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Community Structure and Quality After 10 Years in Two Central Ohio Mitigation Bank Wetlands

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Abstract We evaluate two 10-year-old mitigation bank wetlands in central Ohio, one created and one with restored and enhanced components, by analysis of vegetation characteristics and by comparison of the year-10 vegetation and macroinvertebrate communities with reference wetlands. To assess different measures of wetland development, we compare the prevalence of native hydrophytes with an index of floristic quality and we evaluate the predictability of these parameters in year 10, given 5 years of data. Results show that the mitigation wetlands in this study meet vegetation performance criteria of native hydrophyte establishment by year 5 and maintain these characteristics through year 10. Species richness and floristic quality, as well as vegetative similarity with reference wetlands, differ among mitigation wetlands in year 1 and also in their rate of change during the first 10 years. The prevalence of native hydrophytes is reasonably predictable by year 10, but 5 years of monitoring is not sufficient to predict future trends of floristic quality in either the created or restored wetland. By year 10, macroinvertebrate taxa richness does not statistically differ among these wetlands, but mitigation wetlands differ from reference sites by tolerance index and by trophic guild dominance. The created wetland herbivore biomass is significantly smaller than its reference, whereas detritivore biomass is significantly greater in the created wetland and smaller in the restored wetland as compared with respective reference wetlands. These analyses illustrate differences in measures of wetland performance and contrast the monitoring duration necessary for legal compliance with the duration required for development of more complex indicators of ecosystem integrity.

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Introduction

Wetland mitigation is a firmly established tenet of American environmental policy, and although critics have identified numerous examples of mitigation gone awry (Race and Fonseca 1996, NRC 2001, Turner and others 2001, Zedler 2004), the number of replacement wetlands that meet predetermined criteria for regulatory approval is growing (NRC 2001). One procedural innovation intended to improve the quality of replacement wetlands is mitigation banking—the establishment of large wetlands as prior compensation for multiple permitted alterations to existing wetlands (Federal Guidance 1995, Tabatabai and Brumbaugh 1998). Mitigation banking has grown dramatically in the past two decades and offers apparent economic, ecological, and regulatory advantages over one-to-one mitigation (NRC 2001, ELI 2002). The ecological state of mitigation banks has been systematically evaluated only recently (Spieles 2005), showing that plant community attributes vary widely by mitigation method (i.e., created, restored, or enhanced), by geomorphic setting, and by wetland age. Spieles (2005) also demonstrates that our understanding of mitigation bank development is somewhat limited by the regulatory process, which generally requires only 5 years of annual monitoring for approval. Beyond 5 years, there are few data and hence we have an imprecise picture of the succession and long-term functionality of mitigation banks. Here, we intend to clarify temporal changes in community structure and indicators of biotic integrity through evaluation of two 10-year-old mitigation banks in central Ohio. These mitigation banks differ by mitigation method and location but are similar in construction, performance standards, and monitoring regime and thus provide a case for comparative longitudinal analysis.

Mitigation banks typically take the form of created, restored, or enhanced wetlands. A created wetland is one established where no wetland formerly existed. A restoration is defined as the re-establishment of a particular wetland type where it occurred prior to manipulation, and an enhancement is intended to increase particular values or functions of a wetland currently in existence. Mitigation bank wetlands, under the auspices of a mitigation banking review team, must meet particular performance objectives within a designated time frame in order to be approved for sale to clients with mitigation obligations. Performance standards may include measures of hydrologic regime, soil development, or wildlife but often are heavily dependent upon, or even exclusively composed of, vegetation characteristics (Breaux and Serefiddin 1999, Streever 1999). Monitoring programs are designed to provide data for the regulatory decision-making process and are thus limited in scope. The criteria used to judge success or failure are characteristically structural in nature, intended to ensure that the replacement wetland has the components of a jurisdictional wetland in place prior to approval. In a regulatory sense, this arrangement is both logical and expeditious. In an ecological context, however, it is inadequate, leaving unanswered questions about the quality of different types of mitigation banks and their long-term fate.

We address two aspects of mitigation bank development. First, we consider questions of successional trajectory in created, restored, and enhanced wetlands. Trajectory is the hypothetical path of maturation and changing function a created or restored ecosystem might follow as it undergoes succession (Aronson and Le Floc'h 1996). In reality, successional trajectory may be neither smooth nor rapid, nor toward a particular end point (Zedler and Callaway 1999). Nonetheless, the concept of trajectory has regulatory and ecological significance as a predictor for the degree to which created or restored systems exhibit the characteristics of natural systems over time (Mitsch and Wilson 1996, Simenstad and Thom 1996). In its simplest form, mitigation bank trajectory can be assessed by comparing ecological parameters with performance standards in created, restored, and enhanced wetlands as they age. Previous research (Spieles 2005) shows that some mitigation bank wetlands undergo a period of rapid species turnover in the years after construction or enhancement, as expected in newly disturbed ecosystems (Odum 1989, Mitsch and others 1998). The 5-year monitoring regime gathers data on this period of self-organization but terminates just as the variability in the transitional plant community begins to decrease (Spieles 2005). Furthermore, there is compelling evidence that created, restored, and enhanced systems develop at different rates (Spieles 2005), and thus an important aspect of trajectory may be elucidated by considering different types of mitigation bank wetlands over longer successional periods. Successional trajectory may also be evaluated by comparing replacement wetlands with mature, high-quality wetlands of similar hydrogeomorphic character (Brinson and Rheinhardt 1996). The use of reference wetlands for evaluating mitigation wetlands is not a new idea (Wilson and Mitsch 1996, Fennessy and others 1998b), but longitudinal comparisons of mitigation banks with reference wetlands are absent in the literature.

The second aspect of mitigation banks we examine concerns the metrics by which they are typically assessed. Perhaps because of the ambiguity of functional assessment, and almost certainly for ease of measurement, most mitigation bank performance standards are decidedly structural and seldom include measures of processes, such as nutrient cycling, community energetics, trophic dynamics, or disturbance regime. There is value in structural metrics, for community attributes like species diversity and community complexity are inextricably linked with function (Dobson and others 1997). However, measures of diversity or complexity alone give an incomplete account of biotic integrity. Biotic indices have long been used to evaluate the quality of lakes and streams (Karr and Dudley 1981, Karr 1991), but such integrative assessments have been slower to develop for wetlands (Danielson 1998). Recently, biotic indices have been used to evaluate wetland quality with a number of indicator communities, including plants (Andreas and Lichvar 1995, Wilcox 1995), macroinvertebrates (Kashian and Burton 2000, Spieles and Mitsch 2000), fish (Simon 1998), amphibians (Micacchion 2002), and birds (USEPA 2002). Measures of biotic integrity have not typically been part of mitigation bank performance assessments. We compare the structural measures of vegetation mandated for two mitigation banks with metrics of floristic and macroinvertebrate community quality to evaluate their respective developmental time frames.

The objective of this study is to characterize the post-approval development of two mitigation banks not only in relation to their respective vegetation performance standards but also with indices of biotic integrity and trophic structure. In particular, we assess trajectory by analyzing the predictability of these systems: given 5 years of

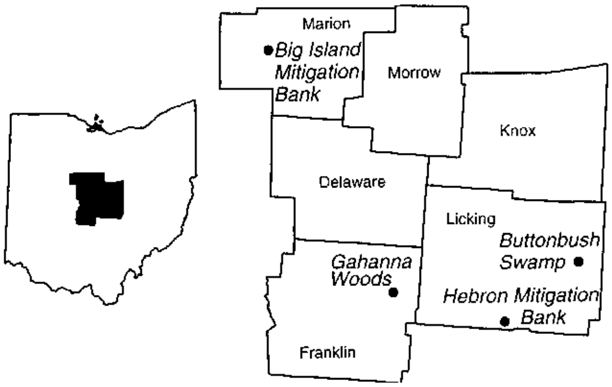

Fig. 1 Location of the created Hebron Mitigation Bank and its reference wetland, Buttonbush Swamp, and the restored and enhanced Big Island Mitigation Bank and its reference at the Gahanna Woods State Nature Preserve in central Ohio

vegetation data, how predictable is the year-10 plant community? Additionally, we test the hypothesis that these 10-year-old systems support plant and macroinvertebrate communities that are similar in structure and quality to reference wetlands. Finally, with various measures of the plant and macroinvertebrate communities, we compare the monitoring duration necessary for jurisdictional compliance with the duration required for equivalence in biotic integrity and trophic distribution. With these analyses, we seek to compare measures of wetland performance and to assess means by which these replacement wetlands might be effectively evaluated.

Methods

Site Descriptions

Created mitigation bank

The Hebron Mitigation Bank is a wetland of approximately 14 ha in south-central Licking County, Ohio (Fig. 1) that was established in 1993 as prior compensation for authorized wetland impacts in central Ohio. The wetland was designed as a depressional system with the construction of low berms and removal of drainage tile across an agricultural area of existing low topography. The wetland was constructed on a Luray silty clay loam, which by 2004 had a bulk density (mean ± SE) of 0.95 ± 0.05 g/cm^3 and a percent organic matter loss on ignition (Carter 1993) of 9.44 ± 0.68 in the upper 10 cm. Approximately 3700 trees and shrubs were planted in 1993, but no herbaceous plants were introduced. Fourteen permanent quadrats (1 m × 2 m) were established for vegetation assessment, which occurred annually through 1998. The hydrologic sources are primarily seasonal precipitation and surface runoff, and thus the hydrologic regime is dominated by seasonal vertical fluctuation. The site supports a mixture of vegetation types, predominantly emergent, floating, and submerged but also including scrub-shrub and forested habitats. Vegetation performance standards for this mitigation bank require that 1) non-wetland plant species be replaced with hydrophytic species, 2) the density of hydrophytic plants increase over time, 3) the plant species composition be predominantly native, and 4) planted woody species show sufficient survival by the end of the 5-year monitoring period (Envirotech 1995a). Qualitative goals for hydrologic regime and wildlife, also part of the monitoring plan, are not considered here. The site is now an Ohio State Wildlife Area managed by the Ohio Department of Natural Resources Division of Wildlife.

As a reference site for the Hebron Mitigation Bank we analyze Buttonbush Swamp, located in the Blackhand Gorge State Nature Preserve in Licking County, Ohio (Fig. 1). This wetland, a 1-ha depressional emergent marsh dominated by buttonbush (*Cephalanthus occidentalis* L.) and spatterdock (*Nuphar advena* (Ait.)Ait. f.), is a Category 3 (highest quality) wetland according to the Ohio Environmental Protection Agency Rapid Assessment Method (Mack 2001). It was formed as a borrow pit on Orrville silt loam from which drainage was blocked by construction of a railroad berm in 1850. By 2004, the top 10 cm of wetland substrate had a bulk density of 0.75 ± 0.04 g/cm^3 and was 11.5 ± 1.12% organic matter. Although it is not a natural wetland, its similarities in geomorphic setting and hydrologic regime, its relative age, and its high quality make Buttonbush Swamp a suitable reference for the Hebron Mitigation Bank.

Restored and enhanced mitigation bank

Big Island Mitigation Bank, located in Marion County, Ohio (Fig. 1) was established in 1994. It consists of 98 ha of restored wetland conterminous with 41 ha of enhanced wetland, constructed with low dams and tile removal on a site that formerly supported agriculture as well as sedge meadow, scrub-shrub, and forested areas. Water is supplied by seasonal precipitation, surface runoff, and groundwater interflow, showing a typical spring high and late autumn low hydrologic regime. This mitigation bank is constructed predominantly on a Latty silty clay with a 2004 bulk density of 1.07 ± 0.07 g/cm^3 and 7.88 ± 0.51% organic matter in the top 10 cm. Approximately 3000 shrubs and trees were planted on site at the time of construction. The enhanced wetland is forested, and there is significant woody cover over much of the restored section. Specific vegetation performance standards mandate 1) that a minimum of 50% of all dominant plant species within the wetland have an indicator status of obligate (OBL) or facultative wetland (FACW) according to Reed (1988), 2) a wetland plant community dominated by native species, and 3) that planted species show "some contribution to the diversity of the site" by the end of the 5-year monitoring period (Envirotech 1995b). Fifty permanent sampling quadrats (1 m × 2 m and 5 m × 5 m) were established (34 in the restored wetland and 16 in the enhanced wetland) and monitored annually for the first 5 years for regulatory compliance. Like the Hebron Mitigation Bank, the Big Island Mitigation Bank is now an Ohio State Wildlife Area.

The reference wetland for the Big Island Mitigation Bank is also a Category 3 wetland. The Gahanna Woods State Nature Preserve in Franklin County, Ohio (Fig. 1) encompasses 22 ha, of which approximately 2 ha are seasonally flooded swamp forest. The substrate is predominantly Carlisle muck, with a 2004 bulk density of 0.97 ± 0.07 g/cm^3 and 9.18 ± 0.65% organic matter in the top 10 cm. Dominant wetland trees include green ash (*Fraxinus pennsylvanica* Marsh), silver maple (*Acer saccharinum* L.), American elm (*Ulmus americana* L.), and pin oak (*Quercus paulstris* Muench.), with an understory of spicebush (*Lindera benzoin* (L.) Blume), buttonbush (*Cephalanthus occidentalis* L.), and diverse herbaceous vegetation. The high quality and extent of woody cover make Gahanna Woods an appropriate reference for the Big Island Mitigation Bank.

Data Collection

Vegetation

We obtained the annual mitigation bank monitoring reports, years 1 through 5, for the Hebron Mitigation Bank and Big Island Mitigation Bank from the United States Army Corps of Engineers, Huntington District. These reports, compiled by Envirotech Consultants, Inc. of Somerset, Ohio and by Geoenvironmental Consultants, Inc. of Westerville, Ohio, include quadrat-specific percent cover of vascular plants. In summer 2004, we replicated the vegetation monitoring procedures at these sites to analyze the plant communities 10 years after construction. Additionally, we established 15 1-m × 2-m quadrats in both reference wetlands by choosing random locations within cells of a pre-established grid. At each quadrat, we note the percent cover of each plant species and identify each according to Crow and Hellquist (2000), Knobel (1980), and Brown (1979) for comparison with similar lists compiled by consultants in years 1 through 5 of the mitigation bank wetlands (Appendix Table A1). We denote each species as native or non-native and assign a coefficient of conservatism value according to Kartesz and Meacham (1999) and Andreas and others (2004). The vegetation communities are compared among sites with a proportional similarity index, $P = \Sigma \text{ minimum}(P_{1i}, P_{2i}) \times 100$, where P equals the percentage similarity between sites 1 and 2, P_{1i} is the frequency of occurrence of species i in community sample 1, and P_{2i} is the frequency of occurrence of species i in community sample 2 (Krebs 1989).

For each quadrat of each wetland, we calculate the prevalence index (PI), a cover-weighted average of the indicator status of all plants present (Wentworth and others 1988). We assign a region-specific indicator value to each species reported in each monitoring report according to Reed (1988), with obligate wetland plants (OBL) = 1.0, facultative wetland plants (FACW) = 2.0, facultative plants (FAC) = 3.0, facultative upland plants (FACU) = 4.0, and upland plants (UPL) = 5.0. No additional weights are given for + or − designations. As a measure of plant community quality, we use the floristic quality assessment index (FQAI). Originally developed for the Chicago region by Wilhelm and Ladd (1988), this index integrates the presence of non-native plant species with the coefficient of conservatism for native species. Coefficients of conservatism are regionally specific numeric rankings of the capacity of each plant species to tolerate environmental perturbation, ranging from zero for invasive species and all non-native species to 10 for native species that are tolerant only to a narrow, specialized niche (Andreas and Lichvar 1995). We use the plant species list developed for Ohio (Andreas and others 2004) to compute an adjusted FQAI for each quadrat of each wetland as follows:

$$\text{FQAI} = \left(\frac{\overline{C} \times \sqrt{N}}{10 \times \sqrt{N + A}}\right) \times 100$$

where C = the coefficients of conservatism per site, N = the number of native species, and A = the number of non-native species (modified from Andreas and others 2004).

To compare plant species richness among mitigation wetlands, we use the same quadrat-based field data to construct a species accumulation curve for each wetland (Palmer 1990, Colwell and Coddington 1994, Ugland and others 2003). In this method, the accumulating area of each quadrat sampled is plotted against the cumulative number of new species identified. We convert this plot for each wetland to a log-log regression using the standard model of $S = cA^z$, where S is the number of species present, c is constant, A is the defined area, and z is the slope of the regression. We then let A = 10 m^2 to interpolate S, the number of plant species present in 10 m^2 of each particular wetland. To remove the confounding factor of sampling order, we randomize the quadrats through 10 iterations to obtain a mean estimate of species richness per 10 m^2 in each wetland. This method allows for the direct comparison of the number of plant species in wetlands of varying area and number of sampling quadrats without requiring frequency data.

Macroinvertebrates

We collected macroinvertebrates in the same four wetlands in the summer of 2004 by deploying aquatic funnel traps (BioQuip Inc.) in standing water near plant sampling quadrats. In each wetland, 12 traps were deployed for two 24-hour periods: one lighted with BioQuip cyalume light sticks, and the other unlighted. The total trapping effort for each wetland thus consisted of 12 lighted trap nights and 12 unlighted trap nights, with traps recovered from Hebron on 5/27 and 6/1, from Buttonbush Swamp on 6/10 and 6/11, from Gahanna Woods on 6/15 and 6/16, and from Big Island on 6/23 and 6/24. For the Big Island mitigation wetland, we did not differentiate between the restored and enhanced habitats. Each collection day, trap contents were filtered through 0.5-mm mesh netting and preserved in ethanol. After identifying macroinvertebrates to the lowest taxonomic unit possible according to Merritt and Cummins (1996), Smith (2001) and Thorp and Covich (1991), we calculate the diversity of each sampling location and for each wetland according to Simpson (1949), expressed as $1 - D$, where $D = \Sigma(P_i)^2$ and P_i is the proportion of the ith taxon of each collection event. To estimate taxa equitability, we calculate Simpson's E as the proportion of the maximum value D could assume if individuals in the community were completely evenly distributed. Equitability takes a value between 0 and 1, with 1 being complete evenness. We compare taxa in both the created and the restored/enhanced wetlands to their respective reference wetlands with a proportional similarity index (Krebs 1989).

To assess macroinvertebrate biotic integrity, we assign each taxon a general pollution tolerance score according to Adamus and Gonyaw (2001). These authors classify wetland macroinvertebrate taxa according to literature-based accounts of their sensitivity to anthropogenic stressors, with 1= very tolerant, 2 = tolerant, 3 = moderately tolerant, 4 = somewhat intolerant, and 5 = intolerant. We weight these rankings by abundance to compute a simple tolerance index for each sampling location and for each wetland (Appendix Table A2). To evaluate the representation of macroinvertebrate trophic groups, we assign each taxon to a feeding guild—detritivore, including collectors and shredders; herbivore, including macrophyte herbivores and algal grazers; or predator, including parasites—after Merritt and Cummins (1996). We compare dry biomass by trophic level, excluding taxa with multiple feeding guilds.

Statistical Analyses

Vegetation

We compare the mean prevalence index, percentage of non-native plant species, and adjusted FQAI among years 1, 5, and 10 with paired sample t-tests (SPSS 12; $\alpha = 0.05$) within mitigation bank wetlands. Mean species richness of each wetland is compared among years with independent sample t-tests. The same plant community characteristics are compared among mitigation bank wetlands at different age intervals with one-way analysis of variance ($\alpha = 0.05$). We quantify the year-to-year variability in all four plant community characteristics by calculating the variance in successive years through years 4 and 5. To test the usefulness of the 5-year data set in predicting year 10 for each category, we perform regression analysis on a log-log linearized model of the curvilinear data. We then use the linearized models to generate a 90% confidence interval for each data category in year 10. We compare the actual year 10 means for created, restored, and enhanced treatments with the predicted 90% confidence interval and with the respective reference wetlands using independent sample t-tests at $\alpha = 0.05$.

Macroinvertebrates

To assess the diversity of macroinvertebrates collected in each wetland, we use both taxa richness, expressed as the mean number of taxa collected per trap location, and taxa diversity, given as the mean Simpson diversity score per trap location. We express biological integrity with the mean tolerance index as weighted by abundance. Mean taxa richness, diversity, and tolerance score for both created and restored wetlands are compared with each other and with the respective reference wetland using independent sample t-tests at $\alpha = 0.05$. Macroinvertebrate mass by trophic guild is compared among mitigation wetlands and their respective reference wetlands with independent sample t-tests at $\alpha = 0.05$. Significant differences in both the macroinvertebrate and vegetation communities are confirmed with the nonparametric Mann-Whitney, Wilcoxon, and Kruskal-Wallis tests, but reported statistics are results of parametric analyses.

Results and Discussion

Mitigation Wetland Performance

The created, restored, and enhanced wetlands in this study all met vegetation performance standards by year 5 (Fig. 2). We express the first vegetation performance criterion, hydrophytic vegetation, as the PI, with lower values indicating a greater incidence of hydrophytes. The PI significantly decreases from year 1 to year 5 in the created wetland ($P = 0.04$, $t_{11} = 2.3$) and remain significantly lower through year 10 ($P = 0.02$, $t_9 = 2.9$). The PI of the restored wetland similarly is significantly lower in year

Table 1 Log-log linearized trend for 5 years of four vegetation characteristics in created, restored, and enhanced mitigation bank wetlands in central Ohio, along with 90% confidence interval (CI) for year 10, the actual value for year 10, and reference wetland value in 2004

	5-year trend	r^2	P	Predicted year 10 90% CI	Actual year 10 (Mean ± SE)	Reference wetland (Mean ± SE)
Created wetland						
Prevalence Index	$y = -0.28x + 0.31$	0.99	<0.01	(1.0, 1.1)	1.5 ± 0.2	1.2 ± 0.2
Richness	$y = -0.11x + 1.15$	0.19	0.46	(7.2, 16.9)	8.0 ± 0.5†	7.5 ± 0.5
Pct. non-native	$y = -0.87x + 1.16$	0.58	0.14	(0.4, 8.5)	3.2 ± 2.2†	6.0 ± 4.8
FQAI	$y = 0.32x + 1.16$	0.68	0.09	(21.3, 46.3)	16.1 ± 1.8*	36.6 ± 2.5*
Restored wetland						
Prevalence Index	$y = -0.16x + 0.36$	0.79	0.05	(1.4, 1.9)	1.6 ± 0.1†*	2.1 ± 0.1*
Richness	$y = -0.31x + 1.19$	0.69	0.08	(5.0, 11.4)	10.4 ± 1.1†	13.0 ± 1.0
Pct. non-native	$y = -0.30x + 1.05$	0.36	0.28	(2.6, 12.3)	3.0 ± 1.0†	3.4 ± 1.8
FQAI	$y = 0.20x + 1.17$	0.92	0.01	(20.3, 26.9)	28.4 ± 1.4	28.1 ± 2.0
Enhanced wetland						
Prevalence Index	$y = 0.001x + 0.25$	<0.01	0.99	(1.4, 2.2)	1.8 ± 0.1†	2.1 ± 0.1
Richness	$y = -0.50x + 0.92$	0.83	0.03	(1.7, 4.2)	11.4 ± 1.0	13.0 ± 1.0
Pct. non-native	$y = -0.26x + 1.07$	0.58	0.14	(4.1, 10.0)	1.9 ± 1.3	3.4 ± 1.8
FQAI	$y = 0.16x + 1.35$	0.77	0.05	(25.7, 39.8)	31.6 ± 2.9†	28.1 ± 2.0

†Indicates actual year 10 values that fall within 1 SE of the predicted year 10 90% CI, and *indicates that actual year 10 value differs from reference wetland value at $\alpha = 0.05$.

zation of native hydrophytes suggests that this criterion may indeed be effectively assessed in 3 to 5 years (Brown 1999, Edwards and Proffitt 2003). There is merit to applying short-term performance standards to the first few years of succession; in some ways they may be the best measure of early community establishment and are indicators of systematic development (Craft and others 2003). However, Zedler (2004) argues that standards consisting of plant species characteristics alone are inadequate measures of replacement wetlands.

The trends of species richness and floristic quality, though not part of the performance standards for these wetlands, further describe the structural development of the resident plant communities (Fig. 2; Table 1). Species richness declines from year 1 to year 5 in the restored and enhanced wetland ($P < 0.01$, $t_{18} = 5.4$), whereas species richness is highly variable in the created wetland without a clear 5-year trend. The decline in species richness during the first 5 years of the restored and enhanced wetlands is in part attributable to a disproportionately large decline in the number of non-native, facultative upland, and upland species. Non-native, facultative upland, and upland species similarly decline from years 1 to 5 in the created wetland, but there is an accompanying influx of hydrophytic vegetation. By year 10, species richness in the created wetland is estimated to be significantly lower than year 1 ($P < 0.01$, $t_{18} = 5.2$) but only marginally different in the restored and enhanced wetlands. Floristic quality also changes after year 5. The FQAI of the created wetland increases significantly by year 5 ($P = 0.02$, $t_{13} = 2.6$) before dramatically declining in year 10. The first-year FQAI is statistically higher in the restored wetland ($P < 0.01$, $t_{27} = 6.5$) and marginally higher in the enhanced wetland ($P = 0.01$, $t_{13} = 2.9$) by year 10.

The year-to-year variance of plant community characteristics in these wetlands differs by mitigation method and by age (Fig. 3). Overall, year-to year variance in all four vegetation community metrics is lowest in the enhanced wetland. Variance in plant species richness and non-native species presence declines dramatically by the third year for all three wetlands, and the PI variance similarly declines by year 5. Variance in FQAI declines by year 4 in the restored and enhanced wetlands, but FQAI remains highly variable in the created wetland through year 5. Plant species richness and floristic quality have been shown to be highly variable in young created and restored wetlands in other studies. Mulhouse and Galatowitsch (2003) found a dramatic increase in species richness from year 3 to year 12 in restored prairie wetlands, and an accompanying influx of invasive perennials consistent with a decrease in floristic quality. Reinartz and Warne (1993) similarly report an increase in plant species richness with constructed wetland age, whereas Campbell and others (2002) find that created wetlands less than 10 years old support a greater species richness than older created wetlands in Pennsylvania. Spieles (2005) reports erratic changes in the species richness of created, restored, and enhanced mitigation banks through the first 5 years. Although the variability of species richness and floristic quality are partially dependent on wetland age, then, it is likely that other factors like geomorphic setting, hydroperiod, and seed source proximity are of critical importance.

Our vegetation analyses may be applied to the common criticism that mitigation monitoring regimes are too short

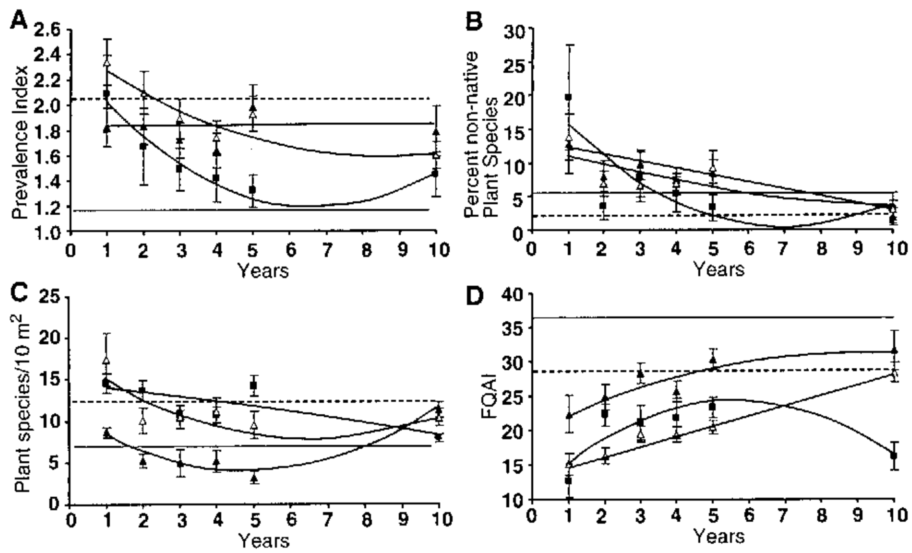

Fig. 2 Ten-year trends of (**A**) Prevalence Index, (**B**) non-native plant presence, (**C**) species richness, and (**D**) FQAI in created (squares), restored (open triangles), and enhanced (closed triangles) mitigation banks, with polynomial trendlines, over 10 years of development since establishment. Symbols indicate mean ± SE for each year in each wetland. The solid horizontal lines denote the 2004 values for Buttonbush Swamp, the reference for the created wetland. The dashed horizontal lines indicate the same for Gahanna Woods, the reference for the restored and enhanced mitigation bank wetlands

5 ($P = 0.04$, $t_{29} = 2.2$) and year 10 ($P = 0.003$, $t_{24} = 3.4$) than it was in year 1. The PI of the enhanced wetland remains statistically unchanged over the 10-year period. During the period of mandatory monitoring (years 1 through 5), the percent cover by OBL and FACW vegetation increases from 60% to 92%, 55% to 71%, and 70% to 76% in the created, restored, and enhanced wetlands, respectively. The second performance criterion requires a predominance of native vegetation. Non-native vegetation comprises an average of 19.7 ± 7.8 (mean ± SE) percent of the created wetland plant species in the first year (Figure 2b). This decreases to 3.3 ± 2.0% by year 5 ($P = 0.08$, $t_{12} = 1.9$) and is estimated to remain at this level through year 10. The non-native vegetation in the restored and enhanced wetlands does not significantly decrease from year 1 to year 5, but year 10 is estimated to be significantly lower than year 1 in both restored ($P = 0.03$, $t_{26} = 2.3$) and enhanced wetlands ($P = 0.01$, $t_{13} = 2.9$). The third vegetation performance standard common to these sites, survival of planted stock, is difficult to judge with quadrat-based data. None of the planted species in the created wetland occur in sampling quadrats in year 5, although some are clearly present at the site in year 10. Four of the 12 planted species are present in the year-5 sample of the restored and enhanced wetlands.

Our analyses indicate that these replacement wetlands remain in compliance with vegetation performance criteria through year 10. All three wetlands have been increasingly dominated by native hydrophytic vegetation since year 5 and by this measure they converge with each other by year 10. In addition, these replacement wetlands do not statistically differ in prevalence index, non-native species presence, or species richness from reference wetlands by year 10. These observations suggest that the wetlands of this study have avoided some common pitfalls of mitigation: insufficient hydrologic regime, invasion by non-native plant species, and dominance by a few aggressive plant species (NRC 2001). Other studies have similarly noted rapid influx of hydrophytes in mitigation wetlands, often within 3 to 5 years after construction (Reinartz and Warne 1993, VanRees-Siewert and Dinsmore 1996, Balcombe and others 2005a) and floristic convergence among created or restored wetlands within 3 (Mitsch and others 1998) to 12 years (Mulhouse and Galatowitsch 2003). Reinartz and Warne (1993) note an increase in percent cover by native wetland vegetation during the first 3 years, a trend that is generally seen in mitigation banks during the 5 to 7 years of monitoring for regulatory compliance (Spieles 2005). The created, restored, and enhanced wetlands in this study all have a year-5 PI and percentage of non-native plant species below the national mean for 5-year-old mitigation banks of each respective category (Spieles 2005), and thus easily achieve their performance criteria. Here, we focus not on compliance but rather on the criteria by which these replacement wetlands were approved, insight that may be gleaned from our 10-year retrospective analysis, and limitations on our understanding of mitigation bank ecology in the current regulatory framework.

As with many mitigation wetlands, these have been evaluated primarily with vegetation parameters based on the observations of a number of different field ecologists. The vegetation performance standards for these wetlands address simple trends in vegetation community structure. These performance criteria are rather vague and not particularly stringent—referring only to an increase in dominance by native hydrophytic vegetation over time—but are similar to the performance standards of many mitigation wetlands (Breaux and Serefiddin 1999). The rapid coloni-

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	5-year trend	r^2	P	Predicted year 10 90% CI	Actual year 10 (Mean ± SE)	Reference wetland (Mean ± SE)
Created wetland						
Prevalence Index	$y = -0.28\ x + 0.31$	0.99	<0.01	(1.0, 1.1)	1.5 ± 0.2	1.2 ± 0.2
Richness	$y = -0.11\ x + 1.15$	0.19	0.46	(7.2, 16.9)	8.0 ± 0.5†	7.5 ± 0.5
Pct. non-native	$y = -0.87\ x + 1.16$	0.58	0.14	(0.4, 8.5)	3.2 ± 2.2†	6.0 ± 4.8
FQAI	$y = 0.32\ x + 1.16$	0.68	0.09	(21.3, 46.3)	16.1 ± 1.8*	36.6 ± 2.5*
Restored wetland						
Prevalence Index	$y = -0.16\ x + 0.36$	0.79	0.05	(1.4, 1.9)	1.6 ± 0.1$^{\dagger *}$	2.1 ± 0.1*
Richness	$y = -0.31\ x + 1.19$	0.69	0.08	(5.0, 11.4)	10.4 ± 1.1†	13.0 ± 1.0
Pct. non-native	$y = -0.30\ x + 1.05$	0.36	0.28	(2.6, 12.3)	3.0 ± 1.0†	3.4 ± 1.8
FQAI	$y = 0.20\ x + 1.17$	0.92	0.01	(20.3, 26.9)	28.4 ± 1.4	28.1 ± 2.0
Enhanced wetland						
Prevalence Index	$y = 0.001\ x + 0.25$	<0.01	0.99	(1.4, 2.2)	1.8 ± 0.1†	2.1 ± 0.1
Richness	$y = -0.50\ x + 0.92$	0.83	0.03	(1.7, 4.2)	11.4 ± 1.0	13.0 ± 1.0
Pct. non-native	$y = -0.26\ x + 1.07$	0.58	0.14	(4.1, 10.0)	1.9 ± 1.3	3.4 ± 1.8
FQAI	$y = 0.16\ x + 1.35$	0.77	0.05	(25.7, 39.8)	31.6 ± 2.9†	28.1 ± 2.0

†Indicates actual year 10 values that fall within 1 SE of the predicted year 10 90% CI, and *indicates that actual year 10 value differs from reference wetland value at $\alpha = 0.05$.

zation of native hydrophytes suggests that this criterion may indeed be effectively assessed in 3 to 5 years (Brown 1999, Edwards and Proffitt 2003). There is merit to applying short-term performance standards to the first few years of succession; in some ways they may be the best measure of early community establishment and are indicators of systematic development (Craft and others 2003). However, Zedler (2004) argues that standards consisting of plant species characteristics alone are inadequate measures of replacement wetlands.

The trends of species richness and floristic quality, though not part of the performance standards for these wetlands, further describe the structural development of the resident plant communities (Fig. 2; Table 1). Species richness declines from year 1 to year 5 in the restored and enhanced wetland (P < 0.01, $t_{18} = 5.4$), whereas species richness is highly variable in the created wetland without a clear 5-year trend. The decline in species richness during the first 5 years of the restored and enhanced wetlands is in part attributable to a disproportionately large decline in the number of non-native, facultative upland, and upland species. Non-native, facultative upland, and upland species similarly decline from years 1 to 5 in the created wetland, but there is an accompanying influx of hydrophytic vegetation. By year 10, species richness in the created wetland is estimated to be significantly lower than year 1 (P < 0.01, $t_{18} = 5.2$) but only marginally different in the restored and enhanced wetlands. Floristic quality also changes after year 5. The FQAI of the created wetland increases significantly by year 5 ($P = 0.02$, $t_{13} = 2.6$) before dramatically declining in year 10. The first-year FQAI is statistically higher in the restored wetland (P < 0.01, $t_{27} = 6.5$) and marginally higher in the enhanced wetland ($P = 0.01$, $t_{13} = 2.9$) by year 10.

The year-to-year variance of plant community characteristics in these wetlands differs by mitigation method and by age (Fig. 3). Overall, year-to year variance in all four vegetation community metrics is lowest in the enhanced wetland. Variance in plant species richness and non-native species presence declines dramatically by the third year for all three wetlands, and the PI variance similarly declines by year 5. Variance in FQAI declines by year 4 in the restored and enhanced wetlands, but FQAI remains highly variable in the created wetland through year 5. Plant species richness and floristic quality have been shown to be highly variable in young created and restored wetlands in other studies. Mulhouse and Galatowitsch (2003) found a dramatic increase in species richness from year 3 to year 12 in restored prairie wetlands, and an accompanying influx of invasive perennials consistent with a decrease in floristic quality. Reinartz and Warne (1993) similarly report an increase in plant species richness with constructed wetland age, whereas Campbell and others (2002) find that created wetlands less than 10 years old support a greater species richness than older created wetlands in Pennsylvania. Spieles (2005) reports erratic changes in the species richness of created, restored, and enhanced mitigation banks through the first 5 years. Although the variability of species richness and floristic quality are partially dependent on wetland age, then, it is likely that other factors like geomorphic setting, hydroperiod, and seed source proximity are of critical importance.

Our vegetation analyses may be applied to the common criticism that mitigation monitoring regimes are too short

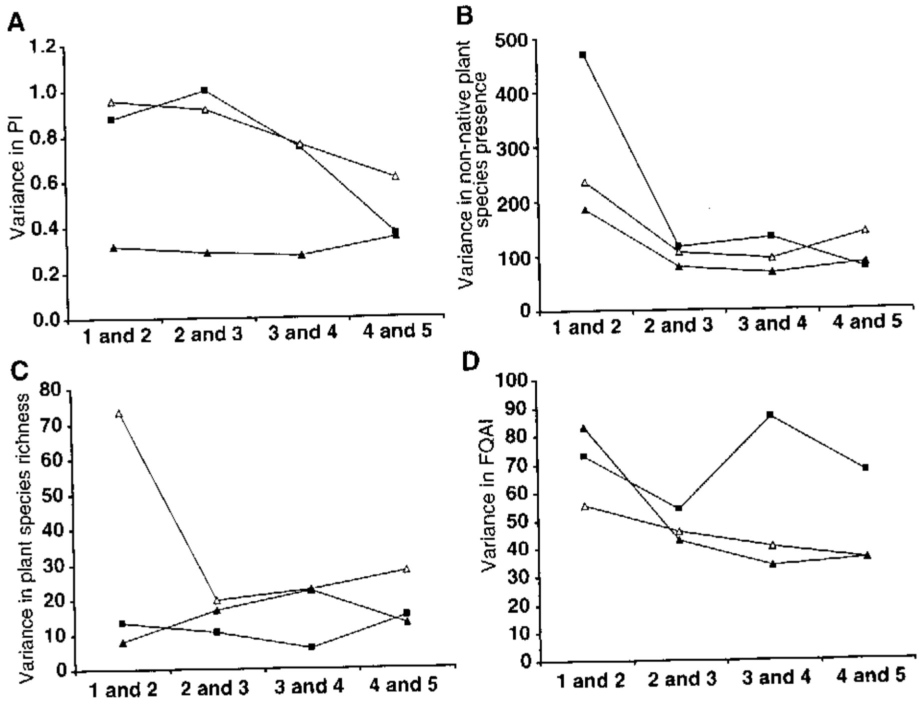

Fig. 3 Year-to-year variance of (**A**) Prevalence Index, (**B**) non-native plant presence, (**C**) Species richness, and (**D**) FQAI in created (squares), restored (open triangles), and enhanced (closed triangles) mitigation banks in central Ohio. PI indicates Prevalence Index and FQAI indicates Floristic Quality Assessment Index

to adequately assess parameters beyond vegetation structure or to characterize long-term successional trajectories (Morgan and Roberts 1999, NRC 2001). First, one may ask whether the standard 5-year monitoring scheme is sufficient to characterize the establishment of a wetland. If native hydrophyte colonization is the sole criterion, 5 years appears to be sufficient for the wetlands in this study. Floristic quality, however, may not be so easily characterized in the short term. The FQAI of the created and restored wetlands had not reached that of their respective reference wetlands by the end of the 5-year monitoring period, and the created wetland remained particularly variable. In this case, the FQAI reveals something that the performance standards do not, and because it is designed as an assessment of ecosystem condition (Fennessy and others 1998a, Andreas and others 2004), the FQAI may provide a more meaningful measure of biotic integrity than simple vegetation inventories do alone. Lopez and Fennessy (2002) report a strong correlation between FQAI and wetland condition, as defined by surrounding land cover and buffer characteristics, hydrologic alteration, and proximity to other wetlands. These authors interpret the FQAI as a "measure of environmental factors that maintain and control plant communities" (Lopez and Fennessy 2002). Our results show that changes in the FQAI over time do not necessarily correspond with changes in individual plant community metrics.

We also note that the plant communities of created, restored, and enhanced mitigation banks in this study change at different rates. The created wetland experienced a departure from the first-year plant community that is both more rapid and more complete than either the restored or the enhanced wetland by year 5 (Fig. 4b). Our study also shows that species richness, non-native prevalence, and floristic quality (Fig. 2) as well as vegetative similarity with reference wetlands (Fig. 4a) differ not only at the outset but also in the rate at which they change during the first 10 years. It seems logical that the time frame for performance standard evaluation should be set according to mitigation method. Race and Fonseca (1996) point out the potential regulatory difficulties with tailored, rather than standard, criteria for compliance and stress that any such scheme must be based on an "acceptable mechanism." Our analyses and corresponding trends in a larger sample of mitigation bank wetlands (Spieles 2005) provide an ecological basis for such a mechanism.

Successional Trajectory

Despite the categorical success of these mitigation wetlands and their vegetative similarity to high-quality reference wetlands, the year-10 vegetation community is not precisely predictable from 5-year data. Table 1 gives the log-transformed linear model for vegetation categories of each mitigation wetland, along with the 90% confidence interval predicted for year 10 by this model. Of the 12 linear regression trendlines of 5-year data, 9 have correlation coefficients greater than $r^2 = 0.5$. Seven regression models

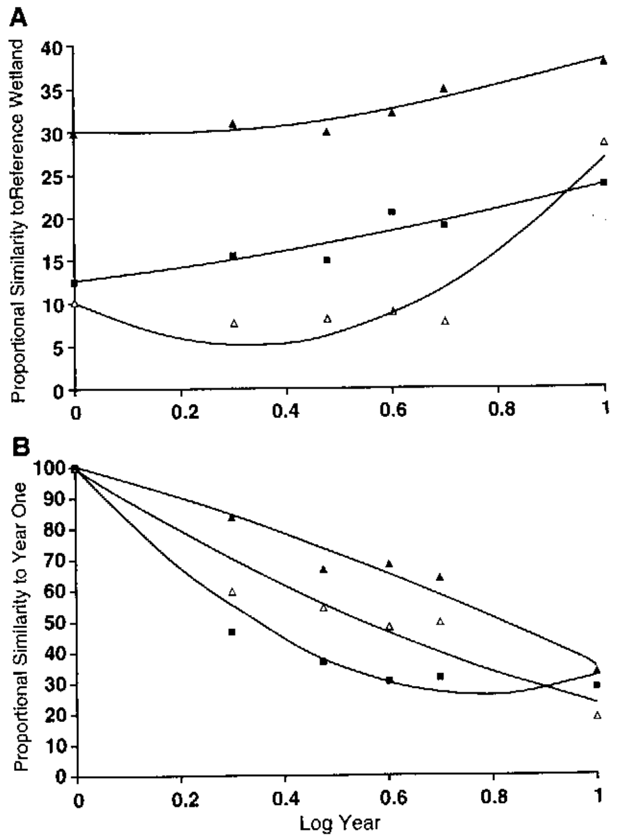

Fig. 4 Proportional similarity (after Krebs 1989) of mitigation bank plant communities as compared with (**A**) their respective reference wetlands and (**B**) with their own year-1 plant community. The created mitigation bank is indicated with squares, the restored wetland with open triangles, and the enhanced wetland with closed triangles

predict a 90% confidence interval that falls within one standard error of the actual year-10 value for that data category. These include two instances each of readily predictable PI, species richness, and percent non-native plant species. Year-10 floristic quality is predictable from 5-year data only for the enhanced mitigation wetland of this study.

Other studies have concluded that short-term patterns in created and restored wetlands are very difficult to extrapolate to functional equivalency with reference wetlands, particularly for complex parameters and stressed sites (Simenstad and Thom 1996, Zedler and Callaway 1999). Here, we assess the trajectories of wetlands in relatively unstressed conditions and conclude that the rate of successional development and our ability to estimate long-term trends with short-term data depend upon the attributes considered. In this respect, our study is in agreement with Craft and others (2003), who hypothesize that parameters of primary production in a created salt marsh may reach equivalence with reference wetlands within 5–10 years. We find the pace of plant community development to be similar in these wetlands, with the caveat that a more complex parameter like FQAI may be less predictable at the outset, particularly in created wetlands. A 5-year monitoring plan, then, may be enough to establish trends of individual measures of plant community structure in mitigation bank wetlands, but it may fall short of predicting the long-term integrity of the ecosystem. This is entirely in agreement with Zedler and Callaway (1999), who suggest that short-term mitigation monitoring data are inadequate predictors of functional equivalency with natural wetlands.

Comparison Among Mitigation Wetlands and with Reference

Comparative analyses of vegetation metrics among our created, restored, and enhanced wetlands at intervals reinforce the idea that development differs by attribute (Fig. 2). The simplest vegetation metrics—PI and percentage of non-native plant species—do not differ among the mitigation wetlands in year 1 and remain statistically similar through year 10. Species richness differs dramatically by method in each interval, with a higher richness in a different wetland in years 1 ($P = 0.02$, $F_{2,27} = 4.97$) 5 ($P < 0.01$, $F_{2,27} = 18.5$), and 10 ($P = 0.03$, $F_{2,27} = 3.90$). Floristic quality is significantly higher in the enhanced wetland during year 1 ($P = 0.04$, $F_{2,52} = 3.56$). By year 10, the created wetland has the lowest FQAI ($P < 0.01$, $F_{2,47} = 13.78$). The plant communities of all three mitigation wetlands are remarkably similar to their respective reference wetlands by year 10 (Table 1). The created wetland differs only in its significantly lower FQAI ($P < 0.01$, $t_{23} = 6.57$). The restored wetland has a lower PI than its reference ($P < 0.01$, $t_{39} = 2.02$) but does not differ in any other vegetative parameter. The enhanced wetland is statistically similar to its reference in all vegetation categories. The plant communities of all three mitigation wetlands become more similar to their reference plant communities over time (Fig. 4).

Comparative analysis of the macroinvertebrate community presents a view of development that parallels the vegetation community. By year 10, the most basic measures of the mitigation wetland macroinvertebrate community, taxa richness and diversity, do not statistically differ among these wetlands (Table 2). The wetlands differ by macroinvertebrate tolerance index, with the created wetland supporting the greatest degree of highly tolerant macroinvertebrates and the restored wetland supporting more intolerant organisms. Furthermore, our analysis of trophic distribution (Fig. 5) suggests that the mitigation wetlands differ from their respective reference sites in terms of guild dominance. The herbivore biomass of the created wetland is significantly smaller than its reference ($P = 0.05$, $t_{22} = 2.0$). Detritivore biomass is significantly greater in the created wetland than its reference ($P = 0.05$, $t_{22} = 2.03$) and smaller than the reference in the restored wetland ($P = 0.03$, $t_{22} =$

Table 2 Characteristics of macroinvertebrate communities in two mitigation bank wetlands and their respective reference wetlands in 2004

	Macroinvertebrate		Taxa/Trap	Simpson		Diversity/Trap	Tolerance Index	Proportional similarity
	Individuals	Taxa	(mean ± SE)	1 − *D*	*E*	(mean ± SE)	(mean ± SE)	
Created wetland	514	29	6.2 ± 0.6a*	0.76	<0.01	**0.55 ± 0.05a***	**2.0 ± 0.2a***	45.2
Reference	209	27	7.3 ± 0.8a	0.90	0.05	**0.74 ± 0.04b**	**2.6 ± 0.1b**	
Rest./enh. wetland	513	26	5.8 ± 0.6a*	0.50	<0.01	0.57 ± 0.07a*	**2.9 ± 0.2a****	26.6
Reference	195	23	6.1 ± 0.5a	0.85	0.03	0.69 ± 0.05a	**2.2 ± 0.1b**	

Data pairs in bold differ significantly at $\alpha = 0.05$, as indicated by letters. Asterisks indicate similarities or statistical differences between the created wetland and the restored/enhanced wetland at $\alpha = 0.05$. Proportional similarities compare the macroinvertebrate communities of each mitigation wetland with its reference wetland

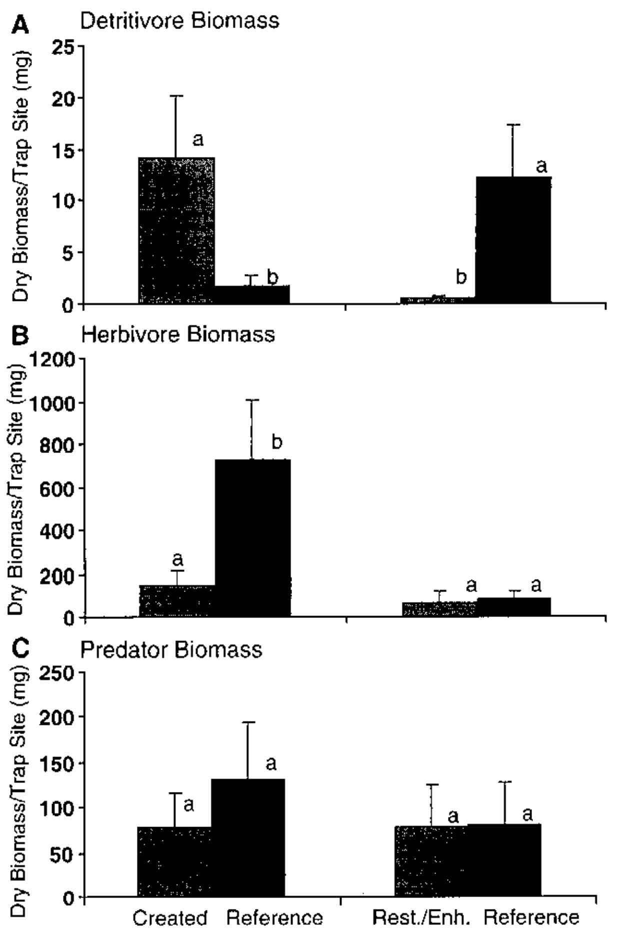

Fig. 5 Dry biomass distribution of (**A**) detritivare, (**B**) herbivore, and (**C**) predator macroinvertebrates in created and restored/enhanced mitigation wetlands along with their respective reference wetlands in 2004. Letters indicate significant differences between mitigation bank and reference site in each trophic category ($\alpha = 0.05$).Note different scales for each graph

2.3). Balcombe and others (2005b) also found similarity in macroinvertebrate richness, diversity, density, and overall biomass among 4–21-year-old mitigation wetlands and natural reference wetlands. Levin and others (1996) similarly found no species richness differences between a 4-year-old created and a natural salt marsh, while Craft and others (1999) found a greater density and richness of benthic fauna in created salt marshes 21 and 25 years old as compared with natural systems. Craft and others (1999) also report that these systems differ by macroinvertebrate feeding guild. Our data suggest that the mitigation banks of this study are equivalent to the reference wetlands in terms of macroinvertebrate richness, but they are not equivalent in community quality by year 10.

Conclusions

Our study is certainly not the first to note the developmental lag in indicators of biotic integrity or to call for increased attention to community quality in mitigation wetlands. A number of studies have indicated that basic measures of the plant community achieve similarity with natural wetlands more quickly than some biogeochemical processes in both freshwater (Erwin and others 1985, Nair and others 2001) and estuarine systems (Craft and others 1999, 2002, 2003). In part, the importance of biotic integrity depends on our definition of successful replacement: can mitigation wetlands be deemed successful without functional similarity to natural systems (Streever 2000)? Many mitigation monitoring regimes omit measures of biotic integrity and direct functional assessment (Richardson 1994) and instead use vegetation parameters relative to a reference standard as indicators of function (Brinson and Rheinhardt 1996). Here, we show in two ways that community parameters do not always develop at the same rate. First, measures of biotic integrity in our study are not predictable in the same time frame as simple structural characteristics. Second, our data suggest that the pace and predictability of development differ by mitigation method. In the absence of direct functional measures, assessment of biotic integrity may be a meaningful addition to mitigation monitoring programs.

At the heart of this issue is the purpose of performance standards and monitoring, articulated nicely by Rolband and others (1999). Most performance standards and monitoring regimes are designed to ensure that the replacement wetland meets jurisdictional criteria. This approach effectively provides information for regulatory approval, but unfortunately it generates limited data for evaluation of ecosystem integrity. Should performance standard monitoring reach beyond the identification of jurisdictional wetland characteristics? We submit that it should, and we present a case for more systemic and longer-term measures of mitigation bank wetland performance. Specifically, we recommend that short-term measures of vegetation structure be supplemented with community criteria that are more holistic, such as indices of biotic integrity. This may become feasible as regional data on species sensitivity and conservatism become more readily available. Second, we suggest that the time frame for monitoring be specific to the mitigation method, with created wetlands requiring longer and later monitoring than restored or enhanced wetlands. Third, our study reinforces the call for extended monitoring duration to better assess successional development and to better judge ecosystem integrity, even if the monitoring is post-approval. The development of meaningful indicators of quality in replacement wetlands is indeed a challenge, and we submit these recommendations as a means of advancing the understanding of replacement wetland performance.

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Appendixes

Table A1 Plant species of restored, enhanced, and created mitigation bank wetlands in central Ohio, along with two reference wetlands—Buttonbush Swamp (BB) and Gahanna Woods (GW)

Species	Indicator status	C	Restored			Enhanced			Created			Reference	
			Y1	Y5	Y10	Y1	Y5	Y10	Y1	Y5	Y10	BB	GW
Acer negundo	FAC+	3	—	—	0.4	—	—	—	7.7	1.4	9.0	3.6	0.7
Acer rubrum	FAC	2	—	—	0.4	21.0	8.5	17.0	—	—	—	7.1	53.0
Acer saccharinum	FACW	3	—	—	—	0.7	—	—	—	—	—	12.0	4.0
Agrimonia parviflora	FAC	2	—	—	—	0.3	0.8	—	—	—	—	—	—
AGROSTIS STOLONIFERA	FACW	0	—	—	6.7	—	—	—	—	—	0.5	—	—
Alisma subcordatum	OBL	2	0.7	3.2	5.0	—	—	8.4	3.5	0.7	—	0.4	—
Ambrosia artemisiifolia	FACU	0	4.5	2.7	—	0.7	—	—	—	—	—	—	—
Ambrosia trifida	FAC	0	—	0.2	—	—	—	—	1.9	—	—	—	—
Apocynum cannabinum	FACU	1	5.0	5.5	—	—	—	—	0.4	1.1	—	—	—
Asclepias incarnata	OBL	4	0.7	1.8	0.4	—	0.8	0.4	0.4	0.7	—	—	—
Aster ericoides	FACU	2	3.3	—	—	1.0	—	—	—	—	—	—	—
Aster novae-angliae	FACW−	2	—	1.9	—	—	—	—	—	—	—	—	—
Aster pilosus	UPL	1	4.2	1.0	—	—	—	—	—	—	—	—	—
BARBAREA VULGARIS	FACU	0	—	—	—	—	—	—	1.2	—	—	—	—
Bidens cernua	OBL	3	—	0.3	—	—	—	—	—	—	—	—	—
Bidens coronata	OBL	3	—	—	0.2	—	—	—	2.3	—	—	—	0.3
Bidens frondosa	FACW	2	0.3	4.4	—	—	—	—	—	—	—	—	—
Boehmeria cylindrica	FACW+	4	—	—	—	—	—	—	—	0.4	0.5	—	2.0
BUTOMUS UMBELLATUS	OBL	0	—	0.3	—	—	—	—	—	—	—	—	—
Calystegia sepium	FAC−	1	4.7	—	—	—	—	—	3.8	—	—	—	—
Campsis radicans	FAC	1	0.7	—	—	—	—	—	—	—	—	—	—
Carex conjuncta	FACW	5	—	0.6	—	—	—	—	—	—	—	—	—
Carex frankii	OBL	2	1.0	0.6	—	1.0	—	—	—	2.1	—	—	—
Carex hyalinolepis	OBL	5	—	—	—	5.0	15.0	—	—	—	—	—	—
Carex lacustris	OBL	5	—	1.0	—	11.0	3.5	—	—	—	—	—	—
Carex squarrosa	FACW	4	1.7	0.5	—	—	0.4	—	—	—	—	—	—
Carex tribuloides	FACW+	4	1.0	5.5	9.4	1.7	0.4	0.4	—	7.5	—	—	—
Carex vulpinoidea	OBL	1	5.5	7.4	0.7	9.0	2.7	—	—	2.9	0.5	0.7	—
Carya laciniosa	FAC	7	—	—	—	1.3	0.8	—	—	—	—	—	—
Cephalanthus occidentalis	OBL	6	—	—	—	—	—	—	—	—	—	34.6	1.3
Ceratophyllum demersum	OBL	2	—	1.9	—	—	0.2	—	—	—	—	—	—
Cicuta maculata	OBL	3	—	—	—	—	0.8	—	—	—	—	—	—
Cinna arundinacea	FACW	4	1.0	1.8	—	8.0	6.9	—	—	—	—	—	—
CIRSIUM ARVENSE	FACU	0	—	—	—	—	—	—	0.4	0.7	—	—	—
CONVOLVULUS ARVENSIS	UPL	0	0.3	—	—	—	—	—	—	—	—	—	—
Cornus amomum	FACW	2	—	0.3	3.0	—	—	—	—	—	—	—	—
Cyperus esculentus	FACW	0	—	—	—	—	—	—	2.3	—	—	—	—
Cyperus strigosus	FACW	1	3.0	—	—	—	—	—	—	—	—	—	—
DAUCUS CAROTA	UPL	0	3.2	0.2	—	—	—	—	—	—	—	—	—
DIPSACUS SYLVESTRIS	FACU−	0	1.3	0.3	—	0.3	—	—	—	—	—	—	—
ECHINOCHLOA CRUSGALLI	FACU	0	—	—	—	—	—	—	—	—	0.5	—	—
Echinochloa muricata	FACW+	3	10.0	0.8	—	2.7	—	—	—	—	—	—	—
Eleocharis obtusa	OBL	1	—	2.9	12.0	—	—	7.7	3.8	—	—	—	—

Table A1 Continued

Species	Indicator status	C	Restored			Enhanced			Created			Reference	
			Y1	Y5	Y10	Y1	Y5	Y10	Y1	Y5	Y10	BB	GW
Elymus canadensis	FACU+	6	0.2	—	—	—	—	—	—	—	—	—	—
Elymus virginicus	FACW-	3	—	0.2	—	—	—	—	—	—	—	—	—
ELYTRIGIA REPENS	FACU−	0	—	—	—	—	—	—	1.2	—	—	—	—
Epilobium coloratum	OBL	1	—	1.0	—	—	0.4	—	—	—	—	—	—
Erechtites hieracifolia	FACU	2	—	—	—	—	—	—	0.4	—	—	—	—
Euthamia graminifolia	FAC	2	—	0.3	—	—	—	—	—	—	—	—	—
Fraxinus pennsylvanica	FACW	3	13.2	4.4	11.0	16.3	7.3	13.0	7.7	0.4	—	—	16.0
Galium palustre	OBL	9	—	—	—	—	—	—	—	0.7	0.5	—	—
Geum canadense	FACU	2	—	0.3	—	—	—	—	—	—	—	—	—
Geum laciniatum	FAC+	2	5.8	3.4	—	0.3	—	—	—	0.7	—	—	—
Gylceria canadensis	OBL	7	—	—	4.4	—	—	—	—	—	—	—	—
HIBISCUS TRIONUM	UPL	0	—	—	—	—	—	—	0.1	—	—	—	—
Hippuris vulgaris	OBL	NI	—	—	0.4	—	—	—	—	—	—	—	—
Hordeum jubatum	FAC	0	—	—	6.1	—	—	3.8	—	—	—	—	—
Impatiens capensis	FACW	2	—	—	—	—	0.8	0.8	—	—	—	—	3.3
Juncus effusus	FACW+	1	2.5	1.1	0.7	9.0	—	—	—	—	—	—	—
Juncus tennis	FAC−	1	0.5	0.3	3.1	1.0	—	0.4	—	—	—	—	—
Leersia oryzoides	OBL	1	—	0.3	—	—	3.8	—	—	0.4	—	—	—
Lemna minor	OBL	3	4.0	1.1	44.5	7.0	6.6	75.0	0.4	11.0	41.0	21.0	48.0
Lindera benzoin	FACW−	5	—	—	—	—	0.4	—	—	—	—	—	19.0
Lindernia dubia	OBL	2	—	—	—	—	—	—	—	0.4	—	—	—
LOLIUM PERENNE	FACU−	0	—	—	—	—	—	—	0.4	—	—	—	—
LONICERA JAPONICA	FAC−	0	—	—	—	—	—	—	—	—	—	—	3.0
Ludwigia alterniflora	FACW+	3	—	1.0	—	—	2.7	—	—	0.4	—	—	—
Ludwigia palustris	OBL	3	—	—	—	—	—	—	—	0.4	—	—	—
Ludwigia polycarpa	OBL	5	0.2	—	—	0.3	—	—	—	—	—	—	—
Lycopus americanus	OBL	3	0.7	1.1	—	—	—	—	0.4	0.4	—	—	—
Lycopus virginicus	OBL	3	—	—	—	—	—	—	—	0.4	—	—	—
Lysimachia ciliata	FACW	4	—	—	—	—	—	—	—	1.4	—	—	—
LYSIMACHIA NUMMULARIA	OBL	0	—	3.5	6.7	14.7	3.8	4.2	1.5	1.4	1.0	0.7	—
Lythrum alatum	FACW+	6	—	—	—	—	—	—	—	0.4	—	—	—
MALVA NEGLECTA	UPL	0	—	—	—	—	—	—	—	—	—	—	0.7
MEDICAGO LUPULINA	UPL	0	0.3	—	—	—	—	—	—	—	—	—	—
Mimulus ringens	OBL	4	—	1.6	—	—	—	—	—	—	—	—	—
Najasflexilis	OBL	5	—	—	10.4	—	—	0.4	—	—	—	—	—
Nuphar advena	OBL	4	—	—	—	—	—	—	—	—	—	29.3	—
Osmunda cinnamomea	FACW	6	—	—	—	—	—	—	—	—	—	—	0.7
Panicum dichotomiflorum	FACW−	0	0.2	—	—	—	—	—	—	—	—	—	—
Panicum virgatum	FAC	4	1.8	7.6	—	0.3	1.5	—	—	—	—	—	—
Parthenocissus quinquefolia	FACU	2	—	—	—	—	—	—	—	—	—	—	2.0
Penthorum sedoides	OBL	2	—	4.2	—	—	2.7	—	—	—	—	—	—
Phalaris arundinacea	FACW+	0	0.5	0.8	—	0.7	—	—	14.0	1.8	30.0	—	—
PHLEUM PRATENSE	FACU	0	0.3	0.2	—	0.3	—	—	1.5	—	—	—	—
Phyla lanceolata	OBL	3	—	—	—	—	—	—	—	1.4	—	—	—
Physostegia virginiana	FAC+	5	—	—	—	0.3	—	—	—	—	—	—	—
Pilea pumila	FACW	2	—	—	—	—	—	0.4	—	—	—	—	3.0
Plantago rugelii	FACU	0	0.3	—	—	—	—	—	—	—	—	—	—
Polygonum amphibium	OBL	4	—	—	—	—	—	—	22.7	3.9	1.5	—	—
Polygonum hydorpiperoides	OBL	6	—	—	17.8	—	—	14.0	—	—	—	13.0	—
Polygonum lapathifolium	FACW+	1	1.7	—	—	—	—	—	—	—	—	—	—
Polygonum pensylvanicum	FACW	0	0.2	1.1	—	—	—	—	—	—	—	—	—
POLYGONUM PERSICARIA	FACW	0	—	0.2	0.4	—	2.3	—	0.4	—	—	—	—
Polygonum sagittatum	OBL	2	—	—	—	—	—	—	—	5.7	—	0.4	—
Populus deltoides	FAC	3	3.7	1.0	4.1	9.3	8.8	—	7.7	—	—	—	—
Potamogeton foliosus	OBL	2	—	0.8	—	—	—	—	—	28.2	—	—	—
Potamogeton nodosus	OBL	3	6.3	4.7	—	—	0.4	—	—	5.4	—	35.0	—
Prunella vulgaris	FACU+	0	0.8	0.3	—	—	—	—	—	—	—	—	—
Quercus alba	FACU−	6	—	—	—	—	—	3.8	—	—	—	—	—
Quercus bicolor	FACW+	7	—	—	—	6.7	3.5	4.2	—	—	—	—	—

Table A1 Continued

Species	Indicator status	C	Restored Y1	Restored Y5	Restored Y10	Enhanced Y1	Enhanced Y5	Enhanced Y10	Created Y1	Created Y5	Created Y10	Reference BB	Reference GW
Quercus palustris	FACW	5	0.2	—	1.9	21.7	6.2	9.2	—	—	—	—	9.0
Quercus rubra	FACU–	6	—	—	—	—	—	20.8	—	—	—	—	—
Ranunculus abortivus	FACW–	1	—	—	—	—	—	0.8	—	—	—	—	1.0
Ranunculus hispidis	FAC	4	—	—	0.7	—	1.2	—	—	—	0.5	—	1.3
Ranunculus sceleratus	OBL	1	—	—	—	—	—	—	—	—	—	0.4	—
ROSA MULTIFLORA	FACU	0	—	—	0.4	—	0.4	—	—	—	—	—	0.3
Rosa palustris	OBL	5	—	—	—	—	0.8	—	—	—	—	—	3.0
Rosa setigera	FACU	4	—	—	—	—	—	—	—	—	—	—	1.3
Rubus allegheniensis	FACU–	1	—	—	—	—	—	—	15.4	3.2	1.0	—	—
RUMEX CRISPUS	FACU	0	0.2	0.2	—	0.7	—	—	—	0.4	—	—	—
Sagittaria latifolia	OBL	1	—	—	—	—	—	—	—	—	0.5	1.4	—
Salix nigra	FACW+	2	—	0.2	2.2	—	—	3.8	—	—	—	2.1	—
Scirpus atrovirens	OBL	1	11.7	5.2	—	6.0	1.2	—	—	—	—	—	—
Scirpus cyperinus	FACW+	1	—	1.6	—	—	—	—	—	—	—	—	—
Scutellaria lateriflora	FACW+	3	—	—	—	—	0.8	—	—	0.4	—	—	—
SETARIA GLAUCA	FAC	0	1.7	0.6	—	0.3	0.4	—	—	—	—	—	—
SETARIA VIRIDIS	UPL	0	1.2	—	—	0.3	—	—	0.8	—	—	—	—
Solatium carolinense	UPL	0	—	—	—	—	—	—	0.4	—	—	—	—
SOLANUM DULCAMARA	FAC–	0	—	—	—	—	—	—	—	—	—	0.4	—
Solidago canadensis	FACU	1	4.3	0.8	—	0.3	—	—	—	—	—	—	—
Solidago gigantea	FACW	3	1.0	0.2	—	—	—	—	—	—	—	—	—
Sparganium eurycarpum	OBL	4	—	—	—	—	—	—	—	0.4	—	—	—
Staphylea trifolia	FAC	6	—	—	—	—	—	—	—	—	—	—	0.3
TARAXACUM OFFICINALE	FACU–	0	—	—	—	—	—	—	1.2	—	—	—	—
Toxicodendron radicans	FAC	1	1.8	1.1	0.4	14.7	6.5	1.5	—	—	—	—	3.7
TRIFOLIUM HYBRIDUM	FACU–	0	—	—	—	—	—	—	0.1	—	—	—	—
TRIFOLIUM PRATENSE	FACU–	0	0.2	0.5	—	2.3	—	—	0.4	—	—	—	—
TYPHA ANGUSTIFOLIA	OBL	0	—	—	—	—	—	—	7.7	2.1	—	—	—
Typha latifolia	OBL	1	—	1.8	7.0	—	—	2.3	1.9	12.0	20.0	0.7	—
Ulmus americana	FACW–	2	3.5	—	—	25.3	11.0	23.0	—	—	—	—	25.0
Ulmus rubra	FAC	3	—	2.7	0.7	—	—	—	—	—	—	—	—
VALERIANELLA LOCUSTA	UPL	0	—	—	—	—	—	—	—	—	0.4	—	—
Vernonia gigantea	FAC	2	—	—	—	—	—	—	1.9	1.8	—	—	—
Vernonia noveboracensis	FACW+	3	—	—	—	—	—	—	—	—	—	—	3.7
Viola sororia	FAC–	1	—	—	—	—	—	—	—	—	—	—	1.4
Vitis riparia	FACW	3	—	0.3	—	—	—	—	—	—	—	—	—
Wolffia columbiana	OBL	3	—	—	—	—	—	—	—	21.1	20.0	—	—

[a]Numbers indicate mean percent cover by quadrat in years 1 (Y1), 5 (Y5), and 10 (Y10) in each mitigation wetland and in 2004 for the reference wetlands. Species in capital letters are considered non-native to Ohio according to Kartesz and Meacham (1999). Wetland indicator status for each species is given according to Reed (1988), and coefficient of conservatism (C) according to Andreas and others (2004).

Table A2 Macroinvertebrate taxa collected in created and restored/enhanced mitigation wetlands and two reference wetlands—Buttonbush Swamp (BB) and Gahanna Woods (GW)—in the summer of 2004

Order	Family	Genus	Trophic guild[a]	Tolerance index[a]	Created	Restored/enhanced	Reference
Amphipoda	Crangonyctidae	Synurella	Collector/gatherer	1.0	34	—	[illegible]
Amphipoda	Crangonyctidae	Crangonyx	Collector/gatherer		—	6	[illegible]
Amphipoda	Hyalellidae	Hyalella	Collector/gatherer	5.0	—	[illegible]	[illegible]
Amphipoda			Collector/gatherer		—	[illegible]	[illegible]
Arhynchobdellida	Erpobdellidae	Erpobdella	Detritivore/predator		4	[illegible]	[illegible]
Arhynchobdellida	Erpobdellidae	Mooreobdella	Detritivore/predator		—	[illegible]	[illegible]
Coleoptera	Anthicidae		Omnivore		[illegible]	[illegible]	[illegible]
Coleoptera	Curculionidae		Shredder		[illegible]	[illegible]	[illegible]
Coleoptera	Dytiscidae		Predator	[illegible]	[illegible]	[illegible]	[illegible]
Collembola	Entomobryidae		Detritivore		[illegible]	[illegible]	[illegible]
Coleoptera	Haliplidae	Haliplus	Herbivore		[illegible]	[illegible]	[illegible]

Table A2 Continued

Order	Family	Genus	Trophic guild[a]	Tolerance index[a]	Created	Restored/ enhanced	Reference	
							BB	GW
Coleoptera	Haliplidae	Peltodytes	Herbivore	5.0	—	5	—	1
Coleoptera	Hydrophilidae	Berosus	Predator		—	—	1	1
Coleoptera	Hydrophilidae	Enochrus	Predator		—	1	—	—
Coleoptera	Hydrophilidae	Lacobius	Predator		—	—	—	1
Coleoptera	Hydrophilidae		Predator	3.0	5	2	3	5
Coleoptera	Noteridae	Hydrocanthus	Predator		—	3	—	—
Decapoda	Cambaridae	Procambarus	Omnivore	2.0	1	1	4	—
Diptera	Chaoboridae	Chaoborus	Predator	2.0	8	—	14	52
Diptera	Chaoboridae	Mochlonyx	Predator		—	2	—	1
Diptera	Chironomidae	Chironomus	Collector/gatherer	2.0	—	—	6	10
Diptera	Chironomidae		Collector/gatherer	2.0	83	4	23	31
Diptera	Culicidae	Culex	Collector/gatherer	1.7	—	—	—	5
Diptera	Culicidae		Collector/gatherer	1.7	4	—	1	—
Ephemeroptera	Caenidae	Caenis	Collector/gatherer	3.5	5	20	—	—
Ephemeroptera	Baetidae		Collector/gatherer		—	2	—	—
Hemiptera	Belostomatidae	Belostoma	Predator	2.0	2	4	4	—
Hemiptera	Corixidae	Dasycorixa	Herbivore		1	—	—	—
Hemiptera	Corixidae	Trichocorixa	Herbivore		6	6	—	1
Hemiptera	Corixidae		Herbivore	2.3	—	23	3	—
Hemiptera	Gerridae		Predator	2.0	1	—	—	—
Hemiptera	Naucoridae		Predator		1	—	—	—
Hemiptera	Nepidae	Curicta	Predator		—	2	1	—
Hemiptera	Nepidae	Ranatra	Predator	2.0	—	2	1	—
Hemiptera	Notonectidae	Notonecta	Predator	5.0	—	—	—	2
Hemiptera	Pleidae	Paraplea	Predator		—	1	—	—
Isopoda	Asselidae	Caecidotea	Shredder		205	1	7	21
Megaloptera	Corydalidae	Chauliodes	Predator	5.0	1	—	—	1
Odonata	Coenagrionidae	Coenagrion	Predator		—	6	—	—
Odonata	Coenagrionidae		Predator	2.0	5	1	4	—
Odonata	Gomphidae	Gomphus	Predator	5.0	1	1	2	1
Odonata	Lestidae		Predator	2.0	2	—	—	—
Odonata	Libellulidae	Sympetrum	Predator		—	—	—	1
Odonata	Libellulidae	Tramea	Predator		1	—	—	—
Odonata	Libellulidae		Predator	2.0	—	—	19	3
Oligochaeta			Collector/gatherer		2	—	—	—
Prostigmata			Parasite	2.0	1	358	1	—
Pulmonata	Limnaeidae		Scraper/grazer		—	—	3	1
Pulmonata	Planorbidae	Menetus	Scraper/grazer	2.5	—	8	3	16
Pulmonata	Physidae	Physa	Scraper/grazer	3.0	69	4	37	20
Pulmonata	Planorbidae		Scraper/grazer	3.0	3	—	27	—
Rhynchobdellida	Glossiphoniidae	Placobdella	Parasite		—	—	8	1
Veneroida	Sphaeriidae		Collector/filterer	1.7	5	—	1	9

[a]Trophic guilds are assigned according to Merritt and Cummins (1996), and tolerance index according to Adamus and Gonyaw (2001), where 1 = very tolerant, 2 = tolerant, 3 = moderately tolerant, 4 = somewhat intolerant, and 5 = intolerant to anthropogenic stressors. Numbers indicate actual numbers of organisms collected in each wetland