ECOLOGICAL ASSESSMENT OF COMPENSATORY WETLAND MITIGATION

Final Report

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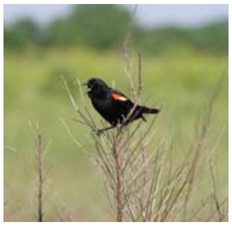







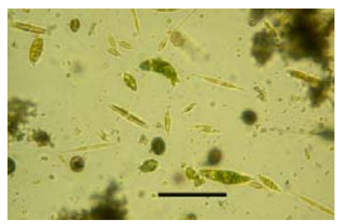

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EXECUTIVE SUMMARY

Intensive biological inventories were used to evaluate ecological performance at 12 Iowa Department of Transportation mitigation wetlands and three reference wetlands in Iowa. Species richness and abundance data were collected on algae, protozoa, aquatic invertebrates, vascular plants, butterflies, amphibians, reptiles, birds and mammals at each site. Species richness and diversity at mitigation sites and reference sites were compared to determine if mitigation wetlands are performing differently than reference wetlands in Iowa. In addition, abiotic factors having the potential to influence biological diversity were also studied, including water quality and physical and landscape characteristics (local and watershed level) at each study site.

The results of this comprehensive study of the ecological performance of wetland mitigation sites suggest that mitigation sites in Iowa are performing similarly to reference wetlands ecologically. Reference wetlands and mitigation wetlands in Iowa are similar in terms of water quality; landscape processes; site conditions; diversity of algae/protozoa/aquatic invertebrates, amphibians, birds, mammals, reptiles; and overall plant and animal diversity. No significant difference was found in overall diversity or within a species group, with the exception of butterflies, as estimated by effective number of species at mitigation and reference wetlands. Because the effective number of species is a measure of the number of common species at a site, this result suggests that the number of common species within each species group is approximately equal between mitigation and reference sites.

Significant differences were found between mitigation and reference wetlands in terms of butterfly diversity and plant composition and floristic quality. Mitigation wetlands were found to have higher butterfly species richness and a significantly greater number of rare butterfly species than reference wetlands. In contrast, reference wetlands were found to have more native plant species, fewer exotic plant species, contained species with wetter indicator status, and more importance of *Carex* species.

The study also evaluated selected existing rapid assessment methods to determine the appropriateness of each for assessing and characterizing ecological performance of mitigation sites and to develop a conceptual framework for developing a new, or adapting an existing, rapid assessment method for use by the Iowa DOT. An existing rapid assessment tool, the *Wetland Mitigation Quality Assessment*, was found to provide the best measure of ecological performance as measured by biodiversity of the four rapid assessment methods evaluated in this study. The WMQA has the potential be used as both a performance measure for wetland mitigation sites and an assessment tool for wetland impact studies.

The results of this study are valuable for building and expanding the tools and knowledge necessary to effectively assess and manage the ecological performance of compensatory mitigation wetlands and improve the ecological effectiveness of wetland mitigation.

CHAPTER 1

INTRODUCTION AND BACKGROUND

The Iowa Department of Transportation (DOT) provides for a safe and reliable transportation system while protecting and enhancing the state's environmental and aquatic resources. As part of this effort, the Department must comply with the Clean Water Act (33 U.S.C. 1344), frequently by constructing replacement wetland areas referred to as mitigation sites.

Compensatory wetland mitigation is traditionally carried out through restoration of a previously-existing wetland, enhancement of an existing wetland, creation of a new wetland, or preservation of a unique wetland. Wetland mitigation is typically performed either on-site or off-site by permittees or through purchasing credits at a mitigation bank or in-lieu fee mitigation program.

Despite these methods and mechanisms for providing compensatory mitigation, scrutiny of compensatory wetland mitigation programs across the county has taken place in recent years (National Research Council 2001; Storm and Stellini 1994). In the late 1990's, the National Research Council concluded that mitigation programs were failing to meet the goal of no net loss of wetlands, that permit conditions fail to clearly define performance expectations, and that the mitigation program lacks a suitable mechanism to assure compliance (National Research Council 2001).

The degree of success of compensatory wetland mitigation programs has been evaluated in several recent studies. For example, Brown and Veneman (2001) found over 50 percent of the 114 mitigation sites sampled in Massachusetts failed to meet permit conditions largely due to acreage shortfalls and out of kind mitigation (e.g., the mitigation wetland was not the type of wetland specified in the permit). Other studies have also documented challenges with meeting permit requirements, including in Tennessee where 75 percent of the study sites failed to meet acreage requirements (Morgan and Roberts 2003) and in Indiana where 44 percent of the sites that did not meet the acreage requirements resulted in a net loss of wetlands (Robb 2002).

In contrast, VanDeWalle et al. (2007) found that 58 percent of the wetland mitigation sites evaluated in Iowa for regulatory compliance were successful when Section 404 permit acreage requirements were used as the criteria for measuring success and when net gain/loss was used as the measure of success, wetland mitigation resulted in a net increase of nearly 44 acres of wetland over what was required by permits.

In response to these challenges, the Environmental Protection Agency (EPA) and the U.S. Army Corps of Engineers (USACE) developed the National Wetlands Mitigation Action Plan to affirm a commitment to the national goal of no net loss of the Nation's wetlands (USACE 2002). The Plan recommended several actions to ensure effective restoration and protection of wetlands including clarifying mitigation guidance, integrating compensatory mitigation into a watershed context, improving compensatory mitigation accountability, clarifying performance standards, and improving data collection and availability.

In March 2008, the USACE and EPA issued a final rule regarding compensatory mitigation for losses of aquatic resources. The rule was intended to improve the planning, implementation, and management of compensatory mitigation by encouraging a watershed approach, requiring performance standards, specifying standard mitigation plan components, and including long-term and responsible party assurances. The rule incorporates many of the National Research Council's (2001) key recommendations. Regardless, more research into the ecological performance of mitigation sites is necessary to better understand these important ecosystems.

In an effort to better understand the ecology of wetland mitigation, the Iowa DOT was awarded a grant from the EPA to investigate the ecological performance of mitigation. Currently, Iowa DOT compensatory wetland mitigation projects are assessed by the total number of wetland mitigation acres attained compared with Section 404 permit requirements. In the absence of an ecological assessment tool to measure wetland condition, little is known about the ecological performance of Iowa DOT mitigation wetlands. By evaluating mitigation wetland condition and establishing ecological performance standards, the Iowa DOT can better monitor and assess compensatory wetland mitigation and gain valuable knowledge toward improving the overall ecological effectiveness of compensatory mitigation.

This project focuses on evaluating the ecological performance of mitigation wetlands and existing monitoring and assessment programs. A tiered approach, including landscape-scale assessment and intensive biological and chemical measurements, was used to evaluate the ecological conditions at 12 wetland mitigation sites and three wetland reference sites. The data were used to summarize the ecological performance of wetland mitigation sites as well as to develop/recommend a tool for assessing ecological performance of wetlands. The research findings have the potential to improve wetland planning and management decisions by building and expanding the skills, tools, and knowledge necessary to effectively assess, plan, and manage the ecological performance of mitigation wetlands through scientific investigations.

The overall goal of this project is to improve the ecological effectiveness of compensatory mitigation. Specific project objectives include, 1) to assess and evaluate the ecological performance of Iowa DOT mitigation wetlands; 2) to develop performance standards for compensatory mitigation based on reference wetlands; 3) to design, or modify an existing, rapid assessment tool to expand wetland monitoring and assessment; and, 4) to facilitate professional development, information exchange, and public education.

The final report is organized into Chapters. Chapter 2 summarizes and compares the ecological condition of mitigation and reference wetlands. Technical reports outlining detailed methods and results from each working group (e.g., abiotic and biotic study variables) and individual data collections that are integrated and analyzed in Chapter 2 are contained in the Appendices included at the end of this report. Chapter 3 is a technical report describing the evaluation and use of several rapid assessment tools for the purpose of evaluating mitigation wetlands. The results of these analyses are integrated and recommendations made for future research in Chapter 4 of this report.

CHAPTER 2

ECOLOGICAL PERFORMANCE OF IOWA DEPARTMENT OF TRANSPORTATION MITIGATION WETLANDS

INTRODUCTION

As a transitional zone linking aquatic and terrestrial environments, wetlands are productive ecosystems with a wide range of functional values including habitat for a variety of plant and animal species dependent on wetland ecosystems for all or part of their life cycle (Wharton et al 1982, Reed 1988, and Tiner 1984). Several studies have estimated the number of wetland acres occupying the North American landscape prior to European settlement at somewhere between 200-220 million hectares (Shaw and Fredine 1956, Tiner 1984, Feierabend and Zelazny 1987). By the mid-1970's however, only a fraction of original wetland acreage remained (Dahl 1990).

A massive conversion to agricultural land use is reported as one of the largest impacts to wetland ecosystems, accounting for 87% of wetland losses in the United States from the mid 1950's to the mid 1970's (Frayer et al. 1983). While ongoing anthropogenic alterations (e.g., road construction and urban development) to natural landscapes continue to impact the remaining wetland ecosystems, along with the plants and animals that depend on them, growing understanding of the functional value of wetlands led to the protection of these ecosystems under federal legislation, specifically Section 404 of the Water Pollution Control Act or Clean Water Act (33 U.S.C. 1344).

The purpose of the Clean Water Act (CWA) is to restore and maintain the chemical, physical and biological integrity of the nation's waters. As a way of meeting this purpose, the CWA requires mitigation for unavoidable wetland losses resulting from conversion of wetland to other uses. Compensatory wetland mitigation is traditionally carried out through restoration of a previously-existing wetland, enhancement of an existing wetland, creation of a new wetland, or preservation of a unique wetland. The national goal of wetland mitigation under the CWA is "no net loss" of wetland acreage and function (USACE 2002). It is clear that impacts to wetland ecosystems have wide ranging implications on wetland dependent biota (Wilcove et al. 1993, Boylan and Maclean 1997, Lehtinen et al. 1999 and Mitsch and Gosselink 2000). What is not clear, however, is whether compensatory mitigation has been successful at addressing ecological concerns.

Compensatory wetland mitigation programs across the country have been increasingly scrutinized in recent years (National Research Council 2001; Storm and Stellini 1994). Studies indicate mitigation programs nationwide often fail to meet the goal of no net loss of wetlands (Brown and Veneman 2001; Robb 2002; Morgan and Roberts 2003). With no net loss of wetland acres being the primary tool for measuring the success of mitigation wetlands nationwide, the ecological performance of mitigation wetlands is largely unknown. Few studies examine the ecological functioning of mitigation wetlands.

The purpose of this study was to evaluate the ecological performance of Iowa Department of Transportation (DOT) mitigation wetlands. Specific objectives of the study were to:

1. Quantify biological diversity at mitigation and reference wetlands.
2. Determine if mitigation and reference wetlands are functioning differently.
3. Determine environmental factors influencing biological diversity.

METHODS

Prior to commencing work on the project, a Quality Assurance Project Plan (QAPP) was prepared and submitted to United States Environmental Protection Agency (EPA) Region 7 for approval. The final QAPP is included in Appendix A.

Study Sites

A total of 15 study sites were evaluated for ecological performance. The location of each site is shown on Figure 1. Both mitigation and reference wetlands were selected using a restricted random method (Hayek and Buzas 1997). Mitigation wetlands (n=12) were restricted based on total size of the site, age of the site (year of construction), and type of construction (restoration versus creation) (Table 1). Reference wetlands (n=3) were restricted based on total size of the site and proximity to mitigation wetlands so that reference wetlands were located in same geographic region as mitigation wetlands (Table 1).

Prior to data collection, wetland areas were delineated at each study site using standard methods defined in the Corps of Engineers Wetlands Delineation Manual (Environmental Laboratory 1987). Field delineations were conducted during August and September 2003 and from May through August 2004. Wetland boundaries were mapped using a Trimble GeoExplorer CE® Global Positioning System (GPS) receiver. Data from the receiver were post-processed using Trimble Pathfinder Office® version 3.00 software for an accuracy of <1 meter.

Data Collection

Field surveys were conducted during the 2005 and 2006 field seasons. The sites were randomly divided and surveys were conducted as follows:

1. 2005 Survey Sites – Jarvis, Grooms, New Hampton, Palisades, Pleasantville, South Point, Hay-Buhr and Engeldinger

2. 2006 Survey Sites – Badger, Boevers, Brush Creek, Dike, Mink, Wickiup Hill, Doolittle Prairie

A 50 m wide survey zone, ranging from 100 to 300 m long, was established at each site (Figures 2a – 2o). At mitigation wetlands, the survey zone was positioned so that it included both the constructed wetland and adjacent upland area at an individual site. Existing wetlands that had been present on a site prior to construction of the mitigation wetland and were preserved as part of the mitigation package were intentionally not included in the survey zone. At reference sites, the survey zone was positioned so that it included both representative wetland and adjacent upland habitats.

Field surveys were conducted primarily within or immediately adjacent to the 50 m survey zone; however, in order to gain a complete picture of the biological diversity of a site, supplemental surveys were conducted outside of the 50 m survey area.

<u>Biological Diversity</u>

Intensive biological inventories were used to evaluate the biological condition at each of the 15 study sites. Species richness and abundance data were collected at each site for nine species groups:

1. Algae
2. Protozoa

3. Aquatic Invertebrates
4. Vascular Plants
5. Butterflies
6. Amphibians
7. Reptiles
8. Birds
9. Mammals

Voucher specimens were taken for difficult identifications under current Iowa Department of Natural Resources Scientific Collecting Permits. Voucher specimens collected for this project were processed following standard methods for each species group (Heyer et. al. 1994; Wilson et. al. 1996; APHA 1998; Deblase and Martin, 2000; Winter 2000; Simmons 2002; US EPA 2002a). Specific survey methods are summarized below. Detailed methods are found in Appendices B - G.

Algae, protozoa, and aquatic invertebrates. Field collection of algae, protozoa, and aquatic invertebrates was done using benthic and surface grab sampling and net sampling for plankton (APHA 1998; US EPA 2002a). Figures 2a – 2o show the location of sampling points at each site. Each of the study sites was sampled over a period of three days during five sampling periods between April and November. An average of four samples was collected at each site on each sampling day and samples from similar habitats at any one site were pooled. Sample analysis included microscopic examination of fresh (or settled) samples and digestion and preservation of diatom samples (APHA 1998).

Vascular Plants. Two methods were employed at each site to survey vascular plants. A floristic inventory that encompassed the entire study site was completed to identify and delineate extant plant communities and develop a qualitative plant species list for each site. Floristic data were collected over the entire growing season during at least three visits to each site. Community maps and species lists were refined during repeated field surveys. The goal of the floristic method was to produce an extensive survey of the site's vegetation, identify its landscape components and community diversity, and generate a plant list for the entire site.

Quantitative methods were used to measure plant species abundance along pre-determined transects with the goal of accurately measuring plant species abundance in the plant communities that represented the mitigated or natural wetland habitat at the site.

Three parallel transects were established within the 50 m wide survey zone beginning at the 10, 25 and 40 m positions (along one of the 50 m ends) (Figures 2a – 2o). Along each transect, two 1x1 m quadrats were randomly located within each 10 m segment, one on each side of the transect and with one side contiguous with the transect. The 1x1 m quadrats were used to measure abundance of all herbaceous species and woody stems less than 50 cm tall. Within each 1x1 m quadrat, four 25x25 cm subquadrats were established, one in each corner of the 1x1 m quadrat. Density of ramets (graminoid tillers, plant shoots, or caudices of acaulescent species) was determined for each species present in the subquadrats. In addition to the density measurements, frequency of plant species was also determined. Frequency was measured at two scales – species presence within the 25x25 cm subquadrats and species presence within the 1x1 m quadrats. A few species, most notably submergent and floating species, could not be assessed with density of ramets. For these species (e.g., *Lemna*, *Potamogeton*, *Ceratophyllum*, *Najas*, *Elodea*, Bryophyta), the percentage area of plant coverage in the quadrat was recorded. The plant community (consistent with the floristic study) at each 1x1 m quadrat was recorded.

The density of shrubs (woody stems greater than 50 cm tall and less than 2 m tall) was measured along each of the three parallel transects using a series of 2x10 m plots. The plots were centered over the transect and placed end to end for the length of the transect. The density of both saplings (woody stems greater than 2 m tall and less than 5 cm diameter at breast height [DBH]) and trees (woody stems greater than or equal to 5 cm [DBH]) was measured in a series of 10x30 m plots established between the two outermost transects (i.e., at the 10 and 40 m positions). These plots were placed side by side (contiguous) for the length of the transect. The DBH of trees was also recorded. A tree was considered to be multiple-stemmed if there was more than one trunk (or low branch) present at breast height. All the stems of multiple-stemmed individuals were measured.

Butterflies. Butterfly data were collected on the species present at each study site every 10 days during the non-frost season (April – October). Standard methods included meander surveys of all habitats and counts of all individuals of each species encountered (Pollard 1991; Pollard and Yates 1993).

Vertebrates. Amphibians were surveyed from April through June to insure sampling of all species present (Heyer et al., 1994; US EPA 2002b). Salamanders were trapped April through June using wire screen funnel traps and hand collected during terrestrial and aquatic searches throughout the surveys. Figures 2a – 2o show tapping locations. Calling surveys and hand collecting of frogs and toads were conducted primarily from April through June during the time that each species is known to breed and continued throughout the survey.

Reptiles were surveyed from May through mid-July when they are most active. Snakes and lizards were documented through meander surveys of the study sites. Aquatic turtle trapping was conducted in all permanent bodies of water possessing suitable turtle habitat using modified fyke nets as described by Legler (1960). Figures 2a – 2o show turtle trapping locations.

The most intensive surveys for amphibians, reptiles and small mammals utilized drift fences as described by Christiansen and VanDeWalle (2000). Drift fence locations and placement of Sherman live traps is shown on Figures 2a – 2o. Drift fence sampling took place from May through September. Sherman live traps were used in conjunction with the drift fence for small mammals.

Migratory birds were surveyed during March and November and breeding bird surveys took place from May through July (Fairbairn and Dinsmore 2001; US EPA 2002c).

Environmental Conditions: Chemical and Physical

In addition to collecting data to evaluate biological condition, abiotic factors having the potential to influence biological diversity were also studied (Table 2). Water quality was assessed as well as physical characteristics (local and watershed level) at each study site. Methods used to perform these assessments are summarized below and detailed methods are found in Appendices F and G.

Water Quality. If surface water was present, a site was sampled every two weeks between May and August and once in either October or November. If surface water was not present at a site during the entire survey period, sampling was limited to those time periods when surface water was present. Grab samples were collected from inlets and outlets when identifiable; otherwise, a grab sample was obtained from a representative location in the wetland.

A YSI Model 556 Multiprobe System was used to measure dissolved oxygen, temperature, pH, and conductivity in the field. The instrument was calibrated each day prior to measurements. A Hach 2100P Turbidimeter was used for turbidity measurements. Calibration was checked each day with Hach Gelex secondary standards. All field equipment exposed was rinsed three times with deionized water after sampling.

In addition to field measurements, water samples were collected just below the surface of the water directly into sample bottles and field-rinsed with sample twice before collection at each site. A 50 mL sample was filtered in the field through a 0.45 mm filter for dissolved reactive phosphorus analysis. All samples for laboratory analysis were immediately stored in a cooler with ice packs until they were transferred to a refrigerator at 4°C. Samples were analyzed the day after collection.

Ion chromatography (Hautman and Munch, 1997) was utilized to measure chloride, nitrite, nitrate, and sulfate concentrations. Spectroscopic methods were used to measure ammonia (Hach 2004a) and dissolved reactive phosphorus (Hach 2004b). Total phosphorus (Hach 2004c) and total nitrogen (Hach 2004d) were measured using a persulfate digestion prior to colorimetric analysis. Dissolved organic carbon was initially assessed using a manganese COD digestion with spectroscopic measurement (Hach 2004e); later measurements used a more sensitive chromium based technique (Hach 2004f). Spectroscopic analyses were carried out on Perkin Elmer EZ150 spectrophotometers and ion chromatographic analyses were carried out on a Dionex DX-80. All chromatographic and spectroscopic analyses utilized a minimum of four standards prepared by dilution of a purchased stock solution (Hach stock solutions for the spectroscopic analyses; Dionex seven-anion standard for the ion chromatographic analysis). Any other reagents used were of reagent grade or higher.

Landscape Assessment. A Geographic Information Systems (GIS) analysis was used to quantify landscape characteristics at three scales. The local watershed of each study site was delineated using 1999 National Elevation Data with a horizontal resolution of 30 meters and a vertical resolution of 15 meters provided by the U.S. Geological Survey and the EROS Data Center in Sioux Falls, South Dakota. Iowa Department of Natural Resources Watershed Initiative Data and Natural Resources Conservation Service 2003 Hydrologic Unit Code (HUC) 12 watersheds were used as base watersheds. Terrain Analysis using Digital Elevation Models (TauDEM), a third-party ArcMap extension developed and distributed by David Tarboton at Utah State University, was used as the watershed-modeling engine.

For each watershed, sediment and nutrient loads were calculated using methods developed by the EPA (US EPA 2002d). In addition, sediment risk was based on the amount of agricultural land cover as well as soil properties. Sediment risk was derived from NRCS, and Iowa Department of Natural Resources-Iowa Geological Survey 1998 Highly Erodible Soil (HEL) data and the same Agricultural land classification as the nutrient load calculations.

Further assessment included classifying land use and land cover at two different spatial extents using remotely sensed imagery in ArcMap (ESRI 2005). The first extent comprised the area within 300 m of the wetland edge and involved quantification of landscape features at a relatively fine grain. This distance was based on the area thought to serve as core habitat for pond-breeding herpetofauna (Semlitsch and Bodie 2002), one of the nine species groups for which species richness and abundance data were collected. In addition, dominant land uses and broad categories of land cover were quantified at a second extent, a 2 km radius of each study site. Ground-truthing of the landscape classifications was conducted from May through July, 2006 and from September through October, 2006.

The intensity of human land use based on the energy use per unit area was measured for each study site (Brown and Vivas 2005). In this method, the intensity of land use is compared to that in an undeveloped landscape and expressed as the Landscape Development Index (LDI). Energy use is weighted depending on factors such as whether or not it is a renewable source. Land use types such as residential and commercial consume more non-renewable energy than land cover types such as pasture. The intensity of all land cover/use types are scaled in reference to natural landscape types, which consume zero energy. LDI calculations were based on land use/land cover within the 2-km buffer of each site.

Site Assessment. In addition to landscape assessment, site specific data were quantified to further describe on-site features having the potential to influence biological diversity. GIS analysis was used to quantify a number of site specific characteristics including, interior edge length, the ratio of interior edge to site area, total edge length, the ratio of total edge to site area, average community size and interspersion (i.e. number of distinct communities at a site).

Data Analysis

Diversity at mitigation and reference sites was quantified using Hill's N1 (Hill 1973) as a representative measure of species diversity. Hill's N1 is given by:

$$N1 = \exp(-\sum p_i \ln(p_i)$$

where p_i is the proportion of a given species found at a site. N1 is one method of calculating the "effective number of species" (MacArthur 1965; Hill 1973). It is the exponential of the Shannon index; unlike Shannon's index, Hill's N1 represents a true diversity that behaves linearly and is therefore easier to interpret ecologically than the Shannon form (Peet 1974). Because it is derived from Shannon's index, it also has the advantage of not emphasizing either rare or common species (Jost 2006).

Species diversity of mitigation sites versus reference sites was compared using the Mann-Whitney two-sample rank-sum test (Mann and Whitney 1947) to determine if mitigation wetlands are performing differently than reference wetlands.

Because of the differing number of mitigation sites (n=12) and reference sites (n=3), species richness of mitigation sites versus natural sites was compared using expected species accumulation curves, i.e., sample-based rarefaction curves (Gotelli and Colwell 2001). The curves were calculated using EstimateS Version 8 (Colwell 2006). This program calculates the expected species accumulation and its associated 95 percent confidence intervals using the methods of Colwell et al. (2004).

For each of the major groups of organisms, observations of species abundance for all mitigation sites were amalgamated into one dataset, and data for reference sites were amalgamated into another. As recommended by Gotelli and Colwell (2001), the expected species accumulation curves and their 95 percent confidence interval curves by individuals were rescaled. By comparing the curves for each group of organisms, species richness between the two groups of sites could be compared based upon the actual number of individuals recovered.

Water quality data were used to compare the overall water quality of study wetlands. For each parameter measured, t-tests were conducted to determine whether or not statistically significant differences existed between mitigation and reference wetlands.

Relationships between effective numbers of species and measured environmental variables (i.e. water quality, were examined using one-way analysis of variance for categorical variables (i.e. restoration type, age class, etc.) and Spearman rank correlation coefficients for other variables.

Several different ordination methods were used and are described in detail Appendices B – G. Ordination attempts to represent sample and species relationships, typically in a two-dimensional space in which similar species or samples are near each other and dissimilar species or samples are far apart (Gauch 1982). Correspondence analysis (CA), an indirect ordination method, was used to evaluate the variation in animal taxa and study sites. Canonical correspondence analysis (CCA; ter Braak 1987), a direct ordination method, was also used to evaluate the variation in animal taxa and study sites, particularly with respect to measured environmental variables. Detrended correspondence analysis (DCA), an indirect ordination method, was used to evaluate the variation in plant taxa and study sites.

RESULTS

Biological Diversity

Over 1,600 species comprising the nine species groups were identified during the study at mitigation and reference sites (see Appendix H for complete list of species). Slightly over 800 species were found at mitigation wetlands and slightly less than 800 species were found at reference wetlands. Of the over 1,600 total species, over half were bacteria, algae, protozoans and aquatic invertebrates. Vascular plants contributed over 500 species.

A summary of species diversity by study site is shown in Table 2 and Figures 3 – 5. As a way of comparing diversity between the 15 sites, overall diversity was calculated using the effective number of species (Hill's N1) for each of the nine species groups to determine an average rank for each site. The sites were then given an overall ranking of 1 – 15 based on the average rank, with 1 representing the highest overall species diversity (Table 2 and Figure 6).

The highest ranking site in terms of overall diversity was South Point, a large (16 ha), created mitigation site that was one year post construction at the time it was surveyed (Table 2 and Figure 6). South Point had the highest diversity of algae, aquatic invertebrates and native herbaceous plants of any site (Table 2 and Figures 3 and 4), the second highest amphibian diversity and the third highest diversity of butterflies and reptiles (Table 2 and Figure 5).

One of the reference sites, the Hay-Buhr Area, a large (46.5 ha) natural wetland, ranked second in overall diversity (Table 2 and Figure 6). The Hay-Buhr Area had the highest diversity of protozoa and reptiles (including one state-listed threatened, one state-listed endangered and one state-listed special concern species) and the second highest diversity of native herbaceous plants, birds (including two state listed endangered species) and mammals (Table 2 and Figures 3 – 5). Although the site ranked high in most animal groups, butterfly and amphibian diversity were low at Hay-Buhr (Table 2 and Figure 5). Even though butterfly species richness is low at the site, the lepidopteran species that are present include a number of wetland obligate species and species that are associated with higher quality wetland habitats.

The site ranking the lowest (15th) in overall diversity was one of the reference sites, Doolittle Prairie (Table 2 and Figure 6). Doolittle Prairie is a small (10.5 ha) native tallgrass prairie remnant with a series of small prairie potholes located across the site that was dedicated as one of Iowa's State Preserves in 1980. Portions of the site have never been plowed, which is reflected in the site having the third highest diversity of native herbaceous plants (Table 2 and Figure 4). However, Doolittle Prairie scored at or near the bottom in every other species group, including the lowest amphibian diversity and the second lowest butterfly diversity (Table 2 and Figure 5). With respect to protozoa and aquatic invertebrates, the only sites with lower diversity than Doolittle Prairie were sites that were dry all, or a large portion, of the year in which they were sampled.

The other reference site, Engeldinger Marsh, ranked sixth in overall diversity (Table 2 and Figure 6) and ranked near the middle in the majority of the species groups, with a couple of notable exceptions. The site ranked the highest (1st) in butterfly diversity with a number of wetland indicator species present that are indicative of high quality wetland habitats. On the other end of the spectrum, the site ranked the lowest (15th) in reptile diversity. This is likely the result of intense agricultural use of the site in the past and the presence of a state highway that bisected the site until recently.

The lowest ranking mitigation site was Jarvis, which ranked 14th in overall species diversity (Table 2 and Figure 6). Jarvis had the lowest diversity of algae, protozoa, aquatic invertebrates, native herbaceous plants and mammals (Table 3 and Figures 3 – 5). The site lacked water during the majority of the year in which it was sampled, resulting in no algae, protozoa or aquatic invertebrates being collected at the site. Nevertheless, surprisingly, Jarvis ranked third in amphibian diversity (Table 2 and Figure 5). One possible explanation for this is the site's location along the Skunk River, which likely provides suitable habitat for a number of anuran species that were heard calling during the survey. The site also ranked surprisingly high for butterfly diversity (4th) (Table 2 and Figure 5) and had several wetland-associated species present. As with amphibians, it is likely that colonization of the site by butterflies may occur from areas to the west and south of the site.

When the effective number of species (Hill's N1) by species group at mitigation sites is compared to that found at reference sites, no significant differences are found within any of the groups (Table 2); however, the *p*-value for native herbaceous plants is suggestive that diversity is higher at reference sites (Table 2). Because the effective number of species is a measure of the number of common species at a site, the lack of a significant difference in effective number of species between mitigation and reference sites suggests that the number of common species within each species group is approximately equal between mitigation and reference sites.

In an effort to further explore the question of whether mitigation sites are performing differently than reference sites, the species richness of mitigation sites versus reference sites was compared using expected species accumulation curves (Figures 7a – 7i).

The species accumulation curves for algae, protozoa, and aquatic invertebrates all show similar patterns (Figures 7a – 7c). Based on the numbers of individuals recovered, the 95 percent confidence intervals for the mitigation sites overlap those of the reference sites for all three groups of organisms. This indicates that insufficient evidence exists to reject the null hypothesis of no significant difference in species richness between the two types of sites given comparable sample sizes. In addition, as more individuals are recovered, the number of species for both mitigation and reference sites do not appear to be converging to an asymptote, indicating that many additional species remain to be recovered. For algae, rarefaction of the mitigation site curve to a sample size of about 2000 individuals (the total for the pooled reference sites) suggests that when the number of individuals recovered is taken into account, "rarefied" species richness at the two types of sites is approximately equal at 200. For aquatic invertebrates at a sample size of about 110 individuals (the total for the pooled reference sites) it is approximately equal at 29 species. Based on a sample size of about 240 individuals (the total for the pooled reference sites), "rarefied" species richness for protozoa ranges from approximately 63 to 69 species at the two types of sites.

The species accumulation curves for native herbaceous plants based on relative frequency are similar to those for algae (Figures 7d). Based on the numbers of individuals recovered, the 95 percent confidence intervals for the mitigation sites overlap those of the reference sites, indicating that insufficient evidence exists to reject the null hypothesis of no significant difference in species richness between the two types of sites given comparable sample sizes. In addition, as more individuals are recovered, the number of species for both mitigation and reference sites do not appear to be converging to an asymptote, indicating that many additional species remain to be recovered. Rarefaction of the mitigation site curve to a sample size of about 6700 individuals (the total for the pooled reference sites) suggests that when the number of individuals recovered is taken into account, "rarefied" species richness at the two types of sites is approximately equal at 160.

The species accumulation curves for butterflies (Figure 7e) indicate that "rarefied" species richness is significantly higher at the mitigation sites than it is at the reference sites. The 95 percent confidence intervals for the two groups of sites diverge at about 1,800 individuals, providing evidence sufficient to reject the null hypothesis of no significant difference in species richness between the two types of sites. The curves for both groups of sites appear to converge to asymptotes (approximately 65 species at mitigation sites and approximately 40 species at reference sites, a difference of 63 percent). Rarefaction of the mitigation site curve to a sample size of about 1,900 individuals (the total for the pooled reference sites) suggests that "rarefied" butterfly species richness ranges from approximately 38 species at reference sites to approximately 46 species at mitigation sites, a difference of 21 percent. This result, in combination with the finding of no difference in the effective number of species between the two groups of sites, suggests that the difference in species diversity is due to the presence of significantly more rare species at the mitigation sites.

Among vertebrate taxa, the species accumulation curve for amphibians (Figure 7f) at mitigation sites is noteworthy because beginning at a sample size of about 1,000 individuals it converges to an asymptote of 11 species, suggesting that all available species have been found at this group of sites. The 95 percent confidence intervals for the mitigation sites overlap those of the reference sites, indicating that insufficient evidence exists to reject the null hypothesis of no significant difference in species richness between the two types of sites. However, at a sample size of about 190 individuals, the curve for reference sites shows signs of beginning to converge to an asymptote at an undefined level lower than that noted for the mitigation sites. This suggests that although additional species remain to be recovered at the reference sites, additional sampling at the reference sites could cause the curves to diverge, with the reference sites possibly being less diverse than the mitigation sites. Rarefaction of the mitigation site curve to a sample size of about 190 individuals (the total for the pooled reference sites) suggests that "rarefied" amphibian species richness ranges from approximately six to seven species between the two types of sites.

The species accumulation curves for reptiles (Figure 7g) are somewhat similar to those for mammals (Figure 7i). No significant difference in species richness was detected between the two types of sites given comparable sample sizes. The curve for reptiles at the mitigation sites appears to be converging to an asymptote, thereby suggesting that most species have been recovered. The curve for reference sites is not converging to an asymptote, indicating that most likely only the most common species have been found and that many additional species probably remain to be recovered. In addition, the 95 percent confidence intervals for the reference sites are very wide, ranging from five to 15 species at a sample size of 21 individuals (the total number recovered from all of the reference sites). This reflects both the small sample size and the high variability in observed reptilian species richness at the reference sites (one species at Engeldinger Marsh, nine at Hay-Buhr, and two at Doolittle Prairie). Rarefaction of the mitigation site curve to a sample size of 21 individuals (the total for the pooled reference sites) suggests that "rarefied" reptilian species richness ranges from approximately seven to 10 species between the two types of sites.

The species accumulation curves for birds (Figure 7h) exhibit patterns similar to those noted for algae, protozoa, aquatic invertebrates and native herbaceous plants. No significant difference in species richness was detected between the two types of sites given comparable sample sizes, and many additional species probably remain to be recovered. Rarefaction of the mitigation site curve to a sample size of about 575 individuals (the total for the pooled reference sites) suggests that "rarefied" avian species richness ranges from approximately 54 to 62 species between the two types of sites.

The species accumulation curves for mammals (Figure 7i) exhibit a slightly different pattern than that noted for birds. No significant difference in species richness was detected between the two types of sites given comparable sample sizes, but the curve for mammals at the mitigation sites appears to be converging to an asymptote of about 25 species, thereby suggesting that all common and most rare species have been recovered. The curve for reference sites does not appear to converge to an asymptote, indicating that many additional species probably remain to be recovered. Rarefaction of the mitigation site curve to a sample size of about 330 individuals (the total for the pooled reference sites) suggests that "rarefied" mammalian species richness ranges from approximately 14 to 16 species between the two types of sites.

Plant species composition was significantly different between mitigation and reference wetlands with respect to the native richness index (p = 0.01), mean native conservatism (p <0.001), and mean wetland indicator status (p = 0.01), indicating that reference wetlands are of higher quality than mitigation wetlands with fewer exotic species, more pristine environments, and species with greater wetland affinity.

Mitigation wetlands had significantly higher ramet (individual plants in a clump, each portion of which is identical with the original parent plant) densities (p = 0.032) due to early successional status. Mitigation wetlands were more species rich (p = 0.014), but reference wetlands were more even (p = 0.055). Three relationships were identified as especially important when comparing mitigation and reference wetlands: 1) reference wetlands had significantly more importance of natives and less exotics than mitigation wetlands (p = 0.030), 2) reference wetlands had significantly more importance of wetland indicators than mitigation wetlands (p = 0.001), and 3) reference wetlands had significantly more importance of *Carex* species than mitigation wetlands (p = 0.013).

Plant data were ordinated using Detrended Correspondence Analysis (DCA). The results indicate that mitigation and reference wetlands are grouped in different parts of the ordination space. The fact that mitigation sites grouped together suggests that these sites are relatively homogeneous when compared to reference wetlands.

No significant relationships were found when plant data were compared with either size or age of mitigation wetlands. However, small sites had species with significantly greater wetland affinity, suggesting that small sites may provide for higher quality wetland restorations.

It is interesting to note that some mitigation sites contained species either not native to Iowa or out of range for the proper area of the state, apparently a product of seeding during the wetland reconstruction process. As noted above, exotic species in mitigation wetlands have likely hindered ecological performance. Therefore, Iowa DOT should review wetland seed mixes for proper design.

Environmental Conditions: Chemical and Physical

Water Quality Assessment

A comparison of the means of the 13 water quality parameters examined shows a number of differences between reference wetlands and the mitigation wetlands (Table 3). For each parameter listed in Table 3, t-tests were conducted to determine whether or not statistically significant differences exist between the two groups. It should be noted that Brush Creek was excluded from these analyses due to unusual water chemistry at the site resulting from a sewage treatment plant located upstream of the site. Averages of pH, turbidity, total suspended solids (TSS) and ammonia (NH_3) at mitigation wetlands were higher than averages observed at reference wetlands, while no statistically significant difference in nutrient levels was found between the two wetland types. Given the importance of nitrate as a pollutant in Iowa, it is interesting to note very similar concentrations of NO3- were found in both mitigation and reference wetlands.

Water quality parameters were compared with biodiversity data to test for any relationships between water quality and species diversity. Significant (p = 0.05) relationships were found between butterflies and total nitrogen, between reptiles and both dissolved reactive phosphate and SO_4, and between algae and total nitrogen. A highly significant (p = 0.01) relationship was found between amphibians and dissolved oxygen.

Levels of dissolved reactive phosphate (DRP) and nitrate-nitrogen (NO_3^--N) were used to calculate a nutrient score (these were selected instead of total phosphorus and total nitrogen because the total unfiltered values may be significantly affected by the presence of algae or other plant life in the samples). It should be noted that Jarvis and Grooms were excluded from these analyses because both sites were dry during the sampling period and no water samples were obtained. To give approximately equal weighting to phosphorus and nitrogen inputs, the DRP value was multiplied by 4 and added to the nitrate value (Nutrient score = (DRP x 4) + NO_3^-N). The results are shown in Table 4 and Figure 8).

The sites clearly divide into two groups, with the seven of the sites with a nutrient score of >4.0 and six of the sites with a nutrient score of <2.0. The sites scoring above 4.0 all have significant nutrient inputs, either from agricultural or point sources, while those scoring <2.0 tend to be more isolated geographically and have fewer direct inputs (see nutrient load discussion below). Wickiup Hill had the lowest nutrient score (0.88) and Doolittle Prairie had the highest (12.0). Two reference wetlands (Doolittle Prairie and Hay-Buhr) are found in the group with scores >4.0 and the remaining reference wetland (Engeldinger) is found in the group with scores <2.0 (Table 4 and Figure 8). The mean nutrient score was 4.1 for mitigation wetlands and 6.6 for reference wetlands. No significant difference was found between the mean nutrient score of mitigation wetlands and reference wetlands.

It is of interest to examine the efficacy of nutrient removal by the study sites. Removal of nitrate is of particular interest, because Iowa rivers have among the highest levels of this nutrient in the nation (Goolsby et al. 1999) and wetlands are often touted as potential treatment options. The efficiency of nitrate removal varied considerably by site and by date (Figure 9). Each of these sites featured one or more inflows of water and a well-defined outlet. Nitrate removal is generally believed to be dependent on the concentration of nitrate in the inflow and the hydraulic retention time (Toet et al. 2005). Of the wetlands studied, Dike was most effective at nitrate removal. The wetland received runoff directly from a waterway draining a cornfield with consistently high concentrations of nitrate. The wetland was relatively large, and drained into a culvert opposite the inflow. This configuration resulted in removal of over 50% of the nitrate concentration during some parts of the summer.

Landscape Assessment

Land Use/Land Cover. The local watershed for each mitigation site includes the area in the immediate vicinity of the site. Local watersheds were highly variable, ranging from 4 ha (Boevers) to 590 ha (Brush Creek). The reference wetland watersheds were 165 ha (Hay-Buhr), 110 ha (Engeldinger), and 59 ha (Doolittle) in size.

When considering an area larger than the local watershed, the landscape surrounding most mitigation and reference sites was dominated by agricultural lands, particularly row crops (Table 5). Within a 2 km buffer, the percentage of agricultural lands ranged from 34% at Wickiup Hill to 89% at Dike. The percentage of wetland within a 2 km buffer was also variable, ranging from 0.3% at Doolittle Prairie to

26% at Wickiup Hill. The percentage of natural areas within 2 km ranged from 2% at Dike to 55% at Wickiup Hill. Interestingly, Wickiup Hill had the highest percentage of wetland, the lowest percentage of agriculture, and the highest percentage of natural land within a 2 km buffer (Table 5).

Nitrogen and phosphorous loadings for all sites ranged from 1.27 to 2.23 and 1.64 to 3.65 respectively. Boevers, with 73% of the land within 2 km of the site in agriculture (Table 6), had the greatest overall nutrient loading. Wickiup Hill, with only 34% of the land within 2 km of the site, had the lowest overall nutrient loading. The mean nutrient loading was 2.42 for mitigation wetlands and 2.27 for reference wetlands. No significant difference was found between the mean nutrient loading of mitigation wetlands and reference wetlands.

Sediment load ranged from 0.0 to 0.46, with Boevers and Mink Creek having the lowest values, primarily due to the lack of Highly Erodible Lands (HEL) within their local watersheds and Brush Creek having the highest value (Table 6). The mean sediment load was 0.14 for mitigation wetlands and 0.02 for reference wetlands. Although reference wetlands have a lower sediment load on mean than mitigation wetlands, the difference is not significant.

Landscape Development Index (LDI), Habitat Development Index (HDI), Road Density, and Connectivity. The landscape development index (LDI) estimates impacts to natural ecosystems from human-development activities, combining land use data with a development-intensity measure derived from energy use per unit area. The index ranges from 0-10, with 0 representing virtually no human-induced impacts and 10 representing a fully developed landscape dominated by high-energy land uses (Brown and Vivas 2005). The LDI for all sites ranged from a low of 2.62 at Wickiup Hill to a high of 5.33 at New Hampton. Reference site LDI scores were 3.41, 4.1, and 3.86 for Hay-Buhr, Engeldinger, and Doolittle Prairie respectively. Mean LDI scores were 3.6 for mitigation wetlands and 3.8 for reference wetlands. No significant difference was found between the mean LDI score of mitigation wetlands and reference wetlands (Table 5).

A habitat diversity index (HDI) using plant data was calculated for each study site (Table 5). The index combines the Simpson and Shannon diversity indices, and is based only on the number and total area of community types (see Appendix C for detailed description). HDI increases as plant communities at a site increase and become more proportional in terms of area. Habitat diversity index scores were highly variable, ranging from 1.59 at Mink Creek to 17.25 at Wickiup Hill. Reference wetland HDI scores were 6.9, 10.9, and 3.3 for Hay-Buhr, Engeldinger, and Doolittle Prairie respectively. The mean HDI score was 6.3 for mitigation wetlands and 7.0 for reference wetlands. No significant difference was found between the mean HDI score of mitigation wetlands and reference wetlands (Table 5).

Road densities within 2 km ranged from 9 m/ha at Grooms to 22 m/ha at Brush Creek. Mean road density for mitigation wetlands was 14.4 m/ha and 14.2 m/ha for reference wetlands. No significant difference was found between the mean road density of mitigation wetlands and reference wetlands (Table 5).

Connectivity is a measure of how connected a site is to other natural habitats. For the purposes of this study, connectivity was expressed as the percentage of natural land within a 2 km buffer of a site. Connectivity ranged from 2% at Dike to 55% at Wickiup Hill, with mean connectivity of 26% for mitigation wetlands and 20% for reference wetlands (Table 5). No significant difference was found between the mean connectivity of mitigation wetlands and reference wetlands (Table 5).

Ordination. Animal, environmental, and site data were ordinated using Correspondence Analysis (CA) (see Appendix G for details). The ordination found that axes 1 and 2 explained 58.9% and 17.6% of the variation in animal species composition respectively. The reference wetlands showed little variability along the first axis. Using Canonical Correspondence Analysis (CCA) with these same data, axes 1 and 2 explained just under 30% of the variation in animal species composition, suggesting that important sources of variability were not captured by the chosen environmental variables. However, three significant environmental variables were identified including, 1) the intensity of row crop agriculture within 2 km, 2) the amount of grassland within 300 m, and 3) the amount of wetland within 300 m. The first axis was positively correlated with potential habitat in the immediate area and negatively correlated with row crops within 2 km (see Figure 2 Appendix G).

Site Characteristics. A number of site characteristics were investigated for differences between mitigation and reference wetlands and for any potential relationship to biodiversity (Table 7). No significant differences were found between mitigation and reference wetlands with respect to any of the site characteristics listed in Table 7. In addition, no significant relationships were found among site characteristics and biodiversity measurements.

DISCUSSION

Many biotic and abiotic factors can affect the ecological performance of a wetland, including geologic history, adjacent land use/disturbance, connectivity to other habitats and time. Data from this study suggest that ecologically, mitigation sites in Iowa are functioning similarly to reference sites. No significant difference in species diversity between mitigation and reference sites was detected. However, the lack of convergence to an asymptote in many of the species accumulation curves suggests that for the most part sampling for many of the sites/species groups (particularly the reference sites) is effectively incomplete, which may explain the inability to demonstrate differences in the effective number of species (Hill's N1) at mitigation versus reference sites. A true difference in species diversity may exist between the two groups of sites; however, due to small sample size and/or a lack of a sufficiently powerful statistical test, a true difference may have gone undetected.

Although, a difference in species diversity between mitigation and reference sites was not detected, differences between individual mitigation and reference sites are apparent from the data. The starkest contrast is between the newly constructed South Point with the highest overall species diversity and Doolittle Prairie State Preserve, which had the lowest overall species diversity (Table 2 and Figure 6). One possible explanation for the difference in diversity between the newly constructed South Point and the remnant wet prairie at Doolittle is the connectivity of each site to other suitable habitat. South Point has a direct connection to a 2,630 ha wildlife area located along the Skunk River, which is home to a diverse collection of woodland, wetland and prairie wildlife. Approximately 43% of the land within 2 km of South Point is natural land, with 8% of that classified as wetland (Table 5).

In contrast, Doolittle Prairie is located in a highly agricultural part of the state and is surrounded by intensively cropped agricultural land. Approximately 83% of the land within 2 km of Doolittle is in agriculture and only 0.3% is classified as wetland (Table 5). The fencerows that formerly bordered the site have been removed and the adjacent fields are plowed right to the edge of the prairie. No direct connection to any large area of natural habitat exists.

Within species groups, a significant difference in species richness between mitigation and reference sites was found only with the butterflies (Figure 7e). No significant difference in the effective number of butterfly species between the mitigation and reference sites was found (Table 2), indicating that both groups contain approximately equal numbers of common species. The 21 percent difference seen in "rarefied" butterfly species richness between mitigation and reference sites therefore suggests that the difference in species diversity is the result of more rare species being found at the mitigation sites. The underlying reason(s) for why mitigation sites may be able to harbor a larger assemblage of butterfly species is beyond the scope of this study, but may be related to such factors as plant species diversity, plant community types, connectivity to other suitable habitat, and/or management.

Although no significant difference in plant diversity as measured by effective number of species was detected, vegetation differences between mitigation and reference wetlands do exist. Specifically, 1) Vegetation at the mitigation wetlands is not of the same quality as that found at reference wetlands, even in sites up to eight years old. Abundant exotic species combined with the lack of *Carex* and other important wetland species seems to be a central difference between mitigation and reference wetlands; and, 2) Mitigation wetlands exhibit very few discernible patterns in vegetation characteristics that could be attributed to size or age effects. For the most part, mitigation wetlands are homogenous, as indicated by the lower mean HDI score for mitigation wetlands (Table 5), whereas reference wetlands tend to have more community heterogeneity. The only effect observed was that small sites have species compositions with greater wetland affinity than do large sites, suggesting that better mitigation can be achieved if the size of the site is kept relatively small.

In general, the study sites exhibited values for the water quality parameters analyzed which are typical of Midwestern surface waters. Compared to all monitoring (streams, lakes, and wetlands) carried out by the Iowa Geological Survey Bureau from 2000-2006 (IGSB 2007a), ammonia, dissolved reactive phosphorus, and total phosphorus averages in this study were at the 75^{th} percentile or above. Chloride, nitrate, total suspended solids, and turbidity averages were in the 25^{th} to 50^{th} percentile of this data set, and the sulfate mean was around the 10^{th} percentile for this time period.

Results were also consistent with a study of Iowa wetlands being carried out by the Iowa Geological Survey Bureau (ISGB) during the same time period (IGSB 2006, IGSB 2007b). In 2005, the ISGB sampled 60 sites in north-central Iowa and resampled 32 of the sites in 2006 along with an additional 40 new sites in north-central Iowa. In 2005, nitrate+nitrite concentrations ranged from 0.05 to 27 mg NO_3^--N/L with a mean of 6.2, total phosphorus ranged from 0.05 to 1.2 mg/L with a mean of 0.27, and dissolved reactive phosphorus (or orthophosphate) ranged from 0.02 to 0.72 mg/L with a mean of 0.16. In 2006, the nitrate range was 0.05 to 9 mg NO_3^--N/L (mean 3.55), the total phosphorus range was 0.05 to 3.1 mg/L (mean 0.38 mg/L), and the dissolved reactive phosphorus range was 0.02 to 0.94 mg/L (mean 0.11 mg/L).

The results reported in this study and the IGSB study both illustrate the spatial and temporal variability found in water quality in wetlands. Nutrient inputs to the wetlands from other surface water sources will vary a great deal over the course of a year in these agricultural watersheds. Other research has indicated that spatial variability with wetlands is particularly important for measurements of both dissolved reactive and total phosphorus (Detenbeck *et al.* 1996). Chloride values may also be indicative of a trend toward salinization of surface waters by road salt reported in the northeastern part of the U.S. (Kaushal *et al.* 2005). Two of the mitigation wetlands with relatively high chloride levels (Dike and New Hampton) are located near four-lane highways; however, the Hay-Buhr reference site is located in a relatively isolated area with respect to roadways and also had relatively high chloride levels.

No statistically significant difference in nutrient levels between mitigation and reference sites was detected, suggesting that mitigation wetlands, although not intentionally designed for nutrient retention, are nevertheless performing that function. Study sites with well defined in- and out-flows exhibited a variety of nitrogen retention behaviors. Although flows were not measured, Dike exhibited higher percent nitrate removal at lower flow (longer retention time) conditions which prevailed later in the summer. New Hampton also was relatively effective at nitrate removal, though the hydrology was more complex due to two small additional inflows. South Point, with low concentrations of nitrate in the inflow, had a low percentage removal. Hay-Buhr typically had a relatively low surface inflow, but percent removal of nitrate was somewhat erratic. Numerous possible explanations exist for this observation, including shallow groundwater flow from the surrounding agricultural areas that could result in a more constant nitrate concentration in this wetland

A number of differences between mitigation wetlands and reference wetlands were found for individual water quality parameters. Higher pH and lower conductivity are consistent with increased photosynthesis and may explain the higher turbidity and TSS in the mitigation wetlands. Higher levels of ammonia may also result from the breakdown of more abundant plant biomass in the mitigation wetlands. While these observations are consistent with increased plant life in the mitigation wetlands, the hydrology of an individual site has a significant influence on the water chemistry observed, and a more detailed study of the sites would be necessary to definitively assess the sources of the observed differences.

The comparison of the water quality in reference and mitigation wetlands in Iowa is complicated by the paucity of “natural” wetlands in the state. Ideally, pairing a reference and mitigation wetland with similar hydrology and geology would allow a more detailed analysis of the effectiveness of the mitigation wetlands with respect to water quality. However, the data set that was gathered in this study allows some comparisons to be made. The differences that were found to be statistically significant were consistent with higher levels of photosynthetic activity in the reference wetlands. In turn, this could be explained by their hydrology, which was either isolated (i.e. Engeldinger and Doolittle) or with relatively low flow (Hay-Buhr). A similar study examining mitigation wetlands in Ohio also found few differences between reference and mitigation wetlands with respect to water quality (Fennessy *et al*. 2004). In comparing 5 reference sites (11 total samples) to 10 mitigation sites (21 samples), the Ohio EPA found significant differences only for pH ($p = 0.05$) and K ($p = 0.024$). While intensive studies of paired wetlands might reveal more subtle differences in water quality, it seems safe to say that occasional grab sampling is unlikely to reveal differences between mitigation and natural wetland sites. Interestingly, the Ohio EPA study did show significant differences in soil chemistry and physical properties between the two sets of study sites.

The results of this study suggest that the effects of landscape (i.e. disturbance, road density, etc.) appear to be the same for both mitigation and reference wetlands. No significant difference was detected between mean LDI, road density, connectivity or nutrient loading for mitigation sites and reference sites, suggesting that both groups are may be equally affected by these factors. However, due to the small sample size in this study a true relationship may have gone undetected.

Recent studies have found correlations between road densities and plant or animal species. Species richness of birds, herptofauna, and plants (but not mammals) in southeastern Ontario wetlands was negatively correlated with road density (Findlay and Houlahan 1997). It is thought that road density may have an effect on amphibians (Fahrig et al. 1995, Hels and Buchwald 2001). In addition, traffic mortality, road avoidance, and road salt runoff can have a negative effect on animal populations (Forman and Deblinger 2000).

The lack of significance between road density and biodiversity in the current study suggests that roads are not having a significant impact on plant and animal biodiversity. Considering the mostly agricultural landscapes in which mitigation and reference wetlands in Iowa are located, it is possible that effects from road density are masked or co-mingled with agricultural effects on species. A true relationship between road density and species diversity may exist, but, due to small sample size and/or a lack of a sufficiently powerful statistical test, a true relationship may have gone undetected.

Table 1. Mitigation and reference wetlands evaluated for ecological performance (2005-2006) Iowa, USA.

Site Name	County	Total Area (ha)	Existing (Planned) Wetland Area (ha)	Ratio Existing Wetland Area to Total Area	Restoration Type	Year Constructed	Age Class	Size Class
Mitigation Sites								
South Point	Polk	16.2	4.0	25%	Creation	2004	1	Large
Pleasantville	Marion	5.3	1.6	31%	Restoration	2002	3	Small
Mink Creek	Floyd	14.6	3.2	22%	Creation	1998	>5	Small
Badger Creek	Warren	55.4	2.4	44%	Restoration	2000	5	Large
Brush Creek	Marion	42	6.4	15%	Restoration	1998	>5	Medium
Wickiup Hill	Linn	19.3	14.6	31%	Restoration	2000	5	Medium
Grooms	Wapello	6.0	2.4	40%	Restoration	2004	1	Medium
New Hampton	Chickasaw	38.7	4.2	11%	Restoration	2002	3	Medium
Palisades	Linn	8.2	1.7	20%	Creation	2000	5	Small
Dike	Grundy	19.4	4.6	25%	Creation	1999	>5	Large
Boevers	Bremer	7.3	1.6	22%	Creation	2005	1	Small
Jarvis	Henry	28.3	18.2	64%	Restoration	2003	3	Large
Reference Sites								
Hay Buhr	Bremer	46.5	39.4	85%	N/A	N/A	>5	Large
Engeldinger Marsh	Polk	19.3	7.0	36%	N/A	N/A	>5	Medium
Doolittle Prairie	Story	9.7	5.7	59%	N/A	N/A	>5	Small

Table 2. Summary of species diversity and diversity rankings as estimated by effective number of species (Hill's N1) at mitigation and reference wetlands

	Effective Number of Species (Rank Corrected for Ties)[1]									
	Algae	**Protozoa**	**Aquatic Invertebrates**	**Native Herbaceous Plants**	**Butterflies**	**Amphibians**	**Reptiles**	**Birds**	**Mammals**	**Overall Site Rank[1]**
Mitigation Sites										
South Point	49.60 (1)	13.67 (9)	18.15 (1)	47.67 (1)	13.99 (3)	4.96 (2)	4.46 (3)	11.28 (8)	4.66 (8)	1
Pleasantville	46.29 (2)	18.81 (4)	11.54 (4)	32.26 (8)	12.34 (7)	2.84 (7)	1.75 (9.5)	4.82 (15)	6.97 (1)	3.5
Mink Creek	39.57 (3)	25.20 (2)	11.44 (5)	35.08 (7)	12.93 (5)	1.99 (9)	1.00 (13.5)	12.28 (7)	6.01 (6)	3.5
Badger Creek	22.02 (8)	14.68 (7)	9.10 (8)	42.32 (4)	14.83 (2)	2.08 (8)	5.86 (2)	7.74 (14)	4.20 (10)	5
Brush Creek	18.79 (9)	20.43 (3)	10.00 (7)	31.76 (10)	9.97 (9)	1.13 (13)	2.05 (5)	15.97 (4)	4.12 (11)	8
Wickiup Hill	18.16 (10)	8.49 (12)	4.11 (13)	36.07 (5)	12.68 (6)	1.89 (10)	1.89 (7.5)	18.02 (1)	6.02 (5)	7
Grooms	14.62 (13)	3.77 (14)	2.55 (14)	20.17 (13)	11.62 (8)	5.59 (1)	2.83 (4)	16.61 (3)	4.48 (9)	9
New Hampton	37.84 (4)	10.62 (10)	14.63 (3)	19.84 (14)	6.17 (15)	1.15 (12)	2.00 (6)	15.01 (5)	2.47 (14)	10
Palisades	30.26 (5)	15.64 (6)	15.00 (2)	22.60 (12)	8.66 (12)	1.11 (14)	1.51 (11)	8.33 (11)	3.52 (13)	11
Dike	14.43 (14)	16.31 (5)	7.02 (10)	24.70 (11)	9.33 (11)	3.43 (6)	1.00 (13.5)	7.88 (13)	6.35 (3)	12.5
Boevers	15.53 (11)	4.21 (13)	4.22 (12)	35.35 (6)	8.39 (13)	3.94 (5)	1.00 (13.5)	10.92 (9)	6.25 (4)	12.5
Jarvis	0.00 (15)	0.00 (15)	0.00 (15)	14.55 (15)	13.93 (4)	4.34 (3)	1.89 (7.5)	14.85 (6)	2.31 (15)	14
Reference Sites										
Hay-Buhr Area	28.30 (6)	28.82 (1)	8.47 (9)	46.02 (2)	9.57 (10)	1.85 (11)	7.56 (1)	17.41 (2)	6.73 (2)	2
Engeldinger Marsh	25.77 (7)	14.53 (8)	10.42 (6)	31.87 (9)	14.95 (1)	4.31 (4)	1.00 (13.5)	10.86 (10)	5.25 (7)	6
Doolittle Prairie	15.31 (12)	8.87 (11)	4.55 (11)	42.38 (3)	7.43 (14)	1.00 (15)	1.75 (9.5)	8.19 (12)	4.06 (12)	15
Mann-Whitney U	19	22	20	28	19	24	18	18	21	19
2-Tailed P[2]	0.95	0.63	0.84	0.18	0.95	0.45	1.0	1.00	0.73	0.89

[1] 1 = Highest Rank; 15 = Lowest Rank
[2] Significant at $\alpha = 0.05$

Table 3. Comparison of water chemistry means at reference vs. mitigation wetlands (grab samples and outflows only)

Analyte	**Mean (mitigation)**	**Mean (reference)**	**Significantly different?**	**P Value**
DO	9.10	10.74	No	0.349
pH	8.17	7.73	Yes	0.038
Conductivity	313	393	Yes	0.037
Turbidity	20.7	8.6	Yes	0.044
TSS	33.1	8.2	Yes	0.034
NH_3	0.18	0.08	Yes	0.031
DRP	0.19	0.45	No	0.222
Total P	0.89	1.13	No	0.556
Total N	3.70	4.75	No	0.432
NO_3^--N	2.08	2.36	No	0.778
SO_4^{2-}	14.6	16.7	No	0.517
Cl^-	14.6	19.1	No	0.122
COD	45.5	66.6	No	0.285

Table 4. Average nutrient concentrations and nutrient score

Site	Site #	DRP	Total P	Total N	NO_3^--N	Nutrient Score[1]
Mitigation Sites						
South Point	2O	0.05	0.50	2.05	0.77	0.95
Pleasantville	3	0.13	0.85	2.57	0.36	0.90
Mink Creek	10E	0.28	1.19	1.90	0.23	1.36
Badger Creek	9	0.39	1.46	BDL	0.21	1.76
Brush Creek	11O	2.23	3.62	4.25	0.76	9.66
Wickiup Hill	*7*	*0.17*	*2.23*	*5.00*	*0.19*	*0.88*
New Hampton	4SO	0.37	0.48	5.83	5.05	6.54
Palisades	6SO	0.09	0.56	2.12	4.51	4.85
Dike	12O	0.05	0.39	11.92	9.43	9.64
Boevers	*8*	*0.06*	*1.53*	*7.75*	*4.20*	*4.46*
Reference Sites						
Hay-Buhr	14SO	0.26	0.43	6.30	5.43	6.48
Engeldinger	13A	0.27	0.49	1.12	0.32	1.41
Doolittle	*15*	*2.97*	*4.29*	*2.90*	*0.10*	*12.00*

[1]Rows in italics were sampled ≤ 3 times.

Table 5. Percent wetland, percent natural areas, percent agriculture, landscape development index (LDI), road density, habitat diversity index (HDI), and statistical comparison of averages at mitigation and reference wetlands. No significant differences were detected.

	Within 2km of a Site[1]					
	% Wetland[2]	% Natural Areas	% Agriculture	Landscape Development Index (LDI)	Road Density (m/ha)	Habitat Diversity Index (HDI)
Mitigation Sites						
South Point	8	43	51	3.05	8.92	6.1
Pleasantville	4	37	60	3.04	10.52	4.9
Mink Creek	0.5	18	78	3.76	12.20	1.6
Badger Creek	2	27	71	3.14	13.38	9.3
Brush Creek	0.8	7	83	4.27	21.96	9.1
Wickiup Hill	26	55	34	2.62	15.59	17.3
Grooms	0.7	32	63	3.00	8.75	2.4
New Hampton	0.8	6	63	5.33	20.45	2.4
Palisades	0.3	22	74	3.64	12.58	3.2
Dike	2	2	89	4.53	19.16	5.2
Boevers	19	25	73	3.59	14.94	3.5
Jarvis	5	36	59	3.21	13.26	11.0
Mean	5.8	24.9	67.2	3.60	14.4	6.3
Reference Sites						
Hay-Buhr Area	9	30	68	3.41	10.60	6.9
Engeldinger Marsh	2	16	75	4.10	18.82	10.9
Doolittle Prairie	0.3	14	83	3.86	13.06	3.3
Mean	3.8	20.0	75.3	3.80	14.20	7.0
Mann-Whitney U	19.5	23	25.5	24	19	22
2-Tailed P[3]	0.84	0.54	0.29	0.45	0.95	0.63

[1] Except HDI scores, which are representative of conditions within the site boundaries rather than within 2km
[2] % Wetland is a subset of % Natural Areas
[3] Significant at $\alpha = 0.05$

Table 6. Summary of nutrient load calculations (EPA)
mitigation and reference wetlands evaluated for ecological performance (2005-2006) Iowa, USA

	Nitrogen Load	Phosphorus Load	Sediment Load	Watershed Area (ha)
Mitigation Sties				
South Point	1.77	2.74	0.11	373.16
Pleasantville	2.04	3.27	0.15	46.97
Mink Creek	2.09	3.37	0.00	165.73
Badger Creek	1.66	2.43	0.43	109.10
Brush Creek	2.02	3.30	0.46	590.17
Wickiup Hill	1.27	1.64	0.16	286.74
Grooms	1.89	2.91	0.04	15.50
New Hampton	2.10	3.47	0.01	360.94
Palisades	1.88	2.90	0.27	22.62
Dike	2.19	3.57	0.02	408.39
Boevers	2.23	3.65	0.00	3.71
Jarvis	1.51	2.18	0.01	130.67
Reference Sites				
Hay-Buhr Area	1.91	3.00	0.01	164.92
Engeldinger Marsh	1.42	1.93	0.04	109.59
Doolittle Prairie	2.06	3.29	0.00	58.72

Table 7. Selected site characteristics at mitigation and reference wetlands. No significant differences were discovered between mitigation and reference wetlands. In addition, no significant relationships were found among these site characteristics and biodiversity measurements.

	Site Characteristics						
	Interior Edge Length (ft.)	**Interior Edge/ Site Area**	**Total Edge Length (ft.)**	**Total Edge/ Site Area**	**Total # of Plant Communities**	**Interspersion[1]**	**Average Community Size (ha)**
Mitigation Sites							
South Point	16051.83	354.23	21683.6	478.51	10	24	3.93
Pleasantville	5836.97	413.35	9392.82	665.16	5	9	1.47
Mink Creek	19300.59	222.59	27638.62	318.75	7	63	10.32
Badger Creek	40596.06	293.87	52653.39	381.15	12	29	4.75
Brush Creek	26497.41	313.36	38388.81	453.99	11	39	6.39
Wickiup Hill	17438.37	365.76	23435.54	491.55	13	30	4.91
Grooms	5455.67	338.76	9414.83	584.59	3	9	1.47
New Hampton	12558.62	315.58	18824.4	473.03	9	17	2.78
Palisades	9600.7	526.79	13306.59	730.13	6	16	2.62
Dike	9551.3	202.14	15589.33	329.93	8	10	1.64
Boevers	4605.82	369.64	7611.99	610.9	7	10	1.64
Jarvis	24880.46	275.73	40077.29	444.15	11	28	4.59
Reference Sites							
Hay-Buhr Area	27180.96	235.84	36261.16	314.62	10	31	5.08
Engeldinger Marsh	11101.14	233.25	17718.34	372.29	10	15	2.46
Doolittle Prairie	7333.51	305.64	11545.35	481.18	4	16	2.62

[1]Number of distinct plant communities

CHAPTER 3

EVALUATION AND COMPARISON OF RAPID ASSESSMENT METHODS FOR USE IN ASSESSING AND MONITORING WETLAND MITIGATION SITES

INTRODUCTION

A large number of rapid assessment methods are currently in use and have been developed for various applications. Fennessy et al. (2007) used several levels of screening to evaluate 40 existing methods for their potential to assess ecological integrity or condition using four evaluation criteria: The method, 1) can be used to measure condition, 2) is truly rapid, 3) includes a site visit, and 4) can be verified. Of the methods evaluated, only six met all four criteria. In addition, Fennessy et al. (2007) identified issues that must be addressed when adopting or developing a rapid assessment method for wetland monitoring and assessment.

Currently, Iowa Department of Transportation (DOT) wetland mitigation projects are assessed by comparing the total number of mitigation acres attained to the number of acres required by Clean Water Act (CWA) Section 404 permits (VanDeWalle et al., 2007). Although this method provides a measure of regulatory compliance, it does not address how a site is performing ecologically. By incorporating an ecological performance measure into the wetland monitoring program, the Iowa DOT may be able to improve the overall ecological effectiveness of compensatory mitigation. In addition, an ecological performance measure that is applicable to both natural and mitigation wetlands would also provide a measure of how effective mitigation wetlands are at replacing the functions and values of impacted wetlands.

The purpose of this study was to evaluate selected existing rapid assessment methods to determine the appropriateness of each for assessing and characterizing ecological performance of mitigation sites and to develop a conceptual framework for developing a new, or adapting an existing, rapid assessment method for use by the Iowa DOT.

METHODS

Prior to beginning the study, U.S. Environmental Protection Agency (EPA) publication EPA 620-R-04-009 (Fennessy et al., 2004) was reviewed for EPA recommended rapid assessment methods. Based on this review, and the Iowa DOT's intended use of the rapid assessment method, three methods were selected for evaluation:

1. Florida Wetland Rapid Assessment Procedure (FWRAP) (Miller and Gunsalus, 1999) – Designed for mitigation projects.

2. Ohio Rapid Assessment Method (ORAM) (Mack, 2001) – Designed for freshwater wetlands.

3. Washington State Wetland Rating System (Western) (WWRS) (Washington State Department of Ecology, 1993) – Designed for freshwater wetlands in western Washington.

In addition to the three rapid assessment methods listed above, a fourth method, the Wetland Mitigation Quality Assessment (WMQA) (Balzano et al., 2002) designed specifically for wetland mitigation sites, was also chosen for evaluation.
Twelve Iowa DOT mitigation wetlands and three reference wetlands in Iowa were selected as study sites to evaluate each of the four rapid assessment methods (Table 1). Each study site was visited once during September and early October 2006 and dataforms for each of the four rapid assessment methods were completed for each study site.

Species richness and abundance data were collected at each of the 15 study sites from 2005 – 2006 using intensive biological inventories for nine species groups, including algae, protozoa, aquatic invertebrates, plants, butterflies, amphibians, reptiles, birds and mammals (see VanDeWalle et al. (2007) for detailed methods). In addition to collection of ecological data, a landscape assessment was completed at each site that included calculation of a landscape development index (LDI), habitat diversity index (HDI), road density and percent natural land within a 2 km buffer of a site, and interspersion (i.e. number of distinct communities at a site).

Data collected during the intensive biological inventories were used as a measure of ecological performance. For the purposes of this study, ecological performance was measured using Hill's N1 (Hill, 1973) as a representative measure of species diversity. Hill's N1 is one method of calculating the "effective number of species" (MacArthur, 1965; Hill, 1973). It is the exponential of the Shannon index; unlike Shannon's index, Hill's N1 represents a true diversity that behaves linearly and is therefore easier to interpret ecologically than the Shannon form (Peet, 1974). Because it is derived from Shannon's index, it also has the advantage of not emphasizing either rare or common species (Jost, 2006).

Site scores for each of the four rapid assessment methods were ranked from 1 – 15, with 1 representing the highest score. Diversity for each of three species groups (algae and aquatic micro-invertebrates, native herbaceous plants and animals [Lepidoptera, amphibians, reptiles, birds, and mammals]) was calculated using the effective number of species and used to determine a rank (1 – 15) for each site for each species group. In addition, overall diversity was calculated for each site using the effective number of species to determine an average rank for each site. The sites were then given an overall diversity ranking of 1 – 15 based on the average rank, with 1 representing the highest overall species diversity. Each site was also ranked based on the landscape assessment measures. Rapid assessment rankings were then compared to ecological performance and landscape rankings to determine which of the four methods best reflects the patterns seen in the ecological and landscape data. Regression analyses were performed using the data analysis extension in Microsoft Excel®.

RESULTS

A summary of rapid assessment scores for each of the 12 mitigation and three reference sites using each of the four methods is shown in Table 2. When all 15 sites are compared using raw scores, the Hay-Buhr Area (reference site) scored the highest under both the ORAM and WWRS methods and tied with South Point (mitigation site) for the highest score under the WMQA method. Engeldinger Marsh (reference site) scored the highest under the FWRAP method with the Hay-Buhr Area a close second. Grooms (mitigation site) scored the lowest under the ORAM, FWRAP and WMQA methods and second to lowest under the WWRS method. Doolittle Prairie (reference site) scored the lowest under the WWRS method and second to the lowest under the FWRAP and WMQA methods. In contrast to the other methods, Doolittle Prairie scored right in the middle of the group under the ORAM method.

When just the mitigation sites are compared, results vary widely between methods. A different site scored highest under each of the four methods: Mink Creek scored the highest under the ORAM method, New Hampton scored the highest under the WWRS method, Dike scored the highest under the FWRAP method and South Point scored the highest under the WMQA method. When the three highest scoring sites under each method are compared, only two sites are found in the top three of more than one method. New Hampton appears in the top three of all four methods and Brush Creek is appears in the top three of two methods (ORAM and WWRS).

Three of the four methods (ORAM, WWRS and WMQA) rate a site based on the raw score that it receives (Table 2). The FWRAP method results in only a raw score for a site. The rating provided by the ORAM method proved to be the least useful for comparing sites. Under the ORAM method, all but one site (Grooms) was rated a Category 3 site (superior habitat or superior hydrological or recreational values, high levels of diversity, high proportion of native species, and includes wetlands which contain or provide habitat for T&E species, bogs, fens, vernal pools or wetlands that are regionally scarce).

Under the WWRS method, all three of the reference sites were rated as Category 1 (documented occurrence of a threatened and endangered species, wetland on record as a Natural Heritage Inventory site, regionally significant waterfowl or shorebird concentration area, wetland with irreplaceable wetland functions (bog, fen, mature forested wetland) or wetland of local significance) (Table 2). No mitigation sites received a Category 1 rating. The largest percentage of the mitigation sites (67%) were rated as Category 2 (documented occurrence of sensitive species, documented occurrence of priority habitats and species recognized by state agencies, wetlands with significant functions which may not be adequately replicated (bogs, fens, etc), freshwater wetlands with significant habitat value or wetlands of local significance) primarily due to a score greater than 21. Four sites (33%) were rated as Category 3 (does not meet any Category 1 or 2 criteria and has a habitat score <21 or is identified a Category 3 wetland of local significance).

The WMQA method rates a site's potential to provide desirable wetland functions and values (i.e. wildlife habitat, etc.) (Balzano et al., 2002). Of the 12 mitigation sites, eight (67%) received an "A" rating (high potential to provide desirable wetland functions and values) (Table 2). The remaining four sites (33%) received a "B" rating (moderate potential to provide desirable wetland functions and values) (Table 2). None of the 12 mitigation sites were rated as having low or poor potential to provide desirable wetland functions and values (C or D rating). When reference sites are compared to mitigation sites, proportionally the two groups score the same. Two of the three (67%) reference sites (Hay-Buhr Area and Engeldinger Marsh) received an "A" rating and one of the three (33%) (Doolittle Prairie) received a "B" rating.

A summary of site rankings for each of the 12 mitigation and three reference sites based on rapid assessment scores, diversity for each of the three species groups and overall diversity is shown in Table 3. The Hay-Buhr Area ranked highest (#1) under the ORAM and WWRS methods and tied with South Point as highest under the WMQA method. Engeldinger Marsh ranked highest under the FWRAP method. When diversity is compared, South Point ranked highest in native herbaceous plant, animal and overall diversity, while Pleasantville and Mink Creek tied for highest diversity of algae and aquatic micro-invertebrates.

No significant correlation was found between either FWRAP or WWRS and diversity of any of the three species groups or overall diversity (Table 4). A highly significant correlation was found between ORAM and algae and aquatic-microinvertebrates (R^2 = 0.37, p = 0.01); however, no correlation was found between ORAM and diversity of either of the other two species groups or with overall diversity (Table 4).

Significant correlations were found between WMQA and algae and aquatic-microinvertebrate diversity (R^2 = 0.27, p = <0.05) and overall diversity (R^2 = 0.29, p = <0.05) (Table 4). The WMQA method calculates individual scores for each of eight field indicators. When the WMQA Wildlife Suitability variable is compared with diversity, a highly significant correlation is found with animal diversity (R^2 = 0.36, p = 0.01) and a correlation is suggestive with overall diversity (R^2 = 0.25, p = 0.055) (Table 4). No correlation was found between the WMQA Vegetation Composition/Diversity – Ground Cover variable and any of the diversity indicators.

Site rankings for each of the 12 mitigation and three reference sites based on rapid assessment scores were compared to site rankings based on landscape factors, including road density within 2 km of a site, HDI, LDI, percent of natural area within 2 km of a site and interspersion (i.e. number of distinct communities at a site). No correlation was found between any of the four rapid assessment methods and road density or HDI, although a correlation is suggestive between the WMQA Wildlife Suitability variable and HDI (R^2 = 0.23, p = 0.07) (Table 4). In addition, no correlation was found between ORAM, WWRS or FWRAP and LDI (Table 4). However, a significant correlation was found between WMQA (R^2 = 0.27, p = <0.05) and the WMQA Wildlife Suitability variable (R^2 = 0.26, p = <0.05) and LDI (Table 4). No correlation was found between WWRS, FWRAP, WMQA or the WMQA Wildlife Suitability variable and percent of natural area within 2 km of a site (Table 4). However, a significant correlation was found between ORAM and percent of natural area within 2 km of a site (R^2 = 0.27, p = 0.05) (Table 4). No correlation was found between ORAM, FWRAP, WMQA or the WMQA Wildlife Suitability variable and interspersion, although a highly significant correlation was found between the WWRS and interspersion (R^2 = 0.38, p = 0.01) (Table 4).

DISCUSSION

Three of the four methods evaluated in this study (ORAM, WWRS and FWRAP) were among the six methods identified by Fennessy et al. (2007) as meeting all four criteria needed for assessing ecological integrity or condition. The WMQA method was not included in the final 16 methods that were evaluated in-depth in the Fennessy et al. (2007) study.

The ORAM and WWRS methods were designed for use with natural freshwater wetlands, and as such, some of the questions on the dataforms for these methods are difficult to apply to constructed mitigation sites, such as questions inquiring about the presence of dikes or weirs on the site or if the site has been graded or filled. In most cases, one or more of these water control structures are present on a mitigation site and/or grading activities have taken place on the site as part of construction. In the context of these methods, these structures or activities are seen as a negative and may lower the overall score. However, in the case of most mitigation sites, these structures or activities are necessary for the success of the site and therefore should not be viewed as negative.

The FWRAP and WMQA methods were designed for use with mitigation wetlands and therefore many of the questions are more suited to mitigation sites. Neither the FWRAP nor WMQA dataforms contain questions specifically inquiring about water control structures or grading activities on the site. Both dataforms do have general questions concerning the level of disturbance on the site. Since WMQA was designed specifically for mitigation sites some of the individual variables in the methodology are not necessarily appropriate for natural wetlands and may need to be revised if used for natural wetlands

(Hatfield et al., 2004). Although, Hatfield et al. (2004) do not recommend deleting any of the variables as they provide valuable information on wetland function that could be useful from a resource management perspective.

Because each of the methods was designed for use in a specific state (Ohio, Washington, Florida or New Jersey), some of the questions on certain dataforms are state specific and are not applicable to Iowa. For example, Metric 5 in ORAM, which gives extra points for special wetlands such as Lake Erie coastal/tributary wetland and Lake Plain Sand Prairie, and the WWRS office form and Question 3 on the WWRS field form which rate a wetland based on a Washington regulatory category. If one of these methods were to be adopted, it would be necessary to revise or eliminate these questions to accurately reflect conditions in Iowa. Questions on both the FWRAP and WMQA dataforms are more general and do not include state specific questions.

The primary objective of this study was to evaluate selected existing rapid assessment methods to determine the appropriateness of each for assessing and characterizing ecological performance of Iowa DOT wetland mitigation sites. Fennessy et al. (2007) established four criteria that a rapid assessment method must meet in order to effectively assess ecological integrity or condition:

1. Can be used to measure ecological condition.

Fennessy et al. (2007) concluded that methods best suited to measure ecological condition do so by providing a quantitative measure of where a wetland lies on a continuum ranging from full ecological integrity (i.e. reference condition) to highly degraded (i.e. poor condition).

A numerical raw score is computed with all four methods evaluated in this study and can be used to compare wetlands to the reference condition (Table 2), although in certain circumstances not all wetlands receive a numerical score with the WWRS method (Fennessy et al., 2007). The FWRAP method allows the user to adjust the score based on site conditions, but is not intended to be used to compare different types of wetlands (Fennessy et al., 2007). The WMQA method requires the reviewer to assign a value (0-3) for each field indicator based on “best fit”, and the reviewer is allowed to assign the highest value (3) if the site does not currently meet a specific field indicator, but it is felt that the site has the potential to develop the field indicator over time (Balzano et al., 2002).

The ORAM, WWRS and WMQA methods use the raw score to derive a final rating (Table 2). As discussed above, the ORAM rating was the least useful, as all but one site was rated Category 3, suggesting that all of the sites were of superior quality, with high levels of diversity and more or less equal in quality or condition. Ecological data collected in the field at each site indicate that this is not the case, and that significant differences in ecological condition do exist between sites.

Using the WMQA rating system, 67% of both the mitigation and reference sites were rated as “A” and 33% of both groups were rated as “B”, suggesting that mitigation wetlands are performing similarly to reference wetlands, a result consistent with that of a comprehensive ecological performance study conducted at the same sites (See Chapter 2 and VanDeWalle et al., 2007).

2. The method must be rapid.

A rapid method must be able to provide an accurate assessment of condition in a relatively short amount of time (Fennessy et al., 2007). In the case of the methods evaluated in this study, instructions for all four methods are, for the most part, easily understood and dataforms for each method can be completed relatively quickly. Regardless of method, the assessment of a site can be completed in less than one day. The speed of completing the assessment increases with the reviewer's experience with the method and familiarity with the site.

3. The assessment must be an on-site assessment.

An accurate evaluation requires a site visit to ensure the method captures the current condition of the wetland (Fennessy et al., 2007). All four methods evaluated as part of this study require a site visit and also include some level of office assessment. Depending on the size and complexity of the site, and the reviewer's knowledge of the site, field dataforms for any of the methods can be completed in <1 – 2 hours with a similar amount of time required for the office assessment.

4. The validity of the method can be determined.

Fennessy et al. (2007) conclude that a central component in the development of a rapid assessment method is to determine its accuracy with more comprehensive ecological assessment data. In this study, diversity, calculated using comprehensive ecological data collected at each site, was used as a measure of ecological performance, and potential relationships between rapid assessment data and ecological and landscape data were explored.

Of the four methods, WMQA had the best fit with the ecological data, showing significant correlations with algae and aquatic micro-invertebrates as well as overall species diversity (Table 4). In addition, the correlation between the WMQA Wildlife Suitability variable and animal diversity was highly significant. ORAM data were correlated only with algae and micro-invertebrates. No correlation was found between WWRS or FWRAP data and the ecological data.

Potential relationships between rapid assessment data and landscape data were also explored with mixed results. WMQA showed significant correlations between LDI and both the overall WMQA score and the WMQA Wildlife Suitability variable score. LDI was not correlated with any of the other methods and no correlation was found between any of the four methods and road density or HDI. ORAM showed a significant correlation with percent natural land within a 2 km buffer of a site and WWRS showed a highly significant correlation with interspersion.

Conclusions and Recommendations

The results of this study suggest that of the four methods evaluated, the WMQA method provides the best measure of ecological performance as measured by biodiversity. The WWRS and FWRAP methods resulted in the worst measure.

The WMQA method was developed to evaluate the potential of a mitigated wetland to function similarly to a natural wetland (Balzano et al., 2002). Hatfield et al. (2004) tested WMQA at 10 mitigated wetlands and 14 natural wetlands in New Jersey and concluded that the method was capable of assessing potential functioning of mitigation wetlands. In this study, a statistically significant correlation was found between WMQA score and overall biodiversity based on comprehensive ecological data collected at each of the study sites, providing further support that WMQA is capable of accurately assessing ecological condition. It should be noted however that sample size in this study was low and further testing would be needed to confirm this relationship.

In addition to providing an accurate measure of ecological condition as verified by comparison with field data, WMQA is also a truly rapid assessment requiring an on-site visit to confirm existing conditions, thereby meeting all four of the criteria established by Fennessy et al. (2007).

Based on the results of this study, it is recommended that the Iowa DOT adopt the WMQA method for use with mitigation and natural wetlands in Iowa. The method already performs well with mitigation wetlands and only slight modification would be needed to adapt the method for use with natural wetlands. Use of the method could benefit the Iowa DOT in two specific areas:

1. As a performance measure for wetland mitigation sites. Currently, Iowa DOT annual wetland mitigation monitoring reports are designed to demonstrate regulatory compliance, as required under CWA Section 404. However, by incorporating the rapid assessment method into the wetland monitoring program, a measure of ecological performance could easily be incorporated into the annual report, demonstrating to resource agencies that a site not only meets permit requirements, but also is functioning similar to a natural wetland and thus replacing lost functions and values.

   By definition, WMQA rates a site's potential to provide desirable wetland functions and values. Therefore, completing the assessment during the first year of monitoring may provide a predictor of a site's future ecological success. If a site's current conditions, versus expected conditions, are evaluated, annual scores could be compared to previous scores to track a site's progress, similar to what is currently done with a site's wetland acreage. If a site scores low or its scores do not improve with time, the assessment may provide an indication of where the site is deficient and remedial measures could be taken to correct the deficiency if needed. If desirable, variables could be added to the method to address regulatory compliance.

2. As a measure of how effective mitigation wetlands are at replacing the functions and values of impacted wetlands. Resource agencies and critics of wetland mitigation often question whether wetland mitigation sites are effectively replacing the functions and values that are lost when a wetland is impacted, even if the acreage is replaced. The rapid assessment could be completed during the initial wetland delineation and used to evaluate the ecological condition of wetlands that would be impacted by the transportation project. Once the mitigation site is completed, the mitigation site could be assessed and the score compared with those from the impacted wetlands to see how the mitigation wetland scores relative to the ecological condition of the impacted wetlands. Because WMQA is a truly rapid assessment, it would not significantly increase the amount of time needed to complete the fieldwork.

Table 1. Characteristics of mitigation and reference wetlands used to evaluate rapid assessment methods.

Site Name	Site Size (Acre)	Wetland Size (Acre)	Year Constructed	Mitigation Type
Mitigation Sites				
Badger	137	60	2000	Restoration
Boevers	18	4	2006	Creation
Brush Creek	104	16	1998	Restoration
Dike	48	12	1999	Creation
Grooms	15	6	2004	Restoration
Jarvis	70	45	2003	Restoration
Mink Creek	36	8	1998	Creation
New Hampton	96	10	2002	Restoration
Palisades	20	4	2001	Creation
Pleasantville	13	4	2002	Restoration
South Point	40	10	2004	Creation
Wickiup Hill	48	15	2000	Restoration
Reference Sites				
Doolittle Prairie	24	14	-	Natural
Hay-Buhr	115	97	-	Natural
Engeldinger Marsh	47.6	17.2	-	Natural

Table 2. Rapid assessment scores for mitigation and reference wetlands evaluated in the study.

	ORAM		WWRS		FWRAP	WMQA	
Site	Score (0 – 100)	Rating[1]	Score (0 – 100)	Rating[2]	Score[3] (0 – 1)	Rating[4]	Score[3] (0 – 1)
Mitigation Sites							
Grooms	27	Category 2	12	Category 3	0.53	B	0.59
South Point	60	Category 3	27	Category 2	0.66	A	0.92
Pleasantville	56	Category 3	19	Category 3	0.80	A	0.80
New Hampton	65	Category 3	38	Category 2	0.81	A	0.88
Jarvis	39	Category 3	33	Category 2	0.66	A	0.80
Palisades	51	Category 3	16	Category 3	0.65	B	0.70
Wickiup Hill	45	Category 3	28	Category 2	0.69	B	0.71
Boevers	40	Category 3	20	Category 3	0.62	B	0.72
Badger Creek	58	Category 3	26	Category 2	0.73	A	0.91
Mink Creek	71	Category 3	29	Category 2	0.67	A	0.81
Brush Creek	62	Category 3	32	Category 2	0.80	A	0.77
Dike	61	Category 3	30	Category 2	0.82	A	0.76
Reference Sites							
Engeldinger Marsh	78	Category 3	24	Category 1	0.92	A	0.91
Hay-Buhr Area	97	Category 3	47	Category 1	0.91	A	0.92
Doolittle Prairie	59	Category 3	10	Category 1	0.55	B	0.67

[1] Category 1 - Hydrologically isolated, low species diversity, no significant habitat or wildlife use, limited potential to achieve beneficial wetland functions and/or a predominance of non-native species.

Category 2 - Support moderate wildlife habitat or hydrological or recreational functions, dominated by native species but no T&E species.

Category 3 - Superior habitat or superior hydrological or recreational values, high levels of diversity, high proportion of native species, and includes wetlands which contain or provide habitat for T&E species, bogs, fens, vernal pools or wetlands that are regionally scarce.

[2] Category 1 – Documented occurrence of a T&E species, wetland on record as a NHI site, regionally significant waterfowl or shorebird concentration area, wetland with irreplaceable wetland functions (bog, fen, mature forested wetland) or wetland of local significance.

Category 2 – Documented occurrence of sensitive species, documented occurrence of priority habitats and species recognized by state agencies, wetlands with significant functions which may not be adequately replicated (bogs, fens, etc), freshwater wetlands with significant habitat value or wetlands of local significance.

Category 3 – Does not meet any Category 1 or 2 criteria and has a habitat score <21 or is identified a Category 3 wetland of local significance.

Category 4 –Wetlands less than 1acre and hydrologically isolated and comprised of on vegetated class dominated by 1 species, wetlands less than 2 acres and hydrologically isolated with one vegetated class and 90% cover of invasive species or a pond less than 1 acre excavated from upland without a surface water connection to a Waters of the U.S.

[3] Score of 1 has the highest functional value.

[4] A (0.75 to 1.0) - High potential to provide desirable wetland functions and values.
B (0.50 to <0.75) - Moderate potential to provide desirable wetland functions and values.
C (0.25 to <0.50) - Low potential to provide desirable wetland functions and values.
D (0.00 to <0.25) - Poor potential to provide desirable wetland functions and values.

Table 3. Site rankings based on rapid assessment method and species diversity.

	Rank[1, 2]							
	Rapid Assessment Method				**Diversity**			
Site	ORAM	WWRS	FWRAP	WMQA	Algae & Aquatic Micro-Invertebrates	Native Herbaceous Plant	Animal	Overall
Mitigation Sites								
Grooms	15	14	15	12	14	13	2	9
South Point	7	8	10	1	3	1	1	1
Pleasantville	10	12	5	5	2	8	8	3.5
New Hampton	4	2	4	3	6	14	13	10
Jarvis	14	3	11	5	15	15	6	14
Palisades	11	13	12	10	4	12	14	11
Wickiup Hill	12	7	8	9	14	5	4	7
Boevers	13	11	13	8	13	6	11	12.5
Badger Creek	9	9	7	2	9	4	6	5
Mink Creek	3	6	9	4	2	7	10	3.5
Brush Creek	5	4	6	6	7	10	10	8
Dike	6	5	3	7	10	11	12	12.5
Reference Sites								
Engeldinger Marsh	2	10	1	2	8	9	7	6
Hay-Buhr Area	1	1	2	1	5	2	3	2
Doolittle Prairie	8	15	14	11	11	3	15	15

[1] 1 = Highest Rank; 15 = Lowest Rank
[2] Duplicate rankings indicate a tie

Table 4. Probability matrix comparing rapid assessment data with ecological and landscape data.

Rapid Assessment Method	*p*-Value								
	Ecological				Landscape				
	Algae & Aquatic Micro-Invertebrates Diversity	Native Herbaceous Plant Diversity	Animal Diversity	Overall Diversity	Road Density	LDI	HDI	% Natural Land Within 2 km	Interspersion
ORAM	**0.01**	0.33	0.66	0.18	0.11	0.22	0.99	0.05	0.12
WWRS	0.70	0.66	0.58	0.95	0.14	0.88	0.23	0.63	**0.01**
FWRAP	0.16	0.93	0.84	0.33	0.11	0.39	0.16	0.16	0.52
WMQA	0.04	0.22	0.16	0.04	0.97	0.04	0.25	0.98	0.13
WMQA – Wildlife Suitability	0.37	0.28	**0.01**	0.55	0.60	0.05	0.06	0.15	0.26

[1] Shaded = Significant ($p \leq 0.05$)
Shaded and Bold = Highly Significant ($p \leq 0.01$)

CHAPTER 4

CONCLUSIONS AND RECOMMENDATIONS

The results of this comprehensive study of the ecological performance of wetland mitigation sites suggest that mitigation sites in Iowa are performing similar to reference wetlands ecologically. Specifically,

1. Reference wetlands and mitigation wetlands in Iowa are similar in terms of water quality; landscape processes; site conditions; diversity of algae/protozoa/aquatic invertebrates, amphibians, birds, mammals, reptiles; and overall plant and animal diversity. No significant difference was found in overall diversity or within a species group, with the exception of butterflies, as estimated by effective number of species at mitigation and reference wetlands. Because the effective number of species is a measure of the number of common species at a site, this result suggests that the number of common species within each species group is approximately equal between mitigation and reference sites. Because all sites are embedded in the same highly disturbed landscape, these results are not entirely surprising.

2. Differences do exist between mitigation and reference wetlands in Iowa in terms of butterfly diversity. A significant difference between mitigation and reference wetland butterfly diversity was found, with mitigation wetlands having a higher species richness and significantly more rare species than reference wetlands. The reasons for this difference are not clear and more investigation is needed to determine the underlying cause.

3. Reference wetlands and mitigation wetlands in Iowa are different in terms of plant composition and floristic quality. Reference wetlands had more natives, fewer exotics, contained species with wetter indicator status, and more importance of *Carex* species. Age was not a significant factor but cannot be ruled out as an important explanatory variable. Small mitigation sites were composed of wetter plant species compared to larger sites.

4. The *Wetland Mitigation Quality Assessment* is an existing rapid assessment tool that provided the best measure of ecological performance as measured by biodiversity of the four rapid assessment methods evaluated in this study. This tool has the potential be used as both a performance measure for wetland mitigation sites and an assessment tool for wetland impact studies.

5. Although the results of this study suggest that mitigation wetlands are functioning similarly to reference wetlands, due to small sample size and/or a lack of a sufficiently powerful statistical test, it is possible that differences between mitigation and reference wetlands do exist and a true relationship may have gone undetected. Therefore, this study should be viewed as a pilot study and additional research is needed to better understand the environmental variables that influence species composition and the development of wetland mitigation sites.

Recommendations

Based on the results of this study, several recommendations can be made to improve the ecological performance of Iowa DOT wetland mitigation sites:

1. Ordination methods revealed a negative relationship between animal diversity and row crop agriculture. It is recommended that, where possible, the Iowa DOT select mitigation sites with minimal amounts of row crop agriculture within 2 km of a wetland mitigation site.

2. When impossible to avoid agricultural areas, it is recommended that vegetation buffers be employed as a design feature to minimize the risk of sediment loading from surrounding agricultural fields.

3. Because small mitigation sites had significantly greater wetland affinity in terms of plant species, design and construction of mitigation sites should seek to mimic smaller wetlands when possible and appropriate, possibly by constructing sub-basins within an otherwise larger site. More work is needed to determine the optimal wetland size for achieving the highest plant species richness and quality.

4. Because reference wetlands were found to have more *Carex* and native species than mitigation wetlands, Iowa DOT seed mixes should be tailored toward providing more *Carex* and native species for seeding and planting. It is also recommended that seed mixes be designed on an individualized basis by persons knowledgeable of local ecotypes.

5. It is recommended that the Iowa DOT adapt the *Wetland Mitigation Quality Assessment* method for use as a performance measure for wetland mitigation sites and as a measure of how effective mitigation wetlands are at replacing the functions and values of impacted wetlands.

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