Comparison of Plant Species Richness, Diversity, and Biomass in Ohio Wetlands

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Abstract. Wetlands are restored and created in many areas of the world to mitigate problems of flooding, pollution and loss of wildlife habitat resulting from urbanization and agriculture. Consequently it is important to understand the factors that determine wetland ecological function as expressed in terms of vegetation. Five different restored, created and unplanned wetlands of young age (1-9 years) in southwestern Ohio were examined for differences in plant species richness and diversity as well as biomass productivity of cattail (*Typha* spp.) and great bulrush (*Schoenoplectus tabernaemontani*). At all sites, the majority of identified plant species were native (77% to 88%), and nearly half of all taxa in each site were wetland indicator species. The proportion of volunteer species in each site ranged from 51% to 100%. Significant differences detected among sites in both species richness and diversity (Shannon-Weiner Index) were solely due to one of the created sites; significant differences were also obtained among habitat types (shore, emergent zone, and open water). In contrast to *Typha*, aboveground biomass of *Schoenoplectus tabernaemontani* differed significantly among sites but not inflorescence biomass. Overall, there were few differences in plant species richness, diversity or biomass among most restored, created and unplanned sites, suggesting that different methods of wetland formation may yield similar vegetative components within the early stages of development.

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INTRODUCTION

The preservation, restoration, and creation of wetlands has become increasingly important to offset ecological problems caused by past and current drainage of wetland areas worldwide. In the United States, for example, more than half of wetlands have been converted largely for agriculture and urbanization (Mitsch and others 1998; Mitsch and Gosselink 2000). This can lead to increased flooding, contamination of nearby waterways and loss of wildlife habitat. The reintroduction and/or creation of wetlands is a possible way of mitigating these problems (Mitsch and Gosselink 2000). For example, flooding can be minimized if runoff from impervious surfaces is collected in nearby wetlands, where water can be evapotranspirated away. Wetland vegetation can also filter excess nutrients and remove contaminants from agricultural runoff before it flows into neighboring watersheds (Farrell and Scheckenberger 2003) and can stabilize soil along river and stream banks (Mitsch and Gosselink 2000).

In many areas of the United States, there is now an enhanced focus on the restoration of former wet areas and the construction of new wetland sites for either retention/detention purposes or to mitigate natural wetland destruction (Wilson and Mitsch 1996; Mitsch and others 1998). Restored wetlands are formed by recreating the natural hydrology of a site (e.g., removing drain tiles in agricultural fields) and in some cases, by planting native species (Klein 1992; Conover and Klein 1994). Created wetlands are formed in various ways, depending upon their intended purpose. Wetlands designed to treat contaminated water are developed in areas near a pollution source and planted with native species. Still other wetlands are designed for flood control and originate as barren retention/detention basins created next to industrial areas. These sites may be colonized by plant species dispersed by visiting waterfowl and wind, or potentially a seed bank (DeBerry and Perry 2000).

Despite these different methods of wetland formation, there have been few comparisons of restored, created, and unplanned wetlands in terms of the establishment of viable and sustainable ecosystems (Mitsch and Wilson 1996). Wetland function is difficult to measure because it can be evident only after a long time (Mitsch and Wilson 1996), such as the development of hydric soils within an area (Atkinson and others 1993). Function has also been quantified in terms of hydrology, water quality, wildlife use, and vegetation (Wilson and Mitsch 1996). The latter is especially important in younger sites, where plant species diversity and biomass are often used as early indicators of function in comparative studies (e.g., Galatowitsch and van der Valk 1996; Whigham and others 2002; Seabloom and van der Valk 2003a; Spieles 2005). In these cases, intact ecosystems are expected to exhibit high species diversity and biomass productivity, as opposed to sites that lack proper functioning ability.

Investigations to date have generally focused on either restored wetlands (e.g., Galatowitsch and van der Valk 1996; Whigham and others 2002; Seabloom and van der Valk 2003a,b) or specific types of created wetlands, often comparing each of these separately to natural wetland ecosystems (Wilson and Mitsch 1996; Cole and Brooks 2000; Cole and others 2001). Unplanned wetlands have remained largely uninvestigated. Compared to natural areas, restored wetlands had lower species richness and vegetative cover (Seabloom and van der Valk 2003a), but this may depend on the type of species examined. Emergent plants, for example, did not differ in species richness between the two wetland types (Galatowitsch and van der Valk 1996). Compared to natural wetlands, created sites may exhibit either lower species diversity (Weller 1987) or higher diversity and richness (Balcombe and others 2005). Created wetlands may also have lower plant biomass than natural wetlands because of lower levels of organic matter in the soil (Cole and others 2001) and wetter hydrology with shorter dry periods (Cole and Brooks 2000). In an investigation of five created wetlands in Ohio, Wilson and Mitsch (1996) reported that at least 50% of plant species recorded at each site were wetland species but the diversity of these species varied by site. In one of the few direct comparisons between restored and created wetlands, Spieles (2005) examined monitoring reports from wetland mitigation banks and found that restored and created wetlands support similar abundance of species, many of which are non-native. Although these investigations have contributed greatly to our knowledge of wetland performance, more comparisons of vegetation are needed among restored, created

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and unplanned wetlands to understand their ability to function as intact ecosystems.

To compare these wetland types, we examined 5 wetland sites in southwestern Ohio that consisted of 2 restored wetlands, 2 created wetlands, and an unplanned wetland. We compared the sites in terms of plant species richness and diversity, as well as biomass of two emergent plant species, *Typha* spp. and *Schoenoplectus tabernaemontani* (previously *Scirpus validus*). We hypothesized that restored wetlands would exhibit greater species richness and diversity as well as higher productivity than unplanned or created sites, especially those that had been unplanted.

METHODS

Site Descriptions

Two restored wetlands were located in New Haven, OH at the Miami Whitewater Forest. The Miami Whitewater Phase 3 wetland (MWW-3) is approximately 20 acres within the Shaker Trace Wetlands (39.28279° N, 84.75006° W). The site once consisted of wetland and wet prairie but was converted to agriculture during the 1800s. It was restored as a wetland in the early 1990s by removing drain tiles and re-contouring the area to a gradual slope (Klein 1992), before planting the area in 1994 with native wetland and prairie species. Rainfall and groundwater support vegetative growth with open water present throughout the entire year. The second site, Miami Whitewater Phase 4 (MWW-4) is located 1 km south of the Phase 3 site (39.27266° N, 84.75269° W). It is a restored 10-acre wetland planted in 1998 with locally available native plant species. The site is very level and the water level changes rapidly with extensive dry seasons.

Two additional wetlands that were examined were created to mitigate specific urban problems. The 319a wetland is located in West Chester, OH (39.33704° N, 84.4616° W) between Mill Creek and commercial development. This site received federal funding under Section 319 of the Clean Water Act in 2000 to reduce nonpoint source pollution. It was designed to treat both the runoff water from a large retention/detention pond immediately north of it and also water from the adjoining Mill Creek during bank full conditions at flood stage. This approximately 5-acre wetland was contoured in 2001 from land previously used for agriculture. It was planted annually from 2001 to 2003 with native species of trees, saplings, shrubs, emergent plant plugs and seed mixtures. An additional site is a created wetland, ChevronTexaco (CT), located in Hooven, OH (39.18708° N, 84.75038° W) at the site of a former petroleum refinery next to the Great Miami River. Construction of the 8-acre pond and wetland complex began in 2001 and the site was planted in 2002 with native species including trees, shrubs, emergent plants and seed mixes. The wetland was designed to remove suspended solids as the last step in a treatment process for hydrocarbon-contaminated groundwater before being discharged to the Great Miami River under a National Pollutant Discharge Elimination System permit. Water depth in the wetland complex is regulated to foster maximum *Typha* and *Schoenoplectus* growth rates, while attempting to eliminate annual terrestrial weeds. The wetland is underlain by an amended (bentonite) soil liner to hold water in a porous and permeable floodplain and prevent conveyance of treated groundwater to the subsurface.

A fifth wetland site, Union Centre (UnCtr), is an unplanned wetland that was initially created as a retention/detention pond in 1999 to control flooding in West Chester, OH (39.32312° N, 84.44463° W). It receives runoff from impervious surfaces of a nearby commercial distribution center and drains directly into the adjoining Mill Creek. Flooding occurs with most high rainfall events and the site receives a high rate of inflow of sediment, largely from surrounding construction sites. The 5-acre site was never planted but has filled in with volunteer plant species.

At each wetland site, elevational surveys were taken during Summer 2003 from the water line on one shore to the water line on the other shore. Water depth and distance to the optical level (Berger SAL automatic level 20x) were taken every 1 m along the transect.

Biodiversity

Within each wetland site, all plants were recorded during Summer 2003 to generate a plant species list in an area that represented shore, emergent and open water habitats where possible. This area was a rectangle defined on one side by 100 meters in length along the shore with the adjoining perpendicular side extending out to the opposite shore. Three people walked throughout the area to document all plant species and then constructed a single list. The only exception to the sampling was the largest site, MWW-3, which had extensive deep water at its center making it impractical to reach the opposite shore; instead, the second transect extended out only into open water. In addition, the UnCtr site did not contain any open water because it had already filled in completely with plant species. Species were recorded and if identification could not be made with certainty in the field, samples were collected for later identification in the laboratory, following Gleason and Cronquist (1991). At each site, the observed species were compared to lists of planted species obtained from site managers to determine the number of volunteer plant species. Species were identified as native or non-native, based on information provided by the Floristic Quality Assessment Index for Ohio (Andreas and others 2004) in which native species are those taxa presumed to be present in the region prior to European settlement.

Wetland Indicator Status (US Fish and Wildlife 1996) was assigned to each plant species as follows: Obligate Wetland species (OBL) occur almost always (>99%) in wetlands, Facultative Wetland species (FACW) usually occur in wetlands (67%-99%) but are occasionally found in non-wetlands, Facultative species (FAC) are equally likely to occur in wetlands or non-wetlands (34%-66%), Facultative Upland species (FACU) are only occasionally found in wetlands (1%-33%), Obligate Upland species (UPL) are almost always found in non-wetland areas, and No Indicator species (NI) are those taxa for which there is not enough information to determine their wetland status. The relationship among sites in terms of species richness of wetland indicator species (OBL and FACW combined) was examined using Nei and Li's (1979) coefficient of similarity to construct an Unweighted Pair Group Method with Arithmetic Mean (UPGMA) phenogram of similarity. Wetland sites with similar species composition would be expected to cluster together in the analysis.

To characterize species richness and diversity within each site, random 1 m^2 quadrats were sampled in 3 distinct zones (shore, emergent, and open water) along transects extending out into the open water. Where possible on each transect, 2-3 quadrats were obtained from each of the 3 zones. At the larger sites (MWW-3 and CT), more than one transect was established. This resulted in 6-15 quadrats per site. Within each quadrat, percent cover of each species (estimated in increments of 10%), water depth, and sediment depth (i.e., depth to which a meter stick could be easily inserted into the sediment) were recorded. Species Richness (S), Shannon-Weiner Index (H'), water depth, and sediment depth were calculated for each quadrat and individually compared across sites and habitat types (shore, emergent, open) using two-way ANOVA tests with Type IV sums of squares. Weighted averages (see Atkinson and others 1993, 2005) were not used because not all plants in all quadrats were flowering and could be identified to species.

Biomass

Aboveground biomass of ubiquitous *Typha* (both *T. latifolia* and *T. angustifolia*) and *Schoenoplectus tabernaemontani* were also collected from 4 wetland sites (MWW-3, CT, 319a, and UnCtr) in June 2003. Presence of both *Typha* species was verified by molecular analysis using Random Amplification of Polymorphic DNA (RAPD) markers (Culley and Thompson, 2003). With the exception of site 319a, 2-3 biomass transects were established from the shore to open water at each site. No transects were set up at 319a because the population size of each species was limited; instead, samples were obtained in areas where the species were found. At the other sites, quadrats (0.06 m^2) were randomly selected along each transect using a random numbers table, and 3 quadrats per transect were sampled for each species. At each quadrat, samples of *Typha* or *Schoenoplectus* were collected at the soil surface (often below the waterline), and water depth and sediment depth were also measured.

The samples were transported to the University of Cincinnati where the number of stems and maximum stem length per species was recorded for each quadrat. Vegetative samples of *Typha* and *Schoenoplectus* within each quadrat were dried at 60 ° for six days to a constant mass and weighed. Inflorescences of *Schoenoplectus* were also counted, dried and weighed separately. Site differences in aboveground biomass for both taxa as well as differences in inflorescence biomass of *S. tabernaemontani* were examined with individual one-way ANCOVAs, using water depth as a covariate and Type IV sums of squares because of unequal samples sizes per site. Across sites, Pearson correlations were used to examine the relationship between dry weight biomass and stem height, water depth and sediment depth of *Typha* and *Schoenoplectus*.

RESULTS

Wetland Characteristics

The 5 wetland sites were significantly different from one another in mean water depth (ANOVA; $F_{4,38} = 39.65$, $P < 0.0001$), due to greater water depth at the MWW-3 site. Sediment depth also differed significantly among the five sites ($F_{4,37} = 18.27$, $P < 0.0001$), largely due to deep layers of sediment at UnCtr. Surveys indicated that 319a had the steepest slope along the transect, followed by MWW-3, CT, UnCtr and MWW-4. There was also a significant difference in water depth among habitat types (shore, emergent, open) ($F_{2,38} = 35.81$, $P < 0.0001$) and in sediment depth ($F_{2,37} = 5.10$, $P = 0.0111$) across sites.

Biodiversity

We sampled 174 plant taxa across all 5 wetland sites and were able to identify 145 (83%) to the species level and the remaining 17% to the genus level; the latter were mainly seedlings that had not flowered by the time sampling took place. At all sites, the proportion of volunteer species was moderate to high, ranging from 50.6% to 100% (Table 1). Of those taxa we identified to species, only 16.9% to 24.6% were non-native (Table 1). Approximately half of all taxa recorded in each site were also those typically found in wetlands more than 67% of the time (OBL and FACW; Table 2). The MWW-3 site contained the highest proportion of wetland species (60.3%) and the MWW-4 site had the lowest proportion of wetland species (43.9%). Twelve plant species were common to all wetland sites (Table 3) and these included both native and non-native species representing every category of wetland indicator status. Five of these taxa were obligate wetland species, consisting of mud plantain (*Alisma subcordatum*), spike rushes (*Eleocharis* spp.), ditch stonecrop (*Penthorum sedoides*), *Schoenoplectus tabernaemontani* and *Typha* spp.

The quadrat data revealed a significant difference among wetland sites in overall species richness (ANOVA; $F_{4,38} = 4.25$, $P = 0.0061$), due to a high number of species per quadrat in 319a. Similarly, there was a significant difference among sites in H’ (ANOVA; $F_{4,38} = 3.90$, $P = 0.0095$), again due to the 319a site. There was also a significant effect of habitat type on species richness (ANOVA; $F_{2,38} = 14.02$, P = < 0.0001) and diversity (H’ ; $F_{2,38} = 14.73$, $P < 0.0001$). Quadrats on the shore had greater species richness (mean S = 8.13) and diversity (mean H’ = 1.58) than quadrats in the emergent (S = 5.79, H’ = 1.02) or open water zones (S = 3.62, H’ = 0.67), where hydrophily is required for species to survive. Of the 5 sites surveyed, 319a had the greatest number of species per quadrat as well as the highest species diversity (Table 4).

In the phenogram constructed from the presence/absence of obligate and facultative wetland species at each site (Fig. 1), the clustering pattern was inconsistent with the type of wetland (restored, created or unplanned). It was, however, consistent with water level fluctuations, specifically by the frequency and extent of drought. Both MWW-3 and 319a typically do not dry out during the season, while UnCtr often experiences periods of prolonged drought followed by short but substantial flooding events. Although CT has regulated water depth, different areas of the wetland are often left dry for extended periods. MWW-4, a shallow site that often dries out completely later in the season, had the least similarity to the other sites in terms of wetland plant species.

Biomass

Aboveground samples of *Schoenoplectus tabernaemontani* and *Typha* spp. were collected from 65 quadrats (0.06 m^2) from the 4 wetland sites. It was subsequently discovered that at the MWW-3 site, *Scirpus cyperinus* had been mistakenly collected in place of *S. tabernaemontani* in several quadrats; thus in this site only 4 quadrats with the correct species were used in the analysis of *S. tabernaemontani* biomass. The maximum stem height of *Typha* was positively correlated with the total dry weight across sites ($r = 0.875$, $P < 0.0001$), but not for *S. tabernaemontani* ($r = 0.232$, $P = 0.311$). No significant relationship of water depth on biomass was found for either *Typha* ($r = 0.135$, $P = 0.469$) or *Schoenoplectus* ($r = 0.387$; $P = 0.068$). Likewise, sediment depth was not correlated with biomass of *Typha* ($r = -0.007$, $P = 0.977$) or *Schoenoplectus* ($r = -0.523$, $P = 0.066$). Significant differences among the sites in aboveground biomass were found for *Schoenoplectus* (ANCOVA; $F_{3,18} = 3.33$, $P = 0.043$; Fig. 2), but not for *Typha* (ANCOVA; $F_{3,26} = 2.19$, $P = 0.113$). A comparison of habitat zones indicated that there was no significant difference in the percentage of *Schoenoplectus* cover across the 3 habitat types (ANOVA; $F_{2,38} = 1.08$, $P = 0.3488$), although this species was found most frequently in emergent and open zones. As expected, *Typha* had the highest percent cover in the emergent zone (ANOVA; $F_{2,38} = 4.00$, $P = 0.0264$).

Inflorescence biomass of *Schoenoplectus tabernaemontani* per quadrat did not differ significantly across sites (ANCOVA; $F_{3,16} = 1.78$, $P = 0.191$). Using a subset of data, a significant relationship was detected between inflorescence biomass and spikelet number ($R^2 = 0.51$, $P < 0.0001$), indicating that greater inflorescence biomass is associated with increased reproductive output via a greater number of spikelets.

TABLE 1

Number and percentage of species found within each wetland site in southwestern Ohio that were either originally planted or were volunteers, and those considered either native or non-native. Age refers to the age of each wetland since completion of its construction. Native status for each species was obtained from Andreas and others (2004). The total number of species in the volunteer and native categories are not the same because some taxa could only be identified down to genera; these genera were known not to have been planted at the sites, but could not be further defined as native or non-native because the genus contained taxa of both types. Sites were Miami Whitewater Phase 3 (MWW-3), Miami Whitewater Phase 4 (MWW-4), 319a, ChevronTexaco (CT), and Union Centre (UnCtr).

		Species Status		Species Type	
Site	Age (yrs)	# Planted(%)	# Volunteer (%)	# Native (%)	# Non-native (%)
Restored					
MWW-3	9	12 (14.3)	72 (85.7)	59 (83.1)	12 (16.9)
MWW-4	5	2 (3.1)	63 (96.9)	46 (76.7)	14 (23.3)
Created					
319a	2	34 (43.0)	45 (57.0)	46 (75.4)	15 (24.6)
CT	1	41 (49.4)	42 (50.6)	52 (75.4)	17 (24.6)
Unplanned					
UnCtr	4	0 (0)	65 (100)	41 (80.4)	10 (19.6)
Mean	—	17.8 (22.0)	57.4 (78.0)	48.8 (78.2)	13.6 (21.8)

Table 2

Number of species in each site in southwestern Ohio according to wetland indicator status (US Fish and Wildlife 1996). Only those specimens identified to species are included. See Table 1 for description of site notation and abbreviations for wetland indicator status are given in the text.

Wetland Indicator Status	MWW-3	MWW-4	319a	CT	UnCtr
OBL	22	12	17	18	17
FACW	19	13	14	13	10
FAC	7	5	9	7	9
FACU	14	17	9	19	12
UPL	1	2	2	4	1
NI	5	8	8	6	1
Wetland Species	41	25	31	31	27
(OBL + FACW)	(60.3%)	(43.9%)	(52.5%)	(46.3%)	(54.0%)
Total Species	68	57	59	67	50

Table 3

Plant species common to all 5 wetland sites in southwestern Ohio, including their status as native taxa (Andreas and others 2004) and wetland indicators (US Fish and Wildlife 1996). See text for description of notation for wetland indicator status.

Species	Native	Wetland Indicator
Alisma subcordatum (mud plantain)	yes	OBL
Ambrosia artemisiifolia (common ragweed)	yes	FACU
Cyperus strigosus (umbrella sedge)	yes	FACW
Daucus carota (Queen Anne's lace)	no	NI
Eleocharis spp. (spike rushes)	yes	OBL
Eupatorium serotinum (late-flowering thoroughwort)	yes	FAC
Penthorum sedoides (ditch stonecrop)	yes	OBL
Polygonum spp.(smartweed)	yes	FACW
Rumex crispus (curly dock)	no	FACU
Schoenoplectus tabernaemontani (great bulrush)	yes	OBL
Trifolium pratense (red clover)	no	FACU
Typha spp. (cattails)	yes	OBL

DISCUSSION

Wetlands are being restored and created in many areas to alleviate problems associated with increased urbanization and agriculture. Thus it is crucial to understand biotic factors that contribute to the ecological function of these areas as expressed in terms of vegetation. Our study indicates that restored, created, and unplanned wetlands only several years old may exhibit differences from one another in terms of plant species richness, diversity, and biomass production. Significant variation was detected among the sites in species richness and diversity, but this was solely due to a single created wetland, 319a. This wetland had the steepest slope which led to a greater number and variety of species identified within a single quadrat than in other sites. Substantial site differences in aboveground biomass were detected for *Schoenoplectus*, but not for *Typha* or for inflorescence biomass of *Schoenoplectus*; this is consistent with Whigham and others (2002) who suggested that biomass measures may be highly variable among sites and across years. As expected, any large difference in species richness or diversity observed in the current study was associated primarily with habitat type (shore, emergent zone, and open water). Overall site differences in vegetational components are not surprising, considering that abiotic characteristics such as slope, water depth, and sediment depth varied significantly among sites. In contrast, Spieles (2005) found that restored and created wetlands were comparable in species richness based on a much larger sample size (19 restored and 17 created wetlands).

The wetlands examined here are still in the early stages of development (1-9 years old) and as such, vegetational response may change as the wetlands continue to mature. Compared to older sites, younger wetlands are expected to contain higher species diversity and richness (Campbell and others 2002) because these newer areas are relatively unstable and can support a variety of species adapted to

Table 4

Site and habitat type comparisons of plant species richness (S), abundance (H'), and the percentage of Typha *spp. and* Schoenoplectus tabernaemontani *within* $1\ m^2$ *quadrats (mean with stderr in parentheses). The quadrat number (N) varies among sites because it was proportional to site area. See Table 1 for description of site notation. Significant differences among sites and habitat types for each trait is indicated by different superscripts (ANOVA,* $P < 0.05$*).*

Comparison	N	S	H'	% *Typha*	% *Schoenoplectus*
Site					
MWW-3	16	4.62 (0.58) ^{a}	0.90 (0.11) ^{a}	24.63 (7.01) ^{a}	0.63 (0.63) ^{a}
MWW-4	9	4.89 (0.82) ^{a}	0.98 (0.22) ^{a}	12.44 (5.68) ^{a}	0.00 (0.00) ^{a}
319a	6	11.00 (1.15) ^{b}	1.89 (0.14) ^{b}	0.00 (0.00) ^{a}	18.67 (11.67) ^{ab}
CT	12	5.17 (1.01) ^{a}	0.98 (0.23) ^{a}	15.42 (8.97) ^{a}	23.25 (11.03) ^{b}
UnCtr	9	6.89 (0.84) ^{a}	1.19 (0.12) ^{a}	18.33 (9.13) ^{a}	18.67 (9.20) ^{ab}
Habitat type					
Open water	13	3.62 (0.43) ^{a}	0.67 (0.11) ^{a}	6.54 (3.51) ^{a}	10.08 (6.78) ^{ab}
Emergent	24	5.79 (0.71) ^{b}	1.02 (0.12) ^{a}	27.71 (6.58) ^{b}	16.79 (6.25) ^{a}
Shore	15	8.13 (0.74) ^{c}	1.58 (0.14) ^{b}	7.07 (3.22) ^{a}	2.33 (2.33) ^{b}

disturbed habitats (Balcombe and others 2005). In addition, percent cover and biomass is expected to increase as sites mature as well as changes in vegetational composition (i.e. annual to perennial species). Results from vegetational surveys of older wetlands have been variable. Both species diversity and vegetative cover increased over 10 years along the margins of an Australian wetland (Garde and others 2004). In created wetlands in Virginia, a transition from annual to perennial graminoid species occurred over a 20 year period (Atkinson and others 2005). Consequently it is likely that diversity, richness, and biomass of vegetation may continue to change in the southwestern Ohio wetlands. The current study represents a baseline investigation of these wetland sites, which will continue to be monitored over time.

The process of succession may take longer than desired for most manipulated wetlands but it can be accelerated by active planting and seeding in addition to natural colonization by plant species. Seabloom and van der Valk (2003a) found that even after 5 to 7 years, unplanted restored wetlands still had lower vegetative cover and species richness than natural wetlands. They concluded that wetland construction and management of hydrologic conditions were insufficient by themselves and recommended active management of the plant community (e.g., planting native species). Natural wetlands are rare in southwestern Ohio and could not be sampled in the current study. However, comparisons of younger planted wetlands with the more established MWW-3 site indicate that active management of the plant community is effective in promoting wetland plant diversity.

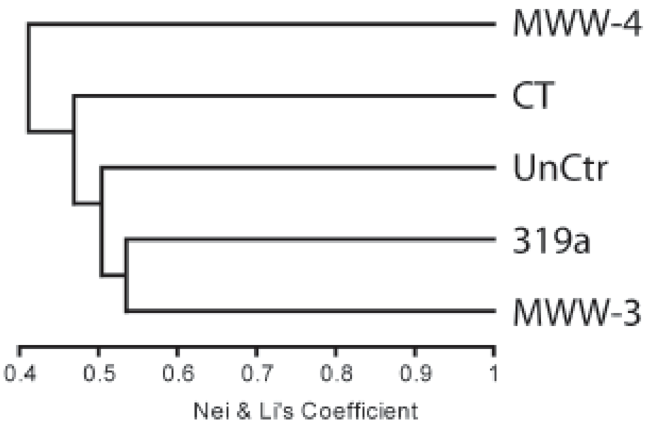

FIGURE 1. Clustering relationships among 5 wetland sites in southwestern Ohio during Summer 2003 based on the presence of obligate (OBL) and facultative wetland (FACW) species, using a UPGMA phenogram and Nei and Li's (1979) coefficient of similarity.

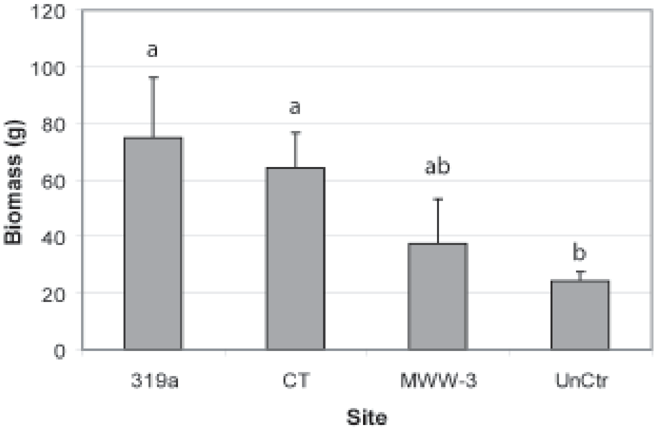

FIGURE 2. Aboveground biomass of *Schoenoplectus tabernaemontani* at 4 sites in southwestern Ohio during Summer 2003: 319a, ChevronTexaco (CT), Miami Whitewater Phase 3 (MWW-3), and Union Centre (UnCtr). Letters denote significant differences among sites ($P < 0.05$).

Planting of native wetland species may offer the advantage of an increased rate of wetland development especially where water conditions support plant growth. For example, CT was less than two years old, but already had a high number of wetland species with fewer volunteer species because it had been planted with typical wetland species and water levels were monitored closely. Species richness and diversity at this location were comparable to other older sites. Interestingly, the same seed mix was used for the shore areas of both 319a and CT and may explain some of their similarities in vegetative cover at those areas. MWW-4 was also planted, but only 4% of the planted species were observed, possibly because of less conducive hydrologic conditions at this location. In contrast, UnCtr (the only unplanted site) consisted solely of volunteers yet contained more wetland plant species than the restored MWW-4. This may reflect greater variation in elevation as well as less severe drying conditions at UnCtr. Furthermore, this site experiences occasional inundations of heavy rainfall and has the deepest sediment levels, enabling it to hold high amounts of moisture for longer periods of time. In contrast, MWW-4 was the driest site, with shallow sediment levels and the most level slope.

The frequency of natural seed and propagule dispersal to sites may also be pivotal in wetland development. Colonizing seeds at a new site may not always disperse from the closest neighboring wetland (DeBerry and Perry 2004), but may arrive through other means. Three of the examined sites (319a, UnCtr, and CT) were connected directly to a waterway, and as such are more likely to receive and exchange floating propagules (Bornette and others 1998). This in part may explain the quick development of thick vegetative cover in the retention/detention basin of UnCtr within only four years. This is further supported by the fact that many currently barren retention/detention basins of the same age near the UnCtr site are not connected to a waterway. Although a waterway connection is lacking for the restored Miami Whitewater sites, substantial revegetation in these two areas (especially Phase 3) may reflect extensive plantings, a predominant seed bank, and use by waterfowl that transport seeds. Both Phase 3 and 4 sites occupy previously wet land that was drained decades ago for agriculture. During restoration of the Phase 3 site in the early 1990s, viable seeds of various wetland plant species were discovered still dormant in the soil (Conover and Klein 1994). Many of these taxa are among the volunteer species now present at the sites. Seed banks have been detected before in wetland sites (DeBerry and Perry 2000), although restored sites often contain fewer species and seeds than in natural wetlands (Galatowitsch and van der Valk 1996). Thus one potential but rarely studied advantage of restored sites over created or unplanned sites is the presence of a seed bank that may promote quick revegetation of the site with native species.

A goal of many restoration projects is to maximize native plant diversity while minimizing the number of non-native species, especially invasive weeds, that often accompany habitat disruptions. We found that native plant species were numerous at all 5 sites we examined, regardless of the different ages and types of wetlands. This included UnCtr that was originally unplanted, suggesting that native plant species are still able to successfully disperse into appropriate habitats in southwestern Ohio and persist. Overall, the Ohio wetland sites examined here contained a slightly higher percentage of non-native species (21.8%) than found in created sites in neighboring West Virginia (17.8%; Balcombe and others 2005), or in restored and created wetlands in mitigation banks in the United States (17.6% and 18.9% respectively; Spieles 2005). Non-native plants considered highly invasive in Ohio (Windus and Kromer 2001) were also observed at several sites, although they were rare. For example, one large individual of purple loosestrife (*Lythrum salicaria*) was found at 319a but was removed shortly thereafter by land managers. Multiflora rose (*Rosa multiflora*) was also observed in small patches at the MWW-3 site but is unlikely to spread because of the dense vegetation already present at the site. Several other invasive plants, such as narrow-leaved cattail (*Typha angustifolia*), Canada thistle (*Cirsium arvense*), and white (*Melilotus alba*) and yellow sweet-clover (*M. officinalis*) were observed at the restored and observed wetlands, but only *T. angustifolia* was found at the unplanned wetland.

Although the number of wetlands sampled in the current study was limited and only involved young sites, the results still indicate that restored and created wetlands have the potential to harbor substantial species richness, diversity and biomass production within only a few years. Even more importantly, given the right environmental conditions such as high inflow and sedimentation, an unplanned wetland such as UnCtr can also flourish and eventually attain equivalent status as restored and created sites. Future investigators should seek out greater numbers of restored, created, unplanned and natural wetlands for comparison so that we can more fully understand the ability of different wetland types to produce high plant diversity and biomass. It would be especially helpful to examine older wetlands sites with well-developed hydric soils because these areas can show changes in vegetation communities with time (e.g., Atkinson and others 2005). In addition, hydrological conditions may vary over time and examination of wetlands over several years should yield further information about the effect of weather variations. This is an important area of research that will greatly benefit our understanding of wetland restoration and creation in an effort to ameliorate the deleterious effects of urbanization.

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LITERATURE CITED

Andreas BK, Mack JJ, McCormac JS. 2004. Floristic quality assessment index (FQAI) for vascular plants and mosses for the State of Ohio. Ohio Environmental Protection Agency, Division of Surface Water, Wetland Ecology Group, Columbus, Ohio.

Atkinson RB, Perry JE, Smith E, Cairns J. 1993. Use of created wetland delineation and weighted averages as a component of assessment. *Wetlands* 13(3):185-193.

Atkinson RB, Perry JE, Cairns J. 2005. Vegetation communities of 20-year-old created depressional wetlands. *Wetlands Ecol. Management* 13:469-478.

Balcombe CK, Anderson JT, Fortney RH, Rentch JS, Grafton WN, Kordek WS. 2005. A comparison of plant communities in mitigation and reference wetlands in the mid-Appalachians. *Wetlands* 25(1):130-142.

Bornette G, Amoros C, Lamouroux N. 1998. Aquatic plant diversity in riverine wetlands: the role of connectivity. *Freshwater Biol.* 39(2):267-283.

Campbell DA, Cole CA, Brooks RP. 2002. A comparison of created and natural wetlands in Pennsylvania, USA. *Wetlands Ecol. Management* 10:41-49.

Cole CA, Brooks RP. 2000. A comparison of the hydrologic characteristics of natural and created mainstream floodplain wetlands in Pennsylvania. *Ecol. Eng.*14(2-3):221-231.

Cole CA, Brooks RP, Wardrop DH. 2001. Assessing the relationship between biomass and soil organic matter in created wetlands of central Pennsylvania, USA. *Ecol. Eng.* 17(4):423-428.

Conover DG, Klein J. 1994. Both planted and non-planted species populate restored wetland (Ohio). *Rest. Management Notes* 12(2):195-196.

Culley TM, Thompson K. 2003. Unpublished data.

DeBerry DA, Perry JE. 2000. Wetland seed banks: Research in natural and created wetlands. *Wetlands Program Technical Report* 00(4):1-6.

DeBerry DA, Perry JE. 2004. Primary succession in a created freshwater wetland. Castanea 69(3):185-193.

Farrell AC, Scheckenberger RB. 2003. An assessment of long-term monitoring data for constructed wetlands for urban highway runoff control. *Water Quality Res. J. Can.* 38(2):283-315.

Galatowitsch SM, van der Valk AG. 1996. The vegetation of restored and natural prairie wetlands. *Ecol. Appl.* 6(1):102-112.

Garde LM, Nicol JM, Conran JG. 2004. Changes in vegetation patterns on the margins of a constructed wetland after 10 years. *Ecol. Management Restoration* 5(2):111-117.

Gleason HA, Cronquist A. 1991. Manual of Vascular Plants of Northeastern United States and Adjacent Canada. New York: The New York Botanical Garden. 910 p.

Klein J. 1992. Drain-tile removal, recontouring, planting characterize wetland restoration project (Ohio). *Rest. Management Notes* 10(2):186-187.

Mitsch WJ, Gosselink JG. 2000. Wetlands, third edition. New York: John Wiley & Sons. 920 p.

Mitsch WJ, Wilson RF. 1996. Improving the success of wetland creation and restoration with know-how, time, and self-design. *Ecol. Appl.* 6(1):77-83.

Mitsch WJ, Wu X, Nairn RW, Weihe PE, Wang N, Deal R, Boucher CE. 1998. Creating and restoring wetlands. *BioScience* 48(12):1019-1030.

Nei M, Li W-H. 1979. Mathematical model for studying genetic variation in terms of restriction endonucleases. *Proc. Nat. Acad. Sci.* 76(10):5269-5273.

Seabloom EW, van der Valk AG. 2003a. Plant diversity, composition and invasion of restored and natural prairie pothole wetlands: implications for restoration. *Wetlands* 23(1):1-12.

Seabloom EW, van der Valk AG. 2003b. The development of vegetative zonation patterns in restored prairie pothole wetlands. *J. Applied Ecology* 40(1):92-100.

Spieles DJ. 2005. Vegetation development in created, restored, and enhanced mitigation wetland banks of the United States. *Wetlands* 25(1):51-63.

US Fish and Wildlife. 1996. National list of vascular plant species that occur in wetlands. Available at: http://wetlands.fws.gov/bha/list96.html.

Weller MW. 1987. Freshwater Marshes: Ecology and Wildlife Management. Minneapolis: University of Minnesota Press. 150 p.

Whigham D, Pittek M, Hofmockel KH, Jordan T, Pepin AL. 2002. Biomass and nutrient dynamics in restored wetlands on the outer coastal plain of Maryland, USA. *Wetlands* 22(3):562-574.

Wilson RF, Mitsch WJ. 1996. Functional assessment of five wetlands constructed to mitigate wetland loss in Ohio, USA. *Wetlands* 16(4):436-451.

Windus J, Kromer M. 2001. Invasive plants of Ohio: a series of fact sheets describing the most invasive plants in Ohio's natural areas. Available from the Ohio Department of Natural Resources.