

Structural and functional vegetation development in created and restored wetland mitigation banks of different ages

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ABSTRACT

Vegetation surveys of seven Ohio wetland sites were conducted from 2008 to 2010 during peak biomass. These seven sites included five created/restored mitigation bank wetlands, a created riverine research wetland, and a natural reference wetland. All of the created/restored wetlands ranged in age from 3 to 18 years. The objective of this study was to examine the development of vegetation structure and function of mitigation bank wetlands less than 20 years of age and to compare these to reference wetlands. Vegetation structure examined included species richness, floristic quality assessment index (FQAI), Shannon–Wiener diversity index (H), and community diversity index (CDI). Functional attributes included aboveground net primary productivity (ANPP) and functional group composition of dominant species. For both structure and function, the reference wetlands were statistically different from the wetland mitigation bank sites ($P < 0.001$, MANOVA). Structurally, there were significant differences of FQAI score ($P < 0.05$) and species richness ($P < 0.05$) with age in the mitigation sites. Functionally, there was a significant difference between ANPP ($P < 0.05$) and age in the mitigation sites. Over the different types of wetlands, the reference wetlands had significantly different ANPP, FQAI scores, and species richness than did the mitigation sites ($P < 0.001$, $P < 0.05$, and $P < 0.001$, respectively). CDI and H were not statistically different between mitigation sites and the reference wetlands. ANPP, FQAI, and species richness tended to be higher in the reference sites than in the mitigation sites. Overall, the mitigation bank wetlands were not statistically similar to the reference sites. Within the mitigation banks, the younger sites had higher values for structural attributes than the older mitigation sites.

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1. Introduction

Throughout much of the humid United States, there has been a significant loss of natural wetlands since European settlement. Most of the Upper Midwestern states have lost more than 80% of their wetlands, while Ohio has lost approximately 90% of its natural wetlands since the 18th century (Dahl, 1990). One way to combat further loss has been through wetland mitigation. Under Section 404 of the U.S. Clean Water Act, a permit is required to drain, damage, or destroy a jurisdictional wetland (legally recognized wetland as determined by the Wetland Delineation Manual published by the Army Corps of Engineers) (Mitsch and Gosselink, 2007; Salzman and Thompson, 2007). The permit holder is then

required to mitigate the loss of the wetlands that were damaged or destroyed, usually at a ratio greater than 1:1, i.e. where more wetland area is created or restored than was lost. The purpose of a greater than 1:1 ratio is to avoid a net loss of wetland area and ecosystem processes. Mitigation can be done on an individual basis or by an organization that specializes in wetland creation and restoration. The second type of mitigation, often referred to as a wetland mitigation bank, is when wetlands are created or restored without regard to a specific permit prior to wetland destruction. Wetland area credits from mitigation banks are then sold to permit holders to fulfill their permit requirements (Mitsch and Gosselink, 2007).

Ideally, the design of a created or restored wetland should be modeled after the natural wetland that was damaged or destroyed in order to replicate the wetland type and ecosystem processes of the natural site. In many cases, particularly with mitigation bank wetlands, information regarding the natural wetland that was damaged or destroyed cannot be used in the planning of the mitigation wetland. Instead, reference wetlands located near the mitigation site are used as a guide for

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mitigation construction and monitoring (LePage, 2011). The use of high quality reference wetlands can illustrate the overall gain or loss of wetland function due to mitigation and can aid in better initial construction of the mitigation wetlands (Brinson and Rheinhardt, 1996).

Monitoring of mitigation wetlands usually lasts for the first five years after creation/restoration, at which point the success of the wetland is determined (NRC, 2001). The success and quality of these created and restored wetlands are determined using hydrologic, vegetation, and soil characteristics, with emphasis being placed on vegetation (NRC, 2001; Spieles et al., 2006). Vegetation parameters commonly used are structural in nature and include such things as species richness, percent ground cover, and percent native plant species. There is some debate as to whether or not current structural parameters are adequate for determining success. It has been suggested that structural parameters have been chosen for speed and cost efficiency and may not reflect ecological processes (Mitsch and Wilson, 1996; Spieles, 2005; Ahn and Dee, 2011). Tilman et al. (1997) found that in grassland ecosystems, functional characteristics (e.g. productivity and functional group composition) are better determinants of ecosystem processes (e.g. photosynthesis, nutrient cycling) than structural characteristics (e.g. species richness, percent cover).

Another issue of concern is the 5-year monitoring period. It has been suggested that 15–20 years of maturation may be necessary before the true success of a wetland can be assessed (Mitsch and Wilson, 1996). Even more time may be needed if the desired result is a forested wetland (Niswander and Mitsch, 1995; NRC, 2001). From a regulatory standpoint, some studies have shown poor wetland structure and function in created and restored mitigation wetlands within the desired time frame of 5 years. Success rate, described as the ability of a site to meet regulatory permit requirements within the allotted monitoring time, range from approximately 40 to 60% in some states (Cole and Shafer, 2002; Reiss et al., 2009). While regulatory success is not the equivalent of ecological success (ability of a mitigation wetland to function as does a natural wetland; see Mitsch and Wilson, 1996), the inability of wetlands to meet permit requirements suggests that mitigation wetlands may not adequately be capturing the ecological processes that were lost during the destruction of natural wetlands or not all wetland ecosystems are capable of achieving ecological success within a 5-year time frame. Atkinson et al. (2005) found that mitigation wetlands, after 20 years, began to reach a state of equilibrium. Additional studies have seen a similar trend, with mitigation wetland vegetation beginning to resemble vegetation of natural wetlands within approximately 20 years (Balcombe et al., 2005; Spieles, 2005; Gutrich et al., 2009).

Much of the mitigation banking within the U.S. relies on creation, restoration, and conservation techniques. Wetland creation tends to have a fairly low rate of success compared to restoration of a site, while conservation does not add any additional wetland area or replace lost ecosystem functions (ELI, 2002; Spieles, 2005). Construction techniques have been linked to problems with wetland development due to such things as improper placement within the landscape and soil removal/compaction by earth moving equipment (Mitsch and Wilson, 1996; Campbell et al., 2002; Bruland and Richardson, 2005; Bantilan-Smith et al., 2009; van der Valk et al., 2009; Ahn and Dee, 2011). Wetland development can be impaired if the wetland is constructed in an area where proper hydrology cannot be obtained or if the wetland lacks connectivity to propagule sources (Mitsch and Wilson, 1996; van der Valk et al., 2009). Soil removal to form wetland basins, as well as soil compaction, can alter the organic matter content and bulk density of the soil. These changes in soil characteristics can inhibit the penetration of roots within the soil and reduce nutrient availability, thus

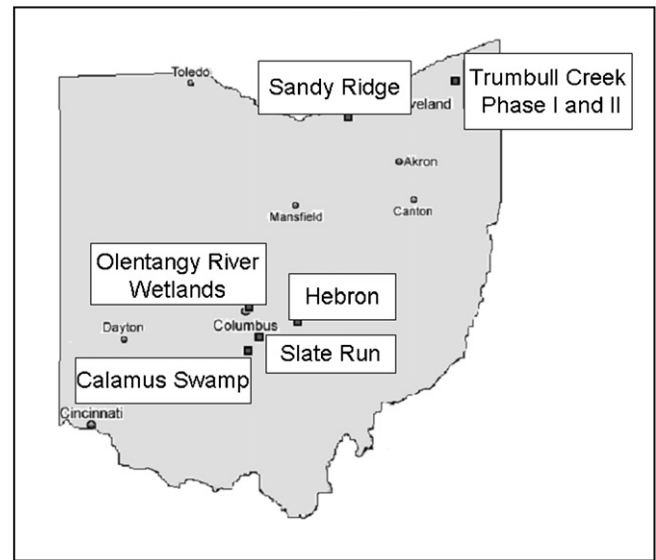


Fig. 1. Location of wetland sites within Ohio, USA. Sandy Ridge, Trumbull Creek Phase I and Trumbull Creek Phase II North and South are located in the northern portion of the state, while Hebron, Slate Run, Olentangy River Wetland Research Park (ORWRP), and Calamus Swamp are located in central Ohio.

hindering the establishment of desired plant species and primary productivity (Campbell et al., 2002; Bruland and Richardson, 2005; Bantilan-Smith et al., 2009; Ahn and Dee, 2011). Delays in wetland development due to construction techniques may account for the inability of some mitigation projects to achieve both legal and ecological success within the allotted monitoring period.

The goal of this project is to examine vegetation of created/restored mitigation bank wetlands from both a structural and functional standpoint and to compare these mitigation bank wetlands to well-maintained research wetlands of a similar age and a natural reference wetland. It is hypothesized that (1) structural and functional parameters will differ between the natural/research reference wetlands and mitigation bank wetlands; (2) older wetlands will have higher productivity, species richness, and diversity than younger wetlands; (3) productivity in wetlands will vary based on location within Ohio; and (4) flow-through wetlands will have higher primary productivity and species richness.

2. Methods

2.1. Study sites

Samples were collected at seven wetland sites throughout northern and central Ohio. Three of the wetland sites are located in northern Ohio, while the remaining four sites are located in central Ohio (Fig. 1). The five mitigation bank sites used in this study are Hebron, Sandy Ridge, Slate Run, Trumbull Creek Phase I, and Trumbull Creek Phase II. The reference wetlands used included the created experimental wetlands at the Olentangy River Wetland Research Park (ORWRP) and Calamus Swamp, a natural wetland in southern Ohio owned by the Columbus Chapter of the Audubon Society. Not only did we want to compare the mitigation sites to a natural reference wetland, we also wanted to compare the mitigation sites to a successful created wetland of similar age. The experimental wetlands at the ORWRP are located on the northern end of The Ohio State University Columbus campus along the Olentangy River. These wetlands, created in 1994, were shown more than a decade earlier (1998) to be comparable to natural wetlands in Ohio based on FQAI scores (structure) and Ohio Wetland

Assessment Method ratings (function) (Elifritz and Fennessy, 1999). The wetlands at the ORWRP developed quickly due to their optimum hydroperiod, hydrologic connection to the Olen-tangy River which acts as a propagule source for the wetlands, and continual exchange of biotic and abiotic factors with the surrounding environment (Mitsch et al., 1998, 2005a,b, 2012; Mitsch and Gosselink, 2007).

All of the created and restored sites contain multiple wetlands with emergent marsh vegetation and are less than 20 years in age (Table 1). The natural reference wetland is a marsh that is beginning a transition along the boarder into a shrub/scrub wetland dominated by *Cephalanthus occidentalis*. Hydrologic conditions varied from site to site and were put into one of three categories; continuously flooded, high spring low fall, and pulsing (Table 1).

2.2. Sampling methods

Sampling was performed at each of the sites in August of 2008, 2009 and 2010. Across the seven sites, 250 vegetation plots were sampled in a total of 52 wetland basins. The number of basins examined per site varied based on the overall size of the site, as well as the size of the individual wetland basins present to ensure similarities in overall area and number of sample plots examined at each site. A basin is defined as a depression within the landscape that meets the criteria of a wetland (proper hydrology, wetland vegetation, and hydric soils) and is hydrologically isolated except under major storm events. A 0.25 m² portable pvc sampling frame was placed in the vegetation plots every 40 paces (approximately 0.5 m per pace) along transects established in each plant community. Data collected at the plots included species present, stem density and average height for each species. These data were used to determine aboveground net primary productivity (ANPP) via regression equations. Data collected from the entire site included area of major plant communities and maximum water depth. The collected data was used to calculate species richness, Ohio floristic quality assessment index (FQAI), Shannon–Wiener diversity index (*H*), community diversity index (CDI), and functional group composition of major plant communities. Additional information on site characteristics was obtained from monitoring reports and previous studies.

Area of each major plant community, as well as total area of emergent vegetation was determined by ground-truthing and GPS. A Thales MobileMapper GPS unit was used to find the area of each plant community and area of open water within each wetland. Data collected was used to determine overall area of the wetlands sampled at each site, to create vegetation maps, and to calculate CDI for each basin. Elevation maps from the mitigation monitoring reports were used to find the approximate location of lowest elevation each wetland. A weighted measuring tape was used to estimate the depth of the water at this location during each sampling visit.

2.3. Structural parameters

FQAI scores were calculated using the methods outlined in Andreas et al. (2004):

$$FQAI = \frac{\sum(C \text{ of } C_i)}{\sqrt{(N)}} \quad (1)$$

where *C* of *C_i* is the coefficient of conservatism for each species and *N* is the number of native species. All species were assigned a coefficient of conservatism value between 0 and 10 which is “an ordinal weighting factor of the degree of conservatism (or fidelity) displayed by that species in relation to all other species of the region” that was determined by Andreas et al. (2004)

to be used within the state of Ohio. Species with lower coefficient of conservatism values are found in highly disturbed areas, whereas higher values are indicative of species that have a relatively narrow ecological niche (Lopez and Fennessy, 2002; Ahn and Dee, 2011). This system does not give additional weight to endangered or rare species. Within wetland sites of Ohio, the FQAI scores for emergent marshes tend to be around 20 or 21, with a range of approximately 11–34 (Lopez and Fennessy, 2002; Andreas et al., 2004).

The Shannon–Wiener diversity index was also used to examine species richness and evenness between the different wetland sites:

$$H' = -\sum_{i=1}^s (p_i \ln p_i) \quad (2)$$

where *H'* is the diversity index score, *s* is the number of species, and *p* is the relative abundance of a particular species (McCune and Grace, 2002). Unlike the FQAI, the *H* includes non-native and invasive species in the calculations and does not include a system to give additional weight to particular species.

The community diversity index (CDI; Mitsch et al., 2005a,b) was calculated using the equation:

$$CDI = -\sum_{i=1}^N (C_i \ln C_i) \quad (3)$$

where *N* is the number of wetland communities and *C* is the relative area of each wetland community. This CDI is a landscape index that estimated the richness and evenness of spatial patterns of vegetation communities, using the area of communities rather than stem counts as the dependent variables. A CDI value of 0.0 represents a monospecific landscapes, whereas higher numbers indicate diverse patterns of several communities.

2.4. Functional parameters

Emergent macrophyte above ground net primary productivity (ANPP) was estimated from peak biomass measurements in August each year with non-destructive sampling techniques for the five mitigation bank sites and the natural wetland site (Thursby et al., 2002). Regression equations from other studies in Ohio (Johnson, 1998) and the United States (Muzika et al., 1987) were used to estimate biomass for most macrophyte species. When equations were not available for a particular species, multiple biomass samples were collected for that species and dried at 105 °C for 48 h. Relationship between biomass dry weight and stem height and stem density were then used to create regression equations for those species (Appendix A). Plot primary productivity data were then averaged to estimate ANPP of each wetland basin. ANPP was weighted based on area of the plant communities within the emergent zone using the equation:

$$WANPP = \frac{\sum(A_i B_i)}{E} \quad (4)$$

where WANPP is the weighted aboveground net primary productivity, *A* is the area of a specific community, *B* is the average biomass for that specific community, and *E* is the total area of all emergent plant communities within the wetland basin. In addition to ANPP, ANPP of emergent vegetation across the entire wetland area was determined, taking into account the amount of open water at each wetland. In rare instances when no equation was available for a plant species, the plot that the species was present in was omitted from calculations and statistics regarding net primary productivity. Data collected from the omitted plots were still used for species richness, FQAI scores, and other appropriate variables.

Table 1

Year of creation, site and wetland areas, hydrology, hydrogeomorphic classification, soil type, and location of Ohio wetland mitigation banks and reference wetlands used in this study (Acton, 2004; Brinson, 1993; Davey, 2007a,b; United States Department of Agriculture Natural Resources Conservation Service, 2011). For hydrogeomorphic classification (HGM), the first letter is the geomorphic settings, the second letter is the water source, and third letter is the hydrodynamics. D: depressional, R: riverine, P: precipitation, S: surface flow, G: groundwater, V: vertical fluctuation, and U: unidirectional flow.

	Mitigation bank wetlands					Reference wetlands	
	Hebron	Sandy Ridge	Slate Run	Trumble Creek Phase I	Trumble Creek Phase 2 North and South	Calamus Swamp	Olentangy River Wetland Research Park
Year created	1992	1998	1999	2000	2005	Natural	1994
Total site area (ha)	13.4	46.5	63.9	56.7	117.8	7.7	21
Total area of wetlands investigated (ha)	8.8	20.7	9.1	6.2	1.2	4.9	2
Wetland hydrology	High spring low fall/continuously flooded	Continuously flooded	High spring low fall	Continuously flooded	High spring low fall	High Spring low fall	Pulsing
HGM classification	DPV	DP/SV	DPV and DP/GV	DPV	DPV	DPV	RSU
Dominant soil type	Luray Silty Clay Loam	Fitchville Silt Loam	Kokomo Silty Clay Loam	Plateau and Sheffield Silt Loam	Sheffield Silt Loam	Montgomery Silty Clay Loam	Ross Silt Loam
Location	Licking County	North Ridgeville, Lorain County	Pickaway County	Geauga-Ashtabula Counties	Geauga-Ashtabula Counties	Circleville, Pickaway County	Columbus, Franklin County

Destructive primary productivity sampling was used at the ORWRP. Vegetation samples were harvested from within a 0.5 m² frame at ground level from 24 plots. Wet weight was determined in the lab shortly after harvesting. Harvested biomass was then separated by species and dried at 105 °C for 48 h or until constant weight. For species that occurred in both the ORWRP and at least one of the mitigation sites, the regression equations used for determining primary productivity for non-destructive methods were checked for accuracy using the data collected at the ORWRP.

The dominant species at each wetland site were categorized into three functional groups based on Boutin and Keddy's (1993) functional classification of wetland plants. These groups were ruderals, interstitial and matrix species. Ruderals consist of obligate and facultative annuals, interstitial included reeds, clonal plants and tussocks, while matrix species were clonal stress-tolerators and clonal dominants. Highly disturbed sites tend to be dominated by ruderals, whereas natural wetlands are dominated by matrix species (Matthews and Endress, 2010).

2.5. Statistical methods

Multivariate analysis of variance (MANOVA) and analysis of variance (ANOVA) were used to test hypotheses. To avoid problems with pseudoreplication due to small wetland size and proximity of the wetlands across the sites, data for the wetlands at Trumbull Creek Phase II South, as well as Trumbull Creek Phase II North were combined to obtain one number for each parameter at these sites. This reduced the data set from 52 to 13 wetland basins. The statistical software program R was used to test assumptions and perform the MANOVA and ANOVA tests (determined assumptions were met using D'agostino skewness test, Anscomb–Glynn kurtosis test, Shapiro–Wilks normality test all at $\alpha=0.1$, normal quantile–quantile plots, and scatter plots) (The R Foundation for Statistical Computing, 2009). Linear regression analysis was then performed to determine the type of relationship (positive or negative) between the structural and functional parameters and age of the mitigation sites.

A detrended canonical correspondence analysis (DCCA) was performed to determine the type of analysis, canonical correspondence analysis or redundancy analysis (CCA and RDA, respectively), to be used for the data set. A gradient length of 2.52 was obtained for the first axis. Since the gradient length is less than three,

using the suggestion from Leps and Smilauer (2003), a redundancy analysis (RDA) was used to analyze the data. A Monte Carlo test was performed to test the null hypothesis that the species data are independent from the environmental variables. The computer software CANOCO was used for DCCA, RDA, and the Monte Carlo test (ter Braak and Smilauer, 2004). The DCCA, as well as the RDA, were performed on species presence/absence data and environmental data for the sites. Categorical environmental variable for type of hydrology and location in Ohio were turned into dummy variables. All of the continuous environmental variables were standardized since they were on vastly different scales.

3. Results

3.1. Structural and functional parameters

Species richness was highest at the reference research site (98 species), Sandy Ridge (102 species), and Trumbull Creek Phase I (109 species). The wetland sites had 2–9 dominant species and 2–13 identifiable plant communities. The Hebron site had the lowest number of dominant species and communities while the Sandy Ridge site had the highest number of dominant species and number of plant communities. FQAI scores for the wetlands ranged from 13.5 to 29.5 at the mitigation banks, 19.9 to 23.8 at the research site, and 26.9 at the natural wetland. The macrophyte *H* ranged from 0.51 to 1.90 at the mitigation bank wetlands and was 1.75 at the natural reference wetland. The CDI ranged from 0.90 to 1.75 for the mitigation sites and from 0.84 to 1.44 for the reference sites (Table 2).

Water depth varied greatly from site to site. Trumbull Creek Phase I North, Sandy Ridge, and two wetlands at Slate Run had more than 1 m of standing water during August and September sampling, whereas many of the wetlands within Trumbull Creek Phase II North and Trumbull Creek Phase II South had no standing water at the time of sampling. The wetlands at Hebron, ORWRP, Calamus Swamp, and the two remaining wetlands at Slate Run had water depths between 10 and 100 cm. Area of emergent vegetation was largest at Sandy Ridge (7.6 ha) and smallest at Trumbull Creek Phase II North (0.23 ha). Hebron and Trumbull Creek Phase I wetlands had the most area of emergent vegetation, while ORWRP, Slate Run, and Trumbull Creek Phase II South had the least.

Table 2

Average \pm standard error of structural and functional parameters at each wetland site. Species richness, number of communities, Ohio floristic quality assessment index (FQAI), Shannon–Wiener diversity index (H'), community diversity index (CDI), and weighted ANPP (WANPP) of all wetland sites over 2008, 2009, and 2010. Calamus Swamp data are for 2010 only. FQAI defined by Andreas et al. (2004). CDI defined by Mitsch et al. (2005a,b). Functional group ratios of ruderal:interstitial:matrix categories for the dominant species at each site were determined using Boutin and Keddy (1993), Keddy et al. (1998), and Lenssen et al. (1999). P -values were determined using ANOVA. All statically significant P -values (≤ 0.05) are in bold.

	Mitigation Bank Wetlands					Reference Wetlands		Hypothesis P -values			
	Hebron	Sandy Ridge	Slate Run	TCPI	TCPII	Calamus	ORW	Mitigation vs. reference	Age	Hydrology	Location
Structure											
Species richness	44 \pm 2.3	91 \pm 6.3	44.6 \pm 2.8	88.7 \pm 10.5	68 \pm 4.3	87	95.5 \pm 1.2	<0.001	0.033	<0.001	0.011
Number of communities	4 \pm 1	11.3 \pm 0.88	5.3 \pm 0.63	8.3 \pm 0.33	7.8 \pm 0.60	7	4.8 \pm 0.54	0.254	0.72	0.367	<0.001
FQAI	15.6 \pm 0.54	22.8 \pm 1.3	17.1 \pm 0.57	27.1 \pm 1.39	20.5 \pm 0.86	26.9	21.7 \pm 0.78	0.042	0.005	0.02	<0.001
H'	1.20 \pm 0.21	1.27 \pm 0.27	1.10 \pm 0.11	1.47 \pm 0.19	1.23 \pm 0.22	1.75	na	0.212	0.209	0.794	0.409
CDI	1.10 \pm 0.13	1.01 \pm 0.15	0.77 \pm 0.09	1.28 \pm 0.11	1.45 \pm 0.14	0.84	1.34 \pm 0.10	0.165	0.187	0.209	0.02
Function											
WANPP	451 \pm 52	452 \pm 44	441 \pm 45	663 \pm 16	440 \pm 51	1093	682 \pm 52	<0.001	0.006	0.026	0.77
Functional group ratio	2:1:4	2:2:1	4:2:5	1:4:2	2:5:3	0:0:4	0:1:5				

WANPP ranged from 201 to 802 g DW m⁻² yr⁻¹ at the mitigation bank sites, 529 to 866 g DW m⁻² yr⁻¹ at the research reference wetlands and 1093 g DW m⁻² yr⁻¹ at the natural reference wetland (Table 2). For functional groupings of the dominant species at each site, the reference wetlands had 84–100% matrix species (Table 3). The remaining dominant species in the reference wetlands were in the interstitial group. In the oldest mitigation bank site (Hebron), matrix species accounted for 57% of the dominant species, while ruderals were 29%. The second oldest mitigation bank site had 20% matrix species and 40% ruderals (Sandy Ridge). The intermediate age wetland had 45% matrix species and 36% ruderals. The two youngest mitigation bank sites had the lowest percentage of matrix species (29–30%). Of all of the sites, the two youngest mitigation banks had the highest amount of interstitial species (50–57%).

3.2. Hypotheses testing

MANOVA was preformed for the four different hypotheses. P -values of ≤ 0.05 were found for comparisons of mitigation vs. reference wetlands ($P=0.003$), age ($P=0.001$), location in Ohio

($P<0.001$), and hydrology ($P<0.001$) for the structural and functional parameters examined. Year sampled had no effect on any of the dependent variables ($P=0.407$).

ANOVA was then used to assess the significance of these environmental variables on the structural and functional vegetation characteristics (Table 2). Significant differences were found between age and WANPP ($P=0.006$), log ANPP over the entire wetland ($P<0.001$), FQAI score ($P=0.005$), and species richness ($P=0.033$). Linear regression analysis was used to determine the type of relationship, positive or negative, between the significant vegetation variables and age of the mitigation sites. A positive relationship was found between age and WANPP (Fig. 2). Negative relationships were found between age and FQAI scores and species richness (Fig. 2). Log ANPP over the entire wetland did not exhibit a linear pattern, but had higher values in the younger and older mitigation sites (Fig. 2). Significant differences were found between type of hydrology and WANPP ($P=0.026$), FQAI score ($P=0.020$), and species richness ($P<0.001$). Pulsed hydrology wetlands had the highest WANPP, with high spring low fall wetlands having the lowest. Continuously flooded wetlands had some of

Table 3

Functional groups of dominant wetland plant species at the five mitigation sites and two reference sites. Functional groups are defined by Boutin and Keddy (1993), Keddy et al. (1998), and Lenssen et al. (1999).

	Mitigation bank wetlands					Reference wetlands	
	Hebron	Sandy Ridge	Slate Run	Trumbull Creek Phase 1	Trumbull Creek Phase 2	Calamus Swamp	ORW
Ruderals							
<i>Bidens cernua</i>		X	X				
<i>Cyperus</i> spp.			X		X		
<i>Echinochloa</i> spp.	X						
<i>Eleocharis obtusa</i>		X	X		X		
<i>Polygonum pensylvanicum</i>	X		X	X			
Interstitial							
<i>Alisma plantago-aquatica</i>			X		X		
<i>Juncus effusus</i>		X		X	X		
<i>Juncus torreyi</i>					X		
<i>Leersia orizoides</i>		X	X	X	X		X
<i>Pontederia cordata</i>				X			
<i>Scirpus cyperinus</i>	X			X	X		
Matrix							
<i>Decodon verticallus</i>						X	
<i>Phalaris arundinacea</i>	X				X		
<i>Phragmites australis</i>							X
<i>Polygonum hydropiperoides</i>	X	X	X	X			
<i>Schenoplectus tabernaemontani</i>			X				X
<i>Scirpus fluviatilis</i>						X	X
<i>Scirpus pungens</i>			X				
<i>Sparganium americanum</i>	X			X	X		
<i>Sparganium eurycarpum</i>			X			X	X
<i>Typha</i> spp.	X		X		X	X	X

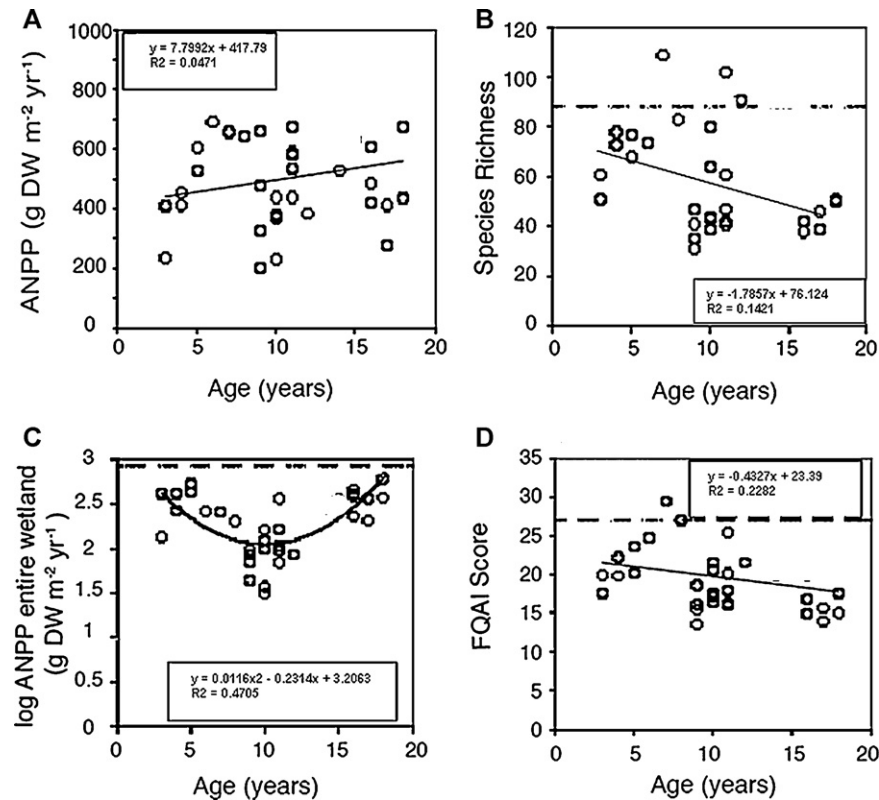


Fig. 2. Scatter plots of (A) aboveground net primary productivity of emergent plant zones (ANPP), (B) species richness, (C) ANPP over the entire wetland, and (D) floristic quality assessment index (FQAI) score with age for 5 mitigation bank wetlands. All wetlands were sampled for three years. Dashed lines in (B)–(D) indicate reference wetland conditions.

the highest FQAI scores, whereas high spring, low fall wetlands had some of the lowest scores. Continuously flooded wetlands had the largest range of species richness across sites, while pulsed wetlands had all higher values of species richness. Additionally, significant differences were found between location in Ohio and FQAI scores ($P < 0.001$), number of communities ($P < 0.001$), species richness ($P = 0.011$), and CDI ($P = 0.02$). Overall, wetlands located in the northern portion of Ohio had higher FQAI scores, number of communities, and species richness. Finally, significant differences were found between the type of wetland (mitigation vs. reference) and WANPP ($P < 0.001$), log ANPP over the entire wetland ($P = 0.013$), FQAI score ($P = 0.042$), and species richness ($P < 0.001$). WANPP, log ANPP over entire wetland, FQAI scores and species richness tended to be higher in the natural and research wetlands than wetlands created for mitigation. There was no statistical difference between mitigation reference wetlands in terms of number of communities present, H , and CDI. While, on average, the mitigation wetlands had lower WANPP, FQAI, species richness, and number of dominant communities, two of the mitigation sites in northern Ohio (Sandy Ridge and Trumbull Creek Phase I) had species richness, FQAI scores, and number of dominant plant communities similar to those found in the reference wetlands.

3.3. Redundancy analysis

The Monte Carlo test found $P = 0.002$ for both the first canonical axes. This suggests that species data are not independent of the environmental variables. For the RDA, the first four axes explained less than 10% of the variance. From the triplot (Fig. 3), the sites separated out with plots from Trumbull Creek Phase 1 and Sandy Ridge on the right hand side of the graph and Hebron, Slate Run, Trumbull

Creek Phase 2 South, Trumbull Creek Phase 2 North, ORWRP, and Calamus Swamp on the left hand side of the graph. Only species with greater than 20% dominance are displayed on the triplot.

Area of emergent vegetation was highest in Sandy Ridge and Trumbull Creek Phase I. These two sites also had the largest

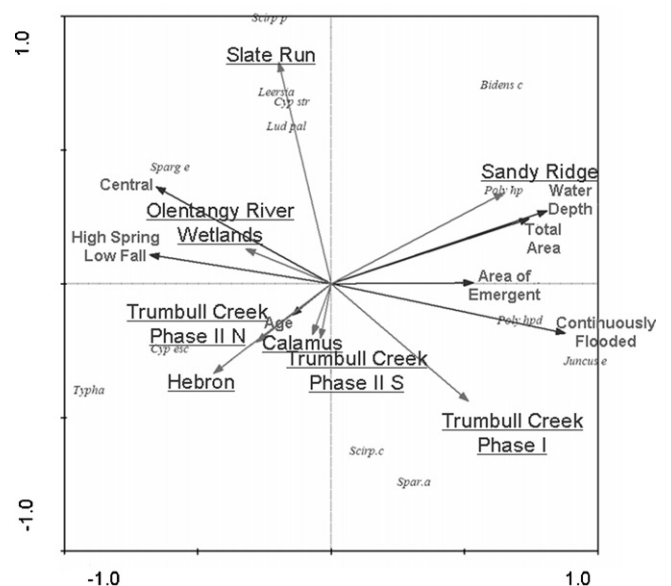


Fig. 3. Redundancy analysis (RDA) triplot of species, environmental variables, and mitigation bank sites. Only the most dominant species are displayed on the triplot, but all species went into calculating the RDA. Species names are italicized and mitigation bank site are underlined.

total area and deepest standing water depths. Species common to these sites were *Juncus effuses*, *Polygonum hydropiperoides*, *Scirpus cyperinus*, *Sparganium americanum*, *Bidens cernua*, and *Polygonum hydropiper*. The youngest site, Trumbull Creek Phase II N seems to be following a similar path as Hebron, the oldest mitigation site. The other side of the youngest site, Trumbull Creek Phase II S seems to be developing along a similar trajectory as Calamus Swamp, the natural reference wetland, based on environmental variables and dominant species at both sites. Vegetation typical of these sites includes *Typha* spp., *Cyperus esculentus*, and *Scirpus cyperinus*. The ORWRP is somewhat similar to Hebron, Calamus Swamp, and both Trumbull Creek Phase II sites. Vegetation characteristic of the ORWRP include *Typha* spp. and *Sparganium eurycarpum*. Slate Run seems to be the most unlike the other sites. Species common here include *Scirpus pungens*, *Leersia oryzoides*, *Cyperus strigosus*, and *Ludwigia palustris*.

4. Discussion

4.1. Structure

Within the mitigation bank wetlands, the vegetation structural characteristics (species richness, FQAI) of the younger sites were on par with the natural and created reference wetlands. The older sites, however, tended to have lower species richness and FQAI than the younger sites. The number of communities, CDI, and *H* for all mitigation wetlands was similar to the reference sites. Other studies have found that mitigation wetlands tend to have high species richness during the monitoring period, but species richness begins to decline with age (Fennessey and Roehrs, 1997; Campbell et al., 2002; Balcombe et al., 2005; Gutrich et al., 2009) and that indicators based on species composition were high shortly after wetland creation, but decreased over time (Matthews, 2008). Data suggests this is occurring along the time gradient in the mitigation bank wetlands used for this study. Caution should be taken when examining multiple sites along a time gradient due to variation in construction techniques and goals.

One explanation for a decrease in species richness and associated parameters over time is Connell's (1978) intermediate disturbance hypothesis. This hypothesis states that the highest diversity of a community is maintained by intermediate disturbances. On the opposing ends of the disturbance continuum, both high and low disturbance will result in low community diversity. With wetland creation, the ecosystem start out highly disturbed with little to no species present and with time become less and less disturbed until reaching a state of equilibrium or another disturbance occurs. Results from this study suggests that peak species richness or the time of intermediate disturbance occurs somewhere between the 4- and 7-year mark after wetland creation. This could be problematic when determining the success of a mitigation bank wetland since many parameters rely on structural vegetation characteristics and monitoring tends to only last 5 years.

Since peak species richness seemed to occur between the 4- and 7-year mark, monitoring time frames of at least 10 years may be more appropriate to capture declines in structural characteristics after the initial peak than current monitoring time frames. As far as these study sites go, additional monitoring is needed to determine what changes, if any, there will be in species richness and related parameters but, since the two mitigation sites are currently higher than the reference wetlands the values may stabilize near those found in the reference sites.

FQAI scores at the mitigation bank sites and the reference sites fell within the range of scores (11–34) found in other studies examining marsh habitats in Ohio (Andreas et al., 2004; Lopez

and Fennessey, 2002). The reference wetlands tended towards the middle and top of this range while the mitigation sites fell throughout. This would suggest that Trumbull Creek Phase I is a relatively good quality wetland given its high FQAI score, whereas the older and the younger wetlands tend to be of lower quality.

4.2. Function

Within the mitigation sites, WANPP increased with age, but was significantly lower in the reference sites. Of the sites used, all of the reference sites, as well as the oldest mitigation bank (2010) had high WANPP, most likely due to the high amount of *Typha* spp. present in the wetlands. Primary productivity is likely dependent on the types of species present, not necessarily on the number of species present. As an example, a wetland dominated by *Phragmites australis* would likely have high primary productivity, but low species richness due to its tendency to form monocultures. Windham (2001) found that *P. australis* had a peak productivity of almost 2000 g DW m⁻² yr⁻¹. This is approximately double what was found at the most productive site used in this study. Since high ANPP is typical of marshes dominated by *Typha* spp. and invasive species, high ANPP at a mitigation site may not be desirable considering monitoring goals that in some cases call for less than 5% invasive species at the sites (Davey, 2007a,b). On the other end of the spectrum, low ANPP is not necessarily desirable since improper hydrology can result in low plant survivorship and thus low NPP (Fraser and Karnezis, 2005). Thus the current amount of ANPP, especially at the middle age range, of the mitigation banks may be desirable.

Functional groups of the dominant species tended towards matrix species as age of the wetland increased. Newly created wetlands, where no or minimal planting has occurred tend to see a successional pattern where ruderals or annual species are the first to dominate at the site. These ruderal species eventually give way to the perennial interstitial and matrix species (Matthews and Endress, 2010). This pattern of succession can be seen across the mitigation bank wetlands with the older wetlands having a mix of ruderals, interstitial, and matrix species, and the younger sites have a higher percentage of interstitial species than matrix. This suggests that all of the mitigation sites are still maturing functionally, but that the older the sites are the closer the functional groups of the dominant species resemble the reference wetlands. This is not surprising given that other studies have found that at about the 20-year mark, mitigation wetlands were reaching a state of equilibrium as far as vegetation goes and that vegetation in mitigation wetlands was similar to natural reference wetlands (Atkinson et al., 2005; Balcombe et al., 2005; Gutrich et al., 2009; Spieles, 2005).

4.3. Mitigation bank and reference wetlands comparison

Looking at a direct comparison of function to structure, in this case productivity to diversity, the reference sites were high productivity, high diversity wetlands in comparison to the sites used in this study (Fig. 4). Of the mitigation sites examined, Trumbull Creek Phase I site was also a high productivity, high diversity wetland. Sandy Ridge and Trumbull Creek Phase II tended to be high diversity, low productivity wetlands, while Hebron and Slate Run were mainly low diversity, low productivity wetlands. Trumbull Creek Phase I most resembles the reference wetlands used in this study. Some studies examining the relationship of species richness to productivity in plants have found that both low and high productivity sites tend to have low species richness, while intermediate productivity sites had the highest species richness (Grime, 1979; Tilman, 1986). Of the sites in this study, the ORW, Trumbull Creek Phase I, and Sandy Ridge fall somewhere in this intermediary

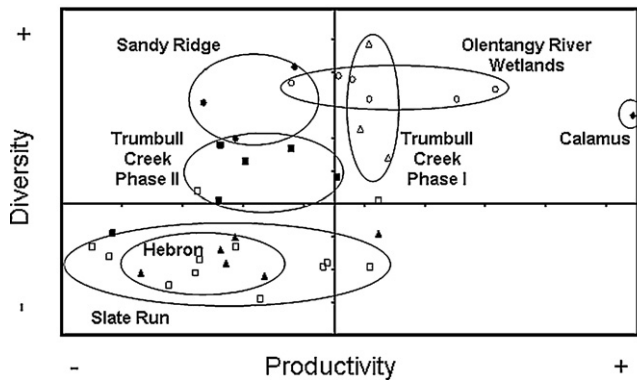


Fig. 4. Diversity–productivity scatter plot divided into four categories for mitigation wetland banks and reference wetlands. The reference wetlands (Olentangy River Wetlands and Calamus Swamp) and one mitigation bank (Trumbull Creek Phase I) were high diversity, high productivity wetlands. Two mitigation wetlands (Sandy Ridge and Trumbull Creek Phase II) were high diversity, low productivity wetlands. Hebron and Slate Run, two of the older mitigation sites, were low diversity, low productivity systems.

productivity range and thus have the highest diversity of the sites. ANPP at these three sites falls within the range of approximately $450\text{--}750\text{ g DW m}^{-2}\text{ yr}^{-1}$ when weighted by community area within the zone of emergent vegetation. Based on the findings of this study regarding ANPP and species richness, diversity is highest under median levels of ANPP, which could be useful in the regulation process when determining target goals in created freshwater marshes to allow for optimum productivity and diversity at the sites.

The younger mitigation banks are structurally more similar to the reference wetlands than are the older mitigation banks. This may be due to better initial construction techniques of the newer sites. The younger sites had more detailed and better thought out goals than the older sites (such as specifics about water depths, allowable % invasive species cover, % cover of specific wetland habitat/wetland communities, and types of specific wetland habitat) (Davey, 2007c), probably due to a combination of learning on the part of the mitigation bank creator, as well as stricter requirement established for wetland mitigation. From a functional standpoint, none of the mitigation sites were statistically similar to the reference sites, but the older mitigation sites were more similar than the younger sites. This suggests that the sites are heading towards functional similarity with the reference wetlands and may need more time to mature before determining functional success.

5. Conclusions

The younger mitigation sites were structurally similar to the reference wetlands while none of the mitigation sites were quite on the same level functionally as the reference sites. Only time will tell if these younger wetlands will remain structurally similar to the reference wetlands and if the mitigation sites will develop functional characteristics similar to those of the reference wetlands. Based on the results of this study, there are three recommendations for improving the restoration/creation and monitoring practices of mitigation bank wetlands:

- Extend monitoring to at least 10–15 years after creation to allow the structural characteristics of the wetland to stabilize before determining if the mitigation project was a success.
- Monitor trends in the succession of functional groups to help determine if the mitigation wetlands are functionally equivalent to natural wetlands.

- Aim for median levels of ANPP within emergent zones for enhanced diversity.

The use of non-destructive ANPP and functional group classifications as monitoring tools requires little additional time in the field and can be derived mainly from currently assessed structural parameters, particularly when regression equations for biomass determination have already been established. Additional research is needed to examine more in-depth the roles of individual species within these sites as well as the environmental variables likely associated with the specific plant assemblages at each site and how this may influence the success of a mitigation project.

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Appendix A. Regression equations

Regression equations created for dominant species. y —biomass per stem and x —average height of the stems within a sampling plot. After using the equation to find y , y is then multiplied by the number of stems in a plot to estimate ANPP of the species. Other equations were obtained from Johnson (1998) and Muzika et al. (1987).

Species	Equation	R^2
<i>Biddens cernua</i>	$y = 0.0084x + 0.0139$	$R^2 = 0.86$
<i>Decodon verticallus</i>	$y = 0.886x - 71.968$	$R^2 = 0.85$
<i>Eleocharis a.</i>	$y = 0.0065x - 0.2201$	$R^2 = 0.77$
<i>Eleocharis obtusa</i>	$y = 0.001x - 0.0025$	$R^2 = 0.67$
<i>Hypericum m.</i>	$y = 0.0136x - 0.4045$	$R^2 = 0.77$
<i>Juncus effusus</i>	$y = 0.0038x$	$R^2 = 0.73$
<i>Penthorum sedoides</i>	$y = 0.0082x - 0.002$	$R^2 = 0.98$
<i>Phalaris arundinacea</i>	$y = 0.0103x - 0.0197$	$R^2 = 0.97$
<i>Phyla lanceolata</i>	$y = 0.0096x - 0.013$	$R^2 = 0.93$
<i>Scirpus cyperinus</i>	$y = 0.0933x - 2.0333$	$R^2 = 0.73$
<i>Scirpus pungens</i>	$y = 0.0034x + 0.1135$	$R^2 = 0.82$
<i>Typha</i> spp.	$y = 0.0472x + 9.2366$	$R^2 = 0.75$

Appendix B. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.ecoleng.2011.11.016.

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