8-6-2021

Carbon Soil Amendment for Promotion of Native Species in Created Emergent Freshwater Wetlands

Taylor E. Williams
tew1745@rit.edu

R I T

Carbon Soil Amendment for Promotion of Native Species in Created Emergent Freshwater Wetlands

By:

Taylor E. Williams

A thesis submitted in partial fulfillment of the requirement for the degree of Masters of Science in Environmental Science

Program in Environmental Science

Thomas H. Gosnell School of Life Sciences

College of Science

Rochester Institute of Technology

August 6, 2021

Committee Approval

Anna Christina Tyler, PhD — Date
Chair of Committee, Thesis Advisor

Carmody McCalley, PhD — Date
Committee Member

Elizabeth Hane, PhD — Date
Committee Member

ABSTRACT

Wetlands created or restored to replace natural wetland often fail to achieve functional equivalence and delivery of similar ecosystem services, especially with regard to plant community structure. In these young ecosystems, native emergent plants may not successfully compete with invasive species that are more rapid colonizers and capable of taking advantage of disturbed systems. This susceptibility to invasion and lack of resilience is exacerbated under conditions of high nutrient availability or extreme variations in weather as a result of climate change. Management of invasive plants in wetlands typically involves costly chemical and mechanical control measures, necessitating development of cost-effective and less environmentally damaging management practices to control invasive plants and ensure successful outcomes of restoration efforts. The addition of carbon to wetland soil has been previously shown to decrease nitrogen availability by promoting microbial processes that immobilize N. We hypothesized that by decreasing nitrogen availability through addition of organic matter, that we would increase the competitive ability of native species that are better adapted to low nutrient environments. To test this prediction, carbon, in the form of readily available leaf litter compost, was added to two created emergent freshwater wetlands in Perinton, NY that differ in hydrology. Compost was applied in large (2 x 30 m) transects and the vegetation community composition and soil characteristics monitored throughout the growing seasons of 2015 and 2016. At the wetter site, compost addition was crossed with manual removal of the dominant invasive plant, *Typha* sp. Weather patterns during the course of this study were variable, including a severe drought during the 2016 growing season. Compost amendments improved soil quality by increasing soil organic matter, soil moisture and bulk density and at the drier site increased the cover of native species. During the drought in 2016, the plant community shifted towards more resilient species, especially the invasive grass *P. arundinacea*, with compost addition providing a minor buffer for the extreme lack of precipitation. The seasonal and interannual variation observed in this study demonstrates the importance of frequent monitoring and implementation of multiple management strategies in created wetlands to ensure desirable outcomes, especially in light of the changing climate.

TABLE OF CONTENTS

FIGURES AND TABLES

Acknowledgements

I am deeply grateful to my advisor Dr. Christy Tyler for her continued help, patience and encouragement during my research and all my years at RIT. She inspired me to continue my education and without her unwavering support, I would not be in the position I am in today. I would also like to thank my committee members Dr. Carrie McCalley and Dr. Elizabeth Hane for offering perspective and guidance during my project.

I would like to thank the Thomas H. Gosnell School of Life Sciences and Waste Management for providing funding for my years at HANA. Also, I'd like to thank Bruce and Mary-Ann Cady for all their help and expertise while working at HANA. Their knowledge and passion for our projects helped keep me motivated.

I'd like to thank the amazing group of students that made all the worst, hot, muggy days spreading compost in wetlands actually enjoyable. Thanks to Kaity Moranz and Lizzy Bruen for showing me the ropes and driving us around blasting the radio in my early days at HANA. I'd also like to thank the students that helped me along the way including Sonia Huang, Rachel Bernard, Delanie Spangler, and Mike McGowan. A special thanks to Kim Lodge who not only accompanied me from day one for long days at HANA and in the lab, but also became a lifelong friend.

Lastly, thank you to my husband, who kept encouraging me through the most difficult days when I wasn't sure this would ever get done. And to my mom, for answering my frantic calls day or night and who showed me from a young age that you can accomplish anything you set your mind to.

1. Introduction

Wetlands: Services and Loss

Wetlands account for only five percent of all land cover in the coterminous United States (Cowardin et al. 1979, Dahl 1986), but are highly productive relative to other ecosystems and offer a multitude of ecosystem services, making them the most valuable per acre of all ecosystems (Costanza 1997, Costanza et. al. 2014). Among the key services provided by wetlands are habitat provision, water quality improvement, and hydrology (NRCS 2001). These important services are threatened by high rates of wetland loss and failed attempts at wetland creation and restoration. In the United States, only half of the original pre-European settlement wetland area remains due to high rates of disturbance caused by land use change that has led to draining, dredging or filling of wetlands for agriculture or development (Dahl 1990). In the absence of complete destruction, degradation occurs through hydrological alterations, nutrient pollution and colonization of invasive species. Directly linked with the degradation and loss of physical wetland area is a decline in the services they provide (Zedler and Kercher 2005).

Created Wetlands

In order to combat loss of wetlands, in 1989 a policy of "No Net Loss" was enacted which requires the restoration or creation of wetlands after the destruction of natural wetlands for development (CWA 1977). One mitigation option is to create wetlands in an area where none existed previously; the replacement wetland is required to mimic its natural counterpart both physically and functionally (Memorandum of Understanding, 1990). Although this is required by law, projects to create wetlands often fail due to poor construction and lack of post-construction monitoring (VanWinkle 2021).

Wetland creation projects typically begin by removing the topsoil layer and then excavating to the water table. The sediment at this layer often differs in texture, structure, chemistry, and biota from the original topsoil or from natural wetland soil, even when the topsoil layer is replaced (NRCS 2001). Soil organic matter in created wetlands is typically much lower than in natural reference wetlands (Campbell 2002). Biogeochemical functions such as nutrient cycling and decomposition suffer in newly created wetlands. Compared to natural wetlands, restored and created wetlands only recovered an average of 74% of their biogeochemical function (Moreno-Mateos et al. 2012). Low soil organic matter has been linked to lower vegetation establishment and growth, poor habitat for macroinvertebrates and altered nutrient cycling (Shafer & Ernst 1999). In a comparison study of created and natural wetlands, created wetlands had slower decomposition rates and soil carbon was up to four times lower, resulting in slowed cycling processes (Fennessy et al. 2008; Wolf et al. 2011). Sequestration of carbon, a major wetland service, is limited by poor soil quality in created systems where the estimated time for equal accumulation of carbon is 300 years (Bouchard, 2010).

Plant communities in created wetlands are also slow to recover and may never mimic the natural wetlands they replaced. In a review of 124 comparative studies on created versus reference wetlands, plants in created wetlands took on average 30 years to become statistically similar to reference sites (Moreno-Mateos et al. 2012). Succession in created wetlands is often unpredictable, oftentimes leading to a different wetland community structure than intended at the outset. The trajectory of succession is highly site-specific and depends on both the methods of construction and prior land use. Even though stability of the plant community in a created wetland can be achieved, the species within that community could be vastly different than reference sites, resulting in an altered ecosystem function (Matthews & Endress 2010). Overall, created wetlands are limited in their ability to provide the ecosystem services of the wetlands they replaced. Poor soil quality and nutrient cycling causes a decrease in biodiversity and may encourage the establishment of less desirable species.

Hydrology

In created wetlands, hydrology plays a critical role in resulting ecosystem function by modifying soil properties and biogeochemical cycling (Calhoun et al. 2014). The alterations to the hydrological regime can result in change in species composition and nutrient cycling, as has been observed in wetlands surrounding Lake Ontario, where historical periodic flooded and dry seasons limited dominant plant species' establishment, but regulating water levels in the lake resulted in extensive stands of these plants (Wilcox & Meeker 1995). Climate change poses additional complications to achieving goals of restoration due to unpredictable weather patterns.

Invasive Species in Created Wetlands

Created wetlands are especially prone to invasion, and 24% of the world's most invasive plants are wetland species (Zedler & Kercher 2004). Due to increased urbanization and agriculture in the watersheds of wetlands, nutrient loads have increased past the natural capacity of wetlands (Brinson & Malvárez 2002). Passing the N threshold can result in a change in the vegetation community within a wetland and dominance by generalist, N-limited, usually invasive species that form dense, impenetrable stands (Bedford et al. 1999). The change in heterogeneity of the vegetation landscape negatively affects biodiversity of other plant and animal species (Zedler and Kercher 2004). Monotypic stands of invasive plants reduce habitat complexity and available niches for native fauna, resulting in lower biodiversity (Weller & Spacher 1965, Keough et. al 1999).

A primary cause for invasion is the functional role of wetlands as stormwater runoff sinks for the surrounding watershed. Wetlands fed by surface water are even more likely to be invaded since water flowing into these types of wetlands can carry seeds, clippings or roots that establish easily in their new environment (Galatowitsch & Van der Valk, 1996). Directly, invasive species form dense monocultures that block native plants from colonizing, decreasing biodiversity of plant and eventually, of animal species. There is a possibility of positive feedback where invasive ability increases as species richness declines, making a wetland less and less suitable for native plants (Zedler & Kercher 2004).

Disturbance can also encourage the invasion of certain pest species in many types of ecosystems (Thompson et al. 2001). When compared to reference wetlands, disturbed environments had more widespread invasions. Once colonization by invasive species was initiated, fewer species of native plants were present and the other species that were found were of lower quality (Zedler & Kercher 2004). Creating wetlands is a disturbance in itself and after construction, invasive species often establish immediately. In a study of Canadian wetlands, it was found that invasive species are more likely to dominate a newly created wetland landscape and have a negative effect on the surrounding native community (Houlahan & Findlay, 2004). The most common invasive species in emergent freshwater wetlands in the northeastern United States can be exotic species like *Lythrum salicaria* (purple loosestrife) and *Typha angustifolia* (narrowleaf cattail), or native species like *Typha latifolia* (broadleaf cattail). Others are of unknown origin like *Phalaris arundinacea* (reed canary grass), from either an introduced European strain or a newer hybrid (Apfelbaum 1987, Zedler & Kercher 2004) or are formed as a hybrid between native and exotic species (*Typha* x glauca) (Frieswyk et al. 2007).

Created wetlands can also suffer from increased N loading as a result of nutrient-rich runoff accumulation. With lower rates of N immobilization than natural wetlands because of low organic matter (Fennessey et al. 2008), reactive N builds in these systems and promotes dominance by invasive plants that thrive in environments with high N availability (Blumenthal et al. 2003, Isbell et al. 2013, Moore et al. 1999). One of the more common invasive species in wetlands throughout North America, *P. arundinacea,* thrives in nitrate-enriched prairie pothole wetlands to the extent that native species biomass may be reduced by one-half (Green & Galatowitsch 2002). Created wetlands face intrusion of invasive species due to the combination of inherent disturbance, poor hydrology and excess nutrients. Together, these factors can threaten the success of created wetland projects suggesting that early management following creation could increase the rates of success.

Traditionally, management of invasive species is undertaken through chemical, mechanical and biological techniques. These traditional control methods are useful for controlling invasions but are costly and have potentially harmful ecosystem impacts. In 2000, the yearly cost to the United States for aquatic plant control was estimated at 145 million dollars (Pimentel et al. 2000). Chemical control utilizes herbicides to eradicate or limit the growth of invasive plants. This is a costly option, but among the most effective (Kettenring & Adams 2011). The effects of the most common herbicide, glyphosate, are still widely unknown but studies suggest negative effects on bees and create herbicide-resistant plant variants (Boily et al. 2013; Bonny 2008). Mechanical control includes cutting or hand pulling the entire plant or cutting off the inflorescence. The drawback of this method is the time and labor intensity, along with the risk of space left behind after removal that can allow for another invasion. Biological control has been successfully used for *L. salicaria* through introduction of *Galerucella calmariensis* (Loosestrife Beetle) that consumes the plant and reduces its ability to survive and reproduce (Mullin 1998). This method of control is not without risk, and is often avoided until extensive research is completed to ensure that the introduced species has no unintended effects.

There has been success in amending wetland soils to make them more suitable for native plants by adding carbon and increasing microbial activity including denitrification, which immobilizes nitrogen. This can promote the growth of plants, mostly native, that are adapted to survive in lower nitrogen environments (Alpert & Maron 2000, Ballantine et al. 2012, Kulmatiski 2011, Perry et al. 2004, Sutton-Grier et. al. 2009).

Development of a more cost-effective and less damaging management strategy for invasive wetland plants is needed. Because biogeochemical cycling controls nutrient availability, which in turn influences plant community dynamics, wetland restoration methods that intentionally alter biogeochemical cycles in the soil could provide a successful and cost-effective method for invasive plant control. C in the forms of topsoil and biochar increases soil C which in turn may enhance heterotrophic microbial activity (Ballantine et al. 2012). The addition of carbon to wetland soil has been previously shown to decrease nitrogen availability. C fuels the immobilization of N by supplying heterotrophic denitrifying bacteria with an organic matter source. When carbon was experimentally added to prairie plots, invasive plant growth was reduced, and native species thrived (Blumenthal et al. 2003). In a greenhouse experiment evaluating competition between native wetland sedge (*Carex hystericina*) and *P. arundinacea*, N availability was lowered using C enrichment. *C. hystericina* was able to outcompete the invasive *P. arundinacea* when the soil was depleted of N (Perry et al. 2004).

Although studies have shown carbon soil amendments alone may reduce invasive plant dominance, combining control techniques could be the answer to prolonged control. In a review study of control themed studies of invasive species, Kettenring and Adams (2011) suggest that when combined, control methods were shown to decrease invasive plant growth more than when alone. For example, in a study looking to prevent *P. arundinacea* reinvasion, it was found that after the removal of the invasive grass, if native grass species were able to colonize, *P.arundinacea* reinvasion was reduced (Iannone & Galatowitsch, 2008). Although isolated studies have demonstrated the potential for carbon addition as a potential restoration tool, large scale, long-term field experiments using practical, commercially available materials are rare (Ballentine et al. 2012, Kettenring & Adams 2011). If carbon addition in the form of accessible, inexpensive leaf litter compost could lead to similar outcomes, it could be incorporated into management strategies.

In this study, I evaluated the potential of a new method of deterring invasive species growth while simultaneously promoting native growth by manipulating resource availability. By amending wetland soil with carbon in the form of leaf litter compost and implementing other traditional control methods, I tested the hypothesis that increasing soil organic matter improves soil quality and alters biogeochemical cycles so that competition between denitrifying bacteria and plants decreases invasive and stimulates native species growth.

The overarching goal of this study was to experimentally determine whether the amendment of created wetland soil with organic carbon was an effective management strategy for the control of invasive species in freshwater wetlands. The findings of this study will inform future management and creation processes in terms of the importance of soil development in successful outcomes. The specific objectives of this study were (1) to determine whether large scale C addition, in the form of leaf litter compost, will maximize native diversity and curb invasive spread in both a large-scale field experiment and a greenhouse experiment, and (2) to evaluate the effectiveness of combining C addition with opening the canopy (by cutting stems of invasive species) on native species cover. We hypothesized that (1) manipulating resource availability through the addition of C would suppress the growth of invasive plants and support that of native for a more diverse community and (2) the effect of C addition would be greater when combined with a more open canopy.

2 Methods

2.1 *Site Description*

This study took place in two created wetlands at the 101 ha High Acres Nature Area (HANA), in Perinton, New York, USA. HANA, which consists of forests, grasslands, created and natural freshwater wetlands, vernal pools and ponds (Figure 1) is owned and managed by Waste Management of New York, LLC (hereafter, WM). An expansion of the adjacent landfill was approved in 2008 and in ordinance with the Clean Water Act and "No Net Loss of Wetlands" requirements the subsequent loss of existing wetlands required the creation of wetlands of equal size and ecological value. The proposed loss of 8 acres of natural scrub/shrub, wooded and emergent wetlands was mitigated with the creation of 6.9 ha of new wetlands at HANA between 2009 and 2012 (Stantec 2009).

The Cady wetlands were created in 2009 and converted 1.8 ha of early successional agricultural field into a forested wetland, an emergent wetland, and eight vernal pools. The wooded and emergent wetland cells are connected by two culverts that run beneath an east-west berm and walking trail; water drains from the northern forested wetland to the emergent wetland to the south (Stantec 2009) (Figure 2). The study took place in the emergent wetland, where in spite of efforts to control invasive plants, the area was rapidly and persistently colonized by emergent wetland species, including *Typha* spp. Standing water was typically present year-round, and the water depth varies seasonally, and the southern half is somewhat deeper than the northern half. Plant species composition at the start of the experiment in 2014 included dominant cover by invasive species, including *T. latifolia*, *T. angustifolia* and *P. arundinacea*, as well as partial cover of native species like *Alisma subcordatum* (American Water Plantain) and *Bidens cernuus* (Nodding Beggar-Ticks).

The second site, the Packard Wetlands, was created in 2012. Prior to creation, the site consisted of upland and an existing pond used to pasture and water cows (Figure 1). The Packard site consists of rehabilitated wet meadow, emergent marsh, forested wetland, and transitional forest and scrub/shrub buffer totaling 1.5 ha (Stantec 2009). This experiment took place in the southeastern wetland cell, intended as forested wetland but at the time of the study was dominated by emergent wetland and wet meadow species, including *Eupatorium perfoliatum* (Common Boneset), *Scirpus validus* (Bulrush) and *A. subcordatum* (American Water Plantain).

Hand-pulling of early colonizing *Typha* spp. seedlings was conducted during spring and summer of 2013, shortly after creation of the wetland, but persistent invasive species like *Typha* spp. and *Lythrum salicaria* (Purple Loosestrife) were common. (Figure 3). The site is flooded in spring, drying completely by early summer. There are dominant stands of *Typha spp*. and *P. arundinacea* in both study sites, so herbicide applications occurred but avoided the area with experimental plots. When spraying, the inflorescences of *Typha* spp. and the base of *P. arundinacea* individuals were sprayed in both areas in mid-summer and fall of 2014, 2015 and 2016.

2.2 *Experimental Design*

In July 2014, five zones, each containing one pair of 30 m x 2 m transects (T1 – T10) separated by a 1 m buffer, were established in Cady (Figure 2). Transects were arranged in a paired block design within zones to accommodate potential spatial differences in water depth across the site. The north-south oriented transects were marked in each corner and at the 15 m midpoint with 1.5 m polyvinyl chloride (PVC) pipe. Within each transect, eight 1 m x 1 m permanent sampling plots were randomly established throughout each transect and marked in the northwest corner using 1 m PVC pipe. Vegetation data was collected in all permanent plots (n=8 per transect) and soil sampling was conducted in odd numbered plots only (n=4 per transect). In May 2015, ten transects in Packard were established using the same design (Figure 3).

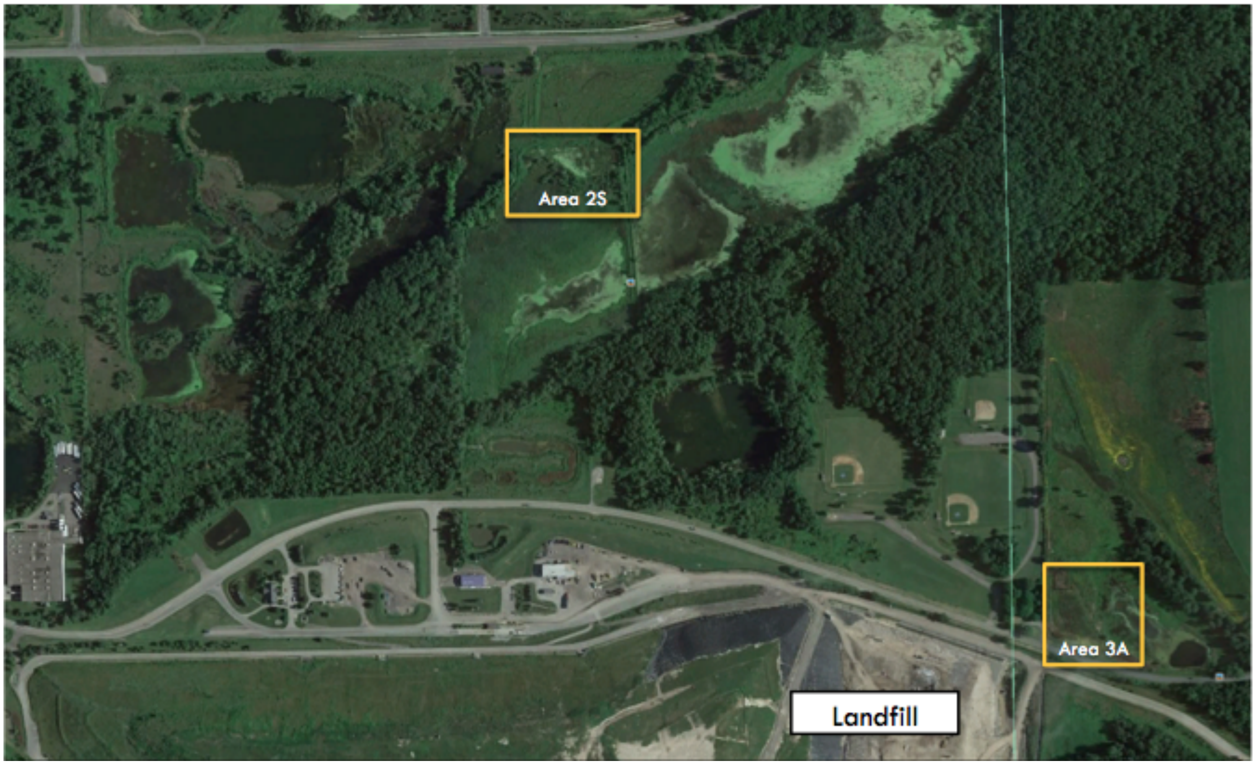

Figure 1. Map of High Acres Nature Area and Landfill in Perinton NY. Experimental sites are highlighted.

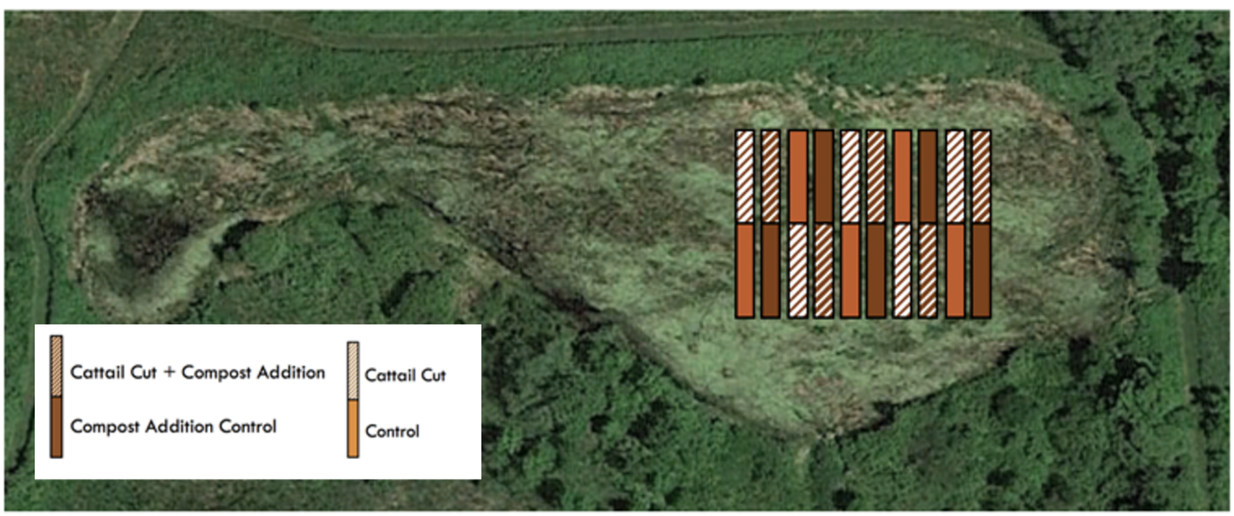

Figure 2. Experimental design at the Cady wetland.

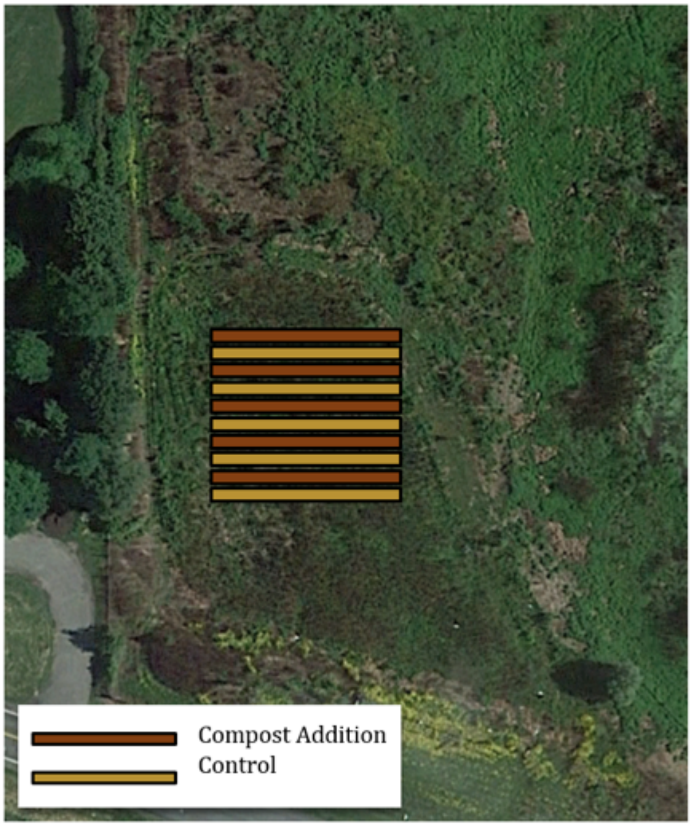

Figure 3: Experimental design at the Packard wetland.

Leaf litter was collected in fall by Waste Management of New York, LLC, from the surrounding community and composted for 1.5 yr in managed rows that were periodically turned. To each of the five odd-numbered transects, approximately 5 cm of leaf litter compost was applied: in June 2014 and May 2015 and 2016 to the Cady wetland, and May 2105 and 2016 to Packard. The C:N molar ratio of compost was ~ 18.7 C:N (28% C, 1.8% N). With plant cover taken into account, the total area of each plot covered by compost was estimated at 50 ± 4%, leading to a supplement of about 2.0 kg cm^{-2} and 0.13 kg N m^{-2} (McGowan 2020).

In 2015, prior to compost addition, cattails at Cady only were cut below the water level in half of each transect to evaluate the combined impact of organic matter addition and mechanical control. To accommodate for small differences in elevation, cutting was conducted in the same half of the transects in each zone: the northern 15 m in Zones 1, 3 and 5 and the southern Zones 2 and 4 (Figure 2). Cattails were left intact in all transects at the Packard site.

2.3 *Evaluating Impact of Management Techniques*

At both sites, vegetation percent cover of each species was recorded in the eight vegetation plots by two observers in July 2014 (Cady only), and in May, July and September of 2015 and 2016 at both sites. Two observers made independent assessments of cover and consensus was reached for each value. Total cover may exceed 100% due to varying plant heights and overlap within the canopy. Species richness was calculated based on the number of species present and diversity was evaluated using the Shannon-Wiener Diversity Index. Water depths were taken from the NW and SE corner of each plot during the vegetation monitoring when standing water was present.

In 2016, prior to compost addition in the spring and again in the fall, two soil cores per plot (2.5 cm x 10 cm deep; 4 per transect) were taken for evaluation of soil organic matter, bulk density, and extractable inorganic nitrogen using a 60 cc syringe corer. These were transported on ice and then frozen at – 20°C until lab analysis. In 2016, pH was measured prior to freezing. Soil pH was measured using a Hach pH probe after mixing soil in a 2:1 (V:V) slurry of deionized water to soil (Gelderman & Mallarino 2012). Soil moisture and bulk density were determined by weighing the sample before and after oven drying at 60°C for 24 hr. Using the same sample, organic matter was measured by loss on combustion at 540°C for four hr (Blake and Hartge, 1986; Heiri et al., 2001). The second soil sample was used for measurement of extractable ammonium and nitrate after KCl extraction. Briefly, the core was split into two 0.5 g samples. Dry weight was obtained for one and the second was mixed with 50 mL of 2M KCl, shaken for 45 min and centrifuged (2000 RPM) for 10 min (Lachat 2003). Ammonium was measured using the phenol-hypochlorite method (Maynard 2008) and nitrate using the cadmium reduction method on a Lachat Quickchem 8500 autoanalyzer (Lachat 2003). Extractable ammonium and nitrate were combined as total inorganic nitrogen (TIN) for statistical analyses and presentation of data. Belowground biomass was measured at the end of the experiment in September 2016 to compare between sites. One core (10 cm diameter x 25 cm depth) was taken just outside of each control soil sampling plot using a soil auger. The samples were washed through a 1 mm mesh sieve to remove soil, and the remaining roots and rhizomes dried at 60°C and weighed.

2.4 *Greenhouse Experiment*

A greenhouse experiment was conducted to further evaluate the effects of carbon addition on invasive species alone and in combination with a common native species. Invasive *T. latifolia* and *P. arundinacea* seedlings were collected from the field and native *A. subcordatum* (American water plantain) was obtained from Southern Tier Consulting, West Clarksville, NY in late June 2016. Each species was planted alone with two seedlings in each pot (*T. latifolia, P. arundinacea and A. subcordatum*) or in paired combinations with one seedling of each species (*T.latifolia x A. subcordatum* and *P. arundinacea x A. subcordatum*).

Seedlings were replanted in 3.78 L plastic planters lined with plastic bags and containing 1.89 L of wetland soil from the Cady wetland mixed with sand at a ratio of 1:9. Like the field experiment, 5 cm of leaf litter compost was added to the surface of half of the pots. Shade cloth was used to minimize extreme temperatures in the greenhouse, and plants were watered every 1-2 days to maintain soil saturation.

Plant heights were measured weekly for four weeks. At the conclusion of the experiment, plants were clipped at the soil level, dried at 60°C and weighed for aboveground biomass. Soil samples were collected for analysis of pH, extractable N and soil organic matter, as described previously. Remaining soil was sieved and roots extracted, dried at 60°C and weighed. Belowground biomass was not separated by species in mixed species pots.

2.5 *Statistical Analysis*

All statistical analyses were performed using JMP 15 Pro. Data were checked prior to analysis, and if not normally distributed, were log transformed to conform to the assumptions of parametric statistical analyses. The alpha level for all analyses was $\alpha = 0.05$. A one-way ANOVA across all seasons in 2016 was used to evaluate site differences in control plots for soil organic matter, total inorganic nitrogen, soil bulk density, pH, soil moisture, and water depth. Total invasive cover, total native cover, species richness, and Shannon-Wiener diversity were evaluated for site differences across 2015 and 2016 for control plots only. Belowground biomass from fall 2016 was also evaluated between sites for control plots.

Soil properties measured in 2016 were evaluated for each site individually using either two-way or three-way ANOVA with season (spring or fall), compost treatment, and cattail removal (Cady only) as fixed factors and a two-way ANOVA with season (spring and fall) and treatment as fixed factors. Zone was included as a random factor to account for potential spatial variability within each site. Biomass and pH, measured one time only, were compared using one-way ANOVA with treatment as the fixed factor. To capture the seasonal dynamics of plant growth and dominance, the plant variables listed above and cover, stem density and height of select invasive plants, were evaluated for each season separately, using ANOVA with year (2015 or 2016), compost treatment and cattail removal treatment (Cady only) as fixed factors and block as a random factor. When significant differences were found, Tukey's HSD post hoc test was performed to further evaluate differences.

For the greenhouse experiment, soil properties, and final plant height and biomass were compared between control and compost amended pots within each species, using t-tests.

3. Results

3.1 *Field Manipulation*

3.1.1. *Hydrology*

The Cady wetland was permanently flooded in 2015 and 2016, drying by fall 2016 due to extreme drought in the region. Water depths ranged from 28 cm during the spring in the south to no standing water in the north at the height of the summer. In contrast, the Packard wetland was seasonally wet and dried by the second seasonal vegetation survey in July both years. Overall, the water level in Cady was significantly higher than Packard ($p<0.001$) (Table 1). In Cady, water depth measured in compost addition plots was significantly shallower ($p=0.04$) than plots with no compost added (Table 3). No standing water was left in either wetland by September 2016.

Table 1. Cross site comparison of control treatments in Cady and Packard. Mean ± SE water depth (cm), soil moisture (%), organic matter (%), total inorganic nitrogen (mg/kg), soil pH, bulk density (g cm^3), belowground biomass (g), species richness, Shannon-Weiner diversity index, invasive plant cover (%), and native plant cover (%) of control plots in each wetland. Results of a one-way ANOVA with significant values highlighted.

	Cady	Packard	F	p
Water depth	4.7 ± 1.6	0.0 ± 0.0	$F_{1,159} = 19.47$	**< 0.0001**
Soil moisture	32 ± 3	47 ± 2	$F_{1,159} = 15.05$	**0.0003**
Organic matter	15.0 ± 1.2	12.3 ± 0.3	$F_{1,159} = 9.84$	**0.0027**
Total inorganic nitrogen	657 ± 194	600 ± 92	$F_{1,159} = 0.09$	0.8
Soil pH	7.60 ± 0.09	8.36 ± 0.02	$F_{1,159} = 116.62$	**< 0.0001**
Bulk density	0.8 ± 0.1	0.9 ± 0.0	$F_{1,159} = 2.84$	0.1
Belowground biomass	15.9 ± 2.8	4.8 ± 0.8	$F_{1,159} = 26.99$	**<0.0001**
Species richness	3.5 ± 0.2	5.2 ± 0.1	$F_{1,319} = 49.16$	**< 0.0001**
Shannon-Weiner	0.8 ± 0.0	1.2 ± 0.0	$F_{1,319} = 51.28$	**< 0.0001**
Invasive plant cover	30.7 ± 2.4	17.5 ± 1.2	$F_{1,319} = 30.80$	**< 0.0001**
Native plant cover	16.1 ± 2.4	26.7 ± 1.7	$F_{1,319} = 12.99$	**< 0.0004**

3.1.2. *Impacts on soil characteristics*

Prior to the compost addition, soil organic matter (OM) was similar between sites. At Cady wetland, OM was 12.7 +/- 0.6% and 12.2 +/- 0.3% at Packard wetland. The comparison of control plots showed significantly higher organic matter and belowground biomass in Cady than in Packard (p=0.0003, p<0.0001) (Table 1). Total inorganic nitrogen was higher in Cady, yet not significant. Soil moisture and soil pH were significantly higher at Packard than Cady (p=0.003, p=<0.0001) (Table 1). Compost addition increased OM by roughly 3% in both areas, although this was dependent on season with higher values at the end of the growing season (Season x Compost interaction, $p < 0.01$ for both sites; Tables 2, 3, 4 and 5). There was no impact of cattail cutting on OM.

Bulk density (BD) was not significantly different between sites (p=0.1; Table 1), but was significantly higher in compost addition plots and in the spring at both areas (Season x Compost interaction, $p < 0.05$ for both sites; Tables 2,3,4, and 5). Soil moisture was higher in spring and with the addition of compost at both sites (Season and Compost main effects, $p < 0.05$ for both sites; Tables 2, 3, 4, and 5). Soil TIN was similar between sites and increased with compost addition at both sites (Table 1), with greater values at Cady in September (1.2 ± 0.1 g N/kg) compared to May (0.2 ± 0.01 g N/kg). At Packard, there was a similar seasonal effect, with September (0.93 ± 0.09 gN/kg) greater than May (0.66 ± 0.09 gN/kg), echoing the increase in OM.

Table 1. Cross site comparison of control treatments in Cady and Packard. Mean ± SE water depth (cm), soil moisture (%), organic matter (%), total inorganic nitrogen (mg/kg), soil pH, bulk density (g cm^3), belowground biomass (g), species richness, Shannon-Weiner diversity index, invasive plant cover (%), and native plant cover (%) of control plots in each wetland. Results of a one-way ANOVA with significant values highlighted.

		Cady					
		TIN	OM	BD	SM	WD	pH
Spring	Control	143 ± 16	15.5 ± 1.8	0.6 ± 0.0	41.4 ± 2.0	9.4 ± 2.3	7.3 ± 0.0
	Compost	194 ± 32	16.8 ± 1.8	0.5 ± 0.0	45 ± 2.8	6.5 ± 0.9	7.2 ± 0.0
	Cattail Cut	172 ± 35	13.1 ± 1.3	0.6 ± 0.0	37.6 ± 2.1	10.1 ± 1.4	7.3 ± 0.0
	Cattail Cut + Compost	173 ± 18	14.5 ± 1.0	0.5 ± 0.0	43.6 ± 2.3	6.7 ± 1.5	7.3 ± 0.0
Fall	Control	1172 ± 305	14.5 ± 1.5	1.0 ± 0.1	22.8 ± 2.2	0 ± 0	7.9 ± 0.1
	Compost	1651 ± 439	20.2 ± 1.0	0.7 ± 0.0	31.5 ± 1.4	0 ± 0	8.1 ± 0.1
	Cattail Cut	550 ± 141	12.9 ± 0.8	0.9 ± 0.1	23.0 ± 2.5	0 ± 0	8.0 ± 0.1
	Cattail Cut + Compost	1321 ± 302	21.9 ± 1.8	0.6 ± 0.0	27.4 ± 2.5	0 ± 0	8.3 ± 0.1

Table 3. Results of a two-way ANOVA examining the effect of season (Spring, Summer, Fall) and treatment on soil characteristics total inorganic nitrogen (TIN), organic matter (OM), bulk density (BD), soil moisture (SM), water depth (WD) and soil pH in Cady wetland. Significant p-values are bolded.

	Cady					
	TIN	**OM**	**BD**	**SM**	**WD**	**pH**
Season	$F_{1,72}$=38.95	$F_{1,72}$= 5.51	$F_{1,72}$= 71.22	$F_{1,72}$= 94.66	$F_{1,72}$= 108.07	$F_{1,72}$= 423.77
	<0.0001	**0.02**	**<0.001**	**<0.0001**	**<0.0001**	**<0.0001**
Compost	$F_{1,72}$= 4.11	$F_{1,72}$=18.85	$F_{1,72}$ =36.48	$F_{1,72}$=12.43	$F_{1,72}$= 4.16	$F_{1,72}$= 4.97
	0.045	**<0.0001**	**<0.001**	**0.0008**	**0.04**	**0.03**
Season x Compost	$F_{1,72}$= 3.47	$F_{1,72}$= 8.96	$F_{1,72}$=14.13	$F_{1,72}$= 0.28	$F_{1,72}$= 4.16	$F_{1,72}$= 13.80
	0.06	**0.0038**	**0.0004**	0.60	**0.04**	**0.0004**
Cut	$F_{1,72}$= 2.11	$F_{1,72}$= 1.33	$F_{1,72}$= 0.18	$F_{1,72}$= 1.81	$F_{1,72}$= 0.07	$F_{1,72}$= 4.11
	0.15	0.25	0.67	0.18	0.78	0.05
Season x Cut	$F_{1,72}$= 2.23	$F_{1,72}$= 1.46	$F_{1,72}$= 3.55	$F_{1,72}$= 0.05	$F_{1,72}$= 0.08	$F_{1,72}$= 1.35
	0.14	0.23	0.06	0.82	0.77	0.24
Compost x Cut	$F_{1,72}$= 0.13	$F_{1,72}$= 0.56	$F_{1,72}$= 0.07	$F_{1,72}$= 0.11	$F_{1,72}$= 0.02	$F_{1,72}$= 0.09
	0.72	0.46	0.78	0.74	0.87	0.76
Season x Compost x Cut	$F_{1,72}$= 0.29	$F_{1,72}$= 0.64	$F_{1,72}$= 0.12	$F_{1,72}$= 1.09	$F_{1,72}$= 0.02	$F_{1,72}$= 0.13
	0.59	0.43	0.73	0.29	0.87	0.71

Table 4. Soil characteristics measured in Packard wetland in control, compost treatments. Mean ± SE total inorganic nitrogen (TIN, mg/kg), organic matter (OM, %), bulk density (BD, g/cm^3), soil moisture (SM, %), pH.

		Packard				
		TIN	**OM**	**BD**	**SM**	**pH**
Spring	**Control**	553 ± 166	12.3 ± 0.4	0.7 ± 0.0	56.2 ± 3.4	8.3 ± 0.0
	Compost	761 ± 139	12.1 ± 0.6	0.7 ± 0.0	59.5 ± 2.7	8.2 ± 0.0
Fall	**Control**	659 ± 85	12.3 ± 0.4	1.1 ± 0.0	37.7 ± 0.9	8.4 ± 0.0
	Compost	1204 ± 125	18.4 ± 0.6	0.9 ± 0.0	46.0 ± 1.9	8.6 ± 0.0

Table 5. Results of a two-way ANOVA examining the effect of season (Spring, Summer, Fall) and treatment on soil characteristics total inorganic nitrogen (TIN), organic matter (OM), bulk density (BD), soil moisture (SM), and soil pH in Packard wetland. All values for water depth (WD) were zero. Significant p-values are bolded.

	Packard				
	TIN	**OM**	**BD**	**SM**	**pH**
Season	$F_{1,76}$= 4.24	$F_{1,76}$= 40.54	$F_{1,76}$= 106.76	$F_{1,2}$= 45.57	$F_{1,2}$= 42.63
	0.04	**<0.001**	**<0.001**	**<0.0001**	**<0.0001**
Compost	$F_{1,76}$= 8.52	$F_{1,76}$= 36.02	$F_{1,76}$= 24.69	$F_{1,76}$= 6.09	$F_{1,76}$= 0.70
	0.004	**<0.001**	**<0.001**	**0.02**	0.70
Season x Compost	$F_{1,76}$ = 1.76	$F_{1,76}$= 42.10	$F_{1,76}$ = 6.31	$F_{1,76}$= 1.10	$F_{1,76}$= 16.20
	0.18	**<0.001**	**0.014**	0.30	**0.0001**

3.1.3 *Impacts on plant communities*

Native plant cover, species richness and plant diversity were significantly higher at Packard than Cady (p=0.0004) (Table 1), with roughly two-fold greater invasive plant cover in control plots at Cady (31 ± 2 %) than in Packard (18 ± 2%) (F=30.8, p=<0.001) (Table 1). The three dominant invasive species evaluated at Cady were *P. arundinacea, T. latifolia* and *L. salicaria,* and *T. latifolia, T. angustifolia* and *L. salicaria* at Packard (Figures 4, 5). Predictably, all plant characteristics varied seasonally, with peak values in July or September, depending on the growth cycle of each species. Belowground biomass was significantly higher in the control plots at Cady than Packard (p<0.0001) (Table 1). The two sites differed in plant species composition and measures of diversity. Packard had a significantly more diverse and rich species composition as well as significantly higher native cover and less invasive cover than Cady wetland (Table1).

At Cady, native species cover was significantly higher during the 2015 growing season in contrast to the dryer 2016 season (p<0.0001) (Table 7, Figure 4). There was a slight interaction p=0.049 between year and compost treatment in the spring at Cady, reflecting the slightly greater native cover in 2016 in the presence of compost, and a moderate increase in native cover with cattail cutting (Table 7, Figure 4). Cutting cattails increased species richness (Table 7) yet, the opening in the canopy allowed for *P. arundinacea*, the dominant invasive plant, to move into the open area (p<0.0001) (Figure 4). In Summer 2015 where cattails were cut, *P. arundinacea* cover was highest (27 ± 9%), but with the addition of compost, was reduced slightly (20 ± 10%) and cover was lower in compost alone plots (8 ± 3%) compared to the control (10 ± 2%) (Figure 4). In the combined management plots, cover was higher (25 ± 9%) than cutting cattails alone (24 ± 5%) (Figure 1). *T. latifolia* cover was not significantly affected by compost addition alone but when cut in the spring, cover remained significantly lower than the control (Figure 4). *L. salicaria* cover was significantly higher in plots where cattail was cut at Cady, but overall cover was low and heterogeneous (p=0.02) (Table 7). Overall, in Cady, invasive species percent cover was higher in areas where cattails were cut (p=0.04) (Table 7).

Table 6. Plant characteristics measured in Cady wetland each season. Mean ± SE species richness (S), Shannon Diversity Index (H), invasive species cover (Inv %) and native species cover (Nat %).

		Cady							
		2015				2016			
		Control	Compost	Cattail Cut	Compost + Cattail Cut	Control	Compost	Cattail Cut	Compost + Cattail Cut
Spring	S	5.0 ± 0.5	2.2 ± 0.2	2.2 ± 0.2	2.4 ± 0.1	1.8 ± 0.1	2.3 ± 0.1	2.5 ± 0.3	2.9 ± 0.4
	H	0.4 ± 0.1	0.5 ± 0.1	0.5 ± 0.1	0.6 ± 0.1	0.4 ± 0.1	0.6 ± 0.1	0.6 ± 0.1	0.8 ± 0.2
	Inv %	13.3 ± 2.4	13.2 ± 2.6	22.9 ± 6.3	14.6 ± 5.7	13.2 ± 1.8	9.4 ± 1.4	13.1 ± 1.6	14.8 ± 1.7
	Nat %	4.9 ± 3.0	1.8 ± 0.6	3.2 ± 1.4	2.2 ± 1.1	0.9 ± 0.4	0.7 ± 0.4	1.3 ± 0.5	4.4 ± 1.6
Summer	S	4.4 ± 0.4	4.4 ± 0.4	5.0 ± 0.5	5.0 ± 0.4	4.8 ± 0.3	5.0 ± 0.2	6.2 ± 0.4	6.2 ± 0.6
	H	1.2 ± 0.1	1.1 ± 0.1	1.1 ± 0.1	1.2 ± 0.1	1.0 ± 0.1	1.1 ± 0.1	1.2 ± 0.1	1.3 ± 0.1
	Inv %	33.0 ± 4.6	48.3 ± 7.7	39.5 ± 12.2	27.7 ± 8.8	32.5 ± 5.9	43.1 ± 5.3	35.3 ± 8.1	34.9 ± 11.9
	Nat %	39.7 ± 6.4	39.0 ± 5.6	31.6 ± 4.9	43.8 ± 11.0	10.6 ± 4.6	8.9 ± 1.5	15.7 ± 2.9	18.6 ± 3.7
Fall	S	4.7 ± 0.5	5.1 ± 0.6	5.6 ± 0.2	5.3 ± 0.2	3.6 ± 0.5	4.0 ± 0.3	3.9 ± 0.3	5.2 ± 0.3
	H	1.2 ± 0.1	1.3 ± 0.1	1.2 ± 0.1	1.3 ± 0.1	0.9 ± 0.2	1.1 ± 0.1	0.8 ± 0.1	1.0 ± 0.1
	Inv %	46.2 ± 4.0	51.5 ± 2.0	54.7 ± 10.6	42.4 ± 10.6	35.0 ± 3.6	35.8 ± 5.1	46.9 ± 5.5	50.2 ± 6.7
	Nat %	36.2 ± 6.7	41.7 ± 6.6	42.7 ± 7.2	50.4 ± 10.7	6.1 ± 1.9	8.6 ± 0.8	13.1 ± 2.9	22.4 ± 5.4

Table 7. Results of a three-way ANOVA examining the effect of year (2015, 2016), season (Spring, Summer, Fall) and treatment on species richness (S), Shannon Diversity Index (H), invasive species cover (Inv %) and native species cover (Nat %) in Cady wetland. Significant p-values are bolded.

		Cady						
		Year	**Compost**	**Year x Compost**	**Cattail Cut**	**Year x Cattail Cut**	**Compost x Cattail Cut**	**Year x Compost x Cattail Cut**
Spring	**S**	$F_{1,152}$= 1.82	$F_{1,152}$= 4.32	$F_{1,152}$= 0.67	$F_{1,152}$= 11.6	$F_{1,152}$= 5.00	$F_{1,152}$= 0.34	$F_{1,152}$= 0.54
		0.18	**0.04**	0.41	**0.0008**	**0.03**	0.55	0.47
	H	$F_{1,152}$= 3.52	$F_{1,152}$= 6.62	$F_{1,152}$= 0.17	$F_{1,152}$= 8.33	$F_{1,152}$= 0.60	$F_{1,152}$=0.08	$F_{1,152}$= 0.36
		0.06	**0.01**	0.67	**0.005**	0.43	0.92	0.55
	Inv %	$F_{1,152}$= 3.36	$F_{1,152}$= 1.75	$F_{1,152}$= 0.61	$F_{1,152}$= 4.19	$F_{1,152}$= 0.55	$F_{1,152}$= 0.08	$F_{1,152}$= 3.22
		0.07	0.18	0.43	**0.04**	0.46	0.77	0.07
	Nat %	$F_{1,152}$= 2.36	$F_{1,152}$= 0.25	$F_{1,152}$= 3.91	$F_{1,152}$= 0.91	$F_{1,152}$= 2.18	$F_{1,152}$= 2.02	$F_{1,152}$= 0.09
		0.13	0.62	**0.049**	0.34	0.14	0.16	0.76
Summer	**S**	$F_{1,152}$= 8.47	$F_{1,152}$= 0.01	$F_{1,152}$= 0.04	$F_{1,152}$=11.2	$F_{1,152}$= 2.04	$F_{1,152}$= 0.18	$F_{1,152}$= 0.14
		0.004	0.91	0.84	**0.001**	0.16	0.67	0.71
	H	$F_{1,152}$= 0.04	$F_{1,152}$= 2.8	$F_{1,152}$= 1.0	$F_{1,152}$= 2.69	$F_{1,152}$= 2.44	$F_{1,152}$= 0.03	$F_{1,152}$= 0.44
		0.84	0.74	0.32	0.10	0.12	0.84	0.51
	Inv %	$F_{1,152}$= 0.05	$F_{1,152}$= 1.11	$F_{1,152}$= 0.15	$F_{1,152}$= 6.36	$F_{1,152}$= 0.43	$F_{1,152}$= 2.76	$F_{1,152}$= 0.85
		0.82	0.3	0.69	0.06	0.51	0.9	0.36
	Nat %	$F_{1,152}$= 41.9	$F_{1,152}$= 0.32	$F_{1,152}$= 0.35	$F_{1,152}$= 0.87	$F_{1,152}$= 1.17	$F_{1,152}$= 0.72	$F_{1,152}$= 0.23
		<0.0001	0.57	0.55	0.35	0.28	0.40	0.63
Fall	**S**	$F_{1,152}$= 13.77	$F_{1,152}$= 1.88	$F_{1,152}$= 2.11	$F_{1,152}$= 6.24	$F_{1,152}$= 0.10	$F_{1,152}$= 0.05	$F_{1,152}$= 2.37
		0.0003	0.17	0.14	**0.013**	0.75	0.83	0.13
	H	$F_{1,152}$= 18.65	$F_{1,152}$= 3.59	$F_{1,152}$= 0.35	$F_{1,152}$= 0	$F_{1,152}$= 0.49	$F_{1,152}$= 0.56	$F_{1,152}$= 0.18
		<0.0001	0.06	0.55	0.99	0.49	0.46	0.67
	Inv %	$F_{1,152}$= 2.56	$F_{1,152}$= 0.02	$F_{1,152}$= 0.42	$F_{1,152}$= 0.33	$F_{1,152}$= 2.65	$F_{1,152}$= 0.01	$F_{1,152}$= 0.77
		0.11	0.96	0.51	0.56	0.11	0.97	0.38
	Nat %	$F_{1,152}$= 53.09	$F_{1,152}$= 1.50	$F_{1,152}$= 0.03	$F_{1,152}$= 5.32	$F_{1,152}$= 0.03	$F_{1,152}$= 0.03	$F_{1,152}$= 0.24
		<0.0001	0.22	0.95	**0.022**	0.86	0.86	0.61

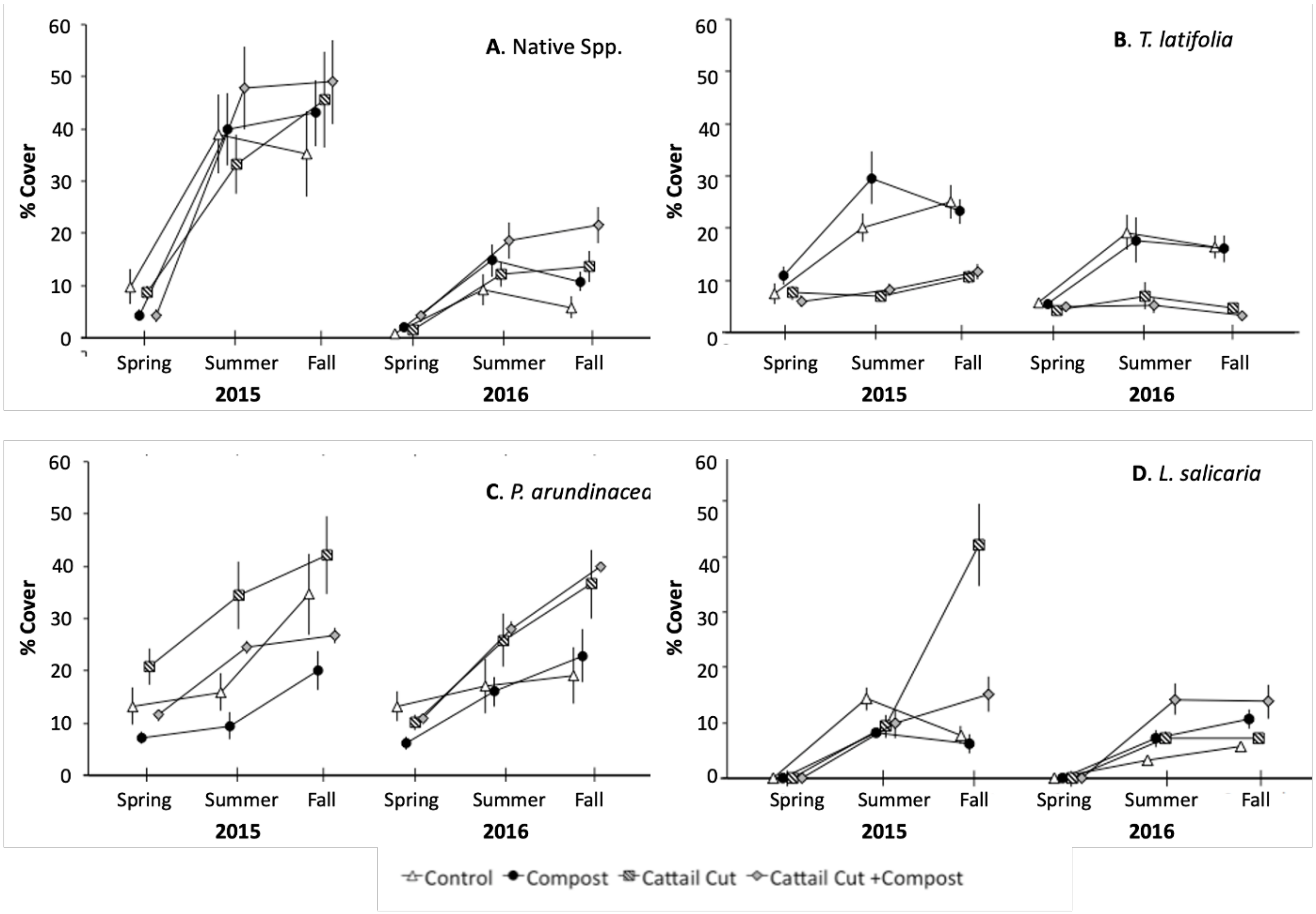

Figure 4. Plant percent cover in the emergent wetland, Cady, in 2015 and 2016. Values are mean +/- SE. Points are off-set slightly on the x-axis to allow for clearer visualization of points.

Table 10. Results of a three-way ANOVA examining the effect of year (2015, 2016) and treatment on plant percent cover of invasive species: *T. latifolia*, *P. arundinacea* and *L. salicaria* in Cady wetland. Significant p-values are bolded.

		Cady						
		Year	Compost	Year x Compost	Cattail Cut	Year x Cattail Cut	Compost x Cattail Cut	Year x Compost x Cattail Cut
Spring	*T. latifolia*	$F_{1,152}$= 12.6 **0.0005**	$F_{1,152}$= 0.44 0.5	$F_{1,152}$= 0.57 0.44	$F_{1,152}$= 3.9 **0.049**	$F_{1,152}$= 0.26 0.61	$F_{1,152}$= 3.1 0.08	$F_{1,152}$= 2.7 0.09
Spring	*P. arundinacea*	$F_{1,152}$= 0.06 0.80	$F_{1,152}$= 3.30 0.07	$F_{1,152}$= 1.64 0.20	$F_{1,152}$= 10.9 **0.001**	$F_{1,152}$= 1.22 0.27	$F_{1,152}$= 0.33 0.56	$F_{1,152}$= 1.3 0.24
Spring	*L. salicaria*	$F_{1,152}$= 32.8 **<0.0001**	$F_{1,152}$= 3.1 0.07	$F_{1,152}$= 3.11 0.07	$F_{1,152}$= 6.41 **0.01**	$F_{1,152}$= 6.79 **0.01**	$F_{1,152}$= 0.14 0.7	$F_{1,152}$= 0.17 0.67
Summer	*T. latifolia*	$F_{1,152}$= 4.86 **0.03**	$F_{1,152}$= 5.03 **0.03**	$F_{1,152}$= 3.89 0.05	$F_{1,152}$= 68.3 **<0.0001**	$F_{1,152}$= 2.74 0.09	$F_{1,152}$= 6.67 **0.01**	$F_{1,152}$= 2.09 0.15
Summer	*P. arundinacea*	$F_{1,152}$= 0.75 0.38	$F_{1,152}$= 0.17 0.67	$F_{1,152}$= 0.63 0.42	$F_{1,152}$= 13.7 **0.0003**	$F_{1,152}$= 0.13 0.71	$F_{1,152}$= 0.04 0.83	$F_{1,152}$= 0.07 0.93
Summer	*L. salicaria*	$F_{1,152}$= 0.01 0.99	$F_{1,152}$= 0.65 0.42	$F_{1,152}$= 1.20 0.27	$F_{1,152}$= 5.82 **0.02**	$F_{1,152}$= 0.02 0.96	$F_{1,152}$= 0.87 0.35	$F_{1,152}$= 0.22 0.88
Fall	*T. latifolia*	$F_{1,152}$= 31.3 **<0.0001**	$F_{1,152}$= 1.15 0.28	$F_{1,152}$= 0.78 0.37	$F_{1,152}$= 96.4 **<0.0001**	$F_{1,152}$= 0.57 0.44	$F_{1,152}$= 2.28 0.13	$F_{1,152}$= 0.46 0.50
Fall	*P. arundinacea*	$F_{1,152}$= 0.55 0.45	$F_{1,152}$= 0.89 0.34	$F_{1,152}$= 1.3 0.24	$F_{1,152}$= 15.4 **0.0001**	$F_{1,152}$= 1.2 0.26	$F_{1,152}$= 0.05 0.93	$F_{1,152}$= 0.25 0.61
Fall	*L. salicaria*	$F_{1,152}$= 0.06 0.93	$F_{1,152}$= 2.8 0.09	$F_{1,152}$= 0.01 0.91	$F_{1,152}$= 10.9 **0.001**	$F_{1,152}$= 0.03 0.98	$F_{1,152}$= 5.08 **0.03**	$F_{1,152}$= 0.02 0.88

At Packard, there was seasonal variation in all plant cover between 2015 and 2016, with lower overall species richness, diversity and cover during the drought of 2016 (p=<0.0001) (Table 9, Figure 5). In Summer 2015, compost addition plots had significantly lower cover of invasive species (25 ± 5%) relative to the control (31 ± 7%; p=0.01) (Table 8). *T. angustifolia* was the dominant invasive plant and reached maximum cover in September 2015 (16.8 ± 1%) (Figure 5). Compost addition had no effect on *T. angustifolia* cover (Table 8). *T. latifolia* followed a similar yearly and seasonal trend as *T. angustifolia* (Table 8). Like Cady, *L. salicaria* cover was low, but significantly lower in the control in Summer and Fall 2015 (p=0.0003) (Table 12).

Table 8. Plant characteristics measured in Packard wetland each season. Mean ± SE species richness (S), Shannon Diversity Index (H), invasive species cover (Inv %) and native species cover (Nat %).

			Packard	
			2015	**2016**
Spring	**S**	**Control**	2.7 ± 0.1	4.5 ± 0.2
		Compost	2.5 ± 0.2	4.1 ± 0.2
	H	**Control**	0.7 ± 0.1	1.1 ± 0.0
		Compost	0.7 ± 0.1	1.0 ± 0.0
	Inv %	**Control**	18.3 ± 3.3	6.0 ± 0.3
		Compost	15.3 ± 4.7	5.2 ± 0.6
	Nat %	**Control**	7.2 ± 1.6	21.2 ± 1.9
		Compost	6.7 ± 1.6	21.8 ± 2.2
Summer	**S**	**Control**	6.3 ± 0.5	4.9 ± 0.3
		Compost	6.4 ± 0.4	4.3 ± 0.2
	H	**Control**	1.4 ± 0.1	1.2 ± 0.1
		Compost	1.4 ± 0.1	1.0 ± 0.1
	Inv %	**Control**	31.4 ± 7.2	11.1 ± 0.9
		Compost	25.7 ± 5.7	7.4 ± 0.8
	Nat %	**Control**	34.6 ± 4.1	20.0 ± 2.0
		Compost	45.8 ± 4.5	20.6 ± 2.0
Fall	**S**	**Control**	7.3 ± 0.5	5.3 ± 0.3
		Compost	6.8 ± 0.5	5.8 ± 0.1
	H	**Control**	1.6 ± 0.1	1.3 ± 0.1
		Compost	1.4 ± 0.1	1.4 ± 0.0
	Inv %	**Control**	32.6 ± 5.0	5.9 ± 1.8
		Compost	25.9 ± 5.4	3.3 ± 0.5
	Nat %	**Control**	52.6 ± 4.7	24.4 ± 3.2
		Compost	67.1 ± 9.3	23.0 ± 2.1

Table 9. Results of a two-way ANOVA examining the effect of season (Spring, Summer, Fall) and treatment on species richness (S), Shannon Diversity Index (H), invasive species cover (Inv %) and native species cover (Nat %) in Packard wetland. Significant p-values are bolded.

		Packard		
		Year	Compost	Year x Compost
Spring	S	$F_{1,152}$= 110.8	$F_{1,152}$= 3.45	$F_{1,152}$= 1.17
		<0.0001	0.06	0.28
	H	$F_{1,152}$= 49.03	$F_{1,152}$= 0.94	$F_{1,152}$= 1.22
		0.01	0.33	0.27
	Inv %	$F_{1,152}$= 40.82	$F_{1,152}$= 1.16	$F_{1,152}$= 0.37
		<0.001	0.28	0.55
	Nat %	$F_{1,152}$= 42.67	$F_{1,152}$= 0.005	$F_{1,152}$= 0.07
		<0.0001	0.98	0.79
Summer	S	$F_{1,152}$= 42.44	$F_{1,152}$= 1.46	$F_{1,152}$= 1.69
		<0.0001	0.22	0.22
	H	$F_{1,152}$= 28.87	$F_{1,152}$= 2.8	$F_{1,152}$= 1.52
		<0.0001	0.09	0.22
	Inv %	$F_{1,152}$= 102.81	$F_{1,152}$= 6.09	$F_{1,152}$= 0.28
		<0.001	**0.01**	0.60
	Nat %	$F_{1,152}$= 26.35	$F_{1,152}$= 2.32	$F_{1,152}$= 1.89
		<0.0001	0.13	0.17
Fall	S	$F_{1,152}$= 28.99	$F_{1,152}$= 0	$F_{1,152}$= 2.61
		<0.0001	1.0	0.11
	H	$F_{1,152}$= 3.02	$F_{1,152}$= 0.005	$F_{1,152}$= 2.98
		0.08	0.95	0.09
	Inv %	$F_{1,152}$= 149.64	$F_{1,152}$= 5.28	$F_{1,152}$= 0.99
		<0.0001	**0.02**	0.32
	Nat %	$F_{1,152}$= 63.07	$F_{1,152}$= 2.06	$F_{1,152}$= 3.05
		<0.0001	0.15	0.08

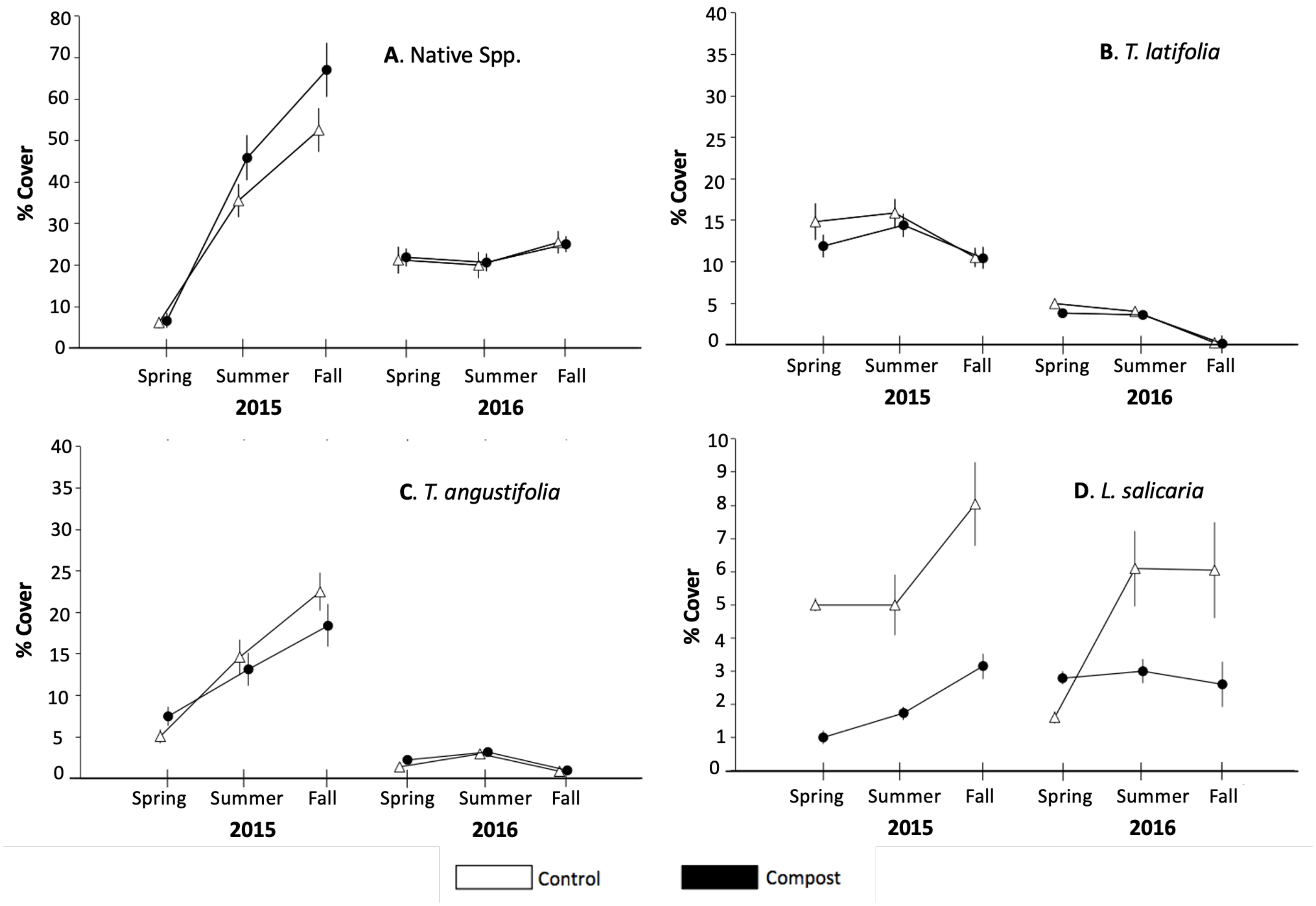

Figure 5: Plant percent cover in Packard in 2015 and 2016

Table 11. Results of a two-way ANOVA examining the effect of year (2015, 2016) and treatment on plant percent cover of invasive species: *T. latifolia*, *P. arundinacea* and *L. salicaria*. Significant p-values are bolded.

		Packard		
		Year	**Compost**	**Year x Compost**
Spring	***T. latifolia***	$F_{1,152}$= 29.6 **<0.0001**	$F_{1,152}$= 4.18 **0.04**	$F_{1,152}$= 1.6 0.19
Spring	***T. angustifolia***	$F_{1,152}$= 28.01 **<0.0001**	$F_{1,152}$= 4.13 **0.04**	$F_{1,152}$= 1.43 0.23
Spring	***L. salicaria***	$F_{1,152}$= 16.67 **<0.0001**	$F_{1,152}$= 1.15 0.28	$F_{1,152}$= 1.15 0.28
Summer	***T. latifolia***	$F_{1,152}$= 89.45 **<0.0001**	$F_{1,152}$= 1.15 0.28	$F_{1,152}$= 0.68 0.62
Summer	***T. angustifolia***	$F_{1,152}$= 64.06 **<0.0001**	$F_{1,152}$= 0.60 0.44	$F_{1,152}$= 0.63 0.42
Summer	***L. salicaria***	$F_{1,152}$= 7.93 **0.006**	$F_{1,152}$= 13.66 **0.0003**	$F_{1,152}$= 1.85 0.17
Fall	***T. latifolia***	$F_{1,152}$= 96.54 **<0.0001**	$F_{1,152}$= 0.08 0.77	$F_{1,152}$= 0.03 0.85
Fall	***T. angustifolia***	$F_{1,152}$= 92.27 **<0.0001**	$F_{1,152}$= 0.43 0.51	$F_{1,152}$= 1.45 0.23
Fall	***L. salicaria***	$F_{1,152}$= 0.03 0.86	$F_{1,152}$= 13.64 **0.0003**	$F_{1,152}$= 0.90 0.34

3.2 *Greenhouse Experiment*

In all species and combination pots, adding compost significantly increased organic matter ($p<0.05$ for all; Table 12). Similar to organic matter, soil TIN also doubled in both combination species and *T. latifolia* alone, driven by the change in nitrate. Compost addition significantly increased belowground biomass and final height in the *T. latifolia* and *A. subcordatum* ($p<0.01$; Table 12), but not any other species. Aboveground biomass was significantly higher when compost was added to *A. subcordatum* alone ($p=0.02$, Table 12) and *P.arundinacea* in combination with *A. subcordatum* ($p=0.005$, Table 13). Final plant height of *T.latifolia* ($p=0.006$) and *A. subcordatum* ($p=0.0003$) alone was significantly higher when compost was added (Table 12).

Table 12. Mean ± SE of soil pH, organic matter (OM, %), total inorganic nitrogen (TIN, mg/kg), belowground biomass (BGBM, g), final height (cm) and aboveground biomass (AG, g) in greenhouse experiment. Average in single species and combination pots. T-test performed for differences between control and compost addition in the single-species and combination pots. Significant p-values are bolded. Final height and aboveground biomass for mixed species pots separated by species in Table 13.

	pH		OM		TIN		BGBM		Final Height		AG	
	Control	+OM	Control	+OM	Control	+OM	Control	+OM	Control	+OM	Control	+OM
T. latifolia	8.1 ± 0.1	8.2 ± 0.02	4.9 ± 0.6	14.4 ± 2.1	548 ± 206	1981 ± 1301	3.9 ± 2.9	7.5 ± 3.0	25.5 ± 21	48.2 ± 15	4.6 ± 1	4.8 ± 2
	t(15)=1.04, p = 0.3		*t*(13)=4.3, **p = 0.0007**		*t*(5)=-2.6, **p = 0.04**		*t*(22)=-3.0, **p = 0.006**		*t*(20)=-3.0, **p = 0.006**		*t*(19)=-2.0, p = 0.8	
P. arundinacea	8.2 ± 0.1	8.2 ± 0.03	3.1 ± 0.5	11.1 ± 1.4	652 ± 374	2253 ± 1751	0.6 ± 0.3	1.3 ± 1.3	37.2 ± 9.3	36.6 ± 9.7	0.5 ± 0.06	0.7 ± 0.1
	t(13)=0.14, p = 0.8		*t*(14)=5.5, **p = 0.0007**		*t*(5)=-2.1, p = 0.08		*t*(12)=-1.8, p = 0.09		*t*(22)=0.1, p = 0.8		*t*(21)=2.0, p = 0.3	
A. subcordatum	8.0 ± 0.1	8.1 ± 0.03	5.0 ± 1.4	8.9 ± 1.0	871 ± 30	1353 ± 608	2.4 ± 1.2	4.6 ± 2.0	8.6 ± 4.2	16.9 ± 3.5	0.8 ± 0.09	1.2 ± 0.1
	t(17)=0.84, p = 0.4		*t*(21)=-2.2, **p = 0.04**		*t*(9)=-1.1, p = 0.3		*t*(18)=-3.2, **p = 0.004**		*t*(21)=-5.2, **p = 0.0003**		*t*(20)=2.0, **p = 0.02**	
T. latifolia x A. subcordatum	9.1 ± 0.1	8.5 ± 0.2	4.2 ± 0.6	8.2 ± 0.7	769 ± 255	2681 ± 703	5.0 ± 2.5	4.6 ± 1.8				
	t(15)=3.3, **p = 0.004**		*t*(22)=-4.4, **p =0.002**		*t*(6)=-6.2, **p = 0.0007**		*t*(20)=0.5, p = 0.6					
P. arundinacea x A. subcordatum	9.1 ± 0.1	8.9 ± 0.1	3.1 ± 0.3	8.9 ± 0.6	280 ± 124	1814 ± 1170	1.7 ± 1.0	1.9 ± 0.7				
	t(22)=2.09, **p = 0.04**		*t*(17)=-8.5, **p = 0.001**		*t*(5)=-3.2, **p = 0.02**		*t*(19)=-0.5, p = 0.6					

Table 13. Mean ± SE of aboveground biomass (AG, g) and final height (cm) for combination pots. T-test performed for differences between control and compost addition in the single-species and combination pots. Significant p-values are bolded.

	AG				Final Height			
	Native		Invasive		Native		Invasive	
	Control	+OM	Control	+OM	Control	+OM	Control	+OM
T. latifolia x A. subcordatum	0.5 ± 0.2	0.7 ± 0.6	1.9 ± 1.7	2.1 ± 1.0	15.8 ± 5	17.5 ± 4.5	27.2 ± 22	24 ± 27
	t(15)=-1.4, p = 0.2		*t*(18)=1.7, p = 0.7		*t*(22)=1.7, p = 0.4		*t*(21)=1.7, p = 0.7	
P. arundinacea x A. subcordatum	0.5 ± 0.2	0.5 ± 0.3	0.3 ± 0.02	0.5 ± 0.03	13.3 ± 2	14.8 ± 4	37.4 ± 13	37.5 ± 7.1
	t(21)=1.7, p = 0.8		*t*(22)=1.7, **p = 0.005**		*t*(17)=1.7, p = 0.2		*t*(17)=1.7, p = 0.9	

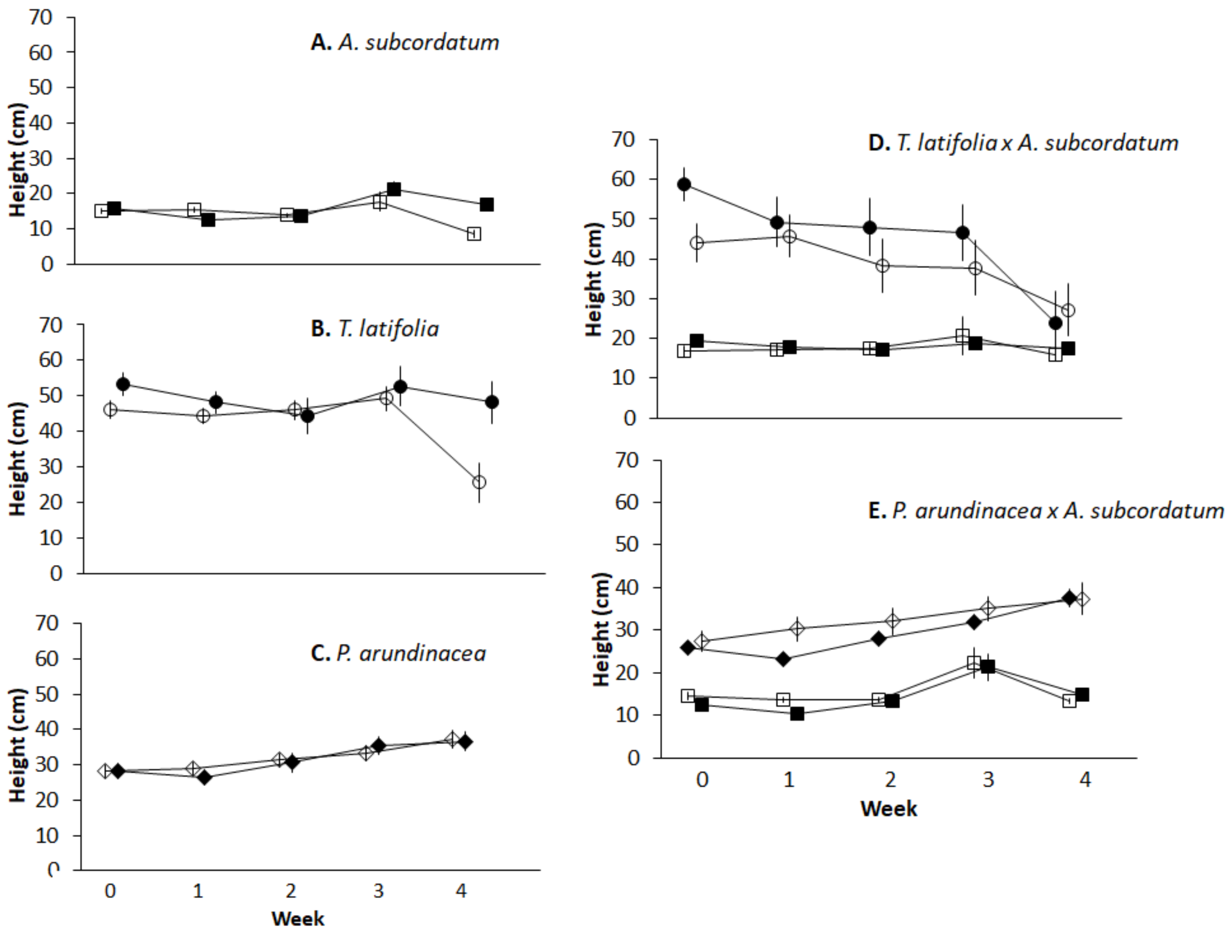

Figure 6. Plant heights in greenhouse experiment. A: *A. subcordatum* (square points) B: *T. latifolia* (circles) C: *P. arundinacea* (diamond) D: *T. latifolia* (circle) x *A. subcordatum* (square) E: Combination *P. arundinacea* (diamond) A. *subcordatum* (square)

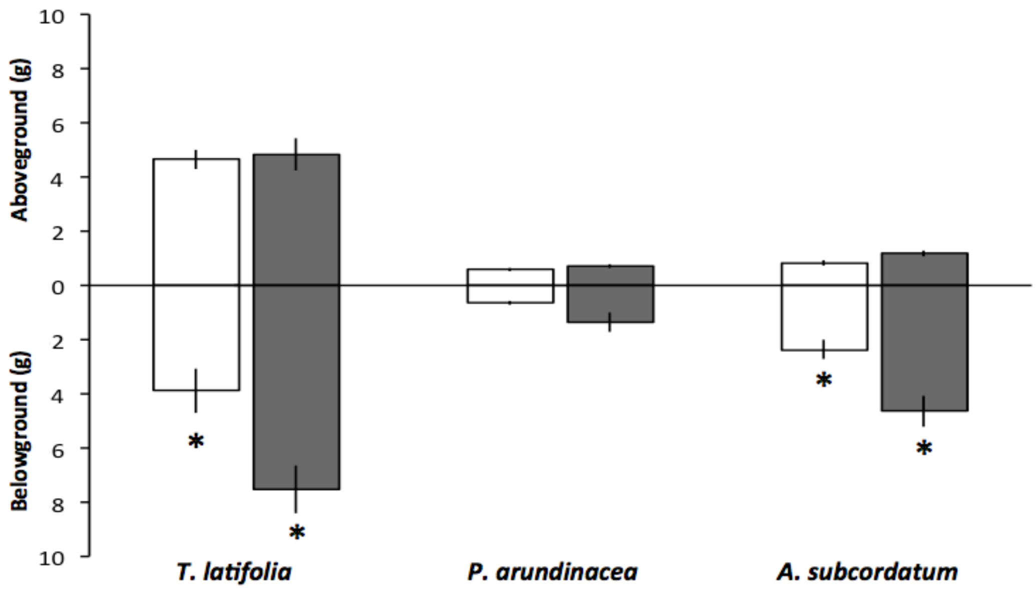

Figure 7. Above and belowground biomass in control (white) and compost amended (gray) single species pots. *indicate significance based on t-test (p<0.05)

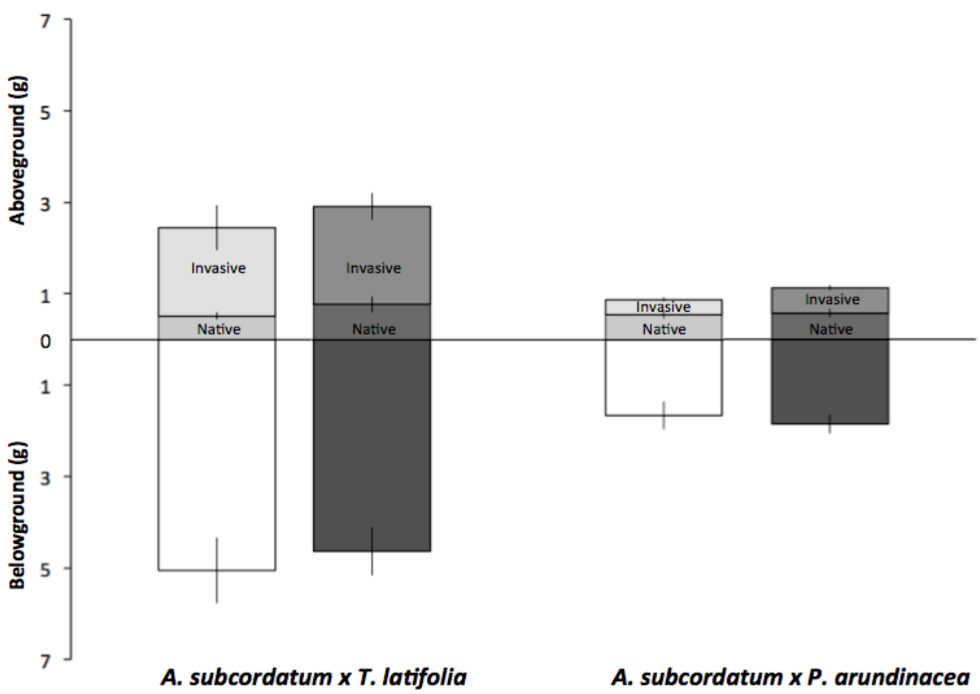

Figure 8. Above and belowground biomass for the species combination pots. Aboveground biomass is split between the native and invasive species in each combination and by treatment: control (light gray) and compost (dark gray).

4. Discussion

Using large-scale compost addition as a management tool helped enhance wetland function through improving soil moisture and organic matter, in turn helping maintain resilience of plants during drought. Combining this strategy with cutting the dominant invasive plant at Cady (*Typha*) helped native plant species, but also led to increased spread of other undesirable species. Drought and site-specific hydrology impacted the extent of soil enhancement. Successful management requires understanding which specific ecosystem functions any one technique impacts.

The two wetland sites were similar in initial organic content and nutrient levels, but different substantially in hydrology. The similarities in nutrient and OM were likely derived from different trajectories, given the dissimilar history of the two sites (pasture versus row crop). The hydrological differences derived from both differences in water table depth and connectivity to other systems. Isolated wetlands are more strongly affected by changes in climate regimes (Middleton et.al 2012). In Packard, altered hydrology due to lack of connectivity to surface and groundwater resources means it's primarily fed by precipitation and some snow melt. Droughts, like the one experienced in 2016, can cause a shift in native plant growth and species composition. Species in Packard shifted from wetland sedge species in 2014 (pers. obs.) to wet meadow species in 2015 and in 2016, with more upland species were recorded than the previous year, suggesting a shift in plant community structure.

Soils are the foundation of wetland ecosystems by providing moisture, structure and nutrients to plant communities. Soil development is often slow in created wetlands so enhancing soil properties such as organic matter, bulk density and soil moisture is key to successful creation efforts. Soil organic matter is often difficult to mimic when creating wetlands where none existed previously, yet this is a crucial element to successful creation. During the early stages of development, organic matter typically increases as plants colonize and senesce over several years (Odum, 1969). Prior land use sets the stage for the levels of organic matter found in the first 10 years after construction (Campbell 2002). In both wetland sites, addition of leaf litter compost was successful in increasing soil organic matter by 3-5% over the course of two to three applications. Low soil organic matter has been linked to lower vegetation establishment and growth, poor habitat for macroinvertebrates and altered nutrient cycling (Shafer & Ernst 1999).

By adding compost, we were able to increase soil organic matter to a level closer to that of natural freshwater wetlands which is between 12 and 20 percent (Campbell 2002, Faulkner & Richardson 1989). Bulk density (BD) was not significantly different between Cady and Packard. Adding compost significantly lowered BD in both wetlands. Created wetlands often have high BD soils that limit the vegetation communities. Amending wetland soils could increase the likelihood of these wetlands reaching full function in 10-15 years and overall, leaf litter compost could be an inexpensive treatment to increase organic matter in created wetlands early in development (Sutton-Grier 2009, Wolf 2011).

Hydrology was the most influential difference between the two wetlands, influencing plant community composition. Drought played a substantial role in the effects seen by adding compost. In 2015, Packard dried by late June and Cady by late July, although the soil remained saturated. In 2016, both wetlands were completely dry by early June. Rainfall in 2015 for the peak growing season (June, July and August) was 40.5 cm compared to only 15.3 cm in 2016 (usclimatedata.com). The extreme summer of 2016 had a negative impact on native plant growth in both areas. There was a 50% decrease in total cover in all treatments between 2015 and 2016 (Figures 1 and 2). In both areas there was a decrease in invasive cover between 2015 and 2016, independent of treatment. At Cady, evaluating the potential impact of compost on *P. arundinacea* was difficult due to the drought. In 2015, *P. arundinacea* cover was significantly reduced in the compost addition plots as compared to the control (Figure 1). In 2016, the effect was minimal due to increased growth in the other treatment plots. Once the soil dried completely, *P. arundinacea* outgrew the other native and invasive species due to higher tolerance to drought, minimizing compost addition effects seen in 2015.

Nitrogen availability in the soil was significantly higher in treated plots in Packard but not significantly influenced in Cady (Table 5). The C:N ratio of the leaf litter compost used for this project was close to that of the initial soil conditions and had a larger quantity of fresh plant material, likely from yard clippings which resulted in a higher amount of available N. The optimum C:N of compost is around 40:1 and the compost used in this experiment was on average 18:1 between the 2015 and 2016. This is consistent with other studies that found that soil amendments with higher N concentrations such as compost and topsoil significantly increased soil N (Bailey et al. 2007, Ballantine et al 2012, O'Brien & Zedler, 2006, Sutton-Grier et al. 2009).

Enriching soil organic matter, moisture and N while decreasing bulk density was potentially responsible for the increased cover of native species in both areas (Figures 1&2). Although not significant, in Cady, the highest amount of soil N occurred in the combination control plots (746.9 ± 209.4 mg/kg) correlated with the highest native species cover (22.9 ± 2.4%). Species richness was also significantly higher using combined management than any other treatment (4.43 ± 0.13 species) and Shannon-Weiner diversity indices followed a similar trend. The greenhouse experiment further validated the effects seen in the large-scale field manipulation. Adding compost increased organic matter by 5-10% resulting in values close to that of natural wetland soil (Table 5). Soil extractable N was also significantly higher in all single species and combination pots. This increase in available N may have caused the slightly higher heights of the native *A. subcordatum* and the invasive *T. latifolia* (Figure 5), suggesting that these species may be most responsive to organic matter addition.

Further, to the benefit of native species, compost addition decreased some invasive plant growth. *P. arundinacea* cover was reduced when plots were amended with compost, in 2015, although this effect nearly reversed in 2016 (Figure 4). Cutting cattails opened the canopy and likely due to lack of rain, drought tolerant *P. arundinacea* filled in space where native plants were unable. Native species cover in Packard followed a similar trajectory between 2015 and 2016 (Figure 5). This suggests that removal of invasive *Typha* spp. combined with adding compost, can lead to a more diverse plant community if wetlands stay flooded (Lishawa et al. 2015, Sutton-Grier et al. 2009) Although species richness and diversity were increased with compost soil amendments, they still didn’t reflect natural values of comparable freshwater wetlands in the northeast (S≅10-12), highlighting the importance of continued restoration and control efforts following construction and past the time frame of typical required monitoring (Balcombe et al. 2005, Campbell 2002).

5. Conclusions

This study demonstrated that amending created wetland soils could be key in helping combat invasive species growth while also curbing the effects of increasingly extreme weather due to climate change. Implementing compost addition as a control measure in areas with high yearly variation may help native species resilience. Compost soil amendments could help in wetland creation by regulating organic matter and bulk density when there is a prolonged lack of precipitation. However, our results indicate that multiple management techniques employed in tandem may be required to promote healthy and functional wetland ecosystems. By removing invasive plants and amending the soil, native species cover increased and was more stable when compost was also added. Amending wetland soil creates a more suitable base for a wider range of wetland plant species are able to colonize, leading to more diverse plant communities that have a better ability to survive potential changes in climate.

6. Literature Cited

Ballantine, K., Schneider, R., Groffman, P., & Lehmann, J. (2012). Soil Properties and Vegetative Development in Four Restored Freshwater Depressional Wetlands. *Soil Science Society of America Journal*, *76*, 1482.

Blake, G.R., Hartge, K.H., (1986). Bulk density. In: Klute, A. (Ed.), Methods of Soil Analysis, Part 1, Physical and Mineralogical Methods, Part I. Physical and Mineralogical Methods, second ed. Agronomy Monograph No. 9, Madison, WI 363–375.

Blumenthal, D. M., Jordan, N. R., & Russelle, M. P. (2003). Soil carbon addition controls weeds and facilitates prairie restoration. *Ecological Applications*, *13*(3), 605–615. doi:10.1890/1051-0761(2003)013[0605:SCACWA]2.0.CO;2

Bedford, B. L., Walbridge, M. R., & Aldous, A. (1999). Patterns in nutrient availability and plant diversity of temperate North American wetlands.Ecology, 80(7), 2151-2169.

Boily, M., Sarrasin, B., DeBlois, C., Aras, P., & Chagnon, M. (2013). Acetylcholinesterase in honey bees (Apis mellifera) exposed to neonicotinoids, atrazine and glyphosate: laboratory and field experiments.*Environmental Science and Pollution Research*, *20*(8), 5603-5614.

Bonny, S. (2008). Genetically modified glyphosate-tolerant soybean in the USA: adoption factors, impacts and prospects. A review. *Agronomy for Sustainable Development*, *28*(1), 21-32.

Bouchard, H. (2010). Soil development and establishment of carbon-based properties in created freshwater marshes. Ecological Applications: A Publication of the Ecological Society of America, 20(2), 539–553.

Brinson, M. M., & Malvárez, A. I. (2002). Temperate freshwater wetlands: types, status, and threats. Environmental conservation, 29(02), 115-133.

Bruland, G. L., & Richardson, C. J. (2004). Hydrologic gradients and topsoil additions affect soil properties of Virginia created wetlands. *Soil Science Society of America Journal*, *68*(6), 2069-2077.

Calhoun, A.J.K., Arrigoni, J., Brooks, R.P. *et al.* Creating Successful Vernal Pools: A Literature Review and Advice for Practitioners. *Wetlands* 34, 1027–1038 (2014).

Costanza, R., d'Arge, R., de Groot, R., Farberk, S., Grasso, M., Hannon, B.,& Raskin, R. G. (1997). The value of the world's ecosystem services and natural capital. *Nature*, 387, 253.

Costanza, R., de Groot, R., Sutton, P., van der Ploeg, S., Anderson, S. J., Kubiszewski, I.,& Turner, R. K. (2014). Changes in the global value of ecosystem services. *Global Environmental Change*, *26*, 152-158.

Cowardin, L. M., Carter, V., Golet, F. C., & LaRoe, E. T. (1979). Classification of wetlands and deepwater habitats of the United States. US Department of the Interior, US Fish and Wildlife Service.

CWA. Clean Water Act of 1972, 33 U.S.C. § 1251 et seq. (2002). Retrieved from http://www.epw.senate.gov/water.pdf

Eschen, R., Müller-Schärer, H., & Schaffner, U. (2006). Soil carbon addition affects plant growth in a species-specific way. The Journal of Applied Ecology, 43(1), 35–42.

Faulkner, S. P., & Richardson, C. J. (1989). Physical and chemical characteristics of freshwater wetland soils. *Constructed wetlands for wastewater treatment*, 41-72.

Fennessy, M. S., Rokosch, A., & Mack, J. J. (2008). Patterns of plant decomposition and nutrient cycling in natural and created wetlands. Wetlands, 28(2), 300–310.

Fisher, J., & Acreman, M. C. (2004). Wetland nutrient removal: a review of the evidence. *Hydrology and Earth System Sciences Discussions*, 8(4), 673-685.

Frieswyk CB, Johnston CA, Zedler JB (2007) Identifying and characterizing dominant plants as an indicator of community condition. J Gt Lakes Res 33:125–135

Galatowitsch, S. M., & van der Valk, A. G. (1996). The Vegetation of Restored and Natural Prairie Wetlands. Ecological Applications: A Publication of the Ecological Society of America, 6(1), 102–112.

Gelderman, R.H., and A.P. Mallarino. Recommended Chemical Soil Test Procedures for the North Central Region." North Central Regional Research Publication 221 (2012).

Green, E. K., & Galatowitsch, S. M. (2002). Effects of Phalaris arundinacea and nitrate-N addition on the establishment of wetland plant communities. *Journal of applied Ecology*, *39*(1), 134-144.

Hey D.L., (1995). Flood Reduction through Wetland Restoration: The Upper Mississippi River Basin as a Case History. Restoration Ecology, 3(1), 4–17.

Houlahan, J. E., & Findlay, C. S. (2004). Effect of Invasive Plant Species on Temperate Wetland Plant Diversity. Conservation Biology: The Journal of the Society for Conservation Biology, 18(4), 1132–1138.

Iannone, B. V., & Galatowitsch, S. M. (2008). Altering Light and Soil N to Limit Phalaris arundinacea Reinvasion in Sedge Meadow Restorations. Restoration Ecology, 16(4), 689–701.

Isbell, F., Reich, P. B., Tilman, D., Hobbie, S. E., Polasky, S., & Binder, S. (2013). Nutrient enrichment, biodiversity loss, and consequent declines in ecosystem productivity. *Proceedings of the National Academy of Sciences*,*110*(29), 11911-11916.

Keough, J. R., T. A. Thompson, G. R. Guntenspergen, and D. A. Wilcox. (1999). Hydrogeomorphic factors and ecosystem responses in coastal wetlands of the Great Lakes. *Wetlands* 19:821–834.

Kettenring, K. M., & Adams, C. R. (2011). Lessons learned from invasive plant control experiments: a systematic review and meta-analysis. *Journal of Applied Ecology*, *48*(4), 970–979. doi:10.1111/j.1365-2664.2011.01979.x

Lishawa, S. C., Lawrence, B. A., Albert, D. A., & Tuchman, N. C. (2015). Biomass harvest of invasive Typha promotes plant diversity in a Great Lakes coastal wetland. *Restoration ecology*, *23*(3), 228-237.

Matthews, J. W., & Endress, A. G. (2010). Rate of succession in restored wetlands and the role of site context. *Applied Vegetation Science*, *13*(3), 346-355.

McGowan, Michael, "Effects of Prior Land Use, Carbon Availability, and Hydrology on Nitrogen Cycling in Created Freshwater Wetlands" (2020). Thesis. Rochester Institute of Technology.

Middleton B.A., Kleinebecker T. (2012) The Effects of Climate-Change-Induced Drought and Freshwater Wetlands. In: Middleton B. (eds) Global Change and the Function and Distribution of Wetlands. Global Change Ecology and Wetlands, vol 1. Springer, Dordrecht.

Mitsch, W. J., & Gosselink, J. G. (2000). The value of wetlands: importance of scale and landscape setting. Ecological Economics: The Journal of the International Society for Ecological Economics, 35(1), 25–33.

Moore, H. H., Niering, W. A., Marsicano, L. J., & Dowdell, M. (1999). Vegetation change in created emergent wetlands (1988–1996) in Connecticut (USA). *Wetlands Ecology and Management*, *7*(4), 177-191.

Moreno-Mateos, D., Power, M. E., & Yockteng, R. (2012). Structural and functional loss in restored wetland ecosystems. *PLoS Biology*, *10*(1), e1001247.

Mullin, B. H. (1998). The biology and management of purple loosestrife (Lythrum salicaria). *Weed Technology*, 397-401.

N.R.C.S., & Committee, O. M. W. L. (2001). Compensating for Wetland Losses Under the Clean Water Act. Washington, DC, USA: National Academies Press. Retrieved from http://www.ebrary.com

Odum, E.P. 1969. The strategy of ecosystem development. Science 164: 262–270.

Perry, L. G., Galatowitsch, S. M., & Rosen, C. J. (2004). Competitive control of invasive vegetation: a native wetland sedge suppresses Phalaris arundinacea in carbon-enriched soil. *Journal of Applied Ecology*, *41*(1), 151–162.

Pimentel, D., Lach, L., Zuniga, R., & Morrison, D. (2000). Environmental and Economic Costs of Nonindigenous Species in the United States. Bioscience, 50(1), 53–65.

Reddy, K. R., Patrick, W. H., & Lindau, C. W. (1989). Nitrification-denitrification at the plant root-sediment interface in wetlands. Limnology and Oceanography, 34(6), 1004-1013.

Rogers, S. D., Beech, J., & Sarma, K. S. (1998). Shoot regeneration and plant acclimatization of the wetland monocot Cattail (Typha latifolia). *Plant cell reports*, *18*(1-2), 71-75.

Shaffer, P. W., & Ernst, T. L. (1999). Distribution of soil organic matter in freshwater emergent/open water wetlands in the Portland, Oregon metropolitan area. Wetlands, 19(3), 505-516.

Shaw, S. P., & Fredine, C. G. (1971). Wetlands of the United States: their extent and their value to waterfowl and other wildlife. US Department of the Interior, Fish and Wildlife Service.

Stantec Consulting. (2009). High Acres Landfill – Wetland Mitigation Report Towns of Macedon and Perinton Wayne and Monroe Counties , New York, 10-26.

Stauffer, A. L., & Brooks, R. P. (1997). Plant and soil responses to salvaged marsh surface and organic matter amendments at a created wetland in central Pennsylvania. *Wetlands*, *17*(1), 90-105.

Stolt, M. H., Genthner, M. H., Lee Daniels, W., Groover, V. A., Nagle, S., & Haering, K. C. (2000). Comparison of soil and other environmental conditions in constructed and adjacent palustrine reference wetlands. *Wetlands*, *20*(4), 671-683.

Sutton-Grier, A.E., Ho, M. & Richardson, C.J. Organic amendments improve soil conditions and denitrification in a restored riparian wetland. *Wetlands* 29**,** 343–352 (2009). https://doi.org/10.1672/08-70.1

Thompson, K., Hodgson, J. G., Grime, J. P., & Burke, M. J. W. (2001). Plant traits and temporal scale: evidence from a 5-year invasion experiment using native species. The Journal of Ecology, 89(6), 1054–1060.

Wilcox, D. A. and J. E. Meeker. (1995) Wetlands in regulated Great Lakes. p. 247–249. In E. T. LaRoe, G. S. Farris, C. E. Puckett, P. D. Doran, and M. J. Mac (eds.) Our Living Resources: a Report to the Nation on the Distribution, Abundance, and Health of U.S. Plants, Animals, and Ecosystems. U.S. Department of the Interior, National Biological Service, Washington, DC, USA.

Wolf, K. L., Ahn, C., & Noe, G. B. (2011). Development of Soil Properties and Nitrogen Cycling in Created Wetlands. Wetlands, 31(4), 699–712.

VanWinkle, Sydney, "Socio-ecological context matters for wetland project outcomes" (2021). Thesis. Rochester Institute of Technology.

Zedler, J. B., & Callaway, J. C. (1999). Tracking wetland restoration: do mitigation sites follow desired trajectories?. *Restoration Ecology*, *7*(1), 69-73.

Zedler, J. B., & Kercher, S. (2004). Causes and consequences of invasive plants in wetlands: opportunities, opportunists, and outcomes. Critical Reviews in Plant sciences, 23(5), 431-452.

Zedler, J. B., & Kercher, S. (2005). Wetland resources: status, trends, ecosystem services, and restorability. Annu. Rev. Environ. Resour., 30, 39-74.