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THE EFFECTS OF RESTORATION TREAMTENTS AND FLOODING REGIME ON PLANT COMMUNITY COMPOSITION IN RESTORED GEOGRAPHICALLY ISOLATED WETLANDS

by

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A thesis

Presented to the faculty of

Towson University

in partial fulfillment

of the requirements for the degree

Master of Science

Department of Environmental Science

Towson University

Towson, Maryland, 21252

(May, 2015)

TOWSON UNIVERSITY
OFFICE OF GRAUDATE STUDIES

THESIS APPROVAL PAGE

This is to certify that the thesis prepared by Kimberley Russell
entitled The effects of restoration treatments and flooding regime on plant community distributions in restored geographically isolated wetlands

has been approved by the thesis committee as satisfactorily completing the thesis requirements for the degree Master of Science

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Acknowledgements

I would like to express my utmost gratitude to my thesis advisor, Dr. Vanessa Beauchamp, whose expertise, patience, and encouragement added immensely to my overall research experience. I would also like to thank the other members of my committee, Dr. Joel Snodgrass and Dr. Roland Roberts, for the assistance they provided throughout the project.

I would also like to thank the Nature Conservancy for allowing me access to the created and restored wetlands within Jackson Lane to conduct my research. In addition, I would like to thank Towson University, the Office of Graduate Studies, the Department of Biological Sciences, and the Department of Environmental Science for their financial support in my research and continuing education.

Finally, I would like to thank my family and friends for all of their loving support throughout the duration of my research. Without their encouragement, I would have never been able to complete my degree and strive for scientific excellence.

Abstract

THE EFFECTS OF RESTORATION TREAMTENTS AND FLOODING REGIME ON PLANT COMMUNITY COMPOSITION IN RESTORED GEOGRAPHICALLY ISOLATED WETLANDS

Kimberley Russell

Wetland plant community diversity is an important structural quality to assess in wetland creation or restoration projects because it is typically used as a proxy for other functional processes that are more difficult to measure. To determine the drivers of plant community diversity, eight wetlands within Jackson Lane, a large scale, fragmented wetland mitigation project, were sampled for species richness and fourteen additional environmental variables almost a decade after mitigation. Results show that size, straw type, and soil chemistry and texture are influential variables on plant species diversity. In addition, differences in average percent vegetative cover, average percent litter cover, coarse woody debris (CWD), and soil chemistry and texture are related to differences in wetland plant community composition. Coarse woody debris application and applications of straw are restoration practices that should be used in any depression wetland mitigation in order to increase plant diversity.

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Introduction

The construction and restoration of wetlands is commonly used to compensate for wetland loss due to any regulated human activity. The mitigation of biologically diverse wetlands is a difficult task that can be accomplished by establishing disturbance regimes that support diverse plant communities and provide connected habitats that facilitate dispersal and re-colonization without the integration of invasive species (Zedler 2003; Fahrig 2003; Thiere et al. 2009). The success of a wetland mitigation project is typically assessed based on its ability to replicate at least some aspects of both the structure and function of natural wetland systems. The structure of wetlands usually refers to the biotic composition of the area, typically plant and wildlife community composition or biomass. Wetland functions are the processes that occur within a wetland (Novitzki et al. 1996), including flooding duration and frequency, nutrient transformation, pollution control, and food chain dynamics (Kusler and Kentula 1990).

A diverse plant community is an important structural quality to have within a wetland because it can also be used as a proxy for the community's functional processes such as productivity, that are more difficult to measure. High plant species diversity has been positively correlated with increases in aboveground and belowground biomass (Zedler et al. 2001), soil microbial productivity (Zak et al. 2003), nutrient retention (Engelhardt and Ritchie 2001), and resiliency (Tilman and

Downing 1994). Plant diversity within wetlands also provides multiple ecosystem services including wildlife refuge, food production, nitrogen and phosphorus storage, and stormwater attenuation (Zedler 2003).

Species richness, the number of species within a given community, is a commonly used metric to quantify biodiversity within a particular wetland and can be indicative of successful restorative practices. Additional metrics such as Shannon and Weaver (1949) and Simpson (1949) indices use species richness and species evenness, the relative abundance of different species within an area, to evaluate the diversity of plant communities (Nagendra 2002). The Shannon-Weaver index places importance on rare species - as well as species richness, whereas Simpson stresses dominant species and species evenness (Nagendra 2002). Both species richness and species diversity indices can be used as a proxy for measuring wetland functions, such as productivity or nutrient retention (Naeem et al. 1996; Zedler et al. 2001; Engelhart and Ritchie 2001), that are more difficult to measure.

Richness and diversity metrics are important measures of wetland plant diversity; however, assessment of wetland mitigation success should not be confined to one diversity value per wetland, as a single wetland is composed of an assortment of heterogeneous patch habitats that can vary greatly - in hydrology, plant community, and soil chemistry (De Steven and Toner 2004). Additive partitioning of species diversity is an effective method of calculating species diversity of landscapes at different spatial scales that have various local and universal environmental variables that influence different habitat patches (Avila et al. 2011). Diversity partitioning quantifies overall species richness (gamma overall diversity) in a diverse habitat like a wetland, and takes into account the richness of the individual patches within the wetland (local alpha diversity) and between patch differences in species composition (beta spatial heterogeneity or species turnover) (Thiere et al. 2009; Whittaker 1972).

Beta diversity indices have been used in several recent studies to measure species turnover and species composition in wetland habitats. For example, the analysis of phytoplankton and macroinvertebrate diversity in streams and wetlands has been used to identify species turnover based on differing hydrologic regimes (Cardoso et al. 2012; Avila et al. 2011). In addition, plant species turnover in restored prairie pothole wetlands has been used to monitor the long-term biotic simplification of wetland flora caused by increases in invasive species and their subsequent competition with native plant species (Aronson and Galatowitsch 2008). Pair-wise beta diversity models can also be a useful strategy for determining which abiotic and biotic variables can generate higher plant species richness in mitigated ecosystems (Cingolani et al. 2010). Wetland restoration studies that use beta diversity as a metric for plant community success have been limited, especially for nontidal or depression wetlands within the Atlantic Coastal Plain; therefore, a more definitive understanding of the effects of local and regional characteristics and restoration methods on constructed and restored wetland vegetation is needed.

An ideal site to look at the effects of various wetland restoration techniques and the effects of biotic and abiotic factors on alpha and beta species richness is the Jackson Lane Preserve on Maryland's Eastern Shore. This 121-hectare conservation site in Caroline County, Maryland contains protected natural forested areas as well as about a dozen geographically isolated depression wetlands. Geographically isolated depression wetlands are temporarily flooded ecosystems that can be found along the East Coast from New Jersey to Florida. These elliptical shaped wetlands vary in size from several thousand square meters to several hundred hectares and are seasonally to perennially inundated depending on the size of the area. The hydrology of these wetlands is mostly groundwater driven with fluctuations influenced by seasonal water table configurations regulated by evapotranspiration, precipitation, and the amount of vegetation in and around the depression. (Phillips and Shedlock 1993; McAvoy and Clancy 1994; Lee 1995).

Specifically in Maryland, these geographically isolated depressional wetlands are referred to as Delmarva Bays, as the majority are located along the eastern shore peninsula shared by the state lines of Delaware, Maryland, and Virginia (McAvoy and Bowman 2002; Stolt and Rabenhorst 1987). These bays are typically smaller than those in other regions, but other than their geographic location and size, Delmarva Bays are not significantly different from Carolina Bays found farther south on the Atlantic Coastal Plain. This subset of wetlands has not been studied to the extent of other types of geographically isolated wetlands, but several reports indicate they provide critical habitat for a variety of fauna and flora. McAvoy and Bowman (2002) show that Delmarva Bays are significant habitat for a diverse number of plant species including about 45 rare and uncommon species, eight of which are deemed globally rare by the Nature Conservancy and one species, Canby Dropwort (*Oxypolis canbyi*), is classified federally endangered by the U.S. Fish and Wildlife Service. Amphibian species and other less mobile taxa have also been known to prefer Delmarva Bay wetlands. For instance, carpenter frogs, an amphibian species that has been given a 'need for conservation' status by the state of Maryland are distributed throughout Delmarva Bays, preferring bay landscapes with intermediate hydroperiods, acidic water chemistry, and a buffer of forest cover (Otto et al. 2007). Similarly, Delmarva Bays provide avian species with refuge during winter months. Diversity of the overwintering bird community is often greater around Delmarva Bays when compared to non-wetland, forested areas (Czapka and Kilgo 2011). The Delmarva wetlands are also habitat for rare and endangered animal species including three species of salamander, three species of treefrog, and three species of damselfly (McAvoy and Clancy 1994). Other ecosystem services provided by Delmarva Bays include erosion control, improved water quality by filtration of excess nutrients and pollutants**,** and flood mitigation through the disbursement and storage of water over a larger area and period of time (Lee 1995).

Natural reference wetlands are typically used in non-tidal wetland mitigation as models to compare mitigated vegetation composition and community structure to one or many natural counterparts (Moorhead 2013; Brinson and Rheinhardt 1996). The use of reference sites is inefficient when mitigating depression wetlands due to the variability within and among wetlands (Sharitz 2003; De Steven and Toner 2004) as well as the variety of abiotic drivers that can influence vegetation composition (Matthews et al. 2009). For example, a comprehensive Delmarva Bay vegetation composition classification study by Berdine and Gould (1999) cited reference wetlands that modeled the characteristics described in each plant community category; however, some of the reference sites contained multiple different classified plant communities. Despite the challenge in comparing constructed Delmarva Bay wetland communities based on the ones of natural reference wetlands, overall zonal vegetation patterns of Delmarva Bays have been more distinctly defined. Typically higher elevations are dominated by wetland shrubs and trees such as *Liquidambar styraciflua* followed by a woodland marsh with an understory of emergent grasses, sedges, and rushes such as *Panicum verrucosum* at mid elevations, and ending with submerged and floating aquatic species at lower elevations in the wetland center (Tyndall et al. 1990; Sharitz 2003).

The importance of these bay habitats is what caused The Nature Conservancy (TNC) to acquire 133 hectares of farmland adjacent to the natural Jackson Lane Preserve in order to design a large-scale wetland restoration project that would completely restore 20-30 hectares of forested and open-canopy wetlands, and have them resemble natural depression wetlands. Before its acquisition in 1999, the farmland had been under cultivation and had been ditched to drain water that could potentially flood crops or affect pasture land. Other goals for the mitigation included restoring the wetland's hydrological regime, adding about 60 hectares of native forest cover, and restoring basic natural wetland functions for native flora and fauna.

Wetland restoration at Jackson Lane took place between 2003 and 2004 and involved creating 23 wetland ponds that were between 0.2 to four hectares in size and less than 0.3 meters to 1.2 meters in depth, which is comparable to natural Delmarva Bay wetlands (Samson 2007). To enable water retention in each wetland cell and re-establish the proper hydrology, drainage tiles and culverts were removed and ditch plugs were formed by piling excavated substrate directly adjacent to the berms of the wetland ponds. In addition, varying amounts of coarse woody debris (CWD) were positioned in and around pond edges, providing topographic diversity and substrate for small organisms. All CWD came from trees that were felled on-site and were left in the wetlands. The practice of adding CWD to depression wetland mitigation sites can increase the species richness and biomass of macroinvertebrate and plant communities, simultaneously increasing overall wetland health (Alsfeld et al. 2009). Wheat straw and barley straw were also deposited in multiple wetlands on bare pond substrates to inhibit the growth of highly competitive wetland plant species, specifically *Typha latifolia*. In a study by Suter et al. (2006), straw deposition used in the restoration of fen meadows was successful in reducing the germination of a competitive, early successional willow species. After the addition of CWD and straw, over 80,000 native tree saplings and shrubs were then planted around wetland pond edges and uplands between wetland cells in order to restore native forest cover (Samson 2007).

Post-restoration studies at Jackson Lane have focused on macroinvertebrates, amphibians and reptiles, hydrology, and water chemistry (Tice 2006; Culler and Lamp 2009; Otto et al. 2007; Samson et al. 2011). Vegetation was studied in six of the created ponds from 2004-2006 (Samson et al. 2011) but that study was not published and no other vegetation studies have taken place in the interim. Plant species diversity and composition plays an important role in the overall characterization of these wetland ecosystems and needs to be better understood within created and restored Delmarva Bays.

The objective of this study is to revisit these wetlands and examine the plant community nearly a decade after restoration. The eight depression wetlands chosen for this project vary in size, depth, hydrology, CWD input, and straw treatment (Table 1). Specifically, the objectives of this study are to characterize the vegetation of these constructed or restored Delmarva bays with respect to flooding/saturation, soil nutrients, and restoration treatments, identify variables associated with high levels of richness and beta diversity at different scales, and to determine which environmental variables are associated with different plant communities within this wetland complex.

The intermediate disturbance hypothesis suggests high diversity is maximized in ecosystems where species composition is continually changing due to disturbances that occur at intermediate levels of frequency and intensity. In response to these intermediate disturbance levels, competitive elimination of opportunistic species should decrease, subsequently allowing for the niche specialization of plant species according to their degree of tolerance to the disturbance (Connell 1978). The primary disturbance in Delmarva Bays is the frequency of wetting and drying that occurs throughout the year depending on precipitation patterns and evapotranspiration rates (Sharitz 2003; Phillips and Shedlock 1993). In depression wetlands, vegetation types are known to be strongly correlated to hydrologic regime and soil type, and not wetland size (De Steven and Toner 2004; Berdine and Gould 1999). Wetland size is instead related to hydrologic regime in that larger wetlands can provide more open-water space, leading to slower drawdowns during dry periods and thereby increasing the opportunity for diverse wetland vegetation (De Steven and Toner 2004).

Using the assumption of the intermediate disturbance hypothesis, wetlands that dry down gradually throughout the season should have higher richness and beta diversity because they will be able to provide the greatest amount of habitat for both flooding tolerant and flooding intolerant vegetation compared to wetlands that are mostly wet or mostly dry for the entire season. Wetlands with less canopy cover that provide more open water habitat should result in higher species richness due to decreased shading. Coarse woody debris treatments were placed within specific wetlands to increase topographic diversity; therefore, we expect that wetlands with the highest amounts of CWD will also have high species richness and beta diversity due to the habitat variability that the treated areas can provide. Finally, we hypothesize that high nitrogen and phosphorus loads in soils should decrease plant diversity since wetlands created on former agricultural land are known to have lower diversity due to excess nutrient loading (Thiere et al. 2009), increased nutrient cycling, particularly of phosphorus, (Verhoeven et al. 2006), and increased competition between species in nutrient rich areas (Tilman 1987; Drexler and Bedford 2002).

Methods

Vegetation Sampling

Restored and created wetland cells used for this study were chosen to encompass a range of sizes, hydroperiods, coarse woody debris application, and straw treatment type (Table 1). All vegetation sampling methods were developed de novo. For each wetland cell, four to six transects were established across the upland-wetland gradient to the center of the wetland. All wetlands less than 1.2 hectares in size were given four transects, and wetlands with greater than 1.2 hectares in size were given six transects. Transect locations were chosen by generating a random number and walking the number of paces produced along the edge of the wetland's tree line. From this position, transects were set up perpendicular to the wetland's center, running from the center of the ponded area to the first five to ten meters of dry, forested upland or impenetrable briar stands situated in uplands, as determined by a change in vegetation community composition. Vegetation sampling was done from June 3-June 16 and August 19-August 30 during the summer of 2013.

Transects were divided into vegetation zones based on plant community composition and local hydrology. Upland vegetation zones along each transect were defined prior to vegetation cover sampling as completely dry, forested areas with mature hardwood species and upland grasses. During vegetation sampling, transition zones were identified as areas with flooding and saturation in combination with hardwood and emergent vegetation cover. Wetland center vegetation zones were identified as open ponded areas that were flooded, contained little to no canopy cover, and were dominated by aquatic vegetation. Vegetation cover was estimated at each wetland site using one meter square plots. Vegetation zones less than five meters wide had one sample plot, zones between six and ten meters wide had two plots, and zones greater than ten meters wide had three plots. For each sampled one meter squared plot, percent cover of all herbaceous and woody vegetation shorter than one meter was recorded and identified to species. To capture additional species that may be infrequent in the area, a belt transect was walked one meter on either side of the transect line to record any plant species not previously encountered in the plot. Woody vegetation greater than one meter tall was counted and identified to species in a five by two meter plot in the middle of each identified vegetation zone.

Environmental Variable Sampling

During each vegetation sampling period, multiple environmental variables were measured in each plot. Percent cover of vegetation, bare ground, and leaf litter were recorded as well as litter depth. Litter depth was measured at three random points in each plot and reported as an average. A soil moisture meter was used to determine the percentage of moisture within the top 15 centimeters of soil. Flooding presence was also determined at each plot and recorded either as dry, saturated, or flooded. In addition, interim soil moisture readings and flooding presence observations were recorded every two weeks, alternating with vegetation sampling periods. Wetland hydrology was calculated by dividing all observations of plot-level flooding and saturation by the total number of observations made at that wetland.

Soil nutrients were sampled on a random subset of half of the transects in each wetland. Soil was sampled to a depth of 15 centimeters in each sample plot along each transect using a hand soil core, and were bulked into the three hydrological categories (upland, transition, wetland center). Soils were analyzed for total carbon and nitrogen (Nelson and Sommers 1996; McGeehan and Naylor 1988), carbon to nitrogen ratio, available nitrogen (Dahnke 1990), total phosphorus (Olsen and Sommers 1982), and soil texture (Hydrometer Method 2002) by a third party company (Brookside Laboratories, New Bremen, OH).

Canopy cover photos were taken in August 2013 using a fish-eye lens camera. Photos were taken at the midpoint of each vegetation zone identified. Once collected, photos were analyzed in Gap Light Analyzer for percent sky area, canopy openness, and site openness (Frazer et al. 1999). Canopy cover was calculated by subtracting percent canopy openness from 100 (Frazer et al. 1999). Elevation for each wetland was obtained using LiDAR data from Maryland Department of Natural Resources (MDNR 2006). The LiDAR obtained was converted to raster files using Grid Batch (Min-Lang 2005) and then exported into ArcGIS. Slope was measured for each spatial scale. Transect slope was determined by calculating the difference between the beginning and end of transect elevations, and then dividing by the total length of each transect surveyed. For wetlands, an average slope of all transects within each wetland was calculated. Coarse woody debris was determined by counting the number of logs observed per wetland and categorized as none (0), medium (15-30), and high (40-60).

Statistical Analysis

Species diversity

Species lists at the plot level over both seasons (June and August) were aggregated into their respective vegetation zones (upland, transition, wetland center) for analysis of transect-level diversity patterns. Transect gamma diversity was defined as the combined total of the number of species found in the plots sampled along each transect in addition the number of plant species identified on the belt transect. Wetland gamma diversity was similarly defined as the sum of all species, including belt transect species, found in the four to six transects sampled. To examine beta diversity within each transect (BWT; upland to wetland gradient) the gamma species richness of each transect was divided by the average richness of each vegetation zone. To look at beta diversity between the sampled transects (BBT) in each wetland, wetland gamma diversity was divided by - transect gamma diversity. Values for transect-level BWT and BBT were averaged within each wetland to generate wetland-level values for these variables. The partition of species diversity into these segments provides an efficient framework for community diversity measures because diversity can be measured at different levels as well as a consolidated unit (Thiere et al. 2009).

Spearman Correlations

The majority of the environmental variables recorded had a non-normal distribution. To investigate relationships between variables, Spearman correlations were run between all vegetation and environmental variables at the transect level (SPSS Ver 21).

Akaike information criterion (AIC) modeling

Akaike information criterion models were formulated at the wetland level in order to determine which environmental variables had the most influence on species richness and beta diversity within and between transects. This type of modeling was chosen because it can account for the complexity of the study. It can also provide inferences for more than one model, which is not an attribute found when using models such as null-hypothesis testing or maximizing fit through adjusted R^2 (Johnson and Omland 2004). All AIC models were performed using R-Studio AGPL Ver 3.

Prior to model selection for AIC analysis, gamma diversity and beta diversity values were analyzed for normality using the Shapiro-Wilk test. All diversity calculations were considered normal distributions. The models run were formulated by setting each diversity statistic equal to one or more of the environmental variables observed. Due to the constraints of the degrees of freedom available for the model, only six environmental variables (wetland size, flooding and saturation scores, coarse woody debris, straw type, available nitrogen, and silt) were chosen to run in model selection. Wetland size, hydrology, nutrient content and soil texture were of interest because they are generally correlated with wetland plant species richness based on scientific literature (De Steven and Toner 2004; Matthews et al. 2009; Berdine and Gould 1999). Coarse woody debris and straw type were of interest because they were used in these wetland creation projects with the intent of increasing species diversity. Available nitrogen was chosen as a surrogate for overall soil nutrient concentration as soil carbon, total N, total P, C:N ratio and available N were all positively correlated with each other (Table 2). Soil texture fractions (% of sand, silt and clay) were also auto-correlated (Table 2) and silt was selected to represent the overall effects of soil texture. After running the formulated models, the AIC values calculated were corrected due to the small sample size using the AICc formula and evidence ratios were calculated in order to identify the models most consistent with the data. Only models with deltaAIC values less than or equal to two and evidence values less than or equal to three were reported (Burnham and Anderson 2010).

Community Composition

Community composition was examined using the maximum cover value of each species in each plot over the June and August sampling periods and averaged across all three surfaces at the transect level. Rare species (occurring only in one or two wetlands) were dropped from the data set. Multiple Response Permutation procedures (MRPP) were used to determine if select restoration treatments or wetland conditions resulted in significant differences in plant community composition. Analyses were run using each wetland as a unique group and then by grouping wetlands by CWD treatment (none, low: 15-30 logs/wetland and high: 40 – 60 logs/wetland), historic hydroperiod (short, medium, long) based on Samson et al. (2011), size (0.83-2.11 acres, 2.74-4.71 acres, 8.17-9.07 acres) and straw type used (wheat or barley). MRPP analyses were run in PCORD 5 with a Relative Euclidian distance measure.

Non-metric multidimensional scaling (NMDS)

Non-metric multidimensional scaling provides a multi-dimensional view of community data and is typically the method of choice when analyzing ecological communities (McCune and Grace 2002). Plant species community composition at the transect scale for the entire sampling season was modeled in response to all environmental variables assessed during the study in an NMDS ordination using the Sorenson’s distance measure. The same species database that was used for the MRPP analyses was also used for the NMDS. Models were run on autopilot and solutions were selected based on multiple runs with real and randomized data. The most influential species and environmental variables were determined by associating their abundance values with each ordination axis using the Pearson correlation coefficient, and only species and variables with an r of 0.20 or greater were reported.

Results

Vegetation Community Characteristics

From June to August 2013 a total of 141 plant morphospecies within 57 families were found. Of these, 130 were completely identified to species, while 11 morphospecies lacked the necessary reproductive structures for identification. According to the USDA plant database (http://plants.usda.gov/java/), 110 of the species identified are classified as native to Maryland and 20 are classified as introduced/exotic species.

Plants were categorized into wetland indicator statuses using the 2014 National Wetland Plant List (NWPL) given by the Army Corps of Engineers (Lichvar et al. 2014). The distribution of the 130 plant species identified across the five indicator categories was as follows: 35 obligate, 28 facultative wetland, 23 facultative, 29 facultative upland, 6 upland, and 8 no indicator (NI).

Wetland-level gamma diversity ranged from 45-78 species with wetland 6 having the least gamma diversity and wetland 2 having the greatest (Table 3). At the transect level, richness increased with bare soil coverage (Table 4) and decreased with higher concentrations of carbon, available nitrogen, total phosphorus, total nitrogen, and greater silt composition proportions (Table 5). Beta diversity within transects ranged from 2.9 to 3.5 with wetland 2 having the least average beta diversity and wetland 3 having the greatest (Table 2). Beta within transects was negatively correlated with flooding and saturation (F&S) scores (Table 4) as well as carbon and total phosphorus

(Table 5), indicating that more species turnover from wetland to upland vegetation zones occurs in areas with greater water level fluctuations and lower nutrient concentrations. Beta diversity between transects ranged from 1.82 to 2.10 with wetland 2 having the least beta diversity and wetland 6 having the greatest (Table 3). Beta between transects showed a strong positively correlated relationship with F&S scores (Table 4).

Tree Community Characteristics

Fourteen tree species over one meter tall were identified in the eight wetlands sampled, with *Liquidambar styraciflua* (sweet gum) and *Acer rubrum* (red maple) dominant in all wetlands. Among the 14 species identified, two were obligate, four were facultative wetland, four were facultative, and four were facultative upland. Most individuals ranged from 2-12 cm in diameter at breast height (DBH). The wetland-level-average DBH ranged from 3.88 cm to 7.52 cm (Table 6). Wetland 1 contained tree species with the greatest average DBH (7.52 cm). The largest hardwoods within wetland 1 were typically located in the wetland center vegetation zone and were greater than 12 cm in diameter at breast height. Tree species richness in each wetland ranged from 4 to 10 species, with the lowest diversity in wetlands 1, 6, and 11 and the highest diversity in wetland 2 (Table 6).

Environmental Variable Sampling

Wetlands 1, 2, 6, and 7 exhibited the highest flooding and saturation (F&S) scores. Higher F&S scores demonstrate that most or all of the plots within a wetland transect are flooded throughout the sampling season, indicating a long hydroperiod for that particular transect. Conversely, low F&S scores identified wetland transects that dried down to a greater extent than other transects sampled (Figure 1). Wetlands 3 and 11 exhibited the lowest F&S scores. Wetlands 15 and 19 had moderate F&S scores that were generally distributed along the range of 0.8 to 0.9, with the exception of two transects within wetland 19 that stayed completely flooded.

Elevations of each wetland ranged from 14-16 meters above sea-level and slopes ranged from 0.009 m/m to 0.03 m/m with wetland 11 being the shallowest wetland observed and wetland 3 being the deepest (Figure 2). Average canopy cover measurements within each wetland ranged from 69-93% with wetland 2 having the least canopy coverage and all other wetlands exhibiting a canopy cover between 81% and 93% (Figure 3). Percent cover observations showed on average that each wetland contained a higher percentage of litter cover than vegetation cover and bare ground cover (Figure 4). Wetland 2 was the only wetland site that contained similar proportions of vegetation and leaf litter cover.

Soil nutrient analysis revealed that wetlands 1 and 6 had large amounts of phosphorus (Figure 5) and nitrogen (Figure 6), while carbon was only high in wetland 1 (Figures 7). Spearman correlations between soil nutrient variables and F&S scores showed that carbon, nitrogen, total phosphorus, total nitrogen, and clay and silt compositions showed a positive relationship with increasing F&S scores (Table 2). Medium and short hydroperiod wetlands followed this trend, exhibiting lower levels of soil nutrients when compared to long hydroperiod wetlands. Wetlands 3, 11, and 19 were primarily comprised of sand (ranging 60-80%) whereas wetlands 1 and 6 had a greater silt composition on average (Figure 8). Wetlands 2, 7, and 15 had soil compositions that were a little over fifty percent sand, thirty five percent silt, and fifteen percent clay on average (Figure 8).

AIC modeling

Plant diversity was considered at two spatial scales (wetland and transect) in determining the variables that drive plant community diversity. Gamma diversity models (richness in the entire wetland) resulted in combinations of the null model, straw, wetland size, silt, and available nitrogen in the top model solutions (Table 7). First-order jackknife estimates of species richness showed that although fewer transects were sampled in the smaller wetlands, richness was adequately characterized and the effect of size on gamma diversity is not due to a difference in sampling effort (Table 8). Beta diversity within transect models presented combinations of straw and silt or available nitrogen as the best solutions (Table 9). Beta diversity models between transects resulted in silt being the best solution (Table 9).

In comparison with the herbaceous vegetation modeling, the tree community modeling responded with mostly null models as being the best solution for gamma diversity and beta diversity between transects (Table 10). Silt was the best solution for beta diversity within transects. The only other variable presented in the top three model solutions was CWD for beta diversity between transects.

Community composition

The MRPP analysis demonstrated that the wetland cell groupings provided the strongest relationship in explaining community composition differences (Table 11). In general, wetland 2 had the most distinctive plant community from all of the other wetlands sampled. In addition, wetland eleven's plant community is most different from wetlands 1, 15, and 6, and wetland one's plant community is most different from wetland 6. Wetlands 3, 6, 7, 15, and 19 were not dissimilar from one another in community composition. Coarse woody debris treatment given to each wetland also showed significant differences between all wetland plant community compositions. Analysis of hydroperiod, wetland size, and straw type also resulted in significant p-values; however, the effect sizes for these variables were not as large as CWD.

The non-metric multidimensional scaling analysis of wetland transects confirm the results seen in the MRPP analysis. Wetland 2 was observed as the most tightly clustered and isolated cell within the data distribution. In addition wetland 11 was separated from cells 1, 15, and 6, and wetland 1 was separated from wetland 6 (Figures 9-10). The variables that were most correlated with plant species composition were average percent vegetative cover, average percent litter cover, CWD, and soil chemistry and texture. Wetlands 1, 3, 6, and 15 have high levels of *Scirpus cyperinus,* which is positively correlated with axis one and negatively corrected with axis 2 (Figure 10). Wetlands 7, 11, and 19 are most correlated with *Acer rubrum*, which is most related to axis two in the distribution (Figure 10). Wetland 2 is highly correlated to *Lemna minor, Ludwigia palustris, Sparangium americanum, and Brasenia schreberi*, which are all specifically correlated to axis three (Figure 10). Wetland 11is most correlated with high average percent litter cover and has low average percent vegetative cover. The opposite can be seen of wetland 2.

In the NMDS distribution that focused specifically on wetland transects sampled for soil nutrients, the soil texture (percent silt and percent sand), total nitrogen, total phosphorus, and coarse woody debris environmental variables were shown to be strongly correlated in addition to average percent litter and average percent vegetation (Figure 11). Wetlands that were more related with high nutrient levels include cells 1 and 6 (Figure 11). High sand soil composition was most related to wetlands 2, 11, and 19 (Figure 11). Higher CWD treatments were correlated with wetland transects with a high average percent vegetative cover and a predominantly sand soil texture. Wetlands 1, 6, and 15 were most correlated with a silt texture, total phosphorus, available nitrogen, and coarse woody debris. Wetland 2 continued to be tightly clustered within the distribution and was correlated with a sandy soil texture. Wetland 3 was the only site that presented a split distribution.

Discussion

The results show that a combination of local environmental variables and restoration techniques can provide high diversity within created and restored wetlands. Our hypothesis for this study was that plant species diversity should be greatest in areas that dry down gradually, which was supported as higher species richness and BWT were correlated with lower flooding and saturation scores. Environmental drivers for plant species richness are shown to be scale dependent. Species richness at the transect level seems to be primarily driven by hydrology and soil nutrients, whereas size, soil nutrients, soil composition and straw type are primarily driving richness at the wetland level.

Influences on species richness and beta diversity at the transect scale

It is well known that the levels of soil nutrients and the frequency of flooding and drawdown within wetlands cause changes in plant diversity between wetlands (Mitsch and Gosselink 1993; Matthews et al. 2009; De Steven et al. 2010; Bedford et al. 1999); therefore, it is not surprising that hydrology and soil nutrients are primary drivers of plant species richness at the transect level. In general, transect species richness is negatively correlated with flooding and saturation scores. Wetland transects that experience a gradual drawdown throughout the season provide a greater range of soil moisture for supporting additional species. In addition, higher sand compositions will support a larger number of plant species since it has a low water holding capacity, allowing it to be more efficient in drying. Increasing bare ground cover will also support higher species richness since it is indicative of low competition between plant species.

Carbon, nitrogen, total phosphorus, and silt composition were also negatively correlated with transect species richness. This relationship is primarily driven by wetlands 1 and 6, which displayed greater amounts of soil nutrients, a low sand composition, and low species richness. Wetland 1 was the only wetland cell within Jackson Lane that was previously a Delmarva Bay wetland. Prior to restoration, wetland 1 had been ditched and drained, but was not tilled for agricultural production. Agricultural tilling typically strips the soil within each field of clays, silts, and soil nutrients such as nitrogen, phosphorus, and carbon; therefore, high soil nutrient availability within wetland 1 was most likely present prior to the restoration process. Wetland 6 also exhibited higher nutrient levels of nitrogen and phosphorus, suggesting that it acts as a nutrient sink as a result of it having one of the lowest elevations within the entire mitigation complex. High nutrient concentrations within wetlands 1 and 6 are detrimental to species richness in that dominant competitor species, are given more opportunity to prevail over other species, thereby decreasing species richness (Drexler and Bedford 2002; Tilman 1987).

Beta diversity metrics provide insight into how species turnover between vegetation zones affects the diversity of the entire wetland. The wetland of Jackson lane exhibit a distinct wetland to upland zonation: aquatic plant species (*Ludwigia sp.*, *Proserpinacea sp.*) in the center, followed by woodland marsh (*Acer rubrum*, *Liquidambar styraciflua*) with emergent sedges and rushes (*Juncus sp. Cyperus sp.*), and finally upland forest with herbaceous species (*Solidago sp*., *Allium vineale*). These zonal patterns are similar to those previously identified by Tyndall et al. (1990) and Kelley and Baston (1955). Flooding and saturation scores were negatively correlated with beta diversity within transects as well as carbon and total phosphorus, suggesting that plant species turnover from wetland center to upland occurs within drier habitats that are less nutrient rich. Beta diversity between transects displayed the opposite effect, presenting a positive correlation with flooding and saturation scores. Unlike species turnover within transects, wetter habitats promote greater species transference from one transect to another through hydrological connection with other transects in the same wetland.

Influences on species richness at the wetland scale

Our results indicate that the null model (model with no variables) explains plant species richness at the wetland scale better than any other model; however, silt composition, available nitrogen, straw type, and size were still included within the deltaAIC and evidence cutoff values. Beta diversity showed similar results, with silt explaining the variation between transects, and either silt, available nitrogen, or straw explaining the variation within transects. Soil texture and nutrient loads reflect the same pattern seen with correlations at the transect scale. There is not much known information on how straw type can affect plant species richness other than being successful in reducing the germination of a competitive, early successional plant species. A study by Suter et al. (2006) confirms this knowledge, showing that straw deposition in the restoration of fen meadows reduced the germination of a highly competitive willow species. Our results suggest that straw type can be a determinant of plant community diversity since wheat treated wetlands were typically more diverse than barely treated wetlands. Out of the 140 plant species identified, 60 were only found in wheat treated wetlands and eight species were only found in barley treated wetlands.

In terms of size, our results suggest that size is a determinant in plant community richness in that bigger wetlands typically provide a greater species richness. Size is not usually an important abiotic driver in Delmarva Bays because hydrologic regime is the primary influencing factor, regardless of wetland size (Matthews et al. 2009; De Steven and Toner 2004; Samson et al. 2011). Samson et al. (2011) indicated similar results that wetland size did not influence the species composition observed in Jackson Lane in 2006 or 2008. Our results show that Wetland 2 drives the relationship between size and species richness since it was the largest wetland observed and contained the highest gamma diversity among all the wetlands sampled. The low canopy cover in Wetland 2 and moderate drying during a wet year increases the ability of understory species to survive and spread, which can ultimately increase species richness within the wetland.

Influences on species composition at the transect scale

Species composition of wetland transects is primarily driven by differences in community composition among individual wetlands and by CWD treatments. Multiple Response Permutation procedures analyses determined that groupings by wetland and CWD contribute the largest differences in species composition between wetland transects. Plant species composition distributions support this analysis (Figures 10-11), displaying wetlands with little or no CWD treatment being grouped separately from wetlands with high CWD.

In addition to wetland groupings and CWD, soil chemistry and texture influence species composition. Soil chemistry is a largely known driver for plant community composition within wetlands (Bedford et al. 1999; Green and Galatowitsch 2002; Matthews et al. 2009; Weiher and Keddy 1995). This is especially expected within wetland 1 since it did not undergo any tillage for over 30 years of crop farming that occurred prior to restoration. The areas with created wetlands, however, experienced frequent agricultural tilling prior to mitigation and have undergone multiple physical and chemical changes as a result (Samson et al. 2011). Generally, wetlands with higher average percent vegetative cover and higher sand composition were separated from those with greater average percent litter cover and increased soil nutrient concentrations. High nutrient concentrations in combination with greater amounts of organic leaf litter cover allows for dominant, nutrient-demanding plant species to outcompete other species with lower differential nutrient uptake (Verhoeven et al. 2006).

Plant community differences between monitoring in 2006-2008 and 2013

Since post restoration monitoring in 2006 and 2008 the plant community within Jackson Lane has undergone significant changes. In a report by Samson et al. (2011), a similar NMDS distribution showed the plant community of the restored wetland (wetland 1) as being distinctive from all other created wetlands within Jackson Lane. Our results indicate that wetland cell 2 has a plant community that is different from any other created or restored wetland, whereas wetland 1 is only separate from wetland cell 6. The environmental drivers of diversity have also changed since 2008. During 2006 and 2008, hydroperiod was the primary variable explaining plant community differences between the wetlands (Samson et al. 2011); however, a combination of CWD, soil nutrients, and soil texture exhibit a stronger explanation of community composition almost a decade after construction.

Potential limitations and weaknesses

One major issue with this study is that restoration techniques varied between each wetland, but treatments among transects within a wetland did not differ. Our results show that the methods of size and straw type are important in determining diversity at the transect scale; however we cannot determine the true importance of these methods since there is no treatment variation within each of the created wetlands. This complication was known prior to conducting the study, but we were limited by the methodology of how the treatments were applied before restoration. Based on this knowledge, large wetlands or wetlands with barley straw treatments could be similar in other unmeasured ways, which could be the true drivers of diversity.

Another potential issue is that data was collected during an unusually wet summer. Typically, depression wetlands go through a greater period of drying in the summer season due to increased evapotranspiration rates and decreased rainfall (Phillips and Shedlock 1993; McAvoy and Clancy 1994; Lee 1995). Between the months of June 2013 and August 2013 total monthly rainfall ranged from 8.64 cm in August to 13.44 cm in July. In comparison to 2013, the summer of 2012 exhibited drought conditions with total monthly rainfall ranging from 2.54 cm to 3.94 cm from June 2012 through August 2012. Prolonged flooding conditions during the sampling period likely caused intermediate drawdowns within each wetland, which could have increased the amount of plant species richness, specifically aquatic or saturation dependent species, observed among wetlands.

Conclusions

Although the wetlands in Jackson Lane were created or restored within close proximity to one another, wetland plant community distribution and plant species diversity were variable. Based on our results, CWD treatments is likely the most effective mitigation practice in providing topographic diversity to promote the establishment of a diverse plant community within wetlands. The application of straw and a larger wetland size are also valid considerations for wetland mitigation methods. The initial purpose for lining straw within the bottom of each wetland cell was to deter broadleaf cattail (*Typha latifolia*) species from outcompeting other wetland species. For this use, straw was effective in minimalizing presence of this species. Broadleaf cattail was sighted and recorded in six out of eight wetlands; however, three out of the six were only recorded as belt transect species, and two only had cattail presence in one plot with no more than 37.5 percent cover. Future research is needed to assess the underlying causes of straw type being a contributor for high species diversity. Creating large restored wetlands would provide greater species diversity based on the species-area relationship, which states that size has a positive correlation with species richness. This relationship is also reflected in the created and restored wetlands observed within Jackson Lane.

Tables and Figures

Table 1. Characteristics of created wetland pond cells and the restored Wetland 1 as described in Samson et al. 2011 (Y=yes, N=no, Medium=1-15 logs, High=16-20 logs, Very High=>20 logs). Hydroperiod was derived based on biweekly measurements of seasonal water level changes between January or March 2005 to April 2010.

Wetland	Size (ac)	CWD added	CWD amount	Straw Type	Hydroperiod
1	8.17	Y	Very High	wheat	Long
2	9.07	Y	High	wheat	Long
3	2.11	Y	High	barley	Short
6	1.41	Y	Medium	barley	Long
7	2.74	N	none	barley	Medium
11	4.71	N	none	wheat	Medium
15	2.88	Y	Medium	wheat	Medium
19	0.83	N	none	wheat	Short

Table 2. Spearman correlations between environmental variables and soil variables. (Hydroperiod=historic hydroperiod observed in Samson et al. 2011, FloodSat=calculated hydrology during sampling period, C=Carbon, N=Nitrogen, TP=Total Phosphorus, AN=Available Nitrogen). Wetland size and CWD were tested quantitatively using observed values and qualitatively using categories based on the observed values. Values of significance are highlighted in bold.

	C	N	C:N Ratio	TP	AN	Clay	Silt	Sand
Size	.090	.133	.408	-.156	.092	.288	.019	-.116
SizeCat	.302	.333	**.599****	.012	.230	.339	.169	-.254
CWD	.343	.353	.283	.142	.399	.364	.418	**-.467***
CWDCat	.325	.336	.218	.118	.441	**.474***	.413	**-.504***
Hydroperiod	**.646****	**.657****	.291	.421	**.706****	**.773****	**.645****	**-.788****
StrawCode	0.000	0.000	**-.558***	.186	.227	.165	.227	-.207
FloodSat	**.469***	**.486***	.285	**.485***	**.651****	**.491***	.428	**-.467***
VegAvg	-.072	-.079	.096	-.096	.184	.102	.121	-.121
LitAvg	.172	.152	.176	.193	-.140	-.144	-.012	.039
BareAvg	-.437	-.385	**-.732****	-.342	-.021	.063	-.137	.093
LDAvg	**.533***	**.495***	**.591****	.332	**.507***	.275	.386	-.389
Slope	-.147	-.131	-.342	-.210	-.068	.150	.004	-.058
Canopy	-.046	-.008	.130	.018	-.253	-.421	-.249	.314
Carbon	1.000	**.986****	**.531***	**.870****	**.737****	**.588****	**.858****	**-.875****
Nitrogen		1.000	**.456***	**.867****	**.788****	**.621****	**.868****	**-.892****
C:N Ratio			1.000	.321	.120	.076	**.213**	-.219
TP				1.000	**.735****	**.518***	**.821****	**-.804****
AN					1.000	**.812****	**.751****	**-.832****
Clay						1.000	**.554***	**-.728****
Silt							1.000	**-.960****
Sand								1.000

Table 3. Gamma and beta diversity results for herbaceous and woody vegetation under a meter within each wetland during the sampling season. Gamma diversity is measured as the total number of species observed within each wetland. Beta diversity within transects (BWT) was measured by dividing the gamma diversity of each transect by the average richness of each vegetation zone. Beta diversity between transects (BBT) was measured by dividing the gamma diversity of each wetland by the transect gamma diversity. Both BWT and BBT were averaged across all transects within each wetland to generate wetland scale values.

Wetland	GDiv	BWT	BBT
W2	78	2.941450522	1.821011673
W1	51	2.764701038	2.095890411
W6	45	2.719945055	2.195121951
W15	57	2.996552209	1.868852459
W7	54	3.385912698	1.87826087
W11	61	3.056746032	1.936507937
W3	56	3.548874224	1.866666667
W19	62	3.434899749	1.984

Table 4. Spearman correlations between richness and all environmental variables, except soils (Hydroperiod= historic categories described in Samson et al. 2011, FloodSat=calculated hydrology during the sampling season, LD=Litter Depth). Significant relationships ($p < 0.05$) are highlighted in bold.

Diversity Measure	Size	CWD	Hydro period	Straw Code	Flood Sat	Veg Avg	Litter Avg	Bare Avg	LD Avg	Slope	Canopy
BetaWithinTransect	-0.219	-0.225	-0.32	0.124	**-.393***	-0.055	0.163	-0.006	-0.05	0.305	0.147
BetaBetweenTransect	-0.018	0.076	0.103	-0.036	**.399***	-0.079	-0.004	-0.319	0.029	0.05	-0.103
Richness	0.304	-0.026	-0.109	-0.271	**-.368***	0.262	-0.225	**.365***	0.001	0.081	-0.157

Table 5. Spearman correlations between richness or beta diversity and soil variables. Significant relationships ($p < 0.05$) are highlighted in bold.

Diversity Measure	Carbon	Nitrogen	C:N Ratio	Total Phosphorus	Available Nitrogen	Clay	Silt	Sand
BetaWithinTransect	**-.465***	-.450	-.398	**-.542***	-.425	-.386	-.433	.454
BetaBetweenTransects	.340	.319	.315	.360	.286	.356	.158	-.240
Richness	**-.680****	**-.664****	-.153	**-.799****	**-.511***	-.379	**-.587****	**.577****

Table 6. Gamma and beta diversity results for woody vegetation over a meter within each wetland during the sampling season. Gamma diversity is measured as the total number of species observed within each wetland. Beta diversity within transects (BWT) was measured by dividing the gamma diversity of each transect by the average richness of each vegetation zone. Beta diversity between transects (BBT) was measured by dividing the gamma diversity of each wetland by the transect gamma diversity. Both BWT and BBT were averaged across all transects within each wetland to generate wetland scale values.

Wetland	Average DBH	Gamma Div	BWT	BBT
1	7.52	4	1.991666667	1.846153846
2	4.48	10	2.17797619	2.4
3	5.26	7	1.709821429	2.153846154
6	5.97	4	2.5625	2
7	4.81	6	1.5	1.846153846
11	4.23	4	1.416071429	1.263157895
15	4.90	5	1.5	1.538461538
19	3.88	7	1.7	2

Table 7. Gamma diversity Akaike Information Criterion (AIC) model results for the sampling season.

Name	Nopar	AIC	AICc	delta_AICc	wt	Evidence
null	1	62.05374139	62.72040806	0	0.154	1
Size+Silt	3	56.74602636	62.74602636	0.025618303	0.152	1.013157895
AvailableNitrogen	2	60.68084431	63.08084431	0.360436248	0.128	1.203125
Silt	2	60.83541285	63.23541285	0.515004794	0.119	1.294117647
Straw	2	61.31427142	63.71427142	0.993863363	0.093	1.655913978
Size	2	61.7660492	64.1660492	1.445641139	0.075	2.053333333
Size+AvailableNitrogen	3	58.4193465	64.4193465	1.698938437	0.066	2.333333333

Table 8. Species area curve analysis. First order jackknife values are based on PCORD 5 output calculations.

Wetland	Size	Species Richness	First order jackknife species richness
W2	3.67	78	74.2
W1	3.31	51	56.2
W6	0.57	45	58.2
W15	1.16	57	58.6
W7	1.11	54	56.1
W11	1.91	61	53.5
W3	0.85	56	54.6
W19	0.33	62	51

Table 9. Akaike Information Criterion (AIC) model results for beta diversity within transects (BWT) and beta diversity between transects (BBT).

Variable	Name	Nopar	AIC	AICc	delta_AICc	wt	Evidence
BWT	Straw+Silt	3	0.254967614	6.254967614	0	0.218	1
	Silt	2	4.168816214	6.568816214	0.3138486	0.186	1.172043011
	AvailableNitrogen	2	4.805987328	7.205987328	0.951019715	0.135	1.614814815
	null	1	7.095128031	7.761794698	1.506827084	0.103	2.116504854
BBT	Silt	2	-13.83041697	-11.43041697	0	0.516	1

Table 10. Tree community Akaike Information Criterion (AIC) model results for gamma diversity, beta diversity within transects (BWT), and beta diversity between transects (BBT).

Variable	name	Nopar	AIC	AICc	delta_AICc	wt	Evidence
Gamma Diversity	null	1	37.50705856	38.17372522	0	0.374	1
BWT	Silt	2	-12.59768939	-10.19768939	0	0.425	-12.59768939
	null	1	-8.846930302	-8.180263636	2.017425752	0.155	-8.846930302
BBT	null	1	8.986587556	9.653254223	0	0.37	1
	CWD	2	9.231771931	11.63177193	1.978517708	0.138	2.68115942

Table 11. Summary of Multiple Response Permutation Procedure (MRPP) analyses between select environmental variables and species composition at the transect scale. Coarse woody debris, hydroperiod, and straw were placed into qualitative categories (CWD= none, low: 15-30 logs/wetland and high: 40 – 60 logs/wetland; historic hydroperiod= short, medium, long; straw type=wheat or barley).

Variable	T	A	P	Differences
CWD	-9.4768121	0.07458943	0.00000000	All
Hydroperiod	-5.8128374	0.04603439	0.00002317	1 vs 3
Size	-5.6070745	0.04413177	0.00003569	3 vs 1 and 2
Straw	-1.9111018	0.01057891	0.04643978	1 vs 2
Wetland	-13.506090	0.21629818	0.00000000	2 vs 1, 3, 6, 7, 19, 15, 11
				11 vs 15, 1, 6
				1 vs 6

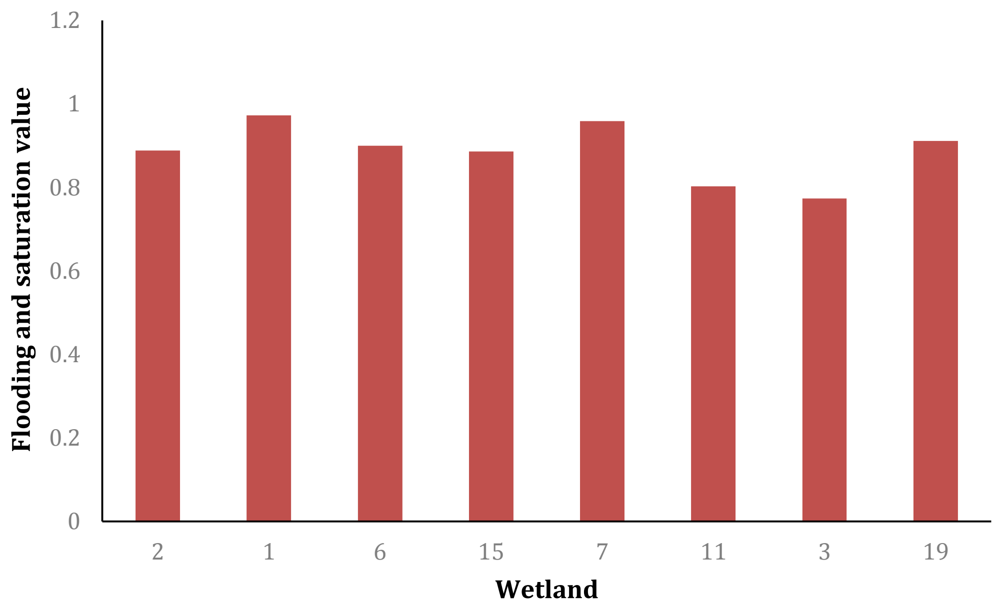


Figure 1. Flooding and saturation (F&S) values within each wetland. Flooding and saturation values were calculated by dividing all observations of plot-level flooding and saturation by the total number of observations made at each wetland.

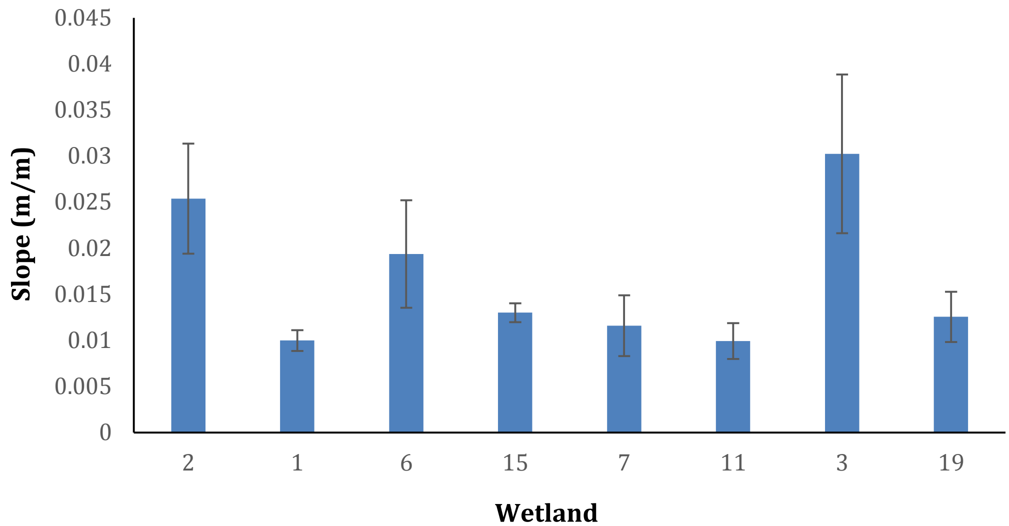


Figure 2. Average slope (± 1 SD) of each wetland. Slope was calculated for each transect by calculating the difference in elevation between the beginning and end of each transect, and then dividing by the total length of each transect surveyed. Slopes were averaged for all transects within each wetland to calculate mean wetland slope.

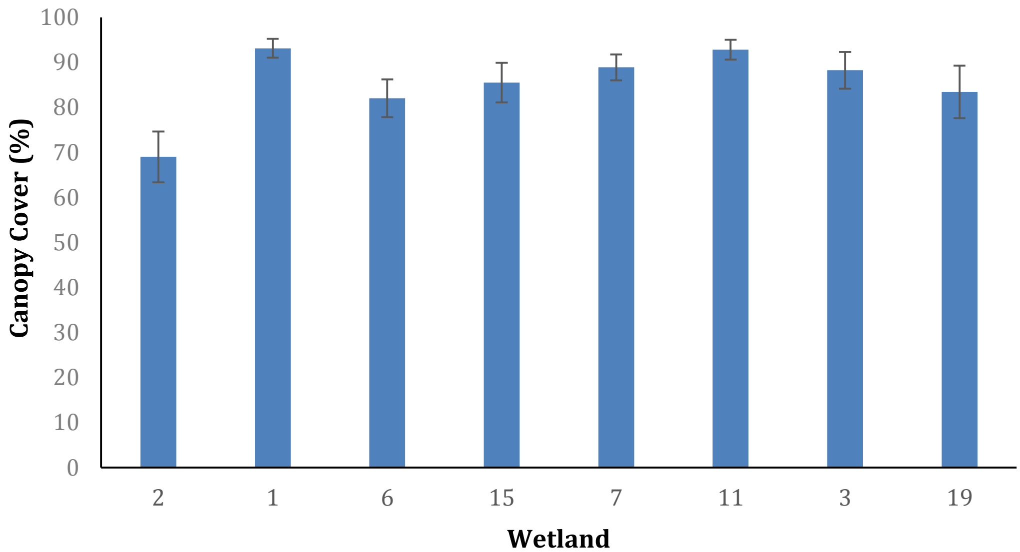


Figure 3. Average percent canopy cover (± 1 SD) for each wetland sampled. Canopy cover was calculated using Gap Light Analyzer and subtracting percent canopy openness from 100.

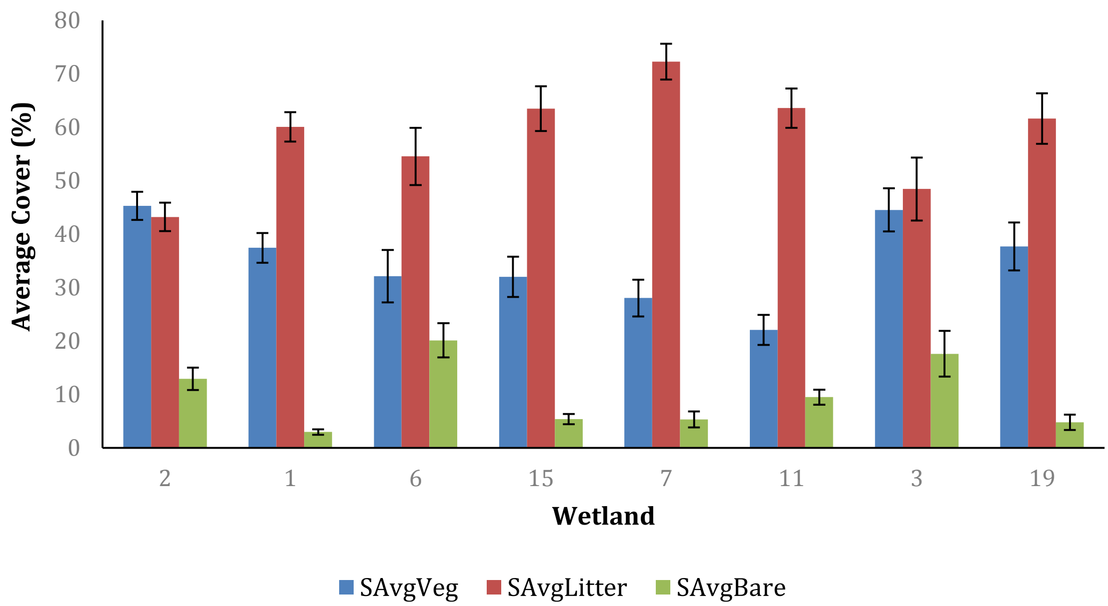


Figure 4. Average percent cover of vegetation, litter, and bare ground (± 1 SD) within each wetland. Percent cover of vegetation, litter, and bare ground were observed at the plot level and averaged for all plots within each wetland.

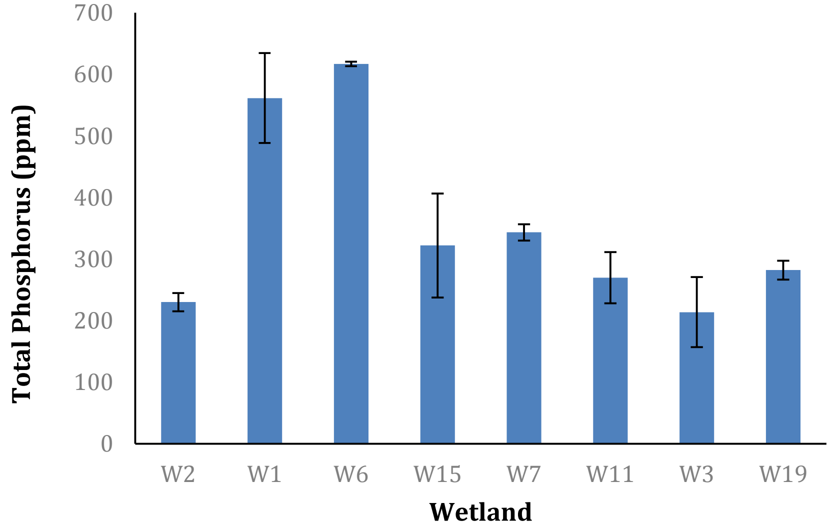


Figure 5. Amount of phosphorus (ppm) (± 1 SD) within the soils of each wetland. Total phosphorus was measured on half of the transects in each wetland and then averaged across transects.

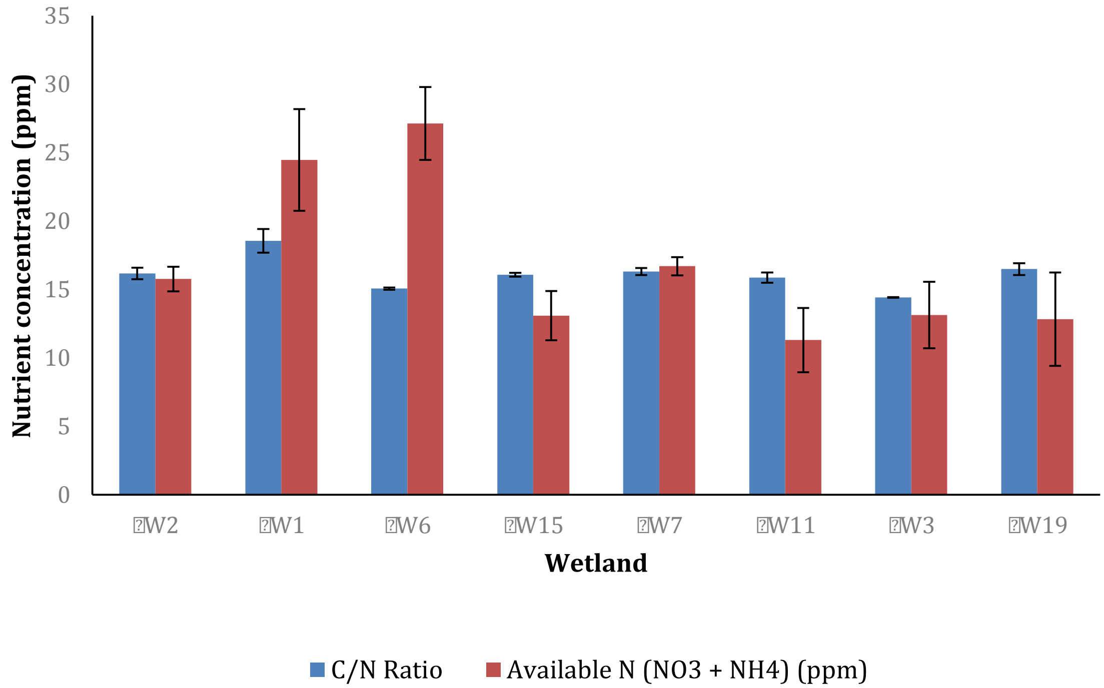


Figure 6. Comparison of C/N ratio and available nitrogen (ppm) (± 1 SD) within the soils of each wetland. Carbon to nitrogen ratio and available nitrogen concentration were measured on half of the transects in each wetland and then averaged across transects.

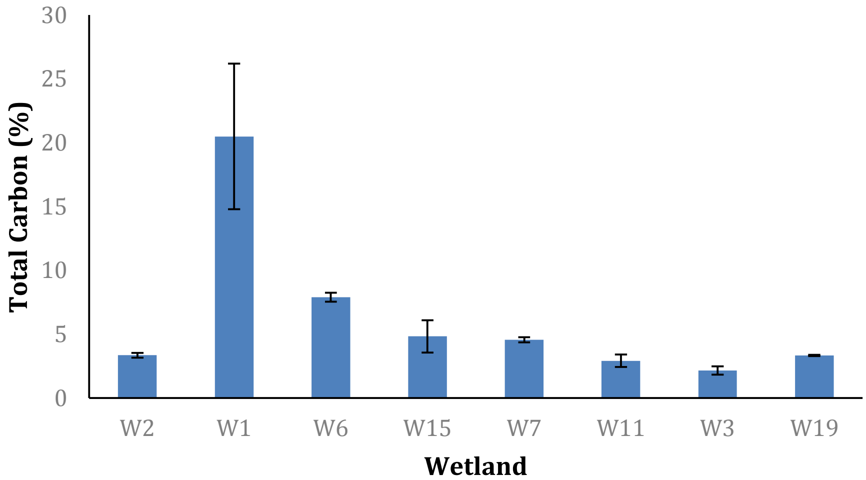


Figure 7. Mean percent total carbon (± 1 SD) within the soils in each wetland. Total carbon was measured on half of the transects in each wetland and then averaged across transects.

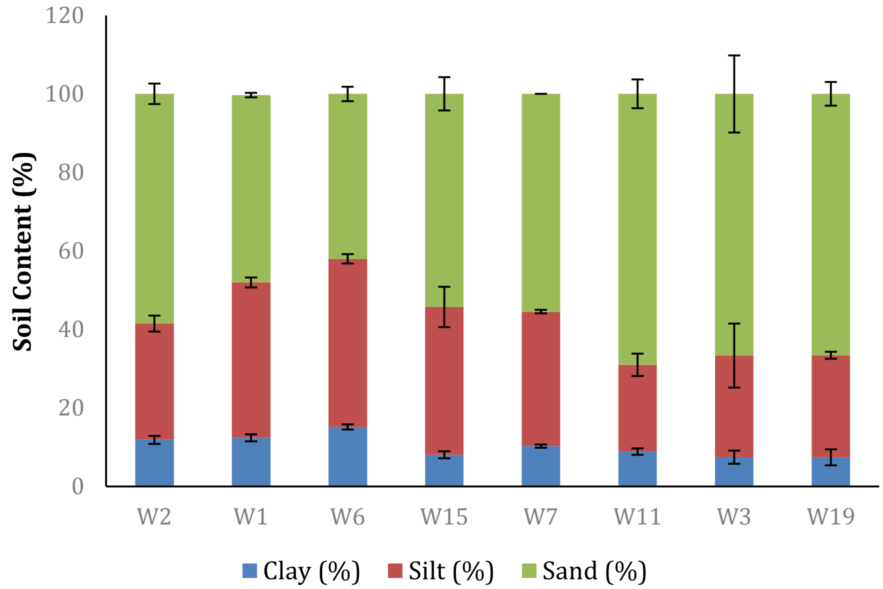
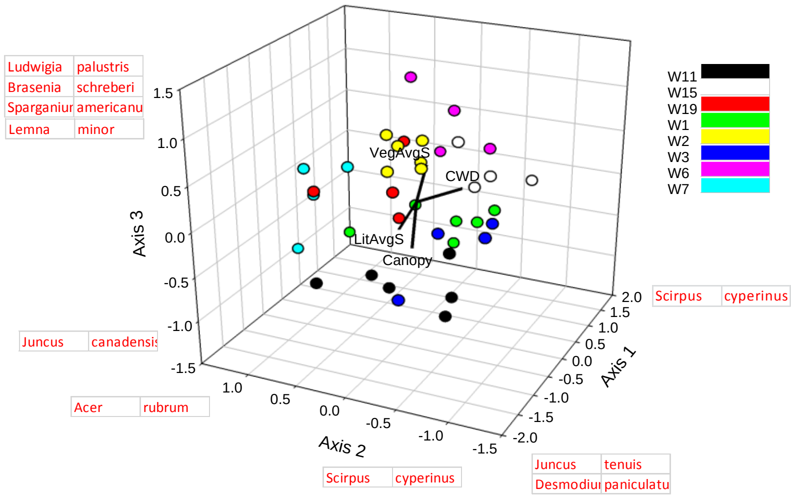


Figure 8. Average percent clay, silt, and sand composition of soils (± 1 SD) within each wetland. Soil composition was measured on half of the transects in each wetland and then averaged across transects.



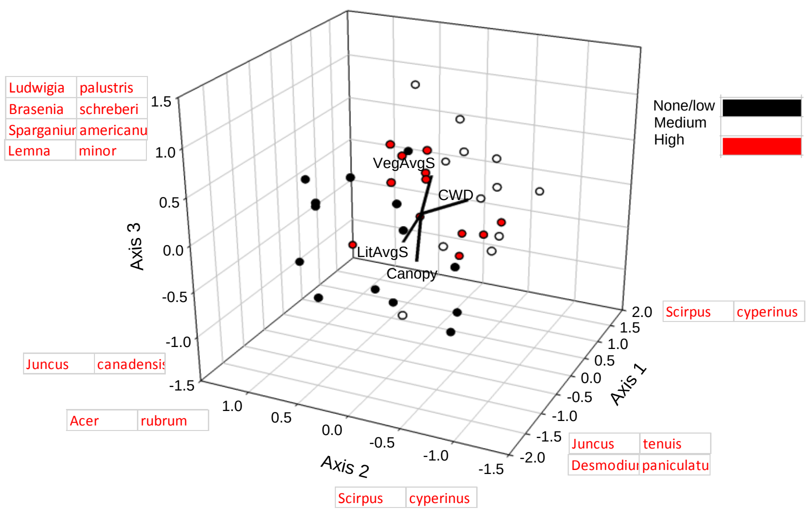
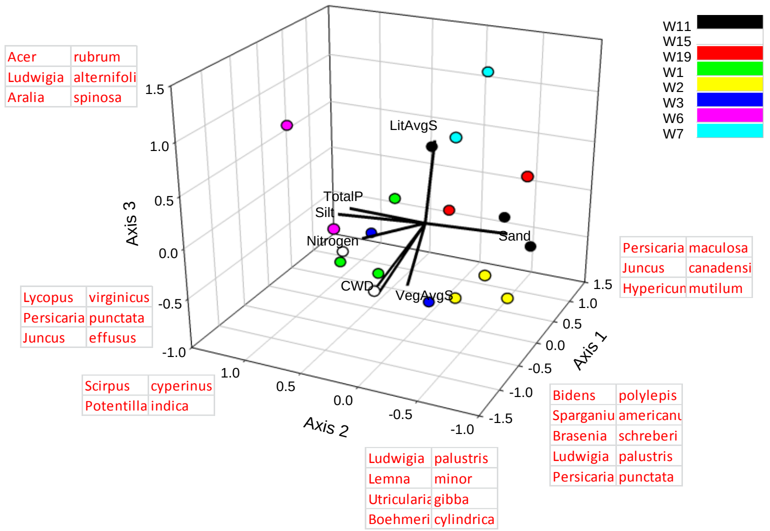


Figure 9. Non-metric multidimensional scaling (NMDS) distributions of wetland plant communities at the transect scale. Each distribution is a 3D depiction of plant community composition based on the average cover of each species along a particular transect. Each transect in the upper distribution is color coded by wetland and each transect in the lower distribution is categorized by coarse woody debris (CWD) amount. Influential plant species identified using the Pearson correlation coefficient, with an r of 0.20 or greater are reported on each axis.



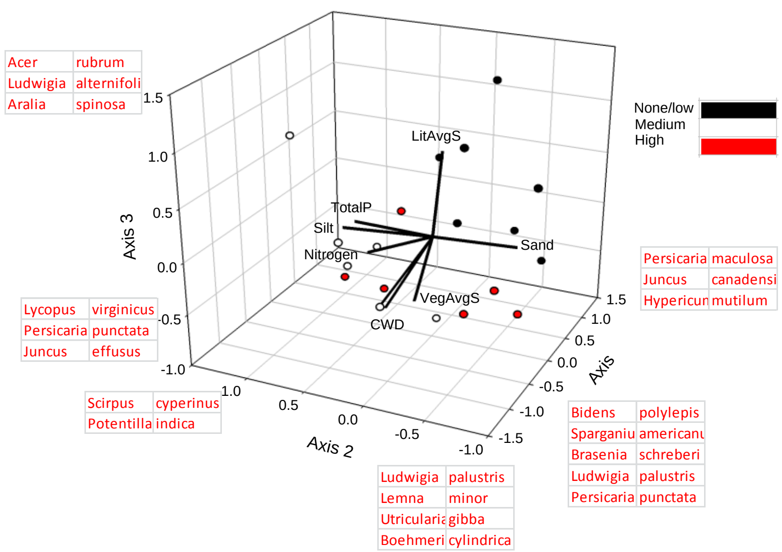


Figure 10. Non-metric multidimensional scaling (NMDS) distribution of wetland plant communities for transects specifically sampled for soil chemistry. Each distribution is a 3D depiction of plant community composition based on the average cover of each species along a particular transect. Each transect in the upper distribution is color coded by wetland and each transect in the lower distribution is categorized by coarse woody debris (CWD) amount. Influential plant species identified using the Pearson correlation coefficient, with an r of 0.20 or greater are reported on each axis.

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KIMBERLEY RUSSELL

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EDUCATION

August 2012 – May 2015 (tentative) Towson University
MS Environmental Science
Subjects included:

- Vascular Plant Taxonomy
- Landscape Ecology
- Advanced Statistics
- Community Ecology
- Environmental Chemistry
- Environmental Law
- Environmental Geology
- Ecosystems Ecology

Thesis. The effects of restoration technique and hydrologic regime on plant community distributions in restored geographically isolated wetlands. Field research involved vegetation and environmental sampling. Statistics used include Spearman correlations, Akaike Information Criterion, Multiple Repeated Permutation Procedures, and Non-metric Multidimensional Analysis. Project has allowed me to develop theoretical and practical research skills including proposal defense, determining appropriate field methodology, conducting research, analyzing and interpreting results, report writing, and oral presentations.

August 2008 – May 2012 BS Biology, Salisbury University

August 2008 – May 2012 BS Environmental Science, University of Maryland Eastern Shore

EMPLOYMENT AND EXPERIENCE

June 2014 – Present, Environmental Scientist, McCormick Taylor Inc., Baltimore, MD

Responsibilities:

- Conduct geomorphic assessments, hydrology and hydraulic analysis, and concept restoration designs within degraded stream reaches
- Conduct wetland delineations within specified areas
- Produce stream concept design and wetland delineation reports

August 2012 – May 2014, Teaching Assistant, Towson University, MD

Responsibilities:

- Teach basic biology lab to non-major undergraduate students

September 2012 – December 2012, Invasive Species Research, Towson University, MD

Responsibilities:

- Conducted a series of field and laboratory experiments that focused on the seed production and dispersal of wavyleaf basketgrass
- Supervised and conducted experiments with a team of undergraduate students (about 6)

September 2010 – May 2012, Wicomico County Creekwaters, Salisbury University, MD

Repsonsibilities:

- Test water that was collected from all creeks adjacent to the Wicomico River for pH, salinity, nitrates, and phosphates
- Create annual reports on results using ArcGIS mapping
- Present scientific research conferences

July 2007 – April 2008, Conservation Planner, Calvert County Soil Conservation, Prince Frederick, MD

Responsibilities:

- Make conservation plans for local community farmers using ArcGIS
- Take Highly Erodible Land determinations
- Survey areas for grassed waterways

QUALIFICATIONS AND CERTIFICATIONS

- Plant identification (on site/taxonomic keys),
- ArcGIS proficient
- Microsoft Office programs

AWARDS

- Graduate Student Association Grant (Towson University, Fall 2012 and Fall 2013)
- Biology Faculty Award (Salisbury University, Spring 2012)