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# VEGETATION DEVELOPMENT IN CREATED, RESTORED, AND ENHANCED MITIGATION WETLAND BANKS OF THE UNITED STATES

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*Abstract:* Wetland mitigation banking is the practice of creating, restoring, enhancing, or preserving large, off-site wetlands to compensate for authorized impacts to natural wetlands. By 2002, there were 219 active mitigation banks in the United States, encompassing 50,000 hectares in 29 states. This study is the first systematic analysis of the ecological quality of these ecosystems; the objective is to determine if mitigation banks are successfully supporting native wetland vegetation and if success differs by mitigation method (created, restored, or enhanced), geomorphic class, age, or area. I obtained monitoring reports from 45 randomly selected mitigation bank wetlands in 21 states to evaluate three measures of ecological status: the prevalence of wetland vegetation, the pervasiveness of non-native species, and plant species richness. Sites range from less than one ha to over 560 ha and include 17 created wetlands, 19 restored wetlands, and 9 enhanced wetlands. Prevalence Index scores (PI; 1.0 for obligate wetland vegetation to 5.0 for upland vegetation) do not differ by wetland area but are significantly lower in created wetlands and significantly decrease from one- and two-year-old created wetlands (PI=2.37±0.15; mean±SE) to those five to seven years old (PI=1.96±0.12). Created and restored wetlands support 12.4 and 12.2 species per 10 $m^2$ respectively, nearly four times more than the 3.2 species in 10 $m^2$ of enhanced wetland. This is in part attributable to a greater incidence of non-native species in created and restored wetlands. The vegetative cover in created mitigation bank wetlands is 18.9±2.8 percent non-native–statistically similar to that of restored (17.6±2.9) but significantly greater than that of enhanced systems (8.7±2.7). Within mitigation methods, there are clear differences among geomorphic and vegetation classes. Depressional systems with a single vegetation class support highly hydrophytic, highly non-native communities with low species richness, while restored and enhanced riverine systems have a greater prevalence of native species. For mitigation bank wetlands in this study, the prevalence of wetland vegetation, the representation of native species, and the plant community homogeneity increase with age, indicating a period of self-organization and a potential trend toward vegetative equivalence with natural wetlands.

*Key Words:* mitigation bank, created wetland, restored wetland, enhanced wetland, prevalence index, non-native species

## INTRODUCTION

Since the 1977 amendments to the Clean Water Act, compensatory wetland mitigation has become commonplace in the United States. The vast majority of mitigation projects have been created or restored wetlands that are the responsibility of a single permitee (National Research Council 2001). A number of researchers have analyzed the regulatory and ecological success of these systems, and in general, mitigation wetlands have fallen short of replacing natural wetlands in terms of both area and function (Roberts 1993, Zedler 1996, Malakoff 1998, Cole and Shafer 2002). More recently, mitigation banks have become established as a means of consolidating the haphazard approach to wetland mitigation. Mitigation banks are defined as ''sites where wetlands and/or other aquatic resources are restored, created, enhanced, or in exceptional circumstances, preserved expressly for the purpose of providing compensatory mitigation in advance of authorized impacts to similar resources'' (Federal Guidance 1995), and perceived advantages in economy, regulation, and ecology have made this an attractive alternative to individual wetland replacement. Third-party mitigation credits became commonly available in the early 1990s (Tabatabai and Brumbaugh 1998), and two major inventories show that the number of mitigation banks has grown rapidly since, with 37 in operation in 1992 and 219 in 2002 (Reppert 1992, ELI 1993, 2002). To date, however, we have no comprehensive understanding of the functional success of these wetlands. This study is the first systematic analysis of the ecological state of mitigation banks.

In theory, mitigation banks have several advantages over individual mitigation projects (NRC 2001). First, because they are initiated prior to need, the temporal uncertainty of successful compensation is alleviated. Second, because they often accommodate multiple permits, mitigation banks encompass larger areas than single mitigation sites. While larger wetlands are not always desirable (Semlitsch 2000), they are in some ways advantageous over many smaller systems–particularly in terms of water quality improvement (NRC 2001). Third, since mitigation banks are typically located off-site, they may be placed in a more ideal ecological and hydrologic setting than smaller mitigation wetlands located on or near the site of development. Finally, mitigation banking allows for a concentration of scientific expertise, financial resources, and regulatory oversight that should provide a functionally sound, cost-effective means of replacing wetlands. Several factors complicate these advantages. A number of the complications are regulatory and economic in nature and are discussed in considerable detail elsewhere (McElfish and Nichols 1996, Shabman et al. 1998, Stein 1999, Stein et al. 2000). Here, I consider the current ecological realities of mitigation banking.

Mitigation bank agreements stipulate performance standards to be monitored for a defined length of time–generally five years, but ranging from three to 50 years (ELI 2002). In almost every case, some measure of vegetation is a performance standard, and in many cases, vegetation is the only performance standard. Common vegetation standards include targets for percent cover of hydrophytic vegetation, limits for nuisance species cover, and goals for survival of planted stock. The National Research Council (2001) cautions that vegetation alone is a poor measure of wetland function, but it is seen as a quick and effective surrogate for the biogeochemical condition of the wetland and is commonly used as a measure of success (Breaux and Serefiddin 1999). Less frequently monitored are hydrologic regime, non-native species, wildlife, and soil development (Breaux and Serefiddin 1999). Progress toward these performance standards are typically submitted in annual monitoring reports, and in many cases (although not all), the reports provide field assessments of performance criteria. Annual reports of vegetation development provide the most complete information on the status of mitigation bank development and are the basis of this study.

One question that may be partially answered through an analysis of vegetation performance criteria concerns the success of different methods of mitigation banking. Do created, restored, and enhanced mitigation bank wetlands achieve the same degree of ecological success at the same rate, or have some consistently performed better than others? Unfortunately, many regulatory decisions have been made in absence of this information (Whigham 1999). Federal mitigation banking guidelines (Federal Guidance 1995) clearly define creation, restoration, and enhancement and, thus, provide a basis for comparison. A created mitigation wetland is one established in an area 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. These three methods are ecologically very different, and it is likely that the plant communities they support are at least initially dissimilar. Indeed, research has shown that vegetation communities of restored and created wetlands are different than natural reference wetlands in terms of species richness and diversity (Galatowitsch and van der Valk 1996, Ashworth 1997, Fennessy and Roehrs 1997).

A second question on the differential success of mitigation bank wetlands concerns placement within the landscape. Mitigation bank wetlands are located in a variety of hydrogeomorphic settings–from hydrologically isolated depressional marshes to riverine bottomland forests–that are generally quite removed geographically from the wetlands they replace (Brown and Lant 1999). The ecological questions here concern the importance of proximity to a source of surface water and propagules–critical factors to the development of some ecosystems (Zedler 1997). Many studies have shown that hydrologic regime (Brinson et al. 1981, Keddy 2000) and hydrochory (Nilsson et al. 1994, Bornette et al. 1998) are directly related to wetland plant productivity, diversity, and distribution. It is likely, then, that mitigation bank wetlands with a relatively closed hydrologic regime will follow a very different successional trajectory than those near a constant source of water and seed. In this way, geomorphic setting is an important qualifier for mitigation banking method. Thus, we may ask how the development of created wetlands is related to the proximity of another body of water, or how creations, restorations, and enhancements perform in different settings.

The mitigation bank data set also provides a useful tool for analyzing questions of age and area. Many researchers have hypothesized that mitigation wetlands will achieve functional equivalency with their natural counterparts given enough time (Mitsch and Wilson 1996), but there is little research on the long-term maturation of mitigation wetlands (Zedler and Callaway 1999) and none on mitigation banks. One might expect the consolidated effort and resources of mitigation banks to produce a clear trend toward diverse, native hydrophytic communities. The decade of mitigation bank reports makes this a testable hypothesis. Similarly, ecological theory holds that area is an important factor in ecosystem development, as larger ecosystems have greater heterogeneity, diversity, and resilience

(Zedler 1997). Mitigation banking attempts to use this advantage of size, although the area of operational mitigation banks covers a wide range–from less than 3 ha to more than 9500 ha (ELI 2002). When considered alongside mitigation method, geomorphic setting, and age, an understanding of the importance of area would be very useful in the design of future mitigation banks.

In this study, I evaluated the success of mitigation bank wetlands by considering three aspects of the plant communities they support: the prevalence of hydrophytic vegetation, the number of species present, and the proportion of non-native species. My objectives are threefold. First, I compare these plant community metrics among created, restored, and enhanced wetlands in mitigation banks to test the hypothesis that they all support similar vegetation. Second, I analyze the vegetation communities among wetlands of different geomorphic setting and vegetation class and consider the interactions of method, classification, age, and area. Third, I assess the trajectory of mitigation bank wetlands by considering evidence for changes in their plant communities over time. My overall goals are to evaluate the vegetative success of mitigation banks and to contribute to a better understanding of how these replacement wetlands perform upon establishment and as they mature.

## METHODS

### Site Selection and Description

Mitigation banks considered in this study were selected from the recent inventory published by the Environmental Law Institute (ELI 2002), in which 219 active mitigation banks are identified. Mitigation banks that were proposed, pending, inactive, or expired (meaning approved but never constructed) at the time of this inventory were not considered. Likewise, umbrella mitigation banks–regional banks with multiple sites–were not included in the study. From the pool of 219 active mitigation banks, I randomly selected 62 and requested the most recent monitoring report for each from the corresponding U.S. Army Corps of Engineers District. Only monitoring reports that included field identification of plant species in multiple plots of defined area qualified for the study. No report was available for 14 of the requested banks, and an additional 12 were excluded, as they reported no vegetation analysis or contained insufficient data. Thus, monitoring reports from 36 mitigation banks were included in the study, representing 21 states (Figure 1). Five banks include a second phase that differs from the first by age or area; these include their own distinct monitoring report and are treated as separate wetlands. These 41 wetlands have an average age since establishment (mean$\pm$SE) of 5.1$\pm$0.4 years and an average area of 6519 ha (Table 1).

Using the monitoring reports and, where necessary, site plans and mitigation banking agreements, I classified the wetlands of each bank as a creation, restoration, or enhancement. In four cases, the banks include more than one mitigation method and supply field data specific to each section; these were also treated as distinct entities. The resulting sample size (N=45) includes 17 created wetlands, 19 restored wetlands, and 9 enhanced wetlands. I also classified each wetland by geomorphic setting according to Brinson (1993). Mitigation bank wetlands in this study occur in three geomorphic settings: depressional, riverine, and lacustrine fringe (Table 1). Wetlands classified as depressional are those topographically isolated from other surface-water bodies and hydrologically dependent upon atmospheric and ground-derived water sources. Riverine wetlands are characterized by periodic fluvial inundation and include floodplains, bottomlands, and riparian wetlands. Lacustrine fringe wetlands occur on the shoreline of large lakes and are hydrologically connected to lake waters. Only a single lacustrine fringe system is part of this sample and was excluded from classification analyses. To allow for habitat heterogeneity, I further divided the depressional group based on vegetation classes—the presence of submerged and floating, emergent, scrub-shrub, or forest habitats. All riverine wetlands in this study include a forested component, and while four also support scrub-shrub or emergent habitat, the riverine group was not further subdivided by vegetation class. Classification analyses are thus based on three groups: depressional wetlands with a single vegetation class (n=16), depressional wetlands with multiple (two or more) vegetation classes (n=17), and riverine systems (n=11). Analyses of these mitigation bank wetlands include only those sections denoted as mitigation wetlands in the monitoring reports; sections specifically denoted as upland buffers were excluded. For age-based analyses, I included the first-year monitoring report for wetlands five years old or older where available, adding seven additional points for temporal comparisons (N=52).

### Data Analyses

The wide variety of field methods and absence of standard procedures in monitoring and reporting mitigation wetlands limited the number of meaningful comparisons that could be made among sites. In some reports, field identifications are reported for several seasons in a given year. For consistency, I analyzed only the data obtained nearest to mid-summer. By including only sites with systematically quantified veg-

Figure 1. Approximate location of 219 operational mitigation banks as of 2002 (after ELI 2002). All circles denote mitigation banks. Gray circles represent banks for which monitoring reports were requested but either were unavailable or contained insufficient data. Black circles represent mitigation banks analyzed in this study.

etation identification, I facilitated comparison of three basic metrics: the prevalence index, the representation of nonnative species, and the overall species richness.

The prevalence index (PI) is a weighted average of the indicator status of all plants present (Wentworth et al. 1988). In this study, I assigned 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 were given for + or − designations. Percent cover data were used to calculate the PI for seventy percent of the monitoring reports considered in the study. Those that provided no percent cover data were weighted by frequency. As a second measure of the state of mitigation bank wetlands, I quantified the presence of non-native plant species at each site. Non-native plants on the total species list of each wetland were identified according to Kartesz and Meacham (1999). The percentage of nonnative species at each site was calculated both by presence and abundance, as weighted by percent cover or frequency.

To compare species richness among mitigation wetlands, I used the same multiple-plot field data to construct a species accumulation curve for each wetland (Palmer 1990, Colwell and Coddington 1994, Ugland et al. 2003). In this method, the accumulating area of each quadrat sampled is plotted against the cumulative number of new species identified. I randomized the order of quadrats and converted 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. I then let $A=10\ m^2$ to interpolate S, the number of plant species present in 10 $m^2$ of each particular wetland. This method is based on the assumption that each species in each quadrat was identified and recorded, and it has the advantage of allowing direct comparison of wetlands with widely ranging numbers (2 to 391) and areas (0.25 to 8296 $m^2$) of sampling quadrats (Table 2).

### Statistical Methods

I used one-way analysis of variance (ANOVA) to test the hypotheses that mean prevalence indices, percent nonnative species, and species richness are similar among mitigation bank wetlands of different mitiga-

Table 1. Location and characteristics of mitigation banks in this study. Method refers to the mitigation category: creation (C), restoration (R), or enhancement (E). Age indicates years since establishment for the monitoring report(s) analyzed. Vegetation classes after Cowardin et al. (1979) include palustrine emergent marsh (PEM), palustrine scrub-shrub (PSS), palustrine floating or submerged vegetation (PAB), palustrine forest (PFO), river floodplain (R2UB), riverbank (R3US), and lake littoral zone (L2AB). Geomorphic settings are according to Brinson (1993).

| Mitigation Bank | State | Method | Established | Age | Area (ha) | Vegetation Classification | Geomorphic Setting |
|---|---|---|---|---|---|---|---|
| Barra Farms Cape Fear | NC | R | 1998 | 1, 5 | 249 | PFO, PSS, R2UB | Riverine |
| Big Island | OH | R, E | 1994 | 1, 5 | 139 | PFO, PSS, PEM | Depressional |
| Big Rivers | MO | E | 1999 | 3 | 44 | R2UB | Riverine |
| Big Rivers Verde | MO | R | 2002 | 1 | 32 | R2UB | Riverine |
| Black River Bottomland I | SC | E | 1998 | 1 | 10 | R3US | Riverine |
| Black River Bottomland II | SC | E | 1998 | 1 | 2 | R3US | Riverine |
| Butterfield Road | IL | C | 1999 | 2 | 22 | PEM | Depressional |
| Callaway Farms | GA | R, E | 1998 | 1, 5 | 45 | PEM, PFO, R3US | Riverine |
| Clay Station Phase I | CA | C | 1994 | 5 | 15 | PEM | Depressional |
| Clay Station Phase II | CA | C | 1999 | 3 | 20 | PEM | Depressional |
| Coulthard | IA | R, E | 2000 | 3 | 19 | PEM, PSS | Depressional |
| Cutler Cranberry Co. | WI | R | 1993 | 5 | 41 | PEM | Depressional |
| Ferson Creek | IL | C | 1996 | 1, 6 | 33 | PEM | Depressional |
| G & L | KY | R | 1998 | 5 | 19 | PEM, PFO | Depressional |
| Hartman Bottoms | AR | R | 2001 | 1 | 64 | PFO, R3US | Riverine |
| Hebron | OH | C | 1993 | 1, 5 | 14 | PEM, PAB | Depressional |
| Hobson Yard | NE | R | 1997 | 3 | 92 | PEM | Depressional |
| Inland Sea Reserve | UT | E | 1997 | 4 | 552 | L2AB | Lacustrine Fringe |
| Johnson County | KS | C | 1999 | 3 | 12 | PEM, PFO, R3US | Riverine |
| Lake Station | IN | R | 2000 | 2 | 81 | PEM | Depressional |
| Limon | CO | C | 1996 | 4 | 23 | PEM | Depressional |
| Marion | OR | R | 2001 | 1 | 24 | PEM, PSS, PFO | Depressional |
| Meadowland | WA | C, E | 1997 | 5 | 5 | PEM, PSS, PFO | Depressional |
| Metz Phase I | VA | C | 1994 | 7 | 1 | PFO, PSS, PEM | Depressional |
| Metz Phase II | VA | C | 1996 | 5 | 6 | PFO, PSS, PEM | Depressional |
| Middle Ouachita River | AR | C | 2000 | 2 | 134 | PEM, PFO | Depressional |
| Middle South Platte River | CO | C | 1999 | 3 | 35 | PEM, PSS, PAB | Depressional |
| Mississippi | MS | R | 2000 | 2 | 562 | PFO, PSS | Depressional |
| Mud Slough | OR | C | 2000 | 1 | 22 | PEM, PSS | Depressional |
| Nelson County | KY | C | 1997 | 5 | 46 | PEM, PFO, R2UB | Riverine |
| North Fork | VA | C | 1999 | 3 | 50 | PEM, PAB | Depressional |
| Oak Creek | OR | R | 1999 | 4 | 35 | PEM, PFO | Depressional |
| Pond Creek | KY | C | 2000 | 2 | 4 | R2UB | Riverine |
| Sauk Trail | IL | R | 1998 | 3 | 26 | PEM | Depressional |
| Schroeder | IN | R | 1999 | 3 | 6 | PEM | Depressional |
| Seifert | UT | R | 1996 | 5 | 21 | PEM | Depressional |
| Stennis Space Center | MS | E | 1996 | 7 | 50 | PFO, PSS | Depressional |
| Shady Valley Orchard Bog | TN | R | 1997 | 5 | 26 | PEM | Depressional |
| Shady Valley Quarry Bog | TN | R | 2000 | 2 | 26 | PEM | Depressional |
| Walkerwin | WI | R | 1996 | 5 | 52 | PEM | Depressional |
| Weathers | OR | C | 1998 | 4 | 24 | PEM, PSS | Depressional |

tion method or geomorphic classification. Individual differences among means were identified with Fisher's pairwise comparisons with a 5% individual error rate. Statistically significant differences were confirmed with the nonparametric Kruskal-Wallis test, but all significance values reported are results of parametric analyses. For comparisons by age, wetlands are grouped into three classes: 1 to 2, 3 to 4, and 5 or more years since establishment at the time of the monitoring report analyzed. Similarly, wetlands are grouped by area into classes of less than 20 ha, 20–49 ha, and 50 or more ha total area. Differences among wetlands of each age and area class are considered individually and as covariates nested with wetland type and habitat class in a general linear model. Significant differences among nested variables were also

Table 2. Values of Prevalence Index (PI), percent non-native plant species, sampling scheme, species richness, and species-accumulation curve slope ($z$) for mitigation bank wetlands. Method refers to the mitigation category: creation (C), restoration (R), or enhancement (E). Age is based on the monitoring report used for the analysis and refers to number of years since the wetland was established.

| Mitigation Bank | Method | Age | Area (ha) | PI | Percent Non-native Presence | Percent Non-native Abundance | Sampling Plots Number | Sampling Plots Area (m2) | Species Richness (per 10 $m^2$) | $z$ |
|---|---|---|---|---|---|---|---|---|---|---|
| Barra Farms Cape Fear | R | 1 | 249 | 1.85 | 0.0 | 0.0 | 34 | 405 | 2.2 | 1.76 |
| | R | 5 | 249 | 2.25 | 0.0 | 0.0 | 34 | 405 | 1.4 | 1.08 |
| Big Island | R | 1 | 98 | 2.25 | 18.4 | 7.4 | 33 | 25, 2 | 19.8 | 1.30 |
| | E | 1 | 41 | 2.00 | 7.4 | 7.7 | 15 | 25, 2 | 9.3 | 0.43 |
| | R | 5 | 98 | 1.98 | 11.5 | 5.4 | 33 | 25, 2 | 20.7 | 0.36 |
| | E | 5 | 41 | 1.81 | 7.5 | 5.0 | 15 | 25, 2 | 2.9 | 1.43 |
| Big Rivers | R | 1 | 32 | 2.61 | 9.5 | 12.6 | 14 | 28 | 2.8 | 1.19 |
| | E | 3 | 44 | 2.48 | 11.5 | 1.9 | 11 | 28 | 3.7 | 1.06 |
| Black River Bottomland | E | 1 | 10 | 2.31 | 3.1 | 1.6 | 6 | 65 | 5.0 | 0.86 |
| | E | 1 | 2 | 2.29 | 0.0 | 0.0 | 3 | 65 | 4.7 | 0.94 |
| Butterfield Road | C | 2 | 22 | 2.77 | 33.1 | 30.3 | 102 | 0.8 | 33.6 | 0.43 |
| Callaway Farms | R | 1 | 32 | 3.13 | 19.0 | 21.8 | 10 | 65 | 2.3 | 1.28 |
| | E | 1 | 13 | 2.62 | 6.7 | 4.8 | 3 | 65 | 2.3 | 1.28 |
| | R | 5 | 32 | 2.51 | 11.8 | 39.6 | 10 | 65 | 1.8 | 1.59 |
| | E | 5 | 13 | 2.53 | 5.3 | 7.7 | 3 | 65 | 2.3 | 1.26 |
| Clay Station | C | 5 | 15 | 1.36 | 33.0 | 33.0 | 21 | 79 | 2.7 | 0.23 |
| | C | 3 | 20 | 2.07 | 31.1 | 37.0 | 151 | 77 | 3.5 | 0.29 |
| Coulthard | R | 3 | 16 | 2.51 | 10.5 | 6.4 | 13 | 7 | 8.6 | 0.82 |
| | E | 3 | 3 | 1.95 | 8.3 | 9.5 | 4 | 7 | 5.5 | 0.90 |
| Cutler Cranberry Co. | R | 5 | 41 | 2.07 | 16.0 | 15.8 | 19 | 445 | 1.0 | 0.58 |
| Ferson Creek | C | 1 | 33 | 2.64 | 25.2 | 28.6 | 85 | 0.8 | 21.3 | 0.58 |
| | C | 6 | 33 | 1.74 | 22.4 | 14.7 | 180 | 0.8 | 12.0 | 0.53 |
| G & L | R | 5 | 19 | 3.02 | 16.7 | 13.1 | 36 | 1, 405 | 19.1 | 0.21 |
| Hartman Bottoms | R | 1 | 64 | 2.69 | 6.1 | 13.1 | 50 | 81 | 1.5 | 1.96 |
| Hebron | C | 1 | 14 | 2.18 | 32.4 | 18.2 | 14 | 2 | 17.5 | 0.76 |
| | C | 5 | 14 | 1.29 | 11.4 | 1.9 | 14 | 2 | 16.4 | 0.97 |
| Hobson Yard | R | 3 | 92 | 2.55 | 12.1 | 20.3 | 98 | 0.8 | 11.8 | 0.71 |
| Inland Sea Reserve | E | 4 | 552 | 1.63 | 27.3 | 5.5 | 4 | 316 | 3.0 | 0.67 |
| Johnson County | C | 3 | 12 | 2.73 | 32.1 | 32.8 | 4 | 283 | 3.3 | 0.94 |
| Lake Station | R | 2 | 81 | 2.69 | 30.1 | 30.5 | 391 | 1 | 21.7 | 0.60 |
| Limon | C | 4 | 3 | 1.88 | 4.8 | 4.3 | 4 | 8296 | 2.3 | 0.91 |
| Marion | R | 1 | 24 | 3.27 | 52.7 | 56.1 | 40 | 314 | 1.2 | 2.47 |
| Meadowland | C | 5 | 1 | 2.39 | 0.0 | 0.0 | 4 | 186 | 1.9 | 1.22 |
| | E | 5 | 4 | 2.89 | 12.0 | 3.0 | 16 | 186 | 1.3 | 0.68 |
| Metz | C | 7 | 1 | 2.13 | 9.7 | 37.7 | 13 | 40 | 3.5 | 0.80 |
| | C | 5 | 6 | 2.09 | 2.3 | 12.9 | 77 | 40 | 2.9 | 1.01 |
| Middle Ouachita River | C | 2 | 134 | 2.59 | 7.0 | 6.6 | 80 | 77 | 5.0 | 0.83 |
| Middle South Platte River | C | 3 | 35 | 1.56 | 21.1 | 12.6 | 100 | 1 | 19.0 | 0.47 |
| Mississippi | R | 2 | 562 | 1.88 | 3.0 | 2.1 | 10 | 931 | 2.0 | 1.34 |
| Mud Slough | C | 1 | 22 | 1.80 | 23.0 | 6.2 | 28 | 29 | 5.0 | 0.80 |
| Nelson County | C | 5 | 46 | 2.74 | 18.4 | 14.1 | 20 | 0.8 | 58.0 | 0.39 |
| North Fork | C | 3 | 50 | 1.77 | 9.5 | 18.4 | 42 | 121 | 2.1 | 1.13 |
| Oak Creek | R | 4 | 35 | 2.79 | 39.3 | 17.6 | 50 | 1 | 21.1 | 0.27 |
| Pond Creek | C | 2 | 4 | 2.26 | 14.6 | 11.5 | 6 | 0.8 | 41.0 | 0.50 |
| Sauk Trail | R | 3 | 26 | 2.47 | 14.8 | 26.6 | 20 | 0.25 | 56.0 | 0.02 |
| Schroeder | R | 3 | 6 | 2.58 | 25.8 | 15.4 | 15 | 1 | 21.0 | 0.56 |
| Seifert | R | 5 | 21 | 2.16 | 8.3 | 1.1 | 27 | 8 | 4.8 | 0.60 |
| Stennis Space center | E | 7 | 50 | 2.27 | 3.5 | 3.5 | 2 | 1862 | 0.2 | 0.20 |
| Shady Valley Orchard Bog | R | 5 | 26 | 1.88 | 20.2 | 2.5 | 18 | 77 | 28.0 | 0.27 |
| Quarry Bog | R | 2 | 26 | 2.78 | 23.8 | 14.2 | 18 | 77 | 4.7 | 0.96 |
| Walkerwin | R | 5 | 52 | 2.56 | 22.0 | 16.5 | 30 | 263 | 2.2 | 0.89 |
| Weathers | C | 4 | 24 | 2.04 | 26.7 | 26.2 | 21 | 29 | 6.4 | 0.66 |

identified with Fisher's pairwise comparison. Changes in species accumulation slope over time were analyzed with a regression of *z* values by each year of mitigation bank age.

## RESULTS AND DISCUSSION

### Descriptive Statistics

This data set provides an overall picture of the state of mitigation banks (Table 2). Based on this sample, a disproportionate amount of the total mitigation bank area is wetland restoration (57% of total area; n=19), while wetland creation (n=17) and enhancement (n=9) account for 16% and 27%, respectively. Wetlands in this survey vary widely by prevalence index values, ranging from 1.3 (strongly hydrophytic) to 3.3 (facultative), with and average of 2.3±0.06 (mean±SE). Non-native plant species account for 16.2±1.6 percent of all plants identified and 14.6±1.8 percent when weighted by abundance. The wetlands support 10.5±1.8 species per 10 $m^2$, meaning that, on average, nearly two non-native species occur per 10 $m^2$. The prevalence of non-native species is also quite variable, ranging from no recorded non-native species to over 50 percent non-native. These data illustrate the wide range of plant communities in mitigation banks wetlands and the necessity for more detailed comparisons of vegetation metrics.

### Mitigation Method and Geomorphic Setting

Some differentiation may be achieved by considering the wetlands as grouped by mitigation method: created, restored, or enhanced (Figure 2). Prevalence indices average between 2.0 and 2.5 for all wetlands, indicating that all three mitigation methods are supporting vegetation in the facultative wetland range. The predominance of PI values below 2.5 is a positive indicator of hydrologic regimes sufficient to support hydrophytic vegetation. Indeed, an inadequate or improper hydroperiod has been identified as a primary failing point for both individual mitigation wetlands (Erwin 1991, Gallihugh and Rogner 1998) and mitigation banks (McElfish and Nichols 1996). Twenty-one of the monitoring reports (40%) indicate PI values greater than 2.5, and three exceed 3.0 (Table 2). Interestingly, created wetlands have a significantly lower PI than restored wetlands (Figure 2; $p=0.014$, $F_{2,42}=4.72$). Conventional wisdom holds that restoration should be favored over creation, as the restored wetland has an existing—although impaired—hydrologic regime and therefore a better chance to develop full functionality (Kusler and Kentula 1990). Average PI values in this study indicate that created wetlands

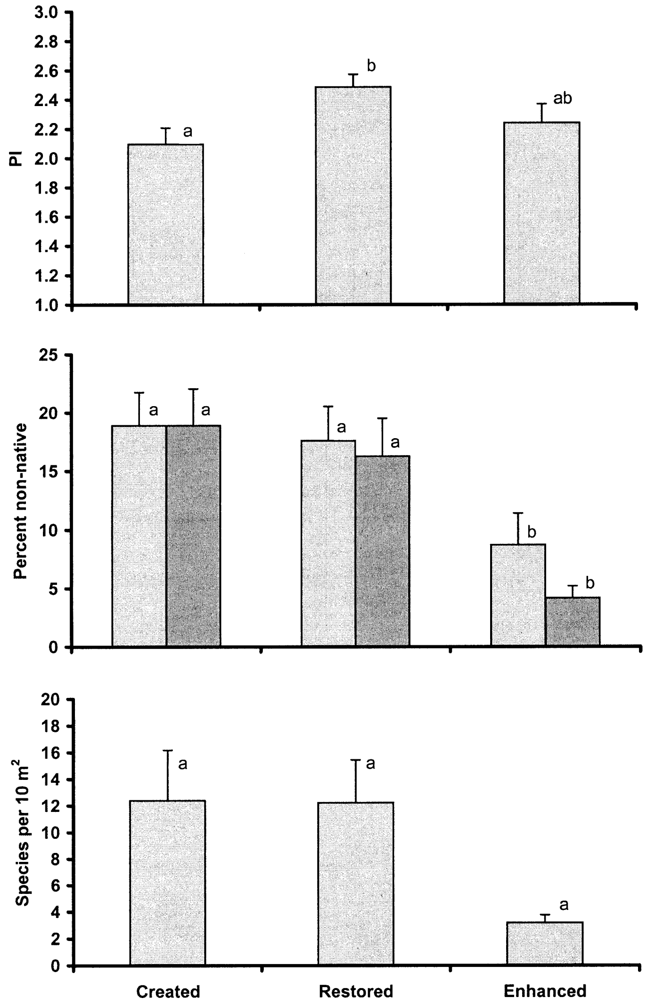


Figure 2. Mean+SE Prevalence Index (PI), percent non-native plant species by presence (light bars) and abundance (dark bars), and plant species richness of created, restored, and enhanced mitigation bank wetlands. Letters indicate significant differences among all treatments ($\alpha=0.05$).

support hydrophytes to a greater degree than restored wetlands, although all mitigation methods appear to have a hydroperiod sufficient to support a predominance of wetland plants.

There is a greater disparity in the number and type of species supported by each type of wetland. Created and restored wetlands support four times as many vascular plant species as enhanced wetlands and also have a significantly greater non-native constituency by both presence ($p=0.10$, $F_{2,42}=2.42$) and by abundance ($p=0.02$, $F_{2,42}=4.34$). This is logical, as created and restored wetlands are converted from non-jurisdictional wetland areas and are subject to temporarily barren areas, a lack of established plants, and a period of species influx. Enhanced wetlands, which by definition have at least some degree of an established hydrologic regime, hydric soils, and hydrophytic vegetation, do not characteristically show an initial influx of species. The differences in non-native species and species richness among created, restored, and enhanced wetlands are consistent with those of individual mitigation wetlands, which have been shown to support more species and a greater proportion of non-natives than natural wetlands (Fennessy and Roehrs 1997). Reinartz and Warne (1993) similarly noted high species richness in one- to three-year-old created wetlands in Wisconsin. They found much greater richness in wetlands seeded with native propagules than in unseeded wetlands. Extensive revegetation efforts were part of more than 80% of the mitigation banks in this study and likely contribute to the high species richness of created and restored wetlands.

These data also provide insight on the effects of geomorphic setting and heterogeneity on wetland development. The off-site placement of mitigation banks is seen both as a serious limitation (Stein 1999) and as a potential advantage (Federal Guidance 1995) for functional replacement. This analysis confirms that site selection plays an important role in mitigation bank vegetation assembly and advances our understanding of two prevailing perceptions. The first is that creation, restoration, and enhancement efforts are not uniformly distributed by geomorphic setting (Gwin et al. 1999). Indeed, depressional mitigation bank wetlands in this study tend to be creations (44%) or restorations (44%), with only 12% enhancements, while riverine and fringe lacustrine systems have more equal representation from all three mitigation methods (25% creation, 33% restoration, 42% enhancement). By comparison, 45% of national mitigation banks of all classifications involve wetland creation, 65% involve restoration, and 62% enhancement (with many banks using more than one method; ELI 2002). In this sample, depressional systems are more frequently creations than enhancements, and the opposite is true for riverine and fringe lacustrine wetlands.

A related perception was articulated by Rogers (1996), who asserted that mitigation wetlands with an adjacent source of native hydrophytic propagules have an improved chance of success over isolated systems, as nearby seed pools may enhance colonization by native wetland plants. The mitigation bank wetlands in this study do not strongly support this hypothesis. Depressional wetlands of all vegetation classes support 10.6±2.1 (mean±SE) species per 10 m$^2$, statistically similar to the 11.5±5.8 species per 10 m$^2$ of riverine wetlands ($p$=0.86, $F_{1,42}$=0.03). There is only a weak difference in percent non-native species by presence between depressional (17.4±2.1) and riverine wetlands (10.2±2.9; $p$=0.08, $F_{1,42}$=3.22). Dividing depressional wetlands by vegetation class allows for further comparison: depressional wetlands with a single vegetation class support significantly more hydrophytic species ($p$=0.04, $F_{1,25}$=4.5) and a greater percentage of non-native species by presence ($p$=0.01, $F_{1,25}$=7.1) than riverine habitats (Figure 3). No significant differences in any category were found between depressional wetlands with a single vegetation class and those with multiple vegetation classes. There is no statistical difference in species richness among mitigation bank wetlands of any geomorphic setting. The only substantial difference in this analysis of geomorphic setting and vegetation class, then, is the significantly smaller presence of non-native species in riverine systems. While this is an intriguing difference, the notion that proximity to natural aquatic ecosystems increases

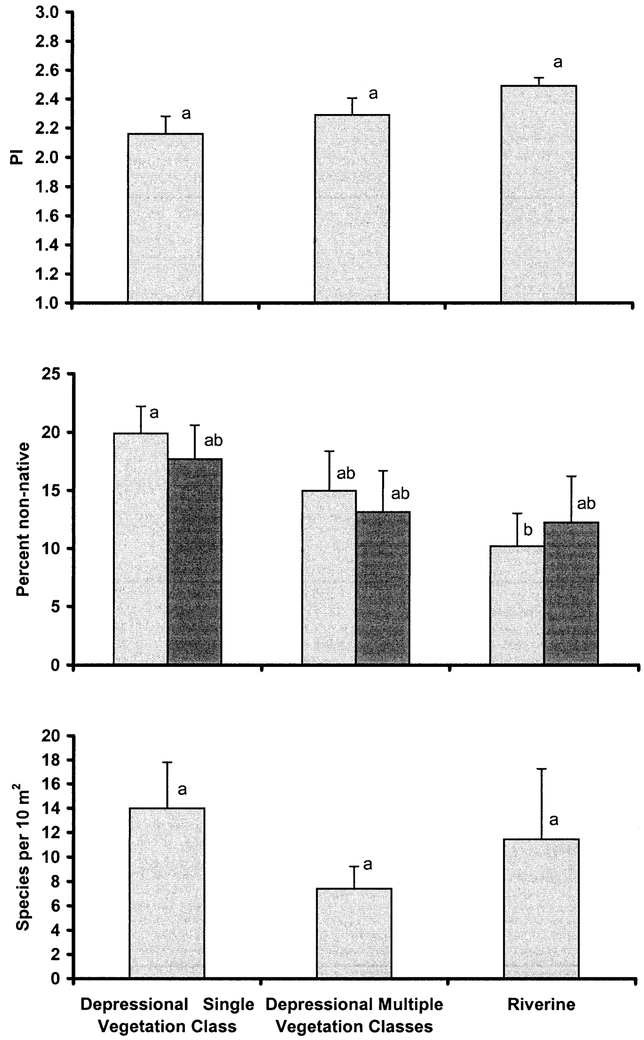


Figure 3. Mean+SE Prevalence Index (PI), percent non-native plant species by presence (light bars) and abundance (dark bars), and plant species richness of mitigation bank wetlands of different geomorphic settings and vegetation class heterogeneity. Letters indicate significant differences among all treatments ($\alpha$=0.05).

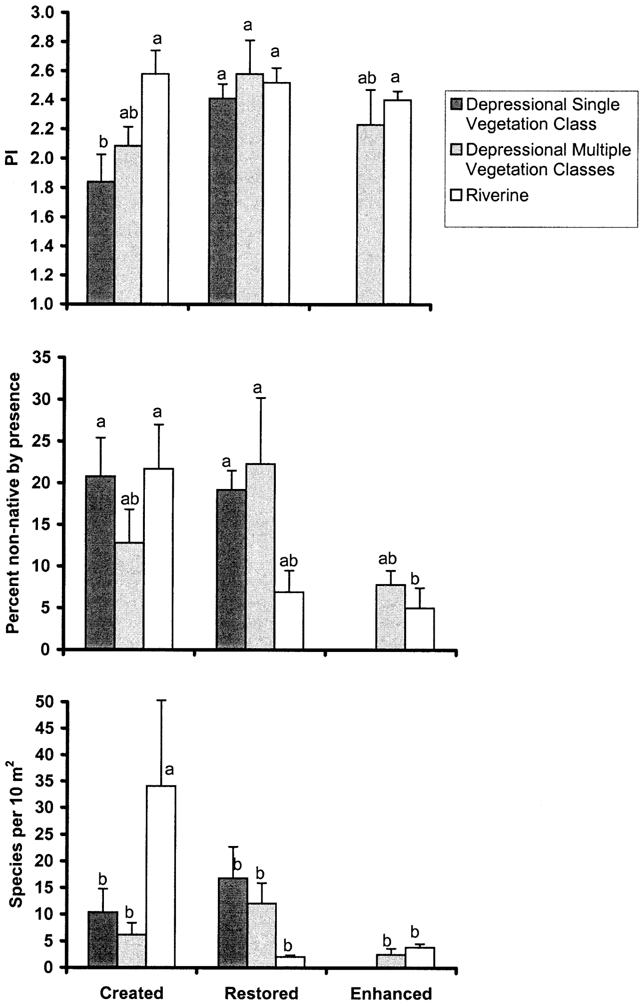


Figure 4. Mean+SE Prevalence Index (PI), percent non-native plant species, and plant species richness of various geomorphic classifications within created, restored, and enhanced mitigation bank wetlands. Letters indicate significant differences among all treatments ($\alpha=0.05$).

recovery of local flora (Cronk and Fennessy 2001) is still open to some debate.

A simple comparison of mitigation banks by hydrogeomorphic class makes it unclear whether it is the mitigation method or the landscape placement that makes these wetlands dissimilar. Nesting wetland class within mitigation method (Figure 4) clarifies this relationship. Created depressional systems with a single vegetation class support highly hydrophytic but low richness, highly non-native communities. Riverine creations support many more species ($p=0.02$, $F_{7,36}=2.68$) but also a significantly greater prevalence index ($p=0.03$, $F_{7,36}=2.7$). Restored wetlands are statistically similar to enhanced wetlands and show a trend of fewer species and a greater prevalence of native species ($p=0.06$, $F_{7,36}=2.1$) in riverine systems as opposed to depressional systems. There is no clear ideal combination of mitigation method, geomorphic setting, and vegetation class heterogeneity—the best choice may well depend on the performance objective. If rapid establishment of a hydrophytic community is deemed a more important goal than richness or native plant presence, created depressional wetlands may be ideal. Riverine creations seem to be particularly susceptible to intense colonization and are likely to achieve high diversity, but include significant representation of facultative and non-native species. If the exclusion of non-native species is the greatest priority, riverine restorations and enhancements seem to have an advantage. In a study of individual mitigation projects in California, Breaux and Serefiddin (1999) noted a vegetation performance standard emphasis on establishment of hydrophytic cover and density, followed by composition requirements, growth measurements, and community structure measurements. These data suggest that performance standards should be tailored to the particular combination of mitigation method and geomorphic setting, thereby requiring a wetland to achieve a target appropriate to its design.

There is considerable concern expressed over the prevalence of non-native species in mitigation banks, with some 30 percent of operational banks specifying performance criteria for non-native species cover (Marsh and Young 1996, ELI 2002). In some cases, mitigation credits are even awarded for control of specific non-native plant species (Albrecht and Wenzel 1996). The ubiquitous presence of non-native plants—averaging 21% of total species of each state of the US according to Kartesz and Meacham (1999)—and the prevalence of non-native species in this study confirms that these concerns are well-founded. Most of the common non-native plants in this study are not obligate wetland plants (Table 3). Only three of the non-native plant species that inhabit more than 20% of mitigation bank wetlands sampled (*Echinochloa crusgalli* (L.) Beauv., *Rumex crispus* L., and *Polygonum persicaria* L.) have an indicator range that includes obligate or facultative wetland; the other ten are facultative to upland plants. This is not to say that non-native obligate or facultative wetland plants like purple loosestrife (*Lythrum salicaria* L.) are absent from mitigation banks, and it does not include undesirable native aquatic species like cattail (*Typha latifolia* L.) and reed canary grass (*Phalaris arundinacea* L). It does imply, however, that the most common non-native plant species are present either because of insufficient inundation or as persistent residents in the new hydrologic regime. To some degree, with a re-established or enhanced hydrologic regime the prevalence of some non-native species may decrease over time.

Table 3. Most common non-native species of mitigation banks analyzed in this study, listed by percentage of banks in which they occur, along with the wetland plant indicator range according to Reed (1988) for the regions in which they were found. Obligate wetland plants (OBL) occur in wetlands >99% of the time, facultative wetland plants (FACW) 67–99%, facultative (FAC) 24–66%, facultative upland (FACU) 1–33%, and upland (UPL) <1%; NI refers to no indicator status.

| Scientific Name | Common Name | Frequency of Occurrence (%) | Regional Indicator Range | | | | |
|---|---|---|---|---|---|---|---|
| | | | OBL | FACW | FAC | FACU | UP |
| *Rumex crispus* L. | Curly dock | 58 | | X | X | X | |
| *Echinochloa crusgalli* (L.) Beauv. | Barnyard grass | 39 | | X | | X | |
| *Trifolium hybridum* L. | Alsike clover | 36 | | | X | X | |
| *Daucus carota* L. | Queen Anne's lace | 28 | | | NI | | |
| *Taraxacum officinale* G.H. Weber ex Wiggers | Common dandelion | 28 | | | | X | |
| *Cirsium arvense* (L.) Scop. | Canada thistle | 28 | | | | X | |
| *Cirsium vulgare* (Savi) Ten. | Bull thistle | 25 | | | | X | X |
| *Trifolium repens* L. | White clover | 25 | | | | X | |
| *Trifolium pratense* L. | Red clover | 22 | | | | X | |
| *Lactuca serriola* L. | Prickly lettuce | 20 | | | X | | |
| *Phleum pratense* L. | Common Timothy | 20 | | | | X | |
| *Polygonum persicaria* L. | Lady's thumb | 20 | X | X | | | |
| *Setaria faberi* Herrm. | Japanese bristle grass | 20 | | | | X | X |

## Comparisons by Area and over Time

The perception that large replacement wetlands are in some ways more desirable than small ones has been fairly consistent throughout the lifespan of mitigation banking (Short 1988, Rogers 1996, NRC 2001). The general argument has been that larger areas provide greater heterogeneity of habitat and successional stages and are thus able to support greater biodiversity and are more resilient to disturbance. The advantages of area are most clear for the support of animals with large ranges, for assimilatory and sequestration capacity of pollutants, and for hydraulic storage capacity, although smaller wetlands can be equally advantageous for providing specialized habitat and biogeochemical efficiency (NRC 2001). Do larger mitigation banks support a plant community that is likely to be more hydrophytic, more diverse, or more native than smaller mitigation banks? Based on the wetlands in this study, it is apparent that they do not. No statistical differences in PI ($p$=0.4, $F_{2,42}$=0.9), representation of native species ($p$=0.37, $F_{2,42}$=1.01), or species richness ($p$=0.11, $F_{2,42}$=2.36) occur among wetlands of different area class. The area of mitigation wetlands may well have a relationship with some aspects of ecological development, but for the vegetation parameters measured here, it is clear that area is less important than mitigation method or geomorphic setting.

Another common perception is that replacement wetlands become more similar to natural wetlands as they age. The hypothetical smooth trajectory toward increasing ecosystem function has been called into question (Zedler and Callaway 1999), and it is likely that functional equivalency is unachievable in all but the lowest stress systems within the short time frame of mitigation monitoring. Data from this study show that mitigation bank wetlands do not develop vegetation at the same rate, but the trajectory over the first five years of development is not entirely clear (Figure 5). Created and restored wetlands show weak trends toward a decreasing PI ($p$=0.10, $F_{8,44}$=1.8) and decreasing non-native presence ($p$=0.12, $F_{8,44}$=1.17). Less clear are the changes in species richness over time, which are highly erratic ($p$=0.06, $F_{8,44}$=2.02). Enhanced wetlands show no significant change in PI, non-native presence, or species richness over five years. These could be positive indicators, as they suggest trends toward a greater presence of native hydrophytes, particularly in created and restored wetlands. However, the high degree of variability makes any trends arguable at best; it seems clear that these plant communities are still in a period of self-organization (Odum 1989) and self-design (Mitsch and Wilson 1996). The successional trajectory and speed of development likely vary among mitigation bank wetlands according to factors like land treatment, soil structure, and seed-bank availability.

A more compelling trend is illustrated in a time-series plot of $z$, the slope of the species accumulation curve for each site (Figure 6). The $z$ value indicates the likelihood that new species will be encountered with increasing area sampled. For the wetlands in this study, $z$ decreased over time and can be described by a logarithmic function ($y=-0.3030\ln(x)+1.0961$; $r^2$=0.73; $p$=0.016, $F_{1,50}$=6.24). A decreasing $z$ over time suggests increasing homogeneity within mitigation wetlands. The average $z$ value in one-year-old

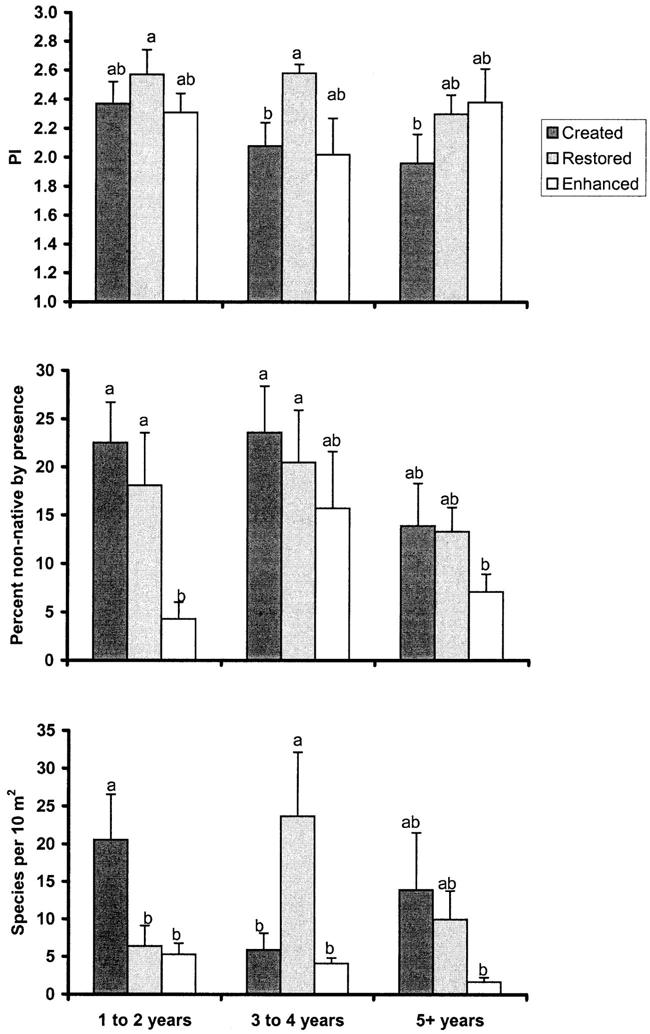


Figure 5. Mean+SE Prevalence Index (PI), percent non-native plant species, and plant species richness of created, restored, and enhanced mitigation bank wetlands as segregated by age since establishment. Letters indicate significant differences among all treatments ($\alpha$=0.05).

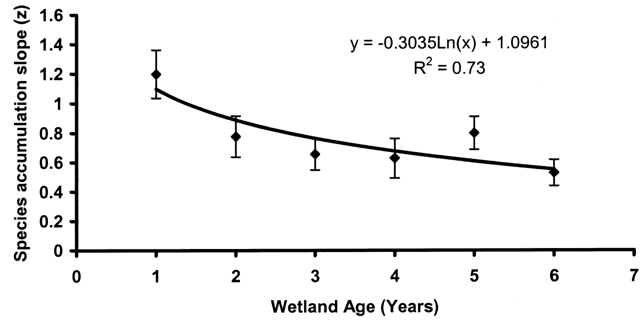


Figure 6. Regression of mean±SE slope values ($z$) for species accumulation curves of mitigation bank wetlands of various ages since establishment. Slope of trendline is significantly different than zero ($p$=0.016, $F_{1,50}$=6.24).

mitigation bank wetlands (1.2±0.16) greatly exceeds the $z$ values reported in natural populations (0.2 to 0.4; Connor and McCoy 1979)—values increasingly approached by older wetlands. Although temporal changes in $z$ have not been thoroughly explored for created or restored wetlands, it has been noted that species accumulation slope for a variety of ecosystems may decrease with source pool size (i.e., the pool of species that are candidates for immigration and establishment) (Martin 1981). In the case of mitigation wetlands (or any newly established ecosystem), decreasing species accumulation slope may simply be a function of decreasing area available for new species establishment and increasing competitive exclusion among species already established. This process may occur in wetland enhancements that are subjected to anthropogenic introduction of species at the outset, but it is particularly likely in created and restored wetlands that experience both the large scale disturbance of the creation or restoration effort and a rapid post-disturbance influx of species. Figure 6 provides further evidence for this early self-organization of mitigation bank wetlands, and it suggests that assessments based only on the first five years of development will not depict the true character of the vegetation community the wetland will ultimately support.

## CONCLUSIONS AND RECOMMENDATIONS

Mitigation bank performance standards include a wide variety of criteria by which the replacements wetlands are to be evaluated. In this study, I used three metrics that should be common goals for all: the establishment of hydrophytic vegetation, the prevalence of non-native species, and the number of species per unit area. I suggest that these fundamental measures can be used to evaluate mitigation banks across ecoregions, mitigation methods, and geomorphic settings, and given the variability in performance standards and monitoring methods, these may be among the only measures common enough for a broad comparison. How successfully, then, are mitigation banks developing wetland vegetation? The answer depends on one's standards for success, for which there is little precedent. Using arbitrary and fairly modest standards of a PI less than 2.75, non-native presence no greater than 21% of total species (after Kartesz and Meacham 1999), and species richness greater than 2 species per 10 $m^2$, twenty-two out of the 45 (49%) wetlands considered in this study are successful. Of those that fail, 26% have a high PI, 70% have excessive non-native presence, and 30% have fewer than 2 species per 10 $m^2$. Five wetlands fail to meet two standards, and one fails to meet all three. Considering the same standards for only those wetlands 5 years old and older (n=19), 63% are successful.

Similar analyses of success based on myriad criteria have been conducted for individual mitigation projects in various regions of the United States. Allen and Feddema (1996) found that 43% of 75 California mitigation wetlands met all criteria for success. In Pennsylvania, 62% of 13 mitigation sites were judged successful by Cole and Shafer (2002). Morgan and Roberts (2003) reported that 40% of 50 mitigation projects in Tennessee met the area requirement, but an abysmal 4% met all permit requirements. Sixty-four percent of mitigation sites across many habitat types in Indiana were considered successful by Robb (2002), and 65% of 114 were in compliance in Massachusetts (Brown and Veneman 2001). Mitigation banks, which ostensibly should be held to higher standards, do not seem to be dramatically more successful than individual mitigation projects. However, this study shows that success, depending on the measure of success, varies by mitigation method, geomorphic setting, and time. With a better understanding of this differential success and a more systematic approach to placing mitigation banks by method and type, we may achieve more effective results.

One may argue that mitigation banks are replacing biogeochemical, hydrologic, and habitat-based functions more effectively than the basic measures of vegetation indicate here, but the data to support such an argument simply do not exist. Indeed, even vegetation data are poorly accounted, with 42% of mitigation banks surveyed having no data available or insufficient field data for this analysis—a reality echoed in analyses of individual mitigation wetlands by Cole and Shafer (2002) and Robb (2002). A quantifiable, field-based vegetation monitoring scheme should be a basic requirement of all mitigation banks. Further, our understanding of the success of mitigation banks is not likely to advance with monitoring regimes that terminate after five years. In this study, over one-third of the banks have been in operation for more than five years, but analysis was limited to the age of the final monitoring report. Successional trajectories clearly extend longer than five years, and in my estimation, we could learn much more about the structure and function of different types of mitigation banks with a relatively small investment by extending monitoring programs. For if we have learned anything about replacement wetlands, it is that mitigation based solely on area is inadequate. With modifications to monitoring criteria, durations, and procedures, functional evaluation can provide a more meaningful assessment of the degree to which we are achieving no net loss.

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