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Agricultural wetland restorations on the USA Atlantic Coastal Plain achieve diverse native wetland plant communities but differ from natural wetlands

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ABSTRACT

Wetland restoration is globally important for offsetting effects of wetland loss and degradation but is not consistently successful. Vegetation studies provide insight into the effectiveness of restoring wetland ecosystem functions. We compared plant community composition in 47 non-tidal wetlands under different management (natural, restored, and former wetlands that had been converted to cropland) in the Atlantic Coastal Plain of the USA. As expected, drained cropland sites were dominated by conventional upland row crops, had low species richness and evenness, and were highly disturbed. Plant communities in restored sites were more like natural sites based on the percentage of species that were native and hydrophytic, plant community evenness, and floristic quality. However, natural sites were forested, while restored and drained cropland sites were primarily herbaceous. Restored sites continued to be impacted by anthropogenic disturbance compared to natural sites. Our findings demonstrate that restored wetlands in agricultural settings can develop diverse native wetland plant communities within a decade but they remain very different from natural wetlands, raising questions about restoration goals, ecosystem service tradeoffs, and our ability to restore wetlands to ecological conditions found in reference sites.

1. Introduction

The wetlands of the world provide more ecosystem services per area than any other habitat type (Costanza et al., 1997; Dodds et al., 2008). They store and clean water, sequester carbon, provide habitat for diverse and often rare plants and animals, and are popular recreation spots (Dodds et al., 2008; Hefting et al., 2013; Hubbard and Linder 1986; Millennium Ecosystem Assessment 2005; Ullah and Faulkner 2005). The loss of ecosystem services when wetlands are degraded or converted to other land use types is well documented, as are global rates of wetland loss that range from 30 to 90% by region (Dahl et al., 1990, 2011; Junk et al., 2012; Zedler and Kercher, 2005).

In order to slow and reverse the rate of wetland loss and benefit from the services they provide, the United States of America (USA) and many other nations have implemented programs to protect and restore wetlands. The USA Department of Agriculture (USDA), for example, provides financial and technical assistance to landowners to help protect, restore, and enhance wetlands, primarily through two voluntary initiatives: the Wetland Reserve Program (WRP) and the Conservation Reserve Program (CRP)–Wetland Initiative. Stated objectives of WRP and CRP-Wetlands Initiative include protecting wetlands; providing habitat for migratory birds and other wetland-dependent fauna and flora; protecting and improving water quality by trapping sediment and removing nutrients; attenuating floodwater; recharging ground water; protecting and improving aesthetics of open spaces; and contributing to education and scientific knowledge. According to the technical guidelines for wetland restoration under these programs, ecosystem services are provided by returning the "soil, hydrology, vegetation and habitat conditions of the wetland that previously existed on the site to the extent

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practicable". These conditions may be determined by historic documentation or through the use of a reference site (USDA NRCS Manual Title 440 Wetland Reserve Program, 2010; USDA NRCS Practice Standard Code 657, 2010).

Conceptually, restoration is the process of returning an ecosystem to a pre-anthropogenic-disturbance state. In practice, specific features or services are targeted for restoration rather than attempting a complete ecosystem restoration. Wetland restoration is a complicated process, in part, because wetlands are regionally distinct and the actions required to restore them to the functional equivalency of natural wetlands have been shown to be difficult to prescribe broadly (Zedler and Callaway, 1999). In many cases, restoration efforts have failed to return the biological and biochemical features to levels found in natural wetlands even after many decades (Benayas et al., 2009; Moreno-Mateos et al., 2012). Restored wetlands also tend to differ physically from their original condition. In the USA, for example, most non-tidal wetland restorations have resulted in the formation of ponds with a fringe of emergent marsh, regardless of what type of wetland they were historically. As a result, few restored wetlands match reference conditions (Cole and Shafer 2002; De Steven et al., 2010; Kentula et al., 1992).

In a typical year, all of the approximately \$500 million WRP budget and more than 11% of the \$1.8 billion CRP budget are spent on wetland restorations (American Planning Association, 2010; personal communication, Alexander Barbarika, Farm Service Agency). The return on these investments can be difficult to determine due to the complexity and cost of measuring ecosystem functions. The USDA's Natural Resources Conservation Service (NRCS) has implemented a national project to assess the effectiveness of conservation practices and programs through the Conservation Effects Assessment Project (CEAP). It is under CEAP that the research in this paper was conducted.

Due to the difficulty and expense of measuring multiple ecosystem functions as metrics of restoration success, rapid field assessment methods have been developed to facilitate quantification of biological indicators of ecosystem integrity (Fennessy et al., 1998, 2004; Lopez and Fennessy, 2002). Karr and Dudley (1981) defined ecosystem integrity as "the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having species composition, diversity, and functional organization comparable to that of natural habitats of the region". Rapid field assessments are used to describe overall ecosystem condition, suggest probable causes of poor conditions, identify human activities that contribute to degradation, monitor wetland restoration trajectories, and set and assess measurable goals (Cronk and Fennessy, 2001; Galatowitsch et al., 1999).

Plants are one of the easiest and most frequently used indicators for assessing the progress of a wetland restoration (Mitsch and Wilson, 1996). They are adapted to natural variations in conditions and can reflect current as well as historic conditions (Bedford 1999; Cronk and Fennessy, 2001; Wilcox et al., 2002). Plant communities also respond to human disturbance in predictable ways. For example, the proportion of weedy species tends to increase with human disturbance and, given extreme disturbance, plants tend to decrease in size of individuals, cover, and lifespan (Karr, 1993). Advantages of using plants as biological indicators include: they are present in most wetland ecosystems; they are relatively easy to identify; sampling methods are well established; and their relative immobility creates a direct link between onsite environmental conditions and plant community characteristics (Cronk and Fennessy, 2001). Because of these traits, plant communities provide a good way to compare wetland conditions under different types of management.

We compared plant communities in two hydrogeomorphic wetland classes (depression and flat wetlands; Brinson, 1993; Brooks et al., 2011) under different management practices on the Atlantic Coastal Plain of the USA. Our goals were to (1) compare plant communities in restored wetlands and natural wetlands as well as in drained croplands that were previously wetlands, and (2) determine the degree to which each of the habitat types were impacted by human disturbance. Specifically, we used plant species composition and indices of diversity, floristic quality, and anthropogenic disturbance to compare management practices. We expected to find differences in plant communities between the habitat types due to their land use history and because the group of restored sites that were chosen were only 3–11 years in age. Even though the restored sites were relatively young, we sought to test the hypothesis that restored ecosystems are on a trajectory to having high ecosystem integrity, based upon the presence of seedlings and saplings of species found in natural sites.

2. Methods

2.1. Study sites

Forty-seven sites were selected for comparison in the USA Atlantic Coastal Plain regions of Delaware, Maryland, Virginia, and North Carolina (Fig. 1). The sites consisted of 14 "natural" wetlands, 16 "drained cropland" sites, and 17 "restored" wetland sites. Natural and drained cropland sites were identified using aerial photography and digital elevation models. They were selected to serve as references and controls for restored sites by minimizing natural differences (i.e., geomorphology, soil, and geographic proximity) and maximizing land use history differences. Natural sites were relatively undisturbed shallow wetlands characterized as either depressions or flats as described by Brinson's hydrogeomorphic classes (1993). They ranged in size from approximately 0.04 to 4.01 ha (mean = 1.58, SD = 1.6; estimates calculated via remote mapping of 7 out of 14 natural sites; calculations were difficult due to canopy cover and in some cases flat terrain). Depressions and flats are seasonally flooded and only occasionally connected to other wetlands via surface water. The drained croplands were once natural depression or flat wetlands, but had been drained hundreds of years prior to restoration for agricultural use. The restored wetlands were drained croplands that had been restored to depression wetlands. They ranged in age from 3 to 11 years since restoration and in size from 0.12 to 1.13 ha (mean = 0.53, SD = 0.38). As part of the restoration process, hydrology was restored either by plugging ditches or by excavating and compacting cropland to create shallow perched water table depressions often with water retention berms. Hummocks or islands were installed in the depressions of most restored sites in order to create within-wetland microtopographic diversity. Some of the restored sites had been planted with trees and most were planted with upland grasses on berms and in buffer areas.

2.2. Plant community survey

Plant community surveys were conducted once in each of the 47 sites from late June through September 2011 at the peak of the growing season to minimize differences due to time of year. The areas sampled in natural and restored wetlands were within the wetland boundary as roughly delineated by the plant community shift from wetland to upland plants. Ponded areas (i.e., standing water) without vegetation were not sampled. Drained cropland sites were sampled within approximately 25 m of the center of the wettest drained area. Given adequate area, three 10 m × 10 m quadrats (adapted from Peet et al., 1998) were randomly selected per plant community at each site. Plant communities within each wetland were visually

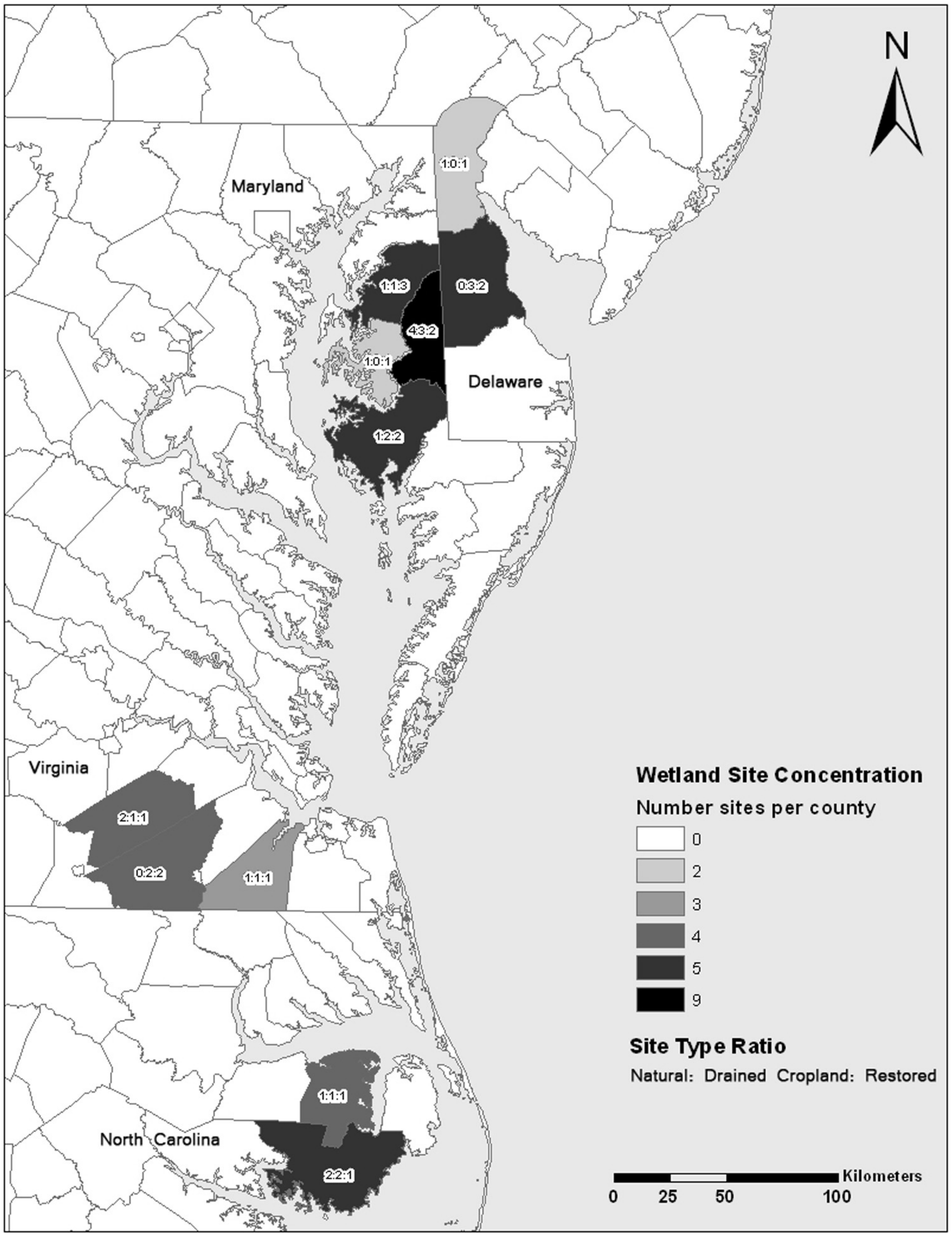

Fig. 1. Map of study sites. Map of study sites by county in Delaware, Virginia, Maryland, and North Carolina on the eastern coast of the United States of America.

determined based on the relative cover of dominant species (e.g., *Phragmites australis* would visibly dominate one plant community and *Xanthium strumarium* another). Where space allowed, quadrat locations were selected using a randomly generated compass point and number of paces from the center of the plant community. If space was limited, quadrats were placed so that they were completely contained within the plant community, which sometimes meant changing the shape of the 100 m^2 sampling quadrat (Peet et al., 1998). When space was very limited, quadrats were placed so as not to be overlapping, which sometimes resulted in fewer than three quadrats per plant community. We sampled all plant communities within each wetland boundary and surveyed the site outside of the quadrats to capture species that had not been sampled in the quadrats. In order to ensure adequate sampling, quantitative cover data for all dominant plants and 90% or more of the species in each site were captured in the quadrats, the latter based on the surveys of species that were at each site but not encountered in the plots.

Each species within a 100 m^2 quadrat was assigned a percent cover class (trace, 0–1, 1–2, 2–5, 5–10, 10–25, 25–50, 50–75, 75–95, or >95; class midpoints were used in data analysis) (Peet et al., 1998). Cover was used rather than number of individuals because it was more equitable when comparing tree and herb species. Any plants that could not be identified to species level in the field were collected, pressed, dried, and later keyed out using Brown and Brown (1972, 1984); Gleason and Cronquest (1991); Radford et al. (1968). Plant nomenclature is according to the USDA Plants Database (USDA NRCS, 2012). Approximately 23% of plant observations could not be identified to species due to a lack of flowering or fruiting material and were not included in calculations that required species level identification.

2.3. Common and dominant species

Common plant species were defined as those that occurred within quadrats at 40% or more of natural, restored, or drained cropland sites. Dominant plant species within each site type were defined as those that had cover of 20% or higher in the sites in which they were found. Percent cover of a species and percent cover of woody and herbaceous species at each site were calculated as the average of the species' cover across all quadrats at the site, with 0% cover assigned in quadrats where the species were not found.

2.4. Indices

Natural, restored, and drained croplands were compared using 9 vegetation indices commonly used to determine differences in wetland condition. The indices (described in Table 1) were calculated from the quadrat cover data and the presence of species found inside and outside the quadrats. The mean of each index was averaged for each type of site.

2.5. Statistical analysis

Statistical comparisons of natural, restored, and drained cropland sites were made using analysis of variance (ANOVA) performed using the MIXED procedure in SAS version 9.2 (SAS Institute, Cary, NC). Significance was assigned for $p < 0.05$. Arithmetic mean and standard error were calculated using MEANS procedure in SAS. Regressions were calculated using the REG procedure in SAS and SigmaPlot (Systat Software, San Jose, CA) comparing (1) the Anthropogenic Activity Index (AAI) to the Floristic Quality Assessment Index (FQAI), (2) AAI to Floristic Assessment Quotient for Wetlands (FAQWet) scores, and (3) time since restoration to percent cover by woody species in restored sites to determine if succession was taking place in restored sites.

Variation in plant communities between sites and their relation to metrics of diversity, quality, and disturbance were also examined using non-metric multidimensional scaling (NMS) analysis. Sørenson distance measures were used and Beal's smoothing was applied to vegetation data to achieve a final stress value near 10 (McCune and Grace, 2002). Significant difference between wetland types was tested using multi-response permutation procedures (MRPP). Both NMS and MRPP analyses were conducted using PC-ORD v. 6 (MjM Software, Gleneden, OR). Joint plots were prepared to visualize site scores relative to vectors of plant community and disturbance metrics.

3. Results

3.1. Plant community composition

A total of 204 species were observed across the three site types with 71 species found in natural sites, 134 in restored sites, and 34 in drained cropland sites. Four species (*Hypericum mutilum*, *Phytolacca americana*, *Diospyros virginiana*, and *Liquidambar styraciflua*) were found at all three types of sites. There was no overlap in dominant species between site types and little overlap in common species (Fig. 2). Woody species accounted for more than 70% of the cover in the natural sites, typically from *L. styraciflua*, *Acer rubrum*, and *Nyssa biflora* (Fig. 2A). These species were also found in a majority of natural sites and shaded an understory shrub and small tree stratum, which contained species such as *Clethra alnifolia, Smilax* species, *Eubotrys racemosa, Magnolia virginiana,* and various *Vaccinium* species.

By contrast, only 10% of cover in restored sites was from woody species and there was more variation in plant community composition between sites. While woody species such as *L. styraciflua* and *A. rubrum* were found in 30–50% of restored sites, they averaged <1% cover. *Echinochloa crus-galli*, *X. strumarium*, *Scirpus purshianus*, *P. australis*, and *Mollugo verticillata* were frequently found in restored sties and tended to have relatively high cover (Fig. 2B). Only 7 herbaceous species were found in both restored and natural sites. Of those seven species, only *Scirpus cyperinus* and *Woodwardia virginica* were found in more than one site of each type. Drained cropland sites were dominated by conventional row crops of *Zea mays*, *Glycine max*, *Gossypium hirsutum*, or *Sorghum bicolor* (Fig. 2C).

Table 1
Summary of vegetation and disturbance indices used.

Index	Description
Shannon–Weiner evenness index	The Shannon–Weiner evenness index was calculated for each site based on the species richness and cover of the species found in quadrats. It is used as an index that combines diversity and evenness. It is calculated as [sum(AC $\times$ ln(TC)]/ln(*SR*); where AC is the average cover of each species found in quadrats at the site with 0% cover assigned in quadrats where the species was not found; TC is the sum of all AC values found per site; and *S* is number of species found in quadrats at the site (Gurevich et al., 2006).
Species richness	The total number of species found in each site.
Wetness coefficient (WC)	Value assigned to plant species between −5 (always occurs in non-wetland upland areas) and +5 (always occurs in wetlands). Based on regional U.S. Fish and Wildlife Service Wetland Indicator Status (Ervin et al. 2006; Lichvar and Kartesz, 2009).
Coefficient of conservatism (CC)	Value assigned to plant species between 0 (non-native or weedy species that tolerate disturbance and are found in wide variety of conditions) and 10 (native species that are only found in specific undisturbed conditions) (Chamberlain and Ingram, 2012).
Proportion of woody species	The percent of species found in each site characterized as woody (as opposed to being an herb or vine) (USDA Plants Database).
Proportion of native species	The percent of plants identified to species found in each site that were native (USDA Plants Database).
Floristic Quality Assessment Index (FQAI)	Site wide index based on native species richness and coefficients of conservatism. A low score indicates low native species richness and/or species that tolerate a wide range of conditions and disturbance. A high score indicates high native species richness and plants that are only found under specific conditions and do not tolerate disturbance. Calculated as FQAI = $R/\sqrt{N}$; where *R* is the sum of the coefficients of conservatism for all species found at a site and *N* is the number of native plants identified to species in each site (developed from Andreas et al., 1995; Ervin et al., 2006; Lopez and Fennessy, 2002).
Floristic Assessment Quotient for Wetlands (FAQWet)	Site wide index based on species richness, species nativeness, and whether the species are more commonly found in wetland or upland areas. A low score indicates low native species richness and/or non-wetland species. A high score indicates high native species richness and plants that are almost always found in wetlands. FAQWet has the advantage of using wetness coefficients, which have been developed for all regions of the USA, rather than coefficients of conservatism, which have not. The other way that FQAI and FAQWet differ is that the FAQWet equation places a heavier weight on non-native plant species. Both indices are influenced by species richness. The FAQWet score for each site was calculated as FAQWet = (SumWC)/($\sqrt{S}$)(N/S); where WC is the wetness coefficient value assigned to each species, *S* is the species richness per site, and *N* is the number of native species at each site (Ervin et al., 2006; Herman et al., 1997; Reed, 1988).
Anthropogenic Activity Index (AAI)	AAI is an index for qualitatively assessing human disturbance based on observations during site visits. The AAI rates wetlands on a scale of 0–3 for five conditions: land use intensity in a 500 m buffer; intactness and effectiveness of a 50 m buffer; hydrologic alteration; habitat alteration; and habitat quality and microhabitat heterogeneity (Ervin et al., 2006).

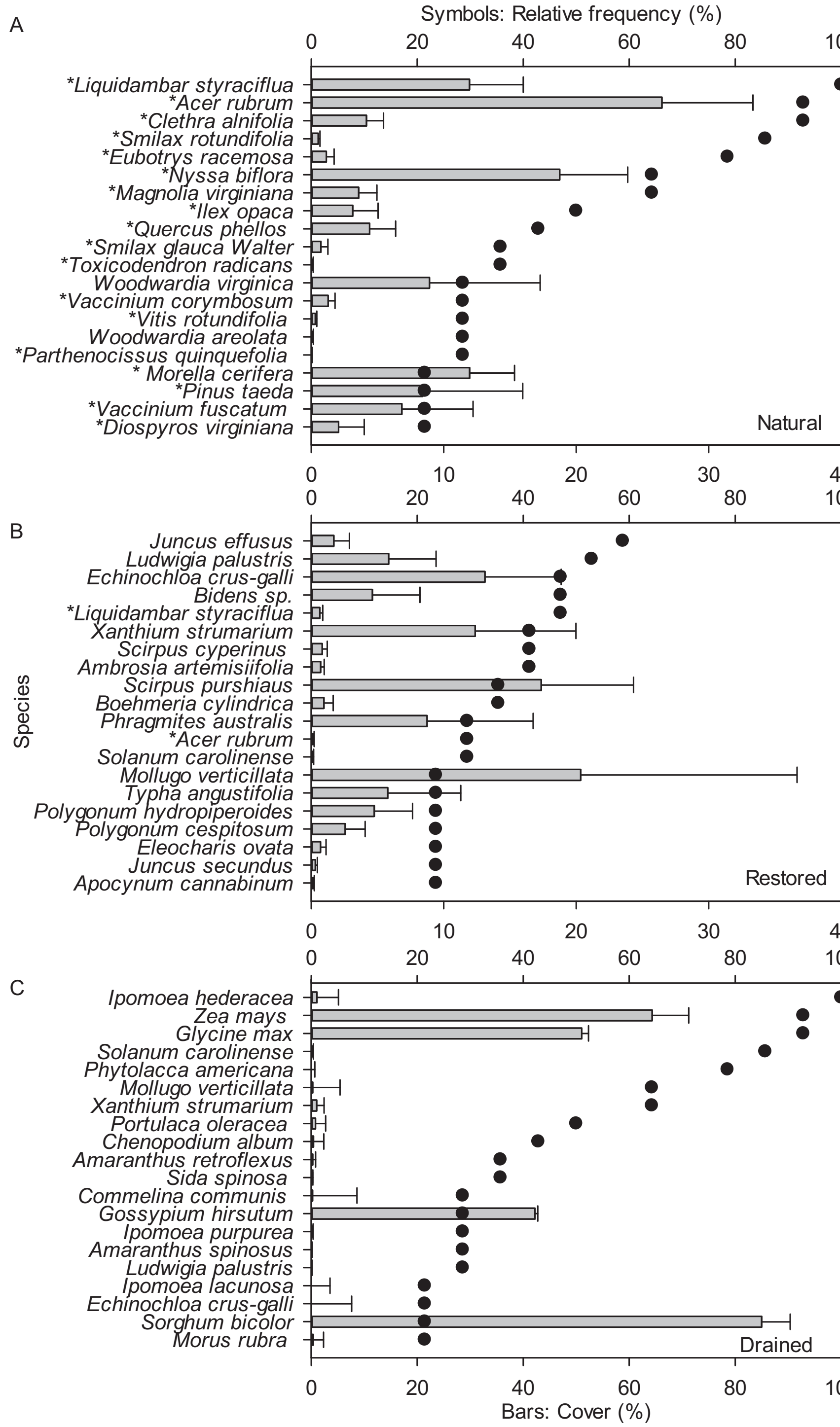

Fig. 2. Common and frequent species. Average cover and frequency of the most common plant species found in (A) natural, (B) restored, and (C) drained cropland depressional wetlands in the US mid-Atlantic Coastal Plain. The plant species listed are the 20 most frequent species with the highest average percent cover in the sites in which they were found. Plotted cover values are mean + 1SE. * woody species. Dots represent frequency and bars represent average percent cover.

3.2. Index variation between site types

For most indices, restored sites were more similar to natural sites compared to drained croplands (Fig. 3). Natural and restored sites had similar proportions of native species and evenness (Fig. 3C and D). Natural sites, however, had a significantly higher proportion of woody species, higher FQAI scores, and lower anthropogenic disturbance (Fig. 3B, G, and I). Restored sites, compared to natural sites, had significantly higher species richness and the plant species were more wetland specific (Fig. 3A and E).

There was a stronger negative correlation between AAI and FQAI than AAI and FAQWet scores, but both were significant ($R^2 = 0.65$ and 0.30, respectively, $p < 0.001$; Fig. 4A and B). The correlation between percent cover by woody species and number of years since restoration in restored sites was not significant ($R^2 = 0.24$, $p = 0.17$; Fig. 5).

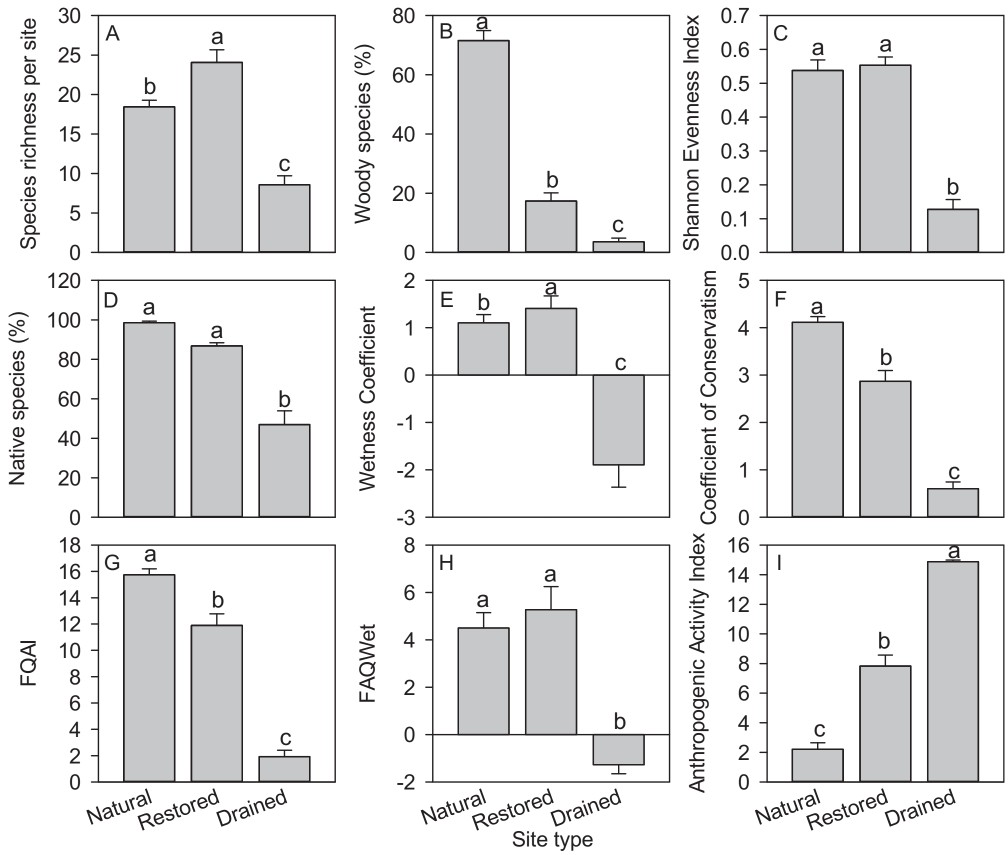

Fig. 3. Indices. Plant community metrics for natural, restored, and drained cropland ("Drained") depressional wetlands of the US mid-Atlantic Coastal Plain. Plotted values are mean + 1SE; means with different letters represent statistically significant differences (Tukey, $p < 0.05$). (A) Species richness per site (number of species) ($F_{2,44} = 38.7$, $p < 0.0001$); (B) percent of species that were woody ($F_{2,44} = 183.8$, $p < 0.0001$); (C) Shannon index of species evenness ($F_{2,44} = 77.15$, $p < 0.0001$); (D) percent of species that were native to the USA ($F_{2,44} = 35.66$, $p < 0.0001$); (E) wetness coefficient (ranges from −5 for species that are always found in wetlands to 5 for species that are only found in wetlands; $F_{2,44} = 35.66$, $p < 0.0001$); (F) coefficient of conservatism (ranges from 0 to 10, with 10 assigned to species least tolerant of disturbed conditions; $F_{2,44} = 97.50$, $p < 0.0001$); (G) Floristic Quality Assessment Index (FQAI) ($F_{2,44} = 115.99$, $p < 0.0001$; (H) Floristic Assessment Quotient for Wetlands Index (FAQWet) ($F_{2,44} = 24.65$, $p < 0.0001$); (I) Anthropogenic Activity Index ($F_{2,44} = 137.46$, $p < 0.0001$).

3.3. Community–index relationships

The NMS ordination indicated that plant communities in the three types of sites were significantly separated from each other (MRPP A-statistic = 0.35, $p < 0.0001$; Fig. 6). Axes 1 and 2 cumulatively explained 87.6% of the variation in species composition. Position and length of disturbance and community metric vectors indicate that the natural sites tended to be undisturbed (low AAI) and dominated by woody species and native plant communities of high floristic quality (high FQAI and coefficients of conservatism). Drained cropland sites were, in contrast, highly disturbed and dominated by non-native, herbaceous, and low-floristic-quality communities (i.e., crop monocultures). Restored sites tended to have more diverse plant communities than natural sites and were clearly dominated by wetland plant species. The restored sites, however, had lower floristic quality index values and had a lower proportion of native species compared to natural sites.

4. Discussion

4.1. Species richness and evenness

Despite the presence of a larger number of non-native and opportunistic species, restored sites had the highest species richness and evenness that matched natural sites, a finding consistent with studies of recently restored freshwater wetlands (Balcombe et al., 2005; Ficken and Menges 2013; Gutrich et al., 2009; Matthews et al., 2009). Higher species richness may be due to the mix of new hydrophytic species and the existing seed bank of weedy species that were present during the period of time that the sites were farmed. Although matching the species richness of reference sites is a common management goal, species identity must be considered to determine whether richness indicates ecological health or degradation. Species richness may be increased by the presence of weedy and invasive species, by removing nutrient limitations through the addition of pollution, or by creating more mico-habitats where upland species can colonize (Ehrenfeld, 2000; Ehrenfeld and Schneider, 1993).

4.2. Floristic quality

Floristic quality indices incorporate plant species identity as well as richness, allowing for meaningful comparisons between sites (Lopez and Fennessy, 2002). Although we recognize that coefficients of conservatism have not been developed for all regions of the USA and are not used in other countries, our findings suggest that the FQAI index is preferable to the FAQWet index for comparing floristic quality between restored and natural depressions. This is because FAQWet scores are based on native species richness and species tolerance for wetland conditions. Like species richness, a higher FAQWet score may indicate a more diverse and native wetland plant community or it may indicate highly altered

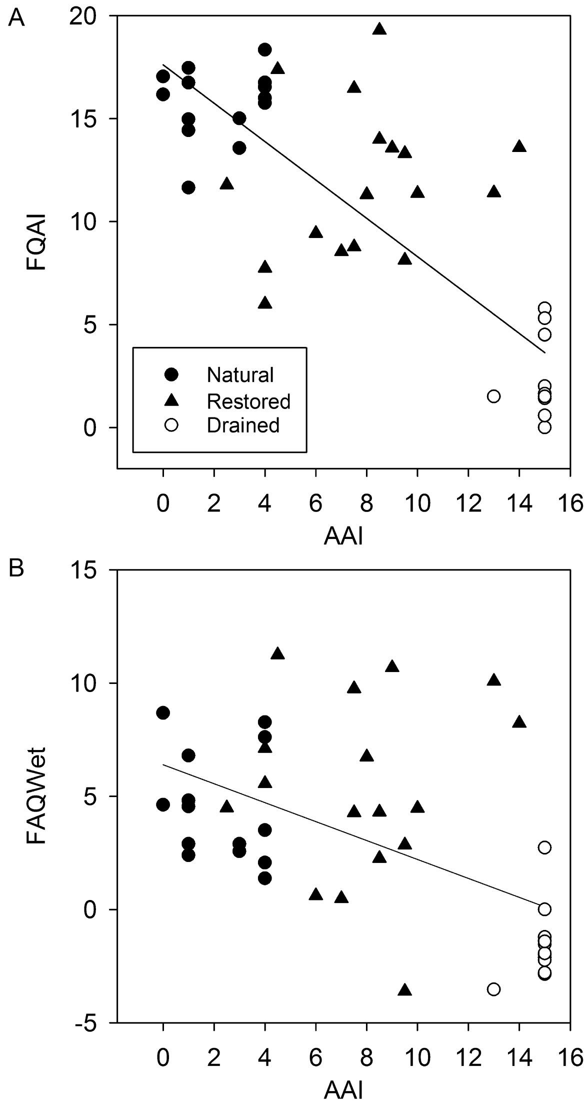

Fig. 4. Disturbance and floristic quality. Relationship between (A) Floristic Quality Assessment Index (FQAI) and Anthropogenic Activity Index (AAI) scores assigned to each site ($p < 0.001$, $R^2 = 0.65$) and (B) Floristic Assessment Quotient for Wetlands (FAQWet) and the AAI ($p < 0.001$, $R^2 = 0.30$).

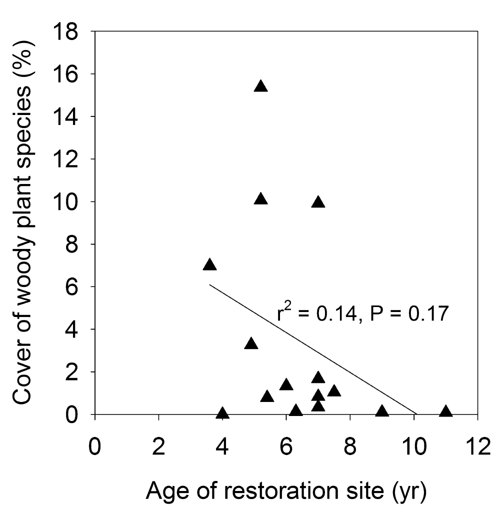

Fig. 5. Restoration age and succession. Relationship between the number of years since restoration and the percent cover of woody species in restored sites.

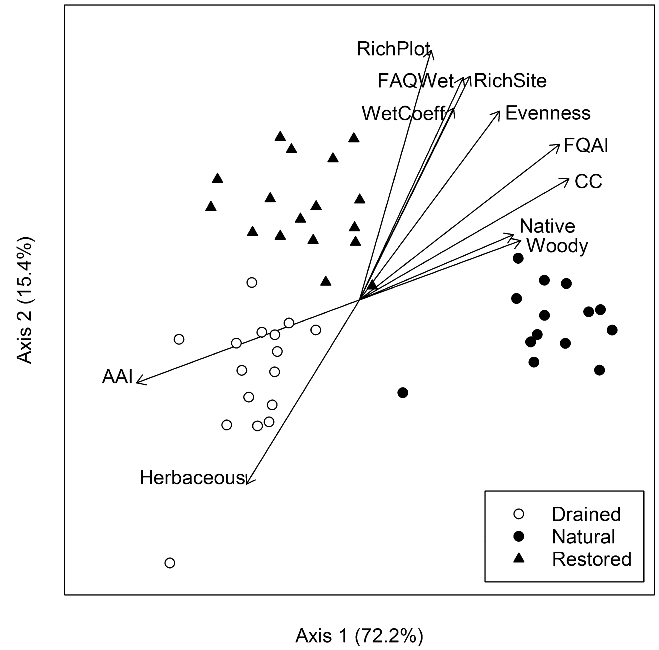

Fig. 6. Non-metric multidimensional scaling. Results of non-metric multidimensional scaling (NMS) ordination of plant communities of natural, restored, and drained cropland wetland sites. Plant communities of the three site types differed significantly (MRPP, $A = 0.35$, $p < 0.0001$). The direction and length of vectors reflect the relationship of plant communities to community and disturbance metrics (AAI = Anthropogenic Activity Index; CC = coefficient of conservatism; evenness = Shannon index of species evenness; FAQI = Floristic Quality Assessment Index; FAQWet = Floristic Assessment Quotient for Wetlands; herbaceous = absolute cover of herbaceous plant (%); native = proportion of species that were native (%); RichPlot = number of species per 100 m^2 plot; RichSite = number of species in the entire wetland site; WetCoeff = wetness coefficient; woody = absolute cover of woody plants (%).

hydrology, where the restored wetland is wetter than natural sites due to anthropogenic manipulation (Ervin et al., 2006), a condition we observed in our study sites.

Another potential problem with FAQWet is that if it was a good indicator of ecosystem integrity, we would expect a correlation between the index scores and the level of human disturbance at a site as indicated by AAI. Similar to Tietjen and Ervin (2007), we found that FAQWet was not significantly correlated with AAI, indicating that FAQWet is not a good indicator of ecosystem integrity. By contrast, FQAI has been found by our study and others to be strongly correlated with anthropogenic disturbances such as agricultural land use, fragmented vegetation buffers, altered hydrology, and long distances between wetlands (Lopez and Fennessy, 2002). This is likely because the coefficients of conservatism used in FQAI are based on how specific the species' habitat requirements are and its tolerance of disturbance rather than whether or not the species is wetland specific as is the case for FAQWet. Thus, FQAI may inherently be a better measure of ecosystem integrity.

4.3. Anthropogenic disturbance

Restored wetlands continue to have more signs of historic and ongoing anthropogenic disturbance than natural sites as evidenced by higher AAI scores. Historic impacts include large areas of open water that resulted from restoration methods and proximity to roads and agricultural fields. Ongoing disturbances observed at restored sites included active mowing and in some cases digging. Ecosystem integrity is thought to be inversely related to human disturbance because disturbances can upset ecosystem balance by changing nutrient cycling, photosynthesis, hydrology, competition, predation, and more (De Leo and Levin, 1997; Lopez and Fennessy, 2002).

4.4. Differences in plant community composition

Given enough time, the structure and function of restored wetlands should become more like natural sites, although this process may take 100 years or more (Mitsch and Wilson, 1996; Moreno-Mateos and Comin, 2010). The restored wetlands sampled in this study were comparatively young (i.e., sampled 3–11 years post restoration), but we anticipated that the oldest would have increased cover and density by the woody species found in nearby reference sites. This assumption was based on a study of restored wetlands that were similar to ours as well as predictions of succession based on the growth rates of common wetland tree species (De Steven et al., 2006, 2010; USDA Plants Database). De Steven et al. found that the cover of woody species in restored wetlands averaged 40% after 5 years and that restored sites had 53% of species in common with forested reference sites. In addition, the common tree species found in the natural sites we sampled can grow 0.6 m or more per year (USDA Plants Database 2014). In contrast, we found that restored sites had very few of the woody species found in natural sites and that there was no correlation between the time since restoration and percent cover of woody species. Several explanations could account for the differences between our predictions and our findings.

Since few of the restored sites were planted with woody species and the majority of the trees that were planted were not the same species found in the natural sites (e.g., *Platanus occidentalis* and *Taxodium distichum* were not found in natural sites), seed banks and seed dispersal remain as the major sources of propagules in restored wetlands. However, the seed banks of farm fields are dominated by herbaceous species and tend not to contain woody species (De Steven et al., 2006; Middleton, 2003). Even if viable seeds of native species were present in the seed bank at the time the sites were restored, many of them would have been removed during restoration, as a common practice is to remove topsoil to create a depressions and use the excavated material to create berms, thus, reducing the potential importance of the seed bank in restoring native species diversity.

Because restored wetlands in agricultural areas are often surrounded by farm fields rather than forested wetlands, dispersal limitation of the propagules of woody species may explain their lower abundance in restored wetlands (Herault and Thoen, 2009; Kettenring and Galatowitsch, 2011; Middleton, 1999, 2003). Depressions and flats are especially isolated because they rarely receive overland flow of water from other wetlands, which leaves wind transport as the major remaining source of woody propagules. Dispersal of seeds from woody species declines exponentially from forest edges into clearings (Greene and Johnson, 1996). Clewell and Lea (1990) suggested that wetland restoration sites within two tree heights of a forest composed of mature trees would have the most successful natural regeneration of early colonizers like *L. styraciflua* and *A. rubrum*. Thus, the availability of propagules in the restored sites we studied was potentially limited by prior land use, restoration methods, and isolation from propagule sources. This suggests that more-active management and planting may be required for plant communities in restored wetlands to more closely resemble natural wetlands.

Hydrologic differences between restored and natural sites may be another explanation for differences in plant communities. Long-term changes in the landscape, like ditching and over-extraction of ground water, may prevent the restoration of hydrology to pre-disturbance conditions (Zheng et al., 1988). As described above, it is common to remove topsoil during restoration and compact subsoil with equipment in order to ensure that the depressions hold water. This practice creates ponded conditions and likely alters surface–ground water connectivity, leading to longer and deeper hydroperiods. Field observations support this hypothesis, as many restored sites had ponded water in the summer when all but the largest natural sites were dry.

Hydroperiod plays an important role in determining species composition. Herbaceous wetland species are only abundant in the natural depression wetlands in our study area when they are large enough to have a deeper central zone where water depths preclude the establishment of woody species (Tyndall et al., 1990; Verhoeven et al., 1994). Forested wetlands exist only where the hydroperiod is long and deep enough to exclude upland species, but not so wet as to kill trees (Lugo, 1990). In one study, higher cover by woody plant species was correlated with shorter hydroperiods 5 years post-restoration (De Steven et al., 2010). Thus, restored sites may have lacked woody species for a variety of reasons (i.e., seed availability, longer hydroperiods with deeper water). If this is the situation, we would expect that woody species would first colonize restored wetlands near the upland–wetland boundary where flooding depth is less and soils are more likely to be exposed during dry periods. This situation has occurred in a separate set of restored wetlands in the Atlantic Coastal Plain that were initially sampled in the mid-1990s and again in 2011 (D. Whigham and L. McFarland, personal observation, and unpublished data).

One final explanation as to why restored sites remained largely dominated by herbaceous species is continued anthropogenic disturbance. Many sites were regularly mowed and some had evidence of recent disturbance by heavy machinery. Regular mowing and soil disturbance are effective methods for preventing the establishment of woody species.

4.5. Restoration and management goals

Most of the restored wetlands sampled were restored under either Wetland Reserve Program (WRP) or Conservation Reserve Program (CRP), both of which use the USDA Natural Resources Conservation Service (NRCS) Practice Standard Code 657. The practice standard defines wetland restoration as the return of vegetation, soils, and hydrology to pre-human disturbance conditions to the extent possible so as to "restore wetland function, value, habitat, and diversity". NRCS Practice Standard Code 657 allows for no more than 30% of the restored wetland area to be a plant community different from what would have existed pre-disturbance, but no timeframe is given as to when this should be reached. None of the restored sites sampled for this study met the desired condition. However, there is some ambiguity to these guidelines because an herbaceous wetland community is considered to be an acceptable precursor to a forested wetland and can be maintained as a herbaceous community for CRP restorations, but not under WRP (personal communication, Steve Strano, NRCS). It is possible that these allowances are made in part due to landowner preference, since the WRP and CRP are voluntary and will tailor implementation to meet landowner expectations within USDA guidelines. For a few reasons, landowners may not be interested in having a forested wetland on their property: (1) it may be considered less aesthetically pleasing and block the view (Clewell and Lea, 1990) and (2) forested wetland habitat may not attract waterfowl for hunting or bird watching as well as an open pond with herb-dominated edges. Thus, it is likely that some of the restored sites were never designed to transition into forested wetlands that are similar to the natural wetlands. For this reason, it may be beneficial to re-examine the definition and goals of USDA wetland restorations to ensure that restoration techniques are tailored to unambiguous, measurable, and realistic restoration goals that have a stated timeframe for being reached.

Although restored sites may not provide the same functions as natural sites due to their structural differences, they are still likely to provide many of the services associated with wetlands and meet broad CRP and WRP goals. As emergent wetlands dominated by herbs, grasses, and sedges, the majority of restored sites provide food and habitat for migratory birds and waterfowl, albeit different species than a forested wetland. The restored sites also support diverse native wetland plant communities. The hydrogeomorphic status (i.e., depressional class) of the restored sites also assures improved water quality via sedimentation and denitrification, but restored sites are less likely to recharge ground water due to their perched water tables (Ator et al., 2013; Bruland et al., 2003; Tanner et al., 2005). The depressional wetlands, through water storage and reduction of flow velocity, would also reduce downstream flooding. Due to their proximity to houses and agricultural fields, restored sites may even provide more of the services valued by landowners than do natural sites.

5. Conclusions

Restored depressional wetlands in agricultural areas of the USA Atlantic Coastal Plain have developed diverse native wetland plant communities that are likely to provide many of the broad functional benefits targeted by the federal programs that supported their implementation. However, we hypothesized that restored sites, especially those that had been restored for a decade, would show signs of woody species establishment, demonstrating that their plant communities were on a trajectory toward that of natural sites. This hypothesis was not supported; after 3–11 years, the tree and shrub species that occurred in natural wetlands were not represented in restored wetlands. Thus, without a change in management, it is unlikely that plant communities of restored sites will converge to be similar to natural wetlands. Ambiguity in the goals and guidelines in the federal restoration documentation make it difficult to determine whether or not these projects were intended to return to pre-anthropogenic disturbance conditions.

It is important to recognize that although these restorations provide important ecosystem services, wetlands of different types inherently provide different services. We need to consider the landscape scale effect of this shift from forested wetland to cropland to herbaceous wetlands on resultant services and then clearly define our restoration goals and objectives based on the needs of a given area. The lack of established goals and success criteria is a well-documented problem in wetland restoration. In order to demonstrate that environmental restoration is a reliable tool, it is imperative that outcomes match stated goals. The goals and objectives of a wetland restoration must be site specific, measurable, and have a stated timeframe for when the goals are expected to be achieved. These goals and objectives should be documented and used to inform the entire restoration process from selecting restoration methods to project evaluation.

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