in the data render this difference insignificant from N2 and C2 (P = 0.654 and 0.931, respectively) for T6-96, and only significantly different from C1 for the same time period (P = 0.015). It is important to note that the direction of net nutrient flux is reversed relative to the constructed wetlands that release nitrate to the water column (Figure 13a).

The phosphate flux rates were consistently greater from 0-6hr in all wetlands except for C1 in the unspiked cores (Figure 14a and b). C2 unspiked cores and N1 spiked cores had a higher rate between 0-6hr than 6-96hr, but variability in the data rendered the differences insignificant. Nitrate flux rates were more variable and so fewer distinct trends emerged (Figure 14 c and d) between T0-6 and T6-96 time points.

DISCUSSION

Vegetation

The lack of clear age-related trends in vegetation zone structure and diversity suggests that there are a number of factors at play in determining the vegetation zone. Despite being slightly older than C1, C2 was consistently the least diverse study site. The majority of the site was a shallow pond, with emergent wetland vegetation limited to the shore periphery, which itself was relatively steep. Many wetland plants have a water depth threshold and will not grow in areas where the water is too deep; a change of only one foot in water depth can limit vegetation distribution. At the time of construction, a wide variety of wet meadow and herbaceous emergent vegetation species were planted (Terrestrial Environmental Specialists 2002), including various grasses, rushes and sedges. The lack of diversity in C2 indicates that these species are not regenerating from season to season, or that invasive species are more aggressive than the originally planted species. Cattails were not included in the planting plan, despite their current dominance. *Typha angustifolia* (narrowleaf cattail), however, is highly invasive in disturbed wetlands (Stevens and Hoag 2011), as are *Phalaris arundinacea* (reed canary grass), and *Phragmites australis* (common reed), all of which were also found in C2 (Table 2). While the *Typha* sp. only saw a slight increase in cover over the course of the growing season, the large increase of *P. arundinacea* cover (4.6% in spring, 19.7% in fall) suggests that other invasive species are capable of competing with *Tyhpa* sp, and together will continue to crowd out native species. One of the three ponds at C2 was altered in 2007 to provide better emergent habitat, but little else has been altered since its creation. Given the aggressive nature of invasive species, this limited management of a freshly disturbed wetland area could encourage invasive species growth.

Though the creation of C1 also resulted in a great deal of disturbance, C1 consistently had the highest vegetation diversity of all four wetlands (Table 2). There was a greater diversity of water depths and habitats at the time of creation at C1, including many more wet meadow and herbaceous emergent regions. This variation in habitat, and the fact that C1 is younger than C2, may contribute to the greater diversity. Invasive species were also present in C1, including *T. angustifolia*, the hybrid cattail, *Typha x glauca*, and *P. arundinacea* (Table 2). *P. australis* was also present at C1, but not within the study area. The presence of dense zones of invasive species that propagated in only three to four years indicates that they may continue to spread. This is corroborated by the increase in percent cover of *T. latifolia* from spring (15.8%) to fall (21.5%), suggesting that the vegetation community at C1 was not stable and may become more fully dominated by monoculture forming species in the near future without intervention. Without continued management and invasive species control, such as physical removal of invasive and replanting of native species, it is likely that C1's diversity could decrease dramatically in as little as five years, much like C2.

Invasive species were also present in N1 and N2 and included *Lythrum salicaria* (purple loosestrife) as well as *P. arundinacea* and *T. angustifolia* (Table 2). *P. australis* was also present in N1, but not within the selected study area. With the exception of *T. angustifolia*, however, the invasive species did not form the dense zones found in the created wetlands. This may be due to the fact that native vegetation was able to adequately establish itself prior to when the nonnative species began to invade. However, in both natural wetlands, the percent cover of *T. angustifolia* increased substantially (1.6 x in N1 and 2.6 x in N2) from spring to fall, with a concomitant decline in *Eleocharis sp.* and *F. rubrus* in N1 and N2, respectively. The spread of invasive species even in the natural wetlands makes evident the need for continued invasive species

management. While created wetlands may be more vulnerable to the colonization of nonnative species, natural wetlands are clearly at risk as well. Similar to management in the created wetlands, physical removal of invasive species may be required.

The invertebrate distribution among wetlands was not conclusive and did not show many trends. Some of the genera found were consistent with other studies of natural and created wetlands, such as *Physidae, Lymnaeidae,* both lunged snails, and *Pisidium*, a fingernail clam (Spieles and Mitsch 2000, Stanczak and Keiper 2004, Stewart and Downing 2008). The presence of only lunged snails may indicate poorer water quality as gilled snails, being unable to breathe air, require higher water dissolved oxygen levels to survive. *Pisidium sp.* were most numerous, almost exclusively appearing in created wetlands, a finding that is contrary to earlier studies (Stanczak and Keiper 2004, Clinton and Whiles 2008). Due to differing sample sizes and the limited time frame of the study, however, these findings are not decisive. That such a limited number of genera were found indicates that further studies should be conducted to better assess the macroinvertebrate communities in these wetlands.

Organic Matter

Organic matter also did not follow clear age-related patterns or consistent trends with vegetation zone structure. N1 did not appear to be representative of a typical natural wetland as it had such low soil OM. In studies, natural wetlands have been found to not only have high OM content, but also to have OM content that is consistently greater than that in comparable created wetlands (Stolt *et al*. 2000, Campbell *et al*. 2002, Bruland and Richardson 2006). Indeed, even within this study N2 had nearly three times the OM content than N1, $13.39 \pm 1.61\%$ and $4.88 \pm 0.31\%$, respectively. The differences in soil type between N1 and the other three wetlands may

play a significant role in this inconsistency. Based upon visual assessment, the soil in N1 was clay-rich and in some areas sandy with little apparent leaf litter, even within the *T. angustifolia* zones. The cattails at this site were relatively less dense than the other sites; this reduced production in turn lead to less leaf litter, and thus likely contributed to the lower soil OM. Natural wetlands often have higher %OM than created wetlands (Bruland and Richardson 2006). The high %OM in C1 soils is similar to both N2 %OM and other comparable natural wetlands (Campbell *et al*. 2002), indicating its success and similarity to a naturally occurring wetland in terms of carbon storage. Despite its low vegetation diversity and younger age, the mean %OM at C2 was higher than N1 (6.83 ± 0.76%) and was similar to the initial, amended concentration at the time of construction (7-8 %OM; D. Harris personal communication April 6 2010) suggesting stability over time.

Despite their overall significant differences, there were no significant trends in soil OM between vegetation zones in N1, N2, or C2 (Figures 7a, b, and d, respectively), though plots that were dominated by grasses did tend to have higher soil OM. Other studies have found that vegetation zones remained similar among different wetlands despite significant differences in soil OM (Bailey *et al.* 2007, Bantilan-Smith *et al.* 2009). These results indicate that vegetation may not have as strong of a feedback impact on soil OM than vice versa. That the *T. latifolia* plots in N2 contained nearly twice the OM of any other *Typha* plot may be due to the fact that N2 simply had the highest overall OM content, rather than an effect of the vegetation. Out of all the study sites, N2 was closest to Red Creek and may have acquired additional organic matter from flooding and run-off events. C1's overall OM was statistically equivalent to that of N2, but its mean % OM was heavily influenced by the *A. stolonifera* and *P. arundinacea* plots. These two grasses had significantly higher OM content than *J. effusus* or *T. latifolia* plots in C1 (Figure

7c) and form a dense root mat that likely contributed to the increased OM content. *P. arundinacea* also produces a dense stem and leaf mat, which may trap additional soil and organic material. These results indicate that for some species, location and other factors influence the soil OM content more than the vegetation zone type alone. Additionally, should this hold true, using hydrophytic vegetation alone as the primary indicator of wetland success (Atkinson and Cairns 2001) may be inadequate. As vegetation zones and soil OM levels do not appear to be mutually exclusive, it would be inappropriate to determine functional success based upon one, vegetation, when the other, high soil OM is a known attribute of natural wetlands (Stolt *et al*. 2000, Campbell *et al*. 2002, Bruland and Richardson 2006, Bantilan-Smith *et al.* 2009). However, with only two species forming dominant zones in more than one wetland, this conclusion is still preliminary and further studies need to be conducted to determine any significant links on a larger scale.

Soil Phosphorus

Biologically available phosphorus is a critical macronutrient for plants and microorganisms (Reddy *et al*. 1989, Schatchman *et al*. 1998). An ecosystem's ability to transform nutrients to biologically available forms is therefore paramount to its functional success. A number of different factors impact phosphorus availability in soil, including pH, the presence of other ions, microbial activity, and even the level of ecosystem succession (Odum 1969, Goldberg and Sposito 1985, Song *et al.* 2007). The trend of increasing soil IP from spring to summer indicates a shift in these processes leading to release of IP. Increased biological, and thus microbial, activity leading to mineralization of P from organic matter could be a large contributing factor to the increased available P in summer as decomposition rates in wetlands are

28

higher during the warmer months (Kirschbaum 1994, Davidson and Janssens 2004). The increase in decomposition is supported in the natural wetlands, where a concomitant decrease in OP was observed from spring to summer (Figure 9b). The slight increase in OP in the created wetlands, however, suggested that either the IP in created marsh soils is from external sources in the summer, or that microbial mineralization is limited by other factors.

The relative proportions of IP and OP among the different vegetation zones within each site reflected the trend observed for total soil OM. IP remained similar among all wetlands, regardless of vegetation type, while OP fluctuated in turn with soil OM (Figure 10). Just as soil OM was not significantly related to vegetation zone, neither was soil phosphorus. However, soil nutrients have been found to be increased by OM loading (Hogan *et al.* 2004, Bailey *et al.* 2007), and the two were found to be significantly correlated in this study (Figure 11). These findings indicate that soil chemistry may exert stronger effects on vegetation zones in a positive feedback loop. As soil OM and soil phosphorus appeared to be linked, soil nutrient content becomes an additional critical factor for determining created wetland functionality. Vegetation alone may not indicate wetland success, but soil nutrients limit vegetation growth (Verhoeven 1996) and thus the ecological success of organisms that feed or otherwise rely on vegetation for habitat.

Nutrient Flux

It appears that even though N1 did not have significantly high soil IP concentrations and even had the lowest OP concentrations (Figure 9 a and b), it still maintains the greatest ability to quickly take in phosphate in flood conditions. Beyond that, however, all wetlands had a statistically equivalent capacity for phosphate uptake in both normal and nutrient loaded conditions. Previous studies provide conflicting results, with some finding that created wetlands

actually had a greater phosphorus sorption and retention capacity than natural wetlands (Mitsch *et al.* 1995, Hogan *et al.* 2004). D'Angelo (2005), however, found that late successional wetland soils had a greater phosphorus sorption capacity than early successional wetland soils. As created wetlands are relatively young ecosystems, they are often in an early successional stage. With age often comes a build up of organic material, and thus greater levels of organic compounds in the soil. A soil's ability to adsorb phosphorus is linked to organically-bound aluminum and iron (Hogan *et al.* 2004, D'Angelo 2005), compounds that may build up over time with the accumulation of OM in soils. As the created wetlands in this study did not show significant long-term differences in phosphorus sorption capacity, it may be inferred that their soils are functionally equivalent to the natural wetland soils in this regard, despite their history as agricultural fields.

Despite the variability in the data, all four wetlands do show an overall capacity for nutrient uptake in extreme loading conditions. N2, C1 and C2 released nitrate under low N loading, but still maintained the capacity to adsorb additional nitrate under pulsed loading conditions. All cores were used intact, and while any surface vegetation was cut to soil level, root stock remained. Microorganisms and burrowing macroinvertebrates were also presumably present as the cores remained unaltered from their collected state. Invertebrates play a key role in nutrient cycling (Blackwell *et al*. 2002, Song *et al.* 2007, Stewart and Downing 2008) as do microorganisms through pathways such as denitrification (Blackwell *et al.* 2002, Reddy *et al.* 1989) and mobilization (Song *et al.* 2007). They, combined with any remaining root systems, may have had a strong impact on these results. Further studies examining the microbial zones of these wetland soils may help to elucidate additional biotic differences that may contribute to nitrate uptake or release rates.

<u>Conclusions</u>

Overall, there were no significant age-related trends among the four wetlands. Contrary to our hypotheses, neither vegetation nor macroinvertebrates were less diverse in the created wetlands, suggesting that their biotic functionality is comparable to that of natural wetlands. The created wetlands were also comparable to the natural wetlands in terms of their geochemical functions. Soil, hydrogeomorphology and vegetation type therefore may be the primary influencing factors on overall ecosystem function. Hydrogeomorphology is a significant factor in determining the biotic structure of a wetland (Lougheed *et al.* 2001, Bailey *et al.* 2007), and so investigation of additional chemical components of the soil, such as pH, salinity, and ion content. The low invertebrate densities at all sites may indicate limited distribution among natural and created sites. ..Vegetation data may have been slightly limited as only selected plots were reassessed throughout the growing season. A complete vegetation survey of all plots would provide a broader image of the vegetation community dynamics over the course of the growing season.

Compared to the other three study sites, N1 was significantly different in several aspects: it had the lowest soil OM, the lowest OP concentrations, the greatest capacity for phosphate uptake in normal, low nutrient loading conditions, and continued to act a nitrate sink while the other wetlands were a source under low nutrient loading conditions. N1 was also the only wetland to be comprised of a different soil type, and thus may not have been an ideal reference wetland.

Soil organic matter content and organic phosphate concentrations appear to be linked, and as seen in C1, vegetation zone type may influence soil organic matter content. However, it is

likely that soil structure and deviation are the driving factor in determining vegetation zone type, as there were few differences among vegetation zones that could not be explained by soil attributes.

Both created wetlands fell within the range of the natural wetlands for all tested parameters, indicating that overall, they may have comparable functionality. In this case, functionality would be based upon similarities in soil chemical structure and the ability to act as a nutrient sink. Based upon vegetation diversity, richness, and evenness, C1 may be functionally comparable to both natural wetlands in its ability to support a diverse vegetation community. However, greater diversity due to the presence of invasive species does not necessarily indicate a healthier ecosystem. The much lower vegetation diversity in C2 indicates that while soil chemical structure may be similar to natural wetlands, it is not supporting a similar biotic community and thus may not be providing similar services and functions in terms of habitat. The lower vegetation diversity in C2 also implies that C1 may become less diverse over time if not properly managed and if invasive species are not continually controlled. The presence of invasive species in both natural and created wetlands proves that regardless of status or age, wetlands are vulnerable to colonization by aggressive, invasive species. The limited diversity in C2 does however indicate that created wetlands may be particularly susceptible to invasion while natural wetlands maintain greater stability over time. Continued monitoring of C1 and additional restoration of C2 are of paramount importance to maintain both created wetlands at their highest biotic functionality. While a greater wetland sample size will help to further support this conclusion, it does appear that created wetlands fall within the functional range of natural wetlands. Invasive species remain a problem for both natural and created wetlands, but with

continued management both wetland types may be allowed to thrive and provide comparable

functionality.

Table 1. Abiotic and biotic characteristics of each study site.

Study Site	Total Wetland Area (ha)	Study Site Area	Habitat Types	Vegetation Zones	Soil Types	Date of Creation	Total Sampling Plots
N1	1.2	0.6	Persistent emergent, wet meadow, herbaceous emergent, shrub	*Aulacomnium palustre, Typha angustifolia, Eleocharis sp.*	Canandaigua and Odessa silt loam	n/a	34
N2	1	0.3	Wet meadow, persistent emergent, shrub	*Typha angustifolia, Festuca rubra, Scirpus cyperinus*	Niagara silt loam	n/a	29
C1	29	1	Wet meadow, herbaceous and persistent emergent, pond, wooded	*Typha latifolia, Phalaris arundinacea, Juncus effusus, Agrostis stolonifera*	Niagara silt loam	2007	65
C2	5.6	1.1	Wet meadow, persistent emergent, pond, shrub, wooded	*Typha latifolia, Typha angustifolia*	Niagara silt loam	2002	42

Table 2. Seasonal vegetation species dominance, wetland Shannon-Weiner diversity index, species richness and evenness. Species whose dominance was less than 1% were included in "Mixed species". A dash (-) indicates that the species was present at greater than 1% in another season or wetland. "Bare ground" was eliminated as a category to compare only vegetated area. Bolded values indicate dominant vegetation zones and * denotes an invasive species.

| Species	N1 % Relative Dominance			N2 % Relative Dominance			C1 % Relative Dominance			C2 % Relative Dominance		
	Spring	Summer	Fall	Spring	Summer	Fall	Spring	Summer	Fall	Spring	Summer	Fall
Aulacomnium palustre	**7.4**	**15.1**	**10.7**	-	-	-	-	-	-	-	-	-
Carex tribuloides	-	-	-	-	-	-	2.8	-	-	-	-	-
Typha latifolia	-	-	-	-	2.3	1.3	**15.8**	**18.7**	**21.5**	**8.4**	**20.8**	**14.5**
Equisetum arvense	-	-	-	3.0	-	-	-	-	-	-	-	-
Carex vulpinoidea	-	-	-	-	-	-	4.5	-	-	-	-	-
Euthamia graminifolia	-	-	-	-	1.8	-	-	-	-	-	-	-
Scirpus atrovirens	-	-	-	-	-	-	2.2	1.4	-	-	-	-
Carex lurida	6.2	4.8	4.0	-	-	-	2.6	-	1.1	-	-	-
Mixed species	9.2	7.0	10.2	5.4	3.7	4.6	12.9	5.4	5.1	13.7	8.3	10.5
Lysimachia nummularia	1.8	-	2.8	-	-	-	-	-	-	-	-	-
*Typha angustifolia**	**16.3**	**22.6**	**26.6**	**10.8**	**22.6**	**28.3**	1.9	-	2.2	**62.7**	**69.4**	**55.3**
Alisma triviale	-	-	-	-	-	-	-	1.0	1.1	-	-	-
*Phragmites australis**	-	-	-	-	-	-	-	-	-	-	1.4	-
Toxicodendron radicans	-	-	-	-	1.4	-	-	-	-	-	-	-
*Lythrum salicaria**	8.3	8.1	9.0	-	2.3	1.3	-	-	-	2.7	-	-
Festuca rubra	-	-	-	**50.2**	**36.4**	**35.4**	-	-	-	-	-	-
*Phalaris arundinacea**	2.1	-	-	7.6	2.3	-	**11.2**	**26.5**	**22.5**	4.6	-	19.7
Equisetum arvense	-	1.6	2.3	-	-	-	-	-	-	-	-	-
Juncus effusus	-	-	-	-	-	-	**27.3**	**23.5**	**19.3**	6.1	-	-
Scirpus tabernaemontani	-	-	-	-	-	-	1.6	-	-	1.9	-	-
Eleocharis sp.	**48.7**	**40.9**	**34.5**	-	-	-	-	-	-	-	-	-
Agrostis stolonifera	-	-	-	-	-	-	**17.3**	**23.5**	**21.1**	-	-	-
Scirpus cyperinus	-	-	-	**22.9**	**27.2**	**29.2**	-	-	6.2	-	-	-
Shannon-Weiner Diversity	1.62	1.59	1.72	1.45	1.61	1.41	2.10	1.68	1.87	1.31	0.85	1.16
Species Richness	9	6	7	9	10	10	15	11	12	8	3	3
Species Evenness	0.70	0.67	0.83	0.60	0.67	0.59	0.76	0.68	0.73	0.60	0.61	0.84

Table 3. Seasonal invertebrate species dominance, expressed as % relative dominance per m^2, wetland Shannon-Weiner diversity index, species richness and evenness. ". A dash (-) indicates that the species was present in another season or wetland.

Species	N1 % Relative Dominance			N2 % Relative Dominance			C1 % Relative Dominance			C2 % Relative Dominance		
	Spring	Summer	Fall	Spring	Summer	Fall	Spring	Summer	Fall	Spring	Summer	Fall
Lumbricus terrestris	**-**	**-**	**-**	8.3	16.7	25.0	**-**	**-**	**-**	**-**	**-**	**-**
Lymnaeidae spp	10.0	6.3	100.0	-	-	-	2.2	0.7	10.0	-	-	-
Physella gyrina aurea	-	12.5	-	11.7	16.7	-	3.3	3.5	1.7	11.1	7.4	-
Physella heterostropha halei	10.0	-	-	-	-	-	-	-	-	5.6	3.7	-
Pisidium compressum	2.5	-	-	-	-	-	14.4	6.7	8.3	33.3	3.7	-
Planorbidae spp	2.5	6.3		-	-	-	-	0.3	-	-	18.5	-
Sphaerium simile	-	-	-	-	-	-	-	1.3	-	-	-	-
Shannon-Weiner Diversity	0.83	1.04	0.00	0.68	1.04	0.00	0.78	0.51	0.92	0.85	1.15	0.00
Species Richness	4	3	1	2	2	1	3	5	3	3	4	0
Species Evenness	0.51	0.75	0.00	0.98	1.00	0.00	0.71	0.32	0.84	0.77	0.83	0.00

Table 4. F statistic and significance P value from one-way ANOVAs determining the differences in soil phosphorus content among vegetation zones within each wetland for both spring and summer. IP = inorganic phosphorus, OP = organic phosphorus. $P < 0.05$ indicates significant differences, in **bold**.

Wetland	Analyte	Season	F statistic	P Value
N1	IP	Spring	0.279	0.766
		Summer	0.347	0.720
	OP	Spring	1.507	0.296
		Summer	1.716	0.257
N2	IP	Spring	**17.328**	**0.003**
		Summer	**33.172**	**0.001**
	OP	Spring	0.957	0.436
		Summer	0.955	0.436
C1	IP	Spring	0.610	0.627
		Summer	1.019	0.434
	OP	Spring	3.915	0.054
		Summer	3.218	0.083
C2	IP	Spring	0.991	0.376
		Summer	0.153	0.715
	OP	Spring	0.000	0.996
		Summer	0.874	0.403
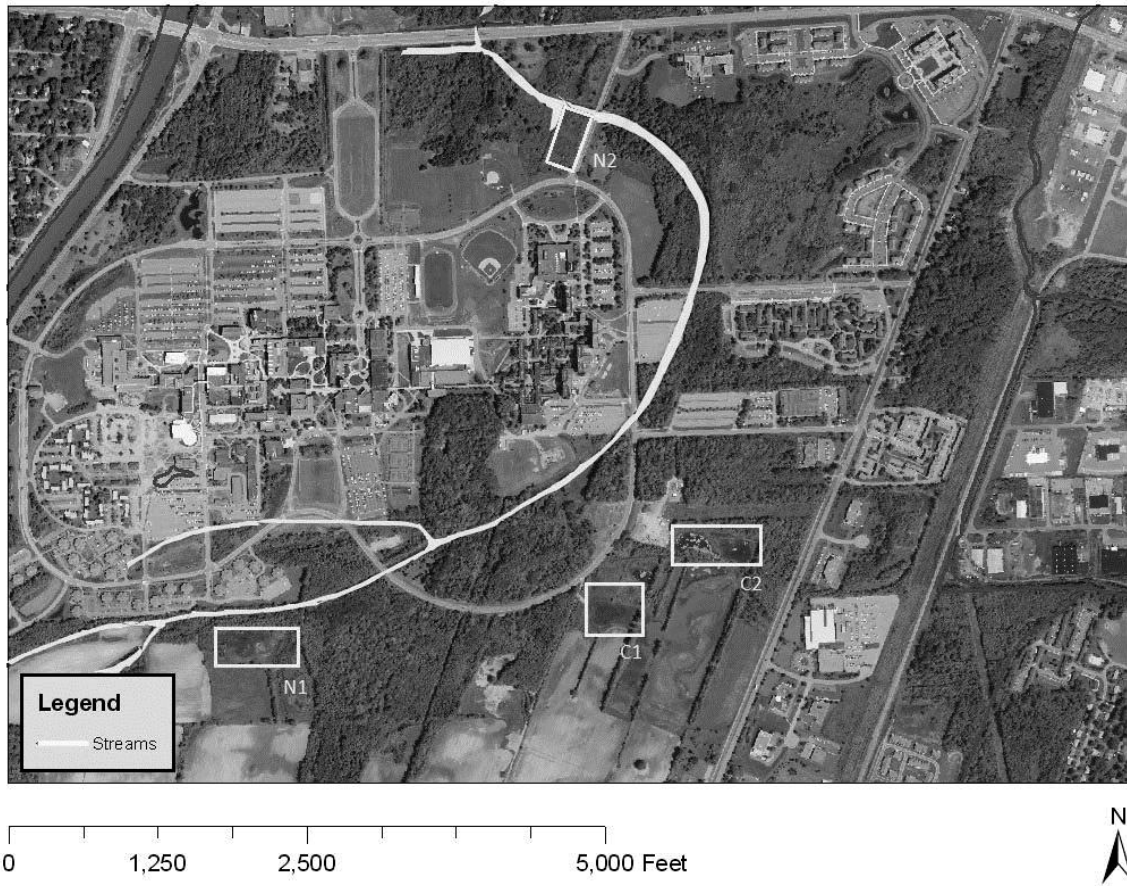
Figure 1. Study sites, N1, N2, C1 and C2 on the RIT campus. Study sites are delineated by boxes.

Figure 2. The delineated study sites of the two created wetlands, C1 and C2

Figure 3. The delineated study site of the natural wetland, N1.

Figure 4. The delineated study site of the natural wetland, N2.
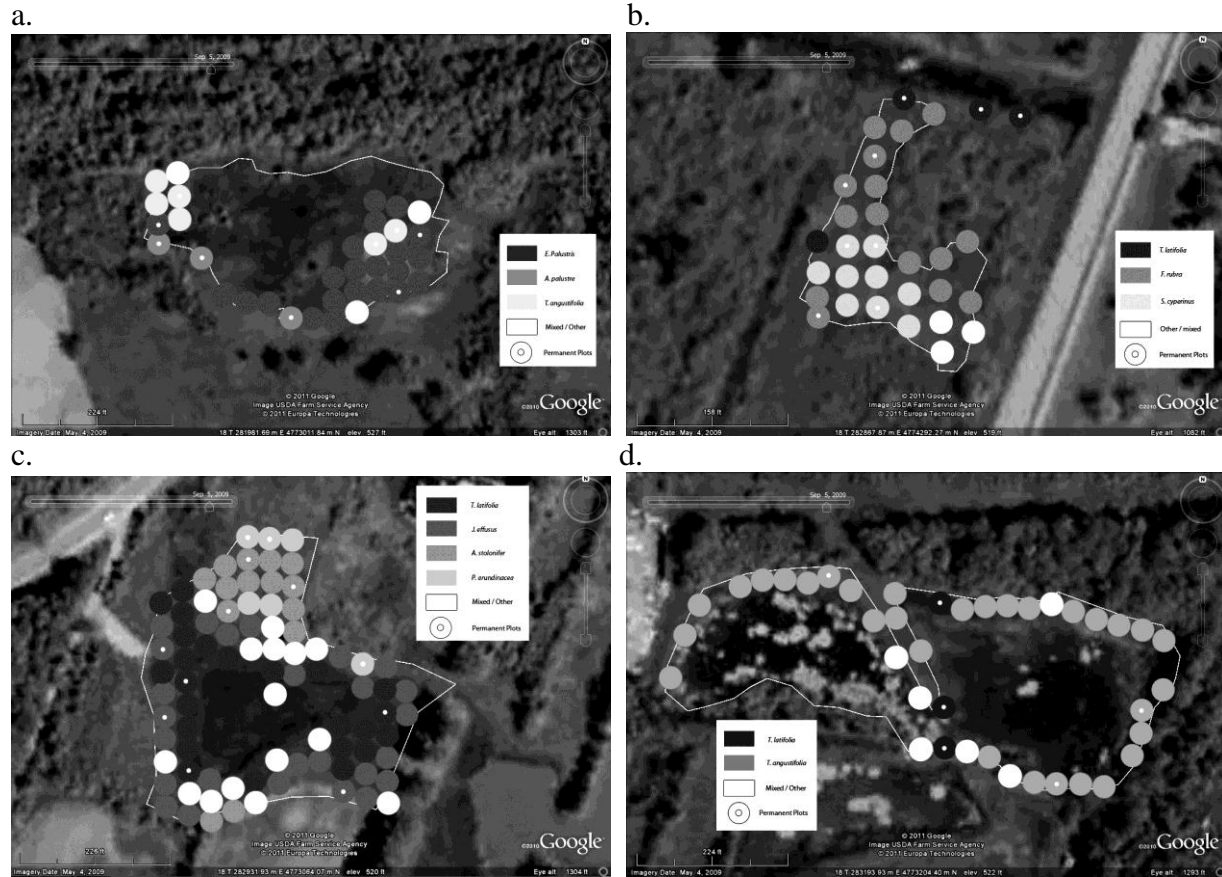
Figure 5. Distinct vegetation zones in all four study sites, N1, N2, C1 and C2 (a., b., c., and d. respectively). Vegetation zones were determined based upon surveys conducted in spring, 2010. Each plot is denoted by a 5 m buffer zone to repesent GPS error.
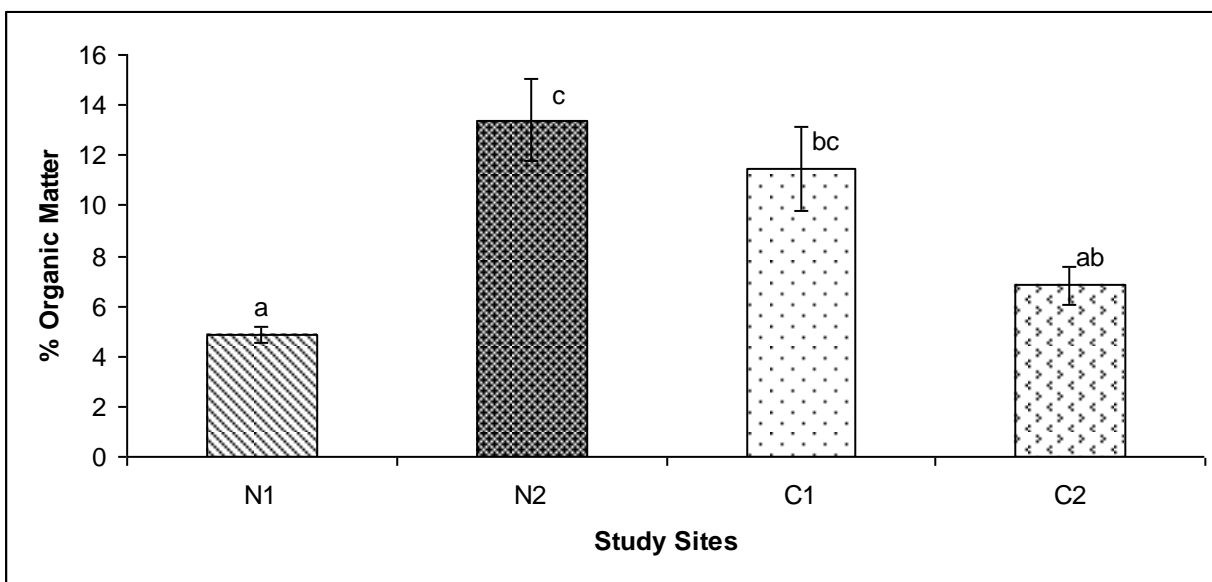
Figure 6. Mean spring and summer % soil organic matter for all four wetlands, ± SE. Identical lower case letters indicate statistical similarity.
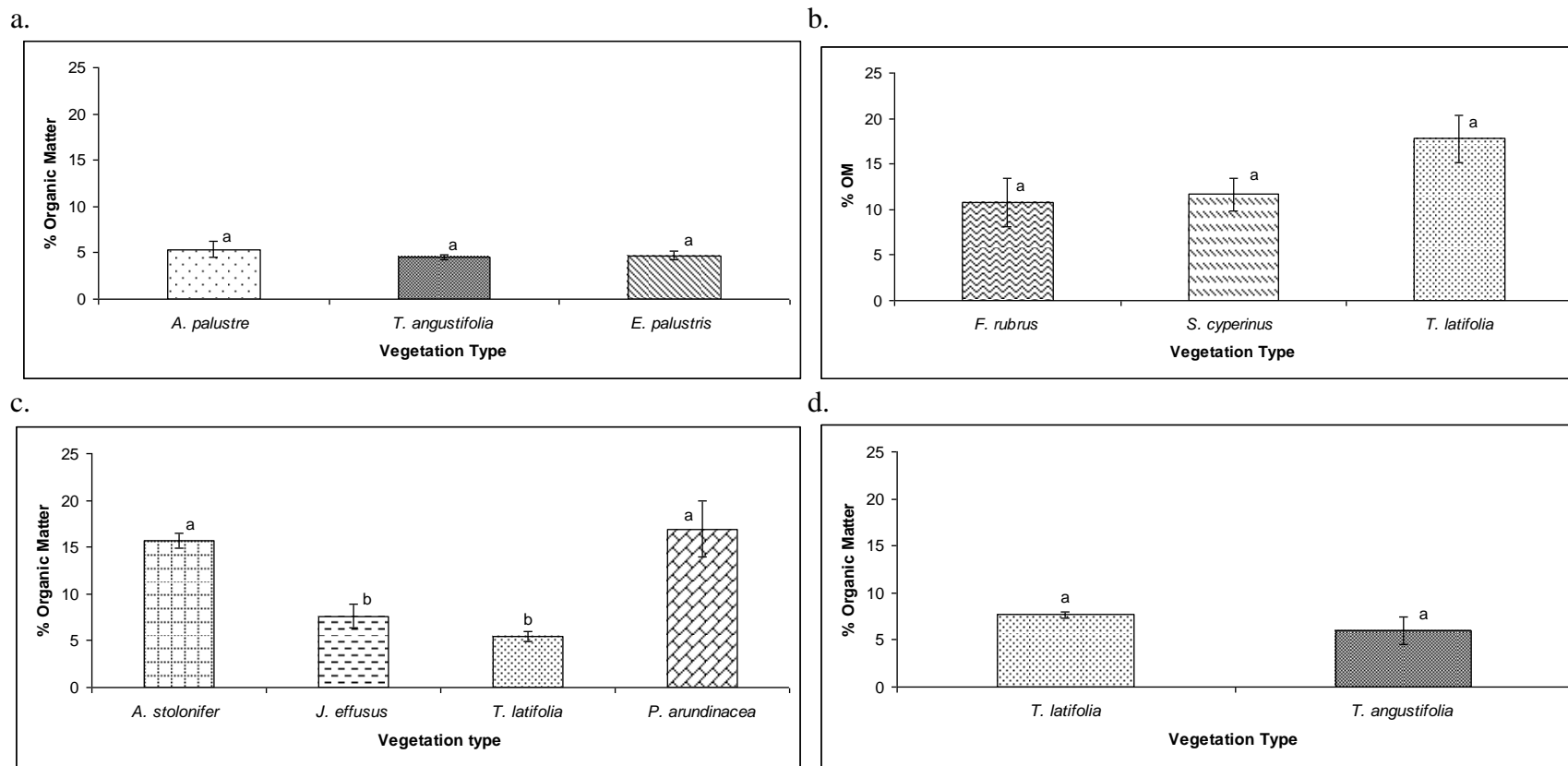
Figure 7. Mean spring and summer OM distribution ± SE in N1 (a), N2 (b), C1(c) and C2(d) among different vegetation zones. Identical lower case letters indicate statistical similarities within each wetland.

44
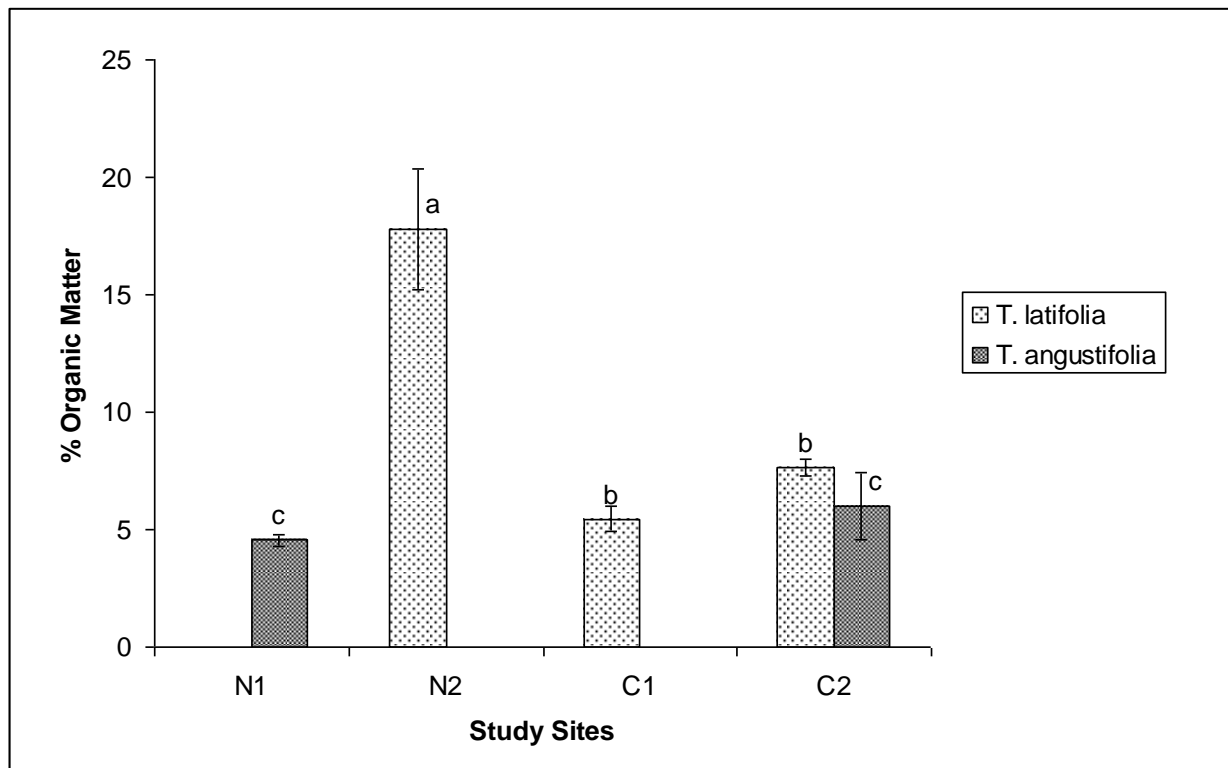
Figure 8. Mean spring and summer OM distribution ± SE in cattail plots between wetlands. Identical lower case letters indicate statistical similarities.

a.
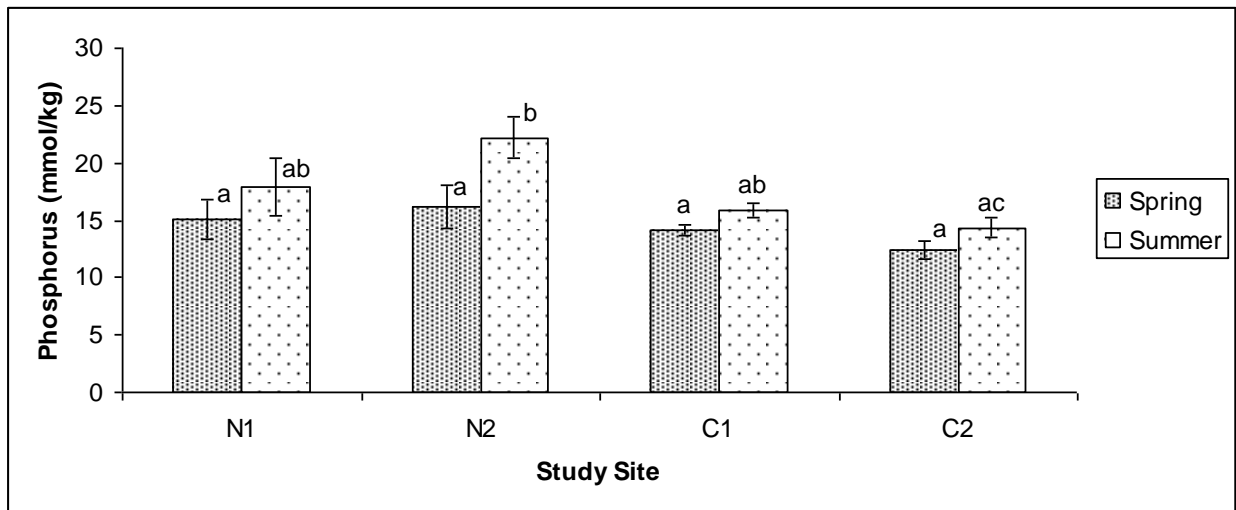
b.
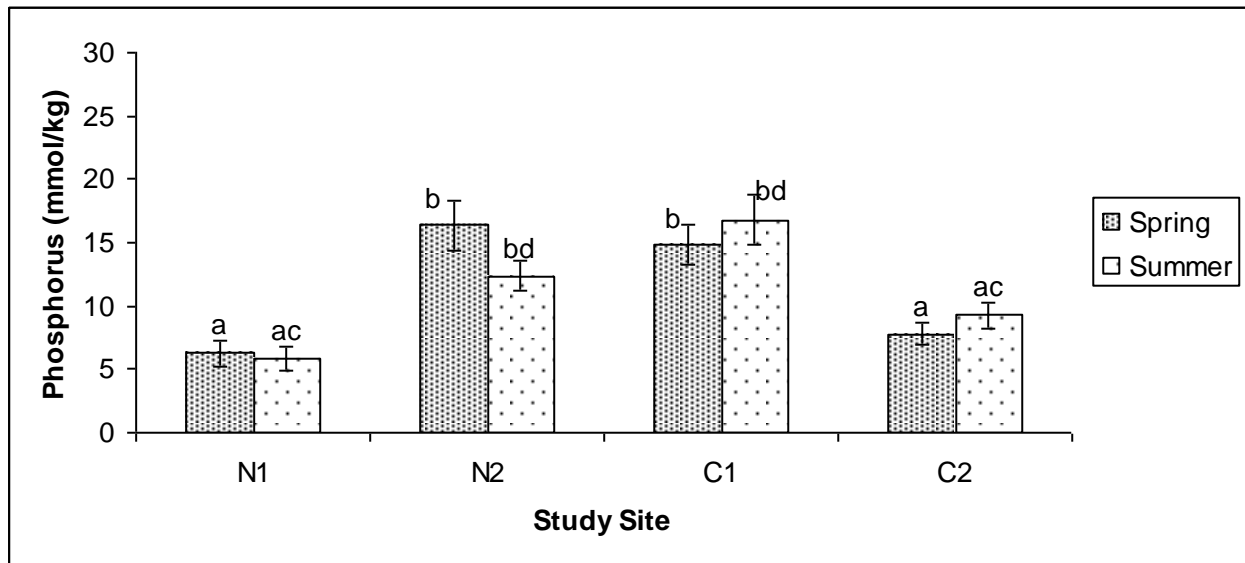
Figure 9. Spring and summer inorganic (a.) and organic (b.) phosphorus concentrations $\pm$ SE for all four study sites. Identical lower case letters indicate statistical similarities.
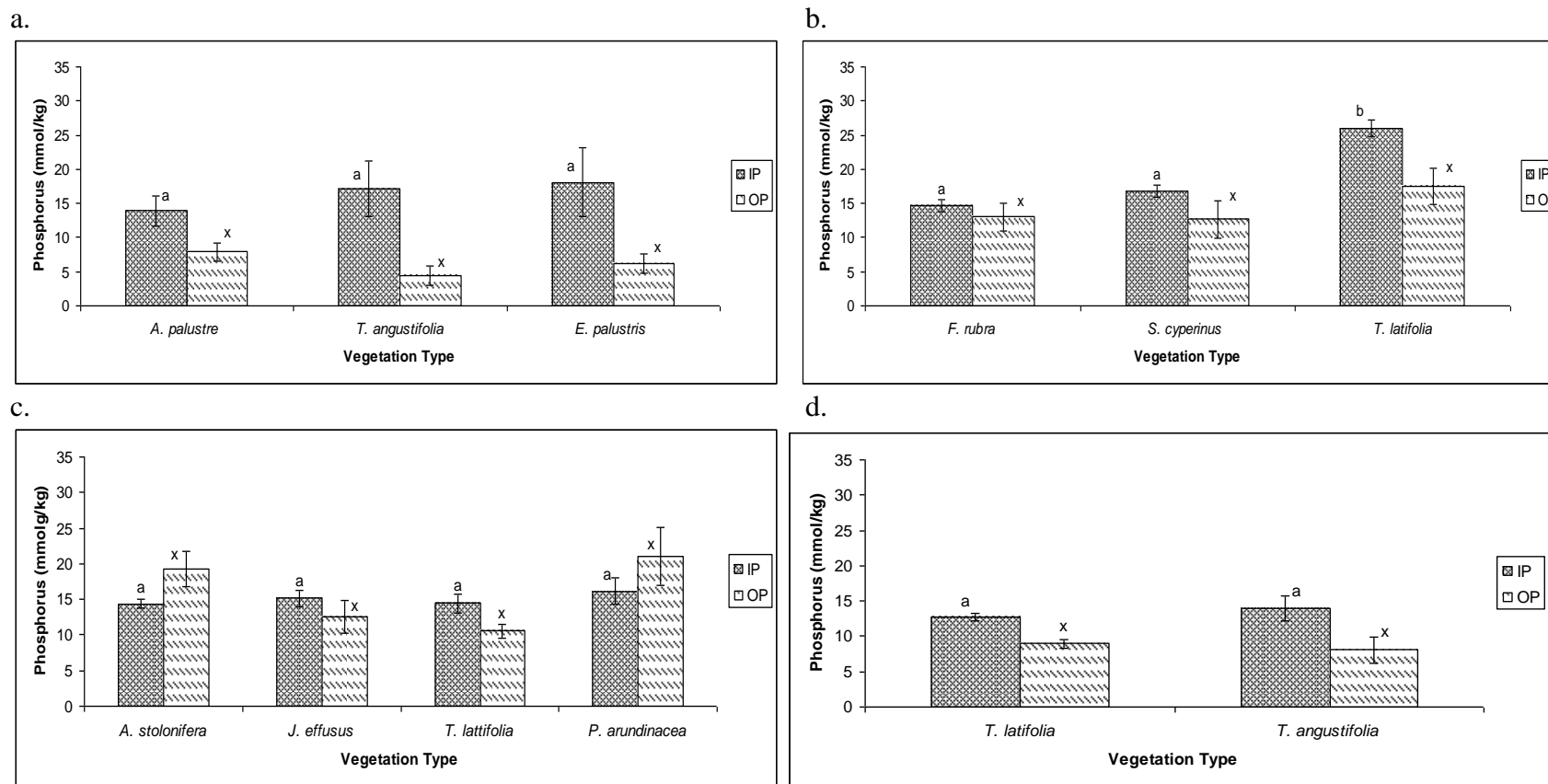
Figure 10. Pooled spring and summer inorganic and organic phosphorus (IP and OP, respectively) concentrations $\pm$ SE in N1 (a), N2 (b), C1 (c) and C2 (d) distributed between vegetation zones. Identical lower case letters indicate statistical similarities within each wetland, with different letter groupings (a,b vs x) for each IP and OP analyses.
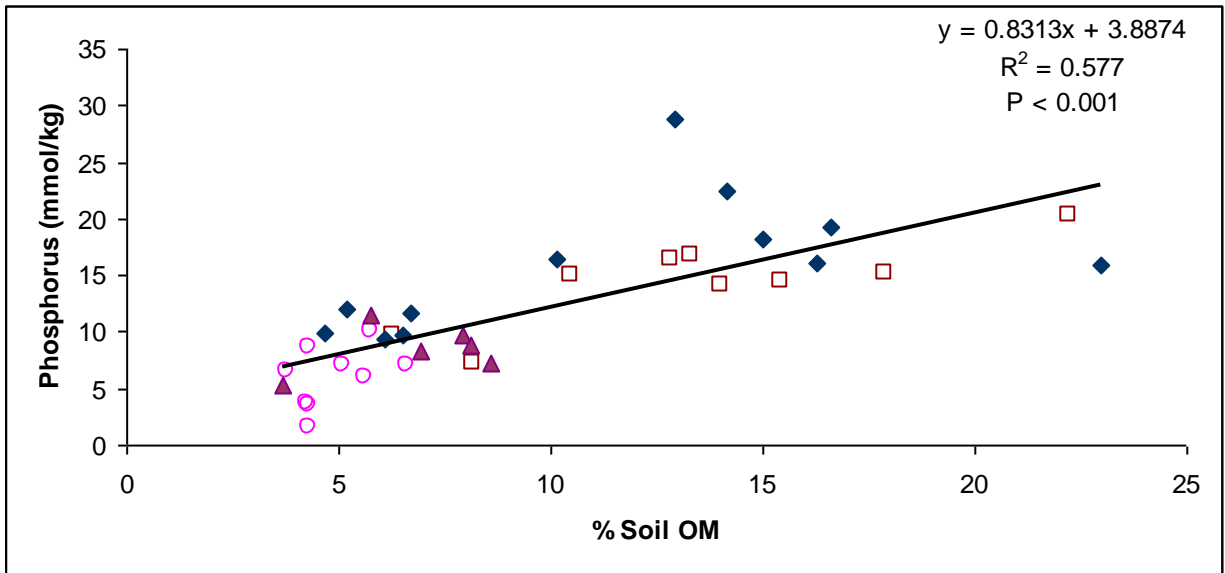
Figure 11. Mean spring and summer organic phosphorus concentrations for all wetlands vs mean spring and summer %OM for all wetlands. ○ = N1, □ = N2, ♦ = C1 and ▲ = C2. The trendline for all data combined is shown with the associated R^2 and equation.
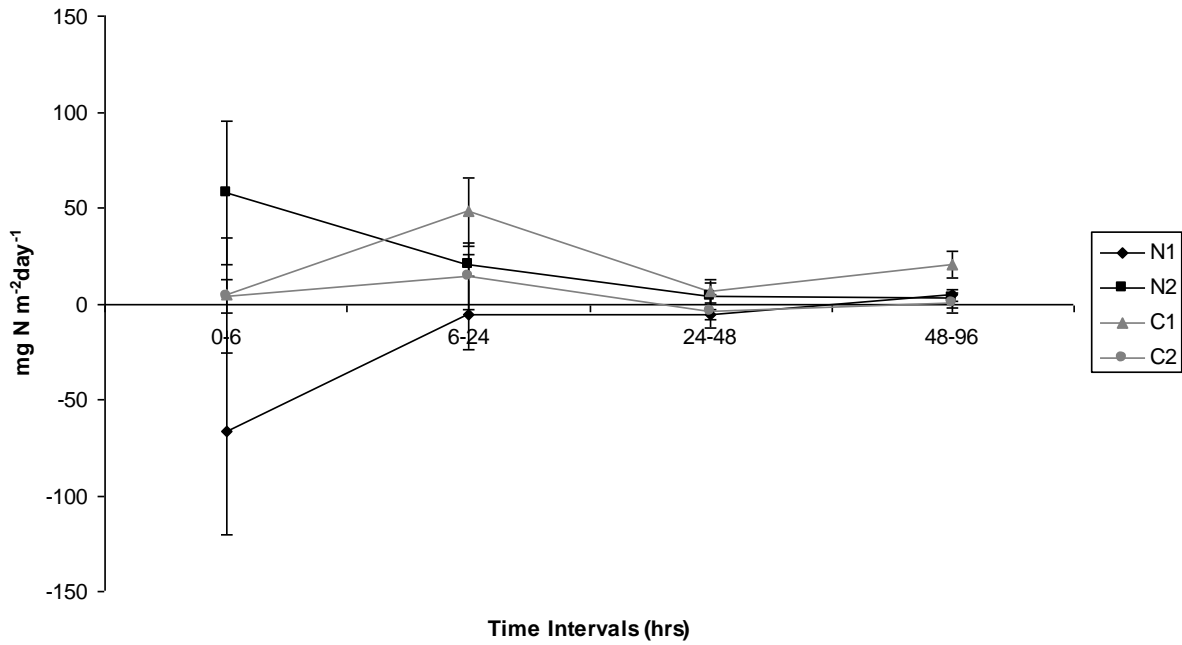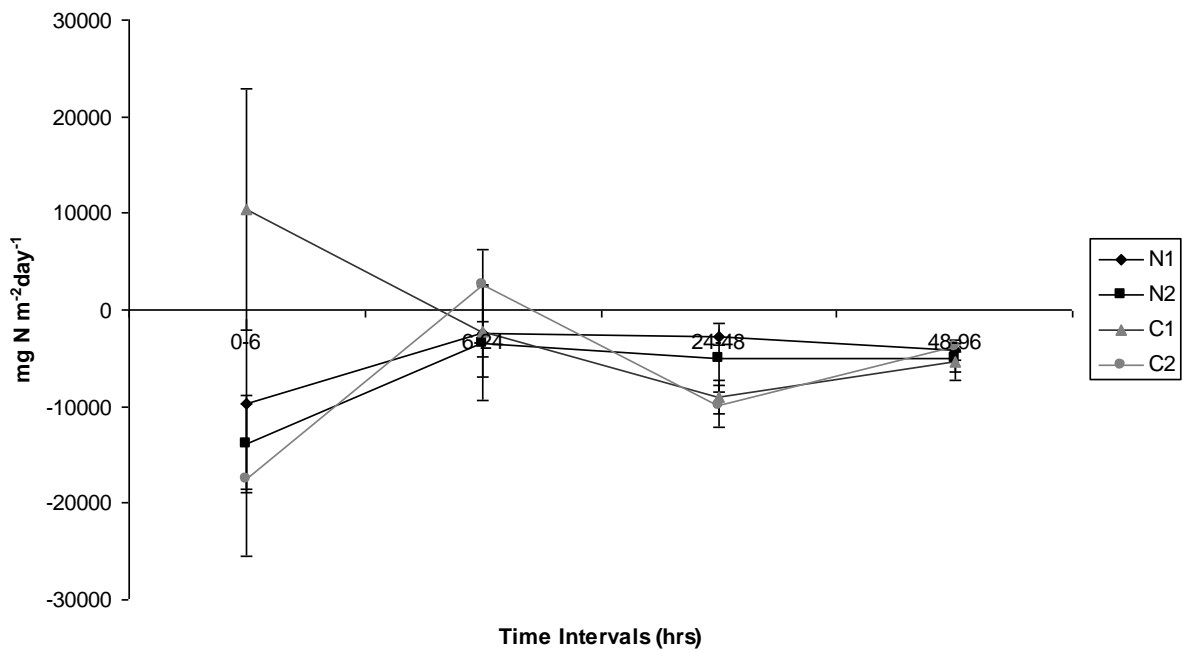
a.
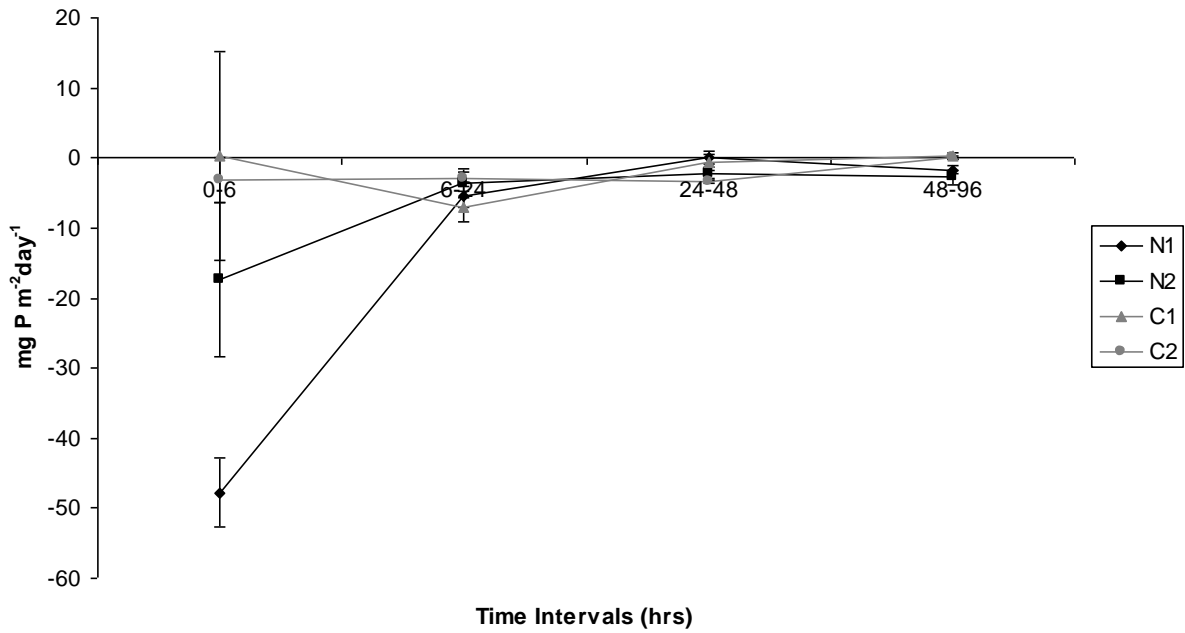
b.
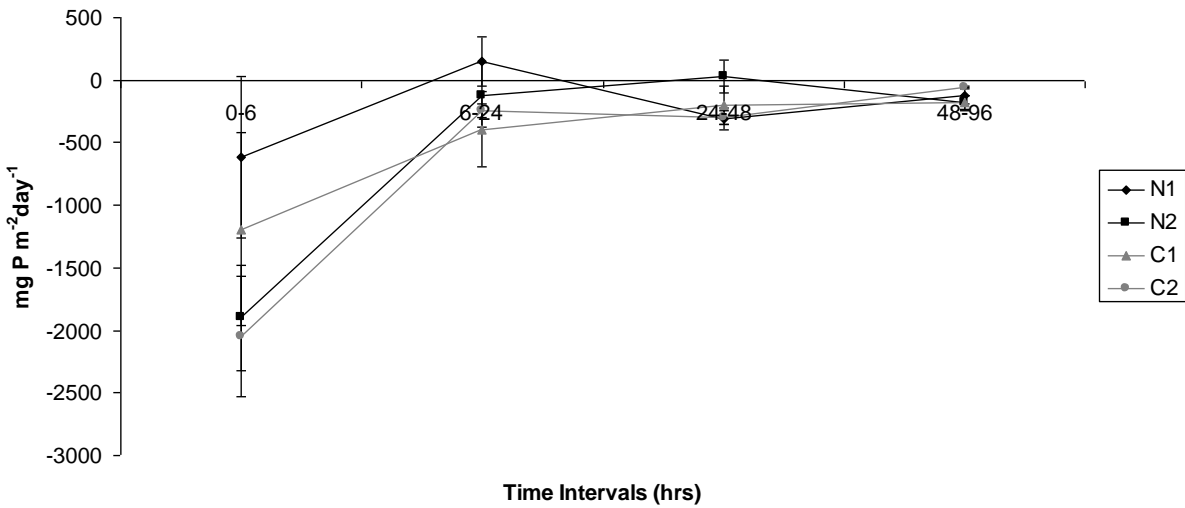
Figure 12. Mean phosphate flux rate for unspiked (a) and spike (b) cores ± SE. Note the difference in scales. Negative values indicate uptake of phosphate from the water column into the sediments; positive values indicate release from the sediment to the water column.
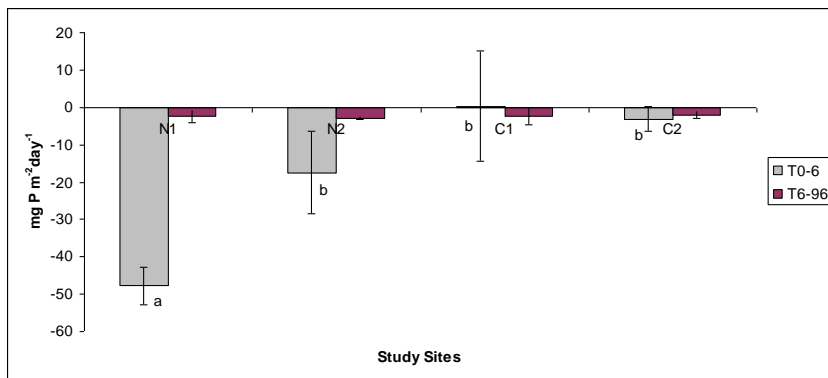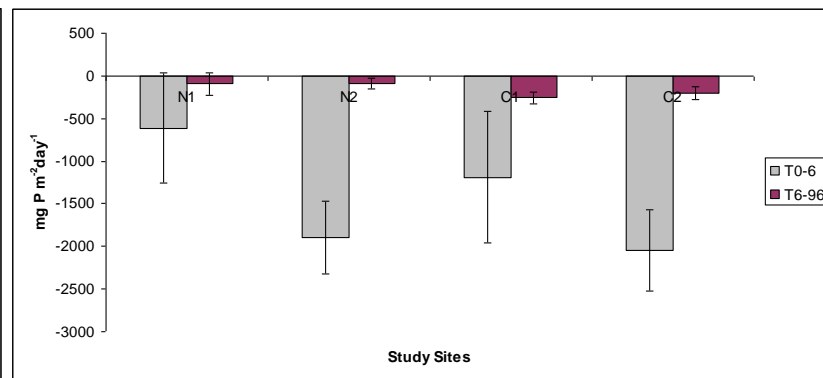
a

b.

Figure 13. Mean nitrate flux rate for unspiked (a) and spiked (b) cores ± SE. Note the difference in scales. Negative values indicate uptake of phosphate from the water column into the sediments; positive values indicate release from the sediment to the water column.
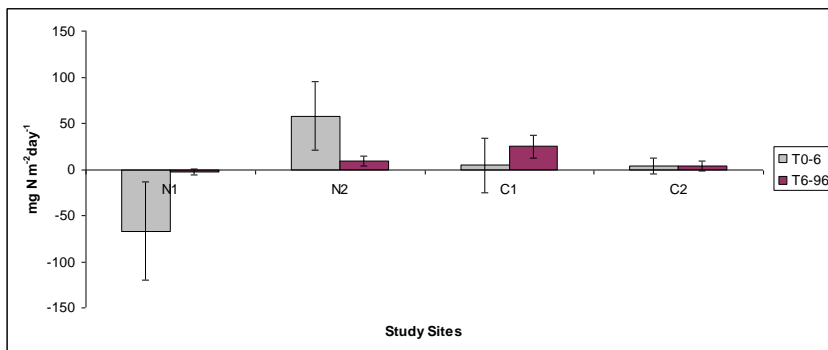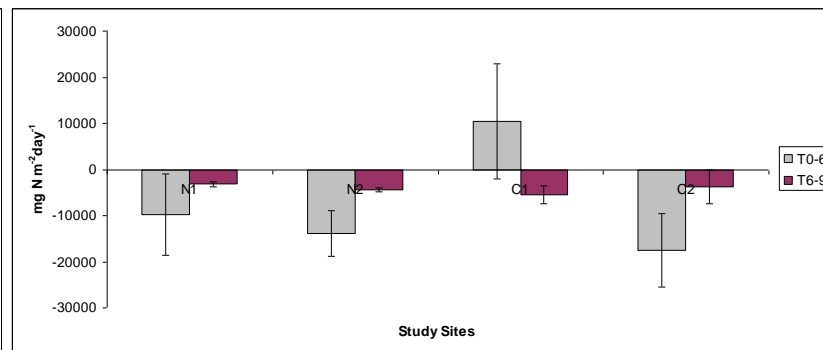
Figure 14. Mean phosphate flux rates ± SE for unspiked (a) and spiked (b) cores, and mean nitrate flux rates ± SE for unspiked (c) and spiked (d) cores among all study sites. Differing lower case letter indicate significant differences, a lack of letters indicates no significant differences among data sets. A positive rate value indicates a release of nutrients to the water column, while a negative rate value indicates an uptake of nutrients by the soil.

LITERATURE CITED

Anderson, Christopher J. and William J. Mitsch. 2006. Sediment Carbon and Nutrient Accumulation at two 10-year old Created Riverine Marshes. *Wetlands*. 26(3), p. 779-792.

Aspila, K. I, Haig Agemian and A. S. Y. Chau. 1976. A Semi-Automated Method for the Determination of Inorganic, Organic and Total Phosphate in Sediments. *Analyst*. 101, p. 187-197.

Atkinson, Robert B. and John Cairns Jr. 2001. Plant Decomposition and Litter Accumulation in Depressional Wetlands: Functional Performance of Two Wetland Age Classes That Were Created via Excavation. *Wetlands* 21(3), p. 354-362.

Bailey, David E., James E. Perry, and W. Lee Daniels. 2007. Vegetation Dynamics in Response to Organic Matter Loading Rates in a Created Freshwater Wetland in Southeastern Virginia. *Wetlands*. 27(4), p. 936-950.

Balcombe, Collin K., James T. Anderson, Ronald H. Fortney, James, S. Rentch, William N. Grafton, and Walter S. Kordek. 2005. A Comparison of Plant Communities in Mitigation and Reference Wetlands in the Mid-Appalachians. *Wetlands* 25(1), p. 130-142.

Bantilan-Smith, Meris, Gregory L. Bruland, Richard A. MacKenzie, Adonia R. Henry, and Christina R. Ryder. 2009. A Comparison of the Vegetation and Soils of Natural, Restored, and Created Coastal Lowland Wetlands in Hawai'i. *Wetlands*. 29 (3), p. 1023-1035.

Blackwell, M. S. A., D. V. Hogan, and E. Maltby. 2002. Wetlands As Regulators of Pollutant Transport. p. 321–39. In P. M. Haygarth and S. C. Jarvis (eds.) Agriculture, Hydrology, and Water Quality. CAB International, New York, NY, USA.

Brinson, Mark M. *et al*. 1981. Primary Productivity, Decomposition and Consumer Activity in Freshwater Wetlands. *Annual Review of Ecology and Systematics*. 12, p. 123-161.

Bruland, Gregory L. and Curtis J. Richardson. 2006. Comparison of Soil Organic Matter in Created, Restored, and Paired Natural Wetlands in North Carolina. *Wetlands Ecology and Management*. 14, p. 245-251.

Campbell, Deborah, Charles Andrew Cole, and Robert P. Brooks. 2002. A Comparison of Created and Natural Wetlands in Pennsylvania, USA. *Wetlands Ecology and Management*. 10, p. 41-49.

Clark, KR and Warwick, RM. 2001. Change in Marine Communities: An approach to statistical analysis and interpretation, 2nd edition. PRIMER-E, Plymouth.

Clean Water Act 33 U.S.C. §1251 et seq. 1977.

Clinton K. Meyer and Matt R. Whiles. 2008. Macroinvertebrate communities in restored and natural Platte River slough wetlands. *Journal of the North American Benthological Society* 27(3), p. 626-639.

Cowardin, L. M., V. Carter, F. C. Golet, E. T. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. U.S. Department of the Interior, Fish and Wildlife Service, Washington, D.C. Jamestown, ND: Northern Prairie Wildlife Research Center Online. http://www.npwrc.usgs.gov/resource/wetlands/classwet/index.htm (Version 04DEC1998)

Dahl, T. E. 1990. Wetland Losses in the United States:1780s to 1980s. U.S. Department of the Interior, Fish and Wildlife Service, Washington DC.

D'Angelo, Elisa M. 2005. Phosphorus Sorption Capacity and Exchange by Soils from Mitigated and Late Successional Bottomland Forest Wetlands. *Wetlands* 25(2), p. 297-305.

Davidson, Eric A. and Ivan A. Janssens. 2006. Temperature Sensitivity of Soil Carbon Decomposition and Feedbacks to Climate Change. *Nature*. 440, p. 165-173.

Eghball, Bahman and John E. Gilley. 1999. Phosphorus and Nitrogen in Runoff Following Beef Cattle Manure or Compost Application. *Journal of Environmental Quality* 28, p. 1201-1210.

EPA. Functions and Values of Wetlands. 2001. EPA 843-F-01-002c http://www.epa.gov/owow/wetlands/pdf/fun_val.pdf

EPA. Importance of Wetlands. 2009 http://www.epa.gov/bioiweb1/aquatic/importance.html

Gutrich, John J. and Fred J. Hitzhusen. 2004. Assessing the Suitability of Mitigation Wetlands for Natural Sites: Estimating restoration lag costs of wetland mitigation. *Ecological Economics*.48, p. 409-424.

Heaven, Jennifer B., Francesca E. Gross, and Andrew T. Gannon.. 2003. Vegetation Comparison of a Natural and a Created Emergent Marsh Wetland. *Southeastern Naturalist* 2(2), p. 195-206.

Hey, Donald L. and Nancy S. Philippi. 1995. Flood Reduction through Wetland Restoration: The Upper Mississippi River Basin as a Case History. *Restoration Ecology*. 3(1), p.4-17.

History of RIT. 2010. Rochester Institute of Technology http://www.rit.edu/overview/history.html

Hogan, Dianna M., Thomas E. Jordan, and Mark R. Walbridge.. 2004. Phosphorus Retention and Soil Organic Carbon in Restored and Natural Freshwater Wetlands. *Wetlands* 24(3), p. 573-585.

Hooper, D. U. F. S. Chapin III, J.J. Ewel, A. Hector, P. Inchausti, S. Lavorel, J.H. Lawton, D.M. Lodge, M. Loreau, S. Naeem, B. Schmid, H. Setala, A.J. Symstad, J. Vandermeer, and D.A. Wardle. 2005. Effects of Biodiversity on Ecosystem Functioning: A consensus of current knowledge. *Ecological Monographs* 75(1), p. 3-35.

Kirschbaum, Miko . F. 1994. The Temperature Dependence of Soil Organic Matter Decomposition, and the Effect of Global Warming on Soil Organic C Storage. *Soil Biology and Biochemistry*. 27(6), p. 753-760.

Lougheed, Vanessa L., Barb Crosbie and and Patricia Chow-Fraser. 2001. Primary Determinants in Macrophyte Community Structure in 62 Marshes Across the Great Lakes Basin: Latitude, land use, and water quality effects. *Canadian Journal of Fisheries and Aquatic Sciences*. 58(8), p. 1603-1612.

McMullen, Joseph M. 2007. *Corrective Action Plan*. Terrestrial Environmental Specialists.

Metcalfe, Janice L. Biological water quality assessment of running waters based on macroinvertebrate communities: History and present status in Europe. *Environmental Pollution*. 60(1-2), p. 101-139.

Mitra S, Wassman R, Vlek PLG. 2005. An Appraisal of Global Wetland Area and its Organic Carbon Stock. *Current Science*.88, p 25–35

Mitsch, William J., Julie K. Kronk, Xinyuan Wu, and Robert W. Nairn. 1995. Phosphorus Retention in Constructed Riparian Marshes. *Ecological Applications*. 5(3), p. 830-845.

Mitsch, W.J. and V. Bouchard. 1998. Enhancing the Roles of Coastal Wetlands of the North American Great Lakes. *Wetlands Ecology and Management* 6, p.1-3.

Mitsch, W.J. and Naiming Wang. 2000. Large-scale Coastal Wetland Restoration on the Laurentian Great Lakes: Determining the Potential for Water Quality Improvement. *Ecological Engineering*. 15, p. 267-282.

Mitsch, William.J. and Renee Wilson. 1996. Improving the Success of Wetland Creation and Restoration with Know-How, Time, and Self-Design. *Ecological Applications*. 6(1), p.77-83.

Odum, Eugene. 1969. The Strategy of Ecosystem Development. *Science*. 164(3877), p. 262-270.

Reddy, K. R., W. H. Patrick Jr., and C. W. Lindau. 1989. Nitrification-Denitrification at the Plant Root-Sediment Interface in Wetlands. *Limnology and Oceanography*. 34(6), p. 1004-1013.

Schatchman, Daniel P., Robert J. Reid, and S. M. Ayling. 1998. Phosphorus Uptake by Plants: From soil to cell. *Plant Physiology*. 116, p. 447-453.

Seabloom, Eric W. 2003. Plant Diversity, Composition and Invasion of Restored and natural Prairie Pothole Wetlands: Implications for restoration. *Wetlands*. 23(1), p. 1-12.

Shukla, M.K., R. Lal, and M. Ebinger. 2006.Determining Soil Quality Indicators by Factor Analysis. *Soil & tillage Research* 87, p. 194-204.

Smil, Vaclav. 2000. Phosphorus in the environment: Natural flows and human interferences. *Annual Review Energy Environment.* 25, p. 53-88

Smith, Fraser. 1995. Biological Diversity, Ecosystem Stability and Economic Development. *Ecological Economics*. 16, p.199-203.

Soil Survey Staff, 2010. Natural Resources Conservation Service, United States Department of Agriculture. Web Soil Survey. Available online at http://websoilsurvey.nrcs.usda.gov/ accessed 3/29/11.

Song, Keun Yea, Kyung-Duk Zoh, and Hojeong Kang. 2007. Release of Phosphate in a Wetland by Changes in a Hydrological Regime. *Science of the Total Environment*. 380 p. 13-18.

Spieles, Douglas J. and William J. Mitsch. 2000. Macroinvertebrate Community Structure in High and Low Nutrient Constructed Wetlands. *Wetlands*. 20(4), p. 716-729.

Stanczak, Marianne and Joe B. Keiper. 2004. Benthic Invertebrates in Adjacent Created and Natural Wetlands in Northeastern Ohio, USA. *Wetlands* 24(1), p.212-218.

Stevens, Michelle and Chris Hoag. 2011. Plant Guide: Narrowleaf cattail. *United States Department of Agriculture Natural Resources Conservation Services*. http://plants.usda.gov/plantguide/pdf/cs_tyan.pdf

Stewart, Timothy W. and John A. Downing. 2008. Macroinvertebrate Communities and Environmental Conditions in Recently Constructed Wetlands. *Wetlands*. 28(1), p. 141-150.

Stolt, Mark H., Michael H. Genthner, W. Lee Daniels, Velva A. Groover, Steven Nagle, and Katie C. Haering. 2000. Comparison of Soil and Other Environmental Conditions in Constructed and Adjacent Palustrine Reference Wetlands. *Wetlands*. 20(4), p. 671-683.

Terrestrial Environmental Specialists Inc. 2001. Wetlands Delineation Report RIT Campus Town of Henrietta Monroe County, New York.

Terrestrial Environmental Specialists Inc. 2002. Plan for Wetlands Creation Rochester Institute of Technology, Monroe County, New York.

Tilman, David. 1996. Biodiversity: Population Versus Ecosystem Stability. *Ecology* 77:350–363

Turner, R. Kerry, Jeroen C.J.M. van den Bergh, Tore Soderqvist, Aat Barendregt, Jan van der Straaten, Edward Maltby, Ekko C. van Ierland. 2000. Ecological-Economic Value of

Wetlands: Scientific integration for management and policy. *Ecological Economics*. 35, p. 7-23.

Verhoeven, J.T.A., W. Koerselman, and A.F.M. Meuleman.1996. Nitrogen- or Phosphorus-Limited Growth in Herbaceous, Wet Vegetation: Relations with atmospheric inputs and management regimes. *TREE*. 11(12), p. 494-497.

Whiting, Gary J. and Jeffrey P. Chanton. 2001. Greenhouse Carbon Balance of Wetlands: Methane Emission Versus Carbon Sequestration. *Tellus B*. 53(5), 521-528.

Wilcox, Douglas A. and Thomas H. Whillans. 1999. Techniques for Restoration of Disturbed Coastal Wetlands of the Great Lakes. *Wetlands*. 19(4), p. 835-857.

Zedler, Joy B. 2000. Progress in Wetland Restoration Ecology. *TREE* 15 (10), p. 402-407

Zedler, Joy B. 2004. Compensating for Wetland Losses in the United States. *Ibis* 146 (Suppl.1), p.92-100.

Zedler, Joy B. and Suzanne Kercher. 2005. Wetland Resources: Status, trends, ecosystem services and restorability. *Annual Review of Environmental Resources* 30, p.39-74.