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Wetland Ecology Group
Division of Surface Water

INTEGRATED WETLAND ASSESSMENT PROGRAM. Part 5: Biogeochemical and Hydrological Investigations of Natural and Mitigation Wetlands

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INTEGRATED WETLAND ASSESSMENT PROGRAM.
PART 5: Biogeochemical and Hydrological Investigations of Natural and Mitigation Wetlands

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ABSTRACT

We performed a comprehensive investigation of the biota (structure) and biogeochemical cycles (processes or functions) of a population of natural (n = 9) and mitigation wetlands (n = 10). Intensive data were collected on various wetland ecosystem components including: hydrology, soil and water chemistry, characteristics of the plant, macroinvertebrate and amphibian communities, biomass production, decomposition, and nutrient cycles. The goals of the project were as follows: 1) to demonstrate the efficacy of floral and faunal community-based indicators in order to assess the performance of mitigation wetlands, 2) determine the links between floral and faunal community structural attributes and ecosystem processes in natural and mitigation wetlands, 3) compare the biological and physical characteristics, as well as patterns of biogeochemical cycling in natural and mitigation wetlands in order to assess their relative condition, and 4) identify simple, cost-effective biogeochemical indicators for use in mitigation monitoring and as performance standards. The biological and biogeochemical characteristics of the natural and mitigation wetlands were substantially different. The mitigation wetlands were generally "dryer" than the natural sites based on measures of ground water. Mean depth to ground water averaged -53.8 $\pm$ 11.1 cm and -25.0 $\pm$ 6.1 cm in the mitigation and natural sites, respectively in 2001 ($p = 0.04$) and -44.5 $\pm$ 9.1 and -25.4 $\pm$ 4.9 in 2002 ($p = 0.09$). Concentrations of soil organic carbon (%OC), %N, and plant available P (µg P g^{-1} soil) were 4.8 times, 4.3 times, and 1.6 times higher in the natural compared to the mitigation sites. Mean values for soil bulk density and percent solids were significantly higher in the mitigation wetlands ($p = 0.001$). These measures quantify the extremely heavy soils found in the mitigation sites that may lead to reduced root growth and limit carbon accumulation. The Vegetation Index of Biotic Integrity (IBI) scores for natural sites ranged from 9 to 82, reflecting the fact that the natural wetlands were selected along a gradient of human disturbance. The range of scores for mitigation wetlands was narrower, ranging from 16 to 50. This

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compression of scores is due in part to the fact that the community composition of the mitigation sites is similar, with a dominance of ubiquitous, tolerant plant species. Mean VIBI scores were more than twice as high at natural wetlands ($p = 0.005$). Aboveground biomass production was also significantly higher in the natural sites ($p = 0.04$) where production averaged 34.7 g 0.1 m^{-2} compared to 20.9 g 0.1 m^{-2}. The invertebrate data showed major differences in the numbers of taxa, abundance of tolerant and sensitive species, and the community metrics in the Wetland Invertebrate Community Index scores between the mitigation sites and natural sites. Taxa richness averaged 46 in natural sites compared to 34 at mitigation sites. Amphibian communities of the mitigation wetlands differed markedly from natural forest and shrub dominated wetlands. However, amphibian communities of natural emergent wetlands and the mitigation sites were similar and factors like permanence of hydrology and presence of predatory fish appeared to be more important in determining amphibian community composition. Despite this, average Amphibian Index of Biotic Integrity scores were 0.3 for the mitigation sites in this study and 6.5 for the natural emergent sites. Both decomposition rates and litter nutrient concentrations were higher in the natural wetlands. The soil and water data demonstrate the low levels of organic carbon contained in the mitigation sites. Low organic carbon levels can limit the activity of decomposers (heterotrophic microbes and invertebrates), limiting diversity and leading to slower rates of decomposition. Multivariate analyses show that the natural and mitigation sites group as two separate populations, indicating that wetland mitigation is currently creating a new subclass of wetlands on the landscape. Based on the results of this study, several indicators could serve as measures of mitigation performance relative to natural wetlands: 1) soil chemical and physical characteristics especially soil organic carbon and soil nitrogen content and percent solids in the soil or bulk density; 2) hydrological characteristics including mean depth to ground water and percent time water is found in the root zone; and 3) multimetric indices developed from natural reference wetland data sets.

INTRODUCTION

In response to Sections 401 and 404 of the Clean Water Act, freshwater wetlands are being created and restored at great frequency in the United States as "replacement" or "mitigation" wetlands that are meant to compensate for wetland loss (Zedler 1996, Race and Fonseca, 1996, Fernandez and Karp 1998, Zedler 2000, NRC 2001). A persistent question has been whether or not these created or restored wetlands are structurally or functionally equivalent to those they replace. In other words, is wetland creation a fair trade? The creation of wetland ecosystems to replace natural ones has been referred to as a large-scale ecological experiment because of the uncertainty about the success of this practice.

Despite the recognized importance of wetland functions for providing services such as water quality improvement, flood control, and aquatic life habitat, there are limited data showing the links between ecosystem structure (which regulations or permits often specify as mitigation goals) and ecosystem functions (which regulations often specify as what should be maintained through the mitigation process). Where performance standards exist, they are typically based on measures that were neither derived nor tested with empirical data relating them to ecosystem processes or to natural, reference wetlands. For example, macrophyte cover and species richness are often used as determinants of legal and ecological success (Figure 1), but these standards are not usually evaluated against reference wetland data sets or related to ecological function (Zedler and Callaway 1999; Mitsch et al. 1998; NRC 2001; Cole 2002). Ecologically sound performance standards are a critical component of an effective wetlands program.

Mitigation projects may be classified as creation, restoration or enhancement projects. Wetland creation is the conversion of an upland whereas restoration is defined as the return of a previous wetland from a disturbed or altered condition. Enhancement involves "improving" the condition of an existing wetland (Mitsch and Gosselink 2000). When replacing wetlands, restoration projects have generally been judged as more successful than creation efforts because of the higher probability that remnant seed banks, natural hydrology, and hydric soils will be present (Kusler and Kentula 1990). Wetlands included in this project include creation and restoration projects; we refer to both as "mitigations" or "mitigation wetlands" throughout this report.

An intensive review and assessment of wetland mitigation projects was recently published by the Committee on Mitigating Wetland Losses conducted by the National Academy of Sciences National Research Council (NRC 2001). The committee concluded that "...the goal of no net loss of wetlands is not being met for wetland functions by the mitigation program, despite progress in the last 20 years" (NRC 2001). It is apparent that the methods of wetland mitigation need to be improved, and these methods applied to projects on the ground. The National Research Council (NRC 2001) made several recommendations in order to improve the outcome of wetland restoration and creation projects: (1) consider both the structure and function of wetland ecosystems, and understand better the relationships between them; (2) reference wetlands should be used as a model for the dynamics of created or restored sites; (3) the science and technology of wetland restoration and creation must be broadened to include sites that differ in degree of disturbance and restoration effort in order to improve the predictability of the outcomes; (4) mitigation wetlands should be self-sustaining; (5) hydrological variability is important in the structure and function of created and restored wetlands; (6) a broader range of functions should be both required and measured for mitigation projects (7) the destruction of wetlands that are particularly hard to restore (e.g. very high quality wetlands) should be avoided.

The goal of this study was to do a comprehensive investigation of the biota (structure) and biogeochemical cycles (processes or functions) of a population of natural and mitigation wetlands using a study design that incorporates five of the recommendations in NRC (2001) (1, 2, 3, 5, and 6):

1. To demonstrate the efficacy of using floral and faunal community-based indicators to assess the performance of mitigation wetlands;

2. To investigate the linkages between flora and faunal community structural attributes and ecosystem processes in natural and mitigation wetlands;

3. To investigate the biological characteristics and biogeochemical cycles of the wetlands in order to assess the condition of mitigation sites as compared to natural sites;

4. To investigate the hydrology of the wetlands in order to assess the hydroregimes of the mitigation sites as compared to the natural sites;

5. To identify simple, cost-effective biogeochemical indicators for use in mitigation monitoring. These measures will then be translated to performance standards.

Intensive fieldwork was conducted at natural and mitigation wetlands in order to collect data on various wetland ecosystem components (e.g. hydrology, soil, plant community composition and productivity, macroinvertebrate and amphibian community composition, decomposition, and nutrient cycling). The data from this large-scale field study provides us with information pertaining to the biological and physical characteristics and biogeochemical cycles of each wetland so that we may assess the condition of mitigation sites as compared to natural sites. If mitigation wetlands are found to be substantially different from natural sites, then characterizing these differences will help diagnose possible causes for the lack of success at mitigation wetlands, and establish ecologically relevant performance goals. The results of this work have been translated into standardized monitoring, design and performance protocols for mitigation wetlands (Part 6 of this series) (Mack et al. 2004).

Wetland Definitions

Wetlands are usually defined by the presence of three parameters: periodic or continuous soil inundation or saturation (wetland hydrology); soils that have developed under anaerobic conditions (hydric soils); and, vegetation that is adapted to anaerobic conditions (hydrophytic vegetation) (Environmental Laboratory 1987). The wetland hydrology criteria requires that the water table is within 30 cm of the soil surface for a continuous period for more than five percent of the growing season (Environmental Laboratory 1987). Hydrophytic (wetland vegetation) occurs when, under normal circumstances, more than 50 percent of the composition of the dominant species from all strata are obligate wetland (OBL), facultative wetland (FACW), and/or facultative (FAC) species (Environmental Laboratory 1987, Reed 1988). Last, a soil is considered "hydric" when it is formed under conditions of saturation, flooding, or ponding, long enough during the growing season to develop anaerobic conditions in the upper part (Environmental Laboratory 1987).

Hydrology

Hydrology is considered the master variable of wetland ecosystems, driving the development of wetland soils and leading to the development of the biotic communities (Mitsch and Gosselink 2000). It can determine plant species composition as well as the distribution of species within a wetland (for example, vegetation zonation with depth in freshwater wetlands), their productivity and capacity for nutrient uptake (Cronk and Fennessy 2001). Despite this fact, quantitative hydrologic data is not often collected as part of mitigation monitoring.

Hydroperiod (the pattern of water levels over time) has been called the most important predictor of future wetland success (Mitsch and Jorgensen 2004). Hydrological modifications (at natural or mitigation sites) can drastically alter ecosystem processes such as primary productivity

(Mitsch 1988) and species composition. Some studies have argued that mitigation wetlands can never achieve parity with natural wetlands if the hydrology is not correct (Magee et al. 1999, Cole and Brooks 2000, Craft et al. 2002). Other studies have found that even if hydrologic parity is achieved, restored wetlands may not develop plant communities similar to natural wetlands (Galatowitsch and van der Valk 1996a, b, c; Mulhouse and Galatowitsch 2003). In addition, inadequate hydrologic restoration or hydrologic disturbance often leads to colonization by invasive species. For example, Owen (1999) discovered that as the result of landscape development and the hydrological changes that occurred, *Carex* spp. (native wetland sedges) were replaced with *Phalaris arundinacea* (reed canary grass), *Typha angustifolia* (narrow-leaved cattail), *T. latifolia* (common cattail), and *T.* x *glauca* (hybrid cattail).

A wetland's hydrogeologic setting is the position of the wetland within the landscape in conjunction with geological characteristics such as topography, slope, thickness and permeability of soils, and the resulting flows of surface and ground water. Hydrogeologic settings have been described as the "templates" for wetland development (Winter 1988, 1992; Bedford 1996; Bedford 1999). The diversity of wetland templates including their type, abundance and spatial distribution) can be summarized in a wetland landscape profile. Templates are the result of hydrologic variables operating at the landscape scale that generate and maintain different wetland types (classes). In this way regional hydrogeologic and hydrogeomorphic settings determine wetland types and locations (i.e., the profile) that are sustainable in a particular landscape. Attempts to restore wetlands that are equivalent to those that were destroyed involves a greater understanding of the landscape and the ways the landscape may affect wetland function (Bedford 1996). Despite the importance of wetland hydrology and its relationship to the surrounding landscape, there is still uncertainty about how watershed position and wetland placement affects the success of restoration efforts (Zedler 2000).

Soils

Hydric soil serves as the physical foundation that can influence the development and maintenance of both ecosystem processes and the composition of the biological communities (Stolt et al. 2000). Many factors affect hydric soil development including hydrology, organisms, topography, climate, parent material, vegetation and time (Mitsch and Gosselink 2000). Water drives the formation of hydric soils, adding material through deposition of eroded sediment, removing solids and dissolved materials, and influencing the breakdown of plant litter into organic matter (Stolt et al. 2000). Wetland soils are formed as the result of periodic to continuous inundation; soil saturation leads to anaerobic soil conditions and reduced decomposition, which results in the build up of organic matter (Craft 2000). As organic matter content in a soil increases, bulk density decreases due to reduced particle density of the organic material compared to mineral soil (Craft 2000). Organic matter plays an important role in plant community dynamics by reducing wind and water erosion, supplying nutrients, retaining moisture and reducing water evaporation.

Currently, there are no methods or indicators that have been proven useful to determine if the soil in newly constructed wetlands will develop characteristics similar to natural wetlands. If hydric soils do not develop, plants suitable or characteristic of wetland environments may not successfully colonize or persist. For example, a San Diego Bay mitigation site that was constructed in order to provide habitat for the endangered light-footed clapper rail (*Rallus longirostris* subsp. *levipes*) failed because soils in the created marsh had a higher percentage of sand compared to natural marshes in the region (Zedler et al. 2001). Because of its physical properties, the soil did not retain or supply sufficient levels of nitrogen for plant growth, nor did it contain the normal level of soil organic matter found in natural marshes. As a result, the created marsh did not develop structural (plant height) or functional (habitat for the endangered clapper rail) properties similar to the natural marsh it replaced.

Vegetation

The assemblage of plant communities within a wetland is determined by the initial conditions of the site (i.e. presence or absence of wetland seeds and propagules) and associated environmental factors (i.e. flooding, temperature, nutrient availability) (Mitsch and Gosselink 2000). Plant community dynamics have been described by the individualistic model of species distribution, where community composition is regulated primarily by physical (allogenic) processes to which each species responds according to its individual life history (Mitsch and Gosselink 2000, van der Valk 1981). This model, based on H.A. Gleason's theoretical definition of succession, states that each individual organism in a community is present due to its unique combination of adaptations to the environment (Cronk and Fennessy 2001, Middleton 1999, van der Valk 1981).

Establishing vegetation in mitigation wetlands is a complex process, involving a basic understanding of each site's underlying ecological processes. One long-standing debate concerns the best technique for establishing vegetation (Middleton 1999; Streever and Zedler 2000). Proponents of the "self-design" theory espouse the idea that over time a wetland will restructure itself around the forcing functions that have been put in place (Mitsch et al.1998, Mitsch and Jorgensen 2004). Others support the "designer" theory which suggests that it is not a matter of time but of intervention that determines the outcome of creation projects (Middleton 1999). Unfortunately, neither theory has been thoroughly evaluated and both fail to address the fundamental issue of how you define success. For example, in Mitsch et al. (1998), two created wetland basins were established to test the self-design theory. One basin was planted (13 species at a density of ~1 plant per 2 $m^{2)}$; the other basin was unplanted. Plant communities converged relatively quickly at both sites with the majority of the species being wetland annuals or tolerant perennials that recruited naturally. Evaluations of both approaches would be well-served by comparison of created sites to biological and biogeochemical characteristics of reference wetland data sets.

Our knowledge of successful revegetation techniques is lacking because many projects are still relatively "young" (constructed over the last 5 to 10 years), studies of mitigation wetlands are often limited to a few sites and types of wetlands, and there are often incomplete monitoring records. Stauffer and Brooks (1997) determined that in some circumstances, planting or constructing wetlands using remnant seed banks or salvaged marsh surfaces, could accelerate the process of vegetation or at least soil development. Reinartz and Warne (1993) concluded that seeding creation projects can limit the invasion of non-native or invasive species. Brown and Bedford (1997) found that soil transplants were effective at improving the establishment of plant communities and preventing the establishment of invasive species (e.g. *Typha* spp.). However, a close review of the species lists in these studies generally reveals that the increases in "diversity" are due to the establishment of upland weeds, wetland annuals, and tolerant wetland perennials. Again, comparison to the biological and biogeochemical characteristics of reference wetland data sets would provide an objective yardstick with which to evaluate the species assemblages established from this practice.

Long-term studies of plant community assembly and succession are essential if we are to understand management options and improve mitigation success. Plants often respond both quickly and visibly to environmental stressors such as an alteration in hydrology, land use, high nutrient input, sediment loads, or herbivory. A wetland's ability to support certain plant species can serve as an indicator of its capability to sustain specific functions and biological processes (Lopez and Fennessy 2002; Cronk and Fennessy 2001; Fennessy et al. 2001). This is the basis for indices such as the Floristic Quality Assessment Index (FQAI; Wilhelm and Ladd 1988) and the Vegetation Index of Biotic Integrity (VIBI) developed by the Ohio EPA (Mack et al. 2000, Mack 2001b, Mack 2004b) and other vegetation-based wetland IBIs (e.g. Gernes and Helgen 1999, Carlisle et al. 2000, Simon and Rothrock 2001).

Faunal Communities: Amphibians and Macroinvertebrates

Amphibians are keystone species that prey on insects, invertebrates, other amphibians and detritus. They also serve as a food source for predacious invertebrates, other amphibians, reptiles, birds, mammals and fish. Additionally, amphibians are well recognized as sensitive indicators of environmental conditions, and many amphibian species are dependent on wetlands to provide habitat for some or all of their life stages (Wyman 1990, Wake 1991, Griffiths and Beebe 1992). The composition of surrounding upland habitats are often just as important to amphibian species as the wetlands themselves (Semlitsch 1998, Porej et al. 2004).

Macroinvertebrates are also important wetland species for many reasons. They are closely tied to wetland habitat, depending on wetland pools for recruitment. Macroinvertebrates are herbivores, detritivors, and predators and are involved with multiple wetland ecosystem processes. Their community composition is responsive to minor disturbances in the ecosystem, making them effective bioindicators (Hecnar and M'Closkey 1996, Adamus et al. 2001, Sparling et al. 2001, Helgen 2002, Lillie et al. 2002).

Despite their importance, there are relatively few studies focusing on the ability of mitigation wetlands to support healthy amphibian and macroinvertebrate communities. The National Research Council (NRC) pointed out the lack of data on most animals in natural wetlands and stated "...biological dynamics must be evaluated in terms of the animal [amphibian and macroinvertebrate] populations present and the ecological requirements of the species" (NRC 2001) In this study we addressed the questions whether amphibian and macroinvertebrate populations in mitigation wetlands differ from natural wetland populations and whether faunal bioindicators (Micacchion 2002, 2004) can be used to evaluate mitigation success.

Wetland Ecosystem Processes

Details on the structural components of wetland ecosystems are covered in great depth in recent texts (e.g. Mitsch and Gosselink 2000; Keddy 2000). Underlying biogeochemical processes ('functions') also play an important role in the development and maintenance of these structural components. Our study was designed specifically to quantify the processes of biomass production, decomposition rates, and nutrient dynamics in order to link structural and functional variables.

Biomass production is a common measure of primary productivity (the conversion of solar energy into chemical energy per unit area per time). It is a useful measure of ecosystem function because it integrates many environmental variables such as vegetation composition, soil nutrient composition, climate, and hydrology (Brinson et al. 1981, Cronk and Fennessy, 2001, Fennessy et al. 2001). The effects of hydrology on primary productivity (i.e. biomass production) have been extensively studied among different wetland types. Water level affects biomass production through changes in the depth, frequency of flooding, duration of flooding, and regularity of inundation (Cruz 1978). In general, wetlands exposed to flow-through conditions have higher levels of primary productivity than wetlands with stagnant conditions (Brinson et al. 1981, Middleton 1999, Craft 2001). In permanently inundated wetlands, compared to wetlands that experience a dry-down period (a period in which there is little to no standing water), the levels of biomass production are typically much lower (Conner and Day 1976, Mitsch 1988).

Cole (1992) studied the biomass production of four wetlands developing on a reclaimed coal-surface mine. The biomass of these created wetlands was low, between 30.6 g m^{-2} and 108.4 g m^{-2}. The highest biomass production occurred in a site dominated by *T. latifolia*. Cole (1992) suggested that low biomass production was the result of low organic matter content and soil moisture. Other studies have hypothesized that soil structure and the availability of soil nutrients influences the biomass production of wetland ecosystems (Cruz 1978). When soils are inadequate, the vegetation community is slow to form, which may lead to low biomass production and the slow recycling of nutrients and organic matter within the system.

Decomposition

Decomposition is a complex biological process, its dynamics are poorly understood (Mitsch and Gosselink 2000). Plant litter decomposition, or the breakdown of vascular plants and woody debris, is one of the least studied functions of wetlands but is vital to biogeochemical cycles. Decomposition represents a crucial feedback loop that recycles and transfers nutrients and mediates the accumulation of organic matter. Three main stages characterize the decomposition of leaf litter into organic matter: an initial rapid loss due to leaching, a period of microbial mineralization, and a period of mechanical and invertebrate fragmentation (Webster and Benfield 1986).

Decomposition is affected by many variables including soil composition, plant nutrient composition (C:N ratio of the litter), frequency of flooding, dissolved oxygen concentration, pH, and temperature (Brinson et al. 1981, Webster and Benfield 1986, Vargo et al. 1998, Battle and Golladay 2001). Biogeochemical properties of plant litter, particularly its nitrogen content or C:N ratio, are known to influence decomposition rates (Day 1982, Valiela et al. 1984; Lee and Bukaveckas 2002). Water column nutrient availability has also been shown to be a significant predictor of decomposition rates (Verhoeven et al. 1996, Lee and Bukaveckas 2002). For instance, Peterson et al. (1993) documented increases in decomposition in whole ecosystem experiments upon the addition of nitrogen and phosphorus.

Flooded or saturated conditions lead to low oxygen availability, thus low redox potentials that slow the process of decomposition, leading to organic matter accumulation (Brinson et al. 1981, Webster and Benfield 1986). The slow breakdown of plant material that is characteristic of anaerobic conditions is also attributed to low levels of microbial activity (Arp et al. 1999). Microbial activity is promoted by aeration of the soil; pulsing conditions are optimal for microbes whereas permanently anaerobic conditions often inhibit microbial activity. Battle and Golladay (2001) found that exposure to anaerobic conditions significantly decreased decomposition in permanently saturated conditions compared to rates under multiple flooding events. Because of differences in wetland structure and the frequency and extent of flooding among wetland ecosystems, generalizations about decomposition in wetlands are difficult to make (Day 1982). The majority of wetland decomposition studies have focused on the decay rates of leaf litter in natural systems so little is know about the decomposition process of mitigation wetlands (Atkinson and Cairns 2001).

The decomposition of plant litter returns nutrients previously bound in organic form to the soil or water column. Plant primary productivity in many ecosystems is largely dependent on this nutrient recycling, particularly if other pathways of nutrient input are low (Aber and Melillo 1991, Gartner and Cardon 2004). Nutrients released through decomposition are also important for use by detritivors whose nutrient requirements (expressed as C:N ratios for example) are higher than plant litter can supply. In the decomposition process nutrients are initially leached due to mechanical breakdown by invertebrates and other organisms, followed by mineralization of organically bound nutrients by microbial activity. Upon release, some proportion of the available inorganic nutrients are absorbed by the remaining litter (nutrient immobilization); this is one useful measure of nutrient availability and microbial activity within a particular wetland ecosystem (Benfield and Webster 1986). To a large extent, net primary productivity and decomposition control nutrient uptake and retention in any ecosystem.

Plant community structure and ecosystem function

Major approaches to wetland assessment like the IBI (index of biotic integrity) and HGM (hydrogeomorphic) methods, assume that if measurable community "structural" attributes deviate little from "reference" conditions, then the functions supporting that structure are also operating at reference levels (Stevenson and Hauer 2003). However, holistic studies on the links between wetland plant community structure and ecosystem function are rare. A notable exception is a series of papers on restoration of prairie pothole wetlands (Galatowitsch and van der Valk 1996a, b, c; Mulhouse and Galatowitsch 2003). While hydrologic conditions (hydroperiod, basin surface area) similar to natural prairie pothole wetlands were usually restored, the surface water of the restored potholes had higher pH and lower alkalinity, conductivity, and calcium and magnesium concentrations than natural reference wetlands; carbon content of soils was lower and bulk density higher in the restored versus natural prairie potholes Galatowitsch and van der Valk 1996c). While initial recolonization by wetland species happened relatively quickly at most sites, species and plant communities (notably sedge meadows) characteristic of prairie potholes did not develop after 3 years Galatowitsch and van der Valk 1996a, b), and 12 years post-restoration, most sites had diverged even further from reference conditions and were often dominated by invasive perennials like *Phalaris arundinacea* (Mulhouse and Galatowitsch 2003). Studies on terrestrial ecosystems have suggested that increasing species richness is correlated with the rates of ecosystem function (Kareiva 1996, Tilman et al. 1996, Schlapfer and Schmid 1999), however, others have criticized these findings by attributing differences to variations in experimental design or questionable data interpretation (Grime 1997, Doak 1998, Allison 1999).

Nutrient cycling, productivity, and decomposition rates have been implicated as responding to plant diversity. Chapin et al. (1997) hypothesized that the relationship between diversity and ecosystem processes is due to the functional traits of the species present which accrue into ecosystem level processes, which in turn feed into regional processes. For example, in wetland systems that have been altered by human disturbance, species composition may shift towards invasive, monoclonal species such as cattails or reed canary grass with concomitant increases in productivity and altered nutrient cycles (Windham and Ehrenfeld 2003). Naeem et al. (1996) provide an example of the importance of species composition in a study of grasslands, finding that the most productive species were 25 times more productive than the least productive species.

Decomposition is also predicted to respond to changes in diversity. For example, higher plant diversity is also expected to lead to higher quality litter caused by the high nutrient retention in plant litter (Hooper and Vitousek 1998), which in turn should support faster rates of decomposition. Odum (1985) proposed general trends that can be expected in stressed ecosystems including an increase in the relative abundance of tolerant species, a decrease in the size of plant species, shortening of food chains due to reduced energy flow at higher trophic levels, a decrease in the lifespan of organisms, and a decline in diversity and associated dominance by a few species. Based on this, we hypothesize that both primary productivity and decomposition will vary with wetland condition. Specifically we expect productivity to decline and decomposition to increase as ecological condition improves.

METHODS

Site selection

Nine natural wetlands and 10 mitigation wetlands located throughout Ohio were selected for this study (Table 1). In the selection of natural sites we considered: the relative degree of disturbance, landscape position, dominant vegetation, and site access. All of the study sites were emergent wetlands that can be classified as "mixed emergent marshes" or "cattail marshes" (Mack 2004a). Natural wetlands were intentionally selected to include highly disturbed, somewhat disturbed, and relatively undisturbed (reference standard condition) sites. The nine natural sites included 3 highly disturbed "nonreference" sites (Dever, Lake Abrams and Lodi North) and six "reference" sites (Baker Swamp, Ballfield, Calamus, Eagle Creek Beaver, Eagle Creek Marsh, and Rickenbacker) ranging in condition from moderately good to excellent.

Nine of the mitigation wetlands were projects constructed pursuant to individual Section 401/404 permits and were selected to represent a range of ages (Table 1). One site (Sacks) was voluntarily constructed as part of the Wetland Reserve Program. The mitigation wetlands ranged in size from 0.15 ha (0.37 a.) to 10.4 ha (25.7 a). Seven of the 10 mitigation wetlands were regularly to permanently inundated, 6 of 10 had large areas of unvegetated open water, and 5 of the 10 had populations of predatory fish.

In order to more extensively test the performance of the mitigation wetlands included in this study, we took advantage of a much larger data set collected at natural wetlands, including emergent marshes, in Ohio over the period 1996 – 2002 (Fennessy et al. 1998, Mack et al. 2000, Mack 2001b, Mack 2004b). This data was collected as part of the development of wetland biological assessment tools for the state of Ohio, and includes data from natural emergent marshes that span the full range of disturbance.

We took an ecosystem level approach in this study, including measures of the components and processes that were most likely to (1) illustrate any differences between the two populations of wetlands, (2) provide us with possible diagnostic capabilities to make recommendations on improving mitigation project success, and (3) derive indicators from this data for use as performance standards.

Hydrology

Shallow ground water level monitoring wells were installed at each site (Model WL-40, Remote Data Systems, Inc.). The WL-40 water level recorder has a built in data logger attached to a 101.6 cm (40 in) long copper wire that is inserted into a slotted well screen. Water level in the well is measured by sending a small electrical pulse down the copper wire. The data logger records the level of water around the wire. Wells were usually placed just up gradient of the areas of standing water at the edge of the wetland pools in locations where inundation of the data logger was unlikely and away from public view to avoid vandalism.

Wells were installed by auguring a hole with a posthole digger, backfilling the hole with a few inches of sand, inserting the well into the hole and backfilling the bore hole with 20/40 sand, and grouting the top of the hole with clay. After installation, the distance between ground surface and the calibration point was measured. Wells were not usually installed as far as the calibration point. Well holes were only excavated until impermeable clay layers were reached in the B or C horizons. Wells were programmed with the Hewlett-Packard HP 48G calculator to record ground water readings every 12 hours (8 a.m. and 8 p.m.). Data was downloaded periodically and transferred into Microsoft Excel™. The mean ground water level and the percent time water was found within the root zone were calculated. The root zone is defined as the top 30 cm of the surface soil layer and is the primary zone for water and nutrient uptake by macrophytes (Mitsch and Gosselink 2000). Hydrographs were constructed and analyzed for each site.

Soil and Water Analysis

Five soil samples were collected at each wetland site using a small stainless steel shovel. Samples were taken to a depth of approximately 10 cm from the surface. The location of each sample depended on wetland morphology and size. Samples were taken in a Y-shaped pattern in order to obtain a representative sample of the wetlands soil characteristics (Figure 3). Soil samples were placed into clean plastic bags, packed in ice, and returned to the lab for analysis. Soil samples were oven dried at 100 °C. Bulk density measurements were taken by collecting soil cores using PVC pipe ($77 cm^3$). Two samples were collected at each site and the contents were dried and weighed to calculate soil bulk density.

Soil samples were sent to Midwest Laboratories, Inc., Omaha, Nebraska for chemical analysis including pH, percent organic matter (Walkly-Black), and exchangeable ions (calcium, magnesium, potassium, sodium), cation exchange capacity, and weak and strong Bray[4] extractable phosphorus using standard agronomic soil testing methods (NCR 1998). Soil subsamples were also sent to The Ohio State University for total organic carbon (TOC) and total nitrogen analysis on a CE Instruments CHN-Analyzer (Model nc-2100). The soil was first tested for inorganic carbon using 4M HCl (Nelson and Sommers 1982). If inorganic carbon was detected, the soil was treated with 5% H_2SO_4.

A Soil sample was also collected from each vegetation plot from the top 10 cm of soil using a 8.25x25cm stainless steel bucket auger (AMS Soil Recovery Sampler) and sent for analysis to the Ohio EPA laboratory. Samples were placed in the butyrate plastic liner that was inserted into the auger. Samples sent to the Ohio EPA laboratory were analyzed for pH, particle size, ammonia-N, total phosphorus, total organic carbon and metals (aluminum, barium, calcium, chromium, copper, iron, magnesium, manganese, lead, nickel, potassium, sodium, strontium, zinc) using standard agency methods.

Grab samples of surface water were collected and preserved in the field, and held at 4 °C for transport to the Ohio Environmental Protection Agency laboratory for analysis according to standard agency procedures for the following parameters: pH, ammonia-N, total Kjeldhal N, Nitrate-Nitrite-N, total phosphorus, total organic carbon, total suspended solids, total solids, chloride and metals (aluminum, barium, calcium, chromium, copper, iron, magnesium, manganese, lead, nickel, potassium, sodium, strontium, zinc).

Vegetation Survey

Vegetation surveys were conducted at each study site in July and August 2001. A 0.1 ha sample plot (20m x 50m), was established using the methods described in Mack (2004c). The vegetation sampling procedures were adapted from methods developed for the North Carolina Vegetation Survey as described in Peet et al. (1998). Ohio EPA has sampled over 250 plots between 1999-2004, including reference wetlands, mitigation banks, and individual mitigation wetlands using this method. The most typical application of the method employs a set of 10 modules in a 20m x 50m layout (Figure 4). At least four 10m x 10m modules are intensively sampled with a series of nested quadrats. Within these "intensive" modules, species cover class values are estimated for the 0.01ha ($100m^2$) area of the each intensive module. Species located outside of the intensive modules (the "residual" modules) are also recorded and percent cover is estimated over the residual area (typically 0.06ha or $600m^2$) of the non-intensive (residual) modules.

Standing Biomass

Standing biomass was sampled by harvesting vegetation to ground level using $0.1m^2$ clip plots. Clip plots were located in the corners of the intensive modules of the vegetation plot (Figure 4) for a total of eight clip plots per site (Mack 2004c). Harvested vegetation was placed in paper bags. Samples were oven dried at 105°C for 24 hours and weighed

Vegetation-Based Indicators

Vegetation community data were used to calculate various plant community attributes and indicators. The Vegetation Index of Biotic Integrity for Emergent wetlands (VIBI-E) was calculated (Mack et al. 2000; Mack 2001b, 2004b). The VIBI is a multimetric index (Table 2) used to describe the condition of the wetland based on plant community characteristics that respond predictably to human disturbance (Mack et al 2000; Mack 2001, 2004b). The VIBI is calculated by converting metric values to standard scores of 0, 3, 7, and 10 and then summing the metric scores to obtain the VIBI score which ranges from 0 to 100 (Mack 2004b).

The Floristic Quality Assessment Index score (FQAI) (Andreas et al. 2004), a metric incorporated into the VIBI, was also calculated.

[4] The standard Bray extraction (P1 or weak Bray) is with dilute acid; the strong Bray extraction (P2) has 4 times the acid concentration of the weak Bray. In agronomic situations, the difference between strong and weak Bray is often considered to be the active reserve of P which becomes available as soils warm up in the spring.

The FQAI is a variant of the weighted averaging ordination technique where species abundances or presence are multiplied by an ecological weighting factor (the coefficient of conservatism, Andreas et al. 2004). The FQAI has been shown to correlate with disturbance (Fennessy et al. 1998, Fennessy et al. 2002, Fennessy and Lopez 2002, Andreas et al. 2004).

A floristic quality index is developed by assigning a numeric score (the coefficient of conservatism or C of C) from 0 to 10 to each plant species growing in a region (Swink and Wilhelm 1979, Wilhelm and Ladd 1988, Andreas et al. 2004). The C of C is an ordinal weighting factor of the degree of conservatism (or fidelity) displayed by that species in relation to all other species of the region. Each C of C is an expression of the taxon's autecology as it relates to narrow or broad habitat requirements with respect to all other taxa in the flora (Andreas et al. 2004). The FQAI metric was calculated by using Equation 7 in Andreas et al. (2004):

$$I = 3\,(CC_i)/\%(N_{\text{all species}})$$

where I = the FQAI score, CC_i = the coefficient of conservatism of plant species *I*, and $N_{\text{all species}}$ = the total number of species both native and non-native.

Macroinvertebrate and Amphibian Sampling

Funnel traps were used to sample macroinvertebrate and amphibian communities at the study sites. Funnel traps were constructed of aluminum window screen cylinders with fiberglass window screen funnels at each end. The funnel traps were similar in shape to commercially available minnow traps but with a smaller mesh-size. The aluminum screen cylinders were 45.7 cm (18 in) long and 20.3 cm (8 in) diameter and held together with wire staples. The bases of the fiberglass screen funnels were 22.8 (9 in) diameter and attached with wire staples to both ends of the cylinder such that the funnels point inward. The funnels had a circular opening in the middle 4.5 cm (1.75 in) diameter.

Each wetland was sampled three times between mid-March and early July spaced approximately six weeks apart. Ten funnel traps were placed evenly around the perimeter of the wetland and the location was marked with flagging tape and numbered sequentially. Traps were set at the same location throughout the monitoring period. The late winter/early spring (mid-March to early April) sample allows monitoring of adult ambystomatid salamanders, early breeding frog species and macroinvertebrates such as fairy shrimp, caddis fly larvae, some microcrustaceans and other early season taxa which are often present for a limited time. A middle spring sample (late April-mid May) was conducted in order to collect some adult frog species entering the wetland to breed, to sample larvae of early-breeding amphibian species and to sample for macroinvertebrates. A late spring/early summer (early June-early July) sampling was performed to collect macroinvertebrates and relatively well developed amphibian larvae.

Activity traps were unbaited and left in the wetland for twenty-four hours in order to ensure unbiased sampling for species with diurnal and nocturnal activity patterns. Upon retrieval, the traps were emptied by everting one funnel and shaking the contents into a white collection and sorting pan. Organisms that could be readily identified in the field (especially adult amphibians and larger and easily identified fish) were counted and released. The remaining organisms were transferred to wide-mouth one liter plastic bottles and preserved with 95% ethanol in the field. Laboratory analysis of the funnel trap macro-invertebrate and fish samples followed standard Ohio EPA procedures (Ohio EPA 1989). Salamanders and their larvae were identified using keys in Pfingsten and Downs (1989) and Petranka (1998). Frogs, toads and tadpoles were identified using keys in Walker (1946).

Macroinvertebrate and Amphibian Indicators

Macroinvertebrate and amphibian community data were used to calculate various faunal community attributes and indicators. The Amphibian Index of Biotic Integrity (AmphIBI) and the Wetland Invertebrate Community Index (WICI) (Micacchion 2004, Knapp 2004) were calculated. Both are multimetric indices used to describe the condition of the wetland based on faunal community characteristics that respond predictably to human disturbance (Micacchion et al. 2000; Micacchion 2002; Micacchion 2004; Knapp 2004). They are calculated by converting metric values to standard scores of 0, 3, 7, and 10, and then summing the metric scores to obtain the index score which ranges from 0 to 50 for the AmphIBI and 0 to 60 for the WICI.

Decomposition (Litter Bag) Study

Decomposition rates were estimated using the litter bag technique. Litter bags were constructed of black mesh fiberglass window screen material with a mesh size of 5 mm (Cornelissen 1996). Each bag was made by folding a piece of screen into a 20cm x 20 cm bag stapled at 1.5 cm intervals on three sides (Deghi et al. 1980). Plant litter for the litter bags was collected in mid-June 2001. *Typha* spp. tissue was collected in approximately 60 cm sections from the top of the leaf in order to minimize variation in the structural and chemical content of the litter. *Juncus effusus* or *J. tenuis* leaves were cut approximately 5 cm from the ground.

Because variable N concentrations and lignin content in plant litter have been shown to influence decomposition rates, we measured decomposition using site-specific litter (on-site) and a litter collected from a neutral site not otherwise included in this study (control litter). This allowed us to evaluate the effects of site type (natural or mitigation) and nutrient availability on decomposition rates of standard materials. On-site litter consisted of plant material collected from a wetland study site and was then deployed in litter bags at the same site from which it was collected. Control litter was collected from a depressional marsh not otherwise included in this study. Mixed litter was used in both cases, consisting of tissue from *Typha latifolia* and *Juncus effusus.* At three sites (Calamus, JMB, Slate Run Bank 3) where no *J. effusus* was found, *J. tenuis* was substituted in the litter bags. Recent research has shown that decomposition rates in single-species litters experiments is typically not equivalent to rates observed in mixed litter, therefore mixed litter may more accurately reflect ecosystem level decomposition rates (Wardle et al. 1997, Gartner and Cardon 2004). The control litter bags containing *Typha latifolia* and *Juncus effusus*, were deployed at all sites in order to control for the effect of litter quality (C:N ratio, lignin content) on decomposition rates.

Each control litter bag contained 10 grams of *T. latifolia* and 2 grams of *J. effusus.* On-site litter bags contained 10 grams of *Typha* spp. and 2 grams of *Juncus* spp. Practical problems with the *Juncus* litter during and after deployment precluded its use in data analysis: 1) during deployment some *Juncus* litter was observed falling out of the litter bags through the mesh; and 2) because of the relatively long deployment of the litter bags (1 year), the *Juncus* litter did not maintain enough physical integrity to be separately removed, washed, and weighed. Therefore, only the results for *Typha* litter are presented. Logistical issues with deploying and collecting the litter bags limited sampling in 2001-2002 to 6 of the natural and 9 of the mitigation sites (Table 3). Control litter was subsequently deployed in 2002-2003 at the Lake Abrams, Eagle Creek Beaver, and Eagle Creek Marsh sites but data was not included in most analyses to control for possible inter-year differences (Table 3). Over the period May 2001 - July 2002, five stations of on-site litter (20 litter bags) and 3 stations of control litter (12 bags) were established except at the three sites noted above where only 3 stations of control litter were deployed (in 2002) for a total of 516 bags deployed. There was some loss of bags or stations due to high water events and beaver activity, so total analyzable bags per site varied. Sample stations were chosen to represent the typical conditions at the site. In order to minimize differences in incubation conditions and water depths, bags were deployed near the “vegetation line” where vegetated areas and open water areas met in each wetland. At each station, four bags were tied with a nylon rope to a wood stake to prevent movement of the bags.

At each collection period, replicate bags were retrieved from each site (for a total of 5 on-site bags and 3 control bags) and stored on ice for transport. At the lab, litter remaining in each bag was removed, briefly washed to remove dirt and debris, separated by species (*Typha* and *Juncus*), and oven dried at 90°C to a constant weight. The plant litter used in the litter bags was analyzed for nitrogen, phosphorus, calcium, magnesium, and potassium content prior to deployment. These are referred to as initial litter nutrient concentrations. Nutrient concentrations were also determined for a total of four on-site plant litter samples (2 *Typha*, 2 *Juncus*), and three control plant samples (2 *Typha*, 1 *Juncus*) from each site. Each time litter bags were collected from the field, three samples of on-site litter and 2 of control litter from each site were selected at random for analysis of the same parameters.

Litter samples were analyzed using standard methods (AOAC 1990). Following microwave nitric acid digestion, elemental analysis (except for %N) was done using Inductively Coupled Plasma Spectroscopy (Method 985.01, AOAC 1990). Nitrogen content (%) was determined using the Dumas Method using a LECO FP-428 Nitrogen Analyzer (Method 968.06 AOAC 1990). Litter sample analysis was done at Midwest Laboratories, Inc., Omaha, Nebraska.

After each collection period, the percent mass lost was calculated in order to determine how much litter was lost during each period. These data were also plotted in terms of percent mass remaining in order to track decomposition over time. Decomposition rates were calculated by determining k, a standard measure of decomposition (Molles 1999). The k-value was determined as follows:

$$M_t = M_o e^{(-kt)}$$

where M_t = mass of litter present at time t, M_o = initial of litter, t = time in days, and k = daily rate of mass loss (Molles 1999). The duration (days) of each incubation period is shown in Table 3. Incubation times vary slightly because field logistics precluded us from deploying or collecting litter bags from all sites on the same day.

By quantifying the nutrient concentration in the decaying litter following each collection period, we were able to determine if differences existed in the nutrient dynamics of natural and mitigation wetlands. Nitrogen immobilization was calculated based on changes in litter N concentrations between each collection date (Windham and Ehrenfeld 2003).

Data Analysis

Minitab statistical software v. 12.0 and StatView v. 5.0 were used for all analyses except macroinvertebrate data analysis where Systat v. 9.0 was used and for multivariate analyses (Detrended Correspondence Analysis (DCA), Principal Components Analysis (PCA), Cluster Analysis) where PC-ORD was used (McCune and Mefford 1999). Descriptive statistics, box and whisker plots, regression analysis, analysis of variance, multiple comparison tests, and t tests were used. Detrended Correspondence Analysis (Hill and Gauch 1980; Gouch 1982) and Cluster Analysis (Sneath and Sokol 1973) were used to evaluate species presence and relative abundance data. Principal Components Analysis was used to evaluate IBI metric performance. For the DCA, Euclidean distance was calculated and rare species were down weighted. For Cluster Analysis, Sorensen similarity and Ward's linkage method were used.

RESULTS

Hydrology

Several hydrological parameters (Table 4) were calculated using data collected from May 1 to September 30, 2001 and from April 1 to September 30, 2002 (records were more complete in the 2002 growing season): including the percentage of time that water remained in the root zone (defined as the top 30 cm of soil); mean and median ground water levels (shown in centimeters below the ground surface); and, the maximum and minimum water levels recorded at each site. Positive values indicate that ground water levels were above the ground surface. A "flashiness" index was developed by averaging the absolute value of the differences between each ground water measurement from the measurement just preceding it (Table 5).

Natural wetlands had water in the root zone ranging from 100 to 23 percent of the time while mitigation wetlands ranged from 96 to 0 percent of the time. On average, water remained in the root zone of natural wetlands 50.7% longer than in mitigation wetlands, although this difference was not significant (p = 0.21). Mean ground water levels ranged from -58.2 to 4.2 cm at the natural sites and -12.2 to -0.7 cm at the mitigation wetlands. The mitigation wetlands were generally "dryer" than the natural sites based on measures of ground water. Maximum ground water depths recorded by the wells (note this is often the lowest reading the well can record, not necessarily the lowest water level actually occurring) was -88.3 cm for the natural wetlands (Lodi) and -106.4 cm for the mitigation sites (Trotwood). Minimum depths recorded were 33.1 cm for the natural sites (Baker Swamp) and 11.7 cm for the mitigation sites (Medallion No. 20).

Box and whisker plots were constructed to compare mean hydrological parameters for the natural and mitigation wetlands in both years, and unpaired t-tests were used to test for differences between means (Figures 5a and 5b). Water was present in the root zone for nearly twice as long in the natural sites during the 2001 growing season, and this difference was significant (31.9 $\pm$ 11.3 percent for mitigation versus 63.9 $\pm$ 9.4 percent for natural; p = 0.04). Similar data were collected in 2002 when water was present in the root zone for 37.2 $\pm$12.1 and 66.0 $\pm$ 7.0 percent of the time for mitigation and natural sites, respectively (p = 0.059). Mean depth to ground water reflects this, averaging -53.8 $\pm$ 11.1 cm in the mitigation sites and -25.0 $\pm$ 6.1 cm in the natural sites in 2001 (p = 0.04), and -44.5 $\pm$ 9.1 and -25.4 $\pm$ 4.9 in 2002 (p = 0.09).

A comparison of mean surface water levels, mean ground water levels, and the percentage of time that ground water was in the root zone (Figure 6), shows that natural and mitigation wetlands have significant hydrological differences. Mitigation wetlands had both deeper surface water and greater mean depth to ground water, leaving a substantial unsaturated zone in the upper soil for most of the growing season. This indicates a 'disconnect' between surface and ground waters at the mitigation wetlands. The heavy clay soils that characterize many of the mitigation sites appear to limit the vertical movement of water through the root zone, making the mitigation sites less hydrologically dynamic (see Table 9) for data on bulk density and percent solids). The combination of deeper surface water and drier soils (lower ground water) has implications for plant growth (water available for root uptake) and biogeochemical processes such as denitrification because the relative lack of water flux also limits the movement of compounds such as nitrate and dissolved organic carbon needed by microbial communities. The functional consequences of this are not known, but appear to create substantial differences in the biogeochemistry of the two types of wetlands.

Hydrologic "flashiness" ranged from 1.0 to 4.6; maximum single day change in water levels ranged from 16.0 cm to 79.2 cm (Table 5). Eagle Creek Beaver had the lowest score due to the moderating influence of ground water on its daily water levels; Lake Abrams, Lodi North, and Trotwood had the highest scores due to high stormwater inputs (Table 5). Sites with very strong depressional hydrology (vertical hydrologic pathway driven by precipitation and evapotranspiration had flashiness scores of 1.0 to ~2.0. Index scores of between 2 and 3 occurred at sites with some riverine association or small to moderate stormwater inputs. Scores greater than 3 were indicative of high stormwater inputs disrupting the natural hydroperiod.

Hydrographs for all wetlands included in this study were constructed (Figures 7 to 12). In general, ground water levels in the mitigation sites declined earlier in the growing season than in the natural sites (for those sites that did dry down), and had less daily variation in water levels.

Several hydrologic signatures can be recognized in each group. In the natural sites there are two basic patterns evident, one in which ground water is a significant influence that maintains relatively constant water levels throughout the growing season (i.e. permanently inundated/saturated sites including Baker Swamp, Ballfield, Eagle Creek Beaver), and one in which there is a dry down through the early summer to some low level later in the growing season. Seasonally flooded wetlands include Calamus, Dever, Eagle Creek Marsh, Lake Abrams, Lodi, and Rickenbacker. Seasonally flooded wetlands are common in Ohio with standing water in the spring and early summer and dry soils in the late summer and early autumn, however in nearly all cases, ground water levels are within the upper portion of the soil that is measurable by the well (greater than ~ 80cm). Only Rickenbacker and Eagle Creek Marsh show long periods (> 1 week) where water levels fell below the bottom of the well.

Several sites (Eagle Creek Marsh, Rickenbacker) show a rewetting during July, 2001 in response to rainfall events. Both sites began to dry down again almost immediately, with Rickenbacker only taking a few days for water levels to drop again below the level of the well. For all sites, water levels at the well locations (near the edge of the wetlands) were very nearly at or above the ground surface early in the growing season.

There are three basic hydrologic signatures observed for the mitigation wetlands (Table 5, Figures 7 to 12): permanently flooded, seasonally flooded, and "dry," where ground water levels are very low (defined here as permanently below the root zone at 30cm) and remain so throughout the growing season. The majority of the mitigation sites show a seasonally flooded hydrologic signature (Big Island Area D, JMB, New Albany HS, Prairie Lane, Slate Run Bank SE). However, unlike the natural wetlands, these still underwent dry down to the extent that ground water levels drop below the level of the well (noted by a flat line where levels are lowest). The dry down curves are generally steeper for the mitigation sites, resulting in water levels that bottom out by June. In some cases, mid-summer precipitation caused ground water levels to rise; in the case of New Albany water levels remain high for the remainder of the growing season, for the others (e.g., Prairie Lane) water levels rise and then fall again below the bottom of the well. By contrast, Medallion and Pizzutti have a permanently flooded ground water signature where ground water levels remain high throughout the growing season. These sites had surface water present throughout the growing season as well.

Bluebird and Trotwood have what can be considered a "dry" ground water signature, one where ground water levels are low throughout the growing season. At Trotwood this "dry" ground water signature occurred even though the site is permanently inundated year round with surface water. Trotwood was also the flashiest of the mitigation sites (Table 5) due to massive stormwater inputs from surrounding shopping centers and suburban development. The surface and ground water at this site appear to be completely disconnected. The hydrograph for Bluebird shows more fluctuation over time, but the soils remain unsaturated above 50 cm for the duration of the growing season.

Water Chemistry

Water chemistry parameters for the natural and mitigation wetlands revealed no significant differences in the availability of nutrients (e.g. ammonia, total P, TKN), cations (Ca^{2+}, Mg^{2+}), or physical measurements (pH, TSS) (Table 6). Mean concentrations of these parameters are typical of those found in freshwater marshes (Mitsch and Gosselink 2000), although for several parameters, average concentrations were higher in the natural sites including Total Organic Carbon (TOC).

Water chemistry of a larger reference wetland data set was also examined in order to place the sites included in this study in the context of wetland types across Ohio. Water chemistry parameters are summarized in Table 7. Mitigation wetlands have median values in the range of concentrations typical of natural depressional and riverine marshes for TOC, Ca, Fe, Mg, Chloride, Ammonia, and total P (Table 7); however, total suspended solids (TSS) at mitigation wetlands was high (33 mg l^{-1}) and similar to values found in riverine mainstem marshes (47 mg l^{-1}), forests (25 mg l^{-1}), and shrub swamps (38 mg l^{-1}), riverine headwater marshes (27 mg l^{-1}), and coastal marshes (77 mg l^{-1}) (Table 7).

Comparing water chemistry of mitigation sites to other wetlands types reveals differences for other parameters. As expected bogs have higher median values for TOC (45-59% higher), as did Lake Plains sand prairies (35%), vernal pools (17-29%), wet woods (39%), and mainstem forests and shrub swamps (25% and 38%) (Table 7). Median Ca concentrations are lower at bogs (3-16 mg l^{-1}) and much higher at ground water driven systems like fens (up to 59 mg l^{-1}) and coastal marshes (58 mg l^{-1}) (Table 7). Chloride, Ammonia, and P concentrations can increase substantially at natural wetlands receiving storm water inputs. The 75th percentile for these parameters at natural marshes can be as a high as 176 mg l^{-1}, 0.24 mg l^{-1} and 0.51 mg l^{-1}, respectively (Table 7).

Soils

Of the 21 common soil parameters shown in Table 8, 19 parameters were significantly different ($p < 0.10$) when average values of natural and mitigation wetlands were compared. Average concentrations of organic carbon (%OC), %N, and plant available P (µg P g^{-1} soil) were 6.2 times, 4.6 times and 1.6 times higher, respectively, in the natural sites compared to the mitigation ones. The average nitrogen content (as %N) in natural wetlands was 1.12% compared to a very low 0.24% in the mitigation systems (Table 8; Figure 13). Organic carbon averaged 15.1% in natural wetlands compared to 2.45% in mitigation systems, and the average concentration of plant available P in the natural wetlands was 11.96 µg g^{-1} and 43.4 µg g^{-1} com-pared to 7.38 µg g^{-1} and 30.0 µg g^{-1} in the mitigation systems for weak and strong Bray analyses, respectively. Total P, which is a measure of all P held in the soil (including that which is not readily available), showed a similar pattern with levels of 1156 µg g^{-1} in the natural soils, nearly twice as high as in mitigation soils, which averaged 669 µg g^{-1}. Soil ammonia was more than three times higher in natural soils, averaging 62.4 µg g^{-1} and 20.5 µg g^{-1} in the natural and mitigation sites.

Exchangeable cations provide an important index of fertility and plant growth. Total and exchangeable K and Mg were higher in mitigation wetlands; total Ca was higher in mitigation wetlands but the proportion of exchangeable Ca was lower. The ratio of Ca^{2+} to Mg^{2+} differed in the two groups, ranging from 6.7 in the natural to 4.4 in the mitigation sites, indicating a relative lack of available Ca^{2+}. Overall, cation exchange capacity was nearly 1.5 times higher at the natural wetlands.

Mean values for soil bulk density, %solids, particle size of midrange (2 - 50 µm) and large (>50 µm) particles, and pH between natural and mitigation wetlands were also significantly different (Figure 14). The pH of natural sites was significantly lower than the mitigation sites, averaging 5.58 and 6.19, respectively. The average bulk density of natural wetland soil was 0.62 g cm^{-3} compared to 1.75 g cm^{-3} in the mitigation wetlands. Values for percent solids also differ accordingly, averaging 42.5% in natural and 73.5% in mitigation sites. Average size of soil particles was lower (31.5%) for midrange particles and higher for larger particles (53.8%) for natural wetlands than for mitigation wetlands (41.9% midrange, 39.7% large). These measures quantify the extremely heavy soils found in the mitigation sites that may lead to reduced root growth and limit carbon accumulation (Tables 8 and 9). In a cluster analysis based on the average nitrogen and carbon content of both natural and mitigation wetlands 9 of the 10 mitigation wetlands grouped together, i.e. mitigation wetlands had soil characteristics unique from the natural sites (Figure 15). The distinct separation of the two groups indicates that there are two ecologically distinct wetland populations based on soils.

Soil chemistry of a larger reference wetland data set was also examined in order to place the sites included in this study in the context of wetland types across Ohio. These values are summarized in Table 10. Median values for Al, Ca, Fe, Mg, K, and P total were similar to median concentrations and ranges found in other natural marshes. Median ammonia concentrations at mitigation wetlands was at the lower end of natural wetland values (24 mg kg^{-1}) and was much lower than depressional marshes (39 mg kg^{-1}), mainstem marshes (54 mg kg^{-1}), or headwater marshes (77 mg kg^{-1}), and was more similar to nutrient poor bogs and fens (Table 10). Striking differences were observed when comparing %solids and TOC of soils in mitigation wetlands to all natural wetlands: mitigation wetlands had the highest median %solids (73.3%) and lowest median TOC (2.0%) of all soil samples analyzed (Table 10).

Vegetation

Because individual plant species are differentially sensitive to environmental stressors, and vegetation is always present in wetlands, it is the most common assemblage used to assess the condition and development of wetland ecosystems (Fennessy et al. 2002; Cronk and Fennessy 2002). In order to determine if differences between the vegetation communities of natural and mitigation wetlands existed, plant-based attributes including the Vegetation IBI and its component metrics (Mack 2004b), plant species richness, the FQAI, and the level of aboveground biomass production were calculated for each site (Tables 11 and 12).

Scores for the VIBI–Emergent (Mack 2004b) were calculated for all sites. Scores for natural sites ranged from 9 to 82, reflecting the fact that the natural wetlands were chosen along a gradient of human disturbance (Table 11). This range encompasses nearly the entire range of scores for emergent wetlands in Ohio. Scores for mitigation wetlands were much more consistent, ranging from 16 to 50. This compression of scores is due in part to the fact that the community composition of the mitigation sites is similar, with a dominance of ubiquitous, tolerant species. Mean VIBI scores were more than twice as high at natural as for mitigation sites, and this difference is highly significant ($p = 0.005$) (Table 12). There was no significant difference between the species richness of natural and mitigation sites, an average of 31.0 species were recorded in the plots at natural sites while an average of 25.6 were found in the mitigation sites ($p = 0.22$) (Table 12).

In contrast FQAI scores were significantly different in the two wetland populations, with an average score of 21.6 for the natural sites and 14.2 for the mitigation sites ($p = 0.004$) (Table 12). The range of FQAI scores in the natural wetlands was 15.8 to 31.0 (reflecting the human disturbance gradient), whereas scores for the mitigation wetlands ranged from 8.8 to 18.1. Despite the fact that the natural sites span a gradient of disturbance, the range of scores for the two populations overlaps only marginally (between 15.8 and 18.1) (Table 11).

Aboveground biomass, a measure of wetland primary productivity, was significantly higher in natural sites which produced an average 34.7 g 0.1 m^{-2} (equivalent to 347 g m^{-2}) while mitigation sites produced an average of 20.9 g 0.1 m^{-2} (209 g m^{-2}) (Table 12). Thus production rates were an average of 1.7 times higher in natural wetlands (unpaired t-test, $p = 0.04$). However, when average standing biomass of mitigation wetlands was compared to natural wetlands in three disturbance categories, mitigation wetlands had standing biomass similar to least disturbed natural sites (Figure 22b). This was due to large areas of unvegetated water or bare ground or soil nutrient limitations at many of the mitigation wetlands.

Finally, all vascular plants recorded in each wetland were categorized according to their wetland indicator status, a measure of their fidelity to wetland habitats (Reed 1988). Natural wetlands contained significantly greater numbers of obligate species (those found in wetlands 99% of the time), and had higher means of FACW and FAC species. This may in part be due to the relatively dry soils found at the mitigation sites (Figure 16).

The VIBI has been shown to respond predictably to human disturbance and has undergone two major evaluations with independent data sets to validate its usefulness as a tool to assess wetland condition (Mack et al. 2000, Mack 2001, Mack 2004b). Several measures of human disturbance have been employed to test the VIBI including the Ohio Rapid Assessment Method (ORAM v. 5.0) (Mack 2001) and the Landscape Disturbance Intensity Index (LDI) (Brown and Vivas 2005). Both of these measures show strong correlations with wetland VIBI scores. We used the larger reference wetland data set to test the relationship between the VIBI and the LDI for natural wetlands. A regression analysis shows that the LDI is highly correlated with VIBI scores across all wetland classes ($R^2 = 43.6\%$, $p < 0.001$) (Figure 17); as predicted, the land use surrounding a site has a profound influence on its ecological condition.

We then compared the LDI of the mitigation wetlands with the LDI scores for each of three regulatory categories that have been established based on ORAM scores by the Ohio EPA. Figure 18 shows that the LDI is highly predictive of wetland category with wetlands located in highly developed landscapes tending to be more degraded. Figure 18 indicates that mitigation wetlands have been placed in relatively developed landscape settings. The LDI scores of mitigation wetlands and Category 1 wetlands are similar (mean LDI score of 5.2 for mitigation and 5.4 for Category 1 sites), suggesting that wetlands located in highly developed areas, be they natural or mitigation, are more likely to be performing at lower levels..

Principal components analysis (PCA) was used to evaluate performance of all VIBI metrics simultaneously. Mitigation wetlands in this study separated clearly from good to high quality natural wetlands and also from disturbed natural wetlands (Figure 19). A similar pattern was observed when DCA was used to evaluate species presence and abundance, with mitigation wetlands and disturbed natural wetlands ordinating together (Figure 20). Analysis of metric values in the VIBI-E showed mitigation wetlands differing significantly from natural wetlands in almost every instance (Figures 21 to 24). Vegetation IBI scores were evaluated by disturbance categories. Mitigation wetlands had significantly lower VIBI scores than medium (2nd ORAM tertile) and low (3rd ORAM tertile) disturbance categories; VIBI scores of mitigation wetlands and highly disturbed (1st tertile) natural wetlands were not significantly different (Figure 25).

Box plots were constructed for VIBI scores by grouping the sites that are considered reference wetlands, nonreference wetlands, and mitigation wetlands (Figure 26). Mean mitigation wetland scores are similar to the nonreference sites, but the range of scores is much smaller for the mitigation wetlands (i.e., they score consistently lower). Mean natural reference wetland scores were significantly higher than either the nonreference sites or the mitigation (mitigation) sites (Figure 26). A similar pattern was observed when LDI scores were compared with mitigation wetlands constructed in predominately intensively developed landscapes and many nonreference sites, and most reference wetland sites located in more natural landscapes (Figure 26).

Macroinvertebrates

Major differences in taxa richness and relative abundance of several invertebrate groups were observed when the natural and mitigation wetlands were compared. Numbers of dytiscid beetle, chironomid, dipteran, and total taxa richness were higher at the natural sites (Figure 27). By contrast, numbers of mayfly and caddisfly taxa were significantly higher at the mitigation sites due mainly to the dominance of two mayfly genera *Caenis* and *Callibaetis,* which were present at most of the mitigation sites, but occurred at less than half of the natural sites. These two taxa are considered facultative to pollution tolerant (Knapp 2004).

The relative abundance of oligochaetes, ostracods, and chironomids/dipterans was higher at natural reference wetlands (Figure 28). The oligochaetes identified belong to the family Naididae which appears to be a relatively sensitive taxa in wetlands (Knapp 2004). Most stream ecologists are familiar with a more tolerant oligochaete, *Tubifex tubifex* (Family Tubificidae) collected in high percentages from polluted rivers and streams. Ostracods also appear to be relatively sensitive taxa in wetlands. Some varieties of ostracods are sensitive to herbicides and pesticides (Thorp and Covich 2001).

Relative abundance of tolerant beetles, corixids, and tolerant snails were higher in the mitigation sites. The adult beetle genera, *Haliplus*, *Peltodytes*, and *Tropisternus* are herbivores. They are commonly found in dense mats of aquatic vegetation or algae mats. At the natural sites, there were higher numbers of dytiscid beetle taxa. The adult dytiscid beetles are predacious. Corixidae abundance was higher at the mitigation sites, especially the genera *Ramphocorixa*, *Sigrara*, and *Trichocorixa*. In the reference sites only the corixid genus *Hesperocorixa* was collected in moderate numbers. The tolerant snail genera *Physella* and *Gyraulus* were more abundant at the mitigation sites.

Box plots of a wetland invertebrate community index (WICI) scores for mitigation, nonreference, and reference sites are shown in Figure 29. Mitigation sites had significantly lower WICI scores (mean of 13) due to high relative abundance of tolerant taxa (with fewer sensitive taxa) than nonreference (mean of 27) and reference (43) wetlands. The observed macroinvertebrate trophic structure differed substantially in the two types of wetlands (Figure 30).

Amphibians

Amphibian community data were used to calculate various faunal community attributes and indicators including the Amphibian Index of Biotic Integrity (AmphIBI) score (Micacchion 2004). Ordinations of the species composition of natural and constructed wetlands were examined. For the amphibian analysis information from a larger natural reference wetland data set was also included.

Nine of the 10 mitigation wetlands had AmphIBI scores of 0 (mean score = 0.3). Slate Run Bank SE was the only mitigation site with a score greater than 0 (AmphIBI score = 3) due to the presence of tiger salamanders (*Ambystoma tigrinum*). This site was built at the edge of an existing forested area that had a breeding population of tiger salamanders. The 9 natural emergent wetlands had AmphIBI scores that ranged from 0 to 17 (mean score = 6.55). The differences in mean AmphIBI scores between natural and mitigation wetlands were significant ($p < 0.05$).

Data collected between 1996-2002 in a larger reference wetland data set (n = 101) was included in the ordination with the 10 mitigation wetlands sampled for this study. Mean AmphIBI scores between reference (least-impacted), nonreference (moderate to severe disturbance) natural wetlands and the mitigation wetlands were significantly different (Figure 31). A PCA of individual AmphIBI metrics showed mitigation sites in a very tight cluster surrounded by other natural emergent wetlands (Figure 32). Both mitigation wetlands and natural emergent wetlands were separated from wetland forests and shrub swamps (Figure 32).

Ordination of species presence and relative abundance using DCA revealed clear differences between natural emergent, forest, and shrub wetlands and the mitigation wetlands studied here (Figure 33). Most good to high quality shrub and forest wetlands grouped together on the far right side of the graph. The mitigation sites formed a group with some natural emergent wetlands and a few of the lower quality shrub and forested wetlands (Figure 33). These groupings were due to the presence and abundance of sensitive, forest dependent amphibian species like wood frog (*Rana sylvatica*) and spotted salamander (*Ambystoma maculatum*) at the good to high quality shrub and forest sites (Figure 33).

When only the mitigation and natural emergent sites included in this study were ordinated, there was no strong separation of natural and mitigation sites. Rather, the sites ordinate more on the basis of the permanency of hydrology and the corresponding presence of predatory fish (data not shown). Those wetlands with the more permanent surface water and presence of predatory fish cluster in the upper right portion of the graph and those with seasonal hydrology and no predatory fish cluster in lower left portion of the graph. Two sites with high toad (*Bufo* spp.)[5] dominance (Prairie Lane, Big Island Area D) are far removed from the other sites in the upper left corner of the graph (Figure 34).

Decomposition

After the first incubation period (~37 days), a significant difference was found in *Typha* decomposition rates in which natural sites lost an average of 4.53 g (or 45.3%) of on-site litter from each bag and mitigation wetlands sites lost an average of 3.92 g (39.2%) ($p = 0.067$) (Tables 13 and 14). The *k*-values calculated for the natural and mitigation sites did not differ significantly, but natural wetlands had a faster average rate of decay as compared to mitigation wetlands (0.0167 and 0.0139) (Tables 13 and 14).

Initial difference in the decomposition rates persisted throughout the study. Mass lost was higher at natural than mitigation sites at the second pick up (~87 days), with a mean loss of 5.86 g and 4.49 g, respectively ($p = 0.05$). At the third pick up, mean differences were 6.26 g compared to 5.11 g lost ($p = 0.05$). Figure 35 shows this data graphically as the percent mass remaining at each collection period. The data indicate that there are distinct differences in both the short-term (approximately 1 month) and longer term (approximately 1 year) decay processes of the on-site litter between natural and mitigation wetlands.

We used control litter of uniform C:N ratio to help isolate differences in decomposition rates and short term nutrient flux in natural and mitigation wetlands other than differences resulting from the chemical composition of the plant material itself. As with on-site litter, mean decomposition rates for control litter were faster in natural wetlands. Differences were significant at the second and third pick-up (Tables 15 and 16). At the third pick-up an average of 7.14 g (or 71.4%) had been lost from natural sites, while only 5.34 g (53.4%) had been lost at the mitigation sites. Thus the natural sites lost nearly 1.3 times more litter than the mitigation sites indicating significantly higher levels of microbial activity and higher rates of nutrient flux (see next section). K values show a similar pattern with highly significant differences at the third-pick up. At the end of nearly one year, natural sites had 29% of the original litter remaining while mitigation sites had nearly half (47%) of the previous year's litter left in the system. Bio-geochemical transformations appear to be happening much more slowly in the mitigation wetlands (Figure 36).

Plant Litter Nutrient Analysis

Initial concentrations of both nitrogen and phosphorus in on-site plant litter was significantly higher in natural wetlands. Litter nitrogen levels averaged 0.4 percent higher at natural sites ($p = 0.09$) (Figure 37). The initial phosphorus content at natural sites averaged 0.29 $\mu g\ P\ g^{-1}$ litter while mitigation sites had a mean concentration of 0.25 $\mu g\ P\ g^{-1}$ litter ($p = 0.01$) (Figure 38). Plant nutrient concentrations show a general pattern of decline due to initial leaching, followed by an increase at the second and third measurements as microbial colonization of the litter progresses (Figures 37 and 38). Despite differences in initial concentrations, N concentrations at the time of first pick-up (~37 d) were essentially equal (2.7 % N and 2.6 % N by weight, respectively (Figure 37), indicating that N was leached more rapidly at the natural sites. Leaching is common in the early stages of decomposition and represents a flush of nutrients that is then available for microbial growth and plant uptake. The amount of N leached at the mitigation sites amounted to less than half that lost at the natural wetlands. At the second pickup (87 d), litter N concentrations had increased rapidly in the natural wetlands (increasing by nearly 20%) while N levels stayed constant in the mitigation sites. This resulted in significantly higher mean concentrations of 3.18 %, compared to 2.66 % in the mitigation sites ($p = 0.07$). N concentrations converged slightly as decomposition progressed as levels increased

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5 American toad (*Bufo americanus*) and Fowlers toad (*Bufo fowleri*) tadpoles cannot be differentiated and results are aggregated.

slightly at mitigation sites while staying relatively constant at natural sites (Figure 37).

Phosphorus concentrations in decomposing on-site litter followed a similar pattern: initial concentrations were significantly greater in the litter of natural wetlands and, after the first incubation period, significantly more P had been leached to the surrounding environment ($p = 0.07$) (Figure 38). In the natural wetlands P losses amounted to 0.18 μg P g^{-1} litter compared to 0.01 μg P g^{-1} litter in the mitigation sites. P then increased between the first and second pick up. No significant differences were found in litter P concentrations after this point (Figure 38).

The initial amount of nitrogen in the control *Typha* was 2.39 % N (Figure 39). This value was found within the range of initial N concentrations for the on-site litter of both natural and mitigation wetlands (1.66 to 3.07 % N by weight). The initial amount of phosphorus in the control litter was 0.195 μg P g^{-1} litter, which was slightly lower than the range of P concentrations found in the on-site litter plant litter of natural and mitigation wetlands (0.20-0.29 ug P g^{-1} litter) (Figure 40).

At the time of the first pick-up (~ 45 days for control litter), no significant differences were found in the N or P concentrations in the natural and mitigation wetlands (Figures 39 and 40). Nitrogen concentrations increased in both wetland types to an average of 2.99 % N in natural and 2.78 % N in mitigation wetlands ($p = 0.26$) (Figure 39). Phosphorus concentrations declined in both wetland types to an average of 0.134 μg P g^{-1} in litter from the natural wetlands and 0.128 μg P g^{-1} of litter from the mitigation sites ($p = 0.62$) (Figure 40). After approximately 110 days in the field, significant differences were observed in N concentrations, with mean N concentrations of 3.22 % N and 2.71 % N in natural versus mitigation sites respectively ($p = 0.059$ (Figure 39). This difference persisted through the end of the study. P concentrations also differed at the second and third collection periods. Mean concentrations were 0.16 μg P g^{-1} litter (natural) and 0.12 μg P g^{-1} litter (mitigation), respectively after an average of 110 days, and 0.17 and 0.14 μg P g^{-1} litter after 325 days ($p < 0.001$) (Figure 40).

Overall, in terms of both percent mass lost and nutrient accumulation, decomposition patterns for the control litter were similar to the on-site litter; the natural sites lost more litter mass and accumulated higher levels of N and P than did the mitigation sites. Both bacteria and fungi are responsible for decomposition and changes in nutrient content in litter, and both the structure and activity of the microbial community has a major influence on litter nutrient content and decomposition rates. There have been few studies to investigate microbial community structure in wetlands, or to identify specific microbes as indicators of wetland condition. Our data indicate the possibility of substantial differences in the microbial communities in mitigation sites, with implications for overall ecosystem function and condition of these replacement wetlands.

Standing Stocks of Nitrogen and Phosphorus

In addition to differences in litter N and P concentrations, the estimated standing stocks of these nutrients (in units of g per m^{-2}) were calculated as the product of their concentrations and biomass at the time of harvest. Standing stocks were significantly higher in natural sites due both to higher concentrations and the influence of biomass accumulation through the growing season. Mean biomass production was 1.7 times higher (Table 12) in natural wetlands, leading to aboveground stocks of N and P that were twice as high in natural as mitigation sites (Figure 41). N stocks were 1.4 g N m^{-2} in the natural sites compared to 0.68 g N m^{-2} in the mitigation sites, while P stocks amounted to 0.12 and 0.05 g P m^{-2} respectively (Figure 41). Since shoot growth is new in each growing season, this accumulation can be interpreted to represent both translocation from belowground tissues and uptake from the soil. These data indicate the greater volume of nutrients cycling through the natural wetlands.

Links between Plant Community Structure and Ecosystem Processes

The relationship between the wetland disturbance categories and mean biomass production (g 0.1 m^{-2}) was investigated using the larger data set available for emergent marshes in Ohio (Figure 42). Biomass production is negatively correlated with VIBI scores (by disturbance category), supporting our hypothesis that productivity will increase as VIBI scores decline. This is an ecologically predicable relationship for wetlands since as a site becomes disturbed by human activities, invasive, highly productive species tend to become dominant. Because the VIBI integrates information about the relative tolerance levels, species composition and, as one of its metrics, biomass production, these two measures are not completely independent. In order to further explore this relationship, a regression of biomass production versus FQAI scores was plotted for the sites included in this study (Figure 43). The slopes of the regression lines for the two wetland populations were very different so each is plotted individually. Both show the same trend: biomass production declines as FQAI scores increase (again an ecologically predictable relationship (Keddy 1993), however the slope of the line for the mitigation sites is much steeper than for the natural wetlands, and intercepts the x-axis at a lower value. At nearly all points along the line, for sites with the same FQAI score, the models predict that the natural wetlands will have greater biomass.

A box plot showing the mean decomposition rates for wetlands by disturbance category shows a trend (insufficient data for statistical testing by category) of increasing rates as a function of ecological condition (Figure 44). In order to test this relationship more fully, and to test decomposition rates as a potential indicator of condition, a more extensive data set will be necessary. As described above, mean rates were lowest overall for the mitigation sites and disturbed natural wetlands. This lends support to the hypothesis that impaired wetlands have slower decomposition rates.

The soil and water data presented earlier in this report demonstrate the low levels of organic carbon contained in the mitigation wetlands. Wetland food webs tend to be dominated by heterotrophs, making the availability of organic carbon an important driver of ecosystem processes. Low organic carbon levels can limit the activity of decomposers (heterotrophic microbes and invertebrates) and lead to lower rates of decomposition in the system. In order to more explore this link, we performed regression analyses of decomposition rates across the range of organic matter available in the natural wetlands (since carbon levels were consistently low in the mitigation sites). Both water and soil organic carbon are highly correlated with decomposition rates (Figures 45 and 46).

DISCUSSION

The biological and biogeochemical characteristics of the natural and mitigation wetlands in this study were substantially different. The natural wetlands had faster rates of decomposition, higher IBI scores, biomass production, soil nutrient concentrations, and plant litter nutrient concentrations. Significant differences in hydrological patterns were also observed.

Hydrology

The hydrodynamics of the two groups of wetlands were significantly different. While ground water levels were significantly lower in the mitigation sites, mean surface water levels were higher. We predicted that mitigation wetlands would have higher ground water levels due to the presence of high, often permanent, surface water levels. Our data indicate a hydrological "disconnect" between the surface and ground water in the mitigation wetlands where soils in the root zone are “dry” even though the site is inundated with large areas of open water. For example, on June 6, 2001, the surface water depth was 90 cm at the edge of the pool at the Bluebird mitigation wetland while the ground water reading at this location was –22.4 cm. The high bulk density of the soils is one probable cause of this hydrological disconnection. Heavy, non-porous soils prevent water movement through the soil. Soils in the mitigation wetlands appear to be acting as a clay pan, keeping surface waters perched. This contributed to other hydrological differences such as the significantly shorter duration of water in the root zone at mitigation sites. This may present limitations to macrophyte growth and reproduction since the majority of vascular wetland plants obtain water and nutrients through their roots.

Cole and Brooks (2000) studied two mitigation wetlands and found that surface water was present for longer periods than in two natural sites; but they also report water in the root zone for longer periods. They suggest that hydrological differences are the result of permit requirements. Mitigation wetlands usually have a hydrological standard that must me met. In order to ensure mitigation wetlands achieve "wetland hydrology" they are often deliberately made overly deep as a type of insurance that water will be present (Cole and Brooks 2000). This results in "wetlands" that are basically shallow ponds with distinct hydrological differences from typical natural wetlands.

Contrary to Cole and Brooks (2000), we found that the depth to ground water was greater and the duration of water in the root zone was significantly shorter in the mitigation sites. Improper hydrology will limit the successive development towards a more natural ecosystem (in terms of both vegetation and animal communities). Natural wetlands had higher average ground water levels during the growing season and many of them experienced a dry down period in which very little or no surface water was present. Dry down periods are important for the rejuvenation of the vegetation community by the dispersal and germination of seeds (Cronk and Fennessy 2001). Wetlands that maintain permanently saturated conditions typically have lower species diversity than those that have fluctuating water levels. A fluctuating hydroperiod also tends to enhance productivity, organic matter accumulation, and nutrient cycling by creating a more species rich environment (Mitsch and Gosselink 2000).

Soil

The soil composition of the mitigation wetlands were confirms the results of other studies: the soil constituents measured were markedly different from natural wetlands soils (Tables 9 and 10). Phosphorus, nitrogen, and carbon were significantly lower in the mitigation sites. Low organic matter content and nutrient availability have been shown to limit the development of wetland creation projects (Gibson et al. 1994, Bishel-Machung 1996, Shaffer and Ernst 1999, Stolt et al. 2000). Bishel-Machung et al. (1996) found that soil organic matter levels (SOM) in natural palustrine wetlands were 6.6% to 16.1% higher than in mitigation sites. Similarly, Shaffer and Ernst (1999) measured SOM in 95 palustrine mitigation wetlands in Portland, Oregon and found that natural wetland soils contained an average of 59.1% more SOM than mitigation sites. Stolt et al. (2000) found that reference palustrine wetlands had 5 to 10 times more organic carbon compared to mitigation systems. In our study, total carbon levels were over 6 times higher in the natural sites. At 9 out of 10 of the mitigation sites, soil carbon (measured as both %OC, %OM) was lower than the minimum value found at any of the natural sites. The New Albany HS mitigation wetland, which was built partially on the site of a former wetland, was the only project with soils with organic matter content approaching the minimum values of natural sites. The low levels of organic matter are reflected in the bulk density and the percent solids of the mitigation wetland soils. Average bulk density of the natural soils was 0.62 g cm^{-3}, significantly lower ($p < 0.01$) than the 1.75 g cm^{-3} in the mitigation soils. Mean percent solids varied similarly.

The soils of the mitigation wetlands included in our study were also deficient in critical plant nutrients including nitrogen, phosphorus and calcium. Nitrogen levels in mitigation and natural wetlands averaged 0.24% and 1.12% N, respectively. Bishel-Machung et al. (1996) and Stolt et al. (2000) also found higher levels of total nitrogen in natural wetlands. Bishel-Machung et al. (1996) report median N values of 0.29% in natural versus 0.11% in mitigation wetlands ($p = 0.04$). In our study, all 10 mitigation wetlands had lower nitrogen levels than the minimum concentration measured at any of the natural wetlands.

Several studies have documented N limitations in restored and created salt marshes (Zedler et al. 2001, Craft 2000, Langis et al. 1991), but this has not been demonstrated in freshwater systems. Nitrogen availability has an influence on many ecosystem processes such as primary production, biomass, diversity, reproductive potential, and plant tissue N content (Langis et al. 1991), therefore the low N supplies in created sites may limit ecosystem function. Furthermore, there is no evidence that N levels are increasing with time, nor do we understand the reasons for such low nutrient retention (Zedler et al. 2001). More studies are needed to determine the mechanism(s) behind nitrogen and organic matter accumulation in mitigation and natural wetlands, and the time that might be required for mitigation soils to develop characteristics of natural wetlands with and without any design interventions, e.g. using donor soils from the impact site or providing soil supplements to mitigation wetlands.

We investigated the relationship between site age and percent carbon and nitrogen, using a regression analysis. A positive relationship would provide evidence that mitigation wetlands were developing structural (and hence functional) equivalence. When the mitigation sites included in this study were arrayed as a chronosequence, there was no relationship between site age and the carbon or nitrogen content of the wetland soil (Figure 47). The soil data collected here does not support the assumption that soil indicators like carbon or nitrogen are increasing over time.

The reasons behind low organic matter accumulation (and low organic C content) in mitigation sites has not been extensively studied. Bishel-Machung et al. (1996) were not able to explain the differences between the organic matter accumulation in natural and created wetlands with measures such as landscape cover or vegetation. Interestingly, Cole et al. (2001) found no relationship between biomass production and the accumulation of SOM in created wetlands. Shaffer and Ernst (1999) did find a relationship between soil organic matter accumulation and the extent of inundation: wetlands that had lower soil organic matter concentrations were saturated for longer periods of time. This is an interesting relationship because saturated conditions generally slow the rates of decomposition and therefore should increase the build-up of organic matter.

The addition of soil amendments from remnant or near-by wetlands is one method that could "jump start" the formation of nutrient rich soil (Brown and Bedford 1997). Stauffer and Brooks (1997) used additions of both leaf litter and soil organic matter in an attempt to accelerate the development of newly created wetland plots. The researchers found that after two years, leaf litter plots contained an average of 8.2% organic matter, which more closely resembles organic matter levels found in of natural wetlands. The use of soil amendments appears to be a method that could be applied to nutrient-deficient mitigation wetlands.

One factor that appears to affect soil properties is the method of wetland construction. Construction techniques have been shown to negatively influence the structure and development of wetland soil. Mitigation marshes are often constructed by some or all of the soil's A-horizon with heavy machinery. The A-horizon is typically high in organic matter and plant materials that are rich in nutrient content (Sposito 1989). When the top layer of soil is removed the underlying soil layer (B-horizon) is exposed. The B-horizon often consists of thick clay materials (Sposito 1989). Stolt et al. (2000) for example, found that when the soil was excavated down to the B-horizon, soil microrelief and plant diversity were reduced. Many of the wetland sites in this study were excavated down to the B-horizon and visibly lacked any build up of organic sediments; time was not correlated with soil improvement. Better design and construction, along with a requirement that replacement wetlands are truly restoration projects (as opposed to created wetlands built in upland locations) are needed to overcome this limitation.

Soil composition can serve as a measure of ecological integrity in order to gauge the development or progress of wetland mitigation projects. The NRC (2000) recently described soil organic matter content (or soil organic carbon) as the "best" indicator of soil quality because it responds to environmental disturbance and influences other functions within an ecosystem (NRC 2000). For example, soil characteristics have been shown to influence the vegetation composition and the susceptibility of a wetland to invasion by non-native species (Bishel-Machung 1996, Shaffer and Ernst 1999, Brown and Bedford 1997). Recent work has even equated soil characteristics, particularly organic carbon levels to wetland condition and surrounding land use (Cohen et al. in press). For these reasons as well as their ease of measurement, we propose that both SOM and N be used as indicators of ecosystem condition and as performance standards.

Vegetation

The vegetation-based bioassessment tools that have been developed for use in monitoring and assessment provided clearly distinguishable differences in the two populations of wetlands. Mean VIBI scores were more than twice as high in the natural wetlands, despite the fact that the natural wetlands were arranged along a gradient of disturbance. The Dever site was the lowest scoring natural wetland with a VIBI score of 9. It is a remnant isolated wetland in a row crop field, receiving agricultural runoff and is dominated by *Typha* and other tolerant species common to agricultural wetlands. Big Island Area D was the highest scoring mitigation site, with a VIBI of 50. This site is a 10.4 ha area that is part of a much larger mitigation bank (105 ha) that was extensively planted and partially located on hydric soils of a former wet prairie complex.

The composition of the communities varied significantly. Mitigation sites consistently differed from good to high quality natural emergent marshes on most VIBI metrics (Figures 21 to 24) and had similar average VIBI scores to highly disturbed natural marshes (Figure 25). Ordinations of plant species presence and abundance and individual metric performance suggests that the mitigation wetlands studied are forming a distinct population of wetlands from natural wetlands. Mitigation wetlands had a higher relative abundance of tolerant species, fewer intolerant species, and fewer species of important wetland plant genera, e.g. *Carex*. Their scores on the FQAI were significantly lower (14.2) than the natural sites (21.6) reflecting the dominance of generalist species favored in early successional or disturbed conditions (Table 12) and this pattern continued when FQAI scores were evaluated against the larger reference data set (Figure 23), where mitigation scores were equivalent to the scores of sites in the 1st ORAM tertile. Lopez and Fennessy (2002), using a transect sampling method, characterized a group of natural wetlands (n = 20) ranging from highly disturbed to relatively pristine and found FQAI scores ranged from 12 to 27. The low FQAI scores found reflect the low plant diversity and the generalist nature of the species present.

Aboveground biomass production varied significantly with natural wetlands producing more biomass compared to the mitigation sites, averaging 34.7 g 0.1 m^{-2} (equivalent to 347 g m^{-2}) and 20.9 g 0.1 m^{-2} (209 g m^{-2}), respectively. These values are low compared to values reported in other studies. Mitsch and Gosselink (2000) describe the range of biomass for freshwater marshes anywhere between 500 - 5500 g m^{-2}. The low values of biomass recorded in our study could be a result of several factors, including site selection in previous studies, and the choices of sampling methods. It should be noted that average biomass of highly disturbed natural wetlands (1st ORAM tertile) was approximately 900 g m^{-2} (Figure 22). Because the soils of mitigation wetlands were extremely deficient in nutrients, we suggest that low nutrient availability has a direct effect on aboveground biomass production. Many of the mitigation wetlands had permanently high water levels. The lack of a distinct dry-down period during the growing season can prohibit seed germination (Cronk and Fennessy 2001, van der Valk et al. 1994), limit species diversity and inhibit the production of above ground biomass.

Only one other study that we are aware of has compared biomass production in natural and created wetlands. In that study, Cole et al. (2001) found that created wetlands had a higher amount of aboveground biomass production compared to natural sites (1548 - 5164 g m^{-2} and 1296 - 4352 g m^{-2}, respectively). They attributed the high range of biomass in the created sites to species composition. *Typha* sp. and *Phalaris arundinacea* dominated the created sites whereas the natural sites were composed of a mixed vegetative community. The mitigation sites in our study often had very low plant cover, low nutrient availability, and large areas of open water, which resulted in low overall biomass production.

Species richness also varied in the two wetland populations although differences were not statistically different. Natural sites had an average of 31.0 species in the sample plots whereas mitigation sites averaged 25.6 species. Species richness has often been used as a measure of wetland mitigation success. Our study suggests that measures such as the VIBI are a more sensitive measure of ecological integrity, in part because simple species richness does not take into account successional state, overall plant community attributes or the type of species inhabiting a wetland, i.e. tolerant versus sensitive species, invasive versus native species, etc. The VIBI has been proven as a useful biological indicator for natural wetlands (Mack et al. 2000, Mack 2001, Mack 2004c). As a multimetric index it includes metrics that cross ecosystem levels from species richness, to community composition, to productivity. The NRC (2001) recommended that mitigation performance and monitoring include measures which evaluate multiple wetland characteristics. Our results show that measures like the VIBI are effective tools for evaluating the condition of mitigation wetlands, especially when coupled with key supplemental biogeochemical indicators like carbon, nitrogen, and hydrology. To the extent that the mitigation wetlands in this study are characteristic of the larger population of mitigation wetlands in Ohio, the structural (VIBI, FQAI) and functional (decomposition, nutrient concentrations, etc.) data collected here make clear that replacement wetlands are not meeting the goal of no net loss of wetland structure, nor, as other data presented here show, fundamental ecosystem processes.

Macroinvertebrates

The invertebrate data showed substantial differences in the numbers and type of taxa, abundance of tolerant and sensitive species, and the metrics and scores of the WICI between the mitigation sites and natural sites in this study. In the reference condition (i.e. relatively undisturbed) wetlands, the trophic structure was dominated with the more sensitive invertebrate herbivores and detritovores (oligochaetes, cladocerans, ostracods, limnephilid caddisflies) feeding directly on the microbial organisms (primary producers, Figure 30). These invertebrates are in turn fed upon by dytiscid beetle larvae and adults. Some of the more sensitive invertebrate taxa have specific wetland plant association requirements and may not be present at degraded sites with limited vegetation diversity.

Studies comparing invertebrate communities in natural and mitigation wetlands need to define the reference condition they can determine if the invertebrate community in mitigation wetland(s) studied is equivalent to natural wetlands of the same type (the same can be said for many (most) other studies of the floral and faunal communities of mitigation wetlands).

Disturbed (nonreference) natural wetlands and the mitigation wetlands in this study exhibited an altered trophic structure (Figure 30). Soils with deficient carbon or nutrient stacks and leaky nutrient processes may reduce or alter the microbial community and encourage blooms of filamentous algae. In response, the invertebrate community shifts in composition to tolerant forms of herbivorous corixids, hydrophilid and haliplid beetles, the tolerant forms of detritivore snails *Physella* spp. and *Gyraulus parvus,* and fish and frogs as the dominant predators (Figure 30).

Amphibians

The National Research Council (2001) pointed out the lack of design and evaluation criteria for animals, except for waterfowl and a few other bird species in the wetland mitigation projects they reviewed. They also observed that no consideration is given to the terrestrial requirements of wetland animals. For amphibian species, which generally can only migrate relatively short distances, the lack of nearby terrestrial and aquatic habitats with existing populations would be a critical limiting factor to the colonization of mitigation wetlands. For example, in the highly agricultural landscape of central and western Ohio, the range of the wood frog has been greatly reduced (Davis and Menze 2000). Historically, this frog species was present over the entire state. However, the intensive land clearing and farming activities in the 19th Century left only remnant woodlots and eliminated the wood frog from much of central Ohio and nearly all of western Ohio. Even though significant tracts of forest have been reestablished in some areas and the wood frog is one of the furthest ranging pond breeding amphibian species, it has not been able to recolonize these areas.

Porej (2004) and Porej et al. (2004) developed predictive models based on landscape (%forest cover) and wetland characteristics (amount of shallow zones in wetland, presence of predatory fish) for Ohio amphibian species using Ohio EPA's existing reference wetland data set (Micacchion 2002) and data collected from 48 mitigation sites. Porej (2004) and Porej et al. (2004) found that the "Amount of forest cover within the core zone [200m of the wetland] was included in the most parsimonious models for overall salamander diversity, and individual models for presence of spotted salamanders, Jefferson salamander complex (*Ambystoma jeffersonianum*), smallmouth salamanders (*Ambystoma texanum*) and wood frogs" (Porej 2004, p. 41). Land use beyond 200m (e.g. %forest, road density, etc.) was also important for overall salamander diversity, red-spotted newts, tiger salamanders, and wood frogs (Porej et al. 2004).

In addition to land use factors, Porej (2004) also found that the absence of "littoral" shallows in the wetland and the presence of predacious fish species altered amphibian populations in natural and mitigation wetlands. Overall amphibian diversity was significantly higher for wetlands *with* shallows and *without* predacious fish than for wetlands that had predacious fish, lacked shallows or had some combination of these factors (Porej 2004). The amphibian community structure was also different with certain species thriving, e.g. bullfrog (*Rana catesbeiana*), green frog (*Rana clamitans*), and toads, and others highly reduced or lacking altogether, e.g. spring peeper (*Pseudacris crucifer*), western chorus frog (*Pseudacris triseriata*), and most salamanders. This makes sense since most pond breeding amphibian taxa are adapted to forested wetlands (or wetlands located in an upland forest matrix) that have an isolated surface water and/or ground water source, dry up seasonally and therefore do not provide habitat for predacious fish (Hecnar and M'Closkey 1997, Kats et al. 1988, Micacchion 2002).

When wetlands are constructed to mitigate for impacts to seasonally inundated depressions, replacement of amphibian breeding habitat requires equivalent landscape features and hydrology. Porej (2003, 2004) surveyed 111 mitigation projects permitted by Ohio EPA. Even though almost 50% of permitted wetland impacts in Ohio are to forested wetlands, virtually all mitigations attempted were emergent communities. Porej (2003, 2004) also found that only 54% of small mitigation wetlands (<1 ha) *had* shallows and *lacked* predatory fish and only 23% of larger mitigation wetlands (> 1 ha) *had* shallows and *lacked* predatory fish. Habitat features like vegetation type and abundance are known to strongly influence amphibian richness and the availability of breeding sites (Richter and Azous 1995) and Pechmann et al. (2001) found that amphibian populations in mitigation and natural wetlands varied as a function of hydrology, substrate conditions, and vegetation.

Prior research in Ohio wetlands has

shown clear differences in the composition of amphibian communities between emergent and forested wetlands (Micacchion 2002, Micacchion 2004, Porej 2004, Porej et al. 2004). In Ohio, most sensitive amphibian species are adapted to living and breeding in forested wetlands. Even intact natural emergent wetlands score significantly lower on the AmphIBI than forested wetlands (Micacchion 2002, Micacchion 2004) for several reasons: 1) the majority of pond breeding amphibians in Ohio are adapted to a landscape that was 95% forested prior to European settlement; 2) emergent wetlands are often surrounded by nonforested uplands; and 3) many emergent wetlands are located in riverine landscape positions and have predatory fish. Emergent wetlands also do not have deciduous tree leaf litter serving as an important base of the food chain (Calhoun 2004). Many pond breeding amphibians are dependent on this food source directly, as is the case with anuran larvae, or indirectly through utilizing as prey those organisms that do. All of these factors and others limit the suite of amphibian species that can be expected in wetlands dominated by emergent vegetation.

Figure 33 illustrates the similarities between the amphibian communities in both the natural and mitigation emergent wetlands and the clear differences between those communities and wetland forests and shrub swamps. The natural emergent sites and the mitigation sites are dominated by three taxa: bullfrogs, green frogs, and toads. The high quality forest and shrub sites differ in the species that comprise their amphibian communities. A significant part of their communities are made up of species usually not encountered at natural emergent sites. These species include wood frog, spotted salamander, marbled salamander (*Ambystoma opacum*) and Jefferson salamander, and to a lesser degree the tiger salamander and smallmouth salamander.

The ordinations show the strong similarity in amphibian community composition between natural and mitigation emergent vegetation wetlands suggesting that it should be possible to reproduce amphibian communities similar to those at natural emergent sites. But, the natural wetlands in this study still scored significantly higher (6.5) on the AmphIBI than the mitigation wetlands (0.3) indicating a lack of functional parity. But, this is still in stark contrast to the almost complete inability of mitigation wetlands, like the ones studied here, to replicate the amphibian communities of natural forest and shrub wetlands. Porej (2004) found equivalent levels of amphibian richness but clear tradeoffs in amphibian assemblages, with the 48 mitigation wetlands he studied virtually lacking in forest dependent amphibian species. Failure to include information from the amphibian communities of natural forest and shrub wetlands in the analysis leads to an incomplete, if not erroneous, picture of mitigation wetland performance.

Decomposition

Decomposition rates are typically fastest in the period immediately following leaf senescence, due primarily to the rapid breakdown of vascular plant material and the loss of soluble organic and inorganic materials shortly after exposure to saturated conditions (Webster and Benfield 1986, Battle and Golladay 2001). In our study more mass was lost during the first collection period for both control and on-site litter in the natural and mitigation wetlands. After approximately 37 days, the natural and mitigation wetlands lost 45.3% and 39.2% of the on-site *Typha*. After approximately 45 days, the natural and mitigation wetlands lost 57.5% and 51.5% of the control *Typha*. Over the next two incubation periods, the amount of mass lost for both natural and mitigation were substantially lower (ranging between losses of 0 to 31%). This points out that in order to more fully capture the initial rates of decomposition (exponential decay), litter bags should be removed with more frequency early in the study (for example, at 2, 4, 6 and 8 weeks and then monthly thereafter for a maximum deployment of 6 to 9 months). We found deterioration of the physical integrity of the litter as well as plant growth on and through the mesh bag made data from samples removed after 9 months highly variable.

Decomposition rates of both on-site and control litter were significantly lower in the mitigation wetlands. Decomposition rates are known to vary for many reasons including hydrological conditions and nutrient availability. To investigate whether hydrology was correlated with differences, we compared the mass loss to the hydrological conditions present at the time of collection. On the day of collection, bags were classified as either dry (no standing water), submerged (bag under standing water), wet (saturated soil but no standing water), or buried (no standing waters but the bags were covered with sediment and/or seedlings). Bags that were submerged initially lost litter significantly faster than dry or buried bags ($p = 0.001$). However, submerged bags were found in both natural and mitigation sites and mean decomposition rates were higher in submerged bags in the natural sites. Therefore hydrology alone does not fully account for differences in decomposition between wetland populations (Figure 48). Several other factors including the activity of the microbial community and litter quality (e.g. nutrient content) can affect the decomposition process. Decomposition rates are typically faster when nutrient availability is higher, either in water, or in the plant litter itself (Lee and Bukaveckas 2002, Peterson et al. 1993, Webster and Benfield 1986). We found a weak but significant correlation between decomposition rates and initial litter nitrogen concentrations of the on-site leaf litter at 6 and 9 months (Figure 49). This provides one explanation for the faster rates in natural systems where N concentrations in plant litter was significantly higher.

Other studies in wetland ecosystems have shown that the nutrient availability within a wetland affects decomposition rates (Arp et al. 1999). Aerts and de Caluwe (1997) found that plant litter in fens decomposed at faster rates because of the high nutrient concentrations present in these wetlands. In our study, the natural wetlands, which had higher litter and soil nitrogen concentration, lost more mass compared to the mitigation sites. The lower nutrient status of the mitigation sites appears to limit decomposition. We are aware of only one other study investigating decomposition in created wetlands (Atkinson and Cairns 2001). In that study, the authors found that rates were higher in 20 year old sites (average of 72% *Typha latifolia* mass remaining) compared to 2 year old sites (80% mass remaining), and rates overall were slower in the created wetlands as compared to natural sites (Atkinson and Cairns 2001). Our data confirms their findings, showing that, with respect to decomposition, functional equivalence between natural and mitigation wetlands has not been achieved.

In spite of the dramatic differences in this key ecosystem process, we feel that there is limited potential for measures of decomposition to serve as an ecological indicator of wetland ecosystem condition. A good ecological indicator is one that is relatively easy to measure (Keddy et al. 1993, Dale and Beyeler 2001). This is not the case for decomposition, which is a time and labor intensive parameter to quantify. Decomposition rates also vary as a function of many environmental variables (e.g. hydrology, temperature, sedimentation), making the results potentially highly variable. Our data does add insight however, into the ecosystem effects of low nutrient availability in mitigation wetlands. While not amenable for every-day use in mitigation monitoring, further studies in other wetland types should be undertaken to quantify this important ecosystem process.

Plant Litter Nutrient Analysis

After approximately 3 months of incubation, the mean nitrogen concentration of the on-site plant litter was significantly higher in the natural wetlands, averaging 3.19 mg N g^{-1} litter compared to 2.67 mg N g^{-1} litter, in mitigation sites (Figure 37). To control for differences in the quality of litter, control litter was also deployed. Despite the fact that the initial litter quality was the same in all sites, N concentrations in the decomposing plant litter were higher at the natural sites at all collection periods (Figure 39). For example, control plant litter averaged 3.44 mg N g^{-1} litter at natural sites and 2.81 mg N g^{-1} litter at the mitigation sites after 330 days, providing evidence that the low nutrient availability in the mitigation wetlands is limiting key ecosystem processes. For example, nitrogen accumulation in decomposing litter is generally attributed to the microbial colonization of litter as it breaks down (Mitsch and Gosselink 2000). Higher nitrogen concentrations tend to be found in samples with a higher density of microbes. We suggest that because there are fewer available nutrients in the soil of the mitigation sites, it is more difficult for a wetland to support an abundant microbial community.

Phosphorus concentrations in the on-site and control litter follow very similar patterns. Initial concentrations are higher in the natural wetlands litter, and much more is leached following the first incubation period. Phosphorus levels then increase and stabilize in the natural wetland's litter, while there is essentially no change in P levels in the litter from mitigation sites (Figures 38 and 40). Changes in P concentrations in the control litter show that following the initial loss of P to the surroundings, P levels climb in the natural sites and show no change in the mitigation sites (Figure 37). P immobilization (calculated as the amount of P increase in decomposing litter) following the first incubation period therefore, is close to zero in the mitigation sites, amounting to approximately 0.07 mg P $m^{-2.}$ y^{-1}. Over the whole study period, P immobilization was negative, i.e. there was a net release of P to the water column by litter in the mitigation sites, while net immobilization was essentially zero in the natural wetlands.

Nitrogen immobilization by the control litter in natural wetlands was approximately 1.7 g N m^{-2} y^{-1} greater than in mitigation sites (total net immobilization equal to 3.4 g N m^{-2} y^{-1} in the natural sites). Windham and Ehrenfeld (2003) found litter immobilization rates of 12.0 g N m^{-2} y^{-1} for *Phragmites australis* litter and 2.0 g N m^{-2} y^{-1} for *Spartina patens* litter using similar techniques in a study of tidal salt marshes (no comparable data is presented for P in their study). The potential effects of higher N concentrations (greater immobilization) include a greater magnitude of internal N cycling, greater soil biota activity (which in turn help drive carbon and nutrient dynamics) and better quality litter which serves as the base of the heterotrophic food web (Windham and Ehrenfeld 2003). Blair et al. (1990) found that increasing plant diversity was correlated with higher initial nitrogen release from litter, and a decrease in subsequent immobilization. Recognizing that the natural wetlands in this study had higher levels of diversity, our data lends support to their first finding, but not for the second. More diverse floral and faunal communities increase the likelihood that a wetland will contain species that affect ecosystem processes such as the ability to capture and use nutrients (Chapin et al. 1997). This is one mechanism by which diversity exerts control over ecosystem function.

A conceptual diagram was constructed to illustrate the potential ecosystem effects of low nitrogen levels in mitigation wetlands (Figure 50) using the patterns observed in this study including, low nitrogen and carbon in soils, lower biomass accumulation, low nitrogen in the decomposing litter, and slower decomposition rates. These data offer quantitative evidence of the innate differences between the natural and mitigation wetlands in this study, and the negative feedback loops that perpetuate and propagate nitrogen deficiency throughout the mitigation wetland ecosystem.

Multivariate Analyses

Experimental data from our field study has shown that the mitigation sites were not similar to the natural sites in terms of plant community composition, biomass production, soil characteristics, decomposition rates and nutrient fluxes. In order to answer our original question, i.e., where along a gradient of natural wetland condition the mitigation sites fall, variables such as soil and vegetation were combined to gauge the similarity between these two populations. To assure that this comparison was not biased by recognizing the constraints of the landscape in Ohio that is highly altered by human activities, we sampled natural sites ranging from highly disturbed to relatively unimpacted. A cluster analysis was done combining measures of vegetation (FQAI score) and soil C and N content (Figure 51). There is a distinct division between natural and mitigation wetlands in which the natural sites are found on the left side of the figure and the mitigation wetlands (9 out of 10) are found on the right, indicating that not only are the mitigation and natural wetlands different in terms of individual components (e.g. soil, vegetation), their entire ecosystem structure is different. The imaginary line that can be drawn between the natural and mitigation wetlands is representative of the reoccurring problem in wetland mitigation.

It is important to note that the order in which these wetlands are shown on the figure does not reflect the actual condition of the wetland (i.e. because Calamus is listed as the first wetland on the graph, it is not the "best" natural wetland). The mitigation site (New Albany HS #13) that is included among the natural wetlands) has relatively high levels of soil C and N. If mitigation wetlands were performing up to the standard of even the most degraded natural wetlands, these two populations would be interspersed along the x-axis.

A second cluster analysis was run based solely on physiochemical aspects such as soil characteristics (carbon, nitrogen, and phosphorus content) and the *Typha* litter nutrient concentrations during decomposition (using data after the second collection period) (Figure 52). This also shows mitigation and natural wetlands grouping separately. However, in this analysis two mitigation sites were grouped among the natural wetlands, New Albany (7) and Sacks (1), due to their relatively high soil nutrient concentrations.

Finally, plant community data (presence, relative cover) and associated environmental variables were evaluated using DCA (Figure 53). The ordination shows a strong separation of natural, high quality wetlands, located to the right of the mitigation sites. Only one natural site is located to the left of the mitigation sites (Dever, the remnant wetland located in a crop field and the most disturbed by human activities of any of the natural sites). The biplot indicates the distribution of sites relative to key environmental variables, confirming that soil fertility (%C and %N) and soil density (% solids) are key variable differentiating the sites.

CONCLUSIONS

Noted ecologist A.D. Bradshaw (1987) described restoration ecology the "acid test" of our understanding of natural ecosystems. If the goal of restoration is to return an ecosystem to a more natural, self-regulating and self-sustaining condition (NRC 1992, Mitsch and Gosselink 2000), wetland mitigation has not yet met this goal. The mitigation wetlands in this study, in terms of their structure and function, group as a separate population from the natural sites, indicating the creation of a new subclass of wetlands on the landscape. Major differences included 1) deeper surface water at the mitigation sites; 2) greater depth to ground water at the mitigation sites; 3) substantially reduced soil nutrient pools at mitigation sites; 4) significantly different movement of nutrients, both in terms of rates and quantities between ecosystem components; and, 5) reduced nutrient availability that propagates throughout the mitigation systems and appears to set a limit on ecosystem development. The use of ecological indicators to measure the condition of wetland ecosystems has proved effective in their ability to reflect ecological condition and will aid in our ability to monitor the outcomes of mitigation projects, helping to improve their success. Based on the results of this study, several indicators could serve as measures of mitigation performance relative to natural wetlands:

1. Soil chemical and physical characteristics especially soil organic carbon and soil nitrogen content and percent solids in the soil or bulk density;

2. Hydrological characteristics including mean depth to ground water and percent time water is found in the root zone (i.e. greater than -30cm) (as compared to a natural reference ecosystem of similar

hydrogeomorphic class); and

3. Multimetric indices developed from natural reference wetland data sets.

A full treatment of performance standards is presented in part 6 of this series (Mack et al. 2004).

After the intensive fieldwork that was conducted for this project and the large quantity of data that were produced, we can make recommendations that might increase the likelihood of "successful" mitigation projects. We understand that creating wetlands that are similar to natural wetlands (in both structure and function) may take time as well as a more sophisticated understanding of individual wetland components and their interactions. However, steps can be taken to employ the knowledge that we do have in order to increase success:

1. The methods of construction seem to be a key factor in the soil characteristics and soil development of newly created wetlands. Excavating the top layer of soil does not appear to serve any benefits for initial wetland development therefore should be avoided whenever possible. If the top layer of soil is excavated it should be replaced The amount of impact to the wetland soil during construction should be minimal. Soil sampling of *in situ* soils prior to site selection or construction would aid in determining whether soil amendments are needed

2. Recreating the hydrology (also related to the landscape position) is essential to the overall development of mitigation wetlands. Research needs to be done prior to site selection on the necessary level and depth of excavation. Mitigation wetlands generally have standing water for longer periods of time as compared to natural wetlands. This could potentially be avoided if the wetland was situated in context to its surrounding landscape.

3. Because soil nutrient availability within mitigation wetlands was so poor, we suggest that measures be taken to enrich these systems after initial construction, particularly if extensive excavation is done. It is apparent that more research needs to be done in this area. A supply of organic matter appears to help vegetation establishment and successional patterns in mitigation projects. We suggest that studies investigating the effects of carbon and nitrogen amendments are needed in order to better understand the mechanism behind C and N dynamics in mitigation wetlands.

4. Wetland placement within the landscape is very important; the majority of the mitigation wetlands did not hydrologically or topographically fit within their surrounding landscape. A simple solution to this would be to use the general HGM guidelines proposed by Brinson (1993), Smith et al. (1995) and Bedford (1996) and the Ohio-specific classification outlined in Mack (2004a) when determining the location of the project as well as consideration of surrounding human land uses.

The results of our multivariate analyses confirm with empirical data what other studies have been suggesting for several years, we have not yet shown that we can create wetlands comparable to natural ones. The laws that govern wetland preservation are still operating under the assumption that it is scientifically possible to recreate, in both structure and function, a wetland with full ecological integrity (NRC 2001). This has not yet been demonstrated in the mitigation wetlands evaluated in this study.

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Table 1. Site description of each wetland including name, type (natural (n) or mitigation (m), location, and age for mitigation wetlands. Mitigation sites are sorted by age.

site	type	age	county	coordinates
Baker Swamp	N	na	Jackson	38° 51' 03" N, 82° 36' 58" W
Ballfield	N	na	Knox	40° 16' 08" N, 82° 16' 59" W
Calamus	N	na	Pickaway	39° 35' 02" N, 82° 59' 53" W
Dever	N	na	Franklin	39° 59' 10" N, 83° 10' 28" W
Eagle Creek Beaver	N	na	Portage	41° 17’ 30” N, 81° 03’ 13”W
Eagle Creek Marsh	N	na	Portage	41° 17' 13" N, 81° 03' 48" W
Lake Abrams	N	na	Cuyahoga	41° 22' 52" N, 81° 50' 13" W
Lodi North	N	na	Medina	41° 02' 08" N, 82° 01' 49" W
Rickenbacker	N	na	Franklin	39° 50' 14" N, 82° 54' 42" W
Bluebird	M	1	Delaware	40° 11' 38" N, 82° 52' 10” W
Slate Run Bank SE	M	2	Pickaway	39° 45' 15" N, 82° 52' 01" W
New Albany HS	M	3	Franklin	40° 05' 13" N, 83° 49' 08" W
Sacks	M	3	Knox	40° 20' 59" N, 82° 19' 33" W
JMB	M	5	Franklin	39° 52' 34" N, 82° 53' 31" W
Big Island Area D	M	6	Marion	40° 34' 17" N, 83° 17' 11" W
Trotwood	M	6	Montgomery	39° 48' 57" N, 84° 17' 09" W
Prairie Lane	M	6	Wayne	40° 44' 02" N, 81 °57' 23" W
Medallion No. 20	M	7	Delaware	40° 09' 46" N, 82° 53' 15" W
Pizzutti	M	9	Delaware	40° 08' 18" N, 83° 01' 14" W

Table 2. Description of metrics used in the VIBI-Emergent (from Mack 2004b).

metric	code	type	metric increase or decrease w/ disturbance	description
number of *Carex* spp.	Carex	richness	decrease	Number of species in the genus *Carex*
number of native dicot spp.	dicot	richness	decrease	Number of native dicot (dicotyledon) species
number of native, wetland shrubs	shrub	richness	decrease	Number of shrub species that are native and wetland (FACW, OBL) species
number of hydrophyte spp.	hydrophyte	richness	decrease	Number of vascular plant species with a Facultative Wet (FACW) or Obligate (OBL) wetland indicator status (Reed 1988; Andreas et al. 2004).
ratio of annual to perennial spp.	A/P	richness ratio	decrease	Ratio of number of nonwoody species with annual life cycles to number of nonwoody species with perennial life cycles. Biennial species excluded from calculation
FQAI score	FQAI	weighted richness index	decrease	The Floristic Quality Assessment Index score calculated using Eqn. 7 and the coefficients in Andreas et al. (2004)
relative cover of sensitive plant spp.	%sensitive	dominance ratio	decrease	Percent coverage of plants in herb and shrub stratums with a Coefficient of Conservatism (C of C) of 6,7,8,9 and 10 (Andreas et al. 2004) divided by total percent coverage of all plants
relative cover tolerant plant spp.	%tolerant	dominance ratio	increase	Percent coverage of plants in herb and shrub stratums with a C of C of 0, 1, and 2 (Andreas et al. 2004) divided by total percent coverage of all plants
relative cover of invasive graminoid spp.	%invgram	dominance ratio	increase	Percent coverage of *Typha* spp., *Phalaris arundinacea,* and *Phragmites australis* divided by total percent coverage of all plants
sum of relative cover of annual spp. and cover of unvegetated areas	%unvegetated	dominance ratio	increase	The sum of the relative cover of annual plant species (percent annual spp. cover divided by total spp. cover) and the percent cover of unvegetated areas.
mean standing biomass	biomass	primary production	increase	The average grams per square meter of clip plot samples collected at each emergent wetland

Table 3. Incubation time for litter bags at each wetland site for each incubation period (days) for ON-SITE and CONTROL litter.

Wetland	1st pickup ON-SITE	1st pickup CONTROL	2nd pickup ON-SITE	2nd pickup CONTROL	3rd pickup ON-SITE	3rd pickup CONTROL
Natural						
Baker	37	58	89	108	344	364
Ballfield	36	55	87	106	307	326
Calamus	37	58	89	109	339	359
Dever	33	53	85	105	296	316
Eagle Creek Beaver (2002)	na	108	na	269	na	290
Eagle Creek Marsh (2002)	na	108	na	269	na	290
Lake Abrams (2002)	na	108	na	269	na	290
Lodi	41	60	87	107	301	321
Rickenbacker	36	58	89	103	299	321
Created						
Big Island	41	60	87	107	301	321
Bluebird	41	55	86	103	na	na
JMB	36	56	84	111	295	322
Medallion	36	56	82	102	298	318
New Albany	42	55	89	103	303	317
Pizzutti	33	53	85	105	294	314
Prairie Lane	41	60	87	106	301	320
Sacks	36	55	87	106	na	359
Slate Run	37	58	89	109	301	321

Table 4. Hydrological attributes of natural and mitigation wetlands included in the study for the 2001 and 2002 growing seasons. The mean percent time water was found in the root zone (30 cm), mean depth to ground water, and the maximum and minimum ground water levels recorded for each site are shown. Two ground water readings were recorded at each site per day except at Slate Run where only one ground water reading was taken per day.

site name	year	date of readings	% time >30cm	Mean depth (cm)	Median depth (cm)	maximum depth to water (cm)	minimum depth to water (cm)
Natural Sites							
Baker Swamp	2001	8/16 - 9/30	99	4.7	5.7	-2.9	33.1
Baker Swamp	2002	4/1 - 8/29	97	-4.5	-1.9	-46.4	22.6
Ballfield	2001	4/1 - 9/30	100	-6.1	-5.2	-32.9	-1.9
Calamus	2001	5/9 - 9/30	57	-19.7	-25.9	-76.5	20.3
Calamus	2002	4/2 - 9/30	53	-32.9	-27.2	-58.0	-0.6
Dever	2001	5/24 - 9/30	23	-53.2	-58.0	-85.4	0.5
Dever	2002	4/1 - 9/30	51	-40.9	-25.8	-86.4	0.5
Eagle Cr Beaver	2001	5/30 - 9/30	96	-12.8	-13.1	-37.0	2.1
Eagle Cr Beaver	2002	4/1 - 9/30	96	-4.1	1.4	-35.2	16.4
Eagle Cr Marsh	2001	5/30 - 9/30	57	-33.9	-26.0	-74.5	-5.9
Eagle Cr Marsh	2002	4/1 - 9/30	54	-37.5	-20.4	-68.7	-10.2
Lake Abrams	2001	5/24 - 9/30	58	-28.6	-24.1	-78.0	25.1
Lake Abrams	2002	5/8 - 9/30	70	-23.4	-13.0	-88.1	26.2
Lodi North	2001	5/17 - 9/30	49	-30.9	-31.9	-65.2	-7.8
Lodi North	2002	4/6 - 9/30	56	-29.3	-22.8	-88.3	-1.0
Rickenbacker	2001	5/14 - 9/30	36	-44.4	-74.0	-78.6	20.2
Rickenbacker	2002	4/1 - 9/30	51	-31.0	-23.8	-79.6	23.5
Mitigation sites							
Big Island Area D	2001	5/27 - 9/30	22	-58.3	-74.9	-81	-0.7
Big Island Area D	2002	4/5 - 9/30	73	-31.1	-21.6	-75.4	11.5
Bluebird	2001	6/5 - 9/30	0	-93.1	-99.3	-102.6	-51.3
Bluebird	2002	4/6 - 9/30	0	-60.0	-48.8	-94.0	-31.8
JMB	2001	5/20 - 9/30	21	-59.3	-76.4	-82.0	3.4
JMB	2002	4/1 - 9/30	47	-42.1	-39.3	-76.9	34.1
Medallion No. 20	2001	5/10 - 9/30	96	-0.7	3.9	-43.9	11.7
Medallion No. 20	2002	4/1 - 9/30	89	-7.3	-1.2	-61.4	3.1
New Albany HS	2001	6/6 - 8/31	69	-28	-12.8	-74.5	-3.4
New Albany HS	2002	5/9 - 9/30	1	-51.7	-73.7	-77.0	-6.1
Pizzutti	2001	5/10 - 9/30	56	-19.3	-22.8	-51.5	11.2
Pizzutti	2002	4/6 - 9/30	45	-26.9	-40.1	-52.3	5.6
Prairie Lane (2001)	2001	5/26 - 9/30	17	-55.8	-65.2	-81.5	-2.0
Prairie Lane (2002)	2002	4/6 - 9/30	43	-42.8	-64.7	-76.4	24.4
Slate Run Bank SE	2001	5/26 - 9/30	6	-67.7	-81.6	-85.4	3.0
Trotwood	2001	6/6 - 8/3	0	-102.2	-103.6	-106.4	-34.5
Trotwood	2002	4/1 - 9/30	0	-93.9	-102.6	-106.4	7.1

Table 5. Flashiness index results for mitigation and natural sites. The flashiness index calculated by averaging the absolute value of the difference between a later ground water measurement from the preceding ground water measurement.

site	type	N	mean inter-reading change (cm) (flashiness index)	std deviation	maximum single day change (cm)
Natural Sites					
Baker Swamp	riverine headwater wetland, with some beaver activity, permanently inundated	464	2.8	4.2	25.7
Ballfield	riverine mainstem with seasonal flooding and groundwater	299	1.9	2.8	16
Calamus	depression	1145	1.6	2.7	41.9
Dever	depression	786	1.5	3	51.8
Eagle Cr Beaver	riverine headwater with seasonal inundation and groundwater	1007	1	1.6	15.7
Eagle Cr Marsh	beaver impoundment on floodplain with seasonal inundation	1009	1.6	3.4	60.2
Lake Abrams	riverine headwater with massive stormwater inputs from surrounding urbanization	821	4.2	6.9	79.2
Lodi North	depression with stormwater inputs	792	3.8	4	51.1
Rickenbacker	depression	1124	2	4.4	65.5
Mitigation Sites					
Big Island Area D	impoundment on former hydric soils	753	1.5	3.3	42.4
Bluebird	depression on nonhydric soils	1109	2	4.5	45.2
JMB	depression on nonhydric soils with some stormwater	792	2.4	6	75.7
Medallion No. 20	depression on nonhydric soils, permanently inundated	1171	1.4	2.9	35.1
New Albany HS	depression on nonhydic soils	953	1.9	5.2	71.4
Pizzutti	impoundment on nonhydric soils with some stormwater input, permanently inundated	992	2.8	4.9	50
Prairie Lane	impoundment on hydric soils of low gradient floodplain with seasonal flooding from stream	768	2	5.4	70.9
Slate Run Bank SE	impoundment on nonhydric soils	232	2	6.9	60.7
Trotwood	impoundment on nonhydric soils with massive stormwater inputs	803	4.6	13.1	112

Table 6. Water chemistry parameters for natural and mitigation wetlands in this study. The number of samples collected per site is shown next to site name. Two-sample T-tests were run to compare mean values for the natural sites versus mitigation sites. Means for all natural versus all mitigation sites were not significantly different except TOC ($p = 0.05$); means for reference natural sites versus all mitigation sites were not significantly different except for pH ($p = 0.05$) and K ($p = 0.024$). "nd" = no data collected for that parameter.

Site	n	Conduc-tivity	pH	TSS mg/l	TOC mg/l	Ca mg/l	Mg mg/l	Mn mg/l	K mg/l	Turbidity ntu	Nitrite mg/l	NH4 mg/l	TKN mg/l	Total P mg/l
Natural Sites														
Baker Swamp	4	149	6.59	66	17	21.8	8.5	0.54	2.8	53.8	0.01	0.025	1.29	0.169
Ballfield	2	1520	7.65	164	18	88.5	23.0	0.87	4.5	111.0	nd	0.025	1.75	0.214
Calamus	1	244	6.50	46	26	27.0	10.0	0.43	2.0	18.4	0.01	0.025	1.61	0.230
Dever	4	nd	nd	9	13	44.3	14.8	nd	nd	nd	nd	0.076	1.12	0.233
Eagle Cr Beaver	2	201	8.58	7	48	23.5	5.0	0.32	2.0	46.0	0.027	0.583	3.77	0.380
Eagle Cr Marsh	2	164	6.60	2	45	16.5	5.0	0.55	2.0	96.8	0.028	0.084	2.56	0.324
Lake Abrams	1	1120	7.94	101	9	66.5	14.0	0.24	10.0	89.1	0.01	0.025	0.63	0.081
Lodi North	2	280	8.90	111	28	38.5	11.5	0.04	1.0	nd	nd	0.094	2.39	0.207
Rickenbacker	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Mean for all natural sites		**442**	**7.33**	**65.9**	**22.6**	**38.1**	**11.7**	**0.44**	**3.5**	**67.0**	**0.016**	**0.101**	**1.72**	**0.215**
Mean for reference sites		**368**	**7.01**	**66**	**26.5**	**32.2**	**10.5**	**0.55**	**2.7**	**63.3**	**0.017**	**0.115**	**1.87**	**0.220**
Mitigation Sites														
Big Island Area D	4	267	7.99	113	30	38.0	11.8	0.66	5.3	97.7	0.05	0.053	1.89	0.212
Bluebird	2	383	7.87	30	38	44.0	19.0	1.02	7.5	17.7	0.01	0.078	2.63	0.145
JMB	2	632	8.99	61	10	52.5	17.5	0.84	5.5	63.9	0.01	0.082	2.95	0.813
Medallion No. 20	1	1650	7.23	2	15	20.0	8.0	0.26	2.0	5.6	0.01	0.059	1.62	0.061
New Albany HS	1	426	7.06		31	60.0	11.0	0.72	7.0	127	0.01	0.222	2.84	1.060
Pizzutti	3	201	8.25	49	13	29.3	9.0	0.43	2.7	22.9	0.01	0.025	1.29	0.055
Prairie Lane	2	342	6.00	13	12	31.0	8.0	5.24	9.0	7.3	0.01	2.590	3.67	0.074
Sacks	2	195	9.41	13	21	19.0	8.5	0.11	2.5	2.7	0.01	0.025	1.52	0.252
Slate Run Bank SE	1	228	6.96	nd	12	36.0	11.0	3.59	1.0	nd	nd	0.025	1.73	0.124
Trotwood	3	678	7.49	31	9	48.0	22.7	0.46	4.0	21.6	0.01	0.056	1.05	0.152
Mean for all mitigation		**481**	**7.83**	**62.4**	**18.9**	**38.1**	**13.5**	**0.952**	**4.6**	**46.5**	**0.016**	**0.186**	**1.93**	**0.260**

Table 7. Comparison of median (25th - 75th percentile) water chemistry of wetlands (n ~ 200 water samples depending on parameter) from Ohio EPA reference wetland data set, 1996-2003. Impoundments included in riverine mainstem or riverine headwater. Notable values underlined. Note chemical similarities of slope systems, coastal marshes, and Lake Plains sand prairies. Note differences in bog systems from other types.

Description	HGM class	pH	TSS mg/l	TOC(%)	Ca mg/l	Fe mg/l	Mg mg/l	Cl^- mg/l	NH4 mg/l	P total mg/l
Bogs	weakly ombrotrophic	---	41.8(---)	59.0(---)	15.5(---)	17.1(---)	4.5(---)	4.3(---)	0.13(---)	0.59(---)
Bogs	moderate to strongly ombrotrophic	---	18.0(3-45)	45.0(16-71)	3.0(2-7)	1.4(0.5-3.7)	1.0(0.5-2.0)	<2.5	0.025(0.025-0.09	0.07(0.025-0.18)
Depressional marshes	depression	6.5(5.3-6.9)	9.5(7-42)	14.0(11-22)	35.0(24-45)	1.2(1.0-2.0)	12.0(7-14)	9.8(3-125)	0.025(0.025-0.12)	0.17(0.09-0.29)
Mainstem marshes	riverine mainstem	7.7(7.0-7.8)	47.0(20-177)	11.5(8-17)	43.0(36-73)	2.2(1.5-9.6)	11.0(9-21)	17.0(12-176)	0.055(0.025-0.241)	0.12(0.07-0.24)
Headwater marshes	riverine headwater	6.9(6.6-7.3)	27.0(17-68)	18.0(8-41)	24.0(17-51)	3.6(1.5-7.4)	6.0(5-17)	9.0(3-24)	0.025(0.025-0.054)	0.19(0.09-0.51)
Mitigation marshes	mostly impoundments	7.6(7.1-8.7)	33.0(12-96)	15.5(12-20)	31.0(20-43)	2.1(1.3-5.4)	10.0(7-14)	10.0(3-17)	0.025(0.025-0.088)	0.14(0.07-0.27)
Coastal marshes	Lake Erie Coastal	8.2(7.8-8.3)	77.0(13-98)	9.5(6-14)	58.0(39-65)	3.2(0.3-4.2)	15.0(12-21)	26.9(17-93)	0.051(0.025-0.074)	0.16(0.05-0.17)
Calcareous fens	slope	7.9(---)	2.5(2.5-2.5)	7.4(3-12)	59.0(46-91)	0.94(0.95-2.0)	30.0(28-38)	19.8(17-45)	<0.025	0.25(0.03-0.51)
Fen meadows	slope	---	18.8(---)	13.6(---)	48.0(---)	3.9(---)	11.5(---)	32.8(---)	0.042(---)	0.12(---)
Forest seeps	slope	---	---	---	---	---	---	---	---	---
Wet meadows	all except slope	---	---	---	---	---	---	---	---	---
Lake Plains Sand Prairies	mostly depressions	7.1(7.0-7.3)	7.0(6-152)	35.0(35-44)	19.0(19-57)	2.2(0.6-2.6)	6.0(5-10)	<2.5	0.025(0.025-0.13)	0.08(0.07-0.15)
Forest vernal pools	depression	7.0(5.4-7.4)	14.5(3-49)	16.5(6-23)	36.5(18-66)	0.80(0.5-1.5)	11.0(3-14)	<2.5	0.08(0.025-0.17)	0.14(0.09-0.28)
Shrub vernal pools	depression	6.8(6.5-7.2)	11.0(5-24)	29.0(15-35)	29.0(16-43)	0.95(0.7-1.9)	8.0(5-14)	2.5(2.5-6)	0.085(0.025-0.20)	0.25(0.10-0.52)
Wet woods	depression	7.4(---)	11.0(8-125)	39.0(33-41)	22.0(5-51)	2.3(1.6-3.1)	5.0(2-8)	2.5(2.5-19)	0.05(0.025-0.286)	0.12(0.09-0.42)
Mainstem swamp forests	riverine mainstem	7.0(6.2-7.2)	25.0(14-59)	18.0(16-23)	42.5(15-61)	1.9(1.4-4.4)	11.0(4-16)	2.5(2.5-42)	0.19(0.07-1.27)	0.48(0.22-0.76)
Mainstem shrub swamps	riverine mainstem	6.8(6.1-6.8)	38.0(27-73)	14.0(10-14)		3.1(2.1-8.8)	5.0(4-8)	2.5(2.5-7.6)	0.025(0.025-0.132)	0.12(0.08-0.23)

Table 8a. Standard agronomic soil chemistry parameters for natural and mitigation wetlands (n = 5). %OM determined by Walkly-Black Method, %OC determined by CHN Analyzer (See methods section). * = low %OM values due to underestimation of %OM by Walkly-Black method in high carbon soils.

site	%OM	%OC	%N	weak P ppm	strong P ppm	K ppm	Mg ppm	Ca ppm	Na ppm	pH	CEC
Natural sites											
Baker Swamp	4.1 + 0.36	5.36 + 1.0	0.49 + 0.07	17.2 + 3.2	71.4 + 6.6	225 + 91	264 + 21	1267 + 251	0	5.20 + 0.37	15.2 + 1.6
Ballfield	7.3 + 0.86	12.0 + 2.0	0.87 + 0.15	8.6 + 1.4	19.0 + 3.7	61 + 14	377 + 69	2229 + 267	61.2 + 37.5	6.40 + 0.24	15.9 + 1.3
Calamus	8.6 + 0.61*	27.7 + 4.2	2.36 + 0.50	12.0 + 4.2	24.0 + 9.1	79 + 24	568 + 144	2164 + 268	0	6.04 + 0.25	18.2 + 2.1
Dever	6.8 + 0.44	9.8 + 1.9	0.80 + 0.14	15.4 + 2.5	44.2 + 6.4	108 + 7	547 + 60	2249 + 147	0	5.66 + 0.32	21.3 + 0.5
Eagle Cr Beaver	8.5 + 0.70*	28.2 + 5.5	2.12 + 0.44	12.8 + 2.9	30.4 + 10.8	44 + 9	263 + 39	2022 + 252	0	5.06 + 0.10	20.2 + 1.9
Eagle Cr Marsh	5.6 + 0.19	6.7 + 0.6	0.54 + 0.04	10.6 + 2.5	37.0 + 6.8	70 + 9	145 + 12	1039 + 115	0	4.86 + 0.05	12.0 + 1.1
Lake Abrams	6.2 + 0.34	11.4 + 2.5	0.78 + 0.22	11.4 + 2.9	86.6 + 22.3	60 + 11	277 + 27	2147 + 192	417.4 + 82.1	5.88 + 0.30	18.7 + 1.4
Lodi North	10.9 + 0.21*	27.4 + 3.1	1.66 + 0.14	10.4 + 1.4	33.0 + 4.4	38 + 10	371 + 26	3052 + 360	0	5.72 + 0.19	23.7 + 2.1
Rickenbacker	5.3 + 0.44	7.4 + 1.7	0.53 + 0.09	9.2 + 1.2	43.8 + 5.3	164 + 18	338 + 38	1734 + 96	0	5.38 + 0.10	16.9 + 1.2
Mitigation sites											
Big Island Area D	3.0 + 0.29	2.5 + 0.3	0.28 + 0.02	4.0 + 0.6	18.8 + 3.1	171 + 23	372 + 31	1653 + 118	0	5.90 + 0.18	14.3 + 0.8
Bluebird	2.2 + 0.36	2.2 + 0.4	0.23 + 0.04	3.0 + 0.5	15.8 + 2.0	106 + 6	317 + 19	1144 + 33	0	5.96 + 0.17	10.4 + 0.2
JMB	2.0 + 0.27	2.5 + 0.3	0.26 + 0.02	9.0 + 0.9	41.2 + 6.2	101 + 7	345 + 28	2046 + 115	30.0 + 30.0	7.10 + 0.10	13.5 + 0.8
Medallion No. 20	1.7 + 0.17	1.9 + 0.2	0.22 + 0.02	1.6 + 0.4	12.6 + 0.8	108 + 2	295 + 36	1208 + 52	0	5.66 + 0.18	11.5 + 0.8
New Albany HS	5.8 + 0.72	4.3 + 0.9	0.32 + 0.02	24.0 + 3.6	62.0 + 8.1	134 + 9	325 + 36	2353 + 266	0	7.38 + 0.10	14.8 + 1.4
Pizzutti	1.6 + 0.21	1.3 + 0.2	0.17 + 0.02	2.0 + 0.4	7.4 + 0.8	64 + 2	199 + 33	984 + 91	0	6.12 + 0.07	7.8 + 0.8
Prairie Lane	2.7 + 0.19	2.4 + 0.2	0.24 + 0.02	5.8 + 0.2	21.6 + 4.2	92 + 14	212 + 12	1395 + 65	0	5.78 + 0.12	11.2 + 0.3
Sacks	3.4 + 0.24	3.6 + 0.5	0.32 + 0.04	14.8 + 3.8	75.2 + 6.8	137 + 14	337 + 26	1617 + 123	0	5.66 + 0.05	14.4 + 0.9
Slate Run Bank SE	2.3 + 0.09	2.1 + 0.3	0.20 + 0.01	2.2 + 0.4	15.0 + 1.6	101 + 14	266 + 23	1398 + 162	0	6.18 + 0.28	10.9 + 0.7
Trotwood	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd

Table 8b. Additional soil chemistry parameters for natural and mitigation wetlands (n = 1) and exchangeable ions (%K, %Ca, %Mg, %H) (n = 5). Data on TOC determined by U.S. EPA 415.1.

site	TOC (%)	%solids	Ammonia mg/kg	exchangeable K (%)	exchangeable Ca (%)	exchangeable Mg (%)	particle size >50um (%)	particle size 2-50 um (%)	particle size <2 um (%)
Natural sites									
Baker Swamp	5	58.6	110	4.1 + 1.9	43.0 + 9.1	15.0 + 2.0	23.7	43	11.2
Ballfield	9	48	22	0.9 + 0.2	69.3 + 4.0	19.4 + 2.4	59.2	37	1.8
Calamus	39	15.4	120	1.0 + 0.3	59.2 + 0.8	24.6 + 3.3	61.3	22.1	0
Dever	12	40	27	1.3 + 0.1	53.0 + 3.9	21.5 + 2.6	58	27	3.6
Eagle Cr Beaver	42	16.2	110	0.6 + 0.1	49.9 + 3.0	10.7 + 0.6	83.6	8.2	0
Eagle Cr Marsh	3	69.7	93	1.5 + 0.1	43.0 + 1.7	10.2 + 0.3	52.3	37.5	2.8
Lake Abrams	13	46.6	16	0.8 + 0.2	57.9 + 5.3	12.5 + 1.3	33.1	46.4	5.8
Lodi North	20	37.9	57	0.4 + 0.1	64.5 + 4.6	13.2 + 0.3	60.7	28.9	4.2
Rickenbacker	11	50.1	7	2.5 + 0.2	51.8 + 2.1	16.5 + 1.1	52.5	33.2	4.2
Mitigation sites									
Big Island Area D	2	73.8	20	3.0 + 0.3	57.6 + 2.6	21.5 + 1.0	29.8	43.7	9.5
Bluebird	1	79.5	35	2.6 + 0.2	55.2 + 2.0	25.5 + 1.7	52.7	30.4	5.7
JMB	2	71.8	5	1.9 + 0.1	75.9 + 1.4	21.3 + 1.0	31.3	48.8	6.6
Medallion No. 20	1	70.6	15	2.5 + 0.2	53.2 + 3.0	21.3 + 2.1	33.5	41.2	4
New Albany HS	2	66.7	32	2.4 + 0.3	79.1 + 1.9	18.5 + 1.7	41	44.8	4.1
Pizzutti	1	74.3	24	2.2 + 0.2	63.6 + 1.7	20.9 + 1.1	48.9	40.2	5.5
Prairie Lane	2	69.2	21	2.1 + 0.3	62.1 + 2.0	15.7 + 0.6	31.2	47	10.2
Sacks	2	70.8	30	2.5 + 0.3	56.1 + 1.1	19.4 + 0.3	37.6	46.4	5.4
Slate Run Bank SE	1	82.2	5	2.4 + 0.3	63.7 + 3.5	20.3 + 0.9	32.6	48.8	5.7
Trotwood	2	76	19	nd	nd	nd	58.5	27.7	1.3

Table 9. Summary of mean soil parameters of natural and mitigation wetlands. Values are average of 9 natural sites and 10 mitigation sites listed in Table 8a and 8b. p-values are results of two sample t-test. SE = standard error of the mean. ns = nonsignificant, i.e. p > 0.1

parameter	units	mean (SE) natural sites	mean (SE) mitigation sites	p
pH	na	5.58 $\pm$ 0.10	6.19 $\pm$ 0.10	< 0.0001
Organic Matter (Walkly-Black)	%	7.03 $\pm$ 0.33	2.76 $\pm$ 0.21	< 0.0001
Organic Carbon (CHN analyzer)	%	15.1 $\pm$ 1.6	2.45 $\pm$ 0.16	< 0.0001
TOC (EPA Method 415.1)	%	17.1 $\pm$ 4.7	1.60 $\pm$ 0.16	0.011
%solids	%	42.5 $\pm$ 6.0	73.5 $\pm$ 1.5	0.0007
Nitrogen, total	%	1.12 $\pm$ 0.13	0.24 $\pm$ 0.01	< 0.0001
Ammonia	ug/g	62.4 $\pm$ 15.0	20.6 $\pm$ 3.3	0.028
P, weak (Bray 1)	ug/g	11.96 $\pm$ 0.89	7.38 $\pm$ 1.20	0.003
P, strong (Bray 2)	ug/g	43.4 $\pm$ 4.3	30.0 $\pm$ 3.7	0.021
P, total	ug/g	1156 $\pm$ 252	669 $\pm$ 91	0.099
K	ug/g	94.5 $\pm$ 13.0	112.5 $\pm$ 5.7	ns
Mg	ug/g	296.4 $\pm$ 12.0	350.0 $\pm$ 27.0	0.074
Ca	ug/g	1989 $\pm$ 109	1533 $\pm$ 74	0.0009
Na	ug/g	53 $\pm$ 22	3.3 $\pm$ 3.3	0.027
CEC	na	18.0 $\pm$ 0.7	12.1 $\pm$ 0.4	< 0.0001
exchangeable K	%	1.47 $\pm$ 0.25	2.40 $\pm$ 0.09	0.0011
exchangeable Mg	%	15.97 $\pm$ 0.90	20.50 $\pm$ 0.54	0.0001
exchangeable Ca	%	54.6 $\pm$ 1.9	62.9 $\pm$ 1.5	0.0007
particle size >50µm	%	53.8 $\pm$ 3.2	39.7 $\pm$ 5.7	0.054
particle size 2µm - 50µm	%	31.5 $\pm$ 3.9	41.9 $\pm$ 2.3	0.038
particle size <2µm	%	3.7 $\pm$ 1.1	5.8 $\pm$ 0.8	ns

Table 10. Comparison of median (25^{th} - 75^{th} percentile) soil chemistry of wetlands (n ~ 250 soil samples depending on parameter) from Ohio EPA reference wetland data set, 1996-2003. Impoundments included in riverine mainstem or riverine headwater depending on their landscape position. Notable values underlined. Note similarities in soil chemistry of slope systems, coastal marshes, and Lake Plains sand prairies.

Description	HGM class	%solids	TOC%	Al g/kg	Ca g/kg	Fe g/kg	Mg g/kg	K g/kg	NH4 mg/kg	P total g/kg
Bogs	bog	18.9 (15-27)	<u>41.7(25-43)</u>	<u>5.9(3-51)</u>	4.7(4-12)	<u>5.2(4-11)</u>	2.1(0.8-4.1)	2.6(2-5)	18.5(8-128)	1070(383-1840)
Depressional marshes	depression	62.4(47-68)	3.6(3-7)	27.2(12-36)	3.7(5-13)	17.1(11-21)	5.1(4-6)	5.4(2-6)	39.0(22-60)	1270(784-1925)
Mainstem marshes	riverine mainstem	46.6(40-59)	3.9(2-8)	37.2(23-51)	3.2(2-9)	28.4(16-36)	3.3(3-6)	6.1(4-10)	54.0(26-91)	790(640-1135)
Headwater marshes	riverine headwater	46.6(41-59)	4.1(2-15)	30.7(18-60)	3.1(1-6)	17.9(13-31)	2.5(2-6)	4.9(3-11)	77.5(22-118)	673(436-1142)
Mitigation marshes	mostly impoundments	<u>73.3(70-76)</u>	<u>2.0(1-2)</u>	33.8(26-41)	3.0(2-5)	25.7(20-31)	4.2(3-5)	7.2(5-9)	24.0(8-40)	592(475-802)
Coastal marshes	Lake Erie Coastal	54.9(42-62)	4.1(3-6)	32.6(25-43)	<u>12.8(7-32)</u>	21.6(19-29)	5.9(5-9)	8.6(5-11)	30.0(23-65)	826(556-1315)
Calcareous fens	slope	30.0(24-41)	<u>24.5(7-27)</u>	<u>3.9(0.6-6.9)</u>	<u>45.6(31-140)</u>	<u>7.5(1-24)</u>	5.1(2-7)	<u>14.6(10-25)</u>	52.0(44-210)	618(346-1340)
Fen meadows	slope	48.0(13-57)	<u>9.0(6-34)</u>	13.5(6-31)	<u>3.9(3-29)</u>	13.9(10-19)	2.7(2-4)	2.9(2.6-3.2)	22.0(16-98)	971(957-1260)
Forest seeps	slope	38.0(31-43)	<u>9.3(6-20)</u>	40.0(13-49)	<u>6.6(3-14)</u>	27.2(13-44)	3.4(2-5)	5.0(2-9)	21.3(8-63)	784(568-1054)
Wet meadows	all except slope	55.9.0(---)	3.1(---)	43.9(---)	4.3(---)	37.6(---)	4.3(---)	7.0(---)	26.0(---)	908(---)
Lake Plains Sand Prairies	mostly depressions	61.0(39-71)	<u>9.5(5-14)</u>	11.4(8-19)	<u>7.8(4-28)</u>	<u>9.5(7-14)</u>	1.6(1-3)	<u>0.76(0.51-1.6)</u>	34.0(21-63)	<u>337(202-555)</u>
Forest vernal pools	depression	58.1(38-68)	5.4(2.9-11.1)	30.0(9-41)	6.5(4-13)	12.8(8-34)	3.7(3-6)	5.1(1-7)	49.0(27-95)	1060(593-2180)
Shrub vernal pools	depression	41.9(32-54)	10.1(6-16)	37.3(11-51)	7.1(4-10)	13.6(6-19)	3.7(2-6)	3.5(1-6)	57.2(29-101)	1490(741-1955)
Wet woods	depression	57.4(48-71)	6.6(4-9)	27.2(9-46	4.8(4-12)	15.0(6-20)	2.6(1-6)	3.2(1-12)	39.5(7-39)	841(401-1215)
Mainstem swamp forests	riverine mainstem	66.1(59-70)	2.9(2-4)	36.5(20-49)	3.2(0.8-7.5)	24.8(3-30)	3.3(3-5)	6.5(4-10)	36.0(21-67)	626(511-988)
Mainstem shrub swamps	riverine mainstem	62.1(52-70)	3.7(3-5)	36.0(18-46)	3.0(0.6-6.7)	20.8(8-28)	4.2(3-9)	5.8(3-8)	56.6(21-79)	904(632-1460)

Table 11. Vegetation community attributes of natural and mitigation wetlands including species richness, Vegetation Index of Biotic Integrity scores (VIBI), Floristic Quality Assessment Index scores (FQAI), and biomass production (mean of 8 samples per site).

Wetland	Type	Species	VIBI	FQAI	Biomass
Natural sites					
Baker	N	31	71	24.7	23.7
Ballfield	N	61	73	31	na
Beaver Pond	N	25	82	25.0	31.9
Calamus	N	24	57	21.5	39.4
Dever	N	26	9	15.8	44.2
Eagle Creek	N	42	81	22.9	52.4
Lake Abrams	N	20	33	17	41.4
Lodi	N	19	45	15.8	33.1
Rickenbacker	N	31	67	17.7	11.7
Mitigation Sites					
Big Island	C	34	50	14.7	30.6
Bluebird	C	33	20	14.4	16.4
JMB	C	29	16	14.5	7.1
Medallion	C	39	46	18.1	5.2
New Albany	C	23	16	12.0	58.6
Pizzutti	C	17	30	14.7	21.2
Prairie Lane	C	28	43	15.0	26.5
Sacks	C	13	19	9.8	20.9
Slate Run	C	27	16	15.4	na
Trotwood	C	13	16	13.6	1.9

Table 12. Summary of mean vegetation attributes of natural and mitigation wetlands. Values recoded are an average of the 9 natural sites and 10 mitigation sites cited above (Table 6a). Error recorded is the standard error. (*Ballfield and Slate Run omitted as a statistical outliers).

Vegetation attributes	Natural	Mitigation	P-value
Species richness	31.0 ± 4.4	25.6± 2.8	0.22
VIBI score	57.6 ± 8.1	27.2 ± 4.6	0.005
FQAI score	21.6 ± 1.7	14.2± 0.7	0.004
Biomass Production (g/0.1m^2)	34.7 ± 3.5	20.9 ± 5.7*	0.04

Table 13. Mean amount of ON-SITE litter lost for each wetland at all collection periods. Values recorded are an average of 5 stations (n = 6 natural and n =9 created wetlands). K values are calculated based on the average mass lost for each wetland.

Wetland	Mass lost (g) 1st (g lost·day^{-1})	Mass lost (g) 2nd (g lost·day^{-1})	Mass lost (g) 3rd (g lost·day^{-1})	k-value 1st	k-value 2nd	k-value 3rd
Natural sites						
Baker Swamp	5.42 (0.143)	6.06 (0.068)	6.13 (0.017)	0.0205	0.0103	0.00276
Ballfield	4.24 (0.118)	6.64 (0.076)	5.96 (0.019)	0.0153	0.0125	0.00295
Calamus	4.92 (0.129)	6.00 (0.067)	6.56 (0.019)	0.0178	0.0103	0.00315
Dever	3.46 (0.105)	5.12 (0.06)	6.40 (0.022)	0.0129	0.0084	0.00345
Lodi	6.30 (0.154)	7.10 (0.082)	6.83 (0.019)	0.0243	0.0142	0.00382
Rickenbacker	2.84 (0.079)	4.24 (0.048)	5.68 (0.019)	0.0093	0.0062	0.00281
Mitigation sites						
Big Island Area D	2.94 (0.072)	2.04 (0.023)	3.38 (0.011)	0.0085	0.0026	0.00137
Bluebird	5.36 (0.141)	5.76 (0.065)	na	0.0202	0.0096	na
JMB	3.18 (0.088)	4.38 (0.053)	5.62 (0.019)	0.0106	0.0069	0.00280
Medallion No. 20	4.82 (0.134)	5.00 (0.061)	6.38 (0.021)	0.0183	0.0085	0.00341
New Albany HS	4.90 (0.129)	6.22 (0.070)	6.30 (0.021)	0.0177	0.0109	0.00328
Pizzutti	5.12 (0.155)	4.92 (0.058)	5.10 (0.017)	0.0217	0.0080	0.00243
Prairie Lane	2.68 (0.065)	2.40 (0.028)	3.67 (0.012)	0.0076	0.0032	0.00152
Sacks	2.60 (0.072)	4.80 (0.055)	na	0.0084	0.0075	na
Slate Run Bank SE	3.72 (0.098)	4.90 (0.050)	5.34 (0.018)	0.0122	0.0076	0.00254

Table 14. Summary of both the mean amount of on-site litter lost and the k-value of natural and mitigation wetlands. Values recorded are an average of the 6 natural sites and 9 created sites cited above (mean ± standard error).

	Natural	Created	P-value
Mass lost (g) 1st	4.53± 0.52	3.92± 0.37	0.07
Mass lost (g) 2nd	5.86± 0.42	4.49± 0.47	0.05
Mass lost (g) 3rd	6.26± 0.17	5.11± 0.48	0.05
Mass lost, g·day^{-1} 1st	0.121± 0.01	0.106±0.01	0.35
Mass lost, g·day^{-1} 2nd	0.067±0.005	0.051±0.005	0.06
Mass lost, g·day^{-1} 3rd	0.019±0.001	0.017±0.001	0.23
k-value 1st	0.0167±0.002	0.0139± 0.001	0.32
k-value 2nd	0.0094± 0.0001	0.0078± 0.001	0.36
k-value 3rd	0.0030± 0.001	0.0025±0.001	0.08

Table 15. Mean amount of CONTROL litter lost for each wetland at all collection periods. Values recorded are an average of 3 stations. Values recorded are an average of 3 stations except for values marked with * where n = 1 (n = 9 natural and n = 9 mitigation wetlands; 6 natural sites sampled in 2001 and 3 in 2002). K values are calculated based on the average mass lost for each wetland.

Wetland	Mean mass lost (g) 1st	Mean mass lost (g) 2nd	Mean mass lost (g) 3rd	k-value 1st	k-value 2nd	k-value 3rd
Natural						
Baker Swamp	6.07 (0.105)	6.63 (0.061)	7.10 (0.020)	0.0161	0.0101	0.0034
Ballfield	4.73 (0.086)	6.80 (0.064)	7.3 (0.022)	0.0116	0.0107	0.0039
Calamus	7.10 (0.122)	8.00 (0.073)	8.13 (0.023)	0.0213	0.0148	0.00467
Dever	5.30 (0.010)	5.90 (0.056)	7.1 (0.022)	0.0142	0.0085	0.00392
Lodi North	7.13 (0.119)	7.70 (0.072)	7.20 (0.018)	0.0208	0.0137	0.00397
Eagle Cr Beaver	4.80 0.044)*	6.20 0.023)	6.43 0.022)	0.0061	0.0036	0.0035
Eagle Cr Marsh	5.35(0.050)	6.38(0.024)	5.95(0.021)	0.0071	0.0038	0.0031
Lake Abrams	4.70(0.044)	4.20(0.016)*	4.80(0.017)*	0.0059	0.0020	0.0023
Rickenbacker	4.17 (0.072)	5.83 (0.053)	6.00 (0.019)	0.0093	0.0079	0.00285
Mitigation						
Big Island Area D	5.73 (0.094)	7.07 (0.066)	5.90 (0.018)	0.0142	0.0115	0.00278
Prairie Lane	4.20 (0.070)	3.43 (0.032)	3.90 (0.012)	0.0091	0.0040	0.00154
Sacks	3.53 (0.064)	6.57 (0.062)	5.05 (0.015)	0.0079	0.0101	0.00208
JMB	3.97 (0.063)	4.93 (0.044)	5.80 (0.018)	0.0090	0.0061	0.00269
Slate Run Bank SE	4.47 (0.077)	4.07 (0.037)	4.37 (0.014)	0.0102	0.0048	0.00179
Pizzutti	5.47 (0.103)	6.37 (0.061)	6.00 (0.019)	0.0149	0.0097	0.00292
New Albany HS	5.85 (0.113)	6.20 (0.060)	5.45 (0.017)	0.0160	0.0094	0.00248
Bluebird	6.60 (0.127)	6.93 (0.064)	na	0.0196	0.0115	na
Medallion No. 20	6.00 (0.107)	6.05 (0.059)	6.25 (0.020)	0.0164	0.0091	0.00308

Table 16. Summary of both the mean amount of control litter lost and the k-values for natural and Mitigation wetlands. Values recoded are an average of the 6 natural sites and 9 created sites cited above (mean ± standard error).

	Natural	Created	P-value
Mass lost (g) 1st	5.75± 0.32	5.14± 0.26	0.15
Mass lost (g) 2nd	6.81± 0.22	5.93± 0.29	0.02
Mass lost (g) 3rd	7.14 ± 0.28	5.34 ± 0.29	0.001
Mass lost g·day^{-1} 1st	0.100±.008	0.091±0.008	0.39
Mass lost g·day^{-1} 2st	0.063±0.003	0.098±0.043	0.44
Mass lost g·day^{-1} 3st	0.021±0.001	0.017±0.001	0.008
k–value 1st	0.0156± 0.002	0.0130± 0.001	0.32
k-value 2nd	0.0102± 0.001	0.0090± 0.001	0.44
k–value 3rd	0.004 ± 0.0003	0.002 ± 0.0001	0.001

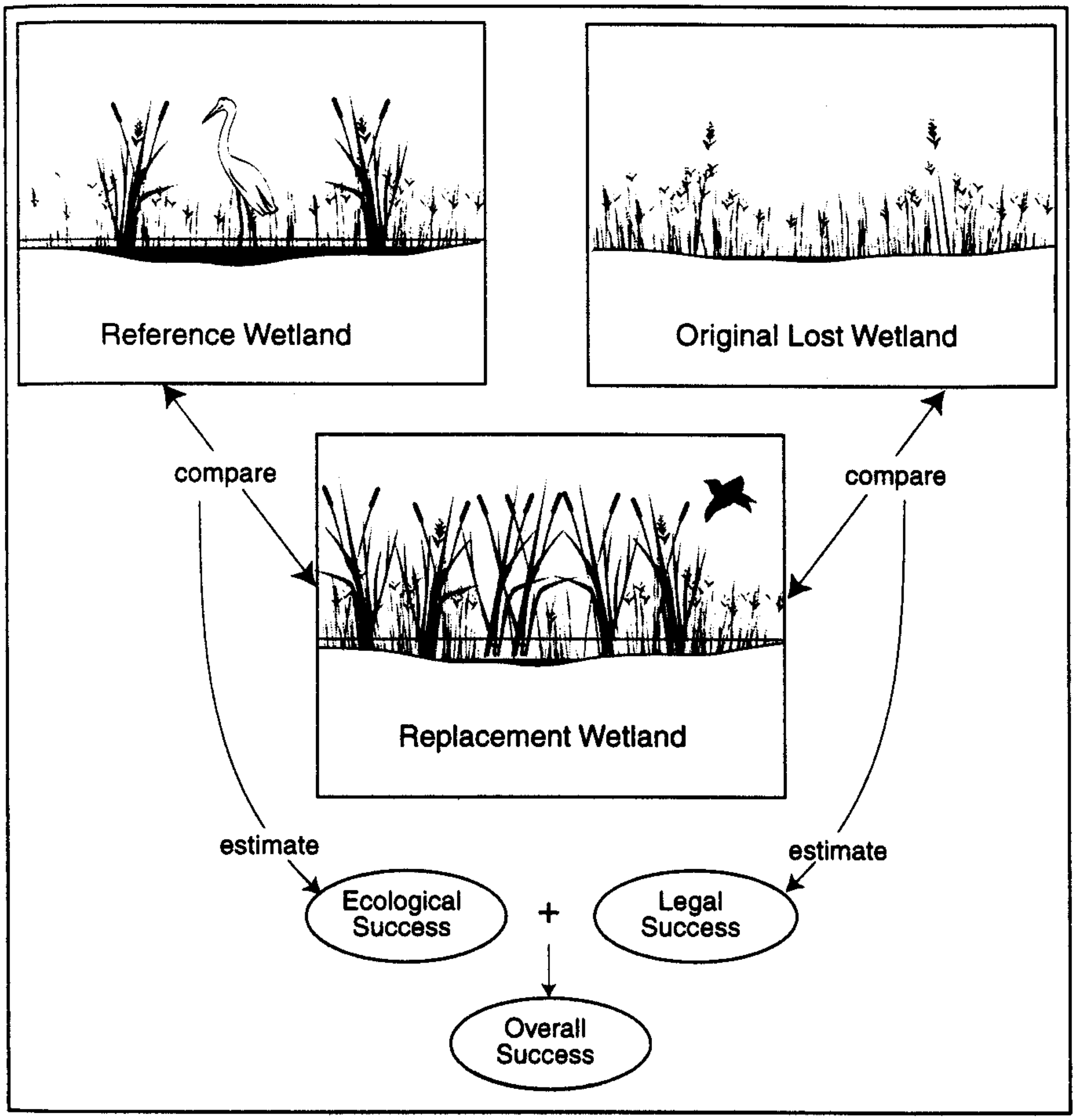

Figure 1. A schematic of the recommended procedure for wetland mitigation. The replacement wetland should not only be compared with the wetland that has been lost (the legal success) but also should be compared to a reference wetland (biological success) (from Mitsch and Gosselink, 2000).

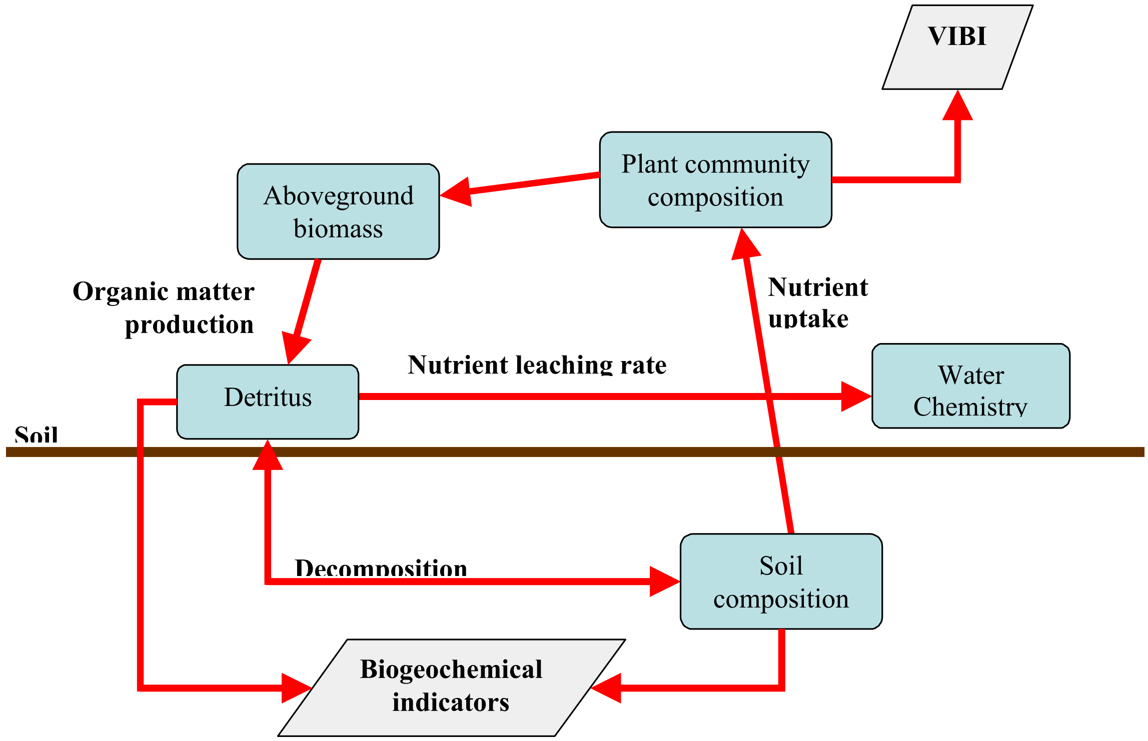

Figure 2. Conceptual model of the ecosystem components included in this study including structural components, processes, and indicators that are empirically derived to indicate ecosystem condition.

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x4              x5      Samples 4 and 5: taken half way between the center and the far edge,
  \            /        at a 45 degree angle from initial line (i.e., line made by samples 1 - 3).
    \        /
       x3               Sample 3: taken and the approximate center of the wetland
       |
       x2               Sample 2:  taken half way between 1 and the approximate center of the wetland
       |
       x1               Sample 1: taken approximately 10 m from wetland boundary, near well
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Figure 3. Sampling scheme used to collect soil samples at all wetlands.

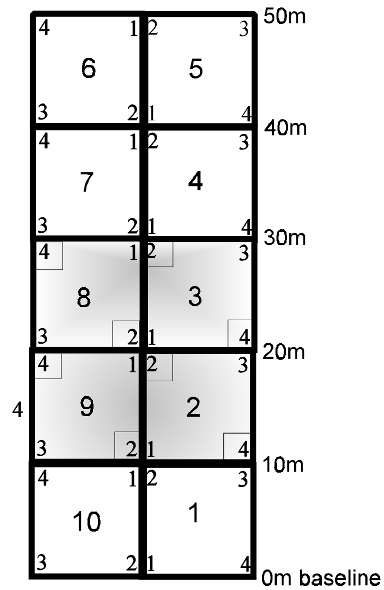


Figure 4. Standard 2 x 5 (20m x 50m) plot with ten modules. Modules are numbered counterclockwise as you move down and back along the long axis of the plot. Module corners are numbered clockwise in direction of movement along the long axis of the plot (down the plot for modules 1 to 5, returning to the baseline for modules 6 to 10). Standard intensive modules are shaded (2, 3, 6, 9). Standard nested quadrat corners of intensive modules are 2 and 4.

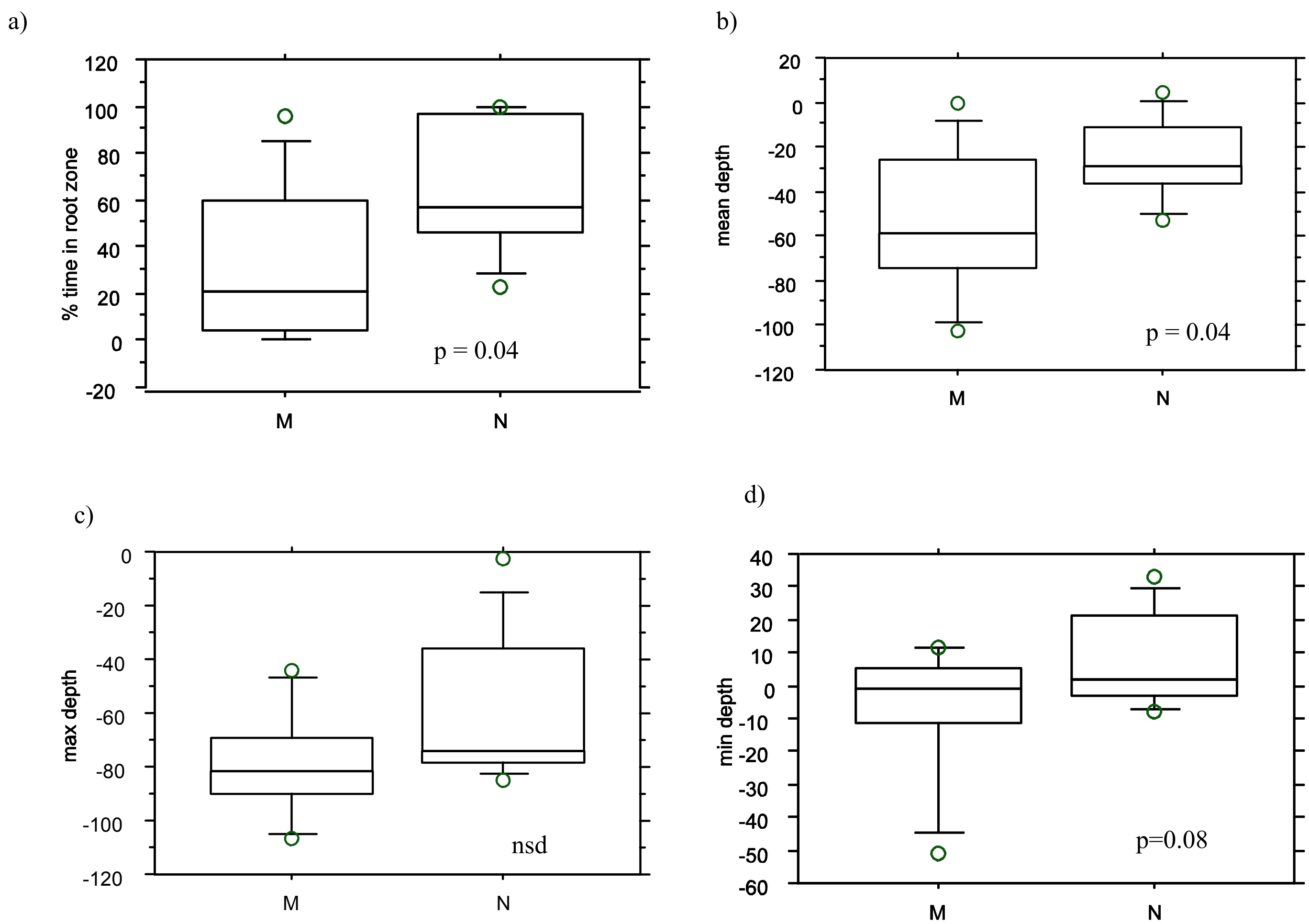


Figure 5a. Box plots for four hydrological parameters for mitigation versus natural wetlands in the 2001 field season. Parameters are a) percent time groundwater was in the root zone (top 30 cm of soil profile); b) mean depth to groundwater (cm); c) mean maximum depth (cm); and d) mean minimum depth to groundwater (cm). Positive values in b – d indicate groundwater levels were above the soil surface. Means were evaluated using unpaired t-tests (p values shown to indicate means that are significantly different).

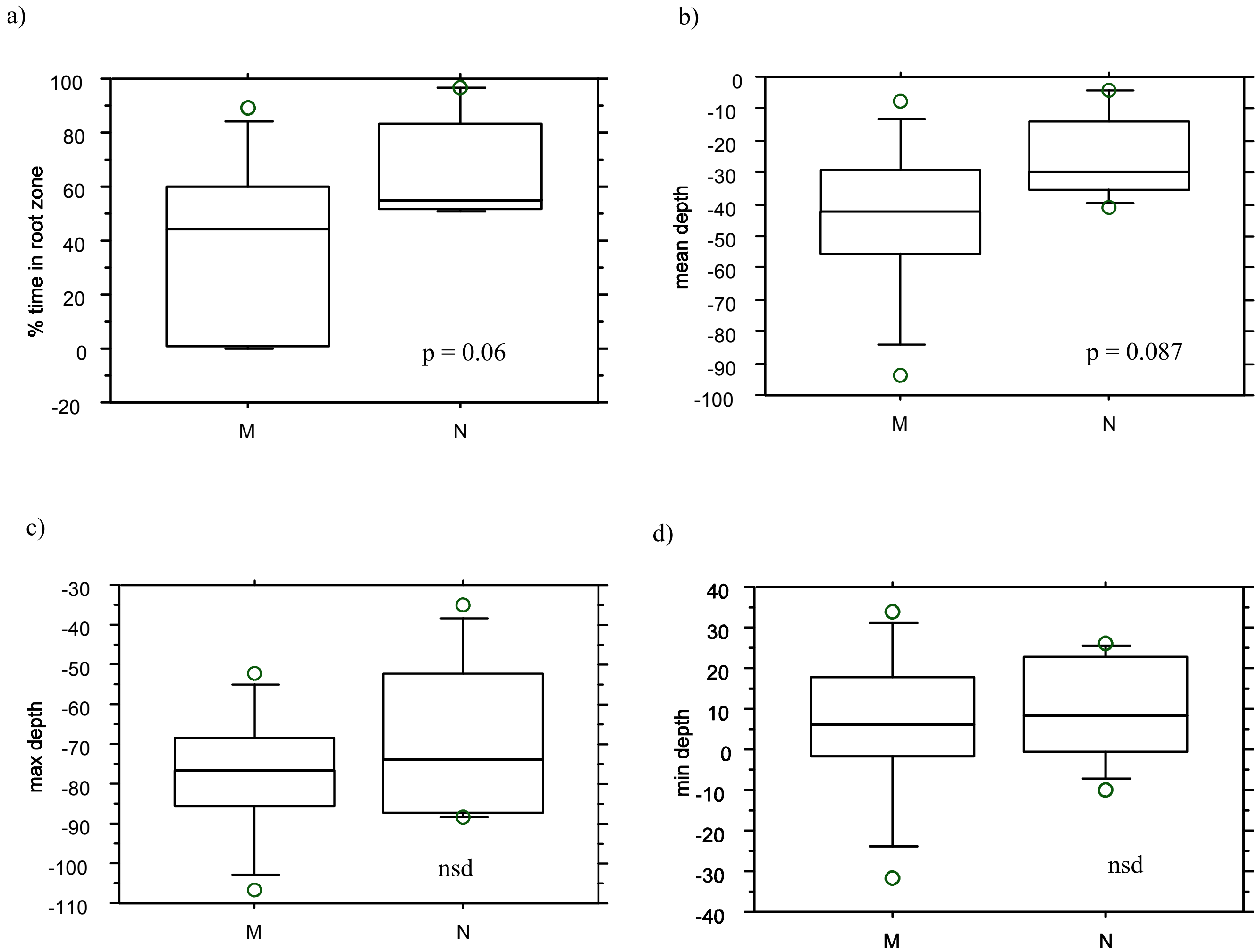


Figure 5b. Box plots for four hydrological parameters for mitigation versus natural wetlands in the 2002 field season. Parameters are a) percent time groundwater was in the root zone (top 30 cm of soil profile); b) mean depth to groundwater (cm); c) mean maximum depth (cm); and d) mean minimum depth to groundwater (cm). Positive values in b – d indicate groundwater levels were above the soil surface. Means were evaluated using unpaired t-tests (p values shown to indicate means that are significantly different). nsd = nonsignificant difference.

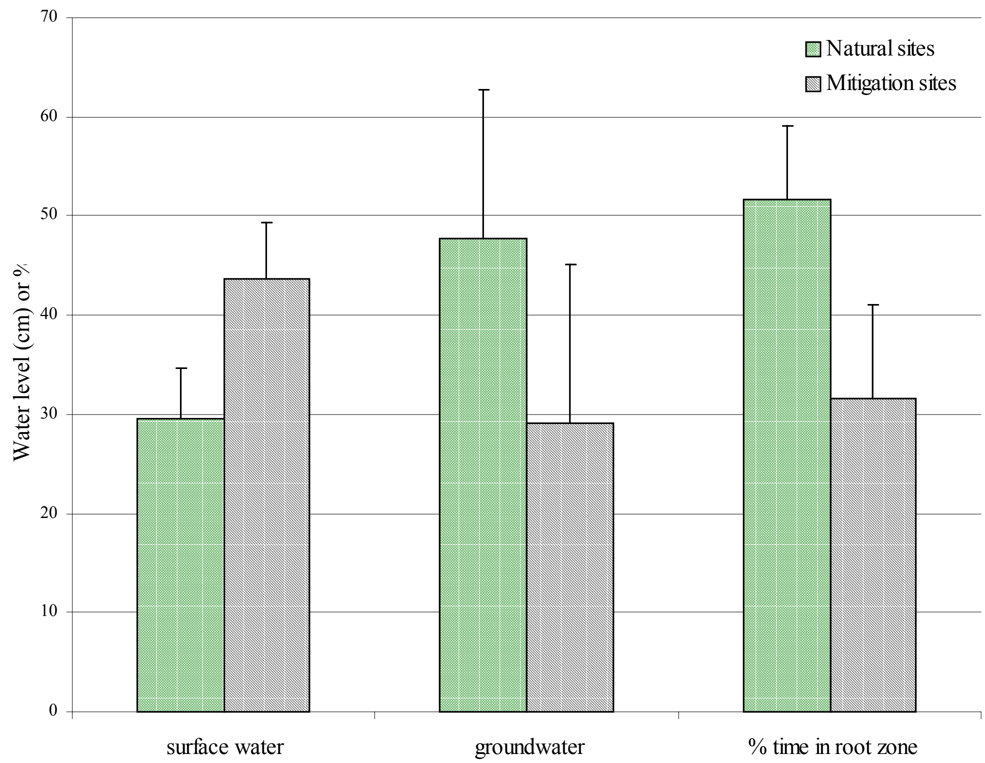


Figure 6. Mean surface water levels, depth to groundwater and the percent time water was found in the root zone for natural and mitigation wetlands in the 2001 growing season (mean ± standard deviation). Mean surface water levels were shallower at natural wetlands, mean depth to groundwater was shallower at natural wetlands, and percent time water was in the root zone was longer at natural wetlands.

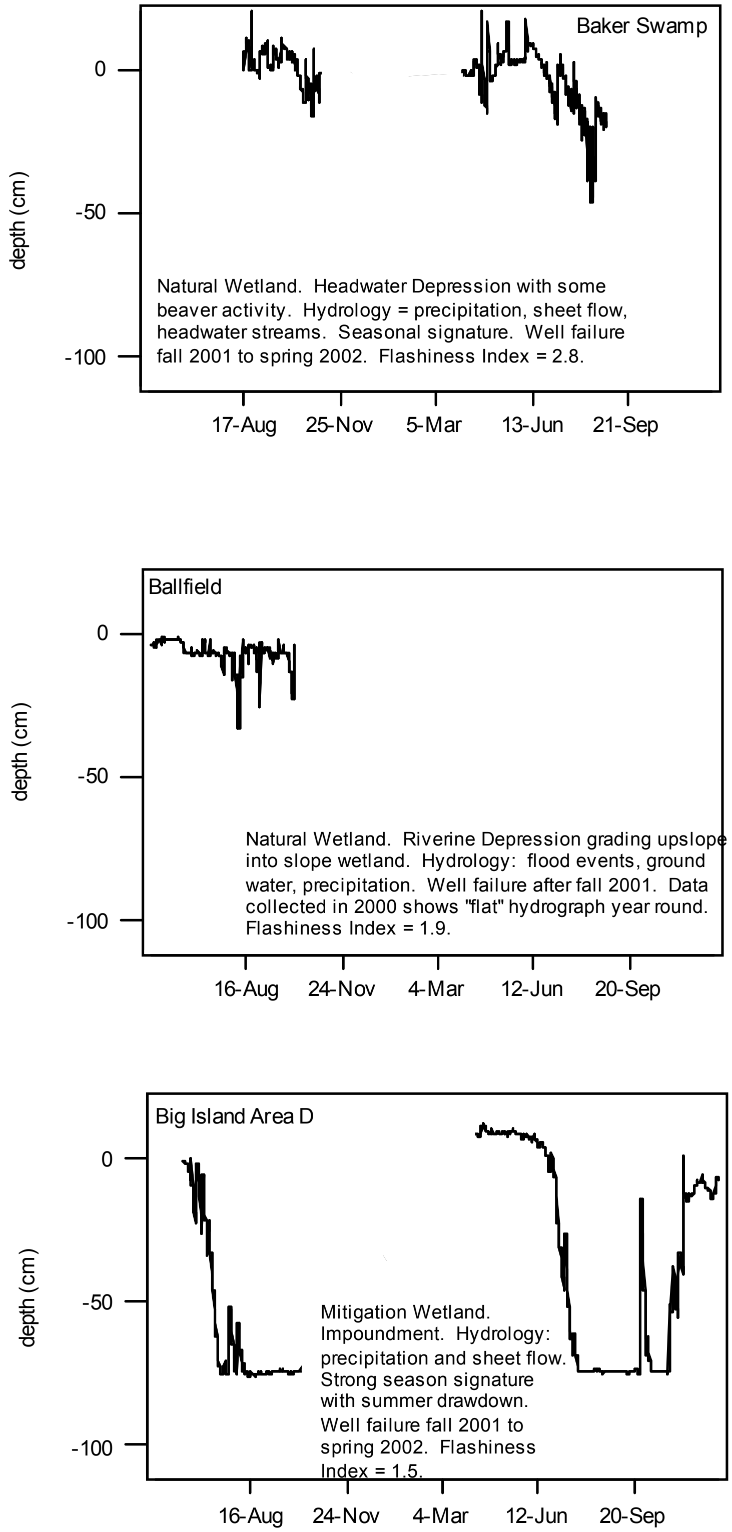


Figure 7. Hydrographs for Baker Swamp (natural), Ballfield (natural), and Big Island Area D (mitigation) sites from June 1, 2001 to October 1, 2002. Gap in hydrograph for Baker Swamp due to vandalization of well. Gaps in hydrographs at Ballfield and Big Island Area D due to failure of well caused by a manufacturing defect.

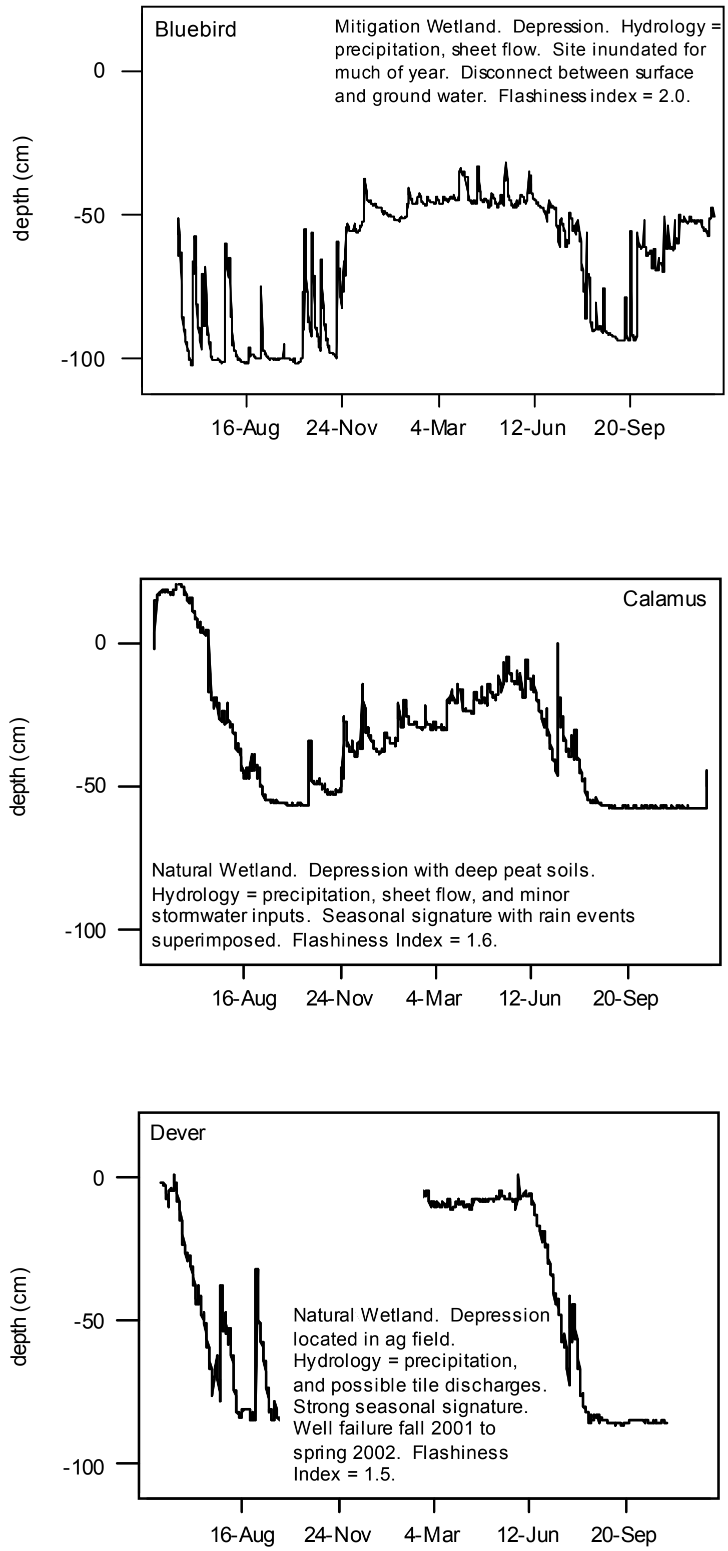


Figure 8. Hydrographs for Bluebird (mitigation), Calamus (natural), and Dever (natural) sites from June 1, 2001 to October 1, 2002. Gap in hydrograph for Dever due to failure of well caused by a manufacturing defect.

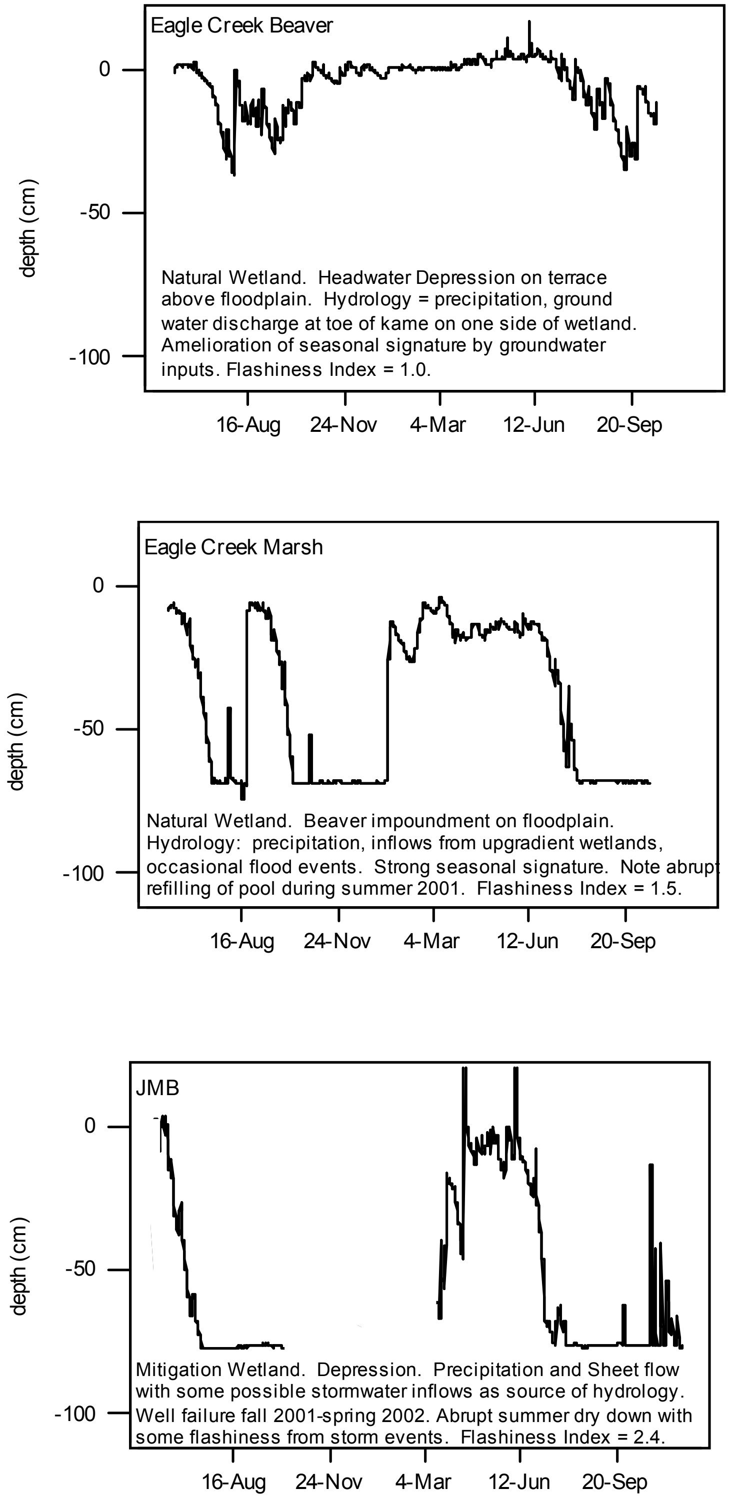
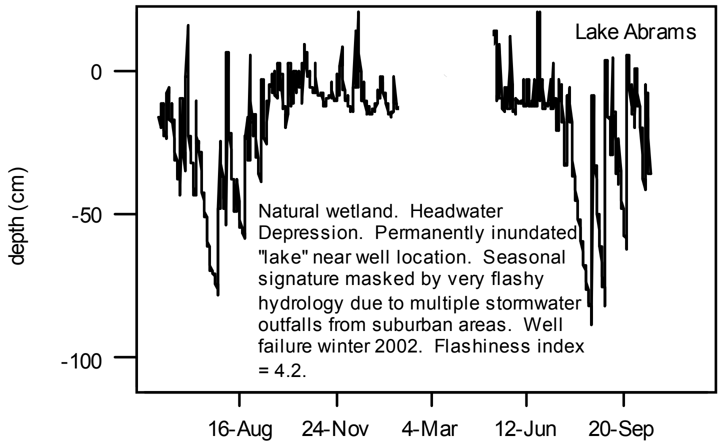
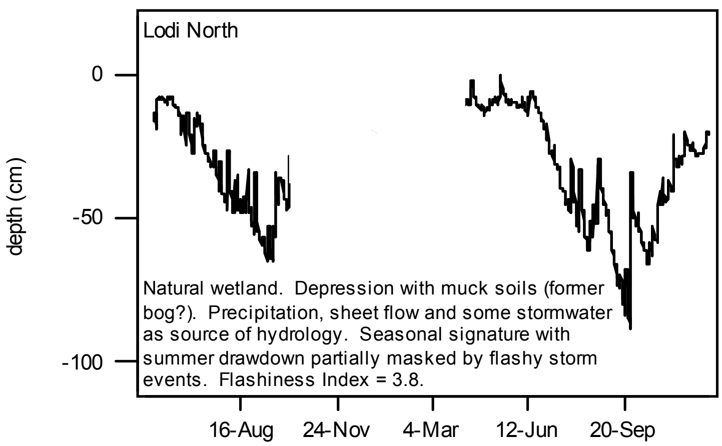


Figure 9. Hydrographs for Eagle Creek Beaver (natural), Eagle Creek Marsh (natural), and JMB (mitigation) sites from June 1, 2001 to October 1, 2002. Gap in hydrograph for JMB due to failure of well caused by a manufacturing defect.





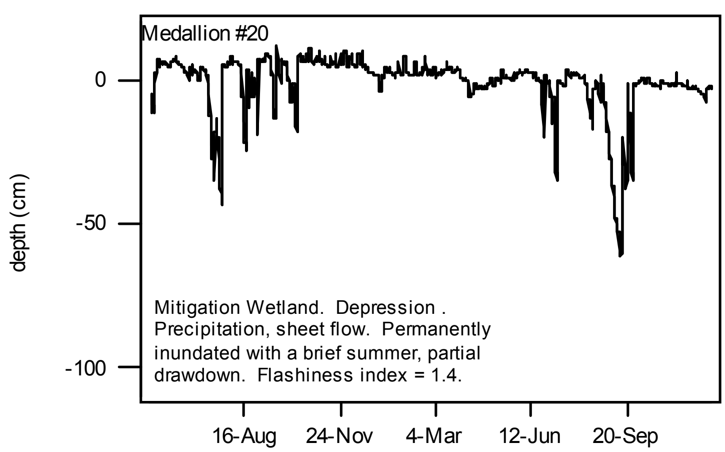


Figure 10. Hydrographs for Lake Abrams (natural), Lodi North (natural), and Medallion #20 (mitigation) sites from June 1, 2001 to October 1, 2002. Gap in hydrograph for Lake Abrams due to a battery failure during extremely cold winter weather. Gap in hydrograph at Lodi North due to failure of well caused by a manufacturing defect.

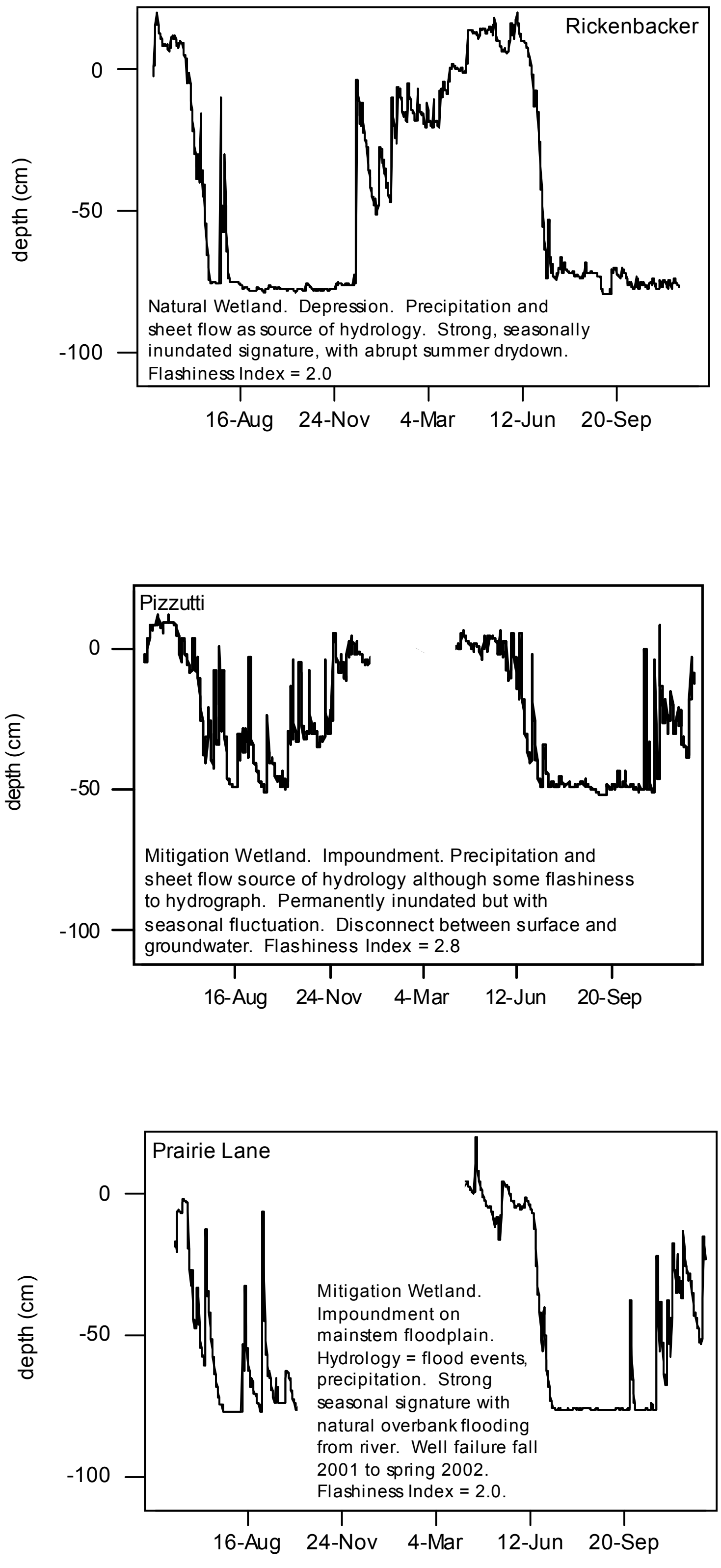
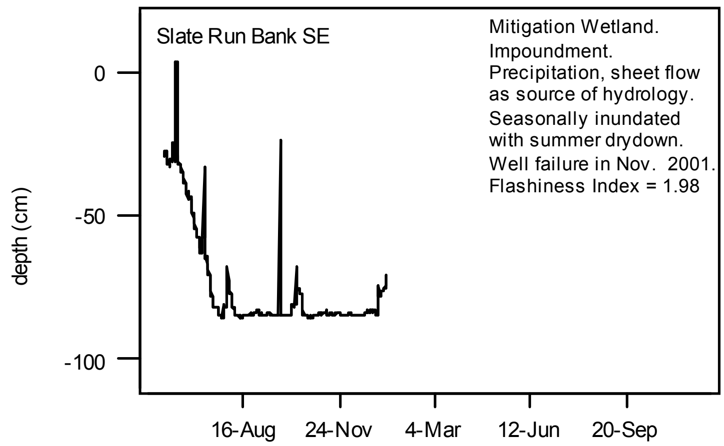


Figure 11. Hydrographs for Rickenbacker (natural), Pizzutti (mitigation), and Prairie Lane (mitigation) sites from June 1, 2001 to October 1, 2002. Gap in hydrograph for Prairie Lane and Pizzutti due to failure of well caused by a manufacturing defect.



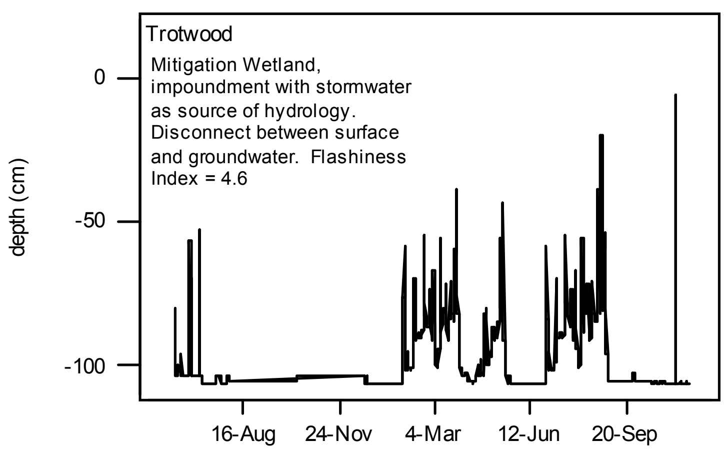


Figure 12. Hydrographs for Slate Run Bank SE (mitigation) and Trotwood (mitigation) sites from June 1, 2001 to October 1, 2002. Gap in hydrograph for Slate Run Bank SE due to failure of well caused by a manufacturing defect.

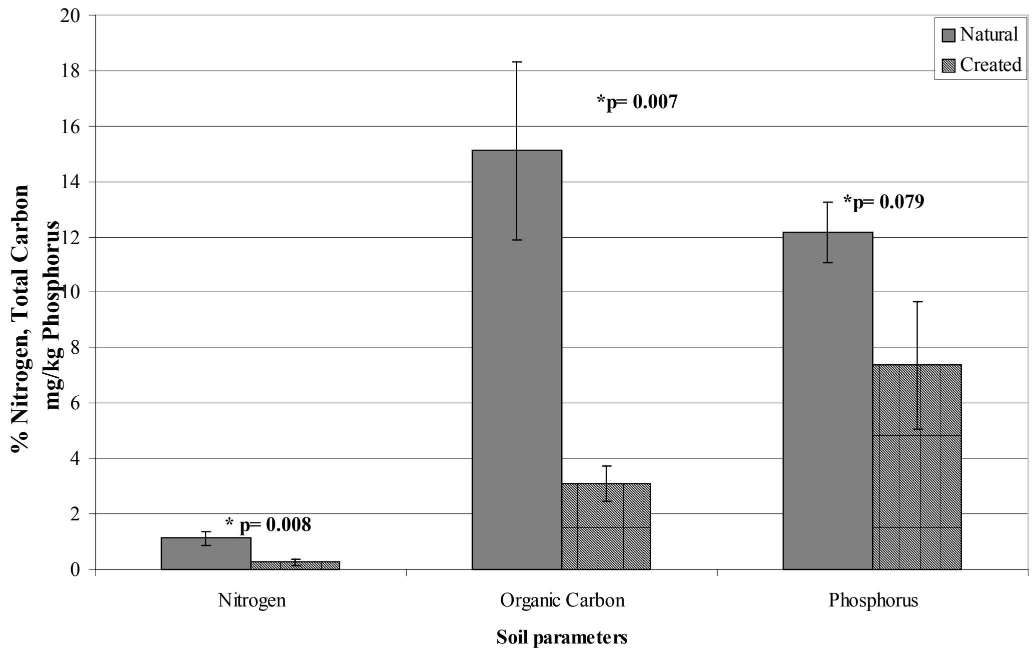


Figure 13. Mean soil parameters for natural and mitigation wetlands. Error bars represent the standard error.

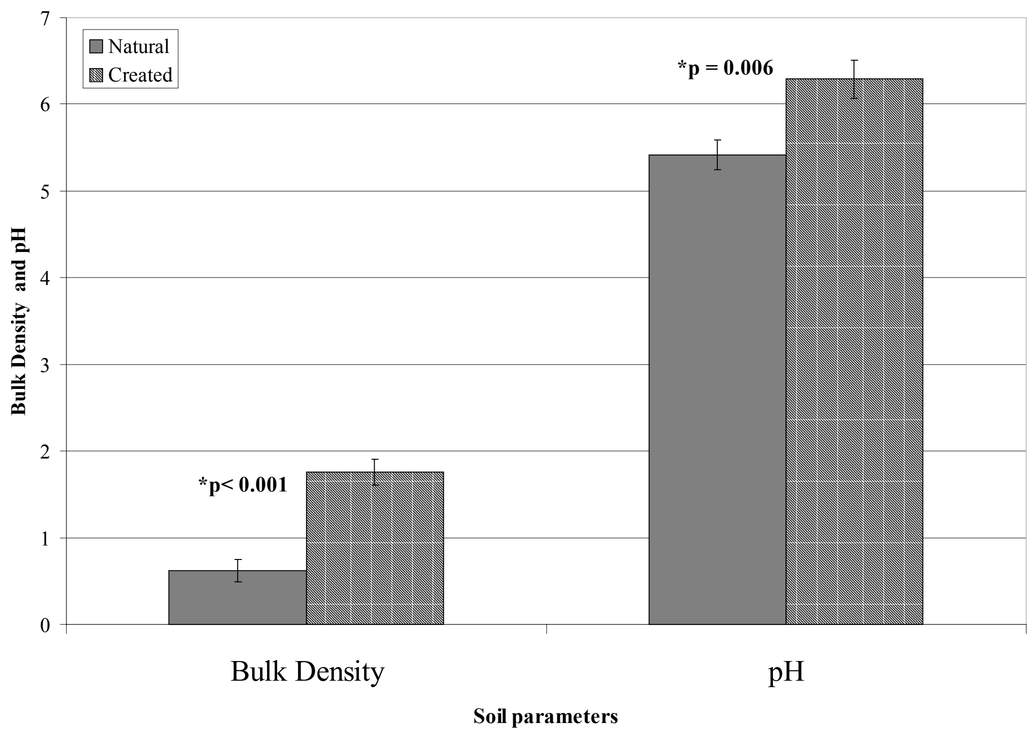


Figure 14. Mean soil bulk density (g cm^{-3}) and soil pH for natural and mitigation wetlands. Error bars represent the standard error.

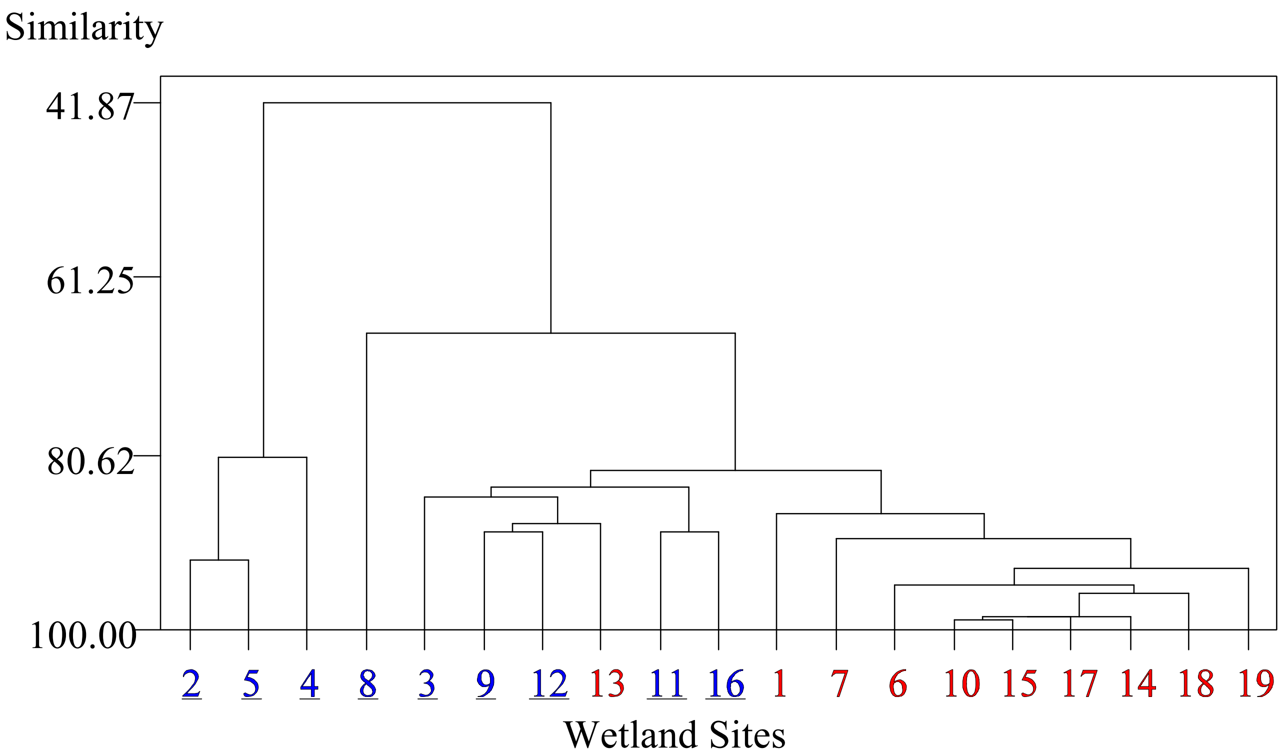


Figure 15. Cluster analysis using soil characteristics (nitrogen and carbon) of wetland ecosystems. Numbers for natural wetlands are underlined (n = 9 natural wetlands and n = 10 mitigation wetlands). Key for wetland sites: 1 = Sacks, 2 = Calamus, 3 = Rickenbacker, 4 = Lodi North, 5 = Eagle Cr Beaver, 6 = Slate Run Bank SE, 7 = Medallion No. 20, 8 = Ballfield, 9 = Dever, 10 = Bluebird, 11 = Baker Swamp 12 = Lake Abrams, 13 = New Albany HS, 14 = Prairie Lane, 15 = Big Island Area D, 16 = Eagle Cr Marsh, 17 = JMB, 18 = Pizzutti, 19 = Trotwood.

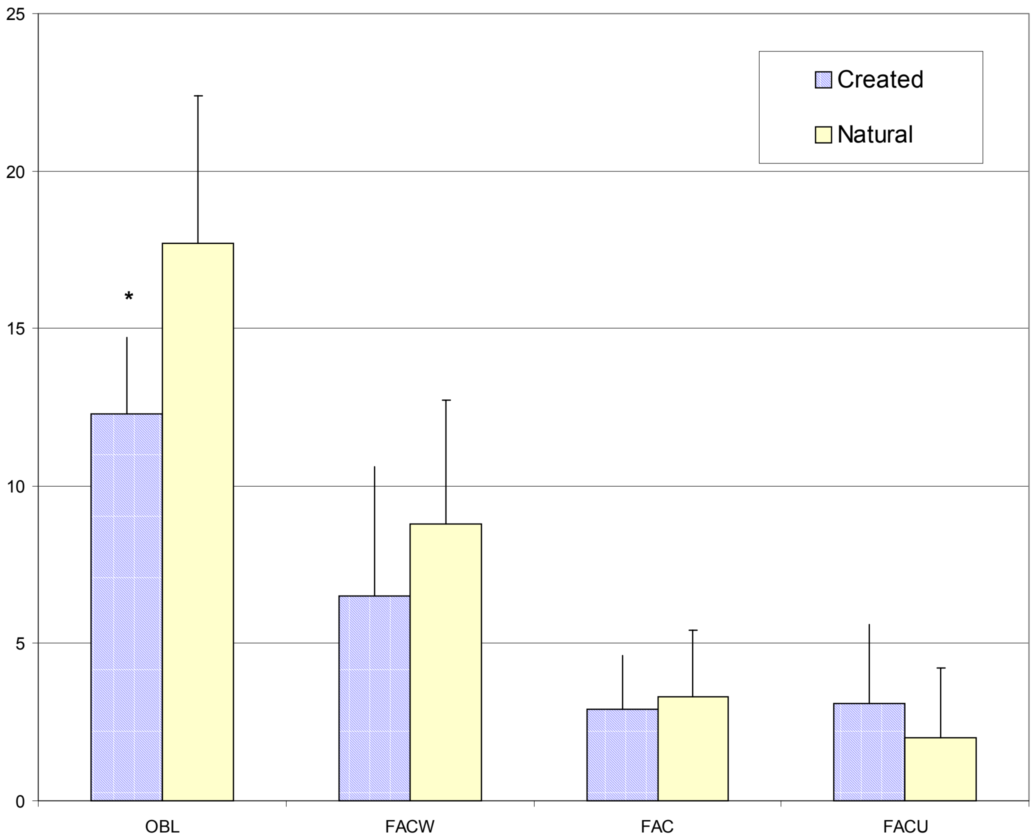


Figure 16. Comparison of the number of obligate (OBL), facultative wet (FACW), facultative (FAC), and facultative upland (FACU) species in the natural and mitigation sites (from Reed 1988). Two sample t-test shows that the number of OBL species is significantly higher in natural sites (p = 0.01), all others nsd).

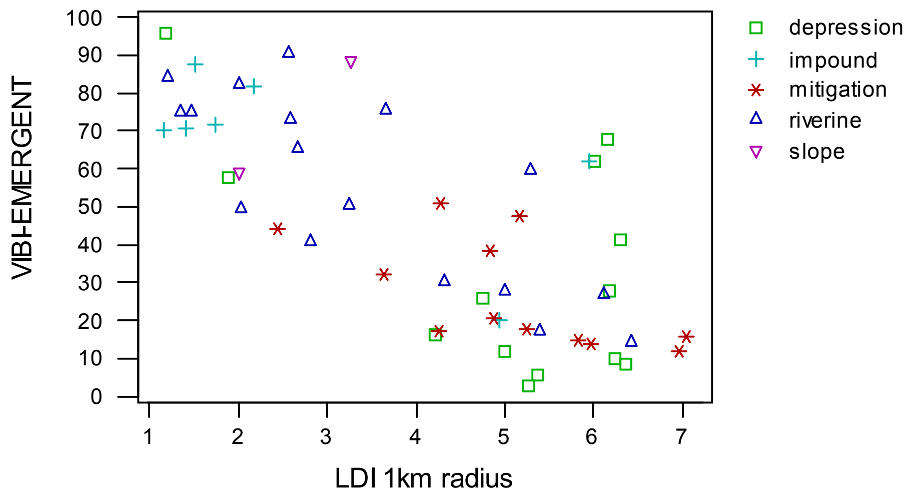


Figure 17. Landscape Development Index scores (LDI) based land use from a 1 km radius circle from a point located in the center of the wetland versus VIBI scores using data from natural emergent marsh wetlands collected from 1996 – 2002 (R^2 = 54.5%, $p < 0.001$).

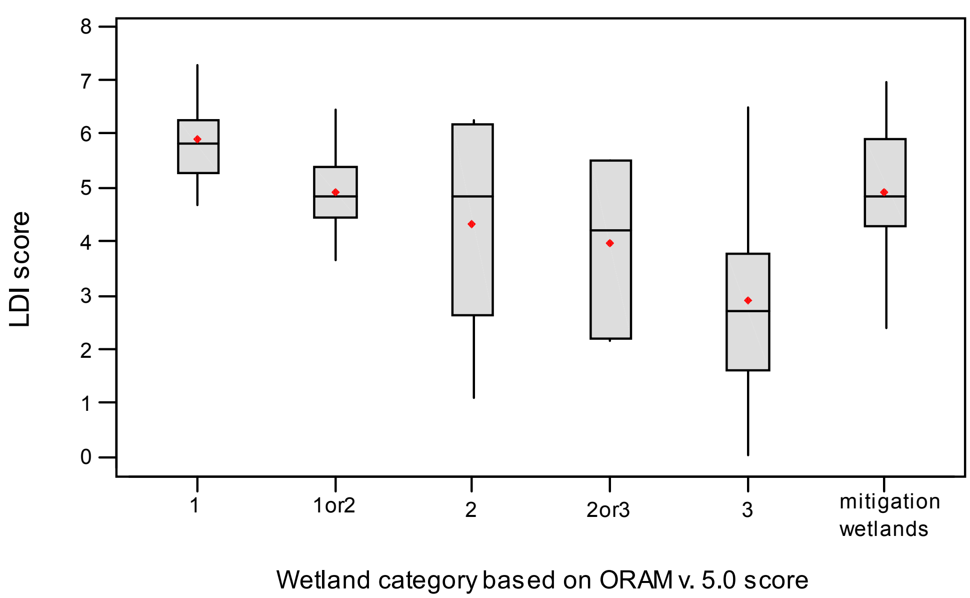


Figure 18. Landscape Development Intensity Index (LDI) score using land use from 1 km radii circle from point in center of the wetland versus wetland regulatory category. Mean LDI scores for Category 1, 2, and 3 wetlands significantly different after analysis of variance followed by Tukey's multiple comparison test ($p < 0.05$). Note that mitigation wetlands tend to be placed in intensively developed landscape positions similar to Category 1 (low quality) natural wetlands. ORAM v. 5.0 scoring categories (Mack 2001a) are Category 1 = 0 - 29.9, Cateogory 1 or 2 = 30.0 - 34.9, Category 2 = 35.0 - 59.9, Category 2 or 3 = 60.0 - 64.9, Category 3 = 65.0 - 100. Category 1 wetlands are low quality with minimal functions, Category 2 wetlands are of moderate quality with moderate functions, and Category 3 wetlands are of high quality with high functions (Ohio Administrative Code Rule 3745-1-54).

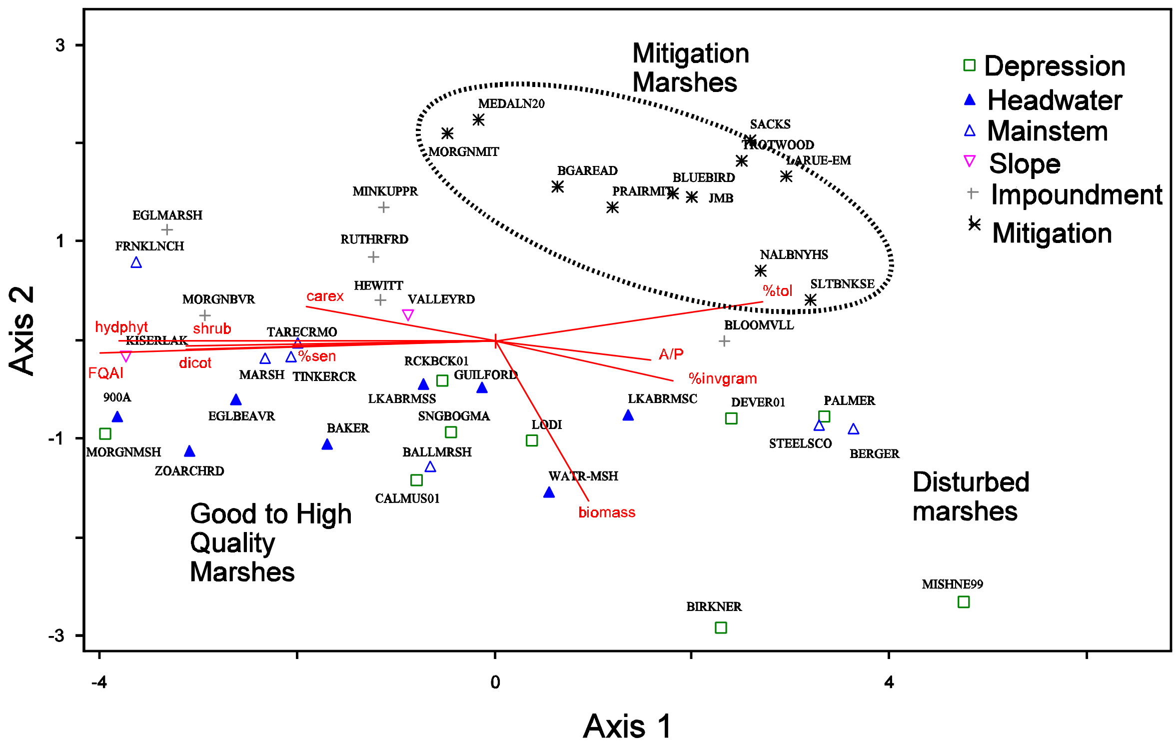


Figure 19. Principal components analysis of VIBI-EMERGENT metrics. Percent variance explained by first three eigenvalues 51.1, 13.8, and 10.0, respectively. Headwater = riverine, headwater; mainstem = riverine, mainstem.

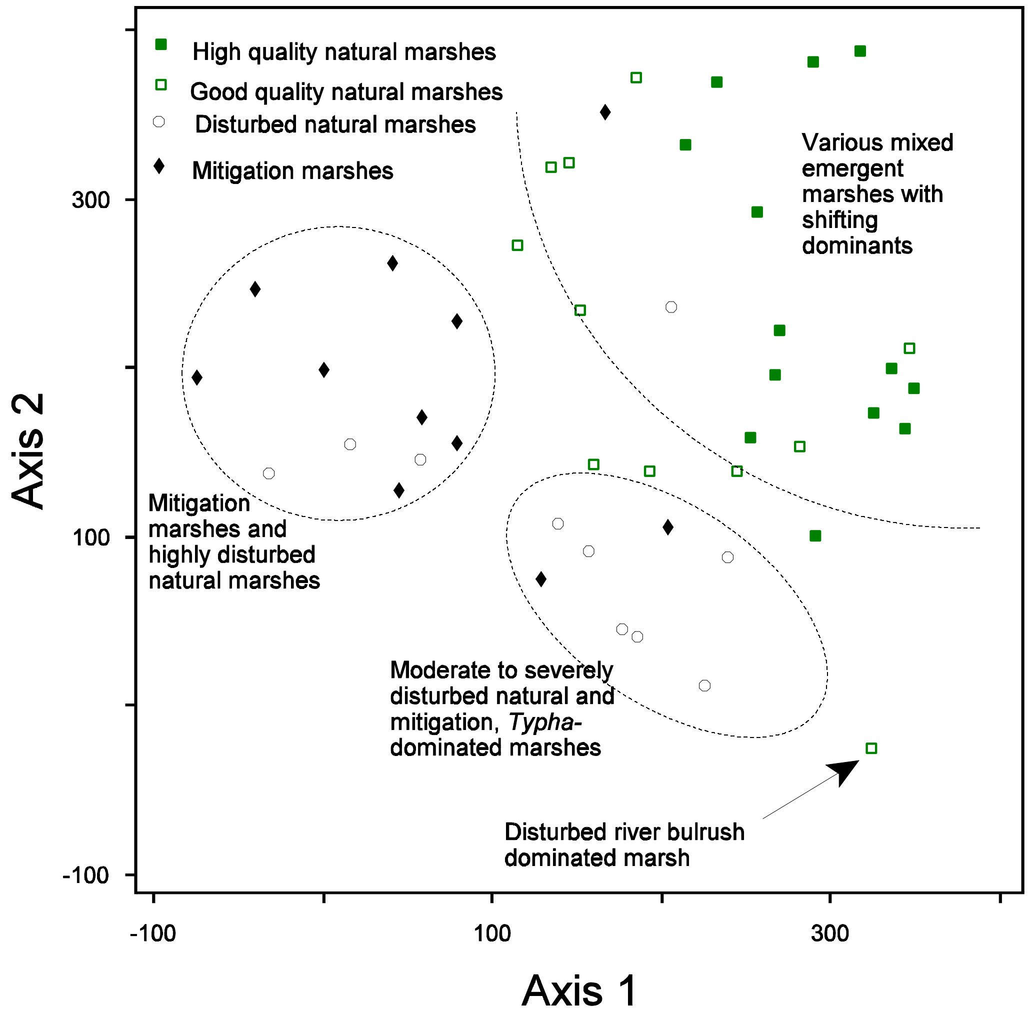


Figure 20. Detrended correspondence analysis of inland marsh (including mitigation marshes) wetland vegetation data from 1999-2002 (n = 47 plots, 213 species). Total inertia (variance) in species data = 10.49; eigenvalues = 0.653, 0.560, 0.409 axes 1, 2, and 3, respectively.

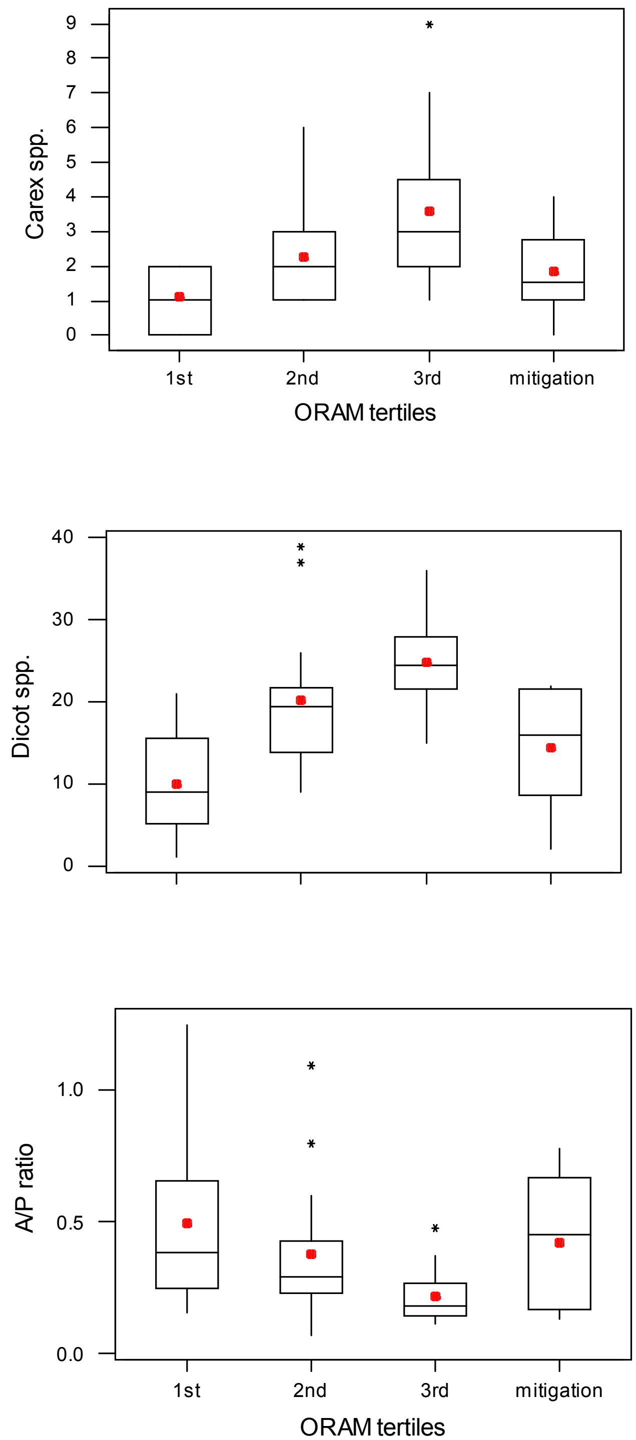


Figure 21. Box and whisker plots of a) *Carex* metric (3^{rd} tertiles significantly different from 1^{st}, $p < 0.05$), b) Dicot metric (3^{rd} tertile significantly different from 1^{st} and mitigation, 2^{nd} significantly different than 1^{st}, $p < 0.05$), and c) A/P metric (3^{rd} tertile significantly different than 1^{st}, $p < 0.05$) of VIBI-Emergent (Mack 2004c).

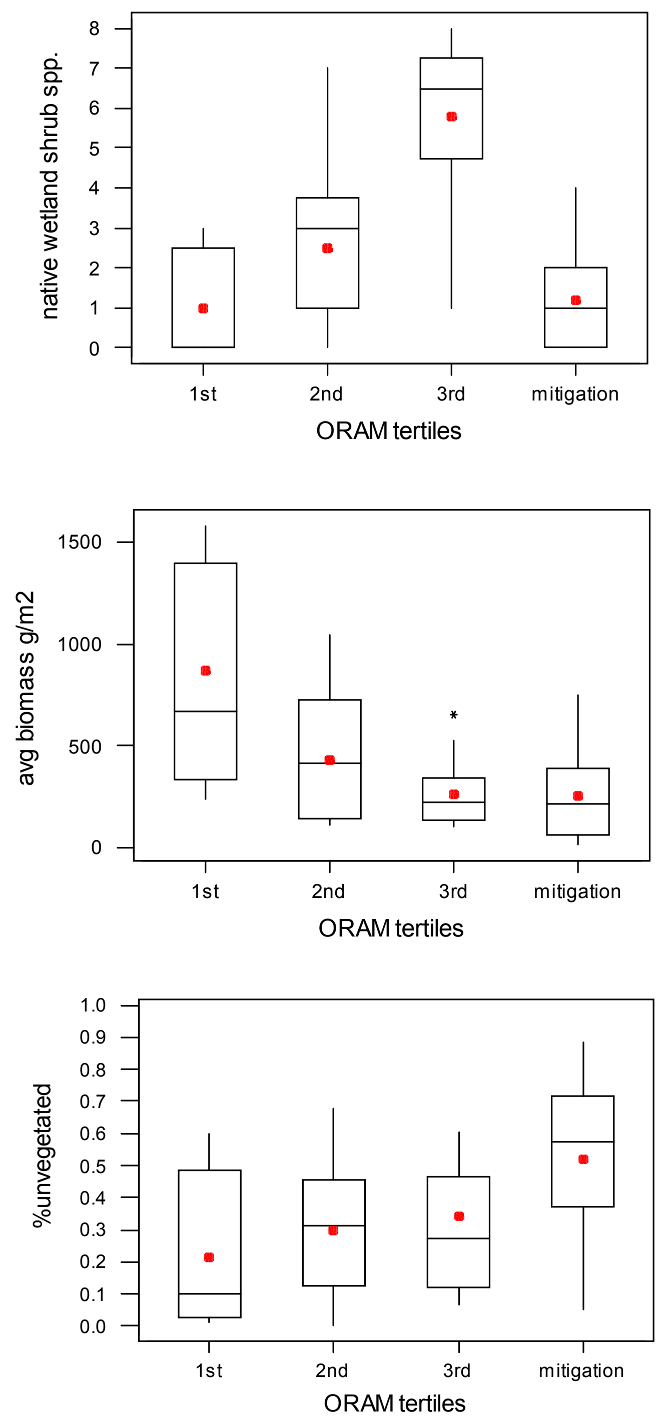


Figure 22. Box and whisker plots of a) shrub metric (3rd tertile significantly different from 1st, 2nd, and mitigation, 2nd significantly different from mitigation, $p < 0.05$), b) average biomass metric (1st tertile significantly different from 3rd and mitigation, and c) %unvegetated metric (mitigation marginally significantly different than natural wetlands, $p < 0.06$) of VIBI-Emergent (Mack 2004c).

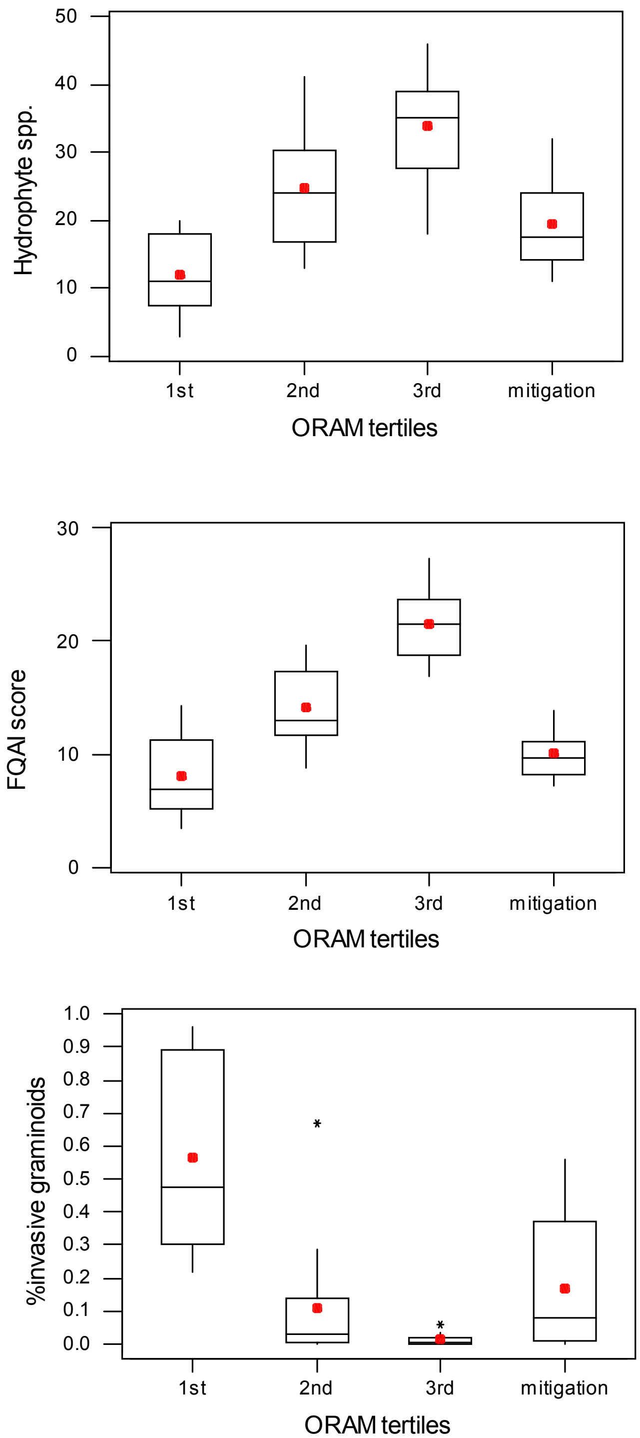
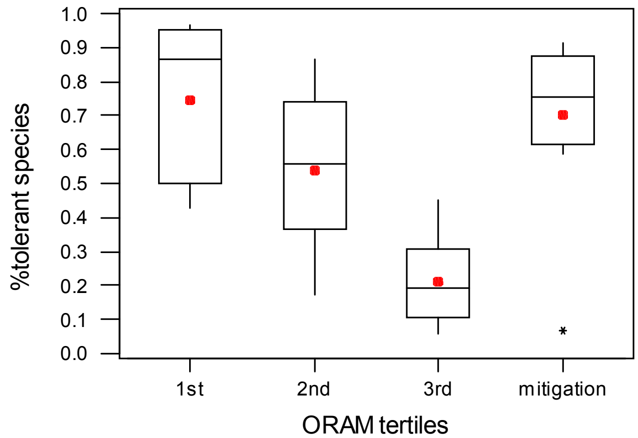


Figure 23. Box and whisker plots of a) hydrophyte metric (3rd tertiles significantly different from 1st , 2nd, and mitigation, 2nd different from 1st, $p < 0.05$), b) FQAI metric (all means significantly different, $p < 0.05$), and c) %invasive graminoids metric (1st tertile significantly different than 2nd, 3rd, and mitigation, $p < 0.05$) of VIBI-Emergent (Mack 2004c).



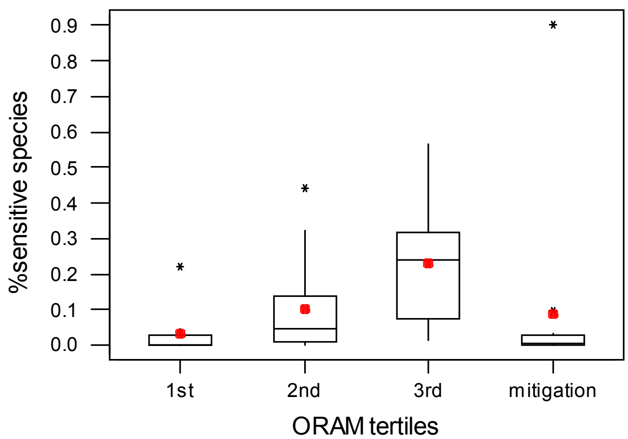


Figure 24. Box and whisker plots of a) %tolerant metric (3rd tertile significantly different from 1st, 2nd, and mitigation, $p < 0.05$), b) %sensitive metric (3rd tertile significantly different from 1st, $p < 0.05$) of VIBI-Emergent (Mack 2004c).

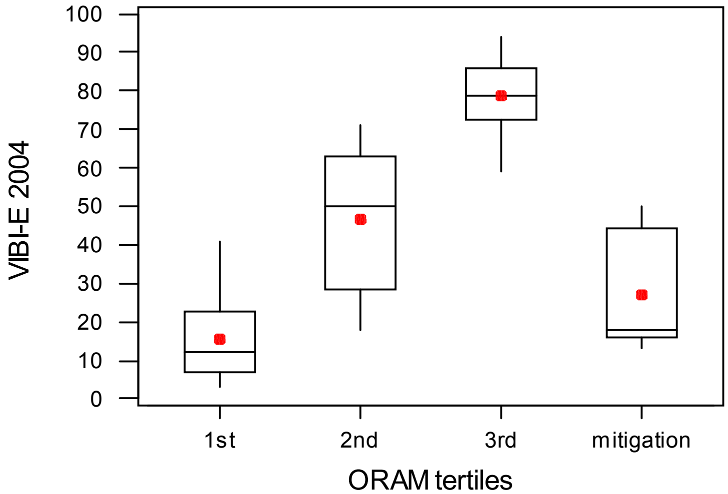


Figure 25. Vegetation Index of Biotic Integrity scores (VIBI) (Mack 2004c). Box and whisker plots represent ORAM score tertiles (thirds). All means are significantly different ($p < 0.05$) except 1st tertile and mitigation.

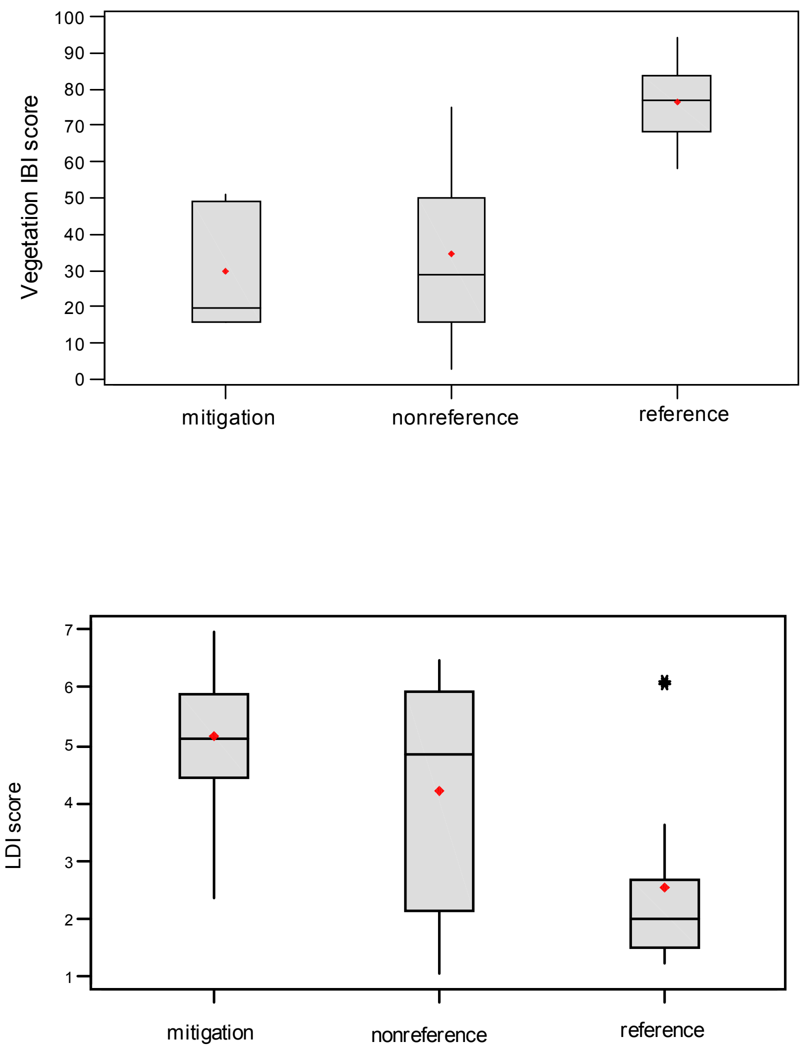


Figure 26. a) Vegetation Index of Biotic Integrity scores (Mack 2004c) versus wetland classified as mitigation (29.9), nonreference (34.6), and reference (76.6) sites. Reference wetlands have significantly higher VIBI scores than mitigation or nonreference wetlands (p < 0.001); b) LDI score versus wetland classified as mitigation (5.2), reference (4.2), and reference standard (2.6) sites. Reference sites have significantly lower LDI scores mitigation or nonreference sites (p < 0.001).

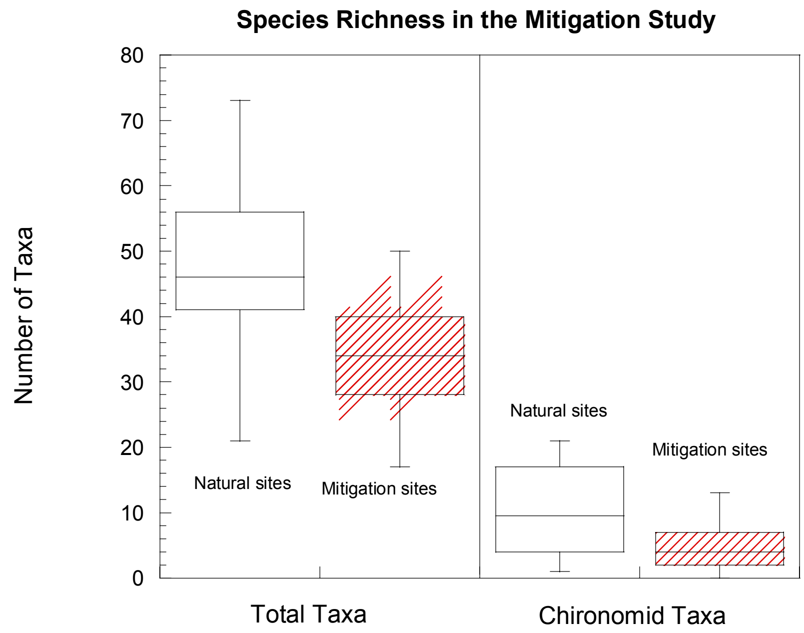


Figure 27. Comparison of total taxa and Chironomid taxa between natural and mitigation wetlands.

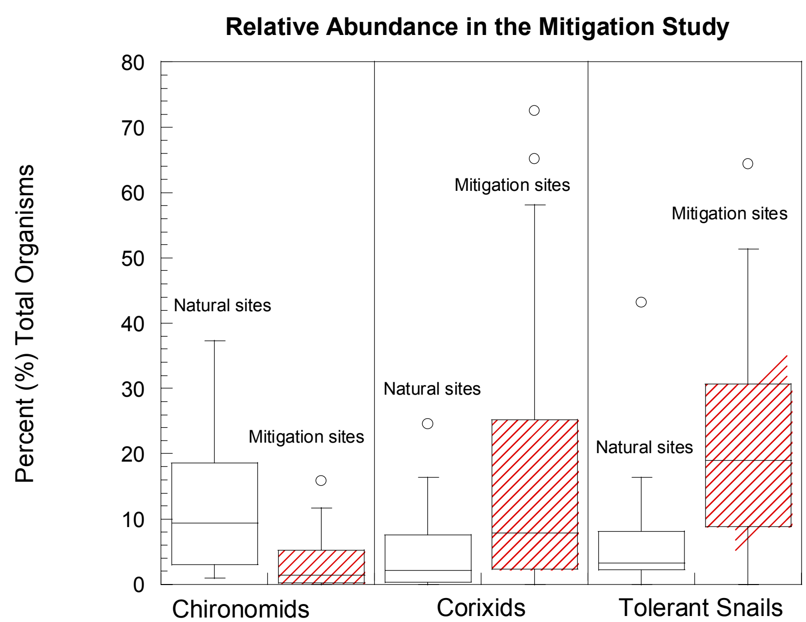


Figure 28. Percent of total organisms at natural and mitigation sites for Chironomids, Corixids, and tolerant snail species.

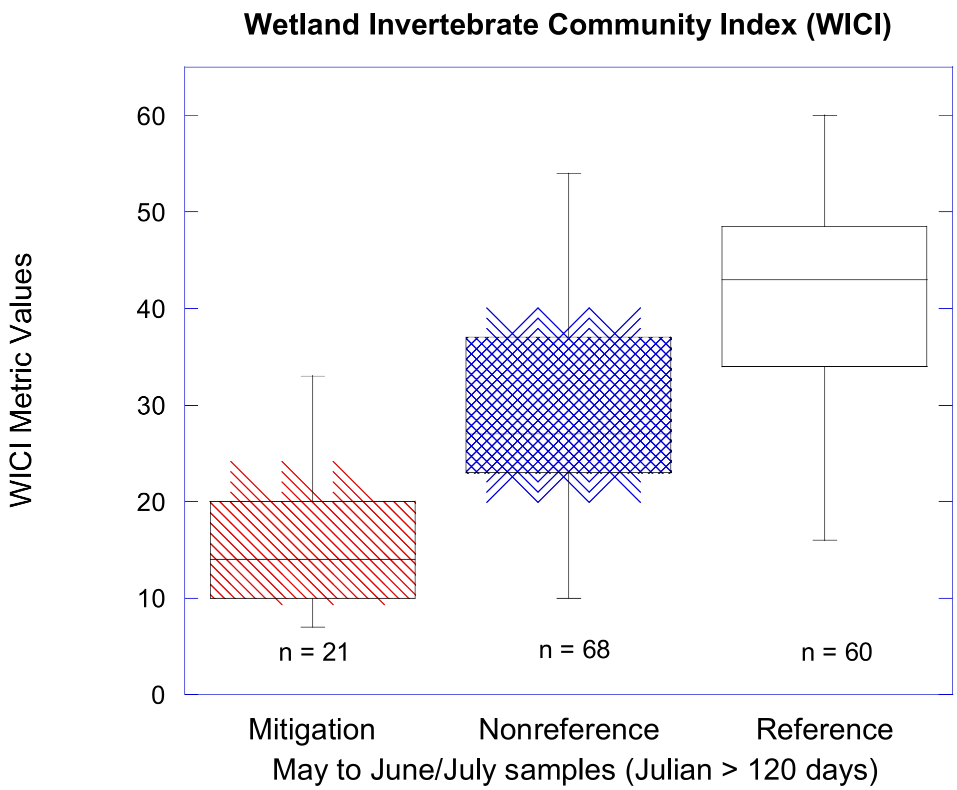


Figure 29. Wetland Invertebrate Community Index (WICI) scores for mitigation, nonreference and reference wetlands. n = number of sample events not number of sites.

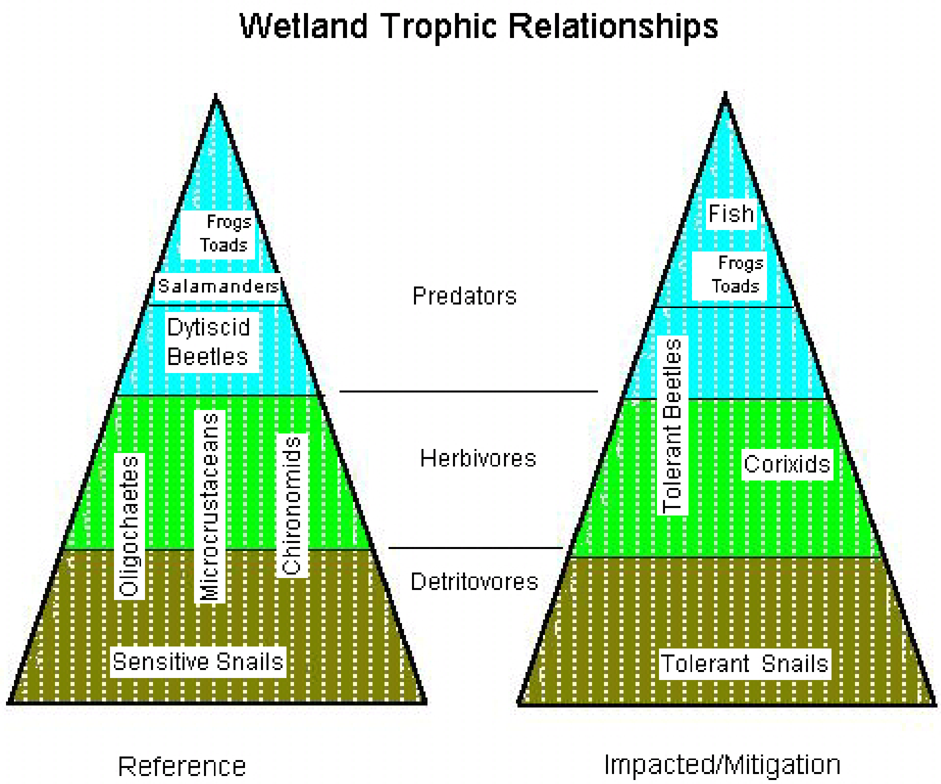


Figure 30. Comparison of wetland trophic relationships between reference and nonreference (impacted) and mitigation wetlands.

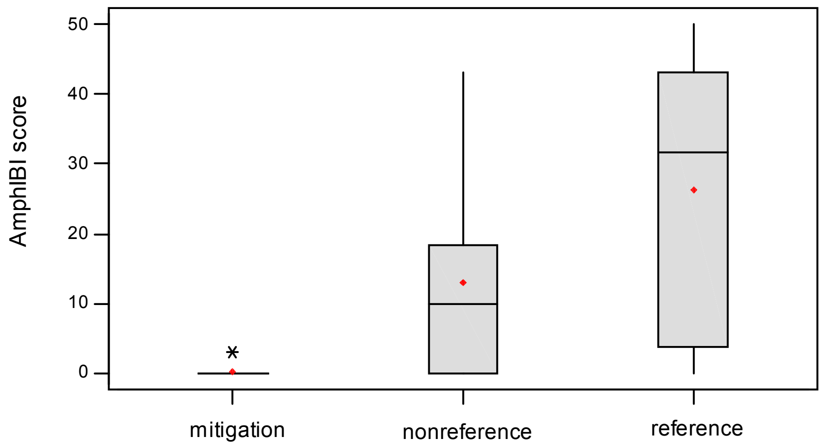


Figure 31. Box and whisker plots of Amphibian IBI score for mitigation, nonreference (moderately to severely degraded) and reference wetlands. Reference standard sites signficantly different from mitigation and nonreference sites ($p < 0.05$). Data includes forested, emergent and shrub dominated wetlands.

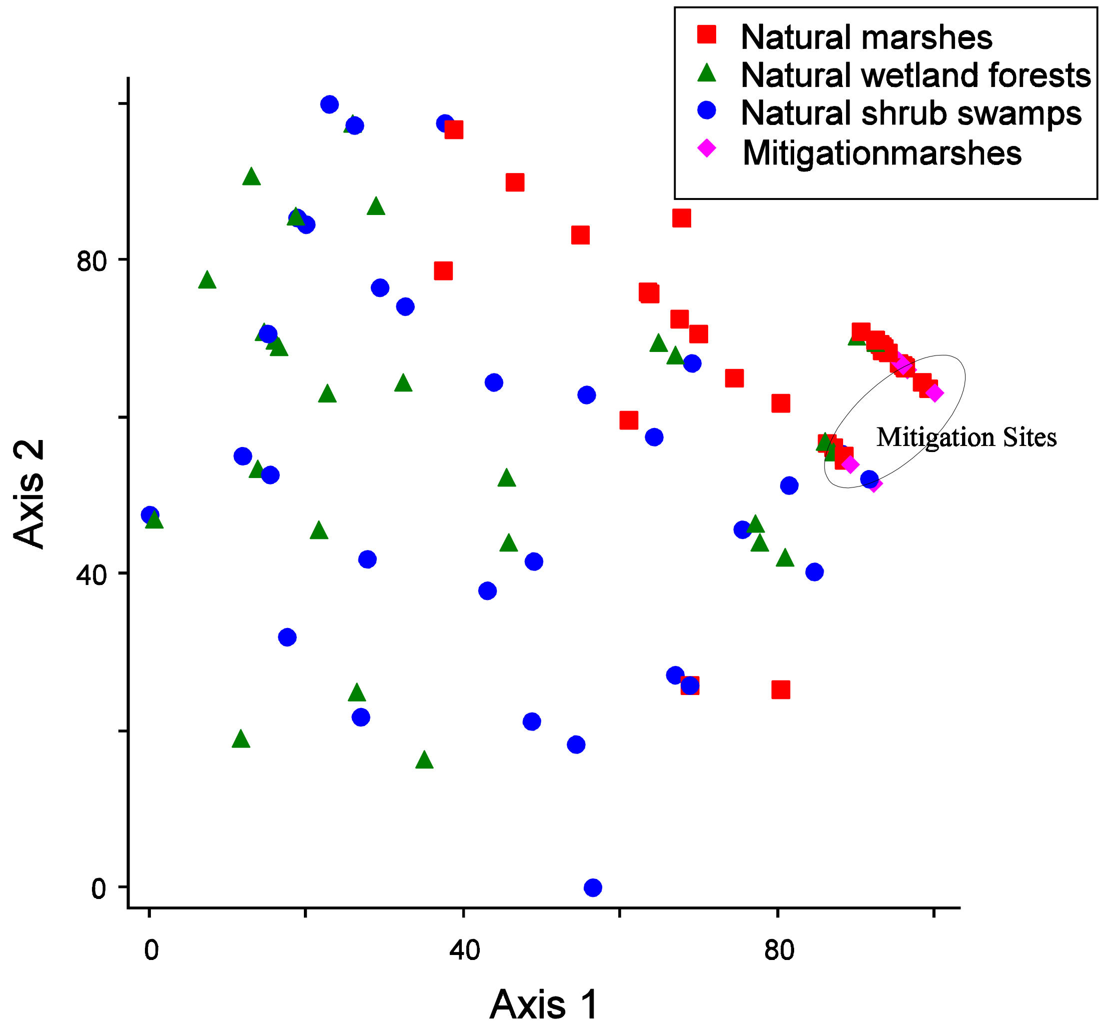


Figure 32. Principal components analysis of AmphIBI metrics. Note extremely tight clustering of 10 mitigation wetlands and separation of emergent from forest and shrub dominated wetlands.

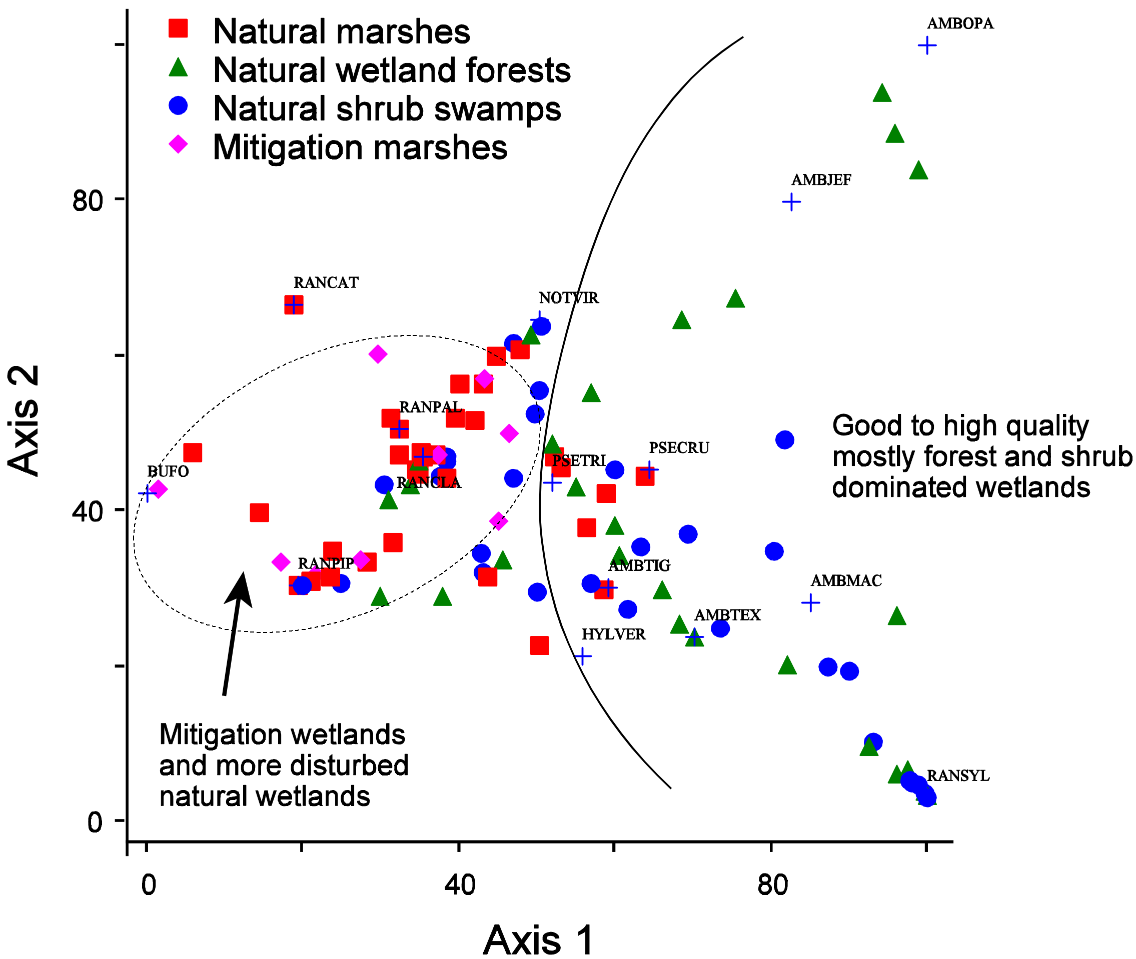


Figure 33. Detrended correspondence analysis of of amphibian presence and abundance. Total intertia (variance) in species data = 6.26, eigenvalues = 0.831, 0.658, 0.436, axes 1, 2, and 3, respectively. Note effect of wood frogs (RANSYL), spotted salamanders (AMBMAC), marbled salamander (AMBOPA) and jefferson salamanders (AMBJEFF) on ordination.

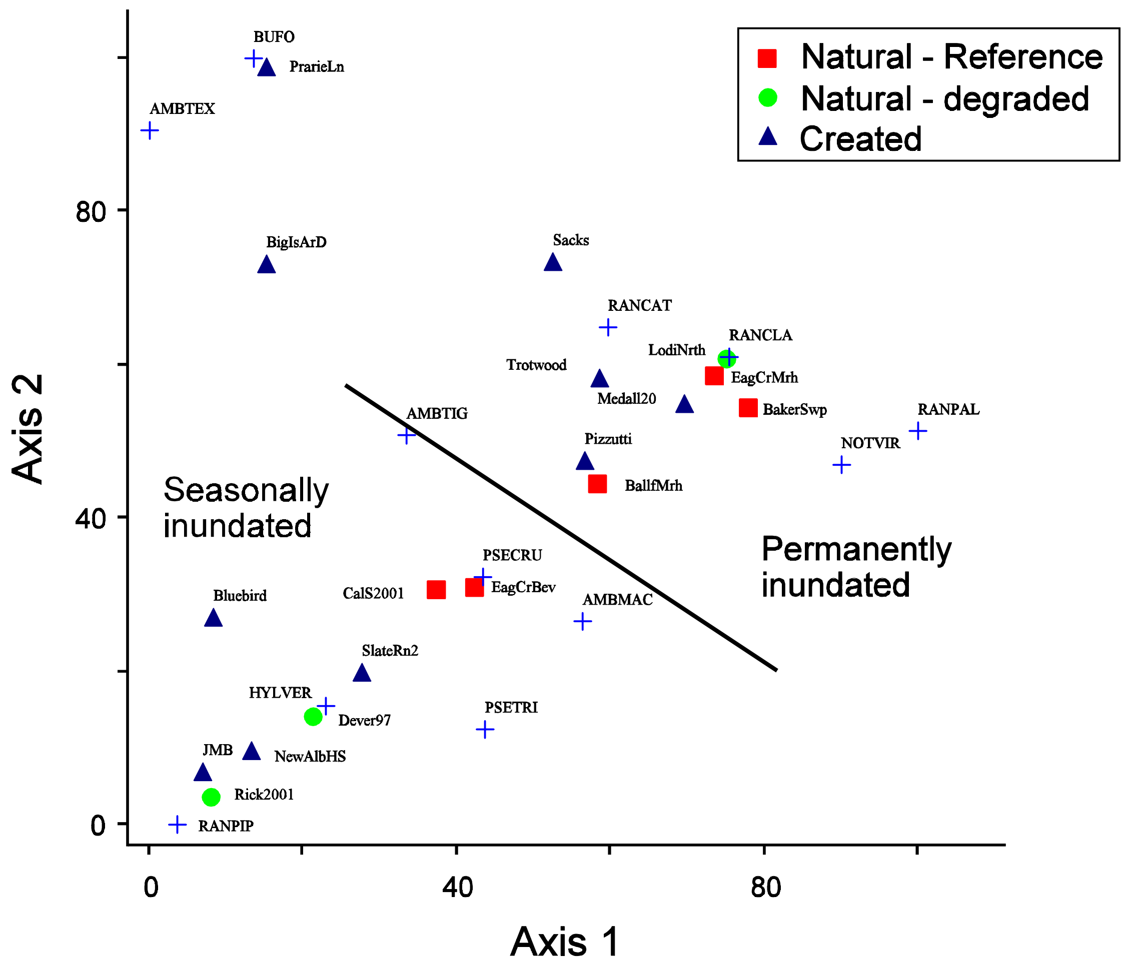


Figure 34. Detrended correspondence analysis of 8 natural and 10 mitigation wetlands. Total inertia (variance) in species data =3.19, eigenvalues = 0.766, 0.387, and 0.168, axes 1, 2, and 3 respectively.

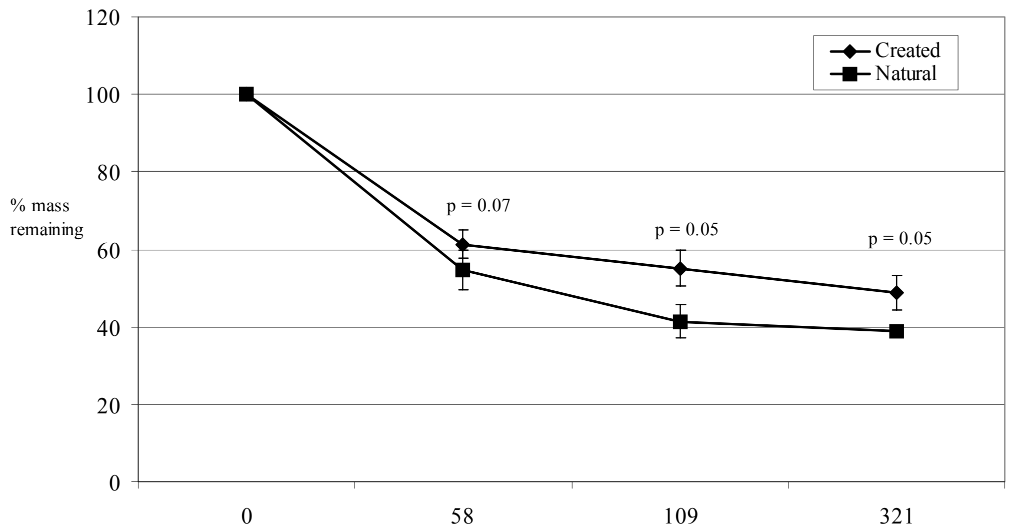


Figure 35. Percent mass remaining of ON-SITE *Typha* over the study period. Each data point is the mean of 5 stations per site per collection period. Error bars are standard error.

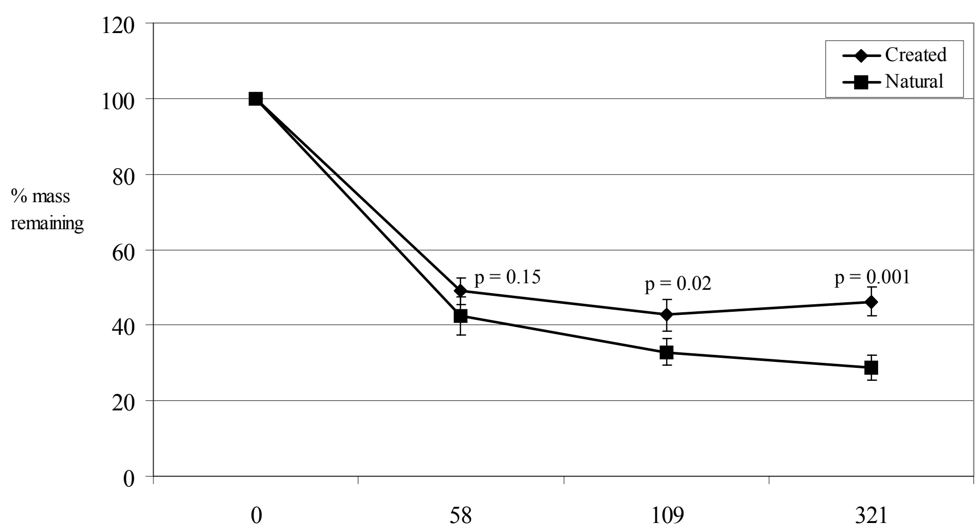


Figure 36. Percent mass remaining of CONTROL *Typha* over the study period. Each point is the mean of 3 stations per collection period per site. Error bars are standard error.

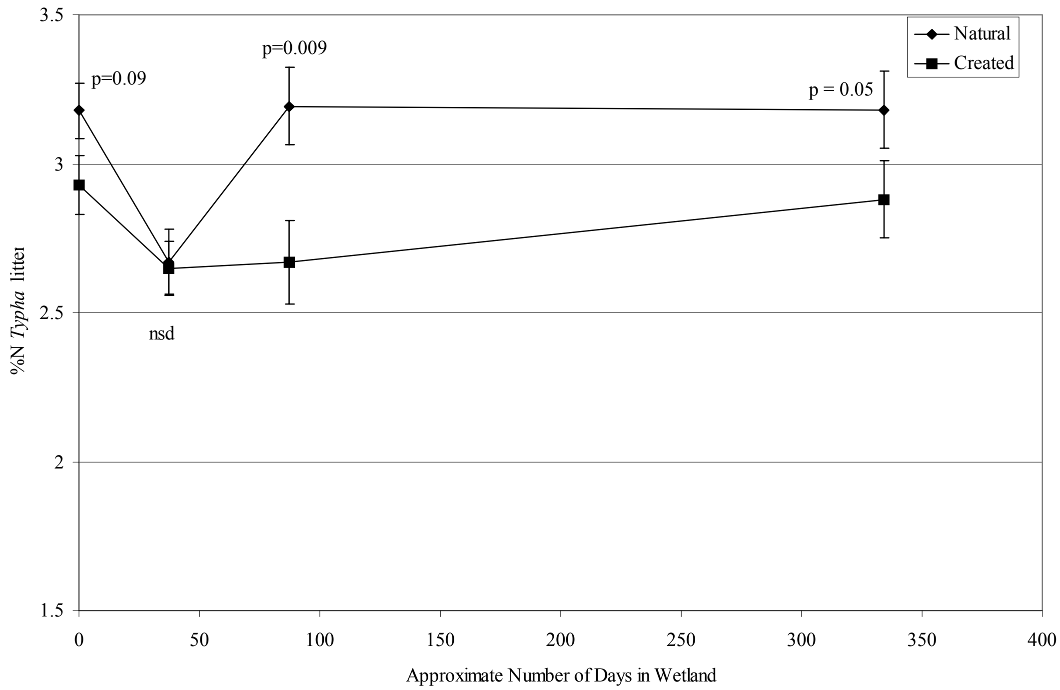


Figure 37. Nitrogen (%) in the ON-SITE *Typha* litter in each decomposition period. The value for each point represents an average of 2 stations per site. Error bars represent the standard error.

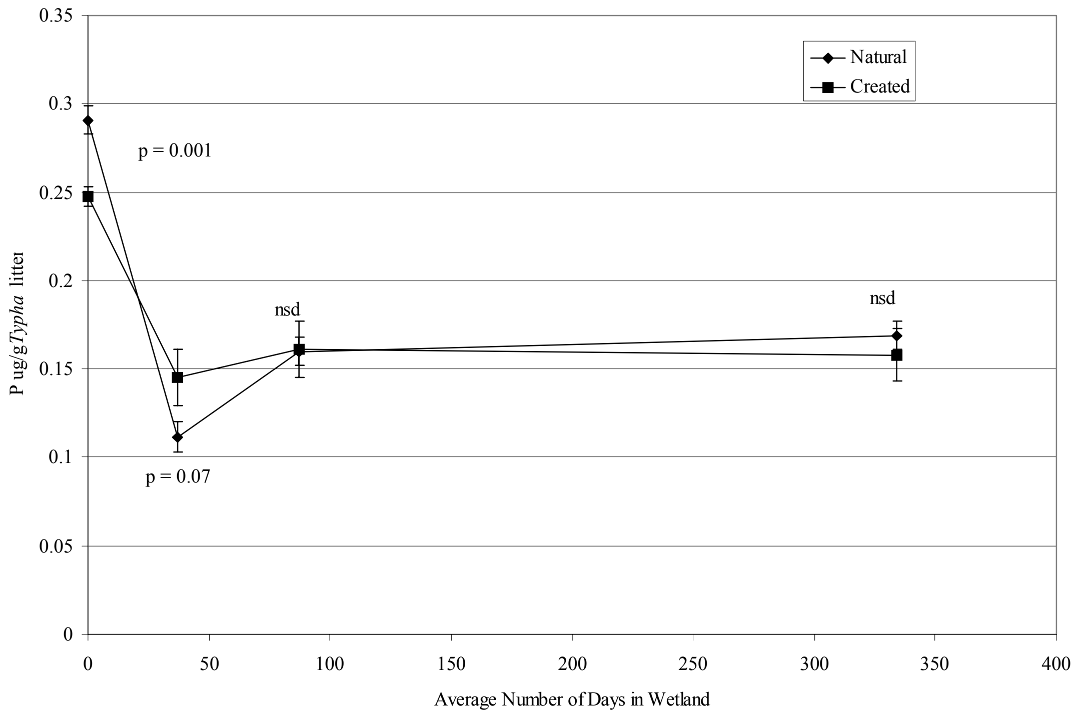


Figure 38. Phosphorus concentrations (µg/g) in the ON-SITE *Typha* litter in each decomposition period. The value for each point represents an average of 2 stations per site. Error bars represent the standard error.

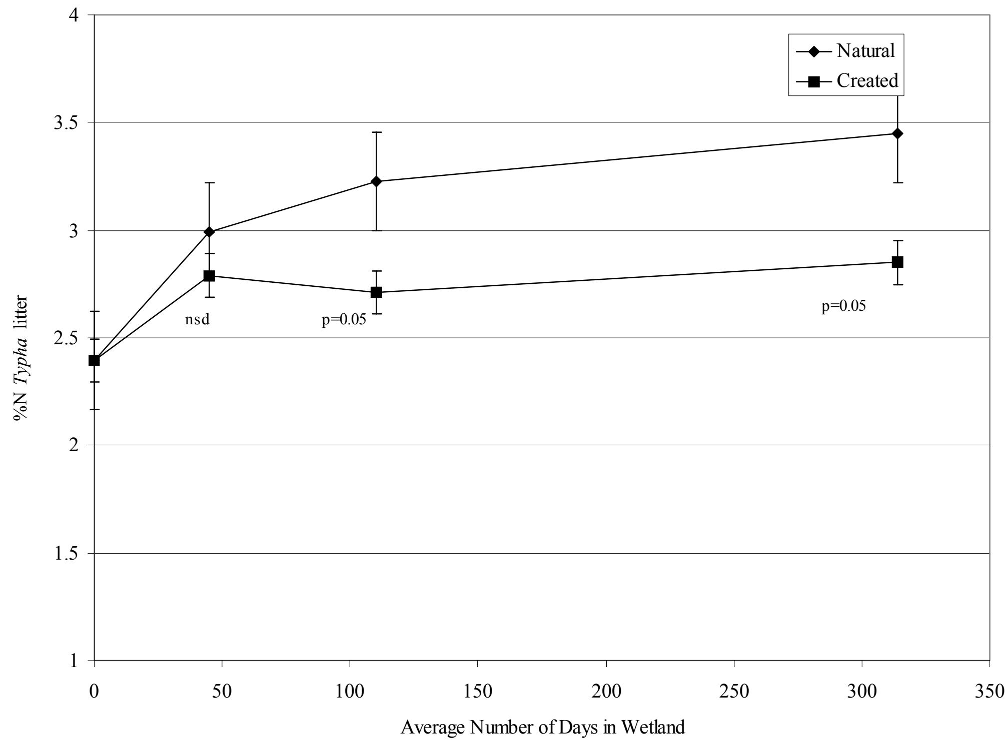


Figure 39. Total N (%) in the CONTROL *Typha* litter in each decomposition period. The value for each point represents an average of 2 stations per site. Error bars represent the standard error.

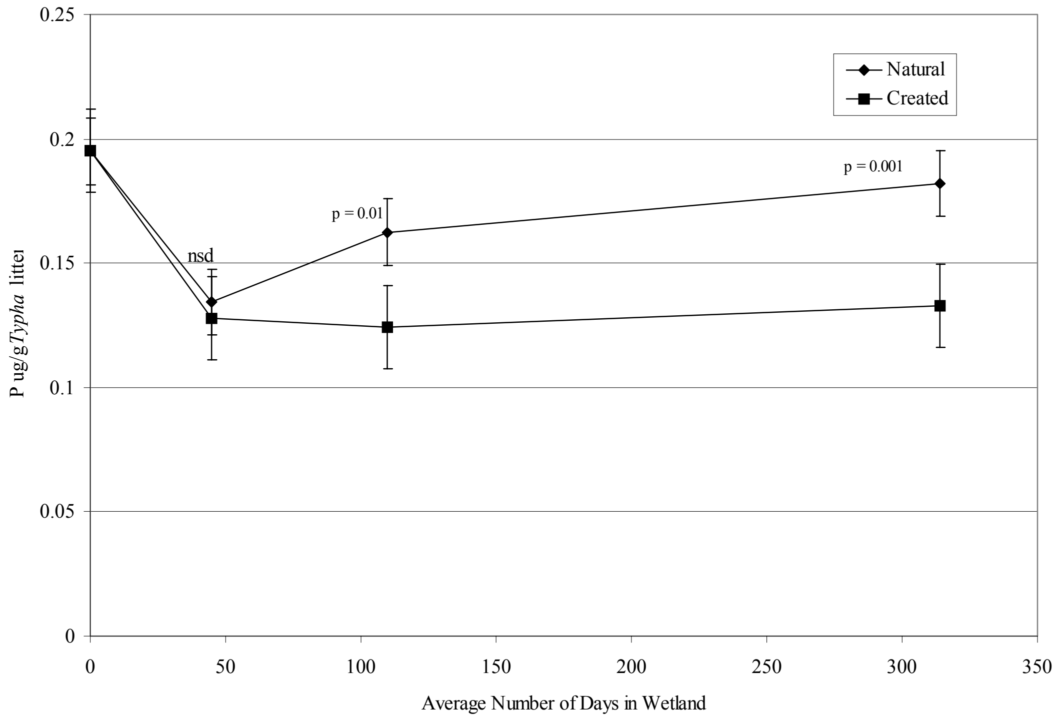


Figure 40. Phosphorus concentrations (µg/g) in the CONTROL *Typha* litter in each decomposition period. The value for each point represents an average of 2 stations per site. Error bars represent the standard error.

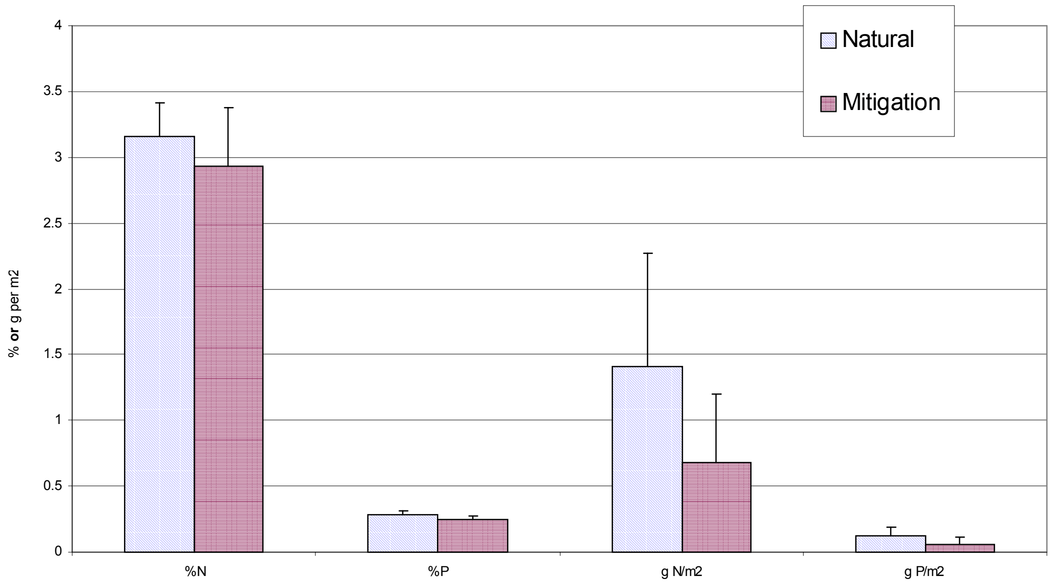


Figure 41. a) Initial litter concentrations of N and P at the time of peak biomass and b) the resulting standing stocks of N and P (g/m2) in each wetland type. Differences were significant in each case (two sample t-test for % concentrations $p < 0.10$, for standing stocks $p < 0.05$).

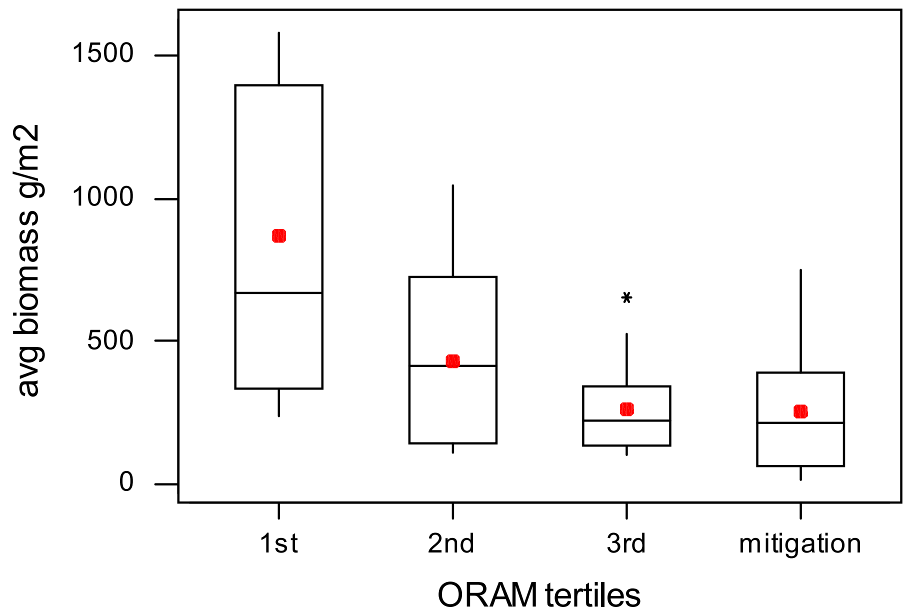


Figure 42. The relationship between the wetland disturbance categories (ORAM tertiles) and mean biomass production (g/m^2) in emergent marshes in Ohio. 1^{st} tertile significantly different from 2^{nd}, 3^{rd}, and Mitigation ($p < 0.05$).

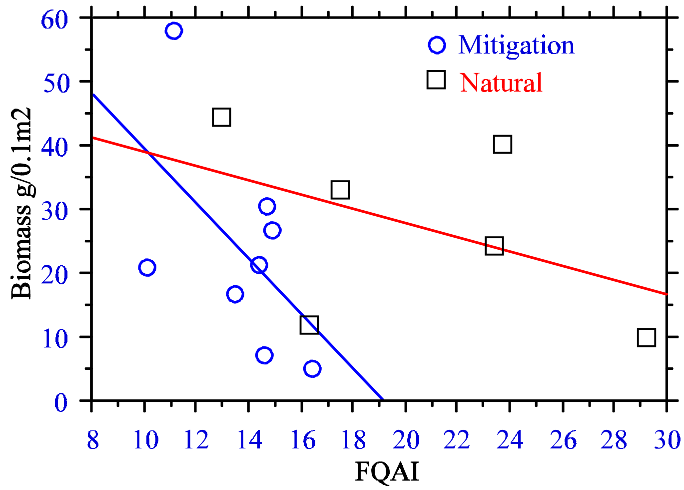


Figure 43. The relationship between FQAI scores and biomass production for the 19 wetlands included in this study. Because the regression models were so different for the population of natural wetlands compared to the population of mitigation sites, each is graphed separately. Both models are significant ($p < 0.10$), where Mitigation biomass = 82.2-4.3*FQAI ($R^2 = 0.30$) and Natural biomass = 50.4 – 1.1*FQAI ($R^2 = 0.22$).

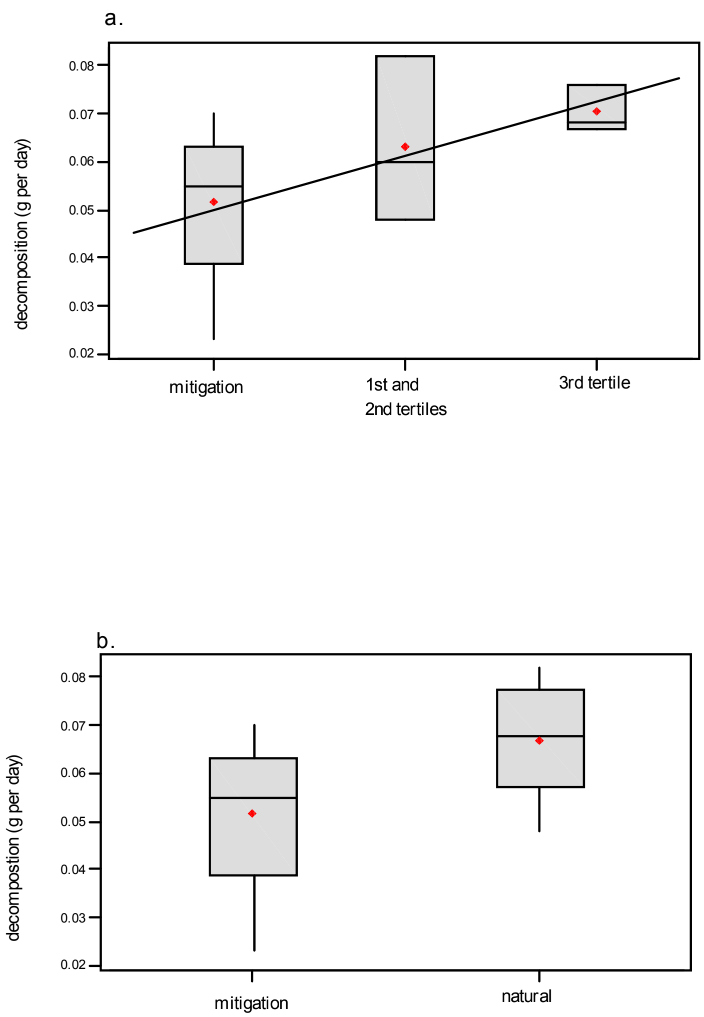


Figure 44. Decomposition rates for ON-SITE litter after 2nd litter bag removal for a) wetland disturbance categories (ORAM tertiles) (1st and 2nd ~ high to moderate disturbance), 3rd ~ low disturbance) and b) by wetland type (natural or mitigation). Line in 44a is line hand fitted to mean values.

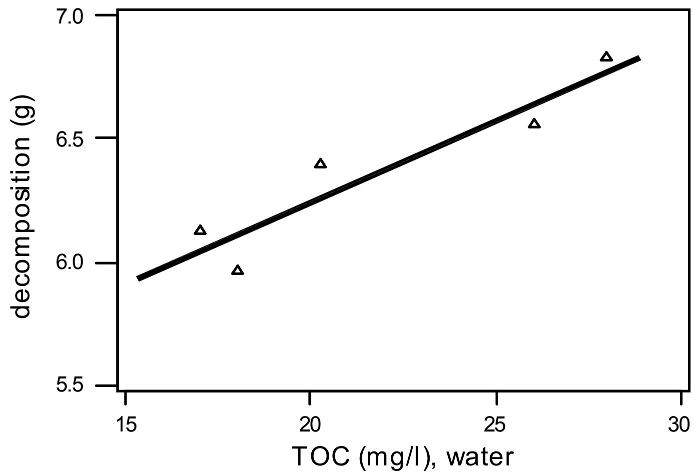


Figure 45. The influence of total organic carbon (mg/L) in the water column on decomposition (total mass lost in g at the 3rd pick-up) (R^2 = 83.7%, p = 0.019) for 5 natural wetlands in this study. No water sample collected at Rickenbacker because of early dry down in 2001. Eagle Cr Beaver, Eagle Cr Marsh, and Lake Abrams excluded due to possible inter-year differences.

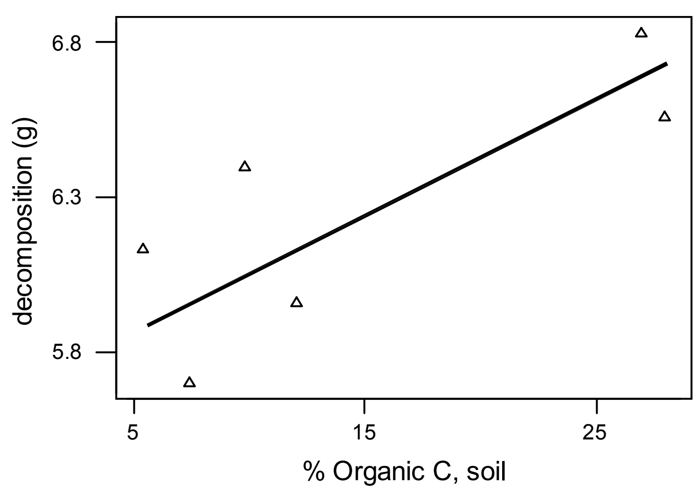


Figure 46. The influence of percent organic carbon in the soil on decomposition (total mass lost in g at 3rd pickup) (R^2 = 63.1% p = 0.059) in 6 natural wetlands included in this study. Eagle Cr Beaver, Eagle Cr Marsh, and Lake Abrams excluded due to possible inter-year differences.

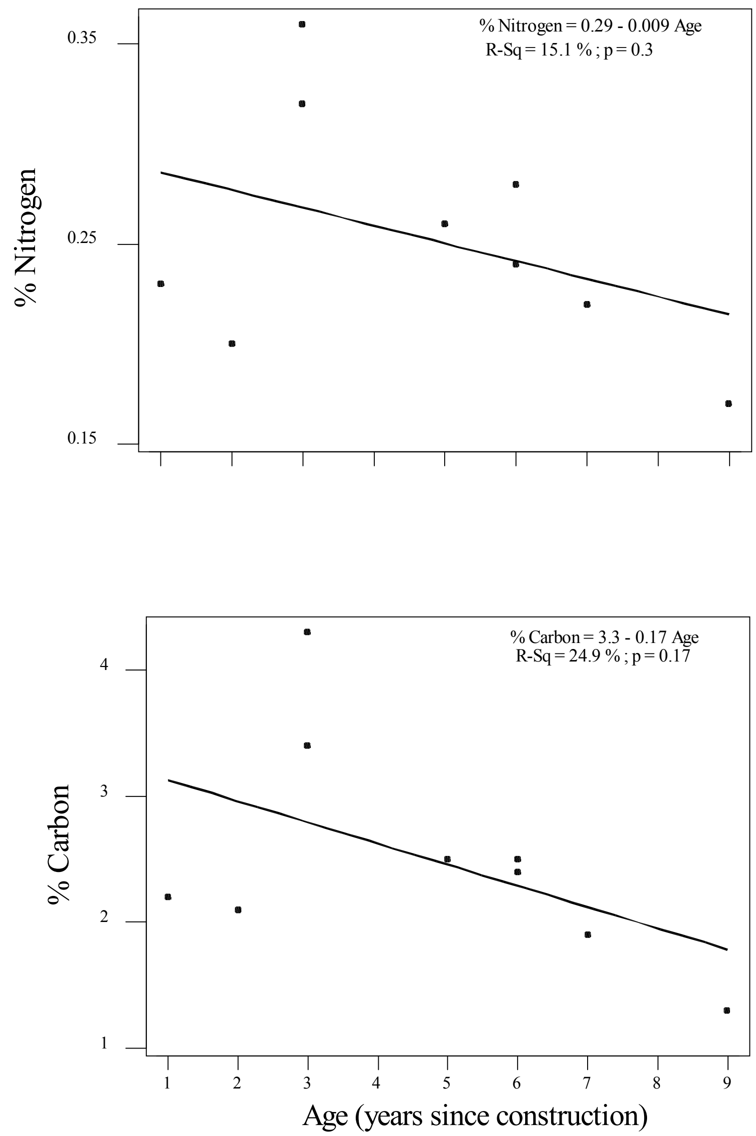


Figure 47. Regression analysis showing the lack of a relationship between soil nitrogen and soil carbon and mitigation wetland age.

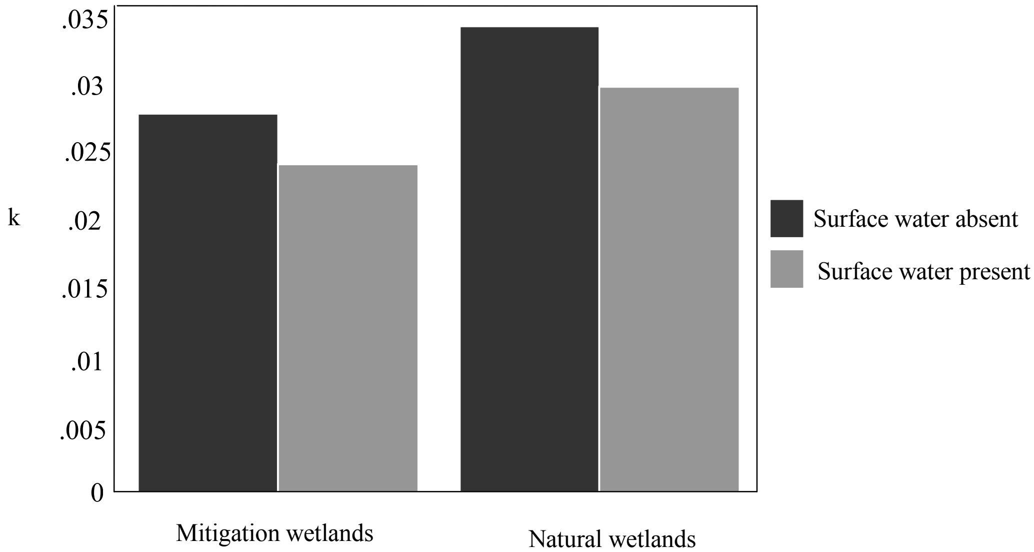


Figure 48. Differences in decomposition rates in mitigation and natural wetlands as a function of water presence at the time of litterbag retrieval.

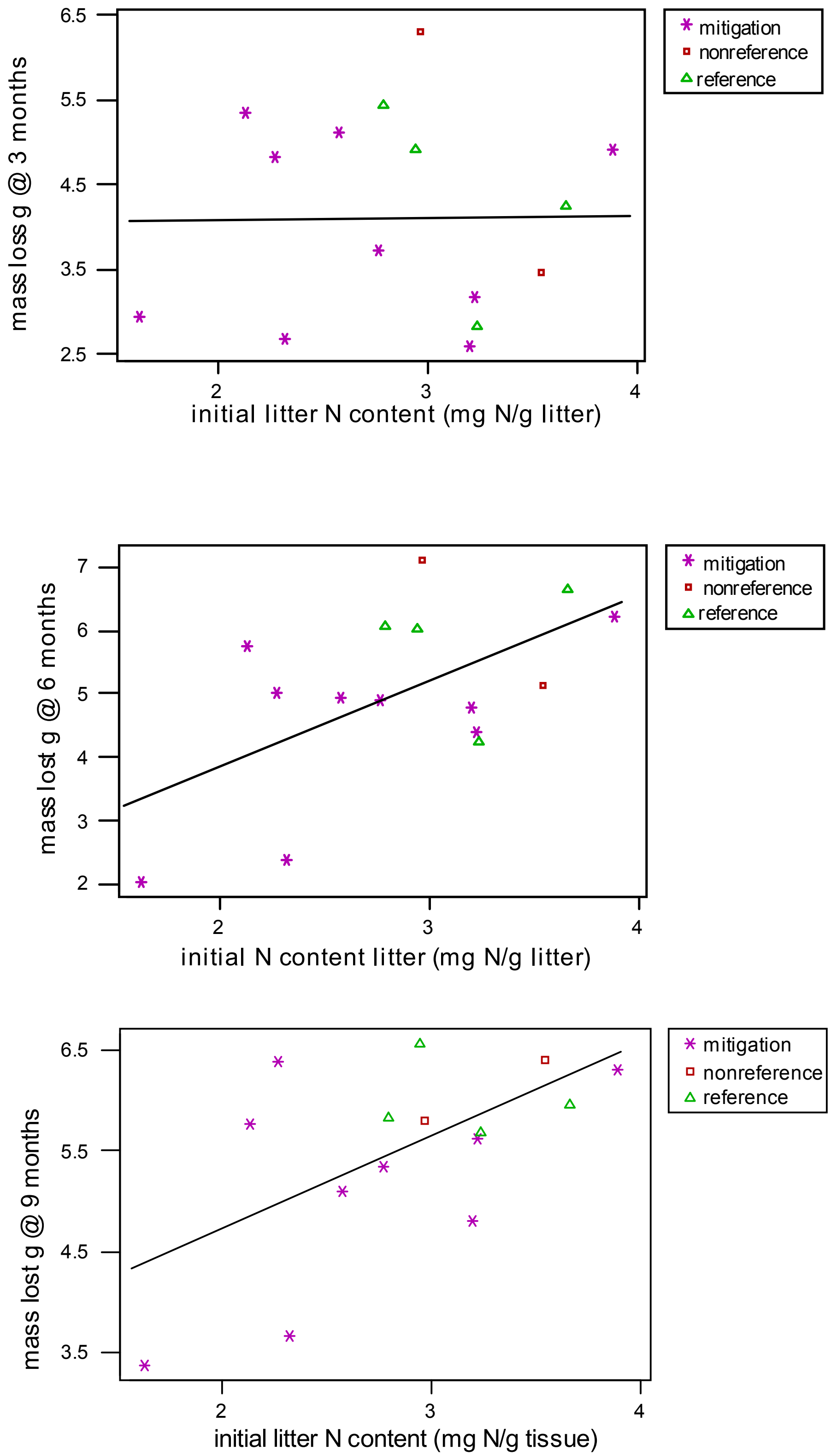


Figure 49. On-site litter mass lost at approximately 3, 6 and 9 months versus initial litter N content (mg N/g litter) for mitigation, nonreference, and reference (least impacted) sites. Lines are fitted regression lines for 3 months ($p = 0.995$, $R^2 = 0\%$), 6 months ($p = 0.030$, $R^2 = 31.4\%$), 9 months ($p = 0.021$, $R^2 = 34.7\%$).

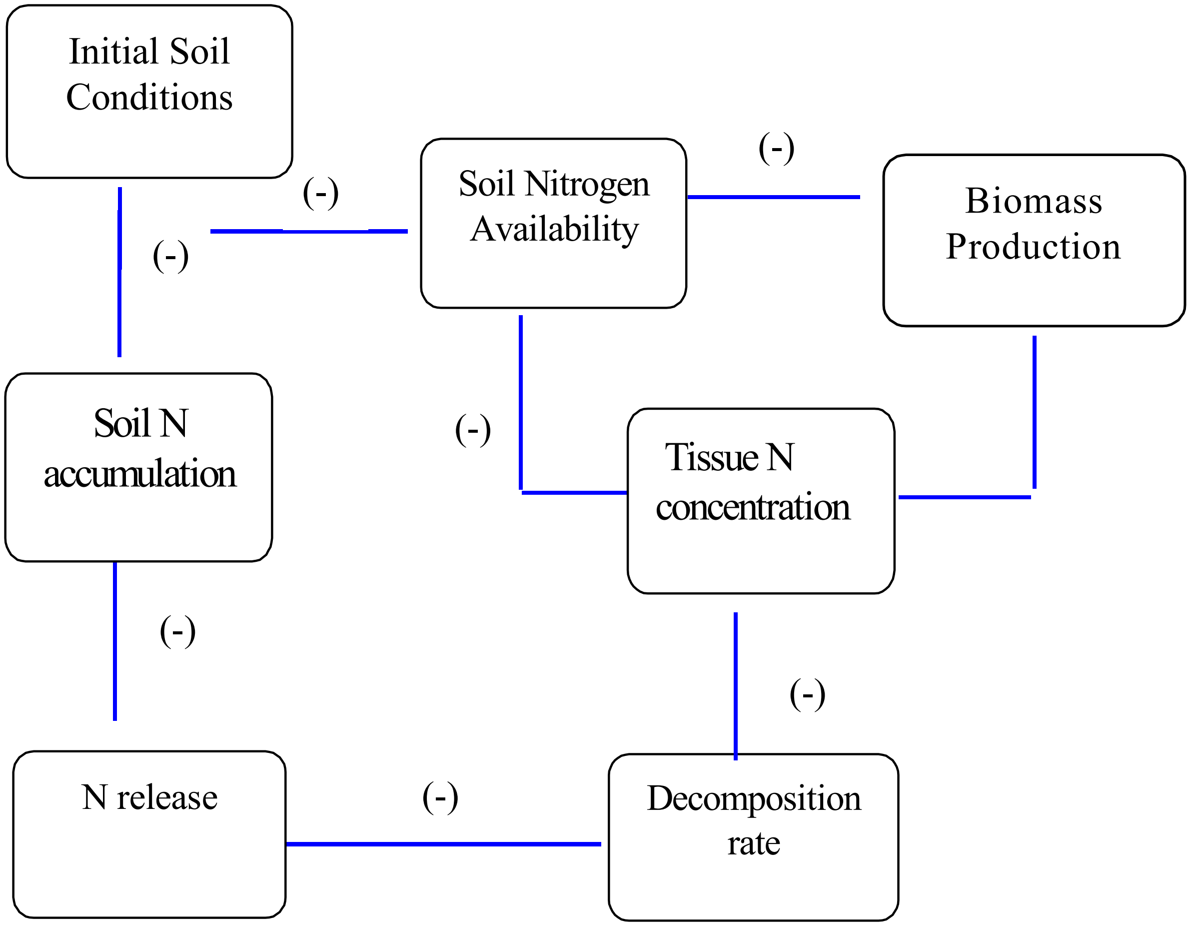


Figure 50. Flow chart of nutrient dynamics in mitigation wetlands relative to natural ecosystems. Low nutrient availability in wetland soils affects both biomass production and the plant litter nutrient concentration. Low litter nutrient concentrations negatively affects the decomposition rates, and slow decomposition rates along with low nutrient accumulation by litter limits the accumulation of nutrients in soil.

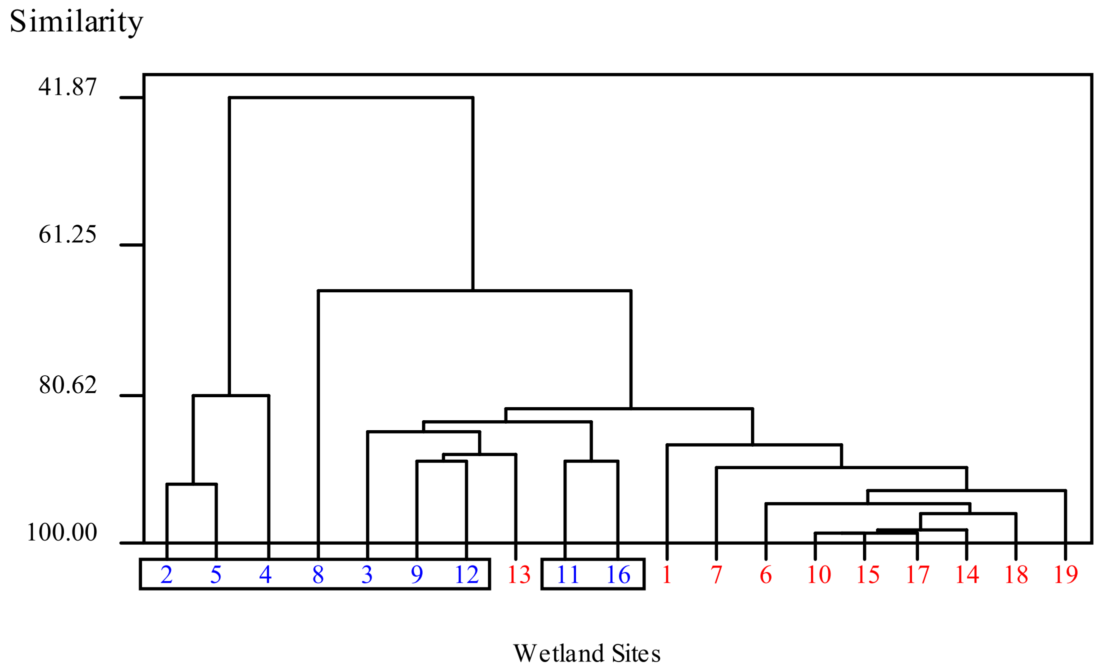


Figure 51. Cluster analysis using a combination of vegetation (FQAI score) and soil characteristics (% total carbon and % nitrogen). Natural wetlands outlined in box. Key for wetland sites: 1 = Sacks, 2 = Calamus, 3 = Rickenbacker, 4 = Lodi North, 5 = Eagle Cr Beaver, 6 = Slate Run Bank SE, 7 = Medallion No. 20, 8 = Ballfield, 9 = Dever, 10 = Bluebird, 11 = Baker Swamp 12 = Lake Abrams, 13 = New Albany HS, 14 = Prairie Lane, 15 = Big Island Area D, 16 = Eagle Cr Marsh, 17 = JMB, 18 = Pizutti, 19 = Trotwood.

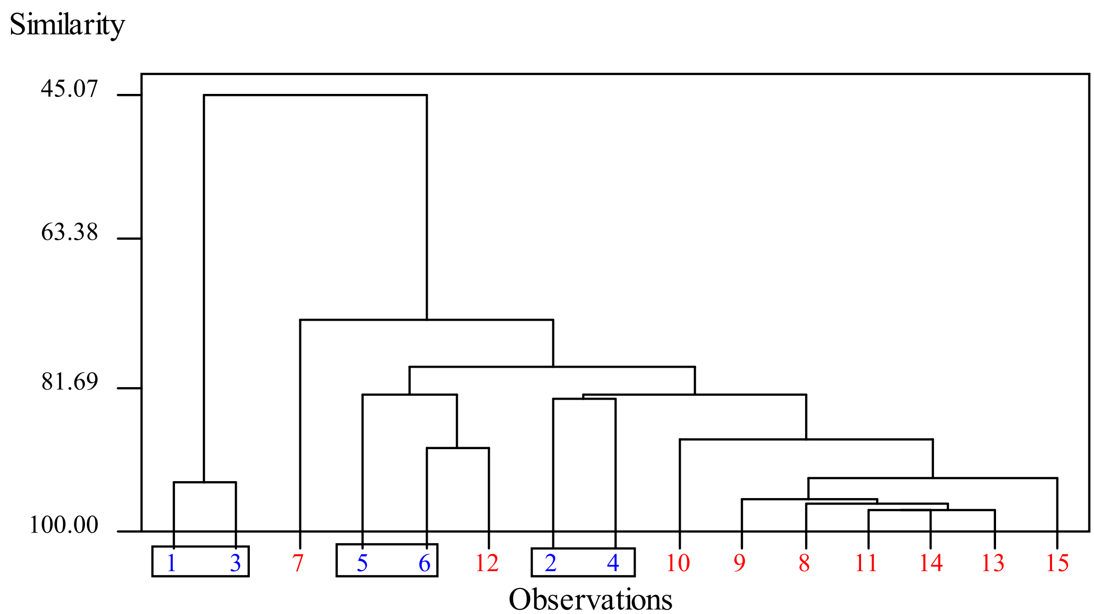


Figure 52. Cluster analysis using the physiochemical variables (soil N and C) and plant litter N concentration. Natural wetlands outlined with box. Key for wetland sites: 1 = Calamus, 2 = Rickenbacker, 3 = Lodi North, 4 = Ballfield, 5 = Dever, 6 = Baker Swamp, 7 = New Albany HS, 8 = Bluebird, 9 = Big Island Area D, 10 = JMB, 11 = Pizzuitti, 12 = Sacks, 13 = Slate Run Bank SE, 14 = Medallion No. 20, 15 = Prairie Lane.

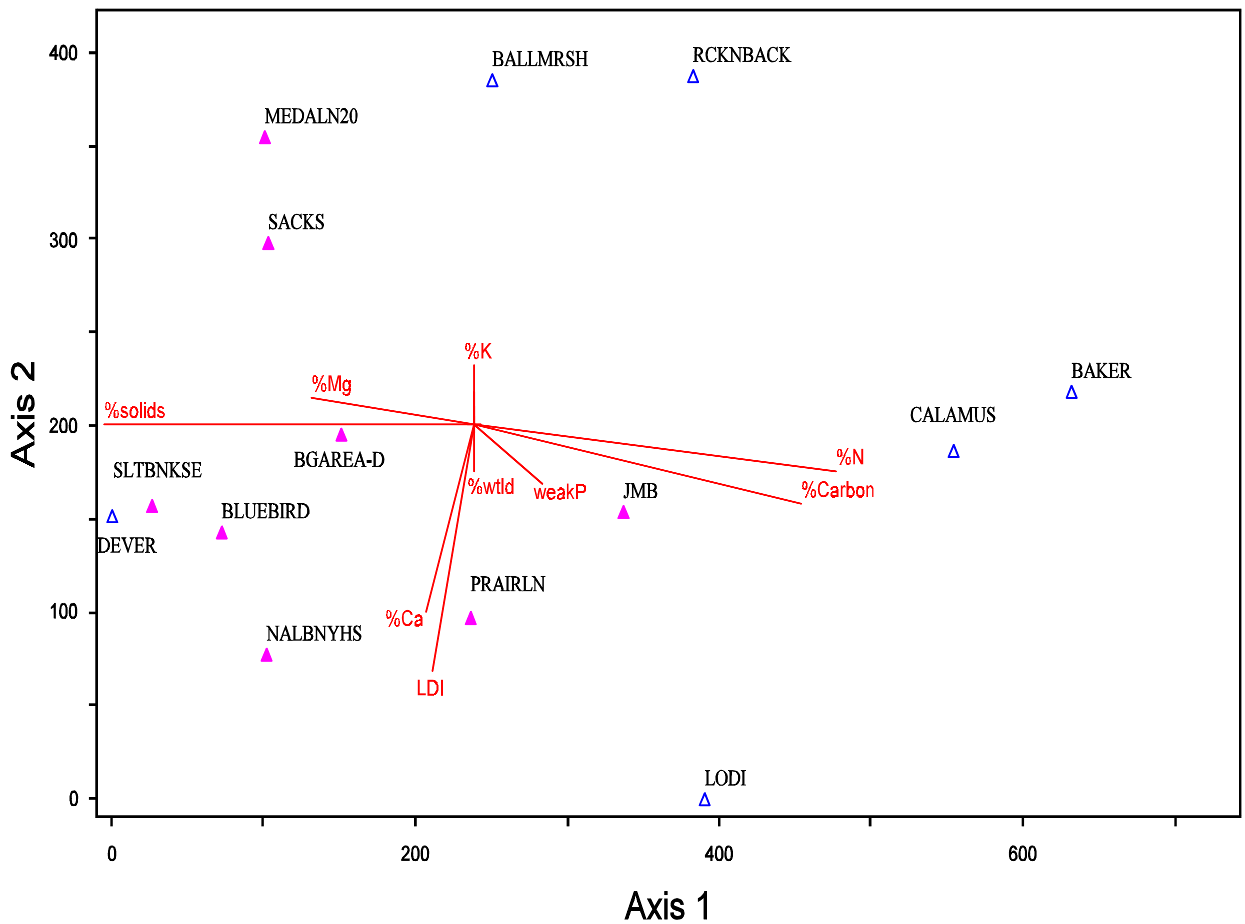


Figure 53. Detrended correspondance analysis of of natural and mitigation wetlands (n = 19 sites, 110 species, 13 environmental variables). Total inertial (variance) in species data = 6.66; eigenvalues = 0.851, 0.795, and 0.744 for axes 1, 2, and 3 respectively. Biplot indicates distribution of sites relative to key environmental variables. All mitigation wetlands ordinate in the left-center of the plot while natural sites are dispersed primarily to the right of the circle. One natural wetland is an exception, grouping closer to the mitigation sites. This site is Dever which was the wetland most degraded by human activities (agriculture). The two natural sites circled on the right are reference standard sites.