Managed disturbance enhances biodiversity of restored wetlands in the agricultural Midwest

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A B S T R A C T

We compared plant diversity in four restored and four natural wetlands in the glaciated interior plains to evaluate how quickly plant biodiversity develops following cessation of agriculture and reintroduction of wetland hydrology. After approximately ten years, restored wetlands had plot and site richness comparable to natural wetlands; 6.2 ± 0.5 species per plot and 33.8 ± 2.3 species per site for restored wetlands versus 5.5 ± 1.3 species per plot and 27 ± 6.4 species per site for natural wetlands. The natural wetlands, however, contained higher quality plant communities based on greater abundance of hydrophytic plant species (OBL, FACW, and FAC) and species with higher Coefficients of Conservatism (C of C). There was no difference in the Floristic Quality Assessment Index (FQAI) values for natural and restored wetlands. The comparable plot and site diversity of restored wetlands is attributed to the use of management tools (such as seeding, prescribed burning, and herbicidal treatments) during restorations that enhance species richness and diversity and shorten the time required for the plant community of restored wetlands to converge with levels in natural wetlands. Further investigation of active management techniques is needed to better understand how they can contribute to the restoration of a diverse assemblage of wetland vegetation.

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1. Introduction

The Glaciated Interior Plains (GIPs) stretch from Minnesota to Ohio, covering an eight state area within the Midwestern United States that encompasses a wide variety of ecosystem types, including wetlands. During the past 200 years, much of the GIP has been drained for agriculture, resulting in a 50–90% net wetland loss (Dahl, 1990) with concurrent loss of habitat and biodiversity. Increased agricultural land-use in the region has led to nutrient enrichment with further declines in plant species diversity, and the loss of rare and uncommon species (Bedford et al., 1999; Craft et al., 2007). These practices have led to a decline in essential wetland ecosystem services, including biodiversity support (Zedler, 2003).

Plant diversity influences ecosystem properties such as wetland productivity, decomposition rate, nutrient cycling (Engelhardt and Ritchie, 2001), and resistance and resilience to perturbations. Plant diversity has also been shown to extend to other trophic levels; it is positively related to mycorrhizal and insect diversity (Knops et al., 1999). Comparative studies have revealed that anthropogenic disturbance has led to species substitutions that alter macrophyte functionality by creating monocultures that may decrease the quality of ecosystem properties (Loreau et al., 2001).

Wetland or hydrophytic vegetation is macrophytic plant life that grows in water, soil or on a substrate that is periodically deficient in oxygen (Mitsch and Gosselink, 2007). Hydrophytic vegetation occurs in areas where the soils are inundated or saturated (hydric) permanently or of sufficient frequency and duration to exert a controlling influence on the species present (USACE, 2010). Elevation and the depth, duration, and frequency of inundation or saturation lead to the formation of distinct vegetative guilds in wetlands, wet-prairie, sedge meadow, shallow emergent, submerged aquatic, and floating annuals, that vary spatially and undergo cyclic changes over time if water level fluctuations occur (Galatowitz and Van Der Valk, 1996; De Steven et al., 2010). Although hydrology is the main factor controlling the presence and distribution of wetland vegetation, wetland size is also an important predictor of plant communities and especially species richness (Houlahan et al., 2006).

The Wetland Reserve Program (WRP) and Conservation Reserve Program (CRP) are legislative acts that were created with the intention of preserving wetland ecosystems and the environmental services they provide. The WRP and CRP are voluntary programs for agricultural landowners offered through the Natural Resource Conservation Service (NRCS) with the opportunity for them to establish
long-term conservation and wildlife practices and protection. Since 1990, these programs have assisted landowners in creating, restoring, and enhancing wetlands by providing technical and financial support (USDA, 2008). Approximately 180,000 acres of wetlands was contracted to be restored under these programs in 2009 alone (USDA, 2009).

One of the objectives of wetland restoration projects is to reproduce the structural and functional attributes of natural wetlands (Zedler et al., 2001). However, in the GIP, many projects consisted only of restoring hydrology to the area by either plugging drainage ditches or destroying drainage tile lines (Galatowitsch and Van Der Valk, 1996). In many cases, no attempts were made to incorporate native species, meaning that the establishment of hydrophytic vegetation relied on the ability of native species to naturally disperse to and recolonize the area (Seabloom and Van Der Valk, 2003). The probability of recolonization occurring through natural processes is low due to the loss of wetlands, a source of propagules, throughout the region (Detenbeck et al., 1999; De Steven et al., 2010). Several studies found that restored wetlands had lower proportions of plant species richness and hydrophytic vegetation than natural wetlands (Galatowitsch and Van Der Valk, 1996; Seabloom and Van Der Valk, 2003). These restorations also lacked the presence of vegetative guilds, particularly wet prairie and sedge meadow species, both in the aboveground plant community and in the seed bank (Seabloom and Van Der Valk, 2003).

Despite the widespread restoration of wetlands in the region, it is unclear whether current ecological knowledge is adequate for restoring natural macrophyte communities to areas that have been converted for agricultural uses (Seabloom and Van Der Valk, 2003). The purpose of this study is to determine if differences in plant communities exist between natural and managed WRP restored wetlands in Northwestern Indiana in the GIP. Our hypothesis is that natural wetlands, having ample time for development and seed dispersal, will have more diverse and higher quality macrophyte communities based on measurements of plot and site richness, the Floristic Quality Assessment Index, overall community quality, and percent of hydrophytic vegetation than the restored wetlands.

2. Methods

2.1. Site selection

Four natural and four restored wetlands were selected from the Kankakee River watershed in Newton County in northwest Indiana (Fig. 1). The natural wetlands were located in the Willow Slough Fishing and Wildlife Preserve and the restored wetlands were on a 7800 acre property owned by The Nature Conservancy (TNC), nine miles to the north. Wetlands were restored under the Wetlands Reserve Program (WRP) between 1999 and 2001 by completely filling in adjacent drainage ditches with sediment (C. O’Leary, pers. comm.). Since the restoration, the property has been actively managed with prescribed burns, seeding of over 350 species annually and herbicidal treatments to eradicate undesirable species in order to achieve a natural grassland mosaic, including marsh vegetation. Natural wetlands that best represented high quality wetland conditions (e.g. wetland hydrology, absence or low density of invasives) were selected as reference sites and included only herbaceous wetlands. All of the wetlands surveyed were classified as depressional areas and were of approximately 0.5 ha in size.
The most prevalent restored wetland soil type found was Conrad series (Mixed, Mesic Typic Psammaquept), a poorly drained soil that is frequently ponded for brief periods of time (Soil Survey Staff, 2010). The most common natural wetland soil type found was Newton series (Sandy, Mixed, Mesic Typic Humaquepts), a very poorly drained soil that is frequently ponded for long periods of time.

2.2. Macrophyte sampling and analysis

In August and September 2010, twenty plots 1 m² were randomly established at each wetland to determine plant diversity. All plant species within each plot were identified and the cover of each species was estimated using a standard cover-abundance scale and then given a ranking in accordance with the Canopy Cover Method (Daubenmire, 1959). This method used a seven point cover scale: (1) 0–2%; (2) 2–5%; (3) 5–25%; (4) 25–50%; (5) 50–75%; (6) 75–95%; (7) 95–100%. The total abundance of a species in a wetland was calculated by summing the cover rankings of a species in the twenty quadrats. The abundance of a species can range from 0 (completely absent) to 140 (a cover ranking of 7 in all 20 quadrats). Plant community richness was measured at the plot and site level by identifying the number of plant species within each plot (plot richness) and across each site (site richness). The United States Department of Agriculture (USDA) Plant Database was used to record the wetland indicator status and native standing of each species (Reed, 1988). The classification system divides vegetation into five species classes according to the frequency with which they are found in wetlands: Obligate Wetland (OBL), Facultative Wetland (FACW), Facultative (FAC), Facultative Upland (FACU), and Upland (UPL).

Following Andreas and Lichvar (1995), the Coefficient of Conservatism (C) for each species was used to calculate the Floristic Quality Assessment Index (FQAI). The C of C is a ten point scale: where (0) is non-native or opportunistic invasive; (1–3) are species tolerant to disturbance; (4–6) are moderately tolerant species; (7, 8) are intolerant species; (9, 10) are sensitive species with a narrow range of ecological conditions. The FQAI for each wetland was calculated as follows:

\[
\text{FQAI} = \frac{R}{N^{0.5}}
\]

\(R\) is the sum of C of Cs for all species found at each site and \(N\) is the number of native species (Andreas and Lichvar, 1995). These measurements rely on the concept of plant species conservatism, the theory that species can be ranked based on their relative fidelity to unaltered natural areas (Matthews et al., 2009).

2.3. Statistical analyses

Differences in plot richness, hydrophytic indicator status, C of C, and FQAI were compared using a one-way analysis of variance (ANOVA) based on natural and restored wetlands (SAS, 1996). All statistical analyses were conducted at a significance level of \(\alpha = 0.05\).

3. Results

There was no difference in plot and site richness between restored and natural wetlands (Fig. 2). Plot richness ranged between 2.3 and 7.7 species per quadrat and site richness ranged between 13 and 43 species per wetland. The Floristic Quality Assessment Index values also did not differ between natural and restored wetlands (Fig. 2). The FQAI values ranged from 11.9 to 20.9, with the minimum and maximum scores found in natural sites. Restored wetlands contained approximately fifteen percent more opportunistic and tolerant species, while natural wetlands contained significantly higher percentages of moderately tolerant species (Fig. 3a). Species composition of both natural and restored sites consisted of mostly tolerant (35–49%) and moderately tolerant (28–54%) species. The only sensitive species documented in this study, Muhlenbergia glomerata, was found on a restored wetland and accounted for 1.2 percent of the total vegetative cover at the site.

When expressed on a percent cover basis, restored wetlands also contained a significantly greater abundance of lower quality (opportunistic and tolerant) species whereas natural wetlands contained more high quality (moderately tolerant) species (Fig. 3b). Opportunistic and tolerant species accounted for approximately 65 percent of the total vegetative cover in restored wetlands as compared to 30 percent in natural wetlands. Moderately tolerant and intolerant species comprised approximately 70 percent of the total vegetative cover in natural wetlands and 35 percent in restored wetlands.

There was little overlap of dominant species between and within natural and restored wetlands. Calamagrostis canadensis was the most abundant species in Natural Sites 1 and 2, while Natural Site 3 was dominated by Boehmeria cylindrica and Natural Site 4 by Scirpus cyperinus. Schoenoplectus pungens was the most abundant species in Restored Site 1 and Scirpus cyperinus was the dominant species in Restored Site 2. Restored Site 3 was dominated by Leersia oryzoides and Restored Site 4 by Solidago altissima.

The percentage of obligate wetland species was significantly greater in natural wetlands, whereas restored wetlands contained significantly higher percentages of facultative upland species (Fig. 4a). Obligate species accounted for 62 percent of the total species found in natural wetlands and only 50 percent of the total species present in restored wetlands. Facultative wetland species were the second most abundant group, 33–37 percent. Facultative and facultative upland species comprised 4–5 and 0–7 percent, respectively, of the remaining species found in natural and restored sites. Upland species comprised less than one percent of the total species in both restored and natural wetlands.

When expressed on a percent cover basis, natural wetlands contained a significantly greater percentage of obligate wetland species, while restored wetlands contained significantly more facultative wetland, facultative, and facultative upland species (Fig. 4b). Obligate species accounted for approximately 67 percent.
of the total vegetative cover in natural wetlands as compared to 52 percent in restored wetlands. Facultative wetland species comprised approximately 36 percent of the total vegetative cover in restored wetlands and 30 percent in natural wetlands. Facultative, facultative upland, and upland species accounted for approximately 3 percent in natural wetlands and 12 percent in restored wetlands.

4. Discussion

After approximately ten years of reflooding, the vegetation of WRP restored wetlands had comparable plot and site richness to natural wetlands (Fig. 2). Species richness and diversity are often higher in restored freshwater wetlands than in natural reference wetlands (Magee et al., 1999; Wissinger et al., 2001; Meyer et al., 2008; Gutrich et al., 2009; Matthews et al., 2009), especially during the early years. Species diversity frequently increases in response to disturbance (Odum, 1985), including the disturbances associated with initial restoration activities. Anderson (2007) and Meyer et al. (2008) found that plant diversity often increases to a maximum several years after the disturbance and then declines to resemble the levels of natural systems.

However, indicators of species composition, such as the Floristic Quality Assessment Index, often do not reach levels equivalent to those of reference wetlands (Seabloom and Van Der Valk, 2003; Fennessy et al., 2004; Gutrich et al., 2009). This contrasts with the results of our study, in which FQAI values were similar in natural and restored wetlands (Fig. 2). Matthews et al. (2009) also found no difference using the FQAI in a study between natural and restored herbaceous wetlands in Illinois. The FQAI, while beneficial for distinguishing between high- and low-quality wetlands, can produce high values for recently restored wetlands. This occurrence is common in metrics that incorporate plant species richness (i.e. the number of species) because it is easily inflated by disturbance generated from the restoration process (Matthews et al., 2009).

The comparable diversity of restored and natural wetland plant communities, measured by plot and site richness, is attributed to management practices that enhance species richness and compress the amount of time required for the plant community to converge with levels in natural wetlands (Seabloom and Van Der Valk, 2003; Meyer et al., 2008; Matthews et al., 2009). In our study, the restored wetlands were actively managed with prescribed burns, annual seeding, mechanical removal, and herbicidal treatments (The Nature Conservancy Staff, 2010). Prescribed burns are performed on an opportunistic basis throughout the property; and, ideally, each unit is burned on a four year interval (Andrea Locke, pers. comm.). Mechanical removal techniques include cutting and mowing, while herbicidal treatments consist
of herbicidal backpack spraying, basal bark treatments, and boom-spraying (Andrea Locke and Ted Anchor, pers. comm.).

The comparable diversity of restored wetlands is also attributed to seeding of the restored wetlands. At Site 3, a record of the seed mixture applied throughout the unit was available (Stephanie Fischer, pers. comm.). The mixture contained 25 species, of which 48 percent were tolerant, 36 percent were moderately tolerant, and 8 percent were intolerant. Thirty-two percent (8 species) of the species introduced by seeding were recorded during field sampling. Species that naturally dispersed (25 species) to the site included more opportunistic (20%), tolerant (44%), moderately tolerant (32%), and intolerant (4%) species (Fig. 5a). Kettenring and Galatowitsch (2011) found that active revegetation practices, such as seeding and planting, not only enhance plant community diversity, but also impede the establishment of invasive species, such as Phalaris spp. Of the 8 introduced species, 75 percent were obligate wetland and 25 percent were facultative wetland. The species that naturally colonized the site contained fewer obligate wetland species (44%), facultative wetland species (40%), and greater amounts of facultative (8%) and facultative upland (8%) species than species that colonized from the seed mixture (Fig. 5b).

The natural wetlands differed from restored wetlands in having plant communities that contained significantly higher percentages of moderately tolerant (Fig. 3) and obligate wetland species and lower percentages of facultative upland species (Fig. 4). The mean Coefficient of Conservatism (C of C) was also significantly greater (p < 0.05) for the natural wetlands (3.4 ± 0.1) than for the restored wetlands (2.9 ± 0.1). Studies of restored wetlands in Illinois and Nebraska reported similar findings as the C of C and hydrophytic vegetation indicator status were lower in restored than in natural wetlands (Meyer et al., 2008; Matthews et al., 2009). These conditions could be linked to site hydrology. Wetland restoration projects often lack surface water and have greater depth to groundwater than natural marshes, which could have reduced the ability of many wetland plants to establish on the site (Meyer et al., 2008). Marton (submitted for publication) found that the subsurface (5–15 cm) soil moisture of the natural sites (36% volume:volume) was two times that of the restored sites (17% volume:volume).

In the GIP, restored wetlands are surrounded by a matrix of agricultural lands and are isolated from natural wetlands (Galatowitsch and Van Der Valk, 1996). Landscape isolation slows the colonization rates of dispersal-limited species and decreases the rate of wetland recovery (De Steven et al., 2010). Restoration efforts are further impeded by the establishment of woody vegetation and invasive species, which quickly re-colonize soils exposed by land contouring during the restoration process (Mittelbach et al., 2001). Therefore, projects that consist only of restoring hydrology to wetlands in the region often produce systems low in species richness and diversity (Galatowitsch and Van Der Valk, 1996; Seabloom and Van Der Valk, 2003). Additional measures to restore native plant communities should include seeding, practices to control invasive species and encroachment of woody vegetation, and periodic disturbance (prescribed fire) to promote the establishment of native species.

5. Conclusions

The restored wetlands had plot and site richness and FQAI values that were comparable to the natural wetlands. The restored wetlands, however, contained species of lower C of C and hydrophytic vegetation indicator status. We attribute that the greater overall species richness of restored wetlands is the result of the ecosystem’s successional stage and of the management techniques (seeding, prescribed fire, and herbicidal treatment) that create moderate levels of disturbance as compared to the natural wetlands. Further investigation of individual and combined management techniques is needed to better understand how they can contribute to the restoration of a diverse assemblage of wetland vegetation.

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