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Effects of Soil Amendments on Community Structure and Function in Created Wetlands

Lauren Saggese
lhs5091@rit.edu

Effects of Soil Amendments on Community Structure and Function in Created Wetlands

By: Lauren Saggese

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Thomas H. Gosnell School of Life Sciences
College of Science
Environmental Science Program

Rochester Institute of Technology
Rochester, NY

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Committee Approval

Anna Christina Tyler, PhD Date

Chair of Committee, Thesis Advisor

Carmody McCalley, PhD Date

Committee Member

Karl Korfmacher, PhD Date

Committee Member

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Abstract

Wetlands provide many unique and important ecological functions, making them one of the most valuable ecosystems. When wetland destruction is unavoidable, creation or restoration of wetlands is a mandatory practice in order to recover lost ecosystem function. However, created wetlands often fail to provide functions and services equivalent to natural wetlands. The addition of carbon-rich plant compost has been shown to alter soil characteristics and biogeochemistry in wetlands. However, the impacts on wetland community structure are unknown. I evaluated the impact of long-term leaf-litter compost addition on soil properties, community structure of invertebrates and plants, algal photosynthesis and soil metabolism in two forested and one emergent created wetland. The three wetlands varied in prior land-use, hydrology, and time since construction and compost addition. Site differences in treatment impacts likely reflected antecedent land use history (row crop or pasture), wetland age (15, 12, 6 yr), hydrology, and time since last compost addition (5, 4, <1 yr). Overall, compost addition effectively increased soil moisture and organic matter while decreasing bulk density. Soil respiration and net metabolism were also enhanced by compost, indicating more rapid biogeochemical cycling, even years after compost addition. Benthic chlorophyll a levels were lower in the compost treatment, but GPP was higher. At some sites, compost reduced plant cover or diversity, indicating potential pitfalls of using compost as a singular management strategy. While macroinvertebrate communities were similar between treatments, there were site and seasonal differences associated with variation in temperature and hydrology. These results suggest that while compost additions can improve some functions associated with soil characteristics and biogeochemistry, integration with other management practices is needed to promote long-term ecological success. Continuous monitoring and adaptive management are crucial for optimizing the benefits of soil amendments and enhancing the ecological performance of created wetlands.

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Introduction

Wetlands and Wetland Loss

Wetlands are among the most valuable ecosystems due to their specific characteristics that enable many important ecosystem functions, including nutrient cycling, flood mitigation, carbon sequestration and habitat support (Clarkson et al., 2013; Liu et al., 2010; N.R.C.S, 2001; Weisner & Thiere, 2010). Mature wetlands also provide unique habitat that promotes high biodiversity due to the abundance of nutrients driving primary production (Meli et al., 2014). The saturated, nutrient rich soils encourage the growth of hydrophytic plants, and the standing water can provide habitat for aquatic and semi-aquatic mammals, reptiles, birds and macroinvertebrates (Carter, 1996; Newman et al., 1996).

While freshwater wetlands only occupy about 7% of terrestrial area, ~30% of all species utilize wetlands as a habitat (LaRoe, 1995). However, more than 70% of global wetland area has been lost since 1900 due to anthropogenic pressures, posing significant risk to the survival of wetland-dependent species and services (Davidson, 2014; Gibbs, 1993; Kingsford et al., 2016). As a result, 25% of wetland-dependent species are threatened and 6% are critically endangered, creating an urgent need for the protection, conservation, restoration, and creation of wetlands (Gardner & Finlayson, 2018).

Created Wetlands

The recognition of wetlands as critical ecosystems began in 1972 with the passing of the Clean Water Act. This was the first major United States federal law to address water pollution and the destruction of wetlands (US EPA, 2013). Specifically, Section 404 regulates dumping of any materials or fill in waterways in the United States, including wetlands (US EPA, 2015). This legislation was further altered in 1978 to emphasize the importance of replacing degraded or filled wetlands (Hough & Robertson, 2009; US EPA, 2022). This amendment serves to reduce the net loss of wetlands by mandating the creation or restoration of wetlands of equal or greater size and function to replace those that are unavoidably lost to anthropogenic development

(Robertson, 2000). Therefore, understanding the function and services of a wetland has become a central focus for creating and restoring wetlands.

Current legislation requires created wetlands to be of equal or greater size and in close proximity to the one lost, but lacks specific requirements for assessing prior function of a wetland slated for destruction, new function delivered by a created wetland, or comparison between the two (US EPA, 2015). Instead, evaluations are done on an individual basis and mainly include permit compliance with soil and vegetation field measurements over time to show successful creation (Brown & Veneman, 2001; Hoeltje & Cole, 2007; Sudol & Ambrose, 2002). Substantial evidence suggests these heavily modified or newly created wetlands do not currently perform to the same standards as naturally occurring wetlands, even when they meet predetermined permitting standards (Brown & Veneman, 2001; Gwin et al., 1999; He, 2019; Moreno-Mateos et al., 2012; Race & Fonseca, 1996; Wilson & Mitsch, 1996; Zedler & Callaway, 1999). Further, after mitigation wetlands have met permit requirements, there is rarely any long term maintenance or monitoring (Sudol & Ambrose, 2002). Therefore, while short term goals are met, this does not necessarily mean that long-term functional goals will be achieved and ultimately a net loss of services is realized at the regional scale.

Created Wetland Management

Because of the inherent complexity of wetlands, it can be difficult to replicate the hydrology, species composition, and soil characteristics of a natural wetland. Created wetlands typically have lower rates of decomposition, plant nutrients and plant production (Atkinson & Cairns, 2001; Fennessy et al., 2008). Additionally, created wetlands are more susceptible to invasive species and have lower plant diversity compared to natural wetlands, often as a result of the disturbed soil and altered hydrological conditions (Lázaro-Lobo & Ervin, 2021).

A variety of intentional and individually tailored management strategies can be implemented to help constructed wetlands reach desired performance outcomes. These can include altering the hydrology, manipulating soil topography or introducing native wetland species, based on the project and individual wetland needs (Moreno-Mateos et al., 2015). The response to management strategies is typically measured by the composition of the plant community, though this measure alone may not accurately reflect the overall wetland response (Matthews & Endress, 2008). Created wetlands have tradeoffs in the services they can provide, and plant community composition may not indicate the overall function. For instance, management focused on increasing biodiversity does not necessarily provide additional ecosystem services, such as nutrient removal or flood control, and is sometimes more negatively correlated with services than areas where management is not focused on biodiversity (Jessop et al., 2015; Naidoo et al., 2008). The effectiveness of the management strategies are also dependent on the specific characteristics of the individual wetland, with larger wetlands in warmer climates often showing quicker success than smaller wetlands in colder climates (Moreno-Mateos et al., 2012). Each strategy has the potential to alter the wetland's structure, which in turn influences the level and types of functions it can perform.

Wetland Structure and Function

Wetland soils are characterized by periods of flooding that create anoxic conditions and promote anaerobic processes. Wetland soil holds a reservoir of available plant nutrients, including nitrogen, carbon, phosphorus and sulfur (Lu et al., 2018). This leads to adaptable vegetation and promotes unique ecosystem services (Trettin et al., 2000). These conditions influence the microbes present in the soil that drive the key biogeochemical processes of the wetland, including nitrogen and carbon cycling (Faulwetter et al., 2009). Heterotrophic and autotrophic nitrogen fixing microbes convert atmospheric dinitrogen gas to reactive ammonium. The ammonium may be taken up by plants, nitrified by aerobic soil microbes, or oxidized back to dinitrogen gas through the process of ANAMMOX (anaerobic ammonium oxidation). Plant uptake of both reactive nitrogen species, ammonium and nitrate, results in storage in the vegetation of the wetland until the plant decomposes and releases the nitrogen back into the soil. Alternatively, nitrate can be converted back to atmospheric dinitrogen gas through denitrification. However, denitrification, which is an anaerobic process, may not be fully carried out in created wetlands where soil heterogeneity may result in incomplete denitrification and subsequent release of nitrous oxide, a potent greenhouse gas, into the atmosphere (Knowles, 1996).

Wetlands are also natural carbon sinks, holding 20-30% of the global soil carbon (Lal, 2008). Highly productive plants sequester carbon dioxide from the atmosphere, resulting in substantial quantities of carbon entering the anoxic soil upon senescence (Kolka et al., 2018; Mcleod et al., 2011). This organic matter is then decomposed, with the rate of decomposition determining overall carbon sequestration and the cycling of other nutrients, as carbon is often the limiting factor for a number of biogeochemical processes in wetlands (Taylor & Middleton, 2004). The hydrology and resulting anoxic conditions of flooded wetlands may cause decomposition to be slower than other ecosystems, further leading to net accumulation of carbon in the soil (Middleton, 2020). Additionally, certain bacteria consume the carbon for use in anaerobic respiration and performing denitrification.

When compared to natural wetlands, created wetlands only achieve around 74% of their biogeochemical functionality, with lower decomposition rates and sequestered carbon, ultimately resulting in reduced nutrient cycling (Fennessy et al., 2008; Moreno-Mateos et al., 2012; Wolf et al., 2011). In created wetlands, carbon availability may be up to four times lower than in natural wetlands, creating another discrepancy between the services provided by created and natural wetlands (Fennessy et al., 2008). Adding carbon rich organic matter to the soil is one management strategy that can help bridge this gap and enhance the services provided by created wetlands (Were et al., 2019).

Wetland Community Structure

The unique vegetation in wetlands is adapted to the saturated soil and abundance of nutrients not commonly found in other ecosystems. The structure of the vegetation community and functional groups present influences the overall functionality of the wetland. Vegetation plays an important role in the uptake of nutrients, acting as an additional temporary carbon sink while helping to facilitate other processes like nutrient removal (Brix, 1994; Kansiime et al., 2007). Different plant species contribute in unique ways to these processes, with some being more efficient at nutrient sequestration while others provide habitat or assist with soil structure. Additionally, when compared to monocultures, higher species richness increases potential nitrogen removal and belowground biomass in created wetlands (Bouchard et al., 2007; Geng et al., 2019). The role of the vegetation in these processes can also be affected by the vegetation type, age, and health, which are typically varying in created wetlands, and ultimately impact the successful restoration of ecosystem functions and services (Bruland & Richardson, 2005).

Another key, but often overlooked, component of the primary producer community in wetlands is the benthic microalgae (BMA). BMA are one of the major primary producers in aquatic ecosystems, with their success affecting other organisms in the wetland (Campeau et al., 1994). The critical ecosystem role of BMA has been well documented in salt marshes, but is lacking in freshwater environments (Benny et al., 2021). Moreover, it is not well known what factors affect BMA growth in a freshwater wetland, and how this, in turn, can impact other aspects of the ecosystem. Algal growth and abundance are dependent on multiple factors including nutrient availability, light, substrate, and disturbances (Dunck et al., 2013), all of which vary substantially among wetland type and between natural and created wetlands.

Of particular importance are the interactions between herbivores and vegetation. Not only does vegetation dictate which herbivores may be present, but herbivores reciprocally impact the vegetation, hydrology and soil characteristics (Lodge, 2017). Herbivores that utilize wetlands include a variety of organisms, from macroinvertebrates, including mollusks and annelids, to larger organisms like waterfowl and mammals. However, very little is known about the impacts of macroinvertebrates on the vegetation and soil function in freshwater wetlands, although invertebrates take on a number of functional roles, including grazing, decomposition of decaying organic material, and support for higher trophic levels (Covich et al., 1999). The macroinvertebrate community structure is impacted by the structure of the wetland with a major factor being hydrology which can vary between natural and created wetlands (Whiles & Goldowitz, 2005). Macroinvertebrates can directly and indirectly influence primary production and plant species composition, which will then create feedback to support higher trophic levels and delivery of desired functions and services (Stewart & Downing, 2008).

As in natural wetlands, the community structure of a created wetland is a dynamic and complex system consisting of reciprocal interactions among trophic levels. BMA abundance is controlled by nitrogen and phosphorus availability, so in created wetlands, a lack of nutrients may limit algal growth and adversely impact wetland structure and higher trophic levels (Mermillod-Blondin et al., 2020). The limited soil nutrients along with environmental stressors such as hydrology changes and physical disturbance also affects plant diversity and structure of created wetlands (Ehrenfeld, 2004, 2008; Fennessy et al., 2008). Certain macroinvertebrate functional groups use detritus from this vegetation as a food source which assists in the decomposition (Wissinger et al., 2021). This can negatively impact the nutrient cycling driven by unique macroinvertebrate and microbial communities (McCary et al., 2016; Santonja et al., 2020).

Created wetlands are highly susceptible to invasive species, often leading to an undesirable monoculture, and decreased habitat for organisms that play key functional roles and lower long-term biodiversity (Zedler & Kercher, 2004). An increase in invasive vegetation may have a negative effect on primary consumers and a positive effect on secondary consumers (McCary et al., 2016), because low diversity and undesirable non-native plants limit herbivores, effectively creating a bottom-up control. Additionally, this change in vegetation has also been linked to an increase in predation pressures on arthropod herbivores as the taller and denser vegetation that is often characteristic of invasive vegetation creates a better environment for arthropod predators, such as web building spiders (Finke & Denno, 2002; Langellotto & Denno, 2004). This creates a feedback loop between decreased herbivore abundance leading to an increase in invasive species. Changes to the vegetation community within a wetland can have cascading effects to other trophic levels, therefore, establishing diverse and resilient plant communities, like those in natural wetlands, is an important objective of creating a wetland (Carvalho et al., 2013).

Soil Amendments

Low soil organic matter can limit the functionality of a created wetland, undergirding many different functions. Organic matter helps to maintain the soil structure, prevent soil erosion, and retain water (Lal, 2020). It also serves as a reservoir of nutrients, including the nitrogen and phosphorus essential for plant growth. However, in created wetlands, carbon storage is about 26% lower than in natural wetlands (Fennessy et al., 2008; Moreno-Mateos et al., 2012; Yu et al., 2017). Due to the importance of carbon in a wetland ecosystem, soil amendments are one way to bridge the gap between created and natural wetlands. Many different types of soil amendments such as straw, topsoil, biochar and plant litter have been used to increase the carbon in wetlands (K. Ballantine et al., 2012; Rubin et al., 2020; Scott et al., 2020).

Soil amendments were first used in agriculture as a natural and sustainable way to improve the soil quality after overuse of the soil lead to erosion and a decrease in crop yield (Bakker et al., 2007; Eden et al., 2017). Soil amendments increase the availability of nutrients, soil moisture, and uptake of nutrients by plants (Oliveira et al., 2017; Tejada et al., 2009; Tu et al., 2006). In an agricultural setting, amendments were also shown to regulate soil pH, improve CO_2 uptake in the soil, and reduce greenhouse gas emissions (Bossolani et al., 2020; Li et al., 2012; Oster, 1982; Sun et al., 2019). Specifically, plant compost increases the soil structural stability, decreases bulk density of soil, improves microbial activity, and increases plant coverage by as much as 87% (Tejada et al., 2009). Thus, the use of soil amendments as a soil remediation tactic has been introduced into other ecosystems, including wetlands.

Soil compost additions have previously shown to positively benefit the wetland community producing similar effects to those observed in agricultural settings. In a wetland, the addition of decomposing leaf litter has increased soil organic matter and soil moisture, decreased bulk density and increased available nitrogen, inorganic phosphate and total carbon (McGowan, 2020; Owens, 2022; Williams, 2021). These increases in available soil nutrients, increases vegetation growth and nutrient cycling (Shaffer & Ernst, 1999). Moreover, soil amendments can promote denitrification without increasing N_2O flux within created wetlands by providing the necessary carbon for this process (K. A. Ballantine et al., 2014; Huang, 2021; McGowan, 2020). The introduction of carbon through leaf litter decomposition as a management strategy could provide support for a more robust community structure. However, the effects of the compost addition are dependent on many factors, including hydrology and prior land use, showing that continued monitoring is beneficial to better understand the lasting impacts of the soil amendments (Williams, 2021).

Overview of Study

This work expands on prior research done on the effects of leaf compost soil amendments at three created wetlands varying in type, hydrology, prior land use and age. The main objective of this study was to encapsulate more of the overall effects the soil amendments have on the community structure and function. This was achieved by evaluating changes in (1) soil properties through soil moisture, organic matter, bulk density and nutrients, (2) vascular plant community through vegetation surveys, and benthic microalgal abundance using chlorophyll *a* content as a proxy, (3) soil macroinvertebrate communities and (4) soil processes through soil respiration and metabolism. Overall, there is a stark gap in knowledge as to how wetland management strategies that aim to improve soil function may influence algal productivity, the nutrients available, and how this extends to have cascading impacts on community structure and this work aims to lessen that gap.

Methods

Site Description

This study took place at three created wetlands in Western New York (Figure 1a). Two wetlands, one emergent and one forested, were at High Acres Nature Area (HANA), in Perinton, NY (Figure 1c). HANA is a 101 ha private conservation site with a variety of different ecosystems and trails throughout that are open to the public. The third wetland, also forested, was in the Frances Willard Conservation Area in Riga, NY (Figure 1b). Frances Willard is also a private conservation site and has 79 ha of wetlands with 35 ha currently being restored. All three wetlands are owned and managed by WM of NY, LLC and were created in accordance with Section 404 of the Clean Water Act as compensatory mitigation for wetlands lost during landfill expansion.

At HANA, two created wetlands have undergone experimental leaf litter compost additions, Cady Wetland South and Packard Wetland cell A. Packard wetland complex at HANA is 1.5 ha and was converted from pasture to a series of forested wetlands and wet meadow in 2012. During the creation of this wetland, soils from an existing wetland were used to fill in where there were insufficient hydric soils. The soil composition in 2020 was 52% sand, 33% silt and 15% clay (Owens, 2022). This study focused on the forested wetland in the southeastern cell which is seasonally flooded, with the majority of water inflow occurring from late fall through spring. The experimental site is dominated by a variety of plants such as *Lythrum salicaria* (Purple Loosestrife), *Acer negundo* (Box Elder), and *Acer rubrum* (Red Maple).

Cady Wetland was created in 2009 after previously being used for row-crop agriculture. Similarly to Packard, soil from a nearby existing wetland was used to fill in the top 30 inches of soil, creating a 15 cm A horizon and a 15 cm B horizon, both classified as hydric. In 2020, the soil at this site showed to be 48% sand, 37% silt and 15% clay regardless of treatment (Owens, 2022). Cady Wetland consists of a forested wetland, vernal pools and an emergent wetland; this study took place at the emergent wetland in the southern portion of the site. This wetland became susceptible to invasive species such as *Typha latifolia* (Broadleaf Cattail), *Typha angustifolia* (Narrowleaf

Cattail) and *Phalaris arundinacea* (Reed Canary Grass) resulting in herbicide application in fall of 2017 and 2021 to eliminate *P. arundinacea*.

Frances Willard Conservation Area was previously used for row crop agriculture prior to transformation into several created wetland areas. Science Hill Wetlands were created as a forested wetland in 2019 and is seasonally flooded (Figure 2a). The soil composition in 2020 at this site was 47% sand, 26% silt and 27% clay (Owens, 2022). Dominant plants include *Typha* spp. (Cattail), *P. arundinacea* (Reed Canary Grass), *Populus deltoides* (Eastern Cottonwood), and *A. rubrum* (Red Maple).

Experimental Design

For each of the sites, leaf litter compost was added to every other transect in a pair-wise fashion. The additions were added to a depth of approximately 5 to 7 cm and had about 250 g C m^{-2}. The compost for past additions was 28 %C, 1.8 %N, with a molar C:N of 18.7 (Owens, 2022). The compost added to Science Hill during this study was 18 %C, 1.05 %N with a molar C:N of 20.3. In each wetland, the pairs of treatment and control transects serve as a 'zone' in a block design, which accommodates for spatial heterogeneity across each site. The leaf litter was collected by WM of NY, LLC and composted for approximately 1.5 years before transport to the wetlands. At Cady wetlands, five pairs of 2 m x 30 m transects were created at this site in Spring of 2014 and 2015 respectively and experimental compost additions were completed annually from 2014 to 2019 (Figure 1e). Similarly, at Packard wetlands, five pairs of 2 m x 30 m transects were created in the Spring of 2015 and experimental compost additions were applied annually from 2015 to 2020 (Figure 1f). At Science Hill wetlands, two pairs of 50 m x 4 m transects were created in 2019 and experimental compost additions were applied annually from 2019-2023 (Figure 1d). Roughly 50 ± 4% of the total soil area was covered with compost, after factoring in the plant cover (McGowan 2020).

At the two HANA sites, 4 pairs of transects were used. In each transect, two permanent plots were selected for sampling. These plots were 10 meters from each end of the transect in order to account for changes across the transect. At Frances Willard, each transect has four permanent plots, located at 10 m intervals from the eastern edge (Figure 1c). Each pair of control and compost addition transects was treated as an experimental block during statistical analysis, and each sampling plot was treated as an independent replicate, given the large size of the transect and spacing across the site.

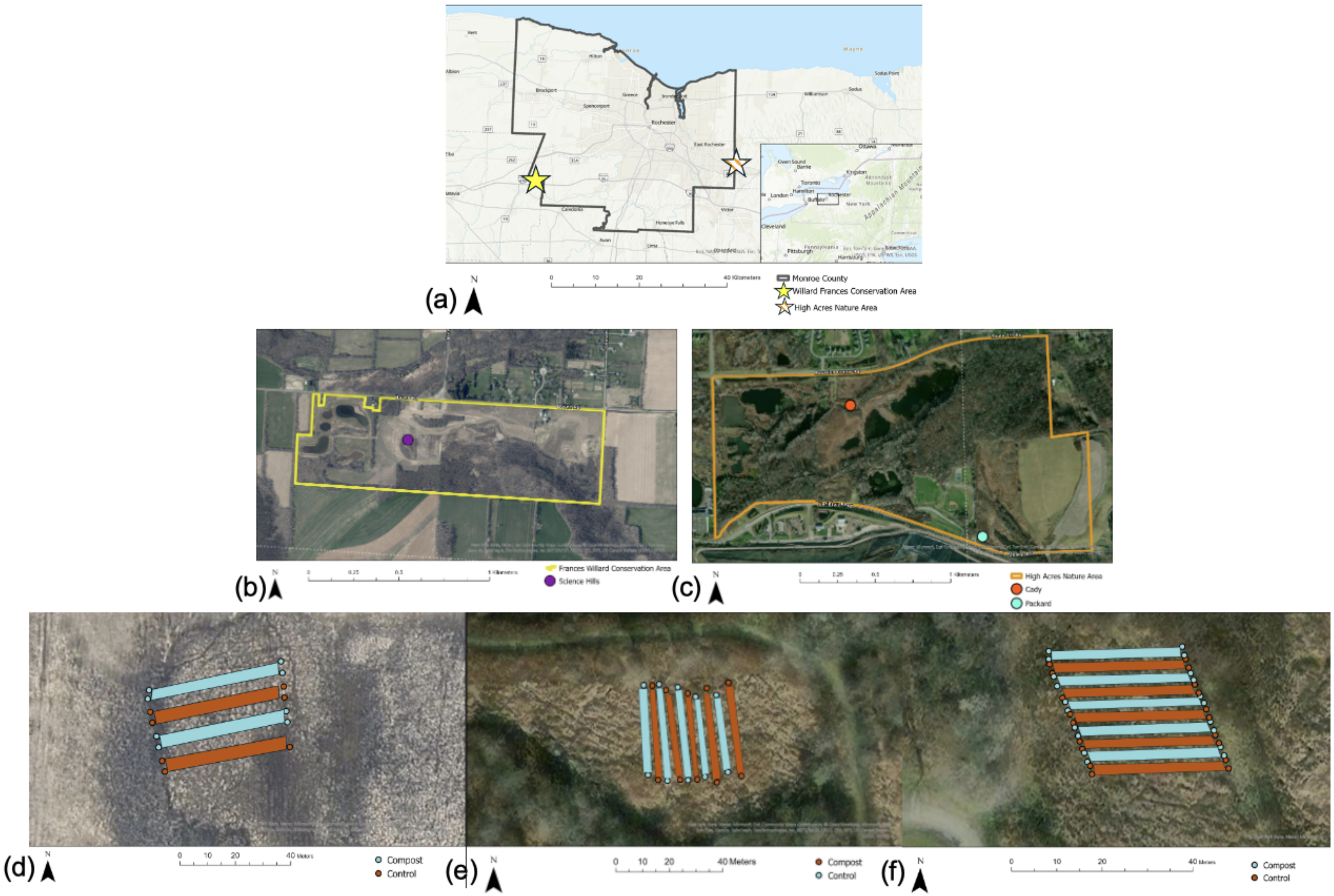

Figure 1. Location of experimental sites in (a) New York State at (b) Frances Willard Conservation Area wetland site; experimental transects at (d) Science Hill Wetlands and the two (c) High Acres Nature Area wetland sites; experimental transects at the (e) Cady Wetlands South and (f) Packard Wetland.

Field and Laboratory Methods

Beginning in May of 2023, one soil core was collected from each plot every four-six weeks through November using a metal tube 9 cm deep and 5 cm in diameter and placed on ice for transport. In June, September and November, the core was divided longitudinally into three parts and used to measure soil moisture (%SM), soil organic matter (%OM), nutrients and soil respiration (R). The nutrients measured were total nitrogen (%N), total carbon (%C), inorganic phosphorus (IP) and total phosphorus (TP) and extractable ammonium and nitrate. In the alternate months of May, August and October, %SM and bulk density (BD) were measured using the whole core. To determine the %SM, the soil was weighed before and after drying at 60º C for 48 hr. The dried soil was then homogenized using a mortar and pestle and combusted in a furnace at 550º C for 4 hr to determine the %OM using the loss-on-combustion method (Heiri et al., 2001). In June, around 10 g of dry soil was set aside prior to combustion for %N and %C as well as IP and TP analysis. %N and %C were measured using a Perkin Elmer 2400 CHNSO Elemental Analyzer. IP and TP content were measured colorimetrically using a Shimadzu UV 1900 Spectrophotometer following extraction using sulfuric acid and colorimetric assay using the ammonium molybdate method (Aspila et al., 1976). Soil for TP analysis was combusted at 550º C prior to acid extraction. The organic phosphate (OP) was calculated as the difference between TP and IP.

To assess extractable nitrate and ammonium, one third of the wet soil sample was frozen after collection and thawed at the point of extraction. Five grams of soil was mixed with50 mL of 2M KCl for 30 min before centrifugation to separate the soil from the extracted nutrients and filter the supernatant (Knepel, 2012). The extractable nitrate was measured at 540 nm following the vanadium reduction colorimetric method (Doane & Horwáth, 2003), and the extractable ammonium was measured at 630 nm using the sodium hypochlorite method (Solórzano, 1969).

Vegetation composition was measured in June and August in each plot. Two person teams determined the percent cover of each plant species within a 1 m^2 PVC frame set around each plot. The Floristic Quality Index (FQI) was calculated for the plant communities using NYSFOLA and NYBG guides for determining FQI (Atha & Boom, 2018; NYSFOLA, n.d). Plant alpha diversity was calculated using the Shannon-Wiener index.

Soil chlorophyll *a* (Chl *a*) was used as a proxy for benthic algal abundance. Two 1.1 cm diameter x 1 cm deep soil samples per plot were taken using a 5-cc syringe corer, placed in test tubes, immediately wrapped in aluminum foil to prevent exposure to light, and placed on ice until returned to the lab where they were stored at -80 °C in the dark until analysis. Chlorophyll was extracted by sonication in 90% acetone followed by incubation at -20 °C. The samples were then shaken, centrifuged, and the absorbance of the supernatant quantified using a Shimadzu UV 1900 Spectrophotometer at 665 nm and 750 nm before and after the addition of 1M HCL (Strickland & Parsons, 1972). Final concentrations of Chl *a* and phaeopigment were calculated using the Lorenzen (1967) equations.

Soil R rates were measured by placing ~25 g of wet soil into a wide mouth mason jar within 24 hr of collection with a paired sample being dried to have the corresponding dry weight. The mason jar was sealed with an airtight lid fitted with tubing connected to a small air pump and a LI-COR LI-820 CO_2 Gas Analyzer to recirculate headspace gasses. The CO_2 concentration was measured once per plot for approximately 10 min, or until a consistent slope of CO_2 increasing over time was observed. The headspace was subsequently measured by filling the jar with water and measuring the mass of the added water.

An additional soil core was collected at half of the plots in June, September and November using a 9.5 cm diameter polycarbonate core to a depth of 10 cm to measure primary production and ecosystem metabolism by quantifying the change in CO_2 concentration in a sealed headspace within 24 hours of collection. In lieu of technical issues that compromised data quality, data from June and September were ultimately discarded. These cores were used in the analysis of macroinvertebrate communities at all three time points. The core was left inside overnight and then wrapped in aluminum foil to keep all light out before measurements were taken. A sealed lid with inlet and outlet tubes was placed on the top of the core and connected to the LiCor LI820 Infrared CO_2 Gas Analyzer. The airflow into the sealed tube was approximately 500 milliliters per minute (ml min^{-1}), ensuring that the full volume was recirculated roughly every 2 min. The CO_2 concentration in the headspace was recorded every sec for 5-7 min in the dark. Then the chamber was unwrapped, opened and placed in direct sunlight outdoors for a 30 min equilibration period. The CO_2 concentrations were then measured for another 5-7 minutes. Headspace volume was measured using the height of the core and the connected tubing. The change in CO_2 concentration over time in the light and the dark was used to determine the areal CO_2 flux, and calculate the gross soil primary production, net soil ecosystem metabolism and soil R. The areal R was calculated using the dark flux values. Hourly soil Gross Primary Production (GPP) was calculated based on the difference between light and dark fluxes. Net Soil Ecosystem Metabolism (NEM) was calculated by multiplying the number of hours of light and dark (10 hr and 14 hr, respectively) in November when measurements were made.

In June, September and November, following metabolism measurements, cores were used to determine the macroinvertebrate abundance and composition. In the plots where soil was not collected using the polycarbonate tube, a 5 cm diameter (10 cm deep) core was used to take a soil sample to determine macroinvertebrate abundance. Each core was sieved (355 μm) to isolate macroinvertebrates. Isolated specimens were preserved in 70% ethanol and identified using a dissecting microscope to the lowest practical taxonomic level with each specimen identified to at least class. After identification, abundance was scaled to a m^2 based on area of cores and these values were used to calculate alpha diversity using the Shannon-Wiener index (H').

Statistical Analysis

Statistical Analysis was performed using JMP Pro 16 and RStudio. Two-way analysis of variance (ANOVA) with treatment, time and their interaction as fixed factors was used to assess the impact of compost and season for each site separately. Block was included as a random factor to account for any spatial differences among transects at each site. All variables were assessed prior to testing to ensure compliance with the assumptions of ANOVA (normality and homogeneity of variance). When there was significance indicated for time or the treatment x time interaction, a post-hoc Tukey (HSD) test was used. For Areal R, GPP and NEM that were measured only once in

November, a one-way ANOVA was used to assess treatment differences at each site. Beta diversity for the plant and macroinvertebrate communities was evaluated using Non-metric Multidimensional Scaling (NMDS) and the Bray Curtis dissimilarity index. Lastly, a principal components analysis (PCA) was conducted to evaluate overall similarities among treatments across all sites. The PCA was conducted on the overall plot mean across all time points for each variable on a reduced set of variables to reduce collinearity. A Pearson correlation was run on the full data set, and for pairs of variables with high correlation coefficients (> 0.85), one of the two variables was eliminated.

Results

Soil Characteristics

Sites varied in soil characteristics, with generally higher %OM and %SM at Cady than other sites (Table 1). Likewise, the effects of compost addition on soil characteristics varied by site. The %OM at Packard was significantly higher in September (23%) than in June (17%) and was on average 21% higher in the compost (22%) than the control (18%) (p=0.010, p=0.03 respectively; Figure 2a; Table 1, Table 2; Table A1). %SM varied seasonally with higher soil %SM in the Spring than in the Fall (p<0.001; Figure 2b; Table 1, Table 2, Table A2). %SM was on average higher in the compost than the control at Packard (p=0.04; Table 1, Table 2, Table A2). Soil bulk density was significantly lower (11%) in the compost than in the control and did not vary seasonally (p=0.04, p=0.17 respectively; Table 1, Table 2, Table A2).

Table 1. Mean ± SE of %OM, %SM, BD (g cm^{-3}), NO_3^- (mg N kg^{-1}), NH_4^+ (mg N kg^{-1}), NO_3^-:$NH_4^\pm$, TP (mg P kg^{-1}), IP (mg P kg^{-1}), OP (mg P kg^{-1}), IP:OP, C:N, %C, %N, GPP (µg C cm^{-2} hr^{-1}), NEM (µg C cm^{-2} day^{-1}), Areal R (µg C g $soil^{-1}$ hr^{-1}), R (µg C g $soil^{-1}$ hr^{-1}), and Chl *a* (mg m^{-2}) measured in Packard, Cady and Science Hill in compost and control treatments averaged across all time points.

	Packard		**Cady**		**Science Hill**	
	Compost	**Control**	**Compost**	**Control**	**Compost**	**Control**
OM	22 ± 1	18 ± 1	32 ± 2	24 ± 2	30 ± 2	26 ± 2
%SM	41 ± 2	38 ± 2	59 ± 1	52 ± 1	45 ± 3	45 ± 3
BD	0.57 ± 0.02	0.63 ± 0.02	0.43 ± 0.02	0.51 ± 0.02	0.39 ± 0.02	0.47 ± 0.02
NO_3^-	16 ± 1	11 ± 1	5 ± 0.5	4.4 ± 0.5	7.2 ± 2	6.9 ± 2
NH_4^+	20 ± 1	16 ± 1	15 ± 1	14 ± 1	10 ± 2	10 ± 2
NO_3^-:$NH_4^\pm$	2 ± 1	5 ± 1	52 ± 13	38 ± 13	12 ± 6	3 ± 6
TP	1,502 ± 43	1,468 ± 43	1,584 ± 56	1,491 ± 56	1,550 ± 57	1,384 ± 57
IP	988 ± 23	942 ± 23	894 ± 13	870 ± 13	958 ± 26	889 ± 26
OP	515 ± 30	526 ± 30	690 ± 52	621 ± 52	592 ± 47	495 ± 47
IP:OP	2.3 ± 0.3	2.0 ± 0.3	1.4 ± 0.1	1.5 ± 0.1	1.9 ± 0.2	2.2 ± 0.2
C:N	16 ± 0.2	16 ± 0.2	17 ± 0.8	17 ± 8	19 ± 0.4	20 ± 0.4
%C	9.7 ± 1	7.5 ± 1	13 ± 1	11 ± 1	13 ± 2	12 ± 2
%N	0.70 ± 0.06	0.54 ± 0.06	0.92 ± 0.08	0.77 ±0.08	0.78 ± 0.10	0.75 ± 0.10
GPP	-8 ± 3	-4 ± 3	-0.9 ± 1	-0.9 ± 1	-12 ± 4	-2.2 ± 2
NEM	563 ± 118	274 ± 118	197 ± 6	194 ± 6	745 ± 67	304 ± 67
Areal R	26.9 ± 5.5	13.0 ± 5.5	8.6 ± 0.7	8.4 ± 0.7	36.1 ± 3.2	13.6 ± 3.2
R	11 ± 2	9 ± 2	50 ± 8	29 ± 8	17 ± 2.0	9 ± 2
Chl *a*	33 ± 2	43 ± 2	36 ± 4	60 ± 4	26 ± 3	39 ± 3

At Cady, there was an interaction between season and treatment with %OM significantly higher in the compost (38%) than the control in November (19%) (p=0.011; Figure 2c; Table 1, Table A1). On average, there was a 24% increase in the %OM from control to compost (p=0.007; Table 1, Table 2). There was a significant interaction between season and treatment for %SM in August and November where the compost was higher than the control (p=0.012, Table 2d). %SM was on average significantly higher in the compost than the control (p<0.001; Figure 3d; Table 1, Table 2). The bulk density was significantly lower (18.6% decrease) in the compost (0.43 g cm^{-3}) than in the control (0.51 g cm^{-3}) and did not vary seasonally (p=0.03, p=0.67 respectively; Table 1, Table 2, Table A3).

At Science Hill, there were no treatment or seasonal differences in %OM (p=0.24, p=0.31 respectively; Figure 23; Table 1, Table 2). The %SM at Science Hill varied seasonally and overall decreased from Spring to Fall (p<0.001; Figure 2f; Table 1, Table A4). There was no significant difference in the %SM between treatments (p=0.77; Table 2). The bulk density was 21% lower in the compost (0.39 g cm^{-3}) than in the control (0.47 g cm^{-3}l) (p=0.03; Table 1, Table 2). There was also seasonal variation in the bulk density at Science Hill, with higher values in May (0.52 g cm^{-3}) and August (0.46 g cm^{-3}) than October (0.32 g cm^{-3}) (p=<0.001; Table 2, Table A4).

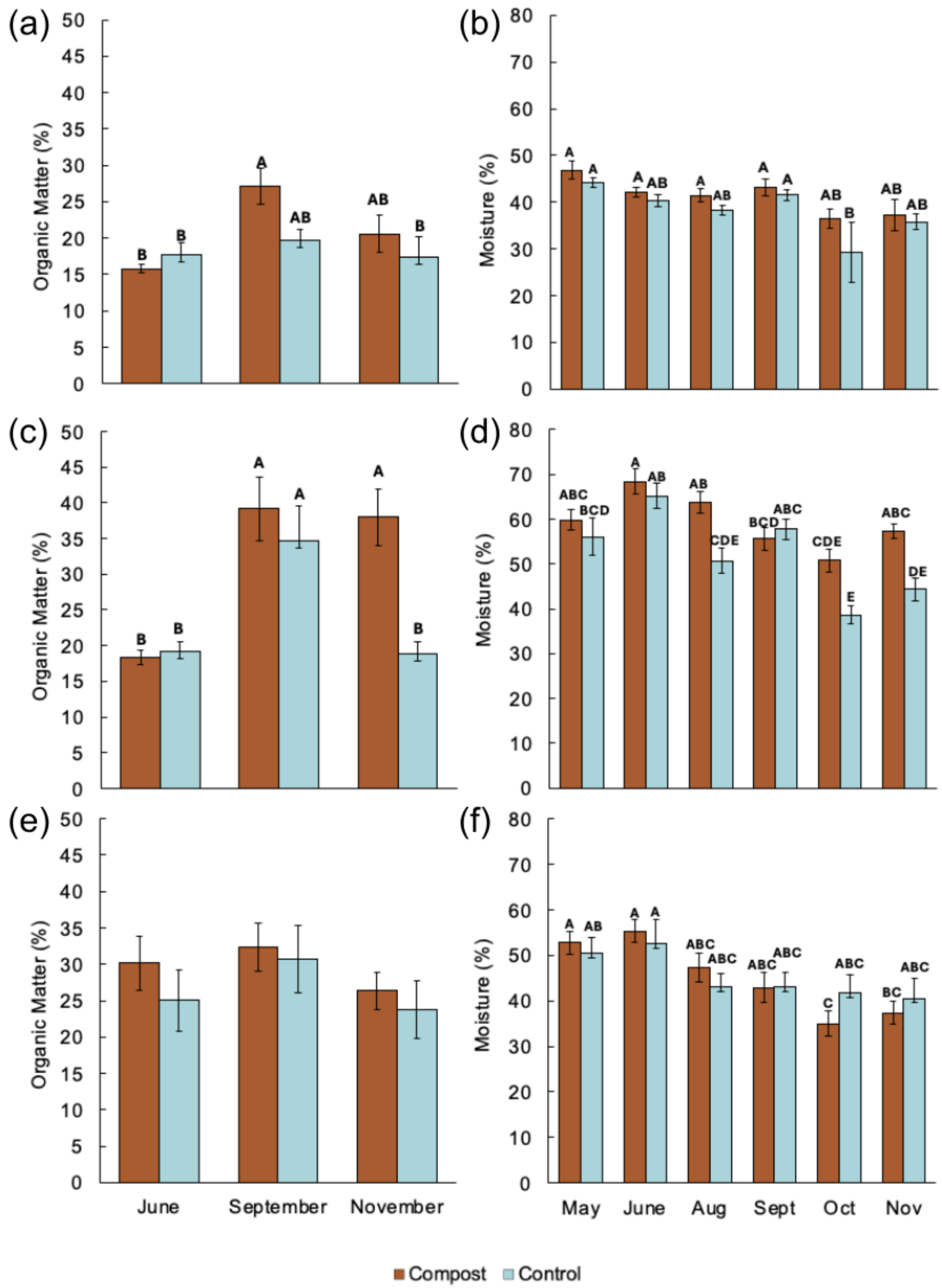

Figure 2. %OM and %SM, respectively, in the compost and control treatments at (a, b) Packard, (c, d) Cady, and (e, f) Science Hill. Values are mean ± SE, n=8. Values that do not share a letter are significantly different from one another.

Table 2. Results of two-way ANOVA examining the effects of time and treatment on soil characteristics. Significant p-values are bolded and p-values approaching significance are italicized.

		Packard			Cady			Science Hill		
		Season	Comp	Season x Comp	Season	Comp	Season x Comp	Season	Comp	Season x Comp
OM	*p*	**0.010**	**0.028**	*0.056*	**<0.001**	**0.007**	**0.011**	0.24	0.31	0.83
	$F_{6,42}$	5.2	5.2	3.1	15.4	8.1	5.0	1.5	1.1	2.4
%SM	*p*	**<0.001**	**0.037**	0.430	**<0.001**	**<0.001**	**0.022**	**<0.001**	0.77	0.89
	$F_{12,80}$	24.0	0.5	1.0	6.7	7.9	3.1	59.0	0.2	0.1
BD	*p*	0.17	**0.039**	0.33	0.67	**0.032**	0.28	**<0.001**	**0.029**	–
	$F_{6,42}$	1.8	4.5	1.2	0.4	4.9	0.3	11.7	5.1	–
NO_3^-	*p*	**<0.001**	*0.088*	0.69	**<0.001**	0.71	0.11	**<0.001**	0.84	–
	$F_{6,42}$	9.3	3.1	0.4	11.8	0.1	2.3	13.3	0.04	–
NH_4^+	*p*	**<0.001**	*0.094*	**0.010**	**<0.001**	0.75	0.96	**<0.001**	0.67	0.99
	$F_{6,42}$	17.8	2.9	5.2	35.7	0.1	0.04	123.5	0.2	0.01
NO_3^-: NH_4^+	*p*	**<0.001**	*0.052*	*0.091*	**<0.001**	0.5	0.7	0.21	0.32	0.20
	$F_{6,42}$	8.7	4.0	0.1	12.9	0.4	0.3	1.6	1.0	1.7
TP	*p*	**0.002**	0.59	0.43	0.80	*0.065*	**0.025**	**<0.001**	**0.001**	0.37
	$F_{6,42}$	7.6	0.3	0.9	0.2	3.6	4.03	9.6	12.4	1.0
IP	*p*	**<0.001**	0.25	0.25	0.46	0.32	0.45	**<0.001**	**0.046**	**0.007**
	$F_{6,42}$	19.8	0.3	0.2	0.8	1.0	0.5	41.2	4.2	5.7
OP	*p*	0.20	0.80	0.74	0.5	0.17	*0.064*	0.18	*0.069*	**0.003**
	$F_{6,42}$	1.7	0.1	0.3	0.8	1.9	2.9	1.8	3.5	6.8
IP: OP	*p*	0.34	0.31	0.53	0.57	0.46	0.37	**0.023**	0.30	0.53
	$F_{6,42}$	1.1	1.0	0.6	0.6	0.5	1.0	4.1	1.1	0.7
C:N	*p*	–	0.70	–	–	0.67	–	–	0.62	–
	F_{15}	–	0.2	–	–	0.2	–	–	0.3	–
%C	*p*	–	0.12	–	–	0.21	–	–	0.84	–
	F_{15}	–	2.7	–	–	1.7	–	–	0.0	–
%N	*p*	–	*0.097*	–	–	0.2	–	–	0.84	–
	F_{15}	–	3.2	–	–	1.8	–	–	0	–
R	*p*	**0.010**	0.67	0.82	**0.002**	*0.068*	0.96	**0.014**	**0.006**	–
	$F_{6,42}$	5.2	0.2	0.2	7.1	3.5	0.05	4.7	8.6	–
GPP	*p*	–	*0.08*	–	–	0.9	–	–	**0.004**	–
	F_3	–	4.5	–	–	0.0	–	–	21.3	–
NEM	*p*	–	0.30	–	–	0.77	–	–	**0.004**	–
	F_3	–	3.0	–	–	0.1	–	–	21.5	–
Areal R	*p*	–	0.12	–	–	0.89	–	–	**0.003**	–
	F_3	–	3.2	–	–	0.0	–	–	24.8	–
Chl a	*p*	**<0.001**	**0.003**	**0.004**	**<0.001**	**0.002**	**0.002**	**<0.001**	**<0.001**	**0.041**
	$F_{12,80}$	34.4	9.3	3.8	19.4	10.8	4.1	50.1	16.0	2.4

Soil Nutrients

Soil nitrate increased from Spring (0.62 mg N kg^{-1}) to Fall (0.71 mg N kg^{-1}) at Packard, but did not differ between treatments (p=0.001, p=0.08 respectively; Table 1, Table 2, Table A1). There was a significant interaction between season and treatment in ammonium content, with ammonium being higher in the compost (17 mg N kg-1) than the control (7 mg N kg-1) in November (p=0.01; Table 2, Table A1). Total phosphate and inorganic phosphate content was higher in September and November than in June (p=0.002, p=<0.001 respectively; Table 1, Table A1). However, there was no treatment effect for inorganic phosphate or total phosphate (p=0.3, p=0.6 respectively; Table 2). There were no differences between season and treatment in organic phosphate (p=0.2, p=0.8 respectively; Table 2). There were no significant differences between the compost and the control for molar C:N, %C and %N (p>0.05; Table 1, Table 2).

At Cady, the soil nitrate increased from June (0.00 mg N kg^{-1}) to September (0.35 mg N kg^{-1}) and November (0.52 mg N kg^{-1}) and had no significant difference between treatments (p=<0.001, p=0.71 respectively; Table 2, Table A1). The soil ammonium decreased from June (19 mg N kg-1) and September (25 mg N kg^{-1}) to November (0.02 mg N kg-1) and had no significance between treatments (p=<0.001, p=0.75 respectively; Table 2, Table A1). The interaction between season and treatment for the total phosphate was significant at Cady, with the compost (1,690 mg P kg^{-1}) treatment being different from the control (1,417 mg P kg^{-1}) treatment in November (p=0.025; Table 2, Table A1). There was no significance in season or treatment for the inorganic phosphate (p=0.46, p=0.32 respectively; Table 2) or for organic phosphate (p=0.5, p=0.17 respectively; Table 2). There were no significant differences between the compost and the control for molar C:N, %C and %N (p>0.05; Table 1, Table 2).

At Science Hill the nitrate was higher in September (0.34 mg N kg^{-1}) and November (0.86 mg N kg^{-1}) than June (0.04 mg N kg^{-1}) and had no significance between treatments (p=<0.001, p=0.84 respectively; Table 2, Table A1). The soil ammonium was higher in September (27 mg N kg^{-1}) than in June (3 mg N kg^{-1}) and November (0.013 mg N kg^{-1}) and had no treatment significance (p=<0.001, p=0.67 respectively; Table 2, Table A1). There was a significant interaction between treatment and month for the inorganic phosphate but there were no pairwise differences ($p=0.007$; Table 2). There was no seasonal significance for the OP and the treatment difference was approaching significance with the compost (592 mg P kg^{-1}) being higher than the control (495 mg P kg^{-1}) ($p=0.18$, $p=0.07$ respectively; Table 1, Table 2). The ratio of IP to OP was similar between treatments at all sites but did vary seasonally at Science Hill, with June having higher levels than November ($p>0.05$, $p=0.023$ respectively; Table 2, Table A1). There were no significant differences between the compost and the control for molar C:N, %C and %N ($p>0.05$; Table 1, Table 2).

Soil Processes

Soil respiration assessed as a stand-alone measurement from homogenized soil samples was higher in June than in November at all the sites ($p<0.05$; Figure 3; Table 2) but higher in the compost (17 μg C g $soil^{-2}$ hr^{-1}) than the control (9 μg C g $soil^{-2}$ hr^{-1}) only at Science Hill ($p=0.006$, Figure 3c, Table 1, Table 2). At Cady, the treatment effect approached significance with the compost (50 μg C g $soil^{-2}$ hr^{-1}) higher than the control (29 μg C g $soil^{-2}$ hr^{-1}) ($p=0.068$; Figure 3b, Table 1, Table 2), but there were no differences at Packard ($p=0.67$; Figure 3a; Table 2). For the process measurements assessed on intact soil cores in November (only), the areal respiration rate had a treatment effect at Science Hill where the compost (36 μg C cm^{-2} hr^{-1}) was higher than the control (14 μg C cm^{-2} hr^{-1}) ($p=0.003$; Table 1, Table 2), but there were no differences at Packard ($p=0.12$; Table 2) or Cady ($p=0.89$; Figure 4a;Table 2). At Science Hill there was significantly higher soil/algal GPP in the compost (-12 μg C cm^{-2} hr^{-1}; negative sign indicates gross uptake of CO_2 across the soil interface) than the control (-2 μg C cm^{-2} hr^{-1}; $p=0.004$; Figure 4b; Table 1, Table 2). At Packard the GPP was approaching significance with the compost (-8 μg C cm^{-1} hr^{-2}) higher than the control (-4 μg C cm^{-2} hr^{-1}; $p=0.08$, Figure 4b; Table 1, Table 2). There was no difference in GPP between treatments at Cady in the fall ($p=0.9$; Figure 4b; Table 2). The NEM was significantly higher in compost (745 μg C cm^{-2} day^{-1}) than the control (304 μg C cm^{-2} day^{-1}) at Science Hill ($p=0.004$; Figure 4c Table 1, Table 2), but no differences were found at Packard or Cady ($p>0.05$; Table 2; Figure 4c).

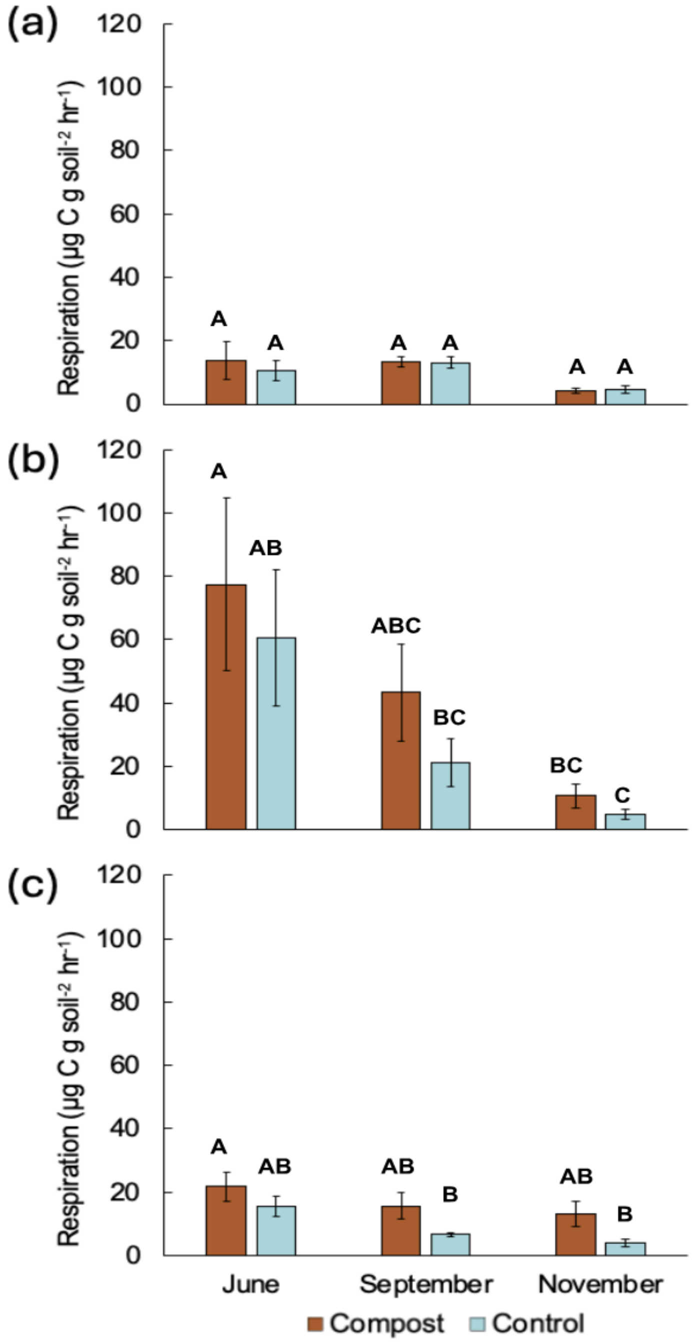

Figure 3. R rates in the compost and control treatment at (a) Packard, (b) Cady, and (c) Science Hill. Values are mean ± SE, n=8. Values that do not share a letter are significantly different from one another.

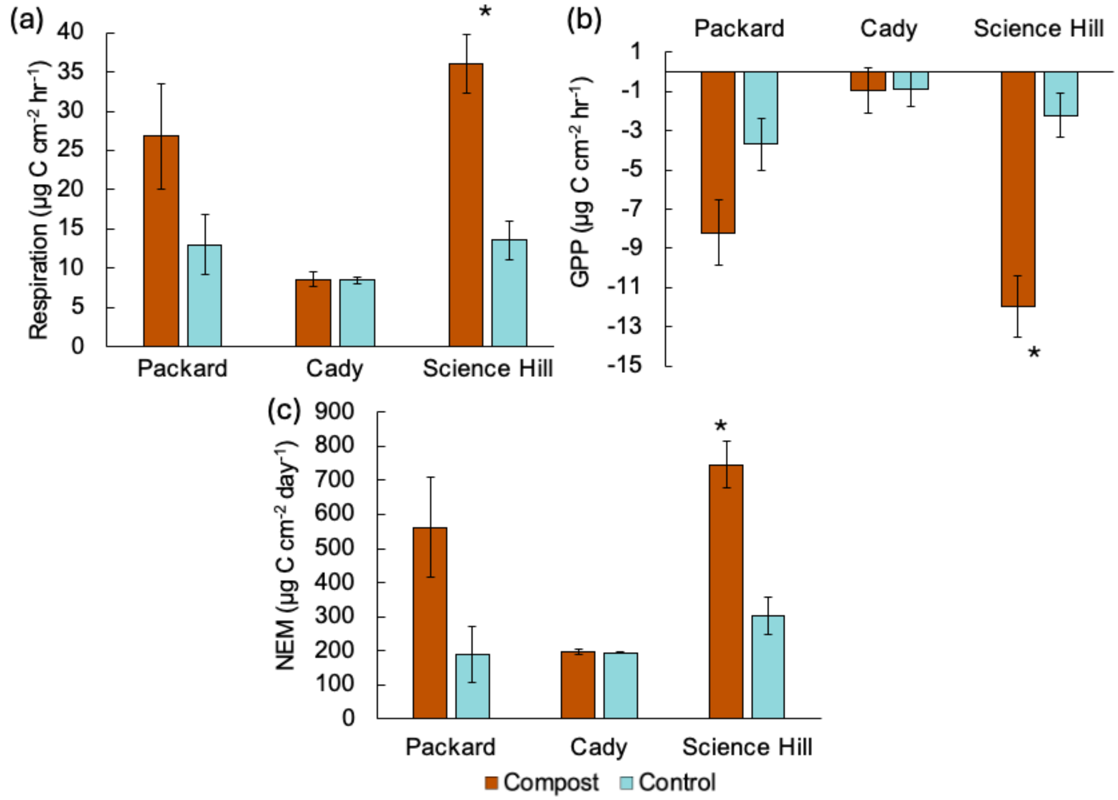

Figure 4. The (a) Gross Soil Primary Production (GPP), (b) Net Soil Ecosystem Metabolism (NEM) and (c) soil respiration the compost and control treatment measured in intact soil cores at Cady, Packard, and Science Hill in November 2023. An average day was estimated to have 14 hours of dark and 10 hours of light. Values are mean ± SE, n=4. Values that do not share a letter are significantly different from one another.

Community Structure

Average Chlorophyll *a* content was reduced in the compost treatment by 23%, 40%, and 33% at Packard, Cady and Science Hill (Table 1) and generally increased from spring through fall (Figure 5; Table A2, Table A3, Table A4). However, a time x treatment interaction at all sites was likely driven by the smaller treatment differences in spring and very large difference in the fall (p=<0.05 for time x treatment interaction; Figure 5; Table 2).

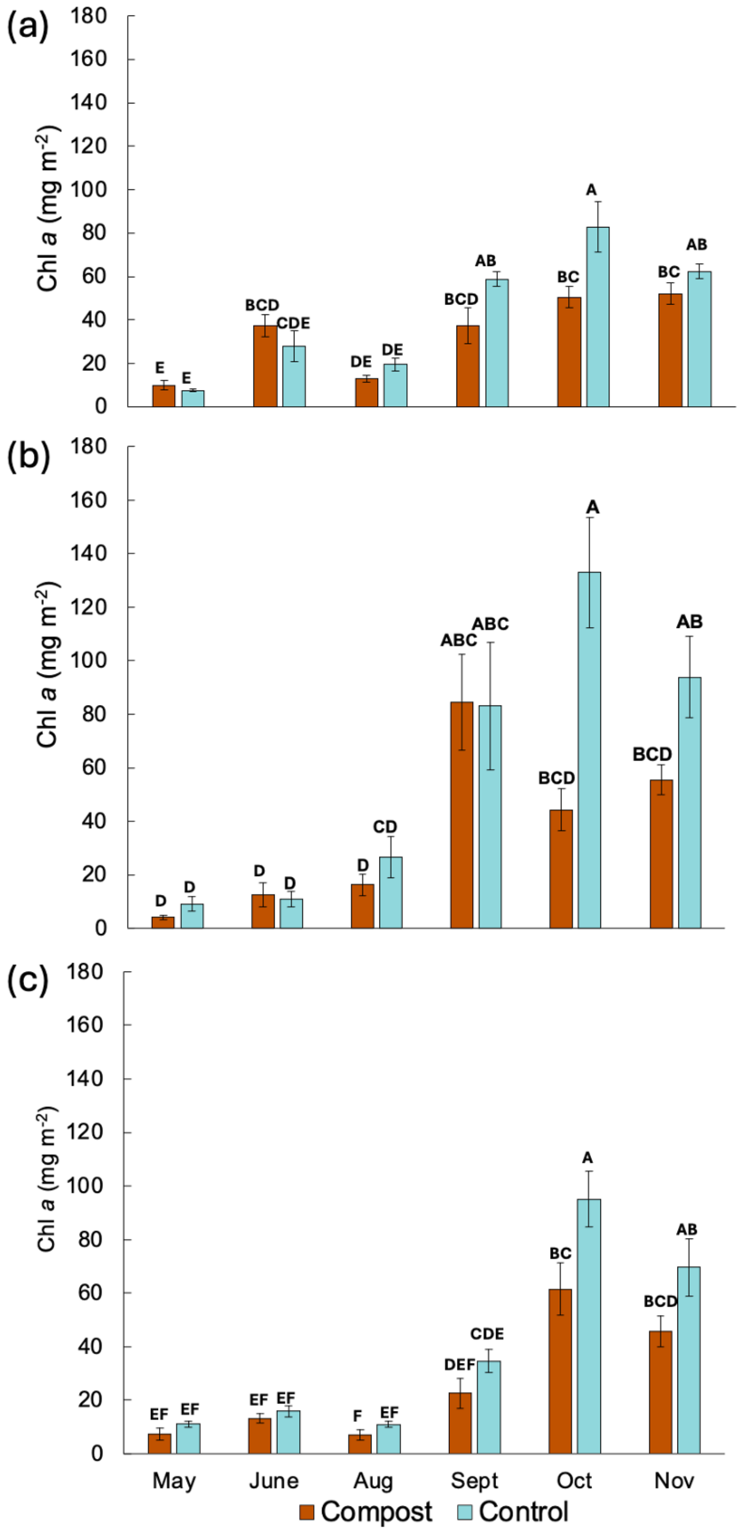

Figure 5. Chlorophyll *a* concentrations in the compost and control treatment at (a) Packard, (b) Cady, and (c) Science Hill. Values are mean ± SE, n=8. Values that do not share a letter are significantly different from one another.

At Packard, species richness for the vegetation was higher in August (4.8) than in June (3.8), but did not vary across treatments (p=0.035, p=0.28 respectively; Table 4, Table A2). The H' also increased from June (0.9) to August (1.3) but had no difference in the treatment (p=0.01, p=0.8 respectively; Table 4, Table A2). The FQI was not statistically significant for treatment or season (p>0.05; Table 4). The compost treatment had significantly higher percent plant cover in the control (112%) than the compost (90%) and increased from June (90%) to August (112%) (p=0.026, p=0.023, respectively; Figure 6a; Table 3, Table 4, Table A2). At Cady there was no seasonal variation for the species richness but was higher in the control (4.8) than the compost (3.6) (p=0.26, p=0.020 respectively; Table 3, Table 4). The H' increased from June (0.8) to August (1.2) and was lower in the compost (0.7) than the control (1.2) (p=0.04, p=0.008 respectively; Table 3, Table 4, Table A3). The FQI was not statistically significant for treatment or season (p>0.05; Table 4). The control (131%) had a higher percent plant cover than the compost (108%) and had no seasonal changes (p=0.007, p=0.5 respectively; Figure 6b; Table 3, Table 4). At Science Hill there was no significance in species richness for season or treatment (p>0.05; Table 4). The H' was not significant for season or treatment (p>0.05; Table 4). The FQI was not statistically significant for treatment or season (p>0.05; Table 4). The effects of treatment were approaching significance for total plant cover with the control (102%) being higher than the compost (92%) and there was no seasonal significance (p=0.09, p=0.3 respectively; Figure 6c; Table 3, Table 4). The NMDS of the vegetation community had a stress value of 0.208 showed samples clustered by site more so than treatment (Figure 7).

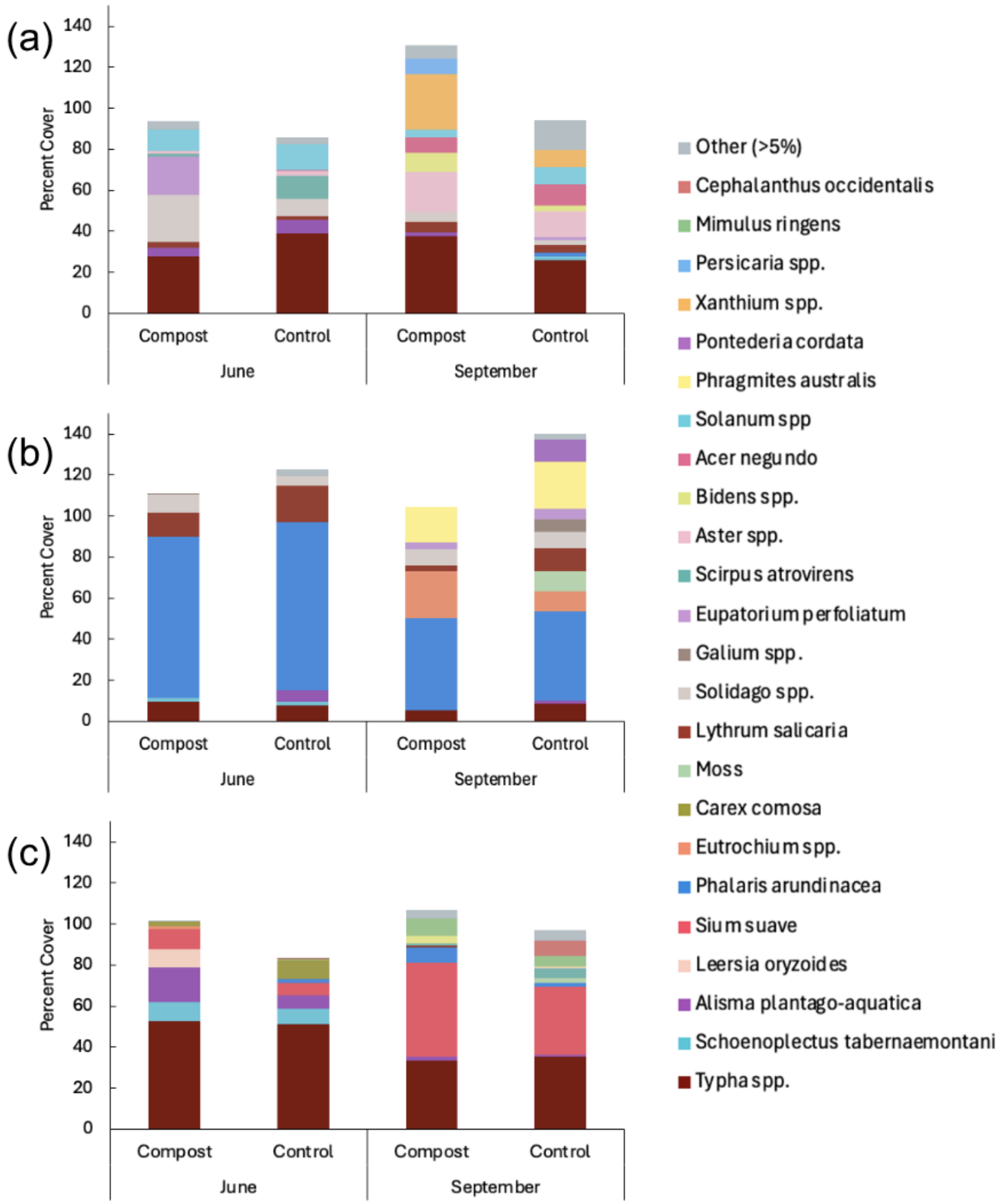

Figure 6. Percent vegetation cover in the compost and control treatment at (a) Packard, (b) Cady, and (c) Science Hill. Percent cover can exceed 100% due to overlapping layers of vegetation. Values are mean for n=8

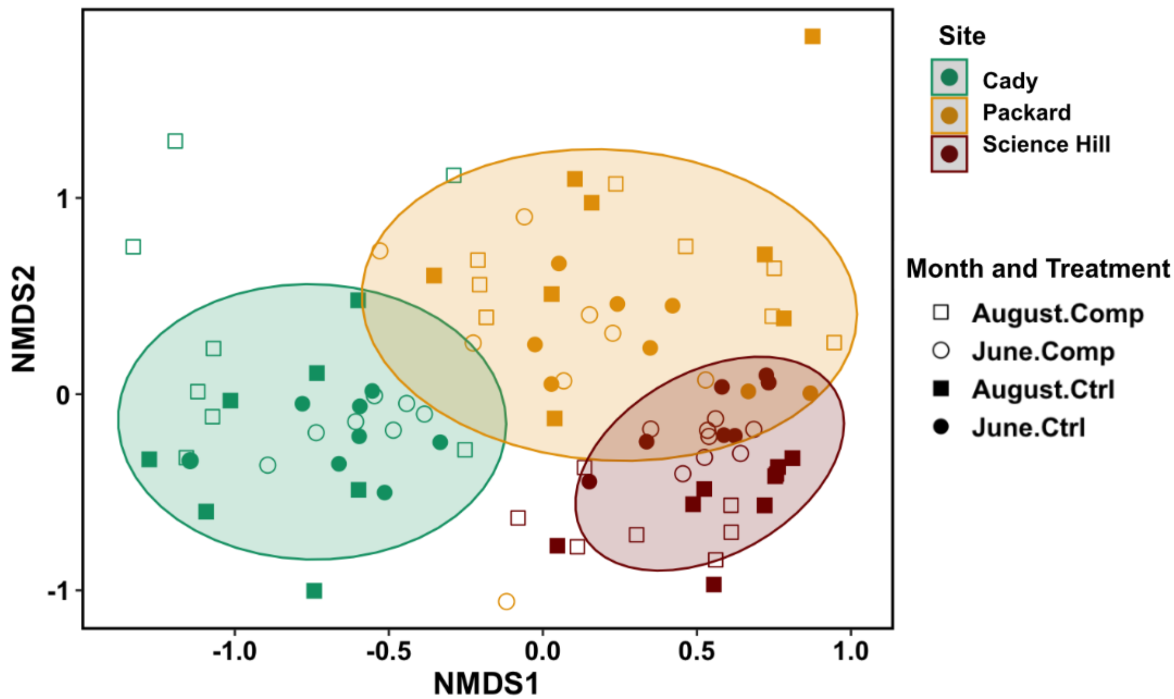

Figure 7. Non-metric Multidimensional Scaling (nMDS) of vegetation community based on Bray-Curtis dissimilarities at Cady, Packard and Science Hill. Shaded ellipses indicate 90% confidence intervals based on each site and a stress value of 0.208.

Table 3. Species richness (S), Shannon diversity (H'), FQI and total vegetation cover (TC %) measured in Packard, Cady and Science Hill in compost and control treatments averaged across all time points. Values are mean ± SE.

	Packard		Cady		Science Hill	
	Compost	**Control**	**Compost**	**Control**	**Compost**	**Control**
S	4.5±0.4	4.0±0.4	3.6±0.3	4.8±0.3	4.9±0.4	4.1±0.4
H'	1.1±0.1	1.1±0.1	0.7±0.1	1.2±0.1	1.2±0.1	1.0±0.1
FQI	10.3±1.4	9.2±1.4	4.9±1.7	9.0±1.8	19.4±1.8	16.1±1.8
TC	90±5	112±5	108±10	131±10	93±6	102±6

Table 4. Results of two-way ANOVA examining the effects of season (Spring, Summer) and treatment on vegetation species richness (S), Shannon diversity (H'), FQI and total cover (TC). Significant p-values are bolded and p-values approaching significance are italicized.

		Packard			Cady			Science Hill		
		Season	Comp	Season x Comp	Season	Comp	Season x Comp	Season	Comp	Season x Comp
S	*p*	**0.035**	0.28	0.69	0.26	**0.020**	0.24	0.72	0.33	0.60
	$F_{4,28}$	4.9	1.2	0.69	1.3	6.3	1.5	0.1	1.0	0.3
H'	*p*	**0.011**	0.76	0.70	**0.037**	**0.008**	*0.076*	0.26	0.12	0.22
	$F_{4,28}$	7.4	0.10	0.15	4.9	8.3	3.5	1.3	2.6	1.6
FQI	*p*	0.82	0.56	0.52	*0.090*	0.12	0.49	0.14	0.20	0.16
	$F_{4,28}$	0.05	0.35	0.43	3.1	2.6	0.5	2.3	1.7	2.1
TC	*p*	**0.023**	**0.026**	0.15	0.511	**0.007**	0.143	0.25	*0.09*	0.60
	$F_{4,28}$	5.8	5.6	2.3	0.4	8.7	2.3	1.4	3.1	0.3

There was no treatment effect on H' for macroinvertebrates at the Class level at any site, but there were some seasonal patterns. The H' increased significantly from June to November at Packard and from June to September and November at Cady but was not significant at Science Hill (p=0.01, p=0.0001, p=0.24 respectively; Table 6, Table A1). The species richness increased significantly from June to September and November at Cady and from June to September at Packard (p=0.006, p=0.03 respectively; Table 6, Table A1), but not at Science Hill (p=0.12; Table 6). The abundance of macroinvertebrates was not significant at any site for season or treatment (p>0.05; Figure 8; Table 6). The NMDS of the macroinvertebrate community had a stress value of 0.198 and showed substantial overlap among all of the sites (Figure 9). Although not as evident in the NMDS, the Bray-Curtis dissimilarity analysis shows clustering by season rather than site. Within each season, sites tended to be closely related rather than treatments (Figure 10).

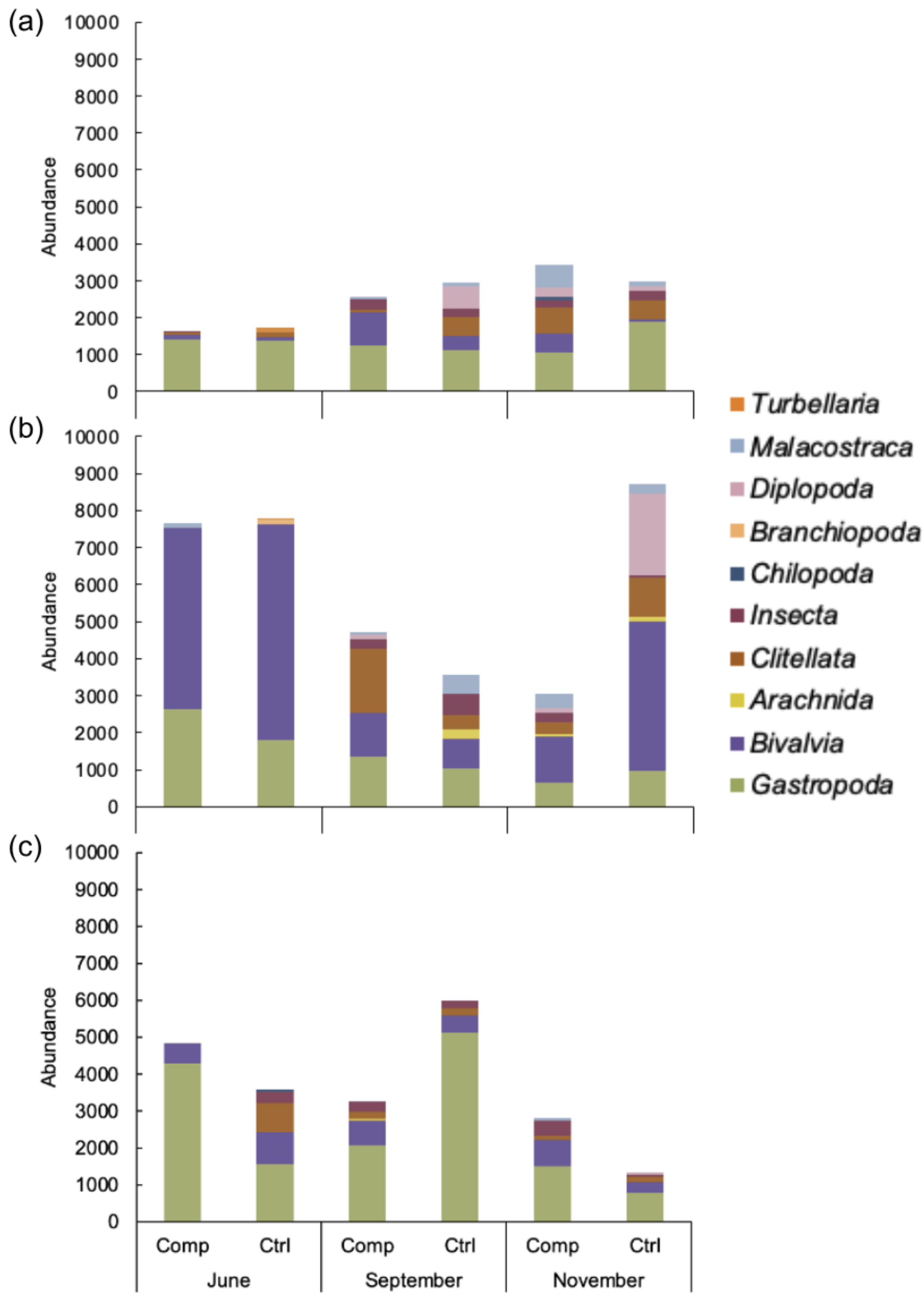

Figure 8. Average abundance of macroinvertebrates per m^2 to class level in the compost and control treatment at (a) Packard, (b) Cady, and (c) Science Hill. Values are mean, n=8.

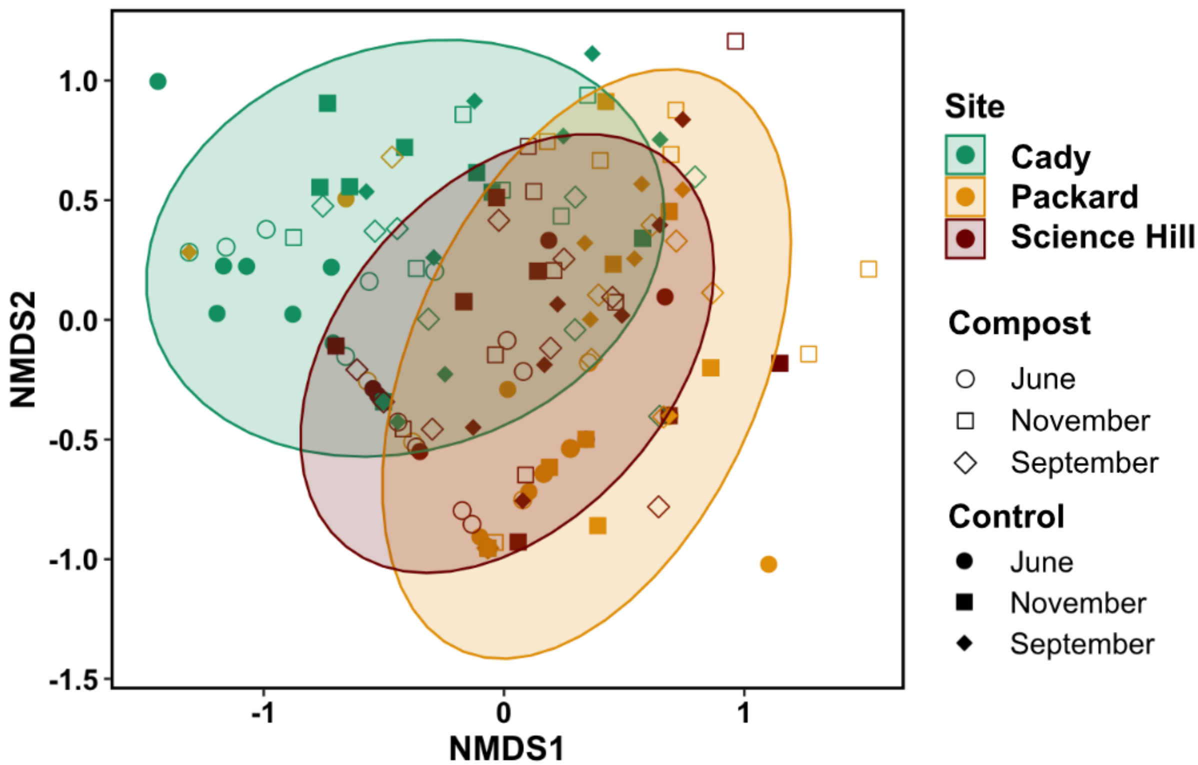

Figure 9. Non-metric Multidimensional Scaling (NMDS) of macroinvertebrate community based on Bray-Curtis dissimilarities at Cady, Packard and Science Hill. Shaded ellipses indicate 90% confidence intervals based on each site and a stress value of 0.198.

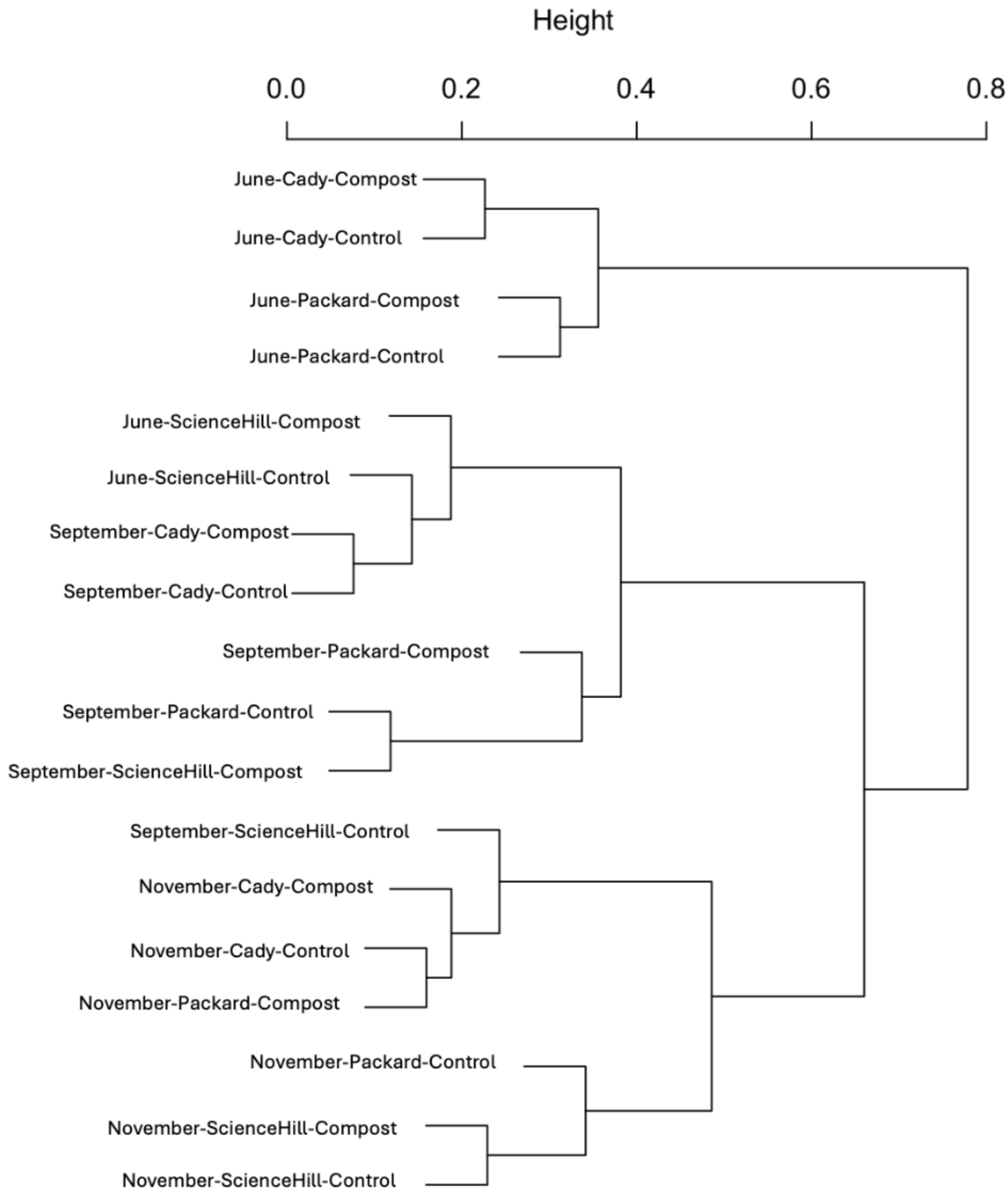

Figure 10. Cluster analysis based on Bray-Curtis dissimilarity of macroinvertebrate community composition at each time point for each site and treatment.

Table 5. Macroinvertebrate abundance, Shannon diversity (H) and species richness measured in Packard, Cady and Science Hill in compost and control treatments averaged across all time points. Values are mean ± SE.

	Packard		Cady		Science Hill	
	Compost	**Control**	**Compost**	**Control**	**Compost**	**Control**
H’	0.74±0.11	0.71±0.11	0.80±0.07	0.81±0.07	0.67±0.04	0.71±0.04
Richness	2.9±0.3	2.9	3.1±0.4	3.6±0.4	2.6±0.2	2.6±0.2
Abundance	20.8±3.0	18.3±3.0	31.4±9.7	20.2±9.7	26.9±3.7	22.8±3.7

Table 6. Results of a two-way ANOVA examining the effects of season (Spring, Summer, Fall) and treatment on macroinvertebrate abundance (A), Shannon diversity (H’), and species richness (S). Significant p-values are bolded and p-values approaching significance are italicized.

		Packard			Cady			Science Hill		
		Season	**Comp**	**Season x Comp**	**Season**	**Comp**	**Season x Comp**	**Season**	**Comp**	**Season x Comp**
H’	*p*	**0.010**	0.92	0.54	**0.0001**	0.80	0.55	0.24	0.53	*0.06*
	$F_{6,42}$	5.2	0.0	0.6	11.3	0.1	0.6	1.5	0.4	3.0
S	*p*	**0.029**	0.78	0.60	**0.006**	0.10	0.87	0.12	1.00	0.49
	$F_{6,42}$	3.9	0.1	0.5	5.8	2.8	0.1	2.2	0.0	0.7
A	*p*	0.93	0.56	0.94	0.24	0.25	0.15	*0.088*	0.51	0.62
	$F_{6,42}$	0.07	0.35	0.94	1.5	1.4	2.0	2.6	0.4	0.5

Principal Component Analysis

The Principal Component Analysis (PCA) yielded two main components that in combination explained 44% of the variability in the data (27% Component 1; 17% Component 2; Figure 11). Key variables (> 0.5) on the first component included a variety of soil-associated characteristics, including %SM, BD, OM, nitrate, OP, R, % C and % N (Table 7). The second component was defined by a combination of soil and primary producer variables, including ammonium, chl *a*, FQI and molar C:N (Table 7). There were distinct groupings by site, with each of the sites separating from each other. Packard separated from Science Hill and Cady on the first component and Science Hill and Cady separated on the second component. Within Science Hill and Cady, the compost and control each clustered together in their respective sites. At Cady, they separated on the first component and at Science Hill they separated on the second component.

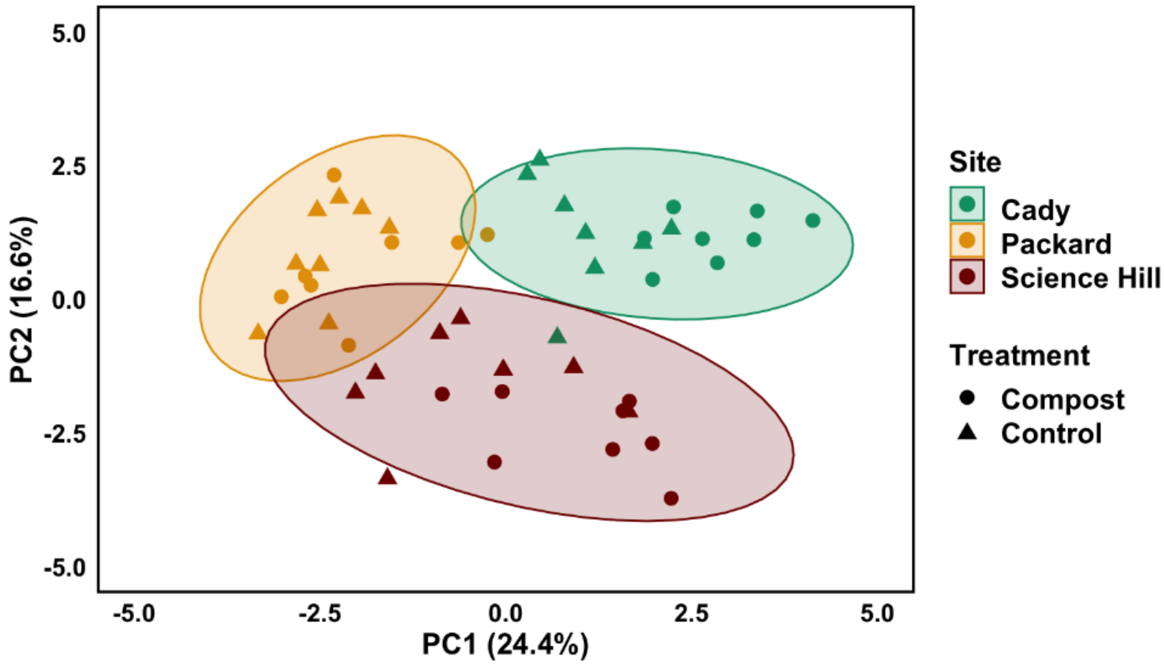

Figure 11. Principal components analysis biplot of the first two Components. Components 1 and 2 explain 24.4% and 16.6% of variability in the data, respectively. Ellipses indicate 95% confidence intervals based on site.

Table 7. Principal Components analysis factor loading from the two strongest principal components. Values with an asterisk load at 0.5 or higher.

	Component 1	**Component 2**
Moisture	0.84*	0.23
Bulk Density	-0.67*	0.46*
Organic Matter	0.71*	-0.22
Ammonium	-0.21	0.50*
Nitrate	-0.55*	0.05
Inorganic Phosphate	-0.42	-0.22
Organic Phosphate	0.72*	0.25
Chl *a*	-0.22	0.56*
Respiration	0.68*	0.48
Vegetation Diversity	-0.30	-0.35
% Plant Cover	0.21	0.32
FQI	-0.16	-0.84*
Macro Diversity	0.02	-0.11
Macro Abundance	0.20	0.14
C:N	0.12	-0.68*
%N	0.60*	-0.16
%C	0.60*	-0.39

Discussion

The use of compost addition as a management strategy can assist in increasing wetland function. But, as highlighted by these results, it is important to take into consideration the characteristics of the individual wetland when employing new management strategies. In this study, which employed three wetlands with varying prior land use, hydrogeomorphic classification, and resulting vegetation communities, we observed some similarities, but also substantial differences in the impact of compost addition on ecosystem structure and function. Yearly fluctuations in precipitation and temperature may also alter the effects of compost addition in ways not captured here because of the limited duration of the measurement period (Williams, 2021).

Overall, variations in the abiotic and biotic ecosystem structure among the three sites were more pronounced than those between treatments (Figures 7 and 9). The first component of the PCA (Figure 11), which distinctly separated Cady from Packard, primarily reflected abiotic drivers with soil moisture, bulk density, organic matter, nitrate, organic phosphate, %C, %N and soil respiration causing the variation among the sites (Table 7). Within sites, there was somewhat of a clustering of treatments that reflects the impact of compost on soils. High organic matter co-occurs with high moisture and nutrients and low bulk density and, as a result, respiration is increased. The second component, which was driven by a combination of abiotic and biotic factors, including ammonium, chlorophyll *a*, FQI and molar C:N, showed similarities between Cady and Packard, but a distinct separation with Science Hill (Table 7). These patterns illustrate the inherent complexity both within and among wetlands that requires an understanding of site-specific drivers of structure and how management may influence emergent functions.

Saturated, high organic soils are characteristic of wetlands. A natural freshwater wetland has a soil organic content of 12-20% and often fully saturated soils (Campbell et al., 2002; Faulkner & Richardson, 2020; Schlesinger & Bernhardt, 2013). In contrast, created wetlands typically have altered soil characteristics with lower organic matter (Campbell et al., 2002), increased bulk density (Hunter et al., 2008) and decreased soil moisture than natural wetlands (Moser et al., 2009). Compost addition successfully lowered the bulk density at all sites and increased organic content and soil moisture at Packard wetland and Cady wetland (Figure 2). While the soil %OM and moisture were not increased significantly at Science Hill, the control plots had organic matter levels comparable to that of a natural freshwater wetland (mean = 26%) and saturated soils (mean = 50%). Once these thresholds have been met, the processes may reach equilibrium beyond which no further increase is feasible.

The soil compost addition did not have a large effect on the available dissolved nitrogen across sites, with similar pools of both nitrate and ammonium in both treatments. This contrasts with prior work (Williams 2021) that showed higher N in compost treated plots at Packard. This prior study was during a drought year, suggesting that differences may be more pronounced both earlier in the restoration process and during drought conditions when nutrient recycling may be slower. Phosphorus was only impacted at the youngest site, Science Hill, where higher concentrations of both TP and IP were found in compost treated plots, but at a similar ratio of IP: OP. This may be a function of age, with less time to build up P stocks in the soil as this effect was not seen at older sites.

Each of the three wetlands had a unique plant community with little overall difference between treatments at any sites (Figure 7), with *Typha* dominant at Packard and Science Hill for most of the summer (Figure 6). In September, *Sium suave* was also common at Science Hill (Figure 6c). Cady was dominated by *Phalaris arundinacea* across both seasons and treatments (Figure 6b), with a decrease in total cover in compost treated plots. The decrease in cover observed here and also at the other sites may be attributed in part to the disturbance of the compost addition itself, which may bury or shade seedlings and limit total cover. All sites had cover at or above 100%, so it is unlikely to have contributed to a significant long-term impact. The higher diversity and species richness in the control at Cady suggest that perhaps the compost, with its additional nutrient value, promoted the presence of dominant, opportunistic, monoculture forming species. A prior study done in Cady in 2015-2016 saw a similar trend with the compost decreasing some native plant growth (Williams, 2021), but no changes to native plant cover was found in the current study at this site (Table A5). Using the scale that FQI between 1 and 19 is low and 20 or greater is high (Ortiz-

Burgos 2016, US Fish & Wildlife Services 2019), the FQI at Cady is relatively low, likely as a result of the high cover of reed canary grass. The lack of a difference in FQI suggests a similar lack of overall plant desirability across treatments (Bourdaghs et al., 2006) and highlights the need for additional management to control invasive plants after the permitted monitoring period has ended.

The Packard wetland, a wooded wetland somewhat younger than Cady, had different patterns that may reflect both age and the intended wetland type. At this site, there was also a decrease in overall cover with compost addition, but this coincided with a higher cover of native species (Figure 6; Table A2, Table A5). This suggests that in contrast to the Cady wetland, where compost enhanced non-native species, at this site the addition of compost may decrease overall cover somewhat, but in the process enhances native species. The impacts at Science Hill on overall plants community structure were minimal, but two individual species were impacted: *Alisma plantago-aquatica* was higher in the compost and *Penthorum sedoides* was higher in the control (Table A7). *Penthorum sedoides* grows well in areas with low nutrients while *Alisma plantago-aquatica* is known to require high nutrient soil which would be more abundant in the compost (Milligan et al., 2008; Moravcová wt al., 2001). At this site, the fifth year of compost addition coincided with the current study and occurred between the June and September plant assessments. There was no apparent interaction between season and treatment, suggesting that the disturbance of the addition was minimal at this site, and that these impacts may appear longer after treatments cease, as seen in the other two sites that were treated years earlier.

The macroinvertebrate communities were most strongly influenced by season, with some variation in functional group dominance among sites (Figure 8), but little impact of treatment (Figure 9). The seasonal clustering in the Bray-Curtis dissimilarity analysis, and seasonal differences in the H' and S suggest seasonal shifts in life cycles as well as hydrology (De Szalay & Resh, 2000) (Figure 10). In general, however, at Packard and Science Hill the dominant macroinvertebrates were mobile grazers in the class Gastropoda (Figure 8). In contrast, the dominance of filter feeding Bivalvia at Cady (Figure 8) suggests a distinct difference in the abiotic community with greater water and hence algal availability. These factors are supported by the higher soil moisture, seasonal standing water, and algal abundance at this site than the others (Hillebrand & Kahlert, 2001) (Figure 2 and Figure 5).

The macroinvertebrates observed in this study are able to move freely in and out of the transects and some have an individual range >1 m, suggesting that they could easily travel between transects and our static measurements do not indicate selective residency within a treatment (Ahrens & Kraus, 2006; Jactel & Gaillard, 1991; Nuutinen & Butt, 2005). Further, although there are no studies, to our knowledge, evaluating the impact of organic matter addition on invertebrate communities, given prior research showing little difference in macroinvertebrate species richness and diversity between created and reference wetlands (Balcombe et al., 2005), our lack of a significant influence on mobile grazers is not surprising. Management strategies that successfully bring overall function of a restored wetland closer to that of a natural wetland might not have any impact on the macroinvertebrate community.

However, the negative influence of compost on algal abundance at all sites suggests a more substantial and surprising impact on benthic ecosystem function (Figure 5). Additionally, while moss was avoided as much as possible in the sampling, it was present in small quantities at all sites and may have contributed additional chlorophyll beyond BMA alone (Marschall & Proctor, 2004). The algal abundance also showed a distinct seasonal trend, with lowest values in spring and summer and highest values in fall. Shading by emergent macrophytes or grazing by gastropods and bivalves earlier in the growing season may limit algal growth. Interestingly, though, the GPP measured in November, when there were differences between treatments at all sites, showed an inverse pattern between algal abundance and GPP, with the control having lower GPP than the compost treatment (Figure 4b). The latter was anticipated if higher nutrient availability in the compost plots acted to fuel algal production. This suggests, however, that the photosynthetic efficiency may be lower in the compost plots (although this was not significant; data not shown) or that the compost creates greater surface area for algal growth, but self-shading or other factors limit overall GPP.

As expected, all sites were net heterotrophic (Figure 4), but both surface (areal) and homogenized soil respiration rates were significantly influenced by compost addition. Compost, likely by increasing both organic matter and to a lesser extent nutrient availability, had a strong positive impact on R. The positive impact on R at Science Hill was not surprising, given the recent addition of organic matter. However, this impact was retained at depth only at Cady, and at the surface only at Packard, even though a number of years have passed since the last addition. The difference in treatment effect between the areal whole-core measurement (R increased with compost at Packard and Science Hill) and the homogenized soil measurement (R increased with compost at Cady and Science Hill) is interesting, and suggests potential for greater retention of organic matter within the soil at wooded sites. At these two wooded and somewhat drier sites, the GPP and NEM are also higher than at Cady, especially in compost treated plots (Figure 4). This suggests more active metabolism and biogeochemical cycling, perhaps promoted by the slightly drier conditions in the soil and higher nutrient availability. These data were taken only in November, and a greater temporal range is needed to make additional conclusions about patterns in soil PP and metabolism

Compost addition as a management tool can effectively alter the structure of a created wetland, but the effects vary based on individual wetland characteristics. The wetlands tested in this study varied in age, time since the last compost addition, wetland type, hydrology, and vegetation communities. These difference likely influence responsiveness to the treatment. Interestingly, the one emergent wetland and the wetland with the longest duration of time since the last addition, retained many differences in soil and plant characteristics. This suggests that perhaps emergent wetlands are most responsive to organic matter addition, and that the more persistent flooding and shifted soil biogeochemistry may lead to lower decomposition and loss of additional organic material.

Conclusion

This study shows compost can be helpful in improving the wetland structure and function and driving wetland function closer to that of a mature or natural wetland. However, we also demonstrated that the impacts are not universal, and that the wetland type is a significant factor in the overall response. The results of the compost addition were dependent on the individual wetland. Compost had the greatest effect on vegetation at Cady, the emergent wetland, and most impacted soil at Science Hill, the youngest wetland with the most recent compost addition. Compost successfully decreased bulk density and increased soil organic matter, phosphate and increased benthic photosynthesis. Although in some cases the treatment effect was lower, many of these same shifts in soil characteristics were retained at sites where compost had not been added for multiple years. However, there were also some unintended consequences such as decreasing species richness, diversity and native plant cover at some sites. As a result, compost can help to alter some of the functions and community structure but should be used in conjunction with other management strategies.

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Appendix

Table A1. Averages for each month for all variables measured in June, September and November at Packard, Cady and Science Hill in the compost, control and the average of the entire site (total).

		Packard			Cady			Science Hill		
		June	Sept	Nov	June	Sept	Nov	June	Sept	Nov
R	Comp	0.004	0.008	0.002	0.022	0.006	0.001	0.004	0.004	0.014
	Ctrl	0.004	0.001	0.002	0.017	0.004	0.001	0.002	0.003	0.004
	Total	0.004	0.005	0.002	0.019	0.005	0.001	0.003	0.003	0.009
NEM	Comp	-	-	562	-	-	196	-	-	745
	Ctrl	-	-	254	-	-	193	-	-	303
	Total	-	-	408	-	-	195	-	-	524
GPP	Comp	-	-	8.19	-	-	1.09	-	-	11.97
	Ctrl	-	-	3.69	-	-	0.88	-	-	2.21
	Total	-	-	5.94	-	-	0.980	-	-	7.09
NO_3^-	Comp	0.88	0.83	1.59	0.001	0.36	0.63	0.04	0.64	0.86
	Ctrl	0.36	0.60	1.40	0.009	0.35	0.40	0.03	0.62	0.81
	Total	0.62	0.71	1.50	0.00	0.35	0.52	0.04	0.63	0.83
NH_4^+	Comp	28	19	17	20	25	0.02	4	24	0.013
	Ctrl	23	14	7	18	25	0.01	2	29	0.012
	Total	26	17	12	19	25	0.02	3	27	0.013
TP	Comp	1669	1561	1579	1551	1510	1690	1492	1514	1548
	Ctrl	1557	1643	1497	1478	1577	1417	1141	1287	1225
	Total	1613	1602	1538	1515	1544	1553	1316	1401	1387
IP	Comp	1171	1013	1094	930	863	890	883	928	863
	Ctrl	1100	1048	1051	877	881	852	699	759	720
	Total	1135	1031	1073	903	872	871	791	844	792
OP	Comp	497	548	485	621	648	800	609	586	686
	Ctrl	458	595	446	601	696	565	442	528	505
	Total	478	571	465	611	672	683	525	557	595
OM	Comp	16	27	21	18	39	38	30	32	26
	Ctrl	18	20	17	19	35	19	25	31	24
	Total	17	23	19	19	37	28	28	32	25
Macro. S	Comp	22	20	21	48	27	19	36	25	20
	Ctrl	17	17	20	40	32	48	29	29	10
	Total	20	19	21	44	30	34	33	27	15
Macro. H’	Comp	0.50	0.74	0.97	0.52	1.00	0.87	0.54	0.85	0.63
	Ctrl	0.47	0.82	0.83	0.63	0.90	0.89	0.62	0.68	0.82
	Total	0.49	0.78	0.90	0.58	0.95	0.88	0.58	0.77	0.42
Macro. S	Comp	1.9	3.1	3.8	2.0	3.9	3.5	2.0	3.3	2.6
	Ctrl	2.0	3.4	3.3	2.6	4.0	4.3	2.4	2.8	2.8
	Total	2.0	3.3	3.6	2.3	4.0	3.9	2.2	3.05	2.7

Table A2. Averages for each month for all variables measured in May, June, August, September, October, and November at Packard in the Compost, Control and the average of the entire site (total).

		May	June	August	Sept	Oct	Nov
%SM	Comp	47	42	41	43	36	37
	Ctrl	44	40	38	42	29	36
	Total	46	41	40	42	33	37
Chl *a*	Comp	47	195	75	60	50	52
	Ctrl	40	176	100	59	83	62
	Total	43	185	88	59	67	57
BD	Comp	0.62	-	0.52	-	0.59	-
	Ctrl	0.66	-	0.63	-	0.60	-
	Total	0.64	-	0.58	-	0.60	-
TC	Comp	-	94	130	-	-	-
	Ctrl	-	85	94	-	-	-
	Total	-	90	112	-	-	-
Plant S	Comp	-	4.1	4.9	-	-	-
	Ctrl	-	3.5	4.6	-	-	-
	Total	-	3.8	4.8	-	-	-
Plant H'	Comp	-	0.9	1.3	-	-	-
	Ctrl	-	0.9	1.2	-	-	-
	Total	-	0.9	1.3	-	-	-
FQI	Comp	-	10.8	9.9	-	-	-
	Ctrl	-	8.4	10.1	-	-	-
	Total	-	9.6	10.0	-	-	-
Invasive TC	Comp	-	2.5	5.6	-	-	-
	Ctrl	-	1.9	5.6	-	-	-
	Total	-	2.2	5.6	-	-	-
Native TC	Comp	-	91.3	125.1	-	-	-
	Ctrl	-	83.8	88.6	-	-	-
	Total	-	87.6	106.9	-	-	-

Table A3. Averages for each month for all variables measured in May, June, August, September, October, and November at Cady in the Compost, Control and the average of the entire site (Total).

		May	June	August	Sept	Oct	Nov
%SM	Comp	60	68	64	56	51	57
	Ctrl	56	65	51	58	39	44
	Total	58	67	57	57	49	51
Chl *a*	Comp	27	60	91	91	44	65
	Ctrl	56	54	130	117	133	90
	Total	42	57	110	104	89	78
BD	Comp	0.48	-	0.37	-	0.45	-
	Ctrl	0.50	-	0.54	-	0.47	-
	Total	0.49	-	0.45	-	0.46	-
TC	Comp	-	111	104	-	-	-
	Ctrl	-	123	140	-	-	-
	Total	-	117	122	-	-	-
Plant Richness	Comp	-	3.4	3.3	-	-	-
	Ctrl	-	4.1	5.5	-	-	-
	Total	-	3.8	4.4	-	-	-
Plant H'	Comp	-	0.7	0.8	-	-	-
	Ctrl	-	0.9	1.5	-	-	-
	Total	-	0.8	1.2	-	-	-
FQI	Comp	-	1.9	8.0	-	-	-
	Ctrl	-	7.6	10.3	-	-	-
	Total	-	4.8	9.2	-	-	-
Invasive TC	Comp	-	90.6	58.4	-	-	-
	Ctrl	-	100.0	69.4	-	-	-
	Total	-	95.3	63.9	-	-	-
Native TC	Comp	-	20.6	50.4	-	-	-
	Ctrl	-	22.6	48.1	-	-	-
	Total	-	21.6	49.3	-	-	-

Table A4. Averages for each month for all variables measured in May, June, August, September, October, and November at Science Hill in the Compost, Control and the average of the entire site (total).

		May	June	August	Sept	Oct	Nov
%SM	Comp	53	55	47	43	35	37
	Ctrl	50	53	43	43	42	41
	Total	52	54	45	43	39	39
Chl *a*	Comp	38	96	41	23	61	46
	Ctrl	61	77	77	35	95	70
	Total	49	87	59	29	78	58
BD	Comp	0.49	-	0.40	-	0.30	-
	Ctrl	0.55	-	0.53	-	0.34	-
	Total	0.52	-	0.46	-	0.32	-
TC	Comp	-	102	107	-	-	-
	Ctrl	-	83	97	-	-	-
	Total	-	93	102	-	-	-
Plant S	Comp	-	4.9	4.4	-	-	-
	Ctrl	-	3.6	4.5	-	-	-
	Total	-	4.3	4.5	-	-	-
Plant H’	Comp	-	1.2	1.2	-	-	-
	Ctrl	-	0.9	1.2	-	-	-
	Total	-	1.1	1.2	-	-	-
FQI	Comp	-	23.2	15.7	-	-	-
	Ctrl	-	16.3	15.9	-	-	-
	Total	-	19.8	31.6	-	-	-
Invasive TC	Comp	-	0.25	8.1	-	-	-
	Ctrl	-	2.2	1.9	-	-	-
	Total	-	1.2	5.0	-	-	-
Native TC	Comp	-	102.3	98.9	-	-	-
	Ctrl	-	81.1	95.3	-	-	-
	Total	-	91.7	97.1	-	-	-

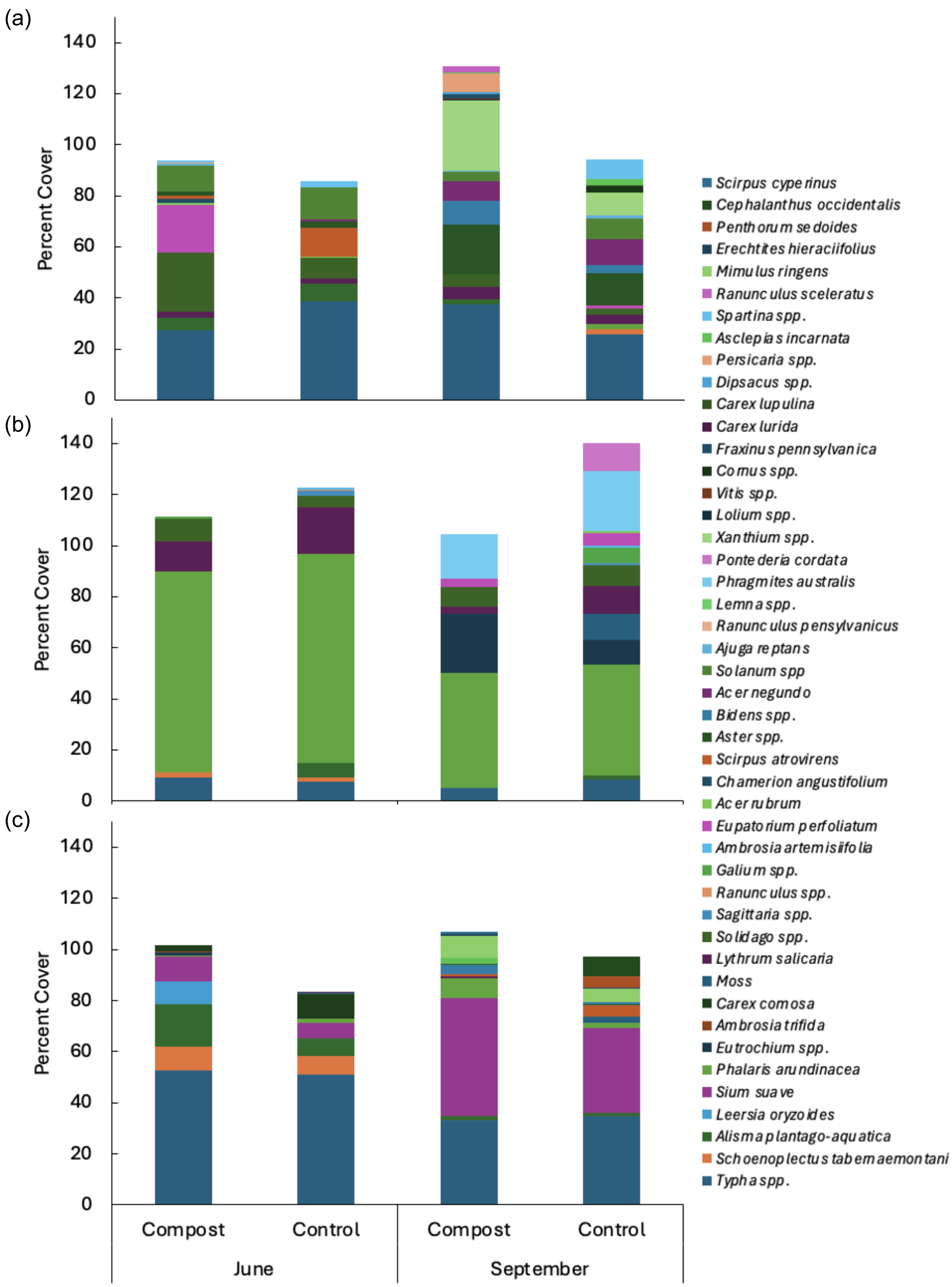

Figure A1. Percent vegetation cover in the compost and control treatment for all plant species at (a) Packard, (b) Cady, and (c) Science Hill. Percent cover can exceed 100% due to overlapping layers of vegetation. Values are mean, n=8.

Table A5. Results of two-way ANOVA examining the effects of season (Spring, Summer) and treatment on percent cover of invasive (INV) and native (NAT) vegetation cover. Significant p-values are bolded and p-values approaching significance are italicized.

		Packard			Cady			Science Hill		
		Season	**Comp**	**Season x**	**Season**	**Comp**	**Season x**	**Season**	**Comp**	**Season x**
INV	*p*	0.12	0.89	0.89	0.015	0.41	0.95	0.24	0.26	0.34
	$F_{4,28}$	2.5	0.0	0.0	6.8	0.7	0.0	1.4	1.3	1.0
NAT	*p*	**0.037**	**0.020**	0.11	**0.0021**	*0.068*	0.96	0.50	0.13	0.28
	$F_{4,28}$	4.8	6.2	2.7	7.1	3.5	0.05	0.5	2.5	1.2

Table A6. Results of a two-way ANOVA examining the effects of season (Spring, Summer, Fall) and treatment on macroinvertebrate abundance Shannon diversity (H'), and species richness (S) when classified to phylum and lowest distinguishable identification (LDI). Significant p-values are bolded and p-values approaching significance are italicized.

		Packard			Cady			Science Hill		
		Season	**Comp**	**Season**	**Season**	**Comp**	**Season**	**Season**	**Comp**	**Season**
Phylum H'	*p*	**0.0405**	0.71	0.72	**<0.001**	0.89	0.0173	0.47	0.31	0.79
	$F_{6,42}$	3.5	0.1	0.3	16.8	0.9	4.5	0.8	1.0	0.2
Phylum S	*p*	0.30	0.58	0.73	**0.018**	0.19	0.28	0.06	0.59	0.91
	$F_{6,42}$	1.3	0.6	0.7	4.4	1.8	1.3	3.0	0.3	0.1
LDI H'	*p*	0.13	0.70	0.89	**0.0032**	0.79	0.90	0.23	0.97	0.61
	$F_{6,42}$	2.2	0.2	0.1	6.6	0.1	0.1	1.5	0.0	0.5
LDI S	*p*	0.28	1.00	0.45	0.32	*0.56*	0.68	**0.035**	0.40	0.70
	$F_{6,42}$	1.3	0.0	0.81	1.2	0.4	0.4	3.7	0.7	0.4

Table 7. Pairwise comparison of each vegetation species in the compost and control at each site and the results of a one-way ANOVA examining the effects of treatment. Significant p-values are bolded and p-values approaching significance are italicized. $F_{4,28}$

Site	Month		*Typha spp.*	*Schoenoplectus tabernaemontani*	*Alisma plantago-aquatica*	*Leersia oryzoides*	*Sium suave*	*Phalaris arundinacea*	*Eutrochium spp.*	*Ambrosia trifida*	*Carex comosa*	Moss
Packard	June	Compost	27.5	0.0	4.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	38.8	0.0	6.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	August	Compost	37.5	0.0	1.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	25.9	1.9	0.0	0.0	0.0	1.9	0.0	0.0	0.0	0.0
		Treatment *p*	0.98	0.33	0.94	-	-	0.33	-	-	-	-
		F	0.0	1.0	0.0	-	-	1.0	-	-	-	-
Cady	June	Compost	9.4	1.9	0.0	0.0	0.0	78.8	0.0	0.0	0.0	0.0
		Control	7.5	1.9	5.6	0.0	0.0	81.9	0.0	0.0	0.0	0.0
	August	Compost	5.0	0.0	0.0	0.0	0.0	45.0	23.3	0.0	0.0	0.0
		Control	8.3	0.0	1.7	0.0	0.0	43.3	10.0	0.0	0.0	10.0
		Treatment *p*	0.95	0.98	0.12	-	-	0.82	0.38	-	-	0.31
		F	0.5	0.0	2.4	-	-	0.1	0.9	-	-	1.1
Science Hill	June	Compost	52.5	9.4	16.9	8.9	9.6	0.3	1.3	0.6	2.5	0.0
		Control	51.3	7.1	7.0	0.0	5.9	1.9	0.0	0.0	9.5	0.4
	August	Compost	33.1	0.0	1.9	0.0	46.3	7.5	0.0	0.0	0.0	0.0
		Control	35.0	0.0	1.3	0.0	33.1	1.9	0.0	0.0	0.0	2.5
		Treatment *p*	0.96	0.63	**0.027**	*0.08*	*0.08*	0.38	0.33	0.33	0.43	0.27
		F	0.0	0.2	6.0	2.9	2.2	0.7	1.0	1.0	0.6	1.3

Site	Month		*Lythrum salicaria*	*Solidago spp.*	*Sagittaria spp.*	*Ranunculus spp.*	*Galium spp.*	*Ambrosia artemisiifolia*	*Eupatorium perfoliatum*	*Acer rubrum*	*Chamerion angustifolium*	*Scirpus atrovirens*
Packard	June	Compost	2.5	23.1	0.0	0.0	0.0	0.0	18.8	0.6	1.9	1.3
		Control	1.9	8.1	0.0	0.0	0.0	0.0	0.0	0.6	0.0	11.3
	August	Compost	5.0	5.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	3.8	2.5	0.0	0.0	0.0	0.0	1.3	0.0	0.0	0.0
		Treatment *p*	0.66	0.21	-	-	-	-	0.12	1	0.33	0.13
		F	0.2	1.5	-	-	-	-	2.7	0.0	1.0	2.4
Cady	June	Compost	11.9	8.8	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0
		Control	18.1	4.4	2.3	0.4	0.0	0.6	0.0	0.0	0.0	0.0
	August	Compost	2.8	7.5	0.0	0.0	0.0	0.0	3.3	0.0	0.0	0.0
		Control	10.8	8.3	0.8	0.0	5.8	0.8	5.0	0.8	0.0	0.0
		Treatment *p*	0.12	0.17	0.22	0.38	0.31	0.21	0.49	0.31	-	-
		F	2.4	1.8	1.8	0.8	1.0	1.9	0.5	1.1	-	-
Science Hill	June	Compost	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	August	Compost	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.9
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.4
		Treatment *p*	0.59	-	-	-	-	-	-	-	-	0.18
		F	0.3	-	-	-	-	-	-	-	-	2.0

Site	Month		*Aster* spp.	*Bidens* spp.	*Acer negundo*	*Solanum* spp	*Ajuga reptans*	*Ranunculus pensylvanicus*	*Lemna* spp.	*Phragmites australis*	*Pontederia cordata*	*Xanthium* spp.
Packard	June	Compost	1.6	0.0	0.0	10.0	0.6	0.6	0.0	0.0	0.0	0.0
		Control	2.5	0.0	0.6	12.5	0.0	0.0	0.0	0.0	0.0	0.0
	August	Compost	19.4	9.4	7.5	3.8	0.6	0.0	0.0	0.0	0.0	27.5
		Control	12.5	3.1	10.3	8.1	1.3	0.0	0.0	0.0	0.0	8.8
		Treatment *p*	0.51	0.28	0.71	0.58	1	0.33	-	-	-	0.18
		F	0.4	1.2	0.1	0.3	0.0	1.0	-	-	-	1.9
Cady	June	Compost	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	August	Compost	0.0	0.0	0.0	0.0	0.0	0.0	0.0	17.5	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	23.3	10.8	0.0
		Treatment *p*	-	-	-	-	-	-	-	0.78	0.34	-
		F	-	-	-	-	-	-	-	0.1	1.0	-
Science Hill	June	Compost	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	August	Compost	0.0	3.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.6	0.6	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0
		Treatment *p*	0.33	0.26	-	-	0.33	-	-	-	-	-
		F	1.0	1.3	-	-	1.0	-	-	-	-	-

Site	Month		*Lolium* spp.	*Vitis* spp.	*Cornus* spp.	*Fraxinus pennsylvanica*	*Carex lurida*	*Carex lupulina*	*Dipsacus* spp.	*Persicaria* spp.	*Asclepias incarnata*	*Spartina* spp.
Packard	June	Compost	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.5
	August	Compost	0.3	0.6	0.0	1.3	0.3	0.0	0.6	7.5	0.3	0.0
		Control	0.0	0.0	2.5	0.0	0.0	0.6	0.0	0.0	2.5	7.5
		Treatment *p*	0.33	0.33	0.33	0.33	1	0.33	0.33	0.33	0.24	0.24
		F	1.0	1.0	1.0	1.0	0.0	1.0	1.0	1.0	1.4	1.5
Cady	June	Compost	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	August	Compost	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Treatment *p*	-	-	-	-	-	-	-	-	-	-
		F	-	-	-	-	-	-	-	-	-	-
Science Hill	June	Compost	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	August	Compost	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	2.5	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Treatment *p*	-	-	-	-	-	0.33	-	-	0.33	-
		F	-	-	-	-	-	1.0	-	-	1.0	-

Site	Month		*Ranunculus sceleratus*	*Mimulus ringens*	*Erechtites hieraciifolius*	*Penthorum sedoides*	*Cephalanthus occidentalis*	*Scirpus cyperinus*
Packard	June	Compost	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0
	August	Compost	2.5	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0
		Treatment *p*	0.33	-	-	-	-	-
		F	1.0	-	-	-	-	-
Cady	June	Compost	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0
	August	Compost	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0
		Treatment *p*	-	-	-	-	-	-
		F	-	-	-	-	-	-
Science Hill	June	Compost	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0
	August	Compost	0.0	8.8	0.9	0.0	0.0	0.6
		Control	0.0	5.0	0.6	4.4	7.5	0.0
		Treatment *p*	-	0.36	0.78	**0.047**	0.33	0.33
		F	-	0.9	0.1	3.9	1.0	1.0